

WATER QUALITY

Prevention,

Identification,

and

Management

of

Diffuse Pollution

ADMINISTRATIVE RECORD
INDEX - DOCUMENTS
WATER QUALITY
FOLDER 9, 1104 # 3

Vladimir Novotny / Harvey Olem

R0022966

WATER QUALITY

WATER QUALITY

Prevention, Identification, and Management of Diffuse Pollution

Vladimir Novotny

*Marquette University
Milwaukee, Wisconsin*

Harvey Olem

*Center for Watershed Protection
Olem Associates, Inc.
Herndon, Virginia*



VAN NOSTRAND REINHOLD
New York


R0022968

Copyright © 1994 by Van Nostrand Reinhold

Library of Congress Catalog Card Number 93-14571

ISBN 0-442-00559-8

All rights reserved. No part of this work covered by the copyright hereon may be reproduced or used in any form or by any means—graphic, electronic, or mechanical, including photocopying, recording, taping, or information storage and retrieval systems—without the written permission of the publisher.

 Van Nostrand Reinhold is an International Thomson Publishing company.
ITP logo is a trademark under license.

Printed in the United States of America

Van Nostrand Reinhold
115 Fifth Avenue
New York, NY 10003

International Thomson Publishing GmbH
Königswinterer Str. 418
53227 Bonn
Germany

International Thomson Publishing
Berkshire House, 168-173
High Holborn, London WC1V 7AA
England

International Thomson Publishing Asia
221 Henderson Bldg. #05-10
Singapore 0315

Thomas Nelson Australia
102 Dodds Street
South Melbourne 3205
Victoria, Australia

International Thomson Publishing Japan
Kyowa Building, 3F
2-2-1 Hirakawacho
Chiyoda-Ku, Tokyo 102
Japan

Nelson Canada
1120 Birchmount Road
Scarborough, Ontario
M1K 5G4, Canada

BBR 16 15 14 13 12 11 10 9 8 7 6 5 4 3 2

Library of Congress Cataloging-in-Publication Data

Novotny, Vladimir, 1938-

Water quality: prevention, identification, and management of
diffuse pollution/Vladimir Novotny and Harvey Olem.

p. cm.

Includes bibliographical references and index.

1. Water—Pollution. 2. Water quality management. I. Olem,
Harvey. II. Title. III. Title: Diffuse pollution.

TD420.N69 1993

628.1'68—dc20

93-14571

R0022969

Contents

Preface, ix

1 Introduction	1
Historical Perspectives and Trends in Environmental Degradation, 1	
Definitions, 11	
Environmental Systems and Their Changes, Interactions, and Impact on Water Quality (Pollution), 22	
Pollution by Toxic Chemicals and Metals, 41	
Important Examples of Systems Affected by Diffuse Pollution, 46	
2 Laws, Regulation, and Policies Affecting Water-Pollution Abatement	69
Social Causes of Pollution, 70	
Environmental Policies of Diffuse Pollution Abatement, 74	
Water Uses and Water Rights, 83	
Laws Affecting Pollution Abatement, 85	
3 Hydrologic Considerations	101
Precipitation-Runoff Relationship, 102	
Rainfall Excess Determination: Surface Runoff, 129	
Overland Routing of the Precipitation Excess, 144	
Ground-Water Systems, 171	
4 Atmospheric Deposition	185
Interdependence between Air and Water Quality, 186	
Toxic Compounds in the Atmosphere, 217	
5 Erosion and Sedimentation	237
Definition and Description of the Erosion Process, 238	
Sediment Problem, 250	
Estimating Sediment Yield, 251	
Sediment Delivery and Enrichment Processes during Overland Flow, 270	
Sediment Transport in Streams, 292	

6	Pollutant Interaction with Soils and Sediments	311
	Loading Functions, 317	
	The Fate of Contaminants in Soils: Three-Phase Approach, 323	
	Specific Pollutant Interactions, 337	
7	Ground-Water and Base-Flow Concentration	387
	Ground Water (Base Flow) and Diffuse Pollution, 387	
	Ground-Water Movement, 392	
	Origin of Natural Ground-Water (Base-Flow) Quality, 406	
	Impact of Diffuse Pollution on Ground Water and Base Flow, 415	
	Ground-Water Quality Models, 433	
8	Urban and Highway Diffuse Pollution	439
	Land-Use Effects on Urban Nonpoint Pollution Loads, 439	
	Urban Land Use and Magnitude of Diffuse Pollution, 443	
	Individual Sources of Pollution, 451	
	Runoff Pollution Generation Process, 461	
	Sources of Specific Pollutants in Urban Runoff and CSOs, 476	
	Statistical Quality Characteristics of Urban Runoff, 484	
	Combined Sewer Overflows, 495	
9	Modeling and Monitoring Diffuse Pollution	507
	General Concepts, 507	
	Simple Statistical Routines and Screening Models, 519	
	Deterministic Hydrologic Simulation Models, 528	
	Stochastic and Neural Network Models, 546	
	Real-Time Control Models for Urban Sewerage Systems, 552	
	Calibration and Verification of Deterministic Models, 554	
	Monitoring and Data Acquisition, 556	
10	Control of Urban Diffuse Pollution	573
	Pollution-Control Measures, 573	
	Urban Drainage and Runoff Pollution Control, 576	
	Source-Control Measures, 577	
	Collection System Control and Reduction of Delivery of Pollutants, 602	
	Detention-Retention Facilities, 616	
	Treatment, 637	
	Efficiency of Best Management Practices for Control of Priority Pollutants, 642	
	Real-Time Control of CSOs and Pollution Load Trade-Offs, 649	
11	Agricultural Issues	673
	Extent of American Agriculture, 675	
	Agriculture and Its Effect on the Environment, 681	

	Best Management Practices: Their Implementation and Effectiveness, 689	
	Roles, Responsibilities, and Programs of Federal, State, and Local Agencies and Groups, 711	
	Balancing Agricultural Production with Environmental Protection, 711	
12	Receiving Water Impacts	735
	Assessment of Water Quality Problems, 735	
	The Aquatic Ecosystem, 740	
	Dissolved Oxygen Problem, 764	
	Nutrient Problem of Lakes: Eutrophication, 783	
	Modeling Fate of Conventional Pollutants and Nutrients, 796	
	pH and Acidity, 802	
	Background (Natural) Water Quality, 806	
13	Toxic Pollution and Its Impact on Receiving Waters	817
	Toxicity Concepts, 817	
	Toxicity and Its Measurement, 821	
	Water Quality and Sediment Criteria and Standards for Toxicity, 838	
	Modeling Fate of Toxic Compounds in Receiving Water Bodies, 852	
14	Wetlands	861
	Definitions and Types of Wetlands, 863	
	Wetland Function, 869	
	Constructed Wetlands, 889	
	Aquatic Plant Systems, 917	
15	Management and Restoration of Streams, Lakes, and Watersheds	927
	Restoration Versus Reclamation, Rehabilitation, and Management, 928	
	Restoration Techniques for Rivers and Streams, 932	
	Restoration Techniques for Lakes and Reservoirs, 949	
	Integrated Aquatic Restoration: Case Studies, 972	
16	Integrated Planning and Control of Diffuse Pollution—Watershed Management	981
	Pollution Load Trade-Offs and the Permit Process, 983	
	Water Uses and Use Attainability, 985	
	Financing Diffuse-Pollution Abatement Programs: Who Should Pay? 993	
	Institutional Issues, 1002	

viii Contents

Epilogue 1011

Appendixes 1017

Index 1035

Preface

Man has lost the capacity to foresee and to forestall. He will end by destroying the earth.

A quote by Albert Schweitzer in Rachel Carson's *Silent Spring*

The nation behaves well if it treats the natural resources as assets which it must turn over to the next generation increased and not impaired in value.

Theodore Roosevelt

In the 1960s the environmental awareness of the U.S. population was greatly awakened. Many attributed this to a book by Rachel Carson entitled *Silent Spring* (Houghton Mifflin, Boston, 1961) that described, among other hazards, the effect of emissions of chemicals and other pollutants on the environment and the potential destruction of the ecology and mankind. But for whatever reasons, the environmental era of the late 1960s was a wide, grassroots movement of activists, many of them ill-informed but extremely enthusiastic. The movement caused several environmental protection legislative acts, of which the 1972 Water Pollution Control Act Amendments was the most far-reaching law (see Chapter 2 for pertinent water quality laws dealing with diffuse pollution). Earth Day of 1970 was the culminating event of this period.

Some well-known traditional environmental scientists and engineers called the environmental period of the 1960s and early 1970s "The Age of Unreason" or "an ecology binge in which relatively few activists aroused the concerns of millions but hopelessly ill-equipped Americans over the future of our environment and indeed our own existence" (Schroepfer, *Journal of the Water Pollution Control Federation*, September 1978). Some of this "unreason" was incorporated into laws that called for "zero pollution discharge,"* but in actuality advocated shifting the pollution

* It is interesting to note that the calls for zero discharge, this time on a more rational and scientific basis, have been renewed in mid-1990s.

disposal from surface water resources onto soils, and eventually into the ground water. Indeed, this environmental period of the 1960s and early 1970s was replaced in the 1980s by consumerism and a general lack of enthusiasm of the population about the state of the environment. In the same period, however, the global problems, unseen but predicted to some degree by Carson and others, became widely publicized. The first large-scale global environmental problem of diffuse nature was acid rainfall and its detrimental effects on North American and Scandinavian lakes (see Chapter 4 for a detailed discussion of acid rainfall). Even though natural rainfall is acidic due to the dissociation of dissolved carbon dioxide from the atmosphere (pH of rainfall in equilibrium with the atmospheric CO_2 is about 5.6), rainfall with much lower pH—down to less than 3 in some areas—is now falling over larger geographical areas of North America and Europe. This elevated acidity is due to sulfuric and nitric acids formed from fossil-fuel burning (sulphur-containing coal) and emissions from all kinds of motorized vehicles. There is now even evidence that, in addition to acidifying lakes, acid rainfall is also damaging soil fertility to such an extent that in some parts of the world it poses a threat to future agricultural production and its sustainability (see Chapter 6). Toxic compounds present in water are more toxic in acidic water than in neutral and slightly basic water bodies (see Chapter 8). Other problems of global proportions include the “greenhouse effect”—that is, the increased atmospheric content of carbon dioxide and other gases—and the ozone-hole problem caused by the discharge of fluorocarbons into the atmosphere. Fluorocarbons, which originate mostly from leaking cooling systems, are also a water quality problem. Consequently, solving the ozone-hole problem by banning the production and use of damaging fluorocarbons may also have a water quality benefit.

Also since the 1970s, beaches of the North, Mediterranean, and Adriatic seas became clogged by algae to the point that swimming was not possible except in swimming pools that the hotels and casinos had to build.

On a smaller but still widespread scale, soil loss from farms and construction sites is alarming as well as being detrimental to surface waters. In order to replace the lost plant nutrients and to protect the monocultural crops from insects and weeds, farmers use more and more chemicals, a practice that results in severe ground and surface water contamination. In many places, ground-water resources have been contaminated by nitrates and pesticides to a point where they cannot be used as a drinking water supply, with the result that potable water must be trucked in or provided in bottles.

As the countries of eastern Europe opened their doors to the West,

environmental problems on a monumental scale were discovered. It was known that these countries were a major cause of some global problems, such as acid rain and atmospheric PCB emissions, but the extent of the damage to their soil, air, and water resources had not been known and was kept secret. The most serious pollution problems are diffuse and widespread in some countries (central and eastern Europe and some developing countries), where they have reached locally catastrophic proportions.

These large-scale environmental scares, plus a still very remote but real possibility that man actually can bring about his own destruction by contamination of the environment and the atmosphere, have generated a new environmental awareness in the 1990s. How correct was Rachel Carson decades ago.

This book does not want to promote gloom and doom, nor does it want to cause unreasonable limitations on resources. The objective of this book is rather to identify the environmental problems caused by pollution, some of them of a global nature, and to suggest possible feasible and economical solutions. The authors wanted to approach the task of preparing the book in a positive, problem-solving fashion, with emphasis on the sustainable use of water and soil resources, their protection, and rehabilitation. As man can destroy the earth, he can also save it and live in harmony with the environment.

This approach requires some rethinking of the philosophies with which environmental engineers and scientists used to approach such problems. For example, a change is needed from the traditional "sanitary engineering," mostly structural approaches that advocated the removal and transport of pollutants and excess water from the affected areas in the fastest way possible to less structural and more ecologically oriented approaches that rely on water conservation and retention. This rethinking will involve a shift from traditional curb-and-gutter storm sewer drainage of urban and urbanizing areas to drainage maximizing the use of natural (grassed) waterways, retention, and infiltration; a change from draining wetlands for monocultural agriculture and urban development to preserving and retaining them; from lining urban and rural streams with concrete to preserving the streams in their natural state or restoring them so they can support an aquatic habitat; from deforestation to forest preservation and reforestation; from intensive agriculture relying on the heavy use of chemicals to sustainable agriculture.

A new branch of environmental engineering called *ecological engineering* is emerging. An ecological engineer knows how to balance pollution discharges with the waste-assimilating capacity of the environment. Then the environment can receive the residual waste loads without harm to the

ecology, aquatic biota, and beneficial downstream uses of water resources. If this capacity of water bodies to receive pollutants is not sufficient, engineering techniques can be employed to both reduce the destructive quantity of the pollutants and to increase the ability of the environment to accept potential pollutants without harm. This ability of the environment to accept limited amounts of pollutants without harm is a great economic asset that must be included in all considerations of environmental protection and restoration, but it must not be exceeded. Most environmental restoration and protection projects do not require a zero-pollution discharge approach.

This book essentially is a follow-up of the *Handbook of Nonpoint Pollution: Sources and Management* by Novotny and Chesters (Van Nostrand Reinhold, 1981). The authors followed also the ideas and solutions presented in others of their previous books (Krenkel and Novotny: *Water Quality Management*, Academic Press, New York, 1980; Novotny et al.: *Karl Imhoff's Handbook of Urban Drainage and Wastewater Disposal*, John Wiley & Sons, New York, 1989; Olem: *Liming Acidic Surface Waters*, Lewis Publishers, 1991). However, the emphasis on the sustainable use of water resources and the ecological approaches to solving the problems of diffuse pollution are somewhat new in the field of environmental engineering and pollution control.

The situation today is different from what it was at the end of the 1970s. Prior to the mid-1970s, diffuse pollution was an unknown phenomenon to the general population and its representatives. Environmental engineering and science was almost exclusively oriented toward wastewater conveyance, treatment and disposal, and water supply. Urban engineering was promoting such approaches as curb-and-gutter storm sewers, sewer separation, and lining streams for drainage. Today, however, there are very active and quite large groups of professionals interested in solutions for urban or agricultural diffuse pollution that would be harmonious with the ecological principles and would lead to the preservation and enhancement rather than the destruction of ecosystems. Many excellent examples of developments that have incorporated the nature and protected the habitat have recently emerged. Specialized groups have been formed by most major environmental professional associations, including the International Association on Water Quality (formerly the International Association for Water Pollution Research and Control), the Water Environment Federation (formerly the Water Pollution Control Federation), the American Water Resources Association, the American Society of Civil Engineers, the American Society of Agricultural Engineers, the Soil Science Society of America, the North American Lake Management Society, and the American Society of

Agronomy. Their journals and proceedings are now a major source of information on topics related to the diffuse-pollution problem and its solutions, and the authors wish to acknowledge the positive role these associations and their publications are playing in the recognition of the problems in this area, as well as to their solution. The authors are also indebted to several foundations and other sponsoring agencies (The National Science Foundation, the Water Environment Research Foundation, the U.S. Environmental Protection Agency, among others) that provided funding for research and enabled the authors to gather the knowledge necessary for preparing this book. The environmental education efforts on diffuse pollution by some of these professional organizations (AWRA, Air and Waste Management Association) have reached elementary and secondary schools throughout the country.

The authors would also like to recognize the important contributions of the reviewers of the manuscript of the book, Professor Peter A. Krenkel of the University of Nevada and Thomas Davenport of the U.S. Environmental Protection Agency. Credit should also be given to Paul and Eric Novotny for their computer art work.

The book is primarily for graduate students and practitioners in the environmental areas. Unlike the previous books by the authors, this one should be considered both a textbook and a handbook. It is intended to teach the recognition of problems and the finding of solutions as well as presenting facts and methodologies. Metric (SI) units are used throughout the book. Conversions into the customary U.S. units are included in Appendix A.

Introduction

Originally all pollution was of nonpoint (diffuse) nature. It became "point" pollution when years ago people in urban and industrial areas collected urban runoff and wastewater and brought it, at a great expense, to one point for disposal.

Paraphrase of a statement made by a well-known urban environmental economist (Gaffney, 1988) that introduces the topics to be presented in this chapter.

HISTORICAL PERSPECTIVES AND TRENDS IN ENVIRONMENTAL DEGRADATION

From the Romans to Earth Day

It is an irony of history that semi-desert conditions now prevail in much of the region once known as the Fertile Crescent. . . . Moreover, the earlier peoples had on the whole a higher standard of living than most of the present inhabitants. The degradations of the region came about almost entirely because of human discord and neglect. The ancient people had ingeniously developed the lands of the Fertile Crescent by intelligent use of meager water resources. . . . Then invaders laid waste to the region and a long decline set in. A succession of indolent and mutually intolerant people allowed the cisterns and reservoirs to fall into ruin, the irrigation channels and terraces to crumble, the trees to be cut down, the low vegetation to be destroyed by sheep and goats and the land to be scoured by erosion. (*Copyright © 1965 by Scientific American, Inc. All rights reserved.*)

These statements are a portion of the introduction by Maurice A. Garbell (1965) in his discussion of "The Jordan Valley Plan." The history of the

Middle East shows that if land stewardship is absent, the well-being of the people who misuse the land and water resources declines. At some point these adverse effects and deterioration become irreversible.

However, in the eighteenth century when the first Europeans arrived in the Piedmont area of the American Southeast they found rivers and lakes "crystal clean," without visible pollution, water transparent and abundant with fish (Clark, Haverkamp, and Chapman, 1985). At about the same time and earlier throughout the Middle Ages salmon migrated during the spawning season all the way into the headwater streams of central Europe to the delight of fishermen in Prague, in the present Czech Republic (located in the very center of Europe, several hundred kilometers from the North Sea into which the rivers flow).

One would be greatly mistaken if these statements about the cleanliness of the rural, mostly uninhabited environment were taken as a general rule about the environment of the ancient world and the Middle Ages. The pristine clean state of the rural environment centuries ago was contrasted by the filth and uncleanness of urban centers. The streets of medieval cities (and the same may be true for large urban centers of ancient Rome and other great historical centers) were covered by garbage, manure, and human excreta. In medieval Paris and other cities piles of garbage and manure in the streets were one meter or more high. The smell was strong and nauseating. Terrible epidemics plagued medieval cities, and even the rural population was not spared. There is no doubt that the medieval urban governments developed some kind of street-sweeping-cleanup or disposal services, and cleanliness or filth varied from city to city. Many urban dwellers themselves also tried to keep the streets in front of their houses clean. In addition the water supplies used by urban dwellers were much smaller than they are today, resulting in less pollution generation. However, rainfall and urban surface runoff were the primary and sometimes the only means of disposal of accumulated street surface pollution. Evidently, problems with urban runoff are not new. Consequently, storm sewers were built, primarily for storm water disposal.

When in the middle of the 1800s it was realized that the filth of the cities and contamination of the water supplies were the major reasons for such waterborne epidemics as cholera and typhoid fever, the first major period of environmental awareness was born. It was born because life in growing industrialized urban centers with medieval drainage became unbearable to the population and its governments. The first urban sewer system in the United States was planned in Chicago in 1855, although sewers had been built in Europe decades and in ancient Rome thousands of years before. The mixture of urban runoff and wastewater was brought by the sewers to the nearest watercourse, and the dilution by the flow of

the receiving water body was considered satisfactory for controlling pollution. It is interesting to note that until the 1950s many European receiving water standards were based on dilution (for example, according to the British water quality standards, one part of untreated sewage discharge required 500 parts of receiving stream flow). Many rivers soon became heavily overloaded and gave off an obnoxious stench, which was caused by the anoxic decomposition of sewage and garbage.

The period between 1880 and 1920 marks the beginning of major concerns about water quality, especially drinking water. In 1910, in Essen, Germany, one of the most industrialized areas of the world, the first water quality management agency was established to provide safe urban runoff and wastewater disposal. A few communities added treatment plants at the end point of their sewer systems to purify the discharged sewage. Almost all sewer systems built in this period carried a mixture of sewage and urban runoff. These systems are called *combined sewers* in contrast to newer and more expensive *separate sewer systems*, which employ dual sewers, one for sanitary sewage and the other for urban runoff (see Chap. 8 for discussion of urban drainage and its water quality impact). Even though the sewer systems were called combined, they were designed to carry primarily sewage and industrial wastewater—so-called *dry-weather flow*. A typical design capacity of combined sewers was six to eight times the dry-weather flow. However, this excess capacity was greatly insufficient for storage and conveyance of rainfall-generated runoff. Similarly, treatment plants were designed mostly for the dry weather flow (a typical design capacity was about four times the dry-weather flow). When, as a result of a rainfall event, the capacity of the sewers or of the treatment was exceeded, an untreated mixture of sewage and rain water was allowed to discharge into the nearest watercourse.

In rural areas, family farming using organic (manure) fertilizers flourished until the middle of the 1900s. This type of farming, in spite of its appearance and sometimes odor, causes less harm to surface and ground-water resources, although localized pollution problems from barnyard wastes were common. Farmers did not use chemical insecticides and fertilizers until the late 1950s.

After the epidemics of the Middle Ages were largely eliminated public interest in the environment subsided until the late 1960s. Meanwhile, however, pollution of the environment in the first half of the 1900s increased rapidly. The pollution of many urban rivers was again becoming unbearable. For example, every summer from the nineteenth century to the middle of the twentieth century the stench of the Thames River in London became so unbearable that the British Parliament recessed during the affected periods.

Man-made chemicals were introduced in the middle of the 1900s and

many of them have found their way into the environment where they caused great and almost irreparable harm. Such were the cases of DDT (dichloro-diphenyl-trichloro-ethane), which was originally heralded as a savior from malaria and every possible obnoxious insect, including lice, and PCBs (poly-chlorinated bi-phenyls), which is a group of very useful industrial chemicals. Both types of chemicals were later found to be greatly damaging, persistent, and bioaccumulating environmental contaminants. Many other chemicals were developed during and since World War II, which now have contaminated soils, water, and the air. The spread of man-made toxic chemicals and the potential dangers led to the second environmental activist period, the impetus for which was the book *Silent Spring* by Rachel Carson (1962). In 1970, the Earth Day celebration emphasized public concern about the state of the environment and initiated calls for action. This period was also marked by the rapid expansion of personal and commercial vehicular traffic, spurred by the building of freeways. Automobile and truck traffic is a major source of toxic chemicals as well as the activities associated with the expansion of the freeway system and suburban developments (urban sprawl).

In the United States and elsewhere, however, some progress in the abatement of municipal wastewater collected by sewers was made, and between 1920 and 1970 treatment plants were built at a rapid pace. By 1977 in the United States, 95% of the (156 million) people residing in sewered communities received some form of treatment of their wastewater while 70% received secondary biological treatment predominantly of the dry weather—sewage and wastewater—flows (Schroepfer, 1978). By 1970, the River Thames in London was alive again and fish have been caught there since. In 1972, the United States enacted the Water Pollution Control Act Amendments, which was the most far-reaching environmental legislative act to solve environmental problems.

However, even in the United States, and more so in Europe, many rivers and lakes could still not support viable fishery, being so polluted that fish were absent and their bottoms were covered with mud contaminated with toxic substances of unnatural man-made origin. For example, Lake Erie of the Great Lakes system was dying. Even where fish were present, carcinogenic compounds discharged into the receiving waters in the post-World War II period had stressed the aquatic population, fish had become unfit for human consumption, and water recreation had been reduced or had ceased.

In addition to pollution, activities that lead to habitat destruction should also be considered and remedied. Typically in the past, in the jargon of water resource developers, "channel improvement" meant

lining a stream with concrete and cutting down stream-bank vegetation, and "beneficial use of water" meant diverting flows from streams and lakes to the point that no flow was left during some periods. These activities caused severe damage, if not elimination, of aquatic habitat.

Wetland Drainage

Until recently in the United States as well as throughout the world, wetlands were considered as a source of disease (malaria) and as an obstacle to man's use of land resources for growth, agriculture, and economic development. Early "wetland management programs" in the United States, both governmental and private efforts, concentrated on drainage, filling, and conversion to agriculture and urban uses. Furthermore, most extensive insecticide applications (DDT spraying) on wetlands were to control malaria and mosquitoes, which are vectors of the disease. The "success" of these earlier programs has been documented by the U.S. Fish and Wild Life Service (a government agency under the U.S. Department of the Interior), which estimated that more than one-half of the approximately 1 million wetlands in the conterminous 48 states have been lost between the arrival of the first settlers and the present time.

However, it is known today that wetlands are ecological and hydrological assets and should be protected and restored rather than drained and destroyed. They provide habitat for waterfowl, animals, and wetland vegetation. Wetlands have the capability of storing water and of purifying polluted waters. Control of mosquitoes can be achieved by planting mosquito-eating fish (*Gambusia affinis*) or by fish management.

Fortunately, Congress and the federal government have realized the very high ecological, hydrological, and pollution-control benefits of wetlands and have enacted both federal and state wetland protection and rehabilitation acts. For example, in Florida developers must now reestablish 2 hectares (ha) of artificial or restored wetland for each hectare of natural wetland lost due to development. Draining or filling wetlands is now prohibited in many states, and efforts to reestablish formerly drained wetlands are pending. Valuable wetlands cannot be destroyed even if a replacement is offered.

Present Status of Water Quality Abatement

Since the passage of the Water Pollution Control Act Amendments in 1972, hundreds of billions of dollars have been spent on the cleanup of pollution, primarily that caused by sewage and industrial wastewater

discharges. But at the same time as the money was beginning to be spent on this type of cleanup, it was realized that these efforts may be insufficient. In addition to pollution from sewage and industrial wastes, pollution from land and from the activities by man occurring on the land would cause the cleanup goals not to be met in spite of the vast expenditures.

The land-use activities that create pollution cause other damage in addition to pollution. For example, soil losses that cause the pollution of receiving waters by sediment and associated pollutants, diminish the agricultural productivity of soils. An authoritative publication by the Conservation Foundation (Clark, Haverkamp, and Chapman, 1985) estimated that damage due to erosion and soil loss of cropland amount to 2.2 billion per year (in 1980 dollars).

Great Lakes Studies

In 1972 the Pollution from Land Use Reference Group (PLUARG) of the International Joint Commission (IJC) was established for the purpose of determining the levels and causes of pollution from land-use activities. A large group of scientists in Canada and the United States studied the pollution of surface runoff and found that indeed the land runoff, in addition to pollution from the atmosphere and from the traditional wastewater sources, was a significant and often major source of pollutant loads to the Great Lakes. PLUARG was a major international cooperative effort undertaken from 1972 to 1978. The resulting studies provided the most exhaustive review conducted up to that time, and to date it remains the most definite data base and reference source for many aspects of diffuse pollution in the Great Lakes and elsewhere (Nonpoint Source Control Task Force, 1983; Novotny and Chesters, 1981).

These studies have found that the most serious pollution of the Great Lakes arises from land areas of intensive agriculture and urban use. The most significant and damaging pollutants to the lakes from these sources were phosphorus, sediments, and pesticides, in addition to a number of toxic industrial compounds. Phosphorus is of concern in the Great Lakes because it is the principal factor causing and controlling accelerate eutrophication (Chapter 12), symptoms of which are the excessive development of algae, increase of turbidity, and general water quality deterioration. Land-use activities contributed from a third to a half of the total phosphorus loads (IJC, 1980). Also, pollution by toxic and hazardous substances from land drainage is an equal if not greater threat to the Great Lakes ecosystem. About 400 organic toxic compounds were identified in the Great Lakes ecosystem, including persistent pesticide compounds; specifically, aldrin-dieldrin and chlordane continue to appear

in the Great Lakes biota in spite of the fact that their use in the Great Lakes basin has been banned or severely restricted. In addition to land drainage and wastewater sources, significant loads of many pollutants are transported to the lakes via the atmosphere (IJC, 1980).

In 1978, the governments of the United States and Canada revised and signed a Great Lakes Water Quality Agreement. In this agreement the governments reaffirmed their determination to restore and enhance water quality in the Great Lakes system.

US EPA Activities

Based on PLUARG and several other studies, and recognizing the fact that abatement of traditional municipal and industrial wastewater sources will not alone achieve the water quality goals of the Clean Water Act, Congress requested the U.S. Environmental Protection Agency to prepare a comprehensive report on the sources of pollution other than traditional wastewater sources—the nonpoint sources.¹ The report was submitted to the Congress in 1984 (U.S. EPA, 1984). The report pointed out that virtually every state identified some sort of nonpoint pollution problem. The principal sources of nonpoint pollution were identified as *agricultural activities*—including those resulting from tillage practices and animal waste management—which were the most pervasive polluting activities reported from every part of the United States, followed by *urban runoff* and pollution from *mining*. Nonpoint pollution from *silvicultural activities* was considered substantial but localized. The large amounts of sediment associated with *construction activities* were recognized as causes of localized water quality problems in those parts of the United States experiencing significant development pressures (Southeast, mid-South, and Northwest). The EPA report to Congress emphasized voluntary approaches to the abatement of nonpoint sources.

In 1976 the Chesapeake Bay Program of the EPA (see the section “Important Examples of Systems Affected by Diffuse Pollution” in this chapter) was established. ~~The National Urban Runoff Project (NURP)~~ also sponsored by the EPA, ~~was a large research project carried out in 25 localities throughout the United States from 1978 to 1983 (U.S. EPA, 1983). This program had the following objectives:~~

- To investigate and establish quality characteristics of urban runoff, and similarities or differences at different urban locations;
- To identify the extent to which urban runoff is a significant contributor to water quality problems across the nation;

¹ See the following section for the definition of nonpoint sources of pollution.

To establish performance characteristics and the overall effectiveness and utility of management practices for the control of pollutant loads from urban runoff.

From its findings, it concluded that

- Urban runoff contains high concentrations of toxic metals; the so-called "priority pollutants" (toxic, mostly organic chemicals) were also detected in significant quantities.
- Urban runoff is contaminated by coliform and pathogenic (disease causing) bacteria;
- Urban runoff carries high quantities of sediment.

Other pollutants, such as those causing depletion of the dissolved oxygen or algal growths in the receiving waters, were present but were less significant. The NURP program also tested various abatement measures, some of which they found to be effective and therefore recommended, such as the use of retention ponds and grassed waterways (swales), while some were found to be ineffective for water quality control, such as street sweeping (U.S. EPA, 1983).

Rural Programs

The Clean Water Act of 1972 authorized the EPA to carry out demonstration projects of pollution-control technologies in the Great Lakes basin. Several experimental watersheds were established, from which the Black Creek basin (Allen County, Indiana) received the highest funding. The program demonstrated and addressed in a research fashion several pioneering pollution-control technologies, such as conservation tillage, including no-till, and animal-waste management as well as soil conservation.

The Black Creek project paralleled the PLUARG studies conducted in the neighboring experimental watersheds of the Great Lakes basin. The program spent more than \$800,000 as a cost-sharing for conservation treatment on the 4000-ha watershed, but could not show significant water quality improvement (Humenik, Smollen, and Dressing, 1987). The experience clearly indicated that the traditional "first come-first serve" cost-sharing incentive programs are not efficient for water quality improvement. The major reason was the fact that many soil-conservation and erosion-control practices do not yield significant water quality improvement (see Chapter 5). Consequently, the project managers and other researchers participating in the PLUARG projects developed a concept of targeting the areas that needed management that would yield the greatest water quality improvement benefit. The Black Creek project

also demonstrated institutional aspects of the nonpoint pollution programs, that is, that the Soil Conservation Service of the U.S. Department of Agriculture was capable of administering a rural nonpoint pollution-abatement program (Humenik, Smollen, and Dressing, 1987).

The Rural Clean Water Program (RCWP), which began in 1980, has funded 21 watershed projects whose objectives were to improve water quality, to help agricultural homeowners and operators to use pollution-control practices, and to develop programs, policies, and procedures for the control of agricultural nonpoint source pollution (*Federal Register*, 1979). It is a long-term (10 to 15 years) demonstration project, so the critical targeting of lands producing pollution is required, water quality objectives are clearly defined, and water quality is monitored. Hence, the program has a much greater water quality emphasis than previous programs, which focused primarily on soil conservation. Approved management practices include water management systems, animal-waste management systems, fertilizer and pesticide management, in addition to erosion control and soil conservation.

At the end of the 1980s, approximately seven to ten years after the start of the program, most projects exceeded their goals of implementing pollution management practices in 75% of the critical watershed areas. Projects that have achieved a high level of farmer participation have been successful because they offered cost sharing for practices farmers want, such as animal-waste storage, conservation tillage, and irrigation system improvement. It should be pointed out that the willingness of farmers to install pollution abatement under voluntary cost-sharing scenarios required that these practices include other benefits, such as the reduction of the cost of fertilizers and chemicals, which would compensate the farmer. Water quality improvements have been documented in several RCWP projects; however, it commonly took more than 5 years to document water quality improvement in these pilot watersheds.

Present and Future Trends and Needs

In 1990 there was worldwide recognition of the twentieth anniversary of the Earth Day. This recognition renewed environmental awareness in the public. The Earth Day movement of the 1960s and 1970s had an impact in the United States, and produced a landmark legislation, the 1972 Water Pollution Act Amendments. It also initiated extensive environmental remediative actions. The present environmental movement is concerned with global as well as local pollution problems and environmental contamination. Also the attitude of lawmakers is changing worldwide. The emergence of Green political parties in Europe which, although still small, keep the balance of power in some European countries. Public

pressures have also led to the further enactment of environmental protection laws.

In the United States it was realized that despite requirements incorporated in the 1972 and 1977 versions of the Clean Water Act, progress in reducing diffuse pollution has been extremely slow. It was demonstrated that in the 15-year period since the 1972 passage of the Water Pollution Control Act Amendments very little had been accomplished in reducing the diffuse pollution problem (Thompson, 1989). As a matter of fact, some problems, especially those related to the use of agricultural and industrial chemicals, have intensified. Therefore Congress in the 1987 Water Quality Act shifted from 15 years of nonpoint source pollution planning, studying, and problem identification to a new National Nonpoint Pollution Control Program (U.S. EPA, 1989). It stated that:

It is the national policy that programs for the control of nonpoint sources of pollution be developed and implemented in an expeditious manner so as to enable the goals of this act to be met through the control of both point and nonpoint sources of pollution.

Far-reaching regulations are being implemented to control diffuse pollution, including urban and industrial runoff and agricultural diffuse pollution. Federal and state laws are also being implemented that would stop the loss of wetlands that are so vital in the ecological balance and for the control of diffuse pollution.

In spite of some monumental efforts in the United States and elsewhere to remedy pollution, environmental degradation is still continuing. The diffuse-pollution problem is only one component of the overall problem with the earth's ecological system. The *State of the Environment* (OECD, 1991) pointed out three major problems now facing the earth's ecosystem:

- Stratospheric ozone depletion;
- The greenhouse effect;
- The global spread of pollution, primarily by atmospheric currents.

These global issues are linked in a number of ways. The causes and transport of diffuse pollution by surface water bodies may not be separated from these global issues, but are a part of the overall system.

As forests shrink at an alarming rate thus contributing to the greenhouse effect by emissions from slash burning, soil is simultaneously eroded from the deforested lands, causing siltation of bodies of water and other water quality problems. As the environment is being filled with man-made chemical compounds, such as fluorocarbons and other

halogenated hydrocarbons, many of them carcinogenic, these emissions are also contributing to the depletion of the protective ozone layer. Nitrous oxides emitted from vehicular traffic and other sources contribute to both the greenhouse effect and the pollution of surface and ground water.

There is no doubt that in spite of the expenditures on pollution, the worldwide state of the environment in the 1990s is worse than it was during the Earth Day of 1970. In the *State of the World*, the Worldwatch Institute (Brown et al. 1990) points out that only a monumental effort can reverse the deterioration of the planet. As the East-West ideological conflict vanishes, time and energy of the population and its political leaders can be concentrated on environmental threats to security. The present and future years will be a period of reordering priorities, providing resources to reforest the earth, and conversion to sustainable development with less harmful pollution problems, as well as avoiding pollution catastrophes during the second half of the twentieth century.

In the past and even today, economic development has caused environmental degradation. While on one hand food supplies and economic output have grown tremendously since the end of World War II, at the same time the world has lost nearly one-fifth of the topsoil from its cropland and one-fifth of its tropical rain forest plus tens of thousands of its plant and animal species (Brown et al., 1990). Rapid increases in pollutant levels in ground water, particularly nitrate and pesticide pollutants, are directly tied to the increase in crop yields that has been stimulated by agricultural chemicals. However, at least in some parts of the world, the reverse is beginning to emerge, that is, environmental trends are beginning to shape economic trends.

This combination of public and government awareness of environmental dangers of diffuse pollution and their willingness to do something about it and pay for it, gives environmental engineers and scientists an opportunity to design abatement programs that would bring about true progress in the reversal of past degradation trends.

DEFINITIONS

Water Quality and Pollution

Any treatise dealing with water quality and pollution needs definitions. In the minds of the public water quality is often synonymous with pollution and, similarly, water quality management, including that related to diffuse sources, is equated with pollution control.

Water quality reflects the composition of water as affected by natural

causes and man's cultural activities, expressed in terms of measurable quantities and related to intended water use. Water quality is perceived differently by different people, for example, a public health official is concerned with the bacterial and viral safety of water used for drinking and bathing; fishers are concerned that the quality of water be sufficient to provide the best habitat for fish; and aquatic scientists are concerned with the health of aquatic habitats, including fish, plankton, and other plants and organisms. For each intended use and water quality benefit, different parameters express best water quality.

The term *pollution* is derived from a Latin word (*pollu'ere*), which means "to soil" or "to defile." The terms *pollution*, *contamination*, *nuisance*, and *water (air, land) degradation* are often used synonymously to describe faulty conditions of surface and ground water. Various definitions have been offered to define pollution and other related terms (Krenkel and Novotny, 1980; Vesilind, 1975; Henry and Heinke, 1989). However, these definitions are not identical and, in a legal sense, not even similar. Probably the definition of pollution most accepted by scientists is "unreasonable interference with the beneficial uses of the resource." However, the perception of beneficial use is again different to different people, which could be a problem. For example, from the economic standpoint, the greatest "beneficial use" of water and air resources is to provide an inexpensive way to dispose of wastes, in which case fishing and swimming may be perceived by these "economical users" as interfering with their "beneficial use." Indeed, during discussions of the implications of the Water Pollution Control Act Amendments of 1972 some people with good intentions tried to put dollar values on the cost of reducing pollution versus the market value of fish in the receiving water body.

Fortunately, such interpretations are not acceptable now, but they show the possible problems with simple definitions and with perceptions. Today's interpretations put a very high value on the protection of the environment, and supersede any economic savings that might be achieved by allowing injurious discharges of pollutants.

The statutory definition of pollution is included in the Clean Water Act, Sec. 502-19 (U.S. Congress 1987):

The term "pollution" means man-made or man-induced alteration of chemical, physical, biological, and radiological integrity of water.

The term ~~integrity~~ means "being unimpaired;" therefore, *alteration of integrity* means impairment or injury. An alternate working definition (the one that will be followed and expanded on in this book) is:

Pollution is a change in the physical, chemical, radiological, or biological quality of the resource (air, land or water) caused by man or due to man's activities that is injurious to existing, intended, or potential uses of the resource.

According to this definition, pollution is differentiated from changes in the quality of the environment due to natural causes, such as volcanic eruptions, deposition of fly ash from natural forest fires, natural erosion, weathering of rocks, and natural elutriation of minerals, even though these events may have the same actual or potential adverse impact on water use or of a resource as does pollution. These changes could be considered *nuisance* or *undesirable quality modifications* or even *Acts of God* by the legal profession, but in a technical engineering sense they mostly do not require abatement or abatement is not technically and economically feasible.

Consider as an example the Rio Puerco in New Mexico (Fig. 1.1). This river drains a highly erodible, sparsely populated arid watershed in the northwestern quadrant of the state. Consequently, extremely high concentrations of suspended solids exceeding 200,000 mg/l are measured during erosive flows. There is no doubt that man has contributed to these high loads. Before ranchers moved in more than 100 years ago, the



FIGURE 1.1. Rio Puerco in New Mexico. (Photo: V. Novotny.)

watershed was more stable. However, overgrazing on the fragile arid lands of the Rio Puerco for the last 70 to 100 years has been a big contributor to the high "natural loads."

Similarly, the Yellow River in China, the Nile River in Egypt, and many other rivers in arid areas of the world are known to carry high amounts of sediments and other water quality constituents. Yet the authorities do not plan any remedial actions at this time, mainly because of the extremely high cost of the watershed reclamation and a lack of concern among the local population. On the other hand, sediment concentrations two orders of magnitude lower due to farming and urbanization in the Great Lakes region and elsewhere are not being tolerated, and plans are on the way to remedy the situation. As a matter of fact, the building of the Aswan High Dam on the Nile River drastically reduced the natural sediment loads of the river, thus depriving the farmers in the Nile Delta region of the natural fertilizer brought onto their fields in flood waters, which resulted in the subsequent damage done to them and to the economy of Egypt. Similarly, the present decrease in sediment loads in the Mississippi River caused by settling of sediment in upstream navigation pools is causing increased erosion and loss of land along coastal Louisiana. The present rate of wetland loss in Louisiana, caused primarily by the reduction of sediment input from the Mississippi River, represents about 80% of coastal wetland losses in the United States (Rooney, 1989).

Other examples of undesirable natural water contamination, but not pollution by man, include the high carbon dioxide content of some ground water which is injurious to building materials and elutriation of humic organics from decaying aquatic vegetation which impairs the suitability of water for water supply. Hence, water quality composition contains both constituents that may be pollution and constituents from natural sources (Fig. 1.2). The use of the term *background pollution* for natural water quality composition is not correct because this pollution is not caused by man.

Pollution may result from causes other than discharges of wastewater or soil losses, the two most commonly mentioned causes. Cutting down a forest, channel modifications (lining), draining wetlands, and similar activities commonly result in undesirable changes in water quality. Hence, such activities create pollution.

Water quality and pollution are determined and measured by comparing physical, chemical, biological, microbiological, and radiological quantities and parameters to a set of standards and criteria. The difference between the *standards* and *criteria* should be explained.

A *criterion* is basically a scientific quantity upon which a judgment can

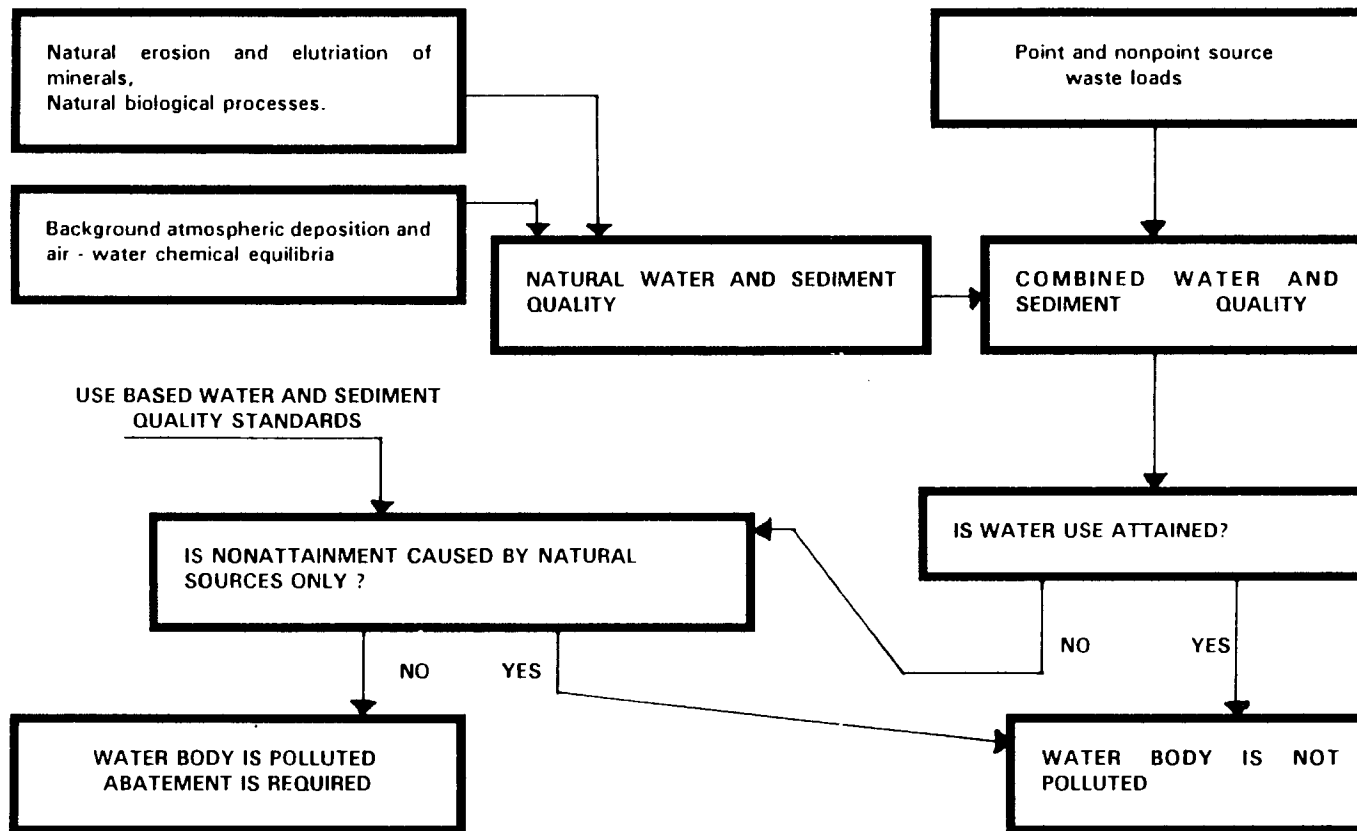


FIGURE 1.2. Concept of water quality.

be based. It is usually developed from scientific experiments. A water quality criterion can be based on morbidity or chronic toxicity of various substances to man or aquatic life, or it can be related to technical methods of removing the substances from water. A *standard* applies to a definite rule, principle, or measure established by an authority.

The water quality criteria and standards currently used by water-pollution-control authorities throughout the world, as well as by engineers and scientists, are either *stream standards* or *effluent standards*. The effluent standards determine how much pollution can be discharged from municipal and industrial wastewater sources and by some types of diffuse pollution. *Performance standards*, which are equivalent to effluent standards for the control of pollution from lands, are used to control pollution from subdivisions, construction sites, and mining. The *stream standards* can be related to the protection of aquatic habitats, which is one of the most important beneficial uses of water, and/or to other existing or intended uses of the water resource. Actual numerical values of water quality standards are given in Appendix 1, and their fundamentals are described in Chapters 12 and 13. The designated use of a water body must be attainable (see Chap. 16 for a discussion of use attainability). Standards and criteria may be numerical, chemical-based, or narrative or based on the toxicity of the entire effluent or water body.

In water quality planning and evaluation, exceeding the water quality parameters over one or more standards (criteria) implies an injury to the water use for which the standard was issued. Consequently, a wastewater discharge that does not result in a violation of a standard may be considered noninjurious, as it does not cause pollution. The quantity of potential pollutants that can be discharged into the environment (receiving water body, atmosphere, or land) is then called the *waste-assimilative capacity*. Determining the waste-assimilative capacity of the receiving water or air body is one of the most important steps in any environmental protection study. Not taking the waste-assimilative capacity into consideration during the planning of pollution abatement or water quality restoration would lead to uneconomical wasteful approaches or even ineffective solutions. The concept of waste-assimilative capacity and its determination is shown on Figure 1.3. Typically, the waste-assimilative capacity of surface water bodies might be higher for decomposable organic matter, but it is very low to nil for some toxic chemicals that bioaccumulate in tissues of aquatic organisms and become injurious to animals and man using them as food. Consequently, this detrimental capacity is reflected in the magnitude of the standards. Determination of the waste-assimilative capacity is a part of the water quality that is based on pollution control, and is now

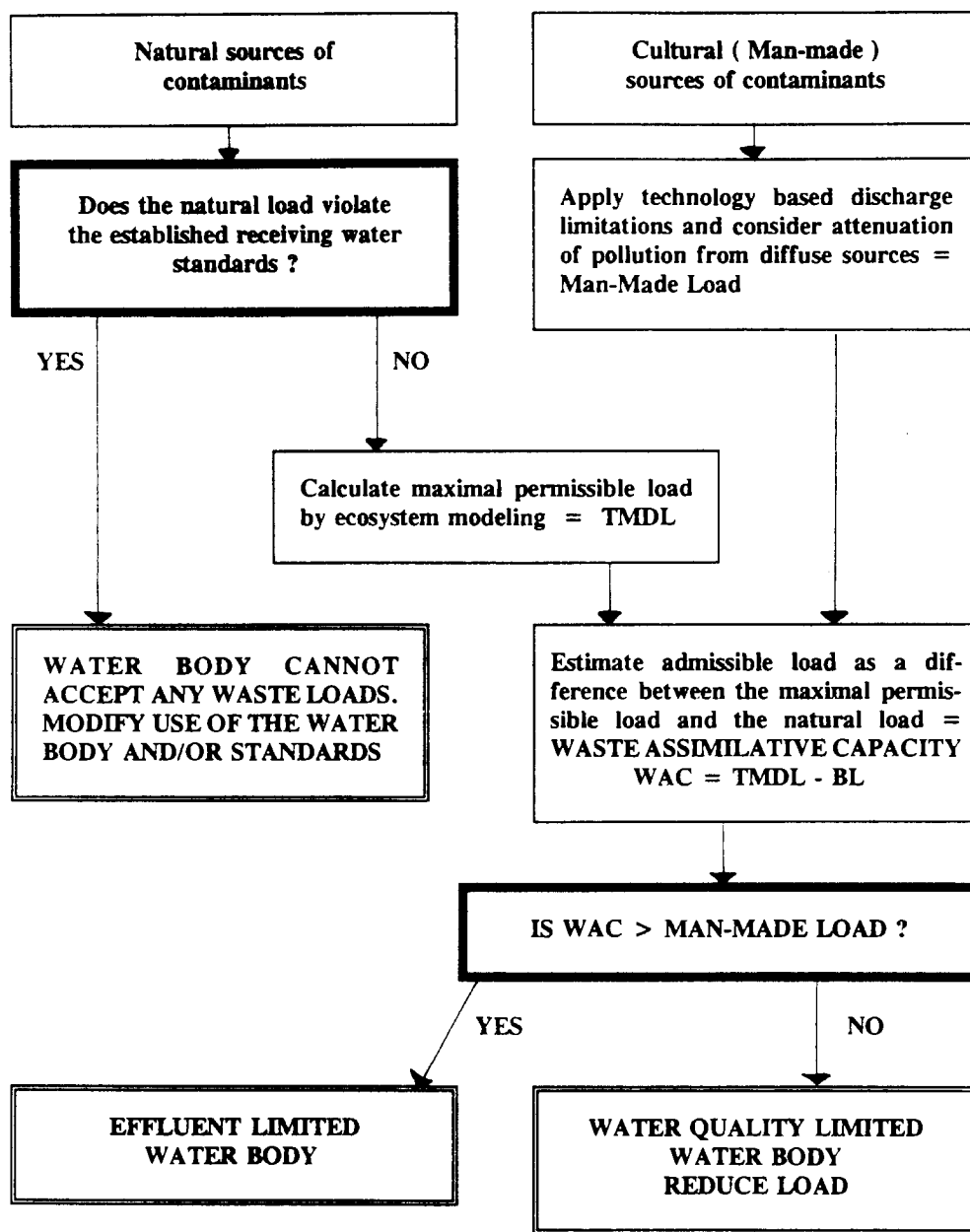


FIGURE 1.3. Waste-assimilative capacity concept.

becoming the most significant part of pollution-control efforts. In water-quality-based controls, effluent standards are tied to the the waste-assimilative capacity of the receiving water body. The process of integrated water-quality-based (point and nonpoint) pollution abatement is presented in Chapter 16. Existing waste-assimilative capacity of a water body can be increased by water body restoration and other measures (see Chap. 15).

According to these definitions a waste constituent becomes a pollutant if it is discharged into the environment in quantities that are injurious to or impair the beneficial uses of environmental resources. Many waste constituents in small quantities are not injurious, and some of them may even be beneficial in low quantities, becoming injurious (toxic) only in quantities that exceed the waste-assimilative capacity. For example, some metals that in higher concentrations are known to be toxic, such as zinc, are in smaller trace quantities, necessary nutrients for aquatic life.

Best management practices (BMP) are methods, measures, or practices selected by an agency to meet its nonpoint (diffuse) source control needs. BMPs include, but are not limited to, structural and nonstructural controls and operations and maintenance procedures. BMPs can be applied before, during, and after pollution-producing activities to reduce or eliminate the introduction of pollutants from diffuse sources into receiving waters.

Definition of Pollution Sources

From the previous discussion it follows that pollution is caused by man and results in an undesirable or harmful change in the quality of the resource, whether water, soil, or air. The sources or causes of pollution can be classified as either *point* or *nonpoint sources of pollution*. During publishing of the previous books by the first author on this topic (Krenkel and Novotny, 1980; Novotny and Chesters, 1981) the term *diffuse pollution* was synonymous to nonpoint pollution. After the U.S. Congress passed the Clean Water Act in 1977 and the Water Quality Act of 1987, it became necessary to redefine these characterizations.

Point sources of pollution were originally defined as pollutants that enter the transport routes at discrete, identifiable locations and that can usually be measured. Major point sources under this definition included sewerage municipal and industrial wastewater and effluents from solid waste disposal sites. Nonpoint sources were simply "everything else," and included diffuse, difficult to identify, and intermittent sources of pollutants, usually associated with land or land use. These definitions led to some legal ramifications for abatement efforts. According to the U.S. Constitution, the government can mandate the control of point sources that enter so-called navigable waters,² while use of land is considered sacred making enforcement of nonpoint pollution control impossible. Hence the new definitions broadened the category of *point sources*.

² Under legal interpretations in the United States, any body of water on which a canoe can float, even potentially, is considered navigable.

Today's statutory definition of point sources is as follows (Water Quality Act, Sec. 502-14, U.S. Congress, 1987):

The term "point source" means any discernable, confined and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, concentrated animal feeding operation, or vessel or other floating craft from which pollutants are or may be discharged. This term does not include agricultural stormwater and return flows from irrigated agriculture.

This definition does not leave much space for "everything else" being nonpoint sources. This ambiguity or lack of definition led the National Resources Defence Council to call nonpoint pollution "poison runoff" (Thompson, 1989).

Rather than looking for some kind of exact legal definition of point and nonpoint sources we attempt to categorize the sources of pollution into point and nonpoint categories according to the latest statutory regulations (see the next chapter for details). The statutory point source category in the United States today includes the following sources:

- Municipal and industrial wastewater effluents;
- Runoff and leachate from solid waste disposal sites;
- Runoff and infiltrated water from concentrated animal feeding operations;
- Runoff from industrial sites not connected to storm sewers;
- Storm sewer outfalls in urban centers with a population of more than 100,000;
- Combined sewer overflows;
- Leachate from solid waste disposal sites;
- Runoff and drainage water from active mines, both surface and underground, and from oil fields;
- Other sources, such as discharges from vessels, damaged storage tanks, and storage piles of chemicals;
- Runoff from construction sites that are larger than 2 hectares (5 acres).

Bypasses of untreated sewage (when the capacity of a treatment plant would be exceeded) are not allowed by law, and hence cannot be considered to be a legal source of pollution.

Two common characteristics of these statutory point sources are that they do indeed enter the receiving water bodies at some identifiable single- or multiple-point location and that they carry pollutants. Three more common characteristics of these point sources are that in the United States they are regulated, their control is mandated, and a permit is

required for the waste discharges. The statutory nonpoint sources (“everything else”) then include:

- Return flow from irrigated agriculture (specifically excluded from point source definition by Congress);
- Other agricultural and silvicultural runoff and infiltration from sources other than confined concentrated animal operations;
- Unconfined pastures of animals and runoff from range land;
- Urban runoff from sewered communities with a population of less than 100,000 not causing a significant water quality problem;
- Urban runoff from unsewered areas;
- Runoff from small and/or scattered (less than 2 hectare) construction sites;
- Septic tank surfacing in areas of failing septic tank systems and leaching of septic tank effluents;
- Wet and dry atmospheric deposition over a water surface (including acid rainfall),
- Flow from abandoned mines (surface and underground), including inactive roads, tailings, and spoil piles;
- Activities on land that generate wastes and contaminants, such as:
 - deforestation and logging
 - wetland drainage and conversion
 - channeling of streams, building of levees, dams, causeways and flow-diversion facilities on navigable waters
 - construction and development of land
 - interurban transportation
 - military training, maneuvers, and exercises
 - mass outdoor recreation.

Some of these “nonpoint” sources are either locally or federally regulated. For example, in many states developers are required to implement erosion-control practices, wetland protection laws regulate drainage of wetlands, before individual septic tank systems can be installed local or state public health and/or land-use authorities must issue permits. In the United States no potentially polluting activities on navigable waters (such as dredging, channel construction, dams) can proceed until the U.S. Army Corps of Engineers has issued the necessary permits. This agency is also responsible for issuing permits for any alterations to wetlands.

From the previous discussion and definition of pollution, one might also conclude that not all lands and land-use activities are polluting. For example, diffuse-waste emission from low-density suburban developments are very close to the background emissions of most important constituents. Waste emitted from some disturbed lands located far from a

watercourse is redeposited between the source and the recipient, hence the land may not cause impairment of water quality (pollution). For this reason some soil-conservation practices might not have a significant water quality improvement benefit. The lands that are most polluting within a watershed (and require pollution elimination) are called *hazardous* or *critical lands*. Determination and location of hazardous (critical) lands is one of the most important tasks in planning a diffuse-pollution abatement.

As can be seen, the division of sources into "point" and "nonpoint" is not as straightforward as it might have been during the 1970s when the *Handbook of Nonpoint Pollution: Sources and Management* (Novotny and Chesters, 1981) was written. Today these definitions have more statutory and legal than technical meanings.

The fact remains, however, that the "traditional" point sources of wastewater, that is, municipal, industrial, and agricultural (farm), are different from *diffuse* sources, which according to the statutory definition, may be both point and nonpoint.

The traditional point sources strictly include wastewater effluents from municipal and industrial areas. The flow and pollution loads from these sources vary; however, in most cases they are continuous, uninterrupted discharges, variability is not greatly related to meteorological factors, and the variability is not great. The primary parameters of interest for control and regulation are the degradable organics (measured as BOD₅ or COD), pH, suspended solids, nitrogen, and phosphorus and toxic compounds (both organic and inorganic). The prevalent method of control is treatment.

Diffuse sources can be characterized as follows:

- Diffuse discharges enter the receiving surface waters in a diffuse manner at intermittent intervals that are related mostly to the occurrence of meteorological events.
- Waste generation (pollution) arises over an extensive area of land and is in transit overland before it reaches surface waters or infiltrates into shallow aquifers.
- Diffuse sources are difficult or impossible to monitor at the point of origin.
- Unlike for the traditional point sources, where treatment is the most effective method of pollution control, abatement of diffuse land is focused on land and runoff management practices,
- Compliance monitoring is carried out on land rather than in water.
- Waste emissions and discharges cannot be measured in terms of effluent limitations.

- The extent of diffuse waste emissions (pollution) is related to certain uncontrollable climatic events, as well as geographic and geologic conditions, and may differ greatly from place to place and from year to year.
- The most important waste constituents from diffuse sources subject to the management and control are suspended solids, nutrients, and toxic compounds.

ENVIRONMENTAL SYSTEMS AND THEIR CHANGES, INTERACTIONS, AND IMPACT ON WATER QUALITY (POLLUTION)

Ecological Systems

There are five biological systems—croplands and grasslands, forests, urban, air systems, and aquatic systems—that support the world economy and provide the means for sustaining biological life on the earth. In addition, all biological life takes place in the atmosphere (including that of soil) or in water. These five systems are interconnected in a balance that is often disrupted by overuse of the resources and/or by their loss and degradation. Figure 1.4 shows the interconnections and pathways of

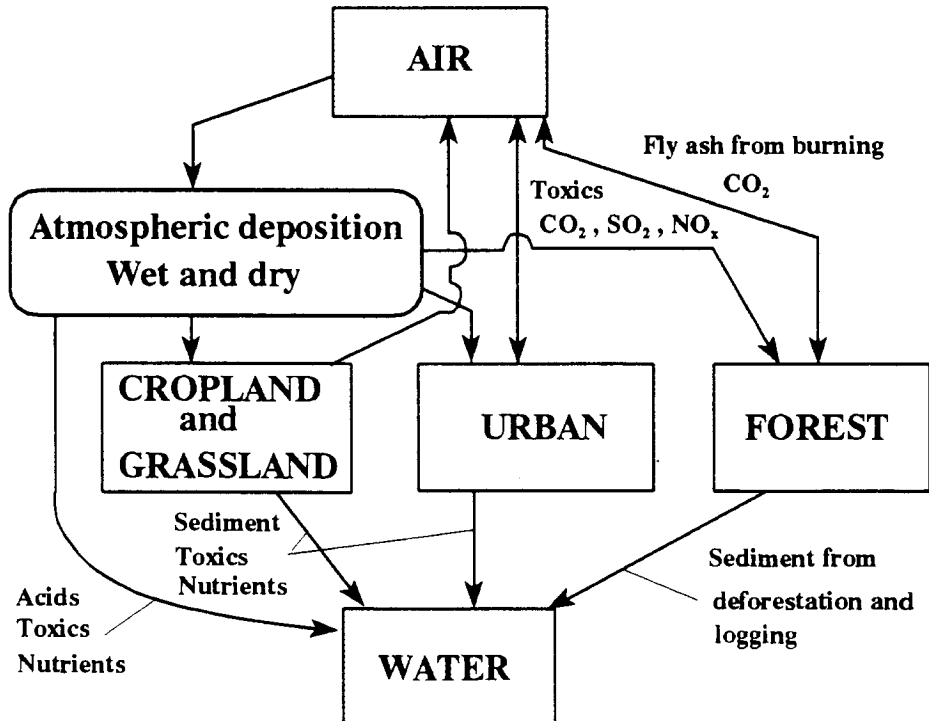


FIGURE 1.4. Pathways of pollutants between basic ecological systems.

contaminants between the five basic biological (ecological) systems. Atmospheric pollution impacts on water quality, not counting the harm done to human health, deforestation (loss of land resources) has a tremendous effect on water resources as well as on the atmosphere (greenhouse effects by slash burning and emissions of CO₂); urbanization (another loss of resources) is a source of many types of pollution, including contamination of air, water, and soil as well as soil loss. Soil loss from agriculture and construction diminishes the use of water resources by clogging the receiving water bodies with excessive sediments and depriving the aquatic biota of their natural habitat. Sediment is also a carrier of many pollutants, including those that cause excessive algal growths (nutrients) and those that cause toxic contamination.

These five systems are not stagnant, but continuously evolve and change. Until the twentieth century most of the changes that adversely impacted the environment were of natural origin, such as the erosion of land, volcanic eruptions, and natural flooding. There have been incidences throughout history where man has adversely and detrimentally impacted entire regions. For example, the rich Mesopotamian civilization flourished until its vast irrigation systems were destroyed by sediment from erosion when the forests on the surrounding hills were cut down and the fertility of the soil was destroyed by salts deposited by the irrigation water. Syria, which is now mostly arid or desert, thousands years ago was forested and enjoyed more humid hydrological conditions. The first deforestation of southern Europe took place during the Roman period, followed by reforestation after the fall of the Roman Empire in the fifth century. Then in the Middle Ages deforestation was repeated by Venetians who cut down the forests of Dalmatia on the Adriatic Sea (present Croatia), which resulted in tremendous soil loss and changed the entire hydrology of the region. However, these instances were mostly localized. As long as the equilibrium between the five ecological systems is not disrupted, the adverse ecological consequences are minimal.

Several major factors caused the environmental (ecological) balance to be severely disrupted in the second half of the twentieth century, resulting in accelerated increases of environmental pollution. They are

- population increase (sometimes termed explosion),
- deforestation,
- conversion of land to intensive agriculture,
- urbanization and industrialization,
- increased living standard, resulting in an increased per capita use of natural resources.

TABLE 1.1 World Population Growth

Year	Population (billions)	Population Increase by Decade (millions)
1650	0.5	—
1800	1.0	—
1930	2.0	—
1950	2.515	—
1960	3.019	504
1970	3.698	679
1980	4.450	752
1990	5.292	842
2000	6.251	959

Source: From United Nations, Department of International Economic and Social Affairs, New York (1988).

Population Increase

Table 1.1 shows the approximate world population. It can be seen that it took about one million years for the world population to reach the first one billion. About the same population increase may occur in the 10 years between 1990 and 2000. The population increase is not evenly distributed throughout the world. While in most developed, industrialized countries population growth is declining or the population is remaining stagnant, most of the growth is occurring in less developed countries. Because the less developed countries lack adequate sanitation, the increase in population results in the increase of diffuse pollution. The population pressures are also detrimental to forests and other resources that are diminishing at an accelerated rate, which results in more pollution.

In the context of sustainable resources development and containment of environmental damage, each additional person represents additional demands on productive resources, and hence additional wastes, plus wastes caused by the maintenance of the life processes.

Increased population also forces accelerated land-use changes. Since rural areas cannot economically sustain all families, the excess rural population is directed toward urban areas in search of employment and even, as in developing countries, for survival.

The truth is, however, that in many developing countries, and to some extent even in developed countries, urban areas cannot absorb the influx of population plus their own population increase, which leads to homelessness and shantytowns that, without sanitation, create high levels of diffuse pollution. The population increase of some world urban centers

has been phenomenal and has greatly outpaced world population growth. For example, according to United Nations statistics, the population of Sao Paulo in Brazil, which in 1950 was about 2.4 million, is now over 20 million; Mexico City's population increased by 19 million between 1950 and 1990; the population of Paris increased by 4 million (from 7.4 million in 1950 to about 11 million in 1990); and the population of London remained almost constant.

Watershed in Transition

Land Use and Diffuse Pollution

It has to be realized that in most cases it is not the land or land use *per se* that causes pollution. Land use is a simple term to describe the prevailing activity occurring in the area. As such it bears little relationship to the pollution generated from the area. Land-use zoning and separation of activities on land is more unique to the United States than to other countries. In many European and world urban centers, use of urban land is mixed, allowing commercial and sometimes industrial activities to exist side-by-side with residential developments.

Many land uses, such as for economic and living purposes and for development, are justified. In most cases, it is the misuse or excessive use of land that causes excessive pollution.

In the United States, land uses have been generally divided into the following categories (Novotny and Chesters, 1981):

General Urban Lands

- Residential (low, medium, and high density)
- Commercial
- Industrial
- Other developed (large parking areas, sports complexes)
- Open (parks, golf courses)
- Transportation (airport, rail, vehicular)

Rural (Nonurban) Lands

- Cropland (irrigated and nonirrigated)
- Improved pasture and rangeland
- Woodland and silviculture
- Concentrated animal feedlots
- Idle land (includes open water surfaces and native lands)
- Land used for waste disposal.

This "static" categorization of land uses has been questioned as insufficient for estimating nonpoint loading (Novotny, 1985; Marsalek,

1978). Wide variations of loads within each land-use category have been observed (IJC, 1980; Marsalek, 1978; Novotny and Chesters, 1981; Beaulack and Reckhow, 1982; Novotny, 1985).

Since it is the use of land by man and his/her associated activities that generate pollution, diffuse pollution has been related to the land-use type and to the transition from one land use to another.

Land Use and Land Transformation

There are four basic types of native undisturbed lands (excluding mountains): arid land (including deserts), prairie, wetland, and woodland. These four types have various forms, depending on the geographical location and the elevation.

The transition process, which today is mostly cultural (anthropogenic), is depicted in Figure 1.5. It begins with one of the four natural lands and progresses through several intermediate stages to a fully urbanized, sewerage watershed, which represents the end of the transition process. Most of this transition is irreversible. Just as the natural morphological processes that carve the basins by erosion and the elutriation of minerals are responsible for the background water quality composition, cultural processes of land changes are responsible for pollution. These cultural land-use modifications and land development processes are also illustrated on Figure 1.5.

Deforestation

Deforestation is an environmentally devastating and polluting land-use activity, which today is recognized as having very serious global consequences. Ancient examples are the Middle East, where deforestation changed the woodlands of Syria and other Middle East countries into desert, and forest cutting by the Romans and Venetians, which changed most of Dalmatia into a barren, soil-poor country. These examples are now being followed on a much larger scale by the disappearance of the tropical rain forest.

Deforestation deprives the soil of its protective, vegetative cover, and hence increases soil loss through erosion by several orders of magnitude. As a consequence, the quality of receiving water bodies is also dramatically reduced by sediments, organic compounds, nutrients (nitrogen and phosphorus), and possibly other pollutants (Fig. 1.6).

Slash burning practices used today to clear rain forests for farming and rangeland also add significantly to the carbon dioxide levels in the atmosphere, thus increasing the *greenhouse* phenomenon. Brazil, for example, contributes some 336 million tons of carbon to the atmosphere each year through deforestation. This contribution also increases

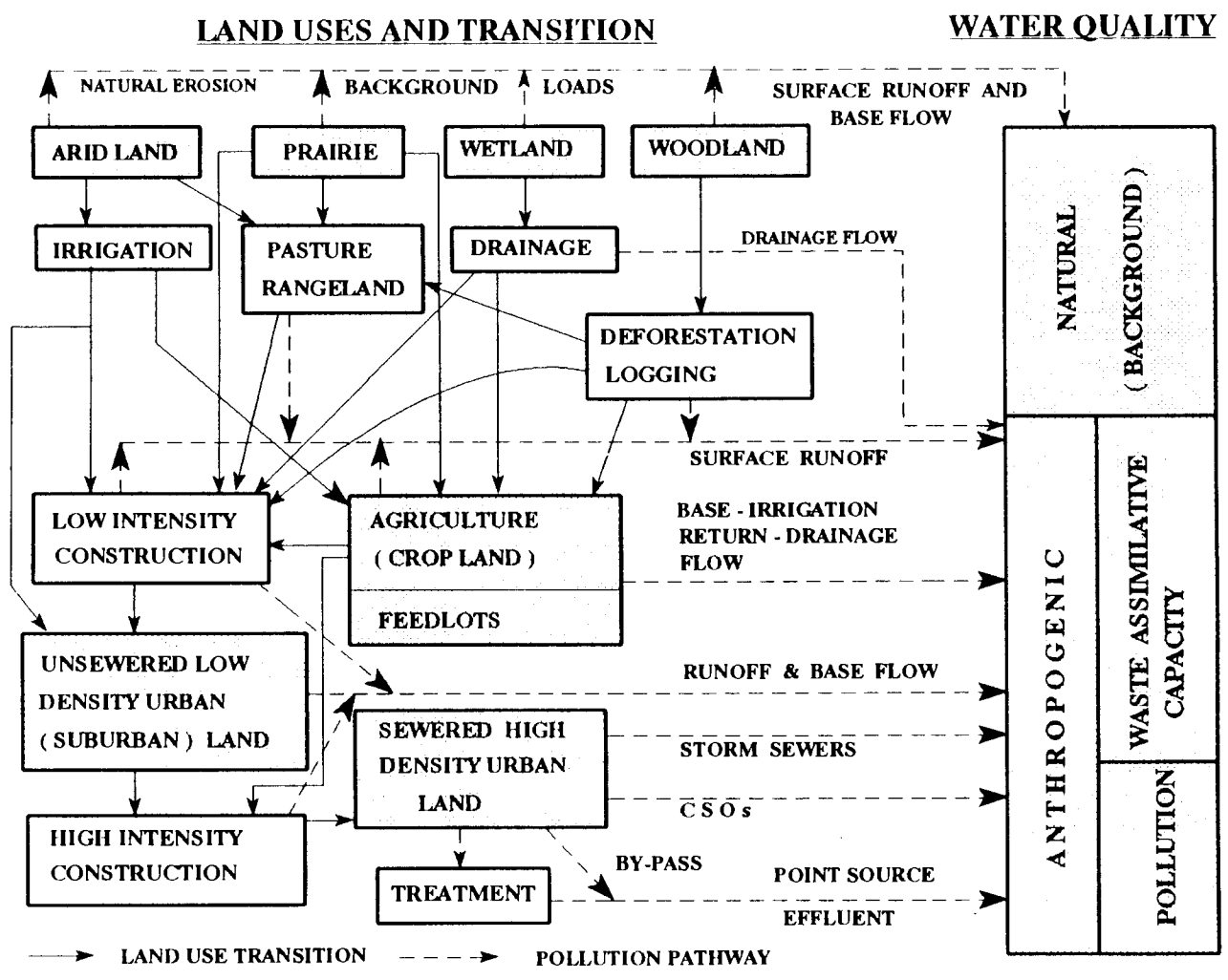


FIGURE 1.5. Land-use transformation and the generation of pollution.

R0023005



FIGURE 1.6. Deforestation.

atmospheric pollution, which by dry and wet deposition then affects the quality of water and terrestrial resources. It should also be pointed out, however, that the United States and Soviet Union are the largest sources of atmospheric emissions of carbon (1224 and 1000 million tons per year, respectively, in 1987), mostly from burning fossil fuel and vehicular traffic. These emissions far exceed those from the destruction of rain forests in developing countries (Brown et al., 1990). Developed countries are also guilty of large-scale deforestation. For example, the disappearance of forests in Florida due to urban development is of the same order of magnitude as the deforestation of the Amazon rain forest in Brazil.

Wetland Alteration

About 6% of the total land surface is wetland (Mitsch and Gosselink, 1986); however, like rain forest, wetlands are disappearing rapidly due to drainage and conversion to agricultural and urban uses. Wetlands

represent a natural sink for various potential pollutants, including sediment, nutrients, and organic compounds.

Wetland alteration or destruction can take several forms; however, drainage and filling are the most common. The fertile soil of drained prairie wetlands provided excellent crop yields, but also deprived the watershed of natural hydrologic buffers and water quality preservation. In addition to drainage for agricultural uses, other activities that adversely alter or destroy wetlands include their conversion to urban uses, transportation, peat mining and mineral extraction, flood control, navigation, and industrial activities.

Construction

Newly developing urban lands should receive special attention. For all land uses, this stage of land is characterized by high production of suspended solids caused by erosion of unprotected exposed soil and soil piles. Extremely high pollutant loads (see the subsequent sections) are produced from construction sites if no erosion-control practices are implemented. Therefore, in establishing pollutant loading related to land uses one must first determine whether the area is fully developed or if it is a developing area and/or if significant construction activities are taking place therein. An area is considered fully developed and established one year after completion of the development.

Conversion to Intensive Agriculture

The first conversion of American prairies to an agricultural use was the use of the land for *pasture* or *range land*. It should be noted that the native prairies (as well as prairie wetlands and woodlands) were actually sinks of potential pollutants. Timmons and Holt (1977) studied organic pollution and nutrient (nitrogen and phosphorus) losses from native prairies in west central Minnesota and found that annual losses of nutrients in runoff were less than the nutrient inputs in precipitation.

Today about 40% or 3.6 millions km², of the land area of the United States is used for grazing livestock. These are mostly unconfined systems. Although the trend is to confine cattle operations, unconfined production is expected to continue to predominate the beef and sheep industries.

The differences between *confined* and *unconfined animal operations* are the same as the division of pollution sources into point and nonpoint. Unconfined operations, that is, operations where cattle graze more-or-less freely over an extended pasture area are strictly nonpoint, diffuse sources. Confined operations are statutory point sources; however, the largest waste loads occur during storms and are carried by surface runoff, which is a characteristic of diffuse sources.

As grazing animals traverse pasture and rangeland, the top soil becomes more compacted. Trampling by cattle may reduce infiltration and increase surface runoff (Robbins, 1985); however, such increases were not significant for pastures with light and moderate intensities of grazing (Moore et al., 1979). In his studies Robbins (1985) also claimed that distinguishing between the water quality of streams affected by light to medium grazing and unaffected streams is difficult or impossible. He concluded that when management is directed to optimize forage production, pollutant yields from unconfined animal operations are not greater than would have occurred under native conditions.

However, it is not the grazing intensity on the land that determines the degree of pollution from rangeland, it is the unrestricted access of cattle to the watercourses that has the major impact. This is particularly true in high-quality cold water fishery areas of the Rocky Mountains and in slow-moving warm streams of the deep southwest. For example, a 100-hectare ranch with 200 steers that are excluded from the stream may have much less impact than a 100-hectare hobby farm with 50 head of cows/calves and steers that have a direct stream access.

Until the 1950s the expansion of *agricultural land for crop production* has more or less kept pace with the population increase. Most farming in the United States and the world was done on small family farms without an excessive use of chemicals. However, around 1950 farming began to qualitatively change. Dramatic changes happened in Eastern Europe, which followed the Soviet example of a forceful change to large collective farming, and also in the United States, where small family farming began to change into larger monocultural enterprises. Concurrently, farming began to rely more and more on the use of chemical fertilizers to increase plant yields and on pesticides for insect and weed control. Monocultural planting requires more chemical use than rotational planting.

Figure 1.7 shows a typical trend in U.S. agriculture. Until 1950 most increases in agricultural production were due to the conversion of native lands to agriculture. However, since 1950 the cultivated land area in Iowa has remained about the same, while in some other states it has decreased due to urban pressures. The increased farm output was solely due to the increased use of chemicals.

The primary agricultural activity that causes the elevated emissions of potential pollutants is the practice of disturbing the soil by tillage, which increases sediment losses, in comparison to the original native lands, by several orders of magnitude. Increased use of chemicals results in more losses into the environment. As noted by Alberts, Schuman, and Burnwell (1978), over 90% of nutrient (nitrogen and phosphorus) losses are associated with soil loss. Although nutrient losses represent only a

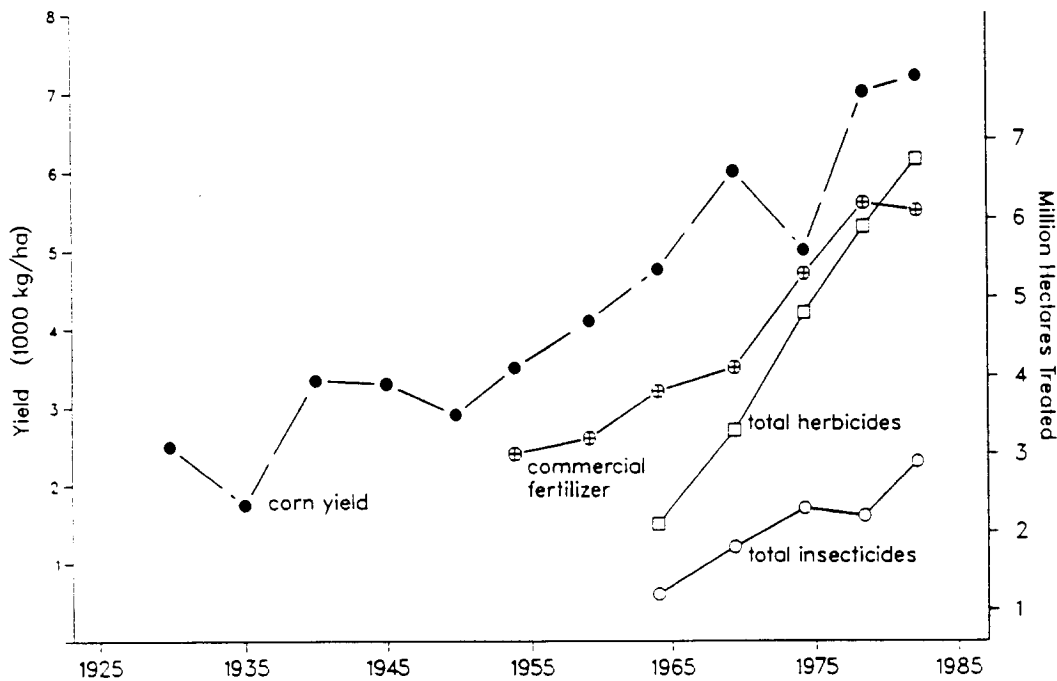


FIGURE 1.7. Trends in the use of chemicals in the agriculture (corn yield) of Iowa.

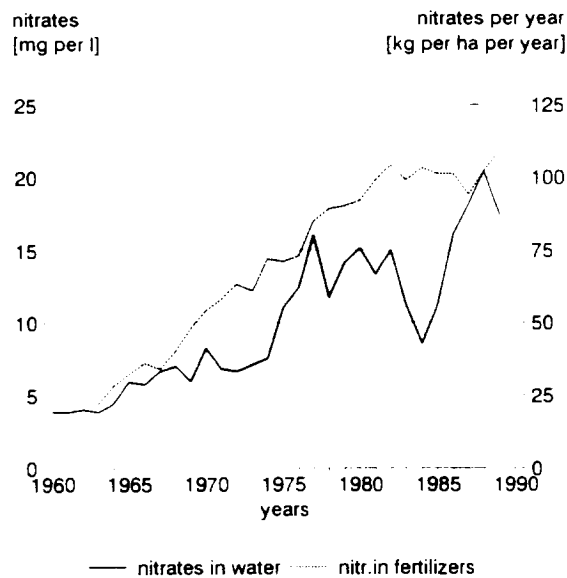


FIGURE 1.8. Relation of nitrate concentration (such as NO_3^-) in the Vltava River in the Czech Republic and fertilizer use in the watershed. (From Czechoslovak Academy of Science, 1992.)

small portion of the applied fertilizer, their contribution to the receiving waters almost always exceeds the standards accepted for preventing accelerated eutrophication of surface waters. Furthermore, the loss of nitrates into the ground water in areas of intensive agriculture are

TABLE 1.2 Typical Concentrations of Pollutants in Rural Runoff in the Midwestern U.S.

	Suspended Solids (mg/l)	BOD ₅ (mg/l)	COD (mg/l)	Total Nitrogen (mg/l)	Total Phosphorus (mg/l)	Total Coliforms (MPN/100 ml)
Background levels ^a	5-1000	0.5-3	NA	0.05-0.5	0.01-0.2	10-10 ²
Cropland ^b	(780)	NA	(80)	(9)	(1.2)	NA
Grazed pasture ^c	NA	(13)	NA	4.5	(7)	10 ⁵
Feedlots ^d	(30)	1000-11,000	31,000-41,000	920-2100	290-380	NA

Note: () = mean; NA = data not available or insufficient.

^aLager and Smith (1974).

^bWisconsin Priority Watersheds, Wisconsin Department of Natural Resources.

^cRobbins (1985).

^dLoehr (1972).

substantial and can make ground water unsuitable for drinking. Alarming levels of nitrate contamination of ground- and surface-water resources have been reached in Eastern and Central Europe. Figure 1.8 shows trends of nitrate concentrations in the Vltava River upstream of Prague, Czechoslovakia. The agricultural use of pesticides (atrazine in Central Wisconsin, the Po River valley in Italy, and elsewhere) is also responsible for contamination of ground and surface receiving waters. Table 1.2 presents typical concentration ranges of pollutants in agricultural runoff.

According to the U.S. EPA (1984), at the beginning of the 1980s cropland, pastureland, and rangeland contributed over 6.8 million tons of nitrogen and 2.6 million tons of phosphorus to conterminous U.S. surface waters per year. The Corn Belt (Illinois, Indiana, Iowa, Missouri, and Ohio) used 39% of the nation's phosphorus fertilizer and 32% of its nitrogen fertilizer.

Delivery losses of pesticides applied to agricultural lands to control insects and weeds amount on average to about 5% of the applied pesticide. Most of the loss ends as pollution of ground and surface waters. However, if rainfall occurs shortly after a pesticide is applied, losses can be substantial and can result in fish kills. Herbicides (weed control) are the most used pesticides in U.S. agriculture. In 1980, farmers used about 200,000 tonnes of herbicides and 140,000 tonnes of insecticides. Since 1980, these uses have doubled. Figure 1.9 shows total trends in pesticide use in the United States.

Agricultural use of arid lands requires irrigation. Civilizations flourished and vanished because of pollution due to irrigation. The problem is the salt content of irrigation water and the elutriation of

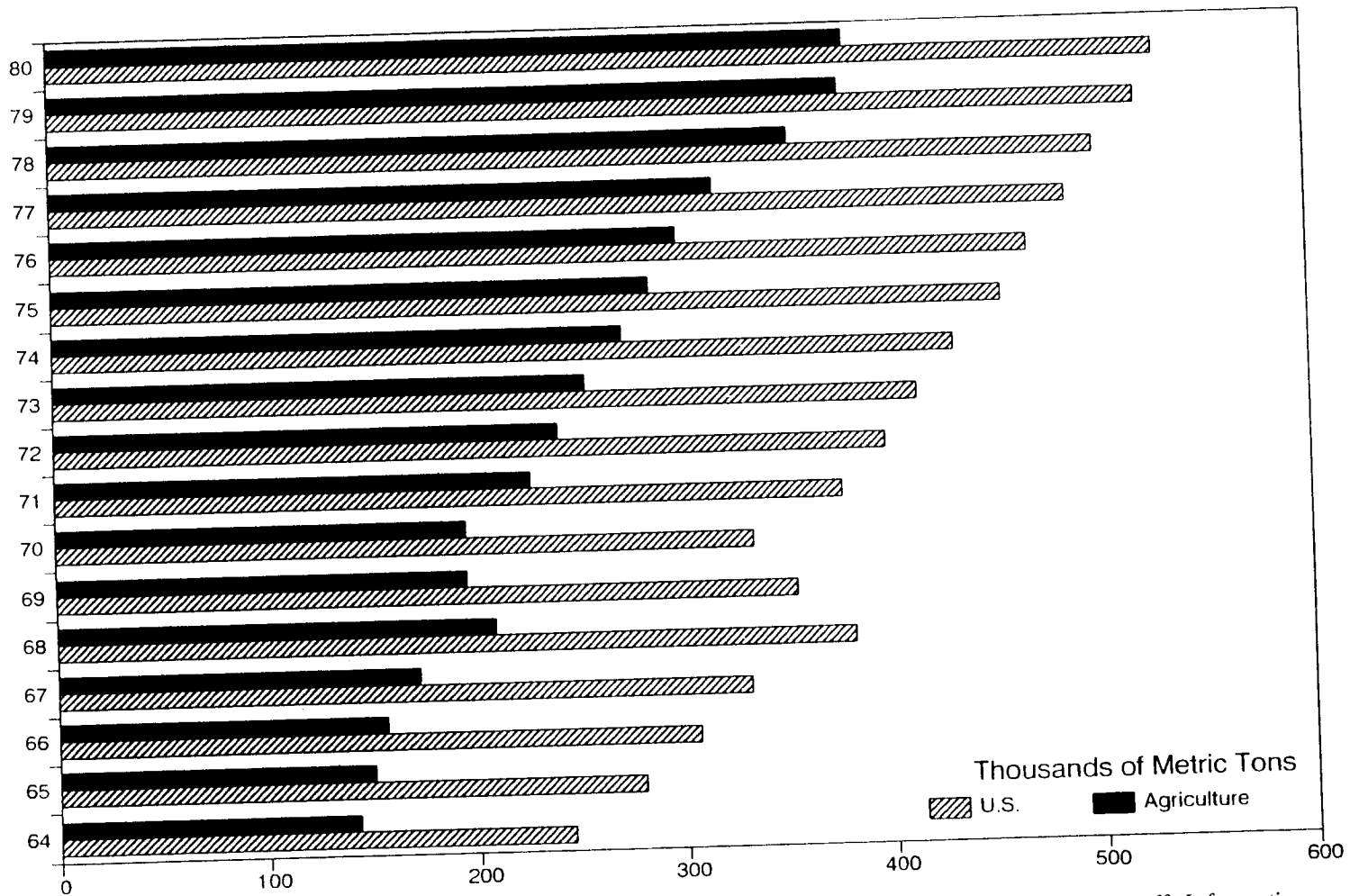


FIGURE 1.9. Total trends in pesticide use in the United States, 1964-1980. (From *Nonpoint Source Runoff: Information Transfer System*, EPA, Office of Water, July 1983, p. 2.7.)

minerals from soils during irrigation. Water lost by evapotranspiration into the atmosphere does not contain salt, hence in order to control salt buildup in soils, excess irrigation water over evapotranspiration losses must be applied. This water excess, called *irrigation return flow*, is considered pollution that gets deposited in surface or ground waters. In some irrigated parts of the arid southwest (such as the lower Colorado River Basin), salinity and pollution of streams receiving irrigation return flows have reached such high levels that the water is not suitable for further use. By an agreement between the United States and Mexico, excess salt from the Colorado River must be removed at great expense to the U.S. taxpayers.

Feedlots and Barnyards

Feedlots and barnyards, statutory agricultural point sources, are the land uses that exhibit the highest amounts of diffuse waste loads, mostly carried by surface runoff. With the advent of improved feeding techniques, farmers prefer not to put cattle to pasture, but to hold them in relatively small areas. The high organic content of the surface crust protects against erosion, and consequently sediment losses from feedlots are lower than those from uncompacted bare land surfaces. Nevertheless, as shown in Table 1.2, barnyard and feedlot runoff has extremely high concentrations of BOD₅, COD, organic nitrogen, and phosphorus. Even with these extremely high concentrations, land management techniques typical for diffuse sources and some "point source" structural abatement (lagoons and manure storage basins) are a more effective means of control than a point-source-type treatment would be (see Chapter 11).

Urbanization

Probably the greatest adverse change in water quality is due to urbanization. The population pressures mentioned previously, migration of population, and economic development result in urbanization.

Urbanization, that is, the transformation of land use from natural or agricultural lands, occurs in several steps. Urbanization changes atmospheric composition, the hydrology of the watershed (Fig. 1.5), receiving streams, and other water bodies, and soil. Native ecological systems are replaced by urban ecology. Waste emissions increase dramatically, and the sources of these contaminants are diverse, such as industries, household heating, transportation, sewage conveyance and disposal, garbage collection and disposal (landfills, incinerators), litter deposition, fallen leaves on impervious surfaces, and street salting. Chapter 8 discusses these sources in detail.

Watershed during Urban Development (Urbanization)

The soil loss from construction sites can reach magnitudes of over 100 tonnes/ha/year. In urbanizing watersheds, a few percent of the watershed under construction can contribute a major portion of the sediment carried by the streams. Often the streams themselves are affected and irreversibly changed by urbanization, regulation, straightening and lining, which destroys their natural habitat so that they can no longer sustain fish and other biotic populations. More surface runoff and hence flooding results from increased imperviousness that also makes surface runoff more flushy and higher in volume. On the other hand, ground-water recharge is reduced. Other profound and adverse changes in hydrology and water quality are caused by draining wetlands. Pollution is mostly nonpoint, unless a sewer system is in place and the drainage is located in a large urban area and/or industrial zone, which would legally reclassify the runoff pollution as a point source.

Unsewered Urban Development

Disposal of sewage into soils (septic tanks or small Imhoff tanks followed by infiltration) eliminates or reduces only pollutants that can be filtered out, decomposed, and/or adsorbed onto soil particles. Mobile pollutants such as nitrates can cause severe contamination of ground water and, subsequently, of the base flow of the streams and other water bodies. When the adsorption capacity of the soil-disposal system is exhausted, contamination of surface waters by organics and pathogenic microorganisms by surfacing sewage may occur and be severe. Failures of older septic tank systems are common. However, in the absence of such failures, the pollution potential of established low-density residential and commercial areas without storm or combined sewer systems is generally low and not much above the background water quality contributions from a natural prairie watershed.

Pollution from Fully Developed, Sewered Urban Watersheds and Transportation Corridors

As the imperviousness of the urban watershed increases, the watersheds become more hydrologically active, which will impact both pollutant loadings and flooding potential. The surface runoff events that carry the heaviest pollution loads become more frequent. Pollutants that accumulate on the surface from traffic, litter, street dust, and other sources are then washed off by the surface runoff into the drainage (sewer) system.

Novotny and Chesters (1981, 1989) have pointed out that the type of drainage can greatly affect the pollution load within the same land-use category. Residential areas with natural (swale) drainage produce

TABLE 1.3 Comparison of the Strength of Point and Nonpoint Urban Sources

Type of Wastewater	BOD ₅ (mg/l)	Suspended Solids (mg/l)	Total Nitrogen (mg/l)	Total Phosphorus (mg/l)	Lead (mg/l)	Total Coliforms (MPN/100 ml)
Urban storm-water ^a	10–250 (30)	3–11,000 (650)	3–10	0.2–1.7 (0.6)	0.03–3.1 (0.3)	10 ³ –10 ⁸
Construction site runoff ^b	NA	10,000–40,000	NA	NA	NA	NA
Combined sewer overflows ^a	60–200	100–1100	3–24	1–11	(0.4)	10 ⁵ –10 ⁷
Light industrial area ^c	8–12	45–375	0.2–1.1	NA	0.02–1.1	10
Roof runoff ^c	3–8	12–216	0.5–4	NA	0.005–0.03	10 ²
Typical untreated sewage ^d	(160)	(235)	(35)	(10)	NA	10 ⁷ –10 ⁹
Typical POWT effluent ^d	(20)	(20)	(30)	(10)	NA	10 ⁴ –10 ⁶

Note: () = mean; NA = not available; POWT = Publicly owned treatment works with secondary (biological) treatment.

^aNovotny and Chesters (1981) and Lager and Smith (1974).

^bUnpublished research by Wisconsin Water Resources Center.

^cEllis (1986).

^dNovotny, et al. (1989).

pollution loads that are approximately one order of magnitude lower than pollution loads from similar land with storm (separate) sewers.

Sewer systems can be either *combined* or *separate*. The pollution potential of urban runoff carried by separate storm sewers is similar to treated sewage, while that of combined sewer overflows is between treated and untreated municipal wastewater (Field and Turkeltaub, 1981). Rohmann, Lyke, and Hoban (1988) claimed that urban nonpoint source released 760 times more lead than the load from point sources. Table 1.3 shows the approximate concentration ranges for some components of diffuse urban sources compared to the strength of municipal wastewater.

In combined sewer systems, wet-weather pollution loads are divided into two components. The first component is the load that is conveyed by the combined sewer interceptors (due to their excess capacity over the dry-weather flows) toward the treatment plant. At the plant this load represents a point source and should not be considered in nonpoint pollution studies. The second component is conveyed by combined sewer overflows (CSO) that occur during wet weather when the interceptor capacity is exceeded. The load conveyed by overflows, referred to as *wet-weather* load, is of diffuse (nonpoint) nature and is considered in this study. Legally, however, the U.S. Environmental Protection Agency and state agencies consider the combined sewer overflows as a point source, for which a National Pollution Discharge Elimination System (NPDES) permit is required.

A part of the wet-weather pollution carried by combined sewers is diverted to the treatment plant. Even though the concentrations of pollutants in CSOs are greater than those in flows from separate storm sewers, the volume of discharged CSOs is smaller. Hence, European experience and measurements indicate that overall, the total pollution loads—including dry-weather point loads—from urban zones with separate and combined sewers are about the same (Novotny et al., 1989b).

In addition to the traditional pollutants reported in Table 1.3, urban runoff contains a variety of toxic (so-called priority) pollutants, such as oils, polyaromatic hydrocarbons (PAH), PCBs, and lead. With the ban on the use of leaded gasoline in the United States, the lead level in urban runoff has dropped significantly. Urban diffuse sources have been identified as a major cause of pollution in surface water bodies by the US EPA (Myers et al., 1985).

Using a hypothetical example, Pitt and Field (1977) showed that in an urban area of 100,000 people, the COD contribution from urban runoff is approximately 50% of that from raw sewage. These results, together with

TABLE 1.4 Comparison of Areal Loadings of Pollutants from a Hypothetical American City of 100,000 People (tonnes/year)

Pollutant	Storm Water	Raw Sewage	Treated Sewage
Total solids	17,000	5,200	520
COD	2,400	4,800	480
BOD ₅	1,200	4,400	440
Total Phosphorus	50	200	10
Total Kjeldahl Nitrogen	50	800	80
Lead	31		
Zinc	6		

Source: After Pitt and Field (1977).

comparisons for some other constituents, are shown in Table 1.4. It is also shown that if sanitary sewage is receiving adequate treatment (90 to 95% removal of solids and organic compounds), almost all—approximately 97%—of the total solids and 70% of the BOD reaching the receiving waters comes from urban runoff.

Example 1.1

Compare volume and pollution load (BOD₅) generated by sewage and by a 50-mm (2-in) 24-hr storm, after it comes from an urban catchment that is 70% impervious. The population density in the catchment is 100 persons/ha, with an average per capita sewage flow of 300 l/(cap-day).

Referring to Table 1.3 the average BOD₅ of urban runoff is 30 mg/l, and that for raw municipal sewage is 160 mg/l. Treated sewage has a BOD₅ of about 20 mg/l. Then the flow and volumes are:

Sewage

$$\begin{aligned}\text{Flow} &= 300 \text{ l/(cap-day)} \times 100 \text{ persons/ha} \\ &= 30,000 \text{ l/(ha-day)} = 30 \text{ m}^3 \text{/(ha-day)}\end{aligned}$$

$$\begin{aligned}\text{Raw sewage BOD}_5 &= 30 \text{ m}^3 \text{/(ha-day)} \times 160 \text{ g of BOD}_5 \text{/m}^3 \\ &= 4800 \text{ g/(ha-day)}\end{aligned}$$

$$\text{Treated sewage BOD}_5 = 30 \times 20 = 600 \text{ g/(ha-day)}$$

Stormwater

$$\begin{aligned}\text{Flow} &= 0.7 \times 50 \text{ mm/day} \times 10,000 \text{ m}^2 \text{/ha} \times 0.001 \text{ m/m} \times 1000 \text{ l/m}^3 \\ &= 350,000 \text{ l/(ha-day)} \\ &= 350 \text{ m}^3 \text{/(ha-day)}\end{aligned}$$

$$\begin{aligned}\text{Stormwater BOD}_5 \text{ load} &= 350 \text{ m}^3 \text{/(ha-day)} \times 30 \text{ g/m}^3 \\ &= 10,500 \text{ g/(ha-day)}\end{aligned}$$

The stormwater BOD load from this storm would be twice as large as the 24-hr BOD load by raw wastewater.

Pollutant Loads from Combined Sewer Overflows

Combined sewers have commonly been designed with a capacity large enough to accommodate flows that are 4 to 8 times the peak dry-weather flow. In Europe, combined sewer designs are now being implemented to handle a critical rainfall of 15l/sec-ha (5.4 mm/hr) plus the dry-weather flow. If the rainfall exceeds the critical rate, overflow from the sewer system is anticipated. To mitigate pollution by the combined sewer overflow (CSO), over 10,000 storage tanks have been built in Germany alone and another 10,000 are planned (Geiger, 1990).

Using this criterion for a typical city located in Switzerland, out of about 1000 hours of rainfall in a typical average year, overflows would occur for about 80 hours (Krejci, 1988). In Milwaukee, Wisconsin, in an average year there were over 40 overflows from combined sewers in the older sections of the city. A large tunnel-interceptor to be completed in 1994 will store the overflows for subsequent treatment.

The pollution strength (Table 1.3) of the overflows is about the same or only marginally less than the strength of raw sewage. However, due to the larger volumes when compared to typical dry-weather sewage flows, overflow from a larger storm represents a pollution shock to the receiving water body that may greatly exceed, during the time of overflow, the load by sewage.

As stated by Imhoff and Imhoff (1990) and documented in Table 1.5, which is taken from Waller and Hart (1986), suspended-solids loadings from separate and combined sewers are about the same, while nutrient loadings from an urban watershed with combined sewers and a treatment

TABLE 1.5 A Comparison of Pollutant Loadings (kg/ha-yr) from Separate and Combined Sewer Systems for Ontario Urban Areas

Pollutant	Separate Systems			Combined Systems		
	Dry-weather STP Effluent	Wet-weather STP Effluent	Surface Runoff	Dry-weather STP Effluent	Wet-weather STP Effluent	Combined Sewer Overflows
Suspended solids	194	24	553	383	66	490
Total Nitrogen	133	16	11	253	44	25
Total Phosphorus	8	0.9	1.1	15	2.5	4.5

Source: After Waller and Hart (1986). (Copyright © 1986 by Springer Verlag; reprinted with permission.)

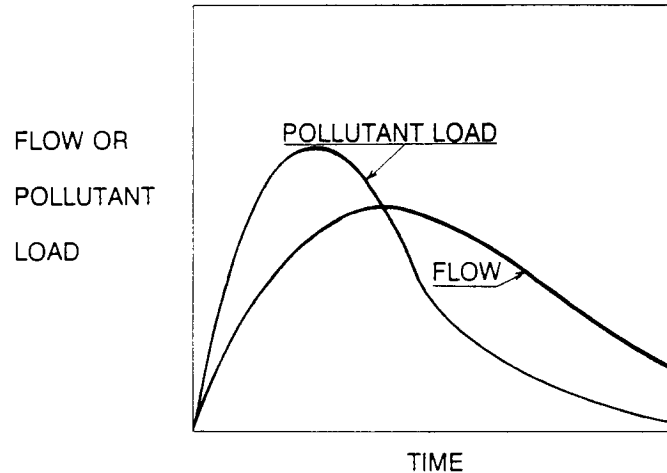


FIGURE 1.10. The first flush concept.

plant, but without storage for the overflows, are about twice that if the same watershed has separate sewers. Sewage solids that accumulate on the bottom or as a slime on the walls of the combined sewers during a dry period preceding a storm may contribute a higher pollutant load to the first portion of the overflow, the so-called “first flush.” If the drainage system has a significant supply of readily washable solids located especially in the sewer system, the peak concentrations and pollutant loads will precede the peak flow and volume (Fig. 1.10). However, Ellis (1986), in summarizing primarily the European experience, stated that the occurrence of first flush is not a consistent feature of either the separate or combined sewer systems. Nevertheless, controlling pollution in the first portion of the runoff hydrograph makes sense (see Chap. 10).

Mining Nonpoint Sources

Unlike construction activities, mining cannot be viewed as a homogenous source of nonpoint pollution. The most common minerals extracted by mining are coal and metallic ores. Mining nonpoint sources include discharges from inactive mining operations as well as runoff from roads, old tailings, and spoil piles. Active mines are considered as point sources for which a discharge permit is required.

Although mining is not as widespread as agriculture, water quality impairment resulting from mining is usually more harmful; sediment discharges and concentrations from mines can be extremely high. Furthermore, entire streams may be biologically dead as a result of acid mine drainage (U.S. EPA, 1984). Erosion and sedimentation problems are associated with almost every abandoned surface coal mine. Other

pollutants associated with mining operations can have an even more serious water quality impact than those associated with sediments.

POLLUTION BY TOXIC CHEMICALS AND METALS

Distribution and Types of Toxic Chemicals

Diffuse pollution may be responsible for the major part of the contamination of the environment by toxic pollutants. Toxic chemicals and components contaminating the environment are either *inorganic* or *organic*. Inorganic contaminants are mostly in a category of trace metals, which may be natural or anthropogenic (man-made) or both. Other inorganic nonmetallic toxic compounds detected in the aquatic environments are unionized ammonia (NH_3), cyanides, asbestos, hydrogen sulfide (H_2S), and low or high pH. Most of the organic toxic contaminants are anthropogenic, a category that includes organic chemicals such as pesticides, PCBs, organochlorine chemicals, solvents, and polyaromatic hydrocarbons (PAH).

As is pointed out in detail in Chapter 13, almost any compound may become toxic to aquatic biota, water fowl, or man when a certain tolerance threshold level is exceeded. From the many thousands of potentially toxic compounds and elements, the category of the so-called *priority pollutants* includes the toxic compounds that can be found in the environment in concentrations and quantities that can be toxic. These pollutants represent a risk to ecology and human health. The priority pollutants category includes both inorganic and organic (the majority) toxic compounds. Many but not all priority pollutants are carcinogenic.

Marsalek (1986) studied the distribution of toxic compounds in Ontario. He found that in or near urban areas trace metallic elements were the most prevalent toxic contaminants in runoff and stream sediments. The most frequently detected elements were zinc (98% of all water samples), copper (93%), nickel (87%), and lead (78%). Similar results were also found by the National Urban Runoff Project in the United States (Table 1.6). In both studies the frequencies of detection of trace elements in sediments were higher than those in water columns.

Among organic chemicals, pesticides were most frequently detected in the Canadian and U.S. studies. Marsalek (1986) found that in Ontario two organochlorine compounds, α -BHC, and γ -BHC (lindane) were found most frequently (98% and 86% of all water samples, respectively). The detection frequencies for chlorinated benzenes varied from 3% to 64% of all water samples. The detection frequencies of PAH compounds

TABLE 1.6 Most Frequently Detected Toxic (Priority Pollutants) in NURP Urban Runoff Samples

Detection rate ^a	Inorganic	Organic
Detected in 75% or more of NURP samples	Lead (94%) Zinc (94%) Copper (91%)	None
Detected in 50%–74% of the NURP samples	Chromium (58%) Arsenic (52%)	None
Detected in 20%–49% of NURP samples	Cadmium (48%) Nickel (43%) Cyanides (23%)	Bis(2-ethylhexyl) phthalate (22%) α -Hexachlorocyclohexane (20%)
Detected in 10%–19% of the NURP samples	Antimony (13%) Beryllium (12%) Selenium (11%) γ -Hexachlorocyclohexane (Lindane) (19%) Pyrene (15%) Phenol (14%) Phenanthrene (12%) Dichloromethane (methylene-chloride) (11%) 4-Nitrophenol (10%) Chrysene (10%) Fluoranthene (16%)	α -Endosulfan (19%) Pentachlorophenol (19%) Chlordane (17%)

Source: U.S. EPA (1983).

^a Percentages indicate frequency of detection, not concentrations.

varied from 2% to 19% in water samples and from 1% to 37% in sediment samples. The high frequency of detection of some of these toxic organic and inorganic chemicals today confirmed Rachel Carson's worries in *Silent Spring*. The origins of these toxic compounds are very diversified. With the exception of trace elements that have a natural occurrence, organic toxic chemicals are man-made and entered the environment mostly after World War II.

The example of PCBs shows how a chemical component, originally considered to be very useful for a number of applications and that represents only a minor traditional point source problem, could contaminate the environment and become a serious diffuse-pollution problem. A similar fate has occurred to asbestos, which until the 1960s, was a material widely used for insulation, fire protective materials, tires, sewer pipes, brake linings, and many other applications. Both compounds

have subsequently been recognized as both a carcinogenic threat to humans and a persistent widespread pollutant.

Sources and Properties of PCBs

PCBs are in a class of chlorinated organic compounds. These substances have been manufactured in the United States since 1929 by Monsanto Chemical Company. Other PCB-producing nations included Germany, France, Japan, Italy, the USSR, and Czechoslovakia.

PCBs are prepared individually by the reaction of biphenyl with anhydrous chlorine, and are available as liquids, resins, or solids. The most important physical properties of PCBs are their low vapor pressure (high boiling point), low water solubility, and high dielectric constant. They can be dissolved in many organic solvents, including some humic acids. The PCBs manufactured in the United States were sold under the trade name Aroclor. Table 1.7 shows some physical properties of PCBs as compared to another environmentally serious contaminant—DDT. Since 1929 over 600,000 tonnes of PCBs have been produced in the United States, with annual production reaching its maximum of 38,000 tonnes in 1970. Figure 1.11 shows the approximate distribution of PCBs that are said to have contaminated the environment.

From an environmental viewpoint, the commercial use of PCBs can be divided into three categories:

- *Controllable closed systems:* PCBs used as dielectrics in transformers and large capacitors have a life equal to that of the equipment, and with proper design environmental contamination should not occur.
- *Uncontrollable closed systems:* PCBs are used in heat transfer and hydraulic systems that permit leakage.

TABLE 1.7 Physical Properties of Some PCBs and DDT

	AROCLOR				DDT
	1242	1248	1254	1260	
Molecular weight					
Range	154–338	222–358	290–3982	324–460	352
Average	262	288	324	370	352
Chlorine (%)	42	28	54	60	50
Solubility in H ₂ O (µg/l)	200	100	50	25	0.7
Vapor pressure at 20°C mmHg	10 ⁻⁴	3 × 10 ⁻⁵	3.6 × 10 ⁻⁶	1.5 × 10 ⁻⁷	

Source: After Nisbet and Sarofim (1972).

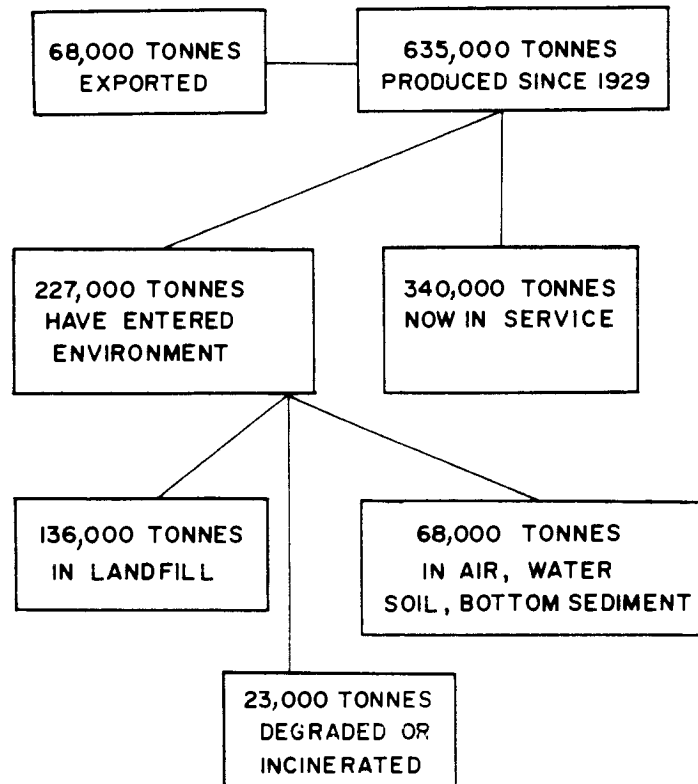


FIGURE 1.11. Distribution of PCBs. (From the Committee on the Assessment of PCBs in the Environment, 1979.)

- *Dissipative use:* PCBs in paint, lubricants, and plasticizers are in direct contact with the environment.

The major source of PCBs that have contaminated the environment are (Fig. 1.12):

- Leaks from sealed transformers and heat exchangers;
- Leaks of PCB-containing fluids from hydraulic systems that are only partially sealed;
- Spills and losses in the manufacture of either PCBs or PCB-containing fluids;
- Vaporization or leaching from PCB-containing formulations;
- Disposal of waste PCBs or PCB-containing materials and the migration of PCBs from landfills and freshwater sediments.

The chemical property of extreme stability that made PCBs an ideal industrial and commercial compound also made them persistent and cumulative toxic components in the environment. Because PCBs are soluble in lipid tissue, these components have been found to accumulate

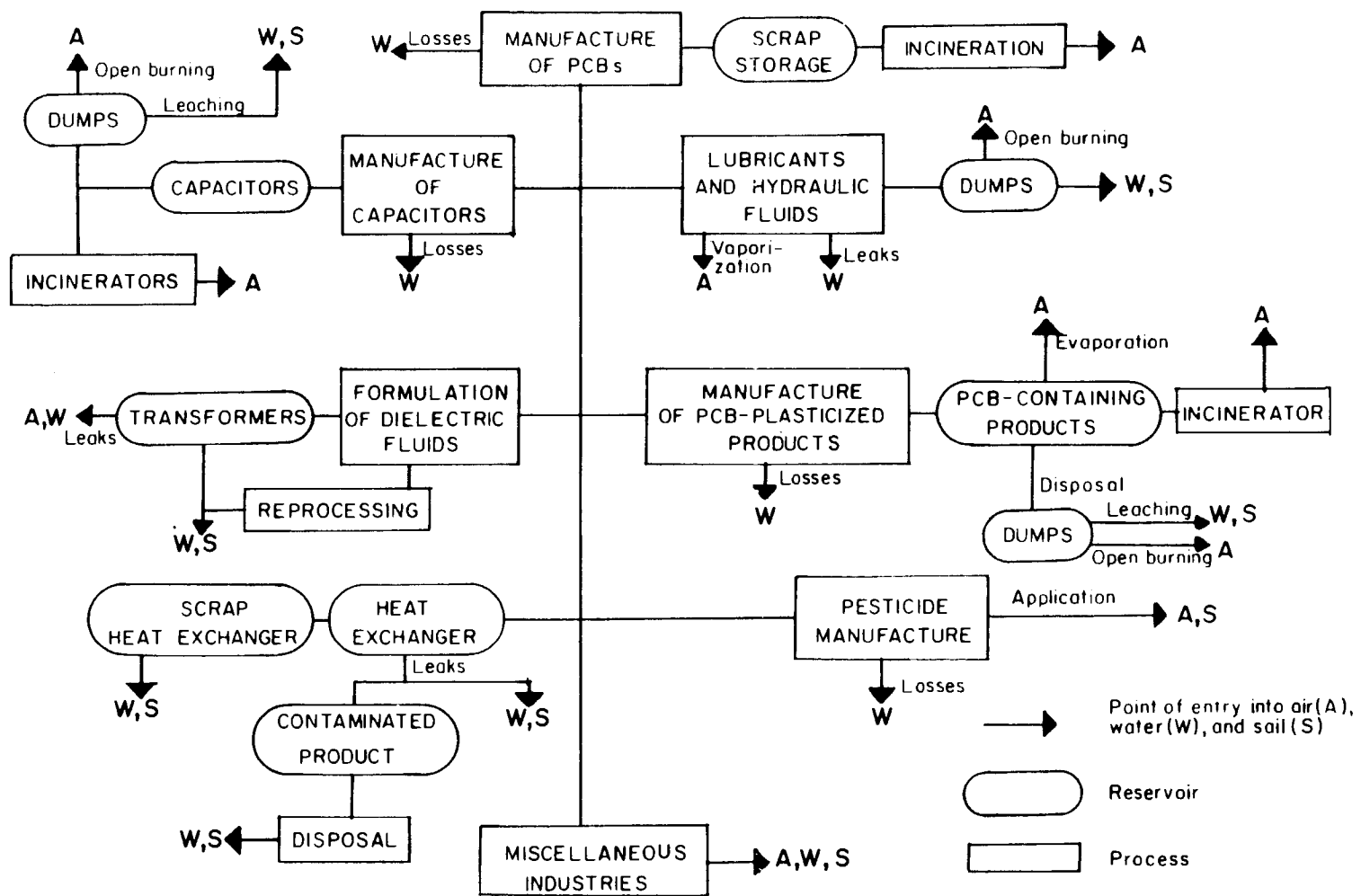


FIGURE 1.12. Pathways of PCBs (From Nisbet and Sarofim, 1972.)

in the fat of living organisms including humans. Salmonoid fish are very susceptible to PCB accumulation because these species have high body fat. Because of the toxic (carcinogenic) nature of PCBs, the U.S. Food and Drug Administration established a tolerance level of 2 mg/kg of PCB in fish tissue. Fish testing has shown that most large Lake Michigan fish species (trout and salmon) contain PCB levels exceeding this limit.

PCBs, which are in a way similar to DDT in the 1950s and 1960s, have entered into global transport routes and can be detected in extremely remote areas. Their removal rate from the environment is very slow, and the primary sink in aquatic ecosystems is burial with organic sediments.

IMPORTANT EXAMPLES OF SYSTEMS AFFECTED BY DIFFUSE POLLUTION

Chesapeake Bay

The Chesapeake Bay is the largest estuary on the east coast of the United States. It has been one of the world's most productive water bodies, providing habitat to fish and shellfish. However, water quality in this relatively shallow body of water has been declining. Submerged aquatic vegetation has been disappearing; fishers have been landing fewer of certain freshwater spawning fish; and oyster harvests have been declining. These problems were traced to excess levels of nutrients and toxic pollutants in the bay system. A 1983 EPA study concluded that these contaminants were also causing, among other phenomena, depressed oxygen concentrations in the water column, algal blooms, increased turbidity, and high concentrations of heavy metals in sediments. The study also found that diffuse (nonpoint) sources of pollution were among the chief causes of the bay's decline.

As a result of these developments, the governors of Pennsylvania, Maryland, and Virginia, the Mayor of the District of Columbia, and the Administrator of the U.S. Environmental Protection Agency in December 1983 pledged to address the problem of nonpoint pollution as well as other sources of pollution in order to restore and protect the Chesapeake Bay. This commitment, known as the Chesapeake Bay Agreement of 1983, established the Chesapeake Executive Council to coordinate all bay cleanup efforts undertaken by the signatories of the agreement. Implementing programs to reduce nonpoint source pollution is one of the most significant elements of the cooperative effort.

The Problem

Excessive nutrients appeared to account for much of the decline in living resources as well as many of the trends in water quality deterioration

TABLE 1.8 Nutrient Loads Reaching the Chesapeake Bay from Diffuse Sources

Land-use Type	Total Nitrogen from Diffuse Sources (%)	Total Phosphorus from Diffuse Sources (%)
Cropland	45-70	60-85
Pasture	4-13	3-8
Forest	9-30	4-8
Urban/suburban	2-12	4-12
Subtotal for Agriculture (Cropland + Pasture)	49-83	63-93

Source: U.S. Environmental Protection Agency (1988).

specified in the preceding paragraph. A watershed model was used to identify the sources. The model calculations indicated that in a year of average rainfall, diffuse (nonpoint) sources contribute 67% of the nitrogen and 39% of the phosphorus entering the bay and that the traditional point sources (wastewater effluents) account for the rest. Runoff from cropland was identified as the largest diffuse source of nutrients in the bay (Table 1.8). Urban runoff loads were minor and caused only localized problems. However, due to urban development pressures, the urban runoff contribution is expected to increase.

The watershed model, along with other information, has provided the basis for understanding the relative contributions of point and nonpoint by major river basins, and linked the nutrient loadings with specific areas where nutrient and dissolved oxygen concentrations potentially limited the aquatic resources.

The inorganic toxic loads to the bay system are also of concern. These toxics affect the aquatic life and habitats, especially near urban centers. The first EPA study (1983) found that in some areas toxic concentrations in sediments had reached magnitudes that reduce the hatching and survival of aquatic organisms, cause gross effects such as lesions or fin erosion in fish, and eventually could destroy an entire population of some sensitive species. Toxic discharges of metals from traditional point sources and from urban runoff appear to be most significant in urbanized/industrialized areas such as Baltimore, Maryland, Norfolk, Virginia, and Washington, D.C. As far as pollution by organic chemicals is concerned, it was concluded that herbicides were not the primary culprit in the decline of the bay ecological system.

Abatement Program

The Chesapeake Bay Agreement has established a framework for cooperation. Concurrently, the U.S. Environmental Protection Agency,

which is a partner in the program, has established a liaison office in Annapolis, Maryland. The federal support for the program provided implementation funds to each state and to Washington to carry out the program. The bay states and Washington, D.C., have developed a variety of approaches to address the diffuse pollution problem; however, their approaches reflect the diverse problems and priorities in each of the jurisdictions. Furthermore, each jurisdiction began with a different base of laws and regulations for the control of the sources of pollution and with varying amounts of resources.

In the agricultural sector, the Chesapeake Bay states have been relying primarily on voluntary cost-sharing programs to carry out their program objectives. These programs are helping farmers reduce soil and associated nutrient losses into the bay. The programs actually build upon the soil-erosion control programs begun in the 1930s by establishing soil- and water-conservation districts. All states targets the diffuse sources at several levels:

- First, the states have targeted general geographic areas where each will emphasize implementation of agricultural diffuse-pollution controls.
- Second, once a general area has been identified, all the states have procedures to target the critical areas and management needs within that area.
- Third, state and local staff identify cost-effective, site-specific management practices for individual landowners and users.

Urban programs to control diffuse sources within the bay watershed lean more toward regulation. The urban sediment control programs in the area have existed since the 1970s. Later sediment control regulations for developing areas were added.

Progress in the 1980s has been slow. For example, the percentage of highly erodible lands on which best management practices were implemented in the 1985/1986 period ranged from 2% to 12% between the states. Installations of animal-waste management (mostly winter storage) in the same period ranged from 0.5% in Pennsylvania to 10% in Maryland. By 1990 Maryland had shown the highest reduction of nutrient inputs (nearly 20% for both nitrogen and phosphorus), while reduction in the other two states was significantly lower. These reductions in nutrient loadings were insufficient to yield significant water quality improvement at the time this book was being written.

Wisconsin Priority Watershed Program

In 1978 the Wisconsin legislature created and funded the Wisconsin Nonpoint Source Water Pollution Abatement Program. The basic

purpose of the program was to systematically control nonpoint source pollution so that the surface-water and ground-water quality goals specified by the Clean Water Act were met. The program was designed to deal with the varying nature of nonpoint pollution problems throughout the state. These problems include pollution from croplands, construction sites, stream-bank erosion, and nutrient loads from barnyard runoff, cropland erosion, manure spreading on croplands, and runoff from city lawns and streets (Konrad, 1985). In 1993, funding for this program was increased by the Wisconsin legislature.

The Wisconsin program concentrates available funds into selected hydrological units (watersheds) that exhibit large problems due to diffuse-pollution inputs. Such units are then called *priority watersheds*. This program allows all categories of urban and rural diffuse sources within the watershed area to be addressed and solved in a comprehensive and coordinated effort. Specific areas within the priority watershed that contribute significant amounts of pollutants to lakes and streams are collectively called *priority management areas*. This approach enables effective targeting of most significant pollution sources. The Priority Watershed Program concentrates available educational, financial, and technical resources in those critical areas where maximum water quality benefits will result from investing financial and human resources.

The selection of a Priority Watershed Project is followed by an 8- to 9-year planning and implementation process. An implementation plan, based on a detailed inventory and assessment of critical source areas in the watershed and the program objectives, is prepared, usually within the first year of the project. The plan guides the Priority Watershed Project and spells out procedures and responsibilities. Central to each Priority Watershed Project are the water quality objectives identified for the lakes and streams to be protected. The determination of critical pollutants, significant sources, the desired level of diffuse-pollution control (pollutant reduction), and the measurement of results are all based on the specific water quality objectives for each individual watershed.

Some of the water quality objectives identified for the Priority Watershed Projects are:

1. Protection of the near-shore waters of Lake Michigan;
2. Rehabilitation of warm-water fishery;
3. Rehabilitation of cold-water fishery, such as the upgrading of a trout stream through habitat improvement;
4. Protection of a desired warm-water fishery;
5. Protection of a desired cold-water fishery;
6. Inland lake rehabilitation;
7. Protection of an inland lake.

The State provides financial support in three major categories: (1) cost-share (50% to 70%) for landowners and municipalities to install management practices; (2) aids for local governments to fund additional technical assistance, education and information, and financial and project management; and (3) administrative and planning funds for state administration and the preparation of priority watershed plans.

The Milwaukee River Priority Watershed Program

The Milwaukee River Priority Watershed Program is an example of a watershed-wide approach to solving excessive nonpoint pollution problems (Gayan and D'Antuono, 1989). The Milwaukee River is the largest of the three tributaries of the Milwaukee, Wisconsin, Harbor, which is a freshwater estuary of Lake Michigan. The drainage area of the Milwaukee River watershed is about 1800 km², with a resident population of about 500,000. The average flow of the river at the entrance into the harbor is 5.6 m³/sec, and the low flow characteristics for pollution abatement studies and planning (so-called Q₇₋₁₀, that is, the 7 days duration for 10 years expectancy of low flow) is 0.72 m³/s.

The upper two-thirds of the watershed are mostly composed of agricultural and mixed land uses, while the lower third is made of suburban and urban (Milwaukee County) land uses. All sewage flow from the city of Milwaukee and its sewered suburban sections is collected and diverted to two regional treatment plants located outside the watershed boundary. As a result, biologically treated sewage (point pollution) from only about 31,000 inhabitants is currently discharged into the river. Phosphorus is removed from the point-source effluents to meet the standard of 1 mg P/l for sources located in the Great Lakes Basin. Hence, most of the pollution of the river is of nonpoint origin. In the lower urban portion and in the harbor the river receives combined sewer overflows (CSO) from about 40 km² of the older section of the city of Milwaukee. Most of the city and all of sewered suburbs (more than 90% of the area served by the Milwaukee Metropolitan Sewerage District) are drained by a separate sewer (storm and sanitary) system. A large tunnel-interceptor, providing about 1.5 million m³ of storage space for interception of CSO and infiltration flow excess, has been built in the dolomite formation 100 meters below the city surface. The total cost of the point source abatement program (deep tunnel, treatment plant, and sewer system rehabilitation) is over \$2.5 billion.

With the upstream point sources greatly reduced or eliminated, the existing water quality problems must be attributed to diffuse sources. These problems include excessive algal growth in the upper reaches of the



FIGURE 1.13. The Milwaukee River is affected by nutrient discharges.
(Photo: V. Novotny.)

river (Fig. 1.13) and dissolved oxygen (DO) depletions in the impounded lower reaches of the river.

The mayor of Milwaukee, Wisconsin, found that after spending more than \$2 billion (mostly subsidized by the federal government) on sewer rehabilitation and wastewater treatment, including expensive means of storage and treatment of overflows and by-passes of untreated wastewater, the receiving water bodies might not be clean and suitable for most of the uses specified by the Clean Water Act due to upstream mostly unquantified diffuse discharges of pollutants. He found this situation politically damaging and irresponsible to the taxpayers.

As a result of serious nonpoint pollution problems, caused by upstream nonpoint sources, that could diminish the water quality benefits of the very expensive point source pollution abatement program, the state legislature declared the Milwaukee River a priority watershed. The Priority Watershed Program focuses on the control of pollution loads caused by soil loss and from concentrated animal operations. The integrated water resource management plan has nine components and goals:

- Reduction of nutrient and sediment contributions to surface and ground water throughout the watershed by 30%. This goal is being accomplished by soil-conservation practices such as strip cropping and no-till planting in agricultural zones, erosion control of construction sites, and by sedimentation ponds in urban areas.
- Reduction of toxic contaminants in surface and ground water. Most of toxic contamination originates from abandoned landfills and urban runoff.
- Enhancement and protection of wetlands. Existing wetlands are now protected and previously drained wetlands adjacent to watercourses are being restored.
- Buffer strips along the watercourses. Five percent of existing agricultural lands are being converted to perennial grassland to provide protective buffer strips along watercourses.
- Conversion of highly erodible land to woodland. In addition to the funding provided by the state of Wisconsin from the Priority Watershed Program, additional funding and incentives to farmers to install soil-conservation practices and for barnyard waste management are provided by the Food Security Act of 1985. This act also provides compensation to farmers for taking highly erodible lands and lands adjacent to watercourses out of production.

The combined Milwaukee River water quality enhancement plan consists of:

- *The Priority Watershed Program* aimed at nonpoint sources of pollution in the upper, mostly rural and suburban (unsewered) parts of the basin;
- *Milwaukee Metropolitan Sewerage District Water Pollution Abatement Program*, an ambitious \$2.5 billion action plan focusing on abatement of point source pollution, including CSO;
- *The Milwaukee Harbor Plan*, involving in-stream measures such as low-flow augmentation and aeration during low DO periods.

These programs could be considered an example of an integrated approach to a regional water quality problem.

Lagoon of Venice

The Lagoon of Venice, Italy, is a tidal embayment located in the northern Adriatic Sea. This relatively shallow water body has an area of 500 km². The total surface area of the basin is about 1700 km². The land distribution today is about two-thirds agricultural and one-third urban (metropolitan Venice and surrounding communities), with a resident

population of 1,000,000. During the tourist season the population greatly increases due to influx of tourists.

The historical center of Venice (resident population of about 80,000 people plus tourists) is located on an archipelago of about 120 small islands inside the lagoon and is connected with the mainland by a causeway and railroad bridge.

Evolution of the Water Quality Problem

Until the last century the drainage basin of the Lagoon of Venice was composed primarily of lowland marshes and wetlands transected by canals and small tributary streams. Marshes and wetlands served as sinks for nutrients that promoted the growth of lush vegetation, while at the same time protecting the lagoon and other waterways from today's symptoms of eutrophication and hypereutrophication exhibited by extensive and obnoxious algal blooms throughout the lagoon.

The Lagoon of Venice is a dynamic ecological system that throughout the centuries has been subjected to cultural (anthropogenic) changes and evolution that have included large hydraulic works such as the dredging of navigational canals and the relocation of two major tributaries outside of the lagoon to reduce siltation, as well as daily maintenance of the canals. These works have been carried out by Venetians for centuries.

In some ways the evolution of the water quality problems of the lagoon is a consequence of the watershed transition delineated in Figure 1.5. This transition has been in progress for a period of 100 years, starting from a mostly rural natural lowland-wetland watershed and moving to an urbanized and agricultural drained basin of one million inhabitants.

The historical center city of Venice located inside the lagoon, has an interesting but difficult diffuse-pollution problem. The historic city, which currently has about 80,000 inhabitants, but which during the height of the Venetian Republic 500 years ago, had a population exceeding 250,000, has no sewers. All sewage from the present population and from thousands of tourists visiting this historical treasure, plus urban runoff from the city's almost 100% impervious lands, are discharged by individual houses and street outlets directly into the famous canals (Fig. 1.14). As a result the canals inside the city are and have been for centuries severely polluted, with sludge deposits accumulating on the bottom of the canals. Odors, due to anaerobic decomposition, have plagued the city since the Middle Ages. The introduction of flushing toilets and the influx of tourists in recent years have intensified these problems. Tides provide the only flushing action by which the pollutants can be carried away from the historic city into the lagoon and through the gaps between the beach islands into the Adriatic Sea. In 1990 only about



FIGURE 1.14. A canal in Venice. (Photo: V. Novotny.)

one-half of the urban sewage on the mainland received some treatment; the rest was discharged without treatment into mainland waterways and subsequently into the lagoon.

The drainage work that was begun in the basin of the Lagoon of Venice approximately in 1880 and still may continue, has transformed the wetlands of the basin into agricultural and urban dry lands. A complex network of drainage canals with pumping stations has dropped significantly the ground-water levels throughout the basin and large quantities of drainage and irrigation return flows rich with nutrients and residues of other agricultural and industrial chemicals are now directed toward the lagoon (Fig. 1.15).



FIGURE 1.15. Terra Ferma (mainland) canal draining into the Lagoon of Venice, which is affected by the elevated nutrient concentrations. (Photo: V. Novotny.)

When the wetlands were drained the lagoon was deprived of its natural buffering system for nutrients, which before the drainage work retained them. Increased use of fertilizers by the agricultural sector, especially after the switch from organic (manure) fertilizers to chemicals, is another factor contributing to the greatly increased transport of nutrients from the basin to the lagoon. On top of these loads, the largest Italian producer of chemical nitrogen fertilizers (plus other chemicals) is located on the shore of the lagoon.

Originally, the lagoon itself was surrounded by brackish tidal marshes, which still remain in some parts of the lagoon. These marshes constituted an additional natural buffer, shielding the lagoon from the influx of nutrients by runoff and shallow ground-water flow. Impounding the marshes and forming fish ponds (*valli da pesca*) and draining to reclaim land limited this buffering capacity.

The watershed itself is mainly composed of flat lowlands, hence, erosion and soil losses are minimal (mostly from construction erosion in sewered urban watersheds). Most of the nutrient load from rural watersheds is carried by drainage flows.

Pollution Loads

In the agricultural areas annual average fertilizer applications range from 25 to 280 kg/ha of nitrogen and 15 to 140 kg/ha of phosphorus, depending on the type of crops. These fertilizer loads and their losses resulted in mean concentrations of nitrogen and phosphorus in the mainland drainage canals of between 1 and 20 mg/l of nitrogen and 0.1 and 4.5 mg/l of phosphorus (Bendoricchio, 1988). As a result the mainland canals produce high quantities of algal biomass, as documented on Figure 1.15.

Urban diffuse loads (excluding sewage and industrial wastes) are also significant; however, the greatest loads originate from the mainland industrial area where chemical fertilizers are produced. Unit loadings of pollutants from urban sources and their comparison to sewage loads are given in Table 1.9.

The annual nitrogen and phosphorus load to the lagoon from all sources has been estimated as from 10,000 to 15,000 tonnes of nitrogen/year and 1000 to 2000 tonnes of phosphorus/year from which agricultural nonpoint sources were responsible for about 50% of the load (Bendoricchio, 1988). The lagoon is greatly affected by these loads. It should be realized that, when converted to water-surface loading, these loads would be devastating to an average lake; however, the tidal water exchange with the Adriatic Sea results in a relatively short detention time for pollutants in the lagoon, which used to be adequate to cleanse most of these waters. Since the 1980s the coastal waters of the Adriatic have also become enriched by nutrients, mainly from the Po River, resulting in the rapid deterioration of the water quality of both the lagoon and coastal waters. Massive algal blooms are now an annual occurrence. The tidal exchange between the lagoon and the sea may be further affected by the planned construction of tidal barriers to alleviate the problem of tidal flooding.

TABLE 1.9 Unit Loads of Pollutants from Diffuse Sources in kg/ha/yr from the Mestre-Porto Marghera Urban Catchment of Metropolitan Venice and from Agriculture

Source	Suspended Solids	Nitrogen	Phosphorus
Untreated dry weather wastewater flow ^a	995	939	62
Wet weather diffuse urban loads	1241	223	26
Average agricultural loads		44-66	4-9

Source: After Novotny, Miller, and Zheng (1989).

^aIncludes primarily industrial loads from Porto Marghera. The dry weather flow and a small portion of the wet weather flow receives biological treatment with nitrification and denitrification.

Abatement Program

The Pollution Abatement Program approved for the lagoon by the Italian government envisions broad measures to protect the lagoon ecosystem and to reverse past trends. For the agricultural sector, which is responsible for about 50% of the total nutrient load to the lagoon, the action plan proposes the following:

1. Optimization of fertilizer application rates to minimize losses and matching applications to the nutrient requirement by crops.
2. Optimization of the temporal distribution of the application of fertilizers.
3. Use of soil drainage to increase denitrification of the soil.
4. Limitation and optimization of irrigation to reduce nutrient losses in the irrigation (surface and subsurface) return flow.
5. Substitution of organic slow-release fertilizers.
6. Increasing the organic content of the soil.
7. Soil-conservation practices on erodible lands to reduce loads of phosphorus, soil adsorbed, and organic nitrogen.
8. Suitable choice of the set-aside lands and conversion to less polluting crops according to the guidelines of the European Community.

The Pollution Abatement Program is also using wetlands situated along the boundary of the lagoon for the abatement of diffuse pollution loads. Point source abatement is focused on both dry weather and wet weather (diffuse) point loads. Finally, sewers will be installed in the historic city.

Lake Balaton

Evolution of the Water Quality Problem

Lake Balaton is located in the southwestern part of Hungary. It is the largest freshwater body in central Europe. On summer weekends up to one million tourists come to the shores to enjoy recreation. The lake was reasonably clean until 1965, since when increased agricultural use of chemical fertilizers in the drainage watershed, tourist use of the lake, which also increased pollution loads and overloaded existing sewage disposal systems, and industrial effluents increased the nutrient loads of the lake by an order of magnitude. As a result, algal biomass levels have increased dramatically and the lake has become eutrophic (Somlyódy and van Straten, 1986).

The progression of the intensification of land and lake use along with the phytoplankton biomass is shown on Figure 1.16. There were also

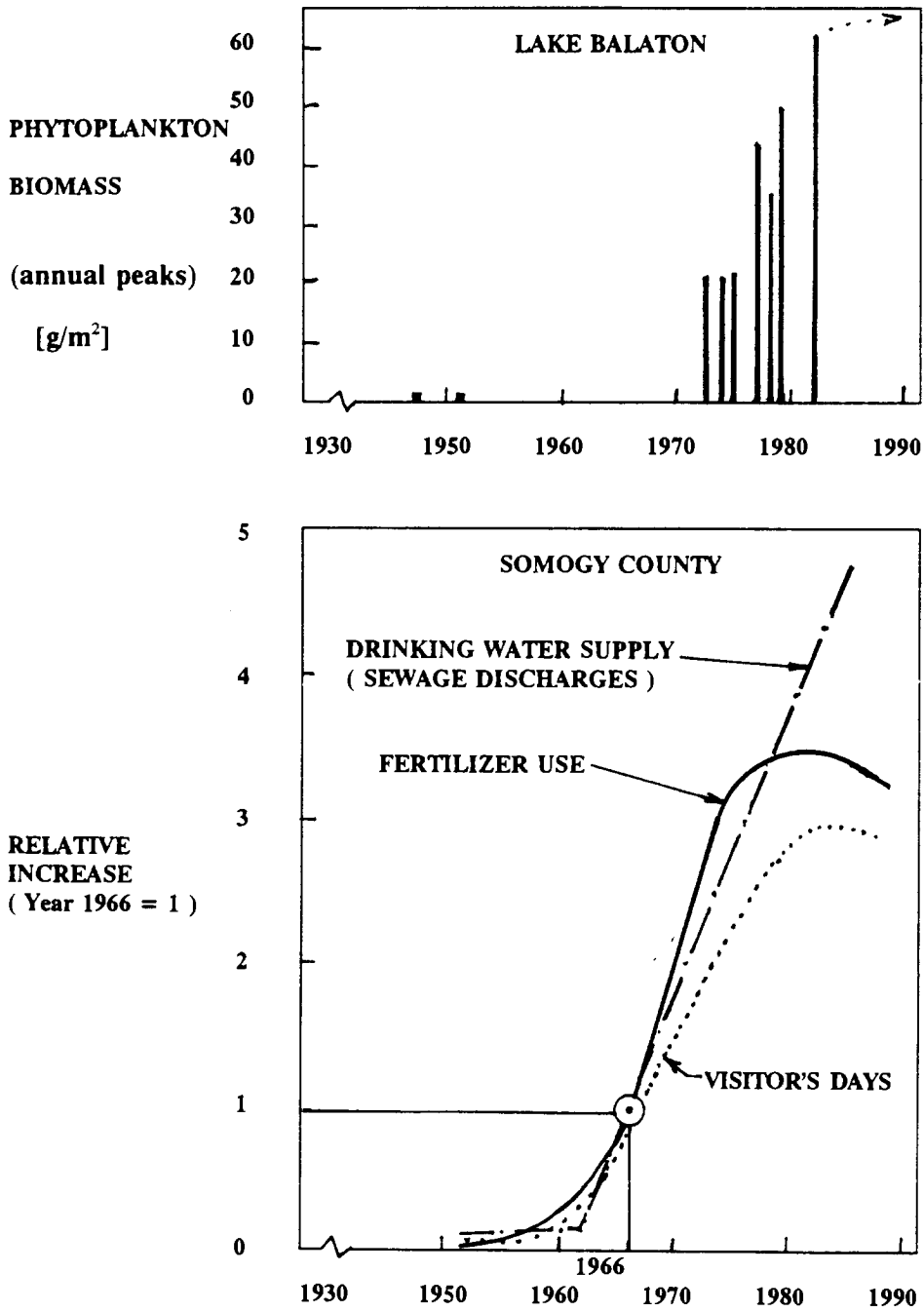


FIGURE 1.16. Nitrates, recreation use, and eutrophication of Lake Balaton, Hungary. (Data from Somlyódy and van Straten, 1986.)

several major fish kills that received wide public attention. The first occurred in 1965 and was most likely the result of fish poisoning by pesticides. The second, which happened in 1975, was indicative of a collapsing ecosystem (Hock and Somlyódy, 1990). This occurrence was

followed by mass proliferation of cyanobacteria in the summer of 1982, and the waters of the lake became mostly unsuitable for contact recreation.

The Action Plan

Following these catastrophic water quality events the authorities in early 1980s recognized the problem, and in January 1982, the Hungarian government decided to launch a series of extensive programs to analyze the problem and then suggest remedial measures. The remedial plan was adopted in 1983.

The goal of the plan is restoration of water quality to the levels that prevailed in the 1960s. This goal is to be achieved in stages, with target A corresponding to the water quality level of late 1970s, target B corresponding to water quality of late 1960s, and finally target C is restoration to early 1960s levels. The plan schedule is 1990 for target A, 1995–2000 for target B, and 2005–2010 for target C.

With the cooperation and sponsorship of the Food and Agriculture Organization (FAO) of the United Nations, Hungarian scientists embarked on an ambitious research project to identify the sources of nonpoint pollution and its management (FAO, 1986). A pilot experimental watershed was established on a small tributary of Lake Balaton. By establishing a data base on pollutant loads from various diffuse and traditional point sources, lake and watershed modeling, and extrapolation, scientists from the Water Resources Management Institute (VITUKI) and other institutes were able to present the authorities with targets, criteria, and possible abatement scenarios.

The first stage of the plan (target A) involves abatement of point sources, such as upgrading of biological treatment plants along the shore of the lake and installing tertiary phosphorus removals, and nonpoint source controls, including construction of sedimentation basins on several tributaries to reduce loads of particulate pollutants, dredging of polluted sediments, control of livestock wastes, improved farming methods, and a complete halt to building of summer homes along the shore.

The Kis–Balaton reservoir network was put in operation by 1990. The reservoir system is designed to filter out the nutrients and pollutants flowing into the western end of the lake from the Zala River. The pollutants are retained in the tributary reservoirs and do not reach the lake. Concurrently, phosphorus precipitation was introduced in 10 regional treatment plants. By 1990, the phosphorus input in the lake was cut in half, resulting in improved water quality in the eastern part of the lake. The rest of the lake, because of internal nutrients storage, will need a longer recovery time for a noticeable improvement.

The Emscher River

The Emscher River is a small river located mostly within the municipal boundaries of Essen in the Ruhr area of Germany. It is a historical river because the first watershed-wide water quality management agency in the world was established there in 1906. Karl Imhoff, a famous pioneer of modern environmental engineering, was then put in charge of the agency.

The drainage area of the Emscher River is 865 km², of which 20% is currently impervious. The present resident population within the Emscher River watershed boundaries is 2.5 million. Most of the pollution entering the river is of urban origin, with only minimal agricultural contributions.

The Emscher River Association (Emschergenossenschaft) is one of several river associations that have been subsequently established in this highly industrialized area of Germany. In order to provide sanitation and a safe water supply by relatively few small rivers for about 8 million inhabitants and about one-third of the German heavy industry, Karl Imhoff and his coworkers devised a plan of primary water uses for each individual river. In this plan, the largest river, the Ruhr, became the primary source of water supply and recreation for the population, while the Emscher River assumed the sad task of conveying mostly untreated or only partially treated sewage and industrial wastewater.

In the late 1800s and early 1900s the Emscher River received mostly untreated sewage from the Essen industrial area. The river was mostly septic with thick sludge deposits on the bottom. In the early 1900s the goal of the new association was to keep the river fresh and avoid further deposits of sewage sludge. However, land subsidence of up to 20 meters due to deep coal mining made drainage by conventional enclosed sewers extremely difficult and costly. Hence, in the first period of the sanitation work in the Emscher River watershed, wastewater effluents received primary treatment (mostly in so-called Emscher or Imhoff primary settling tanks), after which the effluent was discharged into open, concrete-lined channels, and hence into Emscher River. Such open channels allowed for easier correction of the slopes that were disrupted by the subsidence.

To reduce public hazards and increase the aesthetics of these open sewers the channels were fully lined to provide higher velocities that would keep the flow fresh (aerobic) and control odors, the river banks were lined with trees, and the channels were fenced on both sides to prevent access (Fig. 1.17). A treatment plant to treat the entire river flow was built before the confluence with the Rhine River. The river thus became an example of the ultimate conversion of a natural small stream into an open sewer (Anon., 1986).

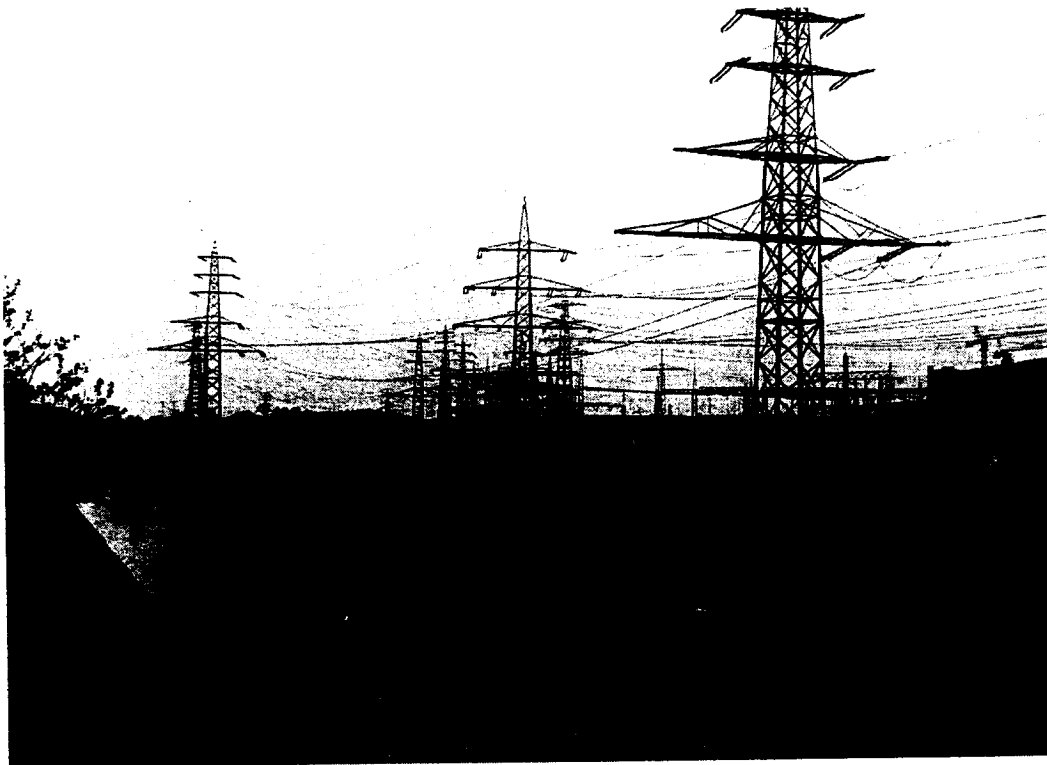


FIGURE 1.17. The Emscher River in Essen, Germany. (Photo: V. Novotny.)

Today, combined sewers replaced most of the open channels; however, until 1990 the river itself was still primarily unchanged. Hence, all of the pollution entering the river has been from combined urban and industrial sewer connections, and of a diffuse nature.

A Plan to "Renaturalize" the Emscher River and Drastically Reduce Pollutant Loads

A new plan for returning the river to a more naturally looking urban stream was prepared by the Emscher Association (Geiger, 1990). In 1990 the annual nutrient inputs in the river were estimated as 13,500 tonnes of nitrogen and 3600 tonnes of phosphorus, respectively. The treatment plant at the mouth of the river reduces these nutrient loadings from the Emscher and Rhine rivers (and the North Sea, which has a severe eutrophication problem caused by nutrient loads, primarily from the Rhine and Elbe Rivers) by about 15%. To reduce the pollutant loads to the river the association proposed an ambitious plan that considered installation of 200 storm-water-detention basins with a total volume of 1.3 mil m³ combined with limited reuse and infiltration of storm water. The individual pretreatment of wastewater discharges should meet the

effluent standards of 15 mg/l for nitrogen and 0.5 mg/l for phosphorus, respectively.

To further reduce flow and pollutant loads to the river, partial infiltration, separation, and reuse of storm water was considered. It was also suggested that storm-water discharges should be diverted to about 500 meters from tributary creeks, which would also enhance infiltration. Through modeling it was found that such measures could reduce peak storm-water flows and volumes by up to 85%. With these measures the nitrogen load from the watershed could be reduced by about 60%, and that of phosphorus by 92%. Along with the flow and pollutant reduction measures the river channel would be partially converted to a more naturally looking stream.

The Experimental Watersheds of the Shirako and Shakiji Rivers

Until 1964 (the year when the Olympic games were held in Japan) most of the watersheds of the Shirako and Shakiji Rivers located in the metropolitan Tokyo area were not sewered. Since 1965 sewerage projects



FIGURE 1.18. Shirako River in Tokyo. (Photo provided by S. Fujita.)

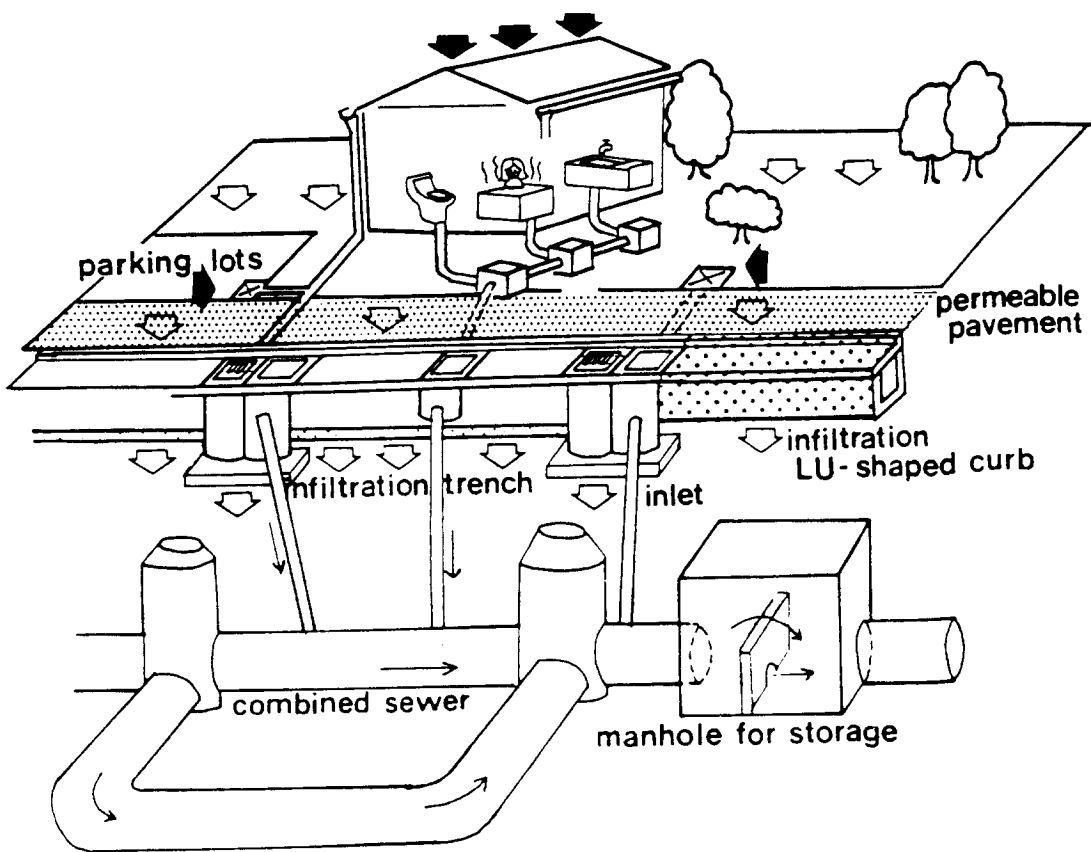


FIGURE 1.19. The ESS concept. (Replotted from Fujita, 1984.)

in Japan have accelerated, with the result that the two watersheds became fully urbanized by the mid-1980s. Consequently, the river channels, limited by urbanization, became insufficient to handle flows (Fig. 1.18). Flooding has thus become a major problem.

In 1982 the Tokyo metropolitan government selected the two rivers as experimental watersheds, in which various infiltration, storage, and other flood-control measures were to be installed and tested. The objective of the measures implemented within the experimental watersheds was to minimize flooding; the pollution control at the beginning of the project was secondary. Unlike U.S. and European practices, which usually do not infringe on private properties and all stormwater control is carried out outside of private homes, the Japanese plan included on-site household water infiltration and heavily relied on it (Fujita, 1984; Fujita and Koyama, 1990). Figure 1.19 shows the concept of the experimental sewer system (ESS). The extent of the ESS system as of 1990 is given in Table 1.10.

The ESS system relies heavily on site infiltration and to a lesser degree

TABLE 1.10 Characteristics and Extent of the ESS System

Parameter/Abatement	Shakiji River	Shirako River	Total
Watershed area (ha)	305	597	702
Resident population	39,600	72,100	111,700
Infiltration inlets	4,778	15,218	19,996
infiltration trenches (km)	43	84	127
Infiltration curbs (km)	22	49	71
Permeable pavements (1000 m ²)	200	408	608
Construction cost bil. Jap. ¥ (mil. U.S. \$)	12.5 (83)	24.3 (162)	36.8 (245)

Source: Fujita and Koyama (1990).

TABLE 1.11 Comparison of Pollutant Loads from Tokyo Urban Watersheds with Combined Sewers with and without the ESS System for an Average Year

Constituent	Without ESS	With ESS	Percent Reduction
Total number of rainfalls	71	71	
Overflow frequency	36	7	81
Overflow loads (kg/ha)			
Suspended solids	223	19	91
BOD	103	4.7	95
Total annual load ^a (kg/ha)			
Suspended solids	1057	589	45
BOD	788	528	33

Source: Fujita and Koyama (1990).

^aIncludes treatment plant effluent with dry-weather flow.

on storage. Again this is in contrast to past sewerage practices that were designed to convey storm water as fast as possible from the site to the watershed outlet. The goal of the ESS system is to retain and infiltrate and minimize conveyance as much as possible. Although the sewer system is combined, wastewater and storm water enter the system separately.

Implementation of a system such as the ESS, which relies heavily on site measures and storm-water disposal, requires the cooperation of homeowners. An extensive educational effort was part of the program.

Water Quality Benefits of the ESS System

As stated the water quality improvement was not the primary objective of the program. However, the ESS system is an example of the dual benefits

of urban storm-water management practices. The pollution loads and the frequency of overflows from the ESS systems, both of which were measured and simulated by a model, were compared to a similar watershed served by combined sewers without on-site storm-water management (Fujita and Koyama, 1990). The results of the comparison are given in Table 1.11.

References

- Alberts, E. E., G. E. Schuman, and R. E. Burnwell (1978). Seasonal runoff losses of nitrogen and phosphorus from Missouri Valley loess watersheds, *J. Environ. Qual.* 7:203–208.
- Anonymous (1986). Karl Imhoff, ein Wegbereiter der Stadtentwässerung und der Gewässerreinigung (Karl Imhoff, pioneer of urban drainage and wastewater treatment), *Herausgeber VDI*, Gesellschaft Bautechnik im Verein Deutscher Ingenieure, Düsseldorf, Germany.
- Bendoricchio, G. (1988). The NPSP abatement program for the Lagoon of Venice, in *Proceedings, Nonpoint Pollution 1988—Policy, Economy, Management and Appropriate Technology*, V. Novotny, ed., American Water Resources Association, Bethesda, MD, pp. 241–260.
- Beaulac, M. N., and K. H. Reckhow (1982). An examination of land use—nutrient export relationship, *Water Resour. Bull.* 18(6):1013–1024.
- Brown, L. R., et al. (1990). *State of the World, A Worldwatch Institute Report on Progress toward a Sustainable Society*, Norton, New York.
- Carson, R. (1962). *Silent Spring*, Houghton Mifflin, Boston.
- Clark, E. E., II, J. A. Haverkamp, and W. Chapman (1985). *Eroding Soils, The Off-Farm Impact*, The Conservation Foundation, Washington, DC.
- Committee on the Assessment of PCBs in the Environment (1979). *Polychlorinated Biphenyls*, National Academy of Sciences, Washington, DC.
- Czechoslovak Academy of Science (1992). *National Report of the Czech and Slovak Republic*. United Nations Conference on Environment and Development.
- Ellis, J. B. (1986). Pollutional aspects of urban runoff, in *Urban Runoff Pollution*, H. C. Torno, J. Marsalek, and M. Desbordes, eds., Springer Verlag, Berlin, New York, pp. 1–36.
- Garbell, M. A. (1965). The Jordan Walley plan, *Sci. Amer.* 221(3):23–31.
- Fed. Regist. (1979). 44:76202–76209.
- Field, R., and R. Turkeltaub (1981). Urban runoff receiving water impact: Program overview, *J. Environ. Eng. Div. ASCE* 107(EE1):83–100.
- Food and Agricultural Organization (1986). *Prevention of Non-point Source Pollution—Hungary*, United Nations, FAO, Rome.
- Fujita, S. (1984). Experimental Sewer System for reduction of urban storm runoff, in *Proceedings, Third International Conference on Urban Storm Drainage*, International Association on Water Pollution Control and Research (IAWPRC)-International Association on Hydrological Research (IAHR), Chalmers University of Technology, Göteborg, Sweden, pp. 1211–1220.

- Fujita, S., and T. Koyama (1990). Pollution abatement and the Experimental Sewer System. in *Proceedings, Fifth International Conference on Urban Storm Drainage*, International Association on Water Pollution Control and Research (IAWPRC)-International Association on Hydrological Research (IAHR), Suita-Osaka, University of Osaka, Japan, pp. 799–904.
- Gaffney, M. (1988). Nonpoint pollution: An economical solution to an unconventional problem, in *Political, Institutional and Fiscal Alternatives for Nonpoint Pollution Abatement Programs*, V. Novotny, ed., Marquette University Press, Milwaukee.
- Gayán, S. L., and J. D'Antuono (1988). The Milwaukee River East–West Branch Watershed Resource Management, in *Proceedings, Nonpoint Pollution 1988—Policy, Economy, Management and Appropriate Technology*, V. Novotny, ed., American Water Resources Research Association, Bethesda, MD.
- Geiger, W. F. (1990). New drain—New dimensions in urban drainage, in *Proceedings, Fifth International Conference on Urban Storm Drainage*, International Association on Water Pollution Control and Research (IAWPRC)-International Association on Hydrological Research (IAHR), Suita-Osaka, University of Osaka, Japan, pp. 33–48.
- Henry, J. G., and G. W. Heinke (1989). *Environmental Science and Engineering*, Prentice Hall, Englewood Cliffs, NJ.
- Hock, B., and L. Somlyódy (1990). *Freshwater Resources and Water Quality*, VITUKI, Budapest.
- Humenik, F. J., M. D. Smollen, and S. A. Dressing (1987). Pollution from nonpoint sources, *Environ. Sci. Technol.* **21**(8):737–742.
- Imhoff, K., and K. R. Imhoff (1990). *Taschenbuch der Stadtentwässerung*, Oldenbourg Verlag, Munich.
- International Joint Commission (1980). *Pollution in the Great Lakes Basin from Land Use Activities*, International Joint Commission, Windsor, Ontario.
- Konrad, J. (1985). The Wisconsin nonpoint source program, in *Proceedings, National Conference on Perspectives on Nonpoint Source Pollution*, EPA 440/5-85-001, U.S. Environmental Protection Agency, Washington, DC.
- Krejci, V. (1988). New strategies in urban and stormwater pollution control in Switzerland, in *Proceedings, Nonpoint Pollution: 1988—Policy, Economy, Management, and Appropriate Technology*, V. Novotny, ed. American Water Resources Association, Bethesda, MD, pp. 203–212.
- Krenkel, P., and V. Novotny (1980). *Water Quality Management*, Academic Press, New York.
- Lager, J. A., and W. G. Smith (1974). *Urban Stormwater Management and Technology: An Assessment*, EPA-670/2-74-040, U.S. Environmental Protection Agency, Cincinnati, OH.
- Loehr, R. C. (1972). Agricultural runoff—characterization and control, *J. Sanitary Eng. Div. ASCE* **98**:909–923.
- Marsalek, J. (1978). *Pollution due to Urban Runoff: Unit Loads and Abatement Measures*, International Joint Commission, Windsor, Ontario.
- Marsalek, J. (1986). Toxic contaminants in urban runoff: A case study, in *Urban*

- Runoff Pollution*, H. C. Torno, J. Marsalek, and M. Desbordes, eds., Springer Verlag, Berlin, New York, pp. 38–55.
- Mitsch, W. J., and J. G. Gosselink (1986). *Wetlands*, Van Nostrand Reinhold, New York.
- Moore, E., et al., eds. (1979). *Livestock Grazing Management and Water Quality Protection*, EPA-910/9-79-67, National Technical Information Service, Springfield, VA.
- Myers, C., J. Meek, S. Tuller, and A. Weinberg (1985). Nonpoint sources of water pollution, *J. Soil Water Conserv.* **40**(1):14–18.
- Nisbet, C. T., and A. F. Sarofim (1972). Rates and routes of transport of PCBs in the environment, *Environ. Health Perspect.* **1**:21–38.
- Nonpoint Source Control Task Force (1983). *Nonpoint Pollution Abatement in the Great Lakes Basin. An Overview of Post-PLUARG Developments*, International Joint Commission, Windsor, Ontario.
- Novotny, V. (1985). Urbanization and nonpoint pollution, in *Proceedings, Fifth World Congress on Water Resources*, Paper No. 17, Aspect 6, International Water Resources Association. Brussels.
- Novotny, V., and G. Chesters (1981). *Handbook of Nonpoint Pollution: Sources and Management*, Van Nostrand Reinhold, New York.
- Novotny, V., and G. Chesters (1989). Delivery of sediment and pollutants from nonpoint sources: A water quality perspective, *J. Soil Water Conserv.* **44**(6):568–576.
- Novotny, V., B. Miller, and S. Zheng (1989). *Urban Diffuse Pollution of Lagoon of Venice*, Technical Report to Venezia Nuova, Department of Civil Engineering, Marquette University, Milwaukee.
- Novotny, V., K. R. Imhoff, M. Olthof, and P. A. Krenkel (1989). *Karl Imhoff's Handbook of Urban Drainage and Wastewater Disposal*. Wiley, New York.
- Organization for Economic Co-operation and Development (1991). *The State of the Environment*, OECD, Paris.
- Pitt, R., and R. Field (1977). Water quality effects from urban runoff, *J. Am. Waterworks Assoc.* **69**:432–436.
- Robbins, J. W. D. (1985). Best management practices for animal production, *Proceedings, Nonpoint Pollution Abatement Symposium*, V. Novotny, ed., Marquette University, Milwaukee, pp. P-III-C 1–11.
- Rooney, P. (1989). Louisiana's wetland calamity, *EPA J.* **15**(5):37–39.
- Schroepfer, G. J. (1978). A philosophy of water pollution control—past and present, *J. WPCF* **50**(9):2104–2109.
- Somlyódy, L., and G. van Straten, eds. (1986). *Modeling and Managing Shallow Lake Eutrophication, with Application to Lake Balaton*, Springer Verlag, Berlin.
- Thompson, P. (1989). *Poison Runoff. A Guide to State and Local Control of Nonpoint Source Water Pollution*, Natural Resources Defense Council, New York.
- Timmons, D. R., and R. F. Holt (1977). Nutrient losses in surface runoff from a native prairie, *J. Environ. Qual.* **6**:369–373.
- U.S. Congress (1987). *Water Quality Act of 1987*, P.L. 100-4, Washington, DC.

- U.S. Environmental Protection Agency (1983). *Chesapeake Bay: A Framework for Action*, U.S. EPA, Region III, Philadelphia.
- U.S. Environmental Protection Agency (1984). *Report to Congress: Nonpoint Source Pollution in the U.S.*, Water Planning Division, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1988). *Chesapeake Bay Nonpoint Source Program*, U.S. EPA Chesapeake Bay Liaison Office, Annapolis, MD.
- U.S. Environmental Protection Agency (1989). *Nonpoint Sources: Agenda for the Future*, Office of Water, U.S. EPA, Washington, DC.
- Vesilind, P. A. (1975). *Environmental Pollution and Control*, Ann Arbor Science, Ann Arbor, MI.
- Waller, D. H., and W. C. Hart (1986). Solids, nutrients and chlorides in urban runoff, in *Urban Runoff Pollution*, H. C. Torno, J. Marsalek, and M. Desbordes, eds., Springer Verlag, Berlin, New York, pp. 59–85.

2

Laws, Regulation, and Policies Affecting Water-Pollution Abatement

Every person learns early in life that "self-preservation" is the first law of nature.

Henry P. Caufield, Jr. (1991)

Often, engineers and scientists dealing with pollution abatement may not realize that they are working and acting within a social and legal system that is relatively complex and that imposes legal constraints on what can and cannot be done to resolve or limit the problem of pollution. These legal problems and social issues have been partially addressed in the previous chapter. The laws and legal rules that affect or may affect discharges of pollution and their abatement and control are numerous. In order to comprehend the complexity and ramifications of the legal rules one should become familiar with U.S. and international legal systems and with policies that rule pollution-abatement programs. Pollution abatement is both a technological and political-economical-legal problem. However, it has become clear that managing water resources and maintaining and achieving acceptable water quality is far less a technological issue than a political, institutional, and economical one. Technical and scientific solutions of the problem and the economical feasibility of various alternatives alone will not automatically lead to implementation of pollution-abatement programs. Although technical solutions to the problem of diffuse pollution have been researched and are available, implementing diffuse-pollution programs is not easy, even if financial resources are available (Novotny, 1988a, 1988b). This book

addresses both technological and nontechnological means of environmental protection and abatement.

The planners, engineers, and scientists involved in pollution abatement must also deal with the legal profession, since environmental law is now a well-established branch of our legal system. Special courses on environmental law are taught in law schools and many law firms now specialize in handling environmental cases. Several international treaties include transboundary environmental issues, including pollution.

SOCIAL CAUSES OF POLLUTION

Living and production to sustain living both produce waste. According to the definition of pollution in the previous chapter generation of pollution results from the use of resources in a fashion that is detrimental to the environment or to the beneficial uses of environmental resources. However, man has three distinct roles in this process. On one side, man is a *producer* (developer) and *consumer*, hence the polluter. On the other side, man and ecology are adversely impacted by pollution, hence man is a *sufferer* of pollution. The conventional explanation for pollution by economists is that it is the least expensive way for consumers and producers to get rid of waste products (Braden, 1988). Excess pollution arises when the waste-disposal capacity of the environment is provided free of charge (Solow, 1971) and/or when the consumers-producers do not incorporate into their economic considerations the cost of the damage caused by pollution. (Recall from Chapter 1 that the definition of pollution implies damage and subsequent cost to the society.)

The basic driving force for production and consumption is sustainment of living processes first and production profit second. In the societies of the Middle Ages and ancient times the producers, consumers, and sufferers of pollution were small groups of people living in a relatively small confined area. Most likely they were unable to recognize the link between pollution and disease, but certainly they could smell and had some sense of aesthetics, therefore, they dumped their waste and that of their domestic animals at some distance from their living areas. The relationship between the living and economical process and pollution unaffected by regulations and pollution-control laws can be illustrated by the following realistic example.

In the historic walled city of Fez in Morocco (one of the world's historical treasures, with about 200,000 inhabitants) time has stopped and people live there in the same way as they did 500 years ago (Fig. 2.1). People living in the city are brought up and educated in the Moslem religion. The oldest Islamic university was founded in the city in the

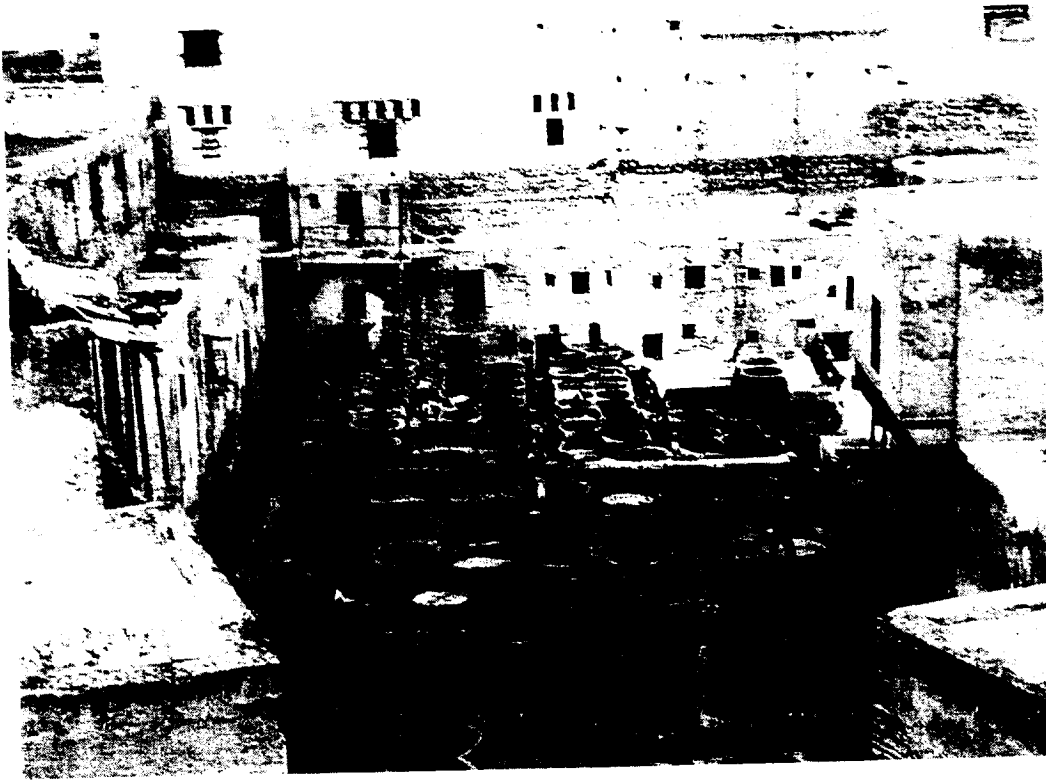


FIGURE 2.1. Historic Fez in Morocco. Famous outdoor tanneries.
(Photo: V. Novotny.)

eighth century, long before the first European university was established in Bologna, Italy (founded in the thirteenth century). The streets of Fez are reasonably clean because homeowners and merchants clean the narrow streets and alleys around their premises. Hence, there is no major urban runoff pollution problem, even though instead of automobiles (which cannot enter the historical city) donkeys and mules are used for transporting goods and people. The simple reason is that the sufferer of pollution is also its cause and he/she may realize the linkage between living and economic activities and the damage done by consequent pollution (for example, no one would come to an artisan shop if the area around it was unclean and smelly). In this way the cost of the damage by pollution is incorporated in the citizens' economic reasoning.

This kind of more or less voluntary participation in cleanup efforts is successful if everybody participates. Let us consider a situation in which several citizens of the city do not participate and let their garbage, refuse, and dirt accumulate. By doing so they cause harm to their neighbors by bad smells, disease, and rats, thus keeping away customers for their products. In the absence of regulations and laws the sufferers have no

legal recourse. This is the problem of economic externalities, which will be explained throughout this book. However, those who want to keep their neighborhood clean can do the following: (1) express their displeasure and try to persuade the polluter not to pollute; (2) put moral and economic pressure on the polluters (for example, boycott their merchandise); (3) go to court; or (4) set up an enforcement scheme by asking their legislative body to pass a law preventing this kind of pollution-generating activity.

In order to set up enforcement they first have to state that pollution is a problem and has to be limited. Hence, they have to formulate a pollution-control policy. In the policy statement they have to say what is acceptable and what is not. Hence, they have to define pollution and pollution-causing activities. They may even say how much pollution is tolerable. Subsequently, they must formulate the penalties for violations and a mechanism for collecting and enforcing the penalties. These are typically the simplest components of a policy to control pollution. Such systems of self-control are feasible only if the sufferers of pollution are a majority in a group of producers and consumers—the polluters and their own pollution directly affects them.

In the same city, however, there is also a small stream that transects it. The stream receives all pollution from households, commerce, and the famous outdoor tanneries, as well as urban runoff. In a section between the city walls about one kilometer long the water quality of the stream changes from marginally good to an awful open sewer, leaving the stream heavily polluted and devoid of oxygen for many kilometers downstream from the city. However, a great majority of the populace of Fez rarely leaves the confines of the walls of the city. Hence, they do not realize the damage done to the stream, nor do they experience any of the economic damage caused by the heavy pollution of the stream. The damage by pollution has been done to downstream farmers who cannot use the stream water for irrigation or drinking, as the water is unfit for any use. The stream is smelly and unsightly, but there is no linkage between the damage done to downstream users and economic production and consumption in Fez where the pollution originates. Consequently, there is no abatement. Again the externality character of pollution prevents abatement if no enforceable regulations or law to control pollution are in place.

One does not have to go to north Africa to see the same effect. In the United States the water quality of Chesapeake Bay has been deteriorating rapidly, causing great economic harm to commercial fishing and the recreation industry. The causes of pollution are farming, point, and nonpoint urban sources throughout the watershed. Yet again there is no

economic linkage nor mechanism by which the sufferers of pollution (fishers and recreationalists) can recover the damage from polluters. A farmer whose motives are economical (profit making) will use man-made chemicals to increase yields and pesticides to control weeds and insects. Consequently, nutrient and pesticide losses degrade both downstream water quality and ground-water resources. Yet those who are impacted by this pollution cannot recover the cost of damage from the farmer, and the farmer himself may not be economically impacted by the pollution. As pointed out in the preceding paragraph, this situation is creating a cost to someone else—including the damage to the resource, cost of resources forgone, and remedial costs—and the costs are transferred by physical means and not by market transaction. This is called *an externality*. The externality problem, especially in diffuse-pollution generation, is pervasive and general, and must be resolved before any meaningful plan of abatement is put in place (Novotny, 1988a, 1988b).

A prominent political economist (Solow, 1971) defined externality as follows: “One person’s use of a natural resource can inflict damage on other people who have no way of securing compensation, and who may not even know that they are being damaged.” Overcoming the externality problem and incorporating the cost of damage caused by pollution in the economic thinking of producers and consumers are the major objectives of pollution-abatement policies. Again quoting Solow: “We would like to insure that each resource is allocated to that use in which its net social value is highest,” which is called by political economists *Pareto optimality*.

Ignoring externality may have serious consequences as it did in the countries of Eastern Europe where these economic principles were disregarded. (To a lesser degree many developing and less developed countries ignore them as well.) The implications of external effects must be traced further than just considering them a cause of pollution. They have secondary effects on production and resource allocation. If, for example, the cost of acid rainfall damage to lakes and soils (forests) is not included in the economic thinking of the producers of electricity, electric power is then “too cheap” to consumers, because they are not charged for the damage. Other commodities that are produced with the help of cheap electricity will also be cheaper, and they will be overproduced (Solow, 1971). This will then result in greater consumption and more damage. In this way, society will subsidize those using a lot of electricity. Similarly, it is known that automobiles cause significant pollution, especially of a toxic and acid nature (see Chapter 8 for details). If the cost of damage by the emitted pollutants was not included in the cost of driving an automobile, then society would subsidize automobile users,

resulting in more automobile use, more urban sprawl, and consequently more urban erosion from building highways and urbanization. Very often the cost is hidden. Using again East Europe as an example, “free” health care and sending children from heavily polluted zones to sanatoria and health care resorts in Czechoslovakia was a substitution for “costly” pollution abatement during the period of the socialist totalitarian regime. Overall, the cost of the damage to the health of the populace and the destruction of the ecology was much greater than the investment in pollution abatement and the curtailment of subsidized but grossly polluting production processes.

ENVIRONMENTAL POLICIES OF DIFFUSE POLLUTION ABATEMENT

In the environmental policy arena there is a difference between the management tool and policy. A *tool* is a single element—effluent standard, enforcement technique, zoning restriction—devised to achieve a specific result or, in a few cases, several results. A *policy* is a set of one or more tools chosen to achieve an overall environmental objective. A management tool can be thought of an action that is taken in the hope of achieving a particular result. Taking the action involves *adverse effects* (cost of implementation, cost imposed on various participating parties), but the expected results will produce *beneficial effects* (improved environmental quality, lower cleanup cost, increased benefits to users) (Boland, 1991).

From the reasons stated in the preceding section, policies for use of resources, waste disposal, and protection of the environment represent a compromise between the producers in the economic production process and consumers on one side and those who suffer from the adverse effects of production and development and waste disposal on the other. This compromise is reached on several institutional levels. All three branches of government (legislative, judicial, and executive) are involved, as are many pressure and lobbying groups. Implementation of successful and efficient diffuse-pollution control programs requires that competing groups (farmers, urban dwellers, urban polluters, developers, industries) cooperate. Pollution abatement often involves conflicts with powerful interests, notably the farm lobby, chemical manufacturers and their lobbying associations, and developers.

Various policy options can be used for enactment and implementation of pollution control and water quality protection and restoration. As shown below, these options range from those that are simple but

ineffective to those that are potentially effective (Carson, 1980; Novotny and Joeres, 1983; Novotny 1988a, 1988b):

- No action
- Moral persuasion and public pressure
- Court litigation:
 - damage payments
 - court established standards
- Regulation (laws and enforcement required):
 - ban on harmful substances and chemicals
 - mandatory control processes and performance standards and permits
 - stream and effluent standards
- Economic incentives (laws and enforcement required):
 - taxes and charges
 - subsidies, tax write-offs, and payments for taking land out of production
 - marketable discharge permits
- Government contributions for research, education, rehabilitation, and preservation

However, the effectiveness of a solution depends not only on technological or legal methodology but also on political conditions and the institutional framework in which the abatement alternatives are implemented.

Most efficient policy alternatives require regulation and enforcement. One may ask why any regulation and enforcement is needed for control and abatement of (diffuse) pollution. The economic system prevalent in the United States and many other countries is based on capitalistic-democratic free enterprises—the market system where market forces determine how much production is needed to satisfy the need of society. The reader should distinguish between the political system of the government (democracy, dictatorship, feudal autocracy) and the economic system (capitalist-free market, socialist, feudal, communist). History and recent social changes in eastern Europe have proved that the capitalist-democratic system is more efficient than any other economic-political systems. Then why not let the same market forces determine the level of pollution and the level of abatement. For one thing, political economists have found, demonstrated, and documented that market forces do not control pollution and do not stimulate pollution abatement (Kneese and Bower, 1968; Bator, 1958; Solow, 1971; Braden, 1988; Baumol and Oates, 1988). Bator describes the market's inability to control pollution as the failure of a more or less idealized system of

price-market institutions to sustain "desirable" activities. Hence, to charge for the use of the environment for waste disposal and to collect damages may require regulation. The free market economy in industrialized countries without environmental regulation and enforcement will lead to a deteriorating environment and to environmental catastrophes as exemplified several decades ago in Japan (Minamata mercury poisoning) and currently in several less developed but rapidly industrializing countries throughout the world. On the other side of the spectrum are the authoritative political regimes, such as those in place until 1989 in Eastern Europe and the Soviet Union, where all activities were subject to some kind of regulation, yet the state of the environment was even more deplorable. The problem in these latter systems was the lack of market forces. Also, the value of environmental damage was not included in the economic thinking of policymakers, leading to the overuse of the environment and to the well-publicized environmental catastrophes primarily caused by diffuse pollution of immense proportions (for example, most soils and ground- and surface-water bodies in former East Germany are heavily contaminated by chemicals, acid rain is widespread throughout central and eastern Europe, irrigation and chemical uses in the Soviet Uzbekistan led to the disappearance of the Aral Sea, and excessive contamination of the entire region by dangerous chemicals, including DDT, which is now an ecological catastrophe).

The process of formulating an environmental policy compromise is neither gradual nor smooth. Throughout history there were a few landmark periods during which most important environmental regulations were enacted, followed by periods of partial regression. In the early 1970s, the first strong environmental legislation was enacted in the United States by a well-organized classic "iron triangle" of legislators, public agency officials, and environmentalists. These three groups are largely responsible for formulating environmental policies today. In Europe the presence of environmental political parties (the Greens) has had a profound impact on formulating environmental policies. These parties, although small in membership, often represent swing votes in the parliaments and cannot be ignored by the larger governing political parties.

Viessmann and Welty (1985) wrote that policymaking in the management of water resources and pollution control is an outcome of political forces operating in different political arenas. Conflict is an inherent element of these political processes, and it serves to ensure that a multiplicity of values is represented. Compromise is a partner to all policymaking, and the art of reaching compromise can be greatly

enhanced by appropriate technological input and the use of state-of-the-art analytical techniques. The formulation of environmental policies has been described by Caufield (1988, 1991). Environmental policy formulation and implementation is a process that is structured and occurs over time; it is societal in that it is a collective process in which many individual decisions play a role; and it is "authoritative" in the sense that outputs of the process are generally accepted and have the force of the law.

Caufield (1991) also described the process of bargaining and compromising among policymakers (legislators) that must take place if an environmental statute is to be enacted. In this process at least one interest group must be served by the proposed legislation, though more often the legislation serves a much wider segment of the population, if not the public as a whole. Most environmental legislation requires the active support of environmental interest groups. Legislators who support the views of the groups and seek to pass environmental legislation engage in vote trading.

Policies are built on tradition, scientific knowledge, and common sense. Some of this knowledge has been incorporated into certain generally accepted rules called *doctrines* or *imperatives*. These imperatives can be technological-physical (such as conservation of mass and/or energy), economic (a project will fail if the cost associated with the project is less than the benefit gains), or political. An example of political rule is the notion of private ownership of land—*Cujus est solem ejus ad collum at ad inferno*, or "He who owns the soil owns it from the heavens to the inferno." Using this doctrine until recently, the courts interpreted all polluting activities on private lands as not being subject to restrictions, which had a profound effect on diffuse-pollution abatement. This decision precluded any effective enforcement of pollution abatement on private lands, which meant that most of the pollution efforts of the past twenty years had to rely on the voluntary participation of landowners.

Other doctrines have been incorporated into legal documents. For example, if pollution discharge is causing harm to a navigable body of water, it can be regulated and/or restricted by the federal government. This right is derived from the Commerce Clause of the U.S. Constitution. The sociopolitical imperatives of diffuse pollution control are introduced later in this chapter. The bulk of this book is devoted to physical-technological rules and imperatives. The last chapter is devoted to institutional imperatives and solutions.

Rogers and Rosenthal (1988) defined several *policy imperatives* for control of pollution. These imperatives are

Equity. No group of individuals in society should bear a disproportionate cost in meeting environmental quality requirements. The levels of environmental quality chosen should be such that no additional benefit can be derived without making one group or individual worse off (Pareto optimality).

Irreversible impact. No actions may be permitted that would irreversibly harm the environment and natural resources. This concern for irreversible environmental deterioration and the consumption of nonrenewable resources has to be guarded against by society at large (intergenerational impact). This imperative is also known and is now being implemented as a requirement for *sustainable economic development*.

Regulations and statutes. Due to the failure of the general market to control the quality of the environment and protect the resources, there must be legislation and regulation. The regulations must be clear and easy to carry out.

Acceptance. There must be concurrence on the part of the people and groups being regulated that they will, by and large, obey the regulations.

The equity imperative is especially crucial to diffuse-pollution abatement. For example, consider two identical farmers, one of whom is located near a watercourse, while the other is far from it. A physical rule that will be described in subsequent chapters states that pollution is attenuated as it travels overland from the source to the receiving body of water. Hence, relating the required pollution abatement to the damage of the receiving body of water would impose a cost on the farmer located near the body of water that the other will not have to bear. A more equitable solution, at least subjectively, would be to share the cost; however, this would be impossible without tax incentives and subsidies, which can only be imposed by regulation.

All pollution management tools will likely redistribute benefits and cost nonuniformly across the population. Benefits might be enjoyed by recreational fishers and boaters, while the cost is borne by the residents in polluting communities installing point source abatement, by farmers implementing soil conservation, and by industries treating their wastewater. The situation where some must bear the cost so that others can benefit is described as a *redistribution of income*. The equity criterion implies that all those affected should bear the cost and share the benefits more or less equally. However, there has been a long-standing policy imperative used in water resource development projects that is also applied to foreign aid to developing countries, stating that those who are

poor should receive proportionally more benefit than those who are more well off (Maas et al., 1970)

In most cases, however, the most equitable resolution of the conflict between the polluters and sufferers of pollution is—by regulation or voluntary participation—to make polluters pay for abatement. In economic terms this means that the price of goods and services should fully reflect both the cost of production and the cost of the resource used, including the use of the environment for waste disposal. In theory and practice, the polluter should pay the full cost of damages caused by his/her activity, which, in turn, will create an incentive for the reduction of polluting activities at least to the level where the cost of pollution reduction equals the cost of damage caused by pollution. This leads to another policy imperative:

Polluter Pays Principle (PPP). Polluters bear the primary responsibility for pollution and its abatement.

The member governments of the Organization for Economic Cooperation and Development (OECD, 1991) agreed to pursue environmental policies that would appropriately use economic instruments, alongside cost-effective regulatory instruments, which would be in accordance with the polluter pays principle. These economic instruments and types of payments for pollution and its abatement are explained in Chapter 16.

Pollution of the environment and adverse water quality are a result of many human activities. Generally, the sources of pollution are classified into point and nonpoint (see Chap. 1). In the past, water quality considerations were fragmented, and almost all financial resources were devoted to point source abatement. Sewage treatment, storm-water management, nonpoint source control, and point source programs were carried out separately from each other, despite the fact that these sources are interrelated and produce combined water quality deterioration. Also water quality has been generally evaluated as if air pollution, solid and hazardous waste disposal, and land- and water-use management and decisions were unrelated. The reverse is true. All these aspects of the pollution problem are interrelated and their combined effect must be considered. This leads to the next policy imperative:

Integrated Approach. The pollution problem must be resolved in an integrated manner, whereby the causes of pollution, all sources, and the combined environmental impact are considered, and the resulting combined solutions are therefore most equitable and efficient.

Single management tools (such as a treatment plant) may have a single medium result (improvement of water quality). Some solutions may produce results in more than one medium. Land-use restrictions, for example, can protect the quality of the air, as well as that of surface runoff and ground water. Phosphorus loads to the receiving water bodies originate from both point and nonpoint sources and to the biota of the receiving water body whose productivity is stimulated by the phosphorus input; it does not matter where the phosphorus load is coming from.

The OECD member governments will also pursue policies that will encourage the progressive adoption of anticipatory, rather than exclusively reactive, instruments of environmental policy. This means that more emphasis should be placed on *pollution prevention* in contrast to pollution abatement, which was prevalent throughout the 1970s and 1980s. Emphasis will be placed on not creating pollutants in the first place. Pollutants that must be created should be removed as close to the source as possible. Care must be taken to avoid shifting pollutants between media (air, water, soil, or underground formations) and to find the most appropriate (smallest risk) media into which unavoidable pollutants should be released. Therefore the last imperative is

Pollution Prevention. Environmental policies will promote economic development that will anticipate potential pollution problems and react to them by political and economic and technological means before they occur.

Criteria for Pollution Abatement Based on Equity and Irreversible Impact

As was pointed out in Chapter 1, two sets of criteria and standards are in force in the environmental area. The first set (the legal difference between standards and criteria was explained in Chap. 1) is designed to protect human health and the well being of fish and aquatic life. We can call them *ecological-toxicological* or *receiving water* standards and criteria, which will be introduced and discussed in Chapters 12 and 13 and in the Appendix.

The second group of standards is not directly related to water quality, though they generally apply to all sources within various source categories. Most of these standards are based on the equity imperative—that is, all polluters should reduce some part of their waste load regardless of whether or not harm is being done by the emissions to the receiving body of water. Other standards and criteria may be based on the avoidance of the irreversible harm to future generations by the

TABLE 2.1 Examples of Technology-based Standards

Land-use Type	Typical Standard	Policy Imperative
Urban runoff	Capture and treatment of first 1.2 to 2.5 cm of runoff ^a	Equity
Pasture	Mandatory street sweeping	Aesthetics
Agriculture	Cattle density limitations, fencing along the streams	Irreversible impact, Public health
	Soil erosion control	Irreversible impact, ^b equity
Construction sites	Soil erosion control	Equity, air pollution, irreversible impact
Suburban lands	Septic tank regulations	Public health, ground-water protection
	Pesticide and fertilizer sale regulations	Ground-water protection, ^b public health
Combined sewer overflows	Restriction on the number of overflows or mandatory capture and treatment of a certain portion of the wet-weather flow	Equity ^b
Surface mining	Land reclamation and restoration	Equity, irreversible impact
BOD and suspended solids standard for municipal point sources	Maximum effluent BOD ₅ and suspended solids limits of 30 mg/l as monthly averages	Equity

^aTypically it controls about 90% of the water and pollution load.

^bIn addition to water quality control benefits.

overuse of the resource, or the standards may be based on aesthetics, air pollution control, ground-water protection, or damages to infrastructures. Most of these standards require the mandatory application of certain technologies, hence, we call them *technology-based* standards. Examples of technology-based standards are given in Table 2.1. Such standards are also known as *effluent standards* and are based on the *best available technology economically achievable (BATEA)* used in point source abatement or on *performance standards* used for diffuse-pollution abatement.

Once issued, technology-based standards are easy to implement and monitor for compliance. Water quality standards, on the other hand, are difficult to enforce because there is rarely a direct and simple relationship between a pollution discharge and the water quality (pollution) of the receiving body of water. Water-quality-based approaches require models that may sometimes be inaccurate and unreliable. For example, nitrogen and phosphorus discharged from point and nonpoint sources cause

Appropriation Exclusive
 Both Doctrines Recognized

Riparian Exclusive

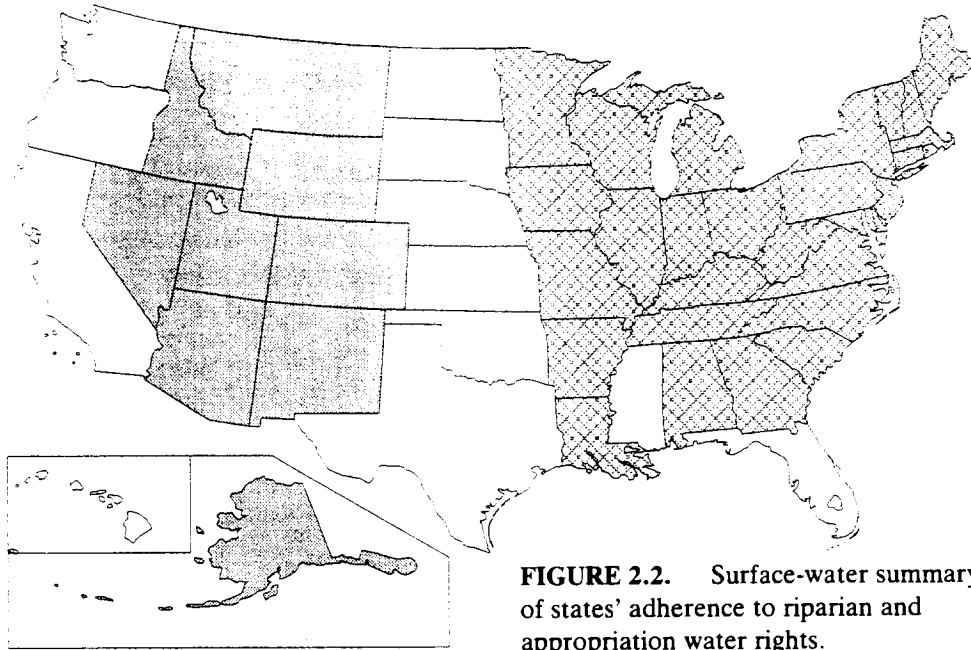


FIGURE 2.2. Surface-water summary of states' adherence to riparian and appropriation water rights.

Common Law Riparian

Correlative Rights

Appropriation

Reasonable Use (riparian)

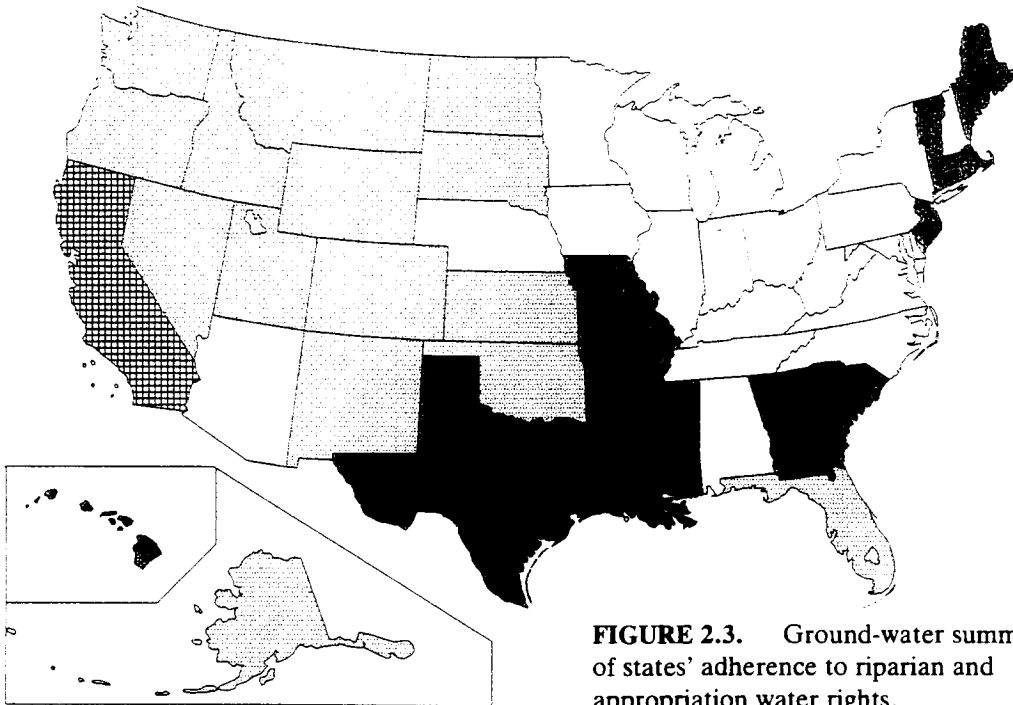


FIGURE 2.3. Ground-water summary of states' adherence to riparian and appropriation water rights.

accelerated eutrophication, which is manifested by algal blooms and other symptoms (see Chapter 12), but the relation between the nutrient levels in the receiving bodies of water and algal biomass is very complex and to some degree speculative. Even more uncertainty is involved in modeling the fate and effects of toxic chemicals (see Chapter 13).

WATER USES AND WATER RIGHTS

The primary objective of all pollution-control efforts is to protect and enhance the use of bodies of water for present and future generations. There is a general consensus that water is public goods and everybody should have an access to and the right to use them. However, in the United States the uses and ownership of water rights (who can use water and in what quantity and quality) differ among the states, because water laws are under the administration of the individual states and consist primarily of two doctrines known as *riparian* and *appropriative rights*, as shown on Figures 2.2 and 2.3. In general, two kinds of laws have developed because of the two major problems occurring with water, use and drainage. The majority of water-use laws have developed in the West, where critical water shortages occur, while the majority of drainage laws have been promulgated primarily by the "wet" eastern states (Krenkel and Novotny, 1980). These two completely different water rights doctrines can make some eastern water laws inoperational in the West and vice versa. For example, maintaining minimum flow for waste assimilation is impossible in the West, while flows are guaranteed in the East. On the other hand, interbasin transfer of water is legally very difficult to impossible in the East while permitted in the West.

Riparian Water Laws

Nearly all of the states east of the Mississippi River follow the riparian doctrine. Its key features are:

The owner of land adjacent to a stream is entitled to receive the full natural flow of the stream undiminished in quantity and unimpaired in quality. The riparian owner has a legal privilege to use the water at any time, subject only to the limitation that the use is reasonable. The right is a natural right that cannot be transferred, sold, or granted to another person as property. The legal body owning this right is then called a *riparian owner* and the property adjacent to the water body is then *riparian land*.

During a time of water shortage, all riparian owners have equal rights to the reasonable use of water and the supply is shared, although

domestic use may have preference over other uses. As pointed out, the riparian rights do not allow transfer of water from one basin to another (a notable exception is a small transfer of flow from the Great Lakes basin to the Illinois River via the Chicago River and connecting canals that was approved by the governors of all the Great Lakes states and by two Canadian provinces and after a succession of litigations was finally approved by the courts). The riparian landowners and users are protected from withdrawals or uses of water that unreasonably diminish its quality or quantity.

The private riparian ownership of water rights may conflict with the rule of water as public goods and with the right of the federal government stemming from the Commerce Clause of the U.S. Constitution. Imagine a situation (that could be very common in the East) where one owner or a group of private owners acquires all riparian lands surrounding a lake or any other body of water. This would preclude public access and public use of that water. The courts, however, have allowed the states to become riparian owners, even by expropriation, of a small piece of land (for example, for a public boat landing) and have upheld the federal government's rights over all navigable waters. Hence, through a public agency becoming a riparian owner of the body of water the public interests are protected and the use of the body is in the public domain.

Use of so-called *riparian buffer zones* and *riparian wetlands* for the abatement of diffuse pollution (see Chapters 10, 11, and 14) necessitates either the acquisition of water rights of a strip of riparian lands along the affected body of water or, by regulation or persuasion, the imposition on the riparian landowners of the abatement in a strip of land adjoining the body of water.

Appropriation Water Laws

The system of water laws adopted by most western states is known as the law of appropriation, which is best stated as: "First in time is first in right." The basic tenets of the system are:

- (1) A water right can be acquired only by the acquiring party diverting the water from the water course and applying it to a beneficial use, and (2) in accordance with the date of acquisition, an earlier acquired water right shall have priority over later acquired water rights. Water in excess of that needed to satisfy existing uses is viewed as unappropriated water, available for appropriation and application of the water to a beneficial use.

The process of appropriation can continue until all the water from the stream is subject to rights of use through withdrawals from the stream. In

times of shortages, the earliest claimants take full share, and others may do without water. If the right is not used, it is lost. In addition, the right is not identified by ownership of riparian land.

There are several problems related to the application of the appropriation doctrine in western states. First, there is no natural flow notion (Beuscher, 1967). The appropriators can take as much water as they are entitled to, even though it exhausts the flow. This leads to situations where the entire river flow may be withdrawn by irrigators and other users, which means that the downstream flow, if any, is then composed of irrigation return flows with greatly elevated salinity and other pollutant content. Some western states, however, permit the states to file for, and ultimately acquire, the right to the unappropriated flow, and thus to preserve such flow, if desired.

Second, because water rights can only be acquired by diversion, protecting in-stream quality values and uses is difficult to impossible. If water is brought from another basin to augment the flow to improve water quality and increase the waste-assimilative capacity of the stream, the increased flow could be appropriated, diverted, and used by others, thus negating the diversion's purpose of improving water quality.

Third, some management practices that are very popular in the East, for example, those relying on infiltration, in the West may be seen as using water that is already appropriated. Hence, implementation of these practices may be prevented.

Fourth, many western rivers are subject to appropriation by Indian Nations (tribes) that are considered to be first claimants whose right to the water precedes any subsequent claims by white settlers. In such situations, any water quality management plans must include consultation, consent, and the cooperation of water managers from the Indian Nations whose water rights would be affected.

LAWS AFFECTING POLLUTION ABATEMENT

The term environmental law refers primarily to that body of law that seeks to prevent adverse environmental consequences by regulating individual, corporate, and governmental behavior. It is also a form of social control whose objective is to regulate the production and consumption of goods in order to preserve ecological balance, natural beauty, and protect public health and endangered species, so as to maintain a stable and satisfactory level of living and quality of life for present and future generations.

The U.S. legal system and the laws protecting the environment and

governing environmental abatement efforts are based on legislative (statutory) enactments by legislative or other authoritative bodies or on judicial decisions in environmental litigations.

Statutory Laws

The first statutory environmental law, passed by Congress in 1899, was The Refuse Act, which stated that it is unlawful to place any material, except sewage and runoff, into a navigable waterway or tributary thereof without a permit. It is interesting to note that in the absence of any meaningful pollution-control legislation until 1972, this archaic statute was used in the late 1960s and early 1970s to control water pollution from industrial sources, although the original intent of the act was to protect navigation and not water quality.

The first Water Pollution Control Act was passed by Congress in 1948 (PL 80-845) and was amended several times between 1948 and 1972. The act and its subsequent amendments authorized the creation of environmental research centers, established a Division of Water Pollution Control within the Public Health Service, authorized grants to build public treatment plants, gave the federal government authority to abate interstate pollution, among other provisions. In general, these laws were ineffective in controlling pollution or enforcing abatement. Responsibility for pollution control, which was originally with the Division of Water Pollution Control, was transferred several times between the departments of the federal government until by the executive order of the president in 1970 the Environmental Protection Agency was created as an independent governmental agency.

The Water Pollution Control Act Amendments of 1972 (PL 92-500)

This grandiose legislative piece known as the Clean Water Act (CWA) had a monumental impact on environmental efforts in the field of water quality control. It was enacted by Congress over a presidential veto (as were several other subsequent amendments).

The declaration of Section 101(a) states:

The objective of this Act is to restore and maintain the chemical, physical, and biological integrity of the Nation's waters.

The objective clearly specifies that ecological objectives and concerns should receive the highest priority. The objectives of the act also stated as

a goal that the nation's (navigable) waters be suitable for contact recreation and provide for the protection and propagation of fish and aquatic wildlife.

Although PL 92-500 was intended to be a comprehensive water quality program, in practice, great emphasis was placed on controlling point sources, while diffuse-pollution control received far less attention and minimal financial resources. The act accomplished three basic tasks: (1) the regulation of discharges of point sources; (2) the regulation of oil spills and other hazardous substances; and (3) financial assistance for wastewater treatment plant construction.

The most significant and revolutionary contribution of the act is the establishment of an enforcement scheme that is built around the National Pollution Discharge Elimination System (NPDES). This system serves as the basic mechanism for enforcing the implementation of pollution abatement of point sources. As shown in Chapter 1 the definition of point sources has been gradually expanded so that many diffuse sources are now legally classified as point sources (animal feedlots, industrial and municipal storm sewers, combined sewer overflows, runoff and acid discharges from active mines) and require an NPDES permit. For point source discharges the permit, among other things, establishes specific effluent limitations and specifies compliance schedules that must be met by the discharger. It also requires compliance with other relevant state and local pollution control laws, if more stringent. The NPDES permit system is the most important tool for implementing the polluter pays principle.

Section 208 of the CWA. Section 208 of the original CWA of 1972 had a far-reaching impact because it enacted a land-use planning process. For the first time, it was realized that the control of point sources would not have solved all the pollution problems in the United States. Many excellent planning reports were produced by designated planning agencies. However, instead of developing a nonpoint source regulatory program that was deemed prohibitively expensive (Billings, 1976), Congress gave an incentive to the development of state- and areawide water quality management plans that would include all sources of pollution and water quality degradation. Incentives for treatment and penalties for noncompliance with the plan were included for point sources (which at that time excluded urban storm water and other diffuse sources currently defined as point sources), while no enforcement tools were available for nonpoint sources. Also after the plans were completed no mechanism for program implementation and maintenance were in place. Consequently, in many cases the plans were not pursued after their initial release and the effort has essentially never fully achieved its potential.

The Clean Water Act Amendments of 1977
(PL 95-217)

A major revision in the law allowed the EPA to add or remove toxic materials without first requiring a formal hearing. New deadlines were included for meeting the point source abatement requirements stipulated by the act.

The Water Quality Act (Clean Water Act) of 1987
(PL 100-24), Sections 319, 402, and 404

These three sections of the act are the most important tools for controlling diffuse pollution. Their highlights and other important issues of the 1987 CWA related to diffuse source pollution management are discussed by Berg (1988).

Section 319 requires the states to prepare State Nonpoint Source Assessment Reports and encouraged states to develop and implement management programs in order to be eligible for federal funds. The deadline for preparation of the reports was February 1989, and most of the states have complied. The management plans developed in the 319 program are now part of the states' water quality management agenda. Federal matching grants are provided to those states that qualify for assistance in implementing their nonpoint source management programs.

Section 402 establishes the permit program for discharges of pollutants from point sources. More specifically, Section 402(p) requires a NPDES permit for separate storm sewers.

Section 404 regulates the discharge of dredged and fill materials into waters of the United States and also establishes a permit program to ensure that such discharges comply with environmental requirements.

The permits for point source discharges under Section 402 are issued by states, while the permits under Section 404 are issued by the U.S. Army Chief of Engineers. Other provisions of the act such as the National Estuary Program (Section 3020), Clean Lakes Program (Section 314), and the Great Lakes Basin and Chesapeake Bay Programs also deal with diffuse source pollution management. The act also reauthorizes funding for the areawide water quality management plans under Section 208.

The Water Quality Act's jurisdiction extends to all waters of the United States. The phrase "all waters" includes waters that are currently used, were used in the past, or may be susceptible to use in interstate or foreign commerce (this jurisdiction is guaranteed by the Commerce Clause of the U.S. Constitution). Generically, such bodies of water include all navigable waters that have been legally understood as waters on which a canoe can be floated. These waters include:

- All waters that are subject to the ebb and flow of the tide;
- The territorial seas;
- Interstate waters and wetlands;
- All other waters (such as intrastate lakes, rivers, streams, and wetlands), if their use, degradation, or destruction could affect interstate or foreign commerce;
- Tributaries to waters or wetlands previously identified;
- Wetlands adjacent to waters previously identified.

Present Regulations for the Control of Urban Diffuse Sources Derived from the Clean Water Act

Control of Combined Sewer Overflows

The U.S. Environmental Protection Agency published its control strategy for combined sewer overflows (CSOs) in 1989 (*Federal Register*, August 10, 1989). The strategy relies on the NPDES permit system. The permit system is aimed at bringing all CSO discharges into compliance with the technology-based requirements of the Clean Water Act and applicable state standards, and to minimize water quality, aquatic biota, and human health impacts from wet-weather overflows.

All permits for CSO discharges should require the following technology-based limitations as a minimum: (1) proper operation and regular maintenance programs for sewer systems and combined sewer overflow points; (2) maximum use of the collection system for storage; (3) review and modification of pretreatment programs to assure that CSO impacts are minimized; (4) maximization of flow to the treatment plant for treatment; (5) prohibition of dry-weather overflows; and (6) control of solid and floatable materials in CSO discharges.

Additional CSO control measures are based on the potential impact on receiving water bodies that would bring CSO discharges in compliance with state standards. Additional control measures include improved operation, best management practices, supplemental pretreatment program modifications, sewer ordinances, local limits programs, identification and elimination of illegal discharges into sewer systems, specific pollutant limitations, compliance schemes, direct treatment of overflows, sewer rehabilitation, in-line and off-line storage, reduction of tide water intrusion, construction of CSO controls within the sewer system or at the discharge point, sewer separation, and new or modified treatment facilities. The compliance monitoring program should be described and included in the permit.

The strategy does not cover treatment plant bypasses, which are considered to be an "intentional diversion of waste streams from any

portion of the treatment facility” that begins at the headwork of the facility. Bypasses are not allowed unless (1) they are unavoidable to prevent loss of life; and (2) there is no other feasible alternative to the bypass.

Storm Water (separate sewers) Permit Regulations

Overview. The storm-water control rules established by the EPA seek to establish NPDES permit application requirements for

- Stormwater discharges associated with industrial activity;
- Discharges from large separate storm sewer systems (systems serving a population of more than 250,000);
- Discharges from medium municipal separate storm sewer systems (systems serving a population of more than 100,000 but less than 250,000).

Discharges from other municipal separate storm sewers, including separate municipal systems serving a population of less than 100,000 and discharges associated with industrial activity connected to these systems are not subject to the permit system.

The rules are to be implemented in a phase-in approach. Storm-water discharges from industrial areas that are subject to the permit system have to comply with Sections 301 and 402 of the Clean Water Act, which require application of the best available treatment technology (BAT). Permits for discharges from municipal storm sewers include controls that reduce the discharge of pollutants to the maximum extent practicable (MEP), as well as a requirement to effectively prohibit discharging nonstorm water (cross-connections) into the storm sewers.

The permit system requires industrial facilities that discharge storm water associated with industrial activity to submit sampling data, a description of storm-water management practices, and certification that the discharge does not contain processed water, domestic sewage, or hazardous wastes. Group applications, industry by industry, are permitted. Indirect discharges to municipal systems serving a population of 100,000 or more generally do not have to submit applications, but do have to notify the municipality of their discharge.

Permits are issued on a systemwide basis for municipal separate storm sewers. Municipalities are first required to describe their existing storm-water management program, identify all known outfalls, and conduct field screening for illicit connections. The municipalities are then required to verify illicit connections, conduct representative sampling, and describe priorities for storm-water management during the 5-year permit term.

The data collected during these phased tasks will allow the permit to be developed for site-specific conditions.

The EPA rules define storm water as follows:

“Storm water” means storm water runoff, snow melt runoff, surface runoff, street wash waters related to street cleaning or maintenance, infiltration (other than infiltration contaminated by seepage from sanitary sewers or by other discharges) and drainage.

Relation to ground-water quality. In Section 319 the Clean Water Act (1987) strengthens the regulatory link between diffuse (nonpoint) pollution and ground-water quality. Under the 1987 amendments, the CWA now specifically requires states to select best management practices, taking into account the impact of the practice on ground-water quality. The Senate Report explained (Thompson, Adler, and Landman, 1989):

States are required to consider impact of management on groundwater quality. Because of the intimate hydrologic relationship that often exists between surface and groundwater, it is possible that measures taken to reduce runoff of surface water containing contaminants may increase transport of these contaminants to groundwater. The State should be aware of this possibility, when defining best management practices, especially in aquifer recharge areas.

The 1972 Coastal Zone Management Act (PL 92-583) programs. This act was established in response to the high rate of development in coastal areas and out of concern about the environmental effects of this growth. These coastal areas encompass land and tidal zones, including those located in the Great Lakes basin. The National Oceanic and Atmospheric Administration (NOAA) is implementing programs mandated by the act.

The act specified that:

The habitat areas of the coastal zone, and the fish, shellfish, other living marine resources, and wildlife therein, are ecologically fragile and consequently extremely vulnerable to destruction by man's alterations (Section 302d);

Important ecological, cultural, historic, and aesthetic values in the coastal zone which are essential to the well-being of all citizens are being irretrievably damaged or lost (Section 302e).

Land uses in the coastal zone, and the uses of adjacent lands which drain into the coastal zone, may significantly affect the quality of coastal waters and habitats, and efforts to control coastal water pollution from land use activities must be improved (Section 302k).

Section 6217 (1990 amendments) of the act then delineates the program for controlling diffuse (nonpoint) pollution. Participation by states in the Coastal Nonpoint Pollution Control Program (CNPCP) is on a voluntary basis; however, the act provides participating states with funding for the program and enforceable policies with penalties for noncompliance. At a minimum, each state program is to provide for the implementation of management measures to protect coastal waters. These programs should contain the following components:

1. Identification of land uses that may individually or cumulatively cause or contribute to degradation of coastal waters.
2. Identification of critical coastal areas that should be subject to management measures.
3. Identification of management measures that are most appropriate for the threatened coastal areas.
4. Delineation of technical assistance.
5. Public participation.

The program is carried out in coordination with the Section 319 programs of the Clean Water Act. Since 1992, about \$12 million per year has been authorized for the program.

The Clean Water Act of 1987 also recognized estuaries and other coastal water as a critical national resource whose health and productivity are increasingly threatened by coastal growth and development. The act then formally established the National Estuary Program (NEP) to demonstrate innovative approaches applicable to coastal areas nationwide. The NEP program covers 12 estuaries, 6 in the North Atlantic region, 1 in the mid-Atlantic zone, 2 on the Gulf Coast, and 3 on the West Coast. Most of the research projects are conducted by universities under NOAA's Sea Grant program. NOAA's Coastal Zone Management and the EPA's National Estuary Programs are coordinated under an agreement between the two agencies.

Wetland Protection

Wetlands have great value for the hydrological-ecological terrestrial system. Their benefits are numerous and include maintaining and storing flows, wildlife habitat, and attenuation of pollutants (see Chapter 14). Wetlands are protected under both Section 402 and 404 permits. The NPDES permit system regulates discharges from point sources, including CSO and storm-water discharges. Section 404 permits regulate drainage and filling of wetlands. These permits, which treat wetlands as receiving bodies of water, do not allow wide use of natural wetlands for treatment

of discharges from diffuse sources, unless they comply with the stream and effluent standards. Specifically, no discharges into wetlands can be permitted that would violate other applicable laws, such as state water quality standards, toxic effluent standards, or the Endangered Species Act. These regulations do not apply to man-made wetlands that can be considered as land treatment.

The U.S. Army Corps of Engineers' evaluation of the Section 404 permit application is a two-part process that includes both the evaluation of the project's eligibility for a permit and environmental assessment. The key policies of the permit evaluation are

- Dredged or fill material should not be discharged into waters of the United States unless it can be demonstrated that such discharges will not have an adverse impact on the aquatic ecosystem.
- From a national perspective, the degradation or destruction of special aquatic sites, such as filling wetlands, is considered to be among the most severe environmental impacts addressed by Section 404.

Under the Food Security Act of 1985 (see the following section) the "Swampbuster" provision (Subtitle III) requires that USDA program benefits be withdrawn from farmers who convert any naturally occurring wetlands to cropland after 1985. Many states have enacted or are considering enacting more stringent regulations protecting wetlands (Florida, Wisconsin, and others).

The Food Security Act (The Farm Bill) of 1985 (PL 99-198). The major deficiency of the Clean Water Act is that it does not provide for enforcement of the abatement of agricultural nonpoint sources, which represent the most significant cause of water quality degradation of many receiving bodies of water. Almost exclusively, programs relied on the voluntary participation of farmers to implement pollution abatement. This lack of enforcement procedures is most likely related to the "sacred" right of unrestricted use of land for family farming. This argument may not be valid today, since more and more small family farms are acquired by large industrial operations. The most significant influences on state programs to control agricultural pollution have to come from federal assistance and land management programs, rather than from the EPA's water quality protection programs mandated by the Clean Water Act.

However, Congress has realized that farming in many cases greatly subsidized by the federal and state governments, does cause pollution. As a consequence, the legislators passed the Food Security Act, which creates two programs that are aimed at reducing pollution from agricultural operations. The Conservation Reserve Program (CRP) gives the Soil Conservation Service the authority to make annual rental

payments for 10 years to farmers who retire highly erodible land and land bordering bodies of water from farming and plant it with such permanent cover crops as grasses, legumes, or trees. The CRP intends to remove more than 30 million hectares of the most erodible cropland from agricultural production. The CRP stream buffers can idle cropland for up to 30 meters (100 ft) from the water's edge. Under "Conservation Compliance" provisions, farmers who plant annually tilled crops on highly erodible lands must implement locally developed and approved conservation plans in order to remain eligible for price support, crop insurance, and other USDA program benefits. Under the "sodbuster" provision (Subchapter II), to retain USDA benefits farmers must follow an approved conservation system when plowing fields that were not in use for crop production between 1981 and 1985.

The CRP programs aimed at excluding highly erodible land have indirect water quality benefit, though not all lands included in the program pose a threat to water quality. On the other hand, buffer strips have a direct water quality improvement benefit. Further modifications of the CRP program to increase water quality benefits were being considered by the lawmakers during the time this book was being written. Future improvements may include adding an additional 10 million hectares to the CRP to address environmental problems, including lands with salinity, selenium, siltation, soil drainage, and other problems. The benefits of the CRP program were estimated by Ogg and Ribaud (1988).

The Conservation Compliance (CC) program requires all farmers with row crops located on highly erodible land to establish a Soil Conservation Service approved plan or lose eligibility for federal support, including supplements and disaster assistance. In the absence of enforcing regulations, this program is the most important tool to persuade farmers to participate.

To implement these programs, employees of the Soil Conservation Service work with local units of state governments of the Soil and Water Conservation Districts (SWCD), which are managed by locally elected, unsalaried citizens (mostly farmers).

Other Federal Laws Affecting Diffuse Pollution and Water Quality Management. Among the most complicating factors in diffuse-pollution abatement and water quality management are the plethora of laws affecting the decision-making process and specifying various sometimes conflicting environmental policies. These legislative pieces include:

Environmental Laws

1. The National Environmental Policy Act
2. The Clean Air Act Amendments

3. The Federal Environmental Pesticide Control Act
4. The Rare and Endangered Species Act
5. The Safe Drinking Water Act
6. The Federal Insecticide, Fungicide, and Rodenticide Act
7. Toxic Substances Control Act
8. The Wild and Scenic River Act

Floodplain Management Laws

9. Flood Control Act and Amendments
10. National Flood Insurance Programs
11. Flood Disaster Protection Act

U.S. Department of Agriculture Laws

12. Rural Development Act

Mining

13. Surface Mining Control and Reclamation Act
14. Federal Land Policy and Management Act

State Statutory Laws

Many states have enacted effective pollution-control statutes and programs. It is beyond the scope of this book to describe the programs in each individual state, though a summary of several state laws may demonstrate some effective and successful approaches to the resolution of the diffuse-pollution problem. These programs are carried in addition to or as a supplement to the federally mandated programs. According to Section 319 of the Clean Water Act (1987 Amendments), all states are required to prepare nonpoint pollution abatement plans that list the state programs. These plans are available from the state pollution-control agencies. A few examples are listed herein.

Florida has strong wetland protection ordinances aimed at the reduction of wetland losses throughout the state. In addition, Florida's storm-water management program, which applies to all new developments, is designed to ensure that the volume, rate, timing, and pollutant load of runoff after development do not cause a violation of state water quality standards. The Florida rule is essentially a performance standard specifying what is expected to achieve compliance with state water quality standards.

Iowa has developed requirements for all confined feedlots that apply to all open feedlots exposed to rainfall and to total confinement facilities where precipitation is not a factor. At a minimum, all open facilities must remove settleable runoff solids.

Maryland began a comprehensive storm-water management program in 1982. This program is administered by the state's Sediment and Stormwater Administration. Depending on the county, mandated local ordinances must at a minimum require that postdevelopment peak discharge for 2-year- and/or 10-year-frequency storm events be maintained at the predevelopment levels. Maryland also participates in the regional Chesapeake Bay clean-up program.

Additional state programs are listed in Chapter 1 (Wisconsin Priority Watershed Programs), in Thompson et al. (1989), CH2M/Hill (1990), and several EPA and state monographs.

Judicial (Judge-Made) Laws

Dating back to the Roman Empire, courts and judges have made rulings that, among other things, affected many aspects of the environment. Most of the water quality litigations are decided by a judge according to the Equity Law, and primarily involve injunction or specific orders from the courts restraining certain types of actions or regulating other actions. These rulings were based on imperatives and doctrines as previously elucidated.

Many court decisions involving water quality are based on the doctrine of nuisance and trespassing. In many cases, judges used these doctrines when actual damage had occurred and was proved by the plaintiff. However, in a landmark case Judge John Grady stated that this may no longer be true because "It is the ability of the courts of equity to give a more speedy, effectual, and permanent remedy in cases of public nuisance. They can not only prevent the nuisance that is threatened, and before irreparable mischief ensues, but arrest or abate those in progress and, by perpetual injunction, protect the public from them in the future" (*Illinois vs. the City of Milwaukee*, Federal Court, Chicago, 1977). In his ruling, Judge Grady issued stringent performance and effluent standards for the clean up of point and diffuse source pollution from the Milwaukee metropolitan area. Some of these standards were almost technically unattainable; however, the ruling was appealed and subsequently overturned by higher courts. If this ruling had been allowed to stand, it would have made a legal precedent for similar cases by other federal and state courts.

In today's legal environment in the United States, litigation is common; however, the large number of statutory laws in place and in force do not allow for frequent judicial rulings outside the statutory legal framework. Many legal cases of violations of pollution-control laws and ordinances are settled out of court.

References

- Bator, F. M. (1958). "The anatomy of market failure," *Quart. J. Econ.*, Aug., pp. 351-379. Included in *Pollution Taxes, Effluent Charges and Other Alternatives for Pollution Control*, U.S. Congress Rep. No. 95-5, 1978, Washington, DC.
- Baumol, W., and W. Oates (1988). *The Theory of Environmental Policy*, 2d ed., Cambridge University Press, Cambridge.
- Berg, N. (1988). "Policies and intergovernmental relations of nonpoint pollution abatement programs," in *Political, Institutional and Fiscal Alternatives for Nonpoint Pollution Abatement Programs*, V. Novotny, ed., Proceedings, Symposium, Marquette University Press, Milwaukee, WI.
- Beuscher, J. H. (1967). *Water Rights*, College Printing and Typing, Madison, WI.
- Billings, L. G. (1976). "The evolution of 208 water quality planning," *Civ. Eng.* 46(11):54-55.
- Boland, J. J. (1991). Unpublished definitions. Panel on policies, institutions and economics, National Research Council, Washington, DC.
- Braden, J. B. (1988). "Nonpoint pollution policies and politics: The role of economic incentives," in *Nonpoint Pollution 1988: Policy, Economy, Management and Appropriate Technology*, V. Novotny, ed., American Water Resources Association, Bethesda, MD, pp. 57-65.
- Carson, W. D. (1980). "Effluent charges for water pollution control in California," State Water Research Control Board, State of California, Sacramento.
- Caufield, H. P., Jr. (1988). "The federal environmental legislative process," in *Nonpoint Pollution 1988: Policy, Economy, Management and Appropriate Technology*, V. Novotny, ed., American Water Resources Association, Bethesda, MD, pp. 1-8.
- Caufield, H. P., Jr. (1991). "Problem of nonpoint source agricultural water pollution: Toward a hypothetical federal legislative solution," *Water Resour. Bull.* 27(3):447-452.
- CH2M/Hill (1990). *Nonpoint Source Impact Assessment*, Report No. 90-5, Water Pollution Control Federation Research Foundation, Alexandria, VA.
- Kneese, A., and B. Bower (1968). *Managing Water Quality: Economics, Technology, Institutions*, Johns Hopkins University Press, Baltimore, MD.
- Krenkel, P., and V. Novotny (1980). *Water Quality Management*, Academic Press, New York.
- Maas, A., et al. (1970). *Design of Water Resources Systems*. Harvard University Press, Cambridge, MA.
- Novotny, V. (1988a) "Diffuse (nonpoint) pollution—a political, institutional, and fiscal problem," *J. WPCF* 60(8):1404-1413.
- Novotny V., ed. (1988b). *Political, Institutional and Fiscal Alternatives for Nonpoint Pollution Abatement Programs*, Proceedings, Symposium, Marquette University Press, Milwaukee, WI.
- Novotny, V., and E. Joeres (1983). "Planning and financing water resources development with emphasis on water pollution control and sanitation, in

- Proceedings, Fourth World Congress IWRA*, Buenos Aires, Argentina, Tycooly Int. Publ., Dublin.
- Ogg, C. W., and M. C. Ribaud (1988). "Economic implications of the Food Security Act of 1985's discretionary environmental provisions," in *Nonpoint Pollution 1988: Policy, Economy, Management and Appropriate Technology*, V. Novotny, ed., American Water Resources Association, Bethesda, MD, pp. 49–56.
- Organization for Economic Cooperation and Development (1991). *The State of the Environment*, OECD, Paris, France.
- Rogers, P., and A. Rosenthal (1988). "The imperatives of nonpoint source pollution control," in *Political, Institutional and Fiscal Alternatives for Nonpoint Pollution Abatement Programs*, V. Novotny, ed., Marquette University Press, Milwaukee, WI.
- Solow, R. M. (1971). "The economist's approach to pollution and its control," *Science* 173:497–503.
- Thompson, P., R. Adler, and J. Landman (1989). *Poison Runoff*, Natural Resources Defense Council, Washington, DC.
- Viessman, W., Jr., and C. Welty (1985). *Water Management—Technology and Institutions*, Harper & Row, New York.

Additional Reading

- Dahlman, C. (1979). "The problem of externality," *J. Law Econ.* 22:141–62.
- Fisher, A., and F. Peterson (1976). "The environment in economics," *J. Econ. Lit.* 14:1–33.
- Freeman, A., III (1979). *The Benefits of Environmental Improvement*, Johns Hopkins University Press, Baltimore, MD.
- Just, R., D. Hueth, and A. Schmitz (1982). *Applied Welfare Economics and Public Policy*, Prentice-Hall, Englewood Cliffs, NJ.
- Kneese, A., and C. Schultze (1975). *Pollution, Prices, and Public Policy*, Brookings Institution, Washington, DC.
- Kneese, A., R. Ayres, and R. d'Arge (1970). *Economics and the Environment: A Materials Balance Approach*, Johns Hopkins University Press, Baltimore, MD.
- Krutilla, J., and A. Fisher (1975). *The Economics of Natural Environments*, Johns Hopkins University Press, Baltimore, MD.
- Majone, G. (1986) "International institutions and the biosphere," in *Sustainable Development of the Biosphere*, W. C. Clark, and R. E. Munn, eds., Cambridge Univ. Press, Cambridge, England.
- Mäler, K. G. (1974). *Environmental Economics: A Theoretical Inquiry*, Johns Hopkins University Press, Baltimore, MD.
- Miller, W. L., and J. H. Gill (1976). "Equity considerations in controlling nonpoint pollution from agricultural sources," *Water Resour. Bull.* 12:253–261.
- Organization for Economic Cooperation and Development (1974). *Problems in Transfrontier Pollution*, OECD, Paris.
- Organization for Economic Cooperation and Development (1976). *Economics of Transfrontier Pollution*, OECD, Paris.

- Organization for Economic Cooperation and Development (1979). *OECD and the Environment*, OECD, Paris.
- Pfeiffer, G. H., and N. K. Whittlesey (1978). "Controlling nonpoint externalities with input restrictions in an irrigated river basin," *Water Resour. Bull.* 14:1387-1403.
- Schultze, C. (1977). *Public Use of Private Interest*, Brookings Institution, Washington, DC.
- Walter, I. (1975). *International Economics of Pollution*, MacMillan, London.
- Williams, A. (1966). "The optimal provision of public goods in a system of local government," *J. Polit. Econ.* 74:18-33.

3

Hydrologic Considerations

He who sees things grow from the beginning will have the best view of them.

Aristotle

Pollution from diffuse sources is driven by meteorological events, that is, precipitation. It is a known fact that there is a correlation between the pollutant loadings from a watershed and rainfall volume, infiltration and storage characteristics of the watershed, permeability of soils, and other hydrological parameters. This distinguishes diffuse pollution from traditional point source pollution, which bears little relation to watershed hydrology.

Diffuse pollution therefore has its beginning in the atmospheric transport of pollutants, and its occurrence and magnitude are closely related to the hydrologic cycle. Consequently, the pollutant load from nonpoint sources has a strong random (unpredictable) component. In addition, hydrologic modifications of watersheds can increase or decrease diffuse-pollution loads.

Most of the models used for simulating diffuse-pollution loadings are basically models of watershed hydrology or are closely related to it. Rainfall energy and the splashing effect of rain droplets liberate soil particles, which then become available for the transport by the overland flow. If, for example, the overland flow is diminished due to higher infiltration, so is the transport of particulate pollutants. If agricultural chemicals or organic fertilizers are placed on the land and surface

overland flow is generated by a storm, a significant portion of these contaminants can be lost in surface waters. Mobile pollutants (that is, pollutants that are dissolved and move with water as ions or salts) can be leached into the ground-water zone and cause ground-water contamination and pollution. The highest pollutant loadings, and in many cases the highest concentrations of contaminants from diffuse sources, occur during high-flow and flood conditions. On the other hand, point source impact and the impact of pollutants carried into surface waters by subsurface flow are most severe during low-flow (drought) conditions.

To control and understand the generation and transport of diffuse pollution, one has to study the hydrologic process causing and contributing to diffuse pollution and to consider the various paths the contaminants travel from the source areas to the receiving water bodies. A large number of basic textbooks and handbooks dealing with hydrological processes and hydrologic cycle are available (Bedient and Huber, 1988; Bras, 1990; Chow, 1964; Chow, Maidment, and Mays, 1988; Linsley, Kohler, and Paulhus, 1982; Hall, 1984; Viessman, Lewis, and Knapp, 1989; Ponce, 1989; McCuen, 1989). The reader is referred to these texts for further reference.

PRECIPITATION–RUNOFF RELATIONSHIP

A classic representation of the rainfall transformation into runoff and the components of the hydrological cycle is shown on Figure 3.1. The first stage of the runoff formation is condensation of atmospheric moisture into rain droplets or snowflakes. During this process, water is in contact with atmospheric pollutants. The pollution content of rainwater can therefore reach high levels. In addition, rain water dissolves atmospheric carbon dioxide, sulfur and nitrogen oxides, and as a weak acid, it then reacts with soil, limestone, and dolomite geological formations. Many ancient statues and historical structures, for example, in Greece and Rome, were made of marble (a form of limestone) and have been deteriorating rapidly in the past few decades as a result of the greatly increased acidity of precipitation (see Chap. 4 for further discussion of the interaction of precipitation with atmospheric pollution).

Runoff formation begins after rain (snow) particles reach the surface. During the winter months runoff formation may be delayed by snowpack formation and subsequent melting. During the initial phase of runoff formation, rain energy liberates the soil particles and picks up the particulate and contaminants deposited on the surface and dissolves salts and other chemicals.

Runoff generated by precipitation has three components:

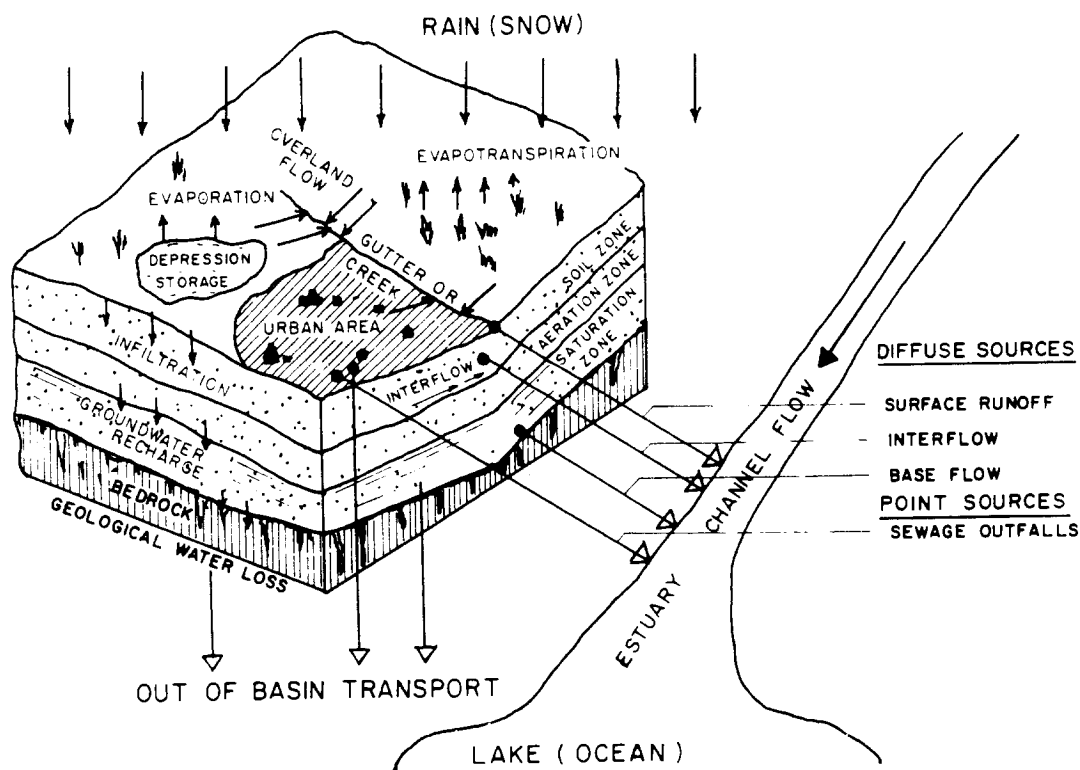


FIGURE 3.1. Schematic representation of watershed hydrology.

1. *Surface runoff* is a residual of precipitation after all losses have been satisfied. Numerical subtraction of the losses yields so-called *excess* or *net rain*. The losses include interception by surface vegetation, depression storage and ponding, infiltration into soils, evaporation from soils and open water surfaces, and transpiration by vegetation. *Evapotranspiration* is a term describing both evaporation from the soil and transpiration by plants. Since surface runoff is a residual of precipitation after all losses have been subtracted, a linear relationship between the volume of precipitation and runoff does not exist. The highest loads of particulate pollutants are carried by surface runoff.
2. *Interflow* is that portion of the water infiltrating into the soil zone that moves in a horizontal direction due to the lower permeability of subsoils. The amount of interflow is again a residual of infiltration after ground-water recharge, soil moisture storage, and evapotranspiration have been subtracted.
3. *Ground-water runoff* (base flow) is defined as that part of the runoff contribution that originates from springs and wells. In sewered urban areas one can also include infiltration inflow into sewers, which can be substantial. Most stream flow during prolonged drought periods can

be characterized as ground-water runoff. In some arid and semiarid regions, the natural base flow may be zero during certain times of the year, and the measured flow in streams may originate from sewage outfalls.

Streams that have measurable flow during the entire hydrological season (a hydrological year begins on October 1, and ends on September 30 of the following year) are called *perennial streams*. *Ephemeral streams* are streams without a measurable flow during certain times of the hydrological year. Urbanization and the accompanying discharges of wastewater effluents can change an ephemeral stream into a perennial one, for example, the Las Vegas, Nevada, Wash draining into the Colorado River is a man-made perennial stream-marsh system. Overuse and mining of ground-water resources will change a perennial stream into an ephemeral watercourse, as has happened in many places, for example, in Tucson, Arizona. From these examples one can see that man has a profound impact on the hydrology of a body of water. In fact, very few streams in populated areas of the world have truly natural flows. Runoff quantity and quality can be dramatically altered by changing the use of land by man, that is, urbanization, deforestation, storage reservoirs, and other land and stream modifications discussed in Chapter 1 (Fig. 1.5).

The quality of surface runoff can be related to the erosion intensity by precipitation and to the quantity of contaminants accumulated on the surface or in the top soils. Interflow and ground-water quality can be related to the amount of contaminants present in the soil, which also reflects the basic chemical composition of the soil, subsoil, and bedrock. Very often pollution of the interflow and ground-water (base) flow results from the excessive contamination of soils; for example, from overloaded septic tank seepage, overfertilization, and excessive chemical use on farms and urban lawns.

Mathematically, the runoff relation to precipitation can be expressed as

Surface runoff

$$R_s = P - \Delta S_i - \Delta S_d - f\Delta t \quad (3.1)$$

Interflow

$$R_i = (f - ET)\Delta t - \Delta S_s - q_g \quad (3.2)$$

Ground-water (base) flow

$$R_g = q_g - \Delta S_g - q_d \quad (3.3)$$

where

R_s = volume of the surface runoff in cm during a time interval Δt

P = precipitation volume (cm)

ΔS_i = change in available interception storage (cm)

ΔS_d = change in available depression surface storage (cm)

f = infiltration rate (cm/hr)

ET = evapotranspiration rate from the soil zone (cm/hr)

ΔS_s = soil moisture storage change (cm)

q_g = ground-water recharge (cm)

R_i = interflow (cm)

R_g = ground-water flow contribution (cm)

ΔS_g = ground-water storage change (cm)

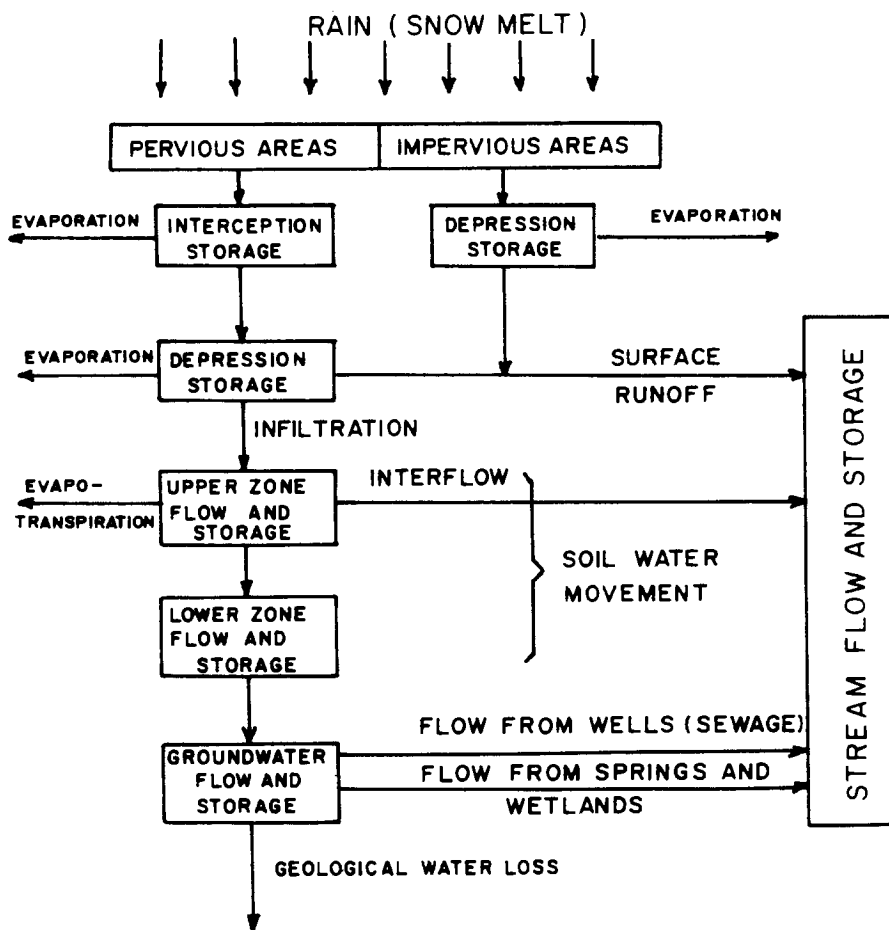


FIGURE 3.2. Block diagram of watershed hydrological processes and storage.

q_d = geological water loss (cm)

Δt = time interval (hr)

A block diagram of the rainfall–runoff transformation process is shown in Figure 3.2.

Components of the Rainfall–Runoff Transformation Process

Interception

A part of the precipitation volume is intercepted by vegetation where it adheres to the surface until a sufficiently heavy film is formed, at which point gravity begins to prevail over adhesion. Interception storage is that part of precipitation that wets or adheres to the surface of aboveground objects and vegetation and is returned to the atmosphere by evaporation. The amount that is intercepted depends on the type and intensity of the vegetation, intensity and volume of the rainfall, roughness of the surface, and the season of the year or growth stage of the vegetation.

The few models for interception storage reported in the literature are crude and inaccurate. Interception can be measured by comparing precipitation in gages or simple open buckets beneath the vegetation with that recorded nearby under the open sky. Generally, about 0.5 to 1.2 mm of rain can be held on foliage before an appreciable drip can take place (Viessman, Lewis, and Knapp, 1989). The total interception by an individual plant is directly related to the amount of foliage (its surface area per unit area of ground surface) and its character and orientation. About 1.2 to 1.8 mm of precipitation can be intercepted by grass and dense shrubbery.

A general form for interception was proposed as (Gray, 1973; Bras, 1990)

$$I = a + bP^n \quad (3.4)$$

where P is precipitation in centimeters. Values of the coefficients for some typical vegetal covers are given in Table 3.1.

Depression Storage

Water reaching the surface must first fill the surface depressions, forming small puddles, ponding, or adding to the general wetness of the area. Water stored in the depression storage either evaporates or percolates into the soil zone. Only when the precipitation rate exceeds infiltration

TABLE 3.1 Coefficients for the Interception Formula (Eq. (3.4))

Vegetal Cover	Coefficients		
	<i>a</i>	<i>b</i>	<i>n</i>
Orchards	0.1	0.18	1.0
Maple, beach, oak in forest	0.1-0.12	0.18	1.0
Beans, potato, cabbage, and other small crops	0.05h ^a	0.45 h	1.0
Forage, alfalfa, etc.	0.025 h	0.30 h	1.0
Small grains, rye, wheat, barley	0.012 h	0.15 h	1.0
Corn	0.012 h	0.15 h	1.0

Source: From Gray (1973).

^ah refers to the height of the plant in meters. Interception is in centimeters for *P* in centimeters.

and all surface storage (depression and interception) is exhausted will surface runoff result.

The character of depression storage as well as its magnitude depends largely on the surface characteristics that can be generally related to land use. The primary factors determining depression storage are surface character, roughness, and slope. An accurate estimation of depression storage is not possible, and little information is available that could serve as a guide for choosing the values of depression storage that would be based on physical measurements in the field.

In hydrological models interception storage is usually lumped together with depression storage into one surface storage (abstraction) parameter, which is determined by calibrating the model. After his first experience with the well-known *Stanford Watershed Model*, Linsley (1967) pointed out that the surface storage parameter is the key element in calibrating of the model for smaller watersheds (less than 50 km²). Some information derived from water balance and/or modeling on the magnitude of the surface storage parameter has been published. For example, Tholin and Keifer (1960) estimated surface storage for Chicago's urban areas as being 6.25 mm ($\frac{1}{4}$ in.) on pervious areas with grass and 1.56 mm ($\frac{1}{16}$ in.) on impervious areas. Figure 3.3 relates surface storage to the slope for various agricultural land uses.

As stated before, the surface storage volume must be exhausted before surface runoff can begin. Therefore, it represents an *initial abstraction* from the gross rainfall input. On the other hand, surface storage is not uniform, even on small watersheds. Hence, for modeling, a range, which is then subtracted from the rainfall as shown on Figure 3.4, may be more appropriate.

Depression storage can be increased by engineering and agronomic

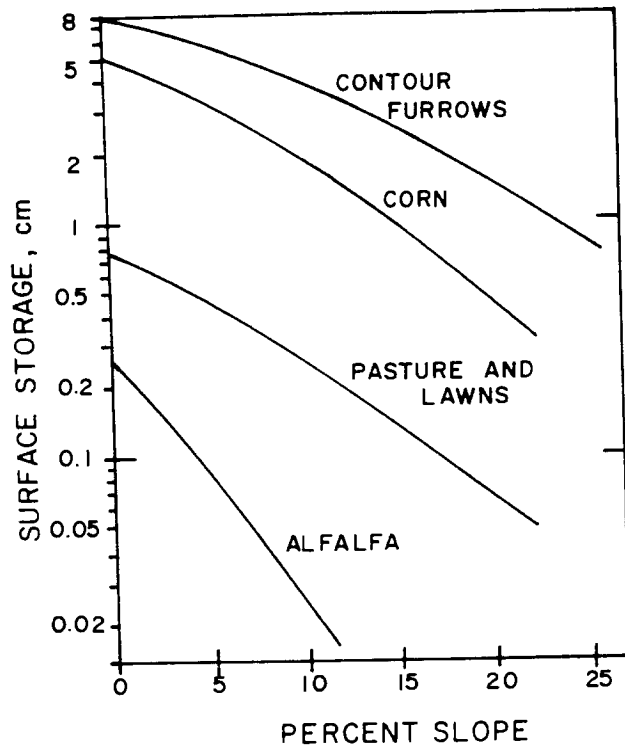


FIGURE 3.3. Relationship of the surface storage parameter to the slope of the land. (After Hiemstra, 1968.)

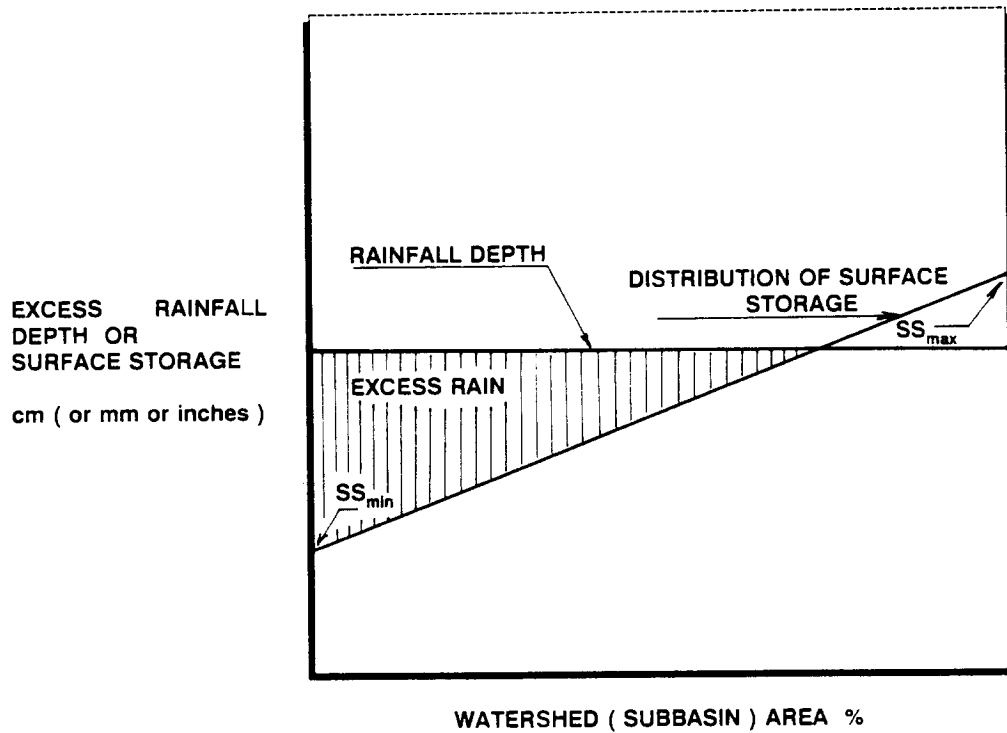


FIGURE 3.4. Initial subtraction of rainfall in a watershed hydrological concept, considering a statistical distribution of surface storage throughout the watershed.

practices, such as plowing, raking of the surface, mulching, and seeding and planting vegetation. Depression storage on agricultural fields is at maximum during planting, but decreases afterwards.

Soil Permeability and Infiltration

Permeability

Infiltration and permeability are not synonymous. Permeability is defined by Darcy's law and denotes the rate of water movement through the soil column under saturated conditions (all voids are filled by water and flow is primarily due to gravitational forces). Infiltration, on the other hand, is the rate at which water percolates from surface storage into the soil zone, and it is governed by the forces of gravity and capillary suction. The permeability of soils depends on such characteristics as texture, compactness, and organic and chemical composition.

As to their permeability and surface runoff potential the soils in the United States have been classified by the Soil Conservation Service (SCS) into four hydrologic groups:

1. Group A are soils with low total surface runoff potential due to high infiltration rates, even when thoroughly wetted. These soils consist chiefly of deep, well to excessively drained sands and gravel.
2. Group B are soils of low to moderate surface runoff potential that have moderate infiltration rates and have a moderately fine to moderately coarse texture.
3. Group C soils have high to moderate surface runoff potential and slow infiltration rates, and consist chiefly of soils with a layer that impedes downward movement of water, or soils with a moderately fine to fine texture.
4. Group D soils have high surface runoff potential and very slow infiltration rates, and consist chiefly of clay soils with a high swelling potential, soils with a permanently high water table, soils with a clay pan or clay layer at or near the surface, and shallow soils over nearly impervious material.

Soil classification and approximate permeabilities can be obtained from the SCS soil maps that are available for most counties of the conterminous United States.

Particle size distribution (texture), arrangement of soil particles, organic matter content, clay mineral content, exchangeable sodium content, and total concentrations of salts are the most important factors affecting permeability (Horn, 1971; Chow, Maidment, and Mays, 1988;

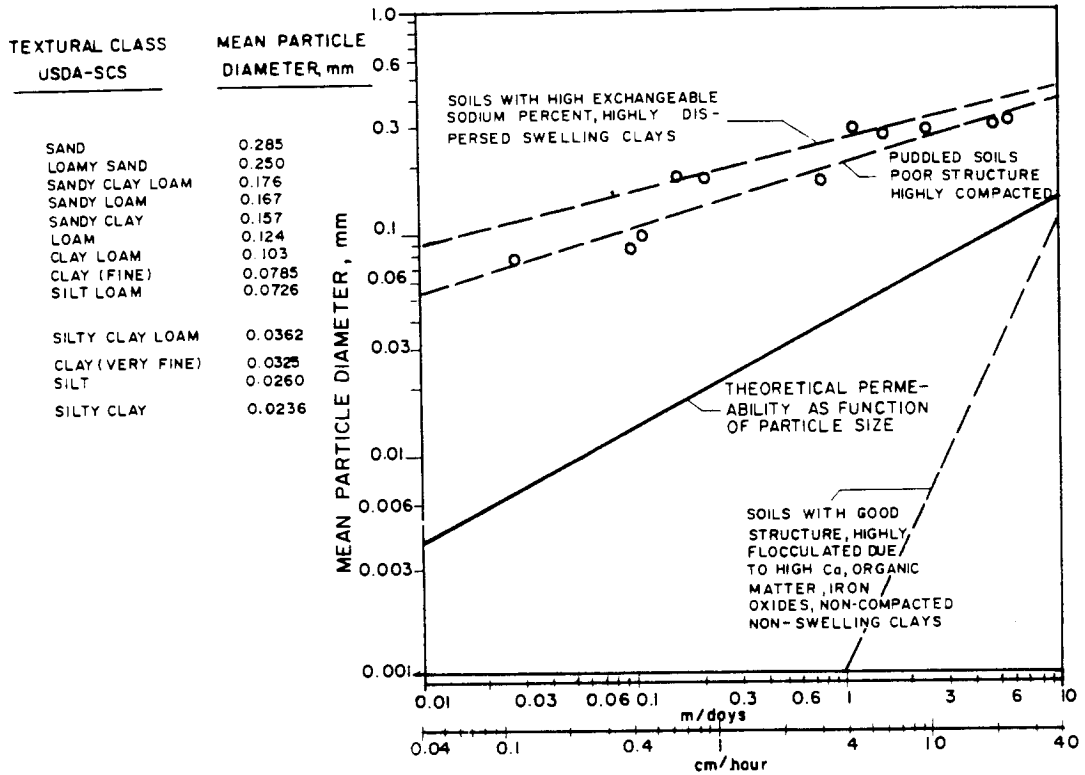


FIGURE 3.5. Relationship of permeability to soil texture. The circled points represent permeability of septic seepage fields after few years of operation. (After Horn, 1971, by permission of ASCE.)

TABLE 3.2 Permeability Classes According to the Soil Conservation Service, U.S. Department of Agriculture

Permeability Class	in./hr	cm/hr	m/day
A Very rapid	>10	>25	>6.2
+B Rapid	3.00-10.00	12.5-25.0	3.1-6.2
B Moderately rapid	2.5-5.0	6.3-12.5	1.5-3.1
+C Moderate	0.8-2.5	2.0-6.3	0.5-1.5
C Moderately slow	0.2-0.8	0.5-2.0	0.12-0.5
+D Slow	0.05-0.2	0.12-0.5	0.03-0.12
D Very slow	<0.05	<0.12	<0.03

Viessman, Lewis, and Knapp, 1989). In addition, permeability rates can be affected by soil compaction, cultivation, vegetation, and land cover.

Most guides developed as aids to estimating permeability rates are based on the relationship of permeability to the soil texture (Fig. 3.5).

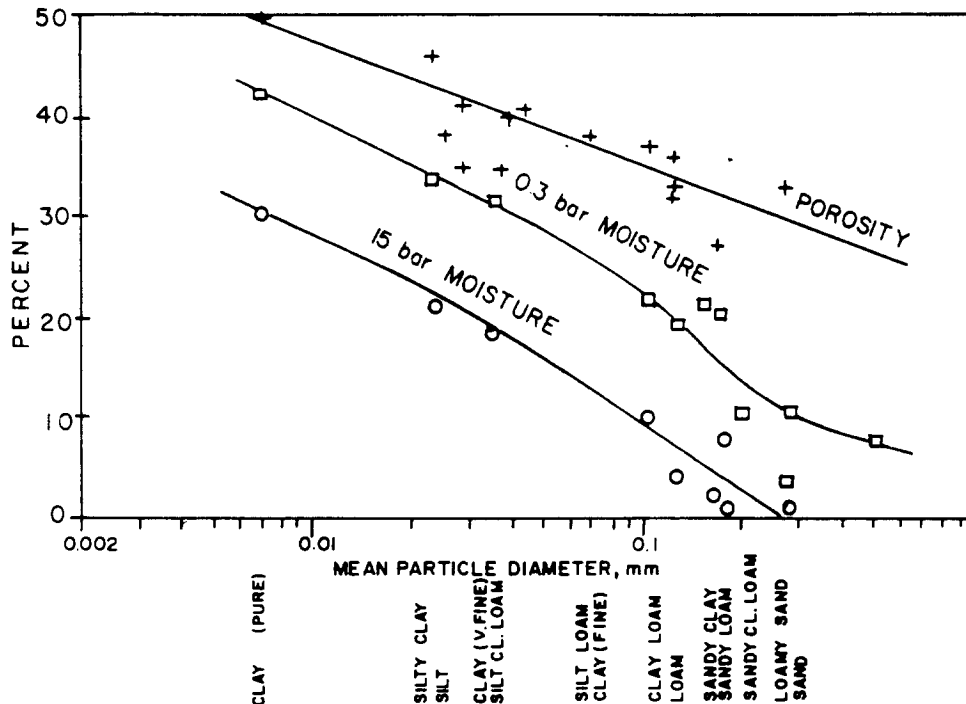


FIGURE 3.6. Moisture characteristics of soils.

Table 3.2 shows the permeability ranges for the hydrologic soil groups recognized by the U.S. Soil Conservation Service.

Soil Water Storage

Storage of soil moisture can be divided into two moisture classes: that held between saturation and 0.3-bar tension, and that held between 0.3-bar tension and 15-bar tension, respectively. The former moisture content is also called *field capacity*; moisture between the field capacity characteristics and full saturation can be drained by gravity. The 15-bar-tension moisture is called *wilting point* and represents the minimum soil moisture content that can be used by plants. Moisture content below 15-bar tension is not available to most crops and plants, and can be reduced only by evaporation (not by transpiration). Gravitational water (*G*) is then determined by subtracting 0.3-bar-moisture volume percentage from the total porosity (in percent). The plant available soil water capacity (*AWC*) is the difference between moisture content at 0.3-bar and 15-bar tensions, respectively. As long as the soil water content is between the field capacity and wilting point, transpiration is not affected. Prolonged saturation of soils may have an adverse effect on transpiration. Figure 3.6 shows the moisture characteristics related to the soil texture.

Infiltration

Infiltration is a function of the permeability of soils and subsoils, soil moisture content, vegetation cover, temperature, and possibly other parameters. During infiltration, water enters from surface storage into soils via the combined effects of gravity and capillary forces. The capillary forces are inversely proportional to the diameter of pores. As the process continues, the pore space becomes filled and the capillary tension decreases. Under saturated conditions, flow is mostly due to gravity.

The distribution of soil moisture within the soil profile during the downward movement of water is shown in Figure 3.7. Apparently, the downward movement of water is related to the advancement of the

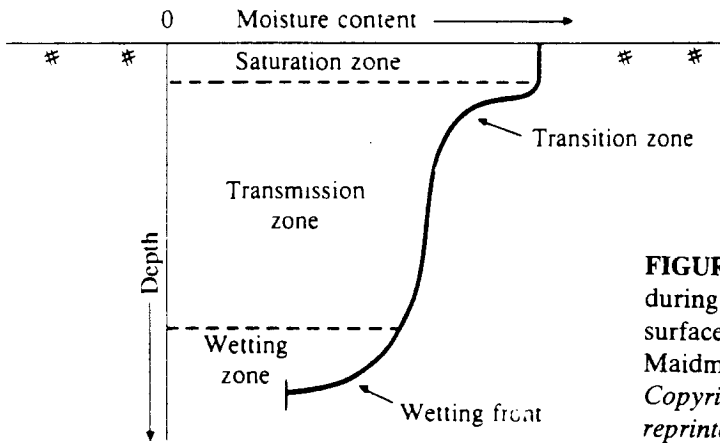


FIGURE 3.7. Soil moisture zones during infiltration from a ponded surface. (Concepts from Chow, Maidment, and Mays, 1988. Copyright © 1988 by McGraw-Hill; reprinted with permission.)

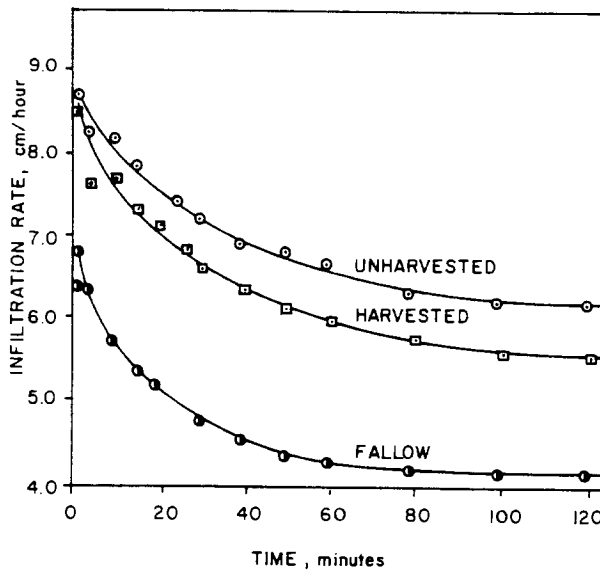


FIGURE 3.8. Characteristic relationship of infiltration rate vs. time of an agricultural field. (Data from Larsen, Axley, and Miller, 1972.)

wetting front, which is exhibited by a sharp and almost discontinuous difference between the advancing moisture driven by infiltration and the moisture below. As water percolates to greater depths, the resistance increases due to the increased length of the channels, decreased pore size from the swelling of clay particles, or the pressure on an impermeable barrier such as rock or clay. Consequently, the infiltration rate decreases as the time from the commencement of the storm increases, as shown on Figure 3.8. Depending on the depth of the soil column and the water supply from rainfall the wetting front may penetrate from a few centimeters to more than one meter into the soil (Hillel, 1980).

The infiltration rate f is expressed in cm/hr (in./hr). Potential infiltration is the maximum infiltration rate that presumes an excess supply of water at the surface, generally exhibited by ponding. Almost all infiltration formulas presented in the literature are for potential infiltration with ponding. Cumulative infiltration, F , is then obtained by integrating the infiltration rate over time, or $F(t) = \int_0^t f dt$.

Horton's Formula

Horton (1939a) more or less intuitively suggested an infiltration formula for exponentially decaying infiltration:

$$f = f_c + (f_0 - f_c)e^{-Kt} \quad (3.5)$$

where

f = rate of infiltration (cm/hr)

f_c = the infiltration rate assumed to be similar to the saturation permeability (cm/hr)

f_0 = the initial rate of infiltration (cm/hr)

K = a constant derived from soil and surface characteristics (hr^{-1})

t = time in hours from the beginning of infiltration

Although this equation at first appears to be a completely empirical model, it was pointed out that it does reflect the laws and basic equations of soil physics (Chow, Maidment, and Mays, 1988). The model assumes that the constant K is independent of the moisture content of the soil.

Holtan's Formula

Holtan (1961) proposed a formula that would relate the infiltration rate to the exhaustion of the available soil moisture storage. The formula was presented as follows:

$$f = a(S - F)^n + f_c = aF_p^n + f_c \quad (3.6)$$

TABLE 3.3 Estimates of the Vegetation Factor a in Holtan's Infiltration Equation for f in Centimeters per Hour and F_p in Centimeters

	Area Rating ^a	
	Poor Conditions	Good Conditions
Fallow ^b	0.07	0.2
Row crops	0.07	0.14
Small grains	0.14	0.20
Hay (legumes)	0.14	0.28
Hay (sod)	0.28	0.40
Pastures (bunchgrass)	0.14	0.28
Temporary pastures (sod)	0.28	0.40
Permanent pastures (sod)	0.55	0.68
Woods and forest	0.55	0.68

Source: From U.S. Department of Agriculture, Agricultural Research Service (1975).

^aAdjustments needed for "weeds" and "grazing."

^bFor fallow land only, "poor conditions" means "after row crop," and "good conditions" means "after sod."

where

S = the volume of soil water storage above the control horizon (cm)

F = cumulative infiltration (cm)

F_p = a measure of the soil moisture remaining in the soil column at any time (cm)

a, n = coefficients

The coefficients a and n were empirically determined. While the value of n was nearly constant and equaled $n \approx 1.4$, the value of the multiplier a was related to the crop cover as shown in Table 3.3.

Equation (3.6) was modified later and included in the USDAHL watershed model in the form (Holtan and Lopez, 1973):

$$f = GI a F_p^{1.4} + f_c \quad (3.6a)$$

where a is the vegetation parameter defined in Table 3.3 and GI is the index in a fraction of maturity. Information on the estimated magnitudes of the growth index for various crops and growing seasons is available from the U.S. Department of Agriculture publication by Holtan and Lopez (1973).

The depth of the control horizon is supposed to coincide with the topsoil zone between the soil surface and the depth of cultivation, or

subsoils for uncultivated soils. The latter corresponds to the thickness of the topsoil horizon A (see Chap. 6 for definition of soil horizons). Although Holtan's model is somewhat more complex than Horton's equation, it appears less physically based, since it relates the infiltration rate to the total moisture content in an arbitrarily chosen control layer and to the advancement of the wetting front in the unsaturated soil zone.

Example 3.1: Infiltration Rate by Holtan's Model

Estimate the infiltration rate curve for ponded soil with a saturation permeability of 2 cm/hr and soil moisture characteristics as follows:

porosity = 45%

0.3 bar moisture (field capacity) = 30%

15 bar moisture (wilting point) = 21%

The depth of the control horizon is assumed to be 50 cm and the antecedent moisture is equal to the field capacity (soil is drained). Then the initial available soil moisture storage capacity becomes

$$F_p(0) = (\text{porosity} - 0.13 \text{ bar moisture}) * 50 \text{ cm} = (0.45 - 0.3)50 = 7.5 \text{ cm}$$

Solution If the land surface is fallow, then the approximate magnitude of the vegetation parameter will be between 0.07 and 0.14. Select $a = 0.1$ and $GI = 1.0$. Then the initial infiltration rate at $t = 0$ from Equation (3.6a) becomes

$$f(0) = 0.1(7.5)^{1.4} + 2 = 3.68 \text{ cm/hr}$$

When $t \rightarrow \infty$ the infiltration rate would approach 2 cm/hr. However, due to the exponent n equaling 1.4 an exact solution of Equation (3.6a) is not possible. A simple numerical solution will be shown here. Since the infiltration rate is a function of the available water storage, a simultaneous solution of the storage equation must accompany the infiltration equation. Hence

$$\frac{dF_p}{dt} = f_r - f$$

where f_r = storage recovery rate.

If the storage recovery rate $f_r = f_c$ (which occurs when the permeability

of subsoils is more than or equal to the permeability of the top layer, and the moisture content is above the field capacity), then

$$\frac{dF_p}{dt} = aF_p^n$$

If the soil moisture content is below the field capacity value (0.3-bar moisture), then the recovery rate equals the evapotranspiration rate.

The preceding differential equation can be solved by simple numerical techniques such as Runge-Kuta, Euler, or Heund's methods. Heund's method would yield an equation

$$F_{p_{t+\Delta t}} = F_{p_t} \frac{\Delta t}{2} (a(F_{p_t})^n + a(F_{p_t} - a(F_{p_t})^n \Delta t)^n)$$

and the solution for $\Delta t = 1$ hr is given in Table 3.4 and plotted on Figure 3.9.

Philip's Equation

Philip's infiltration model (Philip, 1957, 1969, 1983) is based on soil physics. The model and its derivation is very complex; however, its simplified final version has been widely accepted and incorporated into several common watershed hydrological models. The simplified Philip's equation is in the form

$$f(t) = \frac{1}{2} S t^{-1/2} + K \quad (3.7)$$

TABLE 3.4 Solution of Holtan's Infiltration Equation

t (hr)	F_p (cm)	f (cm/hr)
0	7.5	3.68
1.0	5.94	3.11
2.0	4.89	2.92
3.0	3.84	2.66
4.0	3.26	2.52
5.0	2.79	2.42
6.0	2.41	2.34
8.0	1.89	2.24
10.0	1.49	2.17
15.0	0.93	2.09
20.0	0.62	2.05

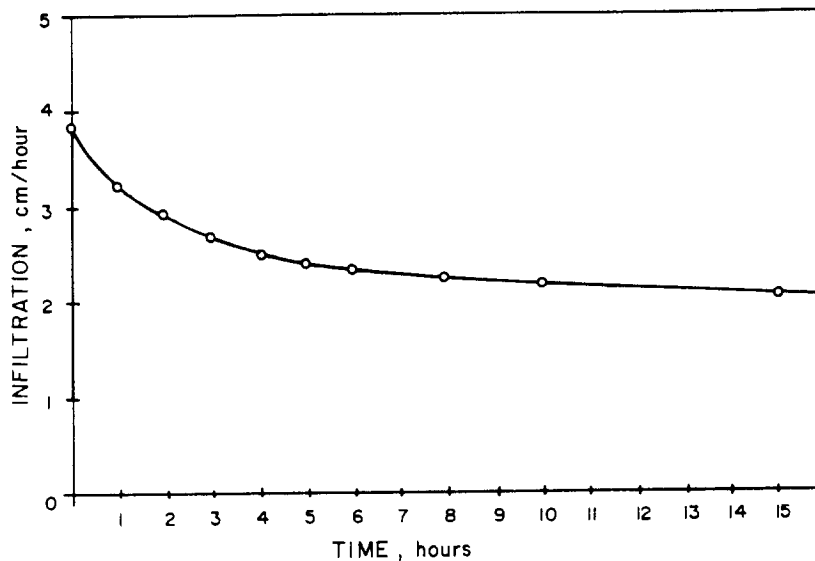


FIGURE 3.9. Infiltration rate curve for Example 3.1.

where

S = sorptivity of the soil

K = conductivity of the wetting front

Sorptivity is computed from soil moisture distribution and is generally difficult to define. As $t \rightarrow \infty$, f approaches K , which is loosely related to the saturation permeability parameter ($K = 0.3$ to 0.5 times saturation permeability). Cumulative infiltration is obtained by integrating Equation (3.7), which yields

$$F(t) = St^{1/2} + Kt \quad (3.8)$$

Since the sorptivity parameter, S , is a function of the soil suction potential, which is in turn related to the dryness of the soil, S will vary with the soil moisture content. Methods of measurement of sorptivity and its relation to the moisture content were presented by Chong and Green (1983); the schematics are shown on Figure 3.10. It was also demonstrated that the sorptivity and conductivity parameters are not uniform, even in small watersheds (Brutsaert, 1976; Sharma, Barron, and Boer, 1983). Bras (1990) included a good discussion and semiempirical formulas for determining the parameters S and K in the Philip's equation. The spatial distribution of infiltration rates, even in small watersheds, are of a statistical nature, probably ranging from close to zero to rates that greatly exceed the mean rate. The statistical distribution, such as is shown

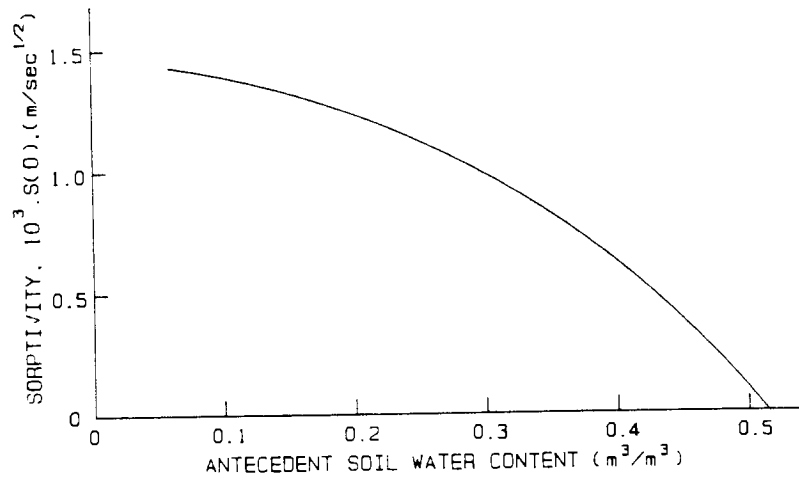


FIGURE 3.10. Relationship of soil sorptivity parameter, S , to soil moisture. (Based on data from Chong and Green, 1983.)

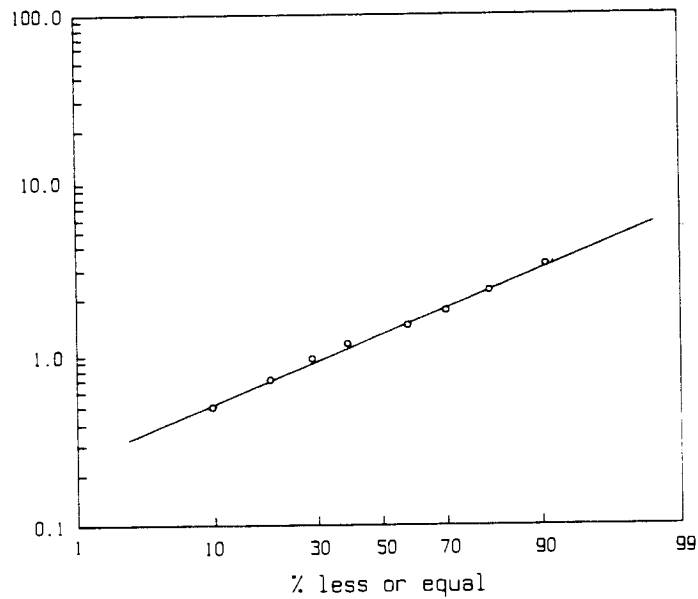


FIGURE 3.11. Typical statistical distribution of infiltration rates. The relation for a watershed is typically log-normal.

in Figure 3.11 (log-normal probability distribution), should be considered in the hydrologic mathematical models.

Green–Ampt Equation

The Green and Ampt (1911) infiltration model assumes that water is moving from a ponded surface downward into the soil as a piston of

saturated moisture. The concept follows the physics of soil water movement through the unsaturated zone shown on Figure 3.7; however, the model assumes saturated soil moisture at the wetting front. For a small ponding depth the Green and Amt equation becomes (Bras, 1990)

$$f = K_s + \frac{K_s S \Psi}{F} \quad (3.9)$$

where

K_s = hydraulic conductivity (which is less than the saturated permeability defined earlier)

Ψ = suction sorptivity

F = total infiltrated water that equals (porosity - initial moisture) * depth of the wetting front

The parameters K_s and Ψ usually have to be determined experimentally or by calibrating of the model. Rawls and Brakensiek (1983) provided procedures, graphs, and tables for determining these parameters from soil data.

Infiltration rates can be partially controlled by engineering and agricultural practices, such as tillage, raking of the surface, enrichment of soils and root systems of vegetation by organic residues, and chemical treatment of soils. Compaction by heavy machinery and by cattle will reduce the permeability and, hence, infiltration.

Infiltration into Frozen Soils

The physics of water movement into unsaturated soils at temperatures below the freezing point is not simple. The presence of ice in soils affects the soil water movement in two ways. First, permeability is reduced due to the reduction in pore size caused by ice crystals, which also cause a reduction in capillary suction. However, studies in Sweden (Lundin, 1989) pointed out several factors that must be considered. (1) Repeated freezing and thawing of clayey soils actually increases infiltration by creating more pore space due to the expansion of frozen water. (2) Water freezing in soils is not the same process as it is on the surface. Below freezing point temperatures, the interactions between the soil matrix and water result in free unfrozen water surrounding the soil particles. Romanov, Pavlova, and Kolyushnyy (1974) and others (Lundin, 1989) pointed out that unfrozen water content of about 10% to 16% may exist in soils at temperatures up to -20°C . The salt content of soil water further depresses the freezing point. The result is a gradual freezing of soil water, starting at temperatures that are below 0°C . Thus, the degree of freezing

point depression depends on the soil characteristics and salinity of soil water. Consequently, in many cases infiltration is reduced by the creation of an ice crust on the surface caused by ponding rather than by a reduction in infiltration rates. If ponding and frozen surface crust do not occur, the speed of infiltration into fine-textured dry soils during subzero temperatures is not drastically reduced when compared to that at above zero temperatures.

Kane (1980) and Romanov, Pavlova, and Kolyushnyy (1974) showed that the amount of infiltration of snowmelt for shallow snowpack (<1-m snow depth) depends on the melt rate, soil type, and soil moisture status at the time of freeze-up. Dry soils exhibit essentially the same or only slightly reduced infiltration rates (after accounting for increased viscosity when compared to those into unfrozen soils). Kane and Stein (1983) then found that the infiltration rate into a dry silt loam frozen soil at 0°C was about 50% of that at 15°C. The 50% decrease reflects the increased viscosity. However, the infiltration rate was reduced by two orders of magnitude when the soil was saturated with water. The authors in another paper (Kane and Stein, 1983b) then concluded that the infiltration rate into frozen soils is inversely proportional to the total moisture content and that the infiltration rate is controlled by the ice content in the upper few centimeters of the soil system.

Burt and Williams (1976) measured hydraulic conductivities of frozen soils. They also noted that frozen soils may contain considerable amounts of unfrozen water. The hydraulic conductivity of frozen soils is about two or more orders of magnitude smaller than that for unfrozen soils, but still enough to produce appreciable infiltration. The measured hydraulic conductivities for several soils ranged from 10^{-8} cm/sec for unlensed silt soil to 10^{-10} cm/sec for unlensed sand. The presence of the lenses reduces the permeability of frozen soils.

Romanov, Pavlova, and Kolyushnyy (1974) noted that an impermeable frozen water layer forms in frozen clayey and loamy frozen soils, when the moisture content reaches 50% to 60% while in sands, the impermeable layer is formed when the moisture reaches 87% to 97% of the total pore volume.

Evaporation and Transpiration

Unlike the previously mentioned losses (surface storage and infiltration), which are directly subtracted from rainfall to produce net (excess) rain and surface runoff, the direct effect of evapotranspiration on the magnitude of surface runoff is not great. A rainfall event usually implies high humidity, which depresses the evapotranspiration rates. On the

other hand, evapotranspiration determines the antecedent moisture conditions of soils and surface storage which, in turn, determine the magnitude of these losses. Hence the knowledge of evaporation and transpiration is important in estimating the rates of recoveries of surface storage and soil moisture storage capacity.

By definition, evapotranspiration represents water loss into the atmosphere by evaporation from both open water surface and soils, while transpiration refers to water drawn from the soil zone by the root systems of plants and vegetation and released to the atmosphere as a part of the life cycle of plants.

Evaporation

Potential evaporation is either measured or computed. Pan evaporation (measured) data are available from the U.S. Weather Service of the National Oceanic and Atmospheric Administration (NOAA). It must be realized that pan evaporation data may differ from the actual evaporation from larger water bodies and soil surfaces. Typically, pan evaporation measurements yield evaporation rates that are 20% to 40% higher than actual evaporation from land or water surfaces (Linsley, Kohler, and Paulhus, 1982).

The models used for estimating potential evaporation are either in the category of energy balance or aerodynamics. The energy balance estimates heat lost from the system by evaporation (about 590 calories are needed to convert 1 gram of liquid water into vapor). This high heat loss is balanced by other heat inputs and changes of temperature within the system. Using the energy balance equation requires measurements of several heat and energy inputs and changes of temperature. Some of these measurements may not be readily available.

The aerodynamic models essentially describe mass transfer across the air-water interface. The rate of vapor transfer by evaporation or condensation is proportional to the difference in vapor content (pressure) in the ambient air above the air-water boundary interface and the saturation vapor pressure in the air immediately at the water's surface. The rate of transfer depends on the degree of turbulence and mixing in the thin boundary layer just above the water's surface. The mixing rate can be determined empirically by relating it to the wind velocity above the surface and other parameters. Over 100 empirical and semiempirical formulas have been published in the literature (Helfrich et al., 1982; Bras, 1990).

A majority of the evaporation formulas are of the type (Bras, 1990):

$$E_v = \frac{Q_e}{L\rho} = \frac{(a + bU)(e_s - e_a)}{L\rho} \quad (3.10)$$

TABLE 3.5 Relation of Saturation Vapor Pressure to Temperature

Temperature (°C)	Saturation Vapor Pressure (e_s , millibars ^a)
-10	2.86
0	6.11
5	8.72
10	12.27
15	17.00
20	23.37
25	31.70
30	42.43

Source: From Linsley, Kohler, and Paulhus (1982).
(Copyright © 1982 by McGraw-Hill; reprinted by permission.)

^aOne millibar = 100 N/m².

TABLE 3.6 Coefficients for Evaporation Rate Formula (Eq. (3.10))

Author or Measurement	a	b
Lake Hefner (Marciano and Harbeck, 1954) (water surface)	0	8.1
Harbeck (1962) for several lakes (water surface) ^a	0	$10.44 \times A^{-0.05}$
Penman (quoted in Priestley, 1959) (land surface)	0	5.87
Zaikov (from Braslavskii and Vikulina, 1963) (water surfaces)	6.45	4.64

^aA = surface area of the lake in hectares.

where

E_v = evaporation rate (cm/day)

Q_e = heat loss by evaporation or gain by condensation (cal/cm²-day)

L = $597.3 - 0.57T$ = latent heat of vaporization (cal/gram of water)

ρ = specific density of water (= 1 g/cm³)

U = wind speed in m/sec measured 2 m above the surface

T = temperature (°C)

e_s = saturation vapor pressure in millibars for the water surface temperature (Table 3.5)

e_a = ambient vapor pressure in millibars measured 2 m above the surface

a, b = coefficients given in Table 3.6

Note that 1 calorie is equivalent to 4.186 joules.

The saturation vapor pressure, e_s , which is a measure of the maximal content of vapor in the air, is related to the temperature as given in Table 3.5.

Example 3.2: Calculation of Evaporation

Compare evaporation rates estimated using the formulas of Zaikov (obtained from large reservoirs in the former USSR) and Harbeck for U.S. lakes. The following daily average values are given:

wind velocity measured at 2 meters above the surface $U = 2.5$ m/sec

relative humidity $r = 50\%$

ambient air temperature $T_a = 25^\circ\text{C}$

water temperature $T_s = 20^\circ\text{C}$

surface area of the lake $A = 500$ ha

Relative humidity is a ratio of the actual humidity of the air to the saturation value. At the ambient air temperature of 25°C the saturation humidity of the air is (Table 3.5) $e_{sa} = 31.7$ mbar and the ambient humidity is $e_a = r \times e_{sa} = 0.5 \times 31.7 = 15.85$ mbar. The saturation vapor pressure at the water's surface with a temperature of 20°C is 23.37 mbar.

Solution The latent heat of vaporization at the water's surface with a temperature of 20°C is

$$L = 597.3 - 0.57 * 20 = 585.9 \text{ cal/g}$$

Zaikov

$$E_v = \frac{(6.45 + 4.64 \times 2.5)(23.37 - 15.85)}{585.9 \times 1} = 0.23 \text{ cm/day}$$

Harbeck

$$E_v = \frac{(0 + 10.44 \times 500^{-0.05} \times 2.5)(23.37 - 15.85)}{585.9 \times 1} = 0.25 \text{ cm/day}$$

Transpiration and Evapotranspiration

Several methods have been developed for estimating the evapotranspiration requirements of crops and forested areas. As defined previously, evapotranspiration is a composite of transpiration by plants and crops and evaporation from the soil.

The actual rate of transpiration is a function of the type and growth stage of the crops, the soil moisture (below or above wilting point), and climatic conditions. Similar factors affect both transpiration and soil evaporation; therefore, evaporation data are sometimes used as a surrogate for the potential evapotranspiration rate. Evapotranspiration can be related to the potential evaporation by a parameter or a function such as that in the *Evaporation Index Method* described by McDaniel (1960):

$$ET = E_p \times KU \quad (3.11)$$

where

ET = crop evapotranspiration requirement on a monthly or shorter period basis

E_p = climatic index, which is identical to the evaporation potential from a shallow hypothetical lake situated at the locality under consideration

KU = crop-use coefficient, which reflects the growth and stage of crops. Average values for crop-use coefficients are presented in Table 3.7

A similar approach has been suggested by the USDA–Agricultural Research Service (Holtan, 1961), which proposed the following equation for the potential evapotranspiration:

$$ET = KU \times (E_p/k) \times \beta \quad (3.12)$$

where

ET = evapotranspiration potential (cm/hour or in./hr)

KU = growth index or crop-use coefficient defined previously and given in Table 3.7

k = ratio of ET to pan evaporation, usually 1.0 to 1.2 for short grasses, 1.2 to 1.6 for crops up to shoulder height, and 1.6 to 2.0 for forests. If evaporation is calculated, $k = 1.0$

E_p = calculated or measured (pan) evaporation potential (cm/hr or in./hr)

β = a moisture stress coefficient that expresses the reduction of evapotranspiration to plain evaporation at or below wilting point soil moisture

$\beta = 1.0$ if $\theta \geq \theta_{15\text{bar}}$ and

$\beta = \left(\frac{\theta}{\theta_{15\text{bar}}} \right)^{AWC/G}$ if $\theta < \theta_{15\text{bar}}$

TABLE 3.7 Crop-Use Coefficients for Use in Evaporation Index Method

Perennial Crops (Northern Hemisphere)												
Average KU values by months												
Crop	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
Alfalfa	0.83	0.90	0.96	1.02	1.08	1.14	1.20	1.25	1.22	1.18	1.12	0.86
Grass pasture	1.16	1.23	1.19	1.09	0.95	0.83	0.79	0.80	0.91	1.91	0.83	0.69
Grapes	—	—	0.15	0.50	0.80	0.70	0.45	—	—	—	—	—
Citrus orchards	0.58	0.53	0.65	0.74	0.73	0.70	0.81	0.96	1.08	1.03	0.82	0.65
Deciduous orchards	—	—	—	0.60	0.80	0.90	0.90	0.80	1.50	0.20	0.20	—
Sugarcane	0.65	0.50	0.80	1.17	1.21	1.22	1.23	1.24	1.26	1.27	1.28	0.80

Annual Crops												
KU values at listed percent of growing season												
Crop	0	10	20	30	40	50	60	70	80	90	100	
Field corn	0.45	0.51	0.58	0.66	0.75	0.85	0.96	1.08	1.20	1.08	0.70	
Grain sorghum	0.30	0.40	0.65	0.90	1.10	1.20	1.10	0.95	0.80	0.65	0.50	
Winter wheat ^a	1.08	1.19	1.29	1.35	1.40	1.38	1.36	1.23	1.10	0.75	0.40	
Cotton	0.40	0.45	0.56	0.76	1.00	1.14	1.19	1.11	1.83	0.58	0.40	
Sugar beets	0.30	0.35	0.41	0.56	0.73	0.90	1.08	1.26	1.44	1.30	1.10	
Cantaloupes	0.30	0.30	0.32	0.35	0.46	0.70	1.05	1.22	1.13	0.82	0.44	
Potatoes (Irish)	0.30	0.40	0.62	0.87	1.06	1.24	1.40	1.50	1.50	1.40	1.26	
Papago peas	0.30	0.40	0.66	0.89	1.04	1.16	1.26	1.25	0.63	0.28	0.16	
Beans	0.30	0.35	0.58	1.05	1.07	0.94	0.80	0.66	0.53	0.43	0.36	
Rice ^b	1.00	1.06	1.13	1.24	1.38	1.55	1.58	1.57	1.47	1.27	1.00	

Source: From *Handbook of Applied Hydraulics*, by Davis and Sorensen (1969). Copyright © 1969 by McGraw-Hill. Used with the permission of the McGraw-Hill Book Company.

^a Data given only for springtime season of 70 days prior to harvest (after last frost).

^b Evapotranspiration only.

θ = soil moisture

AWC = $\theta_{0.3\text{bar}} - \theta_{15\text{bar}}$ = plant available soil moisture capacity

G = $S - \theta_{0.3\text{bar}}$ = gravitational soil water moisture

Other methods for estimating evapotranspiration include the Blaney-Cridle, and the modified Penman and Lowry-Johnson methods. The reader is referred to standard hydrology texts (Bras, 1990; Chow, 1964; Davis and Sorensen, 1969; Gray, 1973; Chow, Maidment, and Mays, 1988) for further reference.

Snowpack Formation and Snowmelt

Knowledge of snow hydrology is important in diffuse-pollution studies. Snow, which in the northern latitudes of North America and Europe may stay on the ground for several months, is a surface trap of pollutants deposited from the atmosphere. In addition, by the process of repeated crystallization and the melting of snow in the snowpack, dissolved pollutants are rejected by the crystals and become available for fast pickup by melt water, resulting in a strong "first flush" effect. Snow precipitation has much less energy than rain droplets. Consequently, the erosion of soils during snowfall or snowmelt is minimal.

The main elements determining the amount of snowmelt are meteorological factors, such as temperature, solar radiation, and wind velocity. An accurate determination of snowpack accumulation and subsequent snowmelt is not simple. For example, in urban areas in addition to meteorological inputs, the heat balance of the snowpack and, hence, the quantity of snowmelt, are also affected by heat losses from buildings and traffic, by deicing operations (street salting), coloration of the snowpack, and by snow removal and dumping.

In the most simple concept, the volume of snowmelt from a snowpack is related to the ambient temperature deviation from a reference equilibrium temperature such as in the following degree-day formula

$$\begin{aligned} \Delta P_s &= -SM = CD \times (T_a - T_r) && \text{for } T_a > T_r \\ \Delta P_s &= P && \text{for } T_a \leq T_r \end{aligned} \quad (3.13)$$

where

ΔP_s = change of water equivalent in the snowpack (cm/day)

SM = snowmelt (cm/day)

P = water equivalent in precipitation (cm/day)

T_a = ambient temperature (°C)

T_r = reference equilibrium temperature ($^{\circ}\text{C}$)
 CD = a proportionality coefficient (also called a degree-day coefficient)
 ($\text{cm}/^{\circ}\text{C}\text{-day}$)

In most technical applications, the equilibrium reference temperature, T_r , is assumed to be 0°C . The proportionality (degree-day) coefficient, CD, is an empirical quantity loosely correlated to the meteorological conditions of the season. Empirical values of the degree-day coefficient reported in the literature ranged from 0.1 to $1\text{ cm}/^{\circ}\text{C}\text{-day}$, with values between 0.2 and $0.5\text{ cm}/^{\circ}\text{C}\text{-day}$ the most common (Bengtsson, 1982; Westerström, 1984; Ellena and Novotny, 1985). The simple degree-day coefficient is also assumed to vary with the season, as shown on Figure 3.12. Due to its simplified nature, the degree-day model has been subjected to critique and modifications. For example, Bengtsson (1984) added a solar radiation component, which, as documented by Ellena and Novotny (1985), has markedly improved the performance of the model.

Advanced models for snowmelt estimation use some kind of the energy balance method. The most comprehensive study of the process of melting and components of the snowpack energy (heat) balance was undertaken by the U.S. Army Corps of Engineers (1956). Anderson and Crawford (1964) incorporated this concept into the Stanford Watershed Model, which is a predecessor to the HSP-F watershed model distributed by the U.S. Environmental Protection Agency (see Chapter 8 for a description of the model).

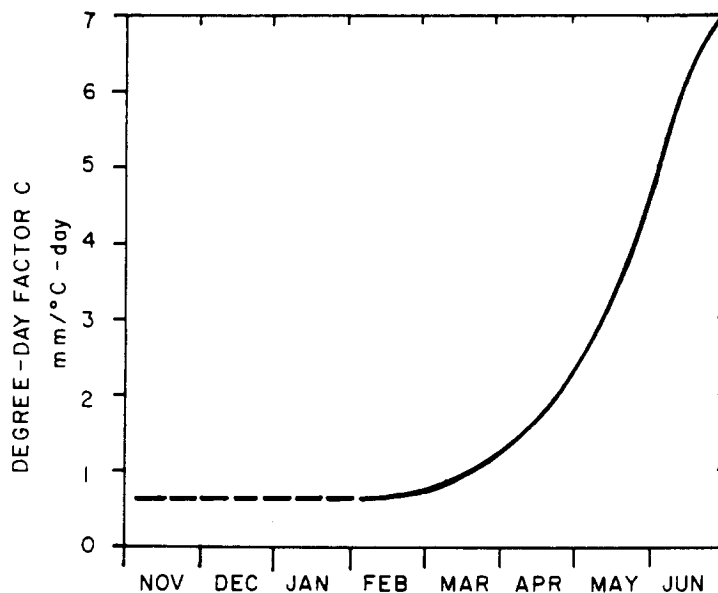


FIGURE 3.12. Degree-day factor for snowmelt computations. (From Gray, 1973.)

The energy balance concept is based on the fundamentals of the physics of melting. The heat balance equation for the snowpack can be written as follows (U.S. Army Corps of Engineers, 1956; Ellena and Novotny, 1985; Novotny, 1988):

$$\Delta Q = Q_{SAB} + Q_{ATM} + Q_E + Q_P + Q_C + Q_G \quad (3.14)$$

where

ΔQ = net change of the heat content of the snowpack

Q_{SAB} = absorbed (net) direct solar radiation

Q_{ATM} = net atmospheric (long wave) radiation

Q_E = latent heat transfer due to the condensation–evaporation–sublimation process, which in the absence of direct measurements, can be estimated by Equation (3.10)

Q_C = sensible heat transfer between air and snow

Q_P = heat content of precipitation

Q_G = heat gain or loss due to other sources or sinks (in urban areas it also includes heat gains from vehicles, as well as from exothermic dissolution of salt for deicing)

The units of the energy (heat) balance equations are calories $\text{cm}^{-2} \text{hr}^{-1}$ or joules m^{-2} . At the melting point, the heat content of the snowpack, Q_m , is

$$Q_m = T_m c_p \rho_s D \quad (3.15)$$

where

T_m = the melting point temperature ($^{\circ}\text{C}$)

c_p = specific heat calories/g or W/kg

ρ_s = specific density (g/cm^3 or kg/m^3)

D = snow depth in centimeters or meters

If, due to increased heat input, the energy (heat) content of the snowpack exceeds the energy needed to melt the snow (Q_m), 1 g of water will melt for each 79.7 calories of the heat excess, or

$$SM = \frac{Q - Q_m}{79.7 Q_t} \quad (3.16)$$

where

SM = resulting melt in centimeters

Q_t = thermal quality of the snowpack (≈ 1.0)

The melting point temperature is a function of salinity or the molar concentration of salts in the water (snow) solution. The melting point is lowered by approximately 0.7°C for each 10 g of salt added to 1 liter of water equivalent of snow. For salt-free snow $T_m = 0^{\circ}\text{C}$, therefore, $Q_m = 0$. Furthermore, salt dissolution is an exothermic process that is heat generated by the dissolution of salt.

The magnitude and models of each component of the snowpack energy balance have been described in detail by Novotny (1985), Ellena and Novotny (1985), U.S. Army Corps of Engineers (1956), and others. As most of the energy components vary throughout the day, the snowmelt rates are also variable, so it may be inappropriate to use average daily values.

Of note is the approximate magnitude of snowmelt rates. Assuming a clear, sunny day in late February, the average incoming solar radiation at latitude 46° is about $20 \text{ cal/cm}^2\text{-hr}$. During clear, dry conditions with the ambient air at 10°C , the net radiation heat loss and condensation heat gain are about equal. The albedo (reflectivity of direct solar radiation) of urban snow varies between 20% for dirty saturated snow to more than 85% for freshly fallen snow. From Equation (3.16) it follows that the maximum runoff rate from melting snow is only about 0.15 cm/hr . This is much lower than runoff rates resulting from typical design storms, and lower than most potential rates of infiltration into dry or moderately wet frozen soils. Hence, surface runoff generation from snowpack over a permeable soil is generally possible only if a frozen layer of solid ice is formed on the surface.

RAINFALL EXCESS DETERMINATION: SURFACE RUNOFF

Design Storm

In most designs of hydrologic systems the designer is faced with the task of selecting the so-called *design storm*. This type of input information is especially important for flood-mitigation projects. It will be pointed out throughout this book, however, that the abatement of diffuse pollution should be focused on precipitation events that are frequent, typically medium magnitude storms with rainfall depths ranging from 1.2 to 3.5 cm (0.5 to 1.5 in.), which would occur several times each year rather than rare large storms.

The isopluvial depths (isohyets) of a typical medium-size one-hour storm with a recurrence time of once per year are shown on Figure 3.13. Such maps for design storms in the United States are prepared by the

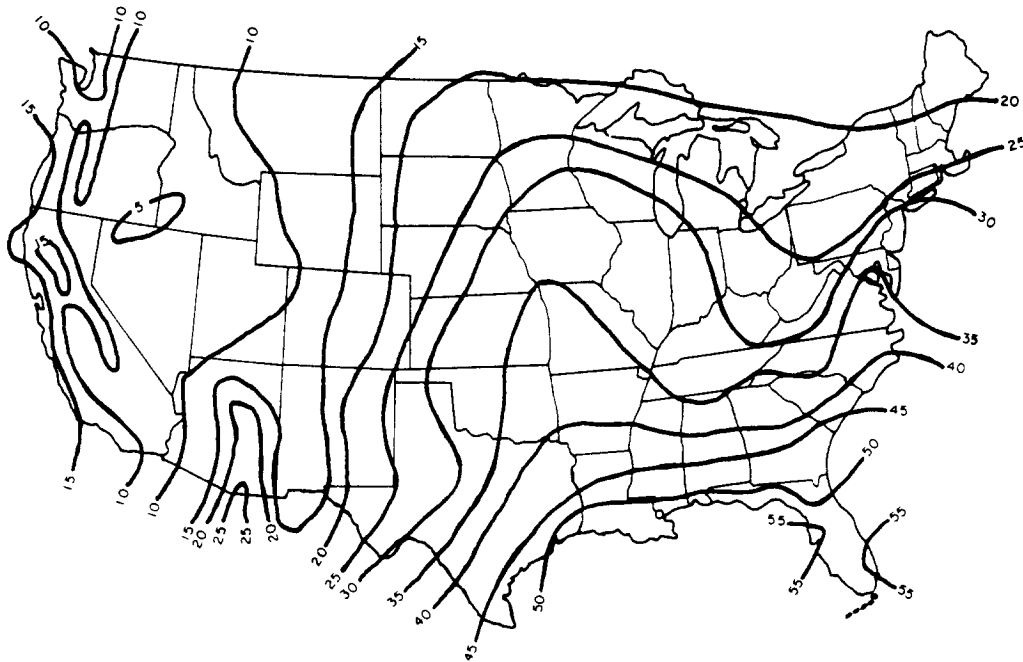


FIGURE 3.13. Isopluvial map of the United States for once-per-year, 1-hr-long rainfall in millimeters, published by the U.S. Weather Service (to convert from mm/hr to l/sec-ha, multiply by 2.78, 1 in. = 25.4 mm).

National Weather Service. Figure 3.14a is a normalized duration-intensity curve. It has been found (first by Karl Imhoff in the 1920s, see Novotny et al., 1989) that the relationship between the average intensity within a storm and the storm duration are similar for most of European and U.S. locations, with the exception of Pacific coastal areas and Hawaii. If the duration-intensity curves are normalized by the one-hour, once-per-year storm depths, a unified duration-intensity curve is obtained. Figure 3.14b is then used to modify the design storm for different recurrence intervals. The use of Figures 3.13 and 3.14 is illustrated on the following example.

Example 3.3: Selection of the Design Storm

Determine the intensity of a twice-a-year design that has a duration of 30 minutes. The watershed is located in the Chicago, Illinois, metropolitan area.

Solution Read the intensity of the standard 1-year, 1-hr-duration storm from the isopluvial map in Figure 3.13. Therein $r_1^1 = 32.5$ mm/hr. To convert this 1-year, 1-hr storm to the desired $\frac{1}{2}$ -year, 30-min design storm, first read the magnitude of the duration multiplier from the upper portion of Figure 3.14. For Chicago $\phi_1 = 1.4$. Then from the lower portion of

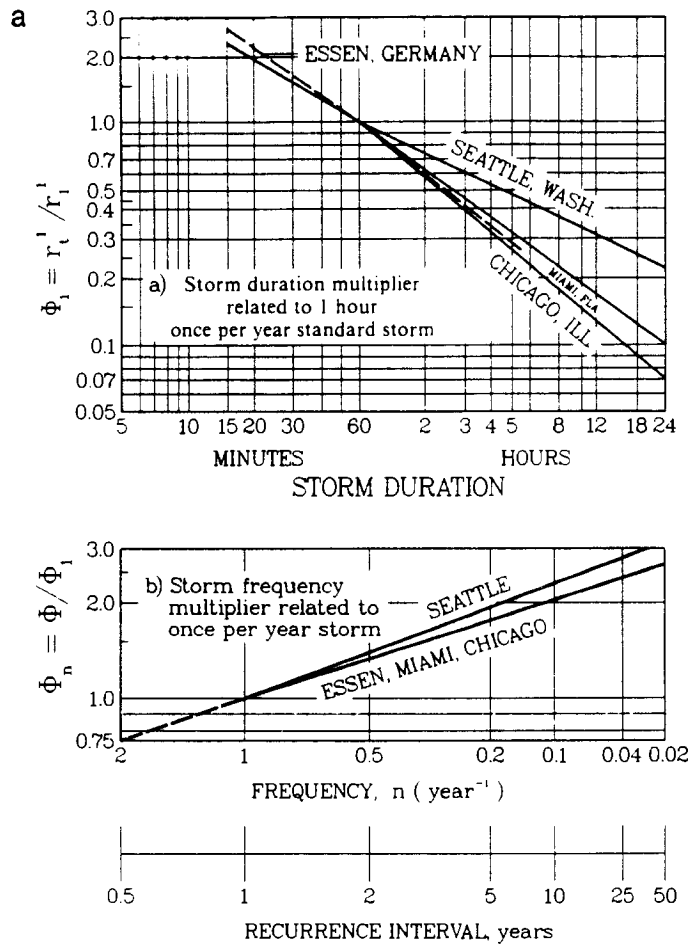


FIGURE 3.14. Normalized storm-duration–frequency–intensity curves for several geographical locations. (From Novotny et al., 1989. Copyright © 1989; reprinted by permission of John Wiley & Sons, Inc.)

Figure 3.14 read the frequency multiplier, ϕ_n , which is $\phi^n = 0.75$. Hence the average intensity of a design storm of 30-minutes duration that occurs approximately twice a year, is

$$I = \phi_1 \phi_n r_1^{-1} = 1.4 \times 0.75 \times 32.5 = 34.12 \text{ mm/hr}$$

Rainfall Excess from Pervious Areas

Rainfall Excess by Subtracting Losses from Precipitation

From the foregoing discussion and referring to Equations (3.1) to (3.3), it is now clear that surface runoff, which is often the most polluted

component of the total runoff, will be generated from a surface only when precipitation or snowmelt is greater than the losses, that is,

$$R_s = P - S_i - S_d - f\Delta t$$

if

$$P > S_i + S_d + f\Delta t$$

and

$$R_s = 0 \quad \text{for} \quad P \leq S_i + S_d + f\Delta t \quad (3.17)$$

The term *rainfall excess* or *net rain* is used to denote the simple numerical subtraction of the losses from the precipitation volume. This differentiates it from surface runoff, which refers to that part of the flow in the receiving body of water that was generated by rainfall excess. The unit for rainfall excess is depth of water on the surface from the excess rain generated during a time interval, while the unit for surface runoff is volume/time. A time lag between maximum rain excess and the peak of the surface runoff is typical for all but very small drainage areas. This time lag (called *peak time*) is due to overland and channel flow routing.

There are several procedures for estimating rainfall excess. The definition of excess rainfall excludes the use of the formulas that are based on a proportionality between the rainfall and runoff, such as the well-known rational formula presented in the next section. Two methods presented here are: (1) numerical subtraction of losses from precipitation, and (2) the Soil Conservation Service Runoff Curve Method. Any infiltration formula can be used for estimating infiltration losses; however, Holtan's infiltration equation is used in Example 3.4.

Example 3.4: Numerical Rainfall Excess Determination by Subtracting Losses

Determine the rainfall excess from a storm with the hyetograph given in the top portion of Figure 3.15. A dry period preceding the storm lasted 5 days. The evaporation rate during the dry period averaged 0.3 cm/day. The area is covered by small grains and has an average slope of 2%. For this slope and surface, the combined depression and interception storage can be read from Figure 3.3 as 0.62 cm. The soil and infiltration characteristics are similar to Example 3.1. Gravitational moisture content (field capacity) is $G = 50$ (porosity - 0.3-bar moisture) = 7.5 cm.

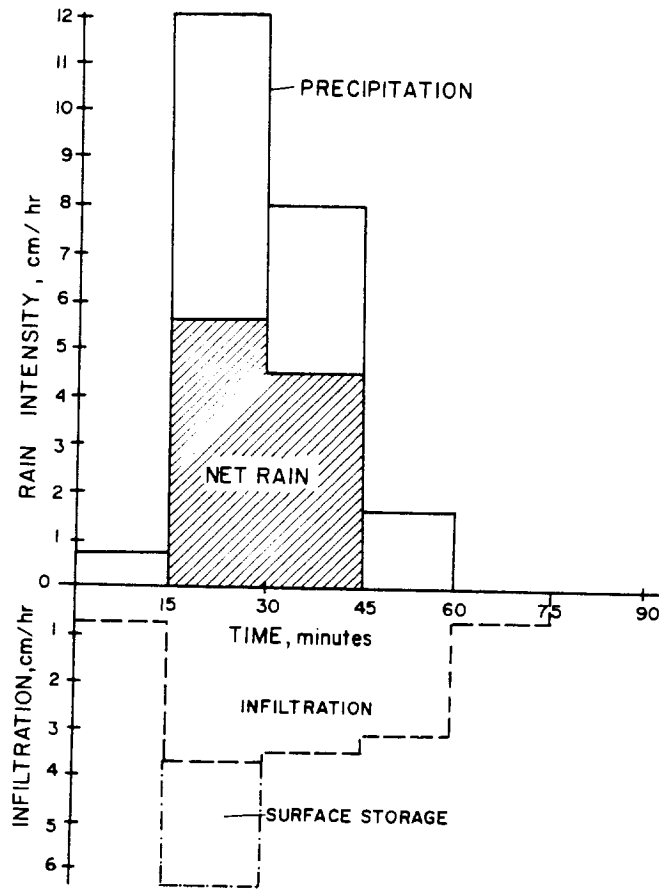


FIGURE 3.15. Rainfall hyetograph, subtractions, and excess rainfall for Example 3.4.

Solution The initial soil moisture and available storage capacity can be computed, assuming that the soil was saturated during the preceding rain and that the crop use factor, KU , is close to one. Since the saturation permeability is 2 cm/hr, the soil moisture reached 0.3-bar moisture shortly after the preceding rain. Hence, the initial available storage capacity becomes

$$F_p(0) = G + ET \times 5 \text{ days} = 7.5 + 0.3 \times 5 = 9.5 \text{ cm}$$

The parameter a for Holtan's infiltration equation can be read from Table 3.3. For small grains the parameter is between 0.07 and 0.14. Select $a = 0.1$. Then for $F_p(0) = 9 \text{ cm}$, the initial infiltration rate is

$$f(0) = 0.1(9.0)^{1.4} + 2 = 4.17 \text{ cm/hr}$$

The excess rain is computed in 15-min time intervals by the simultaneous solution of the following equation (based on Eq. (3.1))

$$R_s = P - S_{id} - f\Delta t$$

$$\text{if } P > S_{id} + f\Delta t \text{ then } S_{id} = 0 \quad \Delta F_p = (f_r - f)\Delta t \quad (3.18)$$

and

$$R_s = 0 \quad \text{if } P \leq S_{id} + f\Delta t$$

with

$$-\Delta S_{id} = P - f\Delta t$$

$$f\Delta t = P + \Delta S_{id} \quad \text{if } (aF_p^{1.4} + f_c) > (P + \Delta S_{id})/\Delta t$$

$$\Delta F_p = (f_r - f)\Delta t$$

and $f_r = 0$, if soil moisture is below 0.3-bar tension (of $F_p > G$).

During the rainy period the recovery of surface and soil water storage by evapotranspiration is minimal. The computation and results are given in Table 3.8 and Figure 3.15.

Rainfall Excess by the Soil Conservation Method

The Soil Conservation Service has developed a method for estimating rainfall excess that does not require computing infiltration and surface storage separately. Both runoff characteristics are included as just one watershed characteristic. The method has evolved from analysis of numerous storms under a variety of soil and cover conditions.

In the SCS method the excess rain volume, Q , depends on the volume of precipitation, P , and the volume of total storage, S , which includes both the initial abstraction and total infiltration, I_a . The relationship between rainfall excess and total rainfall (on a 24-hour basis) is thus (SCS, 1968; McCuen, 1982)

$$Q = \frac{(P - I_a)^2}{(P - I_a) + S} \quad (3.19)$$

The initial subtraction operation is a function of land use, treatment, and condition; interception; infiltration; depression storage; and antecedent soil moisture. An empirical statistical relationship relates the initial subtraction to the total storage as

$$I_a = 0.2S \quad (3.20)$$

TABLE 3.8 Rainfall Excess Calculation

Time Interval (min)	Precipitation Intensity during the Time Interval (cm/hr)	Precipitation Volume, P (cm)	Available Depression and Interception Storage, S_{id} (cm)	ΔS_{id}^a (cm)	Infiltration Volume ^b (cm)	Soil Moisture Storage Capacity, F_p^c (cm)	Excess Rain Volume, R_s (cm)	Net Rain Intensity, R_s/t (cm/hr)
(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
0-15	3.0	0.75	0.62	0.0	0.75	9.0	0.0	0.0
			0.62			8.25		
15-30	12.0	3.00	0.0	-0.62	0.96	7.40	1.42	5.68
30-45	8.0	2.00	0.0	0.0	0.90	7.00	1.10	4.40
45-60	1.6	0.40	0.47	+0.47	0.87	6.63	0.00	0.00
60-75	0.0	0.0	0.62	+0.15	0.15	6.98	0.00	0.00

^a $\Delta S_{id} = f\Delta t - P$, $\min S_{id} = 0$, $\max S_{id} = 0.62$ cm.

^b Infiltration volume = $f\Delta t$ or = $P + S_{id}$, whichever is less.

^c $\Delta F_p = (f_r - f) \Delta t$, $f_r = f_c$ if $F_p < G$, $f_r = 0$ if $F_p > G$; rate of exhaustion of the soil moisture storage capacity reduced at $F_p = 7.5$ cm.

Substituting Equation (3.20) into (3.19) yields

$$Q = \frac{(P - 0.2S)^2}{(P + 0.8S)} \quad (3.21)$$

The cumulative infiltration is also

$$F = (P - I_a) - Q$$

and, after substituting from Equation (3.19),

$$F = \frac{(P + I_a) \times S}{(P + I_a) + S} \quad (3.22)$$

Note that Equation (3.21) has been reduced to only one unknown, the storage parameter S . This parameter (in millimeters) can be obtained from

$$S = \frac{25,400}{CN} - 254 \quad (3.23)$$

where CN is the runoff curve number that can be obtained from Table 3.9. All parameters in Equations (3.19)–(3.23) are in millimeters; P and Q represent daily precipitation and runoff volumes.

The relation between excess rain (runoff) volume, precipitation, and storage for different runoff curve numbers is plotted on Figure 3.16. The watershed soil moisture conditions for determining the runoff curve numbers have been classified by the SCS as follows:

AMC I: A condition of watershed soils where the soils are dry but not to the wilting point, and when satisfactory plowing or cultivation takes place.

AMC II: The average case for annual floods, that is, an average of the conditions that have preceded the occurrence of the annual flood on numerous watersheds.

AMC III: If heavy rainfall or light rainfall and low temperatures have occurred during the 5 days prior to the given storm and the soil is nearly saturated.

Table 3.9 gives the curve numbers for the average antecedent soil moisture AMC II. The corresponding curve numbers for AMC I and

TABLE 3.9 Runoff Curve Numbers for Hydrologic Soil Cover Complexes (Antecedent Soil Moisture Conditions AMC = II)

Land-Use Description and Cover		Hydrologic Soil Groups				
		A	B	C	D	
Residential^a						
Average Lot Size	Average Imperviousness ^b (%)					
0.05 ha (1/8 acre)	65	77	85	90	92	
0.10 ha (1/4 acre)	38	61	75	83	87	
0.15 ha (1/3 acre)	30	57	72	81	86	
0.20 ha (1/2 acre)	25	54	70	80	85	
0.4 ha (1 acre)	20	51	68	79	84	
Paved parking lots, driveways, etc. ^c		98	98	98	98	
Streets and roads						
Paved, with curbs and storm sewers		98	98	98	98	
Gravel		76	85	89	91	
Dirt		72	82	87	89	
Commercial and business	85 (average)	89	92	94	95	
Industrial districts	72	81	88	91	93	
Open spaces, lawns, golf courses, cemeteries, etc.						
Good condition, grass cover on 75% or more of the area		39	61	74	80	
Fair conditions, grass cover on 50% to 75% of the area		49	69	79	84	
		Hydrologic Conditions				
Fallow	Straight row	—	77	86	91	94
Row crops	Straight row	Poor	72	81	88	91
	Straight row	Good	67	78	85	89
	Contoured	Poor	70	79	84	88
	Contoured	Good	65	75	82	86
	Contoured and terraced	Poor	66	74	80	82
	Contoured and terraced	Good	62	71	78	81
Small grain	Straight row	Poor	65	76	84	88
	Straight row	Good	65	75	83	87
	Contoured	Poor	63	74	82	85
	Contoured and terraced	Poor	61	72	79	87
	Contoured and terraced	Good	59	70	78	81
Close-seeded legumes ^d or rotational meadow	Straight row	Poor	66	77	85	89
	Straight row	Good	58	72	81	85
	Contoured	Poor	64	75	83	85
	Contoured	Good	55	69	78	83
	Contoured and terraced	Poor	63	73	80	83
	Contoured and terraced	Good	51	67	76	80

TABLE 3.9 Continued

Land-Use Description and Cover		Hydrologic Soil Groups			
		A	B	C	D
Pasture or range	Poor	68	79	86	89
	Fair	49	69	79	84
	Good	39	61	74	80
Contoured	Poor	47	67	81	88
	Fair	25	59	75	83
	Good	6	35	70	79
Meadow	Good	30	58	71	78
Woods or forest land	Poor	45	66	77	83
	Fair	36	60	73	79
	Good	25	55	70	77
Farmsteads	—	59	74	82	86

Source: After Soil Conservation Service (1968).

^a Curve numbers are computed assuming the runoff from the house and driveway is directed toward the street with a minimum of roof water directed to lawns where additional infiltration could occur.

^b The remaining pervious areas (lawns) are considered to be in good pasture condition for these curve numbers.

^c In some warmer parts of the country a curve number of 95 may be used.

^d Close-drilled or broadcast.

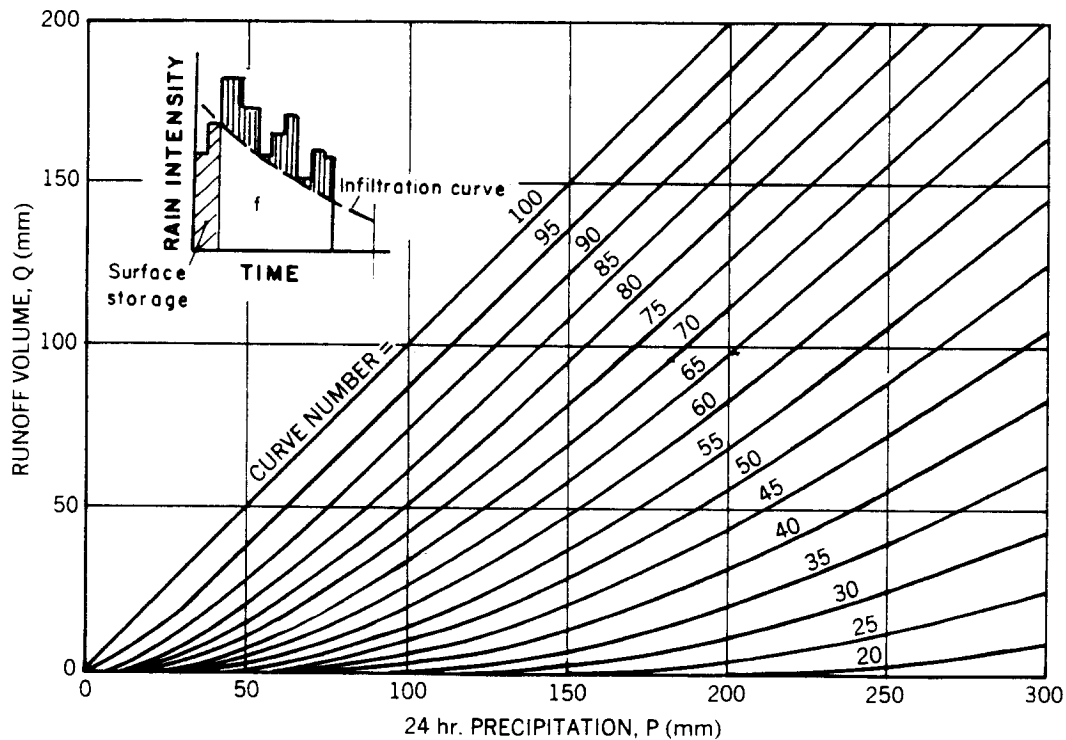


FIGURE 3.16. SCS rainfall-runoff transformation concept and runoff curves.

TABLE 3.10 Runoff Curve Number Correction for AMC I and AMC III

CN for AMC Condition II	Corresponding CN for AMC	
	Condition I	Condition III
100	100	100
95	87	99
90	78	98
85	70	97
80	63	94
75	57	91
70	51	87
65	45	83
60	40	79
55	35	75
50	31	70
45	27	65
40	23	60
35	19	55
30	15	50
25	12	45
20	9	39
15	7	33
10	4	26
5	2	17
0	0	0

Source: After Soil Conservation Service (1968).

AMC III can be read from Table 3.10 if the CN for AMC II are known.

The correction for the curve numbers can also be calculated from empirical equations given by Chow, Maidment, and Mays (1988) as

$$CN(I) = \frac{4.2 CN(II)}{10 - 0.058 CN(II)} \tag{3.24a}$$

and

$$CN(III) = \frac{23 CN(II)}{10 + 0.13 CN(II)} \tag{3.24b}$$

Table 3.11 provides seasonal rainfall limits for the three antecedent soil moisture conditions.

TABLE 3.11 Determination of Antecedent Soil Moisture Conditions

AMC	Total 5-day Antecedent Rainfall (mm)	
	Dormant Season	Growing Season
I	Less than 12.5	Less than 35
II	12.5–28	35–53
III	Over 28	Over 53

Source: After Soil Conservation Service (1968).

Example 3.5: Estimation of Daily Excess Rain by the SCS Method

Estimate the rainfall excess for the storm using the SCS method. The following information is given:

Total precipitation $P = 61.5$ mm

Hydrologic soil group C

Antecedent 5-day rainfall (dormant season) 35 mm

Surface cover—grass

Solution Using the information given in Table 3.11 the antecedent soil moisture conditions are AMC III. From Table 3.9, which gives the information on runoff curve numbers for AMC II the runoff curve number for grass (meadow) and soil hydrologic group C is $CN = 71$. The correction for AMC III is read from Table 3.10. By interpolation, the corresponding curve number for AMC III is 83.

The storage parameter is then (Eq. (3.23))

$$S = \frac{2540}{83} - 254 = 52 \text{ mm}$$

The depth of excess rainfall becomes (Eq. (3.21))

$$Q = \frac{(61.5 - 0.2 \times 52)^2}{(61.5 + 0.8 \times 52)} = 25.3 \text{ mm}$$

Excess rain can also be read from Figure 3.16.

TABLE 3.12 Calculation of Excess Rain from a Hyetograph

Time Interval (min)	Rainfall Intensity (mm/hr)	Cumulative Precipitation (volume, mm)		$f(P_{t-\Delta t})$ (mm)	$f(P_t)$ (mm)	Excess Rain, ΔQ (mm)	Cumulative Q (mm)
		$t - \Delta t$	t				
(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
0-15	30.0	0.0	7.5	0.0	0.0	0.0	0.0
15-30	120.0	7.5	37.5	0.0	9.31	9.31	9.31
30-45	80.0	37.5	57.5	9.31	22.39	13.08	22.39
45-60	16.0	57.5	61.5	22.39	25.33	2.94	25.33

Note: Column (4) = column (3) + column (2) $\times \Delta t$ [hr].

Example 3.6: Excess Rain by the SCS Method from a Hyetograph

The SCS method also enables a detailed but approximate estimation of excess rainfall from a hyetograph, such as the one shown on Figure 3.15. In this example, the excess rainfall in a 15-minute interval will be determined following the relationship expressed by Equation (3.21). From the equation

$$\Delta Q = f(P_t) - f(P_{t-\Delta t})$$

where f is the function expressed by Equation (3.21). The calculation is shown in Table 3.12. In the calculation, $S = 52$ mm (from the previous example) and $Q = 0$ if $P < 0.2S = 0.2 \times 52 = 10.4$ mm.

Rainfall Excess from Impervious Areas

Because of the surface impermeability and small-depression storage, the impervious areas (asphalt and concrete pavements, rooftops, etc.) appear to be 100% active (i.e., they generate surface runoff even during small rains). However, not all of the rainfall excess will appear as surface runoff. Thus the impervious area from which excess rain overflows onto adjacent pervious surfaces and, subsequently, into soils is considered to be not directly connected. Such cases include roof drains and driveways overflowing onto lawns, roadways and other impervious surfaces with poor or no apparent drainage, and parking lots separated by pervious areas. The rainfall excess generated on impervious areas that overflows onto adjacent pervious areas is added to the hydrological balance of the pervious area, that is, the depth of precipitation should be increased by the corresponding amount of overflow from adjacent impervious surfaces.

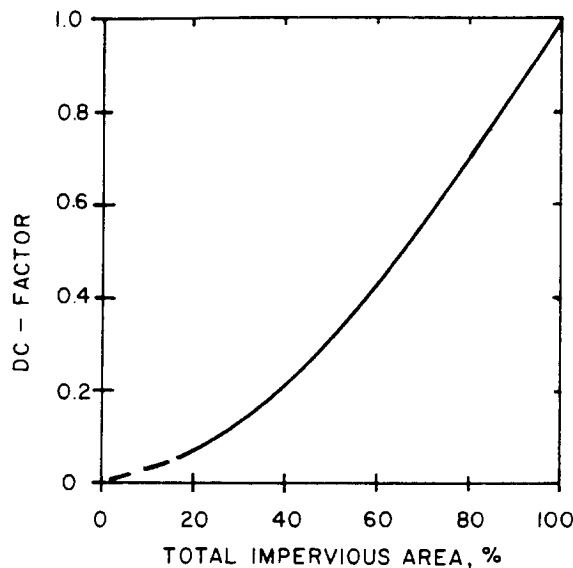


FIGURE 3.17. Relation of the fraction of the impervious surface directly connected to the channel drainage system (DC parameter) to the total imperviousness of the area. (Adapted from Anon., 1972.)

It is apparent that a fraction of the impervious areas directly connected to the drainage system increases with the degree of urbanization reflected in the total imperviousness of the area. Rainfall excess on impervious surfaces in predominantly rural areas will mostly overflow onto adjacent pervious surfaces or pervious road ditches. In densely built-up urban zones, there may not be pervious surfaces available, so all runoff will be connected to a sewer or channel drainage system. An approximate relation of the fraction of the directly connected impervious surfaces (DC) to the total imperviousness of the area is shown on Figure 3.17.

In rural zones, during medium- and low-intensity storms, when most of the surface runoff originates from impervious surfaces, the volume of the surface runoff might be sensitive to the magnitude of the parameter DC. It is therefore recommended that the value of the DC factor be estimated and verified by comparison of the computed surface runoff volumes to measured field observations. Commonly during modeling, the DC factor is a calibration parameter. Areas with storm or combined sewers will have a much higher DC factor than unsewered areas.

Hydrologically Active Areas

Based on the net rain estimation, it is evident that not all areas within the watershed will generate surface runoff and, thus the diffuse pollution associated with it. The areas that will produce surface runoff are called *hydrologically active*, while the rest of the basin contributes only to interflow and base flow (they may also be contaminated by dissolved

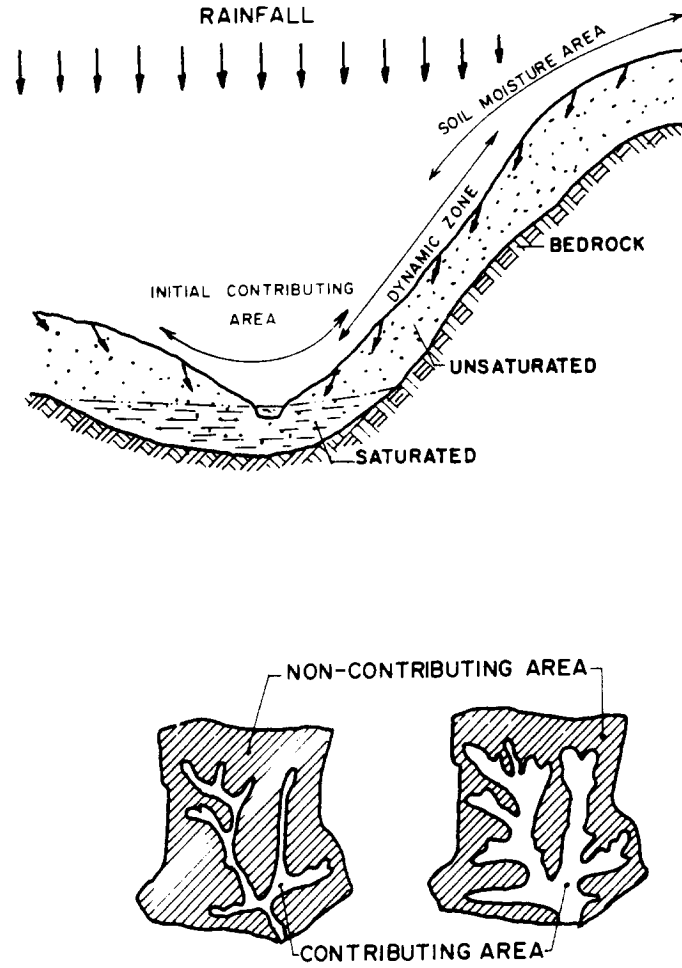


FIGURE 3.18. A hydrologically active area concept for natural watersheds. (After Engman, 1974.)

2-YEARS RECURRENCE INTERVAL 10-YEARS RECURRENCE INTERVAL

pollutants and ions). The areas showing the highest hydrological activity are obviously connected to impervious surfaces (even in this instance the depression storage must be subtracted), followed by clayey soils with low permeability, frozen soils with a high moisture content, soils with a high ground-water table, and highly compacted soils.

Areas with high surface storage, such as woods and flat cropland and, generally, soils with high permeability rates, have the lowest hydrologic activity, and often generate surface runoff only during extreme storms. Remember that the hydrological activity of an area is a stochastic hydrologic phenomenon and that the surface runoff is not generated uniformly over the entire watershed. The extent of the hydrologically active portion of the watershed also changes with the rainfall

characteristics, antecedent soil conditions, and surface characteristics. The contributing hydrologically active area will be smaller for storms with a small recurrence interval and will increase with the magnitude of the storms and their recurrence interval. Figure 3.18 shows an example of contributing areas for 2-year- and 10-year- recurrence storms in a watershed.

Identification of the areas that have a tendency to be hydrologically active is a necessary step in the abatement of diffuse pollution. These areas contribute to sediment and all surface runoff pollution. Hydrologically active areas can be determined by field surveys, plane and satellite photogrammetry and imagery, and by hydrological modeling. Topsoil distribution of a radioactive isotope cesium 137, which originated in the atmospheric nuclear weapon tests of the 1950s, will indicate erosive, hence hydrologically active areas (Ritchie, Sparberry, and McHenry, 1974). Chapter 5 contains a more detailed discussion on the methods of identifying erosive lands.

Reducing hydrological activity, for example, by increasing surface storage or permeability, or by draining high ground-water levels, is one of the most effective measures to abate diffuse pollution potential. Similarly, "disconnecting" the connected impervious areas is one of the most effective measures of controlling both flooding and pollution by urban runoff (Livingston and Roesner, 1991). This control can be accomplished by letting the roof drains overflow onto the adjacent pervious areas, incorporating infiltration into the drainage system, and other means that are discussed in Chapter 10.

OVERLAND ROUTING OF THE PRECIPITATION EXCESS

Excess or net rain can be imagined as the depth of water on the surface contributing directly to surface runoff. *Overland flow routing* is a process by which excess rain is transformed into surface runoff flow. In contrast, *channel flow routing* is a process by which a flow or flood wave is modified as it moves downstream through the channel system. It is defined as a procedure whereby the time and magnitude of a flood wave is determined at a point in a stream from the known or assumed data at one or more points upstream (Chow, 1964).

The watershed size and the length of the overland flow, along with the roughness and slope characteristics, volume and intensity of precipitation, and percent of imperviousness, seem to be the most important factors affecting the shape and magnitude of the surface runoff hydrograph. For larger drainage areas, the hydrograph curve is also affected by channel

routing. The channel portion of the routing process may not be significant for small watersheds of up to 5 km². Channel routing is a process that depends on the hydraulic characteristics of the channel.

Although the hydrograph shape seems to be of lesser importance in diffuse-pollution control than the determination of rainfall excess, the hydrograph evaluation is necessary if the impact of diffuse pollution on receiving waters is studied. Furthermore, it is demonstrated in Chapter 5 that the delivery of pollutants from the source area to a receiving body of water is affected by the characteristics of the hydrograph, namely, by the slope of the receding portion of the hydrograph.

The literature on storm-water and flow routing is quite extensive, and the most recent references include those by Bras (1990), McCuen (1982), Chow, Maidment, and Mays (1988), Gray (1973), Hall (1984), and Viessman, Lewis, and Knapp (1984).

Early overland flow-routing models are known as the *rational formula* and the *unit hydrograph*. Both concepts are oversimplified, though found to be theoretically sound.

Rational Formula

The origin of the rational formula, known in Great Britain as the Lloyd-Davis Formula, can be dated back to the second part of the nineteenth century, since when it has been used for the design of storm and combined sewer and drainage systems. It relates the peak flow of runoff in a sewer or drainage basin outlet to the rain intensity as

$$Q_p = CIA \quad (3.25)$$

where

Q_p = peak runoff discharge (l/min or cfs)

C = a runoff coefficient depending on the characteristics of the drainage area

I = the average rainfall intensity during a specified time interval called the time of concentration (mm/min or in./hr)

A = area (m² or acres).

The time of concentration is the time required for surface runoff to travel from the remotest part of the drainage area to the point of consideration (sewer inlet or sewer or watershed outlet). It consists of the overland flow time (inlet time) and sewer or channel flow time. The flow time in sewers may be estimated from the hydraulic properties of the conduit. The overland flow time is related to the watershed slope, roughness, length of the overland flow, and storm characteristics. The

TABLE 3.13 Runoff Coefficients for the Rational Formula

Description of the Area	Runoff Coefficient
Urban Areas	
Business	
Downtown	0.7-0.95
Neighborhood	0.5-0.7
Residential	
Single family	0.3-0.5
Multiunits—detached	0.4-0.6
Multiunits—attached	0.6-0.75
Residential—suburban	0.25-0.4
Apartments	0.5-0.7
Industrial	
Light	0.5-0.8
Heavy	0.6-0.9
Pavements	
Asphalt and concrete	0.7-0.95
Bricks	0.7-0.95
Roofs	0.75-0.95
Lawns—sandy soils	
Flat, slope 2% or less	0.05-0.10
Average, slope 2%-7%	0.10-0.15
Steep, greater than 7%	0.15-0.20
Lawns—tight soils	
Flat, slope 2% or less	0.15-0.17
Average, 2%-7%	0.18-0.22
Steep, greater than 7%	0.25-0.33
Rural areas	Value of C^a
Topography	
Flat land with slopes less than 1%	0.3
Rolling land with average slopes 1%-3%	0.2
Hilly land with average slopes of 3%-6%	0.1
Soil	
Tight, impervious clay	0.1
Medium, combination of clay and loam	0.2
Open, sandy loam	0.4
Cover	
Cultivated land	0.1
Woodland	0.2

Source: Data for urban areas from American Society of Civil Engineers (1982) and for rural areas from Gray (1972).

^aThe magnitude of the runoff coefficient, C , is obtained by adding values of C 's for each of the three factors (topography, soil, and cover) and subtracting the sum from unity. For example, for flat cultivated watershed with medium soils $C = 1 - (0.3 + 0.2 + 0.1) = 0.4$.

nature of the time of concentration and its relation to the peak time of the hydrograph are discussed in the next section. In seweraged watersheds, however, the overland flow time to the nearest inlet may be very short, ranging between 5 and 30 minutes. In well-developed areas with closely spaced storm inlets, an inlet time of 5 minutes is common, whereas in flat residential districts with widely spaced street inlets, inlet times of 20 to 30 minutes are customary (Novotny et al., 1989).

The runoff coefficient, C , of the rational formula given in Table 3.13 is a ratio of the peak runoff flow to the rainfall intensity of a constant, longer duration rainfall. This is not hydrologically correct since runoff is a residual of precipitation after losses are subtracted, which was documented in the previous section. Thus the coefficients given in Table 3.13 are only approximations. The rational formula estimates directly the peak runoff rate and indirectly rainfall excess, since under certain circumstances the coefficient C also expresses the relation between rainfall excess and total rainfall. Theoretically, C would equal one if rainfall excess during the time of concentration was used instead of the total rainfall.

A good discussion on the nature of the rational formula and its coefficients is given in Hall (1984), who states that the runoff coefficient should be considered as an "impermeability" factor, which is more logical than to consider it as a proportionality factor. An obvious simplifying assumption is to use the ratio of paved (impervious), connected surfaces to the total area as an estimate of the runoff coefficient.

If the drainage area is not homogenous, the area must be first divided into homogenous segments (roofs, streets, lawns) with partial areas A_1 , A_2 , A_3 , etc. The average runoff coefficient, C , can then be computed using the coefficients for the partial surface areas, C_1 , C_2 , C_3 , etc., as follows

$$C = \frac{A_1C_1 + A_2C_2 + A_3C_3 + \dots}{A_1 + A_2 + A_3 + \dots} \quad (3.26)$$

An accurate estimation of the runoff coefficient is the most important task of the entire calculation, as can be seen when considering the large differences between the values in Table 3.13. The values of the coefficients, C , in the table correspond to rather flat catchments; larger runoff coefficients should be selected for steeper catchments.

Example 3.7: Rational Formula

Determine peak runoff and approximate runoff hydrographs from a uniform rainfall with an average intensity of 20 mm/hr, lasting two hours.

Solution The residential watershed with an area of 50 hectares is composed of the following:

Roofs: 25% ($C = 0.9$)

Asphalt pavements: 25% ($C = 0.8$)

Lawns on flat tight soils: 50% ($C = 0.15$)

The watershed is drained by a 0.6-km-long storm sewer with a design flow velocity of 1.5 m/sec.

1. Estimated average runoff coefficient

$$C = \frac{0.25 \times 0.9 + 0.25 \times 0.8 + 0.5 \times 0.15}{0.25 + 0.25 + 0.5} = 0.505$$

2. Estimated time of concentration

Overland flow time: assume 30 min

$$\begin{aligned} \text{Sewer time} &= \text{length/velocity} = 750(\text{meters})/1.5(\text{m/sec}) \\ &= 500 \text{ sec} = 8.33 \text{ min} \end{aligned}$$

Total time of concentration is 38.3 minutes which is less than 2 hrs. The peak flow is

$$\begin{aligned} Q_p &= 0.505 \times 20(\text{mm/hr}) \times (1/60 \text{ hr/min}) \times 50(\text{ha}) \times 10^4(\text{m}^2/\text{ha}) \\ &= 84\,166(1/\text{min})/[60(\text{sec/min})/1000(1/\text{m}^3)] \\ &= 1.402 \text{ m}^3/\text{sec} \end{aligned}$$

The approximate surface runoff hydrograph is plotted in Figure 3.19. Note that the time length of the ascending and recessed limbs of the hydrograph are approximately equal to the time of concentration.

The Unit Hydrograph

The concept of the unit hydrograph was conceived by hydrologists more than 60 years ago (Sherman, 1932), mostly from observations without theoretical mathematical development, which followed later.

Chow (Chow, Maidment, and Mays, 1988) defined two basic differential equations governing overland flow. The first equation is an equation of continuity, such as

$$\frac{dS}{dt} = I - Q \quad (3.27)$$

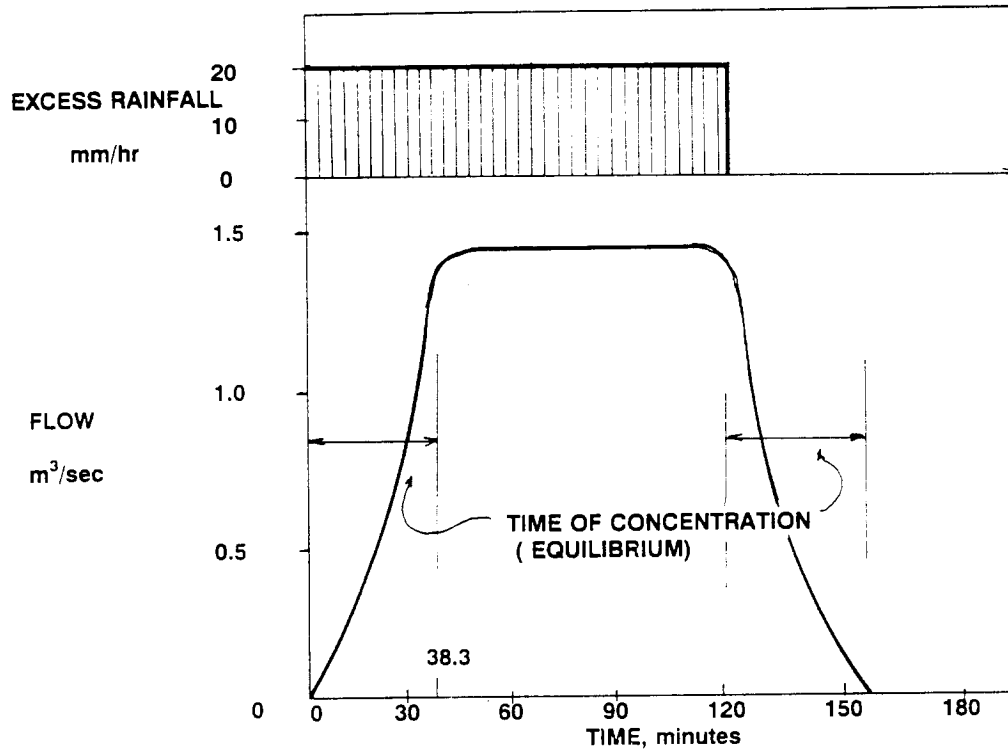


FIGURE 3.19. Simplified hydrograph for Example 3.7 (rational formula).

where

S = water storage within the watershed system

I = input (excess rainfall)

Q = output (runoff flow)

t = time

The second equation relates the outflow rate to the amount of storage and to the inflow. At this point this equation will be represented by a general equation, such as

$$Q = f(S, I) \tag{3.28}$$

Chow and coworkers have then shown that this general storage-input-output relationship can be expressed using the so-called convolution integral

$$Q(t) = \int_0^t h(\tau)I(t - \tau) d\tau \tag{3.29}$$

where

$h(\tau)$ = the ordinate of the instantaneous unit hydrograph (IUH)

τ = lag time

The instantaneous unit hydrograph in Equation (3.29) is the watershed response to a short-duration unit rainfall. Theoretically, to conform with the mathematics, the duration of the rainfall pulse defining the unit hydrograph should be infinitesimally small to instantaneous. The input-output relationship expressed by Equations (3.24) to (3.26) is typical for linear systems, that is, for systems where the shape of the unit hydrograph does not depend on the magnitude of the input or output of the system. The unit hydrograph concept defined herein corresponds to the linear system representation.

Under the assumption of linearity the following two theorems hold:

1. If the input of Equation (3.28) is multiplied by a constant, such as $c \times I$, then the output is also multiplied by the same constant, or $c \times Q$ (*principle of proportionality*).
2. If two solutions $f_1(Q)$ and $f_2(Q)$ of Equation (3.29) are added, the resulting solution $f_1(Q) + f_2(Q)$ is also a solution of the equation (*principle of superposition*).

Based on these two theorems any excess rainfall input can be broken down into the rainfall pulses of the same duration as the unit rainfall input defining the unit hydrograph. The volume of the rainfall pulse is then the multiplier determining the magnitude of the individual response hydrograph. All individual response hydrographs are then summed up as shown graphically on Figure 3.20. The same principle is incorporated into several hydrologic models as an overland flow-routing routine.

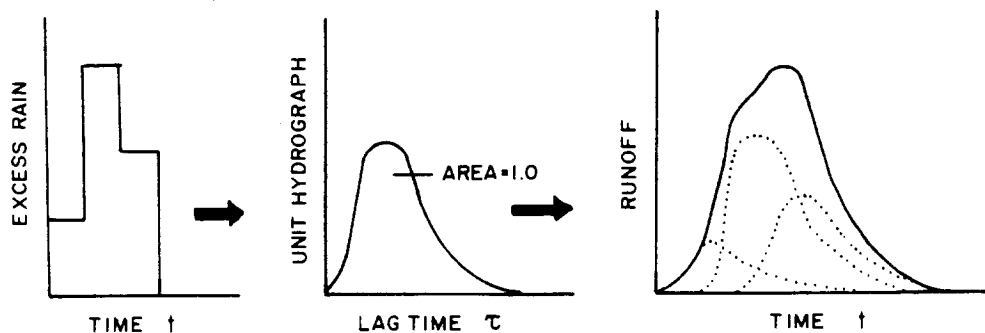


FIGURE 3.20. Graphical convolution of excess rainfall into runoff using unit hydrograph concept.

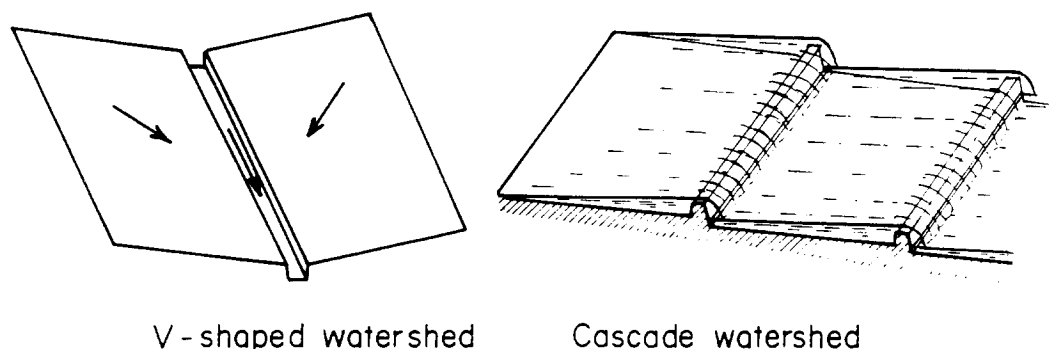


FIGURE 3.21. Representation of watersheds in simple lumped-parameter rainfall-runoff models.

The principle of linearity implied in the unit hydrograph concept was subsequently questioned. Horton (1939b) and Izzard (1946) showed that the ordinates of the unit hydrograph, its peak time, as well as the time of concentration in the rational formula, depend on the intensity of the rainfall excess.

Synthetic Unit Hydrographs

The functions describing the instantaneous unit hydrograph can be based on two simplified watershed representations (Fig. 3.21). The first representation is a two-plane V-shaped watershed, as shown on the left side of Figure 3.21. The hydrograph solution for this watershed can be obtained by kinematic wave approximation (Henderson and Wooding, 1964; Wooding, 1965; Overton and Meadows, 1976). Equations (3.27) and (3.28) are represented in this concept by

$$\frac{\partial H}{\partial t} + \frac{\partial q}{\partial x} = i - f$$

and

$$q = \alpha H^\beta \quad (3.30)$$

where

H = water depth

q = the discharge rate/unit width

$i - f$ = rainfall excess

x = distance measured downstream from the top of the catchment

α, β = empirical coefficients

The kinematic wave model can be applied to small drainage areas of uniform slope and surface characteristics, such as parking lots and highways. The numerical solutions and various types of hydrographs are discussed in Overton and Meadows (1976).

The second watershed representation breaks the watershed into a series of overflowing pool-cascades and uses the continuity equation (Eq. (3.27)) for routing the flow. For a single reservoir the storage-outflow relationship (Eq. (3.28)) is expressed by

$$S = K \times Q \quad (3.31)$$

where K is called the single-reservoir watershed constant. Substituting Equation (3.31) into the continuity equation (Eq. (3.27)) will yield to the following one-dimensional differential equation

$$K \frac{dQ}{dt} + Q = I$$

the solution of which for a unit rainfall input pulse is

$$h(t) = \frac{1}{K} e^{-t/K} \quad (3.32)$$

The cascade watershed instantaneous unit hydrograph function was developed by Nash (1957) as

$$h(t) = \frac{1}{K_N \Gamma(n)} e^{-t/K_N} \left(\frac{t}{K_N} \right)^{n-1} \quad (3.33)$$

where

K_N = multiple basin storage constant

n = a watershed characteristic representing approximately the number of reservoirs

$\Gamma(n)$ = the gamma function of n

Note that if $n = 1$, then the formula will yield the single reservoir model expressed by Equation (3.32). In the watershed modeling process, the principal question is how many equal linear "pools" are needed for an adequate model? Based on the authors' experience, $n = 1$ gives satisfactory results for a small watershed of up to 10 km². For larger watersheds, the ranges of n may be higher; however, as pointed out by Overton and Meadows (1976), the model quickly approaches translation

(that is, the results are not much different from $n \rightarrow \infty$) for $n > 5$. An analysis of a very large number of storm hydrographs by Holtan and Overton (quoted in Overton and Meadows, 1976) indicated that $n = 2$ produced optimum results in fitting computed runoff hydrographs to measured data.

The constants K and K_N and the reservoir characteristics that can be correlated to the travel time of water from the most remote point on the watershed to the watershed outlet. This time parameter is called the *time to equilibrium* or *time of concentration*, t_e . According to Rao, Delleur, and Sarma (1972)

$$t_e = K = n \times K_N \quad (3.34)$$

The time of concentration as previously defined is the theoretically correct time parameter that should also be used in the rational formula discussed in the preceding section. As shown on Figure 3.22, the unit hydrograph concept can be used to derive theoretical justification for the rational formula, which in this case is the watershed response to uniform rainfall that is equal or greater in length than the time of concentration.

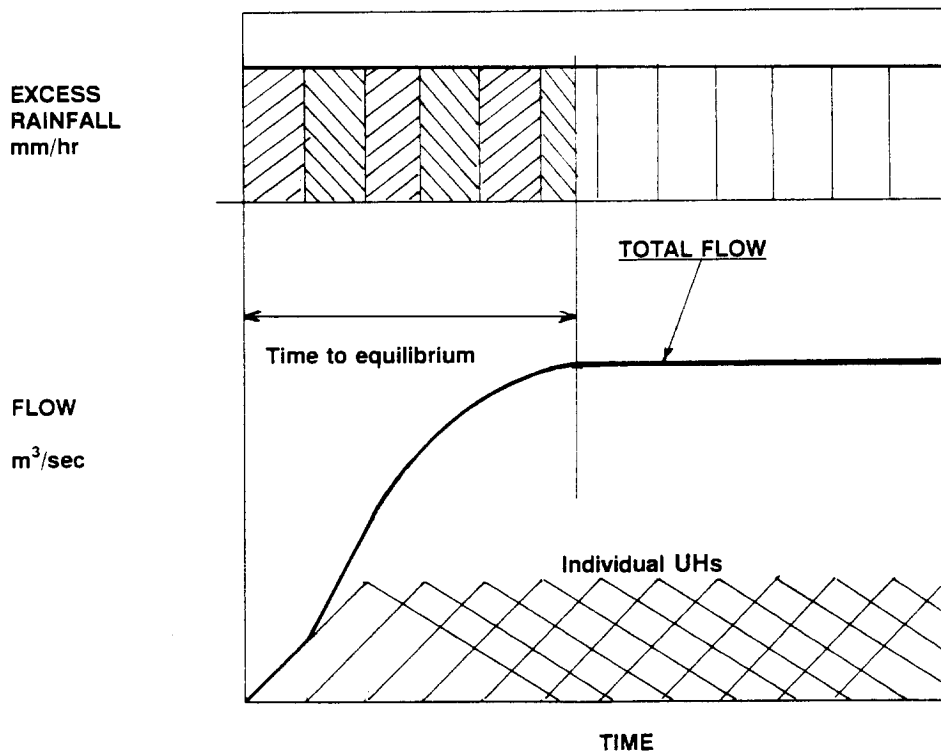


FIGURE 3.22. Rising (S-curve) hydrograph and time to equilibrium estimated by convolution of a long-duration uniform excess rainfall.

The rising limb of the hydrograph (so-called S-curve) is obtained by a summation of individual unit hydrographs multiplied by the average rainfall within the storm. It can be seen that equilibrium is reached when the time approaches the time of concentration.

Henderson and Wooding (1964) used the kinematic wave approximation for a V-shaped watershed and developed the following equation for t_e :

$$t_e = 6.9 \frac{(L \times n_M)^{0.6}}{i^{0.4} S^{0.3}} \quad (3.35)$$

where

t_e = time to equilibrium (min)

L = length of the overland flow (m)

TABLE 3.14 Manning's Roughness Coefficient, n_M for Overland Flow

Ground Cover	Manning's n_M
<i>Urban zones</i>	
Smooth asphalt	0.012
Street pavement	0.013
Asphalt or concrete paving	0.014
Packed clay	0.03
Light turf	0.20
Dense turf	0.35
Dense shrubbery or forest litter	0.40
Short grass	0.03–0.035
High grass—submerged flow	0.025–0.05
Heavy weeds—scattered grass	0.05–0.07
<i>Nonurban zones</i>	
Fallow field	
Smooth—rain packed	0.01–0.03
Medium—freshly disked	0.1–0.3
Rough—freshly disked	0.4–0.7
Cropped field	
Grass and pasture	0.05–0.03
Clover	0.08–0.25
Small grains	0.1–0.4
Row crops	0.07–0.2
Woods	
Light underbrush	0.4
Dense underbrush	0.8

Source: Compiled from several engineering texts and handbooks.

- i = rainfall intensity (mm/hr)
 S = slope (m/m)
 n_M = Manning's surface roughness factor (Table 3.14)

An almost identical formula was developed by Morgali and Linsley (1965). Rao, Delleur, and Sarma (1972) statistically analyzed the hydrograph curves for several urbanizing watersheds. The authors investigated the effects of many variables on the shape of the runoff hydrograph, and only those that were found to be statistically significant were included in the final formula. Based on their work, t_e and the reservoir number n can be estimated from

$$t_e = 304 \frac{(AW)^{0.458}(TR)^{0.104}}{(1 + U)^{1.662}(i)^{0.269}} \quad (3.36)$$

and

$$n = 2.23 \frac{(AW)^{0.069}}{(1 + U)i^{0.155}} \quad (3.37)$$

where

- t_e = time to equilibrium (hr)
 AW = watershed area (km²)
 i = rainfall intensity (mm/hr)
 U = fraction of the impervious area of the total watershed area
 TR = rain duration (hr)

Note that minimum $n = 1$.

Gray (1961) developed the following relationship for the watershed constant, K_N , which is applicable to small rural watersheds (0.6 to 80 km²) in Wisconsin, Illinois, central Iowa, and Missouri:

$$K_N = \frac{t_c}{n} = 7.33 \left(\frac{L}{\sqrt{S_c}} \right)^{0.562} \quad (3.38)$$

where

- t_c = time of concentration (min)
 S_c = average watershed slope (%)
 L = length of watershed (km), which includes overland-flow length and the length of the longest watershed channel

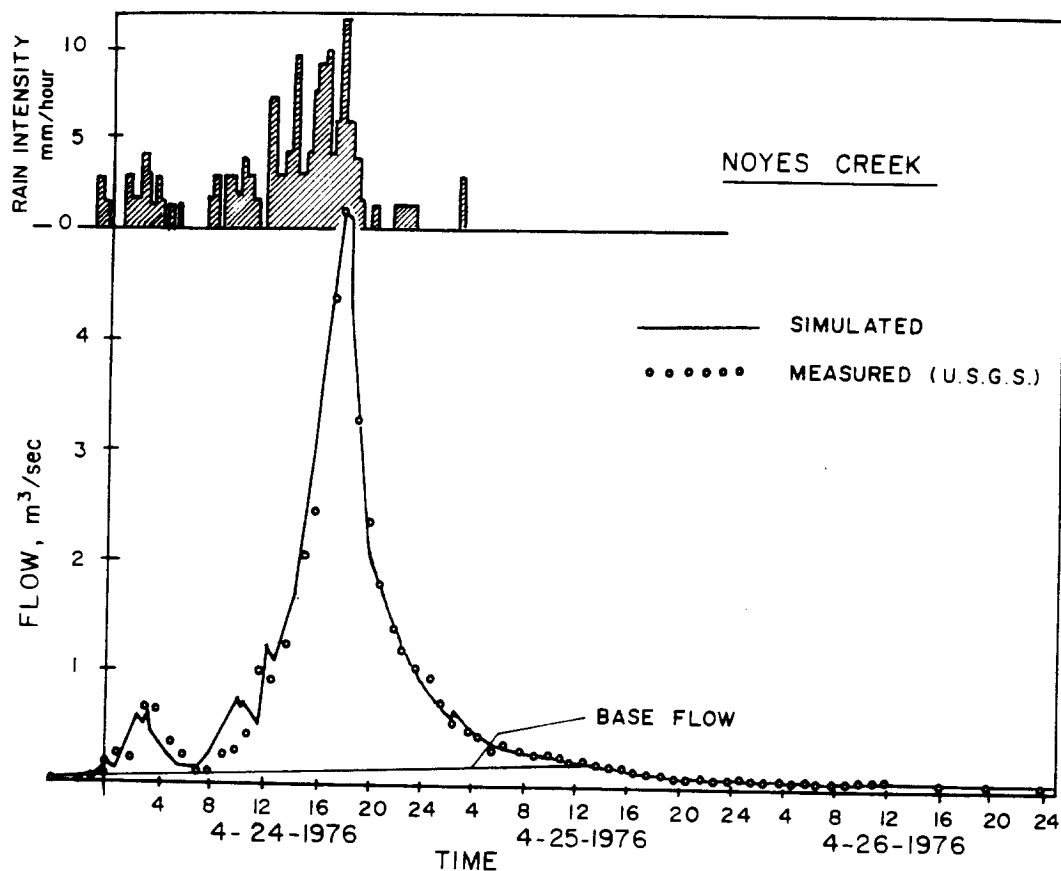


FIGURE 3.23. Comparison of measured and computed flows by a rainfall-runoff model for a small urban (mixed land use) watershed in southeastern Wisconsin.

The formulas for the time of concentration are not overly sensitive to the shape of the watershed.

Equations (3.35) and (3.36), which were obtained by a solution of the kinematic wave equation for the watershed overland flow or by statistically analyzing the hydrographs, indicate that the watershed constants depend on the magnitude of the input (rainfall intensity), which violates the assumption of linearity. It means that each partial rainfall input will be convoluted with different unit hydrographs. This should impose no problem in computer modeling applications and the results of convolution should be fairly accurate for reasonably small watersheds, as shown on Figure 3.23.

Example 3.8: Overland-Flow Routing by a Synthetic Hydrograph

For the rainfall excess computed in Example 3.3 and reported in Table 3.8, estimate the magnitude and shape of the surface runoff hydrograph if the drainage area is 1 km^2 , the slope is 2%, the surface area is covered by grass, and 3% of the watershed is impervious.

Solution Since the watershed size is less than 10 km², the watershed reservoir number characteristic is selected as $n = 1$. This can be checked from Equation (3.37) using the smaller rainfall intensity reported in Table 3.8, column 9. Hence, $i = 44$ mm/hr. Then

$$n = \frac{2.23(1.0)^{0.069}}{(1 + 0.03)44^{0.155}} = 1.2$$

Therefore the single reservoir routing formula (Equation (3.32)) can be used to represent the unit hydrograph. The Manning roughness factor for grass is close to 0.35. For $i = 57$ mm/hr

$$K = t_c = 6.9 \frac{(0.5 \times 1000)^{0.8} 0.35^{0.6}}{57^{0.4} 0.02^{0.3}} = 98.2 \text{ min} = 1.64 \text{ hr}$$

For $i = 44$ mm/hr

$$K = t_c = 6.9 \frac{(0.5 \times 1000)^{0.6} 0.35^{0.6}}{44^{0.4} 0.02^{0.3}} = 99.3 \text{ min} = 1.65 \text{ hr}$$

Since both storage constants are similar, linear response may be assumed. The hydrograph computation using the following equation and convolution are shown in Table 3.15 and are plotted in Figure 3.24.

TABLE 3.15 Hydrograph Calculation for Example 3.8

Time (min) t or τ	Unit Hydrograph at Time τ	Ordinates of the Hydrograph (m ³ /sec)		Total
		$P = 57$ mm/hr	$P = 44$ mm/r at Time t	
0	0	0	0	0
15	0.56	0	0	0
30	0.48	2.22	0	2.22
45	0.41	1.90	1.71	3.61
60	0.35	1.62	1.47	3.09
75	0.31	1.38	1.26	2.64
90	0.26	1.22	1.07	2.29
105	0.23	1.03	0.95	1.98
120	0.19	0.91	0.79	1.70
150	0.15	0.67	0.64	1.31
180	0.11	0.49	0.46	0.95
210	0.08	0.38	0.34	0.72

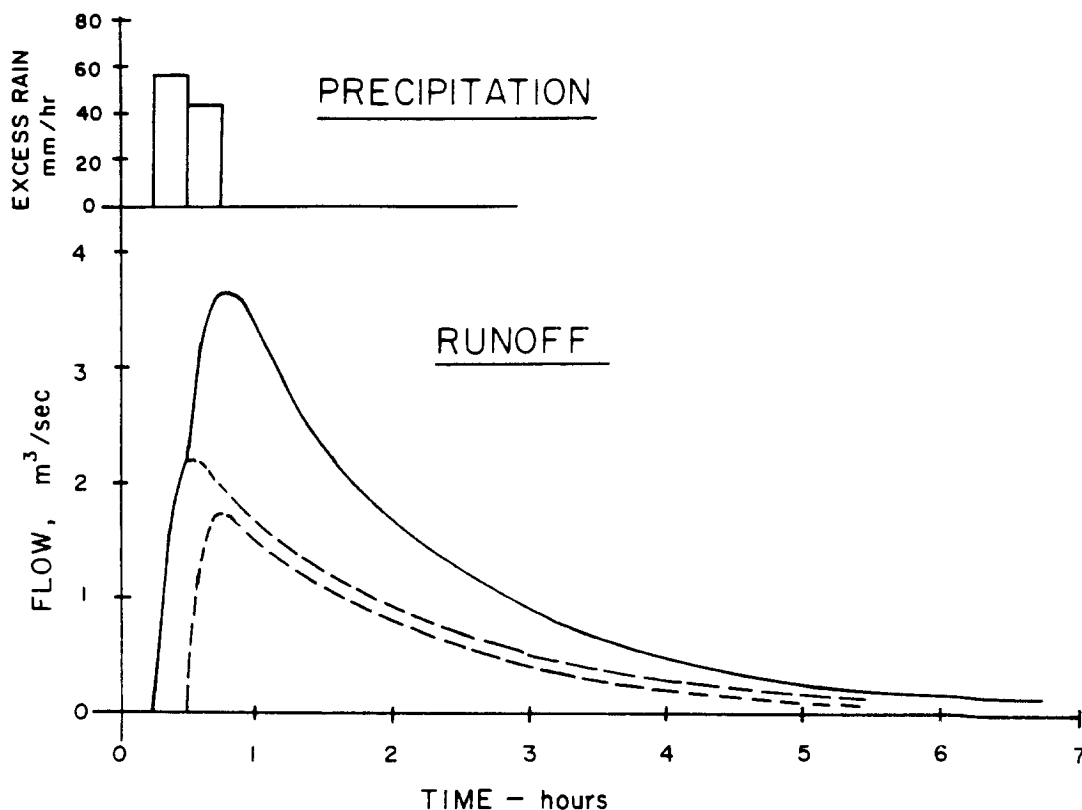


FIGURE 3.24. Rainfall-runoff convolution for Example 3.8.

$$\text{Flow}(m^3/\text{sec}) = \text{Area}(m^2) \times \frac{m}{1000 \text{ mm}} \times \frac{\text{hr}}{3600 \text{ sec}} \times \sum_{r=0}^t [X_{t-\tau} \times h(\tau) \times \Delta t]$$

where

X = rain intensity (P in mm/hr)

h = hydrograph ordinate

Δt = time increment

Overland Routing by the SCS Method

The U.S. Soil Conservation Service hydrologic method (Soil Conservation Service, 1968) provides a methodology for overland routing of excess rainfall. Excess rain is determined, for example, in hourly intervals, as shown in Example 3.6.

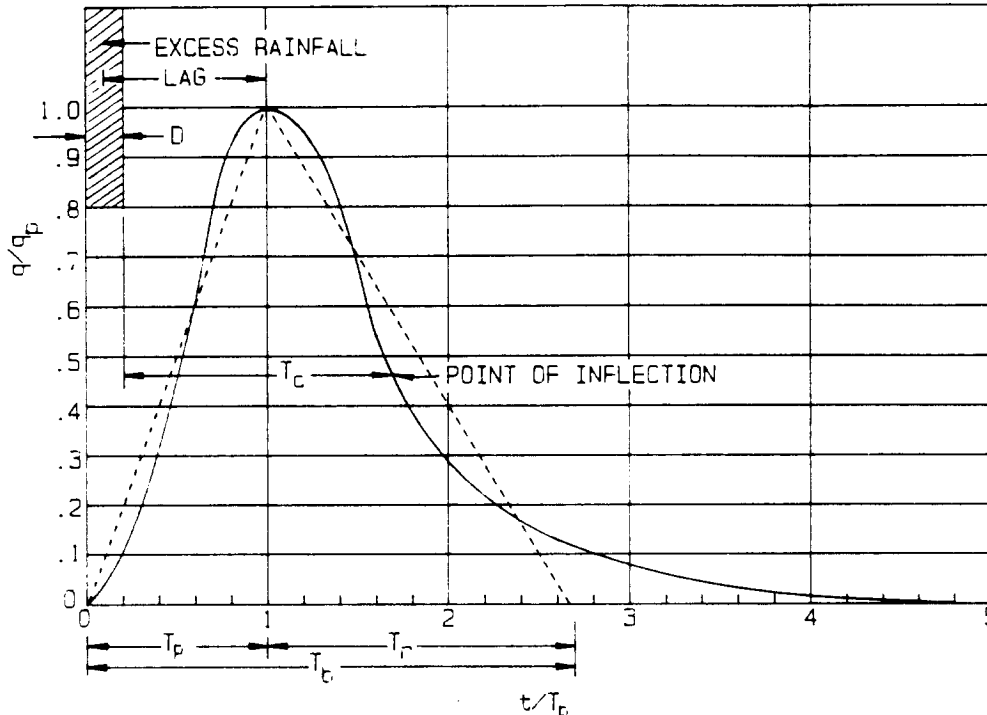


FIGURE 3.25. The Soil Conservation Service dimensionless unit hydrograph.

The Soil Conservation Service (SCS) proposed the dimensionless triangular hydrograph shown on Figure 3.25. This shape is based on the SCS analysis of numerous measured hydrographs under varying watershed and rainfall conditions. The peak time is the only parameter determining the shape of the hydrograph.

The peak time, t_p , according to Figure 3.25, is as follows:

$$t_p = D/2 + t_l \tag{3.39}$$

where

t_l = the lag time from the centroid of the rainfall pulse

D = duration of the unit rainfall pulse

The hydrograph is approximated by a triangle with its peak at t_p and its base $t_b = \frac{8}{3}t_p$, which the SCS suggests is the best approximation of a unit hydrograph for typical rural watersheds. The recession time of the hydrograph is then $t_r = t_b - t_p = \frac{5}{3}t_p$.

The area under the unit hydrograph then equals the unit volume of the rainfall excess, or

$$1 = \frac{1}{2}q_p(t_p + t_r) \quad (3.40)$$

or

$$q_p = \frac{1}{t_p} \left(\frac{2}{1 + t_r/t_p} \right) = \frac{K}{t_p} \quad (3.41)$$

where q_p is the peak runoff rate of the unit hydrograph.

For unit rainfalls given in millimeters and flow in cubic meters/second the watershed constant K has to include the watershed area and a conversion between the units. For watershed area, A , in hectares, rainfall excess in millimeters, t_p in hours, and $t_r/t_p = \frac{5}{3}$, as obtained from the dimensionless unit hydrograph on Figure 3.25, Equation (3.41) for peak runoff of the unit hydrograph becomes

$$q_p = \frac{A}{(480 \times t_p)}$$

The lag time, t_l is estimated from

$$t_l[\text{hr}] = \frac{1}{7053} \frac{L^{0.8}(S + 25.4)^{0.7}}{(Sl)^{0.5}} \quad (3.42)$$

where

L = the length of overland flow to divide in meters

S = watershed storage defined by Equation (3.23)

Sl = percent slope of the watershed

In order to include the effect of imperviousness in urban or suburban watersheds and transportation corridors, the lag time is adjusted (i.e., multiplied) by the lag factor LF defined as

$$\begin{aligned} LF = 1 - \text{PRCT}(-0.006789 + 0.000335 \text{ CN} - 0.0000004298 \text{ CN}^2 \\ - 0.00000002185 \text{ CN}^3) \end{aligned} \quad (3.43)$$

in which CN is the runoff curve number (Table 3.9) and PRCT is the percent imperviousness of the area or percent of the drainage system that has been channelized and lined.

The average lag time is $0.6 \times$ time of concentration or $t_c = 1.666 \times t_l$. Then the duration of the rainfall pulse for convolution can be related to the time of concentration and peak time as follows

$$D = 0.133t_c$$

and

$$D = 0.2t_p$$

Small variations of D are permitted, but they should not exceed $0.25t_p$ or $0.17t_c$ (Viessman, Lewis, and Knapp, 1989).

Example 3.9: Determination of the SCS Unit Hydrograph

Determine the triangular SCS unit hydrograph for a 250-ha watershed that has been developed into a residential subdivision. The flow length is 2000 m, the slope is 3%, the watershed is 25% impervious, and the runoff curve number is $CN = 80$. Determine the uncorrected time of concentration.

Solution From Equation (3.23) $S = 25400/80 - 254 = 63.5$; then

$$t'_l = \frac{(2000)^{0.8}(63.5 + 25.4)^{0.7}}{7053 \times 3^{0.5}} = 0.83 \text{ hr}$$

Correct t_l for imperviousness

$$LF = 1 - 25 \times (-0.006789 + 0.000335 \times 80 - 0.0000004298 \times 80^2 - 0.00000002185 \times 80^3) = 0.848$$

$$t_l = LF \times t'_l = 0.848 \times 0.83 = 0.70 \text{ hr}$$

The time of concentration $t_c = 1.666 \times t_l = 1.17 \text{ hr}$.

The duration of the rainfall pulse for convolution should be about

$$D = 0.1333t_c = 0.1333 \times 1.17 = 0.15596 \text{ hr} = 9.35 \text{ min}$$

Select $D = 10 \text{ min} = 0.16667 \text{ hr}$. The peak time is then

$$t_p = \frac{D}{2} + t_l = \frac{0.1667}{2} + 0.7 = 0.87 \text{ hr} = 52.2 \text{ min}$$

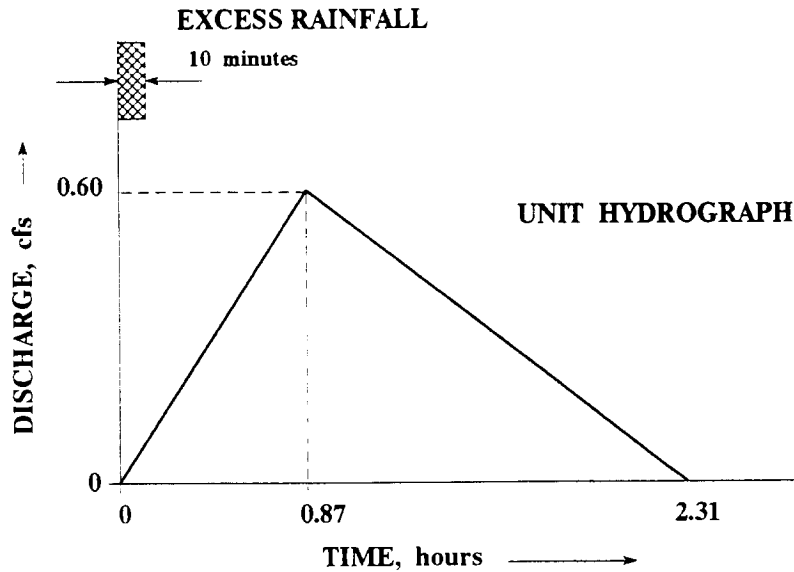


FIGURE 3.26. SCS unit hydrograph for Example 3.9.

also

$$t_r = \frac{5}{3}t_p = 1.44 \text{ hr}$$

$$t_b = \frac{8}{3}t_p = 2.31 \text{ hr} = 138.6 \text{ min}$$

The peak of the unit hydrograph is then

$$q_p = 0.0020833 \times \frac{A}{t_p} = 0.0020833 \times \frac{250}{0.87} = 0.60 \text{ m}^3 \text{ sec}^{-1} \text{ mm}^{-1}$$

The plot of the unit hydrograph is in Figure 3.26.

Example 3.10: Convolution with the SCS Unit Hydrograph

The rainfall hyetograph given below is to be convoluted into surface runoff by the SCS method. The watershed characteristics and the UH are given in Example 3.9. The excess rainfall is estimated in Table 3.16 by Equation (3.21).

From Example 3.9

$$D = 10 \text{ min}$$

$$t_p = 0.87 \text{ hr} = 52.2 \text{ min}$$

$$t_b = 2.31 \text{ hr} = 138.6 \text{ min}$$

$$q_p = 0.6 \text{ m}^3 \text{ sec}^{-1} \text{ mm}^{-1}$$

TABLE 3.16 Excess Rain Calculation

Time Interval, t (min) (1)	Rainfall Intensity (Hyetograph), i (mm/hr) (2)	Cumulative Rainfall, TR (min) (3)	Cumulative Excess Rain, Q (mm) (4)	Excess Rain 10 min Pulse, QRP (mm) (5)
0-10	36	6	0	0
10-20	36	12	0	0
20-30	54	21	0.95	0.95
30-40	60	31	4.09	3.14
40-50	30	36	6.25	2.16
50-60	0	36	6.25	0

Notes: Column (3) TR = i/D ; column (4) Q from Equation (3.21); column (5) $QRP = Q_t - Q_{t-D}$.

The excess rainfall in column (5) is then convoluted with the SCS triangular UH with parameters calculated in Example 3.9. The ordinates of the UH are calculated in Table 3.17.

The hydrograph ordinates in columns (II), (III), and (IV) are obtained by multiplying the ordinates of the UH by the excess rain pulse shifted in time to place the beginning of the hydrograph at the beginning of the excess rain pulse. Column (V) is the summation of columns (I) to (IV).

Statistically Estimated Unit Hydrograph

The most desirable unit hydrographs (UH) are those estimated from monitoring data. Engineering manuals and hydrology textbooks recommend that the hydrograph be determined from hydrographs of several storm events. As pointed out previously, due to the nonlinearity of the process, determination of one uniform UH is not possible. Preferably the UH is determined from a hydrograph resulting from a short-duration, uniform, intense storm. If such a hydrograph is available, then the UH is determined from the measured hydrograph by dividing the ordinates of the hydrograph by the total hydrograph volume, expressed in millimeters (inches), of the runoff volume distributed over the watershed area. Before the UH is estimated, flow components other than those directly driven by the rainfall—including infiltration into sewers, ground-water base flow and interflow—must be subtracted from the measured hydrograph, which is neither easy nor straightforward (see, for example, Linsley, Kohler, and Paulhus (1982) for the procedures of a single storm—single runoff peak UH determination).

Chow, Maidment, and Mays (1988) and Bras (1990) describe several procedures of deconvolution for more complex hydrographs resulting

TABLE 3.17 Calculation of the Coordinates of the Unit Hydrograph and Convolution of Excess Rainfall into Surface Runoff

Unit Hydrograph				Convolution							
				Real Storm Time (min)			Runoff Time (min)	Total Runoff (m ³ /s)			
Lag Time τ (min)	τ/τ_p	q/q_p	q (m ³ /sec-mm)	20-30	30-40	40-50		(I)	(II)	(III)	(IV)
(1)	(2)	(3)	(4)	Excess Rain (mm)							
				Partial Hydrograph (m ³ /s)							
					0.95	3.14	2.16				
0	0	0	0	0	0	0	0	0	0	0	0
10	0.192	0.192	0.115	20	0	0	0	0	0	0	0
20	0.383	0.383	0.230	30	0.109	0	0	0.109	0	0	0.109
30	0.575	0.575	0.345	40	0.218	0.361	0	0.579	0.361	0	0.579
40	0.766	0.766	0.460	50	0.327	0.722	0.248	1.297	0.722	0.248	1.297
50	0.958	0.958	0.574	60	0.437	1.083	0.496	2.017	1.083	0.496	2.017
52.2	1.000	1.000	0.600	70	0.545	1.444	0.745	2.734	1.444	0.745	2.734
60	1.149	0.910	0.546	80	0.519	1.802	0.994	3.314	1.802	0.994	3.314
70	1.341	0.795	0.477	90	0.453	1.714	1.240	3.407	1.714	1.240	3.407
80	1.533	0.679	0.408	100	0.387	1.497	1.179	3.063	1.497	1.179	3.063
90	1.724	0.564	0.338	110	0.321	1.281	1.030	2.632	1.281	1.030	2.632
100	1.915	0.465	0.279	120	0.265	1.061	0.881	2.207	1.061	0.881	2.207
110	2.107	0.333	0.200	130	0.190	0.876	0.730	1.796	0.876	0.730	1.796
120	2.298	0.217	0.130	140	0.123	0.628	0.603	1.353	0.628	0.603	1.353
130	2.490	0.102	0.061	150	0.058	0.408	0.432	0.898	0.408	0.432	0.898
138.6	2.66	0.000	0.000	160	0.0	0.191	0.281	0.472	0.191	0.281	0.472
				170	0.0	0.0	0.132	0.132	0.0	0.132	0.132
				180	0.0	0.0	0.0	0.0	0.0	0.0	0.0

R0023141

from more complex rainfall. Generally, the deconvolution is obtained from a set of equations expressed in matrix form

$$[\mathbf{R}] [\mathbf{h}] = [\mathbf{Q}] \quad (3.44)$$

where $[\mathbf{R}]$ is the vector of rainfall data, $[\mathbf{Q}]$ is the corresponding flow vector, and $[\mathbf{h}]$ is the unit hydrograph expressed here as a polynomial. Several numerical procedures for deconvolution were presented by Bras (1990).

However, Chow et al. (1988) warned that given $[\mathbf{R}]$ and $[\mathbf{Q}]$, there is usually no direct solution for $[\mathbf{h}]$. The solution must be found by minimizing the least-square errors between the measured and computed $[\mathbf{Q}]$; however, the solution is not easy because the many repeated and blank entries in $[\mathbf{R}]$ create computational difficulties.

The limitations and difficulties of the UH determination from measured data were summarized by Novotny and Zheng (1989) as follows:

1. Direct estimation of UH is difficult, especially when the hydrograph contains flow components other than surface runoff. The separation of hydrograph components into surface runoff, interflow, and base and sewage flow is generally inaccurate and arbitrary.
2. The UH is commonly estimated from a few characteristic storms. However, since only net rain contributes to the surface runoff component of the hydrograph, all precipitation losses (infiltration, evapotranspiration, surface storage) must be subtracted or the net rain will be determined from the surface runoff volume only. The losses are generally highly variable, almost random processes; therefore, deterministic (not considering the random component) net rain estimates usually carry large errors.
3. There are also physical inconsistencies of such functions, and some portions of the estimated hydrographs have negative and undulating ordinates.

Somewhat better results can be obtained by using dynamic hydrologic models where the output from the watershed hydrologic model is matched with a time series of measured data. The model must have a synthetic UH function incorporated in the structure of the program, and the parameters of the function are obtained by calibration and verification processes (see Chap. 8 for a description of the models and of their calibration and verification). The estimates of the UH function relies heavily on the adequacy of the function itself, on the accuracy of the

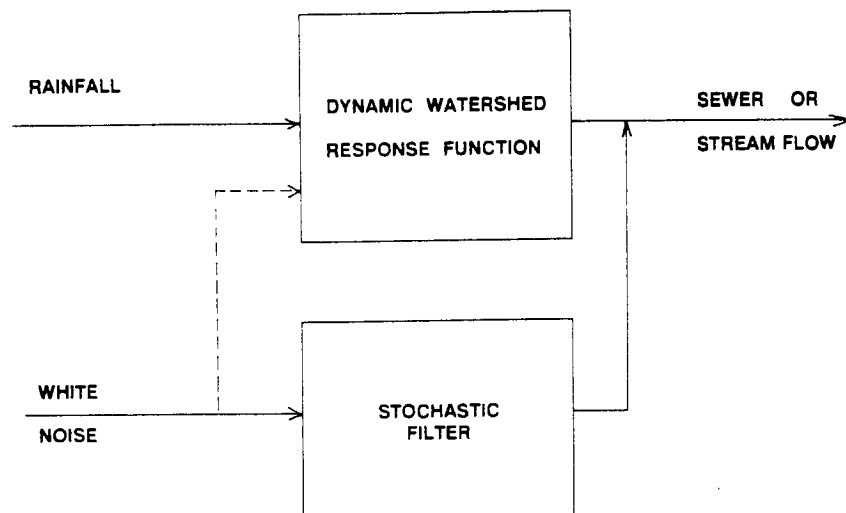


FIGURE 3.27. Stochastic rainfall-runoff transformation process that includes deterministic stochastic transfer function and input random noise.

separation of nonsurface flow contributions, and on the accuracy of the data. Furthermore, calibration and verification, which are often accomplished by eye fitting the measured and computed data, do not give an unbiased UH function estimate.

Novotny and Zheng (1989) proposed an ARMA-transfer-function model to represent the relationship between the rainfall and flow time series. Following standard ARMA modeling concepts (Box and Jenkins, 1976; Bras and Rodriguez-Iturbe, 1985), a standard rainfall-runoff transformation model consists of two parts, as shown on Figure 3.27: (1) a stochastic transfer of the input rainfall into the output hydrograph; and (2) a noise term that is a filtered (transformed) uncorrelated random series generically called *white noise*.

The ARMA modeling uses a backshift operator B introduced by Box and Jenkins for convenience in expressing the time series, such as that for any series $BZ_t = Z_{t-\Delta t}$, $B^2Z_t = Z_{t-2\Delta t}$, and so forth. Then the input-output rainfall-runoff transformation model can be written as

$$Q = h(B)R_t + N_t \quad (3.45)$$

where

$h(B)$ = a polynomial representing the UH function

R_t = the rainfall or excess rainfall series

N_t = the noise component

In this simple representation it is not necessary to focus a priori on the surface runoff component only if it is assumed that N_t is another stochastic model representing flows other than those due to rainfall. In seweraged watersheds, N_t could be considered as dry-weather sewage and wastewater contribution in combined sewers, cross-connections, and illegal dry-weather flow connection in separate sewers, or base flow in unsewered watersheds.

Also R_t does not have to be limited to net rain, since net rain is correlated to the total rainfall by some unspecified transfer function. Furthermore, the amount of infiltration is also correlated to rainfall. Thus, the transfer function $h(B)$ can contain both overland and subsurface routing components that, theoretically, could be separated and identified. Hence, $h(B)$ may or may not be similar to the theoretical synthetic UH formula described previously.

Since N_t is a filtered random series, it contains autocorrelative and random components. These must be filtered out from the measured series before the UH function $h(B)$ can be estimated. Novotny and Zheng (1989) developed a simple multiple-regression procedure that accomplishes filtering. In this procedure the measured hydrograph flows are correlated to rainfall and to the past flow series as follows

$$Q_t = I_1 Q_{t-1} + I_2 Q_{t-2} + I_3 Q_{t-3} + \cdots + I_m Q_{t-m} + \omega_0 R_t + \omega_1 R_{t-1} + \omega_2 R_{t-2} + \cdots + \omega_n R_{t-n} + \Theta_0 \quad (3.46)$$

The constant Θ_0 is included to account for the fact that the noise term (dry weather or base flow) may not have a zero mean. The polynomial of I coefficients is called the inverse Green's function. The size of polynomials I and ω is determined by observing the decrease in the mean residual error and by the requirement that the error series represents an uncorrelated series—white noise. The error series is created by generating a series of one-step-ahead forecasts using the model and subtracting the measured values from the forecasts as described by Box and Jenkins (1976).

The identification of coefficients $I(B)$ and $\omega(B)$ and the constant Θ_0 is done by standard linear multiple-regression routines. Statistical autocorrelation routines are used for the analysis of the series of residuals (errors).

The polynomial of coefficients ω contains the UH function $h(B)$; however, it is "corrupted" by the autocorrelative and moving average terms of the noise. The filtering process involves the division of polynomials $\omega(B)$ and $I(B)$ to yield the UH $h(B)$ polynomial, or

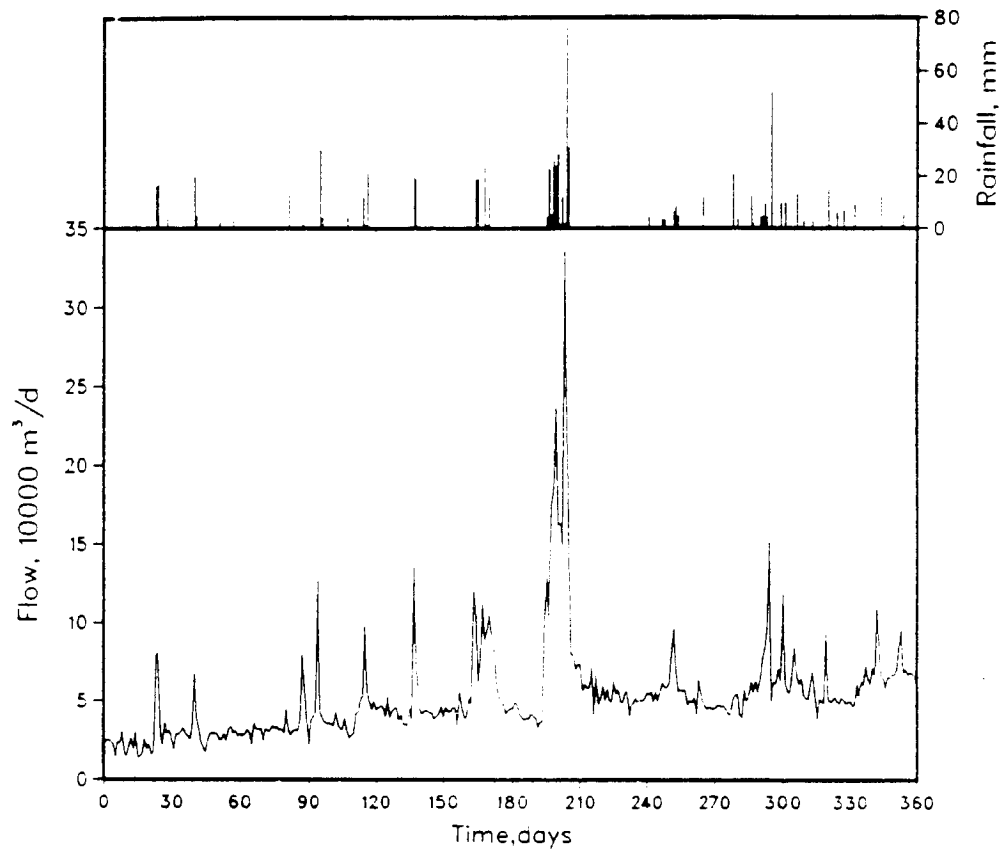


FIGURE 3.28. Rainfall and influent flows time series for an urban watershed with combined sewers (Fusina, Region Veneto in Italy).

$$h(B) = \frac{\omega(B)}{I(B)} \quad (3.47)$$

Example 3.11: Estimation of the UH Function by Stochastic Modeling

The method of Novotny and Zheng is used to estimate the UH function for a combined sewer system. The daily rainfall and corresponding flow series are shown in Figure 3.28.

The multiple regression of the time series of rainfalls and flows was performed (Equation (3.46)) with varying magnitudes of the maximum lags, m and n . In the analysis, m and n were kept equal. Using the F-test the significance of the reduction of the mean square residuals was tested along with the testing of the residual series as to whether they are *white noise*. Table 3.18 shows the results of the estimation. It can be seen that a transfer-function model of the order $m = n = 7$ provides an adequate model. Figure 3.29 shows the autocorrelation function of the residuals.

TABLE 3.18 Coefficient Estimation for Fusina System

Parameters (1)	ARTF Order							
	(1, 1) (2)	(3, 3) (3)	(5, 5) (4)	(7, 7) (5)	(9, 9) (6)	(11, 11) (7)	(13, 13) (8)	(15, 15) (9)
I_0	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
I_1	0.77	0.55	0.52	0.52	0.52	0.51	0.51	0.53
I_2	—	0.16	0.15	0.18	0.18	0.17	0.18	0.15
I_3	—	0.14	0.11	0.15	0.15	0.15	0.13	0.13
I_4	—	—	0.09	0.12	0.10	0.11	0.11	0.12
I_5	—	—	0.08	0.09	0.08	0.07	0.07	0.05
I_6	—	—	—	-0.11	-0.12	-0.13	-0.12	-0.11
I_7	—	—	—	-0.08	-0.07	-0.08	-0.06	-0.06
I_8	—	—	—	—	0.07	0.06	0.03	0.02
I_9	—	—	—	—	-0.03	-0.02	-0.06	-0.03
I_{10}	—	—	—	—	—	0.10	0.07	0.06
I_{11}	—	—	—	—	—	-0.04	-0.07	-0.07
ω_0	0.23	0.23	0.23	0.23	0.23	0.23	0.23	0.23
ω_1	-0.10	-0.06	-0.05	-0.05	-0.05	-0.05	-0.05	-0.05
ω_2	—	-0.04	-0.04	-0.05	-0.05	-0.04	-0.04	-0.04
ω_3	—	-0.05	-0.05	-0.06	-0.05	-0.05	-0.05	-0.05
ω_4	—	—	-0.03	-0.04	-0.04	-0.04	-0.03	-0.04
ω_5	—	—	-0.01	-0.03	-0.03	-0.02	-0.02	-0.02
ω_6	—	—	—	0.04	0.04	0.05	0.05	0.05
ω_7	—	—	—	0.01	0.01	0.01	0.01	0.01
ω_8	—	—	—	—	-0.02	-0.02	-0.01	-0.01
ω_9	—	—	—	—	-0.01	-0.01	0.00	0.00
ω_{10}	—	—	—	—	—	-0.03	-0.02	-0.02
ω_{11}	—	—	—	—	—	0.00	0.01	0.01
Θ_0	0.98	0.62	0.54	0.62	0.56	0.51	0.44	0.43
SSR	499	499	443	424	418	411	396	389
MSR	1.38	1.25	1.24	1.19	1.18	1.16	1.13	1.11
F-test	—	19	3.2	7.8	2.5	2.9	2.1	3.0

Note: The adequate model is ARTF (7, 7).

If the autocorrelation coefficients of the residuals for all lags greater than zero are close to zero, then the series is a white noise. Figure 3.30 then shows the filtered UH function $h(B)$, which has the character of a multiple reservoir function for surface runoff. It is interesting to note that the UH function shows a secondary peak with a much longer response time. Since both surface runoff and infiltration (interflow + ground-water inputs) are correlated with rainfall with different response times, the UH response function is composed of two parts, one with a shorter response

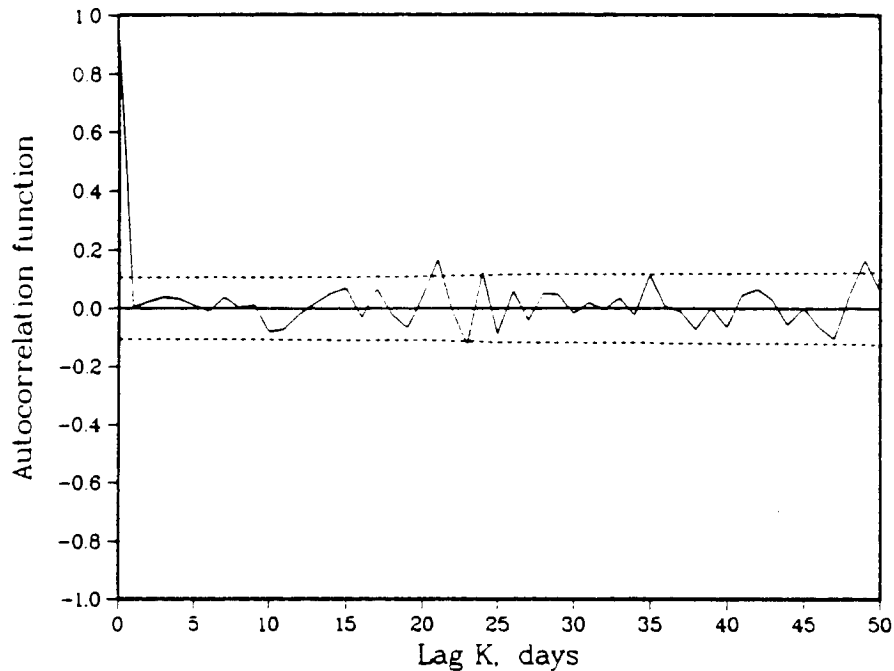


FIGURE 3.29. Autocorrelation function of residuals for Fusina time series, indicating that the residuals are a random white noise.

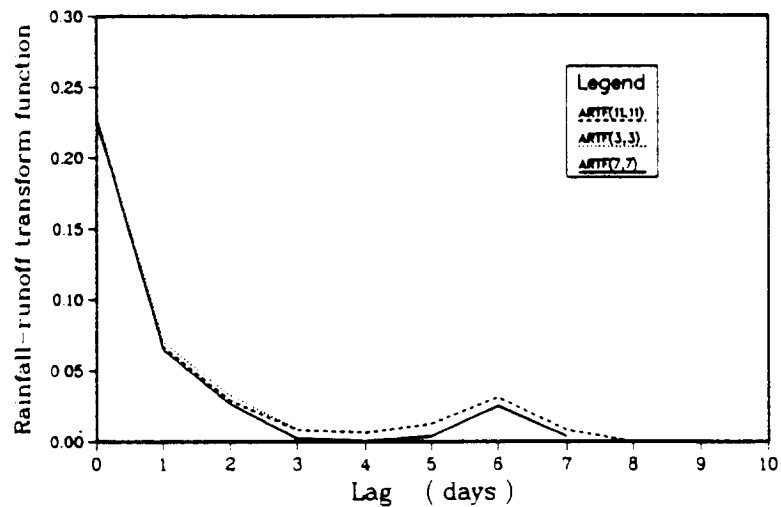


FIGURE 3.30. Filtered unit hydrograph obtained from the Fusina rainfall and runoff time series.

time for surface runoff, and the second, longer response component for infiltration. Hence, the rainfall–runoff transfer function obtained by this method can be used for quantitative separation and evaluation of surface runoff, ground-water inputs, and dry-weather sewage-flow components.

The polynomial $I(B)$ contains most of the information on the dry-weather component that is not related to the rainfall (Zheng, 1989).

III-5 Interflow

Interflow is that part of the subsurface flow that moves at shallow depths and reaches the surface channels in a relatively short period of time. It is therefore commonly considered part of the direct surface runoff (Viessman, Lewis, and Knapp, 1989). Although the quantity of interflow may represent only a small portion of total runoff, for some mobile pollutants its pollution effect may be of the same order of magnitude as the surface runoff. Interflow pollution originates mainly from salts and pesticides deposited in soils. On the other hand, interflow may be free of suspended pollutants.

The occurrence of interflow may be observed in areas where the permeability of the subsoils is less than that of the upper soil zone, causing horizontal movement of water in the upper soil zone. Lateral water movement in soils is especially significant during the snowmelt process, when subsoils are still frozen.

Theoretically, the amount of interflow could be computed from Darcy's law if the depth of the saturated upper zone storage, the saturation permeability, and the slope of the piezometric soil water surface were known. The Stanford Watershed Model (Crawford and Linsley, 1966) approximated the outflow from the interflow storage as

$$\text{INTF} = \alpha \text{SRGX} \quad (3.48)$$

where

SRGX = current volume of water in interflow storage

α = an empirical coefficient

The volume of the interflow storage is continuously calculated by the model using mass continuity equations in which the infiltration from the surface is the source of water, and the ground-water recharge and evapotranspiration are the losses or outflow.

GROUND-WATER SYSTEMS

Ground-water movement and occurrence are an integral part of the hydrologic cycle. Almost all ground water originates from infiltrated precipitation after subtraction of the surface losses—surface runoff, evapotranspiration, and interflow. During prolonged dry periods, most of

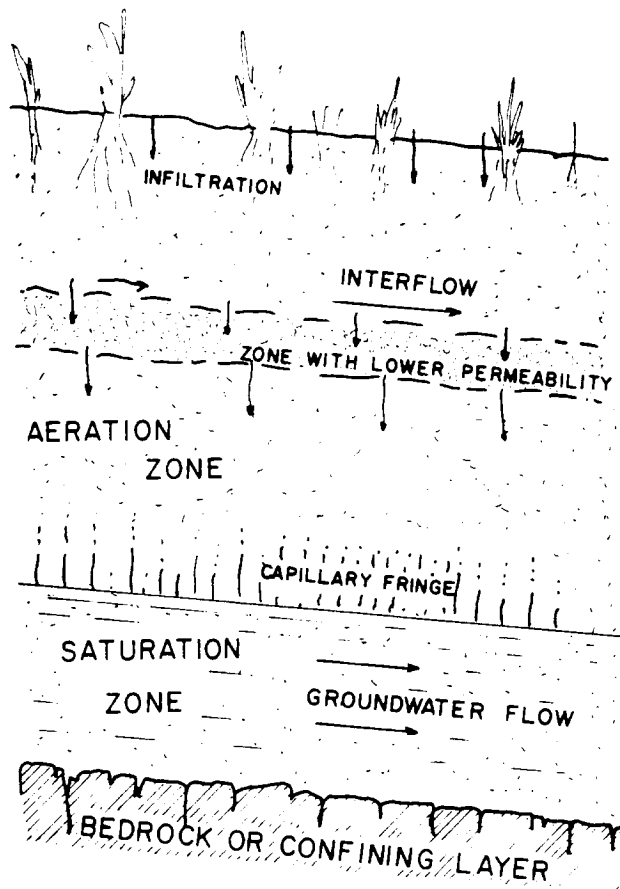


FIGURE 3.31. Soil and ground-water flow zones.

the natural flow in surface waters, as well as infiltration into deep sewers, originates from ground-water systems and is referred to as *base flow* or *ground-water runoff*.

Subsurface Distribution of Water

Water below the ground's surface occurs in four zones, which are shown in Figure 3.31 (Todd, 1959):

1. *Soil water zone*, which begins at the surface and extends downward to the end of the root zone. This zone is commonly unsaturated, except during periods of heavy infiltration. Its depth varies, but is generally from a fraction of a meter to a few meters thick.
2. *Intermediate zone* extends from the bottom of the soil zone to the top of the capillary fringe zone.
3. *Capillary zone* extends from the ground-water table, and its height is

determined by the capillary rise of water in pores of the soil. Due to capillary suction forces, water in this zone occurs at less than piezometric atmospheric pressures.

4. *Saturated zone*. The pores are completely filled with water, and water exists at pressures that are greater than or equal to the atmospheric pressure. The *ground-water table* is defined as the plane of a fully saturated zone where water occurs at atmospheric pressure.

The soil and intermediate zones are called *vadose zones* (in Latin *vadosus* means shallow) or zones of aeration. The zone of aeration contains voids and cracks that are partially occupied by water and air. In the saturated and capillary zones, all interstices are filled by water.

The direction of water movement in the vadose zone is primarily vertical, except when the topsoil zone becomes saturated and the interflow moves in a lateral direction. The direction of water movement in the saturated zone is lateral (nearly horizontal). The top of the saturated zone—the ground-water table—can be found in wells and borings that penetrate into the saturation zone.

The vadose zone is the place where most of the pollutant–soil interactions take place. It is a transition zone between surface contamination and ground-water pollution. Many pollutants are effectively adsorbed by soil particles and/or are decomposed in soils and will not penetrate the vadose zone, but other pollutants will penetrate the vadose zone and may potentially contaminate ground-water resources.

The vadose zone can be absent under the high ground-water conditions exhibited especially in wetland areas, or may be several hundred meters thick in arid and semiarid regions. Part of the water in the intermediate zone is held by hygroscopic and capillary forces. The excess moves downward by gravity. Water that can be drained from soil by gravity is known as *specific yield* and is given as the volume of water that can be drained by gravity to the gross volume of the soils. The specific yield depends on the type of soil.

Saturated Zone, Aquifers, and Aquitards

A geological formation saturated by water that yields appreciable quantities of water that can be economically used and developed is called an *aquifer* (Todd, 1959). An aquifer can be either confined or unconfined (Fig. 3.32). An unconfined aquifer is one in which the upper boundary of the saturation zone is the same as the water table. Confined aquifers, also known as artesian or pressure aquifers, are overlain by an impermeable geological stratum that keeps water under pressure. Water enters an

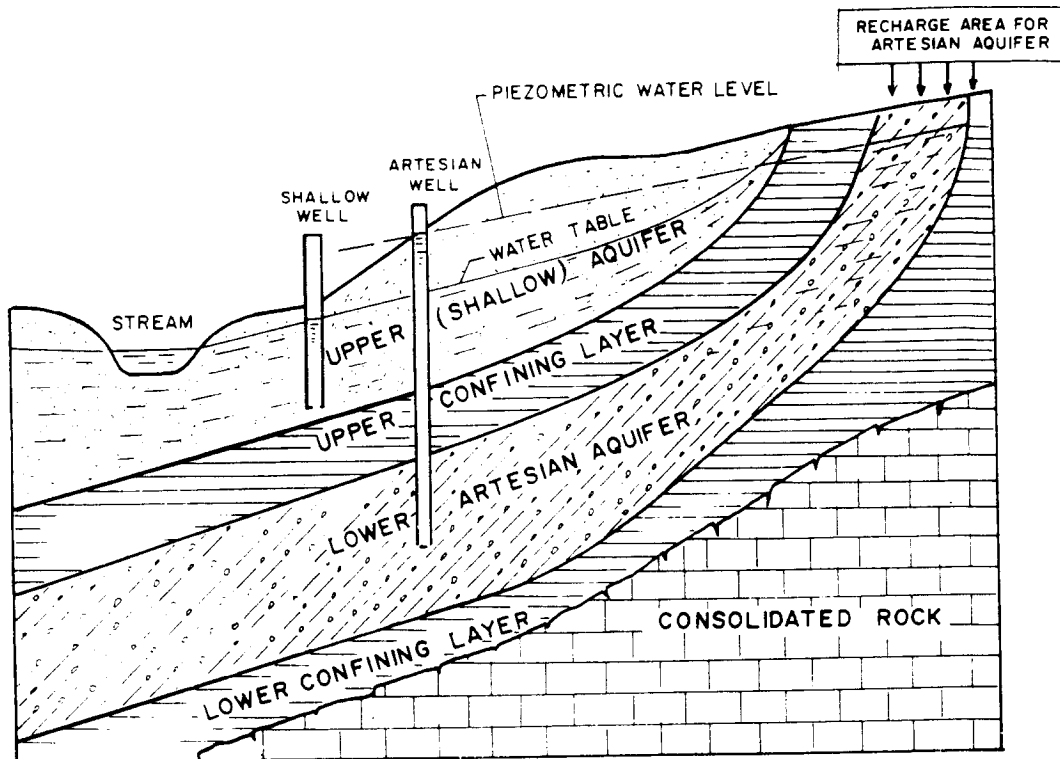


FIGURE 3.32. Representation of aquifers.

aquifer through a *recharge area*, which is an area where the waterbearing stratum is exposed to the atmosphere or is overlain by a permeable zone of aeration.

Aquitards are geological formations that are not permeable enough for the economic development as a ground-water source. An *aquiclude* is a formation that stores water, but is incapable of transmitting (e.g., clays). A solid rock formation that neither transmits nor stores water is an *aquifuge*.

Aquifers and aquitards can exist in layers with an unconfined aquifer on the top, and underlain by one or more confined zones. The top unconfined aquifer, often called a *shallow aquifer*, is most susceptible to diffuse pollution and contamination. Ground water that can be recovered by springs and wells represents the ground-water runoff or base flow.

Relationship between Surface and Ground-Water Systems

Ground- and surface-water systems are interrelated through two processes: *recharge* and *discharge*. There are two major sources of natural recharge to an aquifer. The first is the residual of precipitation that

infiltrates through the unsaturated (vadose) zone, the second is freshwater inflow from surface-water bodies, such as streams, rivers, lakes, and wetlands. In addition, aquifers may be recharged by septic tanks, irrigation, artificial recharge, and sewer leakage. Natural discharge from aquifers occurs through springs, spring-fed lakes, wetlands, and oceans. In addition, large plants known as phreatophytes, whose roots extend to the water table, extract water from the aquifer by the transpiration process. Man extracts water from aquifers by pumping from wells, by intercepting ground water in galleries and drainage pipes, and by other more sophisticated systems. Under natural conditions, natural recharge and discharge are commonly in balance. Perennial streams in more humid regions are connected with ground-water aquifers. In some places aquifers discharge into streams, providing base flow. In some other areas streams and lakes discharge into aquifers. Ephemeral streams in arid regions are not connected with a discharging aquifer, that is, the ground-water table is well below the bottom elevation of the body of water and can only recharge the aquifer.

Depending on where recharge and discharge take place, ground-water flow systems can be divided into three types; local, intermediate, and regional (Toth, 1963). A local flow system has its recharge area in a topographic high and discharges into an adjacent topographic low. The

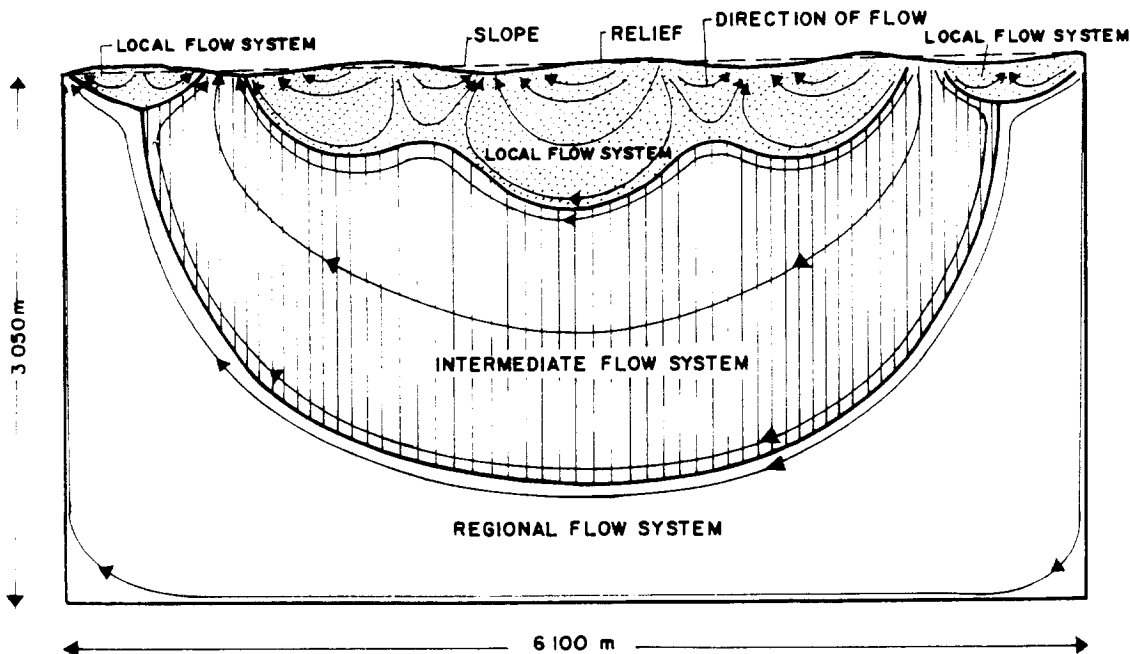


FIGURE 3.33. Theoretical patterns for local, intermediate, and regional ground-water flow systems with recharge and discharge. (Adapted from Toth, 1968. Copyright © 1968 by the American Geophysical Union; reprinted with permission.)

intermediate flow system is viewed as having recharge and discharge areas that are not in adjacent topographic highs and lows. The discharge area for the intermediate system may be located several subbasins downstream. The regional system has its recharge in the regional topographic high, while its discharge occupies the topographic low for the basin (Fig. 3.33).

The flow in ground-water systems is generally slow, and the response to surface and subsurface pollution loadings is often gradual. The residence of water in local ground-water systems may range from days to months, intermediate flow systems may have residence times ranging from months to years, and the residence time of water in regional systems may reach centuries or more. A major part of ground water in regional aquifers of the midwestern United States originates from glaciers that melted during the postglacial period thousands of years ago. In most cases, surface contamination by pollutants affects only local ground-water systems and shallow aquifers.

The three systems have well-defined boundaries that theoretically identify changes in flow patterns, water quality, and the relative rate of ground-water movement. The water movement in a local system is commonly faster, which produces water quality that is different from the regional system. Due to the longer residence time and slower flow rates, regional systems produce ground water with higher concentrations of

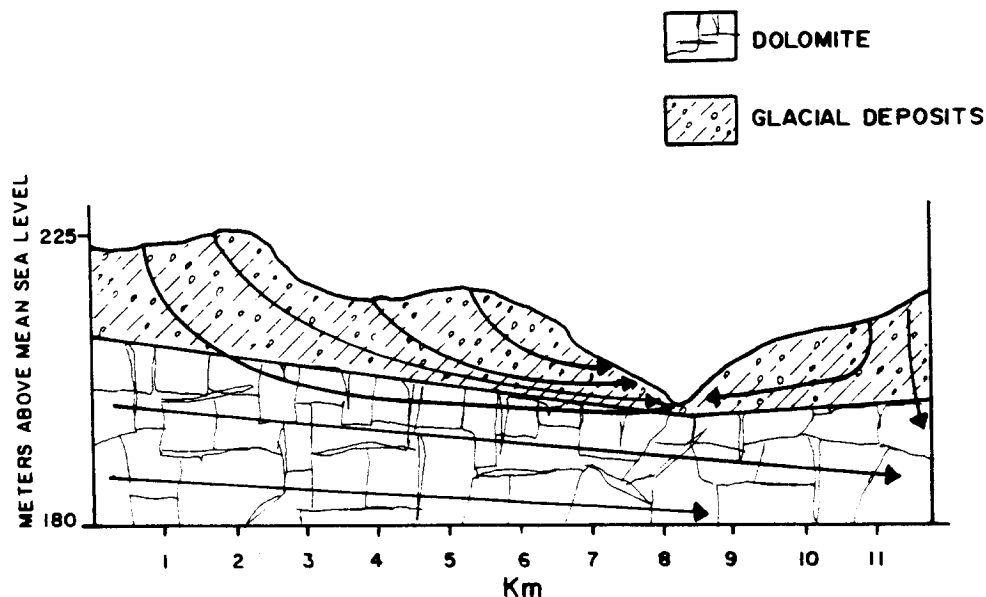


FIGURE 3.34. Generalized flow patterns for local flow systems in a shallow glacial aquifer underlain by a dolomite aquifer. (After Eisen and Anderson, 1978.)

dissolved solids and ions than occur in ground water moving through a local system.

The depths at which the boundaries between the flow systems develop are dependent upon the following geomorphological factors: local topographic relief and its steepness, distribution and depth of bedrock and impermeable geological strata, aquifer thickness, and general topographic and geomorphological characteristics of the basin.

The computer simulation by Toth (1963) indicated that the development, intensity, and depth of local flow systems could be directly related to increased local topographic relief, and inversely to increased regional slope. Strong regional flow systems developed where local relief is negligible and where the ratio of total aquifer thickness to local relief was high. A weak local flow system will have only a portion of the total aquifer contributing discharge to that subbasin. Where there is pronounced local relief, mainly local systems develop. A strong local system has most or all of the aquifer discharging into that subbasin. Figure 3.34 shows typical flow patterns for a glacial aquifer overlaying dolomite and the flow patterns in the two aquifers.

Ground-Water Hydrological Balance

Figure 3.35 shows the relation between the recharge and discharge of a surface-ground-water system. Note that the boundaries of the systems are determined by the extent of the ground-water system. Hence several surface basins may be included for intermediate and regional systems.

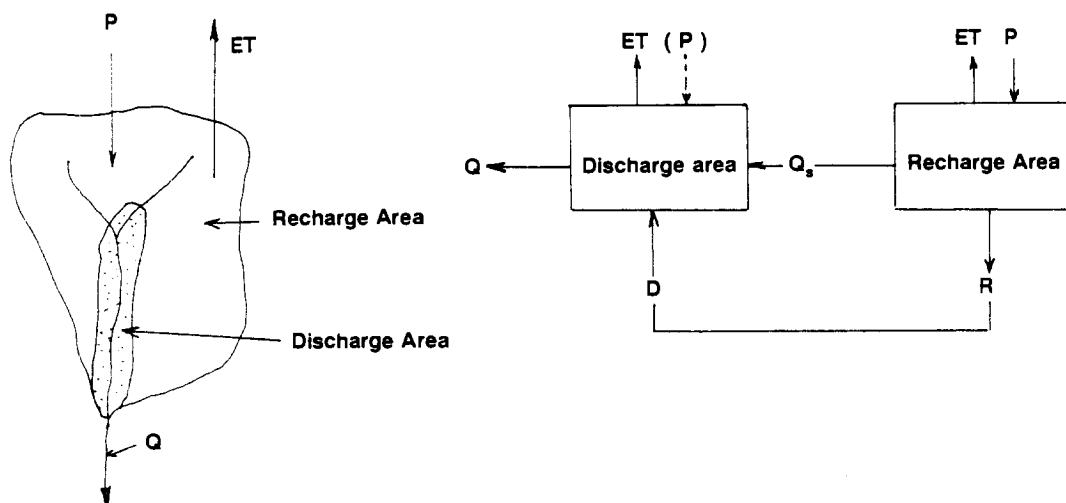


FIGURE 3.35. Recharge-discharge relationships for ground-water systems. (After Freeze and Cherry, 1979.)

The separation of the surface of the system into the recharge and discharge areas enables us to consider the surface and subsurface components of the total discharge from the system (Freeze and Cherry, 1979) and to formulate hydrologic budget for the joint system expressed first in this chapter by Equations (3.1)–(3.3). The overall hydrological balance of the joint system then becomes (all units in centimeters over the watershed area)

$$P = Q + ET + \Delta S_s + \Delta S_g \quad (3.49)$$

where

P = precipitation

Q = runoff (mostly surface, since the area is recharging ground water)

ET = evapotranspiration

ΔS_s = change in surface storage

ΔS_g = change in ground-water storage

If the balance is averaged over a period of several years, then $\Delta S_s = 0$. Hence in the recharge area

$$P = Q_s + R + ET \quad (3.50)$$

where

R = recharge rate (infiltration)

Q_s = surface runoff only because base flow is not discharging

At the watershed outlet in the discharge area

$$Q = Q_s + D - ET + P \quad (3.51)$$

where D is the discharge rate (exfiltration).

By comparing Equations (3.50) and (3.51) with the overall balance (Eq. (3.49)), it follows that $\Delta S_g = R - D$. Under steady-state balanced conditions over a long period of time $\Delta S_g = 0$, and hence, $R = D$. In a dry year or when discharge is increased by man (for example, by excessive withdrawals for irrigation and water supply), $D > R$ and $\Delta S_g < 0$, and the ground-water table is decreasing. If the decrease is caused by excessive use of the ground-water resource by man, then the aquifer is being mined. An amount of water that can be withdrawn from the aquifer while maintaining the aquifer storage constant is called *safe yield*.

By setting $Q_g = D - ET$ as the base flow provided by the aquifer into

the river system, and neglecting P in the discharge area, Equation (3.51) becomes

$$Q = Q_g + Q_s$$

In the analysis of water quality pollution and contamination from diffuse sources, one has to determine whether the source of contamination is surface or ground-water discharge. It is not proper—as was common in nonpoint pollution studies of the 1970s and early 1980s—to focus only on the surface runoff component.

Ground-Water Hydrological Models

The state of the art of ground-water hydrological modeling is well advanced, and numerous models have been developed to represent the flow and quality conditions of aquifers. Some fundamental models are introduced and discussed in Chapter 7. Comprehensive reviews were prepared by Kisel and Duckstein (1976), Anderson (1979), and Bedient, Borden, and Leib (1985), among others. Also most of the basic hydrological texts mentioned in this chapter contain discussions on the fundamentals of ground-water modeling and descriptions of the most common ground-water flow and quality models. These models can be either distributed-parameter flow models or lumped-parameter aquifer models. The lumped systems consider aquifers as homogenous, and the lumped values can be obtained by averaging the aquifer characteristics at a few discrete points or by estimating the parameters from the response of the aquifer to a known input, for example, a dye injection.

Many models are three-dimensional, box-grid representations that use finite-difference approximations of the flow equations (Wang and Anderson, 1982). Due to the linear nature of the basic flow equations, many ground-water problems can be solved by electric analog simulations. For diffuse pollution studies, ground-water flow models provide basic information on the residence times of contaminants in ground-water systems, subsurface flow patterns and dispersion of contaminants, base flow and drainage infiltration pollutant loads, and other valuable data needed in comprehensive diffuse-pollution studies.

References

- American Society of Civil Engineers (1982). *Design and Construction of Sanitary and Storm Sewers*, ASCE-WPCF Manual No. 9, ASCE, New York.
- Anderson, E. A., and N. H. Crawford (1964). *The Synthesis of the Continuous Snowmelt Runoff Hydrographs on a Digital Computer*, Tech. Rep. No. 26, Dept. of Civil Eng., Stanford University, Palo Alto, CA.

- Anderson, M. P. (1979). Using models to simulate the movement of contaminants through groundwater flow systems, *CRC Crit. Rev. Environ. Control* 9(2):97-156.
- Anonymous (1972). *Hydrocomp Simulation Programming Operation Manual*, Hydrocomp International, Palo Alto, CA.
- Bedient, P. B., and W. C. Huber (1988). *Hydrology and Floodplain Analysis*, Addison-Wesley, Reading, MA.
- Bedient, P. B., R. C. Borden, and D. L. Leib (1985). Basic concepts for ground water transport modeling, in *Ground Water Quality*, C. H. Ward, W. Giger, and P. L. McCarty, eds., Wiley, New York.
- Bengtsson, L. (1982). Snowmelt-generated runoff in urban areas, in *Urban Stormwater Hydraulics and Hydrology*, B. C. Yen, ed., Water Resources Publ., Littleton, CO, pp. 442-451.
- Bengtsson, L. (1984). Modeling snowmelt induced runoff with short time resolution, *Proceedings, Third International Conference on Urban Storm Drainage*, P. Balmér, P. A. Malmqvist, and A. Sjöberg, eds., Chalmers University of Technology, Göteborg, Sweden, pp. 305-314.
- Box, G. E. P., and G. M. Jenkins (1976). *Time Series Analysis: Forecasting and Control*, Holden-Day, Oakland, CA.
- Bras, R. L. (1990). *HYDROLOGY: An Introduction to Hydrologic Science*, Addison-Wesley, Reading, MA.
- Bras, R. L., and I. Rodriguez-Iturbe (1985). *Random Functions and Hydrology*, Addison-Wesley, Reading, MA.
- Braslavskii, A. P., and Z. A. Vikulina (1954). *Evaporation Norms for Water Reservoirs*, Gidrometeorologicheskoe izdatelstvo, Leningrad. (English translation available from Israel Program for Scientific Translations, Jerusalem, 1963.)
- Brutsaert, W. (1976). The concise formulation of diffusive sorption of water in a dry soil, *Water Resour. Res.* 12:1118-1124.
- Burt, T. P., and P. J. Williams (1976). Hydraulic conductivity in frozen soils, *Earth Surf. Processes* 1:349-360.
- Chong, S. K., and R. E. Green (1983). Sorptivity measurement and its application, in *Advances in Infiltration*, ASAE Publ. No. 11-83, American Society of Agricultural Engineers, St. Joseph, MI, pp. 82-91.
- Chow, V. T., ed. (1964). *Handbook of Applied Hydrology*, McGraw-Hill, New York.
- Chow, V. T., D. R. Maidment, and L. W. Mays (1988). *Applied Hydrology*, McGraw-Hill, New York.
- Crawford, N. H., and R. K. Linsley (1966). *Digital Simulation in Hydrology, Stanford Watershed Model IV*, Tech. Rep. No. 39, Dept. of Civil Eng., Stanford University, Palo Alto, CA.
- Davis, C. V., and K. E. Sorensen (1969). *Handbook of Applied Hydraulics*, McGraw-Hill, New York.
- Eisen, C. E., and M. P. Anderson (1978). *Field Data Quantifying Groundwater Surface Water Interaction*, Final Report Menomonee River Pilot Watershed Study, International Joint Commission, Windsor, Ontario.

- Ellena, M., and V. Novotny (1985). Computer simulation of the urban snowmelt process, *Proceedings, 1985 International Symposium on Urban Hydrology, Hydraulic Infrastructures and Water Quality*, University of Kentucky, Lexington, KY.
- Engman, E. T. (1974). Partial area hydrology and its application to water resources, *Water Resour. Bull.* **10**(3):512–521.
- Freeze, R. A., and J. A. Cherry (1979). *Groundwater*, Prentice-Hall, Englewood Cliffs, NJ.
- Gray, D. M. (1961). Synthetic unit hydrograph for small watersheds, *J. Hydraul. Div. ASCE* **87**:33–54.
- Gray, D. M., ed. (1973). *Handbook on the Principles of Hydrology*, Water Information Center, Pt. Washington, NY.
- Green, W. H., and G. A. Ampt (1911). Studies in soil physics. The flow of air and water through soils, *J. Agr. Sci.* **4**:1–24.
- Hall, M. J. (1984). *Urban Hydrology*, Elsevier, Amsterdam.
- Harbeck, G. E., Jr. (1962). *A Practical Field Technique for Measuring Reservoir Evaporation Utilizing Mass Transfer Theory*, U.S. Geological Survey Prof. Paper 272-E, Washington, DC.
- Helfrich, K. R., E. E. Adams, A. L. Godbey, and D. R. Harleman (1982). *Evaluation of Models for Predicting Evaporation Water Loss in Cooling Impoundments*, Interim Rep. No. CS-2325, Electric Power Research Institute, Palo Alto, CA.
- Henderson, F. M., and R. A. Wooding (1964). Overland flow and groundwater flow from a steady rainfall of finite duration, *J. Geophys. Res.* **69**:114–121.
- Hiemstra, L. (1968). *Frequencies of Runoff for Small Basins*, Ph.D. thesis, Colorado State University, Fort Collins, CO.
- Hillel, D. (1980). *Applications of Soil Physics*, Academic Press, Orlando, FL.
- Holtan, H. N. (1961). *A Concept for Infiltration Estimates in Watershed Engineering*, USDA Agricultural Research Service, Washington, DC, pp. 41–51.
- Holtan, H. N., and N. C. Lopez (1973). *USDAHL-73 Revised Model of Watershed Hydrology*, Rep. No. 1, USDA Agricultural Research Service, Plant Physiology Institute, Washington, DC.
- Horn, M. E. (1971). Estimating soil permeability rates, *J. Irrig. Drainage Div. ASCE* **97**:263–274.
- Horton, R. E. (1939a). Approach toward a physical interpretation of infiltration capacity, *Proc. Soil Sci. Soc. Am.* **5**:399–417.
- Horton, R. E. (1939b). The interpretation and application of runoff plot experiments with reference to soil erosion problem, *Proc. Soil Sci. Soc. Am.* **3**:340–349.
- Huggins, L. F., T. H. Podmore, and C. F. Hood (1976). *Hydrologic Simulation Using Distributed Parameters*, Purdue University, Water Resources Research Center Rep. No. 82, West Lafayette, IN.
- Izzard, C. F. (1946). Hydraulics of runoff from developed surfaces, *Proc. Highway Res. Board* **26**:129–150.
- Johanson, R. C., J. C. Imhoff, J. L. Kittle, and A. S. Donigian (1984).

- Hydrologic Simulation Program—Fortran (HSPF): User's Manual*, EPA-600/3-84-006, U.S. Environmental Protection Agency, Athens, GA.
- Kane, P. L. (1980). Snowmelt infiltration into seasonally frozen soils, *Cold Region Sci. Tech.* 3:153–161.
- Kane, P. L., and J. Stein (1983). Physics of snowmelt infiltration into seasonally frozen soils, in *Advances in Infiltration*, ASAE Publ. No. 11-83, American Society of Agricultural Engineers, St. Joseph, MI, pp. 178–187.
- Kisel, C. C., and L. Duckstein (1976). Groundwater models, in *Systems Approach to Water Management*, A. K. Biswas, ed., McGraw-Hill, New York.
- Larsen, V., J. H. Axley, and G. L. Miller (1972). *Agricultural Wastewater Accommodation and Utilization of Various Forages*, WRRC A-006-Md, 14-01-0001-790, University of Maryland, College Park, MD.
- Linsley, R.K. (1967). The relation between rainfall and runoff, *J. Hydrol.* 5(4):297–311.
- Linsley, R. K., M. A. Kohler, and J. L. H. Paulhus (1982). *Hydrology for Engineers*, 3d ed., McGraw-Hill, New York.
- Livingston, E. H., and L. A. Roesner (1991). Stormwater management practices for water quality enhancement, in *Manual of Practice—Urban Stormwater Management*, American Society of Civil Engineers, New York.
- Lundin, L. C. (1989). *Water and Heat Flow in Frozen Soils. Basic Theory and Operational Modeling*, Acta Universitatis Upsaliensis, Upsala, Sweden.
- McCuen, R. H. (1982). *A Guide to Hydrologic Analysis Using SCS Method*, Prentice-Hall, Englewood Cliffs, NJ.
- McCuen, R. H. (1989). *Hydrologic Analysis and Design*, Prentice-Hall, Englewood Cliffs, NJ.
- McCuen, R. H., and W. M. Snyder (1986). *Hydrologic Modeling. Statistical Methods and Applications*, Prentice-Hall, Englewood Cliffs, NJ.
- McDaniel, L. L. (1960). *Consumptive Use of Water by Major Crops in Texas*, Texas Water Devel. Board Bull. 6019, Austin, TX.
- Marciano, T. T., and G. E. Harbeck (1954). *Mass Transfer Studies in Water Loss Investigation, Lake Hefner Studies*. U.S. Geological Survey Professional Paper No. 269. Washington, DC.
- Morgali, J. R., and R. K. Linsley (1965). Computer analysis of overland flow, *J. Hydraul. Div. ASCE* 91:81–100.
- Nash, J. E. (1957). The form of the Instantaneous Unit Hydrograph, *Bull. Int. Assoc. Sci. Hydrol.* 111:114–121.
- Novotny, V. (1986). *Effects of Pollutants from Snow and Ice on Quality of Water from Urban Drainage Basins*, Tech. Report., Dept. of Civil Eng, Marquette University, Milwaukee, WI (available from National Technical Information Service, Springfield, VA).
- Novotny, V. (1988). Modeling urban runoff pollution during winter and off-winter periods, in *Advances in Environmental Modeling*, A. Marani, ed., Elsevier, Amsterdam, pp. 43–57.
- Novotny, V., and S. Zheng (1989). Rainfall-runoff transfer function by ARMA modeling, *J. Hydraul. Eng. ASCE* 115(10):1386–1400.

- Novotny, V., K. R. Imhoff, M. Olthof, and P. A. Krenkel (1989). *Karl Imhoff's Handbook of Urban Drainage and Wastewater Disposal*, Wiley, New York.
- Overton, D. E., and M. E. Meadows (1976). *Stormwater Modeling*, Academic Press, New York.
- Philip, J. R. (1957). The theory of infiltration: 1. The infiltration equation and its solution, *Soil Sci.* **83**(5):345-357.
- Philip, J. R. (1969). Theory of infiltration, in *Advances in Hydroscience*, vol. 5, V. T. Chow, ed., Academic Press, New York, pp. 215-296.
- Philip, J. R. (1983). Infiltration in one, two, and three dimensions, in *Advances in Infiltration*, ASAE Publ. No. 11-83, American Society of Agricultural Engineers, St. Joseph, MI, pp. 1-23.
- Ponce, V. M. (1989). *Engineering Hydrology*, Prentice Hall, Englewood Cliffs, NJ.
- Priesley, C. H. B. (1959). *Turbulent Transfer of the Lower Atmosphere*, University of Chicago Press, Chicago, IL.
- Rao, R. A., J. W. Delleur, and B. S. P. Sarma (1972). Conceptual hydrologic model for urbanizing watersheds, *J. Hydraul. Div. ASCE* **98**:1205-1220.
- Rawls, M. J., and D. L. Brakensiek (1983). A procedure to predict Green and Ampt infiltration parameters, in *Advances in Infiltration*, ASAE Publ. No. 11-83, American Society of Agricultural Engineers, St. Joseph, MI, pp. 102-112.
- Ritchie, J. C., J. A. Sparberry, and J. R. McHenry (1974). Estimating soil erosion from the redistribution of fallout of Cs-137, *Soil Sci. Soc. Am. Proc.* **38**:283-286.
- Romanov, V. V., K. K. Pavlova, and I. L. Kolyushnyy (1974). Meltwater losses through infiltration into podzolic soils and chernozems, *Soviet Hydrology: Selected Papers, No. 1*, pp. 32-42.
- Sharma, M. L., R. J. W. Barron, and K. S. DeBoer (1983). Spatial structure and variability of infiltration parameters, in *Advances in Infiltration*, ASAE Publ. No. 11-83, American Society of Agricultural Engineers, St. Joseph, MI, pp. 113-121.
- Sherman, L. K. (1932). Streamflow from rainfall by unit-graph method, *Eng. News Rec.*, p. 501.
- Soil Conservation Service (1968). Hydrology, supplement A to sec. 4, *Engineering Handbook*, USDA-SCS, Washington, DC.
- Tholin, A. L., and C. J. Keifer (1960). *The Hydrology of Urban Runoff*, Trans. ASCE, paper 3061.
- Todd, D.K. (1959). *Groundwater Hydrology*, Wiley, New York.
- Toth, J. (1968). A theoretical analysis of groundwater flow in small drainage basins, *J. Geophys. Res.* **68**:4795-4812.
- U.S. Army Corps of Engineers (1956). *Snow Hydrology*, North Pacific Division, Portland, OR.
- Viessman, W., Jr., G. L. Lewis, and J. W. Knapp (1989). *Introduction to Hydrology*, 3d ed., Harper & Row, Cambridge, Philadelphia.
- Wang, H. F., and M. P. Anderson (1982). *Introduction to Groundwater*

- Modeling, Finite Difference and Finite Element Methods*, W. H. Freeman, San Francisco.
- Westerström, G. (1984). Snowmelt runoff from Porsön residential area, Luleå, Sweden, *Proceedings, Third International Conference on Urban Storm Drainage*, P. Balmér, P. A. Malmqvist, and A. Sjöberg, eds., Chalmers University of Technology, Göteborg, Sweden, pp. 315-324.
- Wooding, R. A. (1965). A hydraulic model for the catchment-stream problem, Parts I and II, *J. Hydrol.* 3:254-282.
- Zheng, S. (1989). Stochastic Modeling of Urban Drainage Systems, Ph.D. thesis, Department of Civil Engineering, Marquette University, Milwaukee, WI.

4

Atmospheric Deposition

The best never let a little rain stand in their way.

Gene Kelly

The atmosphere is the portion of the environment where some of the most severe diffuse pollution problems originate, and, in fact, the magnitude of diffuse pollution often can be correlated with contamination of the atmosphere. Emissions of sulfur dioxide and nitrogen oxides by coal-burning processes and vehicular traffic cause the phenomenon of *acid rain*, which is having a severe adverse effect on many bodies of water throughout the world. Particulate aerosols in the atmosphere contain appreciable quantities of sulfur, toxic metals, pesticides, and other toxic organic compounds, fungi, pollen, soil, fly ash, nutrients, tar; and a variety of other chemical compounds, such as oxides, nitrites, nitrates, chlorides, fluorides, fluorocarbons, ozone, and silicates. Several extensive treatises have been devoted to the air-pollution problem (for example, Stern, 1976).

The adverse effects of the deposition of man-induced emissions to the atmosphere have been recognized for over a century. In the 1872 book *Air and Rain: The Beginnings of a Chemical Climatology* the British chemist Robert Angus Smith wrote that there were "three kinds of air" in and around the industrial town of Manchester, England. He described them as ". . . that with carbonate of ammonia in the fields at a distance, . . . that with sulphate of ammonia in the suburbs, . . . and that with sulfuric acid, or acid sulphate, in the town." He pointed out that acid air in the

town bleached the colors of fabrics and attacked metal surfaces, that the acid rain damaged vegetation and materials, and that substances such as arsenic, copper, and other metals were precipitated with the rain upon industrial regions.

This chapter covers the major pollution sources from the atmosphere that can be transported long distances and fall to earth in rain, snow, mist, fog, and in dry form as gases and particulates. One of the best known pollutants in atmospheric deposition is acidity. Acidity occurs when nitrogen and sulfur oxides are emitted from fossil-fuel combustion. Nitrogen in deposition can also affect the productivity of surface waters, particularly in the coastal zone.

Fossil-fuel burning also emits other substances, such as mercury, that fall to earth and can affect aquatic ecosystems. Other major atmospheric pollutants that affect water quality are trace metals, such as lead and agricultural chemicals, such as pesticides and herbicides.

INTERDEPENDENCE BETWEEN AIR AND WATER QUALITY

Many pollution studies are media specific, dealing with either water or air, but not both. However, there are many instances where the interdependence between media is important. Figure 4.1 illustrates the interdependence between water and air quality. For example, wind, temperature, and mixing patterns of both the atmosphere and a body of water can influence the concentrations of pollutants. Also, large lakes can influence precipitation patterns, and hence the rate of wet deposition of pollutants.

Atmospheric Sources of Water Pollutants

Airborne chemicals come from many natural and human sources. Before the 1800s, natural processes such as photosynthesis, decomposing organic matter, fire, volcanic activity, and wind erosion contributed gases, particles, and other byproducts to the atmosphere. Since that time, fossil-fuel burning, release of other industrial airborne chemicals, automobile exhausts, and intensive agriculture and forestry have emitted more substances to our atmosphere. In 1985, human activity accounted for an estimated 44 million tons of sulfur and nitrogen oxides released into the atmosphere (Irving, 1991). These same sources also release millions of tons of toxic metals and organic substances into the air each year. Air-pollution sources are global—emitted pollutants travel long distances

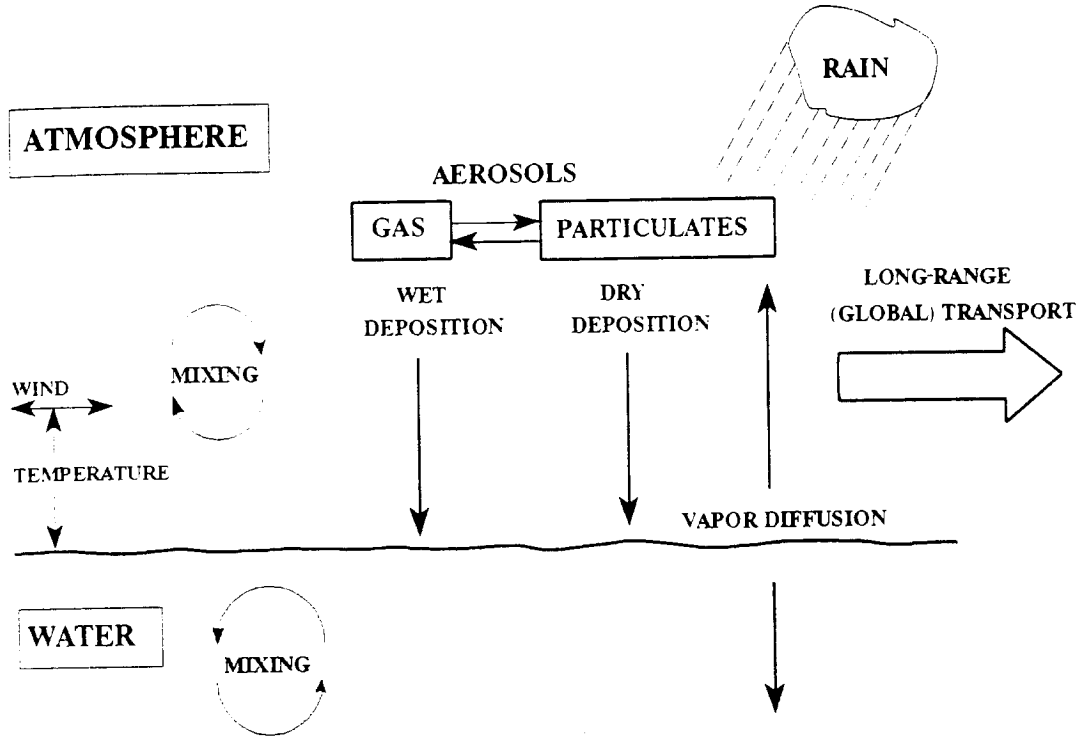


FIGURE 4.1. Water/atmospheric-interacting physical and chemical processes.

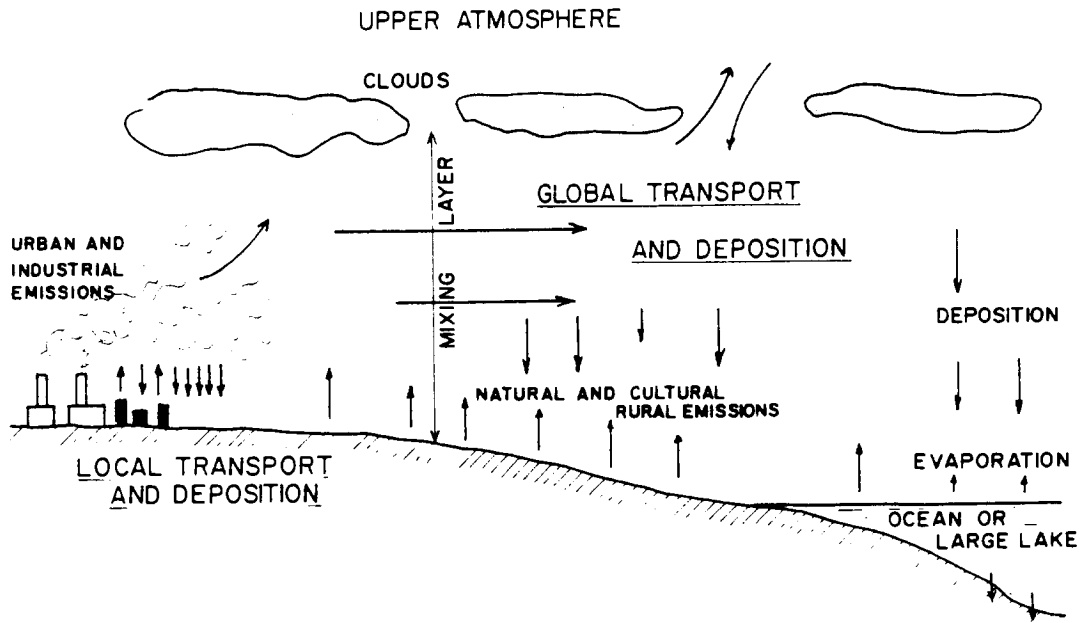


FIGURE 4.2. Local and distant sources of atmospheric deposition.

and deposition occurs over a large regional or global scale—or local (Fig. 4.2).

Major global sources of air pollution include:

1. Urban and industrial emissions resulting from human activities, such as industrial processes and domestic burning.
2. Agricultural and forest emissions resulting from such human activities as:
 - a. Soil erosion by wind during dry weather.
 - b. Slash burning, which in many parts of the world is still recommended to prevent and/or reduce the spread of disease.
 - c. Fertilizer components reaching the atmosphere through wind erosion and/or volatilization (such as the volatilization of ammonium from soils with higher pH).
 - d. Pesticides entering the atmosphere from drift during application, by wind erosion, and by volatilization.
 - e. Decomposing farm wastes and animal operations releasing ammonium, hydrogen sulfide, methane (cows), and mercaptans to the atmosphere.
3. Naturally occurring emissions on a global scale, including:
 - a. Dust blown from arid and desert areas.
 - b. Forest, brush, and grass fires.
 - c. Volcanic eruptions, which are a source of sulfuric compounds and ash.
 - d. Volatile hydrocarbons emitted from forests and other silvicultural activities.
 - e. Sea spray, which is a significant source of salt and other particulates.
 - f. Evaporation from large bodies of water, which can contribute significant quantities of volatile compounds and trace gases.

Local sources of atmospheric pollution and deposition include most of the sources just mentioned plus vehicular traffic. The magnitude of pollution, and hence atmospheric deposition, is magnified by several orders of magnitude in the vicinity of some sources, but fall off rapidly to background global levels as distance from the source is increased. For example, most of lead deposited from automobile exhaust is found within 100 m of the roadway.

Emissions of sulfur dioxide and nitrogen oxides are more thoroughly researched than other pollutants known to affect water quality, such as metals and pesticides. The annual contribution of sources of sulfur dioxide and nitrogen oxides in the United States and Canada in 1985 are

TABLE 4.1 U.S. and Canadian Emissions of SO₂, NO_x and Volatile Organic Carbon from Anthropogenic Sources Based on the 1985 NAPAP Emissions Inventory

Source	Emissions (Tg/yr ^a)		
	SO ₂	NO _x	VOC
<i>U.S. Sources</i>			
Electric utilities	14.6	6.0	neg ^b
Industrial combustion	2.4	2.9	neg
Commercial/residential/other combustion	0.6	0.7	1.7
Industrial/manufacturing processes	2.7	0.8	3.4
Transportation	0.8	8.0	8.0
Other	neg	0.1	6.9
Total ^c	21.6	18.6	20.0
<i>Canadian Sources</i>			
Electric utilities	0.7	0.2	neg
Industrial combustion	0.3	0.2	neg
Commercial/residential/other combustion	neg	0.1	0.1
Industrial/manufacturing processes	2.5	0.1	0.1
Transportation	0.1	1.2	1.0
Other	0.0	neg	0.7
Total ^c	3.7	1.9	2.2
Total United States and Canada	24.7	20.5	22.3

Source: From Irving, 1991.

^aTg = Teragrams = 10⁶ metric tons.

^bNeg = negligible < 0.1.

^cValues may not sum to totals due to independent rounding.

shown in Table 4.1 (Irving, 1991). Total annual U.S. emissions in 1985 were estimated to be 24.7 million tons for sulfur dioxide and 20.5 million tons for nitrogen oxides. Canadian emissions in 1985 were estimated to be 3.7 and 1.9 million tons of sulfur dioxide and nitrogen oxides, respectively. The largest source of sulfur dioxide emissions in the United States is from the electric utility industry (70% of total). In Canada, the largest source is from industrial/manufacturing processes (67%). Nitrogen oxides, on the other hand, are primarily contributed by mobile sources. Transportation accounts for 43% and 63%, respectively, of total emissions in the United States and Canada. Natural sources are estimated to contribute only between 1% and 5% of total U.S. emissions of sulfur dioxide and nitrogen oxides.

The largest regional sources of SO₂ emissions in the United States are from the Northeast (10.72 million tons in the Northeast versus 4.8

TABLE 4.2 Regional Anthropogenic Emissions of SO₂, NO_x, and VOC

Federal Region	Emissions (Tg/yr)		
	SO ₂	NO _x	VOC
1. New England	0.55	0.52	0.86
2. New York/New Jersey	0.77	0.89	1.44
3. Middle Atlantic	2.93	1.97	1.98
4. Southeast	4.76	3.41	3.76
5. Great Lakes	6.46	3.84	3.75
6. South Central	2.21	3.81	3.34
7. Central	1.53	1.26	0.98
8. Mountain	0.65	1.00	0.80
9. West	0.89	1.46	2.27
10. Northwest	0.23	0.48	0.86
Subtotals:			
Northeast (1-3 and 5)	10.7	7.2	8.0
Southeast (4)	4.8	3.4	3.8
West (6-10)	5.5	8.0	8.2
Total	21.0	18.6	20.0

Source: From Council on Environmental Quality (1990).

and 5.5 million tons, respectively, for the Southeast and West). On the other hand, the West proved to be the largest source of NO_x emissions (8.0 million tons versus 7.2 and 3.4 million tons, respectively, for the Northeast and Southeast). The high NO_x emissions for the West represent higher sources of emissions compared to stationary sources (Table 4.2).

Nriagu and Pacyna (1988) calculated the worldwide emissions of trace metals to the atmosphere using emissions factors and statistics on global production or consumption of industrial goods (Table 4.3). Table 4.4 compares atmospheric fallout to aquatic sources of anthropogenic inputs of trace metals into aquatic systems. The atmosphere is an important source of all metals listed when compared to domestic and industrial effluents. Atmospheric inputs are about 10% of the total liquid discharges.

Goolsby (1991) measured herbicide concentrations in rainwater from a 23-state area, principally in the Midwest and Northeast. He found traces of herbicides in all 23 states and in all but two of the 81 collection sites. Pesticide concentrations in wet deposition were measured by Glotfelty et al. (1990) in Maryland. The researchers calculated the projected amounts of pesticides entering the Chesapeake Bay from precipitation.

TABLE 4.3 Worldwide Sources of Trace Elements to the Atmosphere (10^3 kg/yr)

Source Category	Global Production/ Consumption (10^9 kg/yr ⁻¹)	As	Cd	Cr	Cu	Hg	In	Mn	Mo
Coal combustion									
—Electric utilities	[15.5×10^9 MJ]	232–1550	77–387	1240–7750	930–3100	155–542		1080–6980	232–2320
—Industry and domestic	990	198–1980	99–495	1680–11,880	1390–4950	495–2970		1485–11,880	396–2480
Oil combustion									
—Electric utilities	[5.8×10^9 MJ]	5.8–29	23–174	87–580	348–2320			58–580	58–406
—Industry and domestic	358	7.2–72	18–72	358–1790	179–1070			358–1790	107–537
Pyrometallurgical nonferrous metal production									
—Mining		40.0–80	0.6–3		160–800			415–830	
—Pb production	3.9	780–1560	39–195		234–312	7.8–16			
—Cu–Ni production	8.5	8500–12,750	1700–3400		14,450–30,600	37–207	8.5–34.0	850–4250	
—Zn–Cd production	4.6	230–690	920–4600		230–690		2.3–4.6		
Secondary nonferrous metal production			2.3–3.6		55–165			1065–28,400	

R0023168

TABLE 4.3 (Continued)

Source Category	Global Production/ Consumption (10 ⁹ kg/yr ⁻¹)	As	Cd	Cr	Cu	Hg	In	Mn	Mo
Steel and iron manufacturing	710	355–2480	28–284	2840–28,400	142–2840				
Refuse incineration									
—Municipal	140	154–392	56–1400	98–980	980–1960	140–2100		252–1260	
—Sewage sludge	3	15–60	3–36	150–450	30–180	15–60		5000–10,000	
Phosphate fertilizers	137		68–274		137–685				
Cement production	890	178–890	8.9–534	890–1780					
Wood combustion	600	60–300	60–180		600–1200	60–300			
Mobile sources	647 (gasoline)								
Miscellaneous		1250–2800							
Total, emissions		12,000–25,630	3100–21,040	7340–53,610	19,860–50,870	910–6200	11–39	10,560–65,970	793–5740
Median value		18,820	7570	30,480	35,370	3560	25	38,270	3270
Source Category	Ni	Pb	Sb	Se	Sn	Tl	V	Zn	
Coal combustion									
—Electric utilities	1395–9300	775–4650	155–775	108–775	155–755	155–620	310–4650	1085–7750	
—Industry and domestic	1980–14,850	990–9900	198–1480	792–1980	99–990	495–990	990–9900	1485–11,880	
Oil combustion									
—Electric utilities	3840–14,500	232–1740		35–290	348–2320		6960–52,200	174–1280	
—Industry and domestic	7160–28,640	716–2150		107–537	286–3580		21,480–71,600	358–2506	

Pyrometallurgical nonferrous metal production								
—Mining	800	1700-3400	18-176	18-176				310-620
—Pb production	331	11,700-31,200	195-390	195-390				195-468
—Cu-Ni production	7650	11,050-22,100	425-1700	427-1280	425-1700		43-85	4250-8500
—Zn-Cd production		5520-11,500	46-92	92-230				46,000-82,800
Secondary nonferrous metal production		90-1440	3.8-19	3.8-19				270-1440
Steel and iron manufacturing	36-7100	1065-14,200	3.6-7.1	0.8-2.2			71-1420	7100-31,950
Refuse incineration								
—Municipal	98-420	1400-2800	420-840	28-70	140-1400			2800-8400
—Sewage sludge	30-180	240-300	15-60	3-30	15-60		300-2000	150-450
Phosphate fertilizers	137-685	55-274		0.4-1.2				1370-6850
Cement production	89-890	18-14,240			2670-5340			1780-17,800 1200-6000
Wood combustion	600-1800	1200-3000						
Mobile sources		248,030						
Miscellaneous		3900-5100						1724-4783
Total, emissions	24,150-87,150	288,700-376,000	1480-5540	1810-5780	1470-10,810	3320-6950	30,150-141,860	70,250-193,500
Median Value	55,650	332,350	3510	3790	6140	5140	86,000	131,880

Source: From Nriagu and Pacyna (1988).

TABLE 4.4 Anthropogenic Inputs of Trace Metals into the Aquatic Ecosystems (10^6 kg/yr)

Source Category	Annual Global Discharge ^a (10^9 m ³)	As	Cd	Cr	Cu	Hg	Mn	Mo	Ni	Pb	Sb	Sc	V	Zn
Domestic wastewater^b														
—Central	90	1.8–8.1	0.18–1.8	8.1–36	4.5–18	0–0.18	18–81	0–2.7	9.0–54	0.9–7.2	0–2.7	0–4.5	0–2.7	9.0–45
—Noncentral	90	1.2–7.2	0.3–1.2	6.0–42	4.2–30	0–0.42	30–90	0–1.8	12–48	0.6–4.8	0–1.8	0–3.0	0–1.8	6.0–36
Steam electric	6	2.4–14	0.01–0.24	3.0–8.4	3.6–23	0–3.6	4.8–18	0.1–1.2	3.0–18	0.24–1.2	0–0.36	6.0–30	0–0.6	6.0–30
Base metal mining and dressing	0.5	0–0.75	0–0.3	0–0.7	0.1–9	0–0.15	0.8–12	0–0.6	0.01–0.5	0.25–2.5	0.04–0.35	0.25–1.0	—	0.02–6
Smelting and refining														
—Iron and steel	7						14–36			1.4–2.8				5.6–24
—Nonferrous metals	2	1.0–13	0.01–3.6	3–20	2.4–17	0–0.04	2.0–15	0.01–0.4	2.0–24	1.0–6.0	0.08–7.2	3.0–20	0–1.2	2.0–20
Manufacturing processes														
—Metals	25	0.25–1.5	0.5–1.8	15–58	10–38	0–0.75	2.5–20	0.5–5.0	0.2–7.5	2.5–22	2.8–15	0–5.0	0–0.75	25–138
—Chemicals	5	0.6–7.0	0.1–2.5	2.5–24	1.0–18	0.02–1.5	2.0–15	0–3.0	1.0–6.0	0.4–3.0	0.1–0.4	0.02–2.5	0–0.35	0.2–5.0
—Pulp and paper	3	0.36–4.2	—	0.01–1.5	0.03–0.39	—	0.03–1.5	—	0–0.12	0.01–0.9	0–0.27	0.01–0.9	—	0.09–1.5
—Petroleum products	0.3	0–0.06	—	0–0.21	0–0.06	0–0.02	—	—	0–0.06	0–0.12	0–0.03	0–0.09	—	0–0.24
Atmospheric fallout ^c		3.6–7.7	0.9–3.6	2.2–16	6.0–15	0.22–1.8	3.2–20	0.2–1.7	4.6–16	87–113	0.44–1.7	0.54–1.1	1.4–9.1	21–58
Dumping of sewage sludge^d	$[6 \times 10^9$ kg]													
Total input, water		0.4–6.7	0.08–1.3	5.8–32	2.9–22	0.01–0.31	32–1.06	0.98–4.8	1.3–20	2.9–16	0.18–2.9	0.26–3.8	0.72–4.3	2.6–31
Median value		12–70	2.1–17	45–239	35–90	0.3–8.8	109–414	1.8–21	33–194	97–180	3.9–33	10–72	2.1–21	77–375
Median value		41	9.4	142	112	4.6	262	11	113	138	18	41	12	226

Source: From Nriagu and Pacyna (1988).

^a The discharges given represent contaminated process waters, and do not include cooling waters.

^b The wastewater production figure corresponds to about $60 \text{ m}^3/\text{capita}/\text{yr}$ multiplied by the 2.4×10^9 residents in urban and rural areas of the world. The other discharge figures likewise have been derived from the reported water demand per unit tonne of metal smelted or goods manufactured.

^c We have assumed that 70% of each metal emitted to the atmosphere is deposited on land, and that the remaining 30% is deposited in the aquatic environments.

^d Worldwide sewage sludge production is estimated to be 30 million tonnes, assuming average sludge production rate of $30 \text{ g}/\text{capita}/\text{day}$ in urban and rural communities. It is believed that 20% of the municipal sludge is directly discharged or dumped into aquatic ecosystems, about 10% is incinerated, and the rest is deposited on land.

Long-Term Trends in Atmospheric Emissions

Total emissions of sulfur dioxide and nitrogen oxides generally increased in the United States during the period from 1900 to 1970 (Fig. 4.3). During this time, annual emissions of sulfur dioxide increased by a factor of 3 and nitrogen oxides increased almost tenfold.

Tall stacks constructed by electric utilities and smelters in the last 25 years have caused atmospheric pollutants to be transported long distances from their emission sources by emitting them into the upper air currents. Prior to 1970, there were only two stacks in existence taller than 150 meters (500 ft); in the United States today there are more than 175 such stacks. These stacks were built to reduce ambient air-pollutant concentrations in the vicinity of the stacks, but their net effect has been to spread air pollutants over long distances.

Emission-control plans adopted since passage of the U.S. Clean Air Act in 1970 have been effective in reducing the growth of national emissions of sulfur dioxide and nitrogen oxides (Fig. 4.3). Sulfur dioxide emissions decreased an estimated 30% between 1970 and 1988, while nitrogen oxide emissions were about 10% lower in 1988 than their peak in 1978.

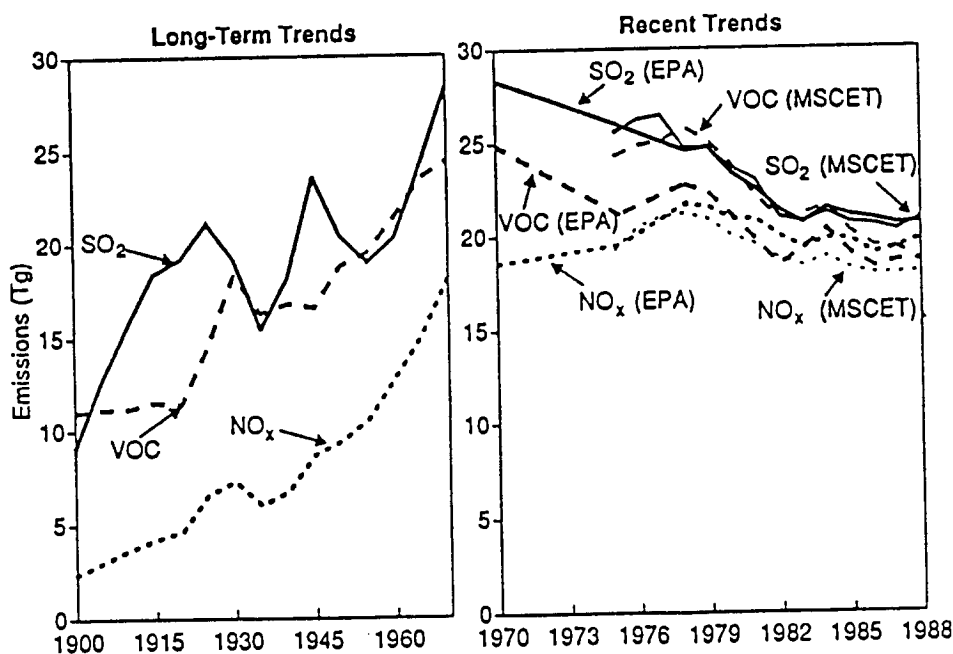


FIGURE 4.3. Historical trends in SO_2 , NO_x , and volatile organic carbon emissions. Sources for 1940–1970 trends: Gschwandtner, Wagner, and Husar 1988; for 1970–1987 trends: U.S. EPA, 1990; for 1975–1988 trends: Kohout et al., 1990. (From Irving, 1991.)

It is expected that the Clean Air Act Amendments passed by the U.S. Congress in 1990 will result in regulations that lower sulfur dioxide and nitrogen oxides emissions even further. Also, the act offers a comprehensive plan for significantly reducing emissions of hazardous air pollutants, also referred to as air toxics. The law lists 189 pollutants and requires major sources to reduce emissions according to a prescribed schedule. Unfortunately, we do not know as much about air toxics and their effects on bodies of water compared to our knowledge of acidification.

Global Transport of Pollutants

Measurements at the most remote points on the earth, such as Antarctica, indicate that many pollutants, such as organic chemicals (e.g., PCBs), enter the global cycle and are deposited in appreciable quantities anywhere on the earth. Most of the discussion on global transport is taken from Junge (1977).

The concentration of pollutants in the atmosphere is determined by the mass balance between global sources and sinks of the pollutant. Mathematically, one can write that

$$\frac{dM}{dt} = Q - S(M) \quad (4.1)$$

where

M = the global mass of the pollutant in the atmosphere

Q = the global source strength for the pollutant

$S(M)$ = the global sink of the pollutant

Under a steady-state assumption, which can be applied only to time intervals of more than one year, and fairly steady inputs, the left side of Equation (4.1) becomes zero and

$$Q_{ss} = S(M) \quad (4.2)$$

The sinks of atmospheric pollutants include:

1. Deposition (wet and dry) on land and sea surfaces.
2. Adsorption on land and sea surfaces.
3. Decomposition by atmospheric chemical and photochemical processes.
4. Emissions into the stratosphere.

The global removal (sink) rate is a function of the mass of pollutants present in the atmosphere (or its concentration), and if deposition prevails as in the cases of some relatively inert components (DDT, PCBs), it can be approximated by

$$S(M) = v_d \times C_M \quad (4.3)$$

where

v_d = depositional velocity (m/day)

C_M = average global (background) concentration of the pollutant, typically measured at some remote point unimpacted by cultural emissions

The average residence time of a pollutant in the atmosphere under steady-state conditions is given by

$$T = \frac{M}{S(M)} = \frac{C_M V_A}{v_d C_M A_G} = \frac{H}{v_d} \quad (4.4)$$

where

A_G = global surface area

V_A = volume of the atmosphere within the mixing layer

H = average depth of the surface air boundary layer (typically of the order of about 1000 m)

The most effective natural removal process is the attachment of pollutants to atmospheric aerosols and their subsequent removal by dry and wet fallout on land and sea surfaces. For many pollutants, the sea is the final sink, since pollutants deposited on land can be reentrained or can reenter the atmosphere by volatilization, wind erosion, and other processes described previously.

If the input of a pollutant into the global transport system is instantaneous, as occurs during an explosion, volcanic eruption, or one-time widespread pesticide application, and if the sink function is linearly proportional to the mass of the constituent in the atmosphere ($S(M) = S_0 \times M$), Equation (4.1) can be solved to yield

$$M(t) = \frac{Q_0}{V_A} e^{-S_0 t} = \frac{Q_0}{V_A} e^{-t/T} \quad (4.5)$$

where Q_0 = the mass of the instantaneous input.

Example 4.1: Global Pollution Transport

Background (steady-state) concentrations of an “inert” pesticide in the atmosphere are measured at about $C_M = 0.1 \text{ ng/m}^3 = 10^{-10} \text{ g/m}^3$. The worldwide production of the pesticide is about 1000 tonnes/yr = 10^9 g/yr from which about 40% is lost to the atmosphere during and after application. Estimate the average residence time and deposition velocity of the pesticide. Assume an average depth of the mixed air boundary layer of $H = 3000 \text{ m}$.

Solution Atmospheric input of the pesticide:

$$Q = 0.4 \times 10^9 \text{ (g/yr)} = 4 \times 10^8 \text{ g/yr}$$

To determine the amount of the pesticide in the atmosphere it is necessary to know:

Volume of the atmospheric mixing layer (earth radius $r = 6.3 \times 10^6 \text{ m}$):

$$V_A = 4\pi r^2 H = 4\pi(6.3 \times 10^6)^2 \times 3000 = 1.5 \times 10^{18} \text{ m}^3$$

Mass of the pesticide in the atmosphere:

$$M = C_M V_A = 10^{-10} \text{ (g/m}^3) \times 1.5 \times 10^{18} \text{ (m}^3) = 1.5 \times 10^8 \text{ g}$$

Average residence time:

$$T = \frac{M}{Q} = \frac{1.5 \times 10^8}{4 \times 10^8} = 0.375 \text{ yr} = 137 \text{ days}$$

Deposition velocity:

$$v_d = \frac{H}{T} = \frac{3000}{0.375} = 8000 \text{ m/yr}$$

Entry of Atmospheric Pollutants into Surface Waters

Atmospheric pollution consists of gases and aerosols or atmospheric particulates. The particulate matter ranges in size from 6×10^{-4} to $10^3 \mu\text{m}$. The term aerosol should be differentiated from dust. Dust contains particles that are mostly insoluble, while aerosols contain also water-soluble materials (about 50 according to Paterson and Junge

(1971)). Besides direct emissions from terrestrial sources, aerosols can be formed in the atmosphere by precipitation, absorption, and chemical reactions.

Removal of particles (aerosols and dust) from the atmosphere is due to:

1. Dry deposition by sedimentation.
2. Removal by rainfall and snowfall.
3. Dry deposition by impact on vegetation and rough surfaces.

Removal of gases occurs primarily by:

1. Removal during periods of precipitation.
2. Absorption at the earth's surface.
3. Adsorption of aerosolic particles and subsequent deposition.

Dry Deposition

Dry deposition is an important process for the removal of gases and airborne particles. While data on wet deposition are relatively abundant, dry deposition data are sparse. Sisterson et al. (1990) evaluated a number of data sources on the dry deposition of sulfur and nitrogen species. Wet deposition was found to account for most of the total deposition of sulfur and nitrogen species for regionally representative sites. Individual sites that were more heavily impacted by sulfur emissions tended to have a relatively larger dry deposition in relation to the total contribution than sites not impacted by local emissions. This means that dry deposition is probably much greater than wet deposition in and near urban areas. When urban areas are excluded, dry deposition of sulfur species is estimated to be 30% to 60% of the total (wet plus dry) deposition. Under the same conditions, the percentage for nitrogen species ranges from 30% to 70%.

The rate of dry fallout from the atmosphere is primarily determined by the force of gravity, but other effects, such as surface impaction, electrostatic attraction, adsorption, and chemical interactions, may explain why the deposition rate of small particles (order of magnitude of 1 μm or less) onto the ground is often greater than can be expected from the pull of gravity.

The rate of deposition of aerosol particles can be related to their average above-ground concentration:

$$D_d = v_d C(x, y, z) \quad (4.6)$$

where

D_d = amount of aerosols removed per unit area per unit time
(e.g., g/m²-day or tonnes/km²-month)

$C(x, y, z)$ = average concentration of aerosols at x , y , and z locations
from the source or coordinate origin (g/m³)

v_d = deposition velocity of particles (m/day or m/month)

Depositional velocity differs from the physical settling velocity expressed by Stokes' law. Figure 4.4 shows the depositional velocity of particles as estimated by measurements. MacMahon, Dension, and Fleming (1976) reported that the depositional velocities of gases range from 0.5 to 2.5 cm/sec, depending on the ground surface.

Wet Deposition and Composition of Precipitation

Due to the fact that precipitation scavenging is one of the most effective processes for cleansing the atmosphere, rain and snow contain many pollutants in quantities that may be harmful to terrestrial and aquatic ecosystems. Pollutants included in wet precipitation are acidity, toxic metals, organic chemicals, phosphates, and nitrogen compounds. In some areas pollution from rain has been devastating to surface water biota and often leads to acidification of lakes, fish kills, and severe reduction in the productivity of lakes. Acid rainfall also leaches cations from soils (such as aluminum) and from urban infrastructures (damage to concrete and elutriation-corrosion of metals).

Contamination can be incorporated into precipitation within or below clouds. In-cloud scavenging is called *rainout* or *snowout*, the below-cloud process of enrichment is called *washout*.

Washout Function. In many cases the amount of pollutants deposited by wet fallout can only be estimated from known atmospheric concentrations, because the rainwater concentrations are either not known or are unreliable. The process of scavenging of pollutants by raindrops during washout or rainout (snowout) is basically an exponential function (Slade, 1968):

$$C_w = C_{w,0} \exp(-\lambda t) \quad (4.7)$$

where

C_w = the atmospheric concentration of the contaminant after the rain

$C_{w,0}$ = the atmospheric concentration of the contaminant before the rain

t = duration of the rain

λ = the washout coefficient

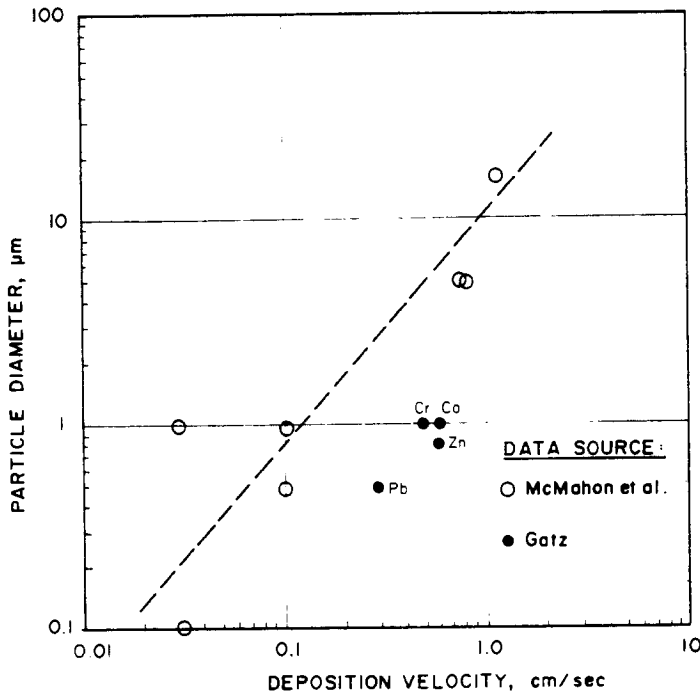


FIGURE 4.4. Deposition velocity for particles $<10 \mu\text{m}$. Sources: \circ McMahon, Dension, and Fleming, 1976; \bullet Gatz, 1975.

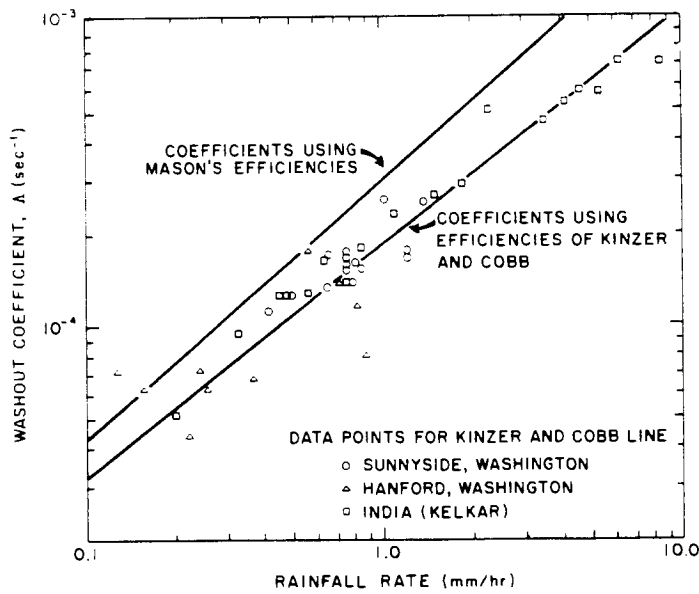


FIGURE 4.5. Rain washout coefficient λ related to rainfall intensity. (From Slade, 1968.)

The wet fallout per unit area, D_w , is then

$$D_w = (C_{w,0} - C_w)H = C_{w,0}(1 - \exp(-\lambda t))H \quad (4.8)$$

where H = the depth of atmosphere through which the pollutant plume is mixed. The magnitude of the washout coefficient, λ , is of the order 10^{-4} sec^{-1} and is a function of rain intensity (Fig. 4.5). The magnitudes of λ are similar both for gases and particulates.

Example 4.2: Concentration of Pollutants in Precipitation

The atmospheric concentration of phosphate before rain was estimated or measured as $C_{w,0} = 10 \mu\text{g}/\text{m}^3$. The depth of the mixed atmospheric layer extended about 1000 meters above the ground surface. Estimate the amount of wet fallout during a storm with a volume of 20 mm lasting 2 hours.

Solution From Figure 4.5 the washout coefficient for the storm intensity $i = 10 \text{ mm}/2 \text{ hr} = 5 \text{ mm}/\text{hr}$ is $\lambda = 7 \times 10 \text{ sec}^{-1} = 2.52 \text{ hr}^{-1}$. Then the mass of deposited phosphate becomes (Eq. (4.3)):

$$\begin{aligned} D_w &= C_{w,0}(1 - e^{-\lambda t})H = 10(1 - e^{-2.52 \times 2})1000 \\ &= 9935 \mu\text{g}/\text{m}^2 = 9.935 \text{ mg}/\text{m}^2 \end{aligned}$$

The phosphate concentration in rainwater (rain volume $V_r = 10 \text{ mm} = 0.01 \text{ m}^3/\text{m}^2$) is

$$C_r = \frac{D_w}{V_r} = \frac{9.93 \text{ mg}/\text{m}^2}{0.01 \text{ m}^3/\text{m}^2 \times 1000 \text{ l}/\text{m}^3} = 0.99 \text{ mg}/\text{l}$$

Acidity

All rainfall is by nature somewhat acidic. The principal factor in rainfall's acidity is carbon dioxide. Pure water in equilibrium with the atmosphere would have a pH of 5.6. The accumulation of acidic chemicals in the atmosphere is also contributed to by decomposing organic matter, volcanic eruptions, and movements of the sea.

In some parts of the world the acidity of rainfall is considerably lower than pH 5.6, primarily because of the emission of sulfur dioxide (SO_2) and nitrogen oxides (NO_x). The areas of maximum deposition of chemical species related to acidity (H^+ , SO_4^{2-} , NO_3^-) in North America are located in the northeastern United States and southeastern Canada (Figs.

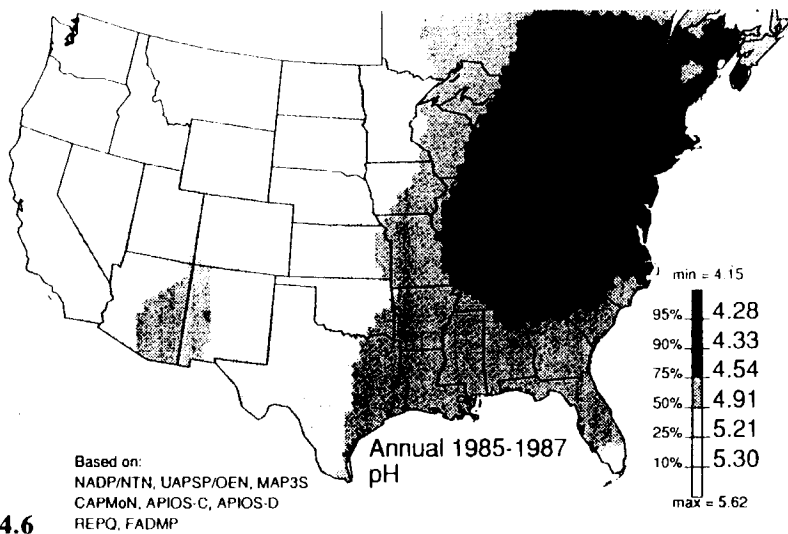


FIGURE 4.6

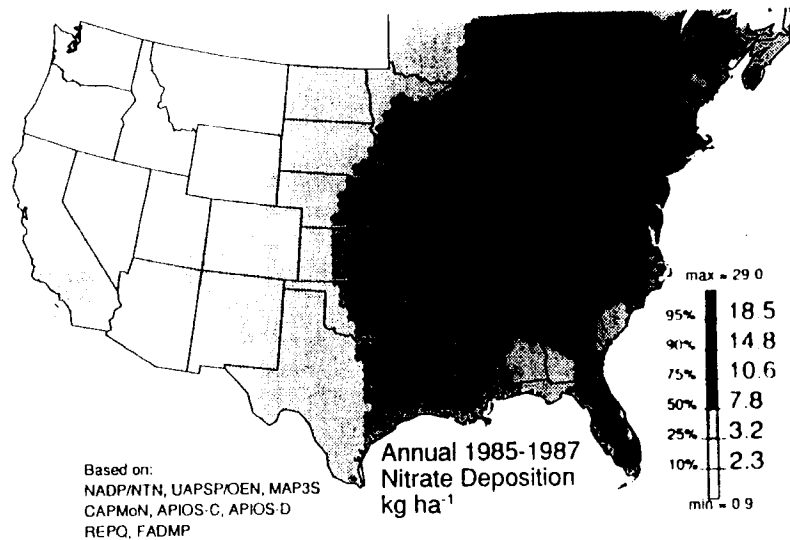


FIGURE 4.8

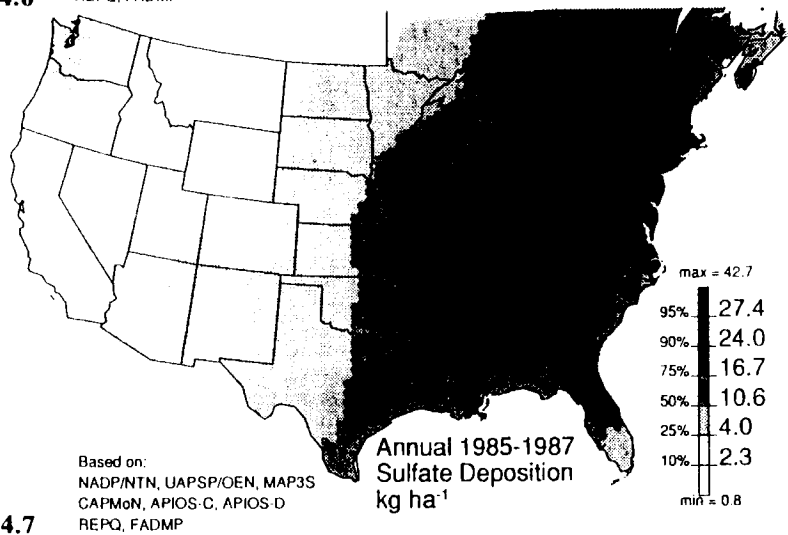


FIGURE 4.7

FIGURE 4.6. Gray scale composite of volume-weighted pH for North America for 1985–1987 for fixed percentile ranking. (From Irving, 1991.)

FIGURE 4.7. Gray scale composite of sulfate ion wet deposition for North America for 1985–1987 for fixed percentile ranking. (From Irving, 1991.)

FIGURE 4.8. Gray scale composite of nitrate ion wet deposition for North America for 1985–1987 for fixed percentile ranking. (From Irving, 1991.)

4.6–4.8). The pattern for annual average pH of wet deposition for the United States shows that large areas in and downwind of emission zones have depositions in the pH range of 4.1 to 4.5. Individual precipitation events can produce rainfall as acidic as pH 3 or below. These areas have considerably higher levels of atmospheric deposition (as measured by acidifying species) than remote parts of the world. Table 4.5 summarizes wet deposition pH values from remote regions of the world. When areas with possible anthropogenic influence are taken into account, it is apparent that the average pH of precipitation in remote regions are closer to 5.0 than 5.6, the pH initially thought to represent “clean” rain.

In areas where precipitation pH is below about 4.6, we also find surface waters that are acidic (considered here to be those with pH values below about 5). Some surface waters are more sensitive to precipitation acidity than others because of differences in the neutralizing capacity of watersheds. The National Surface Water Survey (NSWS) was initiated in 1986 to establish the current chemical status of surface waters in the United States in relation to known deposition levels and the biogeochemical environment (Linthurst, Landers, and Eilers, 1986). The key elements of the survey were three major synoptic surveys of lakes and streams in eight geographic regions of the United States believed to contain the majority of surface waters susceptible to acidification (Fig. 4.9).

The survey found acidic lakes and streams in some of those regions, although some of them were naturally acidic or acidic for reasons other than deposition (Baker et al., 1990). The majority of the surface waters were circumneutral but weakly buffered, with about half the lakes and streams having acid-neutralizing capacities (ANC) $\leq 200 \mu\text{eq/l}$ (Tables 4.6 and 4.7). A relatively small percentage (4.2%) of the 1181 lakes investigated were acidic. Most acidic lakes were found in the Northeast, Florida, and the upper Midwest. Of the total stream length in the survey, 2.7% (5506 km) was acidic. Most of the acidic stream length was in the mid-Appalachian and mid-Atlantic coastal plain regions. The researchers concluded that atmospheric deposition was the dominant source of SO_4 in most surface waters sampled.

Baker et al. (1990) conducted a literature review of acidic lakes in the high deposition areas of Canada, Finland, Norway, Sweden, and the Kanai Peninsula of Alaska, plus several acidic lakes in low deposition areas. As in the National Survey, there was a positive relationship between SO_4^{2-} deposition and lake water SO_4^{2-} concentrations in Canada and Scandinavia. Nitrate in deposition was found to play a more important role in lakes in southern Norway and parts of Europe than was observed in lakes in the United States and Canada.

TABLE 4.5 Summary of Wet Deposition pH Values from Remote Regions of the World

Location	N ^a	Period	pH			Other Chemical Measurements	Sampling Protocol ^a	Possible Anthropogenic Influence
			Min	Max	Avg			
<i>American Tropics</i>								
Rain Forest								
Manaus, Brazil	53	1966–1968	3.6	5.4	4.6 ^b	None reported	UNK	Urban (definite)
Adolfo Ducke Forest, Brazil	2	UNK	—	—	4.38 ^c	None reported	UNK	Urban (possible)
Amazon River from coast of Brazil to Colombia, Peru, Bolivia	31	1976–1977	4.71	5.67	5.03 ^c	Full inorganic	F/B-wet	Unlikely
La Selva, Costa Rica	UNK	1973	4.40	4.90	4.66 ^c	None reported	UNK	Unlikely (volcano influenced)
San Carlos, de Rio Negro, Venezuela	70	1979–1980	4.0	6.7	4.69 ^d	None reported	Bottles	Unlikely
Cloud Forest								
Alto de Pipe, Venezuela	19	UNK	4.21	5.90	5.03 ^c	None reported	UNK	Urban (definite)
San Eusebio, Venezuela	UNK	UNK	3.82	6.21	4.55 ^c	None reported	UNK	Unlikely
Savannah								
Calabozo, Venezuela	151	1981–1983	4.8	6.9	5.8 ^d	NH ₄ ⁺	F/B-bulk	Unlikely (wildfires)
Camburito, Venezuela	18	1983–1984	4.0	5.2	4.4 ^d	None reported	F/B-wet	Unlikely (wildfires)
Joaquin del Tigre, Venezuela	17	1984–1985	4.2	5.8	5.1 ^d	None reported	F/B-wet	Urban (possible)
La Paragua, Venezuela	14	1985	4.0	5.6	4.8 ^d	None reported	F/B-wet	Unlikely (wildfires)
Lake Valencia, Venezuela	92	1977–1978	3.2	7.7	5.9 ^e	Full inorganic	F/B-bulk	Light industry (possible)

R0023182

TABLE 4.5 (Continued)

Location	N ^a	Period	pH			Other Chemical Measurements	Sampling Protocol ^a	Possible Anthropogenic Influence
			Min	Max	Avg			
<i>Greenland</i> East coast	10	1981–1983	4.40	6.00	5.13 ^c	Full inorganic	Bulk (long exposure)	Local source (possible)
<i>Israel</i> Negev Desert	30	1978–1983	—	—	7.9 ^f	All major ions but NO ₃ ⁻ , NH ₄ ⁺ , pH	F/B-wet	Urban (possible)
<i>Portugal</i> Coimbra	195	1978–1980	3.5	7.7	4.75 ^g	Conductance	F/B-bulk	Urban (definite)
<i>Australia</i> Hunter Region, New South Wales	UNK	1984–1986	4.0	7.0	5.0 ^d	Conductance	Bulk/wet only	Unlikely
<i>New Zealand</i> Maimai, South Island	10	1985	6.2	5.2	5.6 ^e	Full inorganic	F/B-bulk	Unlikely

Source: From Sisterson et al. (1990).

Note: Probable local anthropogenic influence category is deduced from information provided in the original studies.

^aN is the number of samples, and UNK is unknown; F/B is funnel/bottle sampling.

^bMedian of simple monthly average pH values.

^cSimple average pH value.

^dVolume-weighted pH value.

^eMedian value.

^fpH calculated from average bicarbonate concentration.

^gpH calculated from average H⁺ concentration.

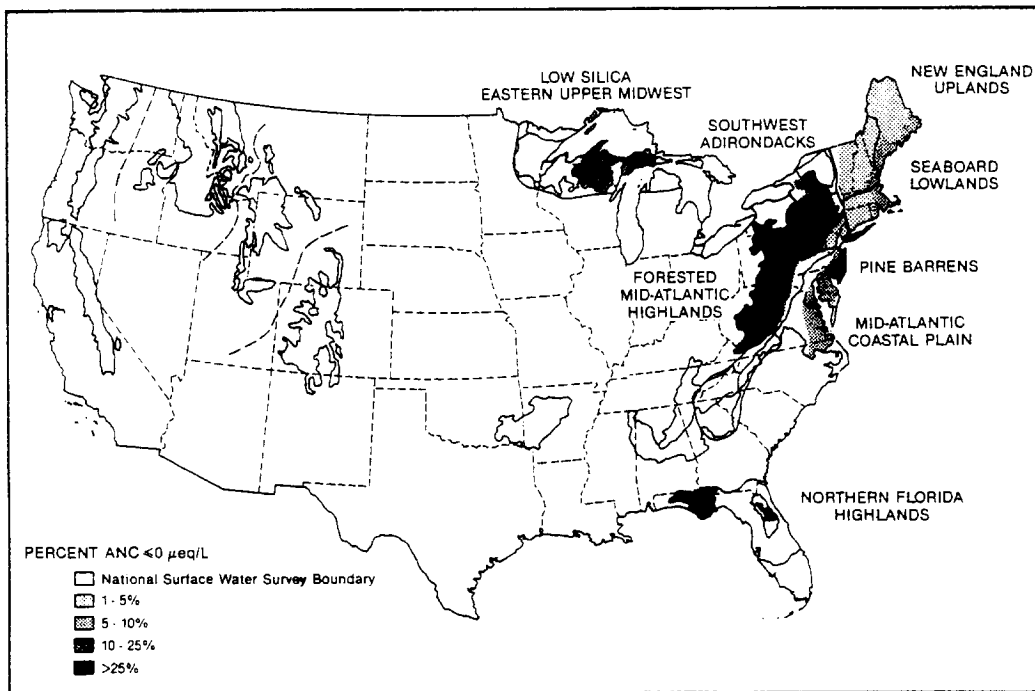
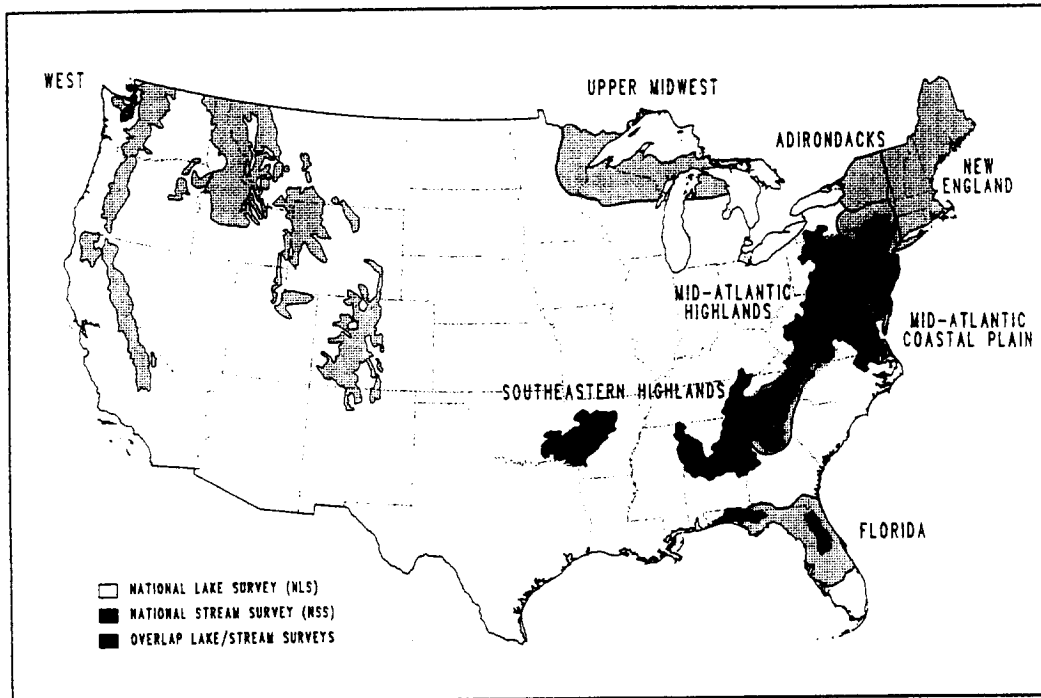


FIGURE 4.9. National Acidic Precipitation Assessment Program—Survey of surface bodies of water; (a) Areas included in the survey; (b) surface waters identified as affected. (From Baker et al., 1990.)

TABLE 4.6 Population Estimates of the Percentage of the Total Number of NSWS Lakes and Streams with ANC, pH, and Al in Reference Ranges^a

Region	Lake or Stream ^b	Total Number	ANC ($\mu\text{eq/l}$)			pH		Al_{MIBK} ($\mu\text{g/l}$) ^c
			≤ 0	≤ 50	≤ 200 %	≤ 5.5	≤ 6.0	> 50
Northeast	L	7,096	6	22	6	19	13	6
	S-u	3,235	6	23	48	7	13	5
	S-d	3,235	1	5	30	<1	3	<1
Mid-Appalachians	S-u	21,527	5	20	49	9	15	6
	S-d	21,527	2	9	43	2	6	2
Mid-Atlantic coastal plain	S-u	11,284	12	30	56	24	49	37
	S-d	11,284	7	20	41	13	22	20
Interior Southeast	L	258	1	1	34	1	<1	<1
	S-u	18,598	1	7	52	2	9	<1
	S-d	18,598	1	6	47	<1	3	<1
Florida	L	2,098	23	40	55	21	33	6
	S-u	1,274	25	66	77	39	69	3
	S-d	1,274	11	59	82	23	59	3
Upper Midwest	L	8,501	3	16	41	4	10	3
West	L	10,393	<1	16	66	<1	1	<1
All NSWS	L	28,346	4	19	56	5	9	3
	S-u	55,917	2	11	44	4	9	5

Source: From Baker et al. (1990).

^aBased on fall index chemistry (lakes) and spring baseflow (streams).

^bL = lakes; S-u = streams—upstream ends of reaches; S-d = streams—downstream ends of reaches.

^cMIBK: methyl-isobutyl-ketone (method that measures total monomeric Al).

Acidic lakes and streams are not common in areas where acidic deposition is low. Where they are found, they are typically associated with areas where there are large amounts of dissolved organic carbon (DOC). Other reasons for highly acidic surface waters in areas of low deposition include acid mine drainage, geothermal springs, high chloride clearwater coastal lakes, and lakes with extremely low conductivity.

As has been pointed out the acidity of precipitation is attributed to the presence of sulfates and sulfides (SO_4^{2-} and SO_3^{2-}) and nitrates (NO_3^-) in the atmosphere. Sulfur is one of the elements that is always found in the atmosphere, and it occurs as SO_4^{2-} and SO_3^{2-} in aerosols and SO_2 and H_2S gases. Hydrogen sulfide in air is normally oxidized to SO_2 , which is then oxidized to SO_3 . The oxidation reaction proceeds quickly if such metallic catalysts as iron and manganese oxides are present (Stern, 1976).

TABLE 4.7 Population Estimates of the Percentage of NWS Lake Surface Area (km²) and Stream Length (km) with ANC, pH, and Al in Reference Ranges

Region	Lake or Stream	Total Resource ^b	ANC (μeq/l)			pH		Al _{MIBK} (μg/l) ^c
			≤0	≤50	≤200 %	≤5.5	≤6.0	>50
Northeast	L	4,279	2	16	68	4	6	3
	S	15,144	4	11	36	6	9	6
Mid-Appalachians	S	54,425	3	11	47	7	13	7
Mid-Atlantic coastal plain	S	40,296	6	24	52	24	47	48
Interior Southeast	L	243	<1	<1	55	<1	<1	<1
	S	86,938	<1	5	48	<1	7	1
Florida	L	662	20	49	64	24	33	9
	S	3,848	12	61	76	44	74	40
Upper Midwest	L	5,015	<1	4	15	<1	2	<1
West	L	1,819	<1	8	35	<1	<1	<1
All NWS	L	12,016	2	11	40	3	5	2
	S	200,652	3	12	48	8	18	13

Source: From Baker et al. (1990).

^a Based on fall index chemistry (lakes) and spring baseflow (streams).

^b Total resource for lakes is expressed as surface area (km²); total resource for streams is expressed as length (km).

^c Total monomeric Al (as measured by the MIBK (methyl-isobutyl-ketone) method; see Section 2.5.3.4 of the full Report).

These metallic compounds are commonly emitted by the burning processes in fly ash. Formation of sulfuric acid is greatly enhanced by the moisture emitted from cooling towers (Fig. 4.10).

The SO₄⁻ and NO₃⁻ anions in the air are balanced by cations, primarily NH₄⁺, Ca²⁺, Mg²⁺, and N⁺. The major sources of these compounds are sea spray, soil dust, and ammonia volatilization from soils. Since Na⁺ from sea spray is already balanced by Cl⁻ in the absence of other buffering agents in the air, there may not be other cations available to balance additional SO₄⁻ and NO₃⁻ ions, so they can only react with H⁺ to produce acid rain (Fig. 4.11).

Example 4.3: Acidity of Precipitation

The ambient concentration of sulfur trioxide and sulfates (SO₃ and SO₄⁻) is about 30% of the ambient SO₂ concentration. Estimate the approxi-

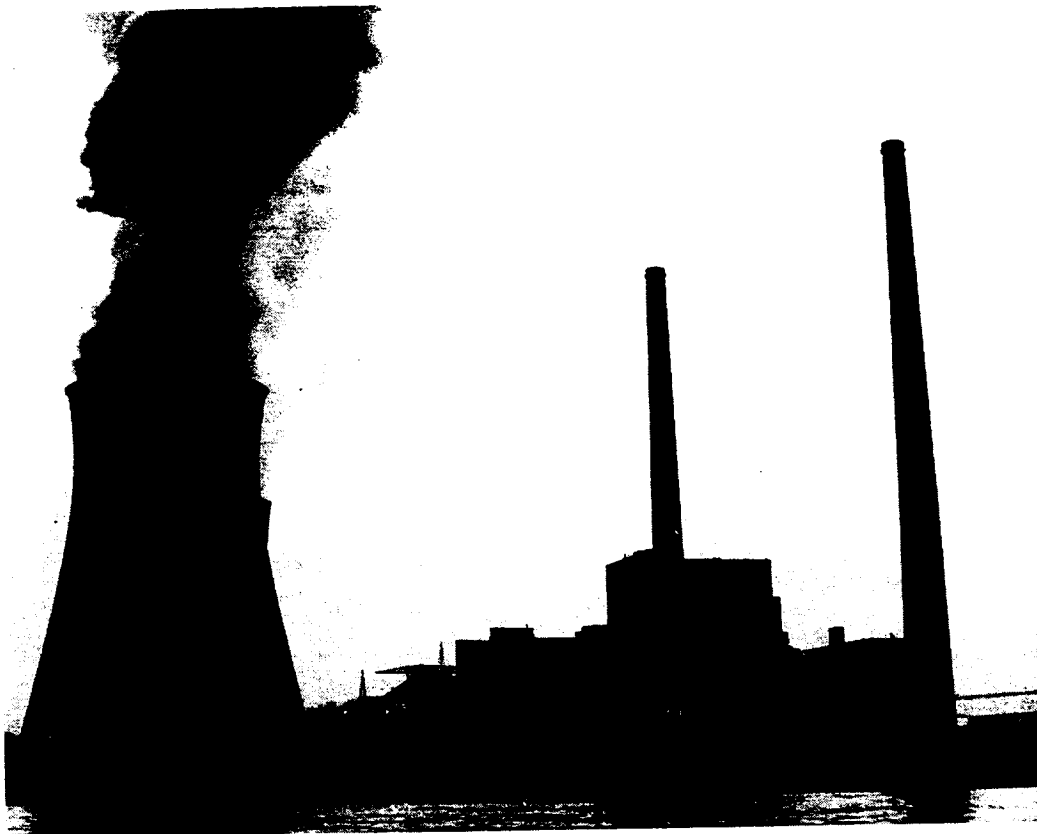


FIGURE 4.10. Formation of sulfuric acid from the SO_2 emissions is greatly enhanced when stack effluent combines with the vapor drift from cooling towers. (Photo: V. Novotny.)

mate pH of rainwater resulting from 5 mm of rain lasting 5 hours if the ambient SO_2 concentrations is $20 \mu\text{g}/\text{m}^3$. The mixed atmosphere depth is $H = 1000$ meters.

Solution From Figure 4.5 the washout coefficient $\lambda = 2 \times 10^{-4} \text{sec}^{-1} = 0.72 \text{hr}^{-1}$. Rainfall volume $V_r = 5 \text{mm} = 51/\text{m}^2$. Mass of sulfates washed out:

$$D_w = (30/100)C_{w,0}(1 - e^{-\lambda t})H = 0.3 \times 20(1 - e^{-0.72 \times 5}) = 5836 \mu\text{g}/\text{m}^2$$

Sulfate concentration in rainwater

$$C_r = \frac{D_w}{V_r} = \frac{5.836 \text{mg}/\text{m}^2}{51/\text{m}^2} = 1.17 \text{mg}/\text{l}$$

Cl^- , SO_4^{2-} NO_3^- , HCO_3^-	H^+	Acid forming
	NH_4^+ , Ca^{++} , Mg^{++} , K^+	
SO_4^{2-}	Ca^{++} , Mg^{++} , K^+	Sea salt
Cl^-	Na^+	

FIGURE 4.11. Atmospheric cation–anion balance resulting in an increase of H^+ ions in rainwater.

Change C_r to equivalent SO_4^{2-} (equivalent weight of $\text{SO}_3 = [32 + 3 \times 16]/2 = 40 \text{ g/eq} = 40,000 \text{ mg/eq}$).

$$\{\text{SO}_4^{2-}\} = \frac{C_r}{EW(\text{SO}_3)} = \frac{1.17 \text{ mg/l}}{40,000 \text{ mg/eq}} = 2.92 \times 10^{-5} \text{ eq/l}$$

Each equivalent of SO_4^{2-} must be balanced by one equivalent of $[\text{H}^+]$ or any other available atmospheric cation (such as NH_4^+). The $[\text{H}^+]$ concentration of rainwater not contaminated by SO_x ($\text{pH} = 5.6$) is $[\text{H}^+] = 10^{-\text{pH}} = 10^{-5.6} = 2.5 \times 10^{-6} \text{ mole/l}$, which is more than one order of magnitude less than the equivalent weight of $[\text{H}^+]$ required to balance SO_4^{2-} ion. Since almost all of the sulfuric acid in water is dissociated, the resulting pH is roughly

$$\text{pH} = \log \frac{1}{[\text{H}^+]} = \log \frac{1}{\{\text{SO}_4^{2-}\}} = \log \frac{1}{2.92 \times 10^{-5}} = 4.5$$

Effects of Precipitation Acidity on Drinking Water

Concerns have been raised about the possibility that acidic deposition might contribute to levels of chemical contaminants in untreated or partially treated drinking water in excess of established or contemplated drinking water standards. Background information is presented on con-

cepts important to the understanding of chemical contaminants in drinking water that may be related to acidic deposition.

Instances of elevated concentrations of chemical contaminants exceeding drinking water standards have been reported for precipitation collected in cisterns, shallow wells and springs, and surface waters. The elevated concentrations are usually the result of pipe corrosion and other materials in the distribution system that come in contact with the water.

In the context of this chapter corrosion is defined as the deterioration of a pipe or fixture by electrochemical reaction with its environment (Patterson and O'Brien, 1979). Water's corrosive tendency depends on its physical, chemical, and biological characteristics and the nature of the material with which it comes in contact. An example of physical action is the erosion or wearing away of a pipe elbow because of excess flow velocity in the pipe. A simple example of chemical action is the oxidation or rusting of an iron pipe. An example of biological action is the release of corrosive by-products by iron-oxidizing and sulfate-reducing bacteria (Singley, Beaudet, and Markey, 1984).

There are several piping materials used in drinking water distribution and home plumbing systems. Common piping materials are composed of copper with lead-based solder, galvanized iron, plastic, and asbestos-cement. Besides the material itself, contaminants in it, such as cadmium and lead, can leach into the water. Even certain plastic piping may contain contaminants that can leach into the water due to corrosion. For example, some plastic pipes have been reported to contain lead stearate as a stabilizing agent in its manufacture (Jacks, 1984). Little is known, however, about the degree of leaching of lead from plastic piping. Further, the brands of plastic that contain lead stearates are trade secrets, thus, the leaching potential of a particular brand cannot be ascertained without experimentation.

The contaminants of interest that may leach from piping due to corrosion are arsenic, asbestos, cadmium, copper, lead, selenium, and zinc. Aluminum, mercury, and nitrate are not significantly leached from piping due to the corrosivity or aggressive properties of water. The results of three U.S. surveys and one in Canada are summarized below for lead, the contaminant of most concern in corrosive drinking water supplies. The impact of rainfall acidity on elutriation of metallic ions (zinc) from tin roofs and downspouts is discussed in Chapter 8.

New England Water Works Association Survey. Taylor et al. (1984) sampled raw and treated surface water and ground water for lead. One of the 328 samples (0.30%) of untreated source waters had lead levels above the current U.S. MCL of 50 µg/l. This was a ground-water

sample. Treated waters were sampled at the treatment facility before distribution through piping and plumbing systems. Of the 134 treated surface water samples two (1.5%) contained lead levels above the current MCL, while one of the 22 treated ground-water samples (4.5%) contained lead levels above the MCL.

National Inorganics and Radionuclides Survey. The EPA collected tap water samples from nearly 1000 ground-water drinking water supplies in the United States on which they conducted analyses for selected inorganics and radionuclides (Logtin, 1988). Of the 983 sites sampled for lead, the concentrations in tap water ranged from <5 mg/l to 240 μ g/l. A total of 55 samples (5.6%) were above the minimum reported detection limit of 5 μ g/l, which is also the proposed MCL. Only four of the 983 samples (0.41%) were above the current MCL of 50 μ g/l. All of the systems exceeding the current MCL were those serving less than 2500 households.

Rural Water Survey. The rural water survey described previously sampled tap water at 2654 households for some constituents, including lead concentrations (Francis et al., 1984). A total of 16.6% of the supplies contained lead levels above the MCL. The median concentration was 8 μ g/l. The highest recorded lead concentration was 970 μ g/l, nearly twenty times higher than the current MCL. The authors attributed the high lead levels to leaching of lead from distribution and plumbing systems, but also noted potential lead contamination from sample bottles.

Canadian Survey. Meranger, Subramanian, and Chalifoux (1979) reported that mean lead levels in treated water samples in 70 Canadian cities were equal to or less than 1 μ g/l, and none of the samples in nine out of ten provinces exceeded 3 μ g/l. The maximum lead concentration found in these source waters was 7 μ g/l, one-seventh the current U.S. MCL.

Based on U.S. surveys of lead in ground-water supplies, the EPA (1988) estimated that approximately 900 ground-water suppliers (about 1%) may have water leaving the treatment plant at levels greater than 5 μ g/l. Based on this information and other surveys, the EPA (1988) estimated that about 99% of the 219 million people in the United States using both surface and underground public water supplies are exposed to distributed water levels between 0 and 5 μ g/l and about 1% (2 million people) are served by water with lead levels greater than 5 μ g/l.

The occurrence of lead as a by-product of corrosion can be an important source of lead in drinking water at the user's tap. Three overall factors are particularly important in determining the degree of corrosion

in the water distribution and home plumbing systems. First, the type and age of piping are important factors in the levels of lead leached from the plumbing system. For instance, the use of copper pipes joined with solder containing lead can result in elevated lead levels due to galvanic corrosion. This corrosion is especially reactive in the case of newly installed solder. Second, the corrosivity of the water toward lead is a major factor influencing the levels of lead at the consumer's tap. The corrosivity of water, in turn, is dependent upon many other factors. Third, regardless of the age of the piping, water that has been in contact with the pipe for a period of time will contain higher lead levels than flushed water from the same pipe.

For this reason, recent surveys of drinking water at the consumer's tap have focused on samples collected in the morning prior to any use of tap water in the house that day and/or samples collected after flushing, usually for 30 to 60 seconds. The first-draw samples provide a worst case estimate of contaminant concentrations, while the flushed samples may be more representative of average concentrations of water consumed during the day. The following three studies reported results of standing water analyses.

Patterson (1981) collected 782 random daytime grab samples flushed for 30 seconds from drinking water taps in 58 cities in 47 states. The percentage of samples collected from each state generally reflected the state population. It was found that 60% of all samples contained lead levels less than or equal to $10\ \mu\text{g/l}$, 84% less than $20\ \mu\text{g/l}$, and 97% less than $50\ \mu\text{g/l}$. Unfortunately, information on lead levels below $10\ \mu\text{g/l}$ were not available from this study, since the reported analytical detection limit was $10\ \mu\text{g/l}$.

Taylor and Symons (1984) sampled 36 treated surface waters and ground waters in seven northeastern states for lead at various points in the system. Samples were collected after standing overnight in plumbing, from the service line, and from the water main. None of the 36 samples from the water main had lead levels above the current MCL. One of the 36 samples from the service line (3%) had lead levels above the current MCL. Treated waters sampled at the tap first thing in the morning exceeded the lead MCL in 6 of the 36 samples (17%). The higher frequency of exceedences for the samples that involved additional contact with plumbing systems indicated corrosion of piping materials.

The American Water Works Service Company (1988) conducted a study of lead levels in 1484 drinking water supplies throughout the American water system. Subsequent evaluation indicated a general association between higher lead levels and several factors, such as pH and

plumbing age. It was also observed that the lead levels at specific sites or within specific systems could not be predicted based on these factors.

Assessment

Although there is some likelihood that metals such as lead can be mobilized by acidity in drinking water systems, Grant et al. (1990) concluded that measurable increases in exposure will occur only under conditions related to specific, isolated exposure scenarios, such as individual drinking water systems that use surface ponds, shallow wells, or cisterns. Even assuming extreme-case conditions, such as acidic depositions resulting in a drinking water pH of 4.5, these researchers concluded that adverse health effects are unlikely except possibly for lead.

In the case of lead, there is a special reason for concern, which is that many individuals are already exposed to lead from multiple sources at levels that cause their blood lead to approach or exceed the lowest observed adverse effect level. As a result, any small increment attributed to acidic deposition might be sufficient to place some people in the range of exposure associated with increased risk of adverse health effects. On the other hand, increments in lead exposure projected to cause health effects are predicated on the assumption that acidic deposition can shift pH levels to the extreme levels necessary for lead mobilization. These extreme pH shifts are not confirmed by current research.

Nitrogen in Atmospheric Deposition

Nitrogen in deposition not only contributes to acidity, but it can also adversely affect the productivity of surface waters, particularly in the coastal zone. This section describes the current understanding of the atmospheric sources of nitrogen and its effects on near-shore waters. There remains considerable uncertainty about the importance of this source of nitrogen.

For many years, scientists believed that the portion of nitrogen entering near-shore waters was trivial compared to nitrogen from other sources, particularly land runoff and sewage treatment plant effluents. In recent years, however, numerous reports have suggested that atmospheric nitrogen contributes significantly to surface water quality.

Fisher et al. (1988) used an empirical approach to describe a cause-effect association between the amount of atmospheric nitrogen and its effects on the Chesapeake Bay. It was estimated that atmospheric nitrate and ammonia contribute 35 and 19 million kilograms, respectively, of biologically available nitrogen to the bay annually. Atmospheric nitrate

and ammonia were estimated to account for 25% and 14%, respectively, of the total nitrogen loading to the bay.

Many scientists initially discounted the validity of the findings because the calculations necessarily required considerable conjecture and numerous simplifying assumptions. However, an objective analysis supported through the National Acid Precipitation Assessment Program reinforced the hypothesis that atmospheric nitrogen is likely a major contributor to surface water quality (Turner et al., 1990). Also, a report by the EPA suggested that at least 13% of the total nitrogen loading to the bay is from atmospheric deposition.

The results from the studies of the Chesapeake Bay may not be indicative of other near-shore waters. The fate of the ecosystem and the effects of nitrogen inputs vary substantially across ecosystems and sites. Information also suggests that a watershed has a high capacity to retain or denitrify the nitrogen entering it, and that this capacity is highly variable among watersheds.

Turner et al. (1990) assessed the effects of atmospheric nitrogen on surface waters according to ecosystem components. A summary is presented below:

1. Forest ecosystems
 - Biological fixation and nitrogen deposition are the two major processes providing nitrogen inputs to forests.
 - Little nitrogen is normally exported from forests because they tend to be nitrogen poor. The majority of atmospherically deposited nitrogen is assimilated by forested watersheds.
2. Agricultural ecosystems
 - Insufficient information is available to determine the effects of a change in agricultural management practices on the export of nitrogen.
3. Wetlands
 - Removal and/or storage processes by wetlands in the Chesapeake Bay could prevent the transport of atmospherically deposited nitrogen from reaching the bay.
 - Approximately 20% of the Chesapeake Bay drainage area passes through wetlands.
4. Freshwater ecosystems
 - A significant amount of surface-water nitrogen is removed within the stream–river ecosystem, mostly through denitrification.

5. Estuaries and near-coastal waters

- The rates of nitrogen processes in these waters increase as nitrogen deposition loadings increase.
- Nitrogen deposition onto watersheds is an important contributor to the total nitrogen loading in estuaries; however, the assumption that nitrogen export from the watershed is proportionate to nitrogen inputs is not necessarily valid.
- There is some uncertainty whether nitrogen is largely retained or largely lost by the mechanisms of denitrification and advective transport.

Although most of the information presented in this section is based on calculations in the Chesapeake Bay watershed, recent evidence suggests that atmospheric nitrogen plays an important role in the water quality of Long Island Sound (New York), the Neuse River (North Carolina), Narragansett Bay (Rhode Island), Ochlockonee Bay (Florida), the Upper Potomac River Basin (Maryland and Pennsylvania), the New York Bight, and the Great Lakes (Turner et al., 1990).

TOXIC COMPOUNDS IN THE ATMOSPHERE

Lead in Atmospheric Deposition

The debate over SO₂ emissions and acidification has slowed since passage of the 1990 Clean Air Act, but the issue of toxic air pollution has intensified. For example, the effects of toxic metals emissions, such as lead, on aquatic ecosystems have received considerable interest recently. Less is known about air toxics and their effects on bodies of water than is now known about acidic deposition and surface-water acidification.

Not only can lead leach from acidified water distribution systems, but there is considerable lead emitted into the atmosphere. Lead (Pb) is a member of the Group IV elements (C, Si, Ge, Sn, and Pb) of the periodic table. Lead is truly metallic compared to carbon and silicon. Lead has stable (+2) and (+4) oxidation states. With the exception of nitrate and acetate, most (+2) lead salts, such as halides, hydroxides, sulfates, and phosphates, are only slightly soluble in water. Lead is one of the oldest metals known to man and, since medieval times lead has been used in building materials, piping, paint solders, ammunition, and castings. In modern times, lead has also been used in batteries, chemicals, and pigments.

Emissions of lead into the atmosphere have increased sharply during

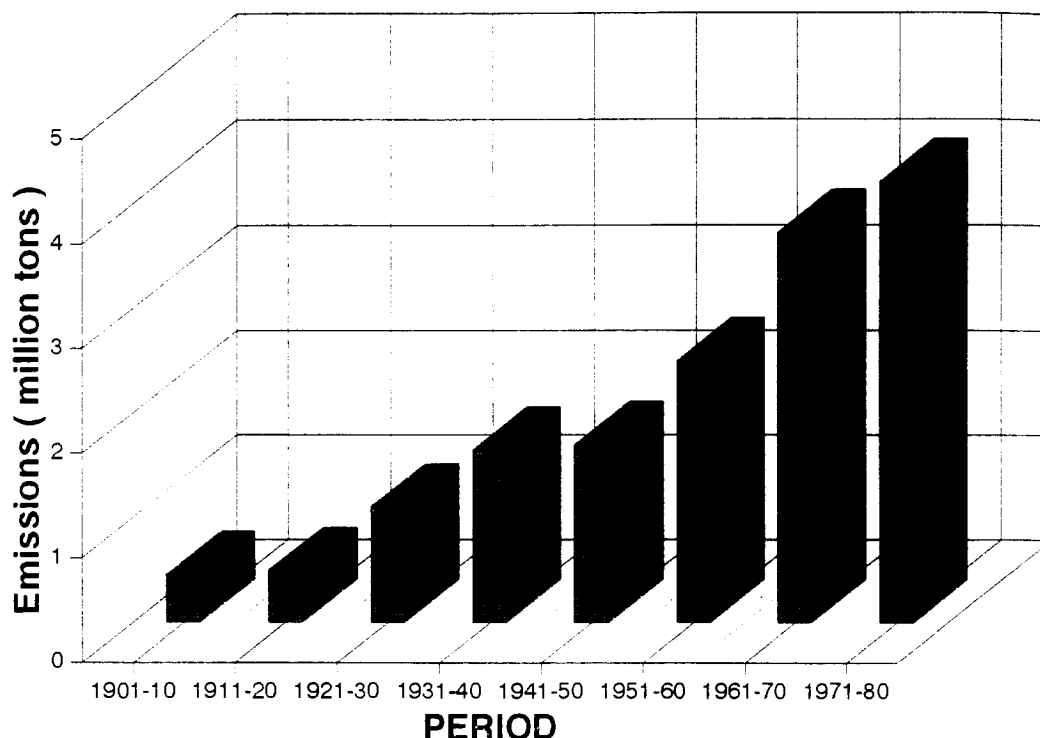


FIGURE 4.12. Global anthropogenic emissions of lead.
(After Nriagu and Pacyna, 1988.)

the twentieth century (Fig. 4.12). Since the peak emissions of the 1970s (4265×10^3 tonnes), lead emissions have declined. Anthropogenic emissions greatly exceed those from natural sources (Table 4.8). Although mining discharges contribute the majority of lead to the land, atmospheric deposition is usually the most important source of lead in surface waters.

The emission of lead into the atmosphere is largely the result of

TABLE 4.8 Comparison of Annual Anthropogenic and Natural Emissions of Lead

Source	Global Production (10^6 metric tonnes)	Global Emissions (10^3 metric tonnes)
Anthropogenic emissions	Not available	4265
Wind-blown dust	500	16
Forest fires	36	0.5
Volcanic particles	10	6.4
Vegetation	75	1.6
Seasalt spray	1000	0.2

Source: From Nriagu (1979).

vehicular emissions, though smaller amounts are emitted from stationary fossil-fuel combustion sources. The emission rates of lead from automobiles using leaded gasoline vary from 30% to 90% of the lead content, depending on vehicle's speed, operation, and exhaust system. Average emission rates for lead are 60% for urban driving and 80% for highway driving (Provenzano, 1978). Lead emissions have decreased dramatically in the United States and other developed countries in the past few years with the ban of lead as an automotive fuel additive. Lead particles generally coagulate to a diameter of 0.1 to 1.0 μm , and thus are capable of being transported long distances.

According to the EPA (1985), atmospheric lead concentrations range from 0.000076 $\mu\text{g}/\text{m}^3$ in remote areas to 10 $\mu\text{g}/\text{m}^3$ near point sources. Average annual lead concentrations in air in most areas were reported to be below 1.0 $\mu\text{g}/\text{m}^3$. The EPA calculated the average intake of lead from respiration to be approximately 1 $\mu\text{g}/\text{day}$. This is very low compared to the maximum drinking water intake, which would be 100 $\mu\text{g}/\text{day}$, assuming there are 50 $\mu\text{g}/\text{l}$ of lead present and a daily water intake of 2 liters.

Wet deposition of lead may roughly equal dry deposition. Lindberg et al. (1982) found nearly equal wet and dry annual fluxes of lead in the Walker Branch watershed, Tennessee. Peirson et al. (1973), however, showed that dry deposition of lead accounts for less than 10% of the total deposition at a rural site in England. This area, however, had much higher precipitation amounts. Lindberg (1982) showed that lead is higher in summer months due to increased air stagnation and a lower precipitation volume. Galloway, Eisenreich, and Scott (1980) reported the results of 32 studies of lead in wet deposition, wherein lead levels ranged from 0.6 to 64 $\mu\text{g}/\text{l}$, with a median of 12 $\mu\text{g}/\text{l}$. Direct atmospheric deposition of lead to a lake has been generally found to exceed watershed sources because lead levels in rocks and soils are generally low and lead is retained in soils by various mechanisms.

A significant amount of the total lead present in surface waters exists as suspended matter. The level of lead drawn from rivers and streams is generally slightly lower than the concentrations present in lakes and ground waters. The EPA (1988) reported the results of a survey by Fishman and Hem, who found that 86% of raw surface waters sampled contained less than 10 $\mu\text{g}/\text{l}$ lead and that less than 1% had over 50 $\mu\text{g}/\text{l}$. Clarkson, Baker, and Sharpe (1984) reported that most natural ground waters have concentrations ranging from 1 to 10 $\mu\text{g}/\text{l}$.

In urban areas lead becomes incorporated into street dust and accumulates within 1 meter of the curb (see Chapter 8 for a more detailed discussion of atmospheric sources of toxic contaminants in urban areas).

Health Effects

The health effects of lead are generally correlated with blood lead levels. High lead exposure is associated with the interference with the heme synthesis necessary for formation of red blood cells, anemia, kidney damage, impaired reproductive function, interference with vitamin D metabolism, impaired cognitive performance, delayed neurological and physical development, and elevations in blood pressure (U.S. EPA, 1988). Human absorption of lead is apparently independent of the chemical form of lead. Effects on children generally occur at lower blood lead levels compared to adults.

Because there are several sources of lead exposure other than drinking water, it is important to examine the relative contribution of water lead to health effects. The EPA (1988) reviewed a number of such studies and suggested that 50 µg/l of lead (the current MCL) in morning first-draw drinking water contributes less than the lowest observed effect level in children, but that this contribution could be important to the total body burden of lead from all sources. Other major sources of lead exposure in humans, including air, soil, food, cigarettes, paint, and other specific sources, can increase the range of mean blood lead levels tenfold.

Mercury in Atmospheric Deposition

Mercury (Hg) is perhaps the toxic pollutant most widely discussed with regard to emissions and effects on bodies of water. One reason is that mercury has a long residence time in the atmosphere and can be transported to remote and pristine ecosystems.

Mercury is a member of Group IIB of the periodic table. It is a silver-white, heavy, mobile liquid at 25°C with a freezing point of -38.9°C. It is the most volatile metal known, with a vapor pressure of approximately 2×10^{-3} torr at 25°C. The principal valence states of mercury are +1 (mercurous) and +2 (mercuric).

Mercury exists in two basic forms, inorganic salt and organic mercury compounds (methylated mercury). The major anthropogenic impact of mercury occurs when inorganic mercury in soils and sediments is methylated to a more toxic methyl-mercury (see Chapter 13).

Sources of Mercury

Mercury is found naturally in soil in the range of 30 to 300 mg/kg (Shacklette and Boerngen, 1984). In addition, there are known areas with substantially elevated mercury in geologic materials. For example, in the United States mercury in soils is elevated in certain parts of California and Nevada.

The major uses of mercury in industry are in electrical equipment (batteries, lamps, switches, and rectifiers) and in the chloralkali industry as a flowing cathode for electrolytic deposition of salt brine into chlorine, sodium hydroxide, and hydrogen. Mercury is used in agriculture as a fungicide. The medical profession uses mercury in dental and pharmaceutical products.

In addition to these sources, mercury may enter the environment from mining, smelting, and fossil-fuel combustion. Mercury is present in coal in the range of 10 to 46,000 mg/kg, though generally it is in the range of 200 to 400 mg/kg (U.S. EPA, 1985). Because of its high volatilization, most mercury in smelting and combustion processes is emitted to the atmosphere. On the other hand, mineral extraction can result in mercury contamination of soil and ground water from improper mining and reclamation practices. Soil and ground water can also become contaminated from improper disposal practices in industries that use mercury and burn fossil fuels. Natural processes such as volcanic activity, geothermal activity, and volatilization from mineral deposits also result in mercury entering the atmosphere.

Most atmospheric mercury has a long residence time in the atmosphere—on the order of months—and consequently is dispersed globally (Lindqvist et al., 1984). Still, a small but significant fraction of what is emitted at a particular source is deposited in its vicinity. Several studies of mercury deposition around chloralkali plants clearly show that deposition is augmented well above the background within 5 or 10 km from the plants. In fact, Lindqvist et al. (1984) calculated that 10% to 20% of the emissions are deposited within 10 km of the plants.

Mercury Exposure to Humans

The most significant pathways of mercury into humans include inhaled air, dust, food, water, and medical care. These exposure pathways are represented in Figure 4.13 and described further in the following list.

Table 4.9 shows estimates of natural and baseline exposures to mercury for children and adults. These estimates are based on consumption factors in inhaled air, dust, food, and drinking water. Estimates for natural exposures represent background exposure to natural global sources of mercury with no input from anthropogenic sources or localized elevated natural sources. The baseline exposures represent background anthropogenic global sources. No estimates are provided for medical exposure because of the specific nature of this route of exposure (for example, medications used, dental fillings present, and other dental treatments).

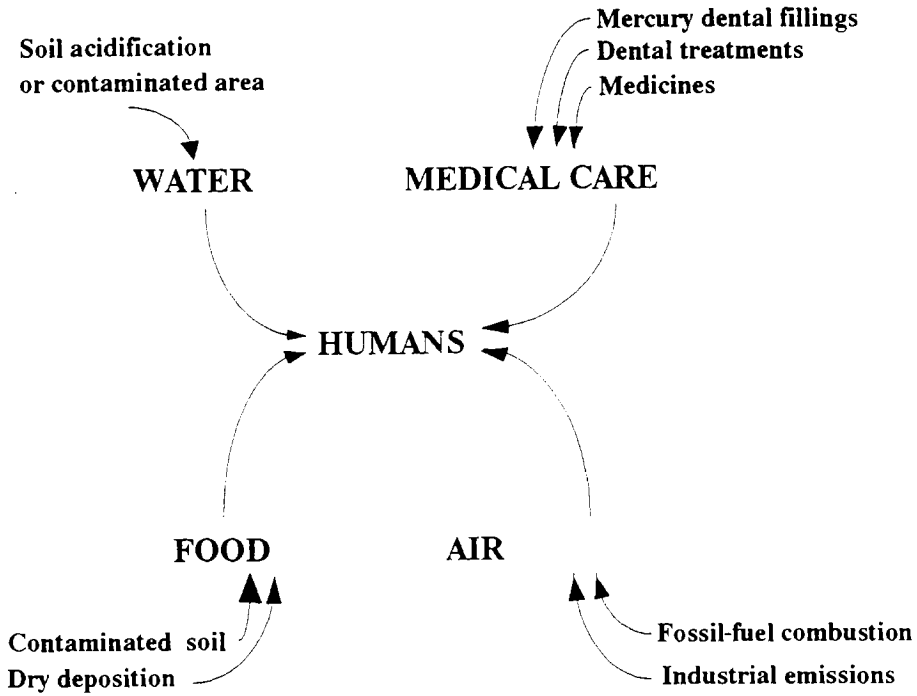


FIGURE 4.13. The quantitatively most significant exposure pathways of mercury in human uptake.

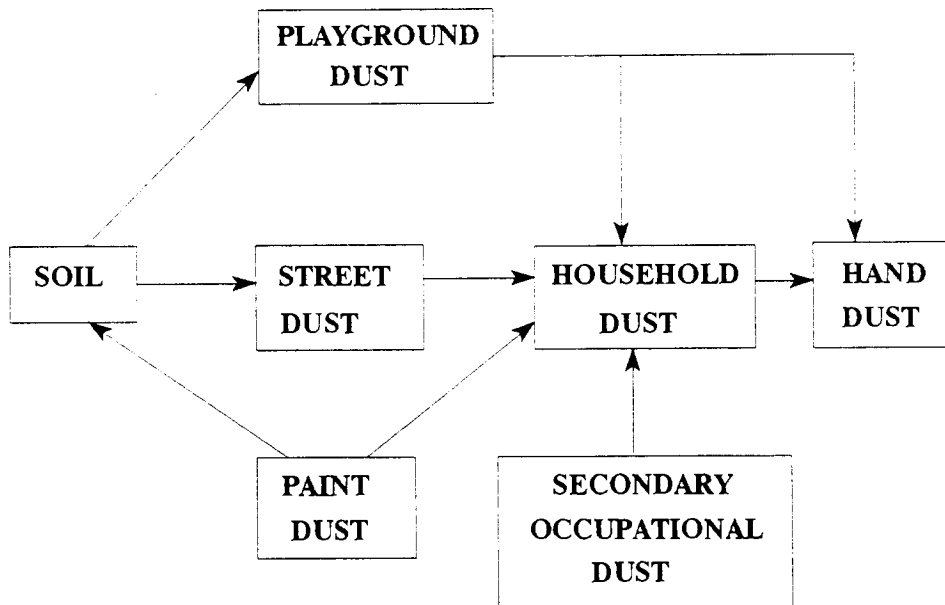


FIGURE 4.14. Intermediate pathways and source of exposure to mercury through dust.

TABLE 4.9 Exposure of Children and Adults of Natural and Baseline Concentrations of Mercury ($\mu\text{g}/\text{day}$)

Source	Children			Adults		
	Factor	Natural	Baseline	Factor	Natural	Baseline
Inhaled air	17 m ³ /day	0.00	0.00	20 m ³ /day	0.00	0.00
Dust	0.2 g/day	0.20	0.80	0.2 g/day	0.02	0.08
Food	1120 g/day	1.12	1.63	2200 g/day	2.20	3.20
Water	0.5 l/day	0.00	0.05	2.4 l/day	0.00	0.10
Total		1.32	2.48		2.22	3.38

Source: After Grant et al. (1990).

Inhaled air. Mercury is introduced into the air from both natural and anthropogenic sources, as described in the previous section. Methylmercury is absorbed from the lungs after inhalation and through the skin. Because mercury and its compounds tend to bind to soil and sediments in the environment, direct inhalation is a minor source of exposure compared to the other sources. Atmospheric concentrations may be as low as 0.01 ng/m³ for remote sources and 0.07 ng/m³ for rural areas. Levels may increase to as high as 4 ng/m³ near industrial emissions, but this level is still relatively low compared to other exposure pathways. Therefore, mercury uptake through inhaled air is considered an unmeasurable source at natural and background exposures compared to the others included in Table 4.9.

Dust. Mercury can enter dust from a variety of sources (Fig. 4.14). The dust is ingested and is absorbed from the gastrointestinal tract. The origin of most dust in the human environment is ultimately the soil. Mercury is deposited mainly as metallic metal and divalent salts to the soil. The divalent salts dominate because they are water soluble and are washed out of the atmosphere with wet precipitation. For children, almost all exposure to dust is by hand-to-mouth activity. Because hand-to-mouth contact is considerably higher in children, they have higher total exposure to mercury from this source than do adults. The exposure levels for dust are an order of magnitude higher for children compared to adults. Therefore, exposure to elevated levels of mercury in dust is an important pathway for children.

Food. Food is a primary source of exposure to mercury. Mercury is ingested with food and absorbed from the gastrointestinal tract. Humans absorb about 90% to 100% of the methylmercury that they ingest. Meat, fish, and poultry are the only important sources of

mercury in food, and the major dietary source appears to be the consumption of fish (Hoffman, 1989). Fruits, vegetables, and dairy products contain relatively little mercury (Gartrell et al., 1986), because methyl mercury is taken up by insects in the terrestrial environment and bottom feeders in the aquatic food chain and are passed up the trophic levels to carnivores and then to humans. Bioaccumulation also occurs along the way. Based on a number of studies, the dietary intake of mercury is estimated to be $3.2\ \mu\text{g}/\text{day}$ for the adult male (excluding beverages), of which about 80% comes from meat, fish, and poultry categories.

Drinking water. In the soil, divalent mercury is absorbed by organic matter in ion exchange sites, because it is easily mobilized during soil acidification. Several biochemical processes subsequently transform the mercury into organically bound forms that are not very mobile. Almost all mercury detected to date in drinking water is in the form of inorganic mercury, and the levels are normally below $0.5\ \mu\text{g}/\text{l}$. The estimated total mercury exposure to adults from drinking water is $0.1\ \mu\text{g}/\text{day}$.

Medical and dental exposure. A number of studies document that dental mercury amalgam fillings can cause substantial leaching of mercury, primarily through vapor emissions that are efficiently absorbed in the lungs or soft tissue (Sverdrup, 1990). There is a great deal of controversy concerning the relation between mercury poisoning and dental fillings. The evaluation and interpretation of all the results reported in the literature is beyond the scope of this discussion. The values reported in the literature indicate that the average mercury uptake to the body due to fillings is in the range of 1 to $50\ \mu\text{g}/\text{day}$. Other medical and dental sources of mercury include pharmaceutical products, such as various ointments and disinfectants for surficial wounds, eyedrops, and vaccines. Several of these products have been recently removed from the market by the manufacturer.

Guidelines and Standards for Maximum Human Intake

Many governments have set occupational exposure limits, drinking water concentration limits, and daily intake limits for mercury. Table 4.10 shows selected limits taken from the U.S. Environmental Protection Agency, World Health Organization, Swedish Food Administration and Environmental Protection Board, and the Japanese Food Administration. The actual limits were adapted as needed to standardize the numbers for comparative purposes. For example, the WHO limits for intake were

TABLE 4.10 Exposure Limits for Mercury Taken from Selected Government Guidelines and Standards

Mercury Form	Organization	Occupational 8 hr/5 day Air Limit ($\mu\text{g}/\text{m}^3$)	Environmental 24 hr/7 day Air Limit ($\mu\text{g}/\text{m}^3$)	Drinking Water Limit ($\mu\text{g}/\text{l}$)	Maximum Daily Uptake Limit ($\mu\text{g}/\text{day}$)	Maximum Daily Intake Limit ($\mu\text{g}/\text{day}$)
Methylmercury	WHO	25	1.3	1	22	25
	EPA	10	0.5	2	22	25
	Sweden	10	0.1	—	30	30
	Japan	—	—	—	35	35
Mercury vapor	WHO	50	—	—	50–80	60–100
	EPA	50	12	—	30	35
	Sweden	50	2.5	—	50–80	60–100
	Japan	50	—	—	—	—

Source: After Sverdrup (1990).

given per unit body weight, and for comparison have been presented here based on an average adult weight.

The toxic effects of mercury are related to the chemical form of the mercury compound. Based on toxicological characteristics, there are three important forms of mercury: elemental, inorganic, and organic compounds. Methylmercury is a highly toxic organic compound and is the chemical species of most concern in terms of potentially increased human health risk due to mercury exposure. Humans absorb about 90% to 100% of the methylmercury that they ingest. Methylmercury is resistant to environmental degradation and is capable of passing through important biological barriers such as the blood/brain membranes and the placenta. Inorganic mercury, on the other hand, is poorly absorbed through the gastrointestinal tract, does not penetrate cell membranes rapidly, and is less toxic than methylmercury.

Summary of lowest observed adverse effect levels. The lowest observed adverse effect levels for adults and for women during pregnancy are summarized in Table 4.11, which is based on the mercury poisonings mentioned previously and other clinical studies. By applying a safety factor of 10, tolerable daily intake becomes $30\mu\text{gHg/day}$ for adults and $8\mu\text{gHg/day}$ for women during pregnancy. As can be seen from a comparison of these figures with the baseline exposures described previously, the tolerable intake level for pregnant women does not allow for a large intake of mercury-contaminated food.

Environmental Effects

Effects on plants. Mercury can be introduced to plants from soils contaminated by atmospheric deposition or the application of wastewater treatment sludge. Fungicides based on mercury were used in the past for treating stored seed grain against molding, and the mercury may remain in the soil in the regions where it was used. No reports could be

TABLE 4.11 Lowest Observed Adverse Effect Levels for Mercury for Adults and for Women during Pregnancy

Mercury Levels (as methylmercury)	Critical Effects	
	Women during Pregnancy	Adults
Exposure ($\mu\text{g/kg/day}$)	0.8–1.7	4.3
Red blood cell mercury ($\mu\text{g Hg/l}$)	40–80	400
Hair ($\mu\text{g Hg/g}$)	10–20	50–100

Source: After Grant et al. (1990).

found of even these high levels of mercury in soils causing effects on plants.

Ecological effects due to soil and ground-water contamination. Typical levels of mercury in the soil are 0.1 to 1.2 mg/kg in the upper soil horizon (O-horizon), 0.0005 to 0.002 mg/kg in the A and B horizons, and as low as 0.003 to 0.008 mg/kg in the mineral soil (Sverdrup, 1990). The high mercury levels in the upper soil horizons indicate that mercury is not highly mobile in soils. This observation also suggests that it is not likely that mercury contamination results in high levels in ground waters.

Low ground-water mercury concentrations have been confirmed through surveys conducted by the U.S. Environmental Protection Agency and others (U.S. EPA, 1985). Compliance monitoring results from thousands of ground-water supplies in the United States show only 12 supplies with levels at any time above the drinking water limit for mercury of 2 µg/l. Of 95 ground-water samples collected by in the mid-1980s in acidified regions of the northeastern United States, the maximum mercury concentration was 1.1 µg/l (Taylor and Symons, 1984).

Pesticides and Other Organic Chemicals in Atmospheric Deposition

Generally, runoff from agricultural areas produces locally high concentrations of pesticides and herbicides, while atmospheric transport and deposition results in low-level but widespread contamination. It is generally accepted that certain organochlorine chemicals, such as DDT, PCBs, and toxaphene, can contaminate surface waters some distance from the source of application (Hites and Eisenreich, 1987; Mackay, 1990). These chemicals evaporate from their point of production or use, are transported in the atmosphere, and then enter surface waters by

- direct absorption
- wet and dry deposition
- washoff from the surrounding watershed.

The atmosphere becomes contaminated with pesticides and other organic chemicals by drift, wind erosion, and postapplication volatilization (Glotfelty et al., 1990). Drift is that portion of the spray that is moved away from the target area by wind. Aerial spraying (Fig. 4.15) contributes a large portion of the atmospheric drift input. Chesters and Simsiman (1974) estimated drift losses for pesticides applied aurally as ranging from 25% to 75% of the quantity applied. The extent of drift



FIGURE 4.15. View of pesticide spray operation by plane. (Photo: USDA.)

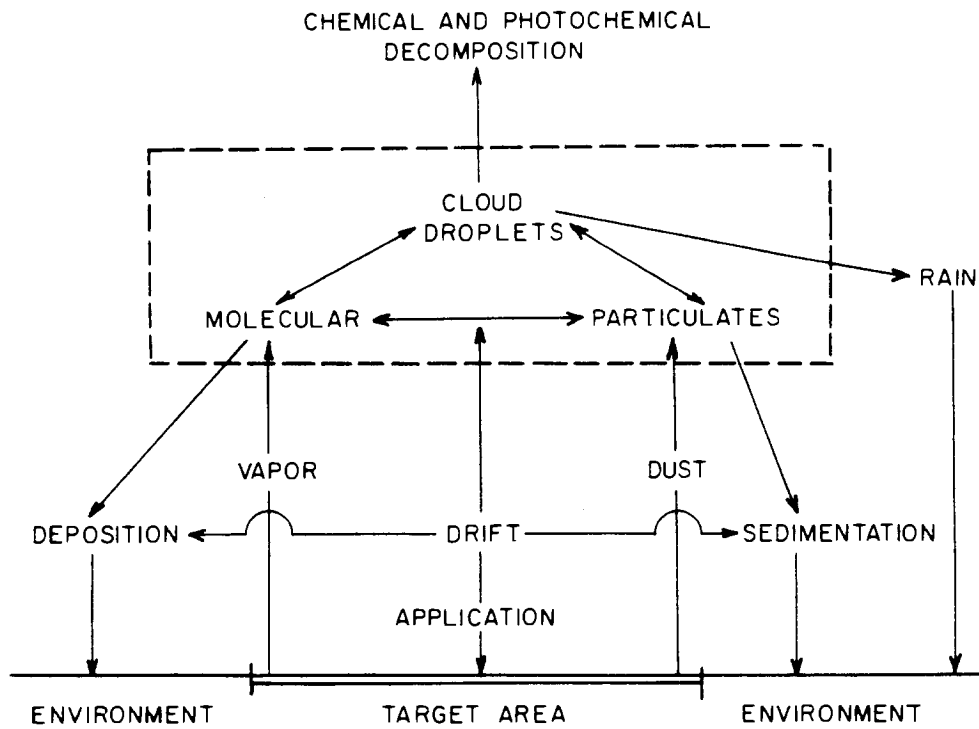


FIGURE 4.16. Pesticide input and removal from the atmosphere. (From Glotfelty, 1978.)

and the subsequent dispersion and transport of pesticides in the air are governed primarily by prevailing atmospheric conditions, the chemical constitution of the pesticide, and the method of application. High temperature and windy conditions (unstable atmosphere) accentuate drifting. The quantity and extent of drift losses can be reduced by considering a combination of interrelated factors such as spray formulation, type of equipment used, meteorological factors, and spray method used.

Volatilization has been recognized as a pathway of pesticide loss from soil, plant, and water surfaces. Once in soil, subsequent losses can differ depending whether the pesticide is surface applied or incorporated into the soil. Chapter 6 describes the interactions of pesticides and organic chemicals with soils. Largely due to their volatility, the chemicals also actively cycle between the atmosphere, soils, and water bodies. The cycle of pesticides between the soil and atmosphere is shown on Figure 4.16.

Although any organic chemical can cause a local diffuse-pollution problem, only persistent compounds such as organochlorine pesticides, DDT and its derivatives, aldrin, dieldrin, and PCBs have global significance. The discovery of DDT and PCBs in Antarctic seals and snow (Peterle, 1969; Sladen, Menzie, and Reichel, 1966; Cramer, 1973; and Rosenbrough et al., 1976) suggests that both components have become a part of global atmospheric pollution and travel with the air to the most remote points of the earth surface. Paasivirta (1990) compared ratios of organochlorine concentrations in fish and predatory birds from the Baltic versus the Arctic in an attempt to rank the organochlorine contaminants from most locally to most globally dispersed in the atmosphere (Fig. 4.17). Toxaphene was found to be the organochlorine compound most globally dispersed, while HexaCN was most locally dispersed. No natural loading exists for these substances.

Goolsby (1991) measured herbicide concentrations in rainwater in a 23-state area, principally in the Midwest and Northeast. He found traces

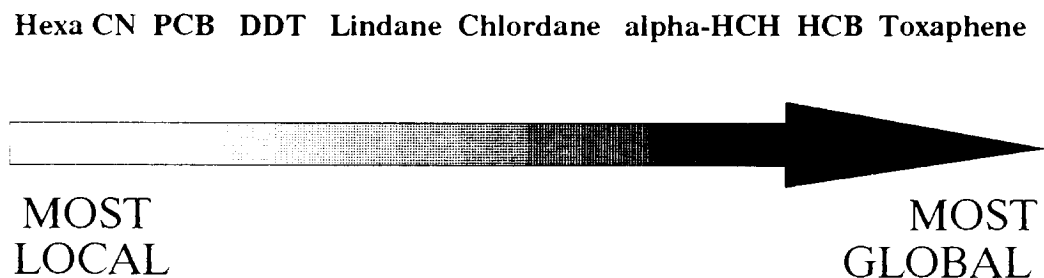


FIGURE 4.17. General ranking of organochlorine contaminants from mostly locally to mostly globally dispersed, based on tissue ratios from fish and predatory bird species from Baltic and Arctic environments. (After Paasivirta, 1990.)

TABLE 4.12 Projected Amounts of Pesticides Entering Chesapeake Bay via Summer (May–August) Rainfall

Pesticide	Wet Deposition (metric tonnes) ^a		
	1981	1982	1984
Atrazine	0.88	1.2	0.64
Simazine	0.13	0.11	0.14
Alachlor	3.7	2.4	9.8
Metolachlor ^b	1.7	1.1	4.6
Toxaphene	0.54	1.1	na ^c

Source: From Glotfelty et al. (1990).

^aPesticide data from Table 6, page 215, of Glotfelty et al. (1990). Area of Chesapeake Bay and tributaries is $11.9 \times 10^9 \text{ m}^2$, from W. B. Cronin, "Volumetric, areal, and tidal characteristics of the Chesapeake Bay estuary and its tributaries," Special Report 20, Johns Hopkins University, Baltimore, Md. (Ref No. 71-2), 1971.

^bOnly a partial set of measurements is available for metolachlor; the projected amounts of metolachlor entering Chesapeake Bay are based upon the relative concentrations of metolachlor and alachlor in the samples for which data are available.

^cna = not analyzed.

of herbicides in all 23 states and in all but two of the 81 collection sites. Pesticide concentrations in wet deposition were measured by Glotfelty et al. (1990) in Maryland. These latter researchers calculated the projected amounts of pesticides entering the Chesapeake Bay from precipitation. Table 4.12 gives the projected amounts of atrazine, simazine, alachlor, and toxaphene that entered the bay with rainfall during the summers of 1981, 1982, and 1984. Quantities of pesticides entering the bay ranged from a low of 110 kg of simazine in 1982 to a high of 9.8 tonnes of alachlor in 1984.

Strachen and Eisenreich (1990) attempted to calculate mass balances for selected chemicals entering the Great Lakes. The atmosphere was found to play an important, and often dominant role in the loading of many toxic chemicals into the Great Lakes. For example, 97% to 98% of the DDT loading into the upper Great Lakes was from the atmosphere. In the lower Great Lakes, with considerable toxic inputs from the land, the atmosphere played a smaller, but still important role, contributing 22% to 31% of the total DDT load.

Rapaport and Eisenreich (1986) reported that toxaphene deposition in peat bogs in northern Minnesota and northern Ontario reached a maximum in the 1970s that corresponded to the maximum production

and use of the pesticide in the United States. Derek et al. (1990) demonstrated that toxaphene is also present in remote lakes in northern Ontario and Manitoba due to long-range atmospheric transport and deposition.

Residence Time of Organic Chemicals in the Atmosphere and their Deposition

In the absence of a strong local source, the transport of persistent chemicals is of global nature. Despite its simplistic nature, the global model (Eqs. (4.1) to (4.4)) can be applied with some accuracy to estimate global (background) loadings, deposition rates, and residence times. For a strong local source a Gaussian dispersion model should be used (see Novotny and Chesters, 1981, or Zanneti, 1992). Atmospheric layers up to the middle latitudes become mixed relatively quickly; air parcels move around the world in 3 to 4 weeks; and a quasi-steady state is reached in months.

As shown previously, the residence time of a chemical component in the atmosphere depends on its form and sinks. There is a great deal of controversy as to the average global residence time and, hence, deposition rates of the most important, persistent chemicals. Because the vapor pressure of these chemicals indicates that they most likely exist in the vapor phase, the residence times were compared to those for gases, which are in years. For example, Woodwell et al. (1971) estimated that the residence time of DDT is 4 years; however, because the ambient concentrations of DDT were very low, Glotfelty et al. (1990) and Orgil et al. (1976) speculated that at least a part of the vaporized chemical becomes adsorbed by atmospheric aerosols and is deposited at a much faster rate and with much shorter residence times.

Junge (1977) estimated that the overall average residence time for atmospheric aerosols is about 7 days, and if ϕ denotes the fraction of an organic chemical present in the air adsorbed on aerosols, the residence time is approximated as $7/\phi$. For DDT the value of ϕ was estimated in the range 0.28 to 0.48, and ϕ for PCBs seems to be <0.01 . This would indicate that while the average residence time for DDT and similar organic components could be in months, the residence time of PCBs may be >1 year.

CONCLUSIONS

There are several major pollution sources from the atmosphere that can be transported long distances and fall to earth in rain, snow, mist, fog, and in dry form as gases and particulates. One of the best-known

pollutants in atmospheric deposition is acidity. Acidity occurs when nitrogen and sulfur oxides are emitted from fossil-fuel combustion. Nitrogen emissions have also been shown to directly affect bodies of water, particularly in coastal areas, through increases in fertilization and resultant eutrophication. The widespread concern for acidification has abated in the United States since passage of the Clean Air Act of 1990. The act has major provisions to control acidifying emissions and the general public believes the problem is being adequately addressed.

Fossil-fuel burning also emits other substances, such as mercury, that fall to earth and can affect aquatic ecosystems. Other major toxic atmospheric pollutants that affect water quality are trace metals, such as lead, and agricultural chemicals such as pesticides and herbicides. There has been less public debate over the issue of air toxics, but these pollutants are not as well understood as those involved in acidification and they may not be adequately addressed in current regulations.

References

- American Water Works Service Company (1988). *Lead at the Tap—Sources and Control. A Survey of the American Water System*. American Water Works Service Co., Voorhees, NJ.
- Baker, L. A., P. R. Kaufmann, A. T. Herlihy, and J. M. Eilers (1990). Current status of surface water acid-base chemistry, in *Acidic Deposition: State of Science and Technology*, Vol. II, *Aquatic Processes and Effects*, P. M. Irving, ed., National Acid Precipitation Assessment Program, Washington, DC, pp. 9-1-9-367.
- Chesters, G., and G. Simsiman (1974). *Impact of Agricultural Use of Pesticide on the Water Quality of the Great Lakes*, Task A-5, Water Resources Center, University of Wisconsin, Madison, WI.
- Clarkson, T. W., J. P. Baker, and W. E. Sharpe (1984). Indirect effects on health, in *The Acidic Deposition Phenomenon and Its Effects*, Committee on Monitoring and Assessment of Trends in Acid Deposition, National Research Council, National Academy Press, Washington, DC, pp. 6-1-6-66.
- Council on Environmental Quality (1990). *Environmental Quality*, 20th annual report of the Council on Environmental Quality, with the President's message to Congress, Washington, DC.
- Cramer, J. (1973). Model of the circulation of DDT on earth, *Atmos. Environ.* 7:241-256.
- Derek, C. G., N. P. Grift, C. A. Ford, A. N. Reiger, M. R. Hendel, and W. L. Lockhart (1990). Evidence for long-range transport of toxaphene to remote arctic and sub-arctic waters from monitoring of fish tissues, in *Long Range Transport of Pesticides*, D. A. Kurtz, ed., Lewis Pub., Chelsea, MI, pp. 329-346.

- Fisher, D., J. Caraso, T. Mathew, and M. Oppenheimer (1988). *Polluted Coastal Waters: The Role of Acid Rain*, Environmental Defense Fund, New York.
- Francis, J. D., B. L. Brower, W. F. Graham, O. W. Larson, III, J. L. McCaull, and H. M. Vigorita (1984). *National Statistical Assessment of Rural Water Conditions: Executive Summary*. U.S. Environmental Protection Agency, Office of Drinking Water, Washington, DC.
- Galloway, J. N., S. J. Eisenreich, and B. C. Scott, eds. (1980). *Toxic Substances in Atmospheric Deposition: A Review and Assessment*, Report NC-141, National Atmospheric Deposition Program, Charlottesville, VA.
- Gartrell, M. J., J. C. Craum, D. S. Podrebarac, and E. L. Gunderson (1986). Pesticides, selected elements, and other chemicals in infant and toddler total diet samples, *J. Assoc. Off. Anal. Chem.* **69**:123.
- Gatz, D. F. (1975). Pollutant aerosol deposition into southern Lake Michigan, *Water Air Soil Pollut.* **5**:239–251.
- Glotfelty, D. E. (1978). The atmosphere as a sink of applied pesticide, *J. Air Pollut. Assoc.* **28**:917–921.
- Glotfelty, D. E., G. H. Williams, H. P. Freeman, and M. M. Leech (1990). Regional atmospheric transport and deposition of pesticides in Maryland, in *Long Range Transport of Pesticides*, D. A. Kurtz, ed., Lewis Publ., Chelsea, MI, pp. 199–221.
- Goolsby, D. A. (1991). *Regional Studies of Agricultural Chemicals in Water Resources of the Midcontinental United States*, progress report, U.S. Geological Survey, Lakewood, CO.
- Grant, L. D., R. A. Goyer, H. Olem, W. J. Nicholson, and R. W. Elias (1990). Indirect health effects associated with acidic deposition, in *Acidic Deposition: State of Science and Technology*. Vol. III. *Terrestrial, Materials, Health, and Visibility Effects*, P. M. Irving, ed., National Acid Precipitation Assessment Program, Washington, DC.
- Gschwandtner, G., J. K. Wagner, and R. B. Husar (1988). *Comparison of Historic SO₂ and NO_x Emission Data Sets*, EPA-600/7-88-009a, U.S. Environmental Protection Agency, Research Triangle Park, NC.
- Hites, R. A., and S. J. Eisenreich, eds. (1987). *Sources and Fates of Aquatic Pollutants*, ACS Advances in Chemistry Series, No. 216, American Chemical Society, Washington, DC.
- Hoffman, S. (1989). *Acid Precipitation and Human Health*, Rep. EN-6492, Electric Power Research Institute, Palo Alto, CA.
- Irving, P. M., ed. (1991). *Acidic Deposition: State of Science and Technology*, Summary Report, U.S. National Acid Precipitation Assessment Program, NAPAP, Washington, DC.
- Jacks, G. (1984). Trace elements in water supplies, in *Trace Elements in Health and Disease*, Skandia International Symposia Proceedings, Almquist and Wiskell International, Stockholm, pp. 40–48.
- Junge, C. E. (1977). Basic considerations about trace constituents in the atmosphere as related to the fate of global pollutants, in *Fate of Pollutants in the Air and Water Environment—Part I*, I. H. Suffert, ed., Wiley Interscience, New York.

- Kohout, E. H., D. J. Miller, and L. Nieves (1990). *Month and State Current Emission Trends for NO_x, SO_x, and VOC: Methodology and Results*, Argonne National Laboratory, Argonne, IL.
- Lindberg, S. E. (1982). Factors affecting trace metal, sulfate and hydrogen ion concentrations in rain, *Atmos. Environ.* **16**:1701-1709.
- Lindqvist, O., A. Jernelov, K. Johansson, and H. Rodhe (1984). *Mercury in the Swedish Environment: Global and Local Sources*, Rep. 1816, Swedish Environmental Protection Board, Solna.
- Linthurst, R. A., D. H. Landers, and J. M. Eilers (1986). *Characteristics of Lakes in the Eastern United States*. Vol. I. *Population Descriptions and Physico-Chemical Relationships*, EPA/600/4-86-007a, U.S. Environmental Protection Agency, Washington, DC.
- Logtin, G. (1988). *National Inorganics and Radionuclides Survey*, U.S. Environmental Protection Agency, Cincinnati, OH.
- Mackay, D. (1990). Atmospheric contributions to contamination of Lake Ontario, in *Long Range Transport of Pesticides*, D. A. Kurtz, ed., Lewis Publ., Chelsea, MI.
- McMahon, T. A., P. J. Dension, and R. Fleming (1976). A long distance air pollution transportation model incorporating washout and dry deposition component, *Atmos. Environ.* **10**:751-761.
- Meranger, J. C., K. S. Subramanian, and C. Chalifoux (1979). A national survey for cadmium, chromium, copper, lead, zinc, calcium, and magnesium in Canadian drinking water supplies, *Environ. Sci. Technol.* **13**:707-711.
- Novotny, V., and G. Chesters (1981). *Handbook of Nonpoint Pollution: Sources and Management*, Van Nostrand Reinhold, New York.
- Nriagu, J. O. (1979). Global inventory of natural and anthropogenic emissions of trace metals to the atmosphere, *Nature* **279**:409-411.
- Nriagu, J. O., and J. M. Pacyna (1988). Quantitative assessment of worldwide contamination of air, water, and soils by trace metals, *Nature* **333**:134-139.
- Orgil, M. M., A. Schnell, and M. R. Petersen (1976). Some initial measurements of airborne DDT over Pacific Northwest forest, *Atmos. Environ.* **10**:827-834.
- Paasivirta, J. (1990). Predicted and observed fate of selected persistent chemicals in the environment, in *Dioxin '90, Short Papers*, Vol. 7, Bayreuth, FRG, pp. 367-374.
- Patterson, J. W. (1981). *Corrosion in Water Distribution Systems*, Report to U.S. Environmental Protection Agency, Office of Drinking Water, Washington, DC.
- Patterson, J. W., and C. E. Junge (1971). Sources of particulate matter in the atmosphere, in *Man's Impact on the Climate*, W. H. Mathew, et al., eds., MIT Press, Cambridge, MA, pp. 310-320.
- Patterson, J. W., and J. E. O'Brien (1979). Control of lead corrosion, *J. Am. Water Works Assoc.* **71**:264-271.
- Peirson, D. H., P. A. Cawse, L. Salmon, and R. S. Cambray (1973). Trace elements in the atmospheric environment, *Nature* **241**:252-256.
- Peterle, T. J. (1969). DDT in Antarctic snow, *Nature* **224**:629-630.

- Provenzano, G. (1978). Motor vehicle lead emissions in the USA: An analysis of important determinants, geographic patterns, and future trends, *J. Air Pollut. Assoc.* **28**:1193–1199.
- Rapaport, R. A., and S. J. Eisenreich (1986). Atmospheric deposition of toxaphene to eastern North America derived from peat accumulation, *Atmos. Environ.* **20**:931–941.
- Rosenbrough, R. W., et al. (1976). Transfer of chlorinated biphenyls to Antarctica, *Nature* **264**:738–739.
- Shacklette, H. T., and J. G. Boerngen (1984). *Element Concentrations in Soils and Other Surficial Materials of the Conterminous United States*, U.S. Geological Survey, Prof. Paper 1270, Denver, CO.
- Singley, J. E., B. A. Beaudet, and P. H. Markey (1984). *Corrosion Manual for Internal Corrosion of Water Distribution Systems*, Rept. No. ORNL/TM-8919, Oak Ridge National Laboratory, Oak Ridge, TN.
- Sisterson, D. L., et al. (1990). Deposition monitoring: Methods and results, in *Acidic Deposition: State of Science and Technology*, Vol. I: *Emissions, Atmospheric Processes, and Deposition*, P. M. Irving, ed., National Acid Precipitation Assessment Program, Washington, DC, pp. 6-1–6-338.
- Slade, D. H., ed. (1968). *Meteorology and Atomic Energy—1968*, U.S. Atomic Energy Commission, Washington, DC.
- Sladen, W. J. L., C. M. Menzie, and W. L. Reichel (1966). DDT residues in Adelie penguins and a crabeater seal from Antarctica, *Nature* **210**:670–671.
- Stern, A. C., ed. (1976). *Air Pollution*, Vol. 1–5. Academic Press, New York.
- Strachan, W. M. J., and S. J. Eisenreich (1990). Mass balance accounting of chemicals in the Great Lakes, in *Long Range Transport of Pesticides*, D. A. Kurtz, ed., Lewis Pub., Chelsea, MI, pp. 291–301.
- Sverdrup, H. U. (1990). *Acidification and Health Effects*, Report, Dept. of Chemical Engineering, Lund Institute of Technology, Lund, Sweden.
- Taylor, F. B., and G. E. Symons (1984). Effects of acid rain on water supplies in the Northeast, *J. Am. Water Works Assoc.* **76**:35.
- Taylor, F. B., J. A. Taylor, G. E. Symons, J. J. Collins, and M. R. Schock (1984). *Acid Precipitation and Drinking Water Quality in the Eastern United States*, EPA-600/2-84-054, U.S. Environmental Protection Agency.
- Turner, R. S., et al. (1990). Watershed and lake processes affecting surface water acid-base chemistry, in *Acidic Deposition: State of Science and Technology*. Vol. II. *Aquatic Processes and Effects*, P. M. Irving, ed., National Acidic Precipitation Assessment Program, Washington, DC.
- U.S. Environmental Protection Agency (1985). National Primary Drinking Water Regulations. Synthetic organic chemicals, inorganic chemicals, and microorganisms; proposed rule, *Fed. Regist.* **50**:46936.
- U.S. Environmental Protection Agency (1988). Drinking water regulations; maximum contaminant level goals and National Primary Drinking Water Regulations for lead and copper; proposed rule, *Fed. Regist.* **53**(160):31516–31578.
- U.S. Environmental Protection Agency (1990). *National Air Pollutant Emission*

Estimates, 1940-1988, EPA-450/4-90-001, U.S. Environmental Protection Agency, Research Triangle Park, NC.

Woodwell, G. M., P. P. Craig, and H. A. Johnson (1971). DDT in the biosphere: Where does it go? *Science* **174**:1101-1107.

Zannetti, P. (1992). *Air Pollution Modeling*, Van Nostrand Reinhold, New York.

5

Erosion and Sedimentation

My earliest lessons on environmental protection were about the prevention of soil erosion on our family farm.

Al Gore

“If we assume that the average worldwide rate of erosion is on the order of, say 2.7 cm per thousand years the average land elevation of the continents (840 m) could be reduced to sea level in about 31 million years” (Leopold, Wolman, and Miller, 1964). This approximate estimate by these well-known geomorphologists (scientists who study the process of the evolution of the earth surface) indicates the scope and magnitude of natural erosion that has been carving the earth for billions of years. If indeed erosion was the only process shaping the earth surface, the whole world would have already been reduced to a featureless marshy flat plain billions of years ago. However, the erosion in the denudation process is countered by tectonic uplifts, hydrostatic buoyancy uplift of land masses, subsidence, and other processes. Furthermore, erosion is not uniform and depends on slope, climate, properties of the parent geological material, and weathering and other factors (Leopold, Wolman, and Miller, 1964). Walling and Webb (1983) estimated the average annual rates of sediment yield (sediment from erosion reaching streams) throughout the world. They reported that the rates of natural erosion and sediment yields range from less than 1 tonne $\text{km}^{-2}\text{y}^{-1}$ for some rivers in Poland and New South Wales in Australia to more than 10,000 tonnes $\text{km}^{-2}\text{yr}^{-1}$ for some tributaries of the Yellow River in China and some rivers in Kenya, New Zealand, Taiwan, Java in Indonesia, and New Guinea. The discharge of a sediment-laden stream into one of the Great Lakes is shown on Figure 5.1.

Geomorphologists have been interested in the process of erosion and deposition for a long time. Only recently, however, have water quality and pollution dimension been added to the science of geomorphology



FIGURE 5.1. Sediment discharge from a stream into one of the Great Lakes. (Photo: University of Wisconsin, Madison, Department of Agricultural Journalism.)

(Wolman, 1977; Walling and Webb, 1983; Walling, 1983; Novotny and Chesters, 1989).

DEFINITION AND DESCRIPTION OF THE EROSION PROCESS

Denudation is a geomorphologic process consisting of weathering or breakdown of the parent rock material, entrainment of the weathered debris, and transportation and deposition of that debris. The word *erosion* is often used synonymously with denudation, although erosion classically applies to entrainment and transportation, but not to weathering (Leopold, Wolman, and Miller, 1964). Portions of this chapter were previously published in Novotny and Chesters (1981).

Erosion and weathering are a result of stresses applied on the surface of the earth. These stresses can be gravitational, molecular, or chemical. Water and air movement over the surface are sources of the gravitational stresses; however, other stresses, such as ruptures, also cause the move-

ment of geological materials. Chemical weathering, for example, due to natural or anthropogenic acidity of the rainfall, is also of great interest, since it is a primary cause of the background chemical quality of ground and surface waters. Molecular and chemical stresses are caused primarily by weathering, while gravitational stresses are generally responsible for entrainment, transport, and deposition. The properties of geological materials considered under gravitational stresses, primarily due to water and air action, range from rigid resistant hard rocks to flowing fluids such as mudflows.

Erosion processes can be divided into sheet and rill erosion (Figs. 5.2 and 5.3), gully erosion (Fig. 5.4), stream and floodplain scour (shown on Figs. 5.5 and 1.1), and shoreline erosion (Fig. 5.6). Typical roadside erosion (Fig. 5.7) could be classified as gully erosion. Erosion from farms, construction, and surface mines is commonly of the sheet and rill type, with gully formation in the drainageways.

Hydrologically, these processes can be classified into sheet (upland) erosion and stream or channel erosion and transport. Thus, the total amount of on-site sheet and channel erosion is gross (potential) erosion.

Erosion, usually expressed in tonnes per km² or hectare per time (year), should be distinguished from *sediment yield*, which is the flow of sediment measured in the receiving body of water in the same time period. As will be described (the section titled "Sediment Delivery and Enrichment Processes during Overland Flow"), significant differences between erosion rates summed up over the watershed area and sediment yield should be expected. A *delivery ratio* (DR) is then a dimensionless ratio of the sediment yield divided by the total potential erosion in the contributing watershed, or

$$DR = \frac{Y}{A} \quad (5.1)$$

where

DR = delivery ratio (dimensionless)

Y = sediment yield

A = gross erosion in the watershed

The ease with which materials give way during erosion is called *erodibility*. Erodibility, as will be shown, depends on both the composition of the geological material and the state of its consolidation. Consolidated rocks obviously have low erodibility. For unconsolidated geological materials (soils, river deposits, sand dunes, etc.) erodibility depends on particle size and the texture of the material, water content, composition



FIGURE 5.2. Agricultural rill and interrill (sheet) erosion. (Photo: University of Wisconsin, Madison, Department of Agricultural Journalism.)



FIGURE 5.3. Sheet erosion of construction sites. (Photo: USDA—SCS.)



FIGURE 5.4. Gully erosion. (Photo: University of Wisconsin, Madison, Department of Agricultural Journalism.)



FIGURE 5.5. Streambank erosion. (Photo: USDA—Soil Conservation Service.)



FIGURE 5.6. Shoreline erosion. (Photo: V. Novotny.)



FIGURE 5.7. Roadside erosion. (Photo: V. Novotny.)

of the material, and the presence or absence of protective surface cover such as vegetation. Leopold, Wolman, and Miller (1964) stated that even a small amount of clay in the unconsolidated mixture markedly improves the cohesion of the soil. Cohesion is also increased by organic materials and by chemical bonding.

Vegetative cover is also extremely important, since it provides additional resistance to shear stresses caused by falling and running water and wind. Water is the primary agent responsible for erosion. As pointed out by Leopold, Wolman, and Miller, only in regions with virtually no precipitation can wind be expected to be the dominant erosive force. Hence there is direct proportionality between the annual precipitation and erosion. However, the intensity of the vegetative cover that reduces erosion is also proportional to the amount of precipitation. This leads to the relation between the annual precipitation and sediment yield published by Langbein and Schumm (1958) shown on Figure 5.8. This figure is only a conceptual schematic since other factors such as the slope of the relief and man-induced changes in vegetation and surface character have not been included. As a matter of fact, Walling and Webb (1983) documented that global data do not follow this concept, and obtained a large scatter of sediment yields on the Lanbein–Schumm chart. However, they have pointed out that the Lanbein–Schumm concept is physically sound and meaningful for an undisturbed watershed in continental climates, such as the United States.

Erosion, sediment transport, and deposition are to a large degree natural processes that have been occurring throughout the geological ages. In many areas the top surface layers are very young, that is, years to thousands of years. These areas are commonly mildly sloping lowlands, alluvial fans, and alluvial deposits located where sediment is being deposited by surface flow or wind. The process of the deposition of sediment and build up of sedimentary deposits is called *aggradation*. Areas on exposed high slopes are the principal sources of sediment. *Degradation* is a process by which sediment is eroded away from the area. The processes of aggradation and degradation are shown in Figure 5.9.

Due to the facts that erosion is a natural process and significant quantities of sediments are being moved as a result of natural denudation, it would be unrealistic to expect or require complete control or elimination of sediment loads to receiving waters. Such “wall-to-wall” control measures would be technically and economically impossible. However, it is feasible to control or manage excessive sediment loadings that have resulted from man’s land use activities that would be detrimental to the quality of the receiving bodies of water and to the aquatic and terrestrial habitat.

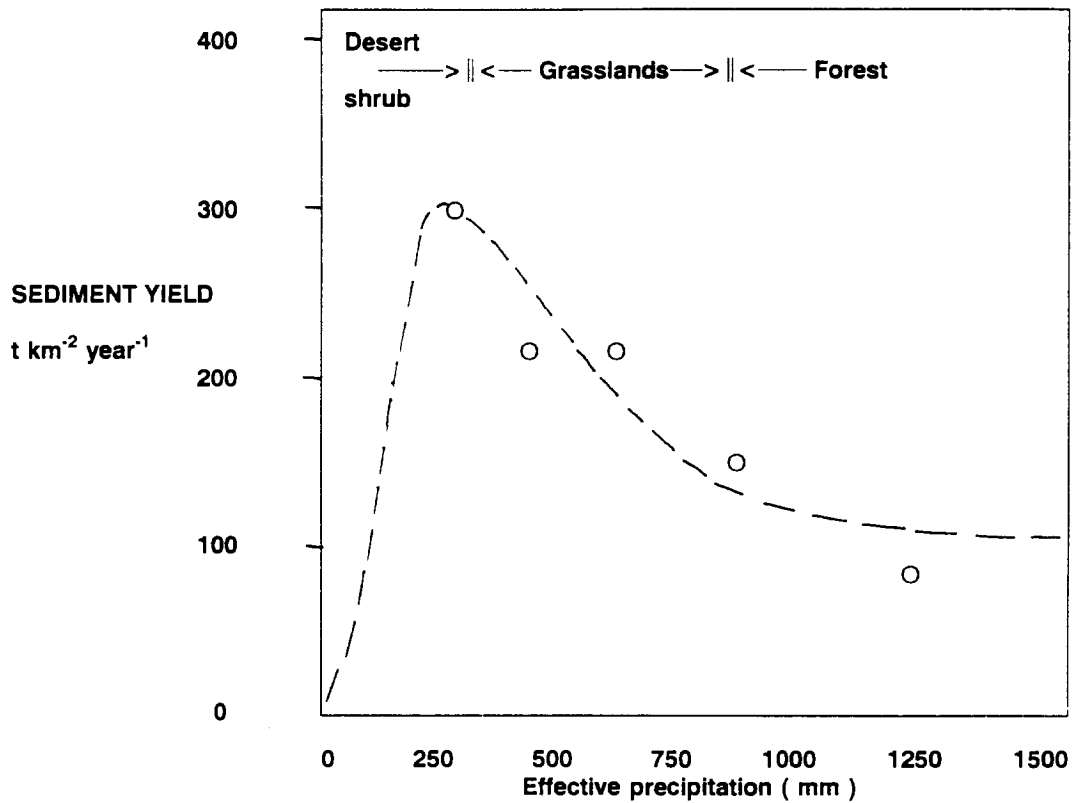


FIGURE 5.8. The relationship between sediment yield and annual precipitation proposed by Langbein and Schumm (1958). (After Walling and Webb, 1983.)

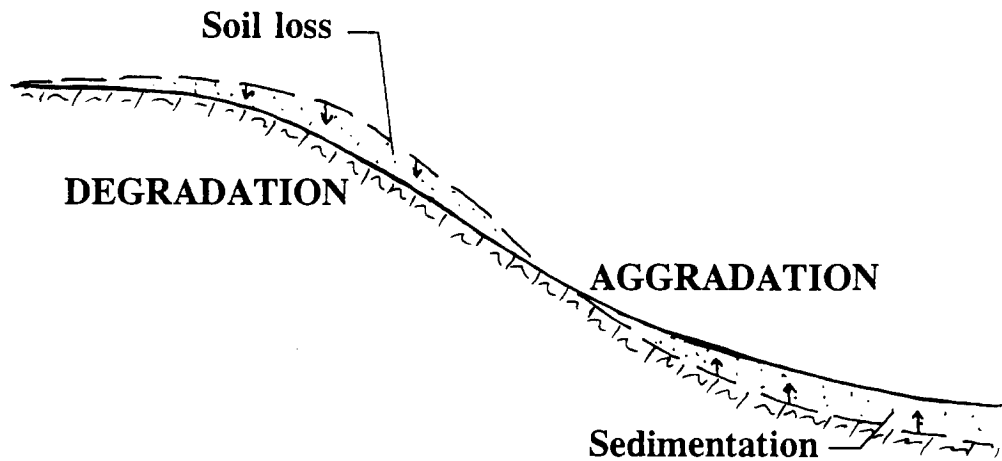


FIGURE 5.9. Watershed aggradation and degradation concept.

Classification of Eroded Sediment

Eroded soil materials are classified according to their grain size in clay, silt, sand, and gravel. Clay particles range from 10^{-4} to 10^{-3} mm, silt ranges from 10^{-3} to 10^{-2} mm, sand from 10^{-2} to 10^0 mm, and gravel particles are greater than 10^0 mm. Table 5.1 provides information on the distribution of sediment particle sizes.

Soil materials are classified as cohesive and noncohesive sediments. Cohesive sediments include clays and other fine particles, including organic materials. These fractions can form bonds between particles that resist scour, cause flocculation during overland and channel flows, and adsorb pollutants. Noncohesive sediments, mostly particles of sand and gravel size, are transported as individual particles, the movement of which depends solely on such particle properties as mass, shape, size, and relative position of particles with respect to surrounding particles. Their sorption potential is small compared to cohesive sediments.

Once in motion (overland or channel flow), sediments transported by the flow follow two modes (Vanoni, 1975; Shen, 1982):

1. as suspended load (*washload*), where the sediment is moved away from the bed and supported mainly by the turbulence of the flow;
2. as *bedload*, where the sediment is moved near the bed and supported most of the time by the bed.

Washload is carried by streams in suspension, while bedload movement consists of shifting bottom sediments due to the intensity of the turbulence and mean stream velocity. It has been found by Gottschalk

TABLE 5.1 Scale of Sediment Particle Sizes

Class	Particle Size (mm)	
	AGU Scale	USDA Scale
Cobbles and boulders	>64	>10
Gravel	2-64	2-10
Very coarse sand	1-2	1-2
Coarse sand	0.5-1.0	0.5-1.0
Medium sand	0.25-0.5	0.25-0.5
Fine sand	0.125-0.25	0.1-0.25
Very fine sand	0.062-0.125	0.05-0.1
Silt	0.004-0.062	0.002-0.05
Clay	<0.004	<0.002

Note: The scale is according to the American Geophysical Union (AGU) and the U.S. Department of Agriculture (USDA).

(1964) and Leopold, Wolman, and Miller (1964) that most of the sediment carried by streams in humid areas originates from sheet erosion and is in the form of fine sediments or washload. Transport of sediments in channels also depends on whether the sediments are cohesive or noncohesive. Cohesive fine sediments, with the exception of impounded sections with very low velocity, are mostly incorporated into washload. There is an exchange of noncohesive sediments between the bottom (bedload) and suspended (washload) forms; however, it is suspected that such continuous exchanges between bedload and washload sediments do not exist for cohesive fine sediments (see the section titled "Sediment Transport in Streams," describing channel transport of cohesive and noncohesive sediments).

Enrichment ratio refers to the difference in particle size distribution and sorbed pollutant content of washload particles and the soils from which the sediment originated. Erosion and sedimentation are the selective processes by which clay and fine particles are first liberated from soils while they are deposited last during redeposition in overland flow.

Erosion control implies the reduction of soil loss by land management and the reduction of the delivery of sediment from the source area to the receiving body of water that can be accomplished by land management, buffer strips, channel modification, sediment traps, and other structural and nonstructural measures and practices. Chapters 10 and 11 contain descriptions and design procedures for sediment management aimed at water quality control. A publication of Goldman, Jackson, and Bursztynsky (1986) and a number of manuals available from the Soil Conservation Service of the USDA, state pollution control agencies, and other sources provide ample information on erosion control. Construction erosion control is now mandated in many states.

Effect of Cultural (Man's) Land-Use Activities on Erosion and Sediment Yields

It has been already pointed out in this book and several other publications (for example, The Committee on International Soil and Water Research and Development of the National Academy of Sciences, 1991) that deforestation in areas of high precipitation and high slope can be devastating both to water quality and flooding. Catastrophic depositions of sediments and increased flooding in Bangladesh are a result of deforestation in the high precipitation regions of the Indian Himalayan mountains.

Changing climate conditions are often related to the amount of disturbance of the watershed. Man-made (cultural) impacts in various portions

of the world may have lasted for the thousands of years (Egypt, Middle East, China, India) or for a hundred years or less (North America). There are historically documented occurrences where man has dramatically increased erosion such as the deforestation of the Middle East, which converted Syria, among other countries, into highly erosive arid regions. Another example is the Adriatic coast of Dalmatia, which was deforested by the Venetians during the Middle Ages. However, the intensity of watershed disturbances has increased between then and now.

Based on measurements by the agricultural research centers in the United States following the devastating "dust bowl" erosion of farmland during the 1930s, which then culminated in the development of the Universal Soil Loss Equation by Wischmeier and Smith (1965), it was found that general land disturbance by agriculture or construction can increase erosion rates by two or more orders of magnitude. Patrick (1975) quoting works by Leopold (1968) and Borman and Likens (1969) pointed out that cutting down a forest—reducing the forest from 80% to 20%—increased sediment yield in a riverine system eight times. Walling and

TABLE 5.2 Increases in Sediment Yield Caused by Land-Use Changes

	Land-Use Change	Increase in Sediment Yield	Source
Rajasthan, India	Overgrazing	× 4–18	Sharma and Chatterji (1982)
Utah, U.S.	Overgrazing of rangeland	× 10–100	Noble (1965)
Oklahoma, U.S.	Overgrazing and cultivation	× 50–100	Rhoades, Welch, and Coleman (1975)
	Cultivation	× 5–32	
Texas, U.S.	Deforestation and cultivation	× 340	Chung, Roth, and Hunt (1982)
Northern California, U.S.	Conversion of steep forest to grassland	× 5–25	Anderson (1975)
Mississippi, U.S.	Deforestation and cultivation	× 10–100	Ursic and Dendy (1965)
South Brazil	Deforestation and cultivation	× 4500	Bordas and Canali (1980)
Oregon, U.S.	Deforestation (clear-cutting)	× 39	Frederiksen (1970)
Ontario, Canada ^a	Urbanization	× 26	Ostry (1982)
	Agriculture	× 14	

Source: After Walling and Webb (1983).

^a Compared to undisturbed forested land in the same area.

Webb (1983) compiled data from selected watershed studies to document the impact of a particular land-use activity on the increase of sediment yields (Table 5.2). Walling (1979), and Ostry (1982) conducted studies on two parallel watersheds, one undisturbed and one disturbed. These studies confirmed that the suspended sediment loads of many rivers may have increased by an order of magnitude or more as a result of cultural land-use changes within the watershed.

The land-disturbing activities were listed in Chapter 1. In addition to deforestation, construction-site erosion and intensive agriculture on highly erodible lands are the activities that will result in the highest rates of erosion.

According to the definition of pollution introduced in Chapter 1, sediment from natural undisturbed watersheds, volcanic eruptions, and the natural weathering of soils and minerals are not considered pollution, although sediment yields may sometimes reach large proportions. One example of this is the sediment load of the Columbia River in the Northwest United States following the eruption of Mount St. Helen in 1980. Pollution in the form of elevated sediment yields results from man-made modification of watersheds.

The following land-use activities by man are producing large sediment loads (Clark, Haverkamp, and Chapman, 1985):

1. Agricultural erosion is a major sediment source due to the large areas involved and the land-disturbing effects of cultivation. Most sediment loading from agricultural zones takes place during the spring, especially when spring rains fall on still frozen soils. Reported erosion rates range from 100 to 4000 tonnes $\text{km}^{-2}\text{yr}^{-1}$ (Brant et al. 1972; Clark, Haverkamp, and Chapman, 1985).
2. Urban erosion. Sediment originates mainly from exposed soils in areas under construction and from street dust and litter accumulation on impervious surfaces. Sediment yields from developing urban areas can be extremely high, sometimes reaching values up to 50,000 tonnes $\text{km}^{-2}\text{yr}^{-1}$ (Brant et al. 1972; Novotny, 1980; Novotny and Chesters, 1981).
3. Highway erosion is associated with the stripping of large areas of their vegetative protection during road construction. Uncompacted or unsettled fills may be subject to intensive rill erosion and, eventually, to landslides. The magnitudes of sediment yields from these areas are similar to those from urban areas under construction.
4. Silvicultural erosion is caused by clear-cutting practices, road building, and timber transportation. Due to the significant quantities of vegetation residues remaining on the surface, erosion rates from silviculture

are deemed to be one to two orders of magnitude less than that from cropland (Brown, 1985).

5. Erosion of strip mines is similar in nature and magnitude to erosion from large construction sites. The potential for erosion is created as soon as vegetation is removed to construct roads and facilitate prospecting and mining. The mining process itself also creates an even greater erosion potential.
6. Stream bank, channel, and shoreline erosion results from concentrated water flows and wave action in channels and floodplains. Channel erosion usually is a source of large-grain-sediments, and in most cases represents a smaller portion of the total sediment loading unless the stream is undergoing structural channelization with excavation. Sediment loadings originating from bank and channel erosion of some streams in arid and semiarid regions (Fig. 1.1) may be quite high (Nordin, 1962).

Steward et al. (1975) listed the following sources and conditions that cause excessive sediment movement:

Farming on long slopes without terraces or runoff diversions
 Row cropping up and down moderate or steep slopes
 Bare soil surface following seeding of crops
 Bare soil between harvest and the establishment of a new crop canopy
 Intensive cultivation close to a stream
 Intensive runoff from upslope pasture or rangeland that traverses areas of row crops
 Poor crop stands
 Gully formation
 Poorly managed idle or wooded land
 Unstable stream banks
 Surface mining
 Feedlots close to a stream
 Long exposure of bare soil resulting from a land use

As was pointed out in the preceding section, sediment produced at the source is not qualitatively or quantitatively the same as sediment measured in the receiving body of water. Many factors, such as distance of the source from the receiving body of water, vegetative buffers, slope and roughness characteristics of the land, and ponding and the presence of depositional areas during overland flow, can affect the delivery of the sediment from the source to the receiving body of water.

SEDIMENT PROBLEM

The publication by Clark, Haverkamp, and Chapman (1985) is an authoritative treatise on the impact of erosion and sedimentation on the environment in general and water quality in particular. Soil erosion is the major cause of diffuse pollution and sediment is also the most visible pollutant. The effects of excessive sediment loading on receiving waters include the deterioration or destruction of aquatic habitat, deterioration of aesthetic value, loss of storage capacity in reservoirs, and accumulation of bottom deposits that inhibit normal biological life. Sediment can destroy spawning areas, food sources as well as directly harming fish, and other aquatic wildlife. Nutrients carried by the sediment can stimulate algal growths and, consequently, accelerate the process of eutrophication. Chapter 12 contains a more detailed discussion of the impact of sediment discharge on water quality.

Sediment per se can be considered a major pollutant of bodies of water. Furthermore, sediment—especially its fine fractions—is a primary carrier of such other pollutants as organic components, metals, ammonium ions, phosphates, and many organic toxic compounds. For example, persistent organochlorine compounds, such as aldrin and dieldrin pesticides, DDT, and PCBs, have low solubility in water but are readily adsorbed by suspended sediment. Phosphates from fertilizer applications and pollution discharges are also adsorbed by soils and suspended sediment. This strong affinity of fine sediments—primarily clay and organic particulates—to adsorb and make biologically unavailable the pollutants is considered by some as a partial water quality benefit of sediment discharges.

Large amounts of particulate materials (sediment) originate from urban areas. Urban sediment sources are known to contribute significant amounts of particulate pollutants containing toxic metals, organochlorine pesticides, asbestos, and other compounds originating from traffic, elevated air-pollution levels, urban litter accumulation, and erosion of pervious urban surfaces, which is accelerated by construction.

Many kinds of environmental damage by sediment are still weakly documented. However, in the United States the damage that has been documented amounts to billions of dollars. Annual in-stream damage by sediments is between \$3.2 and \$13 billion (Clark, Haverkamp, and Chapman, 1985). Recreational uses of bodies of water are most impacted by sediment.

In an extreme, reduction in the discharge of sediment may have a negative effect. For example, farmers in the delta region of the Nile River in Egypt relied on the deposited sediment brought from the Sudan and

Ethiopia for a source of fertilizing nutrients and silt. By building the Aswan High Dam in Upper Egypt, which acts as a sediment trap, significant damage has been done to delta farmers who now, at great expense and greater environmental consequences, must substitute chemical fertilizers. Similarly, building navigation dams on the Mississippi River has dramatically reduced sediment loads into the Mississippi Delta in Louisiana, which is suspected of causing shoreline and wetland losses by subsidence that otherwise would have been compensated for by new sediment brought by the river into the Delta region. The sediment, and especially its clay and organic fractions, can also adsorb and immobilize potentially toxic compounds and make them biologically unavailable. These toxic compounds may then be buried with the sediments (see Chapter 13).

ESTIMATING SEDIMENT YIELD

Sediment yield from a watershed can be estimated by one of the following methods (Gottschalk, 1964):

1. The stream-flow sampling method, which yields a sediment/flow relationship (Fig. 5.10) commonly called the rating curve. Typically, this relationship is plotted on logarithmic paper as the sediment concentration or sediment flow (concentration \times flow) versus discharge. Once the long-term sediment/flow relation is established, it is combined with a long-term flow frequency curve to obtain average annual yields.

A word of caution is needed for the users of standard sediment flow: water discharge curves. These correlations could easily be spurious and misleading since for large discharges and small concentrations the relationship

$$\log Q \text{ vs. } \log(Q \times C) = \log Q + \log C$$

where Q is flow and C is concentration, could easily become a logarithmic regression of Q vs. Q , which will yield very high apparent coefficients of correlation. Yet the relationship between C and Q may be very poor or not exist, so the estimate of the annual sediment discharge would essentially be the average sediment concentration times the average flow.

2. The reservoir sedimentation survey method, which allows sediment input to be determined if accumulated sediment and reservoir trap efficiencies are known and/or measured.

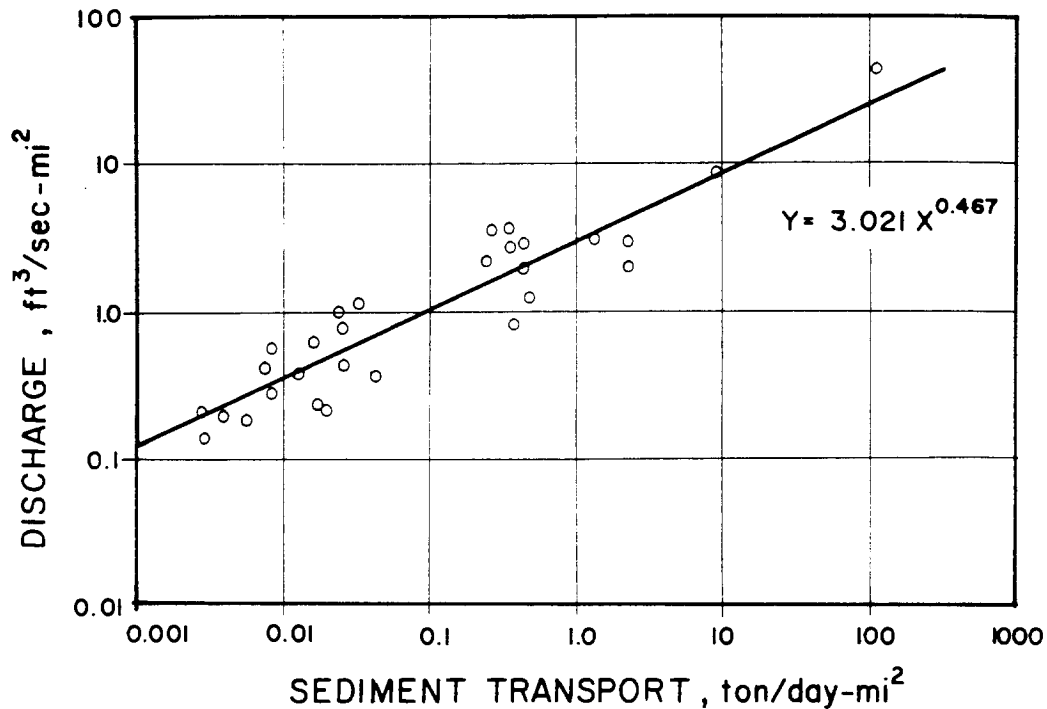


FIGURE 5.10. The relationship between suspended sediment transport and discharge for the Menomonee River in Wauwatosa, Wisconsin. (Compiled from the U.S. Geological Survey, Wisconsin Department of Natural Resources, and Southeastern Wisconsin Regional Planning Commission data. To convert from $\text{ft}^3/\text{sec}\text{-mi}^2$ to $\text{m}^3/\text{s}\text{-km}^2$, multiply by 0.011; to convert from U.S. $\text{ton}/\text{day}\text{-mi}^2$ to SI $\text{tonnes}/\text{day}\text{-km}^2$, multiply by 0.35.)

3. The sediment delivery method, in which sediment yield to some downstream section or deposition point is based on an estimate of the total upstream (upland) erosion factored down to account for the loss (or gain) of sediment during overland and channel transport. This method requires expressing the ratio between the sediment yield and gross upstream erosion, usually based on an empirical or semi-empirical formula. Both determination of the delivery ratio (DR) and upland gross erosion estimates are still unreliable if calibration and verification survey data are not available.
4. Bedload function methods use mathematical equations developed for calculating the rate and quantity of sediment materials in the bed portion of alluvial (sediment carrying and depositing) streams. Application of these equations requires information on sediment particle size, channel gradient and cross sections, and flow-duration curves. The equation can be used when sediment transport is not limited by the upstream supply, but depends solely on the transport capacity of

the flow. These models are mostly applicable to noncohesive, larger grain size sediments.

5. Methods using empirical equations relating sediment yield (directly measured by methods 1 or 2) to watershed hydrologic and/or morphologic characteristics. Most of these empirical equations have severely limited applications, even in the region of origin (Foster, Meyer, and Onstad, 1977).
6. Simulation watershed sediment load models, which usually are attached to a watershed hydrologic model. The watershed models are capable of simulating individual storm events or seasonal water and sediment yields. The hydrologic portion is necessary for determining so-called hydrologically active areas, that is, areas from which most intensive surface runoff and erosion occur. Chapter 9 includes an expanded discussion of mathematical modeling.

Estimating Upland Erosion

Variables influencing upland (sheet) erosion are climate, soil properties, vegetation, topography, and human activities. Rainfall, snowfall, and temperature are the primary climatic factors. Soil particles are detached and transported by the impact of rainfall energy, resulting in eroded materials being carried by surface runoff. Freezing temperatures and snow cover affect permeability and reduce the energy of precipitation. Conversely, spring rains occurring when subsoils are still frozen may cause high sediment yields due to reduced soil permeability.

The major soil properties related to erosion are soil texture and composition. Soil texture determines the permeability and erodibility of soils. Permeability and infiltration determine whether or not the soil surface is hydrologically active (that is, yielding or not yielding surface runoff). Erosion occurs only when surface runoff is generated or when wind picks up loose particles.

The chemical composition of soils (minerals, clay, and organic matter) provide bonding of soil particles, and thus affects erosion rates. Loose soils (silt and fine sand) with low chemical and clay content have the highest erodibility.

Vegetation influences sediment yields by dissipating rainfall energy, binding the soil, and increasing porosity by its root system, reducing soil moisture by evapotranspiration, thereby increasing infiltration. Organic residues also provide better texture and reduce erodibility.

The topographic factors of greatest importance are slope and the path length traversed by sediment generating flow. Human activities are related mostly to agricultural land use and construction.

Since many of the just-mentioned factors are seasonal and vary throughout the year, it follows that sediment yield and its chemical composition from a watershed also vary with time. Walling and Kane (1983) demonstrated that certain mineralogical characteristics of liberated sediment, such as Si, Al, Fe, Ti, and K, remain relatively constant over a range of hydrological conditions, while others, particularly those associated with organic fractions, may exhibit considerable variations.

Universal Soil Loss Equation

The universal soil loss equation (USLE) is the most common and best known estimator of soil loss caused by upland erosion. The equation and its development utilized more than 40 years of experimental field observations gathered by the Agricultural Research Service of the USDA. The USLE, formulated by Wischmeier and Smith (1965), predicts primarily sheet and rill erosion. The equation is

$$A = (R)(K)(LS)(C)(P) \quad (5.2)$$

where

A = calculated soil loss in tonnes/ha for a given storm or period

R = rainfall energy factor

K = soil erodibility factor

LS = slope-length factor

C = cropping management (vegetative cover) factor

P = erosion-control practice factor

The equation expresses soil loss/unit area due to erosion by rain. It does not include wind erosion and it does not give direct sediment yield estimates.

The *rainfall energy factor* (R_r) is equal to the sum of the rainfall erosion indices for all storms during the period of prediction. For a single storm it is defined as follows:

$$R_r = \Sigma[(2.29 + 1.15 \ln X_i)D_i]I \quad (5.3)$$

where

i = rainfall hyetograph time interval

D_i = rainfall during time interval i (cm)

I = maximum 30-min rainfall intensity of the storm (cm/hr)

X_i = rainfall intensity (cm/hr)

Average annual rainfall energy factors were determined for the eastern portions of the United States, and were later developed for the remainder

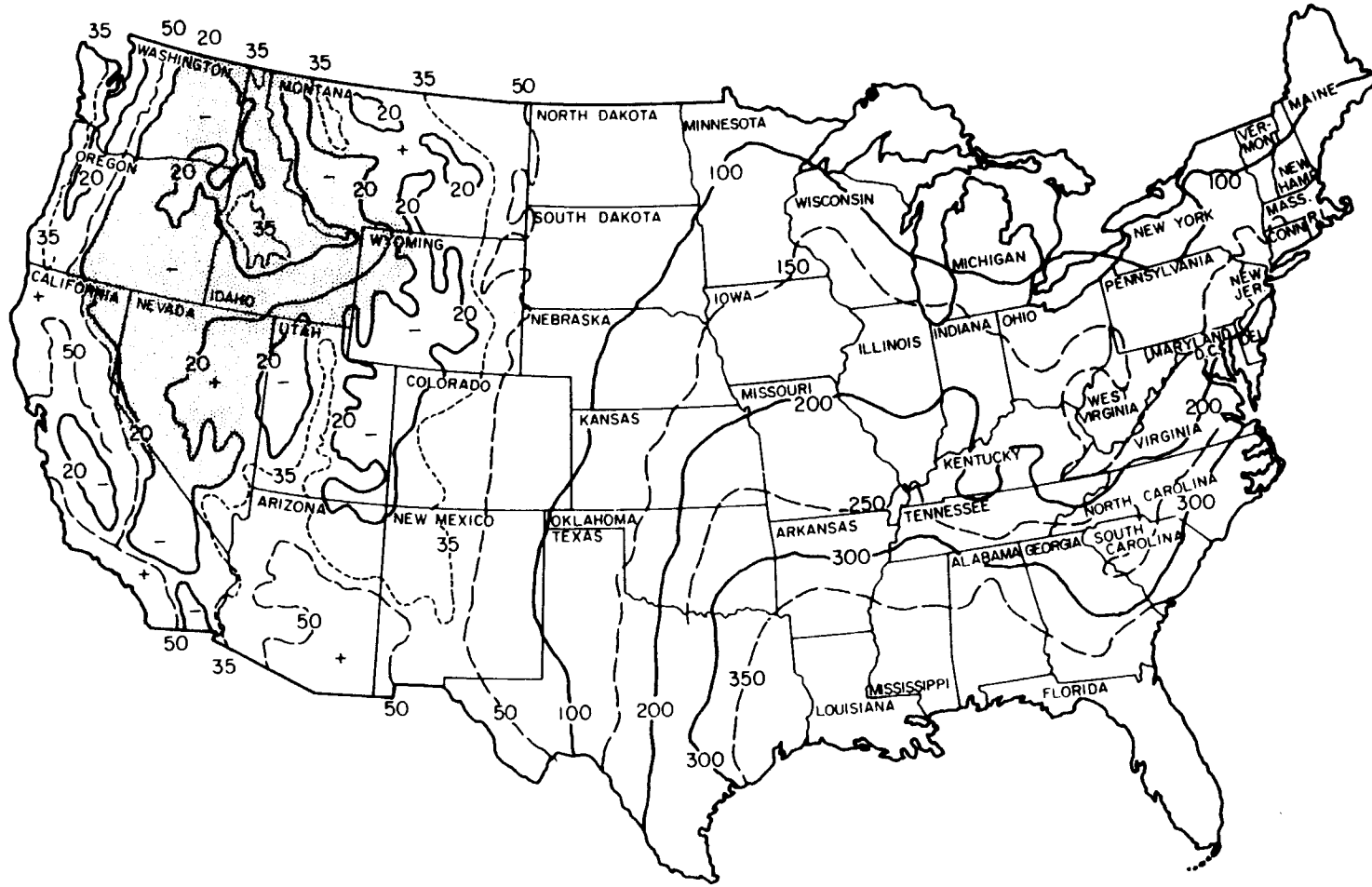


FIGURE 5.11. Values of the annual rainfall energy factor (R) in tons/acre. To convert to tonnes/ha, multiply by 2.24. (From Stewart et al., 1975.)

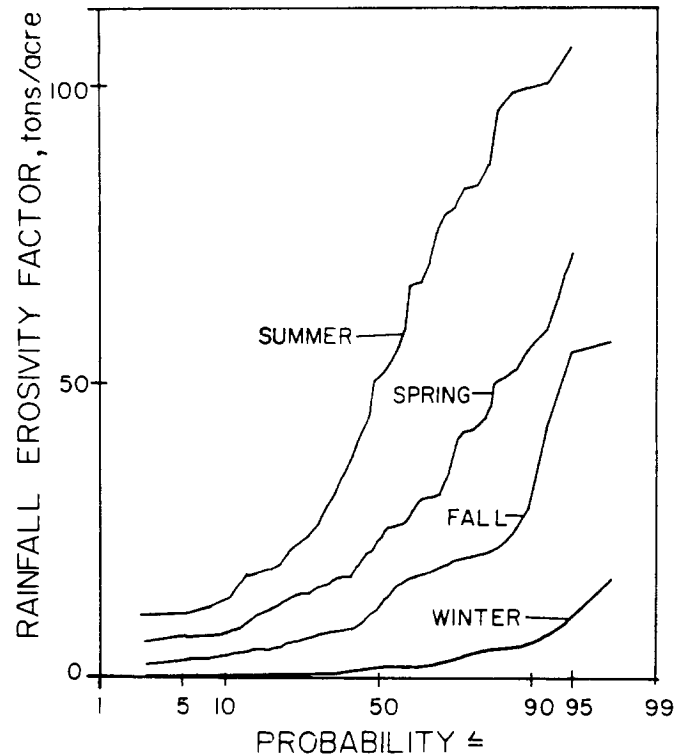


FIGURE 5.12. Seasonal cumulative frequency of the rainfall energy factor (R) for southeastern Wisconsin in tons/acre. To convert to tonnes/ha, multiply by 2.24.

of the country. The distribution of average R , factors for the 48 conterminous states are shown in Figure 5.11. These curves can help as first estimates of the gross erosion potential, but as shown on Figure 5.12, significant yearly and seasonal differences in the magnitudes of the rainfall energy factor R , may be typical. In the midwestern part of the United States, summer rains have the highest rainfall erosive energy, while the effect of winter precipitation on sediment yields is minimal.

The distribution of erosive rain differs significantly for different regions of the country. In the western plains and Great Lakes regions, from 40% to 50% of the erosive rainfall normally occurs during a 2-month period following spring planting when soils have the least protection. In most Corn Belt areas and the eastern parts of Kansas, Oklahoma, and Texas, the value is about 35%, while, for the lower Mississippi Valley and southeastern United States, the value is 20% to 23%. In the dry-land grain-growing region of the Pacific Northwest, about 80% to 90% of the annual erosion occurs in the winter months when the soil has little crop cover, since grain is seeded late in the fall (Steward et al., 1975).

Both erosion by rainfall energy (interrill erosion) and detachment of soil particles by overland runoff (rill erosion) contribute to soil loss (Foster, Lombardi, and Moldenhauer, 1982). Thus, the rainfall factor (R) should also include the effects of runoff. A modification of Equation (5.3) was proposed by Foster, Meyer, and Onstad (1977):

$$R = aR_r + bcQq^{1/3} \quad (5.4)$$

where

a and b = weighting parameters ($a + b = 1$)

c = an equality coefficient

Q = runoff volume (cm)

q = maximum runoff rate (cm/hr)

The weighting factor compares the relative amounts of erosion caused by rainfall and runoff under unit conditions. It was suggested that the detachment of particles by runoff and rain energy is about evenly divided ($a = b = 0.5$). The equality coefficient in metric (SI) units is about 15. Substituting values for a , b , and c into the USLE, the overall rainfall factor (R) becomes

$$R = 0.5R_r + 7.5Qq^{1/3} \quad (5.5)$$

However, the proportion between the rainfall and runoff erosivity may vary greatly between regions. Wischmeier (1976) pointed out that, for example, in the Palouse region of the Pacific Northwest, 90% of erosion is caused by runoff from thaw and snowmelt. A discussion of various functional forms of the rainfall erosivity factors—both rill and interrill effects—is included in Foster, Lombardi, and Moldenhauer (1982).

Williams and Berndt (1977) proposed a modified soil loss equation (MUSLE) that replaced the rainfall energy factor by a runoff energy parameter that is proportional to $(Q \times q)^{0.56}$. Equations (5.4), (5.5), and MUSLE of Williams and Berndt provide soil loss estimates in tonnes per storm (per area). Williams and Berndt pointed out that although q and Q are correlated, the runoff flow rate is more related to detachment, while the peak flow rate defines sediment transport. If flow is retarded by vegetation or other means, the peak flow rate is reduced, thus reducing sediment transport.

The soil erodibility factor (K) is a measure of potential erodibility of soil and has units of tonnes/unit of rainfall erosion index for a 22-m-long overland flow length on a 9% slope in clean-tilled continuous fallow

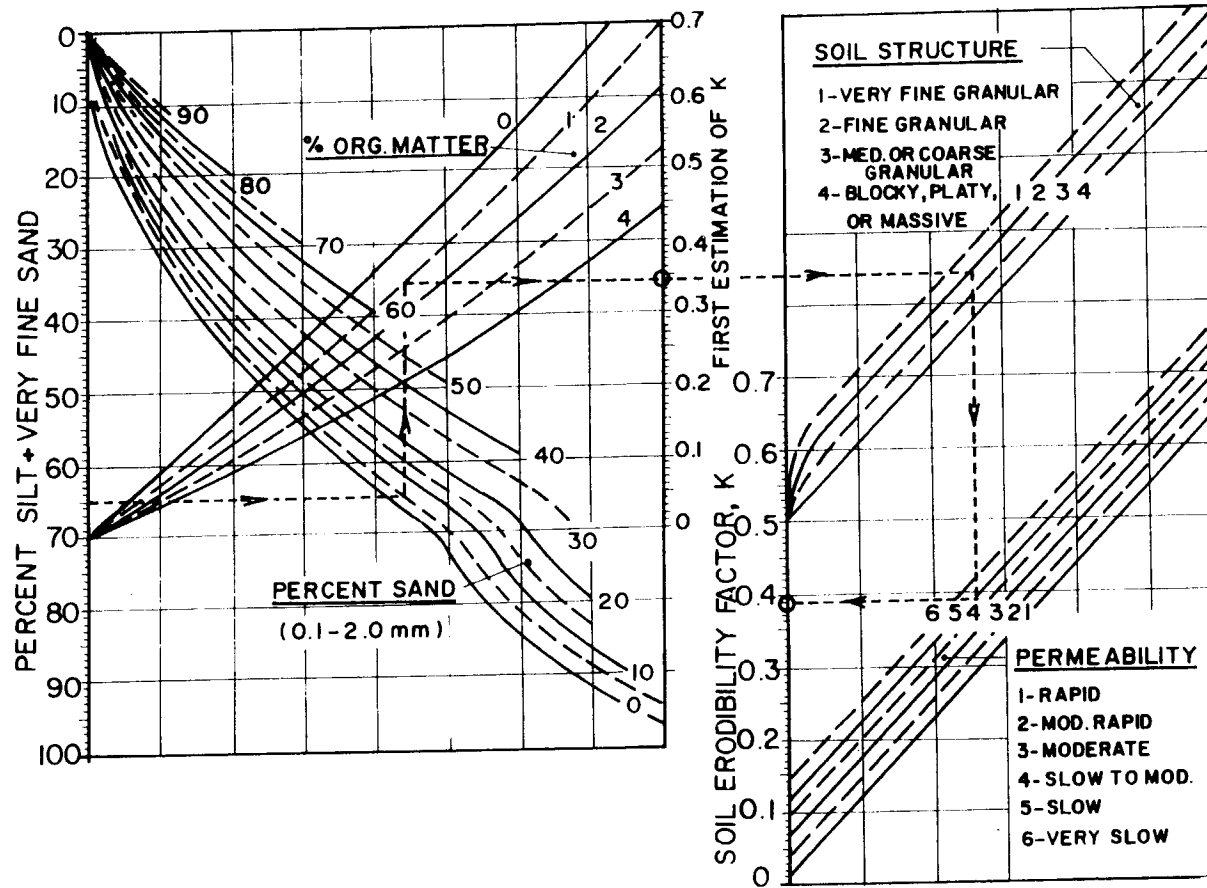


FIGURE 5.13. Soil erodibility nomograph for determining (K) factor for U.S. mainland soils. (Reprinted with permission from W. H. Wischmeier, C. B. Johnson, and B. V. Cross, *J. Soil Water Conserv.* 26:189-193. Copyright © 1971 by the Journal of Soil and Water Conservation.)

TABLE 5.3 Magnitude of Soil Erodibility Factor, K

Technical Class	K for Organic Matter Content (%)		
	0.5	2	4
Sand	0.05	0.03	0.02
Fine sand	0.16	0.14	0.10
Very fine sand	0.42	0.36	0.28
Loamy sand	0.12	0.10	0.16
Loamy fine sand	0.24	0.20	0.16
Loamy very fine sand	0.44	0.38	0.30
Sandy loam	0.27	0.24	0.19
Fine sandy loam	0.35	0.30	0.24
Very fine sandy loam	0.47	0.41	0.35
Loam	0.38	0.34	0.29
Silt loam	0.48	0.42	0.33
Silt	0.60	0.52	0.42
Sandy clay loam	0.27	0.25	0.21
Clay loam	0.28	0.25	0.21
Silty clay loam	0.37	0.32	0.26
Sandy clay	0.14	0.13	0.12
Silty clay	0.25	0.23	0.19
Clay		0.13–0.2	

Source: After Steward et al. (1975).

Note: The values shown are the estimated averages of broad ranges of specific soil values. When a texture is near the borderline of two texture classes, use the average of the two K values.

ground (Wischmeier and Smith, 1965; Wischmeier, 1976). It is a function of soil texture and composition. The soil erodibility nomograph shown on Figure 5.13 is used to find the appropriate values of the soil erodibility factor using five parameters: percent silt and fine sand, that is, 0.05 to 0.1 mm fractions; percent sand >0.1 mm; percent organic matter; textural class; and permeability. The general magnitudes of the soil erodibility factors are given in Table 5.3.

The slope-length factor (*LS*) is a function of overland runoff length and slope. It is a dimensionless parameter that adjusts the soil loss estimates for the effects of length and the steepness of the field slope. The general magnitudes of the LS factor are given in Figure 5.14. For slopes >4%, the LS factor can be estimated as follows:

$$LS = L^{1/2}(0.0138 + 0.00974S + 0.00138S^2) \quad (5.6)$$

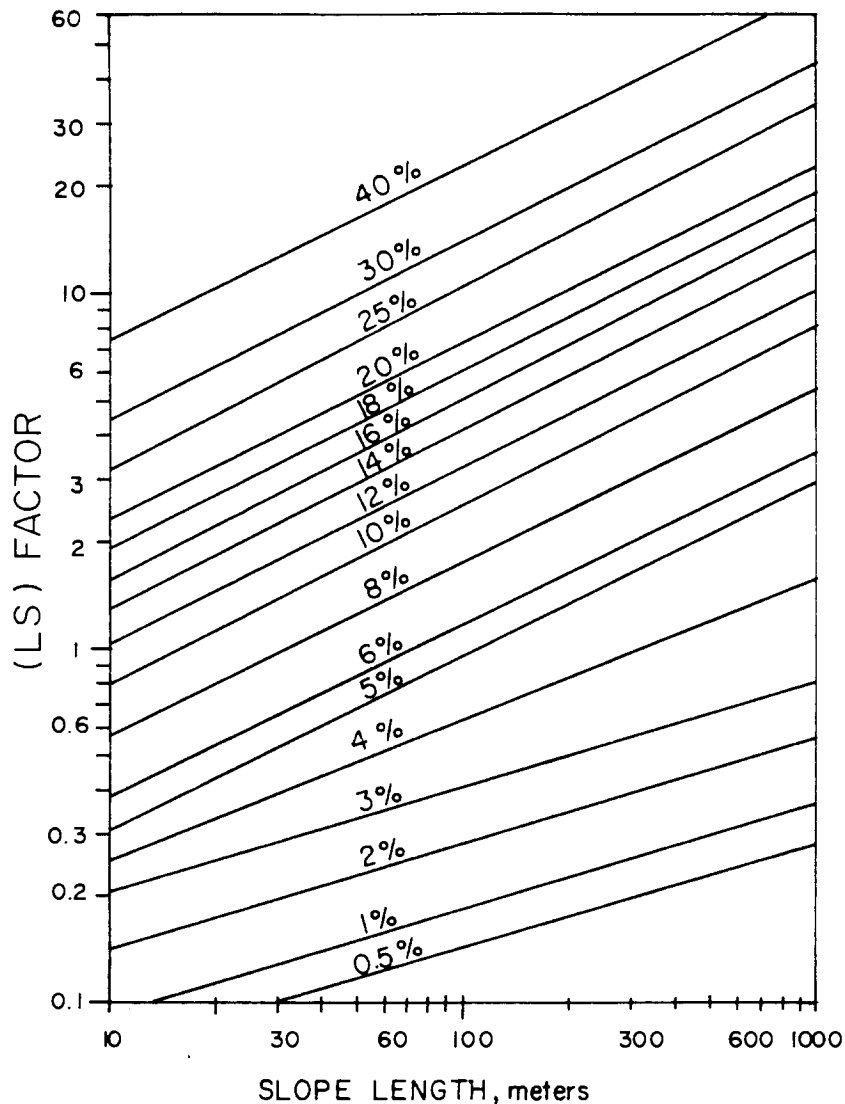


FIGURE 5.14. Slope-length factor (LS) for different slopes.
(From Stewart et al., 1975.)

where

L = length in meters from the point of origin of the overland flow to the point where the slope decreases to the extent that deposition begins or to the point at which runoff enters a defined channel

S = the average slope (%) over the runoff length

Values of the LS factor estimated for length >100 meters or slopes >18% are extrapolated beyond the experimental data from which the magnitudes of the factor was determined.

If the average slope is used in calculating the LS factor, predicted

erosion differs from actual erosion when the slope is not uniform. The equation for LS factors shows that when the actual slope is convex, the average slope will underestimate predicted erosion. Conversely, for a concave slope, the equation will overestimate actual erosion. To minimize these errors, large areas should be broken up into areas of fairly uniform slope. If sediment moves from an area with steep slope to an area of less steep slope, the smaller LS factor will control the amount eroded and the excess sediment is likely to be deposited.

The cropping management factor (C), also called the vegetation cover factor, estimates the effect of ground cover conditions, soil conditions, and general management practices on erosion rates. It is a dimensionless quantity with a value of one corresponding to continuous fallow ground, which has been defined as land that has been tilled up and down the slope and maintained free of vegetation and surface crusting. The effect of vegetation on erosion rates results from canopy protection, reduction of rainfall energy, and protection of soil by plant residues, roots, and mulches.

TABLE 5.4 Values of C for Cropland, Pasture, and Woodland

Land Cover or Land Use	C
Continuous fallow tilled up and down slope	1.0
Shortly after seeding prior to harvesting	0.3–0.8
For crops during main part of growing season	
Corn	0.1–0.3
Wheat	0.05–0.15
Cotton	0.4
Soybean	0.2–0.3
Meadow	0.01–0.02
For permanent pasture, idle land, unmanaged woodland	
Ground cover 85%–100%	
As grass	0.003
As weeds	0.01
Ground cover 80%	
As grass	0.01
As weeds	0.04
Ground cover 60%	
As grass	0.04
As weeds	0.09
For managed woodland	
Tree canopy of 75%–100%	0.001
40%–75%	0.002–0.004
20%–40%	0.003–0.01

Sources: Based on data from Stewart et al. (1975); Wischmeier and Smith (1965), and Wischmeier (1972).

TABLE 5.5 C-Values and Slope-Length Limits (LS) for Construction Sites

Mulch				
Type	Application (tonnes/ha)	Slope (%)	C	LS
No mulch or seeding		All	1.0	
Straw or hay tied down by anchoring and tracking equipment used on slope	2.25	<5	0.2	60
	2.25	6-10	0.2	30
	3.4	<5	0.12	90
	3.4	6-10	0.12	45
	4.5	<5	0.06	100
	4.5	6-10	0.06	60
	4.5	11-15	0.07	45
	4.5	16-20	0.11	30
	4.5	21-25	0.14	23
Crushed stone	300	<15	0.05	60
	300	16-20	0.05	45
	300	21-33	0.05	30
	540	<20	0.02	90
	540	21-35	0.02	60
Wood chips	15	<15	0.08	23
	15	16-20	0.08	15
	27	<15	0.05	45
	27	16-20	0.05	23
	56	<15	0.02	60
	56	16-20	0.02	45
	56	21-33	0.02	30
Asphalt emulsion 12 m ³ /ha			0.03	
Temporary seeding with grain or fast-growing grass with			During first 6 weeks of growth	After the 6th week of growth
	No mulch		0.70	0.10
	Straw	2.25	0.20	0.07
	Straw	3.4	0.12	0.05
	Stone	300	0.05	0.05
	Stone	540	0.02	0.02
	Wood chips	15	0.08	0.05
	Wood chips	27	0.05	0.02
	Wood chips	56	0.02	0.02
	Sod		0.01	0.01

Source: After Ports (1975).

Table 5.4 shows the general magnitudes of C for agricultural land, permanent pasture, and idle rural land. Grassed urban areas have C factors similar to those for permanent pasture. The C factor for construction sites can be reduced if the surface is protected by seeding or the application of hay, asphalt, wood chips, or other protective covers. The effects of these protective practices on C are given in Table 5.5.

The erosion control practice factor (P) accounts for the erosion-control effectiveness of such land treatment as contouring, compacting, established sedimentation basins, and other control structures. Terracing does not affect P because the soil loss reduction by terracing is reflected in the value of LS . Generally, C reflects protection of the soil surface against the impact of rain droplets and subsequent loss of soil particles. On the other hand, P involves treatments that retain liberated particles near the source and prevent further transport.

Values of P for various farm and urban practices are given in Tables 5.6 and 5.7, respectively. It should be pointed out that these coefficients are highly empirical and may be used only as a first approximation. More accurate models are available for several practices, included in the P factor, such as models for the removal efficiency of sedimentation ponds and buffer strips. These concepts are discussed in Chapters 9 and 10, respectively.

Reliability of the USLE. The universal soil loss equation was subjected to a lot of testing and criticizing; however, it withstood the test of time and today it is the only widespread and tested model. It has been used in many applications. The author of the equation (Wischmeier, 1976) reported the results of testing on the reliability of the equation. He pointed out that when the USLE is used for estimating the annual soil loss on many experimental testing plots, the average prediction error (coefficient of variation = deviation/mean estimate) was about 12%. Larger errors

TABLE 5.6 Values of P for Agricultural Lands

Slope (percent) Crops	Contouring	Strip Cropping and Terracing	
		Alternate Meadows	Closegrown
0-2.0	0.6	0.3	0.45
2.1-7.0	0.5	0.25	0.40
7.1-12.0	0.6	0.30	0.45
12.1-18.0	0.8	0.40	0.60
18.1-24.0	0.9	0.45	0.70
>24		-1.0-	

Source: After Wischmeier and Smith (1965).

TABLE 5.7 Values of P for Construction Sites

Erosion Control Practice	P
<i>Surface Condition with No Cover</i>	
Compact, smooth, scraped with bulldozer or scraper up and down hill	1.30
Same as above, except raked with bulldozer and root-raked up and down hill	1.20
Compact, smooth, scraped with bulldozer or scraper across the slope	1.20
Same as above, except raked with bulldozer and root raked across the slope	0.90
Loose as a disked plow layer	1.00
Rough irregular surface, equipment tracks in all directions	0.90
Loose with rough surface >0.3-m depth	0.80
Loose with smooth surface <0.3-m depth	0.90
<i>Structures</i>	
Small sediment basins	
0.09 ha basin/ha	0.50
0.13 ha basin/ha	0.30
Downstream sediment basin	
With chemical flocculants	0.10
Without chemical flocculants	0.20
Erosion-control structures	
Normal rate usage	0.50
High rate usage	0.40
Strip building	0.75

Source: After Ports (1973).

should be expected if the equation is used for predicting the soil loss of individual storms.

The accuracy of the model is increased if it is combined with a hydrological excess-rainfall model. Note that the rainfall erosivity factor, R , has a value greater than zero for every rainfall, hence, erosion and soil loss are anticipated by the soil loss equation for any precipitation. A hydrological excess-rainfall model in combination with the USLE would eliminate erosion by rainfalls with no excess rain.

The USLE was tested and specifically designed for the following applications (Wischmeier, 1976):

1. Predicting average annual soil movement from a given field slope under specified land-use and management conditions and from construction, rangeland, woodland, and recreational areas.
2. Guiding the selection of conservation practices for specific sites.
3. Estimating the reduction of soil loss attainable from various changes that a farmer might make in his cropping system or cultural practices.

4. Determining how much more intensively a given field could be safely cropped if contoured, terraced, or stripcropped.
5. Determining the maximum slope-length on which given cropping and management can be tolerated in a field.
6. Providing local soil loss data to agricultural technicians, conservation agencies, and others to use when discussing erosion-control plans with farmers and contractors.

The USLE will not provide direct estimates of the sediment yield and cannot be used for calculations of soil losses from spring snowmelt.

Example 5.1: Estimation of Annual Soil Loss

An erosive 100-ha farm field in southeast Wisconsin is situated on silt loam soil with a slope classification B (3% to 6% slope). The farmer is growing corn and plowing up and down the slope. Estimate the average annual soil loss per hectare without soil conservation and with contour plowing. The field has a square shape with a drainage ditch located on the side of the field. The overland slope is toward the drainage ditch.

Solution From Figure 5.10 the average annual rainfall erosivity for southeast Wisconsin is $R_r = 125$ U.S. tons/acre $\times 2.24 = 280$ tonnes/ha. From Table 5.3 the average soil erodibility factor for silt loam soil is $K = 0.42$. The slope-length factor (LS) can be read from Figure 5.14 (overland flow length $L = 1000$ m, and average slope $S = 45\%$) as $LS = 1$.

The plowing practice is to till up and down the slope of the continuous fallow field. Consequently the slope has a C factor of $C = 1$ after plowing and 0.1 to 0.3 for corn during the main growing season (Table 5.4), respectively. Average C for no soil conservation planting is assumed to be $C = (1 + 0.3)/2 = 0.65$. Since no erosion control is implemented, then $P = 1$. The average annual soil loss without soil conservation is then

$$A = (R)(K)(LS)(C)(P) = 280 \times 0.42 \times 1 \times 0.65 \times 1 \\ = 76.4 \text{ tonnes/ha}$$

Implementing contour plowing will reduce the P factor to 0.5 (Table 5.7). Hence the soil loss will then be

$$A = 280 \times 0.42 \times 1 \times 0.65 \times 0.5 = 38.2 \text{ tonnes/ha}$$

Example 5.2: Soil Loss from a Construction Area for a Design Storm

A 50-ha land area is to be developed into a single family residential area. The soil map indicates that the soil is loam with the following composition:

- Clay 20%
- Silt 35%
- Fine sand 20%
- (Silt + fine sand) 55%
- Coarse sand and gravel 25%

The organic content of the soil is 1.5%.

The lot has a square shape with a drainage ditch in the center. It has been proposed to replace the ditch with a storm sewer. The average slope of the lot toward the ditch is 2.4%.

Determine soil loss (potential erosion) for a storm for which the hyetograph is given in Figure 5.15. Soil loss should be determined from the pervious areas for the two periods, namely, during construction when all vegetation is stripped from the soil surface (100% pervious) and subsequent to construction when 25% of the area is impermeable (streets, roofs, driveways, etc.).

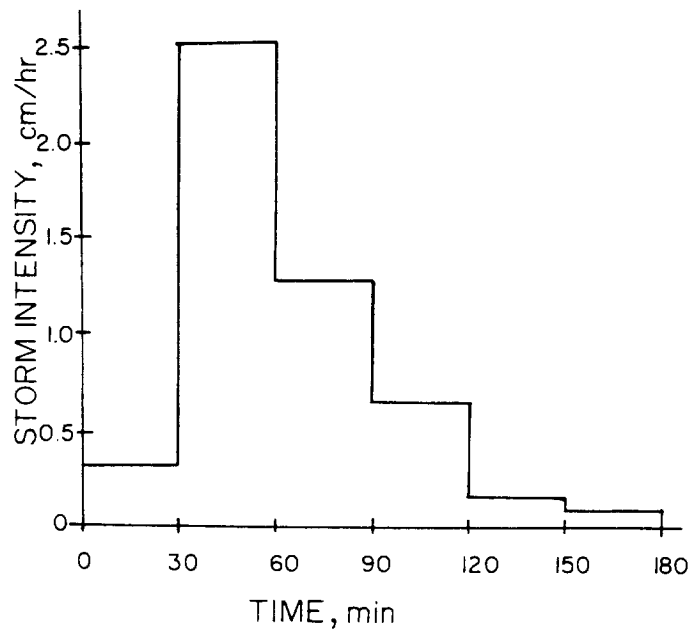


FIGURE 5.15. Storm hyetograph for Example 5.2.

Solution The rainfall energy factor R_r is determined from the hyetograph shown on Figure 5.15. From this information it can be determined that the maximum 30-min rainfall intensity is 2.5 cm/hr.

Utilizing Equation (5.4)

$$\begin{aligned} R_r = & [(2.29 + 1.15 \ln 0.3)0.15 \\ & + (2.29 + 1.15 \ln 2.5)1.25 \\ & + (2.29 + 1.15 \ln 1.25)0.6175 \\ & + (2.29 + 1.15 \ln 0.7)0.35 \\ & + (2.29 + 1.15 \ln 0.2)0.1 \\ & + (2.29 + 1.15 \ln 0.1)0.05]2.5 = 16.4 \end{aligned}$$

The soil erodibility factor is determined from Figure 5.13, assuming that the soil texture is fine grained and the permeability is moderate, giving a K value of 0.33.

To determine the LS factor for a 50-ha area with a ditch or storm sewer in the middle, the length of the overland flow $L = 0.5\sqrt{50 \times 100 \times 100} = 353.5$ m. With the use of Figure 5.14 or Equation (5.6), the LS factor for $L = 353$ and $S = 2.4\%$ becomes

$$LS = (353.5)^{1/2}[0.138 + (0.00974 \times 2.4) + (0.00138 \times 2.4^2)] = 0.47$$

Factors R_r , K , and LS are the same for both alternatives. The remaining factors, C and P must be evaluated for each alternative (P only if erosion-control practices are implemented during construction). For the period during construction (alternative 1), C is estimated assuming no vegetative protective cover and bulldozed soil. In this case, C is approximately the same as for bare fallow ground, that is, $C = 1$. In the absence of erosion-control practices, $P = 1$. Thus soil loss for this storm is

$$A = 16.4 \times 0.33 \times 0.47 \times 1 \times 1 = 2.54 \text{ tonnes/ha}$$

Thus, for 50 ha, total soil loss from the storm is

$$50 \times 2.54 = 127.3 \text{ tonnes.}$$

For the period after construction (alternative 2) and assuming that the pervious areas are covered by lawns, C is reduced to 0.01 and the soil loss/ha is

$$A = 16.4 \times 0.33 \times 0.47 \times 0.01 \times 1 = 0.025 \text{ tonnes/ha.}$$

Given that 75% of the area is subject to soil loss, the total sediment generation from pervious areas is

$$0.75 \times 50 \times 0.025 = 0.93 \text{ tonnes}$$

In order to complete the analysis, sediment generated from connected impervious areas (street dust and dirt—see Chapters 8 and 9) would have to be added to the amount just given. This estimate is also subjected to the condition that the storm will generate appreciable surface runoff.

Modifications of the USLE

The USDA-Agricultural Research Service has improved and simplified the annual form of the USLE by expanding the erosivity map for the western United States, improving information on soil erodibility factor, and by simplifying the slope-length factor (Renard et al., 1991).

For the CREAMS-GLEAMS model (Knisel, 1980), Foster et al. (1981) developed an erosion detachment–redemption model that is derived from the steady-state mass balance equation of the sediment detachment and transfer by overland flow. The sediment movement downslope obeys continuity expressed by

$$\frac{dq_s}{dt} = D_L + D_F \quad (5.7)$$

where

- q_s = sediment load (mass/unit width-time)
- D_L = lateral inflow of sediment (mass/unit area-time)
- D_F = detachment or deposition by flow

Detachment on interrill areas and transport and deposition by flow in rills are then modeled by the following equations:

$$D_L = \beta_1 EI(q_p/Q)(s_o + 0.014)KCP \quad (5.8a)$$

and

$$D_F = \beta_2 \eta Q q_p^{1/3} (/22.1)^{\eta-1} (q_p/Q)KCP \quad (5.8b)$$

where

- D_L = interrill detachment rate (g/m^2 of land surface-sec)
- D_F = capacity rate for rill detachment (g/m^2 of land surface-sec)

- EI = rainfall erosivity factor (defined by Eq. (5.3))
 x = distance downslope (meters)
 η = slope length exponent for rill erosion
 s_o = slope
 Q = overland flow (volume/unit area-unit time = m/sec)
 q_p = peak runoff rate (m/sec)
 K, C, P = factors defined previously

β_1 and β_2 are numerical conversion factors depending on the selected units.

The CREAMS-GLEAMS erosion modeling concept, based on the steady-state mass balance of sediment transport, detachment, and re-deposition in rills and interrills areas was further improved and modified and subsequently incorporated in the Water Erosion Prediction (WEPP) hillside erosion model (Nearing et al., 1989).

The *Negev sediment generation model* (Negev, 1967) simulates the generation and transport of soil by raindrop impact and overland flow. The production of fine soil particles by raindrop splash is determined for each computational interval and unit area as

$$A(t) = (1 - \text{COV}) \times K_N \times P(t)^{RER} \quad (5.9)$$

where

- A = fine soil particles produced during time interval i
 COV = fraction of vegetative cover as a function of the relative value during the growing season
 K_N = the coefficient of soil properties
 P = precipitation during the time interval Δt
 RER = an exponent

The model represents the production of fine soil particles, that is, silt and clay fractions, of the total watershed.

The fine particles produced by raindrop impact are available for transport by overland flow if it occurs during the time interval. If overland flow does not occur, for example, during the initial or final stages of the storm, the fine particles accumulate at the soil surface and represent a reservoir of liberated fine material available for transport by subsequent overland flow. The mechanism is modeled by

$$SER(t) = KSER \times SRER(t - 1) \times ROSB(t)^{SR} \quad (5.10)$$

where

- SER* = fine particles transported during time interval Δt
KSER = coefficient of transport
SRER($t - 1$) = reservoir of deposited fine particles existing at the beginning of the time interval Δt
ROSB = the overland flow occurring during time interval Δt
SR = an exponent

The Negev model was incorporated into the Stanford Watershed Model, and subsequently into the EPA's Hydrocomp Simulation Program (HSPF). This model requires knowledge of several empirical factors and coefficients. Unlike the USLE, the model has not been substantiated by experimental field data and measurements, and the coefficients must be estimated by calibrating the model against an extensive set of field data. Possible conversions between the USLE and parameters of the Negev model have been suggested by Fleming and Leytham (1976).

SEDIMENT DELIVERY AND ENRICHMENT PROCESSES DURING OVERLAND FLOW

As stated previously in this chapter sediment yield measured at a watershed outlet or a point on the water's course is not equal to the upland erosion. The state of the art for estimating delivery ratios has not progressed much beyond the long established empirical relationships, and available lumped models are still inaccurate.

As pointed out by Wischmeier (1976) the universal soil loss equation was developed from measurements on small plots with an average length of 22 meters. Extrapolation of erosion estimates by the USLE beyond this scale was not substantiated by large-scale field data and was not recommended by the authors of the equation. However, in the nonpoint pollution studies of the 1970s nonpoint pollution was equated with soil loss and the delivery ratio parameter obtained by calibration of the watershed models containing the USLE with measured field data, was used to overcome the differences between the upland erosion estimates and measured sediment yields. In this sense the delivery ratio factor became a scaling parameter by which the erosion estimate was adjusted to the measured sediment yields. Hadley and Shown (1976) have documented that only a fraction (about 30%) of the eroded sediment in several tributary basins (0.5 to 5.2 km²) of the Ryan Gulch Basin in northwest Colorado found its way to the main basin and, subsequently, only 30% was transported to the mouth of the 125-km² basin. A nationwide study of 105 agricultural watersheds in the United States un-

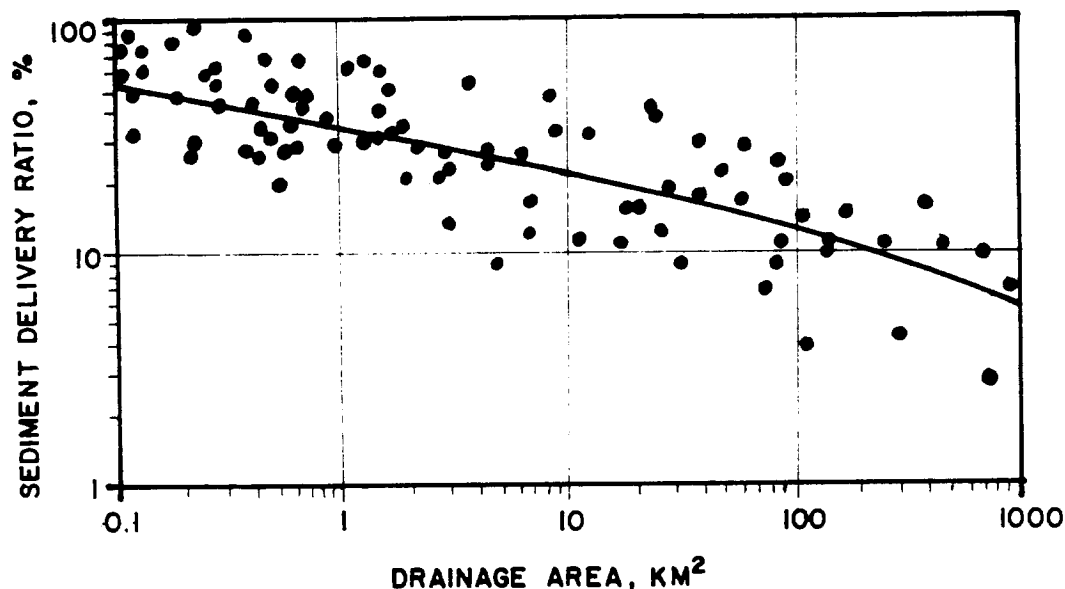


FIGURE 5.16. Relation of sediment delivery ratio to the watershed area. (From Roehl, 1962.)

dertaken by Wade and Heany (1978) reported sediment delivery ratios ranging from 1% to 38% of the gross erosion.

Golubev (1982) provided estimates on the components of the overall delivery process as delineated on Figure 5.9. In an investigation in the Oka River Basin in the central European part of the Russian Republic it was found that only 10% of the gross erosion is transported to the larger rivers and that 60% is deposited on the lower parts of the slopes, 20% in ephemeral channels, and 10% in minor streams of the network.

Other geomorphologists and sedimentologists provided answers on the magnitude of the delivery ratio related to watershed characteristics. The most widely published was a relationship established by Roehl (1962) that relates the delivery ratio to the watershed size (Fig. 5.16). A statistical equation also related the delivery ratio to several geomorphological parameters, such as basin area, relief-to-length ratio, and bifurcation ratio. These relationships were obtained by comparing quantities of sediment deposited in a number of southern reservoirs with upstream erosion potential.

However, Beer, Farnham, and Heinemann (1966) found that for reservoirs draining loessial watersheds located in Iowa and Missouri the drainage area and delivery ratio were poorly correlated. In another geomorphological approach, Hadley and Shown (1976) related the delivery ratio to the degree of gullying, drainage density and soil texture (McElroy et al., 1976), and watershed size and runoff (Mutchler and

Bowie, 1976), while for construction sites the delivery ratio was related to the area under construction (Holberger and Truett, 1976). Williams (1975) related the delivery ratio to the travel time of the sediment in the channels and particle size of the sediment.

Wolman (1977) stated that

The relationship between quantities eroded from the land surface and the quantities delivered to some distance downstream is exceedingly tenuous. The sediment delivery ratio provides a cover for real physical storage processes as well as for errors in estimates of the amount eroded and for temporal discontinuities of the process.

Walling (1983) divided the various approaches to the sediment delivery problem into several types, including a black box approach, by which sediment yield is correlated to upland erosion without considering the processes involved, delivery estimates based on channel characteristics, and estimates based on watershed characteristics. He argued that spatial and temporal lumping (or averaging) of the parameter limits most delivery estimates. As stated by Wolman (1977), the concept of the delivery ratio, as developed by geomorphologists, is indeed a cover for a number of processes that contribute to temporal or permanent deposition of sediments in an eroding watershed. These processes are highly variable and intermittent, and can be described only in the statistical sense. Novotny and Chesters (1989) followed with a comprehensive review of the sediment and pollutant delivery process and of the factors affecting the dynamic magnitude of the delivery parameter as related to water quality.

Factors Affecting Sediment Delivery

Any hydrologic process, such as delivery of sediments or pollutants from diffuse sources, is stochastic (Fig. 5.17). A stochastic process has deterministic and random components. The stochasticity and randomness of the process obviously vary. Flow in a uniform, open channel or runoff formation from a small parking lot are mostly deterministic processes, while a time series of annual rainfalls or stream flows is mostly random with some degree of autocorrelation. A delivery process falls somewhere in between (Fig. 5.18). For example, the correlation of the delivery ratio with runoff coefficients indicates a marked deterministic effect of infiltration and such losses as hydrologic storage and evaporation on the magnitude of the delivery ratio. Yet the scatter of the data around the line of the best fit might indicate a random component. Also, the statistical cumulative probability distribution of the delivery ratio is close to log-normal, which is the most common distribution for several hydrologic processes (Fig. 5.19).

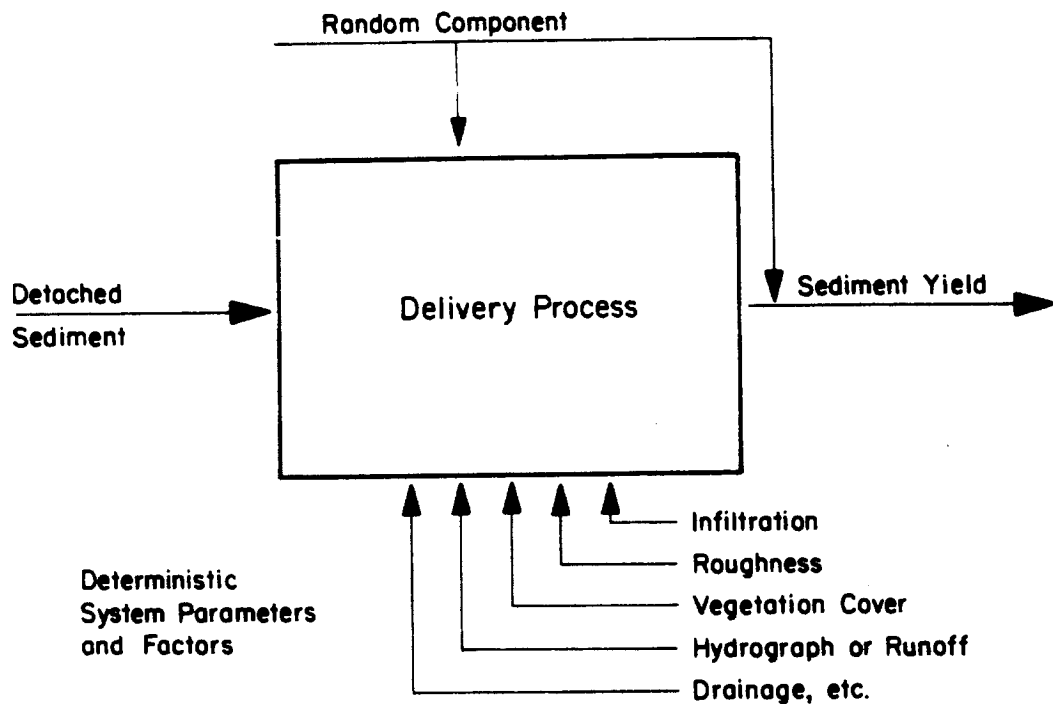


FIGURE 5.17. Stochastic representation of the delivery ratio process.

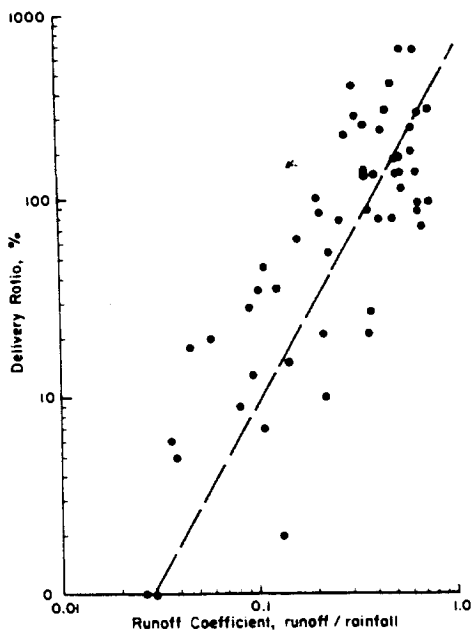


FIGURE 5.18. Statistical relationship of delivery ratio to runoff coefficient. (Data from Piest, Kramer, and Heinemann, 1975.)

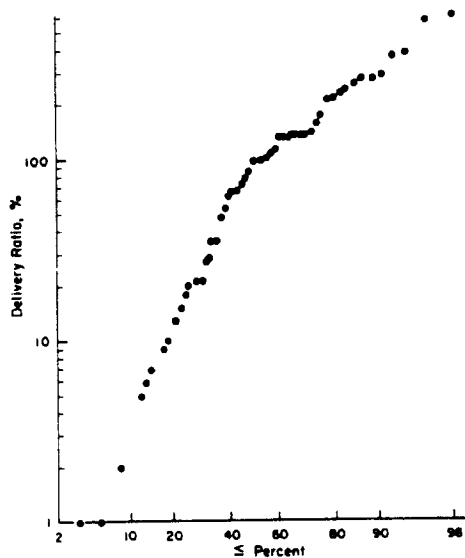


FIGURE 5.19. Statistical cumulative probabilistic distribution of delivery ratio for a loessial watershed in Iowa. (Data from Piest, Kramer, and Heinemann, 1975.)

The hydraulic (deterministic) factors affecting delivery can be divided into two categories: those that affect sediment and pollutant movement during overland flow, and those that affect sediment and pollutant movement during channel flow, including movement and deposition in floodplain.

Overland flow effects. There are six major overland flow factors and processes that affect delivery (Novotny, 1980; Novotny, Simsim, and Chesters, 1986; Novotny and Chesters, 1989).

- *Rainfall impact* detaches soil particles and keeps them in suspension for the duration of the storm. When the energy from rainfall ends or is reduced, the excess particles in suspension are deposited.
- *Overland flow energy* detaches soil particles from small rills and together with some interrill contribution, the particles remain in suspension as long as overland flow persists. Sediment content of overland flow is at or near saturation, and the sediment carrying capacity of overland flow is directly proportional to the amount of flow. If the flow is reduced during the receding portion of the hydrograph, excess sediment is deposited.
- *Vegetation* slows flow and filters out particles during shallow-flow conditions. If the vegetation is not submerged, the flow is mostly laminar, hence its sediment carrying capacity is very low.
- *Infiltration* filters out the particles from the overland flow.
- *Small depressions and ponding* allow particles to be deposited because of reduced flow velocity.
- *Change of slope of overland flow* because of drainage area concavity

often flattens the slope near the drainage channel and steepens the slope uphill. The sediment and pollutant transport capacity of overland flow changes according to slope.

The principal relationship between the detachment, deposition, and sediment carrying capacity of overland flow derived from the steady-state sediment transport mass balance concept (Eq. (5.7)) was defined by Foster and Meyer (1972a) as follows:

$$\frac{D_F}{D_c} + \frac{G_F}{T_c} = 1 \quad (5.11)$$

where

D_F = detachment or deposition rate at a location (mass/unit area/time)

C = coefficient relating rate of deposition or detachment to the difference between transport capacity and sediment load

G_F = sediment load of flow at any location on the slope (mass/unit width/time)

T_c = flow transport capacity at a location on a given land profile (wt/unit width/time)

D_c = the detachment capacity of flow (mass/unit area/time)

Equation (5.11) represents the basis for understanding the process of attenuation and detachment of sediment and particulate pollutants from nonpoint sources. If D_c and T_c are similarly related to a function involving the shear stress of the overland flow and other parameters (that is, if $D_c \approx \alpha T_c$), the detachment or deposition rate defined in Equation (5.11) becomes

$$D_F = \alpha(T_c - G_F) \quad (5.12)$$

where α = a coefficient.

Equation (5.12) essentially states that when the sediment load reaches the transport capacity of flow, there is no detachment, and if G_F by some way becomes greater than T_c , D_F becomes negative (deposition). Consequently if $G_F < T_c$, the flow is erosive.

As shown by Shen (1982) and illustrated in Figure 5.20, the concept can be generalized. Shen called the runoff rate at which no deposition or erosion takes place (at which $T_c = G_F$) "a critical runoff rate." As is shown later, these concepts are applicable primarily to noncohesive sediments. For cohesive sediments and soils the critical flow rates for detachment and deposition may not coincide.

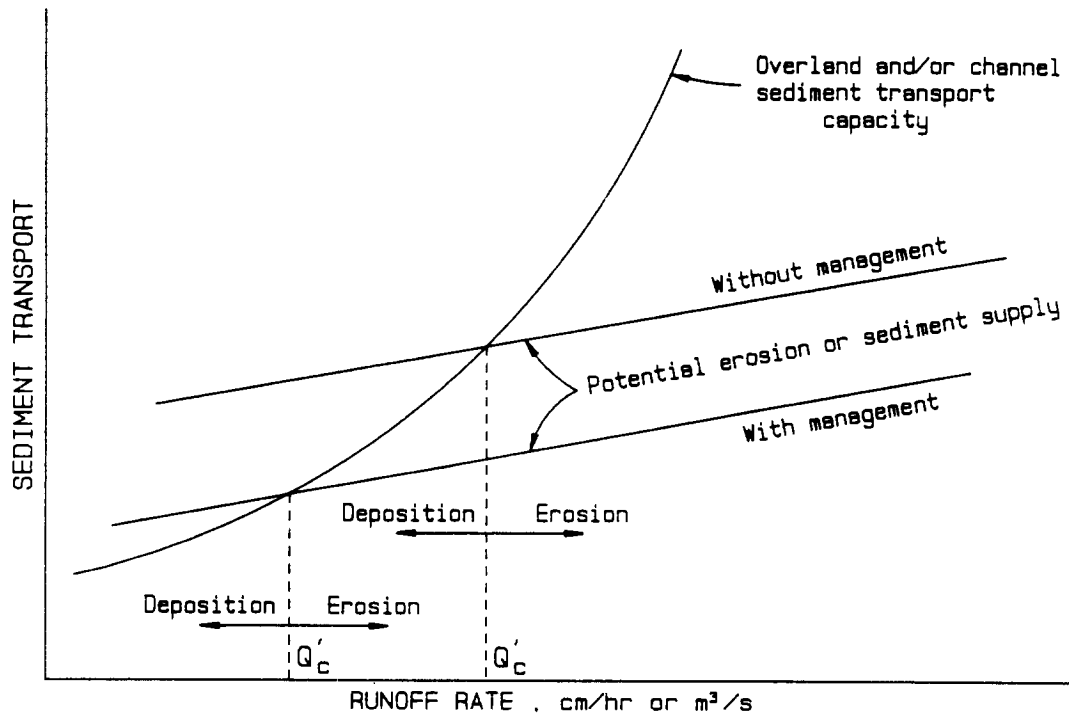


FIGURE 5.20. Shen's (1982) concept of the sediment carrying capacity of flow. (Reprinted with permission from *Water Resources Publications*, POB 260026, Littleton, CO 80126-0026.)

Figure 5.20 and Equation (5.11) point out an interesting fact on the impact of sediment load reduction in a watershed. When the erosion rate is reduced by land management, resulting in a reduced supply of sediment, the erodibility of runoff downstream increases, thus, to some degree, negating the erosion management effort. This conclusion would be only true to the fullest extent for noncohesive sediment, and its applicability to cohesive sediments and to pollutions carried by the sediment is uncertain.

The transport capacity of overland flow during rainfall can be estimated by the USLE (Eqs. (5.2) to (5.5)), and it is believed that the USLE truly represents the sediment transport capacity of flows in small eroded plots. However, with increased source area after the rain ends, a significant portion of the overland flow still remains on the surface and moves to the nearest established channel. This portion is contained in the recession portion of the hydrograph that does not have the same energy as the runoff during the period of the storm. Consequently, the excess sediment content must be deposited. A major portion of this deposition can occur even before the runoff leaves the field (Fig. 5.21).

The sediment carrying capacity of the runoff flow can be expressed best by Yalin's (1963) equation, which was reported in the following form:



FIGURE 5.21. During the delivery process of sediments most of the sediment is redeposited near the source. When the rainfall energy component of the overland flow terminates, the excess sediment settles out. (Photo: V. Novotny.)

$$P = 0.635s \left[1 - \frac{1}{\alpha s} \ln(1 + \alpha s) \right] \quad (5.13)$$

where P is the dimensionless particle transport given by

$$P = \frac{p}{(\rho_s - \rho)Dv_*} \quad (5.14)$$

The variables were defined as follows:

- p = particle transport per unit width of flow (kg/m-sec)
- ρ_s = particle density (kg/m³), typically 2500 kg/m³
- ρ = density of water (kg/m³) = 1000 kg/m³
- D = particle diameter (m)
- v_* = \sqrt{gHS} = shear velocity (m/sec)
- g = gravity acceleration (= 9.81 m/sec²)
- H = depth of flow (m)

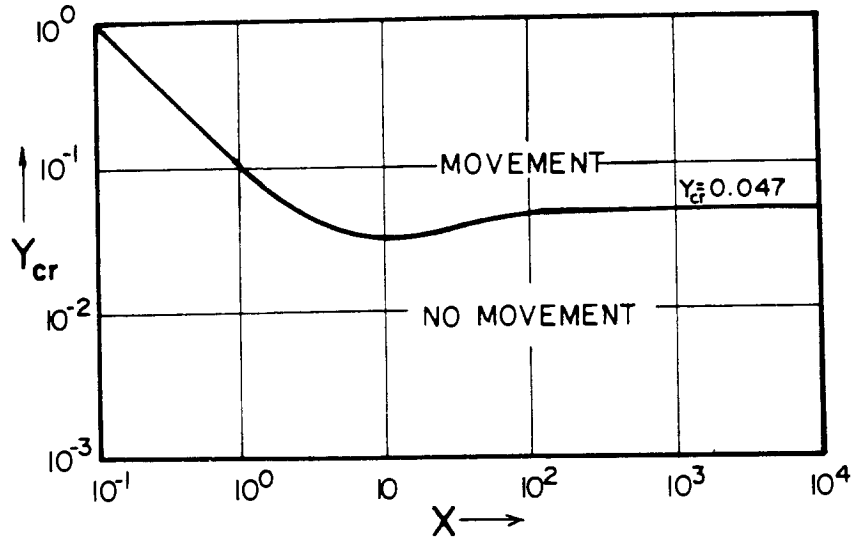


FIGURE 5.22. Shields's (1936) diagram for particle bedload reactive force.

S = slope of energy gradient (m/m)

$s = (Y/Y_{cr}) - 1 =$ dimensionless excess tractive force (when $Y < Y_{cr}$, then $s = 0$)

$\alpha = 2.45 (\rho/\rho_s)^{0.4} \sqrt{Y_{cr}}$

$Y = v_*^2/[(\rho_s - 1)gD]$ is the densimetric particle Froude number

The critical tractive force at which sediment movement begins, Y_{cr} , can be found from the Shields diagram (Fig. 5.22), which is based on the particle Reynolds number

$$X = \frac{Dv_*}{\nu}$$

where ν is kinematic viscosity of the flow (m²/sec).

By rearranging Equation (5.13) and introducing the Manning equation for overland flow rate q

$$q = \frac{1}{n} H^{5/3} S^{1/2}$$

where n is the Manning's roughness coefficient (Table 3.14), and the saturated overland sediment load, p , or concentration, C_e , can be related to the flow rate as

$$p \sim D(qnS^{7/6})^\beta \tag{5.15}$$

or

$$C_e \sim D(nS^{7/6})^\beta q^{\beta-1} \tag{5.16}$$

where the exponent β could theoretically vary in range from 6/5 to 3/4.

Hence, proportionalities (5.15) and (5.16) indicate that the sediment concentration in the receding portion of the flow hydrograph decreases with $\beta - 1$ power as the flow decreases or with $7 \beta/6$ power if the slope decreases.

Yalin's equation was suggested and used by Foster and Meyer (1972b) for estimating soil loss and transport of soil particles when detachment and transport is primarily due to runoff. Yalin himself pointed out that the equation is best used for the estimation of the saturated sediment flow

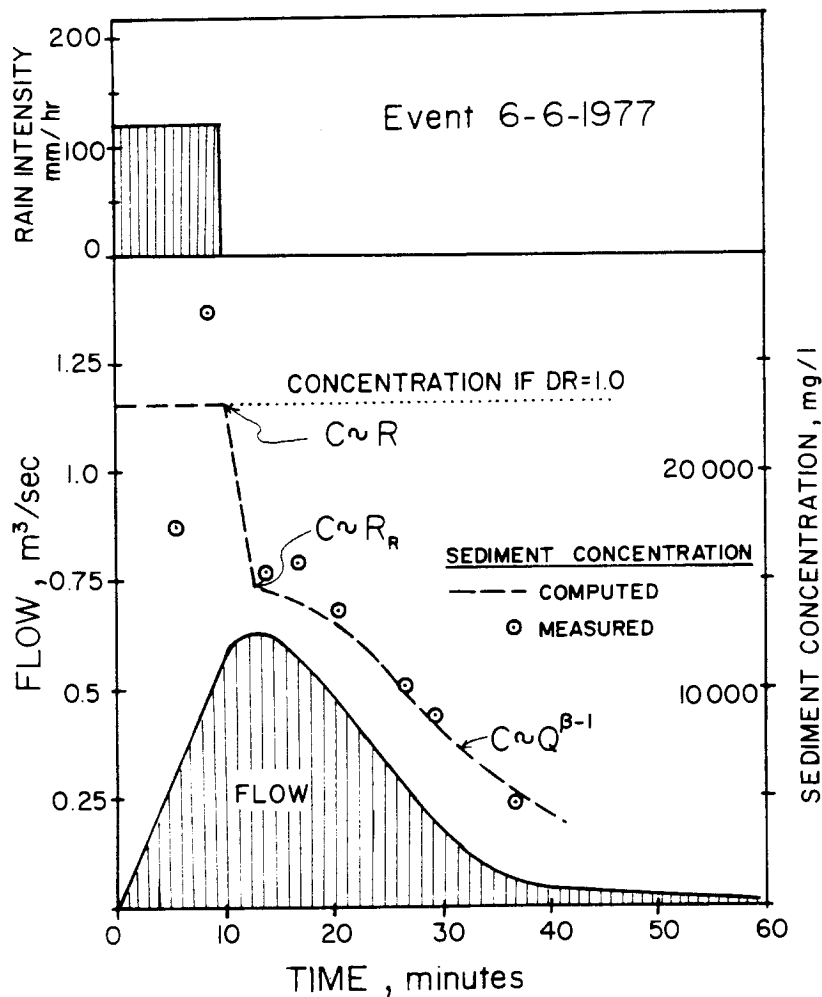


FIGURE 5.23. Runoff hydrograph effect on delivery of sediments and particulate pollutants during overland flow. (From Novotny, 1980, by permission of the American Water Resources Association.)

when most of the particle flow occurs near the bed, which means for the shallow overland and channel flows. It must be also realized that Yalin's equation was developed for fully turbulent flow conditions and for noncohesive sediments. Under shallow flow conditions or when overland flow is through nonsubmerged vegetation, all or a portion of the hydrograph tail can be under laminar conditions (Overton and Meadows, 1976). Under these circumstances, the equilibrium of the sediment concentration, C_e , may approach zero for all fractions. As a result, in a plot of C_e versus q , the exponent can have larger values than the theoretical estimate ($\beta - 1$).

Figure 5.23 and Example 5.3 taken from Novotny (1980) illustrate this concept. In the figure, note that the sediment concentration that would result from both rain and runoff energy can persist only during rainfall. After the rain ends, the excess sediment content settles out, and the sediment concentrations reflect runoff energy. According to the principle of mass continuity it follows that only when the sediment concentration remains approximately constant during and after rainfall (for a uniform rainfall) would the delivery ratio be one.

Example 5.3: Delivery of Sediments from a Construction Site Basin

A 9.7-ha experimental watershed was located in southeastern Wisconsin. A storm with a duration of 10 minutes and storm depth of 2.03 cm occurred on July 6, 1977. The storm hyetograph and resulting flow hydrograph and sediment-graph are shown in Figure 5.23. Estimate the approximate delivery ratio for the storm and calculate the sediment graph.

Solution The parameters for the universal soil loss equation for the construction site basin are

K factor (silt loam) – $K = 0.48$

LS factor for $L = 100$ m and $S = 2.5\%$ – $LS = 0.34$

Cover factor (bare soil): $C = 1.0$

Erosion control practice factor: $P = 1.0$

Infiltration rate: assume 2.5 cm/hr

The storm characteristics are:

Volume of rain: $P_t = 2.03$ cm

Duration: $T = 10$ min

Maximum 30-min intensity: $I_{30} = 4.06$ cm/hr

Calculated rainfall energy factor (Eq. (5.3)): $R_r = 33.20$
 (Rain-infiltration) during 10-min rainfall period = $2.03 - 2.5/6 = 1.61$ cm
 $= 1561$ m³

Runoff characteristics

Excess rainfall = Volume of runoff by integrating the hydrograph

$$Q = 907 \text{ m}^3 / (9.7 [\text{ha}] \times 10,000 [\text{m}^2/\text{ha}]) \times 100 (\text{cm}/\text{m}) = 0.93 \text{ cm}$$

$$\text{Peak flow } q_p = 0.68 \text{ m}^3/\text{s} = 0.68 [\text{m}^3/\text{s}] \times 100 [\text{cm}/\text{m}] \times 3600 [\text{sec}/\text{hr}] / (9.7 [\text{ha}] \times 10,000 \text{ m}^2/\text{ha}) = 2.52 \text{ cm}/\text{hr}$$

$$\text{Runoff erosivity: } R_Q = 0.5 \times 15 \times Q q_p^{0.33} = 9.46$$

Combined rainfall erosivity factor

$$R = 0.5 \times R_r + R_Q = 0.5 \times 33.2 + 9.46 = 26.06$$

Total sediment generated (Eq. (5.2))

$$A_{\text{total}} = 26.06 \times 0.48 \times 0.34 \times 1 \times 1 = 4.25 \text{ tonnes}/\text{ha}$$

For the entire area, assuming that 90% of the watershed contributed the sediment,

$$A_{\text{total}} = 0.9 \times 4.24 \text{ tonnes}/\text{ha} \times 9.7 \text{ ha} = 37.05 \text{ tonnes.}$$

Sediment concentration during the rainfall

$$\begin{aligned} C_{e1} &= \frac{\text{Total sediment loss}}{\text{Volume of rain-infiltration}} \\ &= \frac{37.05 \text{ tonnes} \times 10^6 \text{ g/tonne}}{1561 \text{ m}^3} \\ &= 23,734 \text{ mg}/\text{l} (= \text{g}/\text{m}^3) \end{aligned}$$

This high sediment concentration can be maintained only as long as the rain energy persists. After the rain ends, the total sediment carrying capacity of the runoff drops. At the peak runoff rate, the sediment carrying capacity is approximately proportional to the runoff energy term. Hence, the concentration of sediment becomes

$$\begin{aligned}
 C_{e2} &= \frac{\text{Sediment carrying capacity of runoff}}{\text{Volume of runoff}} \\
 &= \frac{(R_Q)(K)(KS)(C)(P) \times (0.9 \times \text{area})}{Q} \\
 &= \frac{\{9.46 \times 0.48 \times 0.34 \times 1 \times 1 \times 0.9 \times 9.7\}[\text{tonnes}] \times 10^6 \text{ g/tonne}}{907 \text{ m}^3} \\
 &= 14,860 \text{ mg/l}
 \end{aligned}$$

The runoff sediment carrying capacity is decreasing as the hydrograph decreases. Therefore

$$C(t) = \max C_{e2} \times \left(\frac{q(t)}{q_{\text{peak}}} \right)^{\beta-1}$$

The recession coefficient ($\beta-1$) in this case is close to 0.5.

The calculated sediment concentration is plotted as the dashed line on Figure 5.23. The sediment yield can be calculated by integrating the product of multiplying the sediment concentration with the flow. The estimated and measured sediment yield from this watershed during the storm was 16 tonnes. Since the sediment load estimated by the USLE was 37.5 tonnes, the delivery ratio for this storm becomes

$$\text{DR} = \frac{16}{37.05} \times 100 = 43.2\%$$

Effect of drainage on sediment and pollutant delivery. Several small experimental urban and rural watersheds were monitored in southeastern Wisconsin from 1975 to 1980 (Novotny et al., 1979). A hydrological model, calibrated and verified by field measurements from very small, uniform subbasins, was used to estimate upland loadings of sediment and phosphorus. Hydrology (water yield) and sediment and phosphorus yields were calibrated and subsequently simulated. The simulation showed that the delivery ratio for sediment ranged from 0.01 for perviously undeveloped portions of the basin, to 1.0 for completely sewered (separate storm sewers) urban basins. Table 5.8 shows the results of the study. It can be concluded that the delivery ratio depended largely on the degree of storm sewerage.

Sediment flow-discharge relationship. Traditionally, suspended sediment concentration versus flow relationship—known as a rating curve—is obtained by plotting sediment concentrations versus flow on logarithmic

TABLE 5.8 Estimated Sediment Delivery Ratios from Pervious Areas for Various Land Uses in Subbasins of the Menomonee River, Wisconsin

Subbasin Type	Impervious Area (%)	Degree of Storm Sewering (%)	Sediment Delivery Ratio (%)
Agricultural	<5	0	1-30
Developing-construction	<5	20-50	20-50
Low-density residential, unsewered	<20	0	<10
Parks	<10	0	<3
Medium-density residential, partially sewerd	30-50	<50	30-70
Medium-density residential, sewerd	30-50	>50	70-100
Commercial, high-density residential, sewerd	>50	80-100	100

Source: From Novotny et al. (1979).

Note: The delivery ratio for impervious connected areas = 100%.

paper (Fig. 5.9). Generally, according to Walling and Webb (1983), and based on a number of observations of this relationship by the authors, the suspended solids concentration increases with flow, and the logarithmic slope of this relation is between 1 to 2, or, mathematically,

$$\text{Concentration} = a \times \text{flow}^b$$

where b is the slope of the relationship.

However, the pattern of the sediment flow-discharge relationship measured at the flow-quality gauging stations, reflects patterns discussed in Chapter 3 and throughout this chapter. It was shown that the rising limb of the hydrograph is fed by flows during the rainfall event, and the receding limb is due to the remaining overland flows after the rainfall. Hence flows that have generated the hydrographic rise have a higher sediment carrying capacity than do the flows of the end of the hydrograph. This difference between the sediment supply during the rising and ending portions of the hydrograph creates a phenomenon of apparent "first flush" in which the sediment concentrations tend to peak before the peak of the hydrograph occurs (in larger watersheds). This was shown on Figure 5.23. Consequently, the sediment concentration flow relation has a typical "loop" feature, as shown on Figure 5.24. Such hysteric loops have been observed by many observers. Walling and Webb (1983) provided several interpretations of this phenomenon, and based on the work by

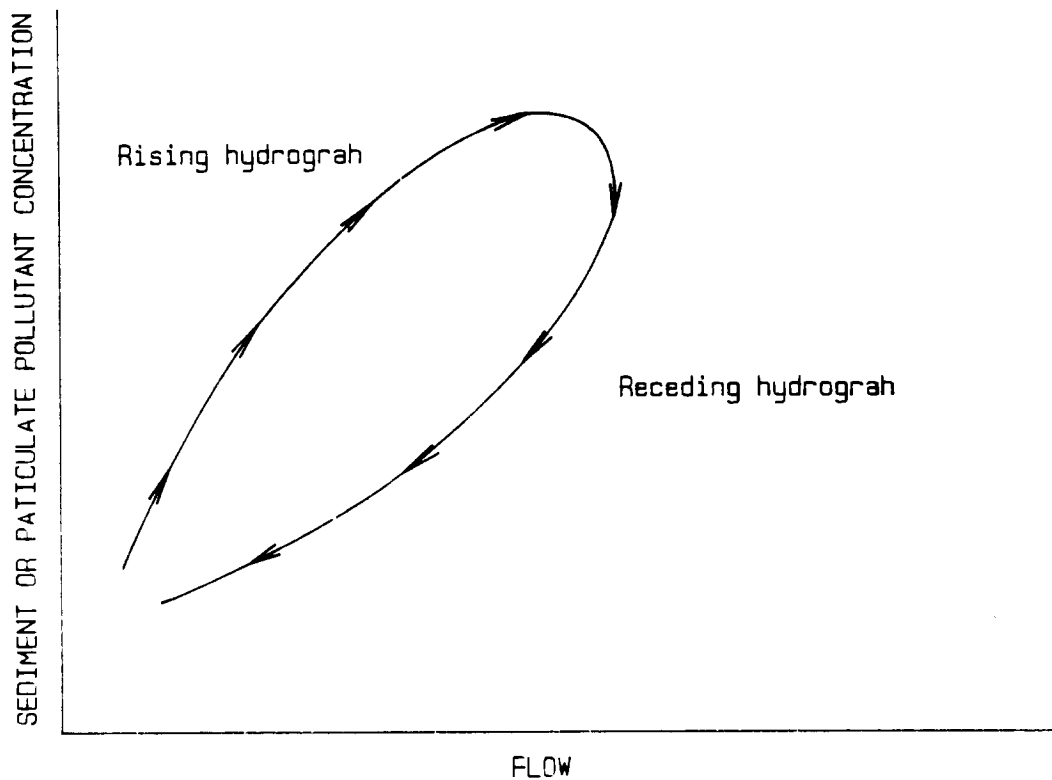


FIGURE 5.24. Hysteric loops in the sediment transport-flow relationship.

Novotny (1980), stated that the hysteric loops of the sediment flow relationship during storm events could be interpreted in terms of reduced detachment or transport of soil particles after the cessation of rainfall. However, Walling and Webb also pointed out that other interpretations are possible, especially during complex multiple and/or prolonged rainfalls.

Long-term measurement of the delivery ratio. Long-term (twenty to thirty years) annual averages of erosion and sediment delivery may be estimated by measuring the cesium 137 isotope concentration in the soil profile (Ritchie, Spraberry, and McHenry, 1974; Ritchie and McHenry, 1975, 1978; Wilkin and Hebel, 1982).

During the period of intensive testing of nuclear weapons by atmospheric detonations, an artificial isotope cesium 137 was deposited more or less uniformly over the earth's surface. This testing period lasted for about twelve years and ended at the beginning of the 1960s. Cs-137 is known to be firmly adsorbed into the soil, and its activity (concentration) can be easily measured. There are no background concentrations of this isotope from a period prior to the nuclear weapons tests. Most of the Cs-

¹³⁷Cs in untilled soil is located in the upper 10 cm. Tillage distributes the isotope uniformly throughout the tillage depth.

Ritchie and coworkers (quoted earlier) pioneered much of the work of using ¹³⁷Cs to quantitatively trace erosion, delivery, and sediment yields. The basic idea is that since the uniform distribution of ¹³⁷Cs during the 1950s the isotope has been redistributed with the sediment. Hence, in areas of higher erosion (degradation), from where the topsoil sediment has been removed, ¹³⁷Cs concentrations are lower than in depositional (aggradation) areas. Alternatively, following Figure 5.8, there will now be less ¹³⁷Cs in areas that were degraded than in aggradation areas. For example, Wilkin and Hebel (1982) found that in eroding areas, ¹³⁷Cs activity was about 0.7 to 0.9 CPMG (count per minute per gram), while in clearly depositional areas of midwestern streams the counts were generally 1 CPMG or greater (these activity counts correspond to a period of sediment redistribution of about 20 years, from about 1960 to 1980, but more profound differences would be found today). The activity of ¹³⁷Cs is determined by gamma-ray spectrometric analyses using a 1024-channel pulse-height analyzer and a solid-state detector.

A regression relationship of the type

$$E = a \times CA - b \quad (5.17)$$

where E is the soil loss in tonnes/ha and CA is the ¹³⁷Cs activity count of the top soil sample in CPMG, has been established by Wilkin and Hebel (1982), who also found that for midwestern watersheds $a = 214$ and $b = 201$. If E is positive, the watershed is eroding, while for depositional areas, E is negative.

For a known or measured change of ¹³⁷Cs Ritchie, Spraberry, and McHenry (1974) found that soil loss can be very closely correlated ($r^2 = 0.95$) to the ¹³⁷Cs activity change

$$A = 0.50 \times \Delta CA^{1.47} \quad (5.18)$$

where

A = total soil loss from 1960 in tonnes/ha

ΔCA = ¹³⁷Cs activity loss $n\%$ of the original input

Using ¹³⁷Cs measurements, Wilkin and Hebel found that much of upland erosion from agriculture was intercepted by upland depressions, grassed border fields, fencelines, hedgegrowths, and roadside ditches. The contribution of floodplains to the sediment problem was also in-

vestigated by this method. Cropped floodplains were eroding at excessive rates and delivering all eroded particles to downstream flow. Forested floodplains, on the other hand, even when grazed, appeared to be removing significant amounts of sediment from downstream transfer.

The Chernobyl nuclear power plant accident (Ukraine Republic) in 1986 caused a deposition of significant quantities of Cs-137 throughout northern and eastern Europe (Haldin, Rodhe, and Bjurman, 1990). Therefore the mass balance of Cs-137 in topsoils affected by the Chernobyl deposition must be recalculated, although the parameters for Equations (5.17) and (5.18) may not apply to the affected areas.

Enrichment of Sediments by Clay and Contaminants

The detachment of sediments and their deposition processes are selective. First, detachment of sediments and contaminants from the parent soil is selective for dissolved pollutants and for the fine soil fractions and pollutants adsorbed in or complexed by them. As will be shown in Chapter 6, most soil contaminants are adsorbed by clay and organic matter because of their high surface area, leading to strong adsorption bonds. When rainwater reaches the surface horizon of the soil, some contaminants are desorbed and go into solution; others remain adsorbed and move with the soil particles. The contaminant content of runoff sediment is then higher than in the parent soil. The difference is termed *enrichment ratio* (ER), and is defined as follows

$$ER = \frac{C_r}{C_s} \quad (5.19)$$

where

C_r = contaminant concentration of the runoff per gram of sediment

C_s = contaminant content of the parent soil per gram

The enrichment concept can be applied to clay, organic matter, and all contaminants adsorbed by soil particles, which include phosphates, ammonium, metals, and pesticides. It is not appropriate to apply the enrichment ratio to contaminants that are mobile in soils, such as nitrates or soluble pesticides. Reviews of the enrichment problems have been published by Novotny and Chesters (1981, 1989) and by Walling (1983).

The enrichment ratio refers to the difference in particle size distribution and associated or adsorbed contaminant content of washload particles and the soils from which the sediment originated. Enrichment of sediments by clay is a two-step process: enrichment during particle pick-

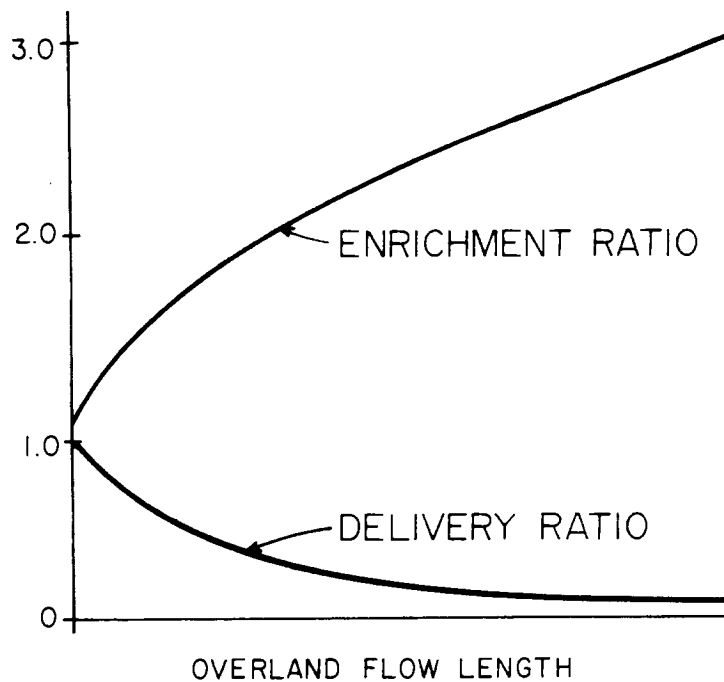


FIGURE 5.25. Relation of the sediment delivery to enrichment of sediment by clays.

up and enrichment during redeposition of the coarser particles during delivery in overland and channel flows. Thus, as the delivery ratio decreases with the increasing watershed area or time of overland flow the enrichment ratio of the washload increases as shown in Figure 5.25.

Pickup of Fines by Erosion

Two variables are necessary for determining the enrichment of sediment picked up by runoff (Free, Onstad, and Holtan, 1975). The variables are the specific surface (SS) of soils or sediment and the clay ratio (CR) defined as

$$SS = [200(\% \text{clay}) + 40(\% \text{silt}) + 0.5(\% \text{sand})]/100 \quad (5.20)$$

and

$$CR = \frac{(\% \text{clay})}{(\% \text{silt}) + (\% \text{sand})} \quad (5.21)$$

Free, Onstad, and Holtan proposed the following relationship, expressing the preceding variables in eroded material and in their soils of origin as

$$SS_e = 14.6 + 0.84SS_m \quad (5.22)$$

and

$$CR_e = 0.021 + 1.08CR_m \quad (5.23)$$

The initial enrichment ratio for sediment picked up by overland flow is then

$$ER = \frac{(\% \text{clay})_e}{(\% \text{clay})_m} \quad (5.24)$$

where the subscripts m and e refer to the soil matrix and eroded material, respectively.

Example 5.4: Computation of the Enrichment Factor

Determine the approximate initial gradation of clay enrichment of the sediment eroded from a loam soil.

Solution The texture of soils can be approximately estimated from maps with the aid of Figure 5.26. In this case, the loam soil has an average composition of clay, 20%; silt, 40%; and sand, 40%. The specific surface according to Equation (5.20) is

$$SS_m = [(200 \times 20) + (40 \times 40) + (0.5 \times 40)]/100 = 56$$

and the clay ratio from Equation (5.21) is

$$CR_m = \frac{20}{40 + 40} = 0.25$$

From Equations (5.22) and (5.23) the specific surface and clay ratios of the eroded material are

$$\begin{aligned} SS_e &= 14.6 + (0.84 \times 56) = 62 \\ CR_e &= 0.021 + (1.08 \times 0.25) = 0.29 \end{aligned}$$

The textural composition of the eroded material can be obtained by solving the three equations

$$[200(\% \text{clay})_e + 40(\% \text{silt})_e + 0.5(\% \text{sand})_e]/100 = 62$$

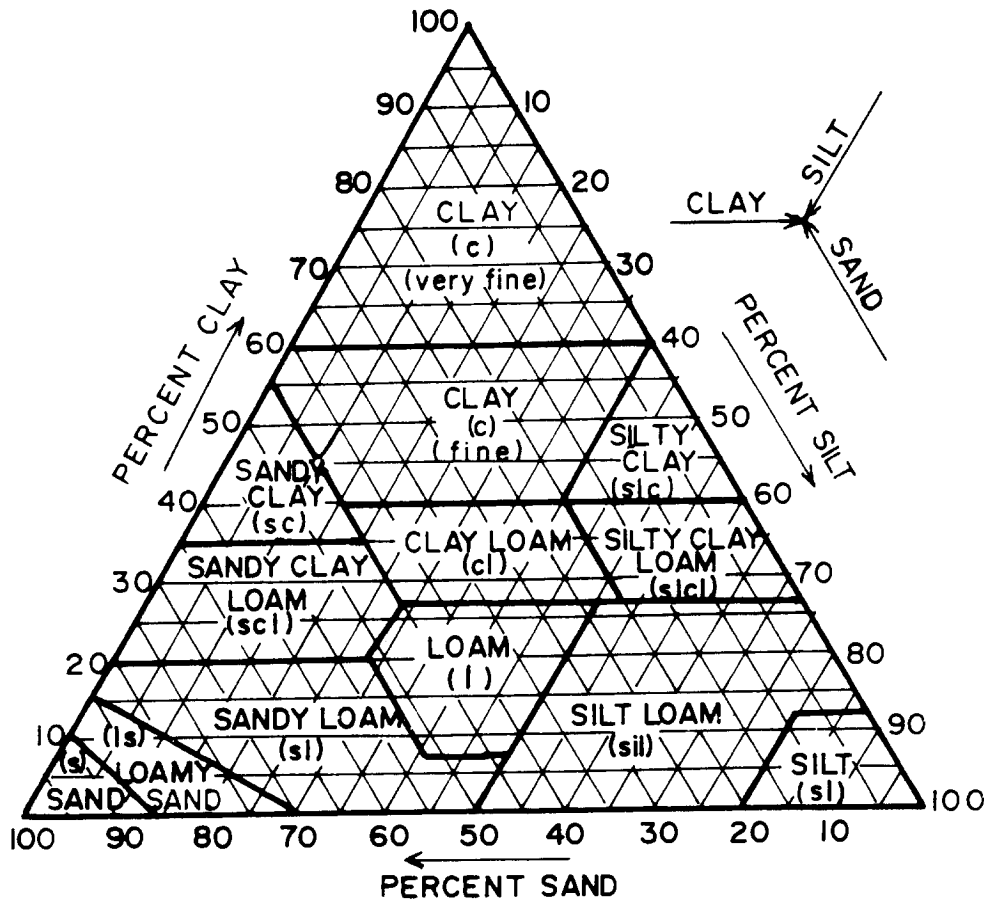


FIGURE 5.26. Guide for the USDA Soil Conservation Service soil texture classification.

$$\frac{(\% \text{clay})_e}{(\% \text{silt})_e + (\% \text{sand})_e} = 0.29$$

$$(\% \text{clay})_e + (\% \text{silt})_e + (\% \text{sand})_e = 100$$

which gives an initial textural composition for the eroded material of clay, 22%; silt, 43%; and sand, 35%. The clay enrichment ratio is then

$$ER = \frac{(\% \text{clay})_e}{(\% \text{clay})_m} = \frac{22}{20} = 1.1$$

Relationship of Enrichment to Delivery

During the delivery process, the texture of the sediment changes; eventually it is enriched by fine fractions. For a delivery ratio DR, a fraction of sediment equivalent to (1 - DR) is redeposited during transport. The redeposition is selective, that is, according to the bottom shear stress of

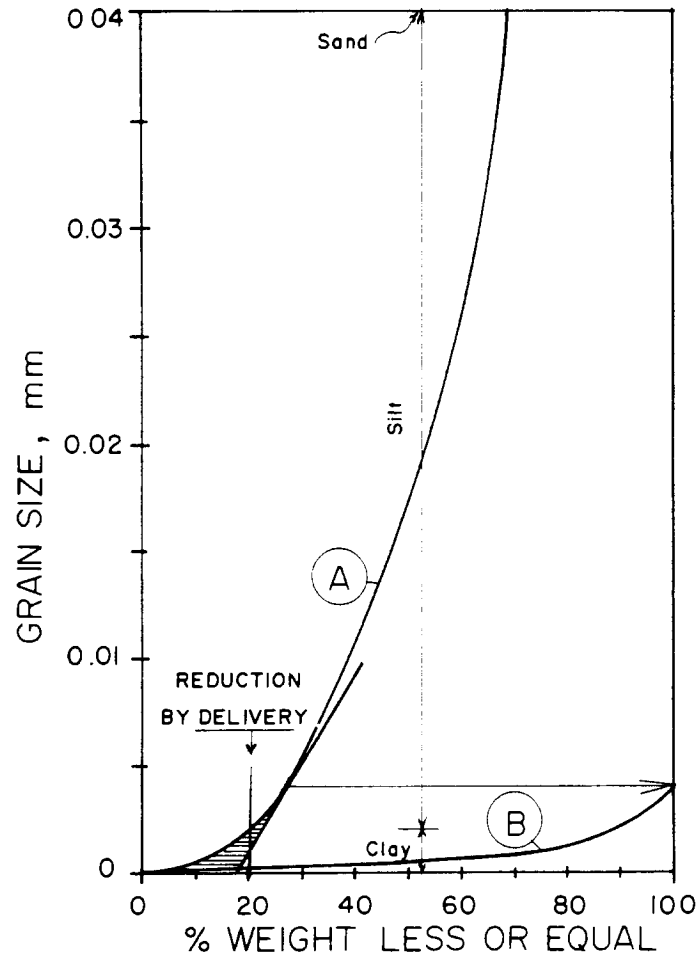


FIGURE 5.27. Graphical determination of washload composition and enrichment ratio from grain-size distribution curve.

flow, sand fractions are deposited first, followed by silt, and then, only when the flow slows down to a very low velocity and low Reynolds number, clays may deposit.

The deposition of clays is enhanced if the clay particles move in the form of aggregates. Dong, Chesters, and Simsman (1983) have proposed a technique for determining an index for clay dispersion. Walling (1983) assumed that the probability of clay deposition is very low, hence the delivery ratio for clays and their enrichment are interrelated as follows

$$DR = \frac{\text{Clay content (soil)}}{\text{Clay content of sediment}} = \frac{1}{ER} \quad (5.25)$$

An approximate graphical procedure based on the grain-size cumulative distribution curve can also be used to determine changes in sediment texture during overland transport, as shown on Figure 5.27.

Foster, Young, and Neibling (1985) defined the enrichment ratio as a ratio of specific surface area in the eroded sediment and parent soils, and related the enrichment ratio, ER, to the delivery ratio, DR, by the following empirical relationship:

$$ER = 1 + \alpha \exp(-\beta DR) \quad (5.26)$$

where the coefficients α and β were dependent on soil texture.

Dong, Simsiman, and Chesters (1983) and Dong, Chesters, and Simsiman (1984) studied particle-size distribution and the composition of soils and sediments from the 350-km² Menomonee River basin in southeastern Wisconsin. Data from a 21-km², predominantly agricultural subwatershed located in the upper reaches of the river provided a direct determination of sediment enrichment by clays and contaminants. Particle-size distribution of soils in the subbasin was 27%, 49%, and 24% for clay, silt, and sand, respectively. Suspended sediment samples collected near the outlet of the subbasin (bedload movement is small compared to washload) during runoff events showed a particle-size distribution 91% clay and 9% silt.

Using the relationship between clay enrichment and the delivery ratio, $DR = 27/91 = 0.29$ and $ER = 91/27 = 3.4$. Delivery ratios for the same watershed established by modeling (Table 5.8) were in the range of 0.2 to 0.3.

The enrichment factor is a more dynamic phenomenon than sediment delivery. In addition to storm-to-storm variations this factor also exhibits seasonal and spatial variations for some constituents, while for some others, such as some metals eroded from soils and minerals, it could remain approximately constant (Walling and Kane, 1982). Most typical potential pollutants, including nitrogen, phosphorus, pesticides, and metals, exist in soils in adsorbed and dissolved forms (see Chapter 6 for details). The adsorbed materials move with sediment, and the fraction dissolved in soil water moves with runoff and/or ground-water recharge. The continuous process of adsorption and desorption–dissolution takes place during the overland and channel flow transport of the constituents. The concept of enrichment is not applicable to some dissolved or dissociated constituents, such as nitrates, which are repulsed by the electrostatic charge of the soil particles.

Enrichment of snowmelt. A different process of enrichment takes place during spring snowmelt. Snow pack is an effective filter of suspended particles; however, the snow crystal lattices during freezing and refreezing effectively reject many dissolved contaminants, which then remain available for pick up by melt water. For this reason, the first portion of snowmelt is highly enriched by dissolved and dissociated pollutants

(nitrates, H^+ , and others) with ER values much greater than one, while particulate pollutants remain in place, which results in ER factors of less than one (Johannessen and Henriksen, 1978; Colbeck, 1981).

Example 5.5 Determination of Texture Changes in Sediments During Delivery

A watershed with a drainage area of 21 km^2 had a measured sediment yield of 10,000 tonnes during a large spring storm. Estimated potential erosion from mostly loam agricultural soils was 50,000 tonnes. Assume that the texture of the soils and the eroded material are similar to those in Example 5.4. The eroded material therefore has the following initial texture: clay, 22%; silt 43%; sand 35%.

Solution The delivery ratio is

$$DR = \frac{Y}{A} = \frac{10,000}{50,000} = 0.2$$

An approximate cumulative grain-size distribution curve of the eroded soil is shown on Figure 5.27 as curve A. The delivery ratio of 20% indicates that most of the washload will be composited from clays. However, some coarser particles from areas closer to the receiving body of water can still remain in suspension. This can be represented by a tangent drawn to curve A, as shown in the figure. The triangular area on the left from the vertical line denoting the 20% delivery and the area between the tangent and the original curve on the right from the vertical delivery line are about the same. The shaded area represents the cumulative percentages of the remaining fractions in the washload, with the tangent as the baseline. To obtain the grain-size distribution curve of the washload, the vertical coordinates of the remaining fractions must be transformed to cover 100% scale (curve B).

From Figure 5.27, the composition of the washload will be: clay, 91%; silt, 9%; sand, not detectable.

The ER for a soil originally containing 20% clay is

$$ER = \frac{91}{20} = 4.5$$

SEDIMENT TRANSPORT IN STREAMS

In channels (ephemeral and perennial streams or impoundments) flow is concentrated and has greater depth and velocity. While the overland

sheet flow is mostly laminar and is able to support concentrated sediment flows only by the combination of rainfall and flow energy, flow in channels is turbulent and able to carry sediment flows on its own.

To a geomorphologist, the channel phase represents only temporary storage of transported sediment. Playfair's Law, one of the basic premises of stream morphology, states that over a long period of time a natural stream must transport essentially all sediment delivered to it (Boyce, 1975). One could argue with these geomorphological postulates because apparent deposition occurs in slow sections of rivers and impoundments, floodplains, and deltas. This results in distinct alluvial deposits, the best example being the mouth (delta) of the Mississippi River. Wilkin and Hebel (1982), using the Cs-137 indicator, found significant floodplain degradation on the Illinois River. Leopold, Wolman, and Miller (1964) concluded that materials eroded from a drainage basin are only temporarily stored in floodplains and that floodplain aggradation is essentially balanced by floodplain degradation. This means that a long-term (say 100 years or more) delivery ratio for streams is essentially one.

However, this is of little concern in pollution and water quality studies. For example, organic materials that are carried by and buried with the sediment in the alluvial deposits become harmless fossils in hundreds of years, long before they may be resuspended by catastrophic floods, yet, meanwhile, they may cause serious degradation of water quality by causing sediment oxygen demand (SOD) and other problems. Thus on the shorter time period (10 years or less) typical for water quality studies, deposition, resuspension of sediments, their contamination, and processes occurring in the deposited sediment layer are important and should be considered. The delivery ratio of many streams is much less than one when considering runoff magnitudes with a recurrence interval of less than 10 years.

Deposition of sediments and particulate pollutants occurs primarily in three places:

- In slow sections of streams, such as impoundments, estuaries, and deltas, and along the stream banks;
- Temporarily in a floodplain. For many pollutants that can be decomposed or degraded, a floodplain is a terminal sink.
- In ephemeral or recharging streams by filtration of particulates during recharge. This storage on the channel bed is only temporary, and the particles remain on the stream bottom available for resuspension by the next runoff event.

Contamination of bottom sediments by toxic and other polluting substances affects the quality of the water column and represents the

entry of toxics into the food chain. It is a known fact that the pollutants that have been previously deposited into sediments contribute to the degradation of water quality and adversely affect aquatic ecosystems in many water bodies, including, among others, Green Bay and other bays of the Great Lakes system, Chesapeake Bay, Long Island Sound, and many other inland and near-shore aquatic systems. Milwaukee (Wisconsin), Grand Calumet Indiana Harbor (Indiana), and Waukegan (Illinois) harbors, as well as many other urban estuaries, exhibit sediment contamination that prevents beneficial uses of these bodies of water. However, due to the binding of many pollutants with the sediments, the adsorbed or complexed pollutants become biologically unavailable and, hence, not toxic. Therefore knowledge of the relation between immobilized (biologically unavailable) and free dissolved (available) fractions of the pollutant in the sediment is important in pollution impact studies. See Chapter 6 for more discussion of this subject.

Bottom sediments become contaminated with toxic substances by the deposition of particulate fractions of pollutants and/or by the deposition of fine sediment particles, both organic and inorganic, that have an affinity to adsorb dissolved pollutants and pollutant ions in the water column.

After deposition, several processes may cause the deposited pollutants to reenter the water column, including the resuspension of sediment particles caused by turbulence and biological benthic activities, diffusion from the bottom layer, uptake and subsequent decomposition of rooted macrophytes, and several other processes. Reentry may be temporal, such as with a particle that had been temporarily resuspended, or semi-permanent, such as by diffusion, in the water column.

In the sediment layer, many pollutants may undergo chemical and biological modifications, including chemical and biological degradation, conversion between inorganic and organic forms, and adsorption (fixation) –desorption reactions.

Sediment Transport Process

Sediment transport, erosion, and sedimentation have been a popular subject of research for more than 80 years (Vanoni, 1975). Sediment in streams is transported either as washload, or suspended sediment, or bedload, or sediment, and it moves near or at the bottom. In the classic representation of sediment movement, the suspended sediment contains mostly fine sediment particles, while normally bedload may be a composite of sand and gravel. Several sedimentologists showed a relation between the washload and bedload (Einstein, 1950; Yalin, 1963; Vanoni,

1975). In this representation there is a continuous exchange of particles between the bedload and the washload, and the vertical concentration distribution of the washload is related to the bedload transport. Most of these classic studies assumed that there is no interaction between the sediment particles other than the impact of collision. Such sediments are called *noncohesive sediments*, and include primarily sand and gravel fractions. Noncohesive sediments have a very low to zero affinity to adsorb pollutants, and transport of pollutants is generally not affected by their presence.

In contrast, *cohesive sediments*, which include primarily clay and organic fractions of the bedload and washload, have a high sorptive capacity for many chemicals and act as carriers for contaminant transport in riverine and estuarine systems. As shown in a publication by Novotny and Chesters (1981), clay and organic matter can effectively adsorb such pollutants as phosphates, ammonia, and a variety of organic chemicals, from which PCBs exhibit the highest affinity for adsorption. In addition, some pollutants, such as organic matter, nitrogen and phosphates, some metals, and some organic chemicals, exist both in solid and liquid phases (precipitated and dissociated), and are an integral part of the washload.

Organic-rich sediments (mud), such as those generated in the productive sections of streams by photosynthesis (Novotny and Bendricchio, 1989), have the most severe implications on water quality conditions. The processes of decomposition that occur in these sediments impose a demand for oxygen on the overlying water. Under anaerobic conditions phosphates and ammonia are released back into the water and chemical adsorption equilibria are modified in the reducing environment of anaerobic sediments.

The classic literature of sedimentology has not addressed the problem and processes associated with the movement of cohesive sediment and their impact on water quality and pollutant transport. Only recently have sedimentologists become interested in the subject.

Transport of Cohesive Sediments

Figure 5.28 shows the interrelationship between the basic transport processes of cohesive sediments in riverine and estuarine systems. While in noncohesive sediment transport, bedload and washload are fairly distinct with a clear boundary interface. Cohesive sediment may exit in the four states shown in Figure 5.28: (1) a mobile suspension where particles move primarily in a horizontal direction; (2) a horizontally stationary high suspension; (3) a consolidating bed; and (4) a settled compacted bed in which movement of particles and water has ceased (Mehta et al., 1989).

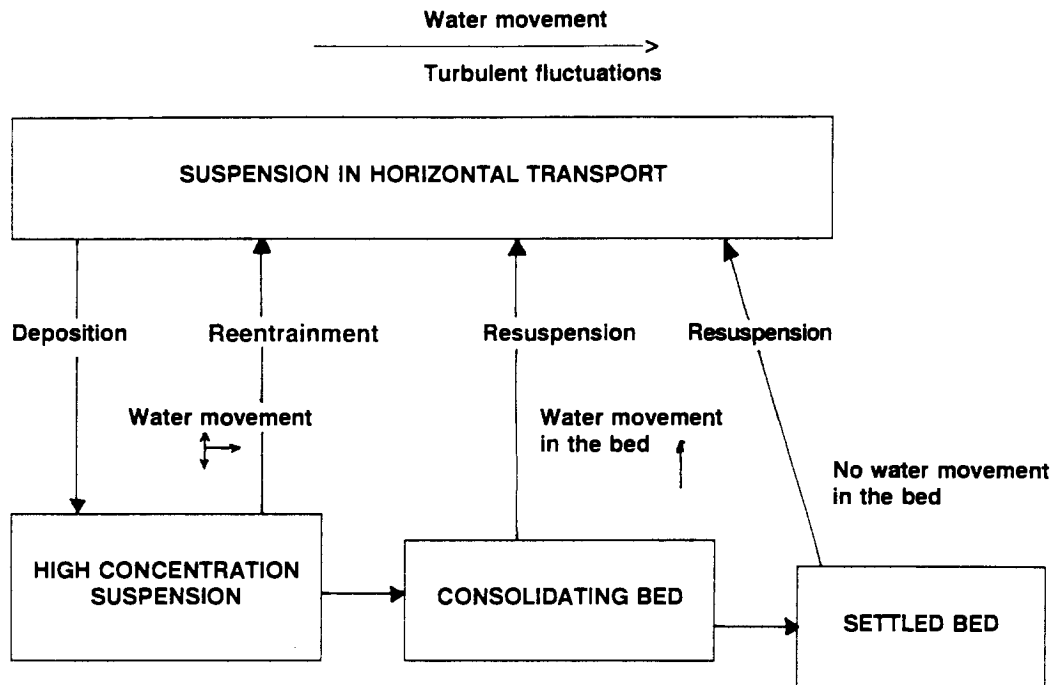


FIGURE 5.28. Physical states and processes governing cohesive sediment transport. (After Mehta et al., 1989, reprinted by permission of ASCE.)

Of interest in this concept is the movement of water. When particles are in suspension, water movement is primarily in a horizontal direction; however, at washload concentrations of solids, turbulent fluctuations may be less than those for clear water (Xingkui and Ning, 1989). In high concentration suspension, the horizontal movement of particles diminishes, and if the net particle flux is down, the replaced water moves up. This replaced water then mixes with water in the flow above, which should be included in the mass balance equations of pollutants. What it means is that if there is net settling of particles, there is a net water movement up, which can carry dissolved pollutants with it. There is similar water movement in the compacting layer where, as the layer reduces its thickness and increases its density, water is pushed upward from the sediment layer. In both cases, the sediment layer is a source of dissolved pollutants.

Higher concentrations of suspended solids modify the density of the sediment-water mixture. Similarly, to heat or salinity transport, suspended sediments in impoundments and estuaries can travel in density currents (Akiyama and Stefan, 1985).

Typically, cohesive sediments (mud) in riverine and estuarine systems are composites of clay, silt, and organic particulates. Cohesion of par-

ticles (floc formation) is more pronounced for clay particles ($<2\ \mu\text{m}$) than for more coarse silt particles (2 to $60\ \mu\text{m}$). In fact, Mehta et al. (1989) speculated that the cohesion of silt particles into larger flocs and compacted matter is by clays. When large quantities of coarse materials (sand, larger organic detritus) are present in the sediment, its behavior and interactions become quite complex, and are currently not well understood (Mehta et al., 1989).

Aggregation of cohesive sediments is a complex problem that depends on several environmental factors. Some of the factors are well understood from water treatment technology, such as the effects of calcium and other cations, anions, total salt concentration, and pH. Other factors, such as the effect of turbulent mixing, settling characteristics of particles in the aquatic environment, and particle interactions are less well understood. The biological effects of bottom dwelling and feeding fish and invertebrates, or the mixing and resuspension of particles by the evolution of gases from the benthic layer are speculated about only qualitatively.

Settling velocity of cohesive sediments. In turbulent flow, suspended particles are supported and distributed in the flow by the mechanism of turbulent exchange, which acts against the gravity force of the particles. The fall velocity of the individual particles is given by the well-known Stokes' law. However, cohesive sediment particles will aggregate under favorable chemical conditions, thus increasing the settling velocity over that given by Stokes' law. Table 5.9 shows the effects of aggregation on the increase of settling velocity.

The preceding results of the effect of aggregation on settling velocity were obtained by experiments in settling tubes (Migniot 1968; Chase, 1979). Reported values of settling velocities for marine and estuarine sediments ranged from 10^{-4} to 10^0 mm/sec.

Effect of hindered settling. At a certain concentration of suspended particles, settling becomes hindered. This is analogous to the classic concept of the settling of flocculent particles in settling tanks and settling

TABLE 5.9 Settling of Individual Particles and Aggregate Settling

Original Particle Diameter (μm)	Stokes Settling Velocity (mm/sec)	Aggregate Settling Velocity (mm/sec)	Aggregate Diameter (μm)
20	$2.4 * 10^{-1}$	$2.7 * 10^{-1}$	88
2	$2.4 * 10^{-3}$	$1.7 * 10^{-1}$	56
0.2	$2.4 * 10^{-5}$	$1.1 * 10^{-1}$	34

Source: After Mehta et al. (1989). (Reprinted by permission of ASCE.)

columns. During hindered settling the probability of particle collision is so high that it slows down settling and the particles may settle as one mass (zone settling). This zone settling pattern may loosely correspond to the high concentration suspension layer depicted on Figure 5.28.

Tesarik and Vostrcil (1970) proposed an equation for the settling velocity of flocculated flocs as

$$W = \frac{\alpha}{c^\beta} \quad (5.27)$$

where α and β are coefficients. The values of the coefficients for the bentonite clay were reported as $\alpha = 12.8$ and $\beta = 2.06$ for W in mm/sec and c in g/l.

The transition between hindered settling and discrete settling occurs approximately at a volumetric floc concentration of 0.005 (Camp, 1947), which is comparable to a concentration of 1000 to 5000 mg/l of silt and clay particles in turbid water.

Generally, the high concentration (hindered settling) zone should exhibit a distinct interface between the more clear turbid water above where settling is discrete. This is due to the fact that if a floc leaves the high concentration layer, the settling velocity ceases to be hindered and increases. Thus the likelihood of the particle to settle back into the high concentration zone also increases. Another reason for a distinct interface is the fact that turbulence fluctuations in the high concentration zone are reduced; therefore, at the interface there is less chance for a particle to move upward from the high concentration zone than there is for it to move downward from the discrete overlaying zone. If the turbulent energy of flow increases above a certain yet unspecified level, this increase may lead to a collapse of the high concentration zone and the flow may become uniformly mixed.

The surface stresses of the high suspension (hindered settling) layer are very small and in general are not measurable (Been and Sills, 1981). Hence, if under some flow conditions (presumably at low energy levels) the probability of particles to remain in the high concentration zone is greater than it is to escape from the zone, the particles will remain in the layer and will require a discrete resuspension energy to rise up from the layer.

What this means is that there appears to be a critical shear stress for formation of the high concentration layer that is an integral component of the deposition process of cohesive sediments. The existence of this layer may also have significant water quality implications. As water that was enriched by the high rate of desorption is pushed out of the high con-

centration layer by the settling of particles, it may become the primary source of contamination in the overlying water.

The energy level can be related to the bottom shear stress of flow. Data in the literature indicate (Mehta et al., 1989) that the shear stress for deposition (deposition of cohesive sediments will not take place if the shear stress is above this value) is about $\tau_c = 0.06$ to 0.08 N/m^2 . This correlates well with the experiments of Partheniades, who found that the critical shear stress for deposition is around 0.1 N/m^2 . For mixed sediments with broad size distribution, Mehta and Partheniades (quoted in Mehta et al., 1989) found that τ_c ranged from about 0.18 to 1.1 N/m^2 . A well-known formula for estimation of shear stress is

$$\tau = \gamma R S_e \quad (5.28)$$

where

γ = specific density of water (N/m^3)

R = hydraulic radius or depth (m)

S_e = the energy gradient of flow (dimensionless)

Below the high concentration (hindered settling) zone is a layer of consolidating sediments. In the theory of settling this layer could correspond to compression settling. The particles in this layer form a solid compacting mass that settles due to the gravity of particles and the water above. This consolidation by compressed settling begins when surface stresses become noticeable. Primary consolidation ends when the excess pore water pressure has completely dissipated. Secondary consolidation, which is the result of the plastic deformation of the bed under the constant overburden, begins during primary consolidation and may typically continue for many weeks or months after primary consolidation ends (Mehta et al., 1989).

Erosion and Resuspension of Cohesive Sediments

The erosion potential of consolidated sediments is again related to bottom shear stress. The critical shear stress for erosion, τ_s , has been related to the specific density of compacted sediments and to the pressure of the overlying water (depth of the water column) as

$$\tau_s = \xi \rho_D^\delta \quad (5.29)$$

If τ_s is in N/m^2 , then the coefficient ξ is about 6×10^{-6} to 8×10^{-6} , and the exponent δ is about 2.3 to 2.4 (Mehta et al., 1989). ρ_D is the dry density of consolidated sediment in kg/m^3 , which is a function of time.

The dry sediment density can be calculated from the bulk density as follows (Mehta et al., 1989):

$$\rho_D = \frac{(\rho_B - \rho_w)\rho_s}{(\rho_s - \rho_w)} \quad (5.30)$$

where

ρ_B = bulk (wet) density of the sediment

ρ_s = sediment density

ρ_w = water density

The critical shear stress for the erosion of consolidated sediments is one to two orders of magnitude greater than that for deposition. The scouring rate of sediment can be related to the bottom shear stress, as pointed out by Mehta et al. (1989) and Partheniades (1965).

Three modes of erosion have been identified: (1) reentrainment of stationary suspension (as discussed earlier), which occurs at shear stress rates between τ_c and τ_s ; (2) aggregate or gradual erosion of a bed at shear stresses greater than τ_s and a certain critical value of bed shear stress τ_b ; and (3) mass erosion of a bed when bottom shear stress energy is sufficient to lift the bulk of the sediments.

Experimental work (Lee, Kang, and Lick, 1981; Ziegler and Lick, 1988) leads to a formulation of a relationship between the erosion of fine-grained cohesive sediment and bottom shear stress as follows (Ziegler and Lick, 1988):

$$\begin{aligned} \varepsilon &= \alpha \left[\frac{\tau - \tau_s}{\tau_s} \right]^m && \text{for } \tau \geq \tau_s \\ &= 0 && \text{for } \tau < \tau_s \end{aligned} \quad (5.31)$$

where

ε = net amount of resuspended sediment in Kg/m^2

$\alpha \approx 0.08$

$m \approx 2$

There are several interesting conclusions from the past research on deposition, formation, consolidation, and erosion of cohesive sediments:

1. The "classic" sedimentology typical of noncohesive sediments is not fully applicable to cohesive sediments. For noncohesive sediments there is a certain limiting shear stress value τ_c above which deposition will not occur.

2. At low shear stresses, formation of a high concentration–hindered settling layer is likely. Even though this layer may not show any measurable shear stress resistance, energy (shear stress) in excess of τ_c is needed to break the high concentration layer into a dispersed flow form. The magnitude of this energy is not known and will most likely be between the critical shear stress for deposition and that for the erosion of a consolidated bottom. At certain small shear stress values above τ_c the erosion of the high concentration layer may be gradual; however, experience shows that it may be possible to anticipate that the surface of the high concentration layer will move upward in response to increased shear stress until the concentrations within the layer cease to be hindered and/or the top of the layer reaches the flow surface.
3. The high concentration layer provides a medium for water to come in contact with contaminated sediments even when resuspension does not occur. Because of the settling of particles there is a distinct upward velocity component within the layer that brings the dissolved pollutants to the overlying water. At some critical yet unknown shear stress the high concentration layer will break and become completely mixed with water.
4. The consolidated layer requires higher energy for breaking and resuspension. Below this energy level, the mechanism of pollutant transfer from (or into) the layer is by diffusion through the boundary layer.
5. At shear stresses greater than the critical erosive shear stress, τ_s , the bottom is resuspended and most likely completely mixed with the overlying water (or hypolimnion of stratified flows). The rate of resuspension is related to the bottom shear stress. At certain high shear stress rates the compacted layer may be eroded at very high rates (Parker, 1982).

Kinetic Model for Sedimentation of Cohesive Sediments

The author previously presented a simple model for the sedimentation of cohesive sediments (Krenkel and Novotny, 1980). The model was based on classic sedimentology and may not be fully applicable today in view of new information published later in the 1980s. However, the basic concepts after modification are applicable. In view of the new information it is necessary to separate the flow of cohesive sediments and pollutant transport into three distinct zones (Fig. 5.28):

1. *Zone of dispersed flow or discrete settling zone in which classic sedimentology is partially applicable.* Turbulence is not greatly affected by

sediment concentrations, particle concentration might follow the classic Einstein distribution (Vanoni, 1975; also published in Krenkel and Novotny, 1980, and Novotny and Chesters, 1981), and dissolved pollutant transport is by convection and turbulent dispersion.

2. *Zone of hindered settling (high concentration) in which classic sedimentology is not applicable.* The turbulence level is affected by particle concentrations, the horizontal movement of particles is diminished, there is a net upward vertical component of water, and water is rich with desorbed pollutants or in a higher equilibrium with adsorbed fractions. This transitional zone may change thickness in response to the shear stress. This layer will exist when shear stress is less than a certain critical value that is smaller than the critical shear stress for erosion of a compacted bed.
3. *Compacting sediment zone.* There is no horizontal movement of sediment; however, there still may be some upward movement of water through the sediment layer during compaction. This convective transport adds to the diffusion of pollutants, which is controlled by diffusion through the boundary layer. When the shear stress exceeds the critical shear for erosion, sediment particles are released from the layer.

Discrete settling zone. In this zone, the classic laws of sediment transport are applicable. Define M as the scour rate of cohesive sediments or the reentrainment rate of particles from the high concentration layer, and W_s as a settling rate under aggregation. Then the concentration distribution can be described by Einstein's equation (see Novotny and Chesters, 1981). Also under steady-state conditions

$$\frac{dg}{dx} = M - W_s C_b \quad (5.32)$$

where

C_b = near bottom concentration of the sediment

g = the sediment flux

However, Partheniades (1977) documented that this equation is not fully valid for the transport of cohesive sediments. It was also documented in the previous discussion that in cohesive sediment transport erosion and sedimentation do not occur simultaneously. Furthermore, according to the Partheniades theory and experimental results the erosion rate of cohesive sediments from the compacted layer should remain constant regardless of the sediment concentration in water. This may lead at high bottom shear stresses to very high sediment concentrations that are

related to the supply of sediment rather than to a saturated sediment flux. Consequently, on the other hand, if the shear stress is below the critical shear stress for deposition, there is no erosion, and hence the minimal concentration of sediments is zero, provided that enough time is available for the particle to settle out.

High concentration zone. The classic sedimentology of discrete non-cohesive sediment flow fails in the high concentration flow, that is, when volumetric concentrations of particulates are above $c = 0.5\%$. It should be pointed out that the high concentration (hindered settling) flow zone can extend all the way to the surface, as documented on flows occurring in some high turbidity streams of some southwestern regions, such as Rio Puerco in New Mexico.

Compaction zone. During deposition solids from the high concentration zone will reach the compaction zone and undergo compressed settling, or at some very low turbulence level the entire high concentration zone may begin to compress. During compressed settling water is released upward.

From the foregoing discussion it appears that no erosion of the compacted zone takes place as long as there is a high concentration zone above, since the collapse of the high concentration zone or its extending to the flow surface may occur at lower shear stress values than the minimum shear stress for erosion of compacted sediment layers.

Furthermore, as pointed out by Partheniades (1977), no simultaneous deposition and erosion of cohesive sediment takes place, and in the absence of erosion the only mechanism by which pollutants can be released from the compacted zone is by diffusion and by convection by water removed from the layer by compaction.

This concept of three-layered cohesive sediment transport and sedimentation, which has been proved by experiments of Mehta et al. (1989) and earlier by Partheniades deviates from some modeling concepts, such as those incorporated in the EPA model WASP (see Chapters 7 and 8), which presumes the existence of a certain unspecified constant layer of sediment that mixes with the overlying water. The variable high concentration layer that can exist only at bottom shear stresses below those for erosion ($\tau < \tau_s$) is the nearest approximation of such a mixing layer. At shear stresses above τ_s it is likely that there is no mixing layer that the entire bulk of the water may have a uniform concentration of sediment, and that, the flux of pollutants from the bottom sediment layer is related to the rate of erosion. This high concentration mixing layer may reappear when the sediment concentration increases to a level at which turbulence is reduced and/or the flow rate drops so that the bottom shear stress is reduced below that which allows deposition.

Example 5.6: Erosivity of a Channel Flow

A wide (width $>10 \times$ depth) rectangular channel is carved in cohesive sediments. The depth of flow in the channel is $H = 0.1$ meters and the velocity is $v = 0.3$ meters, the Manning roughness factor for the channel is $n = 0.02$. Determine whether the flow is erosive or depositional.

Solution In order to estimate the bottom shear stress one has to calculate the energy gradient of the flow. From Manning's equation and assuming that for a wide channel the hydraulic radius approximately equals the depth of flow, the energy gradient, S_e , is

$$S_e = \frac{(v \times n)^2}{H} = \frac{(0.3 \times 0.02)^2}{0.1} = 0.00036$$

The bottom shear stress (Eq. (5.28)) is (for $R = H$ and $\gamma = 9810 \text{ N/m}^3$)

$$\tau = \gamma H S_e = 9810 \times 0.1 \times 0.00036 = 0.35 \text{ N/m}^2$$

The shear stress of flow is above or near the critical shear stress for deposition ($\tau_c \approx 0.1 \text{ N/m}^2$); however, it appears to be below the critical shear stress for scouring. The critical bottom shear stress for scouring can be estimated by using Equation (5.29). For example, if the specific density of the bottom sediment is $\rho_s = 2500 \text{ kg/m}^3$, the specific density of water $\rho_w = 1000 \text{ kg/m}^3$, and the bulk density of the sediment $\rho_B = 1200 \text{ kg/m}^3$, then the dry density of the sediment is (Eq. (5.30)) $\rho_D = (1200 - 1000) \times 2500 / (2500 - 1000) = 333$. Assuming that $\xi = 7 \times 10^{-6}$ and $\delta = 2.4$, then the critical bottom shear stress for scouring is

$$\tau_s = 7 \times 10^{-6} \times 333^{2.4} = 7.94 \text{ N/m}^2$$

One could conclude that the flow is either very mildly depositional or that no deposition or scour will occur. The delivery ratio would be near unity.

References

- Akiyama, J., and H. Stefan (1985). Turbidity current with erosion and deposition, *J. Hydraul. Eng. ASCE*, 111(12):1473–1496.
- Anderson, H. W. (1975). Sedimentation and turbidity hazards in wildlands, in *Proceedings, Symposium of the Irrigation and Drainage Division*, ASCE, New York, pp. 347–376.
- Been, K., and G. C. Sills (1981). Self-weight consolidation of soft soils, *Geotechnique* 31(4):519–535.

- Beer, C. E., C. W. Farnham, and H. G. Heinemann (1966). Evaluating sedimentation predictions in western Iowa, *Trans. ASAE* 8(9):828–833.
- Bordas, M. P., and G. E. Canali (1980). The influence of land use and topography on the hydrological and sedimentological behavior of basins in the basaltic region of South Brazil, in *The Influence of Man on the Hydrological Regime with Special Reference to Representative and Experimental Basins*, Publ. No. 130, International Association of Hydrological Sciences, Washington, DC, pp. 55–60.
- Borman, F. H., and G. E. Likens (1969). Biotic regulation of particulate and solution losses from a forested ecosystem, *Biosciences* 19(7):600–610.
- Boyce, R. C. (1975). Sediment routing with sediment delivery ratios, in *Present and Perspective Technology for Predicting Sediment Yields and Sources*, ARS-S40, U.S. Department of Agriculture, Washington, DC, pp. 61–65.
- Brant, G. H., et al. (1972). *An Economic Analysis of Erosion and Sediment Control for Watersheds Undergoing Urbanization*, Dow Chemical Co., Midland, MI.
- Brown, G. W. (1985). Controlling nonpoint source pollution from silvicultural operations: What we know and don't know, in *Perspectives on Nonpoint Source Pollution*, Proceedings, National Conference, Kansas City, EPA 440/S-85-001, U.S. Environmental Protection Agency, Washington, DC, pp. 332–334.
- Camp, T. R. (1947). Sedimentation and the design of settling tanks, *Trans. ASCE* 3:895–958.
- Chase, R. R. P. (1979). Settling behavior of natural aquatic particulates, *Limnol. Oceanogr.* 24:1417–1426.
- Chung, M., F. A. Roth, and E. V. Hunt (1982). Sediment production under various forestsite conditions, in *Recent Development in the Explanation and Prediction of Erosion and Sediment Yield*, D. E. Walling, ed., Publ. No. 137, International Association of Hydrological Sciences, Washington, DC, pp. 13–23.
- Clark, E. H., Jr., J. A. Haverkamp, and W. Chapman (1985). *Eroding Soils. —The Off-Farm Impacts*, The Conservation Foundation, Washington DC.
- Colbeck, S. C. (1981). The simulation of the enrichment of atmospheric pollutants in snow cover runoff, *Water Resour. Res.* 17(5):1383–1388.
- Committee on International Soil and Water Research and Development (1991). *Toward Sustainability. Soil and Water Research Priorities for Developing Countries*, National Academy Press, Washington, DC.
- Dong, A., G. Chesters, and G. V. Simsiman (1983) Soil dispersibility, *Soil Sci.* 136(4):208–212.
- Dong A., G. Chesters, and G. V. Simsiman (1984). Metal composition of soil, sediment, and urban dust and dirt samples from the Menomonee River Watershed, Wisconsin, U.S.A., *Water, Air, Soil Pollut.* 22:257–275.
- Dong, A., G. V. Simsiman, and G. Chesters (1983). Particle size distribution and phosphorus levels in soil, sediment and urban dust and dirt samples from the Menomonee River Watershed, Wisconsin, *Water Res.* 17(5):569–577.
- Einstein, H. A. (1950). *The Bed Load Function for Sediment Transportation in*

- Open Channel Flow*, Tech. Bull. No. 1026, U.S. Department of Agriculture, Washington, DC.
- Fleming, G., and K. M. Leytham (1976). The hydrologic and sediment processes in natural watershed areas, in *Proceedings, Third Federal Interagency Sedimentation Conference*, Denver, CO, Water Resources Council Sedimentation Committee, Washington, DC, pp. 1-232-1-246.
- Foster, G. R., and L. D. Meyer (1972a). A closed form erosion equation for upland areas, in *Sedimentation*, H. W. Shen, ed., Fort Collins, CO.
- Foster, G. R., and L. D. Meyer (1972b). Transport of soil particles by shallow flow, *Trans. ASAE* **13**(1):99-102.
- Foster, G. R., F. Lombardi, and W. C. Moldenhauer (1982). Evaluation of rainfall-runoff erosivity factors for individual storms, *Trans. ASAE* **25**(1):124-129.
- Foster, G. R., L. D. Meyer, and C. A. Onstad (1977). An erosion equation devised from basic erosion principles, *Trans. ASAE* **20**:678-682.
- Foster, G. R., et al. (1981). Estimating erosion and sediment yield on field-sized areas, *Trans. ASAE* **24**(5):1253-1262.
- Foster, G. R., R. A. Young, and W. H. Neibling (1985). Sediment composition for nonpoint source pollution analyses, *Trans. ASAE* **28**(1):133-139.
- Frederiksen, R. L. (1970). *Erosion and Sedimentation Following Road Construction and Timber Harvest on Unstable Soils in Three Small Western Oregon Watersheds*, Research Paper, PNW 104, U.S. Forest Service, Washington, DC.
- Free, M. H., C. A. Onstad, and H. N. Holtan (1975). *ACTMO—An Agricultural Chemical Transport Model*, Report No. ARS-H-3, U.S. Department of Agriculture, Washington, DC.
- Goldman, S. J., K. Jackson, and T. A. Bursztynsky (1986). *Erosion & Sediment Control Handbook*, McGraw-Hill, New York.
- Golubev, G. N. (1982). *Soil Erosion and Agriculture in the World: An Assessment and Hydrological Implications*, Publ. 137, International Association of Hydrological Science, Washington, DC.
- Gottschalk, L. C. (1964). Sedimentation. Part I. Reservoir Sedimentation, in *Handbook of Applied Hydrology*, V. T. Chow, ed., McGraw-Hill, New York.
- Hadley, R. F., and L. M. Shown (1976). Relation of erosion to sediment yield, in *Proceedings, Third Federal Inter-agency Sedimentation Conference*, Denver, CO, U.S. Water Resources Council, Washington, DC, pp. 1-132-1-139.
- Haldin S., A. Rodhe, and B. Bjurman (1990). Urban storm water transport and wash-off of Cesium-137 after the Chernobyl accident, *Water, Air Soil Pollut.* **49**:139-158.
- Holberger, R. L., and J. B. Truett (1976). Sediment yield from construction sites, in *Proceedings, Third Federal Inter-agency Sedimentation Conference*, Denver, CO, U.S. Water Resources Council, Washington, DC, pp. 1-47-1-58.
- Johannessen, N., and A. Henriksen (1978). Chemistry of snowmelt—Changes in concentration during melting, *Water Resour. Res.* **14**(4):615-619.
- Knisel, W. G. (1980). *CREAMS: A Field Scale Model for Chemicals, Runoff, and Erosion from Agricultural Management Systems*, U.S. Department of Agriculture, Conservation Research Report No. 26, Washington, DC.

- Krenkel, P. A., and V. Novotny (1980). *Water Quality Management*, Academic Press, New York.
- Langbein, W. B., and S. A. Schumm (1958). Yield of sediment in relation to mean annual precipitation, *Trans. Am. Geophys. Union* **39**:1076–1084.
- Lee, D. Y., S. W. Kang, and W. Lick (1981). The entrainment and deposition of fine-grained sediments, *J. Great Lakes Res.* **7**:224–233.
- Leopold, L. B. (1968). Hydrology for urban planning—a guidebook on the hydrologic effects of urban land use, *Geol. Surv. Circular 584*, U.S. Geological Survey, Washington, DC, pp. 1–18.
- Leopold, L. B., M. G. Wolman, and J. P. Miller (1964). *Fluvial Processes in Geomorphology*, W. H. Freeman, San Francisco.
- McElroy, A. D., et al. (1976). Loading functions for assessment of water pollution from nonpoint sources, EPA 600/2-76-151, U.S. Environmental Protection Agency, Washington, DC.
- Mehta, A. J., et al. (1989). Cohesive sediment transport. I: Process description, *J. Hydraul. Eng. Div. ASCE* **115**(8):1076–1093.
- Migniot, C. (1968). A study of physical properties of different very fine sediments and their behavior under hydrodynamic action, *La Houille Blanche* **7**:591–620.
- Mutchler, C. K., and A. J. Bowie (1976). Effect of land use on delivery ratios, in *Proceedings, Third Federal Inter-agency Sedimentation Conference*, Denver, CO, U.S. Water Resources Council, Washington, DC, pp. 1-11–1-21.
- Nearing, M. A., G. R. Foster, L. J. Lane, and S. C. Finkner (1989). A process-based erosion model for USDA-Water Erosion Prediction Project technology, *Trans. ASAE* **32**(5):1587–1593.
- Negev, M. (1967). A sediment model on a digital computer, *Tech. Rep. No. 62*, Dept. of Civil Eng., Stanford University, Palo Alto, CA.
- Noble, E. L. (1965). Sediment reduction through watershed rehabilitation, in *Proceedings, Federal Inter-agency Sedimentation Conference 1963*, Misc. Publ. 920, U.S. Department of Agriculture, Washington, DC, pp. 114–123.
- Nordin, C. F. (1962). Study of channel erosion and sediment transport, *J. Hydraulic Div. ASCE* **90**(Hy4):172–192.
- Novotny, V. (1980). Delivery of suspended sediment and pollutants from nonpoint sources during overland flow, *Water Resour. Bull.* **16**(6):1057–1065.
- Novotny, V., and G. Chesters (1981). *Handbook of Nonpoint Pollution: Sources and Management*. Van Nostrand Reinhold, New York.
- Novotny V., and G. Chesters (1989). Delivery of sediment and pollutants from nonpoint sources: A water quality perspective, *J. Soil. Water Conserv.* **44**(6):568–576.
- Novotny, V., and G. Bendoricchio (1989). Water quality: Linking diffuse pollution and deterioration, *Water Environ. Technol. WPCF* **1**(3):400–407.
- Novotny V., G. V. Simsim, and G. Chesters (1986). Delivery of pollutants from nonpoint sources, in *Proceedings, Symposium on Drainage Basin Sediment Delivery*, Internat. Assoc. Hydrol. Sci., Publ. No. 159, Wallingford, England.
- Novotny, V., et al. (1979). *Simulation of Pollutant Loadings and Runoff Quality*, EPA 905/4-79-029, U.S. Environmental Protection Agency, Chicago, IL.
- Ostry, R. C. (1982). Relationship of water quality and pollutant loads to land uses

- in adjoining watersheds, *Water Resour. Bull.* **18**(1):99–104.
- Overton, D. E., and M. E. Meadows (1976). *Stormwater Modeling*, Academic Press, New York.
- Parker, G. (1982). Condition for the ignition of catastrophically erosive turbidity currents, *Marine Geol.* **46**:307–327.
- Partheniades, E. (1965). Erosion and deposition of cohesive soils, *J. Hydraul. Div. ASCE* **91**:105–139.
- Partheniades, E. (1971). Unified view of wash load and bed material load, *J. Hydraul. Div. ASCE* **103**(HY9):1037–1057.
- Patrick, R. (1975). Some thoughts concerning correct management of water quality, in *Urbanization and Water Quality Control*, W. Whipple, ed., American Water Resources Association, Bethesda, MD.
- Piest, R. F., L. A. Kramer, and H. G. Heinemann (1975). Sediment movement from loessial watersheds, in *Proceedings, Present and Prospective Technology for Predicting Sediment Yield and Sources*, ARS S40, U.S. Department of Agriculture, Washington, DC, pp. 162–176.
- Ports, M. A. (1973). Use of universal soil loss equation as a design standard, Water Resources Engineering Meeting, ASAE, Washington, DC.
- Ports, M. A. (1975). Urban Sediment Control Design Criteria and Procedures, paper presented at the Winter meeting, ASAE, Chicago, IL.
- Renard K., G. Foster, G. A. Weesies, and J. P. Porter (1991). RUSLE—Revised universal soil loss equation, *J. Soil. Water Conserv.* **46**(1):30–33.
- Rhoades, E. D., N. H. Welch, and G. A. Coleman (1975). Sediment-yield characteristics from unit source watersheds, in *Present and Prospective Technology for Predicting Sediment Yield and Sources*, ARS Publ. No. ARS-60, U.S. Department of Agriculture, Washington, DC, pp. 125–129.
- Ritchie, J. C., and J. R. McHenry (1975). Fallout of Cs-137: A tool in conservation research, *J. Soil. Water Conserv.* **40**:283–386.
- Ritchie, J. C., and J. R. McHenry (1978). Fallout Cesium-137 in cultivated north central United States watersheds, *J. Environ. Qual.* **7**(1):40–44.
- Ritchie, J. C., J. A. Spraberry, and J. R. McHenry (1974). Estimating soil erosion from the redistribution of fallout ¹³⁷Cs, *Soil. Sci. Soc. Am. Proc.* **38**:137–139.
- Roehl, J. W. (1962). *Sediment Source Areas, Delivery Ratios and Influencing Morphological Factors*, Publ. No. 59, Internat. Assoc. Hydrol. Sci., pp. 202–213.
- Sharma, K. D., and P. C. Chatterji (1982). Sedimentation in Nadis in the Indian arid zone, *Hydrol. Sci. J.* **27**:345–352.
- Shen, H. W. (1982). Some basic concepts on sediment transport in urban storm drainage systems, in *Urban Stormwater Quality, Management and Planning*, B. C. Yen, ed., Water Resources Publications, Littleton, CO, pp. 495–501.
- Shields, A. (1936). “Änwendung der Ähnlichkeits-mechanik und der Turbulenzforschung auf der Geschiebebewegung,” Preuss. Versuchsanstalt für Schiffbau, Berlin.
- Steward, B. A., et al. (1975). Control of pollution from cropland, U.S. EPA Report No. 600/2-75-026 or U.S.D.A. Rep. No. ARS-H-5-1, Washington, DC.

- Tesarik, I., and J. Vostřil (1970). Hindered settling and thickening of chemically precipitated flocs, in *Proceedings, Fifth International Conference IAWPR*, Pergamon Press, Oxford.
- Ursic, S. J., and F. E. Dendy (1965). Sediment yields from small watersheds under various land uses and forest covers, in *Proceedings, Federal Inter-agency Sedimentation Conference 1963*, USDA Misc. Publ. 970, Washington, DC, pp. 47–52.
- Vanoni, V. (1975). *Sedimentation Engineering*, ASCE, New York.
- Wade, J. C., and E. O. Heany (1978). Measurement of sediment control impact on agriculture, *Water Resources Res.* **14**:1–8.
- Walling, D. E. (1979). Hydrological processes, in *Man and Environmental Processes*, K. J. Gregory and D. E. Walling, eds., Butterworth, London.
- Walling, D. E. (1983). The sediment delivery problem, *J. Hydrol.* **65**:209–237.
- Walling, D. E., and P. Kane (1983). Temporal variation of suspended sediment properties, *Recent Developments in the Explanation and Prediction of Erosion and Sediment Yield*, Proceedings, Exter Symposium, IAHS Publ. No. 137, Washington, DC, pp. 409–419.
- Walling, D. E., and B. W. Webb (1983). Patterns of sediment yield, in *Background to Paleohydrology*, K. J. Gregory, ed., Wiley, New York.
- Wilkin, D. C., and S. J. Hebel (1982). Erosion, redeposition and delivery of sediment to midwestern streams, *Water Resour. Res.* **18**(4):1278–1282.
- Williams, J. R. (1975). Sediment routing for agricultural watersheds, *Water Resour. Bull.* **11**(5):965–974.
- Williams, J. R., and H. D. Berndt (1977). Sediment yield prediction based on watershed hydrology, *Trans. ASAE* **20**:1100–1104.
- Wischmeier, W. H. (1972). Estimating the soil loss equation's cover and management factor for undisturbed areas, Proceedings, Sediment-Yield Workshop, U.S. Department of Agriculture Sedimentation Laboratory, Oxford, MS.
- Wischmeier, W. H. (1976). Use and misuse of the universal soil loss equation, *J. Soil Water Conserv.* **31**(1):5–9.
- Wischmeier, W. H., and D. D. Smith (1965). *Predicting Rainfall-Erosion Losses from Cropland East of Rocky Mountains*, USDA Agricultural Handbook No. 282, Washington, DC.
- Wischmeier, W. H., C. B. Johnson, and B. V. Cross (1971). "A soil erodibility nomograph for farmland and construction sites," *J. Soil Water Conserv.* **26**: 189–193.
- Wolman, M. G. (1977). Changing needs and opportunities in the sediment field, *Water Resour. Res.* **13**:50–54.
- Xingkui, W., and Q. Ning (1989). Turbulence characteristics of sediment laden flows, *J. Hydrol. Eng. ASCE* **115**(6):781–800.
- Yalin, M. S. (1963). An expression for bed load transportation, *J. Hydraul. Div. ASCE* **89**:221–250.
- Ziegler, C. K., and W. Lick (1988). The transport of fine-grained sediments in shallow waters, *Environ. Geol. Water Sci.* **11**(1):123–132.

6

Pollutant Interaction with Soils and Sediments

Future historians may be amazed by our distorted sense of proportions. How could intelligent beings seek to control a few unwanted species by a method that contaminated the entire environment and brought the threat of disease and death even to their own kind?

Rachel Carson, *Silent Spring*

All the human and animal manure, if returned to land instead being thrown into the sea, would suffice to nourish the world.

Victor Hugo, *Les Miserables*

Soils can retain, modify, decompose, or adsorb (immobilize) pollutants. Each year an enormous amount of organic material, atmospheric contaminants, and liquid and solid wastes are deposited and incorporated into soils. As a matter of fact, the focus of early "zero discharge" efforts after the passage of the 1972 Water Pollution Control Act Amendments was to shift the burden of residual waste discharges from water to soil.

Biodegradable organic materials deposited into soils are decomposed largely into such safe products as CO₂, methane, nitrogen, and phosphorus compounds. Due to the large number of microorganisms residing in typical fertile top soils, the decomposition processes under suitable environmental conditions represent the best natural recycling process. A properly balanced and managed soil system does not represent a great threat to water quality. In a balanced system, most of the nutrients and organic matter added to the soil, in amounts normally applied to increase crop

production, will remain in the upper soil layer and/or will be taken up by the crops.

However, it was pointed out by a panel of distinguished soil scientists (Stigliani, 1991) that harmful pollutants due to man's activities had been gradually accumulating in soils and sediments for the past 2000 years. Early sources of such soil pollution were primarily metal mining operations. Since the beginning of the Industrial Revolution, the scale and pace of environmental contamination due to industrial, agricultural, commercial, and domestic activities has accelerated, and in the last 40 years has reached a pace that may be threatening to the health and existence of the human population, at least in some stressed areas of the world.

As stated previously, man always had a tendency to return waste, by-products of living and production processes, and the dead first to the soil and then to the water. The capacity of soil to assimilate pollutants was considered infinite, and discharging waste to soils was deemed beneficial (note the Victor Hugo quote at the beginning of the chapter). However, this capacity though large, is limited.

Today, large amounts of commercial chemical fertilizers are applied to agricultural and grassed urban lands in the United States and elsewhere. Typical application rates range from 20 to 200 kg N/ha and from 10 to 50 kg P/ha. To control pests and weeds farmers and urban dwellers have been using chemical pesticides in amounts that are environmentally damaging and defying the purpose of control. For example, to maintain a "perfect" suburban lawn and eradicate a few dandelions a typical suburban homeowner in the United States uses chemical pesticides and fertilizers in excessive, environmentally damaging quantities. During the 1960 to 1990 period, in order to sustain higher crop yields, agricultural cooperatives in the central and eastern European countries were applying chemicals at such excessive rates that most of the receiving surface- and ground-water bodies in the region had been severely damaged with a consequent impairment of their beneficial uses (for example, unsuitability of water resources for water supply, recreational uses, and severe ecological damages).

Due to the capacity of soils and sediments to store and immobilize toxic chemicals in so-called chemical sinks, the direct effects of pollution may not be initially directly manifested. However, there is now direct evidence that the capacity of soils to safely dispose of pollutants has been exhausted in some parts of the world. Excessive contamination of surface- and ground-water bodies by nitrates, pesticides, and other chemicals is a result of the failure of the soils to which these chemicals were applied to

retain them, modify them, and transfer them to their target media (crops for fertilizing chemicals, weeds and insects for pesticides).

Because loading of the chemicals to the environment may occur long before the adverse consequences are noticed, scientists have used the term *chemical time bomb* to describe the present and future threats by the past, present, and future excessive uses of various chemicals and waste applications to soils.

A chemical time bomb (CTB) is a concept that refers to a chain of events resulting in the delayed and sudden occurrence of harmful effects due to the mobilization of chemicals stored in soils and sediments in response to slow alteration of the environment (Stigliani, 1991; Stigliani et al., 1991).

Typically, the loading of soils by pollutants occurs at lower rates but is widespread (diffuse pollution). In some other instances, chemical loading was very high and localized, such as in the cases of hazardous waste disposal sites and landfills. In many instances, when the delayed environmental effects of the "overload" of soils and sediments with pollutants are manifested, the impact is widespread and may be devastating. Forest diebacks in Europe in the early 1980s, due at least in part to soil acidification by acid rainfall and atmospheric deposition, is a recent example of sudden, unanticipated, and delayed response. Similarly, the rapid explosive eutrophication of lakes (see the example of Lake Balaton in Chapter 1) and ground-water contamination by nitrates in Central Europe and elsewhere are a result of the excessive application of fertilizing chemical at rates that far exceed plant uptake.

When the waste-assimilative or buffering capacity of soils and sediments is gradually exceeded, the result is CTB. The buffering (waste-assimilative) capacity of soils is due to the microbial degradation of chemicals and pollutants, plant uptake and removal with crops, immobilization by adsorption and complexation on soil particles and colloids, chemical reactions, volatilization, and other processes by which pollutants are removed, degraded, and immobilized.

Contrary to the sudden occurrence of the time bomb, which is the effect of an exceedence of the waste-assimilative (buffering) capacity of soils and surface-water bodies, remediation and restoration of damaged ecosystems is gradual and may last years and even centuries. There are no lasting "quick fixes."

As the contaminated soils are eroded, aquatic sediments are also being contaminated at a rather fast pace. Aquatic sediment may also interact with the pollutants dissolved in water in the same manner as soil would

with pollutants contained in soil moisture. As a matter of fact, the processes of pollutant interactions with soils and aquatic and wetland sediments are either the same or similar.

Soil and its buffering capacity is the only barrier between surface contamination and pollutant deposition and ground-water resources for which the buffering is either zero or very small. This difference between the buffering capacities of topsoils and ground water is due to a typically rich microbial population residing in the topsoil layer, higher content of organic matter, a greater proportion of weathered small-grain minerals, such as clays, different chemical environment, and other factors.

The processes and pollutant interactions in aquatic sediments and wetland substrates are similar to those in saturated (wet) soils. Therefore, both media are discussed in this chapter.

The Soil Profile

The soil profile is divided into three layers called horizons A, B, and C (Fig. 6.1). A typical profile may also contain a thin layer of decaying

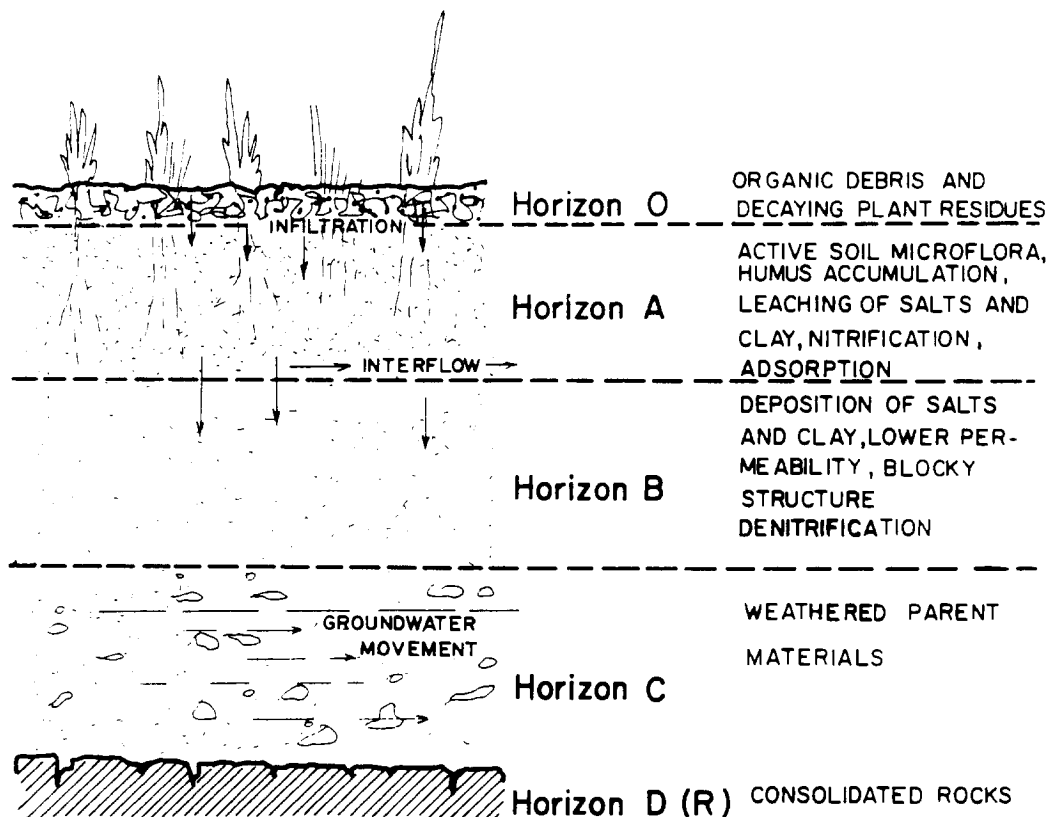


FIGURE 6.1. Soil profile.

organic debris with little soil. The surface layer of organic debris is called horizon O.

The A horizon, usually several centimeters to a fraction of a meter thick, is the soil layer of the greatest concern, since roots, soil microorganisms, and organics can be found there in great densities. It is also a layer of considerable leaching. The B horizon, underlying the A horizon, is a subsoil where most of the leached salts, chemicals, and clay may deposit. It usually has little organic matter and few plant roots (only large plants—macrophytes—have root systems penetrating into subsoils). The C horizon extends from the bottom of the B horizon to the top of the parent bedrock from which the soil evolved by weathering.

The A horizon is of considerable importance in diffuse pollution studies since it is the soil layer where most of the adsorption and biochemical degradation of pollutants takes place. The microbiological processes by which pollutants and nutrients (nitrogen and phosphorus) are decomposed or transformed are mostly confined to the A horizon. Only soluble (mobile) pollutants can penetrate into deeper soil zones and eventually pollute ground water.

Soil nomenclature and soil maps provide a wealth of information on the types of soils, their texture, composition, and properties. The soil texture triangle presented in the preceding chapter (Fig. 5.26) and the U.S. Soil Conservation Service soil maps (Fig. 6.2) are for delineating the distribution of soils. In the classification of soil the name of the soil as reported in the soil map has two parts: (a) a local name that also includes textural classification, and (b) slope. For example, in the soil map of southeastern Wisconsin a code OuB signifies Ozaukee silt loam in slope category B (2% to 6% slope).

Soil (Sediment) Organic Matter

Organic matter is an integral part of soils. The organic matter content varies from <1 to >40% in some organic soils, feedlot soils, wetland, and aquatic sediments. Organic matter content is usually reported as organic matter (%) or organic carbon (%). $\text{Organic matter (\%)} \approx 1.67 \times \text{organic carbon (\%)}$.

The organic content of soils and sediments, often called humus, is a product of the biodegrading processes by microorganisms. It is rich in nutrients and remains for long periods as an important food supply for microorganisms. A significant part of the organic matter is not biodegradable.

Almost all of the organic matter in soils is contained in the A and O horizons of the soil profile. Organic-rich aquatic sediments may be much

deeper. As to their organic content, soils are divided into mineral soils that have low organic fraction and organic soils. A similar classification can also be applied to aquatic sediments. However, for sediments the organic content is affected by the hydraulics of the body of water.

High organic content is not synonymous with fertility in soils. Peat (wetland) soils that have an organic fraction are not suitable for growing most crops (with the exception of rice) that require aerated drained soils, even though wetlands can produce high quantities of organic matter. When peat soils are drained, the excess organic matter is converted under aerobic conditions to CO_2 , and the soil's suitability for growing crops is improved. That was the major reason for draining wetlands during their conversion to agricultural use. Consequently, soil aeration is a factor affecting the soil organic content.

LOADING FUNCTIONS

The Concept of Enrichment

The transport of pollutants from topsoil is affected by the composition of the soil, slope, and other factors. Pollutant transport follows the three transport routes identified in Chapters 3 and 5, depending on whether the pollutants are associated with the sediment or are dissolved in soil water:

1. Soil erosion and overland transport will carry pollutants adsorbed or complexed by soil particulates. Their history of transport is similar (but not identical) to that of soil particles.
2. Pollutants contained in soil water can be transported either by surface runoff or by interflow. These pollutants are in the form of dissolved compounds or ions. For many compounds a dynamic equilibrium exists between the dissolved and adsorbed (complexed-particulate) fractions in soil water and in surface runoff).
3. Only dissolved or ionized pollutants can reach ground-water zones and reappear on the surface with the base-flow-ground-water discharge.

Many contaminants in soils (nutrients and organic chemicals) are subject to bacterial degradation, volatilization, and chemical breakdown. Those that will persist are classified as *conservative* materials, while those that change their total mass with time (excluding the effect of convection and dispersion) are *nonconservative*.

Figure 6.3 shows the major pathways of transport and major transformation processes of soil contaminants between the source and receiving bodies of water. Water is the primary transporting vector.

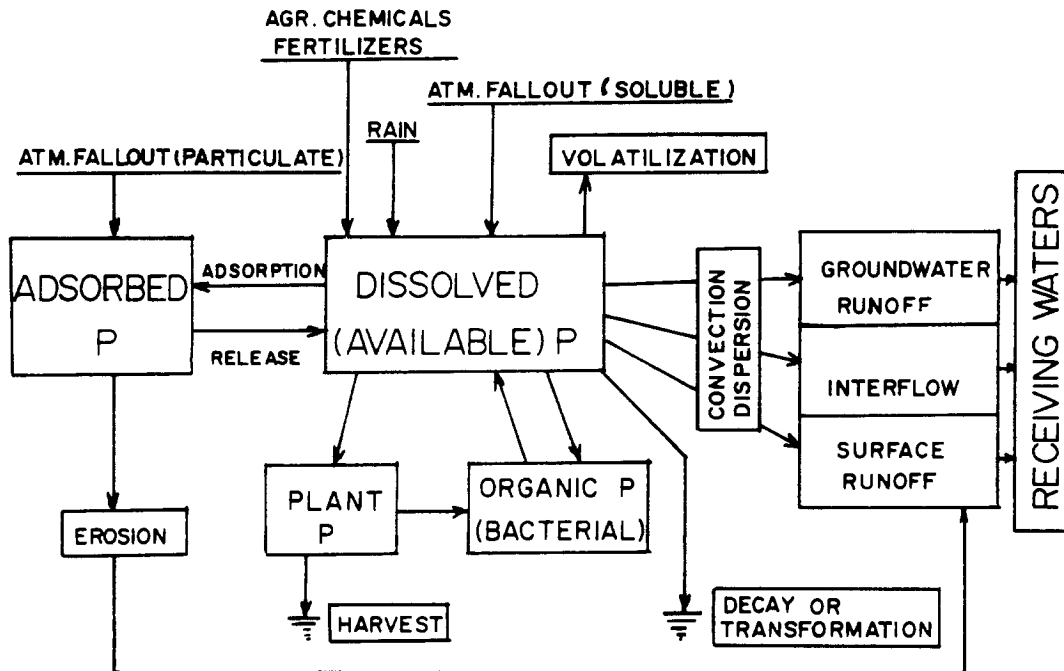


FIGURE 6.3. Pathways of contaminants through soil-environment-soil-contaminant interactions.

Since particulate pollutant transport is a part of sediment erosion and movement processes many studies and models use an arbitrary proportionality factor called the *potency factor* or *transmission coefficient* to equate sediment loading to that of other contaminants. The relationship can be expressed by the equation

$$Y_i = p_i Y_s \tag{6.1}$$

where

Y_i = the loading or concentration of contaminant i

Y_s = the loading or concentration of sediment

p_i = the potency factor for the contaminant

The potency factor can be related to the concentration of the contaminant in the parent topsoil and the enrichment factor for the contaminant, that is,

$$p_i = S_{is} ER_i \tag{6.2}$$

where

S_{is} = the concentration of the contaminant

ER_i = the enrichment ratio for the contaminant between the source and the point of interest or watershed outlet

The enrichment ratio is then the concentration of the contaminant in the eroded material on the sediment in runoff divided by its concentration in the parent topsoil expressed on an over-dried basis.

The use of Equations (6.1) and (6.2) presumes knowledge of the soil-contaminant concentration, S_{is} , and the enrichment ratio, ER_i . The U.S. EPA Screening Procedure (Mills et al., 1985) for estimating pollutant loads from nonpoint sources is based on the preceding concept and the documentation contains information on estimating the needed parameter values. This publication appears to be the most comprehensive treatise on the subjects. Also the publication by McElroy et al. (1976) on which the EPA screening procedure is based contains some information on the magnitudes of potency factors, enrichment ratios, and pollutant content of parent soils. Information may be available for soil content of organic matter, nutrients, and some other contaminants such as metals (Tables 6-1, 6-2, and 6-3); however, the nature of the enrichment factor is not well understood and its estimation is, at best, an approximation. For contaminants associated with fine fractions and particulate organic matter, the enrichment ratio may be inversely related to the delivery ratio as shown for clays in the preceding chapter. Table 6-3 shows the enrichment ratios

TABLE 6.1 Metal Concentrations ($\mu\text{g/g}$) in Surficial Materials in the United States

Element	Geometric Means Arithmetic Analysis		Conterminous United States	West of the 97th the Meridian	East of 97th the Meridian
	Average	Range			
As		0-1000			
Ba	534	15-5000	430	560	30
Cd		0-20			
Ce	86	<150-300	75	74	78
Cr	53	1-1500	37	38	86
Co	10	<3-70	7	8	7
Fe	25,000	100-100,000	18,000	20,500	15,000
Pb	20	<10-700	14	18	10
Mn	560	<1-7000	340	389	385
Ni	20	<5-700	14	16	13
Hg ^a		0.01-0.03			
Cu ^a		2-100			
Se ^a		0.1-2			

Source: After Shacklette et al. (1971).

^a After Lindsay (1979).

TABLE 6.2 Concentrations of Toxic Metals (in $\mu\text{g/g}$) in Unpolluted Parent Soils and Sediments in Rhine River Basin

Metal	Soils	Subrecent Rhine Sediments	Lacustrine Sediments
Cd	0.82	0.3	0.40
Co	12	16	16
Cr	84	47	62
Cu	25.8	51	45
Hg	0.1	0.2	0.35
Mn	760	960	700
Ni	33.7	46	66
Pb	29.2	30	34
Zn	59.8	115	118

Source: From Salomons and Förstner (1984).

TABLE 6.3 Enrichment of Suspended Sediment by Clay and Associated Pollutants

Constituent ^a	Soil	Suspended Sediment	Enrichment Ratio
Clay	27	91	3.4
Total phosphorus	810	1700	2.1
Lead (Pb)	19	39	2.0
Cadmium (Cd)	0.3	0.31	1.0
Zinc (Zn)	69	280	4.0
Copper (Cu)	25	45	1.8
Aluminum (Al)	22,000	49,000	2.2
Iron (Fe)	21,000	46,000	2.2
Manganese (Mn)	700	730	1.0
Chromium (Cr)	29	56	1.9
Nickel (Ni)	17	45	2.6

Sources: Based on data from Dong, Simsiman, and Chesters (1983), and Dong, Chesters, and Simsiman (1984).

^aClay is in percent and phosphorus and metals are in $\mu\text{g/g}$ of sediment.

of contaminants determined from the soil and sediment data of the Menomonee River experimental watershed.

Equations (6.1) and (6.2) are static, that is, they allow neither the prediction of future conditions nor the evaluation of the effects of future management practices. However, in some cases when a contaminant is conservative and immobile, approximate concentrations can be estimated from the soil-contaminant mass balance.

Since almost all particulate contaminants are usually associated with fine soil particles, and since erosion and sediment transport are selective for fine materials, a simple relation or guidelines for determining the potency factor apparently do not exist. Furthermore, the quantity is highly variable, depending on soil characteristics, storm and overland flow characteristics, channel flow hydraulics, the presence of sediment, and pollutant sinks, such as grassed buffer strips, riparian wetlands, forest litter, and the nature of the contaminant. Also, the sorption characteristics and the capacity of soil particles to retain pollutants are different, depending on whether the particles are in soil or are a part of the sediment washload and bedload in streams and other surface-water bodies (McCalister and Logan, 1978; Salomons and Förstner, 1984). Following are several processes that result in higher concentrations of contaminants in the eroded material and aquatic sediments than in the parent soil and, consequently, enrichment ratios of greater than one:

1. Selective removal of fine materials with higher pollutant concentrations than the remainder of the soil material, which is expressed as the clay enrichment ratio (see Chapter 5).
2. Diffusion of pollutants and salts from the topsoils into surface runoff.
3. Desorption of pollutants from soil particles due to low dissolved concentrations in runoff water.
4. Flotation of low-density materials such as organic components from soil into surface water.
5. Deposition of coarse fractions containing few sorbed pollutants during overland and channel flow and in man-made depositional facilities (retention ponds, buffer strips) that are more effective for removal of coarse heavier particles.

The transport and loading of dissolved contaminants is more complex because they can be transported by each of the three hydrological components, namely, surface runoff, interflow, and ground water. It is therefore evident that a simplistic approach to the determination of pollutant loadings from pervious surfaces (soils) may fail to provide adequate results, and prognoses using such approaches must be treated with caution and accepted only as rough estimates. Fortunately, several models have been developed in the past that in a comprehensive manner describe the transport and interactions of several typical contaminants in the soil and water environments. The U.S. EPA's computerized loading procedures (Mills et al., 1985) present methodologies and numerous supporting parameters for estimating loadings from diffuse sources and their impact on receiving bodies of water.

Example 6.1: Estimating Pollutant Concentration in Soils from Mass Balance

Application of 100 kg/ha of a conservative chemical that is immobile in soils was made to an area. Estimate the approximate concentration of the chemical in the soil particles if the pollutant is assumed to be uniformly distributed throughout the top 30 cm of the soil (approximate till depth). The specific density of the soil is 1.8 g/cm³. Hence

$$S_{is} = \frac{\text{Chemical application mass}}{\text{Soil mass}} = \frac{100 \text{ (kg/ha)} \times 10^9 \text{ (\mu g/kg)}}{30 \text{ cm} \times 10^8 \text{ (cm}^2\text{/ha)} \times 1.8 \text{ (g/cm}^3\text{)}} \\ = 18.5 \text{ }\mu\text{g/g}$$

Detachment and Enrichment of Runoff by Soil Organic Matter

Most of the organic matter in soils is in a particulate form and, as such, transport of organic matter from soils and sediments can be related to erosion by applying Equations (6.1) and (6.2). However, values of the enrichment factor are difficult to assess. Massey and Jackson (1952) found that the value of the enrichment ratio (ER) for organic matter increases with sediment concentration in runoff and the rate of erosion. The average enrichment ratio for three silt loam soils in Wisconsin measured by Massey and Jackson (1952) was 2.1. Young and Onstad (1976) proposed the following relationship for the enrichment ratio (ER_{or}) of organic matter based on the regression analyses of Indiana and Minnesota soils

$$ER_{or} = \frac{0.3}{(\% \text{ organic matter})} + 1.08 \quad (6.3)$$

The organic content increases linearly with finer textures, so Equation (6.3) indicates lower ER values for clayey soils and higher values for sandy soils. Since the specific density of organic particles is much less than mineral soil particles, the ER may increase significantly during overland and channel flow.

The organic content of soils lost in agricultural erosion can be restored by the application of organic fertilizers-manure, man-made organic mixtures, sewage sludges, and by keeping plant residues in place. Heavy reliance on fertilizing chemicals only will degrade the soil quality. Increased organic content improves soil permeability and reduces erodibility. As will be shown in later sections of this chapter, a significant portion of the organic matter in fertile soils consists of soil microflora.

Detachment of Dissolved Chemicals: Mixing Layer

Many chemicals, including phosphorus, metals, ammoniacal nitrogen, and organic chemicals exist in soils and sediments in two or even three phases (see the next section for a detailed discussion). The phases are solid—adsorbed or precipitated chemicals, liquid chemicals dissolved or dissociated in pore water, and gaseous chemicals in the vapor phase in aerated soils. When runoff is generated on the soil surface by rainfall, solid-phase chemicals move with eroded sediment, while liquid-phase chemicals in the upper surface layer mix with runoff and are in this way leached from the soil.

Some hydrologic models of soil–water–chemical interaction assume that rainfall and runoff interact within a thin zone of the surface soil. The rainfall is assumed to mix completely and uniformly with the dissolved chemicals in the upper soil zone. The amount of a soluble chemical is then divided proportionately between the soil zone water storage, infiltration, and surface runoff in the water retained in each mode. The thickness of the mixing zone can be obtained by calibration or be fixed. In the ARM model (Donigian et al., 1977) the thickness of the upper soil mixing zone varied between 1.6 and 6 mm, with the smaller values giving better results under field conditions. In the CREAMS-GLEAMS model (Knisel, 1980) the thickness of the mixing depth was selected as 1 cm; however, it is assumed that only a fraction of the chemical in the mixing zone interacts with rainfall water.

Sharpley, Ahuja, and Menzel (1981) and Ahuja et al. (1982) measured and calculated the mixing depth for soils within the range of 2 to 6 mm with typical values between 2 and 3 mm. As expected, the thickness of the mixing depth was proportional to rainfall intensity, and decreased below 1 mm when protective vegetative covers were present.

For aquatic sediments the thickness of the upper mixing zone is affected by the action of benthic organisms. Typically for modeling purposes, the thickness of the interstitial sediment–water mixing zone is in centimeters.

THE FATE OF CONTAMINANTS IN SOILS: THREE-PHASE APPROACH

Chemicals or other pollutants such as nutrients may exist in soils, sediments, and sediment-laden waters in several phases. They can be precipitated and/or strongly adsorbed to particles (solid phase), be dissolved or dissociated in water or soil moisture (liquid phase), volatilize or be gasified. For example, ammonium may exist in soil adsorbed on soil particles, dissociated in water as an ammonia ion (NH_4^+) or as ammonium

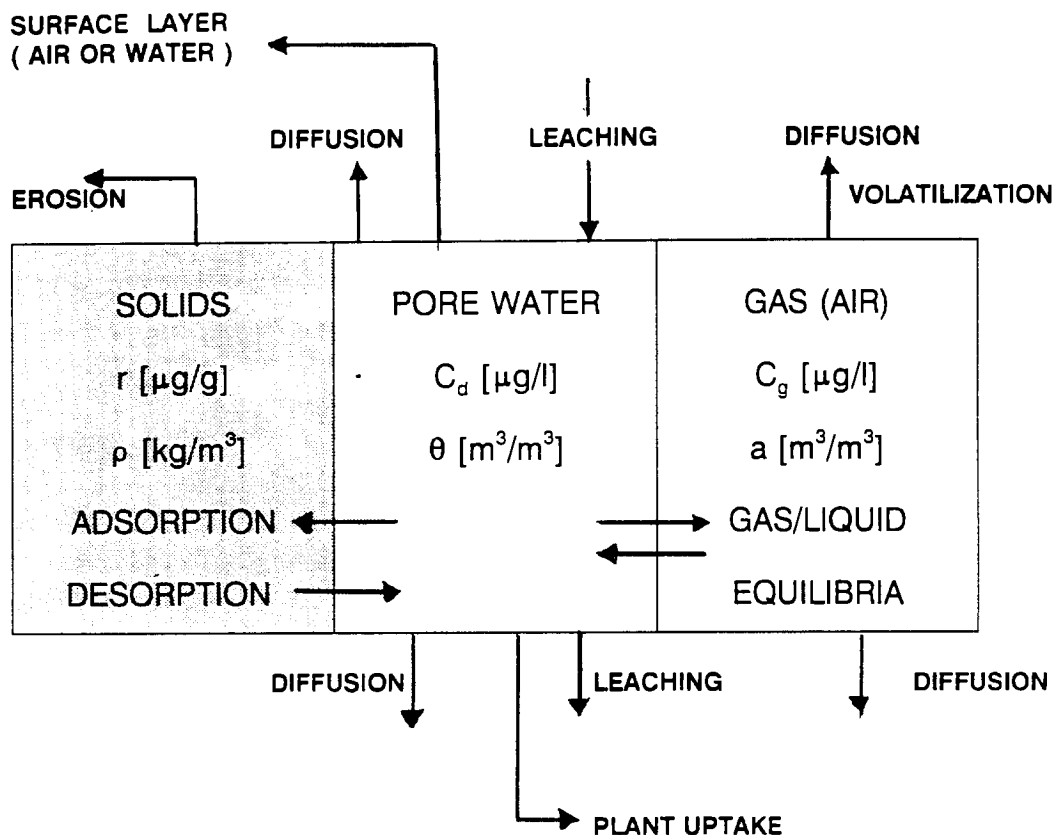


FIGURE 6.4. Compartmentalized mass balance concept of the distribution of a contaminant in soils and sediments. (After Burns et al., 1990.)

(NH_3). Generally, adsorbed or precipitated chemicals are considered immobile in soils and nonmoving sediments and biologically unavailable to plants and microorganisms. Dissolved or dissociated chemicals move with water, hence they are mobile and biologically available.

The fate of a chemical in soil and sediments is determined by the transformation processes. A chemical can volatilize to the atmosphere, be adsorbed, be taken by plants where it may or may not be degraded, and hence it might be returned to the soil-sediment in plant residue, be degraded by chemical, photochemical, and/or microbiological processes, be leached to lower depths by water movement and diffusion, be transported by erosion/runoff, and redeposited on land or reach surface water bodies. In many cases the fate of a particular chemical pollutant is determined by a number of processes (Chesters, 1986).

Figure 6.4 shows the processes that determine the fate of a single chemical in soil or a sediment environment. Typically, mass balance equations must be written for each compartment in order to estimate the overall fate of the chemical (Burns et al., 1990).

Sorption, Precipitation, and Partitioning Processes

The term *partitioning* implies division of the total pollutant mass between the particulate (soil particles) and dissolved (dissociated) fractions in soil or sediment pore water. Adsorption or complexation of the contaminant into the particulate form also immobilizes the contaminants and makes them biologically unavailable in most cases. Dissolved or dissociated (ionized) contaminants are mobile with soil and sediment pore water and can enter the tissues of animal and plant life. Division of the total pollutant mass into its particulate (immobile) and dissolved (mobile) fractions is thus very important in assessing the contaminant impact on the ecosystem and its toxicity and, conversely, it affects the waste-assimilative capacity of the soil-sediment ecosystem to safely accept and dispose of the pollutants and determines the amount of the contaminants that can leach to ground water. A report by Schnoor et al. (1987) describes the processes involving toxic metals and organics and also includes a long list of the numerical values of coefficients and parameters. A book by Salomons and Förstner (1984) is one of the most comprehensive treatises on the fate and transport of toxic metals in the ecosystems.

Generally, water-soluble compounds are weakly adsorbed on soil or sediment particles, hence they have a higher bioavailability and leach more easily into ground water. Due to the ease of leaching these compounds do not persist nor accumulate in soils, sediments, and the tissues of organisms. Water-insoluble compounds, on the other hand, are immobile in soils and sediments; however, they accumulate in soils and sediments and may bioaccumulate in organisms and biomagnify in the food chain (see Chapter 14).

The mobility of a chemical in soils and sediments is related to the so-called *octanol-water partitioning* of the chemical expressed by the *octanol/water partitioning coefficient*, K_{ow} . The coefficient is thus the accepted measure of the solubility of the chemical in water and, consequently, mobility in soils and sediments. The laboratory procedure for measuring K_{ow} is (Lyman, 1982; Schnoor et al., 1987):

1. A chemical is added to a mixture of pure octanol (a nonpolar solvent) and pure water (a polar solvent). The volume ratio of octanol and water is set at the estimated K_{ow} .
2. The mixture is agitated until equilibrium is reached.
3. The mixture is centrifuged to separate the two phases, and the phases are analyzed for the chemical.
4. K_{ow} is the ratio of the chemical concentration in the octanol phase to the chemical concentration in the water phase. From this experiment

K_{ow} is dimensionless, but it may be commonly reported as liters/kilogram.

It has been found that for nonionic chemicals and some other chemicals the adsorption characteristics (partitioning) may be correlated to the octanol partitioning coefficient, K_{ow} . The values of K_{ow} and other important parameters for some environmentally important chemicals (priority pollutants) are given in Table A1 in the Appendix. The report by Schnoor et al. (1987) has an extensive compilation of partition coefficients for many pollutants. The parameter K_{ow} is one of the most important determinants of the fate of a chemical in the soil-sediment environment and its bioavailability. The partitioning coefficients for a large number of polar potentially toxic contaminants have been compiled by the Risk Reduction Engineering Laboratory in a computerized data base.

Sorption results from a variety of different types of attractive forces between the dissolved molecules or ions and the sorbing material (Weber McGinley, and Katz, 1991). These forces are of a chemical, electrostatic, and physical nature. The most significant properties for sorption and desorption of a soil-sediment system are those related to the surface at which accumulation occurs. Quartz particles (sand) do not provide adequate adsorption sites and generally are not considered as sorbents. Clay and particulate organics, on the other hand, provide the most suitable adsorption. In addition, the soil-sediment properties that exert the greatest influence on sorption-desorption processes are pH, temperature, and moisture content (Chesters, 1986).

In precipitation reactions, the proportion of dissolved (ionized) and precipitated fractions of a contaminant is strongly affected by the pH and the presence of complexing compounds—ligands. The precipitating-complexing reactions are important for toxic metals, while organic chemical reactions with soil-sediment particles are mostly adsorption-desorption processes. The presence or absence of complexing ligands is affected also by the oxidation-reduction status in the soil or sediment environment. For example, in anoxic soils and sediments, which occurs primarily when the medium is saturated with water, insoluble metal-sulfide complexes are formed. Such complexes do not form in aerated soils or aerobic sediments, and other precipitating and often less favorable reactions with clays and organic matter dominate. Therefore oxidation-reduction status affects the precipitation process of metals. On the other hand, adsorption of pesticides is not generally affected by the presence or absence of oxygen, and they primarily interact with the soil organic matter. Phosphates and ammonia interact with clays and organic particulates.

The preferred mathematical form for describing the proportion be-

tween the dissolved (ionized) and adsorbed (complexed) particulate fractions is to relate the concentration of the contaminant adsorbed on the soil or sediment particles, r (in μg of contaminant per gram of soil, $\mu\text{g/g}$) to the equilibrium solution concentration, C_e (mg/l) at a fixed temperature (isothermic reaction). Several mathematical formulations of adsorption equilibria (isotherms) have been proposed from which the Langmuir and Freundlich isotherms are most widely accepted.

The Langmuir adsorption model is valid for monolayer adsorption, and is expressed in the form

$$r = \frac{Q^0 b C_e}{1 + b C_e} \quad (6.4)$$

where

Q^0 = the adsorption maximum at the fixed temperature ($\mu\text{g/g}$)

b = a constant related to the energy of net enthalphy of adsorption (l/mg or l/ μg)

r = adsorbed concentration of the contaminant ($\mu\text{g/g}$)

C_e = dissolved (free) concentration of the contaminant water ($\mu\text{g/l}$)

Although the Langmuir isotherm is assumed to be valid for monolayer adsorption, it adequately represents soil adsorption processes for such pollutants as phosphorus and many organic chemicals.

The Freundlich isotherm is useful if the energy term, b , in the Langmuir isotherm varies as a function of surface coverage, r . The Freundlich equation has the general form

$$r = K C_e^{1/n} \quad (6.5)$$

where K and n are constants.

Parameters describing the isotherm equation for various pollutants are statistical quantities. In some cases, laboratory adsorption studies may be available that can provide the magnitude of the adsorption coefficients. In these experiments, soil or sediment samples are put in aqueous solutions with different concentrations of the compounds until an equilibrium between the sorbed and dissolved fractions is reached. Thereafter, the sorbed fraction is $r = (C_{\text{initial}} - C_e)/\text{dry weight of soil}$. To obtain the Langmuir isotherm parameters $1/r$ is plotted versus $1/C_e$ on an arithmetic plot. For the slope of the line of best fit, slope = $1/(Q^0 b)$ and the intercept = $1/Q^0$.

Freundlich isotherm parameters are then obtained by plotting r versus C_e on log-log graph paper. The logarithmic intercept (when X -ordinate

equals unity) is K and the logarithmic slope ($sl = \log\{[r_1 - r_2]/[C_{e1} - C_{e2}]\}$), where 1 and 2 are arbitrarily chosen points on the line of best fit, equals $1/n$.

Linear isotherm. For low concentrations of contaminants, the Langmuir and Freundlich isotherms can be simplified to

$$r = \Pi c_e \quad (6.6)$$

where Π = partition coefficient, l/g.

By comparing the Langmuir isotherm with the linear isotherm for low concentrations of the chemical, it can be seen that $\Pi \approx Q^0 \times b$. The total concentration of the pollutant is then a sum of the dissolved (c_d) and particulate concentrations (c_p). Hence, if $c_p = m_{ss} \times r$ where m_{ss} is the concentration of solids in g/l, then

$$c_T = \theta c_d + c_p = \theta c_d + m_{ss} \times r = c_d(\theta + \Pi m_{ss}) \quad (6.7)$$

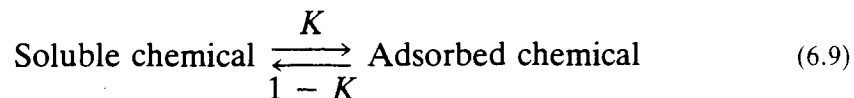
where θ = the water content of the soil or sediment as a fraction of the volume (for water $\theta = 1$).

The relation between the dissolved (pore water) concentration of the chemical and the total concentration in the soil, sediment-laden water, or sediment is then

$$c_d = \frac{1}{\theta + \Pi m_{ss}} c_T \quad (6.8)$$

For some chemicals (for example, metals) the term *precipitation* is commonly used. However, Salomons and Förstner (1984) quoting several other authors pointed out that there should be no difference between *sorption* and *coprecipitation*.

The isothermal concept for soil-sediment chemical interaction presumes either instantaneous adsorption or conditions of equilibria. The instantaneous adsorption assumption is adequate for modeling slowly varying processes in soil. For more dynamic conditions, the kinetic of adsorption can be expressed as



Volatilization

Volatilization is defined as the loss of chemicals in vapor form from soil or water surfaces to the atmosphere. The process is limited by the chemical

concentrations in water and air interfaces above the soil of water surface (Jury, 1986). Based on this definition volatilization does not occur from submerged aquatic sediments.

The potential of volatilization is related to the saturated vapor pressure of the chemical in the air above the interface; however, actual volatilization from soil is also affected by many other factors, such as atmospheric air movement, temperature, and soil characteristics (Spencer, Farmer, and Jury, 1982).

If a chemical exists in the soil and the vapor phase, Equation (6.7) is expanded to include the vapor phase as

$$\begin{aligned} c_T &= \theta c_d + c_p + ac_g = \theta c_d + m_{ss} \times r + ac_g \\ &= c_d(\theta + \Pi m_{ss}) + ac_g \end{aligned} \quad (6.10)$$

where

a = volumetric air content ($a = p - \theta$)

c_g = vapor density of the chemical ($\mu\text{g/l}$ of soil air)

p = volumetric porosity

The relation between the vapor density and corresponding concentration of the chemical in (pore) water solution is given by Henry's law

$$c_d = K_H c_g \quad (6.11)$$

where K_H = Henry's constant for the chemical that in this system is dimensionless. Hence Henry's constant K_H is the partitioning coefficient of the chemical between the air and water (U.S. EPA, 1986).

Substituting Equation (6.11) into Equation (6.10) yields the relationship between the total chemical concentrations in the soil and pore water as

$$c_d = \frac{c_T}{\theta + \Pi m_{ss} + (p - \theta)/K_H} \quad (6.12)$$

When soil dries to a point that only a few monolayers of water remain on the solids, adsorption of chemicals will greatly increase. This is due to the fact that soil particles preferentially adsorb water molecules. Thus in dry regions chemical losses by volatilization are sometimes reduced to insignificant levels. On the other hand, when soil is saturated ($\theta = p$), all transfer of the chemical to the surface of the soil is by diffusion. For this reason volatilization is significant when the soil has an intermediate moisture content (Spencer and Cliath, 1973).

The mathematical description of the vapor flux of the chemical from the soil surface is generally derived from the boundary layer theory, which presumes that at the soil (water)–air interface there is a relatively stagnant layer through which the gaseous chemical must move by molecular diffusion. Using Fick's law, Jury, Spencer, and Farmer (1983) represented this vapor flux away from the soil surface as

$$J = \frac{D_g^{(\text{air})}}{d} [c_g(0) - c_g(d)] \quad (6.13)$$

where

$D_g^{(\text{air})}$ = molecular diffusion coefficient of the gaseous chemical in air

d = thickness of the soil surface–air boundary layer

$c_g(0)$ = vapor phase concentration of the chemical in the upper layer of the soil defined previously

$c_g(d)$ = gaseous concentration of the chemical in the air (above the stagnant boundary layer)

The stagnant boundary layer is a concept. Its statistical thickness depends primarily on the meteorological factors (wind) and the roughness of the surface.

Biological Degradation and Transformation

Biological degradation of a chemical usually implies a breakdown by the living organism to more simple compounds, ultimately to carbon dioxide, water, and possibly other end products. Ultimate degradation means decomposition into mineral products (mineralization), while partial degradation is a decomposition into less complex compounds. *Transformation* on the other hand, means a change to another form, which may or may not be less environmentally hazardous. An example of such a transformation is the biochemical change of DDT to DDE, which is considered just as toxic as the original compound. Chemicals that are not biodegradable are thus called persistent in the environment, and their removal is only by a transfer to another medium (from soil to water or air) or by burial.

Biotransformation of chemicals in soil is accomplished by soil microorganisms and/or fungi (Valentine and Schnoor, 1986). These organisms are mostly *heterotrophic* and require an energy and organic carbon source, and nutrients for their growth. In contrast *autotrophic* organisms derive their energy from sunlight and *chemotrophic* rely on an exothermic chemical reaction as a source of energy. Both autotrophs and chemotrophs use carbon dioxide (alkalinity) as a source of carbon for their growth.

Autotrophic decomposition of chemicals in soil is rare; however, chemotropic transformation of nitrogen in soils is common (see the section titled "Specific Pollutant Interactions" in this chapter). In some cases the chemicals themselves may be the source of organic carbon and energy, in some other instances the chemicals are broken down by enzymatic reactions of microorganisms using a more simple and digestible carbon source (for example, plant residues). To break down some more complex chemicals the organisms require adaptation.

Biodegradation may occur in an *aerobic* (oxygen-rich aerated soils) or *anaerobic* (saturated soils devoid of oxygen) soil environment. Microorganisms that can function in both environments are called *facultative*, while strict aerobes and anaerobes can only function depending on whether oxygen is or is not available. Hence, based on the oxygen availability the biochemical reactions are either oxidation (aerobic) or reduction (anaerobic) processes. As an example, if an organic chemical is made of ring compounds, such decomposition associated with ring cleavage can be accomplished in an anaerobic environment; however, anaerobic biochemical decomposition is generally slower than aerobic breakdown.

Biodegradation kinetics is commonly represented by the Monod's equation derived from the Michaelis-Menton equation for substrate loss in enzyme catalyzed biochemical reactions. The Monod's formula adapted for the substrate variables defined earlier would be

$$\frac{dc_d}{dt} = \frac{-\mu_m X c_d}{K_s + c_d} \quad (6.14)$$

where

μ_m = is the maximum substrate utilization rate ($\mu\text{g/l-day}$)

X = microbial biomass per unit volume of liquid (pore water) ($\mu\text{/l}$)

K_s = half-saturation constant for the chemical ($\mu\text{g/l}$)

As was stated previously, only dissolved (pore water) fractions of the chemical are bioavailable.

For a small concentration of the chemical in the soil ($c_d \ll K_s$) and a sufficient and constant microbial population, the biodegradation equation is converted to a first-order reaction such as

$$\frac{dc_d}{dt} = -k_b c_d \quad (6.15)$$

Major factors affecting the biodegradation of a chemical are (Valentine and Schnoor, 1986) pH, temperature, water content, organic carbon content in the soil, clay content, oxygen availability, nutrients, nature of

the microbial population, acclimation of the microbial population to the chemical, and concentration of the chemical.

Although some texts contain magnitudes of the microbiological decay-decomposition rates such as k_b (see Schnoor et al., 1987), the reported ranges of these rate coefficients are very wide, and generally the reported values are not reliable. Furthermore, microorganisms may have the capacity of adapting to a particular chemical. Biodegradation experiments, both in situ and in the laboratory, may have to be performed with soils and sediments to obtain an insight on biodegradability of the chemical and its reaction-rate type and parameters. Aerobic and anaerobic conditions, soil-sediment organic content, sufficient nutrients, and acclimation of the microorganisms are the factors that may affect the decomposition process. In laboratory experiments, radioactive labeled organic chemicals can be used to estimate metabolic degradation (mineralization) by measuring the $^{14}\text{CO}_2$ carbon production in the reaction. A more detailed discussion on soil microbial populations and decay of pathogens (disease-causing microorganisms) concludes this chapter.

Plant Uptake

Plant uptake of nutrients and chemicals from soil is a form of immobilization of the pollutants. The nutrients and chemicals uptake process is a part of the overall transpiration process of plants. Nutrients and chemicals are transported into plants only from the dissolved or ionic pool of chemicals in pore water. Consequently, chemicals and nutrients adsorbed on soil particles or precipitated in soils (sediments) are not available to plants; thus, only the mobile components of the soil are of concern, and high uptake rates are observed when the mobile component concentrations are high. The total uptake rates and nutrient requirements also depend on crop type and yield.

The annual crop yields and nutrient content of various plants are presented in Table 6-4. The nutrient content of plants in Table 6-4 also represents the nutrient uptake for annual crops. To obtain annual nutrient uptake for perennial plants, divide the nutrient content by the average age of the plants. Dividing the nutrient content into grains and straw or stubble becomes important when planning certain management practices. For example, no-till planting leaves the stalks (and associated nutrients) on the field, while conventional planting removes the residues from the field; hence, nutrients must be resupplied by fertilizing the field.

Other Transformation Reactions

In addition to biochemical reactions a chemical present in soils and sediments can be degraded or transformed by chemical oxidation-reduction

TABLE 6.4 Approximate Yields and Nutrient Content of Selected Crops

Crop		Yield/ha	kg N/ha	kg P/ha
Alfalfa		9 Tonnes	225	30
Barley	Grains	100 bu	39	7
	Straw	2.2 tonnes	17	2
Beans ^a	(Dry)	75 bu	84	11
Cabbage		45 tonnes	165	18
Clover ^a	Red	4.5 tonnes	89	11
	White	4.5 tonnes	145	11
Corn	Grain	370 bu	151	27
	Stover	10 tonnes	111	18
	Silage	56 tonnes	225	34
Cotton	Lint and seed	2.2 tonnes	67	13
	Stalks	2.2 tonnes	50	7
Lettuce		45 tonnes	100	13
Oats	Grain	22 bu	62	11
	Straw	4.5 tonnes	28	9
Onions		17 tonnes	50	9
Peanuts ^a	Nuts	3.4 tonnes	123	7
Potatoes	Tubers	990 cwt	106	14
	Vines	2.2 tonnes	100	9
Rice	Grains	225 bu	62	13
	Straw	5.6 tonnes	34	4
Rye	Grain	75 bu	39	4
	Straw	3.4 tonnes	17	4
Sorghum	Grain	150 bu	56	11
	Stubble	6.7 tonnes	73	9
Soybean ^a	Grain	111 bu	179	18
	Straw	2.2 tonnes	28	4
Sugar beets	Roots	45 tonnes	95	16
	Tops	27 tonnes	123	11
Sugar cane	Stalks	67 tonnes	112	22
	Tops	29 tonnes	56	11
Tobacco		3.4 tonnes	129	11
Tomatoes	Fruit	56 tonnes	162	22
	Vines	3.4 tonnes	78	11
Wheat	Grain	123 bu	73	16
	Straw	3.4 tonnes	22	2
Bermuda grass ^b			540-670	
Fescue ^b			300	
Medium mature forest ^b				
Deciduous			30-60	
Evergreens			20-30	

Source: From Steward et al. (1975), after U.S. Department of Agriculture.

^aLegumes that do not require fertilizer nitrogen.

^bAfter Powell (1976).

reactions, hydrolysis, and photochemical transformation. The last reaction can only occur in the presence of light in a relatively thin surface layer.

Both hydrolysis and the nature and kinetics of the oxidation-reduction reactions are controlled by the moisture content. Other factors, such as pH and oxygen content, which are also important, are also related to the soil moisture.

Photochemical reactions (photolysis) are important for some organic chemicals, such as some pesticides (for example, DDT and parathion), and aromatic hydrocarbons (Schnoor et al., 1987).

Overall Degradation Rates

Typically due to lack of information it is not possible to quantify specific degradation rates. The only information usually available for many organic chemicals is their half-life, which is the mean number of days required for 50% of the original chemical to degrade in soil.

Using the first-order decay reaction (Eq. (6.15)) as the most accepted model to use to obtain the half-life from, the solution of the equation, by substituting overall pollutant concentration, P , for C_d , is

$$P(t) = P(0)e^{-K_t t} \quad (6.16)$$

where

$P(t)$, $P(0)$ = chemical masses at time t and time 0, respectively

K_t = overall degradation rate coefficient

t = time

The half-life is then

$$t_{1/2} = -\frac{\ln(0.5)}{K_t} = \frac{0.69}{K_t}$$

SPECIFIC POLLUTANT INTERACTIONS

Soil Phosphorus

In natural systems, phosphorus (P) occurs as an orthophosphate anion (PO_4^{3-}), which may exist in inorganic or organic forms. The source of all organic P is plant and organic biomass residues. The origin of all inorganic orthophosphate is the class of minerals known as apatites. These minerals are insoluble calcium phosphates existing in several forms, and the orthophosphate ions are liberated by chemical weathering processes.

Phosphorus is an important nutrient to aquatic ecosystems. In excess it causes accelerated eutrophication (see Chapter 13). Unlike nitrogen (N),

phosphorus is not particularly mobile in soils, and phosphate ions do not leach readily. Phosphorus is held tightly by a complex union by the clays and soil particulates, and organic matter. The amount of P in solution is small. Most of the P is removed from soils either by crop uptake or by soil erosion.

Figure 6.5 shows P solubility in soils. At higher pH values—characteristic of calcareous soils—the P precipitates mostly in combination with calcium (Ca). Below a pH of 7, which is characteristic for soils with high clay and organic matter content, Ca rapidly disappears from soils and P reacts predominantly with the Fe and Al ions in soils. From Figure 6.5 it can be seen that the maximum phosphate concentration in the pore water solution is of the order of 10^{-5} mole/l, which corresponds to about 0.3 mg/l. Depending on soil pH, the dissolved P concentrations may decrease to 0.01 mg/l or less.

From the foregoing discussion it can be seen that the P concentrations in the soil (sediment) pore water solution is very low and the excess is fixed by soil particles. Porcella et al. (1974) have reported that almost all P arising from weathering of minerals and/or from P application to soils by atmospheric fallout, commercial fertilizer, plant residues, or animal manure remains near the place of application. The exception is in sandy or peat soils, which exhibit little tendency to react with phosphorus. These authors have also shown that after a normal growing season, fertilizer P applied in the spring is confined to the 5 cm of the surface soil.

The process of fixation of P is controlled by several factors:

1. Al and Fe oxides are responsible for P retention in acid soils (Hsu, 1965; Vijayachandran and Harter, 1975).
2. Calcium compounds control solubility of P in calcareous soils (Hsu and Jackson, 1960; Hsu, 1965).
3. Organic matter contributes to P adsorption.

Figure 6.6 shows typical adsorption isotherms for some selected soils. Several authors have attempted to correlate P sorptivity to various soil parameters. Novotny and Chesters (1981) correlated adsorption Langmuir isotherm parameters to several independent soil parameters for acid soils. The most satisfactory combination of variables found by the authors to assess the adsorption maximum (Q^o) and the energy coefficient (b) was

$$Q^o(\mu\text{g/g}) = -3.5 + 10.7(\% \text{ clay}) + 49.5(\% \text{ organic C}) \quad (6.17)$$

and

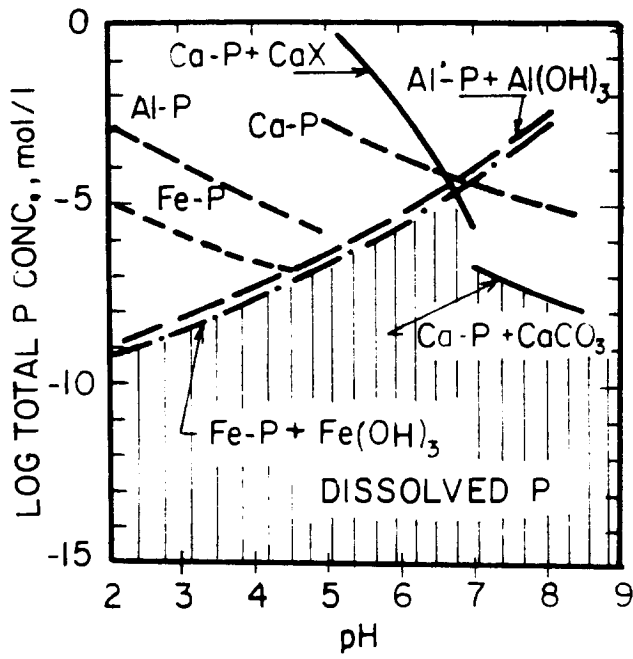


FIGURE 6.5. Solubility diagram for calcium, aluminum, and iron phosphates in soil. (From Hsu and Jackson, 1960, with permission. Copyright © Williams & Wilkins, Baltimore, MD)

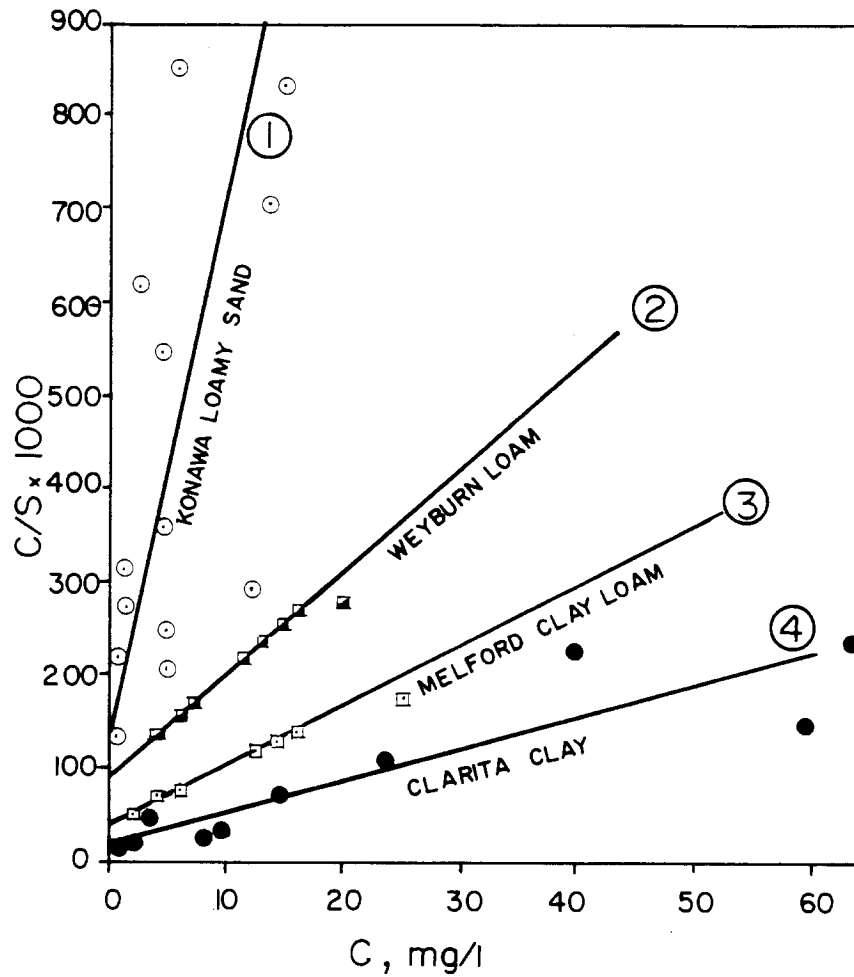


FIGURE 6.6. Langmuir isotherms for phosphate adsorption in soils. (Data 1 and 4 from Rennie and McKercher, 1959; data 2 and 3 from Enfield, 1974.)

$$b(\text{l/mg}) = 0.061 + 170,000 \times 10^{-\text{pH}} + 0.027(\% \text{ clay}) + 0.76(\% \text{ organic C}) \quad (6.18)$$

The coefficients of multiple correlation for Equations (6.17) and (6.18) were 0.83 and 0.53, respectively. The percentage of organic carbon can be roughly calculated from the content of the soil (sediment) organic matter with the use of conversion factors ranging from 0.5 to 0.6. For calcareous soils the distribution of particulate and as dissolved P is governed by the solubility diagram shown in Figure 6.5.

Example 6.2: Distribution of Sorbed and Dissolved Phosphorus in Soil

Fertilizer was applied to a field at a rate of 100 kg/ha. The fertilizer was plowed in and uniformly distributed to a depth of 0.3 m. The soil was a silt loam and contained 20% clay, 55% silt, and 25% sand. The porosity of the soil was 40%, pH was 6.0, organic C content was 1%, and the specific density of the soil sample was 1500 kg/m³ (= g/l). Estimate the approximate ground-water contamination by P during a long period of rain with an average intensity of 1 cm/hr. Assume that all water infiltrated into the soil and that the infiltration rate was slow enough that full equilibrium between the adsorbed and dissolved P was established. Assume also that the antecedent adsorbed P was negligible.

Solution Using Equation (6.17), the P adsorption maximum for the Langmuir isotherm (Eq. (6.4)) is

$$Q^o = -3.5 + 10.7 \times 20 + 49.5 \times 1 = 260 \mu\text{g/g}$$

and the adsorption energy coefficient from Equation (6.18) is

$$b = 0.061 + 170,000 \times 10^{-6} + 0.027 \times 20 + 0.76 \times 1 = 1.531/\text{mg} = 0.001531/\mu\text{g}$$

The total inorganic P content of the soil becomes

$$c_T = \frac{100 (\text{kg/ha}) \times 10^9 (\mu\text{g/kg})}{0.3 (\text{m}) \times 10^4 (\text{m}^2/\text{ha}) \times 1000 (\text{l/m}^3)} = 33,300 \mu\text{g/l of soil}$$

The P concentration is distributed between adsorbed and dissolved (pore water) fractions. Hence

$$c_T = r\rho + c_d\theta = \frac{Q^o b c_d}{1 + b c_d} \rho + c_d\theta$$

where

ρ = the specific density of the soil

θ = moisture content (assume θ = porosity)

Substituting into the preceding equation

$$33,330 (\mu\text{g/l}) = \frac{260 (\mu\text{g/g}) \times 0.00153 (1/\mu\text{g}) \times c_d}{1 + 0.00153 (1/\mu\text{g}) \times c_d} \times 1500 (\text{g/l}) + c_d \times 0.4$$

and solving, c_d is $57 \mu\text{g/l}$.

Note that in the denominator $b \times c_d \ll 1$; therefore, a linear partitioning concept can be used for which the linear partitioning coefficient, $\Pi = Q^o b = 260 \times 0.00153 = 405.61/\text{g}$.

Adsorbed P concentration is

$$r = \frac{260 \times 0.00153 \times 57}{1 + 0.00153 \times 57} = 20.85 \mu\text{g/g}$$

The enrichment ratio (ER) for P is generally higher than that for clay or organic matter due to desorption of P from soil into runoff water and possibly due to other causes. Data by Massey and Jackson (1952), Massey, Jackson, and Hays (1953), and Stoltenberg and White (1953) indicate that the ER for P is about 1.5 to 2 times that for clay or soil organic matter.

The adsorption characteristics for P (and by the same reasoning for other chemicals) adsorbed on soil particles, are different for parent soils and sediments, and when solids are suspended in runoff or surface-water body flow (Green, Logan, and Smeck, 1978; Ryden, Syers, and Harris, 1972; Novotny and Chesters, 1981). An example of an adsorption isotherm of P on suspended solids in runoff is shown on Figure 6.7. The ranges for suspended sediment adsorption maxima, Q^o , in the watershed of the Maumee River in Ohio reported by Green, Logan, and Smeck (1978) were between 500 and 2000 $\mu\text{g/g}$ (average 988 $\mu\text{g/g}$), while the ranges of Q^o for the parent soils were 189 to 287 $\mu\text{g/g}$ (average 257 $\mu\text{g/g}$). Similarly, the adsorption energy coefficient b ranged from 0.00011 to 0.00045 $1/\mu\text{g}$ (average 0.00032 $1/\mu\text{g}$) for the suspended sediment as compared to 0.0009 to 0.0045 $1/\mu\text{g}$ (average 0.00261 $1/\mu\text{g}$) for the parent soil. However, it can be argued that for the linear partitioning coefficient $\Pi = Q^o \times b$ the differences would be much less. Using average values of the Maumee River

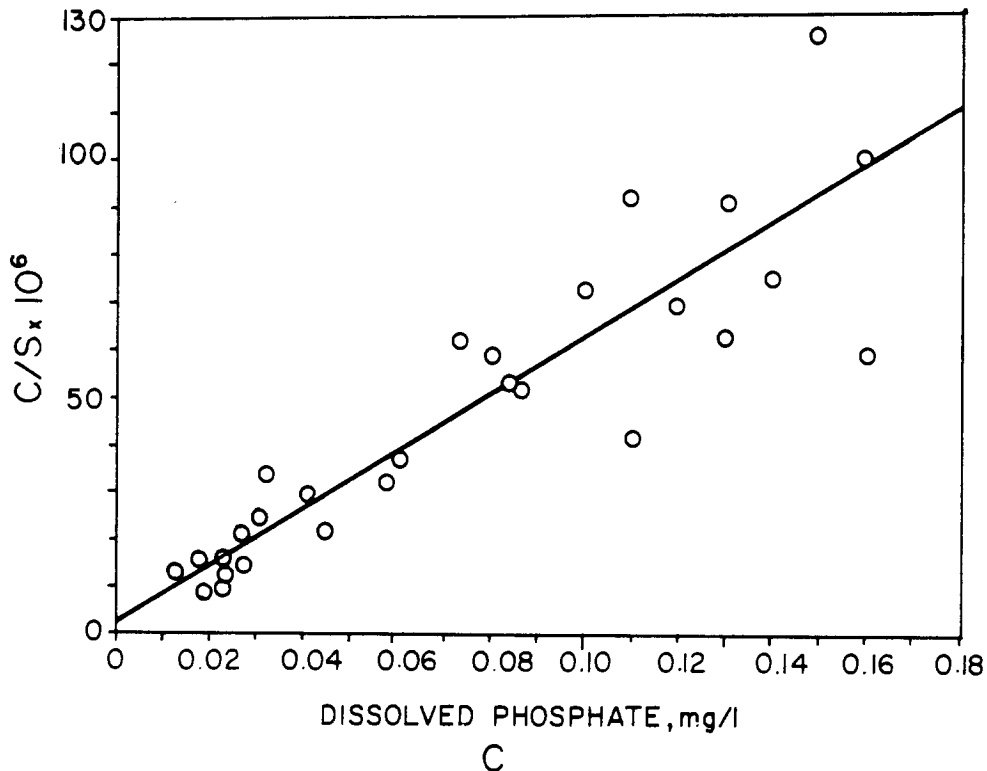


FIGURE 6.7. Isotherm correlation of dissolved and adsorbed phosphates in the suspended solids of a stream. (From Novotny et al., 1978.)

data, the average linear partitioning coefficient for suspended sediment would be $\Pi = 988 \times 0.00032 \approx 0.41/\text{g}$, while that for parent soil is $\Pi = 257 \times 0.0026 \approx 0.661/\text{g}$. Bottom sediments (bedload) exhibited adsorption characteristics between those of suspended sediment and parent soils.

Organic Chemicals

The use of organic chemicals is credited with substantially increasing yields of agricultural crops and assisting in controlling pests and diseases such as malaria. In a situation typical of most modern societies, the availability of agricultural land decreases, giving way to urbanization. On the other hand, severe environmental problems have been created by the use of man-made organic chemicals, which has resulted in ozone depletion, contamination of soils and bodies of water, and chronic damage to animals and humans. The epigraph for this chapter reflects the fears and sometimes realities of the damage done by the improper and excessive use of these chemicals.

Although the use of pesticide chemicals dates back a hundred years,

usage did not become substantial until about 1945 following the commercial manufacture of DDT. Rapid growth of the organic pesticide industry continued for about 25 years. Thousands of chemicals were developed and introduced into the market and subsequently discharged into the environment. Pesticide usage revolutionized agricultural production to the point that most agricultural practices formerly used to control weeds, insects, and disease shifted in favor of chemical control. At the same time, however, disposal of organic chemicals became a serious problem, and unsafe dumping of chemicals into the soil, air, and bodies of water has contaminated them to the point that their use has been severely impaired or even negated. Restoration and "repair" of these systems will be the objective of present and future environmental cleanup efforts.

Organic chemicals used throughout the world that are contaminating the environment can be divided into several categories. *Pesticides* include *herbicides* used for weed control (*algicides* for control of unwanted algal growth in lakes), *insecticides* for control of unwanted insects, while *fungicides* control unwanted fungus that may attack orchards or vegetable farms. The use of pesticides is not limited only to the agricultural sector; urban lawn-care chemicals are in the same category. Other chemicals of concern are *cyclic (aromatic) hydrocarbons*, *fluorocarbons* used in sprays and cooling liquids, *polychlorinated bi-phenyls (PCBs)*, which were widely used in transformers, paper, and other products, *volatile organic chemicals* and *solvents*, and several other categories.

Insecticides include organochlorine, organophosphorus, and carbamate chemicals. Organochlorine compounds, such as DDT, dieldrin, aldrin, heptachlor, and lindane, are essentially conservative chemicals. They can persist in soils and aquatic environments for many years. For example, DDT has been frequently detected 10 years after its application. Lindane can be found in some Great Lakes sediments and soils (Green Bay) 20 years after application in cherry orchards. Eventually, burial by new sediment deposits may alleviate the problem.

Use of organophosphorus and carbamate chemicals increased rapidly following restrictions on the use of organochlorine insecticides. These chemicals are less persistent in soils than the "hard" organochlorine form. An extended discussion on the persistence of chemicals in soils and sediments and their gradual disappearance and/or transformation is presented largely in a subsequent section of this chapter.

Herbicides are less ubiquitous in the environment than organochlorine insecticides. However, such compounds as *s*-triazines, picloram, monouron, and related substituted ureas, and 2,4,5-T often persist in soils for as much as a year following application. Atrazine, alone or in com-

mination, and propachlor are the herbicides most commonly used for weed control on corn in the midwestern United States. The herbicide MCPA and 2,4-D amine are used most frequently on small grains. The carbamate herbicides and 2,4-D are short lived in soil. One has to remember, however, that in sandy soil transport of chemicals into ground water may be relatively fast and the pesticide may reach ground water before soil bacteria can degrade it. Since there are few or no bacteria in ground-water-bearing strata, further biochemical decomposition is stopped and ground-water pesticide contamination arises and persists. Persistent ground-water contamination has been found in many agricultural areas throughout the world, including central Wisconsin and the Po River valley of Italy.

Polyaromatic (cyclic) hydrocarbons (PAHs) are mostly of urban and industrial origin. However, their fate and interactions with soils and sediments are similar.

Mobility of Organic Chemicals in Soils and Sediments

Chemicals deposited on soil surfaces follow the general transport path and can be transported into (1) the atmosphere, (2) ground water, and (3) surface runoff.

A general scheme of the distribution and fate of agricultural chemicals in the biosphere is shown in Figure 6.8. Contamination of the atmosphere by chemicals can occur by drift during application and volatilization, and by wind erosion. Drift is that portion of application that does not reach the target area. The extent of drift losses and the subsequent dispersion and transport of chemicals in air is governed primarily by meteorological conditions and methods of application. Chesters and Simsiman (1975) documented that drift losses range from 25% to 75% of aerially applied chemicals.

Contamination of ground water by organic chemicals can occur through leaching. Downward movement of agriculturally applied chemicals is controlled by soil type and chemistry, pesticide composition, and climatic factors (Hern and Melacon, 1986; Adams, 1973; Helling, Kearney, and Alexander, 1971). The leachability of a compound from soils depends primarily on the degree of adsorption of the chemical on soil particles. The U.S. EPA (1984) developed an approach that incorporates the mechanisms of degradation and sorption of constituents and incorporated it into a procedure of assessment mobility of organic chemicals. Using this procedure, a mobility and degradation index (MDI) for ranking the potential leaching of chemicals was proposed by Mahmood and Sims (1986).

For organic chemicals in soils and sediments, Equation (6.10) describes

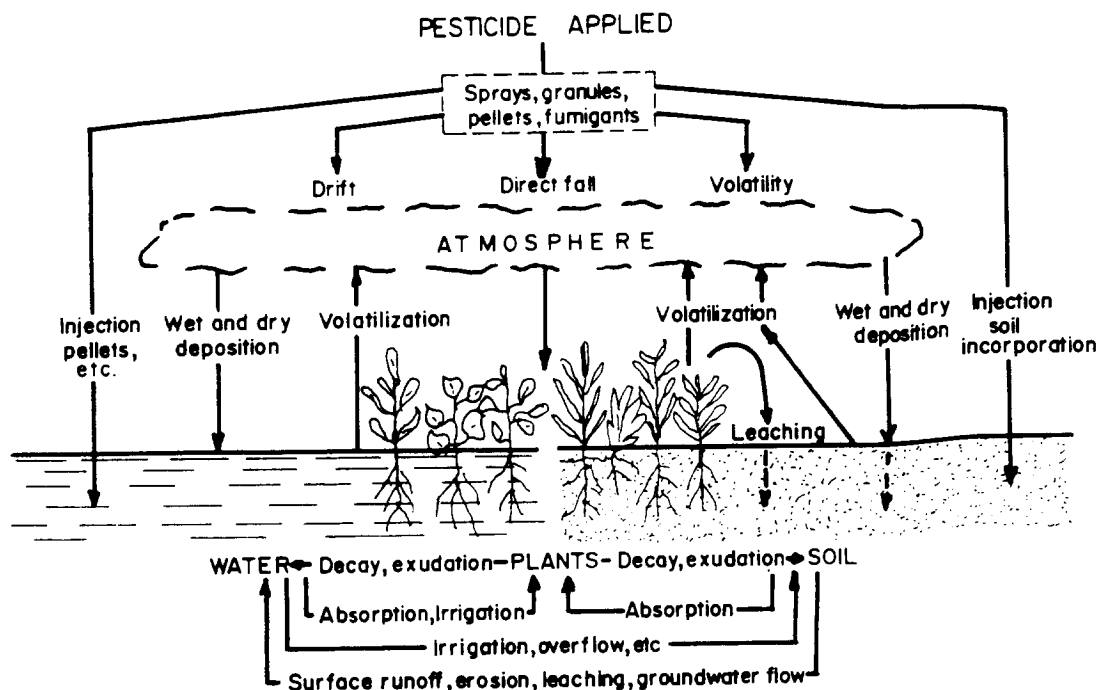


FIGURE 6.8. A scheme showing the possible distribution and fate of agricultural pesticides and their degradation products in the biosphere. (After Foy and Bingham, 1969.)

the distribution of the total chemical concentration between the solid, liquid, and gaseous phases, or

$$c_T = \theta c_d + c_p + ac_g = c_d(\theta + \Pi m_{ss} + aK_H)$$

The adsorptivity of nonpolar organic chemicals is related to their solubility expressed by the octanol partition coefficient K_{ow} and the particulate organic content of the soil (OC) expressed in percent. Then the solid-liquid partition coefficient can be expressed as

$$\Pi \approx K_{oc} \times (\%OC)/100 \quad (6.19)$$

where K_{oc} = the partitioning coefficient normalized by the organic carbon. The coefficient K_{oc} is a hypothetical partitioning coefficient for a sorbent (sediment or soil) composed solely of particulate organic matter. It has been found that for a given chemical, K_{oc} is relatively constant and varies by a factor of only 2 for a wide range of soils and sediments (Schwarzenbach and Westall, 1981; Weber, McGinley, and Katz, 1991). The coefficient K_{oc} can be correlated to the octanol partitioning coefficient, K_{ow} (Kenaga and Göring, 1980; Schwarzenbach and Westall,

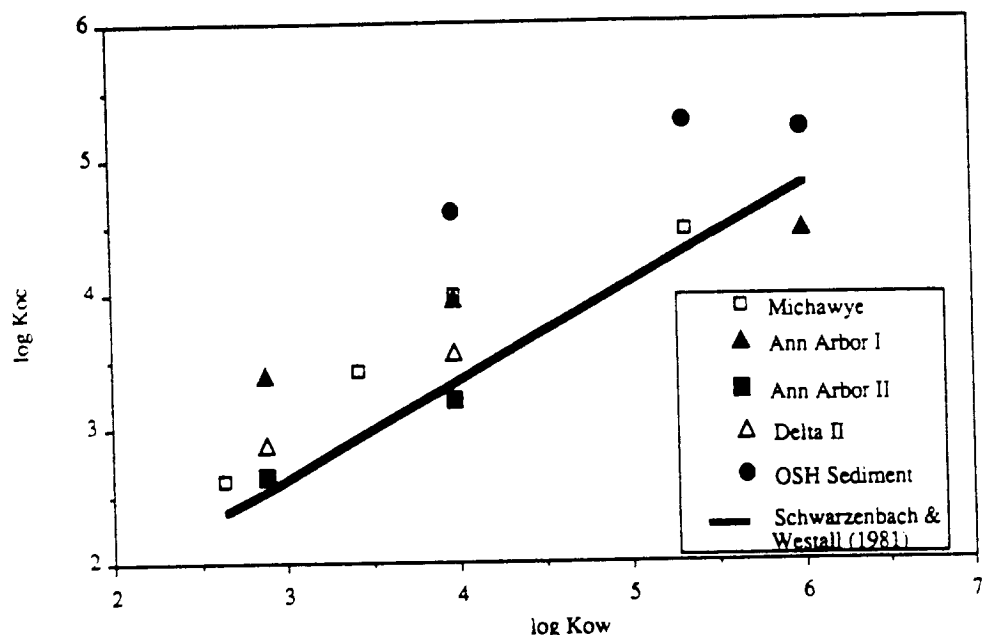


FIGURE 6.9. The relation of organic matter-chemical partitioning coefficient K_{oc} to the chemical's octanol partitioning parameter K_{ow} . (From Weber, McGinley, and Katz, 1991, with kind permission from Pergamon Press, Headington Hill Hall, Oxford, OX3 0BW, UK.)

1981; Weber, McGinley, and Katz, 1991). Several regression equations have been proposed (Karickhoff, 1981; Karickhoff, Brown, and Scott, 1979; Schwarzenbach and Westall, 1981). A relationship of K_{oc} to K_{ow} published by Weber, McGinley, Katz (1991) is shown on Figure 6.9. Values of K_{ow} and K_{oc} for various chemicals are given in the Appendix.

Karickhoff, Brown, and Scott (1979) proposed the following empirical relationships for estimating the partition coefficient for the sorption of aromatic hydrocarbons and chlorinated hydrocarbons on sediments:

$$K_{oc} = 0.63K_{ow} \quad (6.20a)$$

Almost an identical relation was derived by Rao and Davidson (1982), where

$$K_{oc} = 0.66K_{ow}^{1.03} \quad (6.20b)$$

This relationship can be expanded to segregate the influence of particle size as follows (Karickhoff, Brown, and Scott, 1979):

$$\Pi \approx K_{oc} [0.2(1 - f)x_{oc}^s + fx_{oc}^f] \quad (6.21)$$

where for Π in liters/kilogram

f = mass fraction of fine sediments or soil particles ($d < 50 \mu\text{m}$)

x_{oc}^s = organic carbon content of coarse sediment (soil) fraction

x_{oc}^f = organic carbon content of fine sediment (soil) fraction

Example 6.3: Distribution of a Chemical in Sediment

Determine the fraction of benzo(a)pyrene that is contained in pore water of aqueous sediment with a specific density of 1400 kg/m^3 ($= \text{g/l}$). The sediment is 70% fines ($d < 50 \mu\text{m}$) and the weight fraction of organic carbon is 10% of the fines and 5% of the sand fraction. From Table A1 in the Appendix the octanol partitioning coefficient for the compound is 10^6 . Porosity of the sediment is 45%, hence the water content is $\theta = 45/100 = 0.45$. Calculate K_{oc}

$$K_{oc} = 0.63 \times 10^6 = 630,000$$

Solution From Equation (6.21) the partition coefficient becomes

$$\Pi = 630,000[0.2(1 - 0.7)(0.05) + 0.7(0.1)] = 45,000 \text{ l/kg}$$

From Equation (6.10), neglecting the gaseous phase, the distribution of dissolved (pore water) and total concentration is

$$\begin{aligned} \frac{c_d}{c_T} &= \frac{1}{0.45 + 45,000 \text{ (l/kg)} \times 1400 \text{ (kg/m}^3) \times 0.001 \text{ (m}^3/\text{l)}} \\ &= 0.000016 \end{aligned}$$

Almost all of the chemical will be adsorbed on the sediment, and the pore water concentration will be about 1/1000 of a percent of the total concentration of the contaminant.

A brief description of the behavior of organic chemicals in soils and sediments is taken from Moore and Ramamoorthy (1984) and several other sources.

Aliphatic hydrocarbons. These compounds include a diverse group of open chain compounds of carbon and hydrogen, which may be halogenated by chloride, bromide, iodine, or fluoride ions. Halogenation may occur naturally or by adding halogens (for example, by disinfecting organic wastes or wasting halogenated chemicals into receiving bodies of water and soils). From many possible compounds the following priority pollutants are of environmental concern: carbontetrachloride, dichlorobromomethane, chloroethane, dichloromethane, dichloropropane,

vinyl chloride, chloroform, bromoform, tetrachloroethane, trichloroethane, methyl chloride, and methyl bromide. They are in the category of volatile priority pollutants. These priority pollutants have little or no affinity for sorption, and evaporation (volatilization) is the primary mechanism of their loss. Because of their volatility most aliphatics occur at low concentrations in soils and sediments, except in cases of extreme pollution.

Aromatic hydrocarbons. These chemicals have a cyclic six carbon ring. They can be monocyclic, such as benzene and toluene, or polyaromatic (PAHs). Benzene and toluene are examples of monocyclic compounds, while the family of more complex polyaromatic hydrocarbons includes such priority pollutants as anthracene, benzo(a)anthracene, benzo(a)fluoranthene, chrysene, fluoranthene, naphthalene, and pyrene. Many PAHs have been found to be carcinogenic.

Benzene, toluene, and some of their derivatives are moderately soluble in water and moderately volatile, so larger scale sorption on soils and sediments does not occur. Mill (1980) proposed the following equation for predicting the sorptivity of monocyclic aromatics:

$$\log K_{oc} = -0.782 \log\{C\} - 0.27 \quad (6.22)$$

where $\{C\}$ is the solubility of the compound in moles/liter.

PAHs found in soils and sediments are generally of a diffuse nature, and their origin can be traced to automobile use, municipal and industrial wastewater effluents, forest fires, and the combustion of coal. The last two sources are the most significant paths for PAHs entering the atmosphere. These contaminants will subsequently reach terrestrial systems by atmospheric deposition. Automobiles, especially those with diesel engines, were in the past a major source of PAHs. Recent restrictions on emissions have significantly reduced their discharge in countries where such environmental regulations were implemented. Typically, urban runoff contains large quantities of PAHs that are mainly incorporated into sediments (see Chapter 8 for a discussion of urban sources).

PAHs have a larger affinity for adsorption on soils and sediments than do the monocyclic compounds. Soil and sediment microorganisms present are capable of degrading PAHs. Photolysis is an important degradation process for some PAHs (for example, anthracene).

Pesticides. Sorption of nonionic (nonpolar) pesticides—organochlorine and organophosphorus insecticides—in soils and sediments is correlated primarily with organic particulate content and to a much lesser degree with clay (Goring and Hamaker, 1972; Moore and Ramamoorthy, 1984). Organochlorine pesticides—priority pollutants—include DDT and its

derivative DDE, aldrin, chlordane, dieldrin, endrin, endosulfan, heptachlor, and toxaphene. Lindane and mirex are also of concern and have contaminated parts of the Great Lakes and other bodies of water.

Most developed Western countries have curtailed or banned the use of DDT and several other persistent chemicals, yet these chemicals still may be found in the environment. Retention of acidic and basic pesticides in soils and sediments is affected markedly by pH (Goring and Hamaker, 1972). Soil and sediment pH controls the overall charge of the molecule and its adsorptivity to clay and organic matter. The organic cations—diquat and paraquat (not included in the priority pollutant list)—are held strongly by clay minerals and often are adsorbed irreversibly. Weakly adsorbed, water-soluble compounds are desorbed readily by water, and hence possess a greater potential for leaching. None of the acidic or basic pesticides have been included in the priority pollutant list. Organochlorine insecticides, which have low solubility, are least mobile in soils and sediments, followed in turn by organophosphorus insecticides. The water-soluble acidic herbicides are the most mobile. Other pesticides, including triazines, atrazines, phenyl ureas, and carbamates, have an intermediate degree of mobility.

After initial application losses (by drift and volatilization) the remaining pesticide reaches the soil, where loss is evident for surface-applied and soil-incorporated pesticides. In the soil, pesticides are continuously subjected to dissipation processes and their concentration decreases gradually after application. The mechanisms of dissipation include adsorption on soil particles and subsequent loss by erosion, degradation (photochemical and microbial), volatilization, plant uptake, and leaching. By definition, degradation affects only pesticides that are nonconservative. The general magnitudes of the overall persistence of pesticides in soils and aquatic sediments are shown on Figure 6.10.

Photodecomposition occurs only to pesticides located at the soil surface. The practical significance of photodegradation as a means of pesticide removal from soil and aquatic systems has not been determined quantitatively because of the difficulty of interpolating laboratory data to field conditions. In most cases, photodegradation reactions are comparatively slow, and their rate depends on the physical state of the pesticide (vapor, dissolved, adsorbed), light intensity, photolytic efficiency, and other factors. DDT and parathion are examples of pesticides that may degrade by photolysis (Schnoor et al., 1987).

The rate of volatilization of pesticides depends on such factors as temperature, soil moisture content, pesticide vapor density, soil properties, solubility in water, concentration of pesticides, ambient air humidity, and near-surface wind velocity. Factors controlling and mech-

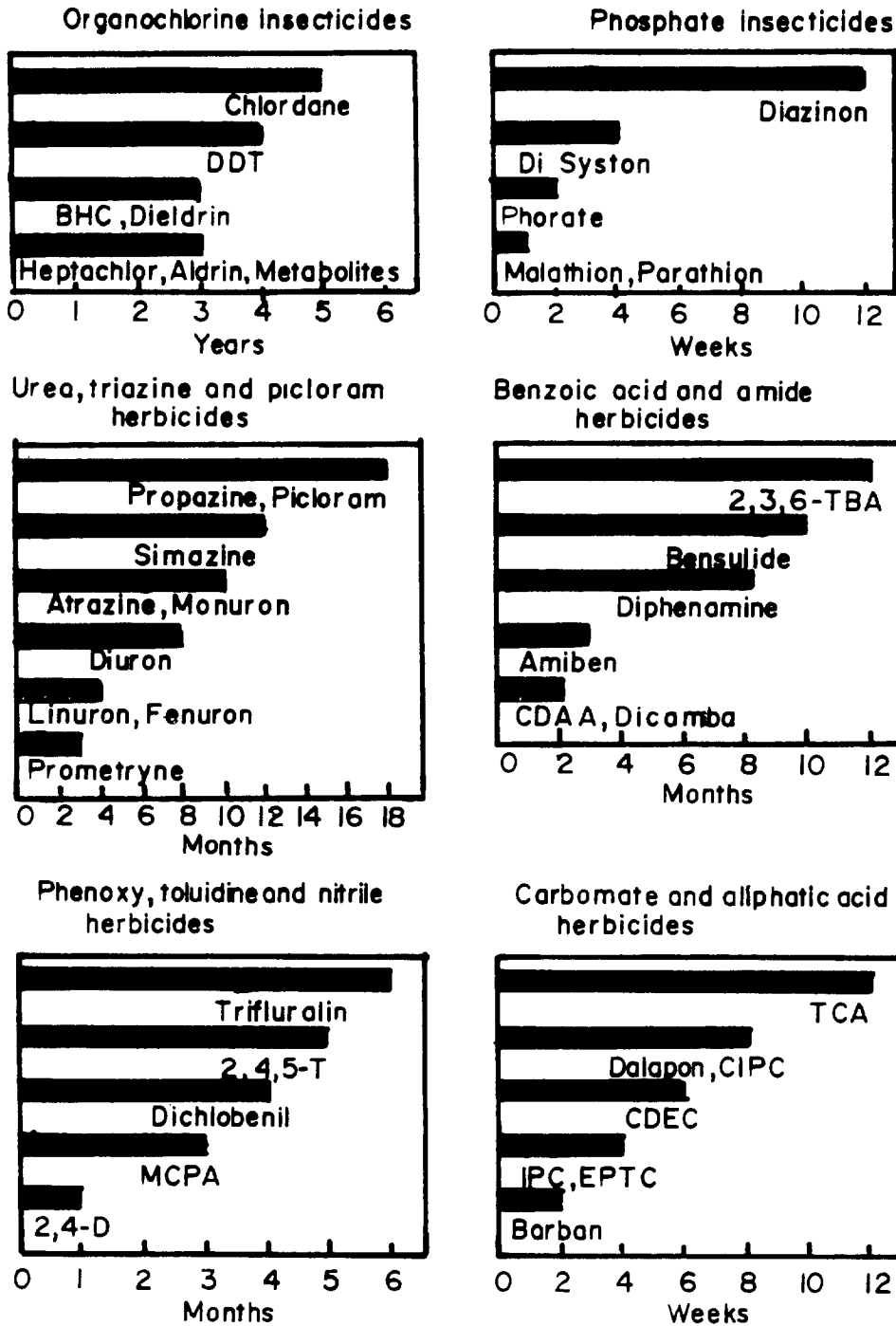


FIGURE 6.10. Persistence of pesticides in soils. (After Kearney et al., 1969.)

anisms of volatilization losses in soils have been discussed in a comprehensive review by Spencer and Cliath (1973). Field measurements indicate that significant volatilization losses may occur if pesticides are not incorporated into the soil.

In a study of pesticide losses from a watershed planted with corn it was found that considerable portions of dieldrin and heptachlor incorporated in the soil were lost by volatilization (Spencer, Farmer, and Cliath, 1973; Cars, Taylor, and Lemon, 1971). Losses in one growing season (from an application of 5.5 kg/ha) amounted to 2.5% to 2.9% for dieldrin and 3.9% for heptachlor. Under field conditions, the volatilization process is continuous, although its highest rate occurs immediately following pesticide application.

Many terrestrial and aquatic plants are capable of adsorbing and translocating pesticides (Edwards, 1970), followed by the possible detoxification of the compound to a less active compound. Several crops are known to adsorb chlorinated insecticides, and evidence of metabolic breakdown was opined for DDT, heptachlor, endrin, γ -BHC, and aldrin. Corn, which is resistant to atrazine and simazine, was able to adsorb these herbicides from soils and metabolize them to nonphytotoxic compounds. Dissipation of atrazine from soils through uptake by corn, sorghum, and other crops has also been indicated by experiments, but the extent of detoxification by plants is small.

Investigations indicate that chemical breakdown may play a significant role in the dissipation of soil-adsorbed organochlorine insecticides during dry periods (aerobic conditions), while the dominant degradation mechanism for these pesticides in moist or ponded soils and submerged sediments is by microbiological degradation. Chemical hydrolysis of the 2-chloro-*s*-triazines in soils and sediments has been reported by Armstrong and Chesters (1968). The hydrolysis of atrazine to nonphytotoxic hydroxyatrazine is enhanced by atrazine adsorption possibly to the carboxyl groups present in the organic components of soils and sediments.

Microbial metabolism is considered to be the major pathway of degradation of many pesticides in soils and sediments. The efficiency of this mechanism depends on such environmental factors as temperature, moisture and organic matter content, aeration or saturation, pH, and pesticide concentration. Although intensive studies have been made on the mechanisms by which microorganisms degrade pesticides, the processes are not understood clearly. In general, organochlorine pesticides are the most resistant to microbial attack. As pointed out previously, the partial degradation of DDT results in the formation of TDE (DDD) and DDE. Both components are stable in soils and sediments and have about the same toxicity as the parent compound.

The oxygen status of soils and sediments has a pronounced effect on the microbial breakdown of organochlorine pesticides. In soils and sediments DDT is rapidly converted to TDE (DDD) under anaerobic conditions, and very slowly to DDE in soils and exposed sediments under aerobic conditions (Guenzi and Beard, 1967, 1968). On the other hand, several organochlorine pesticides—heptachlor (Miles, Tu, and Harris, 1969), lindane (Yule and Rosefield, 1964), and endrin (Bouman, Schechter, Carter, 1965)—have been shown to degrade in soils to compounds of lower toxicity and reduced insecticidal activity.

Compared to organochlorine pesticides, other commercial pesticides currently in use are more biodegradable. One of the first extensive reviews of degradability of herbicides was published by Kearney and Kaufman (1964). Paris and Lewis (1973), Paris (1981), Sethunathan (1973), among others have investigated the biodegradability of pesticides in aquatic systems. The newer vinyl phosphate insecticides, such as phosphamidon, chlorfenvinphos, and mevinphos, have half-lives in soils ranging from 1 to 30 weeks. Chlorfenvinphos appears to be the most resistant to microbial degradation.

In spite of the enormous amounts of research on pesticide distribution and transport through the environment, no information is available to accurately estimate enrichment ratios for these chemicals (and by the same reasoning for any other priority organic pollutants). It appears, however, that for those pesticides that are immobile and tightly adsorbed on soil organic matter, the enrichment ratios may be similar to that of organic particulate matter. Conversely, for pesticides that are mobile (mostly dissolved) in soil water the notion of the enrichment ratio is meaningless. An approximate method of computing pesticide loading is shown in the following example.

Example 6.4: Pesticide Load in Runoff

To control weeds, 5.55 kg/ha of atrazine was applied to a field. Estimate how much atrazine will be lost during a 5-cm storm resulting in 1.5 cm of surface runoff and 1.5 tonnes/ha of soil loss due to erosion, which occurred 4 weeks after application. Porosity of the soils is $p = 45\%$, 0.3 bar moisture content is $\theta_{0.3 \text{ bar}} = 35\%$, 15 bar moisture content of the soil is $\theta_{15 \text{ bar}} = 15\%$, and the organic C content is 2.3%. The antecedent soil moisture $\theta = 31\%$ and evapotranspiration are negligible. Specific density of the soil is $\rho = 1500 \text{ kg/m}^3$. Effective moisture $\theta_{\text{eff}} = (31 - 15)/100 = 0.16$

Solution From Figure 6.10 it can be seen that atrazine persists in soils for approximately 10 months, which implies that during the 4 weeks

between application and the storm, approximately 10% of the pesticide was dissipated or decomposed.

Table A1 in the Appendix shows that the octanol partitioning number for atrazine $K_{ow} = 10^{2.68} = 478.631/\text{kg}$. Then the organic carbon partition coefficient becomes

$$K_{oc} = 0.63 \times 478.73 = 3011/\text{kg}$$

and the soil partitioning coefficient is $\Pi = K_{oc} \times f_{oc} = 301 \times 2.3/100 = 6.021/\text{kg}$.

The depth of penetration of the pesticide in the soil can be approximated as 7.5 cm, with uniform distribution throughout the depth. This is an arbitrarily chosen depth that may only be considered as an order of magnitude; however, it has been seen as an average estimate of pesticide penetration in agricultural soils (Bruce et al., 1975). Thus the pesticide concentration per unit volume of soil is

$$Pe = 0.9 \frac{5.55 (\text{kg}/\text{ha}) \times 10^9 (\mu\text{g}/\text{kg})}{7.5 (\text{cm}) \times 10^8 (\text{cm}^2/\text{ha}) \times 0.001 (1/\text{cm}^3)} = 6670 \mu\text{g}/\text{l}$$

The distribution between the adsorbed and dissolved phases can be computed from the volumetric mass balance equation (Eq. (6.8)) as *Dissolved*

$$C_d = \frac{1}{\theta_{\text{eff}} + \Pi\rho} Pe = \frac{1}{0.16 + 6.02 [1/\text{kg}] \times 1500 [\text{kg}/\text{m}^3] \times 0.001 [\text{m}^3/\text{l}]} 6670$$

$$= 726 \mu\text{g}/\text{l} = 0.726 \text{g}/\text{m}^3$$

Adsorbed

$$r = \Pi\rho C_d = 6.02 (1/\text{kg}) 726 (\mu\text{g}/\text{l}) \times 0.001 (\text{kg}/\text{g})$$

$$= 4.37 \mu\text{g}/\text{g} (= \text{g}/\text{tonne})$$

The estimated pesticide loss during the storm assumes that desorption from the sorbed phase to solution is negligible. In this case, the adsorbed pesticide loss becomes (assume that the estimated ER for organic matter is ≈ 1.2)

$$Y_{Pe} = r \times \text{ER} \times Y_{ss} = 4.37 (\text{g}/\text{tonnes}) \times 1.2 \times 1.5 (\text{tonnes}/\text{ha})$$

$$= 7.87 \text{g}/\text{ha}$$

If this pesticide was immobile and tightly bound to soil organic matter, the preceding amount would represent the approximate pesticide loss during the storm. However, atrazine has intermediate mobility, and in this case the dissolved fraction must be estimated as well. As a first approximation assume that runoff completely mixes with the pesticide in a 5-mm-thick upper soil surface mixing layer. Then the mass balance equation for the dissolved layer is

$$0.5(\text{cm}) \times \theta_{\text{eff}} C_d = C_{\text{runoff}}(0.5 \times \text{porosity} + \text{runoff})$$

from where the pesticide concentration in runoff during and after the storm is

$$C_{\text{runoff}} = \frac{0.5 \times 0.16 \times 726}{0.5 \times 0.45 + 1.5} = 32.72 \mu\text{g/l} = 0.03272 \text{ g/m}^3$$

and the dissolved pesticide loss in the runoff becomes

$$\begin{aligned} Y_{PD} &= 1.5(\text{cm}) \times 0.01(\text{m/cm}) \times 0.03272(\text{g/m}^3) \times 10^4(\text{m}^2/\text{ha}) \\ &= 4.9 \text{ g/ha.} \end{aligned}$$

Relating pesticide loss only to erosion for pesticides with medium to high mobility could lead to gross underestimation of the loss.

Since only gravitational water remains in the soil after the storm, the excess water will be leached into the lower soil layer by infiltration. The excess infiltration amounts to

Rainfall depth – runoff + antecedent soil moisture

$$\begin{aligned} - \text{gravitational soil moisture} &= 5 - 1.5 + \theta \times 7.5 - \theta_{0.3\text{bar}} \times 7.5 \\ &= 3.5 + 7.5(0.31 - 0.35) = 3.2 \text{ cm.} \end{aligned}$$

With this amount of excess water the quantity of pesticide leached downward is

$$3.2(\text{cm}) \times 0.01(\text{m/cm}) \times 0.726(\text{g/m}^3) \times 10^4(\text{m}^2/\text{ha}) = 232 \text{ g/ha.}$$

For this particular scenario, more pesticide is leached into ground water than is transported from the field by soil erosion. The pesticide has a potential to severely contaminate ground-water resources.

A simple pesticide load model was proposed by Haith and Tubbs

(1981) and was incorporated into the EPA loading model routines for diffuse pollution (Mills et al., 1985).

Polychlorinated Biphenyls

Polychlorinated biphenyls (PCBs) are man-made chemicals that are alien to nature, and as with most of the previous types of chemicals (rare exceptions are some PAHs), no natural-background concentrations in soils and sediments exist. Most of the environmental mass of PCBs is confined to industrial and urban areas; however, PCB contamination of arctic glaciers has been measured. Many freshwater and aquatic sediments have been heavily contaminated by these compounds (the New Bedford, Massachusetts, and Waukeagen, Illinois harbors, and Green Bay of Lake Michigan are examples of such environmental damage). The occurrence and distribution of PCBs in aquatic sediments indicate regional variations (National Academy of Sciences, 1979). The highest level of PCB contamination of sediments is found in heavily industrialized populous areas, particularly in the eastern part of the United States. The sources of these contaminations were mostly traced to past industrial operations. Contamination of rural soils and sediments draining mostly rural watersheds is relatively low and below the detection limit. Lake Ontario and Lake Erie of the Great Lakes system exhibit the highest contamination of sediments by PCBs, while Lake Superior has the lowest. It is suspected that the major source of PCB input to the Great Lakes is atmospheric deposition.

PCBs have very low solubility, consequently, their octanol partition coefficients are high; typically, K_{ow} would range between 10^4 and 10^6 l/kg (see Table A1 in the Appendix). PCBs are almost conservative components that are not readily decomposed by soil and sediment bacteria, and again biodegradability is inversely correlated to the chlorine content of the PCB molecule. Their removal from soils is primarily by volatilization and biomodification of lower PCBs (Hague, Schmedding, and Freed, 1974).

Toxic Metals

Most natural waters, and even more so, sediments, have a natural capacity to reduce the toxicity of added metals that is attributed to the presence of ligands in waters, to adsorption, and precipitation (Salomons and Förstner, 1984). As it is for adsorbable organic chemicals, metallic toxicity of contaminated sediments is affected by the metal activity in the pore water of the sediment.

When toxic metals are added to water both from natural and man-made

sources, they undergo complexation with ligands. Ligands are chemical constituents that are both organic and inorganic that combine with the metals in a chemical complex. From the basic physical chemistry it is known that metals precipitate because of changes in pH or when oxidized or otherwise change their chemical composition. However, the process of complexation in natural waters is more complex. Major causes for precipitation and metal complexation are (Salomons and Förstner, 1984):

1. Oxidation of reduced components such as iron, manganese, and sulfides
2. Reduction of higher valency metals by interaction with organic matter (selenium, silver)
3. Reduction of sulfate to sulfide (iron, copper, silver, zinc, mercury, nickel, arsenic, and selenium are precipitated as metal sulfides)
4. Alkaline-type reactions (strontium, manganese, iron, zinc, cadmium, and other elements are precipitated by increased pH, usually caused by interaction with alkaline rocks and sediments or by mixing with alkaline waters)
5. Adsorption or coprecipitation of metallic ions with iron and manganese oxides, clays, and particulate organic matter
6. Ion-exchange reactions, primarily with clays.

In aerobic freshwater sediments the sorption sites are provided by organic carbon, clays, and hydrous oxides of iron and manganese. The Fe and Mn oxides also have limited ion exchange capabilities. Hydrous iron oxides strongly adsorb chromium, while manganese oxides adsorb nickel, and calcium phosphate (also present in sediments) adsorbs cadmium, lead, and other metals. Mercury in sediments (in sediments mercury exists mostly as methyl mercury) is strongly adsorbed by organic matter (Langston, 1985). Oxides of iron and manganese are deemed to be more important than organic matter and clays; however, Combest (1991) documented that the Fe and Mn contents correlate with the clay content. Similarly, Summerhayes, Ellis, and Stoffers (1985) analyzed sediment samples of the New Bedford, Massachusetts, harbor, and found that clay fractions of the sediments were enriched 400 times for silver, 250 times for cadmium, 140 times for copper, 90 times for zinc, 45 times for lead, 40 times for mercury, 32 times for chromium, 7 times for arsenic, and 6 times for nickel when compared to the bulk sediment concentrations. The concentrations of chromium, nickel, lead, and copper correlated strongly with the clay fraction, with the coefficients of correlation ranging from 0.77 for copper to 0.92 for nickel (Moore, 1963). In anaerobic sediments sulfide

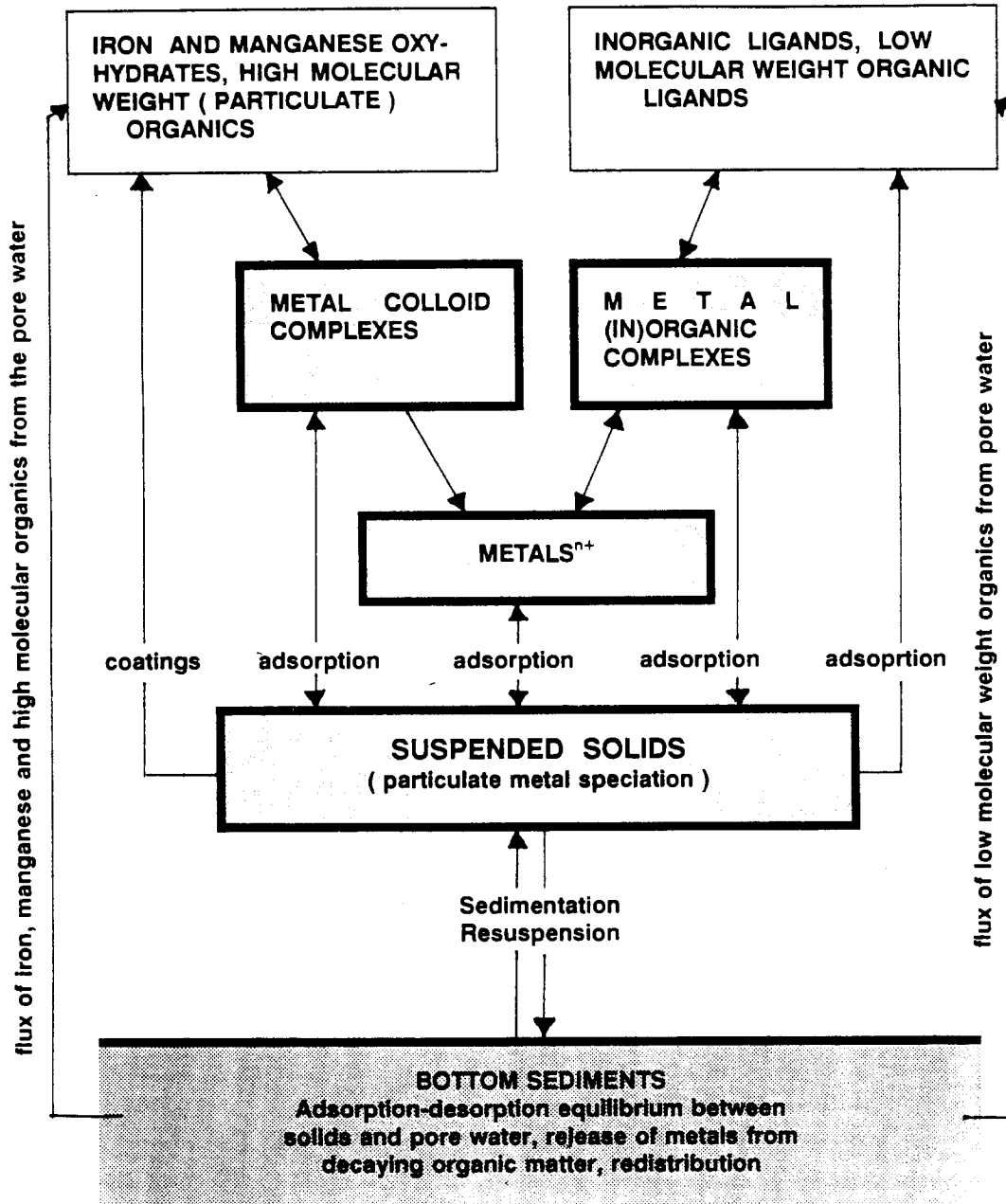


FIGURE 6.11. Major processes and mechanisms in the interaction between dissolved and solid metal species in sediments. (After Salomons and Förstner, 1984, with permission.)

precipitation is important in complexation of toxic metals (DiToro et al., 1989).

It has been found that the free metal ion is the most toxic component for organisms (Gillespie and Vaccaro, 1978; Salomons and Förstner, 1984; DiToro et al., 1989). As a matter of fact Gillespie and Vaccaro and others used a bioassay test with bacteria to detect and measure the

complexing capacity and other parameters of the distribution of metals in natural waters. Exchangeable metallic ions (they adsorbed on organic and inorganic complexes, such as humic and clay particles) are far less available due to the ion exchange between the particles and organisms bonding surfaces (Fig. 6.11). Other metal complexes are not bioavailable. Hence, the presence of compounds that will react with the metal ion and cause its precipitation or adsorption on solids (including iron and manganese oxides, sulfides, clay, and organic matter) will reduce the toxicity of the metals and make them less bioavailable for organisms. The colloidal and ionic compounds that combine within a complex with the metal ion (*ligands*) include organic acids and humic substances, dissolved sulfides, chloride, and OH^- ions.

Hence, the adsorbing and complexing compounds for toxic metals include

1. Particulates: sulfides, iron and manganese oxyhydrates, particulate organic matter, clays
2. Dissolved: sulfides, humic compounds, organic acids, chloride ion, hydroxyl ion

Dissolved metal-organic (ligand) complexes may also be adsorbed by particulates such as iron and manganese oxyhydrates. On the other hand, some dissolved organic compounds, such as detergents, may reduce the adsorptivity of metals, while chlorides may enhance adsorptivity. From the preceding mechanisms, iron and manganese oxyhydrates provide the strongest adsorption sites, followed by particulate organics and clays.

When metals are present in water the metal ions are distributed with the various complexing ligands (Fig. 6.11) and solids. Under equilibrium conditions the total molar concentration of metals in sediment and (pore) water is (Hart, 1981; Salomon and Förstner, 1984; DiToro et al., 1989)

$$[M_T] = (1 - \phi)\Sigma[M \equiv P_i] + \phi\{[M^{n+}] + \Sigma[M \equiv L_i]\} \quad (6.23)$$

where

$[M_T]$ = total (molar) concentration of the metal in the sediment volume

$[M^{n+}]$ = concentration of the free metal ion (toxic)

$[M \equiv P_i]$ = inorganic and organic complexes of the metal adsorbed on the sediment (generally bioavailable)

$[M \equiv L_j]$ = organic and inorganic ligand complexes (less toxic or nonbioavailable)

ϕ = porosity

Assuming that in the aerobic soil environment the primary solids interacting with the metals are iron and manganese oxy-hydrates, particulate organic matter, and clays, the following chemical equilibria and isotherms may be introduced:

$$[M \equiv \text{FeO}_x] = K_{\text{Fe}}[M^{n+}][\text{FeO}_x]$$

$$[M \equiv \text{MnO}_x] = K_{\text{Mn}}[M^{n+}][\text{MnO}_x]$$

$$[M \equiv \text{POC}] = K_{\text{oc}}[M^{n+}]m_{\text{oc}}$$

$$[M \equiv \text{clay}] = K_{\text{clay}}[M^{n+}]m_{\text{clay}}$$

where

K = chemical adsorption equilibria or partitioning constants

$[M \equiv \text{FeO}_x]$, $[M \equiv \text{MnO}_x]$, $[M \equiv \text{POC}]$, $[M \equiv \text{clay}]$ = metal adsorbed on iron manganese oxyhydrates, organic particulate matter, and clay

m_{oc} = organic carbon portion of the sediment

m_{clay} = clay portion of the sediment

DiToro et al. (1989) reported that in sediments the concentration of metal–ligand complexes in pore water is negligible when compared to that adsorbed on the sediments. Then neglecting the ligand–metal concentrations in pore water, the pore water free metal concentration in aerobic sediments or soils becomes

$$M^{n+} \approx \frac{M_s}{K_{\text{Fe}}[\text{FeO}_x] + K_{\text{Mn}}[\text{MnO}_x] + m_{ss}(K_{\text{oc}}f_{\text{oc}} + K_{\text{clay}}f_{\text{clay}})} \quad (6.24)$$

where

$M_s = M_T/(1 - \phi)$ = total sorbed metal per unit dry weight of sediment

f_{oc} = fraction of the sediment that is organic carbon

f_{clay} = clay fraction of the sediment

m_{ss} = sediment dry weight

In anaerobic sediments, iron and manganese oxides are reduced; however, sulfides become the prime complexing ligands. Hence $K_{\text{S}^{--}}[\text{S}^{--}]$ will replace oxide terms in Equation (6.24).

It can be seen that Equation (6.24), which expresses the relationship between the dissociated and precipitated–complexed metal, is similar to the partition relationship between dissolved and adsorbed organic chemicals (Eqs. (6.8) and (6.12)). Hence the denominator of Equation (6.24) could be called a “partition coefficient” for metals, or

$$\Pi_m = K_{\text{Fe}}[\text{FeO}_x] + K_{\text{Mn}}[\text{MnO}_x] + m_{\text{ss}}(K_{\text{oc}}f_{\text{oc}} + K_{\text{clay}}f_{\text{clay}}) \quad (6.25)$$

and similarly to nonpolar organic chemicals $M_s = \Pi_m M^{n+}$

The values of partition constants for various metals and soil/sediment characteristics are given in Table A2 in the Appendix.

From the reactions just listed, the interactions with particulate matter in the water column and, above all, in sediments are most important in determining the toxicity of metals in sediments and water.

The adsorption reactions are strongly pH-dependent, ranging from zero adsorption in low pH to 100% adsorption in higher pH. The change from no adsorption to full adsorption is fairly sharp, usually within two pH units (Salomons and Förstner, 1984). However, at pH > 7 most of metals are complexed, while at a pH of less than 5 the concentrations of free metal ions increase dramatically. This manifests the adverse impact of acid rain and acidity on leaching metals from soils and sediments into the aqueous phase. The leaching of aluminum from acidified soils reduces soil fertility. As pointed out in Stigliani (1991) and Stigliani et al. (1991), in some parts of the world (for example, central Europe) aluminum loss from soil may be a sudden and damaging "time bomb."

The impact of pH can be modeled by including the species MOH^+ in the mass balance equation. The result is that the sorption concentrations are replaced by the following expression (DiToro et al., 1989):

$$K_{\text{Fe}} = K_{\text{Fe}}^{(1)} + K_{\text{Fe}}^{(2)}[\text{H}^+]^{-1} \quad (6.26)$$

In natural waters under oxic condition, iron and manganese oxyhydrates can be present in crystalline forms or as a coating on rocks, sediments, and other solids. Essentially, Fe/Mn oxyhydrates are abundant and present in all parts of the hydrological cycle.

Sulfides, mostly particulate, are also common in sediments and have been found even in sediments with sand and gravel texture that do not resemble the anoxic sulfidic sediments (DiToro et al., 1990). A measure of solid or colloidal sulfides that can combine with metals is *acid volatile sulfides* (AVS), which is sulfide extracted from the sediment by 0.5M HCl in one hour at room temperature. Metals are also extracted by the same treatment of the sediment with acid. These tests provide a quick and inexpensive way of establishing the gross degree by which the sediments are polluted with metals (Salomon and Förstner, 1984; DiToro et al., 1990). The molar concentration of the simultaneously extracted metals (SEM) by hydrochloric acid can then be compared with the molar concentration of the AVS. If the metal toxicity unit (MTU) = SEM/AVS ratio is less than one, then the metals in the sediment are tied in the

sulfide complexes and are not bioavailable, and hence are not toxic. Measured AVS concentration ranged from <0.1 mole/g of sediment to >50 mole AVS/g of sediment (literature sources quoted in DiToro et al., 1990). Acid volatile sulfides are typically found in submerged sediments. When the sediments are exposed, the sulfides are oxidized and may not be available for complexation of metals. Iron and manganese oxyhydrates and organic matter then become the major complexing ligands.

Organic adsorbing particulates are also an essential part of terrestrial waters. Research investigations on reducing (anaerobic) sediments have shown that all of the zinc, almost all of the copper, and most of the nickel and cobalt are bound to humic-type materials (Nissenbaum and Swaine, 1976).

Example 6.5: Metal Distribution in Sediments

Sediment was analyzed by the acid extraction method (metals and sulfides were extracted by 0.5 M hydrochloric acid at room temperature in one hour). The results of the test are given in Table 6.5. Make a judgment whether metal is mobile (precipitated-complexed).

Solution From the table we see that $MTU = 2.2473/5 = 0.45 < 1.0$; therefore, the metals are immobile (complexed).

Methylation of metals. Mercury, arsenic and a few other less important metals undergo biological methylation in the anaerobic sediments. The conversion of inorganic mercury by bacteria residing in the sediments into organic methyl or dimethyl compounds is well known and understood. The case has been widely publicized by the famous Minamata Bay disaster

TABLE 6.5 Test Results

Acid Volatile Sulfides		5- μ mol AVS/g of (Dry) Sediment	
Metals	Atomic Weight	Dry Weight Concentration (μ g/g)	Molar Concentration (μ g/g)
Cadmium	112.4	1.0	0.0089
Copper	63.4	50.0	0.7886
Mercury	200.6	0.1	0.0005
Lead	207.0	300.0	1.4493
Totals		351.1	SEM = 2.2473

sickness in Japan, where people were stricken by the symptoms after ingesting fish contaminated by methylmercury. Later incidents occurred in Niagara Falls (New York) with similar debilitating effects (Krenkel, 1973).

Organic complexed compounds are far more toxic than their inorganic counterparts. This is reflected in current water quality standards, although quality criteria for sediments were evolving at the time the manuscript of this book was being prepared. Methylated compounds are still strongly polar and behave in sediments similarly to free metal ions, that is, they are adsorbed by the sediments and the ligands in the sediment.

Specific Metals

Arsenic (As). Inorganic arsenic compounds have been used in agriculture as pesticides and defoliants for many years. Also lake managers have used arsenic salts to control algal and macrophyte growths. Due to the serious environmental problems caused by these applications, their use was banned in the United States in 1967.

Once incorporated into soils or sediments, As reverts to arsenate, which is strongly held by the clay fraction (U.S. EPA, 1976). Arsenate also forms insoluble salts with Mn^{2+} , Ni^{2+} , and other alkaline cations.

Arsenic organic complexation by bacteria is relatively complex; however, Braman and Foreback (1973) found methylated forms of arsenic in a wide range of surface-water bodies. In anaerobic sediments arsenic is complexed by bacteria to methyl- and dimethylarsenic acids, which in interstitial and overlying water are converted to di- and trimethylarsines (Salomons and Förstner, 1984). These acids can be biologically synthesized. Dimethyl arsenic acid is difficult to oxidize and may contribute significantly to dissolved arsenic concentrations in surface waters.

Desorption of complexed arsenic back to solution in water greatly depends on the strength of the reducing environment (typically expressed as the oxidation-reduction potential). Anaerobic sediments release 10 times more toxic arsenite As^{3+} than aerobic sediment layers (Moore and Ramamoorthy, 1984).

Cadmium (Cd). The divalent cadmium ion Cd^{2+} is the predominant species of cadmium in surface waters and interstitial waters of sediments and soils. The mechanisms by which cadmium (Cd^{2+}) is removed from water are precipitation and adsorption and chemisorption on the surface of solids. Partition coefficients for cadmium in soils were measured by Christenson (1984). The partition coefficient for organic river sediments varied with types of solids and concentration of complexing agents. Organic materials such as humates were the main components of the sediment samples responsible for adsorption of cadmium.

DiToro et al. (1990) pointed out that in anaerobic sediments sulfides are responsible for most of the complexation of cadmium, and the acid volatile sulfides test provides a measure of the immobilization of cadmium.

Adsorption increases with pH. At pH > 7, virtually all cadmium is complexed, with humic acid being the primary ligand. Although cadmium is a rare element the median concentrations in sediments of U.S. surface-water bodies is near 2 mg/kg, which is attributed mostly to anthropogenic activities.

Chromium. In a water environment chromium is stable in six different ionic species. Between pH 5 and pH 9 $\text{Cr}(\text{OH})_2^{2-}$ and $\text{Cr}(\text{OH})_2^-$ predominate. Cr^{2+} species are rapidly oxidized by manganese oxyhydrates and slowly by oxygen. More toxic Cr^{6+} forms (originating primarily from industries) are easily reduced in natural waters and sediments by ferrous and sulfide compounds. Chromium is transported in water primarily by sediments.

Copper. The dissolved phase of copper in water includes a free ion and complexes with organic and inorganic ligands. Humic materials bind most of the total Cu. Fulvic acid forms insoluble complex with Cu^{2+} . Inorganic complexes include carbonates, nitrates, sulfides, chlorides, ammonia, and hydroxide. Copper hydroxycarbonates are slightly soluble. Coprecipitation with ferric oxyhydroxides and other oxides and adsorption on mineral (clay) surfaces are responsible for the immobilization of copper in soils and sediments. Copper is transported primarily in the solid phase. Unpolluted sediments generally contain less than 20 mg/kg of Cu.

Lead (Pb). Lead is an abundant, naturally occurring metal that exists in four stable isotopes. The accepted average value for the Pb content of the earth's crust is 15 $\mu\text{g/g}$, but ranges of 0.8 to 500 $\mu\text{g/g}$ have been reported for arable soils.

In addition to natural background concentrations and the use of leaded gasoline by automobiles, sources of Pb include atmospheric deposition, mining and smelting, and some insecticides (for example, lead arsenate, which is now banned in the United States).

The common dissolved inorganic forms of lead are the free ion, Pb^{2+} , hydroxide complexes, and possibly carbonate and sulfide pairs. At pH > 6 lead is almost totally precipitated or sorbed, hence, lead is immobile in soils and sediments; however, Zimdahl and Hasset (1977) reported that the adsorption capacity is related to the type of complexing minerals and pH. The high degree of Pb immobilization may be related directly to the soil cation exchange capacity (CEC) and inorganic matter content and inversely to pH. A regression analysis yielded the following equation for the Langmuir adsorption maximum, Q^0 , in micrograms of lead per gram of soil,

$$Q^o = 7100 + 1110 (\text{pH}) + 1105 (\text{CEC}) \quad (6.27)$$

Lead forms organic complexes with various ligands, including amino acids, polysaccharides, and fulvic and humic acids. Using data compiled by Schnoor et al. (1987), the partitioning coefficient for adsorption of lead by soil organic humic acids (HA) is about $\Pi_{\text{HA}} = 10^5 \text{ l/kg}$, and that for clays ranges from $10^{3.7} \text{ l/kg}$ for kaolin to $10^{5.5}$ for montmorillonite, respectively.

The transport of Pb from soil to receiving waters occurs almost solely by erosion, with delivery ratios similar to those by clay particles. For all practical purposes, Pb remains only in the few centimeters of soil unless redistributed by plowing.

Mercury (Hg). The background concentrations of mercury in soils range from 0.01 to 0.06 $\mu\text{g/g}$ (Krenkel, 1973; U.S. EPA, 1976). In soils, Hg combines with the exchange complex, forming ionic and covalent bonds. Mercury is strongly adsorbed by clay minerals, and the adsorption process is affected by soil pH. Evidence exists that mercury can be chelated to organic matter. Unlike other metals, mercury may also volatilize.

In sediments, mercury is methylated by bacteria. The final product of microbial methylation of mercury is the dimethyl mercury compound $(\text{CH}_3)_2\text{Hg}$ (Jernelov, 1972; Krenkel, 1973); however, no methylation has been reported in soils.

Zinc. In natural waters zinc species are limited to zinc carbonates, zinc hydroxides, zinc silicates, and the free ion. Zinc also complexes with ammonia, amines, halides, and cyanides. Zinc is more soluble than other toxic metals such as copper or nickel, but its solubility is affected by pH. Acid rainfall may greatly increase dissolution of zinc from soils and from zinc roofing materials.

Nitrogen

Accumulation of Nitrogen in Soils

Nitrogen (N) is one of the four essential elements (carbon, oxygen, hydrogen, and nitrogen) that form the basic structure of proteins. Nitrogen is also the most abundant gas in the atmosphere, accounting for about 80%. Due to its several valence states it can exist in numerous forms and is abundant in the mineral and organic fractions of the soils and sediments. However, not all N forms are related to water pollution or eutrophication of water bodies.

Soils contain 0.07 to about 0.3 of the total N or about 1500 to 6000 kg/ha in the top 15 cm. The source of soil nitrogen include fertilizer ap-

plications (46% in the United States), N fixation from the atmosphere by soil bacteria (20%), manure application (7%), plant residues (17%), and precipitation (10%). In suburban areas without sanitary sewers, significant amounts of N enter soils from the seepage of household septic systems.

In the last decades of the twentieth century, nitrogen contamination of surface and ground water resources has reached alarming proportions (see Chapter 1 for a discussion of nitrate trends). Excess nitrogen fertilizer applications in agriculture and from septic tanks have leached into the ground water, with the result that nitrate–nitrogen loads and consequent concentrations of nitrate nitrogen are steadily increasing. The result is an impairment of the use of water for drinking and the acceleration of eutrophication.

Jenny (1941) noticed that the nitrogen content of native soils depends on the average meteorological factors of the region, namely moisture and temperature. For example, soil nitrogen declines as the temperature rises or moisture declines.

Although virgin lands rich with vegetation (forested areas, swamps, prairies) originally contained high amounts of N, on cultivation the natural supply of N is depleted and fertilization may be needed to supplement soil loss of N. Most soil N (>95%) is contained in soil organic matter, or in the case of ammonium ions, it can be sorbed by clays. In these forms, N may be considered immobile and not available to plants. Nitrogen is mostly lost by erosion and through crop harvesting. However, these immobile forms of nitrogen can be converted to nitrate, which is highly mobile. Also part of the ammonia may exist in soil water as free ammonia ions (NH_4^+), which is mobile. The mobile forms of nitrogen are available to plants and can be transported by soil water and infiltrate into ground water. The time of migration of mobile N components in ground-water aquifers can extend to years and even decades. It is also suspected that the heavy contamination of central Europe by nitrates was caused by nitrate fertilizers instead of the less mobile ammoniacal forms.

Nitrogen Cycle

The behavior and transformation of N in soils, sediments of surface waters, and substrates of wetlands is complex, and the pathways from the soil to surface waters are numerous and not well defined. The process of N transformation in these media can be schematically represented by a N cycle similar to that in Figure 6.12.

In soils and sediments, N exists in four basic forms: ammonium, nitrate, organic phytonitrogen in plants and plant residues, and protein nitrogen in living and dead bacteria and small soil inhabitants. The

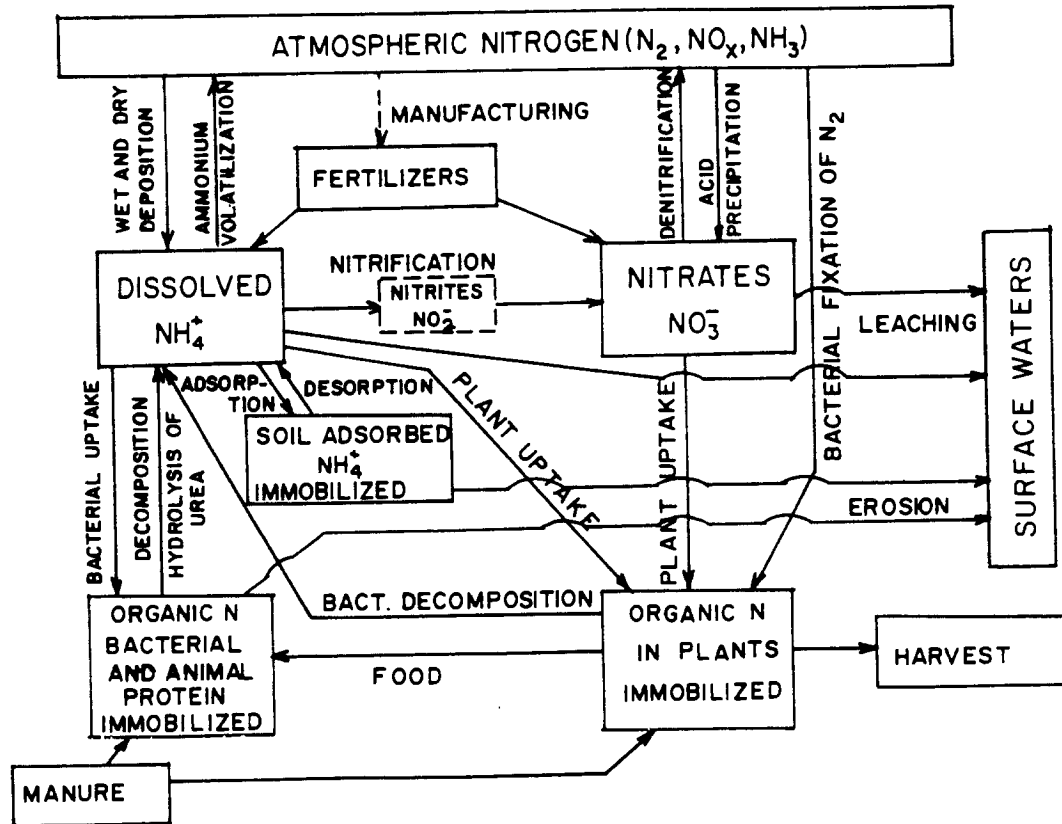


FIGURE 6.12. Nitrogen cycle in soils and sediments.

following definitions are needed for an overall understanding of the N cycle:

Nitrogen fixation is a process by which soil microorganisms in symbiosis with some leguminous plants utilize atmospheric N and change it to an organic form.

Nitrogen accumulation (bacterial uptake) is the conversion of ammonium-N to protein and cell tissue by heterotrophic soil organisms.

Ammonification is a process by which protein and other inorganic forms of N are decomposed to ammonium by a biochemical breakdown of the proteins.

Hydrolysis of urea involves conversion to ammonium ions in the presence of the enzyme urease, which is provided by many heterotrophic organisms.

Nitrification is a complex process by which ammonium-N (NH_4^+) is oxidized to nitrate (NO_3^-) with nitrite (NO_2^-) as an intermediate product. Nitrification is accomplished by two groups of bacteria: namely *Nitrosomonas* (oxidizing NH_4^+ -N to NO_2^- -N) and *Nitrobacter* (oxidizing NO_2^- -N to NO_3^- -N). Both groups of bacteria are chemo-

trophic, utilizing the exothermic reaction as their source of energy and carbon dioxide from the atmosphere or alkalinity of interstitial water as their carbon source. Nitrification is strictly an aerobic process.

Denitrification is a process that occurs under anoxic conditions that usually takes place when water fills most of the available voids and pores in soils. In aquatic sediments and substrates of wetlands anoxic conditions always prevail, so only a very few millimeters thick surface sediment layer can become aerobic. In wetlands rhizomes of some plants have the capability of transferring oxygen down to their roots and creating aerobic pockets near the roots (see Chapter 14). During denitrification NO_3^- -N serves as an electron acceptor and is reduced to gaseous forms including N_2 , N_2O , NO , and NO_2 .

Fixation of ammonium involves the sorption of NH_4^+ in between the layers of expanding clay minerals such as monmorillonite. In this form, ammonia is considered unavailable for plant growth or bacterial uptake.

Ammonia volatilization may occur at high soil pH values when the ammonium ion (NH_4^+) is converted to gaseous ammonium (NH_3), which volatilizes and is lost to the atmosphere.

Most of the reactions of the N cycle (with the exception of NH_4^+ fixation and NH_3 volatilization) are microbial, and thus their rates are sensitive to temperature, moisture content, and aeration. Warm (32°C) and partially moist soils (80% of voids filled with water) are the optimum conditions for N cycling in soils (Steward et al., 1975).

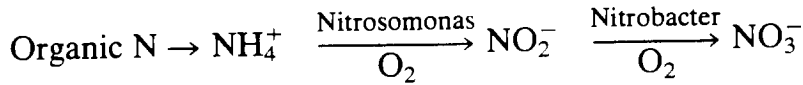
Unlike phosphorus, most of which is adsorbed on soil particles and may be controlled by erosion control and soil conservation, the available N is mobile in soils and may leach to ground water and reappear with ground-water discharge in the base flow of streams. Seepage represents the major pathway of nitrogen loss from agricultural areas and pervious urban and suburban lands. Erosion control may not be effective for reducing nitrogen contamination of surface and ground-water resources.

Since the mobile components— NH_4^+ and NO_3^- —are carried by soil water, downward or lateral N movement occurs only if the moisture content is above that for gravitational water (0.3 bar tension). Maximum movement occurs when soil moisture content is near saturation, and decreases rapidly with decreasing moisture content. Thus, between rains or irrigation, N movement is slow or nonexistent.

Nitrification and Denitrification

Over 90% of the fertilizer used in the United States is in the form of ammonium salts. Most organic N originating from manure application or

from septic tanks can be quickly decomposed to ammonium. In addition, urea is readily hydrolyzed to ammonium. If the ammonium is applied to an aerated, microorganism-rich soil such as farmland, nitrification occurs, resulting in the conversion of NH_4^+ to NO_3^- . In aerated soils or in the top layer of sediments, the nitrification process proceeds as follows:



The last reaction, that is, conversion of NO_2^- to NO_3^- is faster than the conversion of NH_4^+ to NO_2^- . Consequently, very little nitrite accumulates in soils and sediments.

Nitrates can readily move with the soil moisture front. The optimum temperature for nitrification is 22°C (Fig. 6.13), and the rate of nitrification decreases rapidly on both sides of the temperature curve. Stanford, Free, and Swaininger (1973) stated that nitrification essentially ceases below 10°C and above 45°C. Nitrification also depends on the pH of the soil and its moisture content (Figs. 6.14 and 6.15). Since nitrifying bacteria depend on water as their living environment, the nitrification rate decreases with decreasing moisture content.

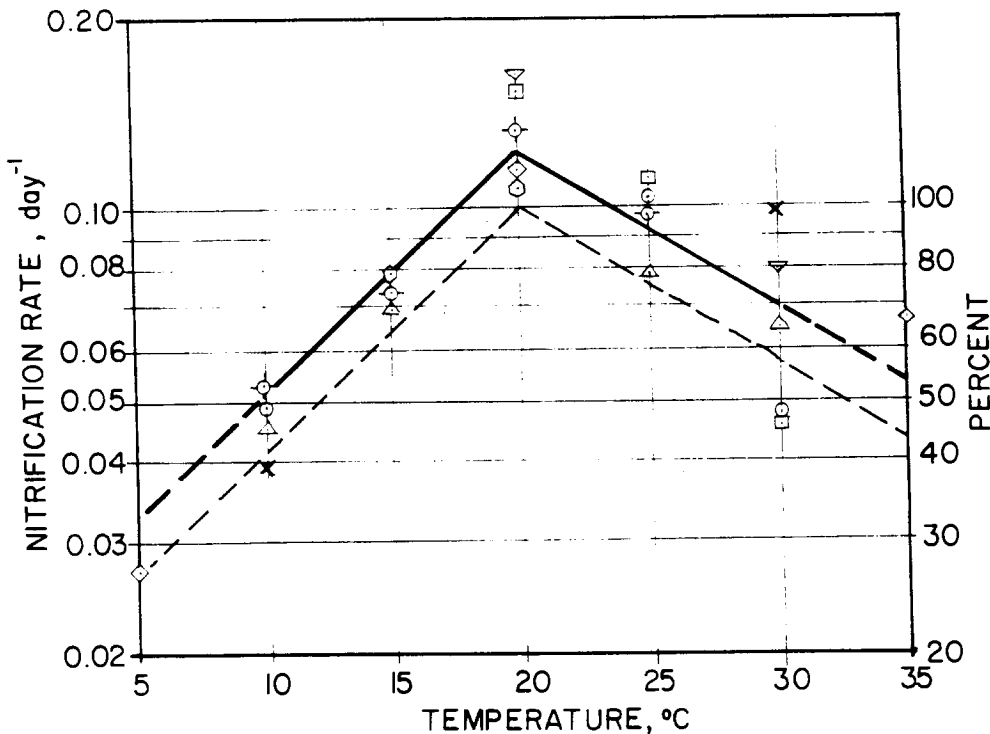


FIGURE 6.13. Dependence of nitrification rate on temperature. (From Zanoni, 1969.)

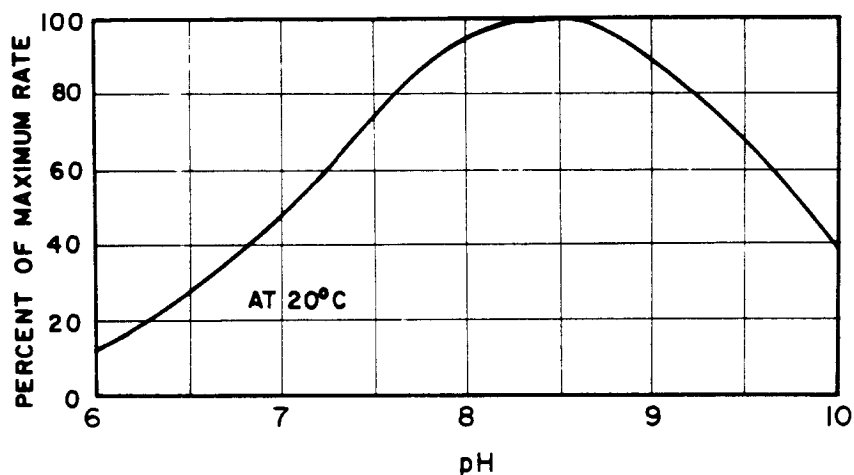


FIGURE 6.14. The effect of pH on the rate of nitrification. (From Wilde, Sawyer, and McMahon, 1971.)

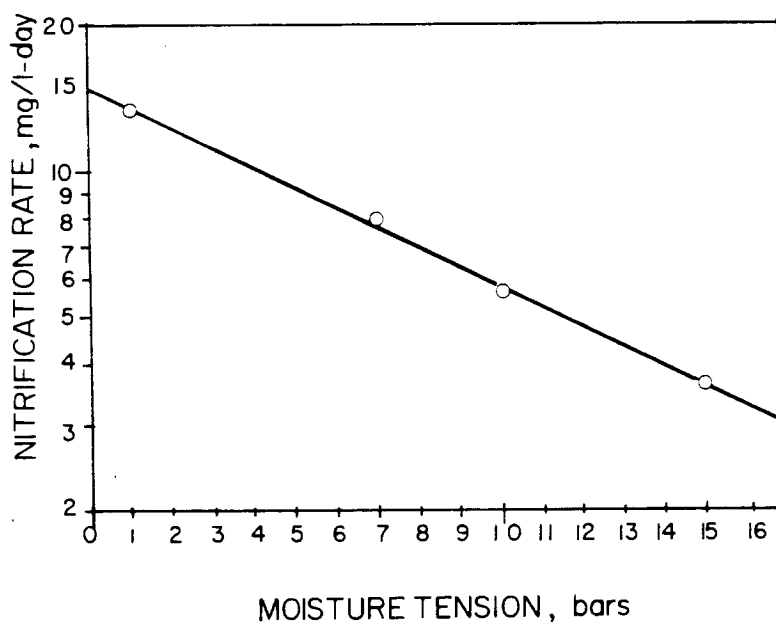


FIGURE 6.15. The dependence of the soil nitrification rate on the moisture tension in soils. For submerged sediments, moisture tension = 0. (After Justice and Smith, 1962.)

By contrast with many heterotrophic microorganisms, the growth of nitrifiers is slow and cell yield per unit of energy source oxidized is low. For practical purposes the NH_4^+ oxidation may be simulated by an equation similar to the Michaelis–Menten–Monod equation (Eq. (6.14)). The value of the rate coefficient for soil at 20°C was suggested as $K_N = \mu X = 0.41 \text{ mg/l-day}$ and the half-saturation constant was $K_s = 1.1 \text{ mg/l}$,

respectively, if the concentration of dissolved ammonia in pore water, C_d , was expressed in milligrams per liter (McLaren, 1969, 1971).

In the absence of oxygen or if the oxygen supply is depleted to a point below the oxygen demand, NO_3^- is reduced mostly to gaseous nitrogenous forms (largely N_2) by denitrification. Denitrifying bacteria are heterotrophic facultative microorganisms, a part of the soil microflora.

The process of denitrification usually occurs in subsoils with low permeability, in soils saturated with water for an extended period, such as in wetlands or after prolonged rainfall. Pratt, Jones, and Hunsaker (1972) reported that denitrification might account for more than 50 percent of N losses in clayey soils. On the other hand, Carter and Allison (1960) concluded that very little, if any, N loss occurred under aerobic conditions, except where higher dosages of dextrose or other carbon-rich compounds were added in the presence of nitrates. Hence they stated that denitrification is of minor importance in soils that are maintained aerobic.

Generally the first-order decay reaction (Eq. (6.15)) is used to describe the reaction where the nitrate concentration in the interstitial water is substituted for C_d . The magnitude of the denitrification coefficient, K_b ,

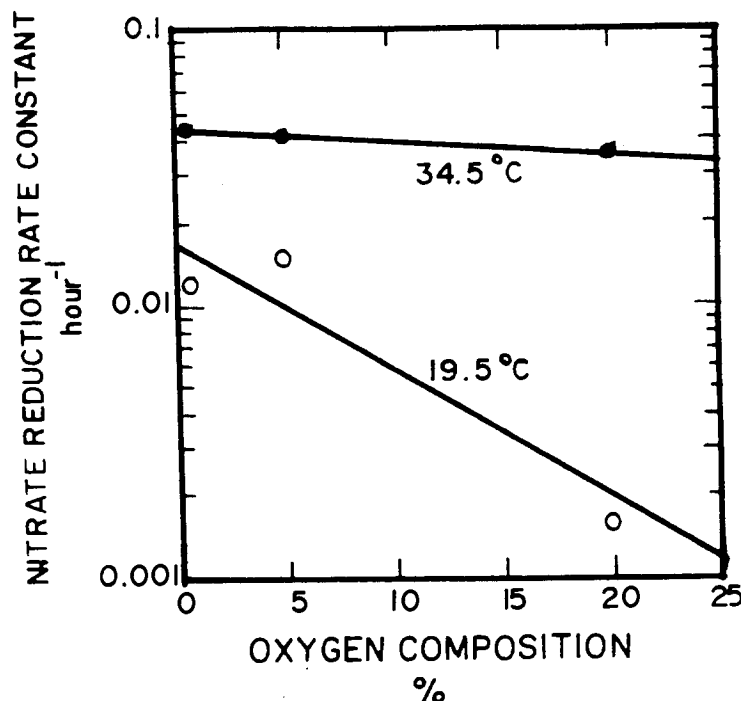


FIGURE 6.16. Dependence of denitrification rate on oxygen composition and temperature. (From Misra et al., 1974. Reprinted with permission of the publisher, Soil Science Society of America, Madison, WI.)

measured by Misra, Nielsen, and Biggar (1974) is shown on Figure 6.16. The denitrification rate is a function of easily decomposable carbon that is derived from the organic carbon sources in the soil/sediment. The rate is expected to decrease with the decrease in the available energy source.

Nitrogen Fixation

Legumes (soybean, peas, beans), as well as some aquatic algae, fix nitrogen from the atmosphere by symbiotic microorganisms. It has been assumed that in soils the rate of nitrogen fixation can be related to root growth (Duffy et al., 1975)

$$N_f = K_f r_g \quad (6.28)$$

where

N_f = rate of fixation of N (mg N/day-cm²)

K_f = a constant (0.011 mg N/cm²)

r_g = the rate of root growth (cm/day)

Ammonification of Nitrogen

Painter (1970) described three ways of producing NH_4^+ from organically bound N:

1. From extracellular organic N compounds (e.g., urea) chemically or biochemically.
2. From living bacterial cells during endogenous respiration when cells are becoming smaller.
3. From dead and lysed cells.

Very few data are available on the breakdown of various N containing components to NH_4^+ .

Hydrolysis of urea is relatively fast, much faster than the rate of nitrification. Its rate is also temperature dependent. The process of hydrolysis of urea is important where manure or urea-containing sewage or septage is applied to field or land. Soil organic N—other than urea—breaks down at a much slower rate. Most of the organic N is not directly available to plants and must be initially converted to NH_4^+ . Straford, Free, and Swaininger (1973) concluded that in the mineralization of soil organic N other than urea, the rate-limiting process is ammonification. Ammonification can be described by the first-order decay equation (Eq. (6.15)); however, with the remaining soil N as substrate, or

$$\frac{dN}{dt} = -K_N N \quad (6.29)$$

Straford, Free, and Swaininger (1973) found that the coefficient, K_N , is statistically uniform for all investigated soils, that is, $K_N = 0.29 \pm 0.007$ weeks⁻¹ at 25°C. As with all biochemical reactions, the coefficient, K_N , is temperature dependent. An Arrhenius' plot yielded the following equation for the overall mineralization of organic N:

$$\log K_N = 6.16 - \frac{2299}{T} \quad (6.30)$$

where T = temperature in degrees Kelvin (= 273 + °C)

Nitrogen Immobilization

Part of the soil or sediment N is dissolved and can move readily with soil moisture and ground water. The dissolved fractions include NO_3^- and NH_4^+ . But a significant portion of n components, especially NH_4^+ and almost all of organic N, can be immobilized. On the other hand, NO_3^- is always dissolved and mobile.

Nitrogen immobilization in soils results from physical-chemical attractions, chemical precipitation, biochemical reactions, and nitrogen uptake by soil organisms and plants. The known nitrogen fixation processes include:

1. Ammonium fixation by clay minerals.
2. Ammonium fixation by lignin-derived substances contained in soil organic matter.
3. Reactions of amino acids derived from plant materials and microbial synthesis with quinones and subsequent polymerization.
4. Biological immobilization, which involves NH_4^+ uptake by heterotrophic bacteria participating in the decay of organic matter in soils.

The immobilized N is not readily available for mineralization and biological uptake, and it appears, as pointed out in the preceding paragraph, that ammonification is much slower than nitrification.

The processes of immobilization and ammonification of soil N depend on the chemical composition of the material undergoing decomposition, primarily its C:N ratio. Plant residues having larger percentage of readily available C stimulate the growth of microbial cells when incorporated in soil under aerobic conditions. Since nitrification is faster than ammonification, the end product of aerobic decomposition is nitrate. Anaerobic degradation of proteinaceous materials, such as those occurring in subsurface sediment layers, is slower and results in the accumulation of ammonia.

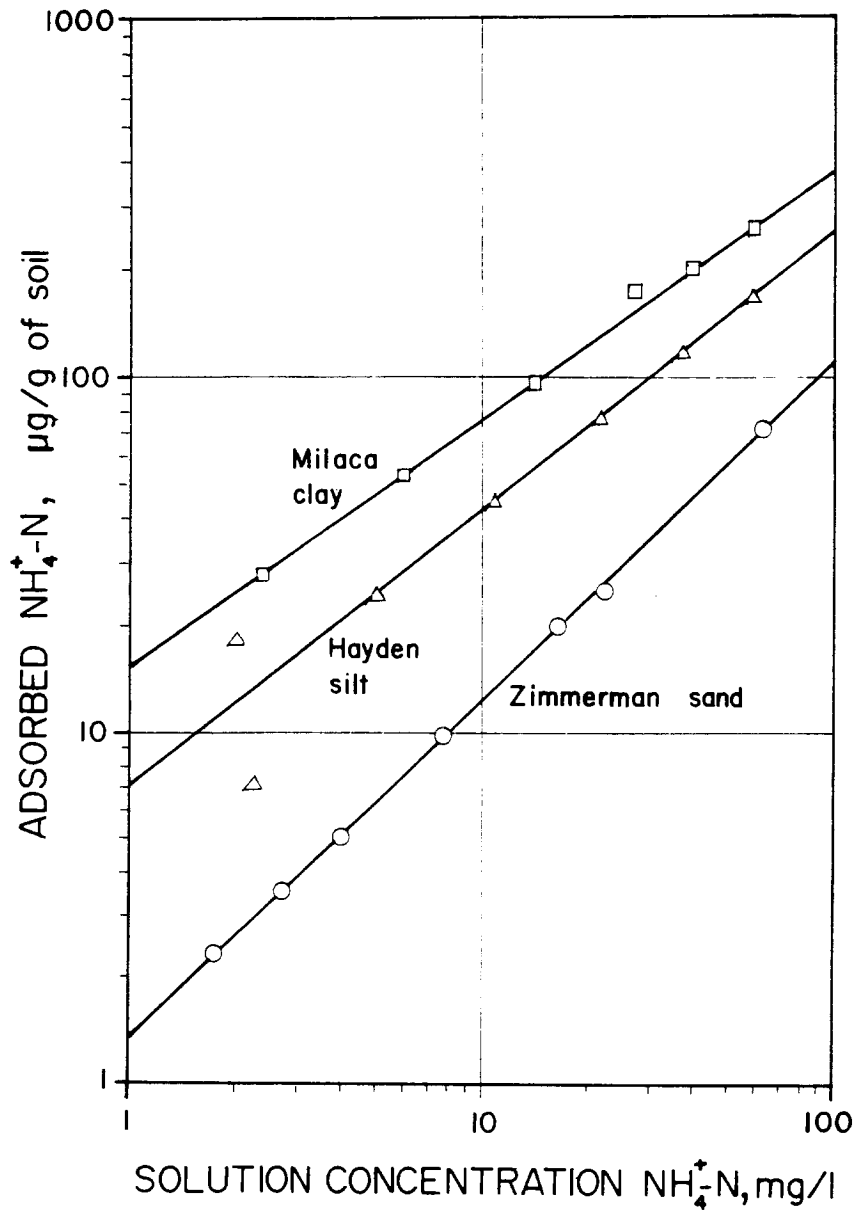


FIGURE 6.17. Freundlich adsorption isotherms for adsorption of ammonia on soils. (From Preul and Schoepfer, 1968.)

Shaffer, Dutt, and Moore (1969) statistically analyzed the rate of NH_4^+ immobilization using the multiple regression technique. The proposed equation was

$$R = \frac{d(\text{NH}_4^+)}{dt} = 0.892 - 0.00216T - 0.027 (\text{organic N}) + 0.392 \log_{10}(\text{NH}_4^+) \quad (6.31)$$

where T is the temperature in $^{\circ}\text{C}$.

Preul and Schoepfer (1968) investigated N adsorption by soils. The Freundlich adsorption isotherms for three soils are shown in Figure 6.17. Bailey (1968) has shown that ammonium is adsorbed by clays in exchangeable form and by organic matter. Much of the NH_4^+ adsorbed by various kinds of organic matter is in a nonexchangeable form and is resistant to decomposition.

Volatilization of Ammonia

For soils of high pH and NH_4^+ content, ammonia (NH_3) can be lost by volatilization. Significant amounts of ammoniacal N (11% to 60%) can be lost from applied sludge, manure, or chemical fertilizers applied to calcareous soils (Ryan and Keeney, 1975; Terry et al., 1978). Losses decrease as clay content increases and more NH_4^+ is immobilized. Since the reaction is pH dependent, all volatilization practically ceases if pH is at or below 7. Note that soil pH is inversely correlated to the soil clay content. Other factors affecting volatilization include moisture content, application rates, and organic content. Due to its effect on pH, liming of soils may enhance ammonium volatilization.

Enrichment and Delivery of Nitrogen during Overland Flow

Due to the fact that N exists in soils in several forms, the notion of enrichment and delivery may be meaningless.

Example 6.6: Distribution between Mobile and Immobile N in Soil

Silt soil contains about 2.5% organic matter and 0.12% total N measured as TKN (total Kjeldahl nitrogen = organic N + NH_4^+ - N), of which about 75% is organic N. Estimate the approximate magnitudes of mobile (available) and immobile N. Effective soil moisture $\theta_{\text{eff}} = 20\%$ and the specific density of the soil $\rho = 1500 \text{ kg/m}^3 (= \text{g/l})$.

Solution From Figure 6.17, the Freundlich isotherm for silt has the approximate form

$$r = 7.0 C_d^{0.8}$$

Organic N constitutes part of the soil organic matter, and for the most part no equilibrium exists between the dissolved and particulate fractions. The dissolved fraction of TKN is a part of the NH_4^+ content, since ammonia can exist both dissolved and adsorbed. Since total NH_4^+ is $(1 - 0.75) \times 0.12\% = 0.03\%$ or $300 \mu\text{g/g}$ of soil, the volumetric concentration is $300 \mu\text{g/g} \times 1500 \text{ g/l} \times 0.001 \text{ mg}/\mu\text{g} = 450 \text{ mg/l}$. Then

$$450 \text{ (mg/l)} = \rho r + \theta_{\text{eff}} C_d = 1500 \text{ (g/l)} \times 7.0 \times C_d^{0.8} \\ \times 0.001 \text{ (mg/}\mu\text{g)} + 0.2 \times C_d$$

Solving this equation gives $C_d = 103.5 \text{ mg/l}$. Also the adsorbed ammonia concentration is $r = 7.0 \times 103.5^{0.8} = 286 \mu\text{g/g}$.

The ratio of available NH_4^+ to TKN is

$$100 \times \{103.5 \text{ [mg/l]} \times 0.2 / (0.0012 \times 1500 \text{ [g/l]} \times 1000 \text{ [mg/g]})\} = 1.15\%$$

Example 6.7: Nitrogen Loss Estimation during a Storm

A 3-cm storm resulted in 1 cm of surface runoff and 1.3 tonnes/ha of soil loss. Estimate N loss using the N distribution computed in the previous example. Porosity of the soil is $\rho = 45\%$ and $\theta_{15\text{bar}} = 10\%$. Assume that the enrichment ratio for organic nitrogen is the same as that for organic matter. For this particular example assume that $\text{ER}_{\text{OR}} = 1.2$.

Solution Recall that organic N is 75% of the TKN content and that the adsorbed ammonia is $286 \mu\text{g/g}$ ($= \text{g/tonne}$). The total TKN content is $0.12\% = 1200 \text{ g/tonne}$, θ_{eff} is 0.2, and dissolved NH_4^+ is 103.5 mg/l . Then the fixed N loss

$$Y_{SN} = (0.75 \times 1200 \text{ [g/tonne]} + 286 \text{ [g/tonne]}) \times 1.3 \text{ [tonnes/ha]} \times 1.2 \\ = 1850 \text{ g/ha}$$

In order to determine the loss of dissolved N, the depth of mixing of the surface runoff with soil moisture containing the dissolved fraction must be known or assumed. Using the same reasoning as in Example 6.4, assume the mixing depth as 5 mm. Then from the mass balance

$$(\theta_{\text{eff}} + \theta_{15\text{bar}}) \times 0.5 \text{ (cm)} \times C_d = \{\text{porosity} \times 0.5 \text{ (cm)} + 3 \text{ (cm)}\} C'$$

from where

$$C' = C_d \{(0.2 + 0.1) \times 0.5\} / \{0.45 \times 0.5 + 3\} \\ = 103.5 \times 0.0465 = 4.81 \text{ mg/l } (= \text{g/m}^3)$$

and the dissolved N loss in the surface runoff is

$$Y_{DN} = 4.81 \text{ (g/m}^3) \times 1 \text{ (cm)} \times 0.01 \text{ (m/cm)} \times 10,000 \text{ (m}^2\text{/ha)} = 481 \text{ g/ha}$$

The total N loss will

$$Y_N = Y_{DN} + Y_{SN} = 481 + 1850 = 2331 \text{ g/ha.}$$

Example 6.8: Mineralization of Ammonia

How much nitrogen will be mineralized during a 2-week dry period if the temperature is 22°C and the soil pH is 6.8? Assume average soil moisture tension close to 0.3 bar. Use the N values estimated in the previous examples.

Solution Only dissolved ammonia can be mineralized. Adsorbed or organic N must first be released into the solution before it can be mineralized.

Since the saturation constant for mineralization of ammonia is about $K_s \approx 1.1 \text{ mg/l}$, and the dissolved ammonia concentration is $C_d = 103.5 \text{ mg/l}$, it is evident that the reaction described by the Michaelis–Menton–Monod equation (Eq. (6.14)) will follow a zero-order pattern and will, for most of the time, not depend on the NH_4^+ concentration in the soil water solution. Then

$$\frac{dC_d}{dt} = \frac{K_N C_d}{K_s + C_d} = \frac{0.41 \times 103.5}{1.1 + 103.5} = 0.41 \text{ mg/l/day}$$

Since the temperature is near optimum, no thermal correction is necessary as indicated from Figure 6.13. However, Figure 6.14 shows that at pH 6.8 the reaction will proceed only at about 50% of the optimal rate. Because the moisture tension is close to 0.3 bar, there is no correction for moisture. Then

$$\frac{dC_d}{dt} = 0.2 \text{ mg/l-day}$$

and in 14 days about 2.8 mg/l of the dissolved NH_4^+ will be mineralized to NO_3^- . This form as well as the dissolved NH_4^+ can be taken up by plants and move with the soil water front.

Simple Steady-State Loading Function for Nitrate Loading to Ground Water

Mills et al. (1985) published a simple steady-state loading function that presumes a constant annual load of nitrogen fertilizer or waste water in the form of organic N. The steady-state loading function is

$$L = 1000N \times [1 - v(1 - F)] - Cu \quad (6.32)$$

where

L = annual steady-state nitrate–nitrogen load to ground water (kg/ha)

X = average annual solids waste–organic fertilizer application rate (tonnes/ha)

N = nitrogen fraction of applied compound

F = organic fraction of applied compound

v = fraction of applied nitrogen that volatilizes (a soil-pH-dependent variable)

Cu = average crop uptake of nitrogen

Table 6.4 provides the values of nitrogen uptake by crops (note that only nitrogen that is removed with crops is considered).

Soil Microorganisms

Soils contain enormous densities of microorganisms that decompose soil organic materials and/or are an integral part of the soil organic composition. Most of the soil microbial population is contained in the A horizon.

Soil contains five major groups of microorganisms (Alexander, 1977): bacteria, actinomycetes, fungi, algae, and protozoa. These organisms as well as their organic and inorganic food are a part of the soil ecosystem.

Bacteria are the most abundant group of microorganisms. They perform most of the decomposing processes in the soils. As pointed out they can be divided into autotrophic (algal) and heterotrophic groups, and based on their oxygen environment, they may be aerobic, anaerobic, or facultative. Based on their origin they can be an autochthonous species, which are the true residents of soils, or allochthonous species or invaders that have entered soils from precipitation or sewage effluents, manure, and sludge applications and from septic tanks. The allochthonous species are the most important in the soil-decomposition processes. The latter group of bacteria (allochthonous) may persist in soils for some time, but do not contribute significantly to any activity of the soil microflora. However, if soil has a low microbial population, application of organic microorganism-rich compounds, such as manure or sewage, may help restore the soil microbial population.

According to Alexander (1977) plate number counts of bacterial densities in soils may range from a few thousand to up to 200 million bacteria per gram of soil. However, microscopic counts have revealed that the microbial densities can be higher, probably up to 10^{10} of micro-

organisms/gram of soil. The bacterial population represents between 0.015% and 0.05% of the total mass of fertile soils, or some 300 to 3000 kg live weight of soil bacteria per hectare.

Most of the bacteria in soils are facultative heterotrophs. During dry, well-aerated conditions these bacteria decompose organic matter as aerobes. During flooding or periods of high moisture content, the facultative bacteria or strict anaerobes use other compounds as electron acceptors, including NO_3^- , SO_4^{--} , and Fe^{3+} ions.

The final product of aerobic decomposition is CO_2 . Thus the atmosphere of aerated soils and exposed sediments during intensive decomposition processes has depleted O_2 and increased CO_2 . This results in increased partial pressure of CO_2 in the soil atmosphere, which in the air above ground level is about 1 mbar, while that in soil may reach 100 mbars. Consequently, by the interaction of the CO_2 with interstitial water the pH may be reduced by 1 pH unit or more when saturated soils or submerged sediments are exposed to the atmosphere and become aerated.

Under unaerobic conditions both CO_2 and methane (CH_4) are produced in various proportions. Also the microorganisms in an anaerobic environment may be capable of decomposing such compounds as cellulose and cyclic organics, reduce nitrates to nitrogen gas, and convert DDT to DDE. However, as a rule, aerobic decomposition is more rapid and complete for most other organic and inorganic biodegradable compounds.

Pathogenic Microorganisms in Soils

Various pathogenic (disease-causing) microorganisms (bacteria and viruses) can be found in soils. Their survival in soils depends on many environmental factors and, often, on the availability of an acceptable host organism. Because of the contamination of soils with animal droppings, manure, sewage and sludge, and the tissues of diseased plants, the survival of these organisms in soils has been investigated (Lance, 1978; Loehr et al., 1979; Schaub and Sorber, 1977; Pettygrove and Assano, 1985; Reed, Thomas, and Kowal, 1980).

Pathogenic bacteria. Pathogenic bacteria may survive in soil for a period of from a few hours to several months, depending on the type of organisms, type of soil, the moisture capacity of the soil, moisture and organic content of the soil, pH, temperature, sunlight, rain, and predation by the resident microflora in the soil. In general, enteric bacteria persist in soil for 2 to 3 months; however, under certain favorable conditions pathogenic microorganisms may actually multiply (Loehr et al., 1979). Factors that influence the survival of bacteria in soil are listed in Table 6.6, while the average survival of selected pathogens are presented in

TABLE 6.6 Factors that Affect the Survival of Enteric Bacteria and Viruses in Soil

Factor		Comment
pH	Bacteria	Shorter survival in acid soils (pH 3 to 5) than in neutral and calcareous soils
	Viruses	Insufficient data
Predation by soil microflora	Bacteria	Increased survival in sterile soil
	Viruses	Insufficient data
Moisture content	Bacteria and viruses	Longer survival in moist soils and during periods of higher rainfall
Temperature	Bacteria and viruses	Longer survival at lower temperatures
Sunlight	Bacteria and viruses	Shorter survival at the soil surface
Organic matter	Bacteria and viruses	Longer survival or regrowth of some bacteria when sufficient amounts of organic matter are present

Source: After U.S. EPA (1977).

TABLE 6.7 Survival of Selected Pathogens in Soils

Organism	Range of Survival Time (days)
<i>Salmonella</i>	15 to more than 200
<i>Salmonella typhi</i>	1 to 200
Tubercle bacilli	More than 200
<i>Entamoeba histolytica</i> cysts	6 to 8
Enteroviruses	8
<i>Ascaris</i> ova	Up to 7 days
Hookworm larvae	42

Source: After Parson et al. (1975).

Table 6.7. For example, the survival of *E. coli*, *S. typhi*, and *M. avium* is greatly enhanced in moist rather than dry soil. Survival time is less in sandy, permeable soil than in soil with a greater water holding capacity such as clayey soil and peat (Loehr et al., 1979).

Both pathogens and indicator microorganisms (coliforms) survive longer under lower temperatures. Figure 6.18 shows typical die-off curves for fecal coliforms and fecal streptococci under summer and winter conditions. The competition and predation by resident soil bacteria is another important factor. Organisms invading sterilized soils survive longer than they would in unsterilized soil.

Bacteria move in the soil with soil water. However, since fine soil particles can effectively adsorb bacteria, the soil is, in general, a very effective filtering medium. Bacteria are more mobile in sandy soils under

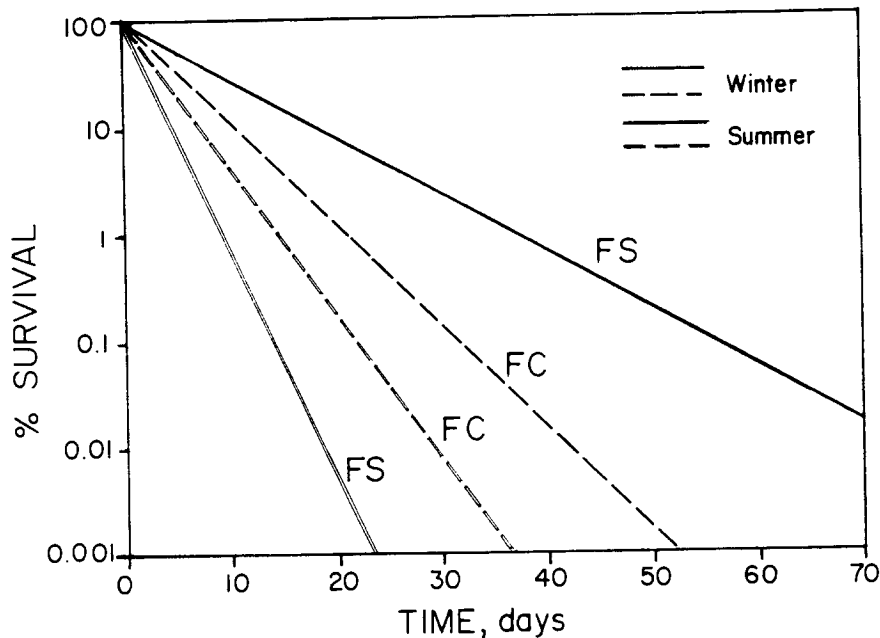


FIGURE 6.18. Die-off curves for fecal coliforms (FC) and fecal streptococci (FS) under winter and summer conditions. (From Von Donsel in Loehr et al., 1979.)

high moisture conditions than in loamy or clayey soils. In fine-textured soils, bacteria can be effectively filtered out by 1 to 2m of soil. Soil containing clay remove most microorganisms through adsorption. Sandy soils remove them through filtration (Lance, 1978).

Viruses. Viruses in soils are also considered invaders. Loehr et al. (1979) summarized the available information on virus survival as follows:

1. Virus survival in soil depends on the nature of the soil, temperature, pH, moisture, and possibly antagonism from soil microflora.
2. Viruses readily adsorb on soil particles. However, such viruses bound to soil particles are as infectious as free viruses.
3. Viruses survive from as short as 7 days and as long as 6 months in soil. Climatic conditions, particularly temperature, have a major effect on survival time.
4. Enteric viruses can survive from 2 to more than 188 days in fresh water, temperature being the most important factor, with survival greater at lower temperatures.
5. Virus survival in crops is shorter than in soil, because viruses are more exposed to deleterious environmental effects.
6. Contamination of crops most commonly occurs when wastewater comes into contact with the surface of the crop.

TABLE 6.8 Soil Adsorption Characteristics for Viruses: Freundlich Adsorption Isotherms

Soil	pH	Cation Exchange Capacity (meq/100 g)	Specific Surface Area (m ² /g)	Organic Carbon (%)	Freundlich Adsorption Coefficients ^a	
					K _F	1/n
Aastad clay loam	6.9	35.2	160	3.02	72.5	0.945
Kranzburg silt loam	6.2	31.4	154	2.38	161	0.908
Palouse silt loam	6.0	22.6	89.2	1.82	45.7	1.24
Parshal silt loam	6.8	13.7	67.6	1.32	4.6	0.92
Quinici loamy sand	7.2	8.9	35.3	0.35		

Source: After Burge and Enkiri (1978), with permission of ASA, CSSA, and SCS.

^aIn number of viruses per gram of soil C_d in number of viruses per milliliter.

7. In rare cases, the translocations of animal viruses from the roots of plants to the aerial parts can occur.
8. Sunlight is believed to be a major factor in killing viruses.
9. Viruses cannot reproduce at all in the soil, and they slowly die off.

Unlike bacteria, for which filtration appears to be the main factor limiting their movement through the soil, viruses are most effectively removed by adsorption on soil particles. The process of adsorption is strongly affected by soil pH. The best adsorption of viruses is achieved at pH 7, but it decreases on both the acid and alkaline sides of the pH scale (U.S. EPA, 1977). Adsorption of viruses does not mean their complete immobilization, since desorption of viruses can occur when pH or other environmental conditions change. Generally, virus adsorption follows the Freundlich isotherm model (Table 6.8). Viruses can be desorbed from soil particles following a heavy rain.

Scheuerman et al. (1979) showed that in organic soils, water-soluble humic substances interfere with the sorptive capacity of soil toward viruses. Wetland soil, such as peat and mulch, were not effective in retaining viruses, and significant viral concentrations were observed in the leachate after land disposal of sewage. One could suspect that similar conclusions can be applied to organic sediments. Sandy soils or sediments with little water-soluble humic substances may retain most or all of the viruses applied.

References

- Adams, R. S. (1973). Factors influencing soil adsorption and activity of pesticides, *Residue Rev.* 47:1-54.

- Ahuja, L. R., et al. (1982). Modeling the release of phosphorus and related adsorbed chemicals from soil to overland flow, in *Modeling Component of Hydrologic Cycle*, Water Resources Publ., Littleton, CO.
- Alexander, M. (1977). *Introduction to Soil Microbiology*, Wiley, New York.
- Armstrong, D. E., and G. Chesters (1968). Adsorption catalyzed chemical hydrolysis of atrazine, *Environ. Sci. Technol.* **2**:683-689.
- Bailey, G. W. (1968). *Role of Soils and Sediment in Water Pollution Control. Part I, Reactions of Nitrogenous and Phosphatic Compounds with Soils and Geological Strata*, FWQA (now EPA) South-east Water Laboratory Report, Athens, Georgia.
- Bouman, M. C., M. S. Schechter, and R. L. Carter (1965). Behaviour of chlorinated insecticides in a broad spectrum of soil types, *J. Agr. Food Chem.* **13**:360-365.
- Braman, R. S., and C. C. Foreback (1973). Methylated forms of arsenic in the environment, *Science* **182**:1247-1249.
- Bruce, R. R., et al. (1975). A model for runoff of pesticides from small upland watersheds, *J. Environ. Qual.* **4**:541-548.
- Burge, W. D., and N. N. Enkiri (1978). Virus adsorption by five soils, *J. Environ. Qual.* **7**:73-76.
- Burns, L. A., et al. (1990). *PIRANHA—Pesticide and Industrial Chemical Risk Analysis and Hazard Assessment-Version 1.0*, Environmental Research Laboratory, U.S. Environmental Protection Agency, Athens, GA.
- Cars, J., A. W. Taylor, and E. R. Lemon (1971). Measurement of pesticide concentrations in air overlying field, in *Proceedings, International Symposium on Identification and Measurement of Environmental Pollutants*, Ottawa, Ontario, pp. 72-77.
- Carter, J. N., and F. E. Allison (1960). Investigation of denitrification in well aerated soils, *Soil Sci.* **90**:173-177.
- Chesters, G. (1986). *Pesticide Transformations and Movement in Soils*, Water Resources Center, University of Wisconsin, Madison, WI.
- Chesters, G., and G. V. Simsiman (1975). *Impact of Agricultural Use of Pesticides on Water Quality of the Great Lakes*, Task A, Water Resources Center, University of Wisconsin, Madison, WI, and International Joint Commission, Windsor, Ontario.
- Christenson, T. H. (1984). Cadmium soil adsorption at low concentrations: Effect of time, cadmium load, pH, and calcium, *Water, Air, Soil Pollut.* **21**:105-114.
- Combust, K. B. (1991). Trace metals in sediments: Spatial trends and sorption process, *Water Resour. Bull.* **27**:19-28.
- DiToro, D. M., et al. (1989). *Briefing Report to the EPA Science Advisory Board on the Equilibrium Partitioning Approach to Generating Sediment Quality Criteria*, EPA 440/5-89-002, Office of Water Regulation and Standards, U.S. Environmental Protection Agency, Washington, DC.
- DiToro, D. M., et al. (1990). Toxicity of cadmium in sediments: The role of acid volatile sulfide, *Environ. Toxicol. Chem.* **9**:1487-1502.
- DiToro, D. M., et al. (1991). Technical basis for establishing sediment quality

- criteria for non-ionic organic chemicals using equilibrium partitioning, *Environ. Toxicol. Chem.* **10**(12):1541–1583.
- Dong, A., G. Chesters, and G. V. Simsiman (1983). Soil dispersibility, *Soil Sci.* **136**(4):208–212.
- Dong A., G. Chesters, and G. V. Simsiman (1984). Metal composition of soil, sediment, and urban dust and dirt samples from the Menomonee River watershed, Wisconsin, U.S.A., *Water, Air Soil Pollut.* **22**:257–275.
- Dong A., G. V. Simsiman, and G. Chesters (1983). Particle size distribution and phosphorus levels in soil, sediment and urban dust and dirt samples from the Menomonee River Watershed, Wisconsin, *Water Res.* **17**(5):569–577.
- Donigian, A. S., D. C. Beyerlein, H. H. Davis, and N. H. Crawford (1977). *Agricultural Runoff Management (ARM) Model Version. II: Refinement and Testing*, EPA 600/3-77/098, U.S. Environmental Protection Agency, Athens, GA.
- Duffy, J., C. Chung, C. Boast, and M. Franklin (1974). A simulation model of bio-physicochemical transformation of nitrogen in tile drain corn belt soil, *J. Environ. Quality* **4**:477–485.
- Edwards, C. A. (1970). *Persistence of Pesticides in the Environment*, CRC Press, Cleveland, OH.
- Enfield, G. G. (1974). Rate of phosphorus sorption by five Oklahoma soils, *Soil Sci. Soc. Am. Proc.* **38**:404–410.
- Foy, C. L., and S. W. Bingham (1969). Some research approaches toward minimizing herbicidal residues in the environment, *Residue Rev.* **29**:105–135.
- Gillespie, R. A., and R. F. Vaccaro (1978). A bacterial bioassay for measuring the copper chelating capacity of sea water, *Limnol. Oceanogr.* **23**:543–548.
- Green, P. B., T. J. Logan, and N. E. Smeck (1978). Phosphate adsorption-desorption characteristics of suspended sediments in the Maumee River Basin of Ohio, *J. Environ. Qual.* **7**:208–212.
- Goring, G. A., and J. W. Hamaker (1972). *Organic Chemicals in the Soil Environment*, Marcel Dekker, New York.
- Guenzi, W. D., and W. E. Beard (1967). Anaerobic conversion of DDT and DDE in soils, *Science* **156**:1–2.
- Guenzi, W. D., and W. E. Beard (1968). Anaerobic conversion of DDT to DDE and aerobic stability of DDT in soils, *Soil Sci. Soc. Am. Proc.* **32**:522–524.
- Hague, R., D. W. Schmedding, and V. H. Freed (1974). Aqueous solubility, adsorption, and vapor behavior of polychlorinated biphenyl arochlor 1254, *Environ. Sci. Technol.* **8**:139–142.
- Haith, D. A., and L. J. Tubbs (1981). Watershed loading functions for nonpoint sources, *J. Environ. Eng. Div. ASCE* **106**(EE1):121–137.
- Hart, B. T. (1981). Trace metal complexing capacity of natural waters: A review, *Environ. Technol. Lett.* **2**:95–110.
- Helling, C. S., P. C. Kearney, and M. Alexander (1971). Behavior of pesticides in soils, *Adv. Agron.* **23**:147–240.
- Hern, S. C., and S. M. Melacon (1986). *Vadose Zone Modeling of Organic Pollutants*, Lewis Publ., Chelsea, MI.

- Hsu, P. H. (1965). Fixation of phosphate by aluminum and iron in acidic soils, *Soil Sci.* **99**:398-400.
- Hsu, P. H., and M. L. Jackson (1960). Inorganic phosphorus transformation by chemical weathering in soils as influenced by soil pH, *Soil Sci.* **90**:16-24.
- Jenny, H. (1941). *Factors of Soil Formation*, McGraw-Hill, New York.
- Jernelov, A. (1972). Factors in the transformation of mercury to methyl mercury, in *Environmental Mercury Contamination*, R. Harrig and B. D. Dinman, eds., Ann Arbor Science Publ., Ann Arbor, MI.
- Jury, W. A. (1986). Volatilization from soils, in *Vadose Zone Modeling of Organic Pollutants*, S. C. Hern and S. M. Melacon, eds., Lewis Publ., Chelsea, MI, pp. 159-177.
- Jury, W. A., W. F. Spencer, and W. J. Farmer (1983). Use of models for assessing relative volatility, mobility, and persistence of pesticides and other trace organics in soil systems, in *Hazard Assessment of Chemicals*, J. Saxena, ed., Vol. 2, Academic Press, New York.
- Justice, J. K., and R. L. Smith (1962). Nitrification of ammonium sulfate in a calcareous soil as influenced by combination of moisture, temperature, and levels of added nitrogen, *Soil Sci. Soc. Am. Proc.* **26**:246-250.
- Karickhoff, S. W. (1981). Semiempirical estimation of sorption of hydrophobic pollutants on natural sediments and soils, *Chemosphere* **10**:833-846.
- Karickhoff, S. W., D. S. Brown, and T. A. Scott (1979). Sorption of hydrophobic pollutants on natural sediments, *Water Res.* **13**:241-248.
- Kearney, P. C., E. A. Woolson, J. R. Plimmer, and A. R. Isensee (1969). Decontamination of pesticides in soils, *Residue Rev.* **44**:137-149.
- Kearney, P. C., and D. D. Kaufman, eds. (1964). *Degradation of Herbicides*, Marcel Dekker, New York.
- Kenaga, E. E., and C. A. I. Göring (1980). Relationship between water solubility, soil sorption, octanol-water partitioning, and biocentration of chemicals in biota, in Proceedings, Third ASTM Symposium on Aquatic Toxicology, *ASTM Special Tech. Pub.* **707**:78-115.
- Knisel, W. J., ed. (1980). *CREAMS: A Field Scale Model for Chemicals, Runoff and Erosion from Agricultural Management Systems*, Conservation Research Report No. 26., U.S. Department of Agriculture, Washington DC.
- Krenkel, P. A. (1973). Mercury in the environment, *CRC Crit. Rev. Environ. Control* **3**(3):303-373.
- Lance, J. C. (1985). Fate of bacteria and viruses in sewage applied to soils, *Trans. Am. Soc. Agr. Eng.* **21**:114-117.
- Langston, W. J. (1985). Assessment of the distribution and availability of arsenic and mercury in estuaries, in *Estuarine Management and Quality Assessment*, J. G. Wilson and W. Halcrow, eds., Plenum Press, New York, pp. 131-146.
- Lindsay, W. L. (1979). *Chemical Equilibria in Soil*, Wiley, New York.
- Loehr, R. C., W. J. Jewell, H. D. Novak, W. W. Clarkson, and G. S. Friedman (1979). *Land Application of Wastes*, Van Nostrand Reinhold, New York.
- Lyman, W. J. (1982). Octanol/water partitioning coefficient, in *Handbook of Chemical Property Estimation Methods*, W. J. Lyman et al., eds., McGraw-Hill, New York, pp. 2-1-2-52.

- McCallister, D. L., and T. J. Logan (1978). Phosphate adsorption-desorption characteristics of soils and bottom sediments in the Maumee River Basin of Ohio, *J. Environ. Qual.* **7**:87-92.
- McElroy, A. D., A. D. Chiu, S. Y. Nebgen, J. E. Aleti, and F. W. Bennett (1976). *Loading Functions for Assessment of Water Pollution from Non-point Sources*, EPA-600/2-76-151, U.S. Environmental Protection Agency, Washington, DC.
- McLaren, A. D. (1969). Steady state studies of nitrification in soil: Theoretical considerations, *Soil Sci. Soc. Am. Proc.* **33**:273-276.
- McLaren, A. D. (1971). Kinetics of nitrification: Growth of the nitrifiers, *Soil Sci. Soc. Am. Proc.* **35**:91-95.
- Mahmood, R. J., and R. C. Sims (1986). Mobility of organics in land treatment systems, *J. Environ. Eng. ASCE* **112**(2):236-245.
- Massey, H. F., and M. L. Jackson (1952). Selective erosion of soil fertility constituents, *Soil Sci. Soc. Proc.* **82**:353-356.
- Massey, H. F., M. L. Jackson, and O. E. Hays (1953). Fertility erosion of two Wisconsin soils, *Agron. J.* **45**:543-547.
- Miles, J. R. W., C. M. Tu, and C. R. Harris (1969). Metabolism of heptachlor and its degradation products by soil microorganisms, *J. Econ. Entomol.* **69**:1334-1338.
- Mill, T. (1980). Data needed to predict the environmental fate of organic chemicals, in *Dynamic Exposure and Hazard Assessment of Toxic Chemicals*, R. Hague, ed., Ann Arbor Science Publ., Ann Arbor, MI.
- Mills, W. B., et al. (1985). *Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants*, EPA-600/6-82-004a and b, Vol. I and II, U.S. Environmental Protection Agency, Athens, GA.
- Misra, C., D. R. Nielsen, and J. W. Biggar (1974). Nitrogen transformation in soil during leaching: III—Nitrate reduction in soil, *Soil. Sci. Soc. Am. Proc.* **38**:300-304.
- Moore, J. B. (1963). Bottom sediment studies, Buzzards Bay, Massachusetts, *J. Sediment. Pet.* **33**:511-538.
- Moore, J. W., and S. Ramamoorthy (1984). *Organic Chemicals in Natural Waters*, Springer-Verlag, New York.
- National Academy of Sciences (1979). *Polychlorinated Biphenyls*, National Academy of Sciences, Washington, DC.
- Nissenbaum, A., and D. J. Swaine (1976). Organic matter-metal interactions in recent sediments: The role of humic substances, *Geochim. Cosmochim. Acta* **40**:809-816.
- Novotny, V., and G. Chesters (1981). *Handbook of Nonpoint Pollution: Sources and Management*, Van Nostrand Reinhold, New York.
- Novotny, V., et al. (1978). Mathematical modeling of land runoff contaminated by phosphorus, *J. WPCF* **90**:101-112.
- Painter, H. A. (1970). A review of literature on inorganic nitrogen metabolism in microorganisms, *Water Res.* **4**:393-450.
- Paris, D. F., and D. L. Lewis (1973). Chemical and microbial degradation of selected pesticide in aquatic systems, *Residue Rev.* **45**:95-124.

- Paris D. F., et al. (1981). Second-order model to predict microbial degradation of organic compounds in natural waters, *Appl. Environ. Microbiol.* **41**(3): 603-609.
- Parson, D., et al. (1975). *Health Aspects of Sewage Effluent Irrigation*, Pollution Control Branch, British Columbia Water Research Service, Dept. of Lands, Forests, and Water Research, Parliament Bldg., Victoria, B. C.
- Pettygrove, G. S., and T. Assano, eds. (1985). *Irrigation with Reclaimed Municipal Wastewater: A Guidance Manual*, Lewis Publ., Chelsea, MI.
- Porcella, D. B., et al. (1974). *Comprehensive Management of Phosphorus Water Pollution*, EPA/600/5-74/010, U.S. Environmental Protection Agency, Washington, DC.
- Powell, G. M. (1976). *Land Treatment of Municipal Wastewater Effluents. Design Factors II*, U.S. Environmental Protection Agency, Washington DC.
- Pratt, P. F., W. W. Jones, and V. E. Hunsaker (1972). Nitrate in deep soil profiles in relation to fertilizer rates and leaching volume, *J. Environ. Qual.* **1**:97-102.
- Preul, H. C., and G. J. Schroepfer (1968). Travel of nitrogen in soils, *J. WPCF* **40**:30-48.
- Rao, P. S. C., and J. M. Davidson (1982). *Retention and Transformation of Selected Pesticides and Phosphorus in Soil-Water Systems: A Critical Review*, EPA 600/3-82-060, U.S. Environmental Protection Agency, Athens, GA.
- Reed, S., R. Thomas, and N. Kowal (1980). Long-term land treatment; Are there health or environmental risks? in *Proceedings, ASCE National Convention*, Portland, OR.
- Rennie, D. E., and R. B. McKercher (1959). Adsorption of phosphorus by four Saskatchewan soils, *Can. J. Soil Sci.* **39**:64-75.
- Ryan, J. A., and D. R. Keeney (1975). Ammonia volatilization of surface-applied wastewater sludge, *J. WPCF* **50**:386-393.
- Ryden, J. C., J. K. Syers, and R. F. Harris (1972). Potential of an eroding urban soil for the phosphorus enrichment of streams, *J. Environ. Qual.* **1**:430-434.
- Salomons, W., and U. Förstner (1984). *Metals in the Hydrocycle*, Springer Verlag, Berlin, New York.
- Schaub, S. A., and C. A. Sorber (1977). Virus and bacteria removal from wastewater by rapid infiltration through soil, *Appl. Environ. Microbiol.* **33**:609.
- Scheuerman, P. R., G. Bitton, A. R. Overman, and G. E. Gifford (1979). Transport of viruses through organic soils and sediments, *J. Environ. Eng. Div. ASCE* **40**:R257-271.
- Schnoor, J. L., C. Sato, D. McKechnie, and D. Sahoo (1987). *Processes, Coefficients, and Models for Simulating Toxic Organics and Heavy Metals in Surface Waters*, EPA 600/3-67/015, U.S. Environmental Protection Agency, Athens, GA.
- Schwarzenbach, R. P., and J. Westall (1981). Transport of nonpolar organic compounds from surface water to groundwater, laboratory sorption studies, *Environ. Sci. Technol.* **5**:1360-1367.
- Sethunathan, N. (1973). Chemical and microbial degradation of ten selected pesticides in aquatic systems, *Residue Rev.* **45**:95-124.

- Shacklette, H. T., J. C. Hamilton, J. G. Boergnagen, and J. M. Bowles (1971). *Elemental Composition of Surficial Materials in the Conterminous United States*, U.S. Geological Survey Prof. Paper 574-D, Washington, DC.
- Shaffer, M. J., G. R. Dutt, and W. J. Moore (1969). Predicting changes in nitrogen compounds in soil-water systems. Collected papers: Nitrates in agricultural waste waters, FWQA (now EPA) WPC Series, 13030 ELY 12/69, pp. 15-28.
- Sharpley, A. N., L. R. Ahuja, and R. G. Menzel (1981). The release of phosphorus to runoff in relation to the kinetics of desorption, *J. Environ. Qual.* **10**(4):386-391.
- Spencer, W. E., and M. M. Cliath (1973). Pesticide volatilization as related to water loss from soil, *J. Environ. Qual.* **2**:284-289.
- Spencer, W. F., W. J. Farmer, and M. M. Cliath (1973). Pesticide volatilization as related to water loss from soil, *J. Environ. Qual.* **2**:284-289.
- Spencer, W. F., W. J. Farmer, and W. A. Jury (1982). Review: Behavior of organic chemicals at soil, air, water interfaces as related to predicting the transport and volatilization of organic pollutants, *Environ. Toxicol. Chem.* **1**:17-26.
- Stanford, G., M. H. Free, and D. H. Swaininger (1973). Temperature coefficient of soil nitrogen mineralization, *Soil Sci.* **115**(4):321-328.
- Steward, B. A., et al. (1975). *Control of Pollution from Cropland*, EPA 600/2-75/026, U.S. Environmental Protection Agency—U.S. Department of Agriculture, Washington, DC.
- Stigliani, W. M., ed. (1991). *Chemical Time Bombs: Definition, Concepts, and Examples*, Ex. Report No 16, International Institute for Applied System Analysis, Laxenburg, Austria.
- Stigliani, W. M., et al. (1991). Chemical time bombs—predicting the unpredictable, *Environment* **33**(4):4-9.
- Stoltenberg, N. I., and J. L. White (1953). Selective loss of plant nutrients by erosion, *Soil Sci. Soc. Am. Proc.* **27**:406-410.
- Summerhayes, C. P., J. P. Ellis, and P. Stoffers (1985). *Estuaries as Sinks for Sediment and Industrial Waste—A Case History from the Massachusetts Coast*, E. Schweizerbart'sche Verlagbuchhandlung, Stuttgart.
- Terry, R. E., D. W. Nelson, L. E. Simons, and G. J. Meyer (1978). Ammonia volatilization from wastewater sludge applied to soils, *J. WPCF* **50**:2657-2665.
- U.S. Environmental Protection Agency (1976). *Application of Sewage Sludge to Cropland. Appraisal of Potential Hazard of Heavy Metals to Plants and Animals*, EPA 430/9-76-103, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1977). *Process Design Manual for Land Treatment of Municipal Wastewater*, EPA 625/1-77/008, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1984). *Review of In-place Treatment Techniques for Contaminated Surface Soils, Background Information for In Situ Treatment*, Vol. 2, EPA 540/2-84-003b, U.S. EPA, Cincinnati, OH.
- U.S. Environmental Protection Agency (1986). *Permit Guidance Manual on*

- Hazardous Waste Land Treatment Demonstrations*, EPA 530/SW86-032, Office of Solid Waste, U.S. EPA Washington, DC.
- Valentine, R. L., and J. L. Schnoor (1986). Biotransformation, in *Vadose Zone Modeling of Organic Pollutants*, S. C. Hern and S. M. Melacon, eds., Lewis Publ., Chelsea, MI, pp. 191-222.
- Vijayachandran, P. K., and R. D. Harter (1975). Evaluation of phosphate adsorption by a cross section of soil types. *Soil Sci.* **119**:119-125.
- Weber, W. J., P. M. McGinley, and L. E. Katz (1991). Review paper—Sorption phenomena in subsurface systems: Concepts, models and effects on contaminant fate and transport, *Water Res.* **25**(5):499-528.
- Wilde, H. E., C. N. Sawyer, and T. C. McMahon (1971). Factors affecting nitrification kinetics, *J. WPCF* **43**:1845-1854.
- Young, R. A., and C. A. Onstad (1976). *Predicting Particle Size Composition of Eroded Soil*, Paper No. 76-2052, presented at 1976 American Society of Agricultural Engineers Meeting, Lincoln, NE.
- Yule, W. N., and I. Rosefield (1964). Fate of insecticide residues. Decomposition of lindane in soils, *J. Agr. Food. Chem.* **15**:1000-1004.
- Zanoni, A. E. (1969) Secondary effluent degradation at different temperatures, *J. WPCF* **41**:640-659.
- Zimdahl, R. L., and J. J. Hasset (1977). Lead in soil, in *Lead in the Environment*, W. R. Rodgers and B. G. Wixson, eds., NSF 770214, U.S. Government Printing Office, Washington, DC.

7

Ground-Water and Base-Flow Contamination

You never know the worth of water till the well runs dry.

Benjamin Franklin

GROUND WATER (BASE FLOW) AND DIFFUSE POLLUTION

Extent of the Problem

Approximately one-half of the United States population depends on ground water for its supply of potable water; that is, approximately 36% of all municipal public drinking water supply systems and 95% of the rural population draw potable water from ground-water resources (Conservation Foundation, 1985). As stated in Chapter 3, water recovered from wells is considered hydrologically part of ground-water runoff when it reaches surface drainage systems.

Ground-water and surface-water systems are interconnected by *recharge* of surface-water into ground-water zones and by *discharge* of ground-water into surface systems. In many cases surface-water bodies are an extension of the ground-water systems. Perennial streams may be both recharging ground-water zones or ground water is discharging as base flow into the streams. Ephemeral streams are mostly recharging during the time of flow (see Chapter 3 for a discussion on hydrologic surface-water-ground-water interactions). Hence with ground-water discharge (base flow), diffuse pollution from the land may reach surface-water bodies via ground-water routes.

7

Ground-Water and Base-Flow Contamination

You never know the worth of water till the well runs dry.

Benjamin Franklin

GROUND WATER (BASE FLOW) AND DIFFUSE POLLUTION

Extent of the Problem

Approximately one-half of the United States population depends on ground water for its supply of potable water; that is, approximately 36% of all municipal public drinking water supply systems and 95% of the rural population draw potable water from ground-water resources (Conservation Foundation, 1985). As stated in Chapter 3, water recovered from wells is considered hydrologically part of ground-water runoff when it reaches surface drainage systems.

Ground-water and surface-water systems are interconnected by *recharge* of surface-water into ground-water zones and by *discharge* of ground-water into surface systems. In many cases surface-water bodies are an extension of the ground-water systems. Perennial streams may be both recharging ground-water zones or ground water is discharging as base flow into the streams. Ephemeral streams are mostly recharging during the time of flow (see Chapter 3 for a discussion on hydrologic surface-water-ground-water interactions). Hence with ground-water discharge (base flow), diffuse pollution from the land may reach surface-water bodies via ground-water routes.

The quantity of ground water underlying the conterminous United States is very large, and exceeds all freshwater surface storage. The amount of water within the top 800 meters of earth surface layer is equal to about 35 years of all surface-water runoff (Geraghty et al., 1973; Knox, and Fairchild, 1987). However, as far as water use is concerned, more surface freshwater is used than ground water.

Ideally, ground water should be characterized by clarity, bacterial purity, and constant temperature and chemical quality, and should require very little treatment prior to its use. However, increased usage of ground-water resources and a general increase of inputs of surface pollution into ground-water zones have caused contamination and a general deterioration of ground-water quality in many parts to the world. Specifically, the severe contamination by nitrates in Central Europe and in some parts of the United States, and pesticide contamination are of concern, as both pollutants may render ground water unsuitable for human consumption without expensive treatment. Excessive mining of ground-water resources results in more costly pumping from deeper geological zones, which yields water with high salt and ionic content or sea water intrusion into aquifers in coastal areas.

While the basic chemical composition of ground-water reflects its contact with soils, minerals, and rocks, surface diffuse pollution is often the primary source of ground water-contamination. With an increased emphasis on land disposal of sewage and industrial wastes promulgated by the first amendments of the Clean Water Act, the surface-water pollution problem could have been shifted to ground water. Deep-well injection of wastewater, subsurface disposal of toxic wastes, unsanitary landfills, excessive use of chemicals and fertilizers, and suburban and rural septic tank waste disposal systems are the main causes of ground-water contamination. However, Table 7.1 shows that other sources of ground-water contamination can be also considered.

It should be realized that only a relatively small portion of the ground-water problem has been discovered. Ground-water pollution is not visible and is detected only when a water supply or spring is noticeably polluted or the pollutant is discharged into surface waters. The monitoring of ground-water quality is still inadequate, as it is mostly done using observation wells that are located great distances apart.

All ground water results from surface precipitation. However, in northern North America and Europe, most of the present ground water originated from the melting of glaciers at the end of the glacial period approximately 10,000 years ago, but some ground water is even older. Also, ground-water movement is very slow. Plus the practices and land-use activities that have an impact on ground-water quality have been

TABLE 7.1 Principal Sources of Ground-Water Contamination and Their Relative Importance by Region

	Northeast	Northwest	South Central	Southwest
Septic tanks and cesspools	I ^a	I	I	I
Petroleum exploration and development	II	II	I	I
Landfills	I	II	II	II
Irrigation return flows	IV	II	II	II
Surface discharges	II	I	III	I
Surface impoundments	I	I	II	III
Spills	I	II	II	III
Mining activities	II	I	III	II
Agricultural activities				
Fertilizers	III	II	III	II
Pesticides	II	III	II	II
Feedlot and barnyard wastes	III	III	II	III
Highway deicing	I	III	IV	IV
Artificial recharge	III	IV	III	II
River infiltration	II	II	IV	IV
Land disposal of wastewater and sewage	III	IV	III	II
Underground (leaking) storage of chemicals	II	II	II	II

Source: After Miller and Scalf (1974).

^aI—high; II—moderate; III—low; IV—not significant.

occurring for the past 40 to 50 years. Apparently, this time span is very short in the geological time scale, so most ground-water contamination is still confined near the source.

In industrialized countries, ground-water pollution by toxic chemicals and nitrates has locally and regionally reached alarming levels. For years, industries and municipalities have been disposing of their toxic and other wastes and refuse on land or burying them without regard to the possible contamination of ground water (Fig. 7.1). Examples of widely publicized severe ground-water contamination include the Love Canal site in Niagara Falls, New York, as well as ground-water quality impairment in other parts of the northeastern United States, specifically in the state of New Jersey.

In the Niagara Falls Love Canal incident, a chemical company between 1947 and 1952 used an abandoned river channel to store toxic wastes, mostly solvents. At the time, due to the lack of knowledge on the environmental consequences of such disposal, such containments were common and considered as acceptable. After 1952 the canal was filled

The quantity of ground water underlying the conterminous United States is very large, and exceeds all freshwater surface storage. The amount of water within the top 800 meters of earth surface layer is equal to about 35 years of all surface-water runoff (Geraghty et al., 1973; Knox, and Fairchild, 1987). However, as far as water use is concerned, more surface freshwater is used than ground water.

Ideally, ground water should be characterized by clarity, bacterial purity, and constant temperature and chemical quality, and should require very little treatment prior to its use. However, increased usage of ground-water resources and a general increase of inputs of surface pollution into ground-water zones have caused contamination and a general deterioration of ground-water quality in many parts to the world. Specifically, the severe contamination by nitrates in Central Europe and in some parts of the United States, and pesticide contamination are of concern, as both pollutants may render ground water unsuitable for human consumption without expensive treatment. Excessive mining of ground-water resources results in more costly pumping from deeper geological zones, which yields water with high salt and ionic content or sea water intrusion into aquifers in coastal areas.

While the basic chemical composition of ground-water reflects its contact with soils, minerals, and rocks, surface diffuse pollution is often the primary source of ground water-contamination. With an increased emphasis on land disposal of sewage and industrial wastes promulgated by the first amendments of the Clean Water Act, the surface-water pollution problem could have been shifted to ground water. Deep-well injection of wastewater, subsurface disposal of toxic wastes, unsanitary landfills, excessive use of chemicals and fertilizers, and suburban and rural septic tank waste disposal systems are the main causes of ground-water contamination. However, Table 7.1 shows that other sources of ground-water contamination can be also considered.

It should be realized that only a relatively small portion of the ground-water problem has been discovered. Ground-water pollution is not visible and is detected only when a water supply or spring is noticeably polluted or the pollutant is discharged into surface waters. The monitoring of ground-water quality is still inadequate, as it is mostly done using observation wells that are located great distances apart.

All ground water results from surface precipitation. However, in northern North America and Europe, most of the present ground water originated from the melting of glaciers at the end of the glacial period approximately 10,000 years ago, but some ground water is even older. Also, ground-water movement is very slow. Plus the practices and land-use activities that have an impact on ground-water quality have been

TABLE 7.1 Principal Sources of Ground-Water Contamination and Their Relative Importance by Region

	Northeast	Northwest	South Central	Southwest
Septic tanks and cesspools	I ^a	I	I	I
Petroleum exploration and development	II	II	I	I
Landfills	I	II	II	II
Irrigation return flows	IV	II	II	II
Surface discharges	II	I	III	I
Surface impoundments	I	I	II	III
Spills	I	II	II	III
Mining activities	II	I	III	II
Agricultural activities				
Fertilizers	III	II	III	II
Pesticides	II	III	II	II
Feedlot and barnyard wastes	III	III	II	III
Highway deicing	I	III	IV	IV
Artificial recharge	III	IV	III	II
River infiltration	II	II	IV	IV
Land disposal of wastewater and sewage	III	IV	III	II
Underground (leaking) storage of chemicals	II	II	II	II

Source: After Miller and Scalf (1974).

^aI—high; II—moderate; III—low; IV—not significant.

occurring for the past 40 to 50 years. Apparently, this time span is very short in the geological time scale, so most ground-water contamination is still confined near the source.

In industrialized countries, ground-water pollution by toxic chemicals and nitrates has locally and regionally reached alarming levels. For years, industries and municipalities have been disposing of their toxic and other wastes and refuse on land or burying them without regard to the possible contamination of ground water (Fig. 7.1). Examples of widely publicized severe ground-water contamination include the Love Canal site in Niagara Falls, New York, as well as ground-water quality impairment in other parts of the northeastern United States, specifically in the state of New Jersey.

In the Niagara Falls Love Canal incident, a chemical company between 1947 and 1952 used an abandoned river channel to store toxic wastes, mostly solvents. At the time, due to the lack of knowledge on the environmental consequences of such disposal, such containments were common and considered as acceptable. After 1952 the canal was filled



FIGURE 7.1. Illegal and unsanitary dumps and solid waste disposal sites are the primary sources of ground-water contamination.

with soil and, subsequently, the area was developed into a residential subdivision. (It should be noted, however, that the company gave explicit caveats to the authorities when giving them the land for development.) In the 1970s the chemicals began seeping through the basement walls into household drainage systems. Air pollution monitoring in the basements throughout the contaminated area registered pollution that exceeded 10 to 5000 times the ambient standards. The toxic chemicals that created this extreme diffuse pollution problem included mostly aliphatic hydrocarbons (benzene, toluene, chloroform, etc.). The Love Canal site has been cleaned up at great cost, and in the 1990s it appeared safe for reinhabitation.

Cleanup of Contaminated Ground Water

In many cases, a leachate from a localized point source (a storage lagoon, landfill, leaking petroleum and gasoline storage tanks, scrap yard) becomes a regional diffuse pollution problem. Many such unsanitary and contaminating sites have been classified by the U.S. EPA as hazardous sites, often requiring costly remediation.

The tasks facing those concerned and responsible for ground-water quality are enormous, and the solution expensive or nonexistent. There are several approaches that have been used to clean up contaminated ground water (Canter, Knox, and Fairchild, 1987):

1. *Physical control measures.* These controls include surface-water control, the containment of the sources of contamination by capping, and the use of liners and other barriers to isolate contaminated sites and/or to prevent contaminated surface runoff from entering the ground-water zones. The ground-water level can also be lowered by pumping to a safe depth below surface contamination.
2. *In-situ technologies.* Both chemical and biological treatment may be employed. Chemical treatment may be considered only if contaminants are known and the extent of the contaminated zone is defined. Some of the chemical treatment methods involve oxygen injection that will precipitate iron and manganese. Biological in-situ treatment is based on the same principle as that in biological wastewater treatment systems. The method requires the addition of nutrients and oxygen, and allowing microorganisms to degrade the chemicals.
3. *Ground-water treatment.* The most popular ground-water cleanup measure remains removal and treatment and subsequent return of the treated water into underground zones. Water is removed by wells or by injection/extraction systems. Although a wide variety of technologies exists, the treatment options used for ground-water decontamination are usually limited to air stripping, carbon adsorption, or biological treatment for organic removal, and chemical precipitation for inorganic removal.

High nitrate content caused by fertilizer applications and/or septic tank effluents must be remediated at any cost because it is usually widespread. When such waters are to be used for potable water supply in central Europe, biological denitrification of the extracted water has been seriously considered by the authorities. The problem with this methodology is finding a suitable nontoxic source of organic carbon instead of methanol (toxic to humans), which is commonly used in many denitrifying wastewater treatment plants. Because of the slow movement of water in underground zones and the large volumes of water in ground-water systems, the recovery of a contaminated aquifer is slow, usually lasting many years.

Ground-Water Quality Standards

Ground-water quality should comply with standards set forth by environmental control legislation. In the United States the three acts that have the greatest impact on ground-water pollution control are

1. The Safe Drinking Water Act of 1974 (PL-93-523)

2. The Federal Environmental Pesticide Control Act of 1972 (PL-92-516)
3. The Toxic Substance Control Act of 1976 (PL-94-469)

In addition the Clean Water Act also specifies the quality of water that is to be used for drinking purposes.

It should be pointed out that the standards based on the Safe Drinking Water Act are for water quality at the point of use, not at the point of intake into the system. Water quality standards derived from the Clean Water Act (see the Appendix) represent the mandated raw drinking water quality of the source, be it a surface-water body or a ground-water zone. These standards assume that intake water will be sufficiently uncontaminated so that with the application of the most effective treatment method, a public water supply system would be able to protect public health.

GROUND-WATER MOVEMENT

Aquifers and Aquitards

This section describes water movement, chemical composition and contamination, and the fate of contaminants in saturated ground-water zones. In contrast the preceding chapter dealt with the phenomena occurring in the vadose-aerated zone.

The basic chemical content of uncontaminated ground water can be related to the contact of water with rocks and soils. However, it should be noted that while mineralization (enrichment of water by salts and ions) occurs throughout the entire movement of water in the aquifer, ground-water contamination and pollution mostly occur in the recharge area (with the exception of deep wastewater disposal by wells and buried toxic waste disposal sites).

An *aquifer* was defined in Chapter 3 as a saturated permeable geologic underground stratum that can transmit significant quantities of water. An *aquitard* is a less permeable layer that may be significant for consideration in the regional transport of water, but its permeability is not sufficient enough to permit economical development (Freeze and Cherry, 1979). The most common aquifer composition—about 90%—consists of unconsolidated materials (Todd, 1980). Other materials capable of forming aquifers mainly include sedimentary rocks. Crystalline and metamorphic rocks are relatively impermeable, and can form aquifers only when they are fractured and/or weathered near the surface.

Clay and clayey materials can retain large quantities of water, but they are generally considered impermeable and unsuitable as aquifers. Such

materials and rocks with poorer water-holding and transmitting capacities (shales and dense crystalline rocks) are classified as aquitards. Completely impermeable geological layers (aquicludes) are rare, so hydrogeologists tend to avoid this classification.

Unconsolidated Deposits

The unconsolidated deposits are composed of gravel, sand, silt, or clay particles that may be bound or hardened by mineral content, pressure, or mineral alteration. The three major types of unconsolidated deposits are

1. Glacial deposits formed during the last glacial age, approximately 15,000 to 100,000 years ago.
2. Alluvial (fluvial) deposits that resulted from deposition of sediments by streams and valleys and floodplains.
3. Aeolian deposits that consist of finer soil materials transported by wind.

Glacial deposits are of particular hydrogeologic importance in Europe and the northern part of the North American continent (Fig. 7.2). Water

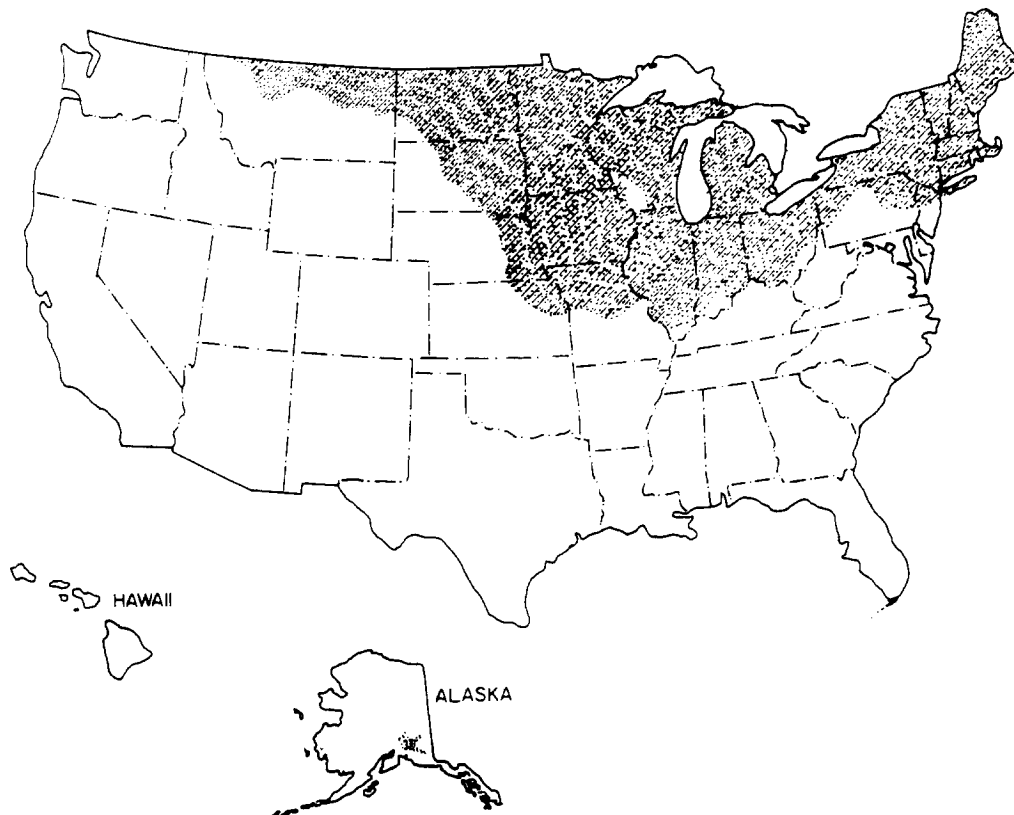


FIGURE 7.2. Extent of glacial deposits in the United States.

in the unconsolidated materials can be divided into four categories: underground water courses, abandoned or buried valleys, plains, and intermontane valleys. Water courses consist of alluvium that forms and underlies stream channels as well as adjacent floodplains. The interconnecting water movement between stream channels depends on the general hydraulic grade. Stream water can be an influent into the ground-water zone (recharging) or vice versa (discharging ground water).

Abandoned or buried valleys are valleys no longer occupied by the streams that formed them. In many areas of the United States, especially the Great Plains and Florida, large regional plain aquifers are located near the surface in unconsolidated deposits. Intermontane aquifers are located in unconsolidated rock deposits, often of considerable thickness, formed by the geological formation of mountains. These are mostly located in the western United States. Water from unconsolidated deposits often has low mineral content; however, due to the higher permeability of these materials and their near-surface location, these aquifers are most commonly affected by surface diffuse pollution.

Sedimentary Rocks

Sedimentary rocks include sandstone limestone or dolomite limestone. Limestone (mostly calcium carbonate) and dolomite (calcium and magnesium dolomite) are also called *carbonate rocks*.

Sandstone. Sandstone is a sedimentary rock with a porosity of 5% to 30%. Its potential as an aquifer is excellent. Sandstone is formed by the binding of sand or gravel by a cementing material such as calcite, dolomite, or clay. These minerals occur as a result of salt precipitation and the filtration of finer clay particles from penetrating water during the geologic times.

Carbonate rocks. Carbonate rocks were formed by the compaction and crystallization of the shells and bones of aquatic animals living in geological oceans and lakes.

Although some of the porosity in carbonate rocks is retained, most of the ground-water movement through them occurs along joints, fractures, and channels formed mostly by the dissolving action of percolating water (recall from Chapter 4 that rainwater is a weak acid that can chemically dissolve calcites and other minerals). The dissolution of limestone formations can reach such proportions that underground streams, caverns, and cavities occur. These limestone formations, called *karst* or *karst limestone*, are especially susceptible to contamination from surface sources. Notable ground-water pollution problems occurred in England (Edworthy, Wilkinson, and Young, 1978) and elsewhere.

Ground-Water Movement

Permeability and Darcy's Law

The flow of water in an aquifer can be expressed by Darcy's law, which relates water velocity in a porous medium to the hydraulic gradient as

$$v = kS = k \frac{\partial h}{\partial l} \quad (7.1)$$

where

v = velocity

S = hydraulic slope of ground water or piezometric table

h = hydraulic head

l = distance in the direction of flow

k = constant of proportionality

In order to compute flow in the aquifer, Equation (7.1) should be multiplied by the cross-sectional area of pores of the media. Hence

$$Q = A_p v = p A k S = K A S \quad (7.2)$$

where

Q = the flow

A_p = the area of voids

A = total cross-sectional area

p = cross-sectional porosity (Table 7.2)

TABLE 7.2 Typical Porosities

	Porosity (%)
Soils	50-60
Clay	45-55
Silt	40-50
Medium to coarse mixed sand	35-40
Uniform sand	30-40
Fine to medium mixed sand	30-35
Gravel	30-40
Gravel and sand	20-35
Sandstone	10-20
Shale	1-10
Limestone	1-10

Source: From Todd (1980).

TABLE 7.3 Typical Hydraulic Conductivities

	Hydraulic Conductivity (cm/sec)
Clay, sand, and gravel mixes (till)	10^{-6} – 10^{-4}
Sand and gravel mixes	10^{-3} – 10^{-1}
Gravel	0.1–1
Coarse sand	0.01–0.1
Medium sand	0.01
Fine sand	10^{-3} – 10^{-2}
Silty sand	10^{-5} – 10^{-3}
Loam soils (surface)	10^{-4} – 10^{-3}
Glacial outwash	10^{-3} –0.1
Deep clay beds	10^{-11} – 10^{-5}
Clay soils (surface)	10^{-2} – 10^{-1}
Volcanic rock	Almost 0–1
Fractured or weathered rock (core samples)	Almost 0–1
Fractured or weathered rock (aquifers)	10^{-6} –0.01
Dense solid rock	$<10^{-8}$
Shale	10^{-10}
Carbonate rocks with secondary porosity	10^{-5} – 10^{-3}
Sandstone	10^{-6} – 10^{-3}

Sources: Based on data from Bouwer (1978) and Fetter (1988).

The coefficient, $K = kp$, is called the *coefficient of permeability* or *hydraulic conductivity*. The coefficient of permeability as defined herein can be visualized as flow velocity in a porous medium under slope that equals unity. Table 7.3 presents coefficients of permeability for various geological materials.

Hydraulic conductivity, K , is a function depending on the porous media and also on the fluid itself. The media characteristics affecting hydraulic conductivity are grain diameter, porosity packing, and the distribution of the material. Fluid characteristics that also affect K are density, viscosity, and its ionic nature. Since K depends on the viscosity of water (the flow is laminar), permeability is affected by temperature.

Example 7.1: Flow Velocity in an Aquifer

An aquifer consisting of sandstone with average slope of the ground-water table of 1% has been contaminated by a highly water-soluble contaminant (it does not adsorb on soil particles). Estimate how fast the contaminated water will move through the aquifer.

Solution This is a typical straightforward application of Darcy's equation. From Equations (7.1) and (7.2)

$$v = \frac{KS}{p} \quad (7.3)$$

and substituting appropriate values for K and p from Tables 7.2 and 7.3, the velocity becomes

$$v = \frac{10^{-4} \times 0.01}{0.15} = 6.67 \times 10^{-6} \text{ cm/sec} = 210 \text{ cm/yr}$$

which is a very slow advancement of the contaminated water front. If the compound has an affinity for adsorption on soil the advancement would be further retarded, as will be shown later.

Typical velocities of ground water may range from less than 1 cm/yr in tight clays to more than 100 m/yr in permeable sand and gravel. Todd (1980) indicated that the normal range for ground-water velocities is 1.5 m/yr to 1.5 m/day. However, highly permeable glacial outwash deposits, fractured basalts and granites, and cavernous limestone formations may allow much higher velocities.

Homogeneity and Anisotropy of Hydraulic Conductivity

Homogeneity

A formation is homogenous if the hydraulic conductivity is uniform at all points within the aquifer. If the hydraulic conductivity varies with location, the formation is heterogenous. Freeze and Cherry (1979) have defined three types of heterogeneity: (1) layered heterogeneity, (2) discontinuous heterogeneity, and (3) trending heterogeneity. Layered heterogeneity is common in sedimentary deposits where different materials have been deposited throughout the geological ages. Discontinuous heterogeneity is caused by the presence of faults or large-size stratigraphic features. Trending heterogeneity results in the progressive change of hydraulic conductivity over a large spatial extent of a geological formation.

Domenico and Schwartz (1990) have pointed out that since the permeabilities in the individual layers of a heterogenous aquifer can vary by several orders of magnitude, the layer with the highest permeability becomes the main route of flow, while the layers of the lowest permeability become essentially aquitards or confining layers. Heterogeneity may also lead to a situation where some layers become saturated with water and some do not. This is called a *perched water table*.

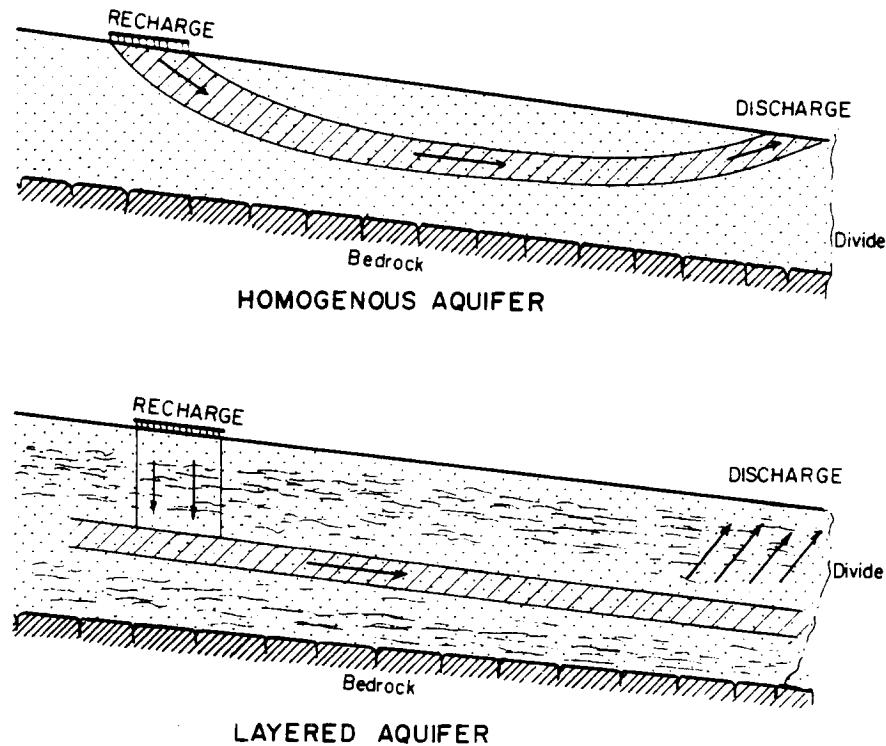


FIGURE 7.3. Flow paths in homogenous and layered aquifers (according to Freeze and Cherry, 1979). (Copyright © 1979 Prentice Hall, reprinted by permission.)

Anisotropy

A geologic formation is isotropic at a given point if the hydraulic conductivity is the same in all directions. The formation is anisotropic if the hydraulic conductivity varies with direction. Heterogeneity and anisotropy can greatly influence flow patterns of contaminants in aquifers (Fig. 7.3).

Dispersion

Average water movement alone does not fully explain the transport and spread of contaminants in ground-water aquifers. The first processes that contribute to forming a concentration field of a contaminant in ground-water zones are advection, dispersion, and diffusion. The second category of processes affecting contaminant movement are adsorption-desorption and chemical precipitation.

Advection is a process in which soluble contaminants are transported by the bulk of the water. The rate of transport can be directly related to the average linear ground-water velocity defined by Equations (7.1) and (7.2). However, the velocity field in ground-water zones is not uniform due to the heterogeneity and anisotropy of the aquifer, which are caused

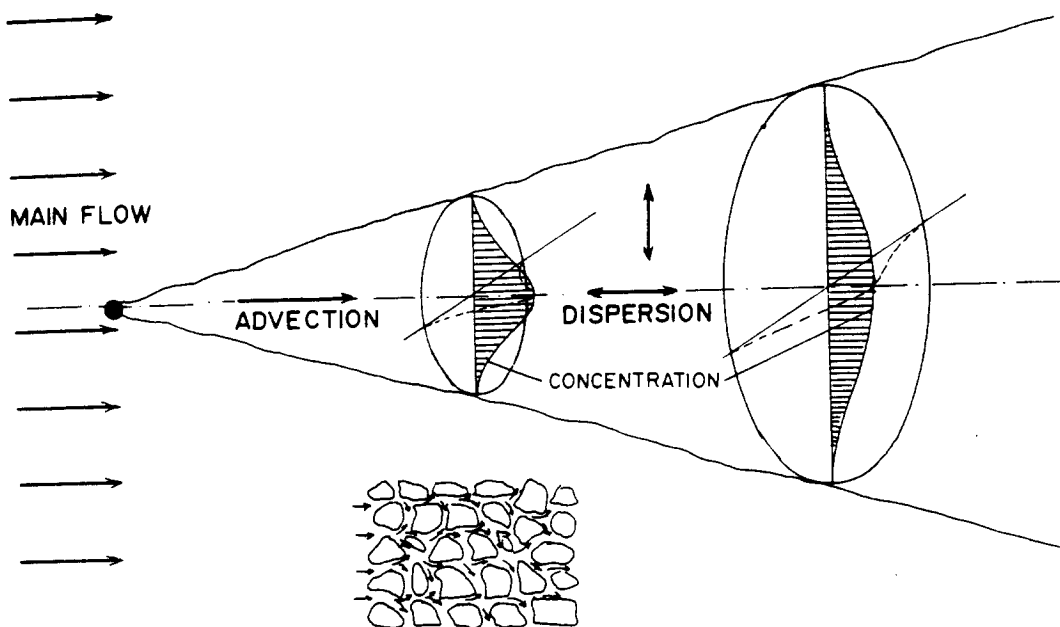


FIGURE 7.4. Contaminant spread in ground-water zones from a continuous source by advection and dispersion.

by differing pore sizes and the branching of pores, plus many obstacles in the path of ground-water movement. These factors cause a spread of mobile contaminants in all directions (longitudinal and transverse), as shown on Figure 7.4. The process is called *hydrodynamical dispersion*. *Molecular diffusion* is a process by which mobile contaminants move as a result of the kinetic activity of molecules and ions in the direction of their concentration gradient. Molecular diffusion can occur independently of ground-water movement. However, normal ground-water velocities and hydrodynamical dispersion are such that molecular diffusion can be neglected (Bouwer, 1978). Hydrodynamical and molecular dispersion cannot be separated and can be included in one dispersion parameter.

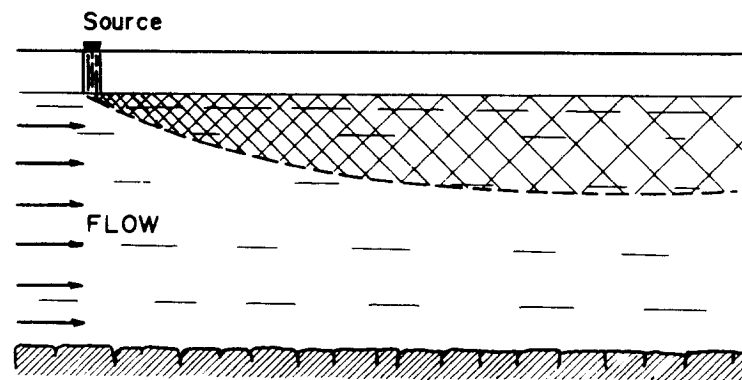
The spread of contaminants by dispersion in ground-water zones has a similar appearance to air pollution plumes or discharges of pollutants in surface-water bodies. It is, basically, a mixing process. Nevertheless, it must be remembered that the dispersion of pollutants in air and water environments is caused primarily by turbulent mixing of the air and water masses, while in ground-water zones, mixing occurs in laminar (viscous) flow patterns as a result of the continuous splitting, slowing down, and deflecting of water particles in the pores.

The spread of contaminants in the direction of the ground-water flow is

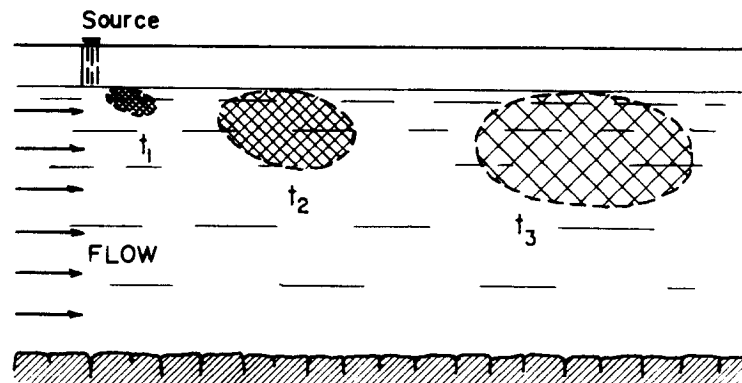
called *longitudinal dispersion*. *Transverse dispersion* is perpendicular to the main direction of flow, and is usually weaker than the longitudinal dispersion. In addition to the movement of water particles, the spread of semimobile contaminants can be caused by adsorption–desorption processes whereby a portion of the contaminant is adsorbed on soil particles at higher concentrations and released back to the solution when the concentration decreases.

Equation for Dispersive Movement

Most of the ground-water contamination problem can be analyzed assuming steady-state flow conditions, which implies that the flow velocity and dispersion characteristics remain constant with time. Furthermore, contamination problems can be limited to the instantaneous injection of



(a) CONTINUOUS SOURCE



(b) INSTANTANEOUS SOURCE

FIGURE 7.5. Spreading of a pollutant in a two-dimensional uniform field in an isotropic sand. (a) Continuous source—leaching. (b) Instantaneous source—spill.

a pollutant such as a spill or a continuous release, that is, leaching (Fig. 7.5). General equations of hydrodispersive movement have been published in many references (Anderson, 1979; Freeze and Cherry, 1979; NCASI, 1985; Canter, Knox, and Fairchild, 1987; Fetter, 1988).

The basic differential equation of the hydrodispersive movement is

$$\frac{\partial C}{\partial t} = \frac{\partial}{\partial X_i} \left(D_i \frac{\partial C}{\partial X_i} - V_i C \right) \quad (7.4)$$

where

X_i = the coordinate

t = time

$C(X_i, t)$ = the concentration of the contaminant

V_i = the velocity in the i th direction

D_i = is the coefficient of dispersion in the i th direction

The solution of this equation for a continuous point source in a one-dimensional flow field (all flow is in the direction of the X -axis) will yield (Ogata, 1970; Fetter, 1988)

$$C = \frac{C_0}{2} \left[\operatorname{erfc} \left(\frac{L - V_x t}{2\sqrt{D_L t}} \right) + \exp \left(\frac{V_x L}{D_L} \right) \operatorname{erfc} \left(\frac{L + V_x t}{2\sqrt{D_L t}} \right) \right] \quad (7.5)$$

where

C = concentration at a distance L from the source at time t

C_0 = concentration at the source

D_L = longitudinal dispersion coefficient

D_T = transverse dispersion coefficient

Due to the slow movement of ground water (a few meters per year or less), most of the ground-water contamination problems could be considered as an instantaneous release. There are two solutions available to the problem. One observes the concentration field from a fixed point, for example, the point of discharge. The other solution views the movement of the contaminant cloud as if an observer were located in the center of gravity of the cloud. The former concept yields the following equation for a two-dimensional (longitudinal and transverse) movement of an inert dissolved contaminant (Peaudecerf and Sauty, 1978)

$$C(X, Y, t) = \frac{m}{4\pi p (D_L D_T)^{0.5} t} \exp \left(-\frac{(X - Vt)^2}{4D_L t} - \frac{y^2}{4D_T t} \right) \quad (7.6)$$

where

m = the mass of the released contaminant per unit of aquifer thickness

p = porosity

The point of the coordinate origin for Equation (7.6) ($X = Y = Z = 0$) is the point of release of the contaminant in a two-dimensional aquifer.

Baestlé (1969) published a solution for the latter case, where the coordinate origin is located at the center of gravity of the contaminated cloud. The cloud is carried away from the source with an average velocity equal to the ground-water movement (Eq. (7.1)). The concentration distribution of the contaminant mass is then

$$C(X, Y, Z, t) = \frac{M}{8(\pi t)^{1/2} p \sqrt{D_x D_y D_z}} \exp\left(-\frac{X^2}{4D_x t} - \frac{Y^2}{4D_y t} - \frac{Z^2}{4D_z t}\right) \quad (7.7)$$

where M = the total mass of the contaminant. In the equation the transformed coordinates are

$$X = x - Vt$$

$$Y = y$$

$$Z = z$$

The maximum concentration occurs at the center of the cloud where $X = Y = Z = 0$. Hence

$$C_{\max} = \frac{M}{8(\pi t)^{1/2} p \sqrt{D_x D_y D_z}} \quad (7.8)$$

Assuming a true dimensionless point of discharge, Equations (7.6) to (7.8) can be used very rarely. For most cases, such as a surface application of a pesticide, the mass term, M , should be replaced by

$$\frac{M}{p} = C_0 V_0 \quad (7.9)$$

where

V_0 = the volume of ground water below the area of application

C_0 = initial concentration

Then Equations (7.6) to (7.8) are applicable where

$$8(\pi t)^{1/2} \sqrt{D_x D_y D_z} \gg V_0$$

Character and Magnitude of the Dispersion Coefficient

The coefficient of hydrodynamic dispersion is the sum of bulk molecular diffusion and mechanical dispersion. In ground-water zones molecular diffusion is not as fast as that in open water, since molecules have to follow longer pathways because the grains block many possible pathways and the pathways are in the pores. Fetter (1988) pointed out that the actual magnitude of the ground-water diffusion coefficient, D , may be only a few percent of that for water. Molecular diffusion becomes the dominant transport process when ground water is not moving. Perkins and Johnson (1963) have shown that molecular diffusive transport is important when the Peclet number $N_{Pe} = Vd_m/D_m < 1$, where d_m is the grain size of the material and D_m is the molecular diffusion coefficient adjusted for ground-water zones.

There are three basic causes of mechanical dispersion: (1) the fluid in the pores moves faster in the center, while the movement near the wall is slowed down by adhesion of water to solid surfaces, (2) water movement takes different pathways, sometimes compared to a so-called "random walk," (3) fluid traveling through larger pores will move faster than that in smaller pores. Lateral dispersion is caused by the fact that the flow paths can split and branch to the side (Fig. 7.4).

Researchers have found that the longitudinal and transverse dispersivity can be related to the average advective velocity, V , as (Fetter, 1988; Domenico and Schwartz, 1990)

$$D_L = \alpha_1 V \quad \text{and} \quad D_t = \alpha_2 V$$

The total magnitude of the dispersion is then

$$D_i = \alpha_i V + D_m \quad (7.10)$$

The proportionality coefficients, α , are called *coefficients of dynamic dispersivity*. These empirical coefficients are a characteristic property of the porous medium, while the molecular diffusion coefficient, D_m , is a property of the contaminant compound. Anderson (1979) compiled the magnitudes of the dynamic dispersivity coefficients. The ranges are given in Table 7.4.

The magnitude of the dispersion coefficient should be measured rather

TABLE 7.4 Ranges of Dynamic Dispersivity in Aquifers

Type of Aquifer	Longitudinal Dispersivity: α_L (m)	Ratio of Transverse Dispersivity to Longitudinal: α_T/α_L
Alluvial sediments	12-61	0.3
Glacial deposits	21	0.2
Chalk	1-3	
Limestone	7-61	0.1-0.3
Fractured basalt	30-91	0.2-1.5

Source: Based on data from Anderson (1979).

than estimated. The measured dispersion coefficients will have the advantage of accounting for the heterogeneity and anisotropy of the aquifer. Peaudecerf and Sauty (1978) described a method for determining the coefficients of dispersion from a dye pulse test by fitting an observed concentration curve to the dimensionless graphical form of Equation (7.6).

Retardation of Contaminants Moving Through an Aquifer

In a fashion similar to contaminant movement in soils, contaminants can be adsorbed on the solids in the aquifer. In some special cases the contaminants may also biodegrade in ground-water zones; however, the microorganism density in the ground-water zones is far less than that in soil. By definition, ground-water aquifers are saturated, hence volatilization does not occur. Similarly, there are no photochemical reactions therein because of the absence of light.

The concept of adsorption-desorption-coprecipitation reactions and models was introduced in detail in the preceding chapter for vadose-soil zones. The same concepts apply to the transport of solutes in ground-water zones. Adsorption is commonly incorporated into ground-water transport models using a retardation factor that is an empirical term derived from the partitioning coefficient, Π defined in the preceding chapter, and other parameters. The retardation factor is (Anderson, 1979; NCASI, 1985; Fetter, 1988)

$$R = \frac{V_x}{V_c} = 1 + \frac{\rho_b}{\rho_e} \Pi \quad (7.11)$$

where

V_x = average convective velocity of water in the aquifer

- V_c = velocity of the solute front where the solute concentration is one-half the original value
- ρ_b = specific density of the porous medium ($\text{kg/m}^3 = \text{g/l}$)
- p_e = effective porosity (dimensionless)
- Π = partitioning coefficient for the contaminant (l/gram)

Recall from Chapter 6 that the partitioning coefficient, Π , for chemical transport is related to the solubility of the chemical and to the properties of the porous medium, such as particulate organic carbon content (non-polar chemicals), and the pH, clay, metal oxide, or sulfide content for sorbable and toxic metals. When a mixture of sorbable and non-sorbable contaminants enters the ground-water zone, each species will travel at a rate that is dependent on its relative velocity, V_c/V_x . It is clear from these concepts that contaminants with higher partitioning coefficients ($>10^3 \text{ l/kg}$) will move at very slow rates, if at all, from the source of contamination if the aquifer is a composite of materials that have adsorbing capacity. It was pointed out in Chapter 6 that some porous media, such as sand and gravel, possess a very low adsorption capability. The concept of contaminant travel and the impact of dispersion and adsorption-desorption is shown on Figure 7.6.

Example 7.2: Movement of a Chemical Through Aquifer

A pesticide with the octanol partitioning coefficient $K_{ow} = 10^4 \text{ l/kg}$ was applied to a 10-ha orchard overlaying an unconfined aquifer. The average depth of the aquifer is 100 m, and the hydrodynamic slope of the aquifer is 0.5%. The aquifer contains primarily alluvial deposits (sand and gravel)

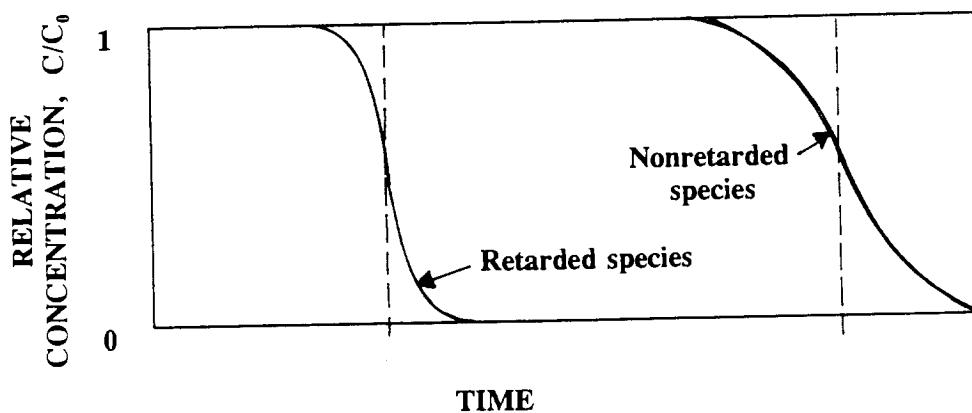


FIGURE 7.6. Influence of retardation on the movement of a solute front in a homogenous aquifer. (After Fetter, 1988.)

with about 0.1% organic matter. The specific volumetric density of the material is 1800 kg/m^3 . Estimate the spatial spread of the pesticide after 10 years in ground-water zones.

Solution

1. Estimate the advective velocity assuming the hydraulic conductivity of the aquifer (Table 7.3) is $K = 10^{-2} \text{ cm/sec}$ and the porosity (Table 7.2) is 30%. Following Example 7.1 $V_x = KS/\rho = 10^{-2} \times 0.005/0.3 = 0.000167 \text{ cm/sec} = 0.144 \text{ m/day}$
2. Estimate the retardation factor. The partitioning coefficient (Eqs. (6.19) and (6.20))

$$K_{oc} = 0.63 \times K_{ow} = 0.63 \times 10^4 = 6300 \text{ l/kg}$$

$$\Pi = K_{oc} \times (\%OC)/100 = 6300 \times 0.1/100 = 6.3 \text{ l/kg}$$

Using Equation (7.11)

$$R = V_x/V_c = 1 + (\rho_b/p_e)K_{oc}$$

$$= 1 + (1800 \text{ [kg/m}^3] \times 0.001 \text{ [l/kg]}/0.3) \times 6.3 = 38.8$$

Hence the velocity of the plume is

$$V_c = V_x/R = 0.144 \text{ [m/day]}/38.8 = 0.0038 \text{ m/day}$$

In 10 years the pesticide will move only $0.0038 \text{ [m/day]} \times 365 \text{ [days/year]} \times 10 \text{ [years]} = 13.8 \text{ meters}$ from the application area (neglecting dispersion).

ORIGIN OF NATURAL GROUND-WATER (BASE-FLOW) QUALITY

The constituents that appear in ground-water can enter the aquifer through the recharge area, as leachate from the upper soil layer and from subsurface disposal of wastewater, or they can originate from dissolution of minerals during ground-water passage through rocks and geological formations of the aquifer itself. As pointed out in Chapter 3 ground water becomes the base flow of surface-water bodies by ground-water discharge. The relation of geology to ground water and the origin of ground-water quality for the regions of the United States are described in Todd (1983). Also the U.S. Geological Survey publication by Hem (1985)

and hydrogeological texts by Freeze and Cherry (1979), Moore and Ramamoothy (1984), Adriano (1986), Fetter (1988), Domenico and Schwartz (1990), and others, discuss this topic in detail.

The dissolved chemical composition of ground water includes *cations*, or positively charged ions, and *anions*, which are negatively charged ions. Calcium (Ca^{2+}), iron (Fe^{2+} or Fe^{3+}), magnesium (Mg^{2+}), and sodium (Na^+) are examples of the most abundant cations. Sulfate (SO_4^{2-}), nitrate (NO_3^-), chloride (Cl^-), and bicarbonate (HCO_3^-) are the most common

TABLE 7.5 Dissolved and Dissociated Constituents in Ground Water Listed According to their Relative Abundance

<i>Major constituents</i> (greater than 5 mg/l)	
Bicarbonate	Calcium
Chloride	Magnesium
Sulfate	Silicon
Carbonic acid	Sodium
<i>Minor constituents</i> (0.01 to 10 mg/l)	
Carbonate	
Fluoride	
Nitrate	
Boron	
Iron	
Potassium	
Strontium	
<i>Trace contaminants</i> (less than 0.1 mg/l)	
Aluminum	Nickel
Antimony	Phosphate
Arsenic	Radium
Barium	Radon
Beryllium	Selenium
Cadmium	Silver
Chromium	Thallium
Cobalt	Thorium
Copper	Uranium
Lead	Vanadium
Manganese	Zinc
<i>Organic compounds</i> (shallow aquifers)	
Humic acids	Tannins
Fulvic acids	Lignins
Carbohydrates	Hydrocarbons
Aminoacids	Total Organic Carbon (TOC)
<i>Organic compounds</i> (deep aquifers)	
Acetate	Propionate

Source: After Davis and DeWiest (1966).

anions. In addition ground water may contain organic compounds, such as fulvic and humic acids, amino acids, tannins, and lignins. Table 7.5 lists some of the typical dissolved constituents.

Processes Controlling Natural Ground-Water (Base-Flow) Quality

Acid-base reactions. Water entering ground-water zones from atmospheric precipitation is generally acidic, that is, it has a pH below neutral. Dissolution of minerals in soil and ground-water zones provides the buffering of acidity.

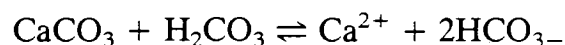
Under normal conditions of an unpolluted atmosphere, the pH of precipitation is in an equilibrium with the saturated CO₂ concentrations in the atmosphere. As reported in Chapter 4, the partial pressure of CO₂ is 0.0003 bars, which will result in the normal pH of unpolluted precipitation of 5.6. However, as a result of the acid rainfall phenomenon, the pH of precipitation may become as low as 3. Also due to CO₂ production by soil bacteria from decomposing soil organic matter, the partial pressure of CO₂ in soils can reach values up to 0.1 bar. This alone can result in pH values of soil and ground water that are significantly below 5.

After entering into soil and ground-water zones, the acidic water dissolves minerals until its dissolving capacity is exhausted. The salt and ionic content of ground water depends on the type of minerals, their solubility, and time of contact. The evolution of chemical ground-water quality begins in the upper (local) ground-water zone (defined in Chapter 3) by the dissolution of soil and subsoil minerals. Due to its elevated acidity, water entering the aeration zone is a reducing agent with the ability to reduce various substances from their oxidized state. For example, ferric iron (less soluble) can be reduced to far more soluble ferrous iron.

The bicarbonate (HCO₃⁻) content of water in the upper zone is a result of the dissolution of limestone (CaCO₃ × nH₂O) and dolomite (CaMgCO₃ × nH₂O) carbonate minerals, as well as silicate minerals, by the acidic action of soil water and ground water. The following simple reactions describe the process for carbonate minerals



and



which indicate that carbonic acid is buffered in the reaction. Similarly, stronger acids ($\text{pH} < 4.5$) in acid rainfall will also undergo a neutralization process in contact with limestone or dolomite, for example,



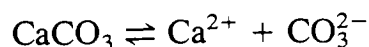
From the preceding reactions it can be seen that the removal of acidity by limestone and dolomite minerals will increase the hardness of ground water (*hardness* is defined as the content of polyvalent cations, such as Ca^{2+} , Mg^{2+} , Fe^{2+} , and Sr^{2+} , commonly expressed on a CaCO_3 equivalent basis).

The bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}) content of ground water represents the basic buffering system for neutralizing acid. This chemical buffering capacity is called *alkalinity*, and is defined as

$$\{\text{Alkalinity}\} = [\text{HCO}_3^-] + 2[\text{CO}_3^{2-}] + [\text{OH}^-]$$

and is expressed as CaCO_3 equivalent (Sawyer and McCarty, 1978).

Dissolution of minerals. Domenico and Schwartz (1990) have stated that dissolution of minerals is probably the most important process by which ground-water chemistry is controlled, and the recharge water derives almost the entire solute content through the dissolution of minerals along the flow path. The solubility of minerals and solids is determined by the dissolution-precipitation reaction and their equilibria. For example, a carbonate mineral such as limestone will partially dissolve into its ions



which is then expressed by the dissolution-precipitation equilibrium

$$K = \frac{[\text{Ca}^{2+}][\text{CO}_3^{2-}]}{[\text{CaCO}_3]}$$

where K = reaction equilibrium coefficient. The magnitude of the equilibrium coefficient for carbonate minerals is about $K = 10^{-8.3}$, indicating very low solubility in the neutral pH range; however, other minerals, such as salt (NaCl , $K = 34.67$) or gypsum (CaSO_4 , $K = 10^{-4.62}$), have relatively high solubility, so when ground water encounters these layers, high concentrations of Na^+ , Cl^- or Ca^{2+} , and SO_4^{2-} are common.

Of interest is the solubility of minerals containing some priority pol-

TABLE 7.6 Sulfides Containing Minerals and Dissociation Reactions

Sulfide	Reaction	Equilibrium Coefficient	Reference
Sphalerite	$\text{ZnS} \rightleftharpoons \text{Zn}^{2+} + \text{S}^{2-}$	$K = 10^{-23.9}$	Domenico and Schwartz (1990)
Galena	$\text{PbS} \rightleftharpoons \text{Pb}^{2+} + \text{S}^{2-}$	$K = 10^{-27.5}$	Domenico and Schwartz (1990)
Hydroxylapatite	$\text{Ca}_5\text{OH}(\text{PO}_4)_3 \rightleftharpoons 5\text{Ca}^{2+} + 3\text{PO}_4^{3-} + \text{OH}^-$	$K = 10^{-55.6}$	Freeze and Cherry (1979)
Gibbsite	$\text{Al}_2\text{O}_3 + 3\text{H}_2\text{O} \rightleftharpoons 2\text{Al}^{3+} + 6\text{OH}^-$	$K = 10^{-34}$	Freeze and Cherry (1979)
Fluorite	$\text{CaF}_2 \rightleftharpoons \text{Ca}^{2+} + 2\text{F}^-$	$K = 10^{-9.8}$	Freeze and Cherry (1979)

lutants (metals). Table 7.6 contains some examples using sulfides. The equilibrium dissolution reaction constant for many minerals and compounds is strongly affected by pH.

Complexation reactions. There are thousands of possible combinations of ions and molecules in water that may produce less soluble compounds. A *complex* is an ion that forms by combining simpler cations, anions, and molecules into a larger, typically less soluble, unit. The cation or central atom is in most cases one of metals. The anions, called ligands, include common inorganic species, such as F^- , Cl^- , SO_4^{2-} , PO_4^{3-} , in reduced conditions S^{2-} or HS^- and CO_3^{2-} , as well as organic molecules, such as humic and fulvic acids. This process of metal complexation was described in the preceding chapter for sediments and soils.

Redox reactions. When oxygen is present in ground water many compounds are present in oxidized and less soluble states. When oxygen is exhausted by the decomposition of organic carbon or other oxygen-consuming reactions redox conditions will convert them to more soluble compounds. However, conversion of sulfate (oxidized) to sulfite (reduced) may cause the more soluble metallic ions to be complexed by sulfides and precipitate as metal-sulfides.

Ion-exchange or surface adsorption-desorption reactions. The ionic species and polar and nonpolar molecules may react with surfaces of solids. *Ion exchange* is a process in which ions in the mineral lattice are replaced by one of the ions in the aqueous solution. In *adsorption-desorption* processes the solid surface attracts and retains a layer of ions or molecules from the solution (Fetter, 1988).

Adsorption-desorption reactions for phosphorus, ammonia, and priority pollutants were described in Chapter 6. Adsorption sites are provided mostly by clays and organic particulates and precipitated metal oxides and sulfide particles.

Ion exchange sites can be found mostly in clay and organic materials; however, Fetter (1988) pointed out that most soil materials have some ion exchange capacity. Both cation and anion exchanges can occur; however, the cation exchange capacity is more dominant. Of concern is the ion exchange process in which sodium from water replaces divalent cations (Ca^{2+} and Mg^{2+}) in soil minerals.

Ground-Water Quality Zones

Mineral ground-water quality is variable. Shallow aquifer water has low mineral content, but often exhibits seasonal or even day-to-day variations. Waters from deep underground zones have high mineral content, but fairly constant quality and temperature. The longer water resides in the underground zones, the higher the measured salt content.

Freeze and Cherry (1979) reported the evolution of mineral ground-water quality known as the Chebotarev or Ignaovich and Souline sequence. According to this concept, ground-water evolution tends to be in the direction of atmospheric to seawater quality. For large sedimentary basins, the sequence can be described in three main zones that correlate well with the depth:

1. The upper zone is characterized by active ground-water flushing through relatively well-leached rocks and soils. Water in this zone has HCO_3^- as the dominant anion and is relatively low in dissolved salts.
2. The intermediate zone has less active ground-water circulation and higher total dissolved solids. Sulfate is normally the dominant anion in this zone.
3. The lower zone has a very sluggish ground-water flow. Highly soluble minerals are commonly present. Very little flushing and leaching by ground water occurs due to the extremely slow movement to water. Dissolved solids are very high and chlorides are common anions.

These three water quality zones are similar to the ground-water zones (local, intermediate, and regional) described in Chapter 3.

Sources of Natural (Background) Ground-Water Quality

Mineral salts and dissolved (ionized) minerals are the most common sources of natural ground-water quality. In terms of contaminant content, natural ground-water contamination may be quite high and sometimes may hinder or even prevent the intended use of water from the source or, after discharge, use of the receiving body of water. For example, streams

draining watersheds containing lead minerals (such as galena) in surface layers may exhibit high natural lead content. Klusman and Edwards (1977) measured toxic metals in ground water from the mineral belt of Colorado and noted that the drinking water standards were violated in 14% of the samples for Cd, 1% for Cu, 2% for Hg, and 9% for Zn. It may be noted that present water quality standards for toxic metals are more stringent than those in use when Klusman and Edwards' study was conducted.

Of the 129 compounds and chemicals on the U.S. EPA's Priority Pollutant List, 13 are inorganic elements that have natural (background) occurrence. These priority pollutants include: antimony, arsenic, beryllium, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, thallium, and zinc. The following list contains the natural sources and minerals from which these elements may originate:

<i>Element</i>	<i>Natural Source of Minerals</i>
Antimony	Stibnite (Sb_2S_3), geothermal springs, mine drainage
Arsenic	Metal arsenides and arsenates, sulfide ores (arsenopyrite), arsenite (HAsO_2), volcanic gases, geothermal springs
Beryllium	Beryl ($\text{Be}_3\text{Al}_2\text{Si}_6\text{O}_{16}$), phenacite (Be_2SiO_4)
Cadmium	Zinc carbonate and sulfide ores, copper carbonate and sulfide ores
Chromium	Chromite (FeCr_2O_4), chromic oxide (Cr_2O_3)
Copper	Free metal (Cu^0), copper sulfide (CuS_2), chalcopyrite (CuFeS_2), mine drainage
Lead	Galena (PbS)
Mercury	Free metal (Hg^0), cinnabar (HgS)
Nickel	Ferromagnesian minerals, ferrous sulfide ores, pentladite ($(\text{Ni,Fe})_9\text{S}_8$), nickel oxide (NiO_2), nickel hydroxide ($\text{Ni}(\text{OH})_3$)
Selenium	Free element (Se^0), ferroselite (FeSe_2), uranium deposits, black shales, chalcopyrite-pentladite-pyrrhotite deposits
Silver	Free metal (Ag^0), silver chloride (AgCl_2), argentide (AgS_2), copper, lead, zinc ores
Thallium	Copper, lead, silver residues
Zinc	Zinc blende (ZnS), willemite (ZnSiO_4), calamine (ZnCO_3), mine drainage

Although nitrate in ground water is primarily of anthropogenic (cultural) origin geologic N, that is, N associated with certain geologic for-

mations of sedimentary origin, is also known (Boyce et al., 1976; Chalk and Keeney, 1971). Chalk and Keeney found widely varying concentrations of NH_4^+ and NO_3^- in Wisconsin limestones and suggested that many limestones are potential sources of NO_3^- in ground water, and hence base-flow contamination.

Kreitler and Jones (1975) reported nitrate levels in the ground water of Runnels County, Texas, that reached values of 250 mg/l, well above the drinking water standard of 10 mg/l. It was found that almost 80% of the nitrate content leached from natural soil nitrogen as a result of cultivation during the last 50 years.

Estimating Base-Flow Quality

The nature and concentrations of dissolved constituents in natural ground water (base-flow contributions) are dependent on the composition of the aquifers through which the ground water flows. The regional characteristics of aquifers are described in a publication by Hem (1985). Depending on the type of geology of the region, water may originate from primarily carbonate formations, crystalline rocks, or sedimentary systems (Freeze and Cherry, 1979). Furthermore, other regional characteristics such as type of surficial vegetation, primarily forestation of the watershed and/or degree of urbanization or agricultural land use, slope and elevation of terrain, soil permeability, and precipitation amount and distribution will affect ground-water and base-flow quality. In agricultural areas with irrigation, ground-water (base-flow) quality is affected by irrigation return flow. The new ecoregional approach to water quality grouping (Omernik and Gallant, 1990; Galant et al., 1989) may provide new relationships for regional base-flow quality determinations.

Mineral loads in base flow can be estimated using an approach developed by Betson and McMaster (1975). These authors correlated the water quality of some undisturbed streams in the Tennessee Valley Authority area using the following logarithmic functional relationship between the quality and flow:

$$C = a(13.66 \times Q/DA)^b \quad (7.12)$$

where

- C = the concentration of a mineral constituent (mg/l)
- Q = the stream flow (m^3/s)
- DA = the drainage area (km^2)
- a, b = empirically determined coefficients

The two coefficients in Equation (7.12) were related to land use, soils, and geological factors by a linear multiregression formula

$$a, b = N_1F + N_2CR + N_3S + N_4I + N_5U \quad (7.13)$$

where

a, b	= the coefficients just defined
F	= the fraction of the watershed area that is forested
CR	= the fraction of the watershed over carbonate rock
S	= the drainage area fraction over shale-sandstone rock
I	= the drainage area fraction over igneous rock
U	= the drainage area fraction over unconsolidated sediments
N_1, \dots, N_5	= regression coefficients

The four independent geological variables simply allocate the drainage area among the rock types present in the watershed and must sum one.

Table 7.7 shows regression coefficients for major mineral constituents obtained by analysis of TVA watersheds. As stated by the authors, the use of Equation (7.13) requires caution since constituent rating curves (Eq. (7.12)) have been found to display a hysteric effect with seasons and with rising and falling stages of the hydrograph (see Chapter 5 for an explanation of the difference in concentrations for suspended solids). Variations among watersheds are also influenced by other factors, as previously mentioned.

Example 7.3: Base Flow Quality

Estimate base background nitrate content in a watershed with the following characteristics:

$$\text{Specific flow } \frac{Q}{DA} = 0.005 \text{ m}^3/\text{sec-km}^2$$

% forest	$F = 25\%$
% carbonate rock	$CR = 5\%$
% sandstone	$S = 35\%$
% igneous rock	$I = 2\%$
% unconsolidated rock	$U = 58\%$

Solution Using the Betson and MacMaster concept, select the coefficients from Table 7.7. Then the coefficients for the rating equation become (Eq. (7.13))

$$\begin{aligned}
 a &= -1.13 \times 0.25 + 3.52 \times 0.25 + 1.02 \times 0.35 \\
 &\quad + 1.3 \times 0.02 + 0.84 \times 0.58 \\
 &= 1.47 \\
 b &= -0.70 \times 0.25 + 0.262 \times 0.25 + 0.899 \times 0.35 \\
 &\quad + 1.063 \times 0.02 + 0.297 \times 0.58 \\
 &= 0.67
 \end{aligned}$$

Hence, the nitrate concentration becomes (Eq. (7.12))

$$C_{\text{NO}_3} = a(13.66Q/DA)^b = 1.47(13.66 \times 0.005)^{0.67} = 0.24 \text{ mg/l}$$

IMPACT OF DIFFUSE POLLUTION ON GROUND WATER AND BASE FLOW

There are many potential sources of ground-water contamination, including septic tank systems, solid waste disposal, land disposal of sewage, industrial wastes and sludge, agricultural and suburban irrigation, fertilizer and pesticide application in agricultural and suburban land uses, leaks and spills of chemicals and oils on the surface and from underground leaking storage tanks, mining, and road deicing. Some of these sources can be viewed as point sources of pollution. They are regulated and may result in only a localized aquifer contamination. Other sources, such as septic tanks and large-scale agricultural practices, are typical nonpoint pollution problems still mostly unregulated. It has been estimated that about 2% of aquifers have been contaminated (Lehr, 1981).

Effects of Septic Tank Disposal Systems

The term septic system is commonly used to describe a subsurface, anaerobic sewage disposal that uses soil filtration and adsorption for attenuating the effluent. Detailed information on ground-water impact of septic tank disposal systems is contained in a publication by Canter and Knox (1985).

Approximately 20 million residents, or 29% of the U.S. population, dispose of their sewage by individual on-site systems. The total amount of sewage and wastewater discharged to the subsurface in the United States is $3 \times 10^9 \text{ m}^3/\text{yr}$ (U.S. EPA, 1977a, 1977b; Canter, Knox, and Fairchild, 1987). Apparently, septic tanks represent the highest total volume of wastewater discharged directly to ground water and are the most frequently recorded sources of contamination of ground water and surface flow.

The amount of discharge from septic tank systems is commonly es-

TABLE 7.7 TVA Base-Flow Mineral Quality Model: Regression with Forest and Geological Variables

Quality Parameter	Regression Coefficient	<i>N</i> ^b	Regression Coefficient Value					Statistics		
			<i>N</i> ₁ <i>F</i>	<i>N</i> ₂ <i>C</i>	<i>N</i> ₃ <i>S</i>	<i>N</i> ₄ <i>I</i>	<i>N</i> ₅ <i>U</i>	<i>R</i>	Standard Deviation	<i>F</i>
SiO ₂	<i>a</i>	64	-1.26	5.42	6.78	10.2	8.95	0.64	1.69	8.11
	<i>b</i>		-0.135	0.051	0.099	0	-0.029	0.40	0.114	2.26
Fe	<i>a</i>	29	0.035	0.020	0.009	-0.008	0.387	0.95	0.004	42.25
	<i>b</i>		-0.173	0.272	0.104	-0.125	0.397	0.38	0.482	0.80 ^a
Ca	<i>a</i>	66	-8.52	53.9	13.4	8.32	8.41	0.85	9.0	31.42
	<i>b</i>		0.064	-0.116	-0.203	-0.229	-0.005	0.32	0.153	1.32 ^a
Mg	<i>a</i>	66	-2.81	11.4	3.41	3.05	2.45	0.75	2.62	15.50
	<i>b</i>		-0.148	-0.145	-0.074	-0.104	0.513	0.67	0.197	9.99
Na	<i>a</i>	44	-1.79	2.23	2.84	3.00	3.74	0.74	0.50	9.45
	<i>b</i>		-0.318	0.079	0.122	0.110	-0.007	0.48	0.138	2.33
K	<i>a</i>	44	-1.08	2.51	1.94	1.58	1.80	0.72	0.47	8.12
	<i>b</i>		-0.152	-0.195	-0.061	0.033	-0.158	0.20	0.254	0.32 ^a
HCO ₃	<i>a</i>	66	-22.8	200	35.3	26.8	21.8	0.86	32.1	34.67
	<i>b</i>		0.110	-0.156	-0.294	-0.355	-0.139	0.35	0.132	1.71 ^a

SO ₄	<i>a</i>	66	-7.41	9.15	12.5	7.90	9.56	0.39	5.35	2.19
	<i>b</i>		-0.302	0.103	0.155	0.272	0.592	0.49	0.274	3.69
Cl	<i>a</i>	66	-1.86	3.21	2.95	2.58	3.81	0.60	0.93	6.68
	<i>b</i>		-0.171	0.010	0.088	0.099	0.067	0.24	0.14	0.72 ^a
NO ₃	<i>a</i>	63	-1.13	3.52	1.02	1.30	0.84	0.80	0.71	20.11
	<i>b</i>		-0.70	0.262	0.899	1.063	0.297	0.26	0.671	0.84 ^a
TDS	<i>a</i>	66	-39.0	195.6	68.5	55.5	57.7	0.84	30.6	28.42
	<i>b</i>		0.016	-0.094	-0.146	-0.142	0	0.26	0.14	0.88 ^a
CaCO ₃	<i>a</i>	66	-33.4	182.8	48.1	34.3	31.5	0.86	29.3	33.06
	<i>b</i>		0.033	-0.150	-0.176	-0.222	0.131	0.49	0.151	3.78
Specific conductance	<i>a</i>	46	-145	357	180	142	128	0.88	54	26.74
	<i>b</i>		-0.015	-0.078	-0.134	-0.095	0.051	0.43	0.106	1.82 ^a
pH	<i>a</i>	65	-0.573	8.37	7.32	7.33	6.86	0.73	0.42	13.53
	<i>b</i>		-0.003	-0.010	-0.003	-0.013	-0.003	0.16	0.021	0.32 ^a
Color	<i>a</i>	63	-1.79	2.50	9.17	9.75	10.8	0.23	8.65	0.62 ^a
	<i>b</i>		-0.376	0.211	0.339	0.204	0.448	0.32	0.364	1.26 ^a

Source: After Betson and McMaster (1975).

^aNot significant at 0.9 level.

^b*N* is number of watershed analyses.

estimated as 280 liter/cap-day (75 gpcd). Septic tank effluents contain 40 to 80 mg N/l, 11 to 31 mg P/l, and 200 to 400 mg/l of BOD₅ (Sikora et al., 1976; Canter, Knox, Fairchild, 1987). Based on reported efficiencies of soil absorption systems, Canter and Knox (1985) reported the following typical concentrations entering the ground water: BOD₅—28 to 84 mg/l; COD—57 to 142 mg/l; ammonia nitrogen—10 to 78 mg/l; and total phosphates—6 to 9 mg/l. Other ground-water constituents of concern include bacteria, viruses, nitrates, synthetic organics, and toxic metals. On the other hand, Brown et al. (1979) and Reneau and Petry (1976) reported that organic matter (BOD₅, pathogenic microorganisms) and phosphates were effectively removed by most properly designed and permitted subsurface disposal systems and did not penetrate more than 1.5 m below the level of discharge or beyond the immediate vicinity of the seepage field.

Nitrification is typically completed in seepage fields located in well drained soils and the mobile nitrate-N will move into ground-water zones. An investigation of nitrogen mass balance in Long Island, New York, revealed that more 20% of the total nitrogen contribution from subsurface disposal systems leached as nitrate-N into ground water (Andreoli et al., 1979).

According to Fetter (1988) septic tanks are most likely to contribute to ground-water contamination in areas where (1) there is a high density of homes with septic tanks, (2) the soil layer over permeable bedrock is thin, (3) the soil is extremely permeable such as gravel, or (4) the water table is shallow, one meter or less below the surface. Typically, local ordinances regulating permits for septic tank disposal do not pay enough attention to the preservation of ground-water resources from pollution by septic tank effluents. The standard percolation test simply favors highly permeable, sandy, and gravel soils. Such soils do postpone the failure of the system by hydraulic overloading; nevertheless, adsorption and the purifying capacity of such soils is greatly reduced and pollutants can move downwards and contaminate the ground water. It also has to be realized that most septic tank systems installed in the 1950–1970 period are now exceeding their design life, which is typically 10 to 15 years. Use of man-made organic chemical additives to prolong the life of the systems and/or improve their impaired function resulted in organic contamination of ground water (U.S. EPA, 1980).

Underground Storage Tanks

Underground leaking storage tanks are now considered to be a major source of ground-water contamination by chemicals, especially those from

petroleum hydrocarbons. Because there are so many underground tanks (about 2 million) and only a few of them are corrosion resistant (Canter, Knox, and Fairchild, 1987), the problem could be considered to be of a diffuse nature that could reach regional rather than localized scope (for example, New Jersey). The problem has received serious attention and concern of the U.S. Environmental Protection Agency which, in 1985 created the Office of Underground Storage Tanks.

Buried gasoline tanks in service stations are the most common source of the problem. The leaking tanks are a source of a number of aliphatic and aromatic hydrocarbons in ground water (see Chapter 6 for a definition). Many of these hydrocarbons are classified as priority pollutants.

Land Application of Water and Wastewater

Sewage and Wastewater Disposal

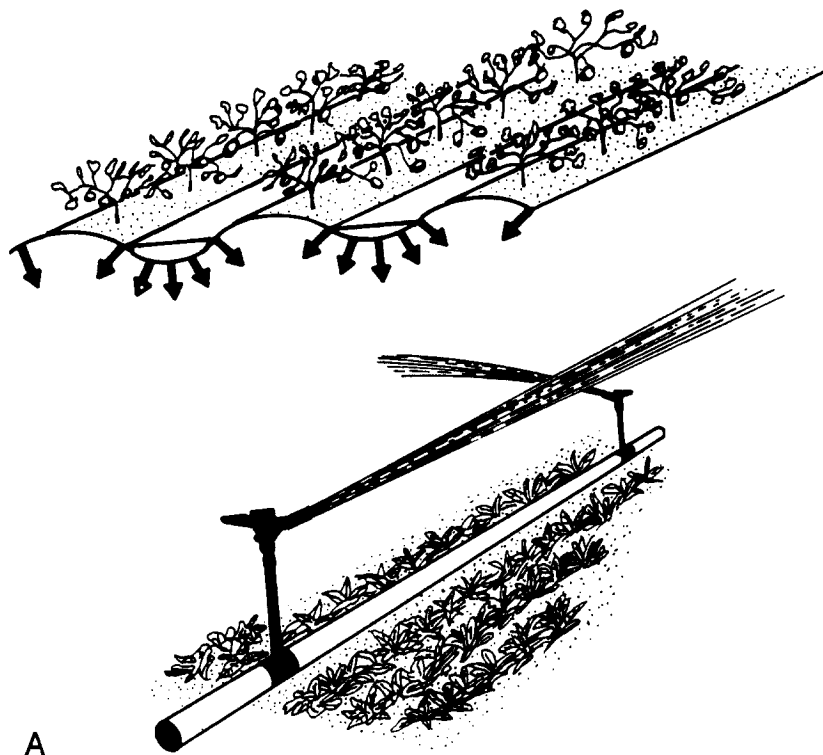
The "no pollution" policy of the Clean Water Act advocated in the 1970s caused an increased interest in land disposal of liquid wastes such as conventionally treated sewage effluents, processing plant wastes, animal wastes and feedlot runoff, and sewage sludge. In many arid regions land application of sewage helps alleviate shortages of irrigation water and is even used to replenish the ground-water aquifers. The city of Tucson in Arizona is reusing almost all of its municipal treatment plant effluent for urban irrigation. The long-lasting drought in southern California in the late 1980s and early 1990s increased demand for land disposal of sewage as a means of water reuse.

Although the soil has a great capacity for attenuating contaminants, the reclaimed wastewater from ground-water zones is obviously not of the same quality as the native ground water (Bouwer, 1974). There are three types of land applications of wastewater (Reed, Middlebrooks, and Crites, 1988; U.S. EPA, 1980, 1984):

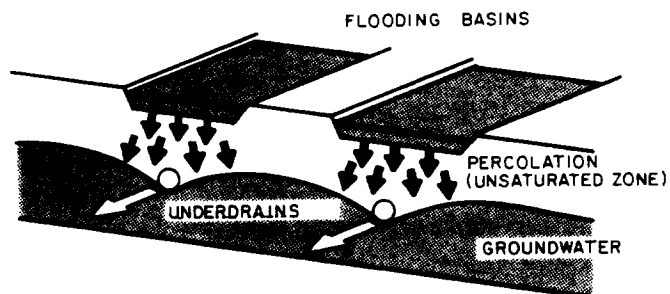
- slow-rate systems
- overland flow systems
- rapid infiltration

The three systems are shown on Figure 7.7.

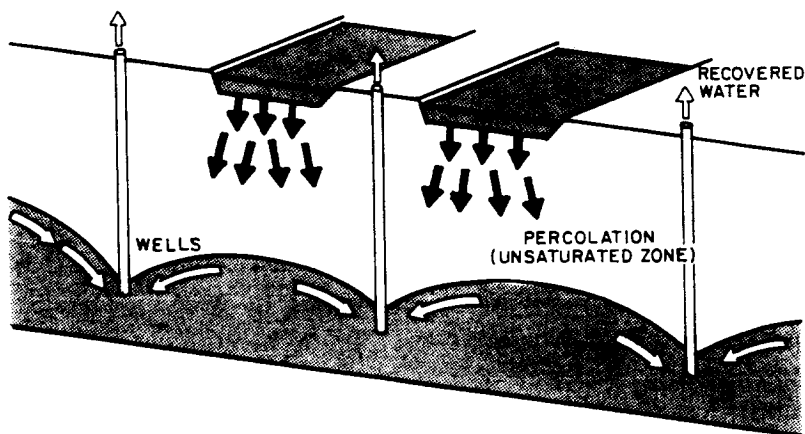
The *slow-rate systems* (SR) are most common in the United States for treatment of municipal wastewater and effluent reuse in arid areas. In Europe, these systems have been in use for centuries. Their hydraulic loading rate is mostly matched to irrigation and nutrient requirements for crops and soil permeability. In arid regions the hydraulic loading is related to the irrigation requirement and prevention of salt buildup in



A



(a) RECOVERY OF RENOVATED WATER BY UNDERDRAINS



B (b) RECOVERY OF RENOVATED WATER BY WELLS

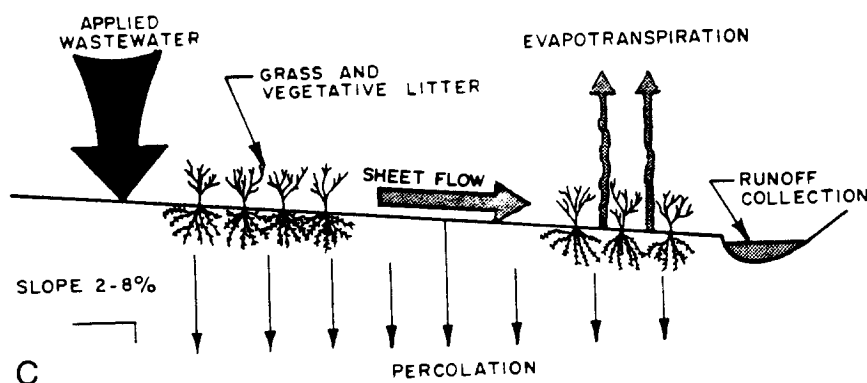


FIGURE 7.7. Land application systems of sewage and wastewater. (A) Slow-rate application. (B) Rapid infiltration. (C) Overland flow systems. (From Novotny et al., 1989.)

soils. These systems are essentially irrigation systems and have similar problems with irrigation return flow and its impact on ground water and base flow (see the next section on irrigation). For the winter nongrowing period, wastewater flows require storage. Of the three systems mentioned the SR exhibits the highest nutrient removal due to the combined effects of nutrient uptake by crops and attenuation by soils.

In *overland flow systems (OF)* wastewater is treated as it moves in graded and maintained grassed and vegetated sloped areas and the treated effluent is collected as residual runoff at the bottom of the slope. Percolation of wastewater is not desirable and should be minimized by the selection of slowly permeable soils, soil compaction, and/or locating these systems over an impermeable subsurface stratum. Under these conditions the impact of these systems on ground-water resources should be minimal. These systems are similar to grassed buffer strips used for the treatment of urban and agricultural runoff (see Chapters 10 and 11). Nitrogen removal is accomplished by nitrification–denitrification processes and depends on the BOD/nitrogen ratio. If the nitrogen in the effluent is primarily in nitrate form, then the removal is minimal (Reed, Middlebrooks, and Crites, 1988).

In contrast to OF systems *rapid infiltration (RI)* systems rely on the infiltration and filtration of wastewater in permeable soils. If subsoils are permeable, the effluent will reach ground water and, if improperly designed, has the potential of becoming a cause of ground-water contamination. Removal of contaminants in the upper soil layer is accomplished by physical–chemical interaction (adsorption) and biochemical degradation (both aerobic and anaerobic). Vegetation and its nutrient

uptake is not considered. Again if most of the nitrogen is in nitrate form, the removal efficiency is greatly reduced.

Application

Sewage can be applied at a low rate in SR systems or a high rate for OF and RI systems. The low application rates—in centimeters week—are based on matching the nutrients from the applied wastewater with their uptake by crops, which reduces nitrate accumulation and pollutant buildup in soils. These systems present the minimal potential for the contamination of ground-water resources and base flow. The disadvantage of low-rate application systems is the large area requirement (approximately 30 ha of land per 1000 m³/day of sewage).

The high-rate systems require only a fraction of the land needed for low-rate systems. Since water in these systems is applied in excess of evapotranspiration rate, a portion of the applied flow will leach into ground-water zones. To minimize the impact on ground-water resources, the land application systems should be designed and operated (a) to obtain recharge water of the best possible quality (particularly with regard to nitrogen), and (b) to restrict the spread of recharged water into the native ground water (Bouwer, 1974; Reed, Middlebrooks, and Crites, 1988; U.S. EPA, 1981, 1984).

Problems and Restrictions

The problems associated with the land application of wastewater, especially RI systems, are similar to septic systems discussed previously; however, much greater volumes of wastewater are concentrated in a smaller area. Mobile pollutants, such as nitrates are of the greatest concern, since evidence indicates that several other common contaminants (BOD, pathogenic microorganisms, and phosphates) remain near the area of application. Bacteria and viruses die off quite rapidly as wastewater passes through the soil material (Bell and Bole, 1978; U.S. EPA, 1980, 1984).

Freeze and Cherry (1979) state that ground-water contamination by mobile organics may become a serious problem. Treated wastewater contains many dissolved organic compounds, and some of the potentially dangerous components such as chlorinated hydrocarbons, may be created by the treatment process itself. Since many of these compounds are not biodegradable and can be partially mobile, their impact should always be evaluated.

As shown on Figure 7.8, land application systems may contaminate all three ground-water systems, that is, local, intermediate, and regional aquifers. The portion of the aquifer that is recharged by treated waste-

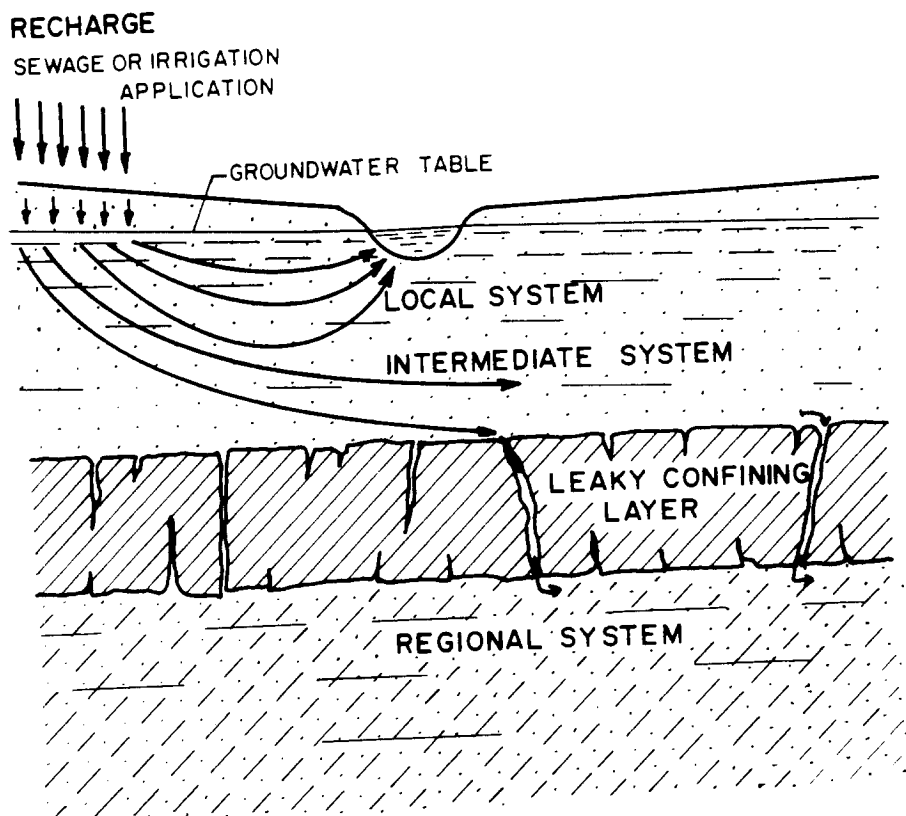


FIGURE 7.8. Possible routes of contaminants in ground-water systems.

water effluents should not be used as a source of drinking water and should be restricted, and water should be taken out at some distance from the recharge area (Fig. 7.9). This distancing could occur naturally as a base flow into a nearby stream or lake, or artificially by drains (shallow aquifer), or by wells (deep aquifer). After collection, water can be reused for irrigation, recreation, aesthetics, or other nonpotable uses, or discharged into receiving streams to augment their flows (Bouwer, 1974). With such aquifer zoning and partitions, the portion of the aquifer between the land application site and the point of discharge is used as a natural filtration system.

Land Application of Sludge

Sludge generated by wastewater treatment facilities is commonly applied to agricultural lands as a fertilizer and soil conditioner. The effect of the land application of sludges on ground-water quality depends on the transformation that occurs within the topsoil horizon and in unsaturated soils. Although most toxic metals will be retained by the topsoil, the toxic

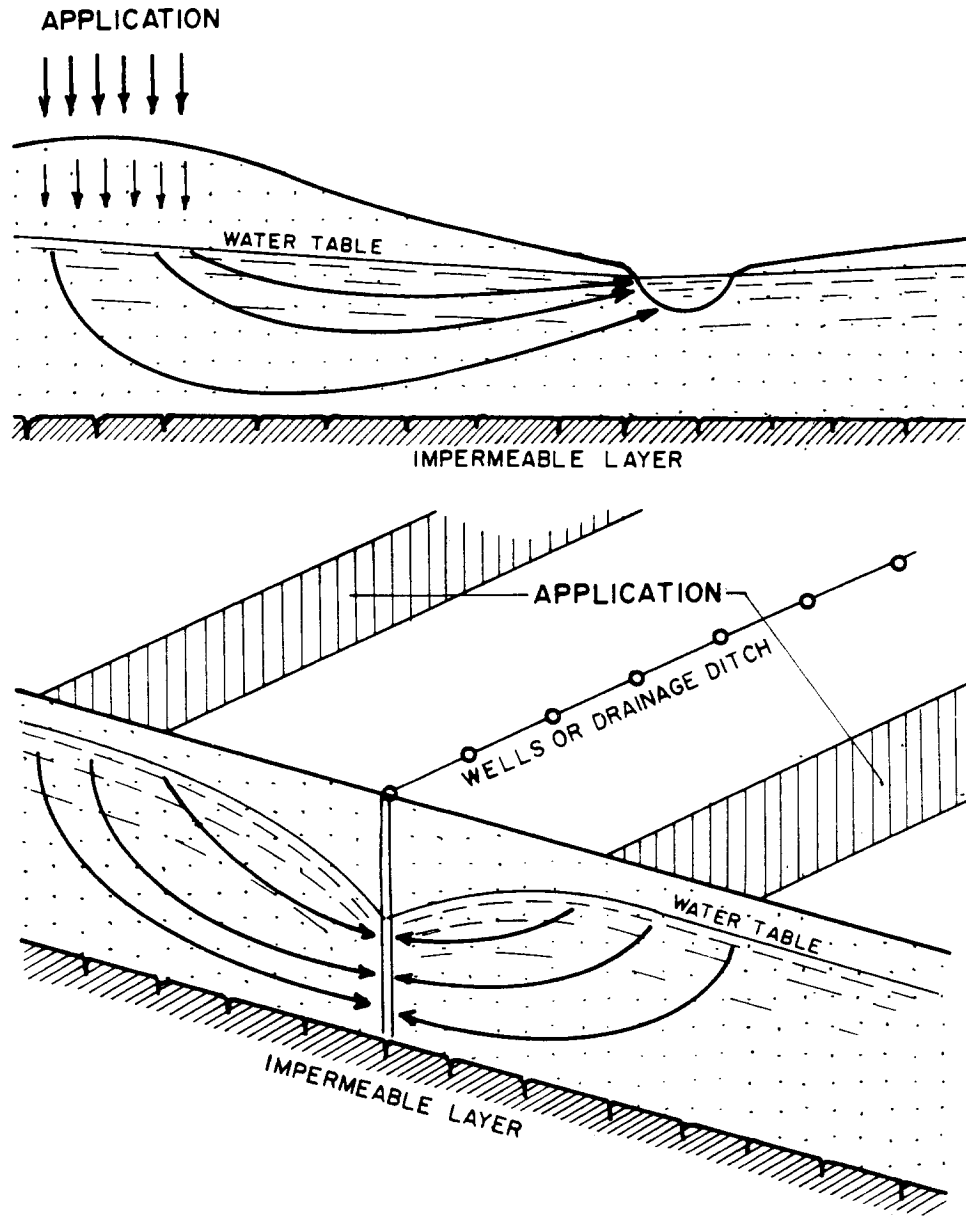


FIGURE 7.9. Restriction of aquifer contamination by sewage application, natural drainage (upper portion), and artificial drainage (lower portion) (according to Bouwer, 1974).

metal content of sludges is of concern. Concentrations of toxic metals in wastewater sludges is much higher than those in raw wastewater.

Irrigation and Irrigation Return Flow

Water applied to land either in the form of a treated effluent or as water withdrawn from a nearby surface-water body contains certain dissolved salts and dissociated ions. A portion of the irrigation water, after appli-

cation on an irrigated area, is returned to the atmosphere by evapotranspiration. Since evaporated or transpired water has no salt content, there is a subsequent salt and contaminant buildup in soils. The portion returned to the atmosphere may range from less than 20% in high-rate application systems in humid climatic conditions to almost 100% in low-rate application systems in arid or semiarid climatic zones.

In order to maintain an acceptable salt content of soils to sustain crop growth and the fertility of the soils, excess irrigation water must be applied if natural precipitation is not sufficient to control salt buildup. The excess water containing increased salinity and leachate from soils is collected by drainage (natural or man-made) and/or will percolate into ground-water zones. The irrigation tail water collected by drainage systems (Figs. 7.10 and 7.11) or leached into groundwater zone is called *irrigation return flow*, and it represents one of the more serious problems associated with diffuse pollution from agriculture.

The concentration of salts in water percolating through the soil root level zone into irrigation return flow or to the ground water can be computed from the following mass balance:

$$C_i D_i = C_d (D_i - D_e) \quad (7.14)$$

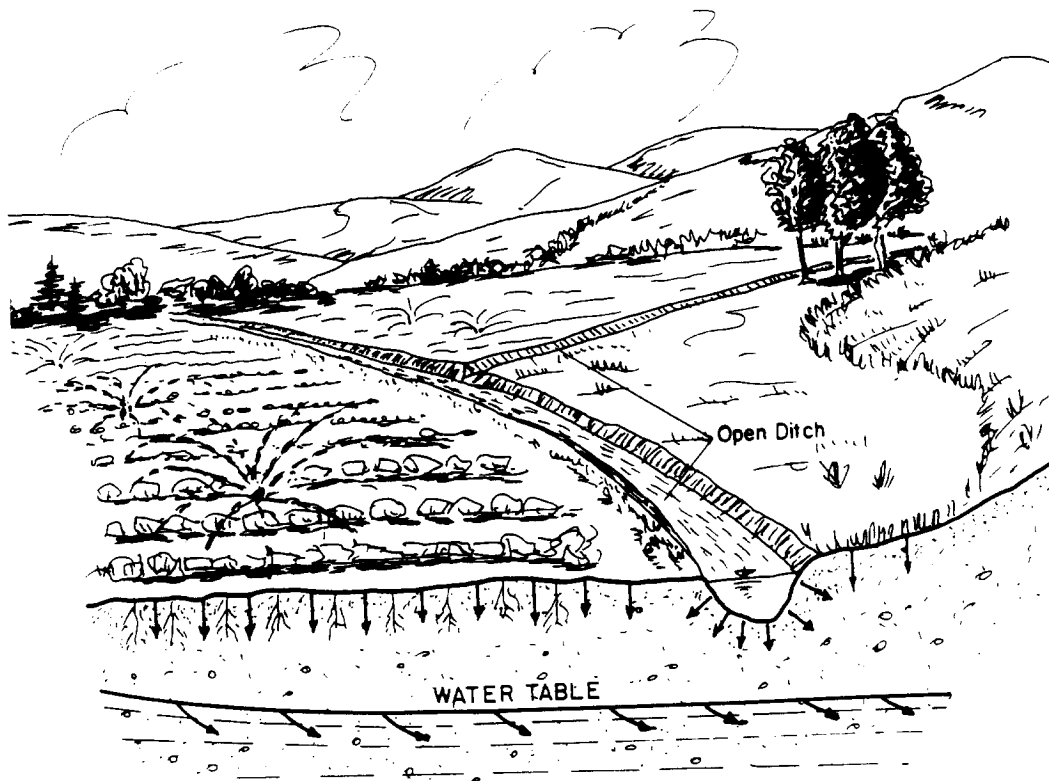


FIGURE 7.10. Irrigation return flow systems.



FIGURE 7.11. Drainage ditch in agricultural fields. Drainage water often contains pollutants leached from soils by percolated water and in irrigation return flows.

where

C_i = salt or conserved contaminant concentration of water or wastewater used for irrigation

D_i = amount of irrigation water, also including effective precipitation (precipitation that is not lost as surface runoff)

C_d = salt or conserved contaminant concentration of water percolating from the root zone downward

D_e = amount of water used by evapotranspiration

The amount of excess irrigation water that has to be applied to control salt or contaminant buildup in soil depends on the tolerance of crops to the compound in soil water, the compound content in irrigation water, evapotranspiration rate, crop uptake, and other losses from the system. The leaching ratio is computed from Equation (7.14) as

$$\frac{D_i}{D_e} = \frac{C_d}{C_d - C_i} \quad (7.15)$$

The salinity of irrigation water is usually expressed as conductivity in micromhos per centimeter ($1000 \mu\text{mho}/\text{cm} \approx 640 \text{ mg/l}$ m of total dis-

solved solids—TDS). The salt tolerance of crops ranges from less than 500 $\mu\text{mho/cm}$ for salt-sensitive crops, such as most fruit trees and some vegetables (celery, strawberries, or beans) to more than 1500 $\mu\text{mho/cm}$ for salt-tolerant crops, such as cotton, beets, barley, and asparagus. Most common grain crops and vegetables have medium tolerance (500 to 1500 $\mu\text{mho/cm}$) to salts. The leaching requirement is then defined as

$$LR = \frac{EC_i}{EC_d} = \frac{C_i}{C_d} \quad (7.16)$$

where

EC_i = electric conductivity of irrigation water

EC_d = salt tolerance of crops or conductivity of drainage water

Combining Equations (7.15) and (7.16), the leaching ratio becomes

$$\frac{D_i}{D_e} = \frac{1}{1 - LR} \quad (7.17)$$

Although the irrigation return flow has been recognized as a water quality problem, the Clean Water Act specifically excluded agricultural runoff and irrigation return flows from the definition of pollution. However, there is no doubt about the pollution effects of these flows. Investigations in central Wisconsin (Saffigna and Keeney, 1977) and elsewhere (Burwell et al., 1976; Brown, 1975) revealed that nitrate concentrations in the subsurface water in agricultural areas receiving irrigation are well above the background level. In the Wisconsin study, nitrate-N concentrations ranged up to 56 mg/l.

However, Kreitler and Jones (1975) and Brown (1975) have documented that the nitrate in the ground water below irrigated fields originated from the natural soil organic nitrogen that was leached from the soil as a result of cultivation and irrigation over a 50-year period. Nitrate levels of the ground water below irrigated fields in Runnels County in Texas reported by Kreitler and Jones have reached average values of 250 mg/l, and over 80% of it was attributed to native nitrogen leaching from the soil. The presence of NH_4^+ in ground water usually indicates incomplete nitrification of nitrogen in sewage. Similar conclusions were reached by Brown for subsurface nitrate content in the irrigated fields of the San Joaquin Valley in California.

The salinity problem is especially troublesome in arid zones of the western United States. It must be realized that water in some streams can be reused for irrigation several times, since downstream users irrigate

mostly with irrigation return flows from upstream farms, with increased salinity and pollution content at each reuse.

Example 7.4: Amount and Quality of Irrigation Return Flow

An agricultural field growing crops was irrigated by a treated effluent with the following quality characteristics:

TDS—500 mg/l ($EC = 781 \mu\text{mho/cm}$)
 nitrogen—10 mg/l ($= \text{g/m}^3$)

The effluent was applied at a rate of 10 cm/week. The evapotranspiration rate during the irrigation period (lasting 2 months) was 5 cm/week and there was no appreciable precipitation during the period. The crop yield was about 5 tonnes/ha and the nitrogen content of the crop was 20 kg/tonne (see Table 6.4 for yields and N contents of specific crops). Estimate the amount and concentration of nitrogen leached into the ground water and the salinity of the soil and drainage water (irrigation return flow). The pH of the soil is around 7, which implies that ammonia volatilization is minimal.

Nitrogen uptake assuming 4 months growing period

$$\begin{aligned} UP_N &= \text{crop yield} \times \text{nitrogen content/growing period} \\ &= 5 \text{ (tonnes/ha)} \times 20 \text{ (kg/tonne)}/16 \text{ weeks} \\ &= 6.25 \text{ kg/ha-week} = 6250 \text{ g/ha-week} \end{aligned}$$

Nitrogen input from the effluent is

$$0.1 \text{ (m/week)} \times 10,000 \text{ (m}^2\text{/ha)} \times 10 \text{ (g/m}^3\text{)} = 10,000 \text{ g/week-ha}$$

Solution The amount of nitrogen leached can be obtained by subtracting the plant uptake from the nitrogen input. For calcareous soil ammonia volatilization should be also considered (see Eq. (6.31)). Hence

$$\text{Nitrogen leached} = 10,000 - 6250 = 3750 \text{ g/week}$$

Nitrogen concentration in the leachate is

$$C_{dN} = \frac{\text{Nitrogen leached}}{\text{Volume of water leached}} = \frac{3750 \text{ (g/week)}}{(0.1 - 0.05) \times 10,000} = 7.5 \text{ mg/l}$$

Estimate the salinity of the leachate. Using Equation (7.17) the leaching requirement factor is

$$LR = 1 - D_e/D_i = 1 - 5/10 = 0.5$$

and by combining Equations (7.16) and (7.17)

$$C_d = \frac{C_i}{LR} = \frac{300}{0.5} = 1000 \text{ mg/l} = 1562 \text{ } \mu\text{mho/cm}$$

Note: This example presumes no leaching of the original nitrogen content of the soil. Examples 6.6 to 6.8 from the preceding chapter may be used to approximately estimate mineralization and leaching of the soil nitrogen.

Ground-Water Pollution from Solid Waste Disposal Sites

In 1970 there were about 20,000 solid waste disposal sites in the United States. However, only 6% were classified as sanitary landfills that do not cause environmental problems and were properly operated. Today there are few licensed and permitted landfills and the suitable landfill site is very difficult to locate. Solid waste disposal sites are now sophisticated engineering operations employing resource recovery (collection of methane and subsequent conversion to energy), collection of liquid wastes (leachate) produced by the landfill with subsequent pretreatment and treatment and daily covering of wastes with soil. A landfill site after ceasing operation should be reclaimed. Figure 7.12 shows an example of a well-operated landfill site in southeastern Wisconsin.



FIGURE 7.12. Omega Hill solid waste (refuse) disposal site in southeastern Wisconsin.

However, for each well-designed and operated landfill there are hundreds of abandoned unsanitary dumps of refuse and toxic chemicals that are now causing ground-water contamination problems. During the decade between 1970 and 1980, a large number of the landfills, including some receiving radioactive wastes, were established. Stored and decomposing wastes and leaching from disintegrating drums left on these sites will represent a serious problem for decades. In the United States, such sites have been inventoried and if severe problems have occurred they were classified by the U.S. Environmental Protection Agency as hazardous waste disposal sites (so-called "Superfund" sites).

Although solid waste disposal sites are considered point sources of pollution, leachate from unsanitary landfills and dumps may have polluted large portions of adjacent aquifers and appear as contaminated base flow in a diffuse manner. Furthermore, some dangerous toxic compounds are commonly a part of the overall composition of the landfill leachate, especially when the landfill is used for disposal of toxic chemicals. Table 7.8 shows the ranges in concentration for various chemical constituents and the physical parameters of typical leachates from municipal solid waste disposal sites. It should be noted that in countries that use coal for household heating, the composition of leachate may be quite different from that typical for U.S. conditions (Johansen and Carlson, 1976).

TABLE 7.8 Leachate Characteristics from Municipal Solid Waste Disposal Sites

Leachate	Median Value (mg/l)	Ranges of All Values (mg/l)
Alkalinity (as CaCO ₃)	3050	0-20,850
Biochemical oxygen demand (BOD ₅)	5700	81-33,360
Chemical oxygen demand (COD)	8100	40-89,520
Copper (Cu)	0.5	0-9.9
Lead (Pb)	0.75	0-2.0
Zinc (Zn)	5.8	3.7-8.5
Chloride (Cl ⁻)	700	4.7-2500
Sodium (Na ⁺)	767	0-7700
Total dissolved solids (TDS)	8955	584-44,900
Ammoniacal nitrogen (NH ₄ ⁺)	218	0-1106
Total phosphate (PO ₄ ³⁺)	10	0-30
Iron (Fe)	94	0-2820
Manganese (Mn)	0.22	0.05-125
pH (pH units)	5.8	3.7-8.5

Source: After U.S. Environmental Protection Agency (1977b).

Leachate Management and Minimization of Ground-Water Impact

There are several general methods for managing leachate: natural attenuation by soils, prevention of leachate formation, collection and treatment, pretreatment to reduce volume and solubility, and detoxification of hazardous wastes prior to landfilling. Leachate undergoes natural attenuation by various chemical, physical, and biological processes as it migrates through soil. Whether natural attenuation will be adequate to prevent ground-water pollution should be evaluated for each site. The generation of leachate can be minimized by restricting water from infiltrating the landfill. This is accomplished by providing appropriate surface drainage and/or placing an impermeable liner over the daily accumulation of refuse. Another method of controlling leachate is to collect it at the bottom of the landfill and treat it before discharging it into surface water or the land. In most cases collected leachate must be pretreated by an anaerobic biological treatment unit (anaerobic filter or suspended growth reactors) before it is discharged into sewers. The high BOD strength of the leachate makes it difficult to treat in conventional aerobic treatment units, and without pretreatment conventional biological treatment plants could become overloaded.

Figure 7.13 shows a well barrier, which prevents leachate from reaching the ground water. Newly constructed landfill requires a clay and geomem-

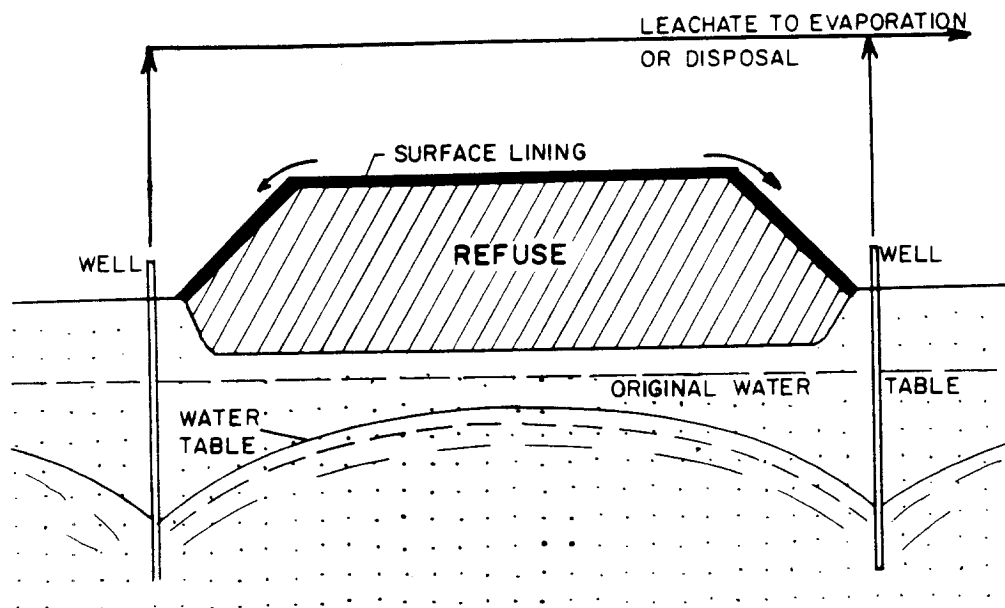


FIGURE 7.13. Control of ground-water pollution from landfills by creating a ground-water depression barrier.

brane lining and a suitable low permeability (clay) substratum to virtually eliminate potential seepage of leachate into ground water.

Most regulations recommend or require that landfill sites are developed on uplands rather than floodplains and in low-permeability soils. Geologically such sites are difficult to find; however, the sites must also be socially and politically acceptable. If the landfill receives hazardous (toxic) waste, a TCLP extraction toxicity analysis must be performed. The solid waste disposal site is then considered toxic (hazardous) if following the application of the TCLP test the leachate (extract) from a representative sample of waste contains any of the 52 listed toxic compounds in concentrations that exceed the limit (see the Appendix for the numerical values of the TCLP criteria). The method and limits have been published by the U.S. EPA (1986).

When leachate from a landfill reaches the saturated ground-water zone it moves as a plume that spreads in the direction of the flowing ground water. This process has been described in the preceding sections of this chapter and shown in Figure 7.5. As the plume slowly moves, the concentrations of contaminants decrease owing to adsorption and retardation, hydrodynamical dispersion, and biochemical degradation.

Tracing contaminated plumes from land disposal sites requires testing the levels of certain water quality parameters. Although the most appropriate parameters may vary somewhat, depending on the types of solid wastes deposited in the landfill, certain parameters have been found to be generally suitable. Key indicators of the presence of leachate that have been suggested by the U.S. EPA (1977c) are specific conductance, pH, temperature, chloride ion, color, turbidity, and COD (TOC).

GROUND-WATER QUALITY MODELS

There are two types of ground-water quality models in present use (Bachmat et al., 1978):

1. Predictive models, which simulate the behavior of the ground-water system and its response to boundary inputs.
2. Resource management models, which integrate prediction with explicit management decision procedures.

Predictive Models

The predictive models are primarily numerical even though electric analogs or even physical modeling can be used for simulation of ground-water quality systems. The numerical models are either *deterministic*

analytical and computer-simulation models or *stochastic* (random walk, etc.) *models*. The predictive models represent the vast majority of models developed for ground-water-related problems (Bachmat et al., 1978; Anderson 1979; Canter, Knox, and Fairchild, 1987).

The ground-water movement models, which are similar to models of surface hydrology and water quality, can be either distributed parameter (two- or three-dimensional) models or lumped-parameter models. A distributed parameter is one in which variables are determined at many discrete locations throughout the system dictated by the breakdown of the system into small uniform segments. In the lumped-parameter models aquifer is treated as one uniform element, and only spatially averaged values are considered.

Analytical models are developed by considering highly simplified conditions or by using simplifying assumptions to obtain a solution of the governing differential equations (hydraulic continuity and motion, and mass continuity). The most common analytical solutions are those assuming steady-state and uniform systems. Some examples were presented in the preceding sections of this chapter or in the literature (Bear 1979; Anderson, 1979). Such models typically use the lumped-parameter concept.

The Gelhar–Wilson (1974) model is presented here as an example of a simpler analytical lumped-parameter model of water quality in an aquifer–stream system. The authors of the model pointed out that when long-term basinwide changes in ground-water quality are desired, spatial variations become less important than temporal variations. The fluctuation of the water table is represented mathematically by the following equation:

$$p \frac{dh}{dt} = -q + \varepsilon + q_r - q_p \quad (7.18)$$

where

h = average thickness of the saturated zone

p = average effective porosity

ε = natural discharge rate

q = natural outflow from the aquifer

q_r = artificial recharge/unit area

q_p = pumping rate/unit area

t = time

It can be demonstrated that $q = a(h - h_0)$, where h_0 is the elevation of the river and

$$a = 3T/L^2$$

where

$T = h_0K$ = transmissivity of the aquifer

K = hydraulic conductivity

L = length of the aquifer

The change in concentration of a nonadsorbable pollutant is represented by an equation of the form

$$ph \frac{dc}{dt} + (\varepsilon + q_r + \alpha ph)c = \varepsilon c_L + q_r c_r \quad (7.19)$$

where

c = concentration

c_L = concentration of the natural recharge

c_r = concentration of the artificial recharge

α = a first-order rate constant that accounts for degradation of the contaminant

Dispersion is assumed to be negligible, which is a reasonable assumption if only regional average concentrations are sought.

The hydraulic response time and the solute response are measures of the lag observed in the response of the system to a given input. Hydraulic response time (t_h) is defined as follows:

$$t_h = p/a \quad (7.20)$$

The solute response time, t_c , is then

$$t_c = ph_0/\varepsilon_0 \quad (7.21)$$

where ε_0 = initial recharge rate. In general, t_c is time dependent, but can be estimated from Equation (7.21). A representation of the model is shown on Figure 7.14.

Gelhar and Wilson (1974) based their model on the concept of a well-mixed linear reservoir. They postulated that aquifer response to a given input would be similar to the response of a well-mixed linear reservoir. They showed that the concentration of water leaving the aquifer is representative of the average concentration within the aquifer. Therefore, such a model is indeed suitable for determining the quality of ground water discharging to surface waters provided that the simulated contaminant is not attenuated by adsorption in the aquifer zone. Such contaminants may include nitrates and chlorides.

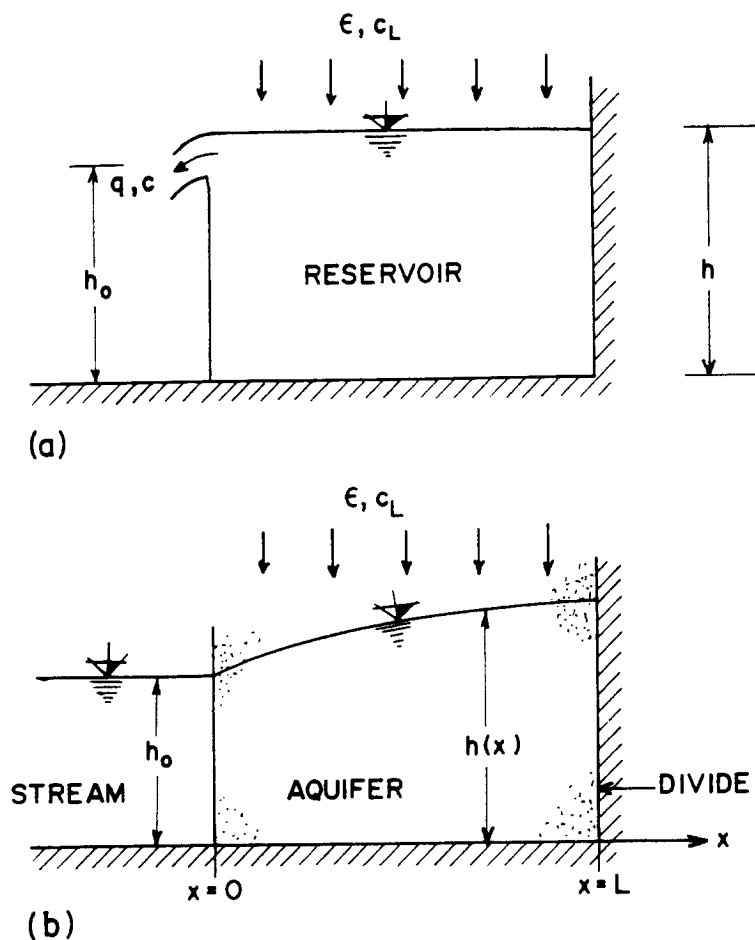


FIGURE 7.14. Representation of the Gelhar–Wilson (1974) model.

Since the Gelhar–Wilson model is based on a lumped-parameter concept, it must be calibrated. For example, historical records of chloride concentrations in wells or base flow can be used for calibration and verification (Anderson, Eisen, and Hoffer, 1978).

Distributed-Parameter Models

The distributed-parameter model differs from the simpler single linear reservoir model by assuming that the aquifer is composed from a number of linear reservoirs that are interconnected. Typically water continuity and mass balance differential equations are written for each cell and simultaneously solved. In almost all cases the equations are solved using finite-difference approximation. A review of such models is included in Canter, Knox, and Fairchild (1987), Anderson (1979), Bachmat et al. (1978), Wang and Anderson (1982), or NCASI (1985), and others.

References

- Adriano, D. C. (1986). *Trace Elements in the Terrestrial Environment*, Springer Verlag, New York.
- Anderson, M. P. (1979). Using models to simulate the movement of contaminants through groundwater flow systems, *CRC Crit. Rev. Environ. Control* 9(2):97-156.
- Anderson, M. P., C. E. Eisen, and R. N. Hoffer (1978). *Groundwater Hydrology*, Vol. 5, Menomonee River Pilot Watershed Study, International Joint Commission, Windsor, Ontario.
- Andreoli, A., N. Bartilucci, R. Forgione, and R. Reynolds (1979). Nitrogen removal in a subsurface disposal system, *J. WPCF* 51:841-834.
- Bachmat, Y., B. Andrews, D. Holtz, and S. Sebastian (1978). *Utilization of Numerical Groundwater Models for Water Resources Management*, EPA 600/8 78/012, U.S. Environmental Protection Agency, Ada, OK.
- Baestlé, L. H. (1969). Migration of radionuclides in porous media, in *Progress in Nuclear Energy. Series XII. Health Physics*, A. M. F. Duhamel, ed., Pergamon Press, New York, pp. 707-730.
- Bear, J. (1979). Groundwater quality problem (hydrodynamic dispersion), in *Hydraulics of Groundwater*, McGraw-Hill, New York, pp. 263-275.
- Bell, R. G., and J. B. Bole (1978). Elimination of fecal coliforms from soil irrigated with municipal sewage lagoon effluent, *J. Environ. Qual.* 7:193-196.
- Betson, R. P., and W. M. McMaster (1975). Nonpoint source mineral water quality model, *J. WPCF* 47:2461-2473.
- Bouwer, H. (1974). Design and operation of land treatment systems for minimum contamination of groundwater, *Groundwater* 12:140-147.
- Bouwer, H. (1978). *Groundwater Hydrology*, McGraw-Hill, New York.
- Boyce, J. S., et al. (1976). Geologic nitrogen in Pleistocene loess of Nebraska, *J. Environ. Qual.* 5:93-96.
- Brown, K. W., H. W. Wolf, K. C. Donnelly, and J. F. Slowey (1979). The movement of fecal coliforms and coliphages below septic lines, *J. Environ. Qual.* 5:121-125.
- Brown, R. L. (1975). The occurrence and removal of nitrogen in subsurface agricultural drainage from the San Joaquin Valley, California, *Water Res.* 9:529-546.
- Burwell, R. E., G. E. Schuman, K. E. Saxton, and H. G. Heineman (1976). Nitrogen in subsurface discharge from agricultural watersheds, *J. Environ. Qual.* 3:325-329.
- Canter, L. W., and R. C. Knox (1985). Groundwater pollution from septic tank systems, in *Septic Tank System Effects on Groundwater Quality*, Lewis Publ., Chelsea, MI.
- Canter, L. W., R. C. Knox, and D. M. Fairchild (1987). *Ground Water Quality Protection*, Lewis Publ., Chelsea, MI.
- Chalk, P. M., and D. R. Keeney (1971). Nitrate and ammonium content of Wisconsin limestone, *Nature* 229:42-47.
- Conservation Foundation (1985). *Groundwater—Saving the Unseen Resource*.

- Proposed Conclusions and Recommendations*, Conservation Foundation, Washington, DC.
- Davis, S. N., and R. J. M. DeWiest (1966). *Hydrogeology*, Wiley, New York.
- Domenico, P. A., and F. W. Schwartz (1990). *Physical and Chemical Hydrogeology*, Wiley, New York.
- Edworthy, K. J., W. B. Wilkinson, and C. P. Young (1978). The effect of the disposal of effluents and sewage sludge on groundwater quality in the Chalk of the United Kingdom, *Prog. Water Tech.* **10**:479–493.
- Fetter, C. W. (1988). *Applied Hydrogeology*, 2d ed., Merrill Publ. Columbus, OH.
- Freeze, R. A., and J. A. Cherry (1979). *Groundwater*, Prentice-Hall, Englewood Cliffs, NJ.
- Gallant, A. L., et al. (1989). *Regionalization as a Tool for Managing Environmental Resources*, NSI Technology Services Corporation, U.S. Environmental Protection Agency, Corvallis, OR.
- Gelhar, L. W., and J. L. Wilson (1974). Groundwater quality monitoring, *Groundwater* **12**:399–408.
- Geraghty, J. J., et al. (1973). *Water Atlas of the United States*, Water Information Center, Syosset, NY.
- Hem, J. D. (1985). Study and Interpretation of the Chemical Characteristics of Natural Water, U.S. Geological Survey Water-Supply Paper 2254, Washington, DC.
- Johansen, O. J., and D. A. Carlson (1976). Characterization of sanitary landfill leachate, *Water Res.* **10**:1129–1134.
- Klusman, R. W., and K. W. Edwards (1977). Toxic metals in groundwater of front range, Colorado, *Groundwater* **15**:160–169.
- Kreitler, C. W., and D. C. Jones (1975). Natural soil nitrate: The cause of the nitrate contamination in groundwater in Runnels County, Texas, *Groundwater* **13**:53–61.
- Lehr, J. H. (1981). A problem yes—A disaster no, *Groundwater* **19**:2–3.
- Miller, D. W., and M. R. Scalf (1974). New priorities for groundwater protection, *Groundwater* **12**:335–347.
- Moore, J. W., and S. Ramamoorthy (1984). *Heavy Metals in Natural Waters: Applied Monitoring and Impact Assessment*, Springer Verlag, New York.
- National Council of the Paper Industry for Air and Stream Improvement (1985) *A Review and Practical Guide for Analytical Groundwater Models Used to Predict Contaminant Transport in Subsurface Systems*, NCASI, New York.
- Novotny, V., K. R. Imhoff, M. Olthof, and P. A. Krenkel (1989). *Karl Imhoff's Handbook of Urban Drainage and Wastewater Disposal*. Wiley Interscience, New York.
- Ogata, A. (1970). *Theory of Dispersion in a Granular Medium*, U.S. Geological Survey Prof. Paper 411-1, Washington, DC.
- Omernik, J. M., and A. L. Gallant (1990). Defining regions for evaluating environmental resources, in *Global Resources Monitoring and Assessments: Preparing for the 21st Century*, Proceedings, International Conference and

- Workshop, American Society for Photogrammetry and Remote Sensing, Bethesda, MD.
- Peaudecerf, P., and J. P. Sauty (1978). Application of a mathematical model to the characterization of dispersion effect on groundwater quality, *Prog. Water Technol.* **10**:443–455.
- Perkins, T. K., and O. C. Johnson (1963). A review of diffusion and dispersion in porous media, *J. Soc. Petrol. Eng.* **3**:70–83.
- Reed, S. C., E. J. Middlebrooks, and R. W. Crites (1988). *Natural Systems for Waste Management and Treatment*, McGraw-Hill, New York.
- Reneau, R. B., and D. E. Petry (1976). Phosphorus distribution from septic tank effluents in coastal plain soils, *J. Environ. Qual.* **5**:34–39.
- Saffigna, P. G., and D. R. Keeney (1977). Nitrate and chloride in groundwater under irrigated agriculture in Central Wisconsin, *Groundwater* **15**:170–177.
- Sawyer, C. N., and P. L. McCarty (1978). *Chemistry for Environmental Engineering*, McGraw-Hill, New York.
- Sikora, L. J., M. G. Bent, R. B. Corey, and D. R. Keeney (1976). Septic nitrogen and phosphorus removal test system, *Groundwater* **14**:304–314.
- Todd, D. K. (1980). *Groundwater Hydrology*, 2d ed., Wiley, New York.
- Todd, D. K. (1983). *Ground-water Resources of the United States*, Premier Press, Berkeley, CA.
- U.S. Environmental Protection Agency (1977a). *Environmental Effects of Septic Tank Systems*, EPA 600/3-77/096, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1977b). *The Report to Congress—Waste Disposal Practices and Their Effects on Groundwater*, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1977c). *Procedures Manual for Groundwater Monitoring of Solid Waste Disposal Facilities*, EPA 530/SW-611, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1980). *Design Manual—Onsite Wastewater Treatment and Disposal Systems*, EPA 625/1-80-012, U.S. EPA, Cincinnati, OH.
- U.S. Environmental Protection Agency (1981). *Process Design Manual for Land Treatment of Municipal Wastewater*, EPA 625/1-81-013, U.S. EPA, Cincinnati, OH.
- U.S. Environmental Protection Agency (1984). *Process Design Manual for Land Treatment of Municipal Wastewater: Supplement on Rapid Infiltration and Overland Flow*, EPA 624/1-81-013a, U.S. EPA, Cincinnati, OH.
- U.S. Environmental Protection Agency (1986). Hazardous waste management system, *Fed. Regist.* **51**(114):21648–21693.
- Wang, H. F., and M. P. Anderson (1982). *Introduction to Groundwater Modeling—Finite Difference and Finite Element Models*, W.H. Freeman, San Francisco, CA.

8

Urban and Highway Diffuse Pollution

As a result of increased traffic, congestion, higher pollution levels, littering, reduced green space and the loss of other environmental amenities, living conditions in many urban areas in the OECD region have become worse in the past 20 years.

*Organization for Economic Co-operation and Development (1991),
The State of the Environment, Paris*

LAND-USE EFFECTS ON URBAN NONPOINT POLLUTION LOADS

In the first half of this century deterioration of water quality due to urbanization and urban sources was associated with point sources from industrial and commercial operations and with domestic sewage. However, not until 1970 was it realized that a significant portion of pollution from urban and urbanizing areas originated from nonpoint diffuse sources such as construction, washoff of dust and dirt from impervious surfaces, or sewage inputs from unsewered suburban areas.

Urban nonpoint sources have been identified as a major cause of pollution of surface-water bodies by the U.S. EPA (U.S. EPA, 1984; Athayde, Myers, and Tobin, 1986; Myers et al., 1985). In the 1988 Report to Congress (U.S. EPA, 1990a) it was stated that urban storm-water runoff is the fourth most extensive cause of the impairment of the water quality of the nation's rivers and the third most extensive source of water quality impairment of lakes. The combined sewer overflows

(CSO's) were the tenth most significant source of impairment for both types of surface-water bodies.

Overflows from both separate storm sewers and combined sewer systems are considered diffuse pollution, although in the United States their discharges are regulated. For example, a permit system is used for point sources (see Chapters 1 and 2 for legal classifications of point and nonpoint diffuse sources and effluent discharge permits).

Diffuse-pollution generation in urban areas is quite different from that in rural lands. Several factors cause the difference (Novotny and Chesters, 1981):

1. Large portions of urban areas are impervious (Fig. 8.1), resulting in much higher hydrological activity. The coefficient of runoff (defined as a ratio of runoff volume to that of rainfall) is generally directly proportional to the degree of imperviousness (Fig. 8.2). Hence, urbanization increases the volume of runoff.
2. The hydrological response of the watershed to precipitation is faster, which decreases the time of concentration. As a result the runoff



FIGURE 8.1. Highly impervious urban areas are one of the largest sources of diffuse pollution.

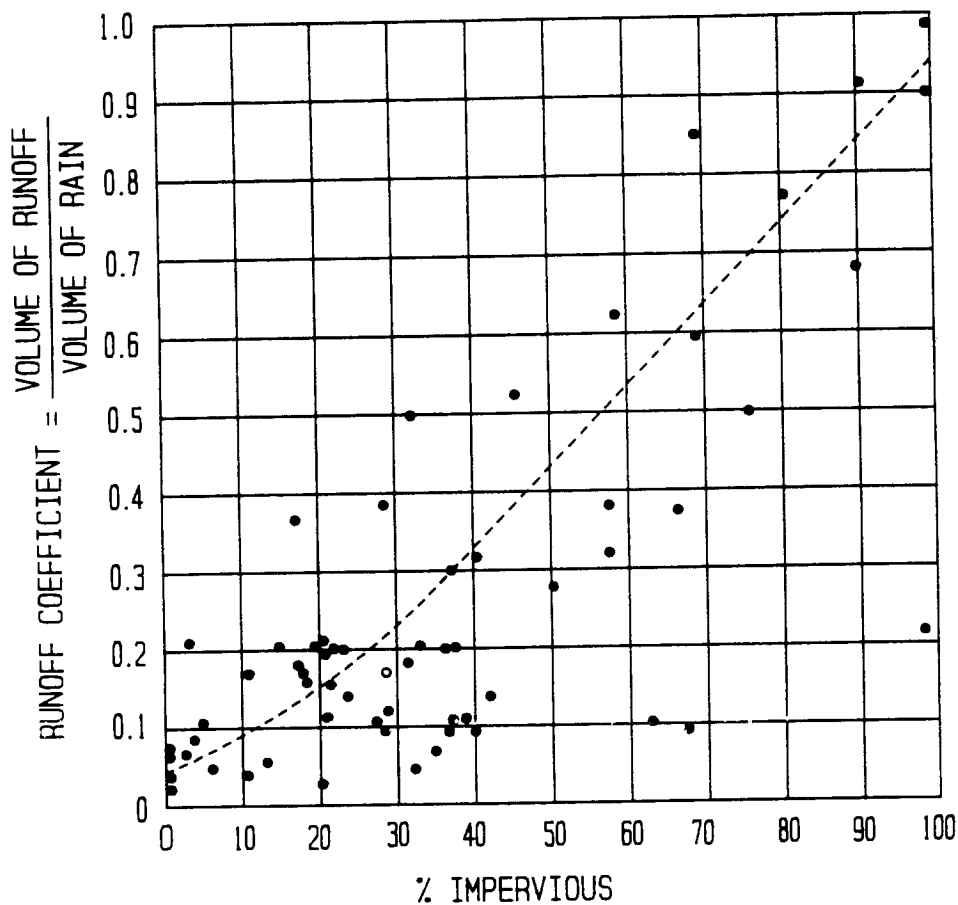


FIGURE 8.2. Relation of the statistical coefficient of runoff for urban areas to imperviousness. (From National Urban Runoff Project, U.S. EPA, 1983.)

- peaks are increased (typically two to five times) over those for the predevelopment period. This may greatly increase the flooding potential, hence, many communities have enacted strict runoff control ordinances (for example, metropolitan Chicago).
3. Urbanization commonly decreases ground-water levels. Ground-water levels are lowered by the installation of sewers and consequent sewer infiltration-inflow inputs, and by drainage below the ground levels of buildings. As a result, base flow of many urban streams may decrease to the point where some smaller headwater urban streams may become ephemeral or the flow in them may consist of predominantly undiluted sewage and wastewater effluents (effluent-dominated streams).
 4. Except for construction sites, most pervious land surfaces in residential areas and in urban areas east of the Mississippi River are protected by lawns and other vegetation and, consequently, erosion is reduced. However, increased volume and peak flow due to urbanization causes

the velocities in the streams to become faster, resulting in increased stream-bank erosion. Excessive fertilizer and pesticide applications onto urban and suburban lawns also represents a water quality problem. In the urban centers located in the arid regions of the world, including southwestern portions of the United States, erosion of pervious lands can be quite intensive, especially when water use restrictions on lawn irrigation are imposed. Rainfall, although rare, may have devastating erosion potential.

5. Over a longer period of time (for example, a year), in areas with storm sewers, all of the pollution deposited on impervious surfaces that has not been removed by street cleaning, wind, or decay will eventually end up in surface runoff. Thus the street solids accumulation, as well as accumulation on other impervious urban surfaces represents a limited source of predominantly particulate pollutants. On the other hand, soil is an infinite pool of sediments and potential pollutants in nonurban and suburban areas and in construction zones. The removal of sediments and pollutants from pervious lands into runoff depends on the energy of the rainfall (in addition to other factors, such as the soil, slope, and vegetation cover characteristics described in Chapter 5). The contributions of pollutants washed off from impervious surfaces and additional loads of overfertilized and contaminated soils change the character and the type of pollution from that of the predevelopment period.
6. The frequency of pollution-carrying runoff events is greatly increased in developed watersheds with higher imperviousness. On the other hand, nonpoint pollution loads from pervious lands (crop lands, woodlands, urban lawns, and parks) occur only during very large storms or, to a lesser degree, during frozen ground conditions in the spring.
7. Polluted runoff from impervious urban surfaces is generated during rainfall that exceeds a certain minimal threshold value of depression storage, which is about 1 to 2 mm in areas drained by separate storm sewers. The threshold value for overflows from combined sewers is somewhat greater due to the fact that a portion of storm runoff is diverted to the treatment plant. In rural areas or areas with large pervious surfaces, polluting surface runoff events are only generated during large hydrologically rare storms and snowmelt.

Urbanization increases both pollution loadings and flooding; consequently, similar practices may be used to remedy both problems.

Comprehensive reviews of sources of pollution of urban runoff and combined sewer overflows were presented by Lager et al. (1977), Ellis (1986), and by Waller and Hart (1986). Ellis compiled information on

both North America and European cases. The edited book by Torno, Marsalek, and Desbordes (1986), which contains the latter two papers, is a comprehensive treatise dealing with the urban runoff problem. Also proceedings of the IAHR-IAWPRC conferences on urban runoff management (Yen, 1982; Balmér, Malmqvist, and Sjöberg, 1984; Gujer and Krejci, 1987; Iwasa and Sueishi, 1990) contain numerous articles dealing with the quality of urban runoff and its characterization.

The pollutant loads from urban areas are strongly affected by drainage. The smallest pollutant loadings are typical for suburban areas with so-called natural surface drainage and sanitary sewers. The highest pollutant loadings are emitted from highly impervious, densely populated (or heavily used) urban centers with separate or combined sewers. As was pointed out in Chapter 1, and will be further elaborated in this chapter, the total pollution impact of overflows from areas served by combined sewers (without CSO abatement) and separate storm sewers may be about the same for many conventional pollutants. This is due to the fact that a substantial portion of storm water in combined sewer systems does not overflow and is conveyed to treatment.

URBAN LAND USE AND MAGNITUDE OF DIFFUSE POLLUTION

As pointed out in Chapter 1, typical urban land uses include

- Residential land (low, medium, and high density)
- Commercial land use
- Industrial land use
- Other developed (large parking areas, sport complexes) lands
- Open lands (parks, golf courses, idle urban lands)
- Transportation (airport, railroad, vehicular traffic)

Since this categorization is insufficient to express the impact of land-use activities on the magnitude of pollution loads emitted from these lands, Marsalek (1978) found it more appropriate to divide the land producing nonpoint pollution into more specific categories, such as:

- *Land-Use Group I: Low Pollution Loads.* This category includes low- and medium-density residential land uses (<125 people/ha) and limited- nuisance industrial activities (wholesale, warehouses).
- *Land-Use Group II: Intermediate Pollution Loads.* Typical land uses in this category include high-density residential (>125 people/ha) and commercial land use.

- *Land-Use Group III: Highest Pollution Loads.* Typical land uses include medium- and high-intensity industrial uses.
- *Land-Use Category IV: Lowest Pollution Potential.* Typical land uses include parks and playgrounds. In many cases pollution loads from these lands are negligible.

The land-use impact applies only to unit loadings of pollutants. A statistical analysis of data collected during the pilot studies of the Nationwide Urban Runoff Program (NURP), sponsored by the U.S. Environmental Protection Agency (1983), found no significant statistical difference between the mean concentrations of pollutants in urban runoff among the typical urban land uses. This finding is elaborated further in the section titled "Nationwide (NURP) Characterization of Urban Runoff Quality."

Sources and Magnitude of Urban Diffuse Pollution

It is not the land itself or land use per se that causes pollution. Pollution is caused by various boundary inputs and polluting processes and activities that occur on the land. The inputs and processes that are the cause of urban diffuse pollution are:

- Pollution contained in precipitation
- Erosion of pervious lands
- Accumulation of dry atmospheric deposits (dust) and street dirt and subsequent washoff from impervious surfaces. The sources of the accumulate pollutants are:

Dry atmospheric deposition

Street refuse accumulation, including litter, street dust and dirt, and organic residues from vegetation and animal population

Traffic emissions

- Solids accumulation and growth in sewers
- Leaching of pollutants from septic systems and other sources, such as landfills, onto surfaces and into ground water and subsequently into storm drainage
- Application, storage, and washoff of deicing and other chemicals
- Application of pesticides and fertilizers onto grassed urban lands
- Discharge of pollutants, such as car oil, detergents, and other household and commercial solvents and chemicals, into the drainage systems
- Cross-connections of sewage and industrial wastes from sanitary sewers, failing septic tanks, and other sources, into storm sewers

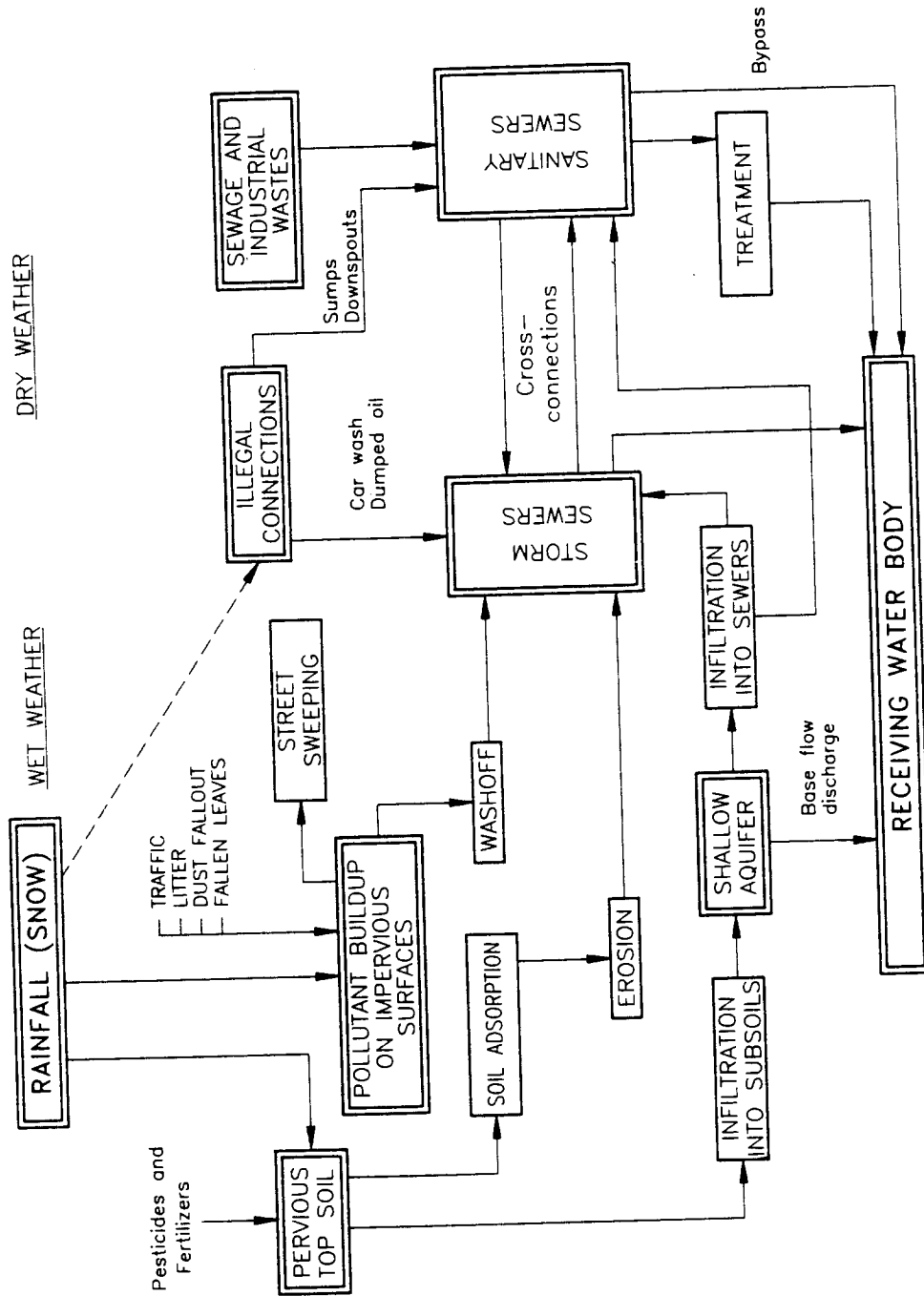


FIGURE 8.3. Schematic diagram of sources of pollution from urban areas.

Figure 8.3 schematically shows the sources of diffuse pollution and the process of accumulation.

Unit Loads of Pollutants

Definition

As first defined in Chapter 1, a unit load is a simple value or function (sometimes termed *export coefficient*) expressing pollutant generation per unit area and unit time for each land use or averaged over the entire contributing basin. The units are usually expressed in kilograms or tonnes per hectare per year or season. In urban areas the unit loads are commonly expressed in mass per unit length of curb ($\text{g/m} = \text{kg/km}$ or lb/ft).

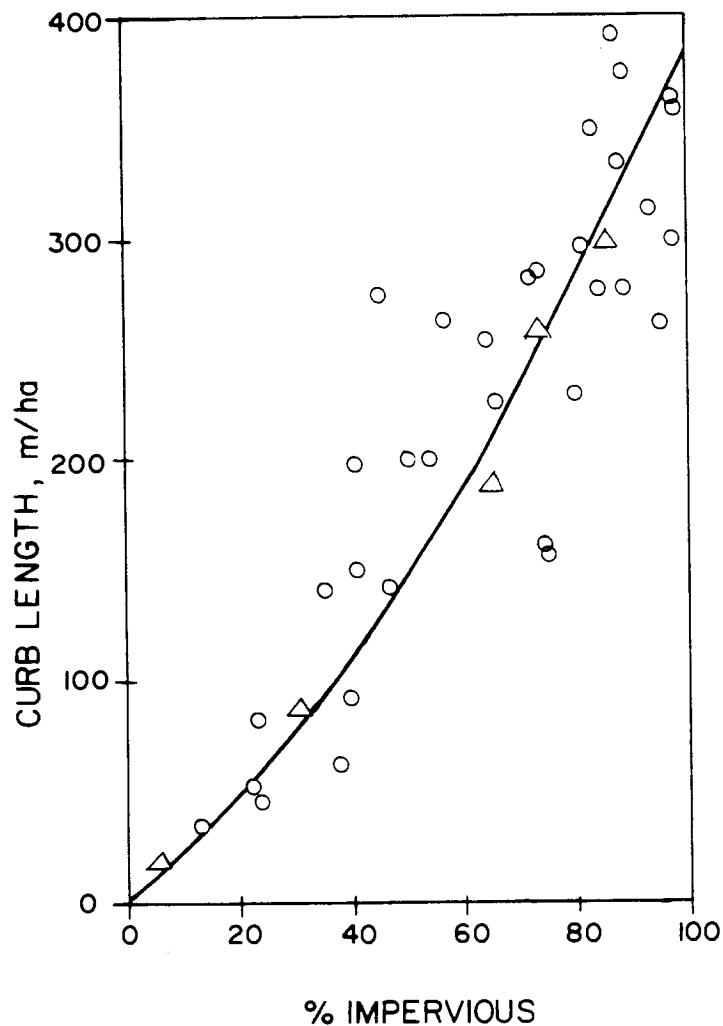


FIGURE 8.4. Relationship of curb length density to total imperviousness of urban areas.

A strong correlation exists between the curb density and the percent of imperviousness of residential areas, as shown on Figure 8.4. The American Public Works Association (1969) developed a regression formula between the curb length of urban areas and population density based on analysis of many American cities. The resulting equation (converted to metric units) was

$$CL = 311.67 - 266.07 \times 0.839^{(2.48 PD)} \quad (8.1)$$

where

CL = curb length (m/ha)

PD = population density (person/ha)

As pointed out by Novotny and Chesters (1981), unit loads established by measurements or by simulation are variable, often expressing averages from a wide range of measured values. The variations are not only caused by randomness of the process but may also include some cyclic (periodic) and systematic factors and trends, such as seasonal variations, geographical differences, meteorological and land cover factors, soil characteristics, and reliability of measured data due to insufficient frequency of measurements and measurement errors. However, the NURP study of the quality of urban runoff was unable to statistically identify a nationwide effect of any systematic factors on the unit loads except imperviousness, which affects the runoff volume and, consequently, the unit loads.

The unit loads can also be estimated by modeling. The same reservation applies to simulated unit loads; however, some variability factors are usually included in the models and are respected.

Measured Unit Loads

A number of studies provided measured unit loads from various types of urban lands. Several studies compiled the measured data into ranges and averages that have been subsequently related to the type of land use.

The first comprehensive and now classic study of urban pollution loadings by storm water and combined sewer overflows was conducted by the American Public Works Association (1969). This study identified and quantified sources of pollution by urban runoff, including air pollution, street refuse, catch basins, and sewer systems themselves. The data analyzed by the study were mostly from Chicago, Illinois.

Sartor and Boyd (1972) conducted a similar study and analyzed data from several U.S. cities. They noted that the pollutant accumulation in curb storage is not constant, but exhibits a decreasing rate of increase, that is, there is a certain limiting value of pollutant accumulation that is

related to land use. The total accumulation of pollutants converted to grams/meter of curb length are presented in the following list (after Sartar and Boyd (1972)):

<i>Land Use</i>	<i>Curb Loading (g/m)</i>
Residential	339
Industrial	790
Commercial	82
Overall	395

Ellis (1986) presented values and ranges of accumulation of street-surface pollutants (Table 8.1). It can be seen that some 20% to 40% of the accumulated material is organic; however, it is not easily biodegradable, being derived from leaf and wood litter, and rubber and bituminous road-surface materials. The high bacterial values recorded in the street-surface solids can largely be ascribed to animal feces. The high metal content of highway solids reflects traffic emissions.

The most comprehensive compilation of unit loads based on measured values to date is the one prepared by the Midwest Research Institute (McElroy et al., 1976). The manual contains estimates and estimation procedures for unit loads of sediment from erosion and impervious urban surfaces, nutrients and organic matter, pesticides, heavy metals, terrestrial disposal, and background emissions of pollutants. For developed urban areas the unit load is related to the curb density factor as

$$Y = CD * L \quad (8.2)$$

TABLE 8.1 Solids Accumulation and Associated Pollutant Concentrations in Urban Areas

Land Use	Residential		Commercial	Light	
	Low Density	High Density		Industrial	Highways
Solids accumulation (g/curb m)	10-182	30-210	13-180	80-288	13-1100
Pollutant					
BOD ₅	5260	3370	7190	2920	2300-10,000
concentration					
COD	39,300-40,000	40,000-42,000	39,000-61,730	25,100	53,650-80,000
(µg/g)					
Tot.N	460-480	530-610	410-420	430	223-1600
Pb	1570	1980	2330	1390	450-2346
Cd	3.2	2.7	2.9	3.6	2.1-10.2
Fecal Coliforms (MPN/g)	60,570-82,500	25,621-31,800	36,900	30,700	18,768-38,000

Source: After Ellis (1986).

where

Y = loading of pollutant in kg/ha per storm event

CD = curb density in km/ha

L = measured or estimated pollutant loading—rate curb load of the pollutant in g/m

The manual by McElroy et al. provides loadings of sediments for typical urban areas located in four different geographical locations in the United States. To obtain loads of pollutants associated with sediments, the equation is multiplied by a given concentration of the pollutant on the sediment. These concentrations have sometimes been called *potency factors*.

Marsalek (1978) compiled the data measured by PLUARG (Pollution by Land Use Activities—Reference Group of the International Joint

TABLE 8.2 Unit Loads of Pollutants from PLUARG Studies

Constituent	Land Use Category I	Land Use Category II	Land Use Category III	Land Use Category IV
Storm Sewers				
BOD	34	90	34	1.12
N	9	11.2	7.8	0.22
P	1.6	3.4	2.2	0.04
SS	390	360	672	11.2
Cd	0.013	0.016	0.024	0.002
Cr	0.026	0.028	0.044	0.003
Cu	0.045	0.049	0.077	0.007
Hg	0.038	0.043	0.065	0.006
Ni	0.029	0.032	0.030	0.004
Pb	0.157	0.174	0.269	0.022
Zn	0.570	0.630	0.980	0.081
Combined Sewers				
BOD	134.0	293	112	1.6
N	31.5	36.5	34.5	1.1
P	10.2	11.6	10.9	0.34
SS	773	672	740	11.2
Cd	0.016	0.017	0.027	0.002
Cr	0.028	0.031	0.048	0.003
Cu	0.064	0.071	0.109	0.009
Hg	0.043	0.047	0.073	0.006
Ni	0.034	0.037	0.057	0.004
Pb	0.162	0.180	0.277	0.022
Zn	0.640	0.703	1.088	0.090

Source: Based on data from Marsalek (1978).

Note: In kg/ha-year.

Commission) and other estimates (such as the APWA study mentioned previously and measured data from the Canadian Great Lakes) and related them to the four land-use categories defined on the preceding pages. Table 8.2 provides some of the unit loads by Marsalek.

Beaulac and Reckhow (1982) compiled, measured, and published unit loads from various nonurban land uses and from general urban land-use types. The ranges of unit loads of phosphorus for urban lands were 0.5 to 6.25 kg/ha-year (average 1 kg/ha-year), and those for nitrogen were 1 to 38.5 kg/ha-year (average 5 kg/ha-year), respectively.

Similarly, Uttormark, Chapin, and Green (1974) compiled ranges of nutrient export from urban watershed that were 2 to 9 kg/ha-year for nitrogen and 1.1 and 1.2 kg/ha-year (only two values reported) for phos-

TABLE 8.3 Unit Loads of Pollutant from PLUARG Experimental Watersheds

Pollutant	General Urban	Residential	Commercial	Industrial	Developing Urban
Suspended solids	200-4800	620-2300	50-830	450-1700	27,500
Total phosphorus	0.3-4.8	0.4-1.3	0.1-0.9	0.9-4.1	23
Total nitrogen	0.2-18	5-7.3	1.9-11	1.9-14	63
Lead	0.14-0.5	0.06	0.17-1.1	2.2-7	3
Copper	0.02-0.21	0.03	0.07-0.13	0.29-1.3	—
Zinc	0.3-1.0	0.02	0.25-0.43	3.5-12.0	—
Chloride	130-750	1050	10-150	75-160	—

Source: After Sonzogni et al. (1980). (Copyright © 1980 by the American Chemical Society, reprinted by permission.)

Note: In kg/ha-year.

TABLE 8.4 Unit Loads of Pollutants from New Jersey Residential Watersheds

	Unit Loads in kg/ha-year	
	Single-Family Housing	Multiple-Family Housing
BOD ⁵	16.7	73.6
Total P	1.2	8.0
Lead	2.3	1.0
Zinc	1.3	2.0
Copper	0.4	0.5
Nickel	0.3	0.12
Chromium	0.11	0.09

Source: After Whipple et al. (1978).

TABLE 8.5 Unit Loads of Pollutants from Homogenous Urban Watersheds Located in Milwaukee, Wisconsin (excluding winter months December to March)

Land Use	Suspended Solids	Total Phosphorus	Total Lead
Freeways	979	1.04	4.96
Industrial	957	1.49	2.70
Commercial	957	1.49	2.70
Parking lots	453	0.78	0.96
High-density residential	487	1.12	0.90
Medium-density residential	216	0.58	0.24
Low-density residential	11	0.04	0.01
Parks	3	0.03	0.006

Source: After Bannerman et al. (1984).

Note: In kg/ha-year.

phorus, respectively. These ranges were consistent with Beaulac and Reckhow's compilation of a large data base. Sonzogni et al. (1980) summarized the PLUARG watershed data and provided the unit loads of pollutants that are included in Table 8.3. Whipple, Hunter, and Yu (1978) measured the unit loads shown in Table 8.4 in residential watersheds located in New Jersey.

The National Urban Runoff Project (NURP) research conducted in Milwaukee, Wisconsin (Bannerman et al., 1984) measured pollutant loads from very small homogenous urban watersheds. The study also attempted to determine the effect of street sweeping on pollutant loads, which was found to be insignificant (see Chapter 10). The measured pollutant loads are given in Table 8.5.

INDIVIDUAL SOURCES OF POLLUTION

At the beginning of the preceding section on unit loads it was emphasized that it is not the land that produces pollutant loadings. Pollution is caused by boundary pollution inputs and by polluting land uses and land-use activities by man taking place on the land. A number of publications describe individual sources of pollution by urban runoff (Lager et al., 1977; Novotny and Chesters, 1981; Ellis, 1986; James and Boregowda, 1986; U.S. EPA, 1984).

Atmospheric Deposition

Atmospheric deposition of pollutants is generally divided into wet and dry or bulk (combined) surface loading. Atmospheric deposition of pol-

lutants is a boundary input caused by local or distant air-pollution emission sources. Both industrial (urban and transportation) and agricultural activities may contribute to the pollution content of atmospheric deposits. As pointed out in Chapter 4, rain droplets and snowflakes absorb pollutants from the atmosphere, including acid-forming components.

As shown in Chapter 4, in most larger cities the deposition rate of atmospheric particulates in wet and dry fallout range from 7 tonnes/km²-month (= gm/m²-month) to more than 30 tonnes/km²-month. As expected, higher deposition rates occur in congested downtown and industrial zones, and lower rates are typical for residential and other low-density suburban zones.

Halverson et al. (1982) measured the pollutant content of precipitation in a nonindustrial urban area in central Pennsylvania (population about 100,000). The loads presented in Table 8.6 were divided into wet deposition (by rainfalls) and bulk deposition (wet + dry deposition). Similarly, Ng (1987) studied the relationship between rainfall chemistry and the

TABLE 8.6 Atmospheric Deposition in an Urban Area in Pennsylvania

Constituent	Wet Deposition	Bulk Deposition
Total N	0.13	0.20
Total P	0.006	0.006
Lead	T	T
Cadmium	T	T
Copper	T	T
Zinc	T	T

Source: After Halverson et al. (1982).

Note: In kg/ha-year.

TABLE 8.7 Atmospheric Deposition in an Urban Area in Ontario

Constituent	Wet Deposition	Storm Runoff
Total N	0.06	0.06
Total P	0.002	0.005
Lead	0.001	0.004
Cadmium	0.0003	0.0005
Copper	0.002	0.001
Zinc	0.004	0.012

Source: After Ng (1987).

Note: In kg/ha-year.

water quality of urban runoff in an urban catchment with prevailing industrial land use located in Burlington, Ontario, Canada. Rainfall was found to be a significant source of nitrogen (nitrite, nitrate, and ammoniacal N), copper, and nickel. For other measured pollutants (phosphorus, cadmium, chromium, lead, and zinc) rainfall contributions were insignificant. Table 8.7 presents the loads measured by Ng.

Extensive information on concentrations and wet and dry depositions of pollutants related to land use are contained in the report by Pitt and Barron (1989). Novotny and Kincaid (1982), and Randall et al. (1982), reported chemical composition of precipitation for various urban and suburban land-use types.

In residential areas, the fugitive dust (dry deposition) mostly originates from surrounding soils, construction sites, refuse disposal sites, and from biological sources (pollen, spores, and other organic residues). The average composition of dustfall and the deposition rate observed in Milwaukee, Wisconsin, experimental watersheds in the early 1980s is given in the following lists (after Bannerman, 1984):

Average Wet Deposition Characteristics

Suspended solids	4.0 mg/l
Organics (VSS)	1.0 mg/l
Organics (COD)	7.0 mg/l
Total nitrogen	0.9 mg/l
Total phosphorous	0.015 mg/l
Total lead	0.012 mg/l
Ph	4.2

<i>Dry Deposition</i>	<i>December–May</i>	<i>June–November</i>
Atmospheric fallout rate (solids), kg/ha-day	0.6	0.43
COD, kg/ha-day	0.23	0.16
Total phosphorus, g/ha-day	0.55	0.47
Total lead, g/ha-day	0.52	0.51
Sulphate, g/ha-day	25.0	20.0

Pitt (1979), after an extensive study in the San Francisco Bay area, concluded that a majority of deposited street dust particles are related to local geological conditions, with added fractions from motor vehicle emissions and road wear. Most of the street refuse particles originate from local erosion of soils and are transported by air.

A study of dry atmospheric deposition in an industrial valley located in Milwaukee, Wisconsin (Anon., 1981) showed that fugitive dust emissions and transport are also the major sources of dry deposition from the atmosphere. The origin of dustfall (particles less than $60\mu\text{m}$ are considered dust in most urban pollution studies) in the central city— downtown locations are mostly from unpaved roads and parking lots, unpaved railroad rights-of-way, uncovered material storage sites, construction and demolition sites, dirty paved roads, urban refuse (garbage) disposal landfill operations, and industrial park emissions.

Lead, which is associated with the combustion of leaded gasoline (in the United States the use of leaded gasoline was phased out during the 1980s), is transported with atmospheric particulate matter. The influence of traffic using leaded gasoline on lead concentrations is profound and a significant reduction has been achieved after switching to unleaded gasoline. Cadmium, strontium, zinc, nickel, and many organic hazardous chemicals are also transported with atmospheric aerosols, and can be found in the atmospheric fallout.

Street Refuse Deposition

Particles that are larger in size than dust ($>60\mu\text{m}$) are considered as street refuse or street dirt. In the NURP studies, these deposits were divided into median-sized deposits (street-dirt particle size ranges $60\mu\text{m}$ to 2mm) and litter ($>2\text{mm}$). Litter deposits contain items such as cans, broken glass, bottles, pull tabs, papers, building materials, plastic, garbage, parts of vegetation, dead animals and insects, animal excreta, and the like. The sources of street dirt are numerous and often very hard to control. Although originally most of the street refuse deposition occurs in larger sizes (greater than 3mm), it is reasonable to assume that a part of dust originates from the mechanical breakdown of larger litter particles.

Vegetation Inputs

In residential areas, fallen leaves and vegetation residues including grass clippings dominate street refuse composition during the fall season (Fig. 8.5). During defoliation in the fall a mature tree can produce from 15 to 25 kg of organic leaf residue (dry weight) that contains significant amounts of nutrients (Heaney and Huber, 1973). During the growing season trees can enrich the throughfall (rainwater that penetrates the canopy) with nutrients and organics (Halverson, DeWalle, and Sharpe, 1984). A typical value of foliage and leaf fallout for a forested area in Minnesota with about 420 trees/ha is about 3.8 tons/ha-year or about



FIGURE 8.5. Leaf fallout in urban areas and its washoff into storm sewers is a source of large urban loads of biodegradable organics. (Photo: V. Novotny.)

9 kg/tree. Of the yearly values, about 65% of the fallout occurs during the fall. The fallen leaves are about 90% organic and contain about 0.04% to 0.28% of phosphorus.

Only a portion of the vegetation residue that accumulates on impervious surfaces is a pollution threat to surface waters. Vegetation fallout on soils becomes an integral part of the soil composition, and in most cases may even improve soil permeability and erosion resistance.

Traffic

Motor vehicular traffic is directly responsible for the deposition of substantial amounts of pollutants, including toxic hydrocarbons, metals, and asbestos, in addition to oils. The particulates contributed by traffic are primarily inorganic. Most of the traffic exhaust pipe emissions are dust size ($<60\ \mu\text{m}$). However, vehicle exhaust pipes are not the only source of traffic-related pollution. Tire wear, solids carried on tires and vehicle bodies, wear and break down of parts, and loss of lubrication fluids add to the pollution inputs contributed by traffic. The subsequent section on

sources of toxics presents various sources of toxic materials in urban runoff, including those from traffic.

Only a small portion (<5%) of traffic-related pollution can be directly traced to vehicle emissions. However, the pollutants that motor vehicles emit are among the most important because of their potential toxicity.

In addition to traffic density and vehicle pollutant emissions, pavement conditions and compactions are significant in determining the traffic impact on pollutant loads. Streets paved entirely with asphalt have loadings of about 80% higher than all concrete streets (Sartor and Boyd, 1972). Streets whose conditions were rated as "fair to poor" were found to have total solids loadings 2.5 times greater than those rated "good to excellent." Similar conclusions were also reported by Pitt (1979).

A study by Shaheen (1975), in the Washington, D.C., metropolitan area quantified the traffic contributions to nonpoint pollution. Table 8.8 shows the composition of traffic-related emissions by Shaheen. Shaheen's study estimated that approximately 0.7 g/axle-km of roadway solids can be directly attributed to traffic. In another EPA study, direct traffic emissions were reported to be 0.2 g/vehicle-km from vehicle exhaust and 0.125 g/vehicle-km from tire wear (U.S. EPA, 1977).

In research sponsored by the Federal Highway Administration, Strecker et al. (1987) summarized data on the quality of runoff from highways. Average concentrations of constituents in highway runoff are given in Table 8.9. Bannerman (1991) has found that the average con-

TABLE 8.8 Traffic Emissions

Pollutant	Percent of Total Solids by Weight
Volatile solids	5.1
BOD ₅	0.23
COD	5.4
Grease	0.64
Total P	0.06
TKN	0.016
Nitrate	0.008
Asbestos	3.6×10^5 fibers/g
Lead	1.2
Chromium	0.008
Copper	0.012
Nickel	0.019
Zinc	0.15
Emission rates of total solids	0.671 g/axle-km

Source: Based on data from Shaheen (1975).

TABLE 8.9 Average Concentration of Pollutants in Highway Runoff

Constituent	Mean Concentration (mg/l)	Coefficient of Variation ^a
Suspended solids	143	1.16
Total Kjeldahl N	1.8	0.97
Lead	0.53	2.01
Zinc	0.37	1.37

Source: Based on data from Strecker et al. (1987).

^aThe coefficient of variation is a ratio of standard deviation to the mean.

centration (event mean concentration—see the section titled “Statistical Quality Characteristics of Urban Runoff” for a definition) of metallic toxic pollutants can be correlated to traffic volume (Fig. 8.6).

Particle-Size Distribution of Street Refuse

Table 8.10 and Figure 8.7 indicate that most street refuse is in coarser fractions, roughly in sizes equivalent to sand and gravel. However, most

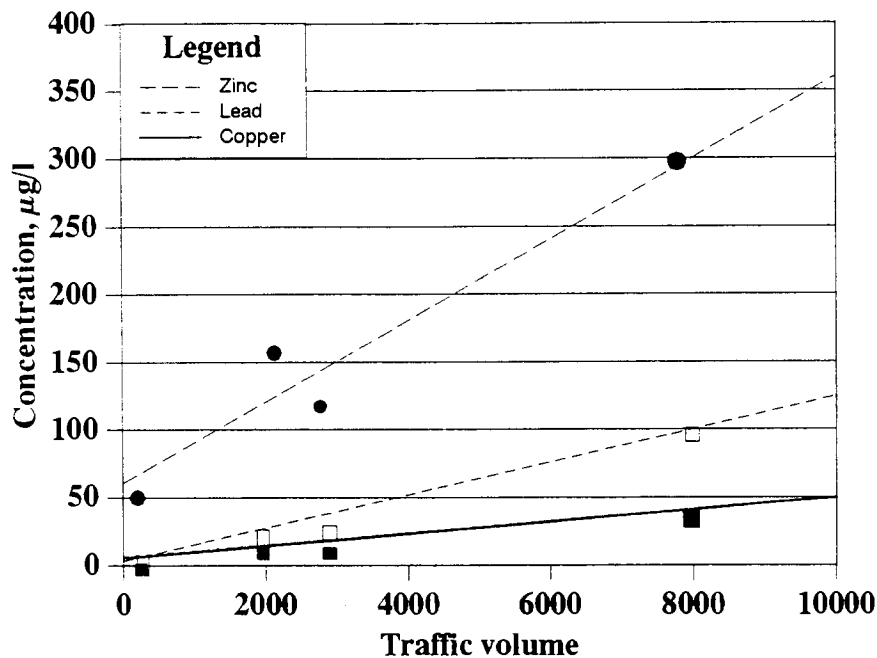


FIGURE 8.6. Relationship of even mean concentrations of some metals to traffic density. (Bannerman, 1991.)

TABLE 8.10 Percent of Street Pollutants in Various Particle-Size Ranges

Pollutant	Particle Size (μm)					
	>2000	840-2000	240-840	104-240	43-104	<43
Total solids	24.4	7.6	24.6	27.8	9.7	5.9
Volatile solids	11.0	17.4	12.0	16.1	17.9	25.6
COD	2.4	4.5	13.0	12.4	45.0	22.7
BOD ₅	7.4	21.1	15.7	15.2	17.3	24.3
Phosphates	0	0.9	6.9	6.4	29.6	56.2
All toxic metals	16.3	17.5	14.9	23.5	—	27.5
TKN	9.9	11.6	20.0	20.2	19.6	18.7
All pesticides		27.0			73.0	
PCBs		66.0			34.0	

Source: After Sartor, Boyd, and Agardy (1974).

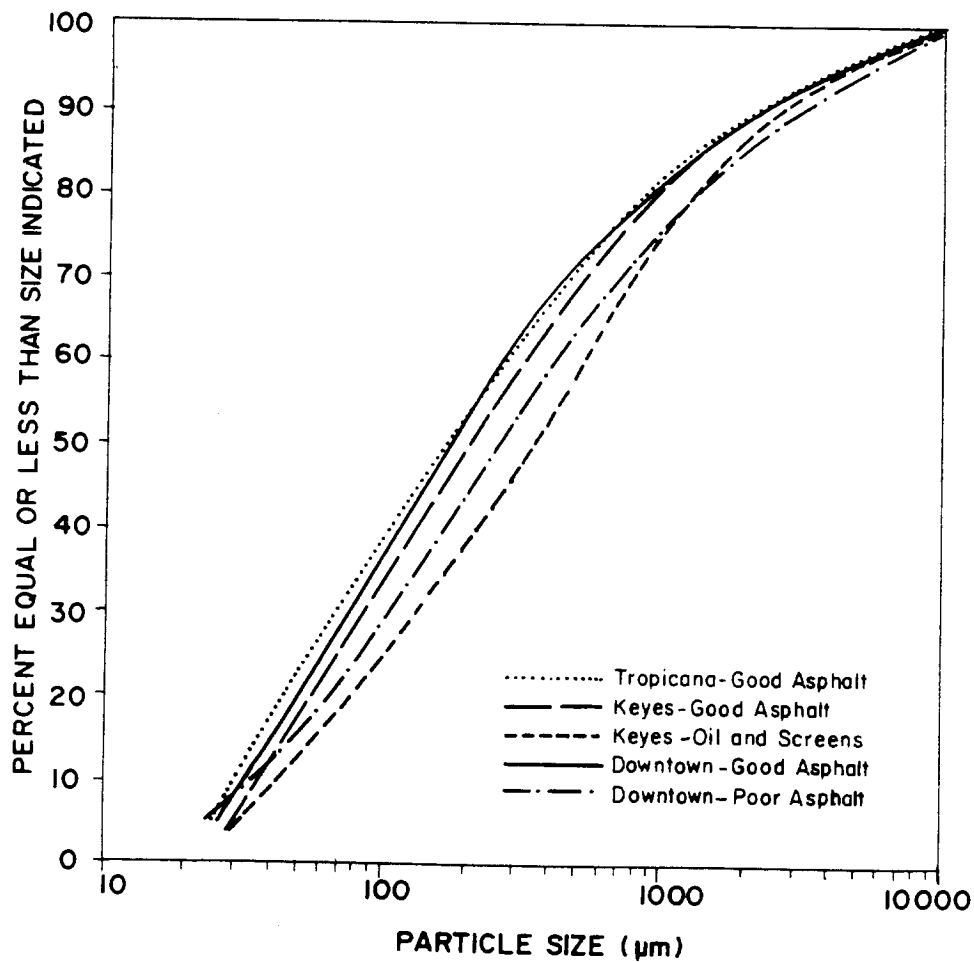


FIGURE 8.7. Particle-size distribution of street refuse in the San Francisco Bay area before sweeping. (From Pitt, 1979.)

pollutants, as they are in soils, are associated with fine fractions. For example, 6% of solids with sizes less than $43\ \mu\text{m}$ (equivalent to clay and silt fraction in soils) contain more than 50% of phosphorus.

Street cleaning practices are selective for coarser particles, while street washing and surface runoff are selective for finer particles. Chapter 10 discusses the efficiency and water quality impact of street sweeping.

Use of Deicing Chemicals

During winter in snowbelt urban areas (approximately the northern half of the United States, all mountain states of the United States, and the northern two-thirds of Europe and the high mountain areas of southern Europe), road deicing salts and sand are applied to road surfaces in addition to snow plowing and removal to provide safe and desirably bare pavement driving conditions. In the United States road deicing salts are applied at rates of 75 to 330 kg/km of highway (street) lane. As an example, in 1984, 40,000 tonnes of salts were applied in Milwaukee, Wisconsin, and during the 1981–1984 period the average seasonal application was 12 tonnes of salt per kilometer of street. Most of the applied



FIGURE 8.8. Urban diffuse pollution during the winter in snowbelt area: Milwaukee, Wisconsin. (Photo: V. Novotny.)

salt in Milwaukee ended up in three relatively small urban streams and ground-water aquifers. Figure 8.8 shows a typical street in Milwaukee after a snowfall and salting.

Salt usage varies among communities. For example, the use of salt in nearby Madison, Wisconsin, is about one-third of that of Milwaukee. The leading states in highway salt consumption in the United States are Minnesota, Michigan, New York, Ohio, and Pennsylvania. Public pressure is directing the municipal and state legislatures to adopt ordinances curtailing salt usage.

Currently, according to information provided by the Salt Institute, the road salt used in most of the northern United States is primarily sodium chloride (about 98%) with added calcium chloride and calcium sulfate. In the past, cyanide compounds and phosphate additives were present in commercial highway deicing salts to control corrosion and to minimize caking; however, due to the severe environmental implications, the use of such toxic and environmentally dangerous additives has been eliminated. Marine salts have proved to be comparable in cost to rock salt and have been used in some New England states (Field et al., 1974).

Extremely high concentrations of salt on street surfaces and in runoff (often more than 20 g/l, which is comparable to or greater than the salt content of sea water) are a major cause of vehicular corrosion and infrastructure deterioration. Most of the metal loss due to corrosion is incorporated in the snowmelt runoff. Other pollution-causing effects include damage to road surfaces and parking lot pavements. High levels of salts are also damaging to soil, vegetation, and ground water. Oberts (1986) noted that significant amounts of phosphorus, lead, and zinc can be found in the sand-salt mixture applied to streets in the Minneapolis-St. Paul area.

Erosion

Urban erosion can be divided into sheet and rill erosion of pervious surfaces or channel erosion. Surface erosion is caused by the energy of rainfall and overland flow, while channel erosion is a hydraulic phenomenon (see Chapter 5 for a discussion). Whipple and DiLouie (1981) discussed the problem of channel erosion, which in urban watersheds, has increased two to three times when compared to predevelopment conditions.

Urban pervious surfaces in humid areas are usually well protected by vegetation and yield pollutant inputs only during large, extreme storm events. In arid areas, recent severe droughts and water shortages have caused a change from urban lawns to a less water-demanding landscape called *xeriscape*, which primarily utilizes native plants and wood or stone

mulches (see Chapter 10 for a discussion of diffuse-pollution abatement in arid regions). Rare but intensive storms in arid regions may result in very large sediment losses from unprotected idle lands.

RUNOFF POLLUTION GENERATION PROCESS

Runoff from urban watersheds originates from both pervious and impervious surfaces. The impervious surfaces are almost always hydrologically active because their depression storage represents only a very small subtraction from rainfall. Thus the amount of pollutants washed off by rainfall from impervious surfaces depends on the amount of pollutants that has accumulated during the preceding dry period and on the energy of the runoff. On the other hand, the hydrological subtraction from rainfall on pervious surfaces (soils) is much larger, and usually only severe storms will yield appreciable runoff and sediment load. Although soils become hydrologically active only for larger rainfalls, they do provide an infinite pool of solids. Suspended solids (SS) concentrations in runoff from eroded soils may be quite high. The sources and processes contributing to urban diffuse pollution are shown schematically in Figure 8.3.

Accumulation on Impervious Surfaces (Buildup)

Theoretically, one can assume that deposition of pollutants from the sources described in the preceding section is randomly distributed over the entire street surface. However, wind- and traffic-induced air turbulence is constantly shifting the particles away from the road surface until they become either reentrained into the air, trapped in zones of more quiescent air, or redeposited on a pervious or hydrologically inactive surface. Because a typical curb or median barrier represents a zone of reduced air turbulence or an obstacle in the path of the shifted solids, almost all street refuse can be found within 1 meter of the curb (Fig. 8.9). On streets with a large number of parked cars, the distribution of solids over a street cross section is more spread out. Nevertheless, it is justifiable to consider the pollutant mass balance to be expressed per meter of curb (see also the definition and magnitude of urban pollutant loads discussed in the first section of this chapter).

The pollutant accumulation in near-curb storage follows a simple mass balance: the change of the mass of pollutants is equal to the sum of the inputs minus losses. The concept is depicted on Figure 8.10. The losses are by washoff by rainfall-generated surface runoff, and, during dry



FIGURE 8.9. Most street refuse remains within 1 meter of the curb.
 (Photo: V. Novotny.)

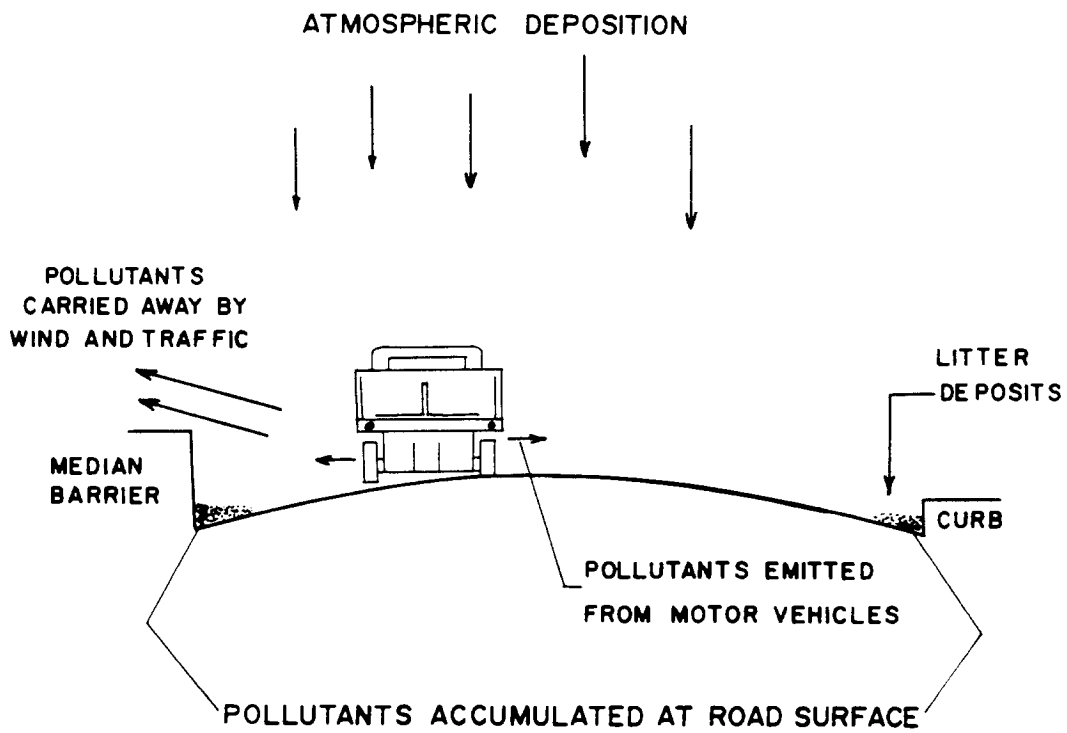


FIGURE 8.10. Schematic of pollutant accumulation on impervious street and highway surfaces.

periods by street sweeping, reentrainment of dust particles, and shifting coarser particles away from the road.

The first urban runoff quantity–quality models, such as the first version of the Stormwater Management Model, SWMM (U.S. EPA, 1971; Heaney and Huber, 1973), and STORM (Hydrologic Engineering Center, 1975) assumed a linear increase in pollutants in curb storage and no losses, whereby each day the amount of solids was increased by a constant increment. This would mean that in the absence of street sweeping, runoff pollution resulting from larger rainfalls would be directly proportional to the number of antecedent dry days. Whipple et al. (1978) criticized the linear accumulation process. Later, theoretical justifications for more complex mass balance models were published by Novotny et al. (1985), James and Boregowda (1986), Huber (1986), and others. The latter two articles contain an extensive discussion of deterministic buildup and washoff models.

The mass balance accumulation function for street solids is

$$\frac{dP}{dt} = I - \xi P \quad (8.3)$$

where

P = the amount of pollutants in curb storage

I = the sum of all inputs

t = time

ξ = a removal coefficient

Introducing this equation into urban runoff models added another variable that had to be estimated by calibration—the removal coefficient. However, it is unwise to speculate that every urban runoff study and permit site will have the extensive data base available that would be needed to calibrate all model parameters. Hence we present information on the approximate magnitude of the removal parameter, as well as the inputs onto the mass balance models.

Several studies have established that wind- and traffic-induced turbulence can dislocate dust particles that may subsequently become airborne or just be shifted away from the road. Findings in the literature on reentrainment of particles from street surfaces were summarized by Pitt (1979). Another study found that the reentrained portion of the traffic-related emissions (by weight) is an order of magnitude greater than the direct vehicle emissions because of exhaust and tire wear (U.S. EPA, 1977).

The reentrained portion of the traffic-related particulate emissions also contains relatively large particle sizes. The median particle size measured in the EPA study was about $15\ \mu\text{m}$, with approximately 22% of the particles greater than $30\ \mu\text{m}$. About 35% of resuspended particles fall out within 30 m of the street. Automobile exhaust and tire wear particulate emissions (by weight) were less than 10% of the total fugitive roadway particulate emissions.

As stated previously, fugitive particulate losses from streets are caused by a combination of wind- and traffic-induced turbulence. Cowherd et al. (1977) stated that wind becomes a factor when the wind speed exceeds a threshold of about $20\ \text{km/hr}$. In most situations, the fugitive particulate losses from the street seem to be due to automobile-induced turbulence.

An equation for the removal coefficient was developed by the author from the statistical analysis of the data published by Shaheen (1975) and NURP study data from Milwaukee. The equation is as follows (Novotny and Chesters, 1981; Novotny et al., 1985):

$$\xi = 0.0116e^{-0.08H}(TS + WS) \quad (8.4)$$

where

H = the curb height in cm

TS = traffic speed in km/hr

WS = wind speed in km/hr

The unit of ξ is day^{-1} . The coefficient of correlation for this equation is $r = 0.86$. Most of Shaheen's data were collected on roads with a very high traffic density in the Washington, D.C., area (75% of traffic counts exceeded 50,000 axles/day).

Another factor that may affect the overall magnitude of the removal coefficient is the presence of sink surfaces in the vicinity of the road. A sink surface is a hydrologically inactive surface (such as a lawn) on which the reentrained particles can settle and be eliminated from the hydrological transport. The most simple measure of this effect is the fraction of the impervious area directly connected to the drainage system (DC). Then in a model the removal coefficient can be corrected by a correction factor that equals $1 - DC$. This term represents a portion of runoff that overflows from an impervious area on an adjacent hydrologically inactive surface.

Integration of the Accumulation Function

Equation (8.3) can be integrated to

$$P(t) = \frac{I}{\xi}(1 - e^{-\xi t}) + P(0)e^{-\xi t} \quad (8.5)$$

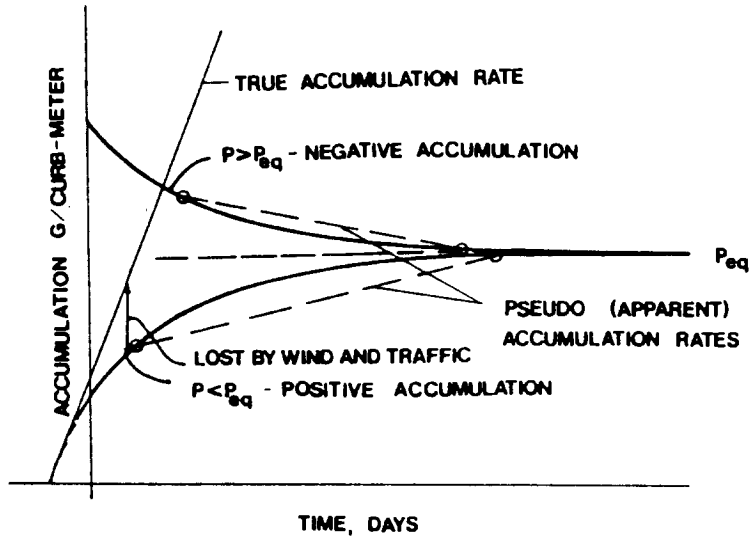


FIGURE 8.11. Street (highway) surface pollution accumulation process.

where in addition to the previously defined variables,

$P(0)$ = initial load of solids

$P(t)$ = the load after time, t

This accumulation function is shown in Figure 8.11. The true input rate, I , is a tangent to the curve at $P = 0$. A variety of apparent pseudoaccumulations can be ascertained by sampling the accumulation pollution loads and by subsequently dividing the difference in loads by the time between the samplings. It also should be noted that in such processes there is always a tendency to attain an equilibrium, whereby

$$\frac{dP}{dt} = 0$$

or

$$I = \xi P_{eq}$$

or

$$P_{eq} = \frac{I}{\xi} \tag{8.6}$$

The accumulation rate then becomes

$$\frac{dP}{dt} = -\xi(P - P_{eq}) \tag{8.7}$$

This means that the apparent accumulation rate of solids in street curb storage can be either positive or negative, depending on whether the initial pollutant load, $P(0)$, is greater or smaller than the equilibrium load P_{eq} . This is also shown on Figure 8.7. A situation where $P(0) > P_{eq}$, which would theoretically result in a negative accumulation (loss) of pollutants from curb storage, is typical for spring, when large amounts of pollutants are left in curb storage after snowmelt. Table 8.1 presented equilibrium curb loads measured by Sartor and Boyd (1972) in their studies of several U.S. cities. Equilibrium curb loadings in residential and commercial sites of the Milwaukee project measured from 1981 to 1984 were about 50% of those presented in Table 8.1.

Statistical evaluation and modeling verifications of the buildup equation were described in Novotny et al. (1985). The results using the Milwaukee NURP study sites showed that the removal coefficient in medium-density residential areas was fairly constant, attaining values around 0.2 to 0.4 day^{-1} , meaning that approximately 20% to 40% of the solids accumulated near the curb on the street surface is removed daily by wind and traffic.

The inputs of solids into curb storage from the three major sources (refuse deposition, atmospheric dry deposition, and traffic) can be combined into one loading estimate

$$\begin{aligned} I(\text{g/m-day}) &= I_r + I_a + I_{tr} \\ &= \text{LIT} + (\text{ATMFL})(\text{SW})/2 + (\text{TE})(\text{TD})(\text{RCC})/2 \quad (8.8) \end{aligned}$$

where

- I_r = litter or street refuse deposition
- I_a = dry atmospheric deposition
- I_{tr} = deposition due to traffic emissions and wear
- LIT = litter and street refuse deposition, g/m-day
- ATMFL = dry atmospheric deposition, g/m²-day (= tonnes/km²-day)
- SW = street width in meters
- TE = traffic emission rate, g/axle-m
- TP = traffic density, axles/day
- RCC = road condition factor

Example 8.1

Estimate equilibrium solids load in curb storage of a medium-density residential area. The following information is given:

Traffic count	TC = 240 cars/day
Traffic velocity	TS = 50 km/hr

Wind velocity	WS = 15 km/hr
Curb height	H = 10 cm
Atmospheric dry deposition rate	100 kg/km ² -day (mg/m ² -day)
Daily street refuse deposition	I _r = 6 g/m-day
Street width	W = 15 meters
Road Condition Index	RCC = 1.0 (average conditions)

Solution Express street surface load per 1 meter of curb length.

Atmospheric contribution

$$I_{\text{atm}} = 100 [\text{mg/m}^2\text{-day}] * 0.001 [\text{g/mg}] * 15 [\text{m}]/2 = 1.5 \text{ g/m-day}$$

Traffic contribution (using information from Table 8.8)

$$\begin{aligned} I_t &= 0.671 [\text{g/axle-km}] * 240 [\text{cars/day}] * 2 [\text{axles/car}] * 0.001 [\text{km/m}]/2 \\ &= 0.32 \text{ g/m-day} \end{aligned}$$

(Division by 2 in the preceding estimates reflects two curbs on each side of the road.)

Total input

$$I = 6 + 1.5 + 0.32 = 7.82 \text{ g/m-day}$$

Removal coefficient from Equation (8.4)

$$\xi = 0.0116e^{-0.08*10}(15 + 50) = 0.34 \text{ day}^{-1}$$

Equilibrium curb load by Equation (8.6)

$$P_{\text{eq}} = I/\xi = 7.82/0.34 = 23 \text{ g/m}$$

Effect of Curb Height on Accumulation

The fact that the rate of pollutant accumulation in curb (median barrier) storage is caused by wind- and traffic-induced air turbulence, leads to the logical conclusion that the height of the curb will affect the accumulation rate, that is, the higher the curb, the more pollutants will accumulate. This effect will be more profound for fine dust particles than for coarser litter particles that are less amenable to becoming airborne. Figure 8.12 shows such a relationship measured in the Washington, D.C., area. The graph indeed indicates that the curb height may have a profound effect on the accumulation of finer particles (<3.2 mm in diameter), while larger

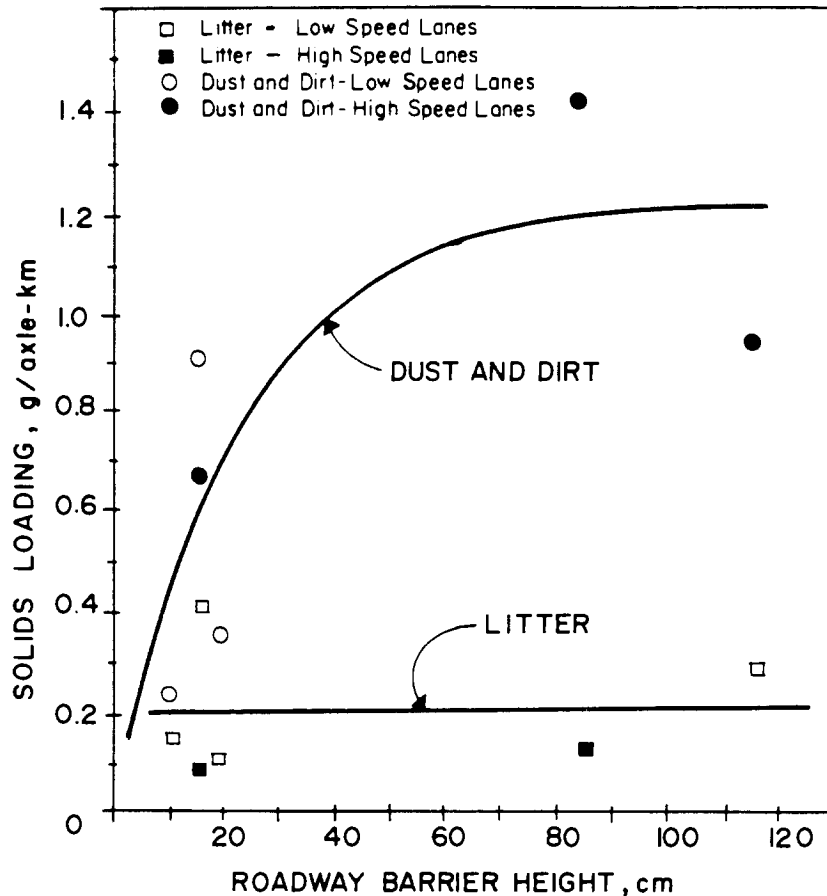


FIGURE 8.12. Effect of curb height on particle loading of street surface. (From Shaheen, 1975.)

particle accumulation will be unaffected. The curve for the finer particles accumulation proves the classic “first-order removal” concept expressed by Equation (8.3), which also implies that the amount of pollutants deposited on the surface or in curb storage will have a decreasing rate of increase approaching an equilibrium.

Example 8.2

What will be the equilibrium of solids accumulation if the curb height in the previous example is lowered to 5 cm?

Solution For $H = 5$ cm the removal coefficient becomes (Eq. (8.4))

$$\xi = 0.0116e^{-0.088 \cdot 5}(15 + 50) = 0.51 \text{ day}^{-1}$$

and the equilibrium accumulation is

$$P_{eq} = 7.82/0.51 = 15.33 \text{ g/m}$$

which represents a 33% reduction.

Winter Accumulation of Pollutants

The concept of translocation and removal of pollutants from curb storage by wind and traffic that determines the magnitude of the curb loading cannot be applied during the winter period in snowbelt urban areas. In addition, snow-removal practices and application of deicing chemicals further complicate the process.

Typically, even after a minor snowfall in most northern U.S. cities snow-removal activities are initiated by the highway and sanitation crews. The snow from the streets and highways is plowed toward the curb and large quantities of salt are then applied to provide bare pavement for traffic. The accumulated snow piles near the curb (Fig. 8.8) may persist for months in northern cities, including Chicago, Milwaukee, Minneapolis, Cleveland, and Buffalo. The snow piles are then effective traps of street pollutants, including large amounts of salt. Also, due to increased heating and energy use, atmospheric deposition from industrial sources is increased. On the other hand, dust from soil erosion is reduced due to the protective snow cover and the freezing of soils. From data by Bannerman et al. (1984) presented in the list on page 453, it can be documented that dry atmospheric deposition in the winter months in Milwaukee (December–May) was 1.5 times higher for COD and solids than in the nonwinter period. Deposition rates for phosphorus, lead, and sulphates were statistically indistinguishable from nonwinter rates.

Because the solids and pollutants are incorporated into snow and are not removed from the snowpack by wind and traffic the accumulation rate is almost linear and, hence, much higher than in nonwinter periods. For this reason the quantity of accumulated pollutants near the curb at the end of the snow period is very high, as shown on Figure 8.13. Bannerman et al. (1984) documented that the street loads of sediment and toxic metals are at their highest level from the onset of the first significant spring melt through the first spring rainfall event. In the Milwaukee research Bannerman et al. reported that 20% to 33% of the annual load of these substances can occur during this period. It should also be pointed out that the possibility of reducing the quantity of street pollutants during the winter is very limited, since the use of sweeping equipment is not possible when snow piles and frozen ice are located on the sides of the streets.

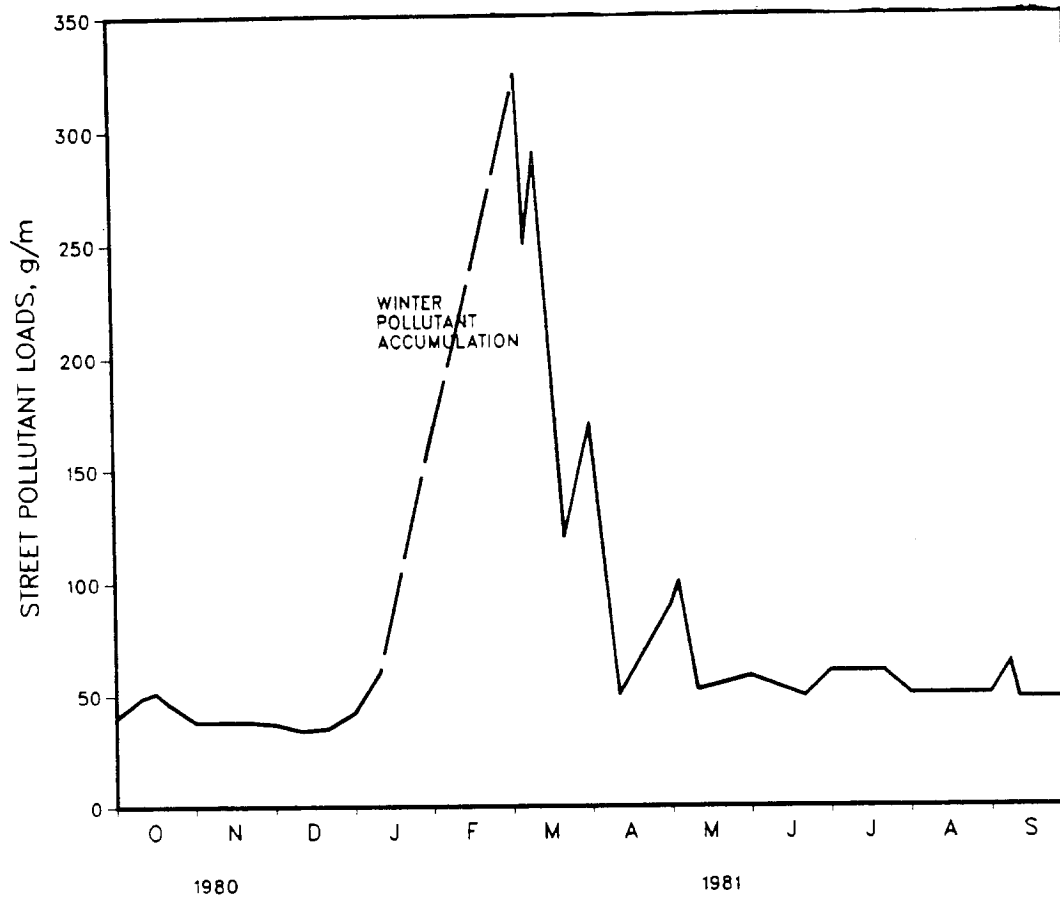


FIGURE 8.13. Annual variation of street accumulation of particulate pollutants in snowbelt urban areas. (Data from Wisconsin Department of Natural Resources.)

Removal of Pollutants from Curb Storage by Precipitation (Washoff)

When surface runoff occurs on impervious surfaces (Fig. 8.14) as a result of either natural storms or street flushing practices, the splashing effect of rain droplets and drag forces of the flow put particles in motion. Many hydraulic models have appeared in the literature on sedimentation that could potentially be applicable to the problem of particle pickup and transport.

From the numerous equations published in the literature, the Yalin equation (Yalin, 1963) (see Chapter 5 for a detailed discussion of the Yalin equation) is viewed by many as the one that best describes the pickup and transport of particles by shallow flows typical of street gutters (Sutherland and McCuen, 1978). Huber (1986) quoted a research investigation by the University of Florida and reviewed several theoretical

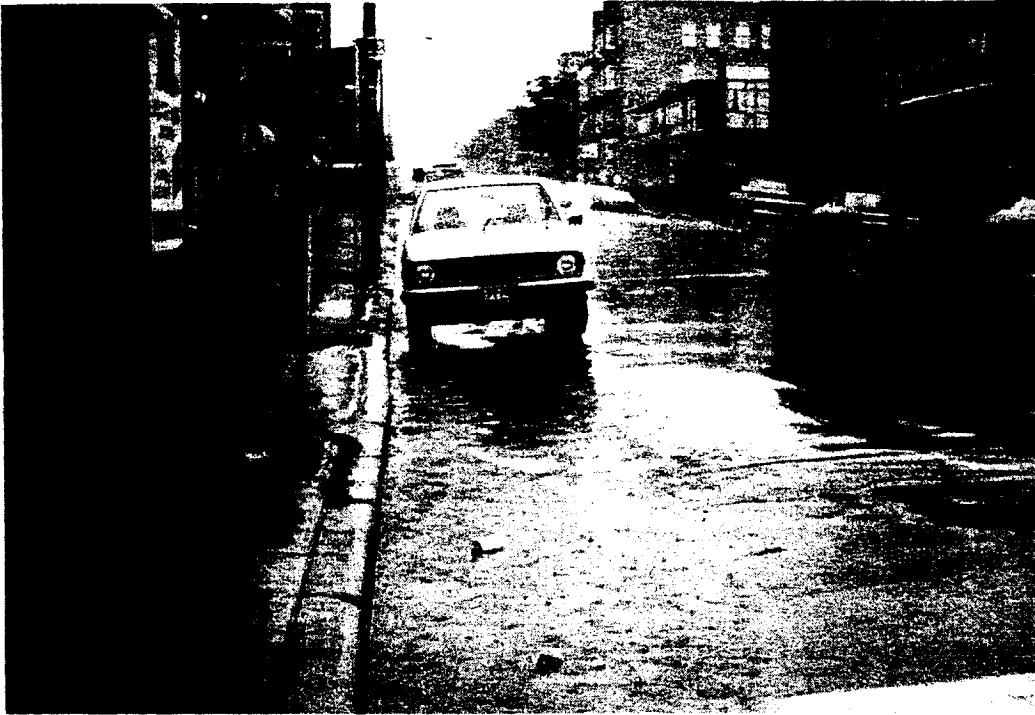


FIGURE 8.14. Washoff of street refuse during a rainstorm. (Photo: V. Novotny.)

approaches to modeling sediment pickup and transport. The study at the University of Florida concluded that although the theoretical sediment transport theory is certainly attractive and worth studying, it is often insufficient in practice because of lack of data for parameter evaluation, sensitivity to time period and breakdown into small discrete computational elements, and uncertain cross-sectional geometry of street gutters and surfaces, and besides which, the simpler semiempirical models work as well or better. Huber pointed out quite correctly that urban runoff modeling is generally crude and always requires calibration regardless of the level of sophistication of individual models and submodels.

The simplest, most widely used but empirical model of washoff was introduced by Sartor et al. (Sartor and Boyd, 1972; Sartor, Boyd, and Agardy, 1974). The authors analyzed the solids washoff data using a simple first-order removal concept

$$\frac{dP}{dt} = -K_u r P \quad (8.9)$$

where

r = rainfall intensity

k_u = a constant called "urban washoff coefficient" that depends on street surface characteristics

P = amount of solids remaining on the surface

t = time

The constant, k_u , was found to be almost independent of particle size within the studied range of 10 μm to 1 mm.

Equation (8.9) integrates to

$$P_t = P_0(1 - \exp(-k_u r t)) \quad (8.10)$$

where

P_0 = the initial mass (weight) of solids in the curb storage

P_t = mass (weight) of material removed by rain with intensity r and duration t

The value of the urban washoff coefficient was almost arbitrarily chosen as 0.19 if the rain intensity is in millimeters/hour. Note that this value has been recommended by most subsequent urban runoff models that utilize this concept. The authors of STORM (Hydrologic Engineering Center, 1975) modified Equation (8.10) by assuming that not all the solids are available for transport. Then

$$P_t = AP_0(1 - \exp(-k_u r t)) \quad (8.11)$$

where

$$A = 0.057 + 0.04(r^{1.1}) \quad (8.12)$$

is the so-called availability factor that accounts for the nonheterogeneous makeup of particles and the variability in the travel distances of the dust and dirt particles. The maximum value for A is 1.0.

Huber (1986) pointed out that the washoff model expressed by Equation (8.11) is not completely without physical basis since the theoretical sediment transport theory predicts dependence of scour upon flow rate (see the Yalin equation in Chapter 5). The Sartor et al. equation and its modifications have been incorporated into most of the widely used urban runoff models presented in the previous chapter.

Example 8.3: Washoff Estimation by the Sartor et al. Model

Estimate solids removal from the curb storage by the Sartor et al. concept for a 15-mm/hr rain lasting 1.5 hours with $P_0 = 50 \text{ g/m}$.

Solution From Equation (8.12) the availability factor is

$$A = 0.057 + 0.04(15^{1.1}) = 0.84$$

Then the amount removed by rain is

$$P_r = 0.84 * 50(1 - \exp(-0.19 * 15 * 1.5)) = 41.42 \text{ g/m}$$

The particle removal by rain is 83%.

To obtain the loading of pollutants, the sediment loads are commonly multiplied by multipliers expressing the pollution content of the sediment. In modeling terminology these multipliers are called *potency factors*. The magnitudes of potency factors for urban and rural areas were investigated by Zison (1980).

Pollutant Removal by Winter Snowmelt

The hydrology and pollutant loads of urban areas located in snowbelt areas is different from that typical for nonwinter periods. The melting of snowpack is a result of heat inputs and heat balance that bring the temperature of the snowpack above the melting point. The melting point of ice or snow with small quantities of pollutants is around 0°C. However, the addition of salt and the exothermic nature of the salt dissolution decreases the melting point of the saline melt below that for clean water.

The heat inputs into the snow pack are solar and atmospheric radiation, sensible heat, and heat gained by condensation of moisture above the snowpack; heat losses include reflectivity (albedo) of the snow, back radiation, and sublimation. Reflectivity (albedo) for urban snow may vary between 20% for dirty saturated snow to about 85% of reflected radiation for fresh fallen snow.

Observation and modeling of winter snowmelt rates in urban areas (Novotny, 1987, 1988; Bengtsson, 1982) showed that melting rates, even during days with higher temperatures, are relatively small, much less than those for a typical nonwinter storm event of the same magnitude (meaning that storm depth is the same as the water content of the snowpack). For example, assuming a clear sunny day in late February and considering the average incoming solar radiation at the 46 degree latitude, dry conditions and a 10°C temperature of the ambient air the maximum runoff from melting snow is only about 0.15 cm/hr. One has to also consider the fact that pervious urban areas in spite of freezing are still permeable enough to absorb the snowmelt from above. Therefore, only impervious urban

areas can be considered in the hydrological models of snowmelt generation from urban areas.

Enrichment of Urban Snowmelt

In Chapter 5 the term *enrichment* referred to the difference between the pollution content of the soil and that of the runoff. For winter snowmelt processes, the enrichment ratio represents the ratio of the pollutant concentration in the snowmelt and that in the snowpack or

$$ER = \frac{\text{Meltwater concentration}}{\text{Bulk snow concentration of prior snowmelt}} \quad (8.13)$$

The enrichment ratio of unity would mean that the pollutant concentrations in the snowmelt (runoff) and snowpack are similar. However, this is rarely the case. During freezing and refreezing, the ice crystals that form the snowpack effectively reject some dissolved pollutants that may then become easily available for transport by subsequent snowmelt water. This phenomenon is especially evident when dealing with pollution and modeling caused by salt applications (salinity and chloride contents). In studies in Milwaukee (Novotny, 1987), enrichment ratios of 6 to 10 had to be used for modeling salt loads to receiving waters. An enrichment ratio of greater than one will result in a "first flush" effect. A similar study with similar results was also performed in Sweden by Westerström (1990), who measured peak enrichment ratios of 4 to 6 for chlorides and nitrates in the first 25% of snowmelt, and up to 8 for sulphates. All snowmelt concentration curves showed a strong peak, followed by a fast decline (Fig. 8.15). For pH the difference between that of snowpack and the beginning of snowmelt was more than one order of magnitude (snowpack pH was one pH logarithmic unit higher than pH in the beginning of snowmelt).

On the other hand, due to the fact that snowmelt must percolate through the snowpack, particulate pollutants are filtered out and remain mostly in the snowpack, and are subsequently deposited into the curbside deposits, as shown on Figure 8.8. Thus for these pollutants the enrichment ratio is less than one. Such pollutants include suspended solids, both volatile and mineral lead, many metals, and phosphates. This was confirmed by, among others, McComas, Cooke, and Kennedy (1976), who compared winter snowmelt and rainfall runoff events in northeast Ohio. The results indicated that while snow was a significant source of water, the snowmelt load of phosphorus was much smaller, insignificant, when compared to a similar rainfall-runoff event. Most of the phosphorus is associated with fine sediments that remain on site and do not move with snowmelt.

SNOWMELT EVENT FEB. 20 - 28, 1985

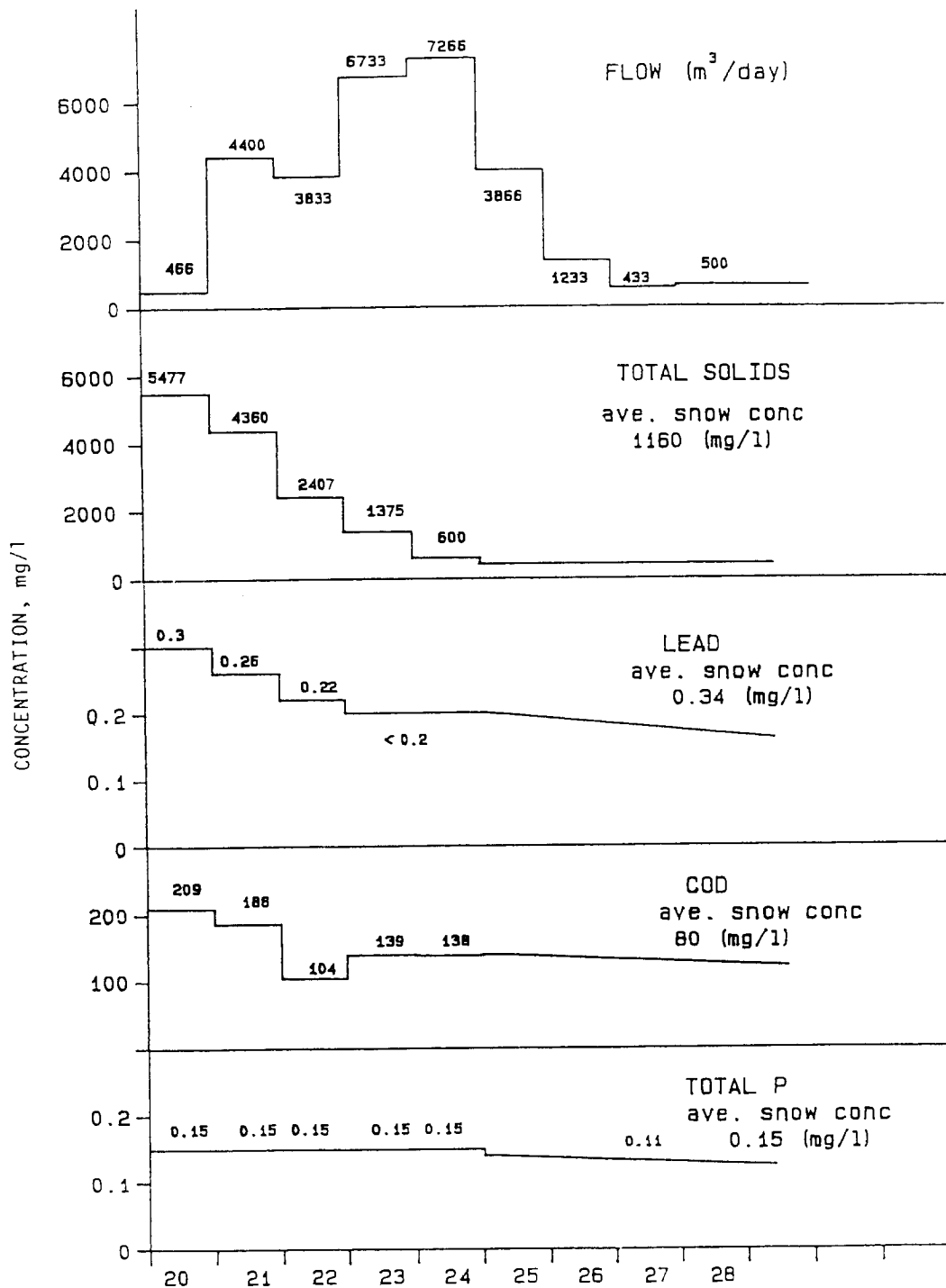


FIGURE 8.15. Snowmelt enrichment by pollutants in urban snowmelt. Data from an urban experimental watershed in Milwaukee. (After Novotny, 1987.)

Oberts (1986, 1990) noted that high concentrations of solids, phosphorus, lead, and zinc in urban snowmelt are due in part to sand and salt spread to improve winter driving conditions. Oberts (1982a) also found that snowmelt is a significant pollution source of urban lakes located in the Minneapolis–St. Paul metropolitan area.

SOURCES OF SPECIFIC POLLUTANTS IN URBAN RUNOFF AND CSOs

Microorganisms

Microbial analyses of storm water in Baltimore, Maryland, performed by Johns Hopkins University (Olivieri et al., 1977) revealed the presence of coliforms, pathogenic organisms, and viruses in both combined and storm sewer flows. The ranges of fecal coliforms found in storm runoff were 200 to more than 2000 MPN/100 ml, with a majority of samples—123 out of 136—having fecal coliform counts greater than 2000 MPN/100 ml. Storm samples with fecal coliform densities greater than 2000 MPN/100 ml were 95% positive for *Salmonella*. Ranges of *Salmonella* densities in urban runoff from Baltimore were less than 1 to more than 11,000 MPN/101. Six storm water flows were examined for viruses and all tested positive.

Significant differences have been found at the NURP test sites between the event mean concentration (EMCs; see the following section, titled “Statistical Quality Characteristics of Urban Runoff,” for a definition) of fecal coliform bacteria between winter (cold) and nonwinter periods (U.S. EPA, 1983). The average value for EMC for fecal coliforms measured at 17 different sites was 21,000/100 ml, while that for cold periods was 1000/100 ml, a 21-fold difference. It was speculated that these significant seasonal differences cannot be related to any comparable magnitude of urban land-use activities taking place in cold and warm periods. The levels of fecal coliform bacteria, especially those in the warm period, are high and could be considered a health risk. However, it was pointed out that indicators such as fecal coliform counts may not be useful in identifying health risks from urban runoff pollution.

Glennie (1984) correlated the density of total coliforms in surface runoff to the population density in the watershed. The resulting equation

$$T. coli \text{ (MPN/100 ml)} = 5700 \text{ (PDE)}^{1.35}$$

where PDE is the population density equivalent in persons per acre, had a coefficient of 0.95. This indicates that the coliform bacteria in urban runoff may be caused by the cross-connection of sewage into storm

drainage. In view of the results of the NURP statistical analysis of a large number of samples from a number of sites throughout the United States the correlation by Glenne may be only site specific.

Literature values refer to the fecal coliforms to fecal streptococci ratio (FC/FS) as an indicator of the origin of fecal pollution. A value of four or greater indicates human source of fecal pollution, while a value of less than one suggest nonhuman origin. In the Baltimore study just quoted, 86% of all storm water samples had a FC/FS ratio of less than one, and 94% of samples had a FC/FS ratio of less than four, giving an indication that most of the fecal contamination is of nonhuman origin. On the other hand, combined sewer overflows had 58% of the samples with a FC/FS ratio of less than one, and 15% with a FC/FS ratio of greater than four. Raw sewage had 50% of samples with a FC/FS ratio greater than four. The authors therefore stated that the FC/FS ratio should not be employed as a magic number to evaluate the source of contamination in a complex system.

Hydrocarbons (PAHs) and Other Toxic (Priority) Pollutants

Based on the analyses of urban storm runoff effluents, Whipple and Hunter (1979) speculated that after implementation of point source abatement, urban runoff will become the major source of petroleum hydrocarbons. The sources of toxics are numerous (Table 8.11).

Inorganic pollutants. As a group, the toxic metals were by far the most prevalent priority pollutant constituent found in urban runoff by the NURP studies (U.S. EPA, 1983). Fourteen inorganic, potentially hazardous constituents (13 toxic metals plus asbestos) were detected in the NURP study sites. Most often detected among the metals were copper, lead, and zinc, all of which were found at least in 91% of the samples. Other frequently detected inorganic pollutants included arsenic, chromium, cadmium, nickel, and cyanide. Pitt (1988) found asbestos in urban runoff in a watershed in the San Francisco Bay area where no natural sources were known; however, no specific single source was identified.

By statistically analyzing the event mean concentrations (EMCs) (see the next section for a definition and methods of estimating EMCs) of urban runoff in Milwaukee, Bannerman (1991) found that metallic toxicity of urban runoff varies with land use. For commercial areas with a high traffic count, acute toxicity criteria for warm-water fish were violated in 73% of the samples for lead and more than 90% of the samples for silver, copper, and zinc. On the other hand, event mean concentrations of urban runoff from a medium-density residential area with low traffic density the

TABLE 8.11 Sources of Toxic and Hazardous Substances in Urban Runoff

	Automobile Use	Pesticide Use	Industrial/Other Use
<i>Heavy Metals</i>			
Copper	Metal Corrosion	Algicide	Paint, wood preservative, electroplating
Lead	Gasoline, batteries		Paint, lead pipe
Zinc	Metal corrosion Tires, road salt	Wood preservative	Paint, metal corrosion
Chromium	Metal corrosion		Paint, metal corrosion, electroplating
<i>Halogenated Aliphatics</i>			
Methylene chloride		Fumigant	Plastics, paint remover, solvent
Methyl chloride	Gasoline	Fumigant	Refrigerant, solvent
<i>Phthalate Esters</i>			
Bis (2-ethylexy) phthalate			Plasticizer
Butylbenzyl phthalate			Plasticizer
D-N-butyl phthalate		Insecticide	Plasticizer, printing inks, paper, stain, adhesive
<i>Polycyclic Aromatic Hydrocarbons</i>			
Chrysene	Gasoline, oil, grease		
Phenanthrene	Gasoline		Wood/coal combustion
Pyrene	Gasoline, oil, asphalt	Wood preservative	Wood/coal combustion
<i>Other Volatiles</i>			
Benzene	Gasoline		Solvent
Chloroform	Formed from salt	Insecticide	Solvent, formed from chlorination
	Gasoline, asphalt		Solvent
Toluene	Gasoline, asphalt		Solvent
<i>Pesticides and Phenols</i>			
Lindane (gamma BHC)		Mosquito control	
Chlordane		Seed pretreatment	
Dieldrin		Termite control	
Pentachlorophenol		Insecticide	Wood processing Wood preservative, paint
PCBs			Electrical, insulation, several other Industrial applications
<i>Asbestos</i>	Break and clutch lining Tire additives		Insulations

Source: After U.S. EPA (1990b).

toxic criteria were exceeded in only 0% to 4% for lead, 0% for silver, and 20% to 30% for zinc and copper. High concentrations of mercury, cadmium, and lead have also been observed in urban soils (Carey, 1979).

Organic pollutants. In the NURP study by the U.S. EPA (1983) organic pollutants were detected less frequently and at lower concentrations than the inorganic pollutants. The most commonly found organic pollutant was the plasticizer bis(2-ethylhexyl) phthalate (22%), followed by the pesticide α -hexachlorocyclohexane (α -BHC) (20%). An additional 11 organic pollutants were reported with detection frequencies between 10 and 20%. Among the PCB group there was only one detected among all the samples.

Marsalek (1986) reported data on priority (toxic) pollutants in urban runoff gathered in Canadian watersheds of the Great Lakes. The highest concentrations in storm runoff were found for trace elements as a group. Among the individual elements the highest mean concentrations were found for zinc (EMC = 400 $\mu\text{g/l}$ in water), copper (19 $\mu\text{g/l}$), lead (90 $\mu\text{g/l}$), and nickel (16 $\mu\text{g/l}$). Much higher concentrations were measured in the sediment. Significant (greater than the detection limit) concentrations were found for PAHs (poly-aromatic hydrocarbons). The highest concentrations were found for 1,2-dichlorobenzene (0.04 $\mu\text{g/l}$), α -BHC (0.02 $\mu\text{g/l}$), and for the total PCBs (0.014 $\mu\text{g/l}$). Again, higher concentrations were found in the sediments, and it was speculated that most of the PAHs transported by runoff occurred most often with the sediment (Marsalek, 1990). The probability distribution of PAH concentrations in runoff was approximated by a log-normal distribution (see the following section titled "Statistical Quality Characteristics of Urban Runoff").

Urban runoff was found to be the source of 48% of the petroleum hydrocarbons, 3% of the lower molecular weight polycyclic aromatic hydrocarbons, 44% of the higher molecular weight polycyclic aromatic hydrocarbons, 65% of lead, 56% of zinc, and 5% of copper entering the Narragansett Bay annually (Hoffman et al., 1984; Hoffman, 1985). Annual loadings of petroleum hydrocarbons measured by Hoffman (1985) were the highest from heavy industrial land uses (14,000 kg/km^2), followed by busy highways (7800 kg/km^2). Loadings from commercial and residential land uses were one order of magnitude less (580 kg/km^2 for commercial and 180 kg/km^2 for residential land uses, respectively). The highest loadings of PAHs were found on the highway (18.1 kg/km^2), followed by heavy industrial lands (8.4 kg/km^2); similarly, PAHs loadings of one order of magnitude lower were measured on commercial (0.59 kg/km^2) and residential (0.27 kg/km^2) lands.

Fam, Stenstrom, and Silverman (1987) found in the San Francisco Bay area watersheds that land uses with high commercial/industrial activity

have much greater aliphatic hydrocarbon emissions than noncommercial ones. The ratio of total extractable organics to total organic carbons was found to vary with land use, with a ratio of six or more indicating significant commercial activity.

Bannerman (1991) found that the EMCs of PAHs in Wisconsin urban runoff (the Milwaukee and Madison areas) violated the human cancer criteria in 60% of samples for chrysene and phenanthrene, 82% for pyrene, and more than 95% of samples for fluoranthene and benzo (GHI) perylene. EMCs of aldrin, toxaphene, and DDT in Wisconsin urban runoff almost always exceeded human cancer criteria. Other pesticides found less frequently in such runoff included endrin, chlordane, malathion, diazinon, and lindane. Annual monitoring of urban soils between 1969 and 1979 has demonstrated that they generally have higher pesticide residue concentrations than agricultural soils in the same (nearby) location (Carey, 1979).

A comprehensive study on toxic contamination of urban runoff and CSOs was prepared for the U.S. EPA by Pitt and Barron (1989). They analyzed about 150 samples of surface runoff (sheet flow) and CSO, and found that about 15% of the samples analyzed were extremely toxic. The remaining samples were evenly split between being moderately toxic and nontoxic. The greatest detection frequencies were for 1,3-dichlorobenzene and fluoranthene, which had detection frequencies of 23%.

A higher percentage of the CSO samples were considered extremely toxic than any of the others followed by parking and storage area surface runoff samples. Roof runoff, urban creeks, and CSO samples had the greatest detection frequencies, while vehicle service and parking areas had the highest observed concentrations of organic toxicants. Roof runoff had the greatest concentrations of zinc, while parking areas had the greatest concentrations of nickel. Traffic- and parking-associated areas had the greatest concentrations of lead, and urban creeks were highest in copper content.

Effect of Acidity of Urban Precipitation on Pollutant Loads

As pointed out in Chapter 4, the acidity of urban precipitation caused by local and regional sources of sulphur oxides and mostly local (traffic) sources of nitric oxides is quickly buffered by overland flow (Novotny and Kincaid, 1982). However, this "buffering" capacity in urban areas is provided primarily by the infrastructure, including buildings, pavements, and sewers. During buffering, the excess hydrogen ions from acid precipitation is buffered by carbonates, which releases the cations, such as

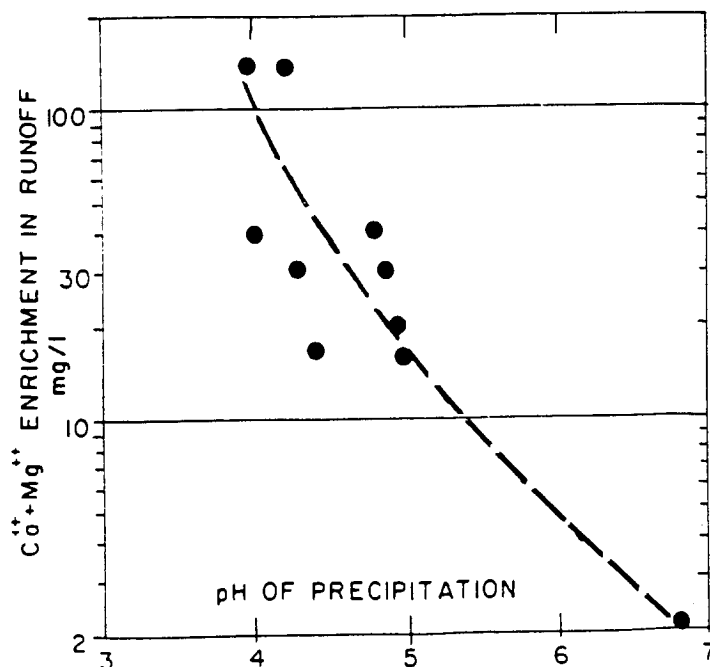


FIGURE 8.16. Effect of the acidity of urban precipitation on the elutriation of minerals and cations from urban infrastructure. (From Novotny and Kincaid, 1982.)

calcium and magnesium (primary components in concrete, mortars, and soil), into the runoff. Figure 8.16 shows the relationship between the acidity of precipitation and the calcium–magnesium content of runoff from a large concrete parking lot basin located in Milwaukee.

Acids in precipitation also dramatically increase the solubility of metals used in cars, zinc gutters, and other places. Förster (1990) showed that urban metal roofs and roof-and-gutter materials (such as zinc) are good buffers for acid rainfall, but metals and other pollutants (for example, PAHs from roofs with tar shingles or cover) are elevated in the runoff from these surfaces. Although zinc roof materials provided very high buffering of the pH of the acid rainfall (pH of runoff was near neutral), dissolved zinc content in the runoff was 450 times greater than that in the rainfall (0.005 mg of dissolved Zn/l in rainfall compared to 2.2 mg dissolved Zn/l in roof runoff from zinc-covered roofs and gutters) when the pH of the rainfall was less than 4.

Growth and Accumulation of Sewer Solids

In the 1970s Pisano et al. (1979) investigated solids accumulation in combined sewers in Boston during dry periods. Research in Switzerland

(Krejci et al., 1987) showed that solids accumulation in sewers has two components: (1) settling of solids in places with low velocity during low-flow, dry-weather flow, and (2) slime growth on the walls of the sewers. A part of the slime growth component is less affected by the hydraulics of the flow and is not easily washable. However, during high-flow storm events scouring of both components will contribute to the solids load of the overflows and may lead to the "first flush" effect.

By statistical analysis, Pisano et al. developed the following equation for estimating solids accumulation (converted and modified for metric units)

$$TS = 11.14S^{-0.4375}q^{-0.51} \quad (8.14)$$

where

TS = solids deposition and growth rate in combined sewers in grams per meter of sewer per day

S = average slope of the sewer, dimensionless (m/m)

q = per capita dry weather waste rate, liters/cap-day

and the coefficient of correlation was 0.85.

The inverse proportionality of the solids deposition rate to the slope reflects the effect of flow velocity that controls the deposition. The per capita waste flow rate may reflect the concentration of solids in the flow. The higher the per capita flow, the more diluted solids may become in the sewer flow, resulting in less deposition.

Research in Switzerland by Krejci et al. (1987) revealed that solids deposition and slime growths in combined sewers represent a significant portion of the total pollution load. They found that the growth of the washable slime layer varied between 10 to 30 g of TSS per square meter of sewer surface per day, and depended on the local dry-weather flow velocity. The sewer slime contained by weight 80% COD, 5% TKN, and 1% phosphate. In the overall balance of pollution carried by the combined wet-weather flow during storm events, the sewer slime-sediment layer contribution represented 55% to 80% of the total load for TSS, COD, TOC, TKN, and total P. This contribution exceeded that by surface runoff and sewage flows during the wet-weather period.

Krejci et al. also found that the length of the dry period preceding the wet-weather flow event had no noticeable effect on the quantity of deposits in sewers, which appears to contradict the results in Boston by Pisano et al. Apparently, the slime-deposit layer is in equilibrium with the scouring energy of the dry-weather flow, and this equilibrium is attained after a relatively short period of time.

Cross-Connections and Illicit Discharges into Storm Sewers

Research and investigations of the impact of illicit and cross-connection on the pollution content of storm water discharges has revealed that these inputs can contribute significant pollutant loads (Schmidt and Spencer, 1986; Pitt et al., 1990a; Pitt, Lalor, and Driscoll, 1990).

Non-storm water discharges in storm sewers originate primarily from sewage and industrial wastewater leaking from sanitary sewers, failing septic systems in storm-sewered areas, and from vehicle maintenance activities. However, almost any type of pollution may find its way into urban storm sewers by illicit discharges and accidental spills. Deliberate dumping into storm sewers and catch basins of used oil or waste paint are especially common and troublesome. Leaking underground storage tanks, leachate from sanitary landfills, and hazardous waste treatment, storage, and disposal sites can also contribute to urban storm water and combined sewer overflow pollution. In the United States the detection, identification, and elimination of such discharges is a major focus of the NPDES Stormwater Permit program (see Chapter 2 for a discussion). The manual by Pitt, Lalor, and Driscoll (1990) provides procedures for identification of sources and magnitudes of non-storm water discharges into storm sewers.

Mobile pollutants such as nitrates may also contaminate the shallow aquifer and discharge into urban drainage systems in the form of base flow in streams and infiltration inflows into sewers. The pollutional impact of these contributions is generally unknown.

Suburban Nonpoint Pollution Loads

Septic systems and construction erosion are the most common and significant sources of pollution in suburban areas. Septic system pollution is due to two pathways: (1) subsurface transport of mobile pollutants (primarily nitrate) via shallow discharging aquifers toward the receiving bodies of water, occurring primarily during base flow (during high-flow runoff events most urban streams may be recharging the aquifer), and (2) from effluent surfacing from failing septic systems.

Suburban nonpoint pollution loads are also affected by the presence or absence of wetlands (Oberts, 1982b; Brown, 1988) within the watershed. Wetlands are known to attenuate pollutants and runoff rates.

STATISTICAL QUALITY CHARACTERISTICS OF URBAN RUNOFF

During the Nationwide Urban Runoff Program (U.S. EPA, 1983) 28 urban sites were monitored throughout the United States. This research effort provided a large data base on the quality and loads by urban runoff (not including the combined sewer overflows). Major conclusions are presented herein. Prior to the presentation of the results and their interpretations, some fundamental statistics have to be reviewed.

Parameters and Statistics of the Urban Runoff Water Quality Evaluation

Event mean concentration. The water quality parameters exhibit large variations throughout each runoff event. Some events exhibit the first flush effect, for some other events the first flush effect has not been noticeable (Ellis, 1986). Thus, the NURP studies focused on evaluating the event mean concentrations, defined as

$$\text{EMC} = \frac{\text{Mass of pollutant contained in the runoff event}}{\text{Total volume of flow in the event}} = \frac{\sum Q_i C_i}{\sum Q_i} \quad (8.15)$$

where

Q_i = discrete flow ordinates on the event hydrograph

C_i = corresponding concentrations on the pollutograph

In most cases the total load by the runoff event is more important than the individual concentrations within the event due to the fact that runoff events are relatively short, the receiving water body provides some mixing, and the concentration in the receiving body of water is a response to the total load rather than to the concentration variability within each particular event. For some pollutants, such as nutrients, the total load is the most important decision variable in determining water quality impact. For this reason the EMC parameter has been found to be the most appropriate variable for evaluating the impact of urban runoff.

Mean, coefficient of variation, and probability distribution. The evaluation of the NURP data base has revealed that the probability distribution of EMCs follows a so-called *log-normal* probability distribution. A probability density or frequency distribution (pdf) expresses the probability that a concentration with a given magnitude falls within a specified interval, or

$$\int_{x_1}^{x_2} \text{pdf}(x) = p(x_1 > x > x_2)$$

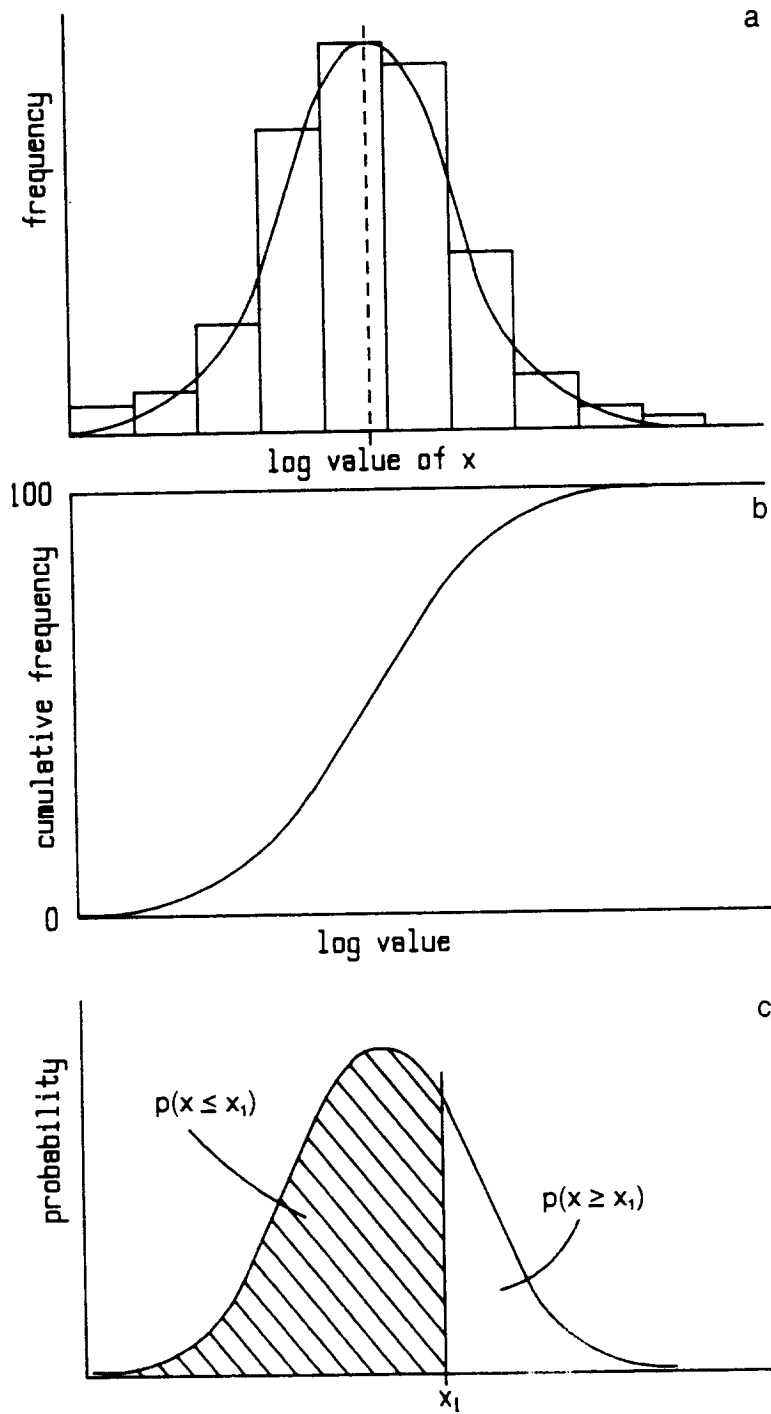


FIGURE 8.17. Frequency and probability distribution functions. (a) Frequency distribution of logarithmic values. (b) Cumulative probability distribution. (c) Probability of exceedence.

where x is the value of the parameter and x_1 and x_2 are the limits of the interval. The frequency distribution of a parameter can be simply found by plotting or tabulating the number or percent of cases for which the value of the parameter falls within each interval (at this point one may select an arbitrary number of intervals between the smallest and the largest value of the record) versus the magnitude of the interval, as shown in Figure 8.17(a).

A cumulative frequency (probability) function (cpf) is thus a summation of all individual probabilities of all x 's that are either less or equal (or greater or equal) than a given value of x (Fig. 8.17(b)). The cumulative frequency curve can also be obtained by integration of the probability density (frequency) distribution function, as shown on Figure 8.17(c). Therein

$$\text{cpf}(x \leq) = p(x \leq X_1) \quad \text{or} \quad \text{cpf}(x \geq) = p(x \geq X_1)$$

Note that $p(x \leq X_1) = 1 - p(x > X_1)$ or a probability of a value being less or equal is one minus probability of the same value being exceeded. This is shown on Figure 8.17c.

If the interval becomes very small, then the probability (frequency) density function becomes continuous and may assume several typical shapes, such as are shown in Figure 8.17(b). There are several mathematical formulations that may describe the shape of these curves, of which Gaussian normal distribution and Pearson Type III distribution are the most common probability density functions used in hydrology. The reader is referred to the several standard texts on hydrology referenced in Chapter 3, or to standard texts on statistics, for more detailed discussion. Probability paper that has one ordinate scaled according to the Gaussian cpf is very convenient for representation of a series of observations such as a sequence of EMCs of surface runoff.

To plot EMCs on probability paper (arithmetic or logarithmic) the concentrations (or any other parameter values subject to statistical analysis, including flows and rainfalls) are arranged according to their order of magnitude, which may be ascending or descending. For pollution studies the ascending (from the smallest to the greatest) order of magnitude is commonly selected; for flood studies the descending order of magnitude is common. The plotting position on the probability paper is then assigned to each parameter value as follows:

$$p(\text{in } \%) = \frac{m}{n + 1} \tag{8.16}$$

where

m = ascending or descending order of magnitude

n = total number of data in the time series

The *mean*, μ , of the series is the arithmetic average of all values, X_i , in the series, or

$$\mu = \frac{\sum X_i}{n} \quad (8.17)$$

To fit the series of EMCs or any other series to the log-normal distribution, a logarithmic transformation of the values must be performed, or $X_i = \ln Y_i$, where Y is the original series and X is the transformed series of natural logarithms of the values of the series.

The *median* of the series is the central value for which one-half of the data values in the series has a magnitude that is less than or equal. If the series fits normal distribution (original or transformed), the median approximately equals the mean. However, if the original series has been transformed into a series of logarithms exhibiting normal (meaning log-normal) distribution, then the median and the mean of the original nontransformed series are different. On a probability plot the median corresponds to a 50 percentile value on the plot.

The *standard deviation* is a measure of the scatter of the values of the series around its mean.

$$s = \sqrt{\frac{\sum (X_i - \mu)^2}{n - 1}} \quad (8.18)$$

The *coefficient of variation* is a ratio of the standard deviation and the mean, or

$$CV = \frac{s}{\mu} \quad (8.19)$$

The *expected value* of the series at any probability or frequency of occurrence (X_α) of a series that has been logarithmically transformed can be determined by

$$X_\alpha = \exp(\mu_{\ln x} + Z_\alpha s_{\ln x}) \quad (8.20)$$

where

Z_α = the standard normal probability

$\mu_{\ln x}$ = mean of log-transformed data

$s_{\ln x}$ = standard deviation of log-transformed data

The standard normal probability value, Z_α , can be obtained from probability tables or calculated from the Gaussian cumulative probability distribution function. For the 90 percentile value (10% exceedence) $Z_\alpha = 1.2817$, for 95 percentile value (5% exceedence) $Z_\alpha = 1.645$, and so forth.

Also X_α can be expressed as a ratio of the median value of the original series by the following equation, which defines the ratio in terms of the coefficient of variation

$$\frac{X_\alpha}{\text{Median}} = \exp(Z_\alpha \sqrt{\ln(1 + CV_{\ln x}^2)}) \quad (8.21)$$

Example 8.4: Statistics and Probability Distribution of Event Mean Concentrations

Nine urban runoff events were measured at a storm sewer outlet location. The event mean concentrations for each event were calculated using Equation (8.15) and are given in column 2 of Table 8.12. The resulting probability plot is in Figure 8.18.

Solution Using the results reported in Table 8.12, the arithmetic mean is

$$\mu_x = 13.08/9 = 1.45 \text{ mg/l}$$

TABLE 8.12 Statistical Evaluation of EMCs (Lead) in Storm Water Runoff

Date of Sampling	X [EMC mg Pb/l]	$\ln X$	Order of Magnitude (m)	Plotting Position $p = 100m/(n + 1)$
3/5/1989	1.41	0.3423	6	60
4/15/1989	2.24	0.8065	8	80
5/2/1989	1.10	0.0953	4	40
5/25/1989	0.90	-0.1054	3	30
6/14/1989	1.27	0.2390	5	50
7/12/1989	1.72	0.5423	7	70
7/20/1989	3.20	1.1632	9	90
9/17/1989	0.50	-0.6931	1	10
10/4/1989	0.74	-0.3011	2	20
$\Sigma X = 13.08$		$\Sigma \ln x = 2.0890$		

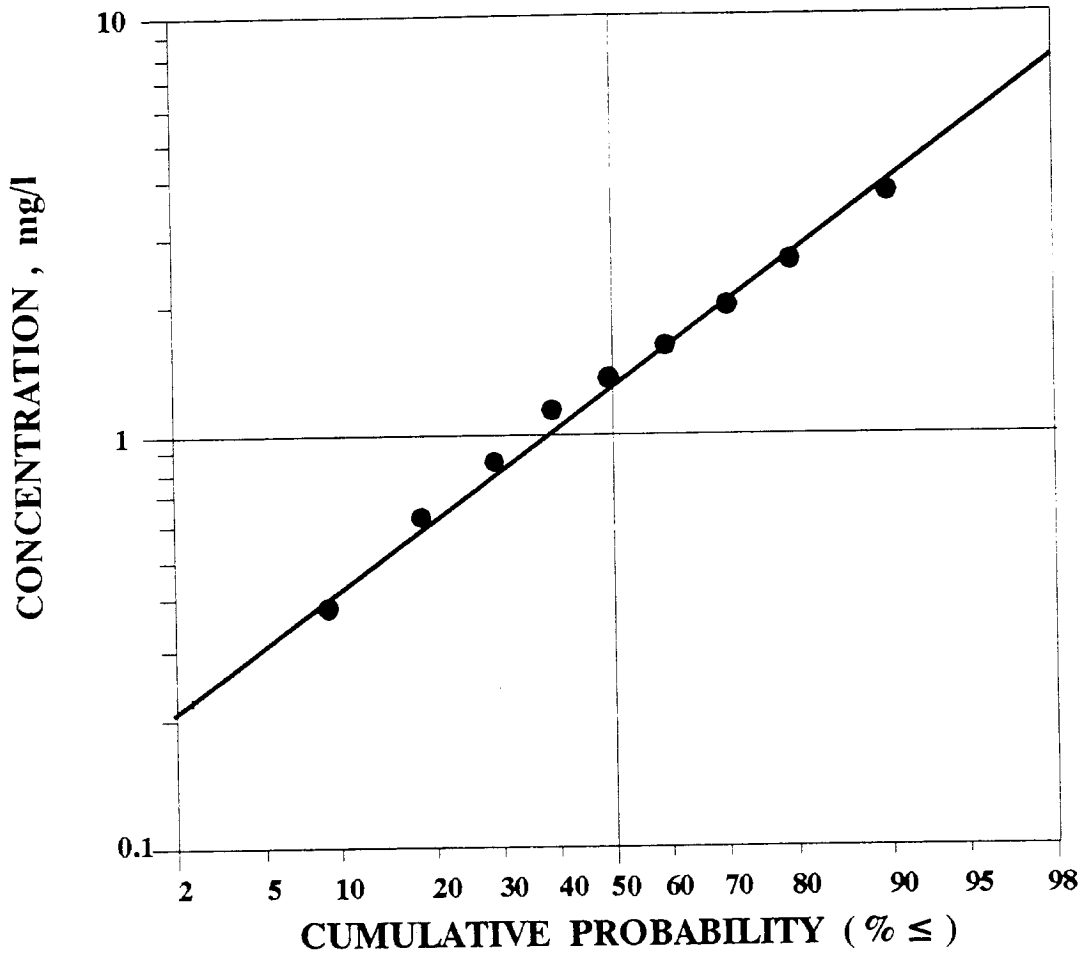


FIGURE 8.18. Cumulative probability distribution for Example 8.4.

The arithmetic median is 1.27 mg/l. Hence, since the mean does not equal the median, the frequency distribution is asymmetrical (skewed).

The logarithmic mean is

$$\mu_{\ln x} = 2.0890/9 = 0.2321$$

The median of the logarithmic transformed series is 0.239. Hence, the transformed series appears to be symmetrical and normally distributed.

Example 8.5: Probability of Exceedence

For the series in the previous example determine a value that would have a 95% probability of not being exceeded.

TABLE 8.13 The Transformation Used to Calculate the Standard Deviation

Data	Value of $\ln x$	$\ln x - \mu_{\ln x}$	$(\ln x - \mu_{\ln x})^2$
1	0.3423	0.1102	0.0121
2	0.8065	0.5744	0.3299
3	0.0953	-0.1368	0.0187
4	-0.1054	-0.3375	0.1139
5	0.2390	0.0069	0.00005
6	0.5423	0.3102	0.0962
7	1.1632	0.9311	0.8669
8	-0.6931	-0.9252	0.8560
9	-0.3011	-0.5332	0.2843
$\Sigma = 2.5781$			

Solution Calculate the standard deviation of the transformed series (see Table 8.13). From the preceding example the mean of the transformed series is $\mu_{\ln x} = 0.2321$.

The standard deviation of the transformed series is

$$s_{\ln x} = [2.5781/(9 - 1)]^{0.5} = 0.5677$$

The coefficient of variation is then

$$CV = 0.5677/0.2321 = 2.44$$

The standard normal probability value for the 95% nonexceedence is $Z_\alpha = 1.645$. Thus the EMC for lead that will have this probability on nonexceedence (and hence $1 - 0.95 = 0.05$, 5% probability of exceedence) is, by Equation (8.20),

$$X_\alpha = \exp(0.2321 + 1.645 \times 0.5677) = 3.21 \text{ mg/l}$$

This checks with the probability graph on Figure 8.18.

The establishment of the fundamental probability distribution of EMCs as log-normal has a number of benefits (U.S. EPA, 1983):

- Concise summaries of highly variable data can be developed.
- Comparisons of results from different sites, events, etc., are convenient and are more easily understood.

- Statements can be made concerning frequency of occurrence. One can express how often values exceed various magnitudes of interest.
- A more useful method of reporting data than the use of ranges is provided; one that is less subject to misinterpretations.
- A framework is provided for examining “transferability” of data in a qualitative manner.

The log-normal distribution is completely specified by the mean and standard deviation of the transformed logarithmic series.

Nationwide (NURP) Characterization of Urban Runoff Quality

The NURP study established that even mean concentrations for total suspended solids (TSS), total phosphorus, total Kjeldahl nitrogen (organic nitrogen plus ammonia), total lead, and total zinc are extremely well represented by the log-normal distribution. For COD and nitrate–nitrite the log-normal distribution fitted the data quite well. For other constituents, including BOD, soluble phosphorus, and total copper, the log-normal distribution cannot be rejected. These were the pollutants of interest analyzed in the NURP study.

Land use effect on EMCs. The nationwide analysis did not find significant statistical correlations of the EMCs to the geographical locations of the site. Also three typical urban land uses, residential, mixed, and commercial, were not statistically different. Only open/nonurban lands were statistically significantly different from the previous three land-use types. Hence the NURP report concluded that

regardless of the analytical approach taken, we are forced to conclude that, if land use category effects are present, they are eclipsed by the storm to storm variabilities and that, therefore, land use category is of little general use to aid in predicting urban runoff quality at *unmonitored* (emphasized by the author) sites or in explaining site to site differences where monitoring data exists.

Correlation between EMCs and runoff volume. A total of 67 sites from 20 of the NURP projects were examined for possible correlation for nine constituents. Of the 517 linear correlation coefficients calculated 116 (22%) were significant at the 95% confidence level and 154 (30%) were significant at the 90% confidence level. Both positive and negative correlations were detected. Hence, the NURP study concluded that there is no significant linear correlation between EMCs and runoff volume.

This finding does have some importance in designing storm water monitoring programs. First, there is natural and appropriate bias that

favors emphasizing resource allocation to larger storm events. Since no significant linear correlation was found, such biases and differences are not expected to influence EMCs comparisons to any appreciable extent. Second, the probabilistic methodologies for examining the receiving water impact of storm water runoff assumes that concentration and runoff volume are independent (i.e., that there is no significant correlation).

Other factors' effects. In deterministic concepts, factors such as slope, soil types, and rainfall characteristics, are all potentially important. However, in a statistical sense derived from a large number of observations at various sites throughout the United States, these factors did not appear to have any real consistent significance in explaining observed similarities or differences among individual sites. However, the sites that were investigated all had storm sewers, hence, the effect of, for example, natural (swale) drainage on pollutant loads cannot be determined. Also no abatement, such as ponds or infiltration, were included in the test sites.

Overall urban runoff quality characteristics. Having determined that geographic location, land-use category, runoff volume, or other factors appear to be of little utility in explaining overall site-to-site, event-to-event variability to predicting the characterization of urban runoff at unmonitored sites, the best general characterization may be obtained by pooling the site data for all sites (other than the open/nonurban ones). The results for the single overall urban land-use category are given in Table 8.14. In the absence of better information, the NURP study recommended these EMC characteristics for planning purposes as the best description of the quality of urban runoff.

TABLE 8.14 Overall Water Quality Characteristics of Urban Runoff

Constituent	Typical Coefficient of Variation	Site Median EMC	
		For Median Urban Site	For 90 Percentile Urban Site
TSS (mg/l)	1-2	100	300
BOD (mg/l)	0.5-1	9	15
COD (mg/l)	0.5-1	65	140
Total P (mg/l)	0.5-1	0.33	0.70
Soluble P (mg/l)	0.5-1	0.12	0.21
TKN (mg/l)	0.5-1	1.50	3.30
NO ₂₊₃ -N (mg/l)	0.5-1	0.68	1.75
Total Cu (µg/l)	0.5-1	34	93
Total Pb (µg/l)	0.5-1	144	350
Total Zn (µg/l)	0.5-1	160	500

Source: From NURP studies, U.S. EPA (1983).

TABLE 8.15 EMC Mean Values for Pollutant Load Estimates

Constituent	Site	
	Median Urban Site	90 Percentile Urban Site
TSS (mg/l)	141-224	424-671
BOD ₅ (mg/l)	10-13	17-21
COD (mg/l)	73-92	157-198
Tot. P (mg/l)	0.37-0.47	0.78-0.99
Sol. P (mg/l)	0.13-0.17	0.23-0.30
TKN (mg/l)	1.68-2.12	3.69-4.67
NO ₂₊₃ -N (mg/l)	0.76-0.96	1.96-2.47
Tot. Cu (µg/l)	38-48	104-132
Tot. Pb (µg/l)	161-204	391-495
Tot. Zn (µg/l)	179-226	559-707

Source: U.S. EPA (1983).

It should be remembered that the values in Table 8.14 are derived from log-normal distribution of EMCs. For estimating loads of pollutants to receiving bodies of water these values should be converted to the mean values, which are given in Table 8.15. The ranges reflect the variability of the coefficient of variation of the EMCs.

Reconciliation of NURP Statistical Results with the Deterministic Urban Runoff Quality Concepts

This somewhat surprising conclusion of the NURP statistical analysis appears, at first, to contradict the deterministic findings presented in the first sections of this chapter. For example, does the poor correlation of EMCs to land use exclude the use of unit loads or more complex deterministic models? The first conclusion would be to throw out these earlier concepts. However, more detailed and thorough analysis leads to a different conclusion.

First, most of the unit loads related to land use and/or other parameters, as well as more complex deterministic models, are site specific, while the NURP statistical analysis encompasses the entire United States.

Second, the NURP conclusions are related to event mean concentrations, while at the same time a relationship between the runoff volume and the degree of imperviousness has been found in the NURP studies (Fig. 8.2). Since the unit load is a product of runoff volume and EMC, there may be a relationship of the unit loads to land use, while a relation to EMCs may not be profound.

Third, site specificity of deterministic hydrologic models is so dominant that conclusions obtained on a continental scale are not applicable to sites with a basin area ranging from a few hectares to hundreds square kilometers. Deterministic models are mostly applicable to site specific designs of abatement for which nationwide statistical averages would not be appropriate.

In the discussion of deterministic buildup and washoff concepts, it was pointed out that the curb solids accumulation (and by the same reasoning curb-side pollutant loads) stay more or less constant. The washoff rate is proportional to rainfall intensity and, in a cumulative manner, to the rainfall depth. This may lead to the conclusion that the EMCs under these circumstances may not vary and, in a statistical sense, may be considered as constant.

Hence the NURP findings do not a priori disprove the deterministic concepts. Furthermore, even though the street loads of pollutants tend to be inversely proportional to the imperviousness of the area, total loads from less impervious areas also include pollutants eroded from pervious surfaces, which again tends to equalize the EMCs and pollutant loads among the land use areas.

The following example documents that there is no major discrepancy between the unit loads presented in the first portion of this chapter and units loads obtained from the statistical nationwide characteristics of urban runoff.

Example 8.6: Unit Loads Based on NURP EMC Data and their Comparison with Other Estimates

Estimate annual unit loads and 90 percentile exceedence event mean concentrations (EMC) of total phosphorus and lead from an urban area. The following information is given:

Land use: residential
Percent imperviousness: 50%
Annual rainfall: 76 cm (30 in.)

From Table 8.14 the median and 90 percentile EMCs for total phosphorus are 0.33 mg/l and 0.70 mg/l, respectively. EMCs for lead are 144 μ g/l (median) and 350 μ g/l, respectively.

Solution The annual load can be estimated using the relation of the coefficient of runoff (defined as a ratio of runoff depth to the total rainfall depth) as determined by the NURP study, which is shown on Figure 8.2.

From the figure for the 50% impervious area, $CR = 0.45$ (recall from the discussion in Chapter 3 that for urban areas the runoff coefficient is approximately a measure of the degree of imperviousness).

For estimating the loads for median and 90 percentile urban sites, the mean EMCs are taken from Table 8.15.

For a median urban site:

$$\text{Average Tot. P EMC} = 0.42 \text{ mg/l}$$

$$\begin{aligned} \text{Tot. P}_{\text{mean}} \text{ (kg/ha)} &= CV * \text{rainfall volume} * \text{EMC} = 0.45 * \\ &76 \text{ (cm)} * 0.01 \text{ (cm/m)} * 10,000 \text{ (m}^2\text{/ha)} * \\ &0.42 \text{ (g/m}^3\text{)} * 0.001 \text{ (kg/g)} = 1.44 \text{ kg/ha} \end{aligned}$$

$$\text{Average Tot. Pb EMC} = 0.182 \text{ mg/l}$$

$$\begin{aligned} \text{Tot. Pb}_{\text{mean}} \text{ (kg/ha)} &= 0.45 * 76 * 0.01 * 10,000 * 0.182 * 0.001 \\ &= 0.62 \text{ kg/ha} \end{aligned}$$

For a 90 percentile urban site

$$\text{Tot. P EMC (90\%)} = 0.9 \text{ mg/l}$$

$$\begin{aligned} \text{Tot. P}_{90\%} \text{ (kg/ha)} &= 0.45 * 76 * 0.01 * 10,000 * 0.9 * 0.001 \\ &= 3.08 \text{ kg/ha} \end{aligned}$$

$$\text{Tot. Pb (90\%)} = 0.443 \text{ mg/l}$$

$$\begin{aligned} \text{Tot. Pb}_{90\%} \text{ (kg/ha)} &= 0.46 * 76 * 0.01 * 10,000 * 0.443 * 0.001 \\ &= 1.55 \text{ kg/ha} \end{aligned}$$

By comparison, the PLUARG data measured in the Great Lakes region given in Table 8.8 reported ranges of annual unit loads for urban areas of 0.3 to 4.8 kg/ha for phosphorus and 0.14 to 0.5 kg/ha for lead. The range of Tot. P unit loadings compiled by Beaulac and Reckhow (1982) for urban lands were 0.5 to 6.25 kg/ha-year (average 1 kg/ha-year). No conclusions can be drawn from this comparison.

COMBINED SEWER OVERFLOWS

Unlike the separate storm sewers that mostly contain “cleaner” surface runoff and infiltrations during wet weather, combined sewers carry a mixture of storm runoff, infiltration, and wastewater (dry-weather) flows. During wet weather, combined sewer systems may overflow,

until recently commonly without any treatment, directly to the receiving waters.

Combined sewers were originally built during the last century by connecting sewage discharges to subsurface and surface storm drainage. By the end of the nineteenth century, the need for wastewater treatment became increasingly apparent, but urban storm water was generally considered to be "clean" and suitable for the dilution of "dirty" wastewater. However, after the turn of the twentieth century most of the urban sewer systems were built separate.

In Europe, the practice of building separate systems was not as widely accepted as in the United States, and many urban sewer systems are combined. Typically in European urban areas, the city centers have combined sewers, to which separate suburban sanitary systems are connected. Because of the period in which the combined sewers were built in the United States, they tend to be located in areas that experienced major growth during the period from approximately 1850 to 1900. Major combined sewer service areas are located along the upper East Coast, in the upper Midwest, and in the far east. There are at least 1 million hectares of combined sewer service area in the United States today, located in 1100 to 1300 distinct collection systems with an average population density of 40 people per hectare and total population served of about 37,000,000 (U.S. EPA, 1978). In total, there are between 15,000 to 20,000 CSOs currently in operation (Association of Metropolitan Sewerage Agencies, 1988).

Sources and Characterization of Combined Sewer Overflows

An overflow from combined sewers is initiated when the mixed flow in the sewer exceeds the sewer capacity. To prevent backwatering into the sewers and for safe conveyance the separation of the flows into an overflow and downstream conveyance occurs in so-called *flow regulators* or *flow dividers* whose schematic is shown on Figure 8.19. There are many types of flow regulators, some of which use simple static weir controls to separate flows and some that use moving parts. Many of the existing regulators became inoperable in the past (Moffa, 1990).

Overflow frequency and duration. Combined sewer overflows (CSOs) occur only during runoff-producing rainfall events which, in general, in an average year range from 200 hours to more than 1300 hours in the United States (U.S. EPA, 1978). In Germany (Ruhr area) there are on average about 700 precipitation hours, and 1000 hours are typical for Switzerland (Novotny et al., 1989). However, during these runoff-producing precipi-

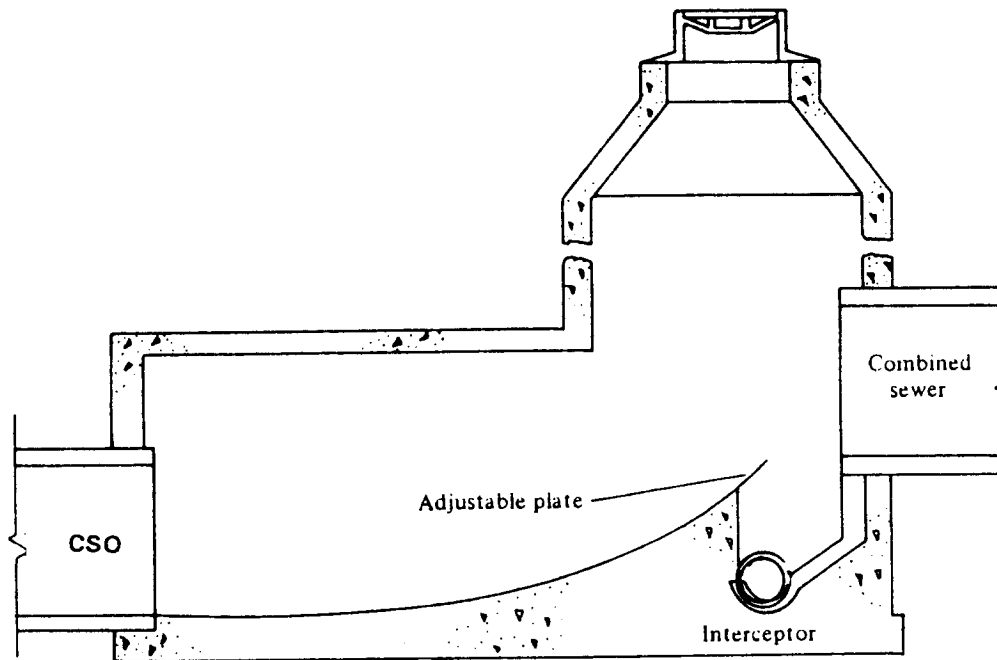


FIGURE 8.19. Leaping weir CSO flow divider (regulator).

tation hours pollutant loads may be extremely large. Combined sewer overflows containing raw wastewater with pathogenic (disease-causing) microorganisms, are repugnant and typically result in unpleasant odors. They are suspected of being a source of objectionable debris, including those illegally dumped into sewers (i.e., medical waste found on the East Coast beaches that is suspected of originating with CSOs).

Typically, urban combined sewage systems are designed to carry flow that is about four to eight times the average dry-weather (sewage) flows. Treatment plants serving these systems built based on past U.S. and European practices, were designed to handle mixed flows that were four to six times the average dry-weather flow. Simply said, any flow in excess of the treatment plant capacity must overflow either in CSOs or in a treatment plant bypass.

The rainfall that initiates overflow can then be related to the hydrological characteristics of the drainage basin, population density (per capita sewage flow), and the excess treatment plant capacity as follows. Assume that the peak surface runoff flow is adequately represented by the rational formula (see Chapter 3) or $Q_{s\text{-peak}} = CAI$, where C is the coefficient of runoff defined in Table 3.13, A is the contributing area in square meters, and I is the rainfall intensity in millimeters/hour, and $Q_{s\text{-peak}}$ is in liters/hour. However, in this concept one has to realize that the minimal CSOs causing rainfalls have relatively low intensity and, in most cases,

pervious areas do not contribute to the runoff. Thus the term $C * A$ should be replaced by the directly connected impervious area $A_{\text{connected}} = DC * (\text{Percent imperviousness}/100) * A$. Then if Q_{dw} is the dry-weather sewage flow and the treatment system capacity is $m * Q_{\text{dw}}$ (where m is typically four to six), then the rainfall intensity at which the overflows occur is

$$I_{\text{crit}} = (m - 1) \frac{Q_{\text{dw}}}{A_{\text{connected}}} \quad (8.22)$$

where I_{crit} is an average rainfall intensity at which CSOs begin in liters/ $\text{m}^2\text{-hr}$ = millimeters/hour.

The typical magnitude of the critical rainfall intensity for initiation of CSOs is evaluated in the following Example 8.7.

Example 8.7: Calculation of the Critical Rainfall Intensity for CSOs

Calculate the critical rainfall intensity for a typical urban area with a population density of 40 persons per hectare and an average dry-weather sewage flow of 300 liters per capita and day. Assume that infiltration inflow equals the dry weather flow. The capacity of the treatment plant is four times the dry-weather flow (including infiltration), average runoff coefficient for the area is 0.7, and the drainage area is 1000 ha.

Solution The dry-weather sewage flow is

$$\begin{aligned} Q_{\text{dw}} &= 40 \text{ [people/ha]} * 300 \text{ [liters/capita-day]} * 1000 \text{ [ha]} \\ &= 12 * 10^6 \text{ liters/day} \\ &= 12 * 10^6 / 24 \text{ (hr/day)} \\ &= 5 * 10^5 \text{ l/hr.} \end{aligned}$$

The total dry-weather flow (including infiltration) is thus

$$Q_{\text{dw}} = 2 * 5 * 10^5 = 10^6 \text{ l/hr}$$

Hence, the capacity of the treatment plant is

$$Q_{\text{cap}} = 4 * 10^6 \text{ l/hr}$$

Assuming $DC = 0.5$ (see Chapter 2 for the magnitude of the directly connected impervious area factor) and 70% average imperviousness, the directly connected area is

$$A_{\text{con}} = 0.5 * (70\%/100) * 1000 \text{ (ha)} * 10^4 \text{ (m}^2\text{/ha)} = 3.5 * 10^6 \text{ m}^2$$

The critical rainfall intensity is then (Eq. (8.22))

$$I_{\text{crit}} = (4 - 1) \frac{10^6 \text{ [l/hr]}}{3.5 * 10^6} = 0.85 \text{ mm/hr (=0.033 in./hr)}$$

The critical CSO-initiating rainfall intensity of about 1 mm/hr may be typical for U.S. conditions. In Europe population density and imperviousness of urban areas are higher. Consequently, the critical rainfall intensity, I_{crit} , is about 3 to 6 mm/hr in Germany and 6 to 12 mm/hr in Switzerland (Novotny et al., 1989). Typically, without CSO control, overflows from combined systems in most urban areas occur on average between 10 and 50 times per year. In Milwaukee a typical number of overflows from the combined sewer area is about 40 per year.

Water Quality Characteristics of CSOs. Although CSOs have received scrutiny by researchers and the EPA, and a large data base of quality data has been assembled, no statistical analysis comparable to NURP studies is available. Hence, only ranges and arithmetic averages have been reported. There is a great likelihood that the statistical characteristics of EMCs from combined sewer overflows may also follow the log-normal distribution, and similar conclusions could be drawn as to the variability as were for storm water quality characteristics. This is especially true for the constituents that almost solely originate from storm water runoff, such as toxic metals and organics. Table 8.16 shows some average quality characteristics of CSOs.

There is ample evidence indicating that the first flush effect is more pronounced in the CSO discharges. Organic solids accumulate and sewer slime grows during dry periods; hence, far more pollutants accumulate in combined sewers than in storm sewers, which are idle and mostly dry between rainfall events.

TABLE 8.16 Nationwide Average Characteristics of CSOs

Parameter	Average Concentration
BOD ₅ (mg/l)	115
Suspended solids (mg/l)	370
Total nitrogen (mg/l)	9–10
Phosphate (mg/l)	1.9
Lead (mg/l)	0.37
Total coliforms (MPN/100 ml)	10 ² –10 ⁴

Source: From U.S. EPA (1978).

Nationwide assessment has also shown that the annual load from CSOs for BOD are about the same as those for the entire year from the secondary treatment plants treating wastewater from the same area. The annual discharges of suspended solids and lead are approximately 15 times higher from combined sewer overflow than from secondary treatment plant effluents. On the other hand, annual loads of total N and phosphates from secondary treatment plant effluents are 4 and 7 times the discharge from combined sewer overflows, respectively.

References

- American Public Works Association (1969). *Water Pollution Aspects of Urban Runoff*, WP-30-15, U.S. Dept. of Interior, FWPCA (now the EPA), Washington, DC.
- Anonymous (1981). *Fugitive Dust Emissions: Their Sources and Their Control in Milwaukee's Menomonee River Valley*, Department of City Development, City of Milwaukee, WI.
- Association of Metropolitan Sewerage Agencies (1988). *Draft CSO Permitting Strategy, August 1988*, Bulletin No. GB88-22, AMSA, Washington, DC.
- Athayde, D. N., C. F. Myers, and P. Tobin (1986). EPA's perspective of urban nonpoint sources, in *Urban Runoff Quality—Impact and Quality Enhancement Technology*, Proceedings, Engineering Foundation Conference, American Society of Chemical Engineers, pp. 217–225.
- Balmér, P., P. A. Malmqvist, and A. Sjöberg, eds. (1984). *Proceedings, Third International Conference on Urban Storm Drainage*, Chalmers University of Technology, Göteborg, Sweden.
- Bannerman, R. (1991). *Pollutants in Wisconsin Stormwater*, unpublished report, Wisconsin Department of Natural Resources, Madison, WI.
- Bannerman, R., K. Baun, M. Bohm, P. E. Hughes, and D. A. Graczyk (1984). *Evaluation of Urban Nonpoint Source Pollution Management in Milwaukee County, Wisconsin*, Report No. PB84-114164, U.S. Environmental Protection Agency, Region V, Chicago, IL.
- Beaulac, M. N., and K. H. Reckhow (1982). An examination of land use—nutrient export relationships, *Water Resour. Bull.* **18**(6):1013–1024.
- Bengtsson, L. (1982). Snowmelt-generated runoff in urban areas, in *Urban Stormwater Hydraulics and Hydrology*, B. C. Yen, ed. Water Resources Publ., Littleton, CO, pp. 444–451.
- Brown, R. G. (1988). Effects of precipitation and land use on storm runoff, *Water Resour. Bull.* **24**(2):421–426.
- Carey, A. E. (1979). Monitoring pesticides in agricultural and urban soils of the United States, *Pesticides Monit. J.* **13**(1):23–27.
- Cowherd, C., Jr., et al. (1977). *Quantification of Dust Entrainment from Paved Roadways*, EPA 450/3-77-027, U.S. Environmental Protection Agency, Research Triangle Park, NC.
- Ellis, J. B. (1986). Pollutional aspects of urban runoff, in *Urban Runoff Pollution*,

- H. C. Torno, J. Marsalek, and M. Desbordes, eds., Springer Verlag, Berlin, New York, pp. 1–38.
- Fam, S., M. K. Stenstrom, and G. Silverman (1987). Hydrocarbons in urban runoff, *J. Environ. Eng. Div. ASCE* **113**(5):1032–1046.
- Field, R., et al. (1974). Water pollution and associated effects from street salting, *J. Environ. Eng. Div. ASCE* **100**(EE2):459–477.
- Förster, J. (1990). Roof runoff: A source of pollutants in urban storm drainage systems, in *Proceedings, Fifth International Conference on Urban Storm Drainage*, Y. Iwasa and T. Sueishi, eds., Osaka University, Japan, pp. 469–474.
- Glenne, B. (1984). Simulation of water pollution generation and abatement on suburban watersheds, *Water Resour. Bull.* **20**(2):211–217.
- Gujer, W., and V. Krejci (1987) *Topics in Urban Storm Water Quality, Planning and Management*, in *Proceedings, Fourth International Conference on Urban Storm Drainage*. IAHR-IAWPRC École Polytechnique Fédérale, Lausanne, Switzerland, pp. 1–20.
- Halverson, H. G., D. R. DeWalle, and W. E. Sharpe (1984). Contribution of precipitation to quality of urban storm runoff, *Water Resour. Bull.* **20**(6): 859–864.
- Halverson, H. G., D. R. DeWalle, W. E. Sharpe, and D. L. Wirries (1982). Runoff contaminants from natural and manmade surfaces in a nonindustrial urban area, in *Proceedings, 1982 International Symposium on Urban Hydrology, Hydraulics and Sediment Control*, University of Kentucky, Lexington, pp. 233–238.
- Heaney, J. P., and W. C. Huber (1973). *Stormwater Management Model. Refinements, Testing and Decision-making*, Department of Environmental Engineering & Science, University of Florida, Gainesville, FL.
- Hoffman, E. J. (1985). Urban runoff pollutant inputs to Narragansett Bay: Comparison to point sources, in *Perspectives on Nonpoint Pollution*, Proceedings, National Conference, EPA 440/5-85-001, U.S. Environmental Protection Agency, Washington, DC, pp. 159–164.
- Hoffman, E. J., et al. (1984). Urban runoff as a source of polycyclic aromatics to coastal waters, *Environ. Sci. Technol.* **18**:580–587.
- Huber, W. C. (1986). Deterministic modeling of urban runoff quality, in *Urban Runoff Pollution*, H. C. Torno, J. Marsalek, and M. Desbordes, eds., Springer Verlag, Berlin, pp. 167–242.
- Hydrologic Engineering Center (1975). *Urban Stormwater Runoff—STORM*, U.S. Army Corps of Engineers, Davis, CA.
- Iwasa, Y., and T. Sueishi, eds. (1990). *Proceedings, Fifth International Conference on Urban Storm Drainage*, IAHR-IAWPRC, Suita-Osaka, University of Osaka, Japan.
- James, W., and S. Boregowda (1986). Continuous mass balance of pollutant build-up processes, in *Urban Runoff Pollution*, H. Torno, J. Marsalek, and M. Desbordes, eds., Springer Verlag, Berlin, pp. 243–271.
- Krejci, V., L. Dauber, B. Novak, and W. Gujer (1987). Contribution of different sources to pollutant loads in combined sewers, in *Proceedings, Fourth Inter-*

- national Conference on Urban Storm Drainage*, W. Guejer and V. Krejci, eds., IAHR-IAWPRC, École Polytechnique Fédérale, Lausanne, Switzerland, pp. 34–39.
- Lager, J. A., W. G. Smith, W. G. Lynard, R. M. Finn, and E. J. Finnemore (1977). *Urban Stormwater Management and Technology: Update and User's Guide*, EPA-600/8-77-014, U.S. Environmental Protection Agency, Cincinnati, OH.
- McComas, M. R., G. D. Cooke, and R. H. Kennedy (1976). A comparison of phosphorus and water contributions by snowfall and rain in northern Ohio, *Water Resour. Bull.* **12**(3):519–527.
- McElroy, A. D., S. Y. Chiu, J. W. Nebgen, A. Aleti, and F. W. Bennett (1976). *Loading Functions for Assessment of Water Pollution from Nonpoint Sources*, Office of Research and Development, U.S. Environmental Protection Agency, EPA-600/2-76-151, Washington, DC.
- Moffa, P. E. (1990). *Control and Treatment of Combined Sewer Overflows*, Van Nostrand Reinhold, New York.
- Myers, C. F., J. Meek, J. S. Tuller, and A. Weinberg (1985). Nonpoint sources of water pollution, *J. Soil Water Conserv.* **40**(1):14–18.
- Marsalek, J. (1978). *Pollution Due to Urban Runoff: Unit Loads and Abatement Measures*, Pollution from Land Use Activities Reference Group, International Joint Commission, Windsor, Ontario.
- Marsalek, J. (1986). Toxic contaminants in urban runoff, in *Urban Runoff Pollution*, H. Torno, J. Marsalek, and M. Desbordes, eds., Springer Verlag, Berlin, pp. 39–57.
- Marsalek, J. (1990). PAH transport by urban runoff from an industrial city, in *Proceedings, Fifth International Conference on Urban Storm Drainage*, Y. Iwasa and T. Sueishi, eds., University of Osaka, Japan, pp. 481–486.
- Ng, H. Y. F. (1987). Rainwater contribution to the dissolved chemistry of storm runoff, in *Urban Storm Water Quality, Planning and Management*, W. Gujer and V. Krejci, eds., IAHR-IAWPRC, École Polytechnique Fédérale, Lausanne, Switzerland, pp. 21–26.
- Novotny, V. (1987). *Effect of Pollutants from Snow and Ice on Quality of Water from Urban Drainage Basins*, PB87-162509, Dept. of Civil Eng., Marquette University, Milwaukee, WI (available from NTIS).
- Novotny, V. (1988). Modeling urban runoff pollution during winter and off-winter periods, in *Advances in Environmental Modelling*, A. Marani, ed., Elsevier, Amsterdam, pp. 43–58.
- Novotny, V., and G. Chesters (1981). *Handbook of Nonpoint Pollution: Sources and Management*, Van Nostrand Reinhold, New York.
- Novotny, V., and G. W. Kincaid (1982). Acidity of urban precipitation and its buffering during overland flow, in *Urban Stormwater Quality, Management and Planning*, B. C. Yen, ed., Water Resources Publ., Littleton, CO, pp. 1–9.
- Novotny, V., K. R. Imhoff, M. Othof, and P. Krenkel (1989). *Karl Imhoff's Handbook of Urban Drainage and Wastewater Disposal*, Wiley Interscience, New York.

- Novotny, V., H. M. Sung, R. Bannerman, and K. Baum (1985). Estimating nonpoint pollution from small urban watersheds, *J. WPCF* 57(4):339–348.
- Oberts, G. L. (1982a). *Water Resources Management: Nonpoint Source Pollution Technical Report*, PUB-10-82-016, Metropolitan Council, St. Paul, MN, Environmental Protection Agency, Chicago, IL, Region V (available from NTIS), 260p.
- Oberts, G. L. (1982b). Impact of wetlands on nonpoint source pollution. *Proceedings, International Symposium on Urban Hydrology, Hydraulics and Sediment Control*, University of Kentucky, Lexington, KY, pp. 225–232.
- Oberts, G. L. (1986). Pollutants associated with sand and salt applied to roads in Minnesota, *Water Resour. Bull.* 22(3):479–484.
- Oberts, G. L. (1990). Design considerations for management of urban runoff in wintry conditions, *Proceedings, International Conference on Urban Hydrology under Wintry Conditions*, Narvik, Norway, March 19–21.
- Olivieri, V. P., C. W. Kruse, K. Kawata, and J. E. Smith (1977). *Microorganisms in Urban Stormwater*, EPA-600/2-77-087, Municipal Environmental Research Laboratory, U.S. Environmental Protection Agency, Cincinnati, OH.
- Pisano, W. C., G. L. Aronson, C. S. Queiroz, F. C. Blanc, and J. C. O'Shaughnessy (1979). *Dry-weather Deposition and Flushing for Combined Sewer Overflow Pollution Control*, EPA-600/2-79-133, U.S. Environmental Protection Agency, Cincinnati, OH.
- Pitt, R. (1979). *Demonstration of Nonpoint Pollution Abatement through Improved Street Cleaning Practices*, EPA 600/2-79-161, U.S. Environmental Protection Agency, Cincinnati, OH.
- Pitt, R. (1988). Asbestos as an urban area pollutant, *J. WPCF* 60(11):1993–2001.
- Pitt, R., and P. Barron (1989). *Assessment of Urban and Industrial Stormwater Runoff Toxicity and the Evaluation/Development of Treatment for Runoff Toxicity Abatement—Phase I*, Storm and Combined Sewer Pollution Program, U.S. Environmental Protection Agency, Edison, NJ.
- Pitt, R., M. Lalor, and G. Driscoll (1990). *Assessment of Non-stormwater Discharges into Separate Storm Drainage Networks—Phase I: Manual of Practice*, Storm and Combined Sewer Pollution Program, U.S. Environmental Protection Agency, Edison, NJ.
- Pitt, R., R. Field, M. Lalor, and G. Driscoll (1990). Analysis of cross-connections and storm drainage, in *Proceedings, Urban Stormwater Enhancement Source Control, Retrofitting and Combined Sewer Technology*, Davos, Switzerland, Engineering Foundation of ASCE, New York.
- Randall, C. W., T. J. Grizzard, D. R. Helsel, and D. M. Griffin (1982). Comparison of pollutant mass loads in precipitation and runoff in urban area, in *Urban Stormwater Quality, Management and Planning*, B. C. Yen, ed., Water Resources Publ., Littleton, CO, pp. 29–38.
- Sartor, J. D., and G. B. Boyd (1972). *Water Pollution Aspects of Street Surface Contamination*, EPA R2-72-081, U.S. Environmental Protection Agency, Washington, DC.
- Sartor, J. D., G. B. Boyd, and F. J. Agardy (1974). Water pollution aspects of street surface contamination, *J. WPCF* 46:458–465.

- Schmidt, S. D., and D. R. Spencer (1986). The magnitude of improper waste discharges in an urban stormwater system, *J. WPCF* **58**(11):744-748.
- Shaheen, D. G. (1975). *Contributions of Urban Roadway Usage to Water Pollution*, EPA 600/2-75-004, Office of Research and Development, U.S. Environmental Protection Agency, Washington, DC.
- Sonzogni, W. C., et al. (1980). Pollution from land runoff, *Environ. Sci. Technol.* **14**(2):148-153.
- Strecker, E. W., E. D. Driscoll, P. E. Shelley, and D. R. Gaboury (1987). Characterization of pollutant loadings from highway runoff in the USA, in *Urban Storm Water Quality, Planning and Management*, W. Gujer and V. Krejci, eds., IAHR-IAWPRC, École Polytechnique Fédérale, Lausanne, Switzerland, pp. 85-90.
- Sutherland, R. C., and R. H. McCuen (1978). Simulation of urban nonpoint source pollution, *Water Resour. Bull.* **14**:409-428.
- Torno, H. C., J. Marsalek, and M. Desbordes (1986). *Urban Runoff Pollution*, Springer Verlag, Berlin, New York.
- U.S. Environmental Protection Agency (1971). *Stormwater Management Model*, EPA 11024D00C 07/71 to 1102D0C10/71, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1977). *Control of Reentrained Dust from Paved Streets*, EPA 905/9-77-007, U.S. EPA, Kansas City, MO.
- U.S. Environmental Protection Agency (1978). *Report to Congress on Control of Combined Sewer Overflow in the United States*, EPA 430/9-78-006, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1983). *Results of the Nationwide Urban Runoff Program. Vol. I. Final Report, Water Planning Division*, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1984). *Report to Congress: Nonpoint Source Pollution in the U.S.*, Office of Water Program Operations, Synectics Group, Inc., U.S. EPA, Washington, DC (available from NTIS).
- U.S. Environmental Protection Agency (1990a). *National Water Quality Inventory—1988 Report to Congress*, EPA 440-4-90-003, U.S. EPA, Office of Water, Washington DC.
- U.S. Environmental Protection Agency (1990b). *Urban Targeting and BMP Selection. An Information and Guidance Manual for State NPS Program Staff Engineers and Managers*, EPA 68-C8-0034, U.S. EPA, Washington, DC.
- Uttormark, P. D., J. D. Chapin, and K. M. Green (1974). *Estimating Nutrient Loadings of Lakes from Nonpoint Sources*, EPA 660/3-74-02C, Office of Research and Monitoring, U.S. Environmental Protection Agency, Washington, DC.
- Waller, D. H., and W. C. Hart (1986). Solids, nutrients, and chlorides in urban runoff, in *Urban Runoff Pollution*, H. C. Torno, J. Marsalek, and M. Desbordes, eds., Springer Verlag, Berlin, New York, pp. 59-85.
- Westerström, G. (1990). Pilot study of the chemical characteristics of urban snow meltwater, in *Proceedings, Fifth International Conference on Urban Storm Drainage*, University of Osaka, Japan, pp. 305-310.

- Whipple, W., et al. (1978). Effect of storm frequency on pollution from urban runoff, *J. Water Pollut. Control Fed.* **50**:974-980.
- Whipple, W., Jr., and J. V. Hunter (1979). Petroleum by hydrocarbons in urban runoff, *Water Resour. Bull.* **15**(4):1096-1105.
- Whipple, W., Jr., and J. DiLouie (1981). Coping with increased stream erosion in urbanizing areas, *Water Resour. Res.* **17**(5):1561-1564.
- Whipple, W., Jr., J. V. Hunter, and S. L. Yu (1978). Runoff pollution from multiple family housing, *Water Resour. Bull.* **14**(2):288-301.
- Yalin, M. S. (1963). An expression for bed load transportation, *J. Hydraul. Div. ASCE* **89**:221-250.
- Yen, B. C. (1982). *Urban Stormwater Quality, Management and Planning*, Water Resources Publ., Littleton, CO.
- Zison, S. W. (1980). *Sediment-Pollutant Relationships in Runoff from Selected Agricultural, Suburban, and Urban Watersheds. A Statistical Correlation Study* (Final rept. Sep. 77-Sep. 78), EPA 600/3-80-022, Tetra Tech, Inc., Lafayette, CA, Environmental Research Lab., Athens, GA (available from NTIS), 150p.

Modeling and Monitoring Diffuse Pollution

Most natural hydrologic phenomena are so complex that they are beyond comprehension, or exact laws governing such phenomena have not been fully discovered. Before such laws can ever be found, complicated hydrologic phenomena (the prototype) can only be approximated by modeling.

Ven Te Chow

GENERAL CONCEPTS

As a result of research in the 1970s and early 1980s, a large number of models have been developed by various agencies, universities, and researchers throughout the world that could be used for modeling diffuse pollution and for assessment of remedial measures. Such models range from simple applications of basic hydrological procedures with added unit pollutant loads (see Chapter 1 for a definition and discussion of the unit load concept), to highly complex hydrological surface and ground-water runoff quantity and quality models. Models that in the past could be run only on large mainframe computers are now available for personal minicomputers. Several are available from agencies (for example, the U.S. EPA Research Laboratory in Athens, Georgia) and software vendors. As a matter of fact, the availability of personal computers that now have the same capability, or even surpass, medium mainframe computers of ten years ago, have meant that almost every engineer or planner can now possess a computer capable of running very sophisticated hydrological models. On-line interactions of user and computer graphics are being

incorporated into the available models. Use of geographic information systems (GIS) and CAD programs for diffuse pollution modeling is now evolving.

Diffuse pollution hydrological models are a part of *loading models* that represent and simulate the generation and movement of water and its pollution content from the point of origin (source area) to a place of treatment and/or disposal (discharge) into receiving waters. These models may be used for planning and design alone, or may interface with receiving water quality models that assess the impact of nonpoint pollution on the aquatic biota and beneficial downstream uses. In most cases, the loading models provide concentrations and flow rates (pollutographs and hydrographs) and/or unit and total loads of pollutants. This chapter briefly describes the concepts and basic approaches of loading models. The processes included in the design of these models are described in pertinent chapters throughout this book.

Stream and receiving water quality models simulate the movement of materials through streams, river impoundments, estuaries, or near-shore ocean dispersion. The concepts of these models are introduced in Chapter 12. Reviews of the available runoff-quality models applicable to diffuse-pollution modeling of urban and agricultural watersheds have been prepared by Haan, Johnson, and Brakensiek (1982), Giorgini and Zingales (1986), Novotny (1986, 1988) and for the U.S. EPA by Donigian and Huber (1990) and Huber (1990).

Types of Models

Every model can be represented by the "black box" concept (Fig. 9.1). The model, just like the real system, produces output to various inputs. The structure of the model is always a very simplified version of the interactions and reactions taking place in the real system. The variables describing the physical state of the system are called *system parameters*; the variables affecting the state of the system are *state variables*. Some input variables may be considered state variables and vice versa. Watershed size, slope and roughness characteristics, erodibility, and texture of soils are examples of system parameters, while temperature and solar radiation and vegetation cover may be considered state variables. Rain, atmospheric fallout, and daily litter inputs on impervious areas can be considered inputs to most diffuse-pollution-water quality models. Note that a time series of unknown random perturbations can also be considered an input series, which is a fundamental premise of *stochastic water* quality models.

In his now classic paper on hydrologic modeling, Chow (1972) clas-

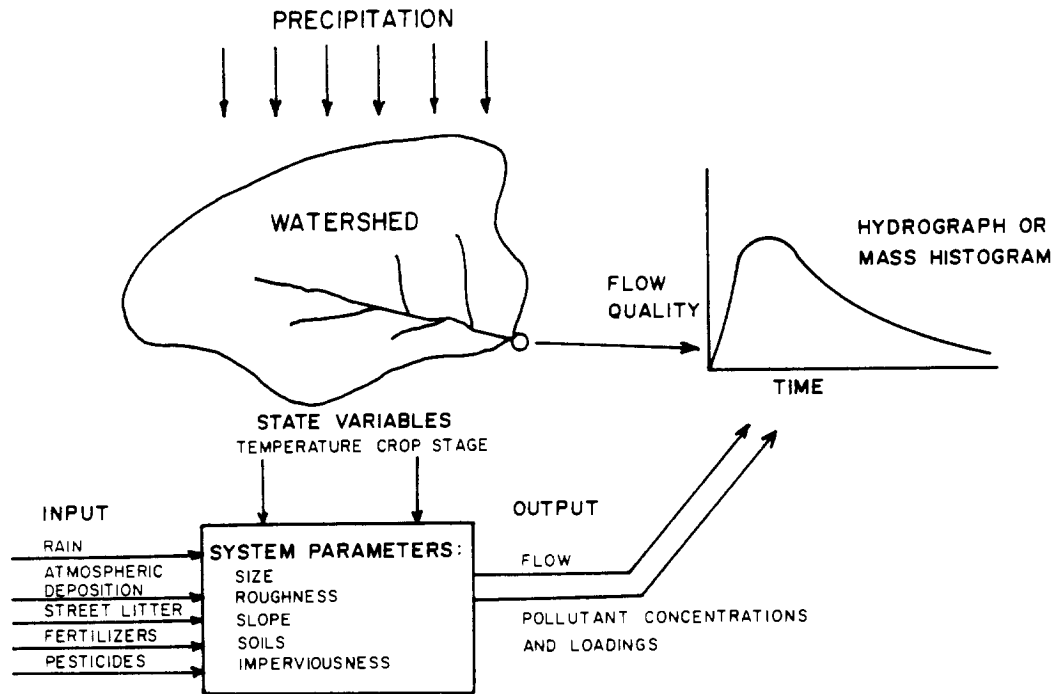


FIGURE 9.1. "Black box" concept of the watershed.

sified models into a number of categories, including physical, deterministic, and stochastic models. Physical models are of little importance to diffuse pollution modeling although they are used extensively for studying individual processes such as infiltration, soil water movement, and adsorption-desorption. Abstract models attempt to represent the prototype theoretically in mathematical form. These models replace the features of the system by relevant mathematical relationships.

Following and expanding Chow's definitions, the diffuse-pollution models can be broadly divided into the following basic groups (Fig. 9.2):

- Simple statistical routines and screening models
- Deterministic hydrologic models
- Stochastic models

A *deterministic model* ignores the input of random perturbations and random variations of system parameters and state variables. Such models for a given set of inputs provide only one set of outputs, while the output from stochastic models is often expressed in terms of mean and probabilistic ranges.

There are basically two approaches to modeling diffuse pollution. The more widely used are *lumped-parameter* models, while some more complex deterministic models are based on the *distributed-parameter* concept.

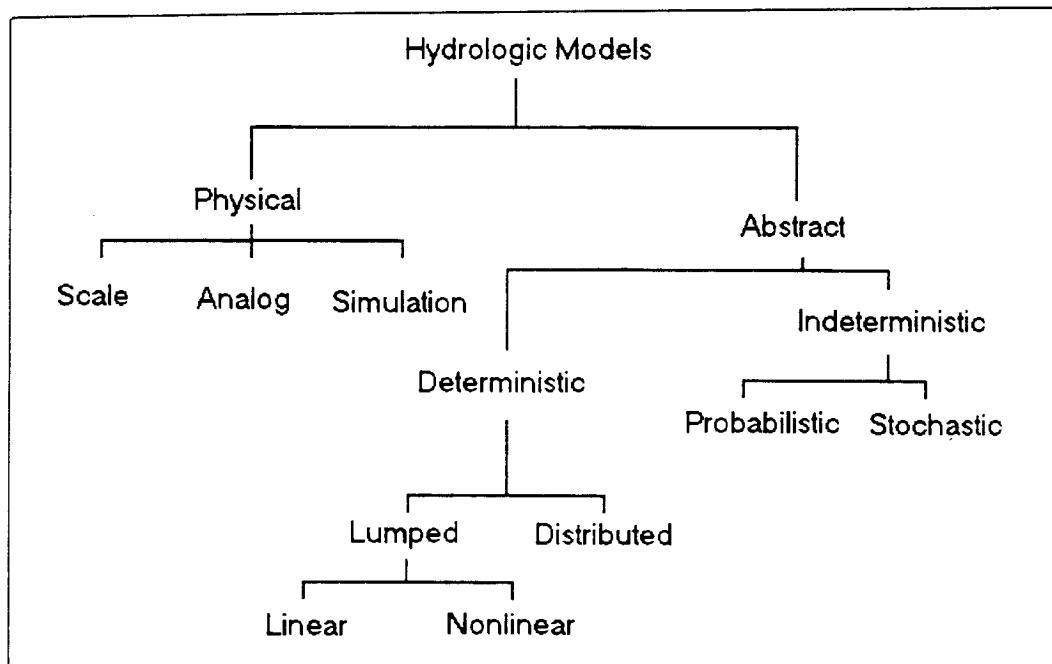


FIGURE 9.2. Classification of watershed hydrologic models. (After Chow, 1972.)

Lumped-parameter models can be both stochastic and deterministic (Fig. 9.3). The lumped-parameter models treat the watershed or a significant portion of it as one unit. The various characteristics of the watershed are often lumped together by an empirical equation, and the final form and magnitude of the parameters are simplified to represent the modeled unit as a uniform system. The coefficients and system parameters for each homogenous unit are often determined by calibration by comparing the response of the model with extensive field data. For such models the input-output relationship can be represented as

$$Y = \phi X \quad (9.1)$$

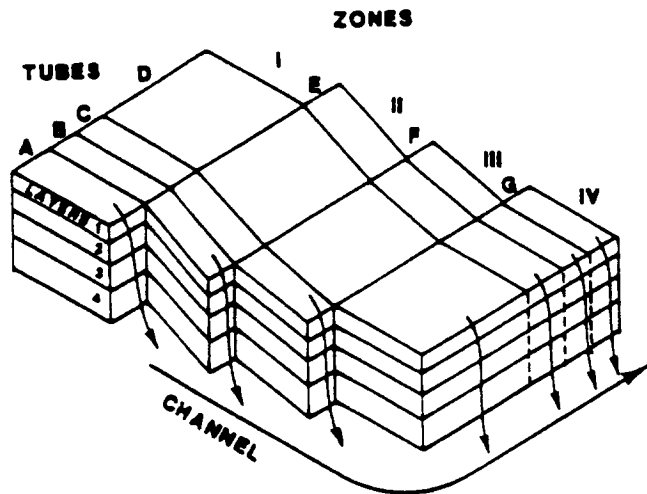
where

X = input vector (single or multivariate)

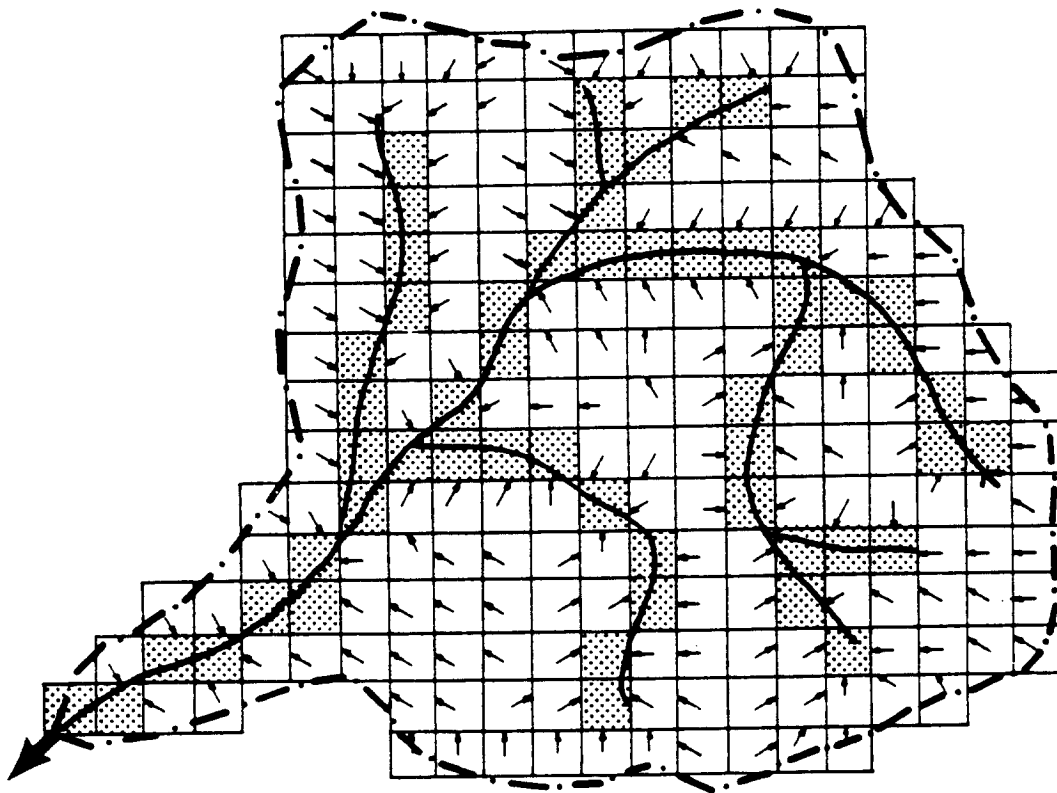
Y = output vector (single or multivariate)

ϕ = simple or complex transfer function

According to this definition, a mathematical model is itself a computerized complex transfer function. Note also that Equation (9.1) does not mean that the output vector, Y , is obtained by multiplication of the input vector, X , by a transfer function, ϕ . This simplified concept may involve complex



Lumped Parameter Model Concept



Distributed Parameter Model Concept

FIGURE 9.3. Modeling concepts.

mathematical operations, such as integrations, finite-difference approximations, and matrix inversions.

The models, once they have been calibrated and verified, can produce long time-series of outputs reflecting different hydrologic and meteorologic conditions (input variables). However, it has to be realized that a true time series of input and output parameters also contains random fluctuations that have been ignored by the deterministic nature of the model. Therefore, using statistical routines to estimate probabilistic characteristics of a time series produced by a deterministic model is inappropriate and may provide erroneous information about the statistical nature of the time series or of the modeled phenomenon. This is explained further in the section on stochastic modeling.

The distributed-parameter approach involves dividing the watershed into smaller homogenous units—areal elements—with uniform characteristics (soil, imperviousness, crop, slope). Each areal unit is described individually by a set of differential mass-balance equations. The input to each unit consists of the distributed inputs, such as rainfall and atmospheric deposition, plus output from upstream adjacent units. The mass-balance equations for the entire system are then solved simultaneously over a small computational time element Δt .

Theoretically, the lumped-parameter model can provide only one output location, while outputs can be obtained throughout the system from distributed-parameter models, that is, from each modeled subunit. Distributed-parameter models require larger computer storage for performing comparable modeling tasks and an extensive detailed description of system parameters, which must be provided for each element. However, changes in the watershed characteristics and their effect on the output can be modeled easily and more effectively. The distributed-parameter models are also more suitable for geographic information systems (GIS) and computer-aided design (CAD) modeling.

Some larger hydrological models allow division of the watershed into a smaller number of homogenous land segments, which then become essentially lumped-parameter subwatersheds. The output is then obtained by summing up the individual outputs from the subwatersheds, and not by simultaneous mass-balance solutions of a large number of areal elements. Nevertheless, the overall character of these more complex, spatially variant models are still more lumped in character than distributed-parameter models.

In size, the watershed models may range from small uniform segments of less than 1 ha to entire watersheds with the area of several hundred km². The U.S. Environmental Protection Agency (Sauders, 1976) has divided the loading models into basin-scale models (>500 km²), areawide

models (55 to 500 km²), and small watersheds or fields (<55 km²). Even the smallest (55 km²) scale may be too large for some more detailed hydrological models, and introduction of a subbasin scale of less than 5 km² would be appropriate. Recall from Chapter 3 that watersheds with an area of less than 5 km² can be represented by a single-reservoir hydrological model, while for larger scale modeling more complex watershed representations are needed. A trade-off always exists between the size of the modeled area and the detail and reliability of the model.

Models can be designed or run on an *event* or *continuous* basis. Discrete-event models simulate the response of a watershed to a major rainfall or snowfall event. The principal advantage of event modeling over continuous simulation is that it requires little meteorological data and can be operated with shorter computer run time. The principal disadvantage of event modeling is that it requires specification of the design storm and antecedent moisture conditions, thereby assuming equivalence between the recurrence interval of the design storm with the recurrence interval of the runoff. This disadvantage is more pronounced when modeling more pervious watersheds than in modeling impervious urban basins.

Continuous process modeling sequentially simulates processes such as precipitation, available surface storage, snow accumulation and melt, evapotranspiration, soil moisture, surface runoff, infiltration, soil water movement, pollutant accumulation, erosion, and possibly other processes. Such models typically operate on a time interval ranging from a day to a fraction of an hour, and continuously balance water and pollutant mass in the system.

The principal advantage of continuous modeling is that it provides long time-series of water and pollutant loadings. Some users of such models attempted to statistically analyze the output time series as to the frequency and occurrence of runoff and pollutant load events. As was pointed out in the preceding discussion and will be further elaborated in the subsequent section on stochastic models, such statistical analyses using outputs from deterministic models are not proper and may provide erroneous statistical characteristics. A principal disadvantage of continuous modeling is that it requires long simulation runs, thus imposing restrictions on the number of alternatives that can be investigated. It also requires historical data on precipitation, often in less than hourly intervals, which is not always available.

Reliability and Usefulness of the Models

The epigraph at the beginning of this chapter introduced the dilemma of hydrologic modeling. It must be understood that as with any simulation of

“real-world” systems, mathematical models are only a rough approximation. A computer model is the formulation into a computer language of the modeler’s concept of the physical system and processes. Models are simplifications of the real system, and the degree of simplification may be the result of the modeler’s understanding of the processes involved in the system or the desired accuracy or the purpose of the model itself. For example, a scientist conducting research on infiltration of rainfall into the soil might very precisely describe mathematically the infiltration process and generation of runoff volume, but might grossly oversimplify the movement of surface runoff rates and the erosion process (Knisel, 1985).

Hence, the accuracy and reliability of models are limited. Although some models represent the best available technology for analysis of environmental systems, a common error made by many planners is that they accept simulation results as true and absolute for unknown conditions. In order to avoid disappointments and court challenges, users should be aware of the limited accuracy of the models. Furthermore, as stated previously, deterministic models that are prevalent in today’s practice inherently neglect random variations in the input and output series. Such variations can sometimes overwhelm deterministic interrelationships. Also the measured data contain inherent (systematic) and random measurement errors. Often measured data are not available at the location of interest and must be obtained by extrapolation from nearby observations.

The most accurate deterministic models (\pm few percent) are hydrologic models simulating runoff from small, uniform, impervious areas; the least reliable (an order of magnitude or more) are water quality models for large watersheds. Figure 9.4 shows the approximate accuracy of modeling water and pollution loads from diffuse sources using deterministic models. It should be noted that for simulating pollutant transport, hydrology must be calibrated and determined first, followed by sediment, and finally pollutant transport. Any errors that appear in the hydrologic or erosion component will be transferred and magnified in all dependent follow-up components. The highest error associated with bacterial estimations is caused primarily by analytical techniques and to a lesser degree by modeling reliability.

Since stochastic models provide a measure of error (a time series of errors between predicted and calculated output series), the reliability of the models can be accurately estimated (an opposite of reliability is the risk of failure of the model, or $p_r = 1 - p_f$, where p_r is reliability and p_f is probability of failure). A similar but inferior series of errors can be obtained by comparing the resulting output time series of deterministic models with the measured data. However, such series may still have

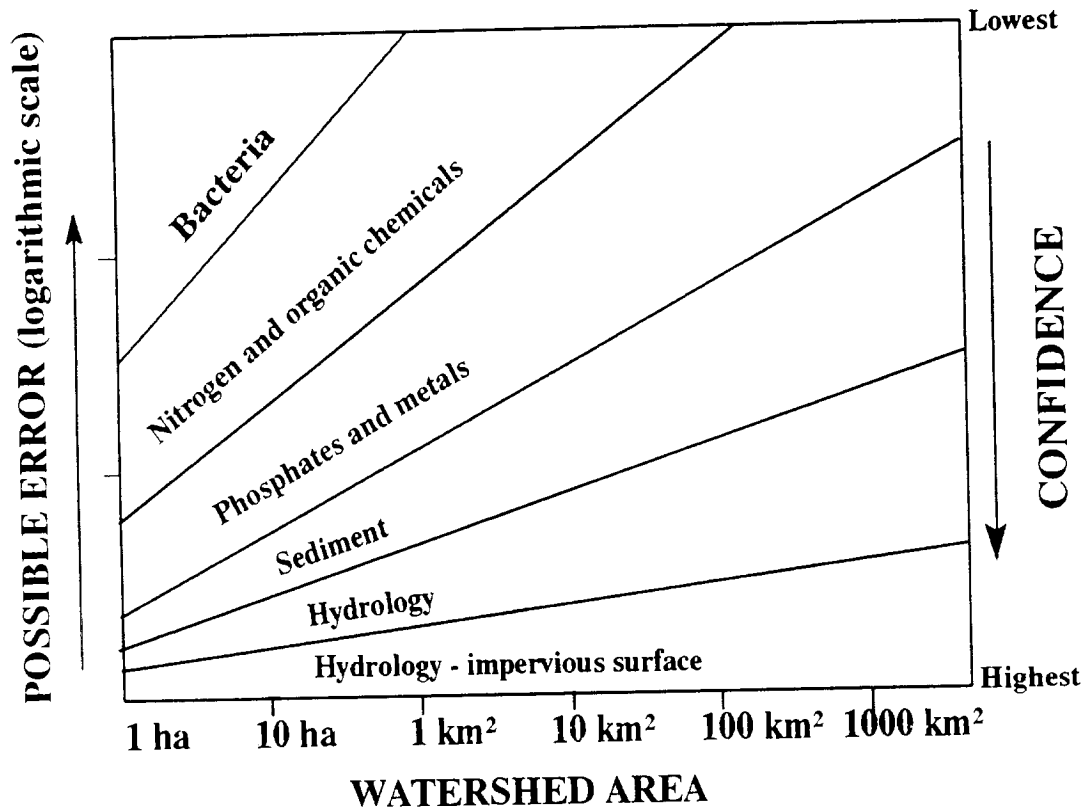


FIGURE 9.4. Relative accuracy and reliability of hydrologic models of diffuse pollution. Accuracy and reliability decreases with the increased complexity and size of the modelled system.

“hidden” deterministic and stochastic subprocesses that have not been included in the model. This will be further elaborated in the section titled “Stochastic and Neural Network Models.”

In spite of the errors involved in modeling such complex environmental systems as diffuse pollution, the model as a planning tool cannot be replaced by any “rule-of-thumb” approach. The use of models is beneficial and greatly enhances the planning process for the following reasons:

1. Models can provide a forecast of impact of planned actions on water quality and pollution loadings.
2. Models provide an understanding of the processes involved in pollution generation from nonpoint sources.
3. The data base necessary to construct and calibrate the model is useful for other planning activities. Many problems will be answered or become clearer just by evaluating the data and compiling them into an appropriate input format.

4. Critical processes and areas of concern can be delineated and detected by modeling.
5. Regulators almost always require proof of water quality impact during conditions for which monitoring data may not be available, especially in cases when some action is in the planning stage and has not been implemented. Such impact can only be established by modeling.
6. Models can be updated continuously according to the state of the art of modeling technology and understanding of the modeled process.
7. Models can generate numerous alternatives and their impact on the environment according to the specifications. Various strategies can be investigated, and the impact of remedial measures can be evaluated.
8. Although the absolute accuracy of the outputs from the model is limited and sometimes even small, a comparison and ranking of outputs for various alternative remedial measures are commonly reliable and in most cases adequate.
9. Models can estimate and analyze trade-offs between planning objectives. A system providing the lowest pollutant load may not be optional for other objectives such as the enhancement or even viability of agricultural production (e.g., limiting or eliminating pesticide and fertilizer use may result in significant yield reduction; urban development versus no future development). If the environmental objective is known, the alternative to achieve it can be measured in terms of economic efficiency by considering the willingness of those involved by the measures to pay for the consequences. If there is a financial limit, it must be treated as a constraint.
10. Water quality loading models are now a required and integral part of the permit application for discharging urban and some agricultural runoff into receiving water bodies, as specified in the United States by the Clean Water Act.

Selection of the Models

As stated before a large number of models were developed in the 1970s and early 1980s. For a review of models developed in this "model creation period" the reader is referred to papers by Renard, Rawls, and Fogel (1982), Huber (1986, 1990), and other reviews by EPA and other agencies. Many mainframe computer models have been converted for use on small personal computers and are available from agencies, universities, and software vendors. However, as pointed out by Knisel (1985), who is one of the most prominent model creators, "the burden is upon the user

to decide which model is appropriate for the problem on hand. The user must learn the model concepts, assumptions, and limitations and whether or not it will adequately treat the problem of concern." Knisel also correctly stated that "a person really cannot learn a model until they actually run it, get familiar with it, and feel comfortable in what they are doing."

Potential users should be well aware of and know by reputation the model creators, and should obtain references just as they would when purchasing any other goods. Purchasing and using anonymous models from software vendors without learning about the model creators and their reputation or acquiring models that have not been extensively tested and used by others is not wise. Model selection is an important part of the modeling process. Barnwell and Krenkel (1982) delineated the following level-of-use for the models: screening, planning, and design. Sutherland (1980) and Novotny (1986) divided diffuse pollution models into five categories:

- I. *Simple statistical procedures and unit loads* with no interactions among the processes
- II. *Simplified procedures* with some interactions among the processes
- III. *Simplified deterministic models*, either event oriented or continuous
- IV. *Sophisticated (detailed) event simulation models*
- V. *Sophisticated continuous models*

Hence, several models may be used in the modeling process, as shown on Figure 9.5. In the process:

1. Overview statistical and simplified screening procedures (Levels I or II) will identify the problem areas.
2. Detailed continuous models (Level III or V) can be used for obtaining time series of simulated water and pollutant loads and for the screening of various management practices and scenarios. Such modeling activities must consider both the source strength (pollutant emission rates) and delivery.
3. A detailed event-oriented model (Level IV) can be used to finalize the design of some technically complex structural and nonstructural management practices and measures selected and evaluated in Phase 2. Event-oriented models should not be used for evaluating the impact of diffuse sources on receiving water bodies since the worst water quality impact may not occur during the high-flow "design" storms.

Models have been designed for specific applications, such as evaluation of pollutant loads and management practices for agricultural operations,

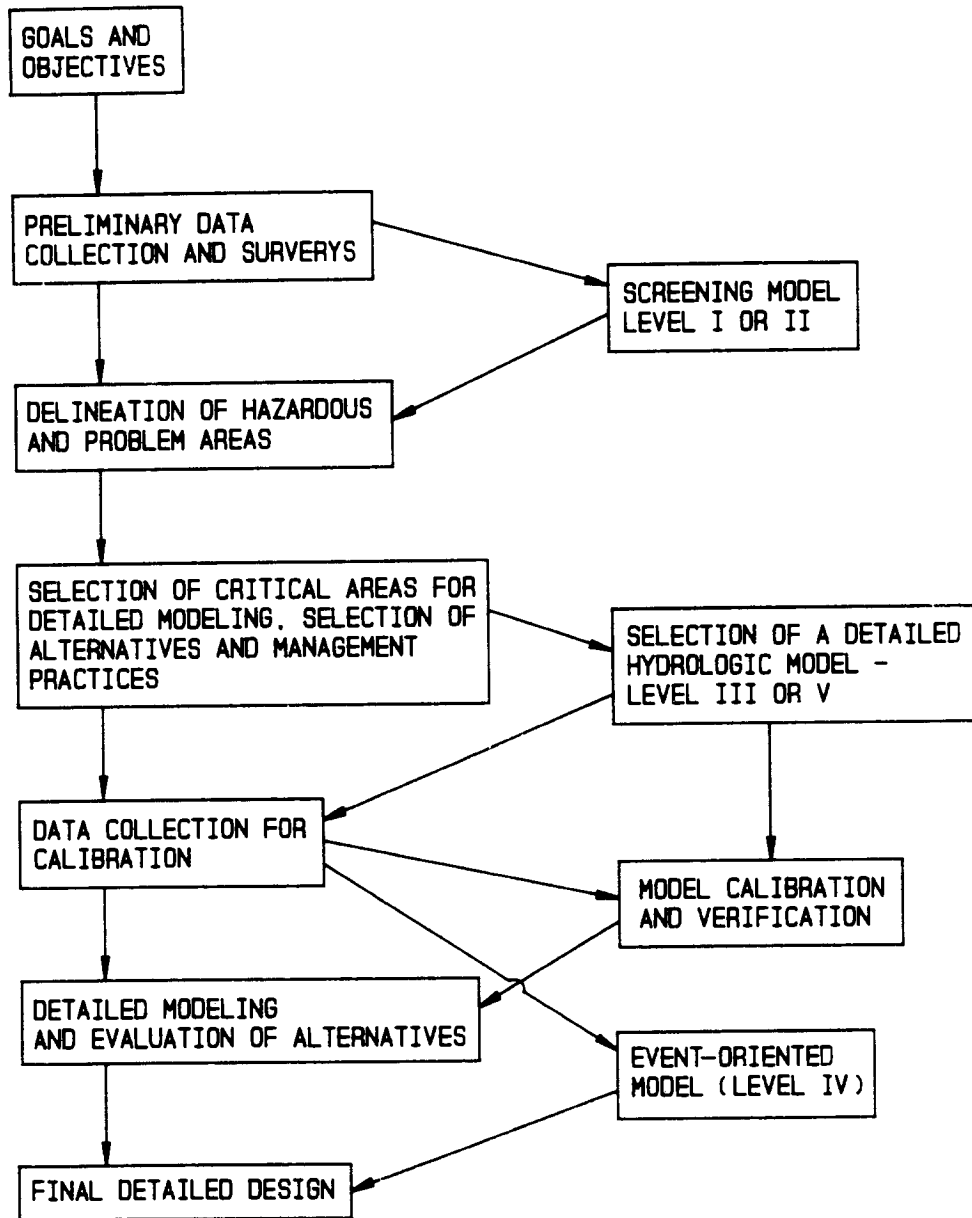


FIGURE 9.5. Schematics of the selection process for using models in diffuse-pollution simulation and abatement design.

urban and highway drainage-water quality models, mining pollution models, watershed models for the evaluation of the effects of atmospheric deposition, subsurface drainage and pollutant transport models, and specific designs of various structural and nonstructural measures, such as detention-retention ponds. There is no "universal" reliable diffuse-pollution model available, and the user must match the selected model with the modeling task.

Calibration and verification of the models with field data may be necessary for Phase 2 modeling activities if accurate loading of pollutants is desired. Calibration is less important if only the relative effect of various management practices is compared. One important consideration in model selection is the experience of others with its use. Few models have adequate documentation, have been extensively tested and applied by others than the model creators, and are continuously supported and maintained.

SIMPLE STATISTICAL ROUTINES AND SCREENING MODELS

Screening models (Levels I and II) evolved primarily from established traditional design procedures and/or the statistical results of monitoring programs. The concepts are the same for both urban and rural (agricultural) land uses, and the land-use type and distribution is generally the most dominant independent input parameter. The hydrologic fundamentals of these models are not complex or may not even be considered. Simple water quality estimation procedures for diffuse-source areas rely primarily on estimating pollutant accumulation in curb storage or use directly established unit loads related to land use for each land use within the watershed.

Diffuse (nonpoint)-pollution unit loads are expressed in mass of pollutant per unit of area per time. Unit loads of pollutants from dry-weather flow in combined sewers are often expressed in so-called population equivalents (pollutant emission rate expressed as mass per capita per day). The pollution load in this simple way is then found by multiplying the unit load by partial area of the land-use category and for sewage by the population number of the contributing basin. Publications by McElroy et al. (1976) and by Chesters et al. (1978) summarized unit loads and functions for nonpoint pollution, including that from urban drainage basins.

The basic characteristic of simplified screening models is that there is very little interaction, if any, among the processes generating the flow and quality. The statistical equation used to derive the unit loads and parameters of the models were based on elemental statistical routines, such as mean, standard deviation, and multiple-regression analyses.

Unit Loads of Pollutants from Impervious Urban Surfaces

Buildup and Washoff. In the late 1960s and early 1970s, investigators (APWA, 1969; Sartor and Boyd, 1972) analyzed distribution of pollutants

on street surfaces and found that the majority of solids accumulated within a 1-m-wide strip on the street side of the curb. Sartor and Boyd (1972) then provided accumulation plots for three major land uses (residential, commercial, and industrial). These findings were then confirmed by a number of studies throughout the United States (for example, Shaheen, 1975, and Bannerman et al., 1984). In such cases, it was convenient to express the accumulated pollutants in grams per unit length of curb. Such curb pollutant loads were provided by McElroy et al. (1976).

Instead of using a constant curb or surface load of "dust and dirt" and associated pollutants, it has been demonstrated that there is a build up of particular pollutants on the impervious surfaces, particularly in the curb storage of urban basins. Originally, it was assumed that this buildup is linear; however, Sartor and Boyd (1972) demonstrated a gradual decrease in the accumulation rates until a certain pseudoequilibrium is achieved. Novotny et al. (1985) showed that this steady-state pollutant load is a result of a quasi equilibrium between the pollutant inputs (atmospheric and litter depositions and traffic emissions) and translocation of pollutants from the curb storage on adjacent pervious and hydrologically inactive surfaces. The steady-state curb pollutant loading may be maintained and has been observed in some experimental urban basins. Typically, these loads were measured between 20 to 100 g/m of curb, depending on the character and "cleanness" of the area. A linear steady buildup of accumulated pollutants on impervious surfaces and curb storage is typical only for urban areas that are nearly 100% impervious. The *buildup* then refers to a complex process occurring during the dry period preceding a rainfall that includes deposition, wind erosion, and street cleaning effects.

Washoff is thus the process by which the accumulated particles are removed from the surface by rainfall-generated runoff. Hence, the quantity of the pollutant in the sewer flow could be computed from:

$$P = CD * L * A * r \quad (9.2)$$

where

P = pollutant load in the sewer or at the basin outlet (kg/storm)

L = accumulated curb load of the pollutant during an antecedent dry period (g/m)

CD = curb density (km/ha)

A = subwatershed area (ha)

r = washoff factor that is related to runoff energy

The magnitudes of the unit loads and simple procedures were compiled by McElroy et al. (1976). The pollutant loads computed by these simple

procedures (using either the literature or local monitoring statistical values) can be complemented by simple flow determination in order to crudely estimate the average concentrations of the pollutant in urban runoff.

The buildup and washoff modeling concept has been used in simple screening models, as well as in more complex hydrological urban runoff pollution models. A more detailed presentation of buildup and washoff processes and their implication on urban runoff quality management was presented in Chapter 8.

Unit Loads of Pollutants from Pervious Surfaces

The annual or seasonal form of the universal soil loss equation (USLE) proposed by Wischmeier and Smith (see Chapter 5 for a detailed description of the USLE with quotations from the authors and developers) is the most widely used and respected model for estimating soil loss from pervious areas. In order to obtain solids load (sediment yield) at the watershed outlet, the potential soil loss estimated by the USLE must be multiplied by the delivery ratio factor, DR . Generally, DR equates the measured sediment yield to the potential soil loss estimated by the USLE. As pointed out in Chapter 5, the DR is a highly variable hydrological parameter that normally is estimated by calibration. For sewered areas, DR is close to unity. The magnitudes of DR for urban areas and different land uses are not known and should be researched. In most cases only simplified morphological estimates of the delivery ratio are considered and included.

To obtain loading of particulates other than sediment, the sediment loads obtained by the USLE are multiplied by *potency factors*, which relate the concentration of the pollutant to that of the sediment. These factors have also been summarized by McElroy et al. (1976) and investigated by Zison (1980). The enrichment is either not considered or considered only superficially.

A number of excess rainfall estimating procedures can be used for calculating flow and, hence, average event concentrations of pollutants. The best known hydrological procedure in the United States was developed by the Soil Conservation Service (SCS) and is called the runoff curve number method, which was introduced in Chapter 3 (Soil Conservation Service, 1972). The SCS method provides a means of estimating surface runoff volume and hydrograph using a semiempirical relationship between rainfall and runoff based on a runoff curve number that depends on land use, soil type, vegetation, and antecedent soil moisture conditions.

For combined sewer systems, the simplified procedures mentioned

earlier provide only the wet-weather water and pollutant contributions by surface runoff. In order to obtain the total pollution load, dry-weather (wastewater) contributions must be added. These contributions are estimated using population equivalent loads such as those reported in Novotny et al. (1989).

Screening Models

The EPA screening procedures (Mills et al., 1982, 1985) evolved from the revisions and expansions of the water quality assessment procedures initially developed for 208 planning areas. The procedures are essentially a compilation of functions and unit loads developed previously by the Midwest Research Institute (McElroy et al., 1976) and provide annual estimates of loads. The nonurban loads are based on the annual form of the universal soil loss equation (see Chapter 5), while urban loads are estimated using the buildup-washoff concept presented in the preceding paragraph. The revised edition of 1985 contains a comprehensive compilation of procedures for estimating loadings of toxic pollutants, including their partitioning between solid and dissolved phases.

The data requirement is minimal and the procedures have excellent user documentation and guidance, including workshops sponsored by the EPA Water Quality Modeling Center, Athens, Georgia. The presentation of loading functions is supplemented with additional parameters and sample calculations.

Model-enhanced unit loads. A large hydrologic-pollution simulation model, calibrated by extensive monitoring data from several experimental watersheds located in southeastern Wisconsin, was used to generate unit loads of pollutants for typical land uses (Novotny et al., 1979; Novotny and Bannerman, 1980). The method called MEUL (model enhanced unit loads) and the generated unit loads were later used by local planning agencies to delineate hazardous lands and watersheds requiring abatement in the Wisconsin Priority Watershed Program and 208 local studies. Figure 9.6 shows the loading diagrams generated by the MEUL method. These unit loads can then be used along with the land-use distribution within the watershed to generate total annual or seasonal watershed loadings.

Cornell University researches (Haith and Tubbs, 1981; Dickerhoff and Haith, 1983; Haith and Shoemaker, 1987) presented generalized loading functions for estimating nutrient loadings. These simple functions describe both rural and urban sources. For urban sources the buildup and washoff concept is used. For rural sources the loading functions have the following form:

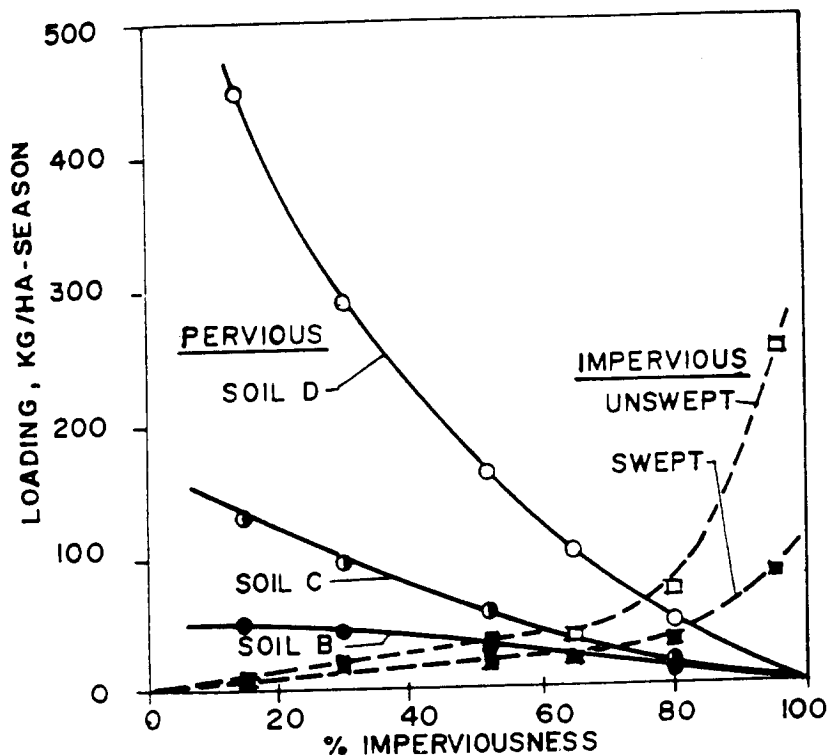


FIGURE 9.6. Unit loads of sediment (suspended solids) obtained by hydrologic simulation (model-enhanced unit loads).

$$\begin{aligned}
 LD_{kt} &= 0.1CD_{kt}Q_{kt}TD_k \\
 LS_{kt} &= 0.001CS_{kt}X_{kt}TS_k
 \end{aligned}
 \tag{9.3}$$

where

- LD_{kt} = unit load of dissolved pollutant from land-use area k during the event on day t , kg/ha
- LS_{kt} = unit load of suspended pollutant from land-use area k during event t , kg/ha
- CD_{kt} = dissolved pollutant concentration of pollutant in runoff, mg/l
- CS_{kt} = solid-phase (adsorbed) pollutant concentration, $\mu\text{g/g}$ of sediment
- Q_{kt} = runoff for the day of the event on day t from land-use area k , cm
- X_{kt} = soil loss from land-use area k due to the event on day t , tonnes/ha
- TD_k, TS_k = transport factors that indicate the fractions of dissolved and solid-phase pollutants that move from the edge of the source area (field) to the watershed outlet.

To obtain the total watershed seasonal or annual (nonwinter) loadings the unit loads expressed by Equation (9.3) are integrated over the entire watershed area and time span of the simulation period as

$$LD \text{ (total)} = \sum_k \sum_t LD_{kt} A_k \quad (9.4a)$$

$$LS \text{ (total)} = \sum_k \sum_t LS_{kt} A_k \quad (9.4b)$$

where A_k = watershed area associated with land use k , ha. Implementation of these loading functions requires estimates of runoff, soil loss, pollutant concentrations, and attenuation and transport from each unit land-use source area within the watershed. Simplified procedures are provided by the authors.

Regional regression models. USGS developed regional regression models for estimating urban runoff quantity and quality (Driver and Tasker, 1988; Driver and Lystrom, 1987; Driver, 1990) in which the nonpoint pollution loads were correlated to the percentage of impervious area, industrial land use, commercial land use, nonurban land use, mean annual rainfall, and mean annual nitrogen-precipitation load. The required input data can easily and inexpensively be obtained from local rain gages, land-use maps, annual weather summaries, and census reports.

The model has the form (Driver and Lystrom, 1987):

$$Y = (B_0 \times X_1^{B_1} \times X_2^{B_2}, \dots, X_n^{B_n}) BCF \quad (9.5)$$

where

- Y = estimated storm-runoff pollutant load or volume
* dependent variable
- X_1, X_2, \dots, X_n = basin morphological or climatic characteristics (independent variables)
- B_1, B_2, \dots, B_n = regression coefficients
- n = number of basins or climatic characteristics in the models
- BCF = bias correction factor correcting the impact that the logarithmic transformation of the variables has on the mean

Independent variables included in the model were:

Basin characteristics

1. Total contributing drainage area
2. Impervious area as a percentage of the total area

3. Industrial land use
4. Commercial land use
5. Residential land use
6. Nonurban land use

Climatic characteristics

1. Total rainfall for each storm
2. Mean annual rainfall
3. Mean annual nitrogen precipitation load

In the analysis the United States was divided into three regions based on mean annual rainfall. The USGS data base from which the models were developed included 1123 storms for 98 urban stations, which were mostly stations of the National Urban Runoff Project, in 21 metropolitan areas. The dependent (modeled) parameters for which adequate regression models were obtained were runoff volume and chemical oxygen demand and nitrogen loads.

Caveats. In spite of their questionable accuracy and reliability, the simple screening procedures have found wide applications. It may be surprising to note that such simple quality estimations have been attached to several complicated hydrological models currently in use. One explanation for this popularity is that these procedures provide a simple mechanism and quick answers to pollution problems of larger watersheds where more complicated models might fail because of the enormous amount of information (not commonly available) required to set up the input data bases. The screening procedures enable identification of hydrologically active areas that generate nonpoint pollution that should then be subjected to further studies and subsequent management.

The effect of various abatement measures on nonpoint pollution loads estimated by the simplified screening procedures can be only approximately determined by using simple rule-of-thumb factors and pollution-removal efficiencies. For example, it can be assumed that an effective street sweeping program will reduce solids loads from impervious urban surfaces by 15% to 25%, or that a properly designed and operated wet detention pond can attenuate particulate pollution by about 50%, or that a certain soil-conservation practice will reduce soil loss as delineated in the guidelines for determining the cover factor, C , and the erosion control practice factor, P , of the universal soil loss equation.

If statistical regression models are selected, the user should be aware that all statistical models reflect and correspond to the data sample from which the model was derived. Using the models for conditions and locales outside the original data is always improper.

Geographical information systems. The unit load models and simpler screening models along with basin attributes can be incorporated in computer-aided design (CAD) and the geographic information systems (GIS) to generate graphic maps of source areas, transport routes, and other components and parameters of diffuse-pollution generation and transport processes, as well as the contaminant (pollutant) loads from these areas. The idea of using GIS for estimating diffuse pollution is not new and was used in the 1970s for calculating basinwide phosphate loads and developing source maps in the Great Lakes basin (Johnson et al. 1978). The GIS have been used in many disciplines, including mapping, military, and water resources-watershed descriptions.

Geographical information systems have a lot in common with the computer-aided design systems used for drafting a wide range of technical objects. Both GIS and CAD need to be able to relate objects to a frame of reference (coordinates), both need to handle graphic attributes and both need to be able to describe topological relations (Burrough, 1986). GIS can be linked to remote sensing, surveying data, and can include functional relationships and unit loads. Geographic information systems can be thought as being both the means of storing and retrieving data about aspects of the earth's surface, and systems by which the data can be transformed and manipulated interactively for studying environmental processes and the impact of planning decisions. They are ideally suited for studying the processes and impacts of diffuse pollution (Connors-Sasowski and Gardner, 1991; DeRoo, Hazelhoff, and Burrough, 1989; Evans and Miller, 1988; Potter, Gilliland, and Long, 1986).

Cline, Molinas, and Julien (1989) described an auto-CAD system that was used in connection with the watershed hydrologic model HEC-1 developed by the U.S. Army Corps of Engineers in the 1970s. They have pointed out that the commercially available auto-CAD system can organize and display spatial watershed data and also has some analytical capability. In a similar application, Stuebe and Johnston (1990) used a public domain GIS system GRASS (available from the U.S. Army Engineering Research Laboratory) for estimating runoff volumes. The GRASS system is the designated GIS of the U.S. Department of Agriculture, Soil Conservation Service.

In a GIS system, information of the spatial characteristics of a geographic area (such as a watershed) can be stored in a square grid system called a *raster* or a *vector-polygon system*. In the raster system, each square may correspond to a dot (pixel) on the computer screen. Information can be stored in different layers (for example, soil information in one layer, elevation in the second layer, and land cover in the third layer. This concept of information storage is shown in Figures 9.7 and 9.8. As

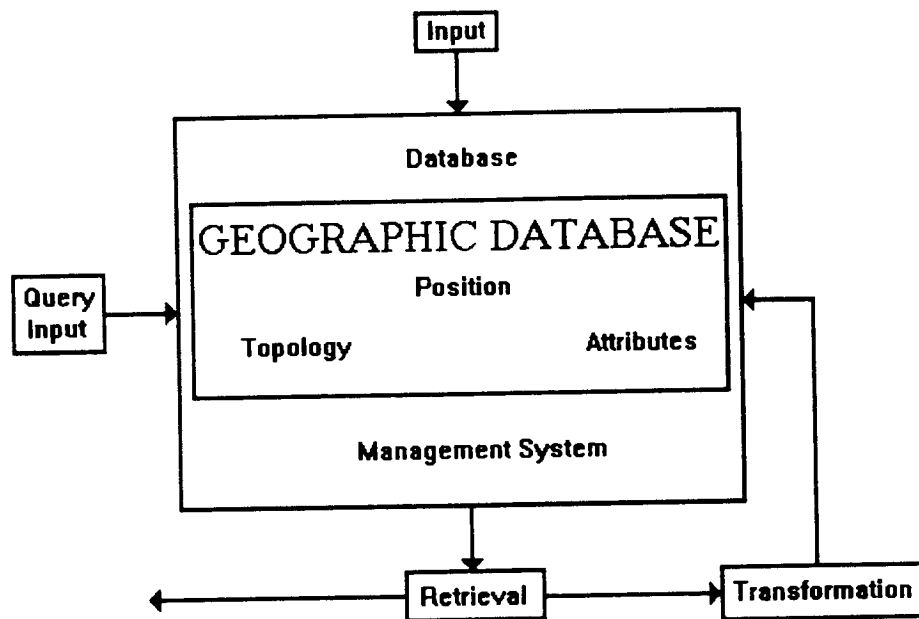


FIGURE 9.7. Schematic of the geographical information systems (GIS). (After Burrough, 1986, by permission of Oxford University Press.)

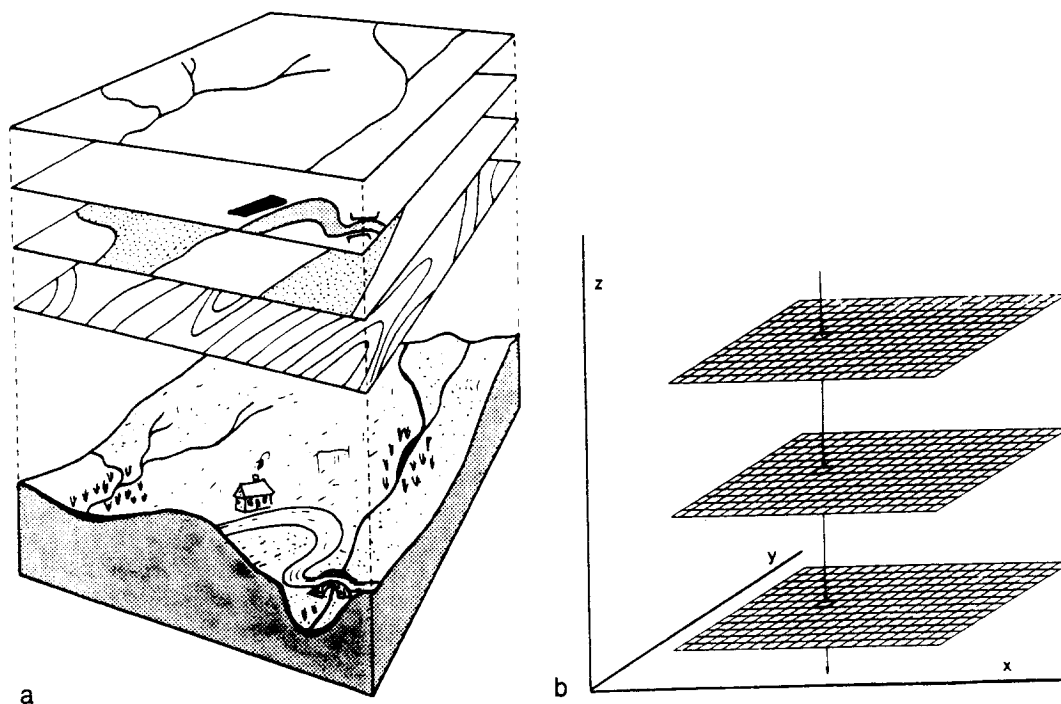


FIGURE 9.8. Overlay and storage of geographic information in a GIS system. (a) The “overlay” concept: the real world is portrayed by a series of overlays, in each of which one aspect of reality has been recorded (e.g., topography, soil type, roads, rivers). (b) Three-dimensional arrays used for coding map overlays in raster data base structures. (After Burrough, 1986, by permission of Oxford University Press.)

soon as the entire watershed information is coded and entered into the GIS system it can be processed using, for example, the universal soil loss equation, to produce source maps of sediment and pollutants. The GIS can also be used for spatial water quality and pollution analyses, as well as for evaluation of the impacts of various land-use changes within the watershed on the water quality of receiving water bodies.

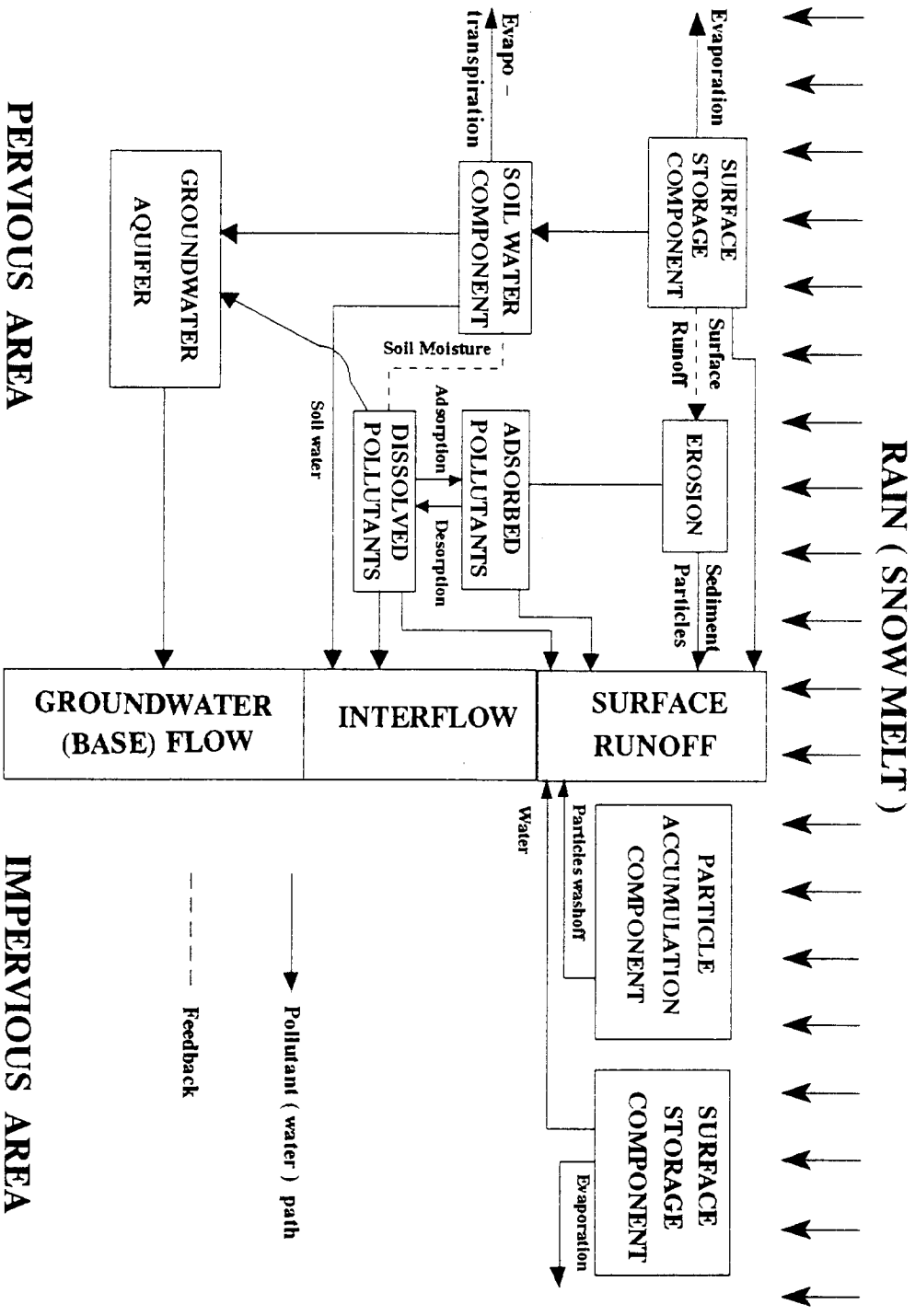
The GIS are used by the agencies to store and manipulate geographical information, including that related to diffuse pollution and water quality. For example, the Soil Conservation Service (SCS) of the U.S. Department of Agriculture has developed the State Soil Geographic (STATSGO) data base (Reybold and TeSelle, 1986; Bliss and Reinbold, 1989). This data base overcame many of the problems associated with previous tedious methods of mapping soils over a larger area. STATSGO data have polygon units that incorporate several components of the soil information system, such as slopes and water capacity. The U.S. Geological Survey was cooperating with the SCS in an effort how to use the soil GIS data base in watershedwide analyses.

Probabilistic statistical methods. The so-called *EPA Probabilistic Method* for analyzing water quality effects of urban runoff is based on the derived frequency distribution of the flows and quality, which is presumed to be log-normal (Driscoll, 1986). When coupled together the distribution of runoff loads can be derived (Driscoll, DiToro, and Thomann, 1979; DiToro, 1984; and Athayde, Meyers, and Tobin, 1986). Further analytical treatment has been developed to account for storage (DiToro and Small, 1979). These models are not identical to the stochastic models discussed in the next section.

DETERMINISTIC HYDROLOGIC SIMULATION MODELS

The more complex hydrologic simulation models in the second category represent a description of the hydrologic rainfall-runoff transformation process with associated erosion, pollution buildup and washoff, and other quality components. In combined sewer systems, sewage flow contributions, infiltration, quality contributions by infiltration (for example, nitrate), and pollutant buildup in sewers are additional sources and processes that should be included.

The basic premise of hydrologic simulation models, in contrast to the previous simplified computational procedures, is an interaction among hydrological and pollution-generation and -transport processes. Obviously, the complexity of the processes that are considered by the models and their feedback and interactions vary from model to model.



PERVIOUS AREA **IMPERVIOUS AREA**

FIGURE 9.9. Components of hydrological—diffuse-pollution models.

Diffuse-pollution hydrological flow-quality models have the following basic components (Novotny, 1986, 1988), which are also depicted in Figure 9.9:

1. A surface-runoff-generation component that computes the transformation into excess (net) rainfall. Modeling surface runoff includes the following processes that were described in Chapter 3:
 - a. Exhaustion of surface storage
 - b. Evapotranspiration
 - c. Snow accumulation and melt
2. The soil and ground-water component (not common to all models) that describes the movement of water through the unsaturated soil zone into the saturated ground-water zone. These submodels balance soil moisture with infiltration rate, evapotranspiration, and water loss into the deep ground-water zone. Since infiltration is a function of soil moisture content, an iterative procedure may be employed. If the soil component is not included in the model, the infiltration rate of surface runoff is estimated by an empirical equation. The most common infiltration models were presented in Chapter 3.
3. Accumulation, removal, and washoff of pollutants from impervious surfaces. This component, typical for urban models, balances the particulate pollutants that deposit onto the impervious surfaces near the curb of paved roads during dry days and estimates their washoff into the runoff during rainfall. The accumulated mass is a result of the deposition of particulate pollutants from various sources (atmospheric dry deposition, litter deposition, traffic emissions, salt application), and their removal from curb storage by wind and traffic-induced turbulence and street sweeping (Novotny et al., 1985).
4. Soil-erosion components, by which soil loss from impervious areas can be estimated. The universal soil loss equation (USLE) is the most common model to represent this process. Soil-erosion models were introduced in Chapter 5.
5. Adsorption of pollutants on soils and urban dust. This component, which in agricultural models is represented by adsorption-desorption relationships presented in Chapter 6, is in most urban models replaced by potency factors that relate the particulate pollutant to that of total suspended solids.
6. Runoff and pollutant routing components transform the excess rainfall into an inlet hydrograph and quality histogram that is then routed through the sewer network. Common techniques for excess rainfall-surface-runoff overland routing (convolution) rely on a synthetic unit hydrograph, in most cases. Routing in sewers and channels is most

often accomplished by kinematic wave formulas. In the absence of a detailed routine for sediment redeposition and trapping during the overland flow, sediment delivery and enrichment factors are used to relate the measured sediment and pollutant yields to their generation at the source.

7. In the combined sewer systems, flow generation and deposition of solids in sewers, as well as the growth of biomass therein, should be considered. The contribution of solids deposited from sewage can be substantial. Dry-weather fraction of the pollution carried by the combined sewers can be estimated by population equivalents, while sewer solids accumulation, growth of biological mass, and first flush effects due to scouring of these materials can only be estimated using highly empirical and unreliable procedures. One can expect that the parameters affecting these processes will include the hydraulic conditions in the sewers and the length of the antecedent dry-weather period.

Discrete event modeling simulates the response of a watershed to a major rainfall. Continuous dynamic models simulate flow and pollutant loading over an extended period of time. However, it is erroneous to expect that continuous models can provide adequate information on the variability of the flow or its quality. By their nature and design, the deterministic dynamic models could be called short-term moving average models without a noise component (this loose classification should not be confused with the moving average ARMA models that are introduced later in this chapter). Therefore, attempts to apply statistical frequency and probabilistic exceedence routines to the outputs from the deterministic models are improper.

Urban Models

All urban runoff and CSO pollution models presented below are lumped-parameter models, although they do allow segmentation of the watershed into uniform subbasins according to land use and drainage patterns. As to their complexity, the urban hydrologic models can be classified into three groups:

Level III. Simplified models, either event oriented or continuous. In these models some hydrologic or water quality components are missing, some others are replaced by a simple relationship. There is some

interaction between the processes, but this is usually done in an approximate way. Water and pollutant routing may not be included.

Level IV. Sophisticated single-event, simulation models. These models provide for extensive interactions among the various processes that are important in the simulation of storm-water quantity, including infiltration, overland and gutter flow, and flow routing. The quality components may be less sophisticated, relying on simpler pollutant buildup and washoff concepts. Street sweeping effects can be included only during a predetermined relatively short antecedent dry period and, in general, such models may not be suitable for evaluation of typical long-term best management practices for controlling urban runoff pollution. Their primary use is in the design and evaluation of urban drainage systems.

Level V. Sophisticated continuous models. These models provide for extensive interactions among the various processes that are important in the simulation of both storm-water quantity and quality in separate and combined sewer systems. Quality and routing of flow are included. Modeling intermittent dry periods accounts for soil moisture, pollutant buildup and removal, and other important processes occurring between rainfalls.

Level III Models

Examples of simplified urban models in Level III include the Corps of Engineers' (Hydrologic Engineering Center, 1975) Storage-Treatment-Overflow-Runoff Model (STORM), Model of Urban Runoff and Sewer Flow (MOURSEF) (formerly the Wisconsin Urban Runoff Model) (Novotny et al., 1985; Novotny and Capodaglio, 1991), the Areawide Stormwater Model (ABMAC) (Litwin, Lager, and Smith, 1981), TVA's model HYSIM (Milligan, Wallace, and Betson, 1984), and a number of proprietary models. Most of these models simulate runoff either by the modified rational formula (STORM) or by the SCS Runoff Curve Number Model (Wisconsin Urban Runoff Model). The models simulate street drainage and storage-treatment systems. The TVA-HYSIM model also incorporated surface-ground-water interrelation due to specific characteristics of the NURP watersheds in Knoxville, Tennessee.

The STORM model is available from the Hydraulic Engineering Center of the U.S. Army Corps of Engineers in Davis, California, but the center does not provide user's support nor further development and maintenance of the program, which was left with private software vendors. The MOURSEF model by the senior author and his coworkers can be provided as supplementary software to the users of this book. Figure 9.10 shows a schematic of the MOURSEF model.

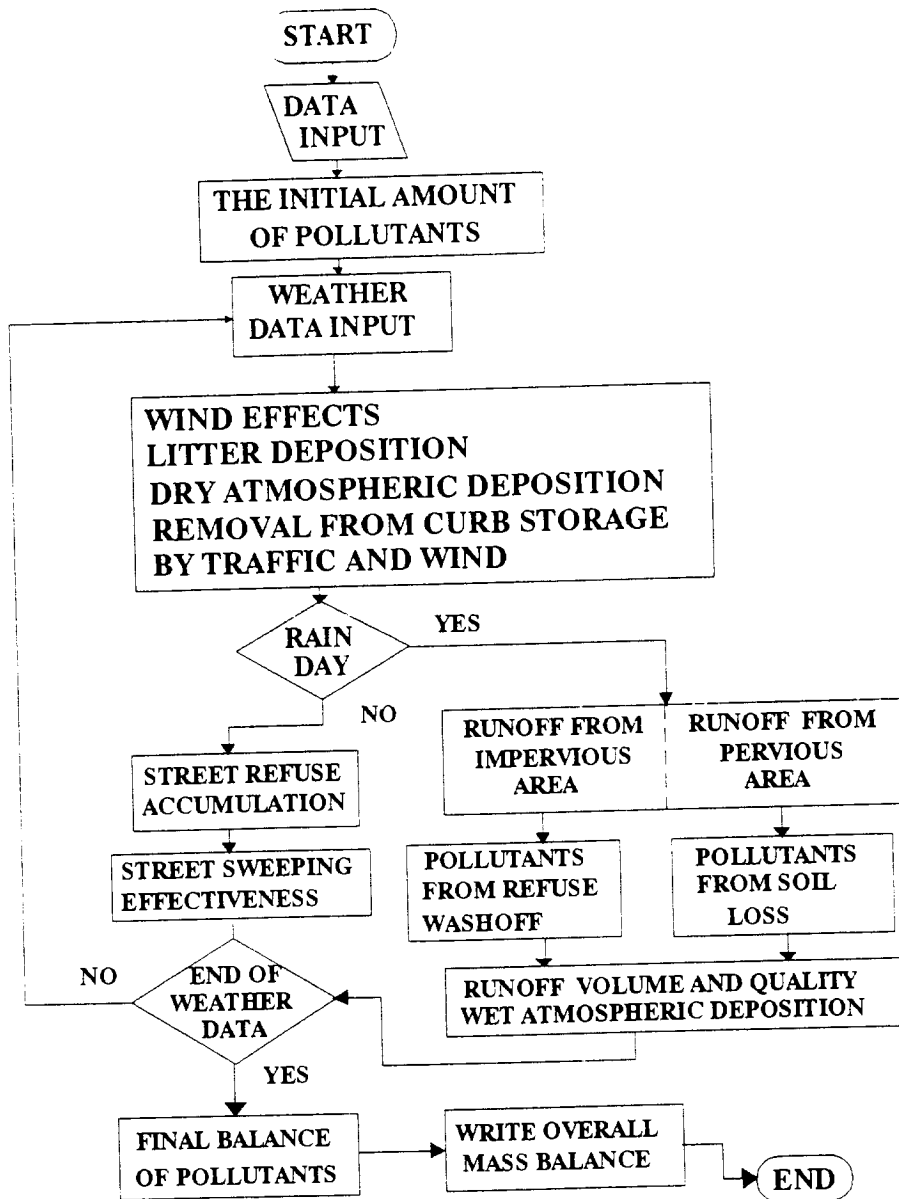


FIGURE 9.10. Schematic diagram and components of the MOURSEF model.

Level IV Urban Models

The best known Level IV models (event-oriented models) are the EPA's original Stormwater Management Model (SWMM) (Huber and Dickinson, 1988) and the Illinois Urban Drainage Area Simulator (ILLUDAS) (Terstriep and Stall, 1974). In Great Britain the WASSP-QUAL model (a proprietary but widely used model) was developed and sold throughout Europe (Henderson and Moys, 1987). Several "synthetic" design storms

have been developed and are available (Soil Conservation Service, 1972, 1985; Chow, Maidment, and Mays, 1988). These design storms can be used as a substitute for measured storm hyetographs in event modeling.

The Storm Water Management Model (SWMM) is a widely used storm-water and combined sewer overflows quantity-quality model that is readily available from the U.S. EPA Laboratory in Athens, Georgia. SWMM, developed in the 1970s (U.S. EPA, 1971), has been applied to urban hydrologic quantity-quality problems in many locations world-

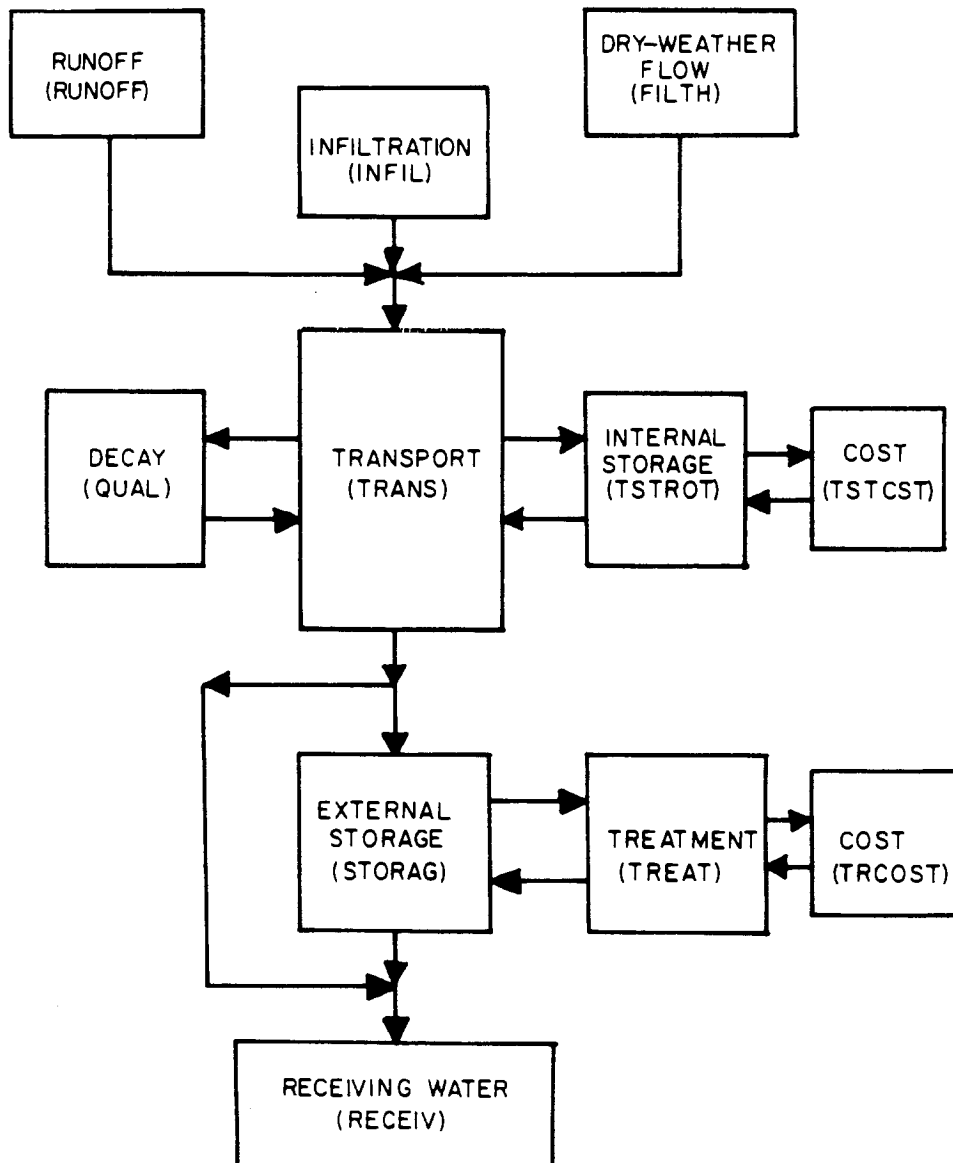


FIGURE 9.11. Components of the Storm Water Management Model (SWMM). Subroutine names are shown in parentheses. (After U.S. EPA, 1971.)

wide. The model is in the public domain, has excellent documentation, and is continuously maintained and updated. The model can analyze single storm events on multiple catchments. Figure 9.11 shows a schematic of the basic components of the program.

Version 4 (Huber and Dickinson, 1988) of the SWMM model performs both continuous and single-event simulation throughout the whole model. It can simulate backwater, surcharge, pressure flows, and looped connections, and has a variety of options for quality simulations, including traditional buildup and washoff functions, as well as rating curves and regression techniques. Subsurface flow routing (constant quality) can be performed in addition to surface quantity and quality routing, and treatment devices may be simulated using removal functions and sedimentation theory. A hydraulic design routine is included for sizing of pipes, and a variety of regulator devices can be simulated, including orifices, weirs, pumps, and storage. This version is essentially a Level III model (Maahel and Huber, 1984).

The volume of data needed to run SWMM varies with the scope of the application, and can be very extensive in some complex situations. At a minimum it requires information on area, imperviousness, slope, roughness, depression storage, and infiltration characteristics. Information on buildup of pollutants is required for quality generation. Precipitation input is in the form of hyetographs for individual storm events (hourly or 15 minute intervals). Channel-pipe data include shapes, dimensions, slopes or invert elevations, and roughness.

Calibration of the model is needed, especially for quality simulation, but it is tedious and an expert computerized system is needed. Even with calibration the performance of the models for quality is relatively weak (Donigian and Huber, 1990; Delleur and Baffaut, 1990). The PC version of the model is not "user friendly;" however, an expert system for calibration of the model is available (Delleur and Baffaut, 1990).

The Highway Drainage Model (Dever and Roesner, 1982, 1983; Roesner, Dever, and Aldrich, 1983; Dever, Aldrich, and Williams, 1983) developed for the U.S. Federal Highway Administration consists of a package of six computer programs. The model has used some components of the SWMM as basic blocs. The general capabilities of the model include statistical analysis of long-term rainfall records, preliminary highway drainage system design, hydraulic analysis of drainage systems during extreme storm events, and simulation of buildup and washoff of pollutants. Its computational capabilities are similar to those of SWMM (Donigian and Huber, 1990).

DR3 QUAL is a version of the USGS Distributed Routing Rainfall Runoff Model that includes quality simulations (Alley and Smith, 1982).

The model is available for general use from the U.S. Geological Survey. Runoff generation is followed by routing with the kinematic wave formula. Quality is simulated using buildup and washoff functions with settling of solids in storage units dependent on particle size distribution. The model has been used in some of the NURP studies that were conducted by the USGS. No microcomputer version is available.

Level V Urban Hydrologic Models

Hydrologic Simulation Program—FORTRAN (HSP-F). The FORTRAN version of the Hydrologic Simulation Program (HSP-F) is an example of a Level V model (Johanson et al., 1982; Barnwell and Kittle, 1984). This model is capable of simulating a hydrologic time series of runoff quantity-quality events, including flows (hydrographs) and conventional pollutants and toxics. The HSP-F incorporates the watershed scale Agricultural Runoff Model (ARM) and urban Nonpoint Simulator (NPS) into a basin-scale analysis framework that includes fate and transport in one-dimensional stream channels. It is the only comprehensive model of watershed hydrology and water quality that allows the integrated simulation of land and soil-contaminant runoff processes with in-stream hydraulic, water temperature, sediment transport, nutrient, and sediment-chemical interactions. The runoff quality capabilities include both simple relationships, such as empirical buildup and washoff functions and constant concentration pollutant inputs, as well as detailed soil process options (including, leaching, soil attenuation, and soil nutrient transformation). The model was originally developed from the Stanford Watershed Model. It is a large model and requires considerable effort when applied to a watershed. The use of this model for combined sewer systems is very limited.

The HSP-F contains three applications moduli—PERLND, IMPLND, and RCHES—and five utility moduli—COPY, PLTGEN, DISPLY, DURANL, and GENER (Fig. 9.12). Basically, the HSP-F performs the simulation in a lumped-parameter mode, whereby the magnitudes of the parameters and coefficients must be determined by calibration.

The PERLND simulates a pervious land segment with homogenous hydrologic and climatic characteristics. Water movement is modeled along three flow paths, overland flows, interflow, and ground-water flow in the manner conceived and incorporated in the Stanford Watershed Model, which is the predecessor of the HSP-F. Erosion is modeled by the Negev model (see Chapter 5). Water quality constituents can be simulated in a simple way by relating them to sediment loads with potency factors or by a more complex adsorption-desorption concept. The former pollutant transport concept is incorporated in the Nonpoint

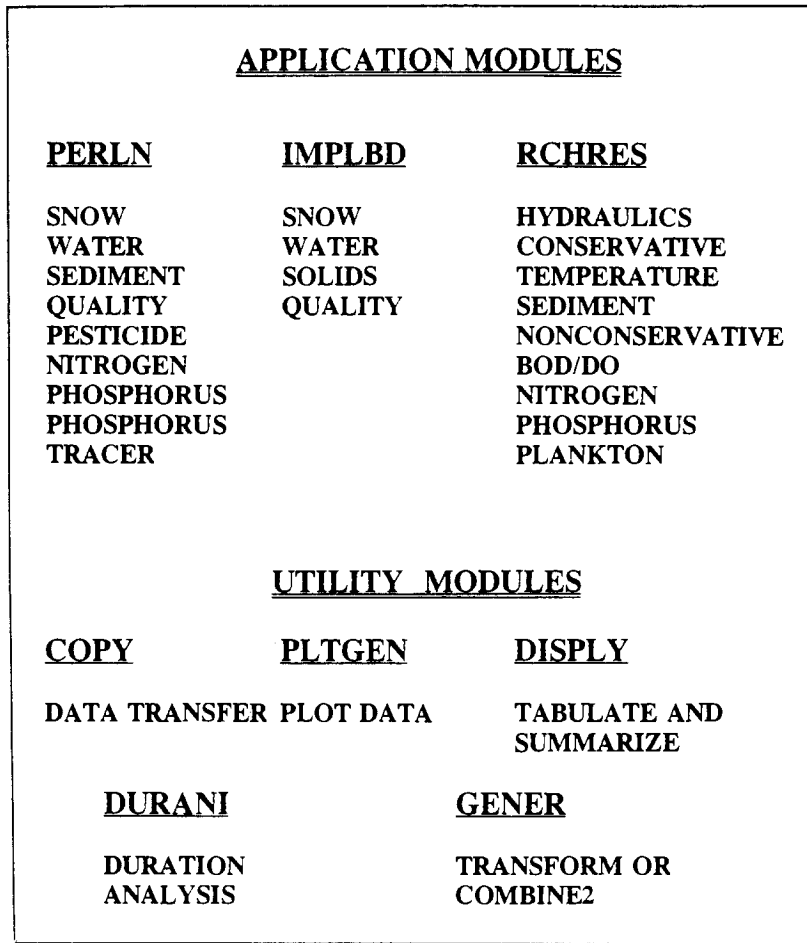


FIGURE 9.12. Components (modules) of the HSP-F watershed model. (After Barnwell and Kittle, 1984.)

Pollution Source (NPS) model, typically used for urban areas, while the latter was incorporated in the Agricultural Runoff Management (ARM) model. The NPS model has been in public use since 1980 and is available for PC computers from the U.S. EPA Environmental Research Laboratory in Athens, Georgia.

Module INPLND is designed to simulate impervious land segments with no infiltration. Water yield and movement is similar to PERLND, except that no water movement occurs by subsurface flows. Solids are simulated using the buildup-washoff concept in a manner similar to other urban models. Water quality constituents are simulated using empirical relationships with solids and water yields.

Module RCHRES simulates the processes that occur in a single reach of an open channel or a completely mixed lake. Hydraulic behavior is modeled by a kinematic wave routing concept (Chapter 3). Water quality

algorithms are similar to many other stream and lake models that have evolved in the past 50 years (see Chapter 12).

The result of the modeling is a time history of runoff flow rate, sediment load and nutrient and pesticide concentrations, along with a time history of water quantity and quality at any point in a watershed. HSP-F simulates three sediment types (sand, silt, and clay) in addition to a single organic chemical and nutrients.

The *WASSP-QUAL* model was developed in Great Britain by Hydraulic Research, Ltd. The model is a result of a joint research effort with the Water Research Centre and other British institutions. The model can be

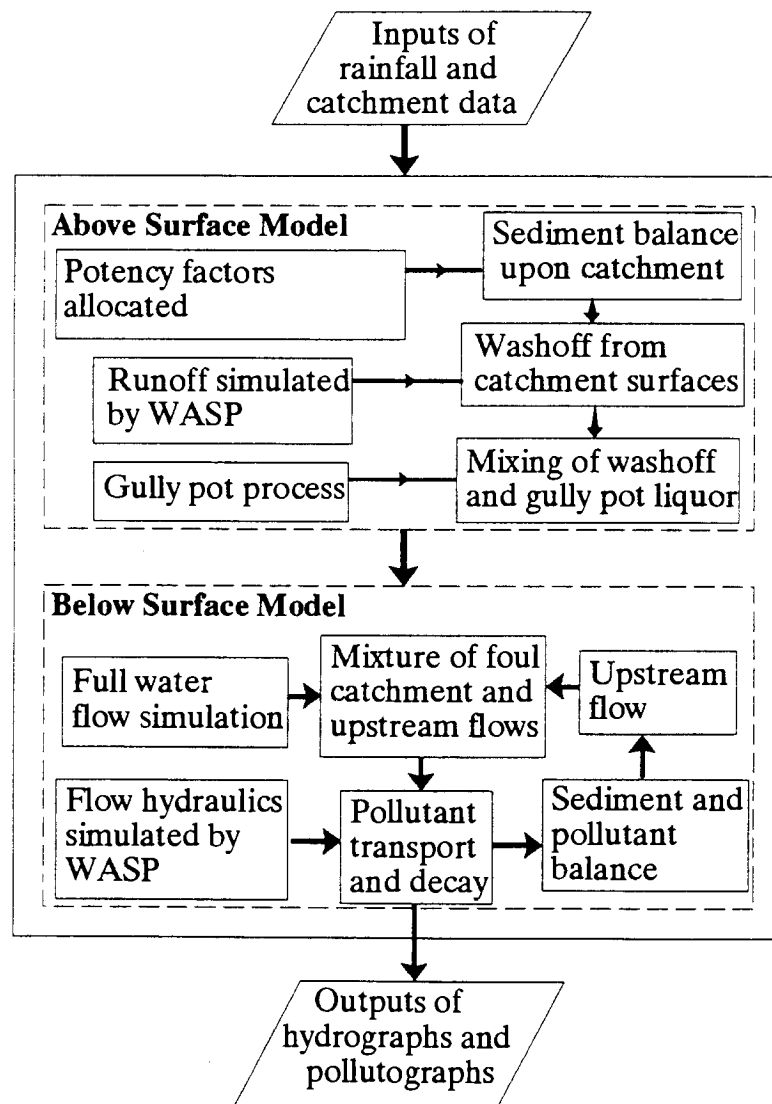


FIGURE 9.13. Overview of the WASSP-QUAL model structure and modeled processes. (After Henderson and Moys, 1987.)

run either in a continuous or single-event mode, and it models the following quality parameters (Henderson and Moys, 1987):

- Suspended solids, both total (TSS) and volatile (VSS)
- Dissolved oxygen
- Oxygen demand (BOD or COD)
- Ammoniacal nitrogen
- Hydrogen sulphide
- Large grain-size sediments

A schematic diagram of the components of the model is shown on Figure 9.13. The model requires calibration, and in some aspects is more complex than the SWMM, but not necessarily more reliable. This model should not be mistaken for the WASP model of the U.S. EPA (see Chapter 12 for a description of WASP).

Operational Urban Models

Huber (1986) listed the six operational urban water quality models that are included in Table 9.1. By *operational* he implied that (1) a user's manual and documentation are available, (2) the model had been used by others than just the model developer, and (3) continuous support was available. Two additional proprietary operational models that fit this definition were also included.

The Proceedings of the International Conferences on Urban Storm Drainage (Yen, 1982; Balmér, Malmqvist, and Sjöberg, 1984; Gujer and Krejci, 1987; Iwasa and Sueishi, 1990) reported numerous modeling efforts throughout the world. Also the NATO Advanced Seminar pro-

TABLE 9.1 Operational Urban Runoff Models

Model	Sponsoring Agency	Year of Origin	No. of Pollutants	Simulation Type
DR3M-QUAL	USGS	1982	4	C, SE
FHWA	FHWA	1979	13	SE
HSP-F	EPA	1976	10	C, SE
STORM	HRC	1974	6	C, SE
SWMM	EPA	1971	10	C, SE
WASSP-QUAL	Proprietary ^a	1987	7	C, SE
MOURSEF	Proprietary ^b	1983	6	C

Source: Adapted from Huber 1986.

Note: C = continuous simulation; SE = single-event simulation.

^a Water Resources Center Engineering, Swindon, England.

^b Available from the senior author.

ceedings (Torno, Marsalek, and Desbordes, 1986) has several overview chapters dealing with modeling. Also see Moffa (1990) for chapters on models dealing specifically with the CSO and their impact on receiving waters.

Rural (Agricultural Models)

Some of the previous hydrological models (for example, HSP-F, MOURSEF) can also be used for simpler modeling of agricultural watersheds, primarily erosion and movement of particulate pollutants. Agricultural models may also be divided into the three levels of sophistication; however, the models presented herein have mostly Level IV or V sophistication. Unlike urban models, which typically use the lumped-parameter concept, both distributed- and lumped-parameter concepts have been used for agricultural models. Their level of sophistication is based on the number of modeled processes and constituents. The agricultural models were reviewed by a number of authors; for example, models for the transport of chemicals were reviewed by Donigian and Dean (1985). The following discussion was adapted from reviews by Novotny (1986) and Donigian and Huber (1990).

Areal, Nonpoint Source Watershed Environment Response Simulation (ANSWERS) was developed by the Agricultural Engineering Department of Purdue University (Beasley and Huggins, 1981; Beasley, 1986). It is a distributed parameter model, primarily event oriented. Currently the model is maintained and distributed by the Agricultural Engineering Department, University of Georgia, Tifton, Georgia.

In order to use the ANSWERS model, the watershed is divided into uniform square elements, as shown in Figure 9.3(b). The elements range from one to four hectares. Within each element the model simulates the process of interception, infiltration, surface storage, surface flow, subsurface drainage, sediment detachment, and movement across the element. The output from one element then becomes a source of input into an adjacent element, which can be either area of the channel segment.

Nutrients (nitrogen and phosphorus) are simulated using correlation relationships between chemical concentrations, sediment yield, and runoff volume. Snowmelt or pesticide movement cannot be simulated. Data needs compose a detailed description of the watershed topography, drainage network, soils, and land use. A single storm hyetograph drives the model. A PC version is available for small watershed applications. For larger watersheds, a mainframe-large memory capacity computer is required.

Agricultural Nonpoint Source Pollution Model (AGNPS) was develop-

ed by the U.S. Department of Agriculture, Agriculture Research Service (USDA-ARS) (Young et al., 1986). The primary emphasis of the model is on nutrients and sediment, and on comparing the effects of various best management practices on the pollutant loadings.

The AGNPS can simulate sediment and nutrient loads from agricultural watersheds for a single storm event or for a continuous simulation. The watershed must be divided into uniform square cells. Grouping of the cells is by division of the basin into subwatersheds. However, flow and pollutant routing is accomplished by a function of the unit hydrograph type, which is a lumped-parameter approach. The model does not simulate pesticides.

AGNPS is also capable of accepting and handling point inputs such as feedlots, wastewater discharges, and stream bank and gully erosion. In the model, pollutants are routed from the top of the watershed to the watershed outlet in a series of steps. The modified universal soil loss equation is used for predicting soil loss in five different particle sizes (clay, silt, sand, small aggregates, and large aggregates). The pollutant transport portion is subdivided into one part handling soluble pollutants and another part handling sediment absorbed pollutants.

The input data requirements are extensive, but most of the data can be retrieved from topographic and soil maps, local meteorological information, field observations, and various publications, tables, and graphs provided in the manual or references. Both mainframe and PC versions of the model are available from the USDA-ARS in Morris, Minnesota.

The Agricultural Runoff Management Model (ARM). This model is a version of the HSP-F model that can be run independently or included in HSP-F. The model simulates runoff (including snow accumulation and melt), sediment, pesticides, and nutrient loadings from surface and subsurface sources (Donigian and Davis, 1978). The ARM model (as the more complex HSP-F model that incorporates ARM) requires extensive calibration. The PC version is only available as a part of the HSP-F.

Chemicals, Runoff, and Erosion from Agricultural Management Systems (CREAMS). This model was developed by the U.S. Department of Agriculture, Agricultural Research Service (Knisel, 1980, 1985; Leonard and Knisel, 1986). CREAMS is a field scale model that uses separate hydrology, erosion, and chemistry submodels connected together by pass files.

The hydrology component has two options, depending upon availability of rainfall data. Option one estimates storm runoff when only daily rainfall data are available. This is accomplished by the SCS Runoff Curve Number model. When hourly rainfall data are available, option two

estimates runoff by the Green-Ampt equation. Both submodels were described in Chapter 3.

The erosion component of the model considers the basic processes of soil detachment, transport, and deposition. Detachment of soil particles is modeled by the modified universal soil loss equation for a single storm event. The transport capacity of the overland and channel flow is derived from the Yalin sediment movement model (see Chapter 5). The basic concepts for nutrient modeling treat their transport as proceeding separately in adsorbed (with sediment) and dissolved (with runoff) phases. Soil nitrate is lost both with surface and subsurface flows. Soil nitrogen is modified by nitrification–denitrification processes and by plant uptake.

The pesticide component estimates concentrations of pesticides in runoff (water and sediment phases) and the total mass carried from the field for each storm during the period of interest. Pesticides in runoff are partitioned between the solution and sediment phases using a simplified isotherm model (Chapter 6). The model has the capability of simulating up to 20 quality components at one time.

CREAMS can simulate user-defined management practices. These activities include aerial spraying (foliar or soil directed) or soil incorporation of pesticides, animal waste management, and agricultural best management practices (minimum or no tillage, terracing, etc.; see Chapter 11).

Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) was developed by the USDA-ARS to utilize the management-oriented CREAMS model. GLEAMS is essentially a vadose zone component for CREAMS (Leonard, Knisel, and Still, 1987). The soil column is divided into three to twelve layers of variable thickness in which pesticide and nutrient mass balance and routing are executed. The input data requirement for CREAMS–GLEAMS simulations are extensive and quite detailed. The maximum size of the watershed is limited to a field plot consisting of a maximum of three segments.

The CREAMS–GLEAMS model can be obtained from the USDA-ARS laboratory in Tifton, Georgia. It is available for PC computers, and there is a very active users group comprised of several hundred users throughout the world. Several other models were developed by adaptation of the CREAMS model, including *Simulator for Water Resources in Rural Basins (SWRRB)* and *Pesticide Runoff Simulator (PRS)*.

The SWRRB model was developed for evaluating basin-scale quality in rural watersheds (Williams, Nicks, and Arnold, 1985). SWRRB operates on a daily time step and simulates meteorology, hydrology, crop growth, sedimentation, flood plain degradation and aggradation, and nitrogen, phosphorus, and pesticide movement. The model was devel-

oped by modifying the CREAMS daily rainfall hydrology model for applications to large, complex rural basins. The PRS objective is to simulate pesticide runoff and adsorption onto the soil in small agricultural watershed.

Pesticide Root Zone Model (PRZM) was developed at the U.S. EPA's Environmental Research Laboratory in Athens, Georgia (Carsel et al., 1984; Muelkey, Carsel, and Smith, 1986; Muelkey, Ambrose, and Barnwell, 1986). It is a one-dimensional, dynamic, compartmental model that can be used for simulation of the movement of chemicals in an unsaturated zone within and immediately below the plant root zone. The model is divided into two major components, namely hydrology (and hydraulics) and chemical transport.

The hydrology and erosion components use the SCS runoff curve number and the universal soil loss equation, respectively. Evapotranspiration is estimated directly from the pan evaporation data and/or by empirical equations. Soil-water capacity terms, including field capacity, wilting point, and water saturation content, are used for simulating water movement through the soil column within the unsaturated zone. The model can also consider the application of irrigation.

Pesticide application onto plant foliage or into soils is also considered. Dissolved, adsorbed, and vapor-phase concentrations in the soil are estimated by simultaneously considering the processes of pesticide uptake by plants, surface runoff losses, erosion, decay, volatilization, foliar washoff, advection, dispersion, and retardation. The model is dynamic, hence pulse (instantaneous) pesticide loads can be simulated.

PRZM is also an integral part of a unsaturated/saturated zone model RUSTIC (Dean et al., 1989). RUSTIC is an acronym for risk of unsaturated/saturated transport and transformation of chemicals concentrations. This shell model links three subordinate models (including PRZM) in order to predict pesticide fate and transport through the crop root zone, and saturate zone to drinking water wells.

Data requirements to run the model are extensive. Predictions are made on a daily basis. The model only simulates organic chemicals in a one-dimensional downward transport. The model is available for PC computers and can be obtained from the Center for Exposure Assessment Modeling of the U.S. Environmental Protection Agency, Athens, Georgia.

Water Erosion Prediction Project (WEPP) Hillslope Profile Model

A new generation water erosion model that was developed by the USDA-ARS National Soil Erosion Research Laboratory (Foster and Lane, 1987;

Lane and Nearing, 1989; Laflen, Lane, and Foster, 1991). It is a continuous simulation model, although it can be run on a single storm basis. By continuous simulation the model "mimics" the processes that are important to erosion prediction as a function of time, and as are affected by management decisions and the climatic environment. The output of the continuous simulation are time-integrated estimates of erosion. The model calculates both detachment and deposition, hence, the delivery process is considered. The output includes both on-site and off-site erosion effects. The on-site effects of erosion include the time-integrated (average annual) soil loss of the net soil loss over the area on the hillslope, which is analogous (but not identical) to USLE estimates. The output also includes deposition of the net deposition over the area on the hillslope. The output describing off-site effects includes sediment loads and particle size information. The output options also include the potential for obtaining monthly or daily (storm-by-storm) estimates of the on-site and off-site effects of erosion.

The model has six components (Fig. 9.14): climate generation

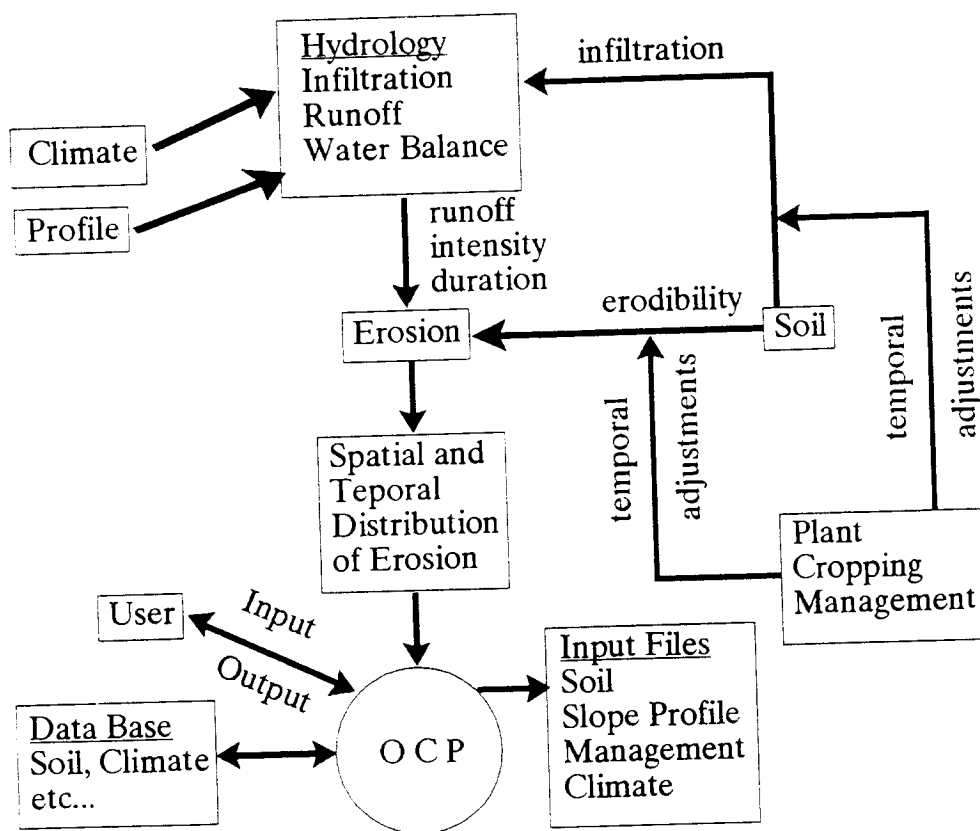


FIGURE 9.14. Flow chart of the Water Erosion Prediction Project (WEPP) model. (After Nicks et al., 1989.)

(CLIGEN), hydrology, plant growth, soils, irrigation, and erosion. The climate generator is run separately from the WEPP model. It generates rainfall amount, duration, maximum intensity, time to peak intensity, maximum and minimum temperature, and solar radiation for the location. The hydrology component calculates infiltration, the daily water balance, including runoff, evapotranspiration, and deep percolation. Infiltration is calculated by the Green-Ampt infiltration equation. Runoff is calculated using the kinematic wave equation or its approximation. The water balance routines are a modification of the SWRRB water balance (Williams, Nicks, and Arnold, 1985) and account for snowmelt, percolation below the root zone, movement of water downward between soil layers within the root zone, and both bare soil evaporation and plant transpiration. The plant growth component calculates the growth, senescence, and decomposition of plant material. This component also calculates leaf area index for transpiration calculations. Grazing and harvest removes material at user-specified intervals. After harvest, decomposition of vegetative residue, if present, is simulated.

Many of the soil parameters that are used in hydrology and erosion calculations change with time as a result of tillage operations, freezing and thawing, compaction, weathering, or history of precipitation. The soil component makes adjustments to soil properties on a daily time step. The irrigation component simulates the effects of irrigation, which are simulated as a rainfall event of uniform density. Several irrigation schemes and options can be accommodated.

The erosion component uses the steady-state sediment continuity equation as a basis for erosion computations. Soil detachment in the interrill areas is calculated from the rainfall intensity. Soil detachment in the rills occurs if the hydraulic shear stress is greater than critical shear and the flow is at less than transport capacity. Deposition occurs when the sediment load is greater than the transport capacity of the flow. The calculated sediment sizes are a function of the original soil material and the preferential deposition of certain sediment-size fractions along the hillside profile.

The erosion predictions from the WEPP profile model are applicable to "field size" areas or conservation treatment units. The maximum modeling unit size is about 270 ha (640 acres). The profile model is also applicable to areas that are farmed over and known as concentrated flow or "cropland ephemeral gullies." The model can also be used for modeling range and forest lands with large concentrated flow channels, and for estimating erosion in terrace channels or grassed waterways on cropland.

The models just discussed are operational models that can be obtained

TABLE 9.2 Summary Overview of Agricultural Models

Model	Modeling Tasks	Event or Continuous Simulation	Lumped or Distributed Parameter	Number and Type of Pollutants Modeled	Agency, Support, Source
AGNPS	SR	C, SE	DP ^a	S, N	USDA-ARS, Morris, Minnesota
ANSWERS	SR	SE	DP	S	University of Georgia, Tifton
ARM ^b	SR	C	L	S, N, 1 OP	
CREAMS- GLEAMS	SR, SW, GW	C, SE	L	S, N, 20 OP	ASDA-ARS, Tifton, Georgia
HSP-F	SR	C	L	S, N, 1 OP	EPA, Athens, Georgia
PRZM	SW, GW	C	L, one-dimensional	S, OP	EPA, Athens, Georgia
SWRRB	SR	C	L	S, N, OP	USDA-ARS, Temple, Texas
WEPP	CL, SR	C, (E)	L	S	USDA-ARS, West Lafayette, Indiana

Note: SR = surface runoff; SW = soil water; GW = ground water; CL = meteorology; C = continuous; SE = single event; L = lumped parameter; DP = distributed parameter; S = sediment; N = nutrients; OP = organic pesticides.

^aDistributed parameter watershed description, lumped parameter routing.

^bARM is a part of the HSP-F model.

either free of charge or for a nominal charge from the agencies that developed and maintain the model. Table 9.2 summarizes some of the most important features of these operational models.

STOCHASTIC AND NEURAL NETWORK MODELS

Stochastic Autoregressive, Moving Average, Transfer Function Models

More than 15 years experience with the development of deterministic urban sewerage flow and quality models, along with their calibration and verification with numerous data, have revealed that:

- Deterministic hydrological models of flow and quality are highly variable and often difficult to reproduce even after extensive calibration of the model

- Increased complexity of the models does not a priori imply a better representation of the process
- The best models can only reproduce general trends in data, leaving a significant variability component aside

The reason for the general failure of deterministic models to represent the variability of the data is, simply said, the deterministic nature of the models themselves, which inherently neglects random and other variations. This defect has been known to researchers for a long time and they have tried to alleviate the problem by Monte Carlo simulations (Brutsaert, 1975). A serious drawback of using deterministic steady-state models and Monte Carlo simulations is the failure to detect the non-deterministic variation of the output. Only the uncorrelated variations of the input and system parameters can be adequately simulated. If these input and system parameters are cross-correlated, the Monte Carlo simulation must be modified to incorporate the cross-correlation, which is tedious and sometimes impossible. The second drawback is the use of steady-state deterministic models. Essentially, the variations of the input and system parameters are filtered by a deterministic model that is not sufficient to identify the entire variability of the output.

Time series analysis techniques have been successfully used in hydrology and water resources for over 20 years. In the earlier applications, one method for analyzing time series was to decompose the series (say, for example, daily or hourly BOD or suspended solids loads in combined sewers) into three components: (1) trend, (2) periodic component, and (3) random fluctuations. A trend was estimated by fitting a straight line or a polynomial to the series, the periodic component was extracted by Fourier analysis, and forecasting was made using the two extracted functions. However, in one of the first papers on the use of stochastic models for wastewater flow analyses, Berthouex et al. (1975) stated that such decompositions can be misleading, even for a deterministic process. Treating a stochastic process (the flow in combined and separate sewers is undoubtedly a stochastic process) the same way is even more dangerous, because it could give highly misleading results.

There are literally dozens of articles in the literature on the application of stochastic models to stream flow or rainfall and other hydrological events; however, only a few articles and reports were published in the last 12 years on applications of stochastic models to sewer flow quantity and quality (Berthouex et al., 1975, 1976, 1978; Barnes and Rowe, 1978; Novotny and Zheng, 1989; Capodaglio et al., 1990; and Novotny et al., 1990a, 1990b). In spite of the apparent lack of interest by researchers, there is no doubt that diffuse flow and quality and, hence, diffuse pollu-

tion are stochastic processes. Stochastic models are highly appropriate and can be successfully used, especially for forecasting and real time control of urban sewerage systems.

The term *stochastic* does not mean “completely random” or “unpredictable.” Box and Jenkins (1976) the pioneers of stochastic modeling, have emphasized that such systems contain both the dynamic (deterministic) and stochastic nature of the underlying processes. There are basically two types of stochastic models:

1. Univariate ARMA
2. Transfer function
 - a. Single input–single output (SISO)
 - b. Multiple input–single output (MISO)

Figure 9.15 presents the concepts of stochastic modeling.

The univariate ARMA models relate the present or predicted value of the parameter to the series of its immediate past values. As shown on Figure 9.15 an uncorrelated random input called “white noise” is the primary driving force of the univariate stochastic model. This random input is “filtered” by the system transfer function to produce the output autocorrelated series. The nomenclature of ARMA modeling makes use of the so-called backshift operator B such as $X_{t-(\Delta t)} = B \times X_t$, $X_{t-2(\Delta t)} = B^2 \times X_t$, and so forth.

The univariate ARMA model is

$$Y(t) = \frac{\theta(B)}{\Phi(B)} a_t = G(B)a_t \quad (9.6)$$

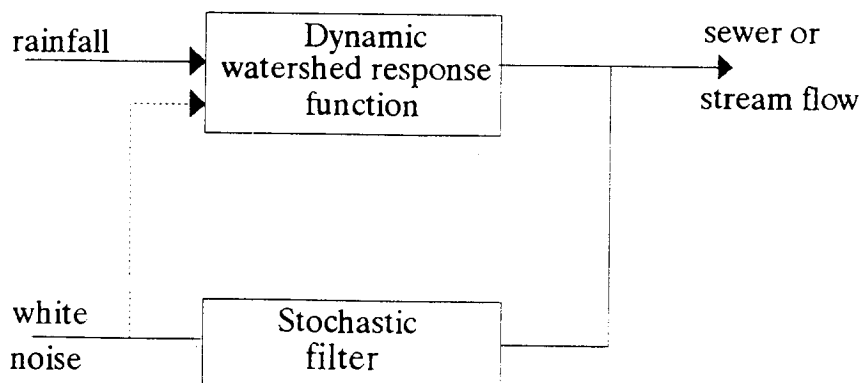


FIGURE 9.15. Representation of a stochastic modeling system. In addition to rainfall, other inputs may also be considered.

where

- Y = the values of the time series
- $\Phi(B)$ = polynomial of autoregression coefficients
- $\theta(B)$ = polynomial of moving average coefficients
- $G(B)$ = Green's function describing the behavior of the series
- a_t = uncorrelated, zero-mean random-shock series

Univariate models were used by Barnes and Rowe (1978) for analyzing combined sewer flow series of Springfield, Ohio, and Anderson, Indiana. For example, the ARMA model of Barnes and Rowe for flow in a combined sewer in Springfield after simplification (Novotny, 1988) is

$$(1 - 0.71B)Q_t = (1 + 0.17B)a_t \quad (9.7)$$

which is a first-order univariate ARMA(1,1) model. The number in parentheses denotes the order of autoregressive (left side of the equation) and moving average (right side) polynomials. Berthouex et al. (1975) found that a first-order univariate ARMA model fits the influent BOD time series of sewer flow in Madison, Wisconsin. Higher order models are needed for simulating quality parameters. Methods for estimating the coefficients of univariate ARMA models were presented by Box and Jenkins (1976), Pandit and Wu (1982), Capodaglio et al. (1990), and others.

The transfer function ARMA models are

$$Y(t) = \sum_j \omega^j(B)X_t^j + N_t \quad (9.8)$$

where

- $\omega^j(B)$ = transfer function polynomial for the j th input series
- X^j = input series
- N_t = noise series described by a univariate ARMA model (Equation 9.6)

Novotny et al. (1990a) documented that stochastic transfer function models can successfully separate dry-weather- (sewage) and wet-weather-driven component flow and quality components in combined sewer systems. Capodaglio et al. (1990) provided a methodology for univariate, MISO, and SISO model identification. Novotny and Zheng (1989) also showed that the stochastic rainfall-runoff transfer function is theoretically identical to the deterministic unit hydrograph, and that the deterministic transfer function could and should be incorporated into the stochastic models.

Figure 9.16 shows a comparison of measured time series of BOD loads in combined sewers and one-step-ahead forecasts by stochastic ARMA-transfer function models. Stochastic models are more crude and can incorporate only a few input and system parameters. In a simplified way, they can have a *deterministic* core model as documented by Novotny and Zheng (1989) and Novotny et al. (1990a, 1990b). Stochastic models can only detect and incorporate the input-output relationships that are not overwhelmed by noise. However, the fact that the stochastic models can differentiate between the deterministic relationships and noise (which can be correlated or uncorrelated) makes them an attractive, unbiased tool for forecasting and control. Stochastic transfer function models are very useful for separation of dry-weather and wet-weather (rainfall-driven) quantity and quality components in combined sewers or storm sewers and surface channels with strong infiltration and/or base-flow inputs. Figure 9.17 shows a rainfall-runoff transfer function for a combined sewer outlet identified by stochastic modeling. The shape of the hydrograph contains faster (presumably surface runoff) and slower (possibly infiltration) components.

The stochastic time series models require for identification an uninterrupted time series in which data are collected at uniform time intervals. Techniques are available for substitutions of missing data.

Neural Network Models

Neural network models represent a new and very powerful modeling technology that, to our knowledge, has not yet been used for modeling urban (or nonurban) nonpoint pollution. They are basically multiple-input-multiple-output complex models that use parallel processing and learning to find a relationship between inputs and outputs (Novotny et al., 1990a). A neural network model consists of a large number of simple *processing elements* that are connected by one directional channels called *connections*. Each processing element can have multiple input signals, but produce only one output. The input signals in each processing unit are weighed by a transfer function that operates on the input signals, weighs them, and modifies the weights according to the strength of the stimulus by the input signal.

There are two operational modes of the neural network—*learning* and *recall*. During learning the system (neural network) adopts connection weights between processing units in response to stimuli being processed at the input and output. These stimuli, in the case of quality and a sewer flow, are the measured time series of, for example, treatment plant influent data or a time series measured by specialized surveys. Such time

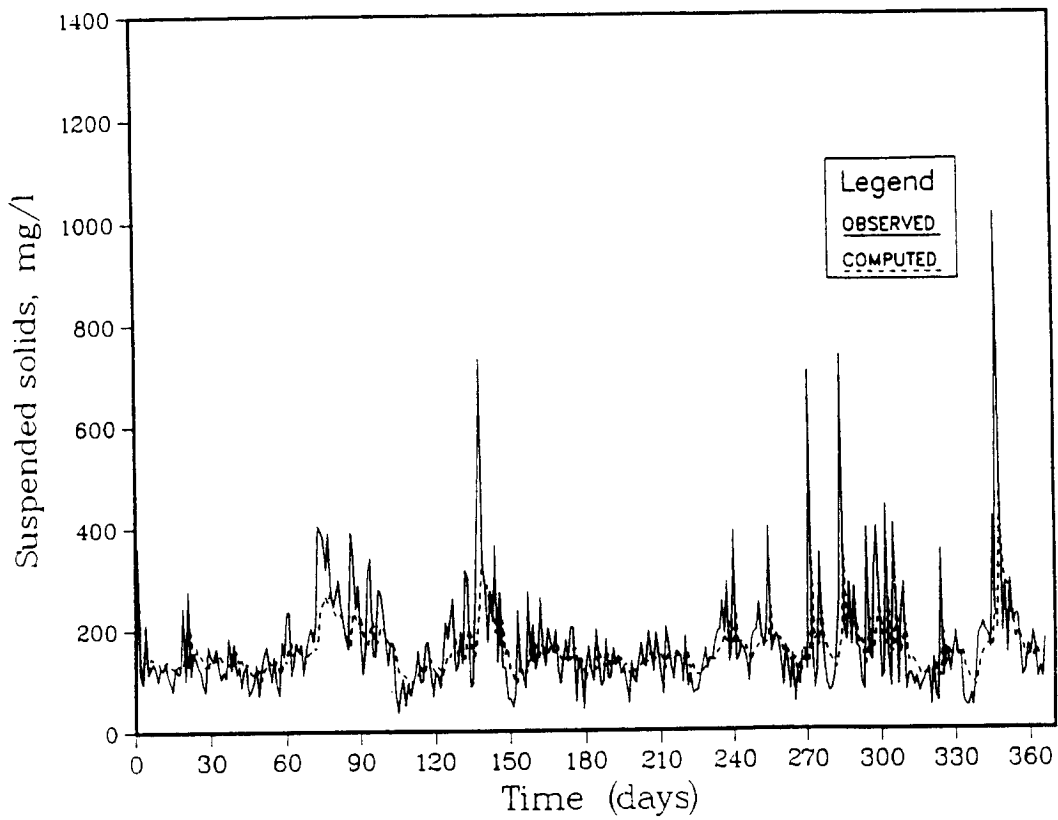


FIGURE 9.16. Comparison of measured BOD concentrations and corresponding one-step-ahead forecasts by an ARMA-transfer function model of a combined sewer system.

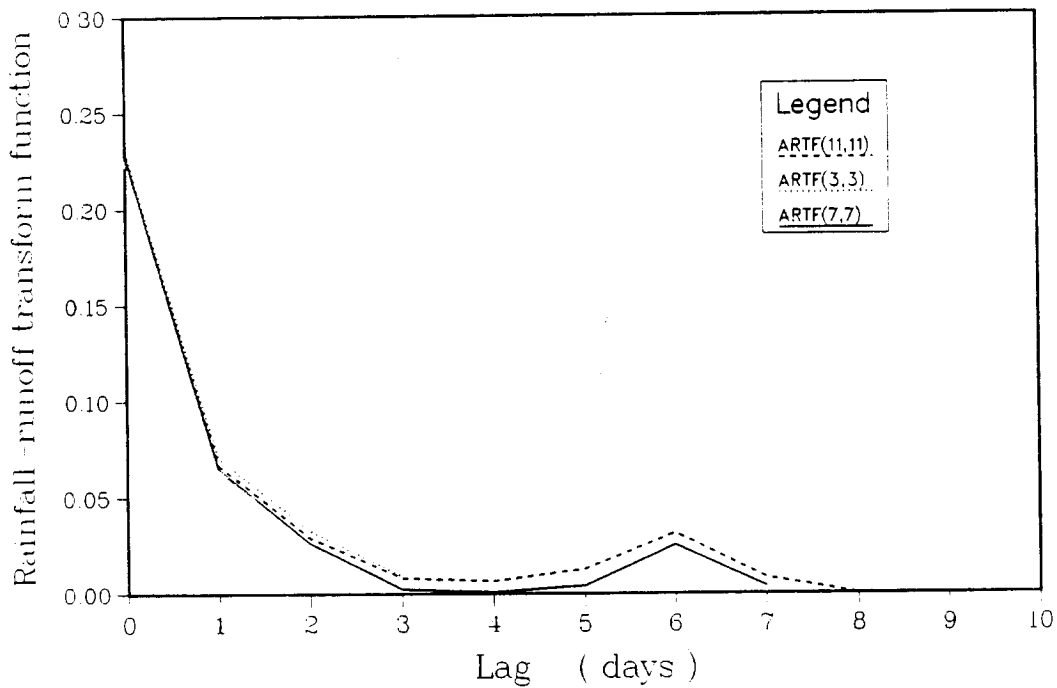


FIGURE 9.17. Unit hydrograph derived from the stochastic model of rainfall-runoff transformation for a combined sewer basin.

series do not have to be continuous as they would be for ARMA models. The input stimuli can be, for example, rainfall, atmospheric deposition (air-pollution levels), traffic density, and similar other parameters used for deterministic modeling. The network learns by implementing a learning rule that determines how the weights will adapt in response to the learning example. The learning process usually involves showing the network learning patterns (examples) many times. The network is said to have *learned* when the error between the measured output series and the output estimated by the network reaches some prescribed value. The type of learning described herein is called *supervised learning*, in which the input and output stimuli are presented to the system.

In the *recall* mode, weights inside the neural network model are frozen and the system produces output from input time series (such as rainfall and other typical inputs mentioned before). The system may be asked to periodically *relearn* the process any time the system parameters or the process has changed. In this way, the neural network is highly adaptive. Although being highly adaptable to modeling any system, including diffuse pollution, neural network models have not been used in this domain at the time of writing this book.

REAL-TIME CONTROL MODELS FOR URBAN SEWERAGE SYSTEMS

Real-time control (RTC) systems actually represent a set of models and concepts that are used for optimal and flexible control of urban sewerage systems (Béron et al., 1984; Schilling, 1989). In a typical sewerage system, especially in those involving combined sewers, there are numerous points that are "bottlenecks." The downstream capacity of the sewer system or the capacity of the treatment plant are potential bottlenecks.

The invariant steady-state model of operation typical for most sewerage systems dictates that when the bottleneck capacity is exceeded, the excess flow is discharged into the receiving body of water, and hence, overflows. The steady-state solution to the problem is to enlarge the bottleneck. This mode of operation is highly inefficient and results, on the one hand, in large quantities of sewage being discharged without treatment into the receiving bodies of water while, on the other hand, there may be unused capacity downstream. A conversion of the system into a dynamic RTC operation may be more economical than conventional structural enlargement of the bottleneck.

In a dynamic RTC scheme, the excesses are temporarily stored in an auxiliary or in-line storage, and the stored flow is then manipulated either by later release for treatment when the flow subsides below the

capacity of the bottleneck, or by a side treatment and release into the receiving body of water if it is permitted by the receiving water standards. Commonly, the waste-assimilative capacity of receiving bodies of water may be greater during wet-weather flows due to higher dilution.

In a real-time control system (RTCS) the following elements apply (Schilling, 1989):

- A (measurement) sensor that is used to monitor an on-going process, for example, water-level gauge, turbidity meter
- A (corrective) regulator that manipulates the process, for example, gates and valves
- A controller that causes the regulator to bring the process back to its desired value (set point)
- A communication system that carries the measured data from the sensor to the controller and the signals of the controller back to the regulator, for example, a telemetry system

The RTC systems do not a priori imply automation. As a matter of fact, significant and substantial operator interface and override are desirable. In either case, computers equipped with RTC software are an integral part of the RTC system.

In the context of RTC, the meaning of "system" is the controllable part of the urban drainage system. A real-time control can be planned and implemented if numerical simulation models (deterministic, stochastic, neural network, etc.) are available and included in the RTC system. The following processes have to be modeled in order to obtain a comprehensive overview of the performance of the system:

- The input to the system: deterministic and/or stochastic models of rainfall-runoff sewer flow and quality models will provide short-term forecasts of the inputs (Labadie, Lazaro, and Morrow, 1981)
- The system response to the input: transfer function models will provide pollutant loads, flows, and water levels at the key bottleneck points of the system
- The total output to the environment: all partial outputs will be summarized in order to minimize, in an optimal way, the total (pollution) output from the system
- The response of the environment to the output from the system: the total output is compared with the waste-assimilative capacity; further restrictions are imposed on the system if the waste-assimilative capacity is exceeded

In a typical application, the input to an urban drainage (nonpoint) system is measured or predicted rainfall. Radar measurements have been

considered for rainfall forecasting. Other inputs, such as traffic density and atmospheric deposition, may also be considered. The input is then transformed by a deterministic or stochastic model (similar to those discussed in the preceding sections) into input parameters, such as predicted flows and pollutant loads. Chapter 10 discusses various strategies employed in RTC systems.

CALIBRATION AND VERIFICATION OF DETERMINISTIC MODELS

Due to the fact that stochastic models are identified from measured time series by an unbiased multiple-regression analysis, further calibration and verification of the models is not needed as long as the series of residuals (a series of one-step-ahead forecasts minus corresponding measured values) is an uncorrelated zero-mean series—the “white noise”—which is the fundamental condition in the identification and design of stochastic models. Similarly, the neural network models are developed from measured input-output multiple time series; however, the white noise residual series is not inherently guaranteed.

Calibration and verification do not make sense for simple unit load models and procedures. A majority of deterministic hydrologic models require calibration and verification of the model parameters during the design and model identification process. Many mathematical equations and formulas used in the deterministic flow-quality models are of an empirical or semiempirical nature, requiring knowledge of a large number of coefficients and reaction rates. Although the model manuals often provide guidelines for a rough estimation of the most important parameters, the ranges are commonly quite wide. On average, twenty or more coefficients must be input for each subunit of a hydrological diffuse-pollution loading model.

As an example consider soil permeability and surface storage, two very simple hydrological parameters. Although the magnitude of surface runoff is very sensitive to these parameters, their exact measurement is tedious and they are not uniform even on very small (less than one hectare) watershed segments. The permeability ranges reported in the U.S. Soil Conservation Service soil maps are broad; for example, silt loam soils are listed as having permeability between 1.6 and 5 cm/hr. In addition, no adequate physical method is available to measure surface storage. Under these circumstances, the only way to arrive at a set of coefficients adequately describing the watershed is by calibrating the model against measured data. Usually, for calibration and verification of event-oriented models (level IV) hydrographs and pollutographs of a few

representative storms should be available. For calibration and verification of dynamic continuous models (Level III and especially, Level V, models), continuous long-term series may be also needed.

Calibration means varying the coefficients of the designed model within acceptable ranges until a satisfactory agreement between measured and computed output values is achieved. Trying to adjust all twenty or more coefficients for each land-use segment at the same time is tedious and often impossible. For diffuse-pollution hydrologic models, the hydrologic components must be calibrated first, followed by erosion, and finally the pollutant component. Also, all models are more sensitive to a few important variables and less to others. As a rule of thumb, the variables to which the model is most sensitive should be calibrated first, with the other coefficients kept at their optimal or average values. The hydrologic components of diffuse-pollution models are sensitive to the magnitude of surface storage and soil permeability, which determine the runoff volume. Surface roughness affects the magnitude and time location of runoff peaks. Other variables—slope, soil moisture characteristics—are less important.

In the erosion component, the factor to which soil loss is most responsive is the vegetative factor, *C*. For example, a typical range of the cover factor for grassed areas is between 0.01 and 0.03, a 300% difference. Uncertainty about the sediment delivery will complicate the calibration. Adsorption characteristics and attenuation rates are most important for the pollutant submodel.

Since calibration is a subjective process requiring experience, Delleur and Baffaut (1990) developed an expert system for calibrating the Storm Water Management Model, and reported improved adequacy of the model.

Once the model has been calibrated, that is, once a satisfactory fit of computed and simulated data has been achieved for one calibration storm or time series, the model must be verified. *Verification* is accomplished by running the model with the coefficients established during calibration and with inputs corresponding to another (verification) storm and/or time series. If a satisfactory fit of computed and measured data is obtained, calibration and verification are accomplished. Very often this is not the case, and calibration and verification must be repeated until a satisfactory fit is found. For some locations, data for calibration and verification of hydrological urban runoff models can be retrieved from data banks such as the EPA's STORET (Huber et al., 1979, 1982).

Caveats. Often the model's structure may have an excessive number of "degrees of freedom." For example, sediment load can be changed by varying three parameters of the universal soil loss equation, slope, and

the delivery ratio. There is no methodology available to distinguish between the parameters, yet selection of the wrong magnitude of the parameter can lead to erroneous conclusions on management alternatives. Therefore, the modeler must be experienced and must be familiar with the watershed. A reconnaissance tour of the watershed by the modeler, who should take photographs and notes, should always be a part of the model design, calibration, and verification process.

MONITORING AND DATA ACQUISITION

Data are collected routinely by agencies for a multitude of purposes. Very often these data bases are suitable for the design and identification of models. For example, sewerage agencies and industrial wastewater sources must routinely and periodically, often daily, collect information on flow and various quality parameters in sewers, including storm sewers. Soil data have been collected throughout the United States for the preparation of soil maps, meteorological data are collected by the National Oceanic and Atmospheric Administration (NOAA), and flow quality data in streams is available from the U.S. Geological Survey, which has offices in every state. Special surveys and monitoring, however, are almost always needed for calibration, and verification of deterministic complex models. This type of data acquisition is different from the routinely performed monitoring by agencies that require preliminary design and a carefully executed monitoring program. Specially designed monitoring programs are also needed for establishing unit loads, quantity and quality characteristics of diffuse pollution, identification of stochastic and regression (statistical) models, and for research of the processes involved in the generation and transmission of diffuse pollution.

The data needed for design (identification), calibration, and verification of hydrologic models can be divided into the following groups, which includes an approximate list of monitored parameters and variables:

1. System parameters
 - a. watershed size
 - b. subdivision of the watershed into homogenous subsegments
 - c. imperviousness of each subsegment
 - d. slopes
 - e. fraction of impervious areas directly connected to a drainage channel or sewer
 - f. maximum surface storage (depression plus interception storage)
 - g. soil characteristics, including texture, permeability, erodibility, and composition

- h. crop and vegetation cover
- i. curb density or street gutter length
- j. sewer system or natural drainage characteristics
- 2. State variables
 - a. ambient temperature
 - b. reaction rate coefficients
 - c. adsorption–desorption coefficients
 - d. daily accumulation rates of litter
 - e. traffic density and speed
 - f. potency factors for pollutants (concentration of pollutants on street dust and dirt and on soil)
 - g. solar radiation (for some models)
 - h. growth stage of crops
- 3. Initialization parameters
 - a. length of antecedent dry period
 - b. initial soil moisture
 - c. depth of snowpack
 - d. initial concentration of pollutants in soil
- 4. Input variables
 - a. precipitation
 - b. atmospheric fallout
 - c. evaporation rates
- 5. Calibration and verification parameters
 - a. storm-water and receiving water flows (hydrographs)
 - b. water quality parameters (quality histograms)
 - c. biological assessment of receiving waters (toxicity impact on biota)

Sources of Data

Walesh (1989) divided the data for planning studies into three groups: (1) completed or ongoing studies; (2) natural resources data; and (3) infrastructure data. In the first category, data for the design of diffuse-pollution models and for planning can be found in the past studies carried out under the mandate of the Section 208 of the 1972 Clean Water Act (208 studies), data obtained by the National Urban Runoff Project by the U.S. Environmental Protection Agency, Flood Plain Management studies by various planning agencies, and numerous other studies. Such data may contain calibration and verification surveys for runoff quantity–quality modeling.

Data related to land and land use can be obtained from maps and/or aerial photographs and remote sensing. The data obtainable by high plane or satellite imagery can be digitized in order to provide more

specific information on the degree of imperviousness, vegetation cover, surface roughness, and soil moisture. Soil data are available from U.S. Soil Conservation Service (SCS) maps. These maps provide information on soil type, texture, slope, approximate ground-water table, permeability, erodibility, soil profile, and other valuable information. An example of the map was shown in Figure 6.2. Land-use data can be obtained from the U.S. Bureau of Land Management, U.S. Census Bureau, and from local and regional planning agencies.

Hydrologic and basic water quality information is collected and published regularly by the U.S. Environmental Protection Agency and by the U.S. Geological Survey. Both agencies have computerized systems for storing and retrieving information on water quality and quantity. Information on dust and dirt accumulation and traffic densities in some urban areas is usually available from the local city engineer's office, sanitation department, and air-pollution control agencies. Meteorological data are routinely measured by NOAA.

It must be emphasized that the format and frequency of data obtainable from various agencies (with the exception of meteorological data) usually do not conform to modeling requirements, and often significant time and financial resources must be spent to get available information into conformity with the requirements of the models. Additional in situ surveys and monitoring are necessary to gather missing information. Expenses associated with data-collection activities often exceed the expenses for setting up and running the model.

Monitoring programs must also be set up to comply with the regulatory requirements of the National Pollution Discharge Elimination Systems (NPDES) permit system. Both combined and separate storm sewer systems in larger urban areas are covered by the NPDES permitting system, which requires monitoring both flows and quality and long-term load estimations by modeling.

Field Monitoring

A typical monitoring station has the following components:

1. Rain gauge
2. Wet- and dry-atmospheric deposition collector
3. Flow monitoring device
4. Quality monitoring device
5. Power source
6. Telecommunication link or data recorder

Rain Gauges and Atmospheric Deposition Monitoring

Rain gauges measure rainfall. Typically the rain gauge is activated when rainfall depth is more than 0.25 mm (0.01 in.). Smaller rainfalls are recorded as *trace precipitation*. Many rain gauges signal each 0.25-mm occurrence of rainfall.

Wet and dry deposition samplers are essentially two collecting containers. One is always open and collects the bulk atmospheric deposition (wet and dry); the other is covered, but the lid opens when a moisture sensor is activated by precipitation. This sampler collects only wet precipitation. The dry deposition is thus the difference between the bulk and wet depositions.

Flow Measurements

Typical monitoring programs include monitoring of flow and its quality. In most cases, measuring flow involves measuring the depth (stage) in the channel or sewer that is then converted to flow using the so-called rating curve for the section where the flow stage is measured.

In the most simple case, the flow stage is measured manually or by a recording float device. In order to avoid backwatering effects, the flow should be impounded. This is best accomplished by standard sharp-crested weirs (triangular, rectangular, trapezoidal), as shown on Figure 9.18, or by narrowing the cross section, as is done in a standard Parshall flume. Rating curves (flow-stage relationships) are available for most standard weirs and flow-measuring flumes. These rating curves must be checked by several in situ flow measurements using simple devices, such as a bucket and stopwatch, or more complex flow-measuring devices. Periodic field checks and manual flow measurements are always necessary and are integral to any monitoring programs.

Velocity sensors can help to alleviate the problem with backwatering effects that could greatly affect the flow-stage relationship of the rating curve. Many newer monitors incorporate both depth and velocity measuring in a single microprocessor that can be telemetrically interrogated. Such sensors must be calibrated by manual measurements at the time of installation. It is important that the point velocity sensor is placed at the right depth, that is, one that corresponds to the average cross-sectional velocity.

Telemetry rather than recording is becoming more common and economical today. Any site that requires permanent NPDES monitoring should consider telemetered monitors. Once installed and calibrated, a telemetered flow monitor should require much less labor to maintain than a portable monitor. On the other hand, a portable manually operated



FIGURE 9.18. Typical monitoring station for a small watershed used for gathering calibration and verification data and/or for storm-water monitoring. (Photo: V. Novotny.)

monitor is more economical for occasional monitoring surveys, such as those required for the calibration of event-oriented models.

In storm-water monitoring stations the flow meters are commonly interconnected with the sampler, hence, they activate sampling during episodes of runoff. They also send a signal to the quality sampler when a sample is to be collected. For example, the flow meter can signal the sampler each time 50 m^3 of flow has passed by the flowmeter. In this way a flow-weighted composite sample can be collected.

Quality Monitoring

Quality monitors are devices that periodically withdraw a small volume of water (or wastewater) from the stream at the monitoring site. This withdrawn volume is then analyzed for various constituents. On-site analyses are typically done for only a few constituents, generally those that can be measured by electrochemical probes such as dissolved oxygen, turbidity, temperature, pH probes, and a few others. This information can be telemetrically transmitted. For most other constituents, the samples collected by the sampler must be transported to a laboratory and anal-

alyzed therein. Sample preservation between the time of the withdrawal from the channel and the time of analysis must be implemented. Many automatic samplers use refrigeration to preserve the sample.

There are basically two types of sampling. *Grab samples*, collected periodically at predetermined intervals and analyzed separately, can document the extent, frequency, and general variability of quality. The sampling interval should be selected according to the rate of quality change or frequency of quality variation. It may range from intervals as short as 5 minutes for some flush storm runoff events and CSOs to as long as 1 hour or more. Daily or even less frequent grab sampling is unreliable and can be only used for monitoring slowly changing streams, such as river flows. When the source composition varies in space, as it is in a typical urban drainage system, sampling should be done at several appropriate locations. Most of the runoff of storm sewer monitoring stations are flow activated, hence, no sampling is conducted when the flow in the conduit is below a prespecified minimum level.

A *composite sample* is a mixture of grab samples or continuously withdrawn flow. The mixing of the volumes withdrawn from the flow is either based on a constant volume–constant time (time composite) or is flow proportional. *Time composite* samples provide the average concentration of the constituent during the time of composition irrespective of flow, or

$$\hat{C} = \Sigma C_i/n$$

where

\hat{C} = time-averaged composite concentration

C_i = individual concentrations of grab samples

n = number of samples

A *flow proportional composite sample* is generated by specifically designed automatic monitoring equipment that measures flow and activates grab sampling when prespecified flow volume passes through the flow-measuring device. Then the concentration of the composite sample can be represented by

$$\hat{C} = \Sigma C_i \times \Delta Q / (n \times \Delta Q)$$

The flow proportional composite sample is representative of the average mass concentration for the sampled runoff event.

There are certain common sense rules related to monitoring. Inadequate frequency of data acquisition and incomplete monitoring may be useless,

and high-frequency monitoring and sampling for many constituents may be costly and will create a backlog of unusable data.

High-frequency grab sampling is used for monitoring flush runoff flow events. Often the period of data acquisition is in minutes; however, high-frequency sampling precludes an extensive variety of monitored constituents. Low-frequency composite sampling is typical for between the event sampling that is to characterize infiltration and base-flow contributions. Low-frequency grab sampling (for example, daily, weekly, or monthly intervals) may not provide representative results and should be avoided (essentially, such sampling may be a waste of funds).

High-frequency or continuous monitoring is also feasible for constituents that can be reliably monitored by electronic or electrochemical probes. Such constituents include pH, temperature, dissolved oxygen concentrations, conductivity (which is related to salt or dissolved solids contents), and turbidity (which can be correlated to suspended solids content). Relatively inexpensive in situ or in-laboratory analyses include organic contents expressed as the chemical oxygen demand (COD), total organic carbon (TOC), total suspended and total dissolved solids (TSS, TDS), volatile suspended and dissolved solids, nitrates, and metals. These analyses can still be performed on grab samples, both filtered and unfiltered. A filtered sample represents dissolved contamination, while the difference between the total and filtered concentrations of the constituent reflects the contamination of the suspended sediment. Analyses for organic chemicals (pesticides, polyaromatic hydrocarbons (PAHs), PCBs, etc.), bacteria, oil and grease, nitrogen compounds other than nitrate (total Kjeldahl nitrogen, ammonia), phosphates, and asbestos are more demanding and costly and are typically performed only on composite samples. In many cases, a few surrogate constituents are selected instead of analyzing a complete spectrum of constituents. For example, in a group of toxic metals, lead and/or zinc and/or cadmium are often selected to indicate pollution by toxic metals. Instead of analyzing for every possible pathogenic microorganism, total and fecal coliform bacteria analyses are commonly used to indicate bacterial and viral contamination. COD or TOC are used in high-frequency grab sampling programs as indicators of pollution by organics. Volatile suspended solids indicate organic contamination of sediments, as does a difference between the total and filtered COD.

Event-oriented models are calibrated and verified using detailed grab sampling over several storm runoff hydrographs. Care should be taken that the grab samples are distributed more-or-less evenly over the rising and receding portions of the hydrograph. Continuous long-term models can be calibrated using daily (wet-weather) composite sampling. How-

ever, long-term sampling must include both work days and weekend (holiday) data.

The guidelines for the NPDES permit for urban storm water require that monitoring data programs provide information on the pollution content of urban runoff, including the first flush portion of runoff. This may require grab sampling programs at various locations of the urban storm drainage system. The new sampling devices available on the market today allow for programmed sample collection in order to characterize separately the more polluted first flush and less polluted tail portion of a storm event hydrograph.

Computer Software. The monitoring station can also be equipped with internal memory and a connection to a computer. The data of a storm event can be printed on the internal chart during the storm event, or stored in the internal memory of the flowmeter for later retrieval by a computer. An internal modem enables convenient retrieval by a remote computer over telephone lines, or a laptop computer can be used for on-site collection of data.

Log-Normal Distribution of Data

Event mean concentration (EMC) and site mean concentration (SMC) are calculated or graphically estimated from monitored grab (within event) or composite (long-term monitoring) data by fitting the data to a log-normal probability distribution function. The mathematical form of the function has been presented in many statistical handbooks, and the concept was also shown in the preceding chapter (Figs. 8.17 and 8.18).

Log-normal distribution has been used and suggested to characterize diffuse-pollution runoff at a sampling site for a large number of samples (Driscoll, 1986). This distribution has a number of important benefits (Woodward-Clyde Consultants, 1989):

- Concise summaries of highly variable data can be developed, and the variability can be identified and dealt with appropriately.
- Comparison of results from different sites, events, etc., are convenient and more easily understood.
- Statements can be made about frequency of occurrence, that is, one can express how often values will exceed various magnitudes of interest.
- A more useful and informative method for reporting data than the use of ranges is provided, one that is less subject to misinterpretations.
- A framework is provided for examining the transferability of data in quantitative manner.

Since most of the deterministic models inherently neglect the statistical variability of the data, statistical evaluation of the monitored data must precede the calibration and verification process in order to minimize subjectivity.

References

- Alley, W. M., and P. E. Smith (1982). *Multi-event Urban Runoff Quality Model*, U.S. Geological Survey Open File Rep. B2-764, U.S. Geological Survey, Reston, VA.
- American Public Works Association (1969). *Water Pollution Aspects of Urban Runoff*, WP-20-15, U.S. Department of Interior, FWPCA (present EPA), Washington, DC.
- Athayde, D. N., C. F. Myers, and P. Tobin (1986). EPA's perspectives of urban nonpoint pollution, in *Proceedings, Urban Runoff Quality-Impact and Quality Enhancement Technology*, Eng. Foundation Conf., American Society of Civil Engineers, pp. 217-225.
- Balmér, P., P. A. Malmqvist, and A. Sjöberg, eds. (1984). *Proceedings, Third International Conference on Urban Storm Drainage*, 4 vol., Chalmers University of Technology, Göteborg, Sweden, 1669p.
- Bannerman, R., et al. (1984). *Evaluation of Urban Nonpoint Source Pollution Management in Milwaukee County, Wisconsin*. Vol. I. *Urban Stormwater Characteristics, Pollutant Sources and Management by Street Sweeping*, U.S. Environmental Protection Agency, Region V, Chicago, IL.
- Barnes, J. W., and F. A. Rowe (1978). Modeling sewer flows using time series analysis, *J. Environ. Eng. Div. ASCE* **109**(EE4):639-646.
- Barnwell, T. O., and J. J. Kittle (1984). Hydrologic simulation program—FORTRAN: Development, maintenance and applications, in *Proceedings, Third International Conference on Urban Storm Drainage*, Vol. 2, Chalmers University of Technology, Göteborg, Sweden, pp. 493-502.
- Barnwell, T. O., Jr., and P. A. Krenkel (1982). The use of water quality models in management decision making, *Water Sci. Tech.* **14**:1095-1107.
- Beasley, D. B. (1986). Distributed parameter hydrologic and water quality modeling, in *Agricultural Nonpoint Source Pollution: Model Selection and Application*, A. Giorgini and F. Zingales, eds., Elsevier, Amsterdam.
- Beasley, D. B., and L. F. Huggins (1981). *ANSWERS Users Manual*, EPA 905/9-82-001, U.S. Environmental Protection Agency, Chicago, IL.
- Béron, P., F. Brière, J. Rouselle, and J. P. Riley (1984). An evaluation of some real-time techniques for controlling combined sewer overflows, in *Proceedings, Third International Conference on Urban Storm Drainage*, Vol. 3, Chalmers University of Technology, Göteborg, Sweden, pp. 1093-1097.
- Berthouex, P. M., et al. (1975). Modeling sewage treatment plant input BOD data, *J. Environ. Eng. Div. ASCE* **101**(EE1):127-138.
- Berthouex, P. M., et al. (1976). The use of stochastic models in the interpretation of historical data from sewage treatment plants, *Water Res.* **10**:689-698.
- Berthouex, P. M., et al. (1978). Dynamic behavior of an activated sludge plant, *Water Res.* **12**:957-972.

- Bliss, N. B., and W. U. Reybold (1989). Small-scale digital soil maps for interpreting natural resources. *J. Soil Water Conserv.* **47**(1):30–34.
- Box, G. E. P., and G. M. Jenkins (1976). *Time Series Analysis, Forecasting and Control*, Holden Day, Oakland, CA.
- Brutsaert, W. F. (1975). Water quality modeling by Monte Carlo simulation, *Water Resour. Bull.* **11**(2):229–236.
- Burrough, P. A. (1986). *Principles of Geographical Information Systems for Land Resources Assessment*, Clarendon Press, Oxford.
- Capodaglio, A. G., S. Zheng, V. Novotny, and X. Feng (1990). Stochastic system identification of sewer flow models, *J. Environ. Eng. ASCE* **116**(2):284–298.
- Carsel, R. F., et al. (1984). *User's Manual for the Pesticide Root Zone Model (PRZM): Release 1*, EPA 600/3-84-109, U.S. Environmental Protection Agency, Athens, GA.
- Chesters, G., et al. (1978). *Pilot Watershed Studies. Summary Report*, International Joint Commission, Windsor, Ontario.
- Chow, V. T. (1972). Hydrologic modeling, *J. Boston Soc. Civ. Eng.* **60**:1–27.
- Chow, V. T., D. R. Maidment, and L. W. Mays (1988). *Applied Hydrology*, McGraw-Hill, New York.
- Cline, T. J., A. Molinas, and P. Y. Julien (1989). An auto-CAD-based watershed information system for the hydrologic model HEC-1, *Water Resour. Bull.* **25**(3):641–652.
- Connors-Sasowsky, K., and T. W. Gardner (1991). Watershed configuration and Geographic Information System parametrization for SPUR model hydrologic simulation, *Water Resour. Bull.* **27**(1):7–18.
- Dean, J. D., et al. (1989). *Risk of Unsaturated/Saturated Transport and Transformation of Chemical Concentrations (RUSTIC)*, EPA 600/3-89/048a, U.S. Environmental Protection Agency, Athens, GA.
- Delleur, J. W., and C. Baffaut (1990). An expert system for urban runoff quality modeling, in *Proceedings, Fifth International Conference on Urban Storm Drainage, Suita-Osaka*, Y. Iwasa and T. Sueishi, eds., University of Osaka, Japan, pp. 1323–1328.
- DeRoo, A. P. J., L. Hazelhoff, and P. A. Burrough (1989). Soil erosion modeling using 'ANSWERS' and Geographical Information Systems, *Earth Surf. Process. Landf.* **14**:517–532.
- Dever, R. J., Jr., and L. A. Roesner (1982). Development and application of a dynamic urban highway drainage model, in *Proceedings, Second International Conference on Urban Storm Drainage*, Vol. I. *Hydraulics and Hydrology*, B. C. Yen, ed., Water Resources Publ., Littleton, CO, pp. 229–235.
- Dever, R. J., Jr., and L. A. Roesner (1983). *Urban Highway Storm Drainage Model*. Vol. 5. *Drainage Design Program User's Manual and Documentations* (Final Report Sep. 76–Mar. 81), Camp, Dresser, and McKee, Inc., Annandale, VA, FHWA/RD-83-045, Federal Highway Administration, Washington, DC, 117p.
- Dever, R. J., Jr., J. A. Aldrich, and W. M. Williams (1983). *Urban Highway Storm Drainage Model*. Vol. 1. *Model Development and Test Application* (Final Report Sep.–Mar. 81), Camp, Dresser, and McKee, Inc., Annandale,

- VA, FHWA/RD-83/041, Federal Highway Administration, Washington, DC, 71p.
- Dickerhoff, L. L., and D. A. Haith (1983). Loading functions for predicting nutrient losses from complex watersheds, *Water Resour. Bull.* **19**(6):951–959.
- DiToro, D. M. (1984). Probability model of stream quality due to runoff, *J. Environ. Eng. Div. ASCE* **110**(3):617–628.
- DiToro, D. M., and M. J. Small (1979). Stormwater interception and storage, *J. Environ. Eng. Div. ASCE* **105**(EE1):43–54.
- Donigian, A. S., and H. H. Davis (1978). *User's Manual for Agricultural Runoff Management (ARM) Model*, EPA 600/3-78-080, U.S. Environmental Protection Agency, Athens, GA.
- Donigian, A. S., and J. D. Dean (1985). Nonpoint source pollution models for chemicals, in *Environmental Exposure from Chemicals*, Vol. II, CRC Press, Boca Raton, FL, pp. 75–105.
- Donigian, A. S., and W. C. Huber (1990). *Modeling Nonpoint Source Water Quality in Urban and Non-urban Areas*, EPA 68-03-3513, Environmental Research Laboratory, U.S. Environmental Protection Agency, Athens, GA.
- Driscoll, E. D. (1986). Lognormality of point and nonpoint source pollutant concentrations, in *Urban Runoff Quality—Impact and Quality Enhancement Technology*, Proceedings. Engineering Foundation Conference, American Society of Civil Engineers, pp. 438–458.
- Driscoll, E. D., D. M. DiToro, and R. V. Thomann (1979). *A Statistical Method for Assessment of Urban Runoff*, EPA 440/3-79-023, U.S. Environmental Protection Agency, Washington, DC.
- Driver, N. E. (1990). Summary of nationwide analyses of storm-runoff quality and quantity in urban watersheds, in *Proceedings, Fifth International Conference on Urban Drainage, Suita-Osaka*, Y. Iwasa and T. Sueishi, eds., University of Osaka, Japan.
- Driver, N. E., and D. J. Lystrom (1987). Estimation of urban storm-runoff loads and volumes in the United States, in *Urban Storm Water Quality, Planning and Management*, Proceedings, Fourth International Conference on Urban Storm Drainage, W. Gujer and V. Krejci, eds., Ecole Polytechnique Federale, Lausanne, Switzerland, pp. 214–220.
- Driver, N. E., and G. D. Tasker (1988). *Techniques for Estimation of Storm-Runoff Loads, Volumes, and Selected Constituents in Urban Watersheds in the United States*, U.S. Geological Survey Open File Rep. 88-191, Denver, CO.
- Evans, B. M., and D. A. Miller (1988). Modeling non-point pollution at the watershed level with the aid of a Geographic Information System, in *Nonpoint Pollution 1988—Policy, Economy, Management, and Appropriate Technology*, American Water Resources Association, Bethesda, MD, pp. 283–291.
- Foster, G. R., and L. J. Lane (1987). *User Requirements: USDA—Water Erosion Prediction Project (WEPP)*, NSERL Rep. No. 1, USDA-ARS, W. Lafayette, IN.
- Giorgini, A., and F. Zingales (1986). *Agricultural Nonpoint Source Pollution: Model Selection and Application*, Elsevier, Amsterdam.

- Gujer, W., and V. Krejci (1987). *Topics in Urban Storm Water Quality, Planning and Management*, in Proceedings, Fourth International Conference on Urban Storm Drainage, W. Gujer and V. Krejci, eds., IAHR-IAWPRC, Ecole Polytechnique Federale, Lausanne, Switzerland.
- Haan, C. T., H. P. Johnson, and D. L. Brakensiek, eds. (1982). *Hydrologic Modeling of Small Watersheds*, Monogr. No. 5, American Society of Agricultural Engineers, St. Joseph, MI.
- Haith, D. A., and L. L. Shoemaker (1987). Generalized watershed loading functions for stream flow nutrients, *Water Resour. Bull.* **23**(3):471-478.
- Haith, D. A., and L. J. Tubbs (1981). Watershed loading functions for nonpoint sources, *J. Environ. Eng. Div. ASCE* **107**(EE1):121-137.
- Henderson, R. J., and G. D. Moys (1987). Development of a sewer flow quality model for the United Kingdom, in *Urban Storm Water Quality, Planning and Management*, Proceedings, Fourth International Conference on Urban Storm Drainage, Ecole Polytechnique Federale, Lausanne, Switzerland, pp. 201-207.
- Huber, W. C. (1986). Deterministic modeling of urban runoff quality, in *Urban Runoff Pollution*, H. C. Torno, J. Marsalek, and M. Desbordes, eds., Springer Verlag, Berlin, New York, pp. 167-244.
- Huber, W. C. (1990). Current methods for modeling of nonpoint source water quality, in *Proceedings, Seminar on Urban Storm Water Quality Management*, American Society of Civil Engineers, Orlando, FL.
- Huber, W. C., and R. C. Dickinson (1988). *Storm Water Management Model. User's Manual*, Version 4, EPA 600/3-88/001a (NTIS PB88-236641/AS), U.S. Environmental Protection Agency, Athens, GA.
- Huber, W. C., et al. (1979). *Urban Rainfall-runoff Quality Data Base: Update with Statistical Analysis*, EPA 600/8-79-004, U.S. Environmental Protection Agency, Cincinnati, OH.
- Huber, W. C., et al. (1982). *Urban Rainfall-runoff-Quality Data Base*, EPA 600/S2-81-238, U.S. Environmental Protection Agency, Cincinnati, OH.
- Hydrologic Engineering Center (1975). *Storage, Treatment, Overflow, Runoff Model, STORM, Users Manual*, Generalized Computer Program 723-S8-L7520, Hydr. Eng. Center, U.S. Army Corps of Engineers, Davis, CA.
- Iwasa, Y., and T. Sueishi (1990). *Drainage Models and Quality Issues*, in Proceedings, Fifth International Conference on Urban Storm Drainage, IAHR-IAWPRC, Suita-Osaka, University of Osaka, Japan.
- Johanson, R. C., J. C. Imhoff, J. L. Kittle, and A. S. Donigian (1984). *Hydrologic Simulation Program—Fortran (HSPF): User's Manual*, Release 8, EPA 600/3-84-006, U.S. Environmental Protection Agency, Athens, GA.
- Johnson, M. G., et al. (1978). *Management Information Base and Overview Modeling*, International Joint Commission, Windsor, Ontario.
- Knisel, W. G. (1980). *CREAMS: A Field Scale Model for Chemicals, Runoff, and Erosion from Agricultural Management Systems*, Conservation Research Rep. No. 26, U.S. Department of Agriculture, Washington, DC.
- Knisel, W. G. (1985). Use of computer models in managing nonpoint pollution from agriculture, in *Proceedings, Non-point Pollution Abatement Symposium*, V. Novotny, ed., Marquette University, Milwaukee, WI.

- Application*, A. Giorgini and F. Zingales, eds., Elsevier, Amsterdam, pp. 383–398.
- Nicks, A. D., V. L. Lopes, M. A. Nearing, and L. J. Lane (1989). Overview of WEPP hillslope profile erosion model, in *USDA-Water Erosion Prediction Project: Hillslope Profile Model Documentation*, L. J. Lane and M. A. Nearing, eds., NSERL Rep. No. 2, USDA-ARS National Soil Erosion Research Laboratory, W. Lafayette, IN.
- Novotny, V. (1985). Role of mathematical models in design and selection of best management practices for control of pollution from urban and urbanizing areas, in *Proceedings, Non-point Pollution Abatement Symposium*, V. Novotny, ed., Marquette University, Milwaukee, WI.
- Novotny, V. (1986). A review of hydrologic and water quality models used for simulation of agricultural pollution, in *Agricultural Nonpoint Source Pollution: Model Selection and Application*, A. Giorgini and F. Zingales, eds., Elsevier, Amsterdam, pp. 9–35.
- Novotny, V. (1988). Modeling of sewer flow quality, *Proceedings, Bilateral Conference US-Italy Urban Stormwater Management*, University of Cagliari, Italy, Water Resources Publications, Littleton, CO.
- Novotny, V., and R. Bannerman (1980). Model enhanced unit loadings of pollutants from nonpoint sources, in *Proceedings, Hydraulic Transport Modeling Symposium*, Publ. No. 4–80, American Society of Agricultural Engineers, St. Joseph, MI.
- Novotny, V., and A. Capodaglio (1991). *Model of Urban Runoff and Sewer Flow (MOURSEF)*, AquaNova International, Mequon, WI.
- Novotny, V., and S. Zheng (1989). Rainfall-runoff transfer function by ARMA modeling, *J. Hydraul. Eng. ASCE* **115**(10):1386–1400.
- Novotny, V., S. Zheng, and X. Feng (1990) Wet weather-dry weather flow and pollutant load separation by ARMA modeling, *Proceedings, Fifth International Conference on Urban Storm Drainage*, IAWPRC-IAHR, Suita-Osaka, University of Osaka, Japan, pp. 403–408.
- Novotny, V., et al. (1979). *Simulation of Pollutant Loads and Runoff Quality*, EPA 905/4-79-029E, U.S. Environmental Protection Agency, Chicago, IL.
- Novotny, V., H. M. Sung, R. Bannerman, and K. Baum (1985). Estimating nonpoint pollution from small urban watersheds, *J. WPCF* **57**(4):339–348.
- Novotny, V., K. Imhoff, M. Olthof, and P. Krenkel (1989). *Karl Imhoff's Handbook of Urban Drainage and Wastewater Disposal*, Wiley, New York.
- Novotny, V., H. Jones, X. Feng, and A. Capodaglio (1990). Time series analysis models of activated sludge plant, *Water Sci. Technol.* **23**:1107–1116.
- Pandit S. M., and S. M. Wu (1983). *Time Series and System Analysis with Applications*, Wiley, New York.
- Potter, W. B., M. W. Gilliland, and M. D. Long (1986). A Geographic Information System for predicting of runoff and non-point source pollution potential, in *Hydrologic Applications of Space Technology*, Publ. No. 160, International Association of Hydrological Sciences, Wallingford, United Kingdom, pp. 437–446.

- Renard, K. G., W. J. Rawls, and M. M. Fogel (1982). Currently available models, in *Hydrologic Modeling of Small Watersheds*, C. T. Haan, H. P. Johnson, and D. L. Brakensiek, eds., Monog. No. 5, American Society of Agricultural Engineers, St. Joseph, MI.
- Reybold, W. U., and G. W. TeSelle (1986). Soil geographic data base, *J. Soil Water Conserv.* **44**(1):28-29.
- Roesner, L. A., R. J. Dever, Jr., and J. A. Aldrich (1983). *Urban Highway Storm Drainage Model*. Vol. 6. *Analysis Module User's Manual and Documentation* (Final Rep. Sep. 76-Mar. 81), Camp, Dresser, and McKee, Inc., Annandale, VA, FHWA/RD-83-046, Federal Highway Administration, Washington, DC, 253p.
- Sanders, W. M. (1976). Non-point source modeling for Section 208 planning, in *Best Management Practices for Non-point Source Pollution Control*, EPA 905/9-76-005, U.S. Environmental Protection Agency, Washington, DC.
- Sartor, J. D., and G. B. Boyd (1972). *Water Pollution Aspects of Street Surface Contaminants*, EPA-R2-72-081, U.S. Environmental Protection Agency, Washington, DC.
- Schilling, W., ed. (1989). *Real-time Control of Urban Drainage Systems—The State-of-the-Art*, Sci. Tech. Rep. No. 2, IAWPRC, Pergamon Press, Oxford.
- Shaheen, D. G. (1975). *Contributions of Urban Roadway Usage to Water Pollution*, EPA 600/2-75-004, Office of Research and Development, U.S. Environmental Protection Agency, Washington, DC, 346p.
- Soil Conservation Service (1972). *National Engineering Handbook 1964*, Section 4, Hydrology, rev. ed., U.S. Dept. of Agriculture, Washington, DC.
- Soil Conservation Service (1985). *Urban Hydrology for Small Watersheds*, Release No. 55, U.S. Department of Agriculture, Washington, DC.
- Stuebe, M. M., and D. M. Johnston (1990). Runoff volume estimation using GIS techniques, *Water Resour. Bull.* **26**(4):611-620.
- Sutherland, R. C. (1980). An overview of stormwater quality modeling, in *Proceedings, International Symposium on Urban Storm Runoff*, University of Kentucky, Lexington, KY.
- Terstriep, M. L., and J. B. Stall (1974). *The Illinois Urban Drainage Area Simulator, ILLUDAS*, Bull. 58, Illinois State Water Survey, Urbana, IL.
- Torno, H. C., J. Marsalek, and M. Desbordes, eds. (1986). *Urban Runoff Pollution*, Springer Verlag, Berlin, New York.
- U.S. Environmental Protection Agency (1971). *Storm Water Management Model*, 4 Vol., EPA No. 11 024DOC07/71 to 11 024DOC10/71, U.S. EPA, Washington, DC.
- Walesh, S. G. (1989). *Urban Surface Water Management*, Wiley, New York.
- Williams, J. R., A. D. Nicks, and J. G. Arnold (1985). Simulator for Water Resources in Rural Basins, *J. Hydraul. Eng. ASCE* **111**(6):970-986.
- Woodward-Clyde Consultants (1989). *A Probabilistic Methodology for Analyzing Water Quality Effects of Urban Runoff on Rivers and Streams*, U.S. Environmental Protection Agency, Washington, DC.
- Yen, B. C., ed. (1982). *Urban Stormwater Quality, Management and Planning*, in *Proceedings, Second International Conference on Urban Storm Drainage*, Water Resources Publ. Littleton, CO.

- Young, R. A., et al. (1986). *Agricultural Nonpoint Source Pollution Model: A Watershed Analysis Tool*. Agricultural Research Service, U.S. Department of Agriculture, Morris, MN.
- Zison, S. W. (1980). *Sediment Pollutant Interrelationships in Runoff from Selected Agricultural, Suburban, and Urban Watersheds. A Statistical Correlation Study*, EPA 600/3-80-022, U.S. Environmental Protection Agency, Athens, GA.

Control of Urban Diffuse Pollution

When a city takes a shower, what do you do with the dirty water?

Paraphrased from Lager et al. (1977)

POLLUTION-CONTROL MEASURES

Urban runoff from separate sewers and combined sewer overflows can be controlled by a variety of measures that can be structurally intensive and expensive or nonstructural and relatively inexpensive. The term *best management practices* usually refers to less structurally intensive modifications of the drainage system aimed at the reduction of pollution loads.

Several treatises and manuals deal extensively with control of urban runoff. The following manuals were reviewed and are recommended for further reference:

DeGroot (1982): Stormwater Detention Facilities

Dorman et al. (1988): Retention, Detention, and Overland Flow for Pollutant Removal from Highway Stormwater Runoff

Lager and Smith (1974): Urban Stormwater Management and Technology: An Assessment.

Lager, Smith, and Tchobanoglous (1977): Urban Stormwater Management and Technology: Update and Users' Guide.

Minnesota Pollution Control Agency (1989): Protecting Water Quality in Urban Areas

Moffa (ed.) (1990): Control and Treatment of Combined Sewer Overflows

Novotny and Chesters (1981): Handbook of Nonpoint Pollution: Sources and Management

Schueler, Kumble, and Heraty (1991): A Current Assessment of Urban Best Management Practices. Techniques for Reducing Non-point Pollution in the Coastal Zone.

Stahre and Urbonas (1990): Stormwater Detention

Torno, Marsalek, and Desbordes (1986): Urban Runoff Pollution

Walesh (1989): Urban Surface Water Management

Whipple et al. (1983): Stormwater Management in Urbanizing Areas

In addition, the *Proceedings of the International Conference on Urban Storm Drainage* (Yen, 1982; Balmér, Malmqvist, and Sjöberg, 1984; Gujer and Krejci, 1987), *Proceedings of an Engineering Foundation Conference* (Urbonas and Roesner, 1986; Roesner, Urbonas, and Sonnen, 1989), and *Proceedings of the National Conference—Perspectives on Nonpoint Pollution*, held in Kansas City in 1985, were also quoted by individual papers and are recommended for further reading. Manuals by the Water Pollution Control Federation (1990), the American Public Works Association (1981), and numerous research reports sponsored by the U.S. Environmental Protection Agency published since the 1970s provide a comprehensive review of various practices and structural measures for control of the quantity and quality of urban runoff.

The National Urban Runoff Project was the most comprehensive urban runoff research effort to date anywhere in the world. Research that addressed issues of the characterization of the quality of urban runoff and the effectiveness of various measures of abatement was carried out at 28 localities throughout the United States (for example, Bannerman, Baun, and Bohn, 1983; Bannerman et al., 1983; Metropolitan Washington Council of Governments, 1983; Pitt and Shawley, 1981; Terstriep, Bender, and Noel, 1982, and others). A summary of the NURP research was published in a U.S. EPA (1983) report.

Review papers on urban nonpoint pollution abatement measures include articles and reports by Field (1984, 1985a, 1985b, 1986, 1990) and Novotny (1984). A report by the U.S. EPA (1990) provides an information and guidance manual for state NPS program staff engineers and planners. Livingston and Roesner (1991) pointed out that runoff quality control (and by the same reasoning, CSO control) is not yet a technical science; rather it is an engineering art, with few design criteria for pollution removal having been established to date. Nevertheless, some empirical rules have been suggested by the authors:

1. The most effective runoff quality controls reduce the runoff peak and volume (these are generally infiltration controls).

2. The next most effective controls reduce the runoff peak. (These controls generally involve storage.)
3. For small storms (those with return intervals of less than two years), the peak rate of runoff should not exceed the peak rate of runoff from a two-year storm in a *preurbanized* condition, in order to control stream erosion.
4. Most obnoxious pollutants in urban runoff can be settled out; however, appreciable amounts of nutrients and some heavy metals are dissolved pollutants and require further treatment.

Urban storm-water management, which has an objective of flood control, must consider storms that have a relatively long recurrence interval. Typically urban storm sewers are designed to carry flows generated by a storm with a recurrence interval of 5 to 10 years, while urban floodplains and other flood conveyance systems should be able to convey without major damage 50- to 100-year floods (Walesh, 1989). On the other hand, storm-water management for *water quality control* is most efficient and economical if *small frequent storms* (smaller than the 1-year storm) events are considered (Livingston and Roesner, 1991).

After an extensive review of urban best management practices, Schueler, Kumble, and Heraty (1991) pointed out that

- Not all urban best management practices (BMPs) to control the quality of urban runoff can reliably provide high levels for both particulate and soluble pollutants.
- The longevity of some BMPs is limited to such a degree that their widespread use is not encouraged. Of particular concern are such infiltration practices as basins, trenches, and porous pavements.
- BMP options are adaptable to most regions of the United States, with the exception of extremely arid regions of the west and the colder climates of the north. In these regions, conventional BMP designs may need to be refined to account for high evaporation rates or subfreezing snowmelt conditions, respectively.
- No single BMP option can be applied to all development situations, and all BMP options require careful site assessment prior to design.
- Several BMPs can have a significant secondary environmental impact, although the extent and nature of these impacts is uncertain and site specific. These impacts can be both positive (such as the creation of a wetland wildlife habitat) or negative (accumulation of toxic compounds in ponds).
- Relatively limited cost data exists to aid in the assessment of the comparative cost-effectiveness of urban BMPs.

It is more difficult to consider pollution abatement in the fast conveyance systems. The delivery ratio for pollutants in these systems is close to one (all pollutants delivered to the systems from the surface will be conveyed to the receiving body of water). Runoff and CSO quality management must then rely on expensive structural measures and treatment.

A potentially more effective system for pollution control is the *storage-oriented* (retardance) system. These systems provide for temporary storage at many points throughout the system, beginning with the source areas, and subsequent release after the flows subside. Pollution can be attenuated better when the flow is slowed down and temporarily stored. Therefore, pollution mitigation is more effective and inexpensive in the storage-oriented systems.

Walesh (1989) pointed out that fast conveyance systems can be implemented in both existing (developed) and newly developing urban areas, while the storage-oriented system, which uses landscape to a great extent, is mostly applicable to new developments. The principal advantages of the storage-oriented approach are possible cost reductions in newly developing areas, far better prevention of downstream adverse flooding and pollution associated with storm-water runoff, and potential for multiple-purpose reuse.

Urban runoff pollution-control measures can be divided into several categories. Such categorization separates the runoff and combined sewer overflow control measures according to where the measure is implemented, namely, into on-site land pollution-control measures, hydrologic modification and land management, reduction of delivery of pollutants in the collection (drainage) system, and end-of-pipe storage and treatment. Typically in a study or a plan, hydrological models or design routines are used for the design and evaluation of the effectiveness of the proposed measures. Many measures have multiple benefits in addition to water quality control, such as flood control, aesthetic enhancement and recreation, remedy of sewer backwatering problems, aquifer recharge, wetland restoration and protection, and water conservation.

SOURCE-CONTROL MEASURES

Source-control measures are most effective for control of urban runoff pollution delivered to receiving waters from separate storm sewers, and the efficiency of such measures may be minimal for controlling pollution by combined sewer overflows. These measures commonly involve reduction of pollutant accumulation on the impervious surfaces of the contributing watersheds or reduction of the erosion of pervious lands.

Source controls are generally difficult to implement because the sources

are diffuse, originating from the atmosphere, streets, parking lots, and so forth, as documented in Chapter 8. Many of these controls can be incorporated into the landscape and, in fact, they can often enhance the aesthetics of the urban landscape (Roesner, 1988). Generally, however, planners have relatively little control over the policies that would implement source controls. Homeowners are not restricted in using chemicals on their properties as long they are approved for sale. Street cleaning practices are mostly for aesthetic purposes, and their frequency depends on the financial resources of each individual community and on the willingness of public officials to implement these practices.

Implementing source-control practices requires extensive public education and grass-roots public effort. Local environmental organizations and schools must be involved in planning, providing educational materials outlining sound environmental uses of chemicals, and promoting clean and well-kept neighborhoods. These organizations and citizen groups must keep pressure on elected officials to implement and enforce controls. Without these efforts implementation of these practices is impossible.

Control of Atmospheric Deposition

After implementation of stricter industrial air-pollution control measures, switching from coal to natural gas as a primary energy source for household heating, and by implementing limitations on leaded gasoline, dry deposition of pollutants has dropped substantially in the United States during the last 30 years. However, other problems, such as acidity of rainfall and deposition of nitric oxides from traffic have not improved. As shown by Novotny and Kincaid (1982), wet-atmospheric deposition (precipitation) in the midwestern U.S. cities is acidic, with pH around 4, and contains appreciable amounts of nitrogen.

Control of atmospheric deposition originating from air pollution by industrial and traffic sources is regulated in the United States by the Clean Air Act, which also includes provisions to control sources of acid rainfall. It was pointed out in Chapter 8 that the acidity of urban precipitation elevates loads of many pollutants that are elutriated during the buffering of acid rainfall from the urban infrastructure, including metal and tar shingle roofs, from vehicular corrosion, and from other sources.

Removal of Solids from Street Surfaces

Litter-Control Programs

Novotny (1984) described the effects of litter-control programs. Litter includes larger items and particulates deposited on street surfaces, such as

paper, vegetation residues, animal feces, bottles and broken glass, and plastics. In the fall, fallen leaves become the most dominant component of street litter. It has been shown that litter-control programs can reduce the amount of deposition of pollutants by as much as 50%, and litter control, especially in watersheds that are highly impervious, may be an effective measure of controlling pollution by storm runoff. During the fall leaf fallout period the organic and nutrient pollution inputs in areas with trees require an effective leaf pickup program to minimize the impact on the receiving waters. As stated in the preceding section, implementation of litter-control programs requires public education and acceptable enforcement.

Street Cleaning—Sweeping

Street cleaning practices include the sweeping of streets and parking lot surfaces by mechanical vehicles or flushing from tanker trucks (Figs. 10.1 and 10.2). Sweeping is more common in the United States, while street flushing is practiced more in Europe. Both practices are used primarily for aesthetics and removal of unsightly litter. The choice whether street sweeping or flushing is to be used for water quality control depends above all on the type of sewer system.

Street flushing. Pravoshinsky (1975) has shown that the quality of runoff from street flushing is very poor and is apparently inferior to storm runoff. However, street flushing is an advantageous and efficient means of control in areas served by combined sewers. Flushing cleans a larger street area (not just a narrow strip near the curb) and is more efficient for picking up fine particles. Because the capacity of flow separators and sewer capacity in the combined sewer system is selected to be about six times the peak dry-weather flow, street runoff generated by this practice during the dry period is generally far below the critical runoff rate that would initiate an overflow, hence, all polluted flow is conveyed to the treatment plant. In addition, frequent flushing can dislocate solids and slime that accumulate in sewers that otherwise would be discharged untreated with the overflow into the receiving waters.

Street sweeping. This method is appropriate for urban areas served by storm sewers. Application of street flushing in these areas would defy the purpose of this activity, that is, all solids washed into the sewer system would be carried directly into the receiving waters, while sweepers physically remove them from the surface, thereby making them unavailable for pickup by subsequent runoff-generating rainfall. Two types of sweepers are currently used to remove solids from impervious surfaces. The most common design (mechanical street cleaners) uses a rotating gutter broom to remove the particles from the street gutter area and place them in the



FIGURE 10.1. Street sweeping vehicle. (Photo: University of Wisconsin.)



FIGURE 10.2. Street flushing vehicle. (Photo: University of Wisconsin.)

path of a large cylindrical broom that rotates to carry the material onto a conveyor belt and into the hopper. Vacuum-assisted street cleaners use gutter brooms to loosen the deposits and move street refuse into the path of a vacuum intake. The vacuum places the debris in the hopper.

Both types of sweepers are relatively ineffective for removing fine particles (American Public Works Association, 1969; Sartor and Boyd, 1972; Sartor, Boyd, and Agarty, 1974; Bannerman et al., 1983). Broom mechanical sweepers are ineffective for particulates in the dust and dirt range (<3.2 mm in diameter), and their overall efficiency is only about 50%. Vacuum sweeper efficiency is higher, but still ineffective for silt and clay size particles (American Public Works Association, 1969; Sartor and Boyd, 1972). Tables 10.1 and 10.2 show typical street cleaner efficiencies for removal of particles of various sizes. If the efficiency is combined with

TABLE 10.1 Mechanical Street Cleaner Efficiencies for Various Equipment Passes (%)

Size Range (μm)	Curb Loading (50–500 g/curb m)		
	1 Pass	2 Passes	3 Passes
<43	15	28	59
43–104	20	36	49
104–246	50	75	88
246–840	60	84	94
840–2000	65	86	96
2000–6370	80	96	99

Sources: Data from Sartor and Boyd (1972) and Pitt (1979)

TABLE 10.2 Removal Efficiencies for Vacuum Street Cleaners at Different Initial Particulate Loadings and for Various Equipment Passes

Size range (μm)	Street Surface Loading and Number of Passes								
	5–50 (g/curb m)			20–200 (g/curb m)			280–2800 (g/curb m)		
	1	2	3	1	2	3	1	2	3
44–74	3	6	9	20	36	49	70	91	97
74–177	25	40	50	40	60	72	75	94	99
177–300	50	75	88	60	84	94	80	96	99
300–750	60	84	94	65	88	96	70	91	94
750–1000	50	75	88	60	84	94	70	91	97

Sources: Data from Clark and Cobbins (1963) and Pitt (1979).

the pollutant distribution on particles of various sizes (Table 8.11), the sweeper removal efficiency for pollutants can be estimated.

A normal sweeping effort was defined by Sartor and Boyd (1972) as 2.56 equipment minutes per 1000 m² of cleaned area, which can be translated to the average sweeper vehicle velocity of about 10 km/hr (6 mph). The information on street sweeper efficiencies in Table 10.2 refers to the normal operating effort. Increased street sweeping efficiency can be achieved by operating the vehicle at a lower speed or by conducting multiple passes. From the work by Sartor and Boyd (1972) it follows that the removal of particles by street sweepers can be approximated by the equation

$$P = P^* + (P_0 - P^*)e^{-kE} \quad (10.1)$$

where

P = the amount of street surface particulates in a given size range remaining after sweeping

P_0 = the initial amount of particulates in the size range

E = the amount of sweeping effort involving (min/1000 m²) or as a relative effort (i.e., actual effort divided by the standard effort)

P^* , k = empirical coefficients depending on sweeper characteristics and design, particle size of particulates, and street surface characteristics

If E is normalized by expressing it as a relative effort, the coefficient k becomes dimensionless. From Sartor and Boyd's data it appears that k can be approximated best by the following equation

$$k = \alpha d^\beta \quad (10.2)$$

where α and β are empirical coefficients and d is particle size. The approximate magnitudes of α and β are around 0.027 and 0.35, respectively, if d is in micrometers. The actual magnitudes of α and β should be determined for each type of equipment by the manufacturer or by a testing laboratory.

Example 10.1: Removal of Pollutants by Sweeping

A street dust and dirt sample had the following particle-size distribution

<i>Particle Size (μm)</i>	<i>% Distribution</i>
<43	5
43-104	10

Particle Size (μm)	% Distribution
104–246	20
246–840	25
840–2000	25
>2000	15

The sweeper efficiency coefficients, α and β , were given by the manufacturer as $\alpha = 0.03$ and $\beta = 0.55$. Determine the overall removal efficiency of the sweeper if it moves at a sweeping speed of 5 km/hr.

Solution The removal efficiency for each particle-size fraction can be computed using Equation (10.1). Then approximately (assuming that $P^* \approx 0$)

$$\text{Eff}(\%) = 100(1 - e^{-kE})$$

where $k = 0.03d^{0.55}$ The normalized sweeper effort variable is

$$E = \frac{\text{Actual effort (km/hr)}}{\text{Standard effort (km/hr)}} = \frac{(1/5 \text{ km/hr})}{(1/10 \text{ km/hr})} = 2.0$$

The solution can be obtained either graphically or numerically as shown in Table 10.3. The calculation may also be incorporated into a computer

TABLE 10.3 Computation of Overall Sweeper Efficiency

Particle Size (μm) (1)	Initial Distribution (%) (2)	k (3)	e^{-kE} (4)	Average Removal $1 - e^{-kE}$ (5)	% Removal (6) = (2) * (5)	% Remaining (7) = (2) - (6)
0		0	1.0			
	5			0.19	0.95	4.05
43		0.24	0.62			
	10			0.46	4.6	5.4
104		0.39	0.46			
	20			0.625	12.5	7.5
246		0.62	0.29			
	25			0.81	20.25	5.75
840		1.22	0.09			
	25			0.945	23.62	1.38
2000		1.96	0.02			
	15			0.98	14.7	0.3
Total	100				76.62	23.38

model (see the Wisconsin Urban Runoff Model in Chapter 8). From Table 10.3, we see that the overall efficiency is 76.6%.

Example 10.2: Lead Removal by Sweeping

By analyzing the lead distribution with particle sizes of the street dust and dirt sample the same as in the previous example, it was found that most lead is associated with fine fractions is as follows:

<i>Particle Size (μm)</i>	<i>% Lead</i>
0–43	48
43–104	25
104–246	20
246–840	16
840–2000	1

Estimate how much lead will be removed by the sweeper.

Solution As seen in Table 10.4, the computation can be arranged in the same fashion as in the previous example. Thus we get that only 48% of lead will be removed, as compared to 76% of solids.

Water quality benefits of the removal of pollutants by street sweeping. Current street sweeping practices are primarily for aesthetic purposes. The results of the National Urban Runoff Project documented the relatively low impact of street sweeping on improvement of water quality on midwestern and eastern U.S. conditions (Bannerman et al., 1983; Terstriep, Bender, and Noel, 1982; Bender and Rice, 1982; Novotny et al., 1985; Field, 1986).

TABLE 10.4 Computation of Percentage of Lead Removed by the Sweeper

Particle Size Range (μm) (1)	Initial Distribution (2)	Average Removal (3)	% Removed (4) = (2) * (3)
0–43	48	0.19	9.12
43–104	25	0.46	11.50
104–246	20	0.625	13.50
246–840	16	0.81	12.96
840–2000	1	0.94	0.94
2000	0		0
Total	100		48.02

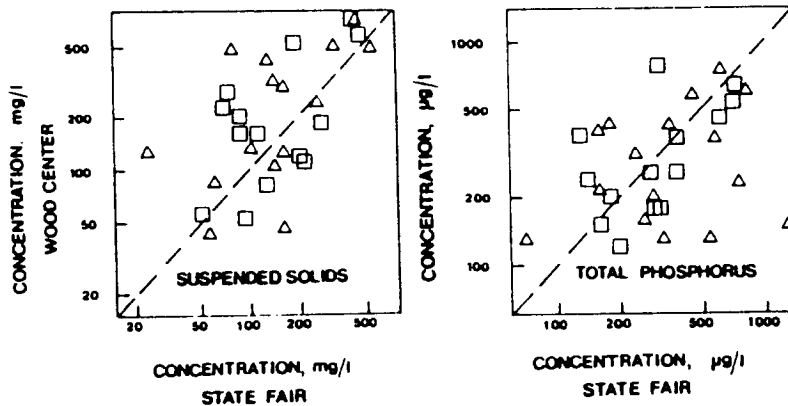


FIGURE 10.3. A comparison of runoff concentrations of suspended solids from paired events from two almost identical small and uniform urban experimental watersheds with predominantly commercial land use. State Fair test site was swept, while Wood Center was kept unswept. (Data from the Wisconsin Department of Natural Resources.)

In the Milwaukee, Wisconsin, NURP project six pairs of small uniform watersheds were investigated. The watersheds in each pair were either identical or similar in land use and surface characteristics. One watershed was swept regularly with relatively high frequency (at least once per week); the other watershed was unswept. The measured results were disappointing. Street sweeper efficiency was generally low and the effect of street sweeping on runoff pollutant concentrations was minimal, as shown on Figure 10.3, in which corresponding concentrations of swept and unswept watersheds in the pair with commercial land use were plotted against each other. Modeling results for these experimental paired watersheds were quite similar (Novotny et al., 1985).

Several hypotheses have been given to explain the reasons for these results (Novotny et al., 1985). First, street sweeper efficiency is relatively low, and sometimes, when initial street loads were low, it was negative. Second, simulations by a mathematical model showed that in areas of lower imperviousness, more particles are blown away by traffic and wind onto adjacent pervious and hydrologically inactive areas than can be removed even by a very vigorous sweeping program. Third, erosion of pervious areas, although not as frequent as the washoff of pollutants from impervious areas, yields unit loadings of pollutants that are often of the same order of magnitude or greater than the corresponding loads from impervious areas affected by street sweeping.

Tests under real conditions in San Jose, California, showed that street cleaning can remove up to 50% of the total solids and heavy metals in urban storm water with very frequent (once or twice daily) cleaning.

Typical cleaning programs of once or twice a month proved ineffective (Pitt, 1979; Field, 1986).

Control of Pervious Areas

Soil loss from unprotected bare soils may be considerable—up to 100 tonnes/ha-year. Temporary or permanent seeding of grass, sodding, and mulching is used to reduce erosion of pollutants. Such measures are important and mandated in some states for control of pollution caused by construction activities. Ports (1975) presented procedures and criteria for the design of urban sediment-control practices. More extensive design procedures for erosion control were included in a publication by Goldman, Jackson, and Bursztynsky (1986).

Chemical stabilization of soils. This is a temporary measure employed on bare soils until permanent vegetation is established or other long-term erosion-control measures are implemented. The chemical emulsions that were previously used for erosion control included polyvinyl acetate emulsions, vinyl acrylic copolymer emulsions, or methacrylates and acrylates. The use of organic chemicals and oil derivatives may not be possible due to suspected surface- and ground-water contamination by carcinogenic priority organic pollutants (vinyl chloride and poly-cyclic aromatic hydrocarbons (PAHs)). Hydrate lime and cement is also used for stabilization of clayey soils.

Mulching and protective covers. Covering an exposed area with any of a number of available mulches generally increases surface roughness and hydrological surface storage, protects the surface against rainfall impact, and subsequently reduces erosion. The effect of land-cover materials is reflected in a reduction of the land-cover factor (C) of the universal soil loss equation (Table 5.5). Straw mulch (small grain) has proved effective on 12% slopes at application rates of 5 tonnes/ha and on 15% slope with an application rate of 10 tonnes/ha. These application rates should reduce erosion by 75% to 80%. Straw mulch application can be combined with grass seeding for more permanent surface protection. As a matter of fact, the most economical, effective, and practical surface protection in construction sites is hydromulching, which is an application of a slurry mixture of seed, mulch, fertilizer, and lime (Fig. 10.4).

Figure 10.5 shows an example of straw mulch application protected by plastic or paper (biodegradable) fibers. Straw mulching loses its effectiveness on steep slopes because of rill formation and its tendency to be washed away by more concentrated and erosive overland flow. Alternate materials include woodchips, crushed stone, and blankets and mats from textile materials (Fig. 10.6). Chapter 5 contains an extensive discussion of erosion-control measures and design parameters with examples.

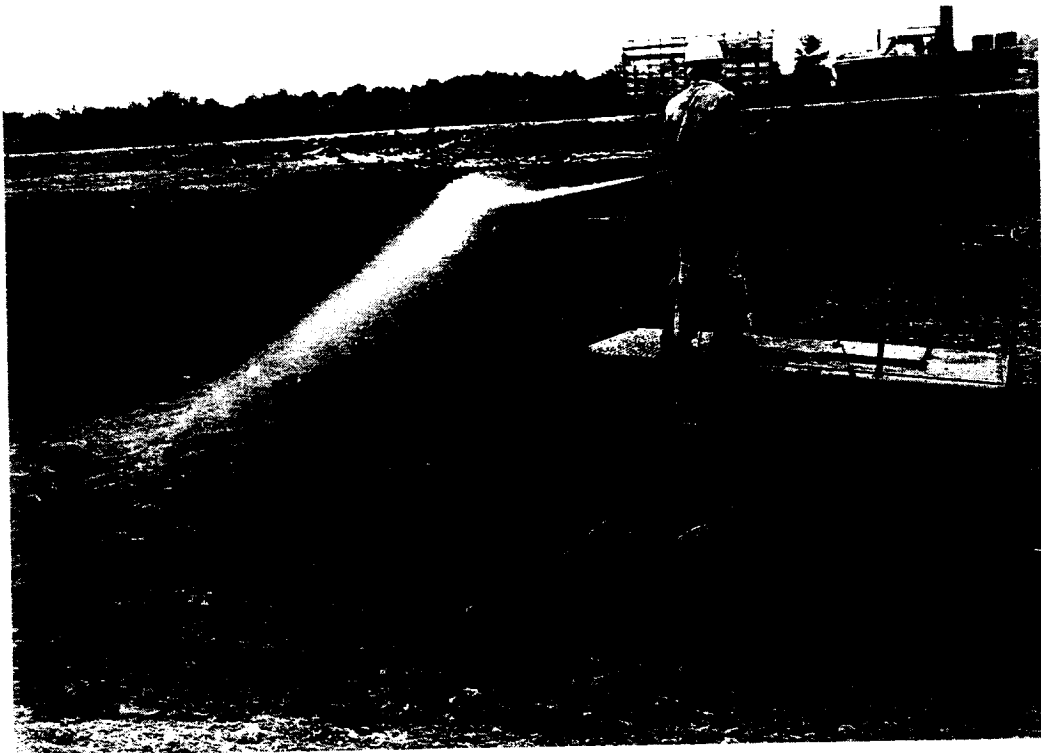


FIGURE 10.4. Hydroseeding of grasses. (Photo: University of Wisconsin.)



FIGURE 10.5. Mulch net applied over straw mulch, seed, and fertilizer. The net is of paper fiber with plastic strands that are pinned down over the straw. (Photo: USDA, Soil Conservation Service.)



FIGURE 10.6. Control of erosion by burlap netting. (Photo: USDA, Soil Conservation Service.)

Control of surface application of chemicals. Measures to control the surface application of chemicals include control of herbicide use on pervious grassed areas (lawns and golf courses) and road deicing salt storage. Studies by the EPA and the Federal Highway Administration quoted in Field (1986) prompted several states to enact legislation controlling salt application and storage. For example, chemical loading/unloading and storage areas should be covered or diked to capture about 1.25 cm (1/2 inch) of runoff, with a closeable outlet so that, if spillage occurs, it will not be washed into the drainage system before it can be cleaned up. In areas served by separate sewers, runoff from washing should be discharged into sanitary sewers (Livingston and Roesner, 1991).

As was pointed out previously, control of pollution caused by chemical use by individual homeowners on their lands is difficult due to a lack of legal instruments to enact regulation. Public education is currently the only possible mechanism to implement some partially effective measures. A switch to *xeriscape*, that is, landscape that incorporates native plants

and shrubs that do not require chemicals and excessive irrigation, should be encouraged and demonstrated to the public. Such changes may even be mandated in the future in some areas that are short of water because of their water-conservation benefits in addition reducing the pollution of the surface- and ground-water resources.

Hydrologic Modifications

Hydrologic modifications of urban watersheds include measures and practices that reduce the volume and intensity of urban runoff entering the separate storm or combined sewer systems. In separate systems it is expected that the pollution load will be proportional to the volume of runoff, and with the volumetric reduction of flow, one may also expect a reduction in pollution load. In combined sewers, reduction of peak flow and volume of flow reduces the frequency of untreated overflows from the sewer systems. Thus hydrological modification practices are effective for the control of pollution from both systems (storm sewers and CSOs) and both have pollution-control and flood-control benefits. A special report by the EPA (1977) deals with practices involving hydrologic modifications of urban watersheds.

The hydrologic measures can be divided into:

1. Practices that increase permeability and enhance infiltration, such as the use of pervious pavements or vegetation filtration strips;
2. Practices that increase hydrological storage;
3. Practices that reduce the size of impervious areas that are directly connected to the sewer system.

Schueler, Kumble, and Heraty (1991) cautioned against the indiscriminate use of infiltration practices in an urban setting. The poor longevity of these BMPs is attributable to lack of pretreatment, poor construction practices, application at unsuitable sites, lack of regular maintenance, and faulty design. The life span of such practices may be increased if local communities adopt enhanced maintenance and inspection programs.

Porous (Pervious) Pavements

Porous pavement is an alternative to conventional pavement. Its use allows rainfall to percolate through it into the subbase. The water stored in the subbase then gradually infiltrates the subsoil.

Porous pavement provides storage, enhancing soil infiltration that can be used to reduce runoff and combined sewer overflows. These pavements are either made from asphalt, in which fine filler fractions are missing, or

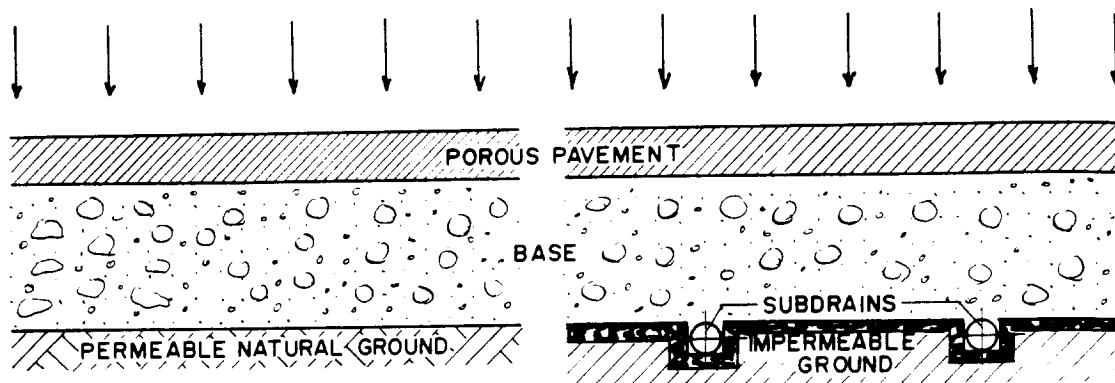


FIGURE 10.7. Porous pavement installed over permeable and impermeable ground.

are modular or poured-in concrete pavements (Fig. 10.7). The primary benefit of porous pavements is a significant reduction or even complete elimination of surface runoff rate and volume from an otherwise impervious area. If the pavement is designed properly, all or most of the runoff can be stored and subsequently allowed to infiltrate into the natural ground. Aquifer recharge by infiltrated water is the second important benefit. The third benefit is the reduced need for storm drainage. As a matter of fact, if subsoils are very permeable, there may be no need for installing storm drainage. However, porous pavement is most feasible when subsoils are permeable and the slopes are relatively flat. If the soils are permeable, porous pavements are feasible even in cold snowbelt areas (see Chapter 3 for a discussion of infiltration into frozen soils).

In areas with poorly draining subsoils or if the porous pavement is installed over an existing impervious base, a drainage system can be installed. The pavement and its base in this case enhances hydrological storage, reduces the peak runoff rate, and provides filtration of pollutants from runoff. Field (1986) summarized the results of several U.S. EPA studies on the experimental applications of porous pavements. Results from a study in Rochester, New York, indicated that peak runoff rates were reduced by as much as 83% where porous pavement was used. The structural integrity of the pavement was not impaired by a heavy traffic load and, if properly installed, porous pavements have load-bearing strength and longevity similar to conventional pavement. Although clogging may occur during construction and during operation, it can be remedied by flushing and sweeping.

Typically, hydraulic conductivity (permeability) of porous pavements is much greater than runoff rates. Hydraulic conductivity measured by Jackson and Ragan (1974) was about 250 cm/hr, which is an order of magnitude higher than a typical catastrophic design storm. This means



FIGURE 10.8. Example of porous pavement installation in an experimental basin in the Tokyo metropolitan area. Before (*top*) and after (*bottom*) installation. (Photo: S. Fujita.)

that infiltration into the base should occur without ponding. An extensive report summarizing EPA research on the hydraulic characteristics of pervious pavements was prepared by Goforth, Diniz, and Rauhut (1984). Porous pavement has an excellent potential for use in parking areas and on side streets. These pavements have been installed in many localities throughout the world (Fig. 10.8) and a design manual has become available in Florida (Florida Concrete Products Association, 1989) and other states (Maryland Water Resources Association, 1984; Schueler, Kumble, and Heraty, 1991).

Longevity of the porous pavement function is a problem (Schueler, Kumble, and Heraty, 1991) since it has a high failure rate. Failure is due to partial or total clogging that occurs during or immediately after construction or over time, when porous pavement is clogged by sediment and oil. For this reason, porous pavements should not be installed where high solids loads from wind erosion or other sources (heavy truck traffic, use of sand for ice skidding control) are expected. The Japanese experience (Fujita, 1984; Fujita and Koyama, 1990) has shown that frequent and proper sweeping by vacuum sweepers resolves the problem of clogging.

Contamination of shallow aquifers by toxic materials attributed to asphalt, vehicular traffic, and road usage, including salt application for deicing, represents a slight to moderate environmental risk that depends on soil conditions and aquifer susceptibility.

The construction cost of porous pavement is about the same or even less than that for conventional pavement when savings on storm drainage are included. A laboratory and economic study on feasibility of porous pavements was undertaken by Thelen et al. (1972). Haak and Oberts (1983) showed that the cost of conventional pavement on streets and highways with drainage can be as much as two to three times higher than the cost of a pervious pavement alternative.

Increasing Surface Storage

Rooftop storage on flat roofs, temporary ponding, and restriction of storm-water inlets are used to control combined sewer overflows and to reduce flooding (American Public Works Association, 1981). As a water quality control measure in areas served by combined sewers, these practices may be effective and may reduce the requirements for in-line or off-line storage. In areas served by storm sewers, they can control flooding and reduce the requirements for detention-retention basin volumes, but alone they do not have a significant water quality benefit.

Decreasing Connected Impervious Area

Directly connected impervious area (DCIA) is defined as the impermeable area that drains directly to an improved drainage system, that is,

paved gutter and, subsequently, sewer. Livingston and Roesner (1991) suggest that the minimization of DCIA is by far the most effective method of runoff quality control because it delays the concentration of flows into the sewers and maximizes infiltration.

Practices used for minimization of DCIA include:

1. Disconnecting roof drains from storm sewers. This practice has a low direct water quality benefit since only wet- and dry-atmospheric contributions are controlled. It may have some impact on the necessary detention volume. Since this practice represents an infringement on private property, it may be difficult to implement.
2. Permitting surface runoff to overflow on adjacent pervious surfaces.
3. Use of dry wells, infiltration basins, and ditches into which storm water is directed.

Porous pavements obviously also reduce the directly connected pervious area.

Filter strips. Filter strips are grassed strips situated along the roads or parking areas between the impervious road surface and drainage sewers or ditches (Fig. 10.9). The filter strips remove the pollutants from runoff by filtering, provide some infiltration, and slow down the runoff flow. Their efficiency is related to the velocity and depth of flow. The higher the flow rate and flow depth, the lower the efficiency. Growing grasses almost guarantee laminar flow conditions of the shallow flow. Typically grass height should be maintained between 15 and 30 cm. If grass is submerged such, as in grassed swales and ditches, the flow is turbulent and efficiency drops significantly. Thus, low velocity and shallow non-submerged flow depth are key design criteria. The Maryland design guidelines recommend that the slope of the strips be less than 5% and overland flow velocities less than 0.75 m/s. Failures will occur if rill erosion creates concentrated flows. Filter strips can effectively reduce particulate contaminants; however, their ability to remove soluble pollutants is variable and depends on the fraction of infiltrated runoff in the strip and adsorption of contaminants on organic matter and soils.

Kao, Barfield, and Lyons (1975) showed by experiments on artificial grass strips that grass filter strips provide excellent trapping efficiency, especially for construction and surface mining sites. Grass strips are also widely used as borders to control soil and pollutant loss from agricultural fields and barnyards (see Chapter 11). Grassed areas are also effective traps for pollutants associated with particulate matter, such as phosphates and pesticides (Asmussen et al., 1977).

The distance at which close to 100% removal of solids is achieved (both by filtering and infiltration) is called the *critical distance*. Wilson

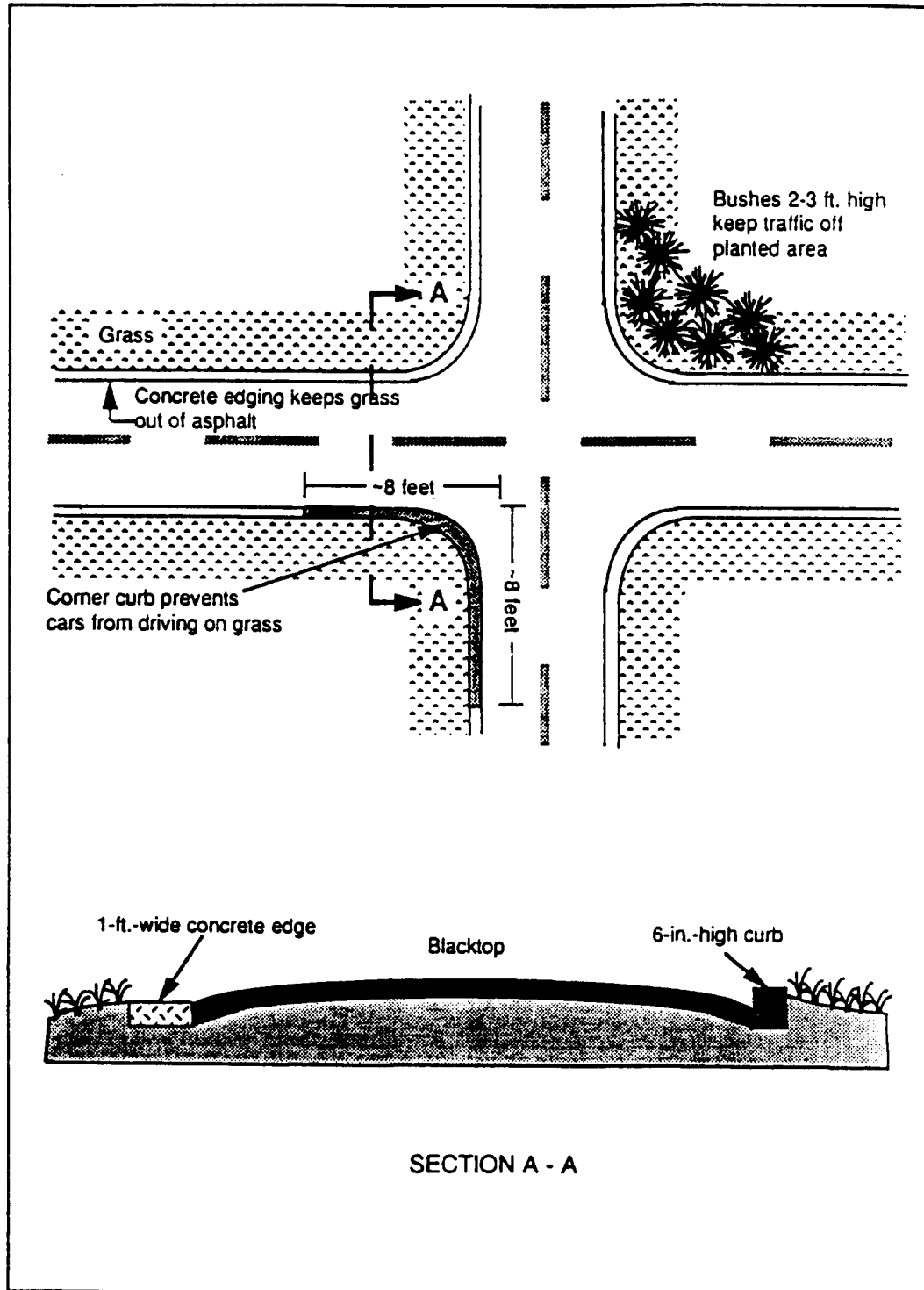


FIGURE 10.9. Simple installation of grass filters along roads. (After Livingston and Roesner, 1991, by permission of ASCE.)

(1967), in an empirical study on Bermuda grass, found that the maximum percentages of sand, silt, and clay were removed at about 3, 15, and 122 meters distances along the test strip. A minimum of 85% sediment removal can be achieved with a 2.5-m-wide grass strip during shallow (nonsubmerged) flow (Kao, Barfield, and Lyons, 1975; Barfield, Kao, and Toller, 1975). This efficiency is increased when alternating grass and bare land are used.

Experimental research at the University of Kentucky yielded an empirical model for particle trapping efficiency of grass filters (Barfield, Kao, and Toller, 1975; Toller et al., 1977). This efficiency was found to be a function of two dimensionless parameters, namely, the flow Reynolds number, Re' , and the particle fall number, N_f (a determination of the probability of how many times a particle can reach the bottom during the flow period). These two variables were defined as

$$Re_t = \frac{v_s R_s}{\nu} \quad (10.3)$$

and

$$N_f = \frac{L_T w}{v_s} \quad (10.4)$$

where

v_s = flow velocity through the grassed media

L_T = overland flow length

R_s = spacing parameter defined as

$$R_s = \frac{s D_f}{2 D_f + s} \quad (10.5)$$

where

s = spacing of grass blades

D_f = depth of flow

ν = kinematic viscosity

w = settling velocity of particles from Stokes law

$$w = \frac{g D^2}{18 \nu} (\rho_s - 1) \quad (10.6)$$

where

D = particle diameter

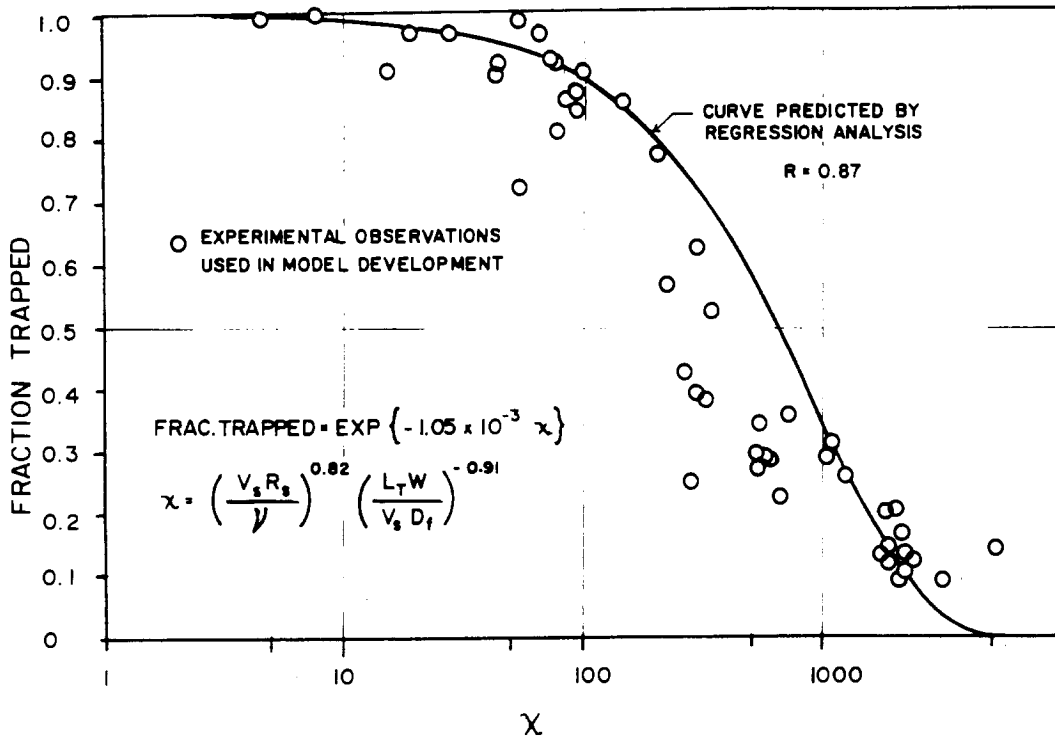


FIGURE 10.10. Relationship of particle removal efficiency of grass filters to Re_s and N_f . (From Kao, Barfield, and Lyons, 1975.)

g = gravity acceleration
 ρ_s = specific gravity of the grain with respect to water

For cohesive sediments (clays) use Table 5.9 for settling velocity of aggregate particles rather than that for settling individual particles according to Stokes' law.

The relationship of the removal efficiency to Re_s and N_f is shown on Figure 10.10. From the figure a removal efficiency approaching 100% can be achieved when

$$\chi = \left(\frac{v_s R_s}{v}\right)^{0.82} = \left(\frac{L_T W}{v_s D_f}\right)^{-0.81} \approx 5 \quad (10.7)$$

Example 10.3: Grass Strip Estimation

What should be the minimum width of a grass buffer strip dividing a construction site from the nearest drainage that would reduce clay loads to the receiving body of water by 90% during a storm that yielded an average surface runoff of $q = 0.003 \text{ m}^3/\text{s-m}$ width? The slope of the strip, S , is 5%.

Solution Assume:

Grass blades spacing $s = 3 \text{ mm} = 0.003 \text{ m}$

Clay particle size $d = 2 \mu\text{m} = 2 \times 10^{-6} \text{ m}$

Clay settling velocity from Table 5.9 $w = 0.17 \text{ mm/sec} = 0.00017 \text{ m/sec}$

Manning roughness factor for grass (from Table 3.14) $n = 0.35$

The flow depth can be estimated from the Manning equation

$$v_s = \frac{1}{n} D_f^{2/3} S^{1/2} = \frac{q}{D_f} \quad (10.8)$$

and assuming shallow flow in a wide channel

$$D_f = \frac{(qn)^{0.6}}{S^{0.3}} = \frac{(0.003 \times 0.35)^{0.6}}{0.05^{0.3}} = 0.0365 \text{ m}$$

Assume that the grass height is greater than the flow depth (nonsubmerged flow conditions). Then the flow velocity becomes

$$v_s = \frac{q}{D_f} = \frac{0.003}{0.0365} = 0.082 \text{ m/sec}$$

The spacing parameter, R_s , is

$$R_s = \frac{sD_f}{2D_f + s} = \frac{0.003 \times 0.0365}{2 \times 0.0365 + 0.003} = 0.00144 \text{ m}$$

Ninety percent removal occurs when the removal parameter, χ , from Figure 10.10 is ≤ 100 . Hence

$$\chi = \left(\frac{v_s R_s}{v} \right)^{0.82} \times \left(\frac{L_m w}{v_s D_f} \right)^{-0.91} \leq 100$$

Solving for L_m , the critical distance for which 90% removal is achieved is

$$\begin{aligned} L_m &= v_s^{1.9} R_s^{0.9} D_f v^{-0.9} w^{-1} 100^{-1.1} \\ &= 0.082^{1.9} \times 0.00144^{0.9} \times 0.0365 \times (10^{-6})^{-0.9} \times 0.00017^{-1} \times 100^{-1.1} \\ &= 7.93 \text{ m} \end{aligned}$$

or about 10 m.

Example 10.4: Efficiency of grass strip

For the critical distance, L_m , calculated in Example 10.3, estimate removal efficiencies for silt and sand fractions.

Solution For silt, assume particle size $d = 0.02$ mm the settling velocity $v = 0.03$ cm/sec = 0.0003 m/s (Table 5.9). Then the removal factor χ is

$$\chi = \left(\frac{0.082 \times 0.00144}{10^{-6}} \right)^{0.82} \times \left(\frac{10 \times 0.0003}{0.082 \times 0.0365} \right)^{-0.91} = 6.15$$

According to Figure 10.10, for $\chi = 6.15$ about 99% or more of the silt will be removed in the 10-m-wide buffer strip. The same is true for sand that will yield lower values of χ .

It is possible to speculate that this estimation of solids removal efficiency is on the conservative side since it does not include the removal of solids by infiltration and the effect of increased infiltration.

Infiltration

Perforated concrete street gutters (Fig. 10.11) and infiltration pipes laid in trenches have been proposed and tested in Japan (Fujita, 1984; Fujita and Koyama, 1990). The infiltration pipes are combined with an infiltra-

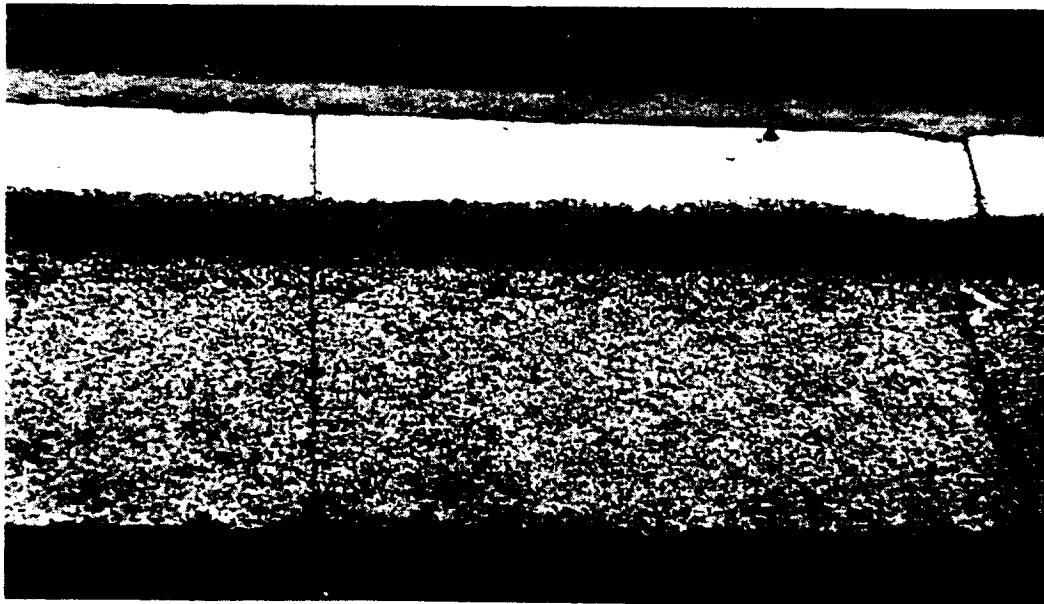


FIGURE 10.11. Example of street gutter modification for enhanced infiltration. Thorough maintenance and frequent solids removal are needed to maintain efficiency of infiltration. (Photo: S. Fujita.)

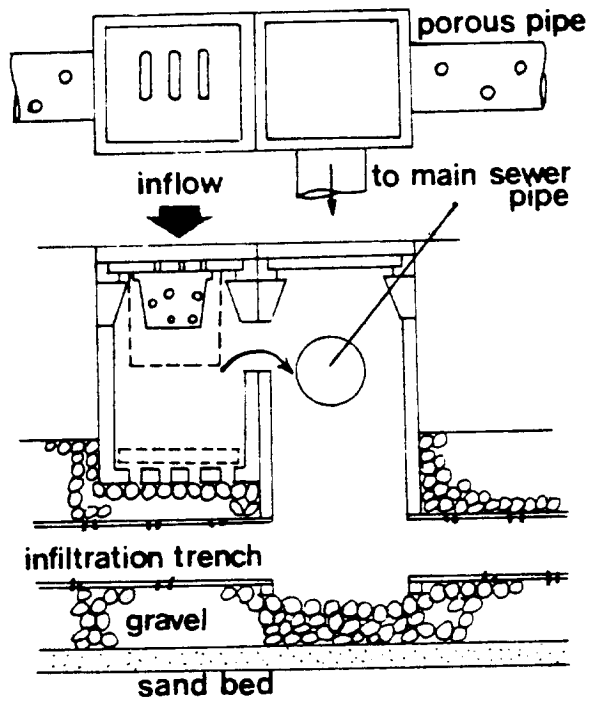


FIGURE 10.12. Example of manhole modification for enhanced infiltration. The manhole has an impermeable bottom and overflow area, not a perforated infiltration pipe laid in an infiltration trench. (From Fujita, 1984.)

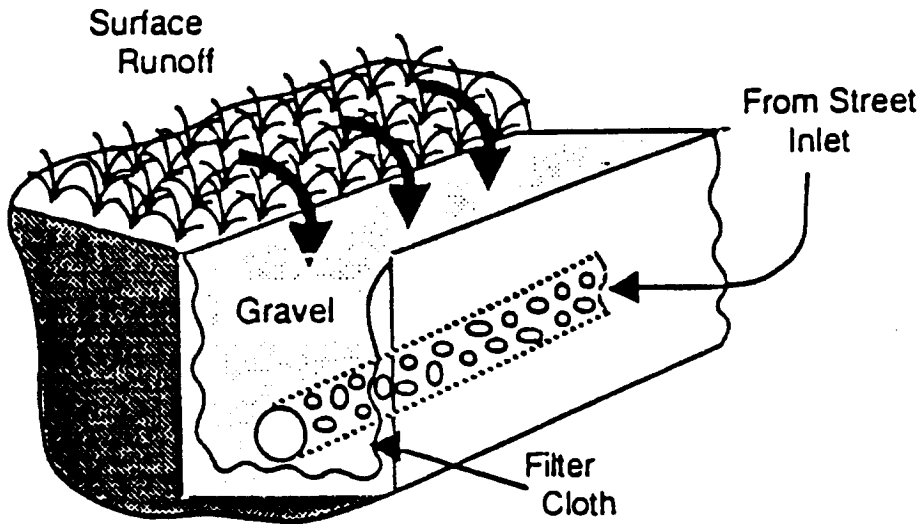


FIGURE 10.13. Infiltration trench.

tion manhole (Fig. 10.12) that has a pervious bottom. The soils in the Tokyo metropolitan area, where the experimental watersheds are located, are mostly composed of volcanic highly permeable turfs.

In the United States, excavated infiltration trenches, basins, and wells are more common (Fig. 10.13). These devices are especially advantageous in areas where ground-water recharge benefits are desired. Properly designed trenches are believed to be highly capable of removing particulate pollutants and moderately capable of removing soluble contaminants. Concerns persist about the possibility of ground-water contamination; however, sources in the literature to date do not indicate a major risk, but have noted migration of chlorides and to a lesser degree of nitrates (Schueler, Kumble, and Heraty, 1991). Generally, the nitrate content of urban runoff is lower and below the human health protection criterion.

An *infiltration trench* is an excavated trench that is backfilled with stone aggregates, gravel, or sand. An *infiltration basin* is made by constructing an embankment or by excavating it down to relatively permeable soils. The basin stores storm water until it infiltrates through the bottom and the sides of the system. The basins that are mostly dry can be incorporated into the landscape design of open areas or even recreational areas, such as sport fields (Roesner, 1988; Livingston and Roesner, 1991). Infiltration trenches can be located along highways and parking lots, in the median strip of divided highways, as a part of residential drainage, and several other possible locations (Harrington, 1989).

Both infiltration trenches and basins are prone to clogging by deposited solids. To increase their life span when sediment-laden storm-water flows are anticipated, sediment traps can be employed (Haak and Oberts, 1983). The Maryland experience (Schueler, Kumble, and Heraty, 1991; Galli, 1992) has shown that about one in five conventional trenches installed in the 1970s and 1980s failed to operate as designed immediately after construction. Furthermore, barely half of all conventional infiltration trenches operated after five years (Galli, 1992). The best pretreatment device is a grassed filter or buffer strip, which should be at least 6.6 m wide, along the trench or basin (Harrington, 1989). If trenches become clogged, the gravel and sand backfill will need to be removed and the trenches washed before they can be reused (Haak and Oberts, 1983), which could be expensive. To minimize this possibility it is necessary that the entire contributing area to the infiltration device be stabilized before construction of the trench or basin begins.

Oil and grease should also be removed before they enter these devices since these contaminants are difficult to remove and pose a danger to ground water. Treatment of runoff contaminated with these pollutants is accomplished through specially designed three-chamber storm-water

inlets. Grit and sand are removed in the first chamber, and oil and grease are removed in the second chamber, from which runoff water is siphoned into the third chamber for final storage and polishing. These devices must also be regularly cleaned and the accumulated sediment removed.

Infiltration devices obviously require permeable soils or subsoils. The minimum infiltration rate of soils underlying the device should be more than 1.25 cm/hr, which corresponds to the Soil Conservation Service's soil permeability classification of C or better (Table 3.2). After taking moderate clogging into consideration, the permeability rates should call for the diverted volume to infiltrate within 72 hours, or within 24–36 hours for infiltration basins that are planted with grasses. The seasonal high water table should be at least 1.2 meters below the bottom of the infiltration device to assure that pollutants present in storm water are removed by vegetation, soil, and soil bacteria before they reach ground water.

Infiltration devices should not be installed in areas with shallow bedrock and in areas with Karst limestone formations, where sinkholes, caverns, and large rock fractures are common. They should also be located at least 6.6 m from buildings to avoid cross-connections with basements.

Both infiltration trenches and basins should be carefully designed. Infiltration trenches can handle runoff from watersheds ranging in size up to 5 hectares. Maryland's design guidelines for sizing infiltration trenches recommend the size to be sufficient to capture the first 1.25 cm (1/2 in.) of rainfall from connected impervious areas (Maryland Water Resources Administration, 1986). The U.S. Environmental Protection Agency suggests that infiltration devices be designed to store and treat flow from an average storm, which typically may have a rainfall depth lower than 1.25 cm. Stahre and Urbonas (1989) presented a methodology for site selection and design of infiltration basins, which for site selection, assigns points to various site, watershed, and soil characteristics.

Dry (French) wells are smaller borings that are filled with gravel to permit infiltration of accumulated storm water. Their best use is for infiltration of rooftop runoff. Again it is important that the well is located several meters from the foundation wall of the house to prevent back entry of water into foundation drain pipes.

Environmental Corridors and Buffer Zones

The corridors—usually a park along a stream, lake, or adjacent to the drainage system—act as buffers between the polluting urban area and the receiving body of water. In most cases the buffer zones also provide storage for flood and pollution control (Wiesner, Kassem, and Cheung, 1982). The corridors lose their efficiency if the storm drainage outlet

bypasses the grassed and vegetated areas and discharges directly into the receiving body of water or into a channel with concentrated flows that is directly connected with the body of water. The "treatment" processes for storm runoff, such as vegetated filters, infiltration basins, detention-retention ponds (dry or wet), and wetlands, are incorporated into the landscape of the corridor.

Buffer strips made of uneven shoreline vegetation may also be used to attenuate runoff pollutants that would otherwise reach the body of water. Woodard (1989) measured the efficiency of buffer strips that had vegetation typical of a Maine lakeshore (mixed growth, uneven age stand, predominantly hardwood, moderate ground cover of shrubs, ferns, etc.). Similar measurements were made by Potts and Bai (1989) in Florida. They found that the critical distance of a grass strip, used for the control of suspended sediment and phosphates, from residential developments was 22.5 meters; however, they concluded that the efficiency of the buffer strips is highly dependent on a sufficient cover of organic matter (natural vegetation) and on the initial concentration of the pollutants or the density of shoreline development. A 30-m-wide buffer strip is recommended for the protection of surface-water reservoirs that are part of the drinking water supply, in states where residential development is permitted in such watersheds (Nieswand et al., 1990). In some countries, urban and agricultural land-use practices are greatly restricted or not permitted at all in watersheds of water supply reservoirs.

Buffer strips are ineffective on steep slopes with loose soils. Also their effectiveness is reduced by exposed soil on any part of the buffer strip, which can actually erode and contribute suspended solids and other pollutants instead of attenuating them. Woodard (1989) recommends that a porous organic "duff" layer and/or a dense growth of underbrush cover the mineral soil if buffer strips are to be effective. Environmental corridors in urban areas are typically a part of the major drainage system.

COLLECTION SYSTEM CONTROL AND REDUCTION OF DELIVERY OF POLLUTANTS

The second group of management practices for urban diffuse-pollution control involves methods and structures for removing pollutants from runoff after they leave the source area. Collection system controls refer to management alternatives for storm-water interception and transport. These include (Field, 1986) improved maintenance and design of catch basins, sewers, in-sewer (in-pipe) and in-channel storage, and elimination

of sanitary and industrial cross-connections from separate storm sewers. Most of these facilities are a part of the minor drainage system.

Urban areas with good separate storm sewer drainage have a delivery ratio (the ratio of a mass of pollutants delivered to a stream divided by the mass of pollutants generated at the source) close to one, while residential areas with natural (swale) drainage have a delivery ratio ranging from several percent to about 50% (Novotny and Chesters, 1989). The objective of management practices is thus reduction of the delivery ratio. In addition to grass filters, buffer strips, and other practices mentioned in the preceding section, such practices in urban basins also include grassed waterways (natural drainage) and catch basins.

In separate sewer systems, sanitary and industrial wastewater cross-connections into separate sewers are a nationwide problem (Pitt et al., 1990a). A manual of practice for identification and control of cross-connections has been prepared (Pitt et al., 1990b). Most of the methods in this category require some engineering structures and design, and are more expensive than source-control measures. In the disturbed areas the polluted waters should be segregated as much as possible from the rest of the runoff. Segregation can be achieved by diversion dikes, culverts, and other drainageways constructed to divert upslope cleaner runoff from the source of diffuse pollution (disturbed area). Diversion structures can be temporary or permanent.

Grassed Waterways and Channel Stabilization

If runoff is allowed to leave the source areas without a conveyance system, rill and gully erosion will create an uncontrolled channel drainage network. Channel erosion may also occur if natural and man-made channels are damaged by the increased flows generally caused by urbanization or by construction activities (Fig. 10.14). To minimize or prevent this occurrence, collection systems must be designed in such a way that the shear stress caused by the flow on the channel bottom and banks is less than the resistance of the channel to erosion. Example 5.6 presented a method of estimating flow erosivity in evaluating the stability of the channels.

Several engineering means are available to mitigate channel erosion. The shear stress caused by flow is proportional to flow velocity, channel slope, and depth. Thus, collection systems should be designed to keep the velocity of the flow or the slope of the channel below a certain critical value representing the resistance of the channel to erosion. The resistance of unprotected earth-dug channels against erosion is very low. Therefore excavated channels and newly created roadside ditches should always be



FIGURE 10.14. Erosion of unprotected channels in construction zones. (Photo: University of Wisconsin.)

protected. Materials used for dry channel protection (such as roadside ditches) are straw mulch, matting (Fig. 10.15), grass (sod), wire mesh with stone, gabions, rip rap, and other less used and more expensive protective measures. These erosion-control measures should be implemented immediately after the channel is excavated.

In extreme, high-slope situations, channels can be lined with concrete or covered concrete pipes can be used. Lining of natural perennial streams with concrete for control of erosion or flooding is not recommended because such methods destroy aquatic habitat and would meet public resistance.

Grassed Waterways

Grassed waterways are probably the most inexpensive but most effective means of conveying water (Figs. 10.16 and 10.17). In a simple case, a grassed roadside swale will perform as well or better than a far more expensive buried storm sewer. If the grassed waterways are designed properly, in-channel erosion should be minimal, and the grass lining may even serve as a trap and treatment for pollutants.

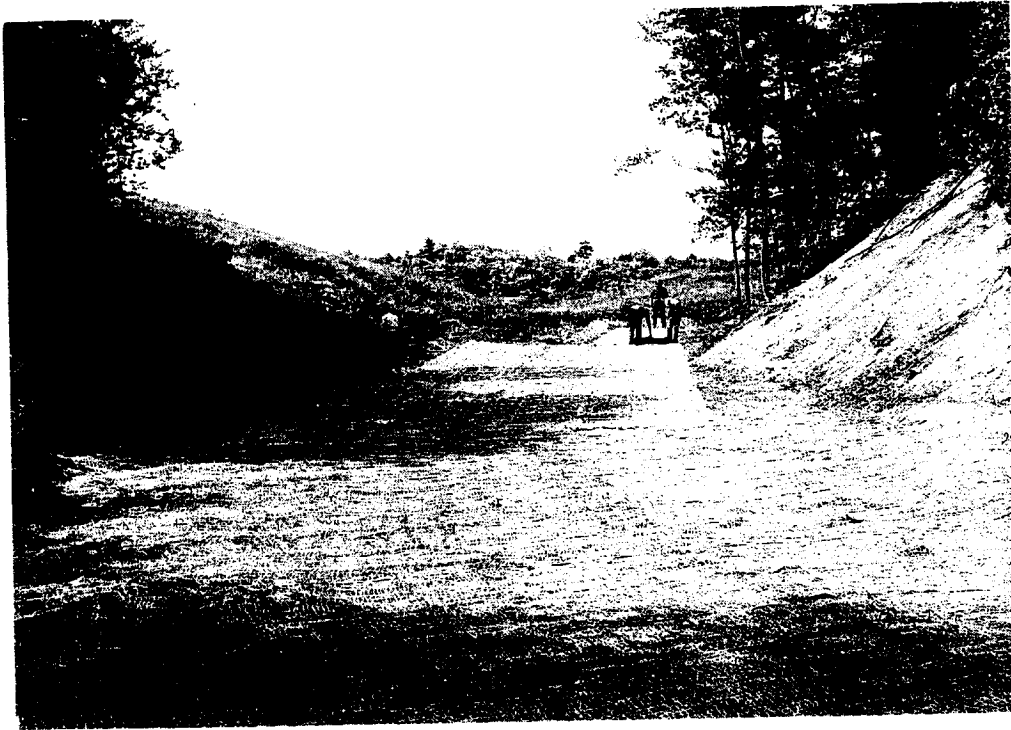


FIGURE 10.15. Adding jute matting to give additional protection in a channel where grass cover has failed during excessive flush runoff events. (Photo: USDA, Soil Conservation Service.)



FIGURE 10.16. Grassed waterway in a residential zone. (Photo: USDA, Soil Conservation Service.)

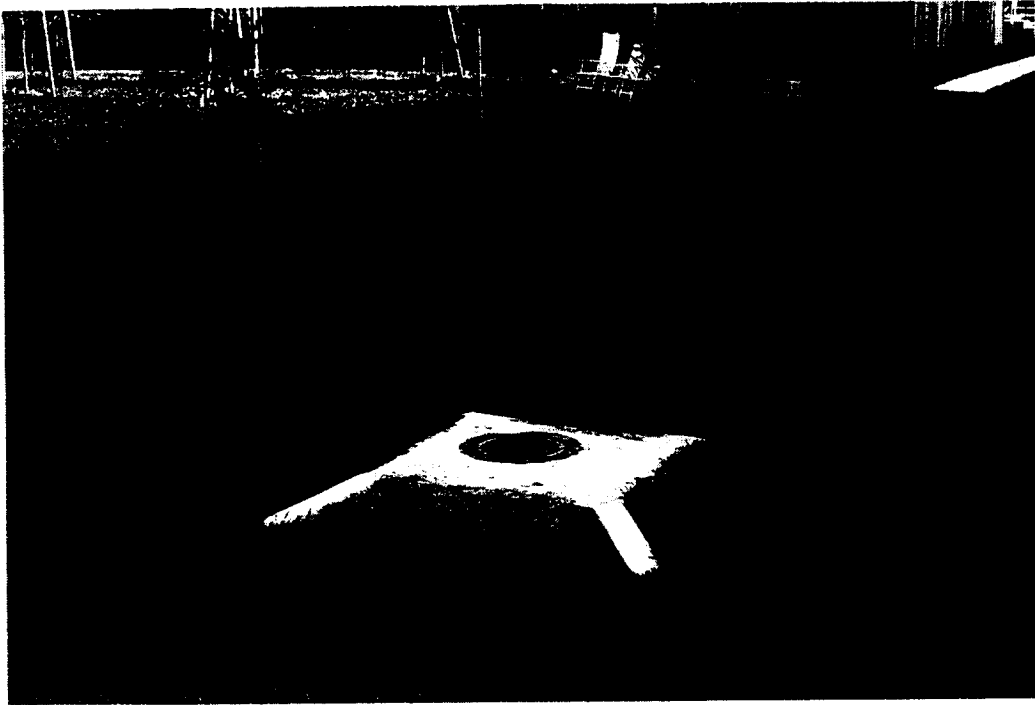


FIGURE 10.17. Urban swale connection to an underground storm sewer. Straw bale provides detention and filtration.



FIGURE 10.18. Temporary sediment traps made of straw bales. Sod is laid immediately after excavation to prevent excessive channel erosion. (Photo: V. Novotny.)

Grassed waterways and roadside swale drainage are man-made channels of parabolic, triangular, or trapezoidal cross section. A typical swale is a shallow trench with side slopes of one (vertical) to three (horizontal). Swales are used solely for the conveyance of surface runoff, hence they are mostly dry and contain water only following a rain. Planted vegetation (primarily grass) is for soil stabilization and storm-water treatment. Grassed waterways are primarily used for the conveyance and treatment of runoff from surface mining and similar operations, and from agricultural and silvicultural lands (Chapter 11). As with grass filters, swales and grassed waterways remove pollutants by slowing down the flow, filtering by grasses, infiltration, and by nutrient uptake by vegetation. In contrast to grass filters, however, the grasses in swales are submerged, flow in the waterways is turbulent, and are therefore less effective in removing particulates than grass filters. The efficiency of grassed waterways (swales) in removing pollutants is about 30% (Oakland, 1983). Generally, the lower the slope and the velocity, the better the treatment performance of swales. Higher slopes can be reduced by check

TABLE 10.5 Maximum Permissible Design Velocities for Waterways

Cover	Range of Channel Gradient (%)	Permissible Velocity (m/sec)
<i>Vegetative^{a,b}</i>		
1. Tufcote, Midland and Coastal Bermuda grass ^c	0-5.0	1.8
	5.1-10.0	1.5
	Over 10	1.2
2. Reed canary grass, Kentucky 31 tall fescue, Kentucky bluegrass	0-5.0	1.5
	5.1-10.0	1.2
	Over 10	0.9
3. Red fescue	0-5.0	0.75
4. Annuals ^d —ryegrass	0-5.0	0.75
<i>Unlined Channels^{e,f}</i>		
5. Fine sand		0.5-0.75
6. Silt loam		0.6-0.9
7. Alluvial silts, colloidal		1.15-1.5
8. Fine gravel		0.75-1.5
9. Coarse gravel		1.2-1.8

^a After Ree and Palmer (1949).

^b To be used only below stabilized or protected area.

^c Common Bermuda grass is considered to be a restricted noxious weed in Maryland.

^d Annuals—use only as temporary protection until permanent vegetation is established.

^e After Goldman, Jackson, and Bursztynsky (1986).

^f Lower velocity is recommended for clean water, higher is allowed for silty water.

dams and/or by temporary barriers made of straw bails or cloth (Fig. 10.18).

The basic design criteria for grassed waterways are given in Tables 10.5 and 10.6 and in Figure 10.19. Table 10.5 shows the maximum flow velocities that would cause minimum erosion for different types of vegetation linings. Table 10.6 classifies vegetative linings into retardance groups, which basically express the hydraulic roughness of the vegetal cover. For design velocity of less than 1 m/sec, seeding and mulching are needed to establish vegetation. For design velocities over 1 m/sec, the waterway or swale should be stabilized with sod (bottom), the seeded sides protected by jute or excelsior matting, or with seeding and mulching, along with diversion of runoff until the vegetation is established.

Channels can be designed using the common Manning's formula

$$Q = A \frac{1}{n} R^{2/3} S^{1/2} = Av \quad (10.8)$$

where

Q = flow rate in the channel (m^3/sec)

TABLE 10.6 Classification of Vegetative Cover in Waterways Based on Degree of Flow Retardance by the Vegetation

Cover	Stand	Condition and Height	Retardance
Reed canary grass	Excellent	Tall (average 1 m)	A
Kentucky 31 tall fescue	Excellent	Tall (average 1 m)	
Tufcote, Midland and Coastal Bermuda grass	Good	Tall (average 30 cm)	B
Reed canary grass	Good	Mowed (30–40 cm)	
Kentucky 31 tall fescue	Good	Unmowed (average 50 cm)	
Red fescue	Good	Unmowed (average 40 cm)	
Kentucky bluegrass	Good	Unmowed (average 40 cm)	
Redtop	Good	Average	
Kentucky bluegrass	Good	Headed (15–30 cm)	
Red fescue	Good	Headed (15–30 cm)	
Tufcote, Midland and Coastal Bermuda grass	Good	Mowed (average 15 cm)	
Redtop	Good	Headed (40–60 cm)	D
Tufcote, Midland and Coastal Bermuda grass	Good	Mowed (6 cm)	
Red fescue	Good	Mowed (6 cm)	
Kentucky bluegrass	Good	Mowed (5–12 cm)	

Source: After Ree and Palmer (1949).

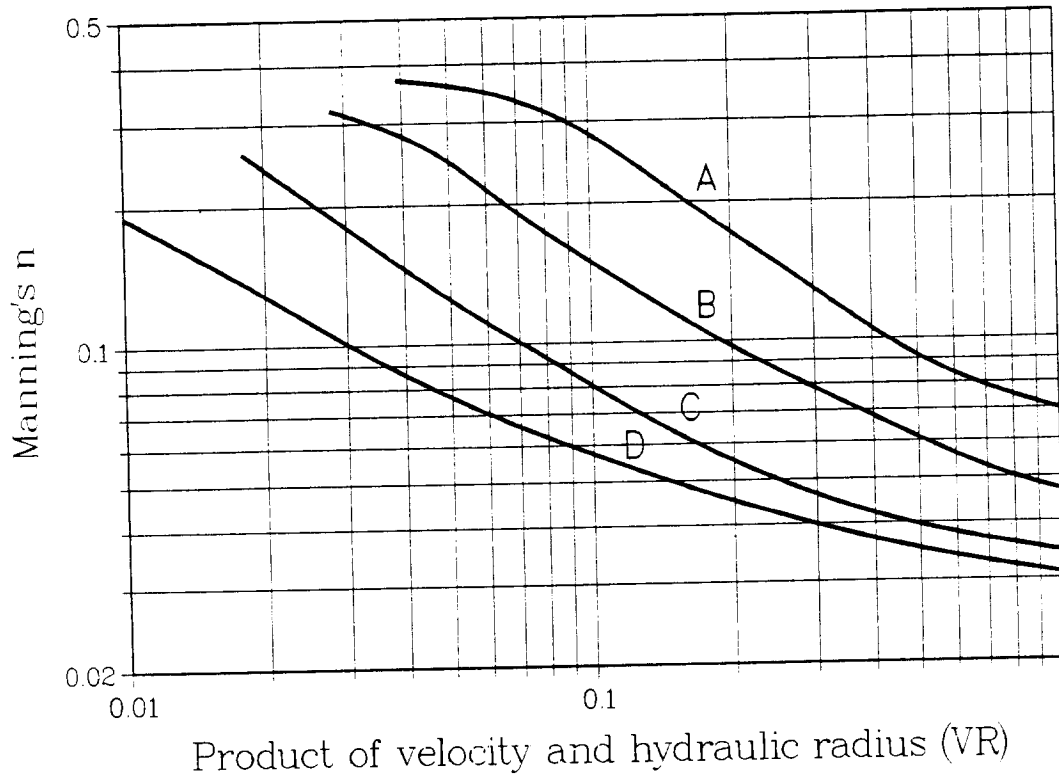


FIGURE 10.19. Manning's n for grassed waterways related to velocity v (m/sec), hydraulic radius R (m), and vegetal retardance. (After Ree and Palmer, 1949.)

- A = cross-sectional area of the channel (m^2)
- n = Manning's roughness coefficient for the channel (Fig. 10.19)
- R = hydraulic radius defined as cross-sectional area divided by the wetted perimeter (m)
- S = slope of the energy line of flow, which under steady conditions equals approximately the slope of the channel bottom (dimensionless—m/m)
- v = mean velocity (m/sec)

The roughness coefficient for grassed waterways depends on flow conditions and vegetative cover. Figure 10.19 shows the magnitude of the roughness coefficient for the four retardance classes defined in Table 10.6. In the figure Manning's roughness coefficient is related to the product of velocity and the hydraulic radius.

Example 10.4: Grassed Waterway Design

Determine the nonerosive velocity and dimensions of a trapezoidal waterway given that $Q = 2 m^3/sec$, the slope of the channel is 5%, the bank

slope is 1 (vertical) to 3 (horizontal), and the vegetation cover is a good stand of headed Kentucky bluegrass (C-retardance curve (see Table 10.6)).

From Table 10.5 the permissible velocity $v_{\max} = 1.5$ m/sec. The solution is a trial-and-error process, since the Manning roughness coefficient in Figure 10.19 depends on the product of the velocity and hydraulic radius.

Solution First, make a first estimate of the cross-sectional area:

$$A = \frac{Q}{v_{\max}} = \frac{2}{1.5} = 1.33 \text{ m}^2$$

Then try depth of flow $H = 0.3$ m and calculate the width of the channel (note that the channel banks have slope of 1:Z, where $Z = 3$. Then the cross-sectional area is

$$A = BH + ZH^2$$

where B is the bottom width, B is computed from

$$1.33 = B \times 0.3 + 3 \times 0.3^2$$

or

$$B = 3.53 \text{ m}$$

The wetted perimeter (P) is then calculated

$$P = B + 2H\sqrt{Z^2 + 1} = 3.53 + 2\sqrt{3^2 + 1} = 5.43 \text{ m}$$

and the hydraulic radius is

$$R = \frac{A}{P} = \frac{1.33}{5.43} = 0.24 \text{ m}$$

Estimate the Manning's roughness coefficient using the product of velocity and hydraulic radius

$$vR = 1.5 \times 0.24 = 0.33 \text{ m}^2/\text{sec}$$

From Figure 10.19, for a value of $vR = 0.33 \text{ m}^2/\text{sec}$ and C-retardance, the Manning factor $n = 0.043$.

Next compute v and compare with the initial assumption:

$$v = \frac{1}{n} R^{2/3} S^{1/2} = \frac{1}{0.043} \times 0.24^{2/3} \times 0.05^{1/2} = 2.03 \text{ m/sec}$$

which is not acceptable since $v > v_{\max}$. Therefore try $H = 0.2$ m.

$$B = \frac{A - ZH^2}{H} = \frac{1.33 - 3 \times 0.2^2}{0.2} = 6.05 \text{ m}$$

$$P = 6.05 + 2 \times 0.2 \sqrt{3^2 + 1} = 7.31 \text{ m}$$

$$R = \frac{A}{P} = \frac{1.33}{7.31} = 0.18 \text{ m}$$

$$vR = 1.5 \times 0.18 = 0.27 \text{ m}^2/\text{sec}$$

From Figure 10.19, $n = 0.048$ and

$$v = \frac{1}{n} R^{2/3} S^{1/2} = \frac{1}{0.048} \times 0.18^{2/3} \times 0.05^{1/2} = 1.5 \text{ m/sec}$$

which is acceptable since the computed v agrees with the initial estimate.

Therefore, for $Q = 2.0 \text{ m}^3/\text{sec}$, design a grassed waterway about 6.1 meters wide at the bottom, which will result in a design flow depth of roughly 0.2 meter and a nonerosive flow velocity of $v \leq 1.5 \text{ m/sec}$.

Ripraps and Gabions

Riprap is a layer of loose rock or concrete blocks placed over an erodible soil surface. *Gabions* are blocks made of wire mesh filled with smaller rocks or gravel. Both ripraps and gabions are primarily used for channel stabilization in high erosion zones such as sharp bends, channel drops, and flow-energy dissipators (stilling basins) below the outlets from sewers, narrow bridges, and near connections with lined high-velocity channels. Note that energy dissipators are permanent structure that require hydraulic design. The Manning roughness coefficient, n , used for determining velocities and flows of riprap- (gabion-) lined channels can be obtained from Figure 10.20.

Sediment Barriers and Silt Fences

Sediment barriers and silt fences are small temporary structures used at various points within, and at the periphery of a disturbed area to detain runoff for a short period of time and trap heavier sediment particles. Sediment barriers are built from many materials (Goldman, Jackson, and Bursztynsky, 1986), such as straw bales (Fig. 10.18), filter fabric attached

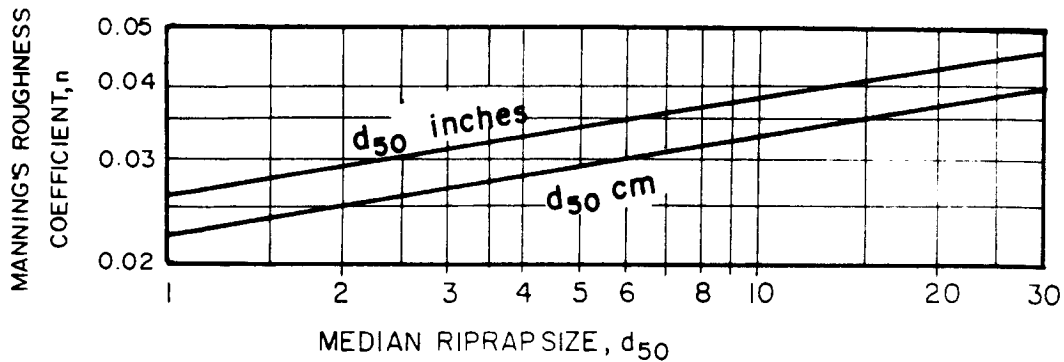


FIGURE 10.20. Manning's n for riprap surfaces. (After U.S. EPA, 1976.)

to a wire or wood fence, filter fabric on straw bales, and gravel and earth berms. These barriers are placed in the path of sediment-laden runoff, commonly from construction sites and surface mining. A sediment barrier should not be placed across a drainageway that carries a large volume of runoff.

Collection Sewer Control

Combined sewer overflows (CSOs) have been recognized as a significant source of diffuse urban pollution since the late 1960s and early 1970s (Sullivan, 1968; Graham, 1978). Initially, sewer separation was considered a solution; however, since then discussions about whether "to separate or not to separate" have continued (see Chapter 1). Nevertheless, combined sewer overflow pollution-control practices use unit processes and units that are either unique to CSO control or common to both types of sewer systems (for example, detention, hydrologic modifications, infiltration).

Catch Basins

A *catch basin* is a chamber or well, usually built at the curbline of a street, through which storm water is admitted into the sewer system. In contrast to a simple inlet chamber, catch basins are equipped to retain grit, detritus, and other sediment. Historically, the role of catch basins has been to minimize sewer clogging by trapping coarse debris and to reduce odor emanation from low-velocity sewers by providing a water seal (Field, 1990). Typically, a catch basin has a sump at its base. This sump should be large enough to provide storage for trapped debris. A Japanese version of catch basins in the Experimental Sewer System (ESS) described by Fujita (1984) includes a perforated removable bucket above the sump for trapping larger sediment and debris and a permeable bottom to the sump for enhanced infiltration.

Lager, Smith, and Tchobanoglous (1977) claimed that catch basins installed in sewer systems were ineffective, as they do not remove appreciable amounts of pollutants and the decaying organics trapped in them are actually a source of pollution. However, in a project conducted in Boston, Massachusetts, catch basins were shown to be quite effective for solids reduction (60–97%). Removal of associated pollutants, such as COD and BOD, were also significant (10–56% and 54–88%, respectively). To maintain the effectiveness of catch basins, they must be cleaned about twice a year, depending on local conditions (Aronson, Watson, and Pisano, 1983; Field, 1986, 1990).

Sewer Flushing

The solids that are deposited on the bottom of sewers and the biological slime that grows on the walls of combined sewers during the dry, low-flow period have been recognized as a problem for a long time. As a matter of fact, as documented by Krejci et al. (1987) this pollutant source may represent a major portion of the total pollution load in CSOs. Estimates of the magnitude of solids accumulation in sewers were presented in the preceding chapter.

Sewer flushing during dry weather is designed to periodically remove the accumulated material and convey it to the treatment plant downstream. Flushing is especially necessary in sewer systems that have low grades, as a result of which velocities during low flow periods fall below those needed for self-cleaning (typically, self-cleaning velocities are above 0.6 to 1 m/sec). It may be convenient to combine sewer flushing with street flushing, although street flushing from tanker trucks (Fig. 10.2) may not provide enough flow and thus should be used for solids removal in smaller-diameter laterals and trunk sewers (Field, 1990). Flow for flushing can also be provided by gates installed in some strategically located manholes (Novotny et al., 1989). Internal automatic flushing devices have also been developed for sewer systems. An inflatable bag is used to stop flow in upstream reaches until a volume capable of generating a flushing wave is accumulated, at which point the bag is deflated with the assistance of a vacuum pump, and the released sewage cleans the sewer segment (Field, 1990).

Regulators, Concentrators, and Separators

Regulators, concentrators, and separators are capable of separating solids from the flow of storm water. The dual-purpose swirl-flow regulator–solids concentrator has shown a potential for simultaneous quality and quantity control (Field 1986, 1990). A helical-type regulator–separator

has also been developed. These devices have been primarily applied to CSO; however, they can also be installed as storm-runoff pollution-control devices. The concentrated flows that may amount only to a few percent of the total runoff flow can be stored and subsequently directed toward sanitary sewers for treatment during low-flow periods. The swirl concentrator was included in a CSO abatement program in Michigan (Pisano, Connick, and Aronson, 1985).

The *vortex solids separator* is a compact solids separation device. As early as in 1932, the idea of separating solids from CSO in a vortex chamber was conceived in England (Brombach, 1987, 1989; Pisano, 1989, 1991). The idea was pursued in the United States in the 1970s, resulting in a device known today as a *swirl concentrator*. Similar devices known as *fluidsep*, developed in Germany, or *Storm King*, developed in the United Kingdom, have also been implemented throughout the United States and Europe.

Vortex separation devices have no moving parts (Fig. 10.21). Dry-weather sewage passes unimpeded through the unit, if used as a combined sewer, in-line regulator. If the device is intended to operate as an

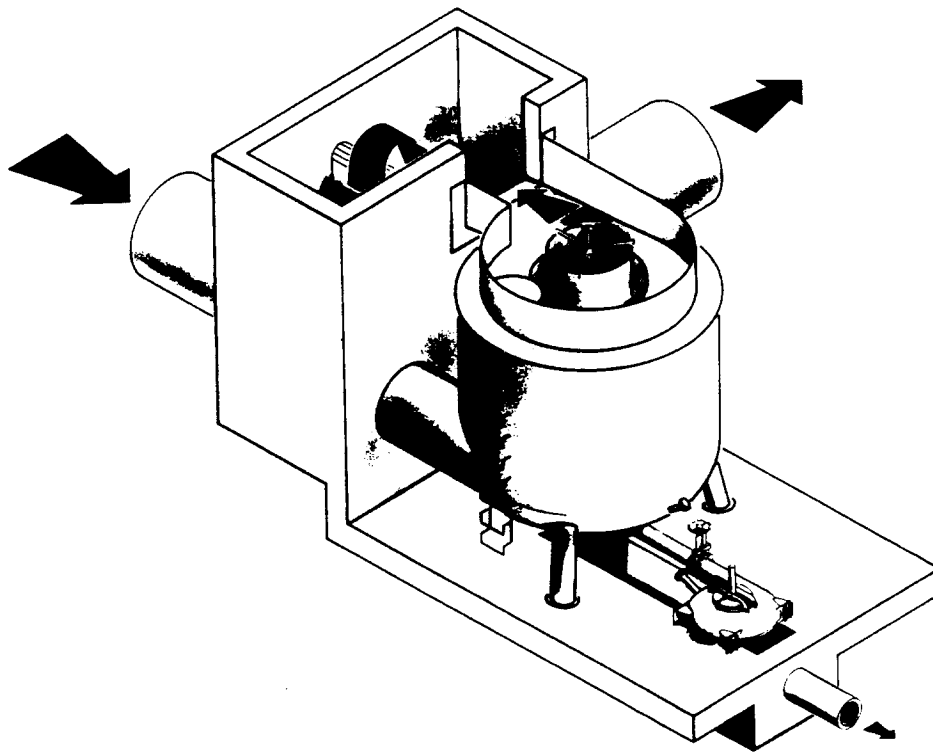


FIGURE 10.21. Fluidsep (vortex) separator. (Courtesy of W. Pisano.)

off-line treatment unit only, then storm flows are deflected into the unit by gravity or by pumps. During wet weather the unit's outflow is throttled (typically only 3% to 10% of the flow passes through the foul sewer outlet toward the treatment plant), causing the unit to fill and to self-induce a "swirling" vortexlike operation. Settleable grit and floatable matter are rapidly removed. Concentrated foul matter is sent to the treatment plant (or sent to temporary storage), while the cleaner, treated flow discharges into the receiving waters.

The three primary determinants governing the process performance of these devices are the hydraulic throughput, amount of internal fluid turbulence generated in the device, and the relative preponderance of particles whose settling velocities exceed 1 mm/sec (Pisano, 1991). Solids removal decreases with an increasing flow rate, and increases with more coarse (gritty) particle grain distribution. Any internal vessel protrusions that disrupt smooth vortex patterns cause decreased performance.

Vortex solids separators remove settleable matter by two mechanisms (Pisano, 1990, 1991; Field, 1990):

1. The sweeping action of solids by secondary vortex flow currents toward the centroidal axis of rotation
2. The transport of particles by gravity in the laminar sheet flow regime on the floor of the unit toward the same axis

There are some differences, none major, between the U.S. "swirl concentrator" and the German "fluidsep" (Pisano, 1990), but both work on the same principle. A design handbook for vortex separators was prepared by Sullivan et al. (1982). A partial list of U.S. installations with experience in their use was presented by Pisano (1989, 1991).

The design overflow rate for vortex separators installed in the United States ranges from 10 to 30 l/sec-m² (swirl concentrator) and 18 to 140 l/sec-m² (fluidsep), respectively. Up to 60% of suspended solids removal can be achieved, but generally the performance of such units is less than that for primary treatment.

A *helical bend regulator/concentrator* induces helical motion in a curved separator with a bend angle of about 60° and a radius of the curvature equal to 16 times the inlet pipe diameter. Dry-weather flow passes through the lower portion of the device to the intercepting sewer. As the flow increases during a wet weather period, the helical motion begins and the particles are drawn to the inner wall and drop to the lower channel leading to the treatment plant. The excess cleaner flow overflows over a weir into a CSO. The removal efficiency of helical bend separators is about the same as that of the swirl concentrator (Field, 1990). The

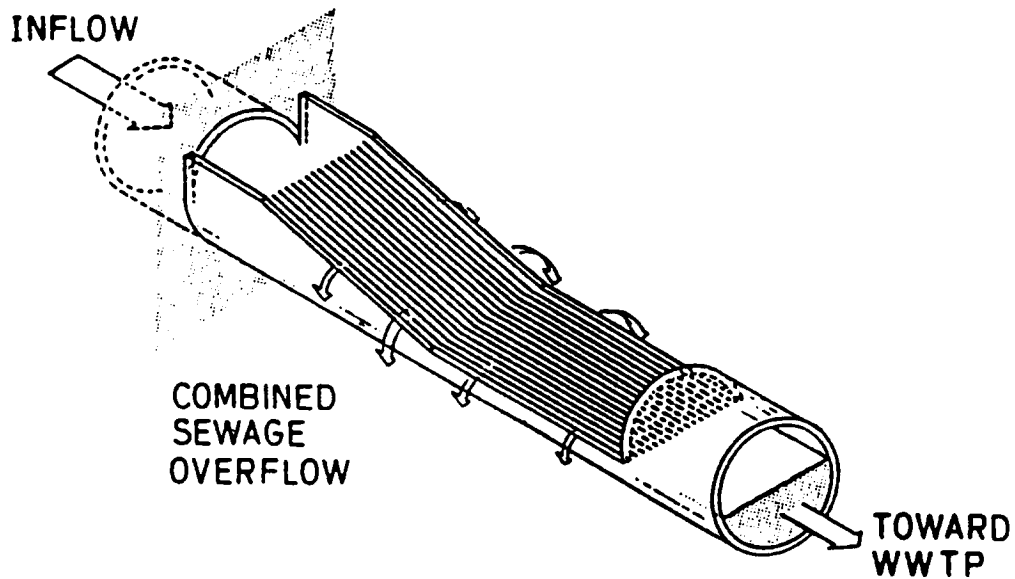


FIGURE 10.22. Self-cleaning screens for the control of pollution in combined sewer overflows and urban storm-water runoff. (After Krejci, 1988.)

handbook by Sullivan et al. (1982) also contains design parameters for helical bends.

Screening

Unlike screens in treatment plants that can be supervised, manually or mechanically cleaned, and where the screenings (material collected on the screens) can be removed and transported away, screens used for removal of solids from CSOs or urban runoff must be self-cleaning and reliable without supervision. An example of a self-cleaning structure developed and implemented in Switzerland is shown on Figure 10.22. In this simple fixture solids that are captured by the screens are washed by the flow that is diverted to the treatment plant. Pilot investigations indicated a treatment capacity of 0.2 to 0.4 m³/sec-m² for screens without clogging (Novak, 1983; Krejci, 1988; Krejci and Baer, 1990).

DETENTION-RETENTION FACILITIES

Urban runoff entering a drainage system is a highly variable, intermittent phenomenon. In this case, storage and flow equalization, which can also be connected with pollution removal, is necessary and storage facilities must be included in the master plan for pollution abatement. Storage facilities also provide for maximum use of existing dry-weather flow treatment facilities, minimize their overload, and allow for the subse-

quent treatment of stored excess flow. Storage facilities range from inexpensive multipurpose terrain depressions with restricted outlets to expensive underground tunnels drilled in underlying bedrock (Milwaukee, Chicago). Wetlands also provide a combination of storage and treatment and are becoming an attractive alternative for storm-water and CSO pollution control. Today, ponds and storage basins are the backbone of urban storm-water quantity–quality management.

Detention Volume for Quality Control

Before the release of current storm-water quality control regulations ponds and detention basins were primarily used for peak attenuation from storm runoff rather than for quality control. As a matter of fact, many dry storage facilities designed previously for large design storms (5-year or greater recurrence interval) are ineffective for quality control. For example, Roesner (1988) has shown that in Cincinnati, Ohio, 74% of the total annual rainfall volume is contained in storms with a rainfall depth of less than 2.54 cm (1 in.). Considering the capture of the first 2.5 cm of rainfall would not only capture the total volume of a majority of storms (94% of storms in Cincinnati are less than 2.5 cm) but also two-thirds of the volume of larger than 2.5-cm storms. Hence, 91% of the runoff that falls on the watershed would be captured and subsequently treated. A 2.5-cm rainfall is substantially less than a 1-year storm. Similarly, Vitale and Spray (1975) reported that an 85% decrease in total BOD load can be realized by capturing the first 0.8 to 2.5 cm of the runoff. Hence, most guidelines for storm-water quality control call for the capture of rainfall depths of less than 2.54 cm.

As a rule-of-thumb calculation, consider a 2.5-cm rainfall falling on a residential watershed that is 50% impervious. Only flow from impervious areas will be considered because in most cases surface runoff is not generated from pervious areas by smaller storms. If surface storage of impervious surfaces is assumed to be about 0.15 cm (see Chapter 3), the flow generated from a 1-ha (10,000 m²) catchment will be

$$0.5(2.5 \text{ cm} - 0.15 \text{ cm}) \times 0.01 \text{ m/cm} \times 10,000 = 117 \text{ m}^3$$

If one assumes further that the typical depth of the water in the pond is about 1 meter, then the planimetric extent of the pond surface area is 117 m², or slightly above 1% of the catchment area, or 2.2% of the connected impervious area. Typically, about 2% to 4% of the connected impervious area should be devoted to management of urban diffuse pollution by detention–retention facilities. The 90% capture and treat-

ment of the total load from sewered urban areas is mandated in Germany. However, this volume, designed to capture small storms and the first flush of larger storms, is not sufficient to remove compounds such as nutrients that require longer detention times.

Sizing the CSO and Storm-Water Storage–Quality Control Basins

Kuo and Zhu (1989) presented designs for diversion systems for first flush overflow control. Griffin, Randall, and Grizzard (1980) developed a method by which it is possible to size the detention basin based on a predetermined portion of runoff (first flush) to be stored in the basin.

The sizing of storage basins can best be accomplished by continuous simulation models, since it involves the optimization (minimization of cost) of two components of the system, that is, the storage volume and the excess treatment plant capacity (the rate at which the storage basin is gradually emptied during the subsequent dry period). Few dynamic models are available for modeling the transient phenomenon of filling and emptying the storage tanks. Of the models discussed in Chapter 9, the Storm Water Management Model (SWMM) is capable of computing storage volumes; however, in most applications the SWMM is an event-oriented model. Lessard and Beck (1991) presented a conceptual model for dynamic simulation of the dynamic performance of storm retention tanks. Four modes of behavior have been identified: fill, draw, dynamic sedimentation, and quiescent settling.

Using popular so-called “design storms” to size the CSO storage facilities may not be appropriate, since in the United States the constraining parameter is the number of overflows allowed per year, while in Europe total stored and treated volume is the controlling variable.

These control parameters do not render themselves to the “excess design runoff” storage capacity determination that is used for flood control. The difference in storage capacity determination between flood control and water quality control is shown on Figure 10.23. However, these two constraints are similar, and their application may lead to a unified methodology, as pointed out by Roesner, Burges, and Aldrich (1991).

By continuous simulation modeling Roesner, Burges, and Aldrich have shown that the basin storage volume needed to capture a certain portion of runoff and the number of overflows are interrelated, as shown in Figure 10.24 and Table 10.7. To a lesser degree the storage volume is related to the excess treatment plant capacity, which regulates the outflow from the storage. The pumping or release rate of stored runoff (CSO) from the basin should provide a minimum of 24 hours retention (Grizzard et al., 1986; Schueler, Kumble, and Heraty, 1991). The storage volume

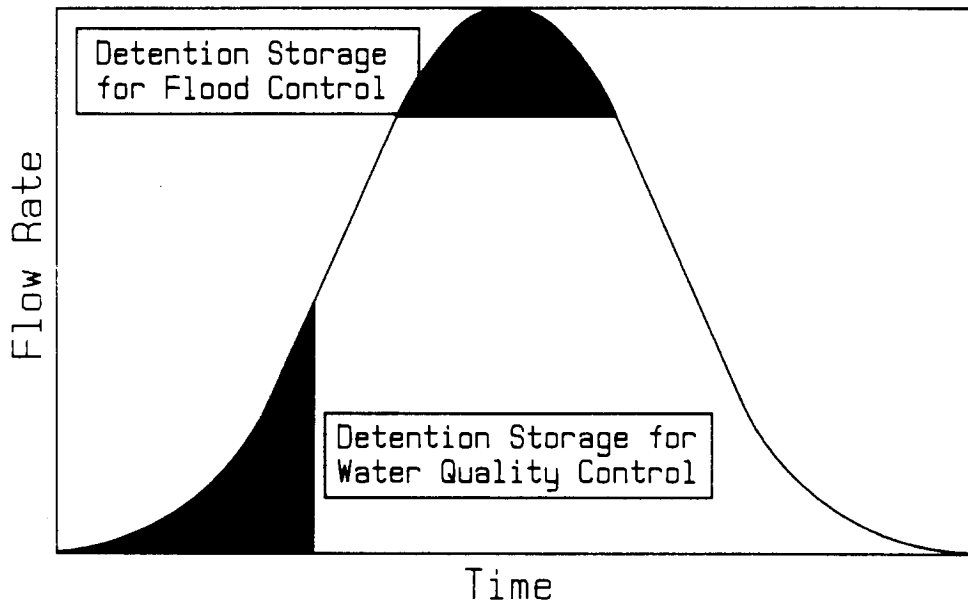


FIGURE 10.23. Storage strategies for urban flood control and pollution capture. (After Roesner, Burges, and Aldrich, 1991.)

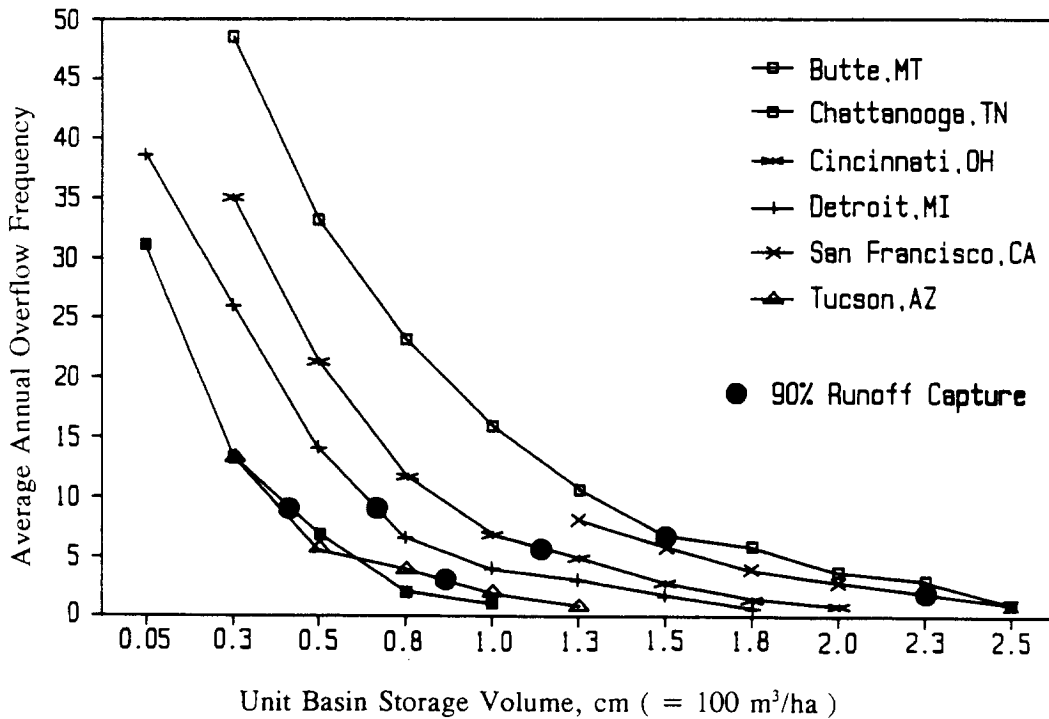


FIGURE 10.24. Relation of overflow frequency to unit storage volume. (After Roesner, Burges, and Aldrich, 1991.)

ment of the total load from sewerage urban areas is mandated in Germany. However, this volume, designed to capture small storms and the first flush of larger storms, is not sufficient to remove compounds such as nutrients that require longer detention times.

Sizing the CSO and Storm-Water Storage—Quality Control Basins

Kuo and Zhu (1989) presented designs for diversion systems for first flush overflow control. Griffin, Randall, and Grizzard (1980) developed a method by which it is possible to size the detention basin based on a predetermined portion of runoff (first flush) to be stored in the basin.

The sizing of storage basins can best be accomplished by continuous simulation models, since it involves the optimization (minimization of cost) of two components of the system, that is, the storage volume and the excess treatment plant capacity (the rate at which the storage basin is gradually emptied during the subsequent dry period). Few dynamic models are available for modeling the transient phenomenon of filling and emptying the storage tanks. Of the models discussed in Chapter 9, the Storm Water Management Model (SWMM) is capable of computing storage volumes; however, in most applications the SWMM is an event-oriented model. Lessard and Beck (1991) presented a conceptual model for dynamic simulation of the dynamic performance of storm retention tanks. Four modes of behavior have been identified: fill, draw, dynamic sedimentation, and quiescent settling.

Using popular so-called “design storms” to size the CSO storage facilities may not be appropriate, since in the United States the constraining parameter is the number of overflows allowed per year, while in Europe total stored and treated volume is the controlling variable. These control parameters do not render themselves to the “excess design runoff” storage capacity determination that is used for flood control. The difference in storage capacity determination between flood control and water quality control is shown on Figure 10.23. However, these two constraints are similar, and their application may lead to a unified methodology, as pointed out by Roesner, Burges, and Aldrich (1991).

By continuous simulation modeling Roesner, Burges, and Aldrich have shown that the basin storage volume needed to capture a certain portion of runoff and the number of overflows are interrelated, as shown in Figure 10.24 and Table 10.7. To a lesser degree the storage volume is related to the excess treatment plant capacity, which regulates the outflow from the storage. The pumping or release rate of stored runoff (CSO) from the basin should provide a minimum of 24 hours retention (Grizzard et al., 1986; Schueler, Kumble, and Heraty, 1991). The storage volume

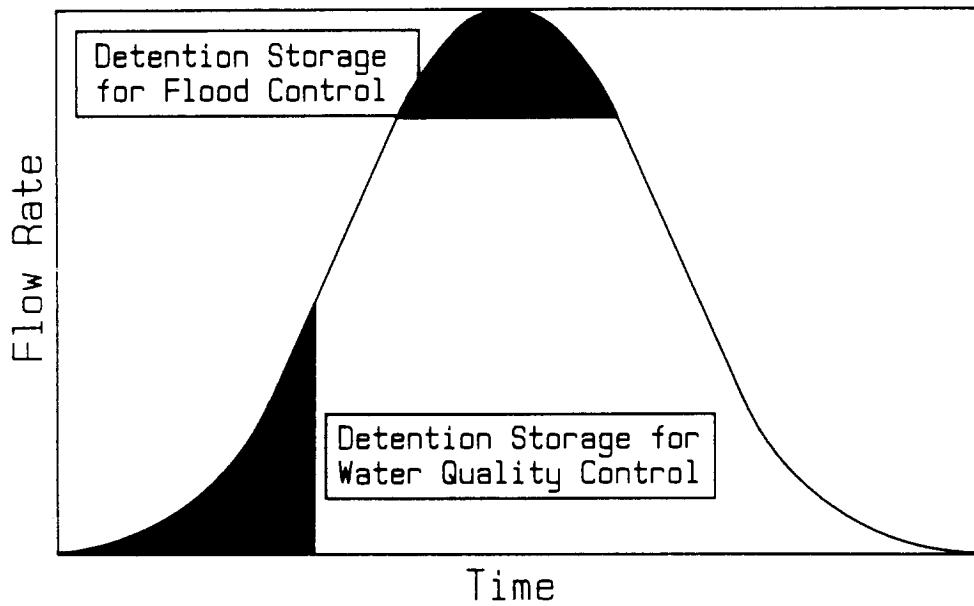


FIGURE 10.23. Storage strategies for urban flood control and pollution capture. (After Roesner, Burges, and Aldrich, 1991.)

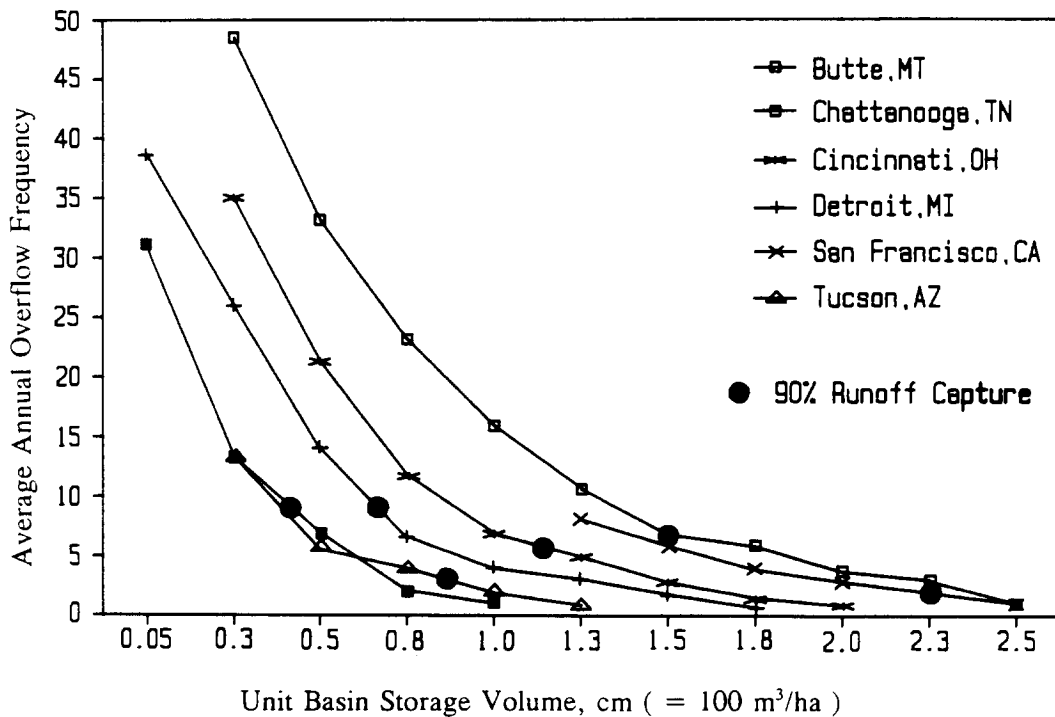


FIGURE 10.24. Relation of overflow frequency to unit storage volume. (After Roesner, Burges, and Aldrich, 1991.)

TABLE 10.7 Relation of Overflow Frequency, Storm Recurrence Interval, and Storage Volume Needed to Capture 90% of the Flow

City	Storage Volume Required for 90% Runoff Capture (cm) ^a	Overflow Frequency (times/year)	Recurrence Interval for Design Storm ^b
Butte, Montana	0.46	6	2 month
Chattanooga, Tennessee	1.52	10	1.2 month
Cincinnati, Ohio	1.14	8	1.5 month
Detroit, Michigan	0.69	12	1 month
San Francisco, California	2.29	4	3 month
Tucson, Arizona	0.89	3	4 month

Source: After Roesner, Burges, and Aldrich (1991).

^aDepth of rainfall uniformly distributed over the catchment area. To convert to cubic meters of storage per hectare of catchment area, multiply by 100.

^bThe design storm would be a single storm that would completely fill the initially empty storage basin without an overflow.

of the sediment-control basins must also be sized to accomplish two functions, namely, to effectively remove a certain percentage of the suspended sediment, and to provide sufficient storage capacity for the removed sediment during a period of about five years or more between cleaning.

The planning and design of storm-water detention basins were covered extensively in the following publications:

American Association of State Highway and Transportation Officials (1980): Design of sedimentation basins

DeGroot (1982): Stormwater detention facilities

Ormsbee (1984): Systematic planning of dual purpose detention basins

Urbonas (1984): Summary of findings by ASCE Task Committee on detention basins

Stahre and Urbonas (1990): Stormwater detention for drainage, water quality, and CSO management

Schueler, Kumble, and Heraty (1991) A current assessment of urban best management practices. Techniques for reducing non-point source pollution in the coastal zone

Ponds and Detention Basins

Two types of detention basins are used for quality control of urban runoff. The first type is the *wet detention pond*, which maintains a permanent pool of water with additional storage designated to capture

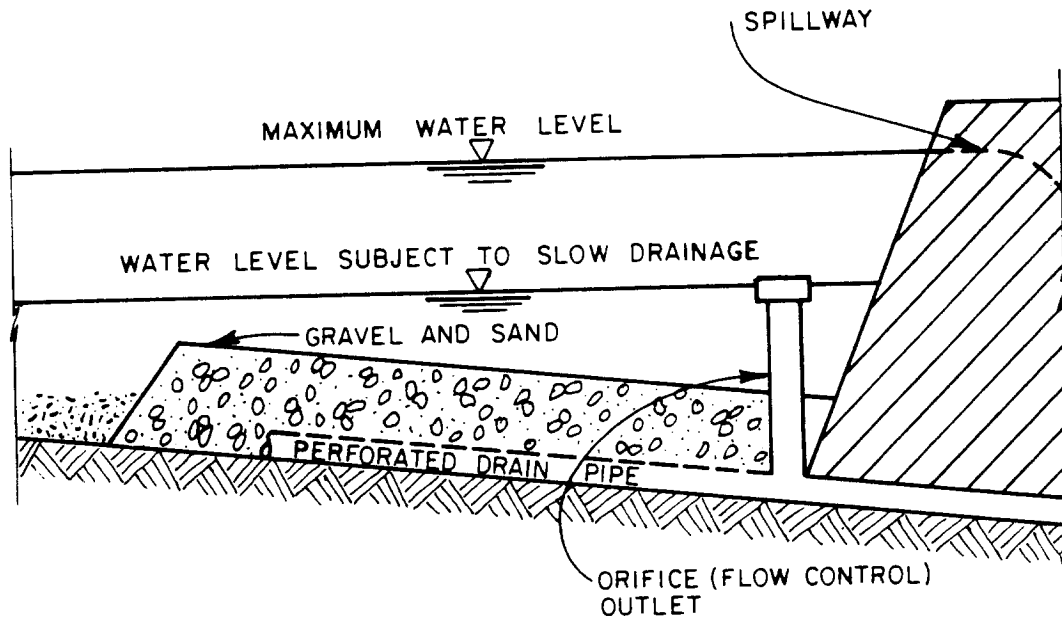


FIGURE 10.25. Modified (extended) dry pond.

transient storm runoff. The second type is the *extended* or *modified dry pond*, which provides a part of its storage capacity for enhanced settling of solids and auxiliary removal of pollutants by filtering.

The *dry pond* is a storm-water detention facility that is normally empty and is designed to hold storm water temporarily during high peak flows. The outlet is restricted and the storage volume is filled only during flows that exceed the outlet capacity. The pond also has a safety overflow spillway for the conveyance of very high flows when the storage capacity of the pond is exhausted. Since the outlet of dry ponds used for flood control is typically meant for large storms, smaller but polluting runoff events will pass through them without appreciable attenuation of the pollution load. Hence, such dry ponds are ineffective for urban runoff quality control. However, by combining the dry detention pond with an infiltration system located at the bottom of the pond, its pollution-control capability is enhanced. Modified dry ponds are effective pollution-control devices in both sewer systems (Fig. 10.25).

A *wet detention pond* has a permanent pool of water. This type of pond acts as a settling facility with relatively low efficiency (Fig. 10.26). Accumulated solids must be removed (dredged out) in order to maintain the removal efficiency and aesthetics of the pond. Improper design and maintenance can make such facilities an eyesore and a mosquito-breeding mudhole. On Figure 10.26 one can see that the pond located in a park is already partially filled with solids after about 10 years of operation and



FIGURE 10.26. Storm-water wet detention pond in Wauwatosa, Wisconsin. Note large sediment deposits in the middle of the basin, which should be periodically dredged. (Photo: V. Novotny.)

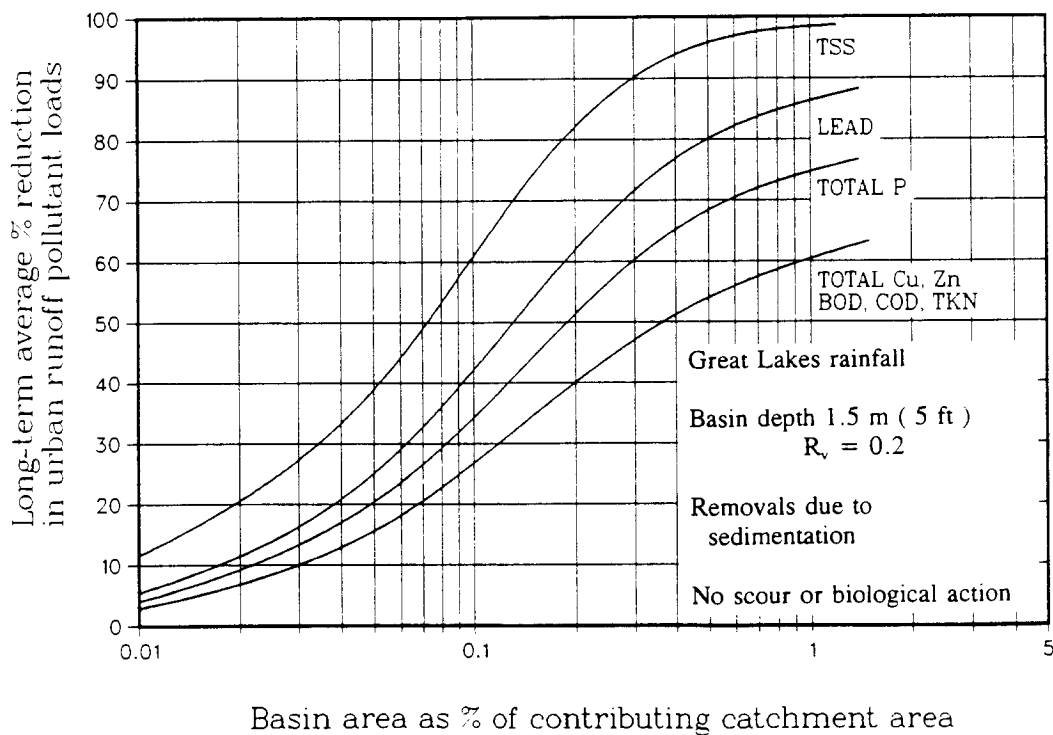


FIGURE 10.27. Approximate removal efficiencies of conventional wet detention ponds. (After Driscoll, 1988.)

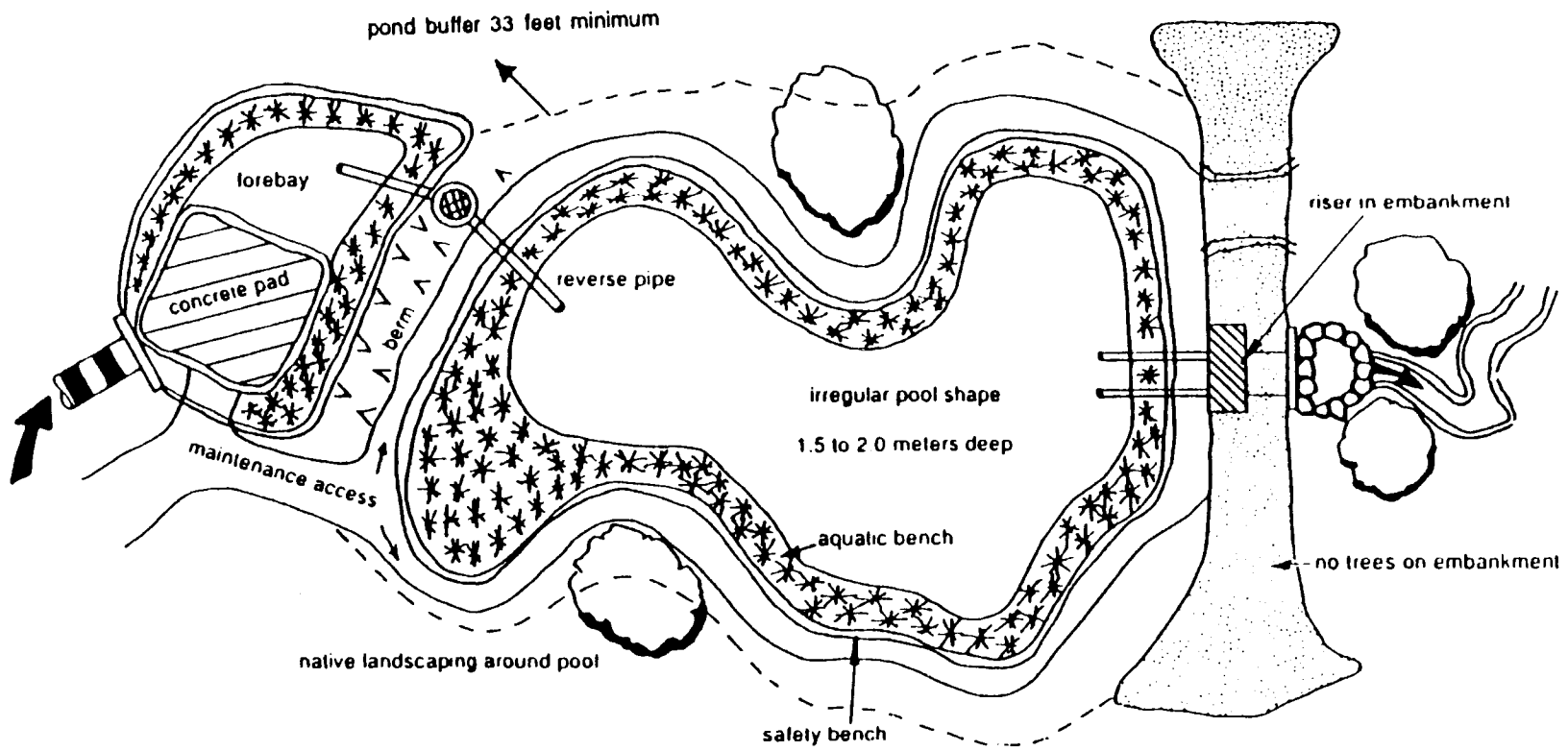


FIGURE 10.28. Enhanced high-efficiency wet pond system. (After Schueler, Kumble, and Heraty, 1992.)

should be cleaned out. The removal efficiency of wet ponds for constituents, obtained by a statistical analysis of NURP study sites is shown on Figure 10.27.

A well-designed wet pond consists of (1) a permanent water pool, (2) an overlying zone in which the design runoff volume temporarily increases the depth of the pool while it is stored and released at the allowed peak discharge rate, and (3) a shallow littoral zone acting as a biological filter (Fig. 10.28). Wanielista et al. (1982) described a system consisting of a detention pond with effluent infiltration, which increased the otherwise low efficiency of the detention pond. Similarly, Urbonas and Ruzzo (1986) stated that in order to achieve a 50% removal rate for phosphorus, a properly designed wet detention pond must be followed by filtration or infiltration. Schueler and Helfrich (1988) described an improved design for detention ponds that includes a permanent wet pool, extended detention storage, and storm-water storage. The perimeter wetland area created by the extended detention and stormwater storage provides additional water quality improvement (Schueler, Kumble, and Heraty, 1991).

The simplest design method for determining the size of sedimentation basins is based on the classic "overflow rate" theory of settling-tank design. The overflow rate is defined as the flow rate divided by the surface area of the pond. A particle is removed if

$$OR = \frac{Q}{A_s} \leq w \quad (10.9)$$

where

- OR = overflow rate (m/day)
- Q = inflow in the basin (m^3/day)
- A_s = surface area of the basin
- w = particle settling velocity

The particle settling velocity is estimated from Stokes' equation, or

$$w = \frac{gD^2}{18\nu}(\rho_s - 1) \quad (10.10)$$

where

- D = particle diameter (m)
- ρ_s = specific gravity of the particle with respect to water
- g = gravity acceleration (m/sec^2)
- ν = kinematic viscosity (m^2/sec)

Under the “ideal settling basin” assumption, all particles with settling velocities greater than OR will be trapped in the basin. The depth is determined from the required trap volume of the basin, which should be enlarged (at least doubled) to provide storage for the trapped sediment.

Example 10.5: Sedimentation basin design by overflow method

Design a settling basin that would remove sediment with particle sizes up to 0.03 mm. The basin receives flow from a 60% impervious watershed and should be large enough to store flow from a 2.5-cm rainfall that will reach the basin within one hour. Excess rainfall over 2.5-cm depth might be allowed to overflow. The average residence time of runoff in the basin is one day. The catchment area is 10 hectares. Kinematic viscosity is $\nu = 10^{-6} \text{ m}^2/\text{sec}$, and the specific gravity of the particle is $\rho = 2.5$.

Solution Estimate flow in the basin

$$\begin{aligned} Q &= 0.6 \times 10 [\text{ha}] \times 10,000 [\text{m}^2/\text{ha}] \times 0.025 [\text{m of rainfall}] \\ &= 1500 \text{ m}^3/\text{hr} \end{aligned}$$

From Stokes' law (Eq. (10.10)) estimate the settling velocity of the 0.03-mm- (0.00003 m) diameter particles:

$$w = \frac{gD^2}{18\nu} (\rho_s - 1) = \frac{9.81 \times 0.00003^2}{18 \times 10^{-6}} (2.5 - 1) = 0.000736 \text{ m/sec}$$

$$w = 0.000736 \text{ m/sec} \times 3600 \text{ sec/hr} = 2.65 \text{ m/hr}$$

The basin's surface area size is then (Eq. (10.9))

$$A_s = \frac{Q}{w} = \frac{1500}{2.65} = 566 \text{ m}^2$$

To account for nonideal settling, increase the calculated area by 20%, or

$$A_{\text{adjusted}} = 1.2 \times A_s = 1.2 \times 566 = 679 \text{ m}^2$$

Then the depth of the basin is $H = 1500/679 = 2.2 \text{ m}$. This depth may be too large if safety considerations are included. Therefore the depth should be reduced and the surface area increased proportionally, which may improve the treatment efficiency of the basin.

The design of the settling basin based on the overflow rate is simple and has been used for some time for the design of sedimentation basins in sewage and wastewater treatment plants and for the control of sediment problems from strip mines (U.S. EPA, 1976). This method presumes uniform flow conditions at the outflow, so it may be inappropriate for the design of settling basins for storm water when discharge is not uniform and the surface area of the basin varies considerably during the storm event.

A better method of sizing sedimentation basins is based on investigations by Chen (1975). Figure 10.29 shows trap efficiency curves for a settling reservoir related to the reciprocal of the overflow rate. Note that the overflow rate is based on the outflow discharge, which is commonly less variable than the inflow rate. The plot also indicates that the earlier overflow models overestimated the trap efficiency for coarse materials.

Example 10.6: Efficiency of Sedimentation Basin for Solids Removal

Using Figure 10.29, estimate trap efficiency of a sedimentation basin with surface area $A_s = 680 \text{ m}^2$ and average inflow rate of $1500 \text{ m}^3/\text{hr}$. The particle size of the sediment is 0.03 mm :

$$A_s/Q = 680 [\text{m}^2]/(1500 [\text{m}^3/\text{hr}] = 0.4533 [\text{m}^2/(\text{m}^3/\text{hr})] \times 3600 [\text{sec}/\text{hr}] \\ = 1632 \text{ m}^2/(\text{m}^3/\text{sec})$$

From Figure 10.29 the removal efficiency of particles with 0.03 mm diameter is approximately 60%. Complete removal is roughly achieved

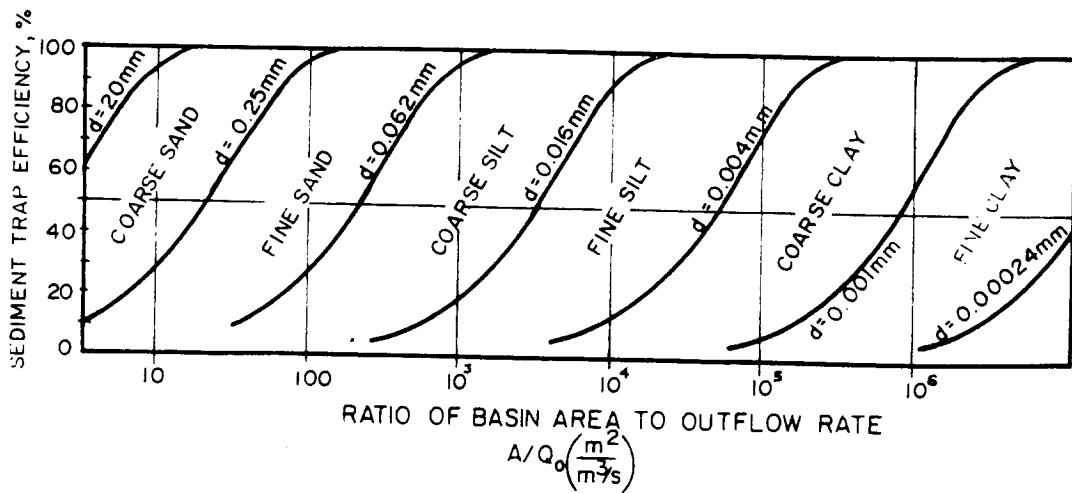


FIGURE 10.29. Settling-trap efficiency curves for detention basins related to top overflow rate. (From Chen, 1975.)

for particle sizes >0.062 mm. It thus appears that the 'ideal settling basin' concept overestimated the removal efficiency.

Whipple and Hunter (1980), reported settleability of some pollutants associated with urban runoff. By laboratory settling experiments they found that lead and hydrocarbons settled out at 60% to 65% in 32 hours. Slightly higher settleability was observed for suspended solids. The removal rates for BOD, zinc, and copper were smaller. Coliform bacteria were reduced by an order of magnitude. Driscoll (1988) evaluated and summarized pollutant trap efficiencies of wet ponds measured by the NURP studies (Fig. 10.27). Trap efficiency of storm-water management basins for pollutant attenuation was reviewed by McCuen (1980). Data collected in Maryland showed that a detention basin can trap as much as 98% of pollutants associated with the sediment. This high efficiency observed by McCuen contradicts the evaluation of detention-retention pond performance by Driscoll (1988). Efficiencies of detention ponds for quality improvement and settleability of various pollutants were also extensively covered by Ellis (1985).

Hartigan (1989) compared the removal efficiencies of modified (extended) dry ponds and wet ponds. He reported that removal of total phosphorus in wet ponds is 2 to 3 times greater than in modified dry ponds (50–60% versus 20–30%) and 1.3 to 2 times greater for total nitrogen (30–40% versus 20–30%). For other pollutants, the average removal rates for wet detention basins and extended dry detention basins were very similar: 80–90% for all the dissolved solids, 70–80% for lead, 40–50% for zinc, and 20–40% for BOD and COD. Negative removal efficiencies caused by resuspension of sediment during runoff events were observed by Dally and Lettenmaier (1984). Similarly, Mulhern and Steele (1988) measured negative removals for phosphorus in wet ponds during base-flow conditions.

A small lake in the Chicago area (a wet pond type) removed 91–95% of suspended sediment and 76–94% of copper, iron, lead, and zinc. The accumulated sediment in the lake was progressively reducing the available storage space and interfered with the aquatic habitat (Striegl, 1987). Nix and Tsay (1988) conducted an experiment in which storm-water flows were partially diverted at various levels into an off-line detention basin. The experiment showed that off-line storage basins are an effective means of controlling peaks and pollution by urban runoff. DiToro and Small (1979), Ferrara and Hildick-Smith (1982), Kuo and Ni (1984), Nix, Heaney, and Huber (1988), and Loganathan, Delleur, and Segarra (1985) presented mathematical models used for estimating the trap efficiencies of detention basins.

The "ideal solids settling" models are inappropriate if removal of pollutants that are partially dissolved is required. Such is the case of nutrients (nitrogen and phosphorus), which exist in urban runoff in both dissolved and particulate forms. For such designs Hartigan (1989) recommends using "a small lake eutrophication" model (for example, a model by Walker (1987)). The solids settling model is appropriate for simulation of removal efficiency for total suspended solids, a majority of toxic metals and organic toxic compounds that have a strong affinity for adsorption on suspended solids.

Typical Sizes of Ponds Combining Storage and Treatment

Hartigan (1989) pointed out that the size of wet detention ponds (which are apparently more efficient for removing nutrients) is 2 to 5 times the storage area of an extended dry pond. The storage volume of wet detention ponds is based on an average hydraulic detention time (typically two weeks), while the dry pond is designed to capture small rains and a first flush of larger rainfalls, as delineated in the previous section. Wet detention ponds are therefore recommended and cost-effective primarily for nutrient control. As an example, based on a typical average 2-week detention period for wet detention ponds and first flush capture for extended dry detention ponds located in Northern Virginia (Washington, D.C., metropolitan area), the sizes of the ponds are given in Table 10.8 (Hartigan, 1989). The depth of wet ponds should range between 1 and 3 meters. It is important that the side slope of the basin be very mild (5 to 10 horizontal to 1 vertical) to minimize the danger of drowning.

TABLE 10.8 Sizes of Wet and Dry Detention Ponds in Northern Virginia

Land Use		Percent of Imperviousness	
		Wet Detention (cm) ^a	Extended Dry Detention (cm) ^a
Low-density residential	20	1.8	0.25
Medium-density residential	50	2.5	1.0
Industrial/office	70	3.0	1.25
Commercial	80-90	3.3	2.0
Forest/park/undeveloped	0	1.25	0

^aOne cm of captured runoff for a watershed is equivalent to 100 m³ of pond volume per one hectare of the basin area.

Dual Use of Detention Basins for Flood and Quality Control

From the foregoing discussion it follows that the design of storage facilities for flood and quality controls use different objectives and design criteria. Hence, facilities designed solely for flood control using statistically rare design storms may be ineffective for quality control. It is, however, possible to retrofit existing flood-control storage facilities, for example, by installing an additional small orifice and implementing a dual-level control strategy. This was shown in Figure 10.25.

Storage Basins for CSO Control

For CSO control, retention basins are often used to catch the first, most polluted portion of the overflow (the first flush). Storage facilities that have been employed for this purpose can be either *in-line* or *off-line* (Fig. 10.30) and include ponds or surface basins, underground tunnels, excess sewer storage, and flexible or collapsible tanks located in the receiving body of water. The in-line storage basins do not need pumps and the outflow is controlled by restricting the outlet. From off-line basins wastewater is pumped at a rate determined by the capacity of the next treatment facility for disposing of the stored overflows.

Off-line storage facilities are dry between rainfall events and typically collect the “first flush,” more polluted portion of the overflow during large rainfalls or the entire overflow during smaller rains. The overflow occurs when the downstream capacity of the interceptor is exceeded. The flow is regulated in the flow divider (FD). After the basin is filled during longer rainfalls, the safety overflow of the basin (BO), located upstream or on the inflow is turned on and the rest of the overflow is then discharged into the adjacent receiving body of water (disinfection of the overflow may be required). Such basins are advantageous if the sewer system is not overloaded and the time of concentration is less than 15 minutes.

In-line flowthrough basins can have an additional clarifying overflow weir (OW) over which, after the basin is full, the mechanically clarified mixture of runoff and sewage is discharged into the receiving water. When incoming flow exceeds a predetermined critical flow (by regulating agencies and/or by design), the excess flow can be diverted in an upstream flow divider (BO).

Most of the sewers in a typical combined sewer system are designed with a capacity to handle 5- to 10-year storms; however, interceptors that carry water toward the treatment plant have a capacity of 4 to 6 times the

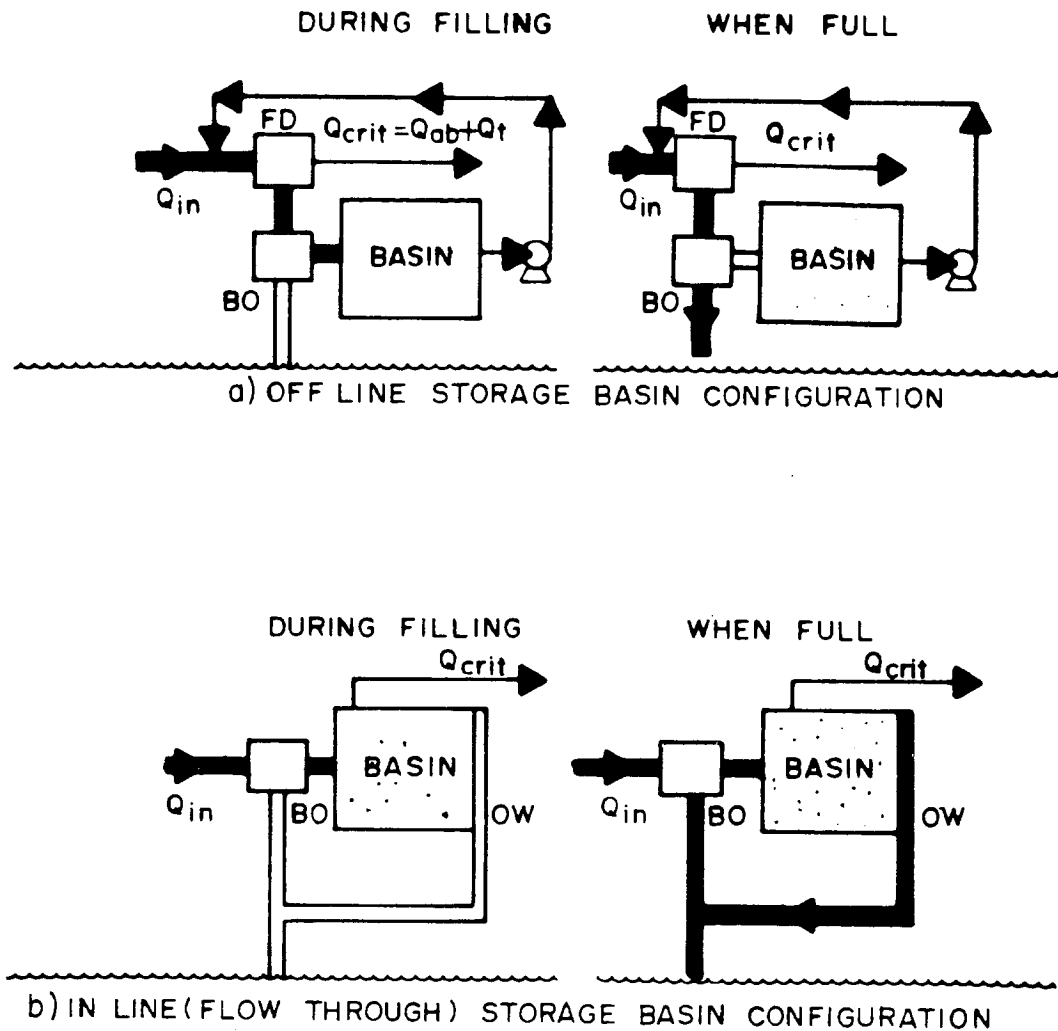


FIGURE 10.30. In-line and off-line storage basins for first flush control of CSOs. (After Novotny et al., 1989.)

maximal dry-weather flow. Hence sewers may have some unused capacity that can be used as in-line or in-pipe storage.

Ponds and surface storage are used to collect and store primarily "cleaner" and less malodorous storm water before it enters the collection system, as described in the preceding section. Once the runoff enters the combined sewer system and mixes with sewage, open air storage may not be recommended unless sufficient distance from inhabited areas is available for locating the storage basins or the basins are aerated and a part of the treatment plant. The storage basin must be surrounded by fences. Covered or underground storage basins are more appropriate but, consequently, far more expensive than open surface storage basins.

When geological underground formations are composed of soft but



FIGURE 10.31. Deep tunnel pumping station in Chicago, Illinois. (Photo: V. Novotny.)

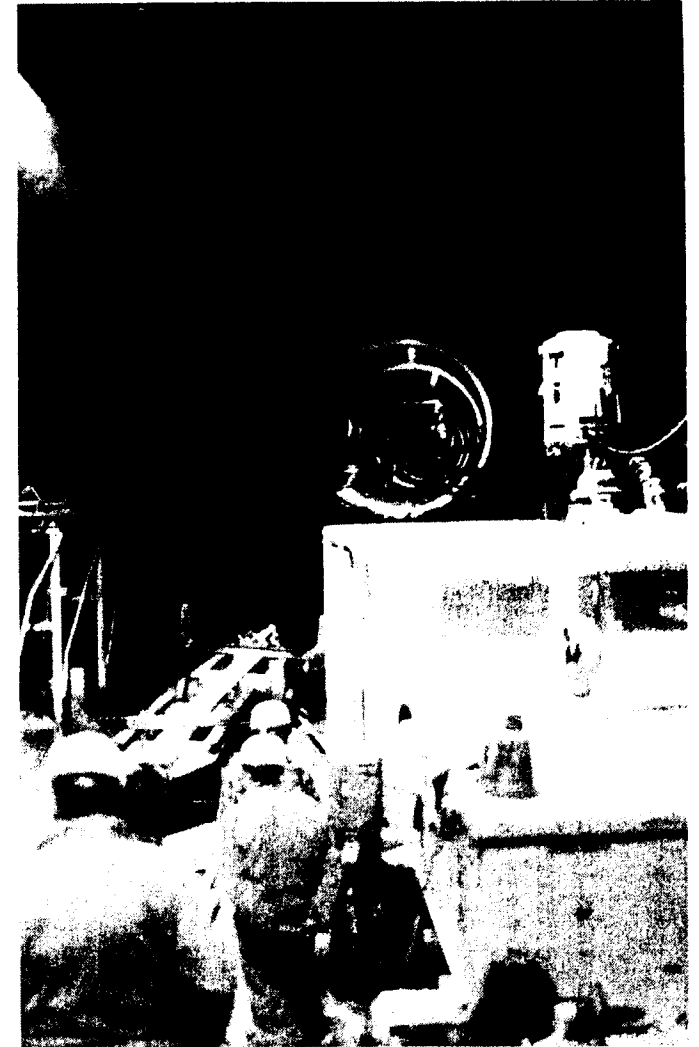


FIGURE 10.32. Deep tunnel construction in Milwaukee, Wisconsin. (Photo: V. Novotny.)

solid rocks (such as limestone or dolomite), the drilling of underground storage tunnels and caverns may be economical. However, cavernous (Karst) limestone formations should not be considered for underground storage of CSOs. A system of underground tunnels for storage of CSOs has been implemented in Chicago, Illinois. Located 100 meters below the surface, the TARP (Tunnel and Reservoir Project) system, when fully completed, will have 190 km of 10-m-diameter underground tunnels for storage of CSO. The CSO can be subsequently pumped to one of the largest treatment plants in the world (Fig. 10.31). An almost identical but smaller tunnel system is in Milwaukee, Wisconsin (Fig. 10.32), and similar systems are now being considered in other U.S. cities (Cincinnati, Ohio; Boston, Massachusetts; Cleveland, Ohio). Instead of conventional blasting, these tunnels have been drilled by a special giant drilling machine and grouted for minimization of clean ground-water inflows. Up to 30 meters of tunnel can be drilled in one day by the machine.

Tunnels in fractured rocks or purely consolidated geological materials may require lining by concrete, which will escalate the cost. When constructing the so-called "deep tunnel" in Milwaukee, soft rock formations were encountered, which resulted in ground water flooding the construction site (the tunnel was 100 meters below the ground-water table in most locations, resulting in high static pressures) and large remedial expense. Pockets of underground methane are also a problem, as they were in Milwaukee and Chicago, and all mining safety precautions must be taken to prevent explosions.

Sizing of First Flush CSO Control Retention Basins

In a typical routine application, the retention basin is designed to capture the more polluted, first flush portion of the combined sewer overflow. Thousands of such first flush capture basins have been installed in Germany and other parts of Europe. U.S. CSO control regulations require larger capture volume, and the captured flow must be subsequently treated in the main or satellite treatment plant.

A simple method of determining the size of the capture basins was developed in Germany (Novotny et al., 1989). In this method first calculate the specific capacity, r_{ab} , of the treatment plant or of the downstream interceptor

$$r_{ab} = \frac{Q_{ds}}{DC \times A_{imp}} \text{ (l/sec-ha)} \quad (10.11)$$

where

Q_{ds} = capacity of the flow that is allowed downstream from the storage basin, l/sec

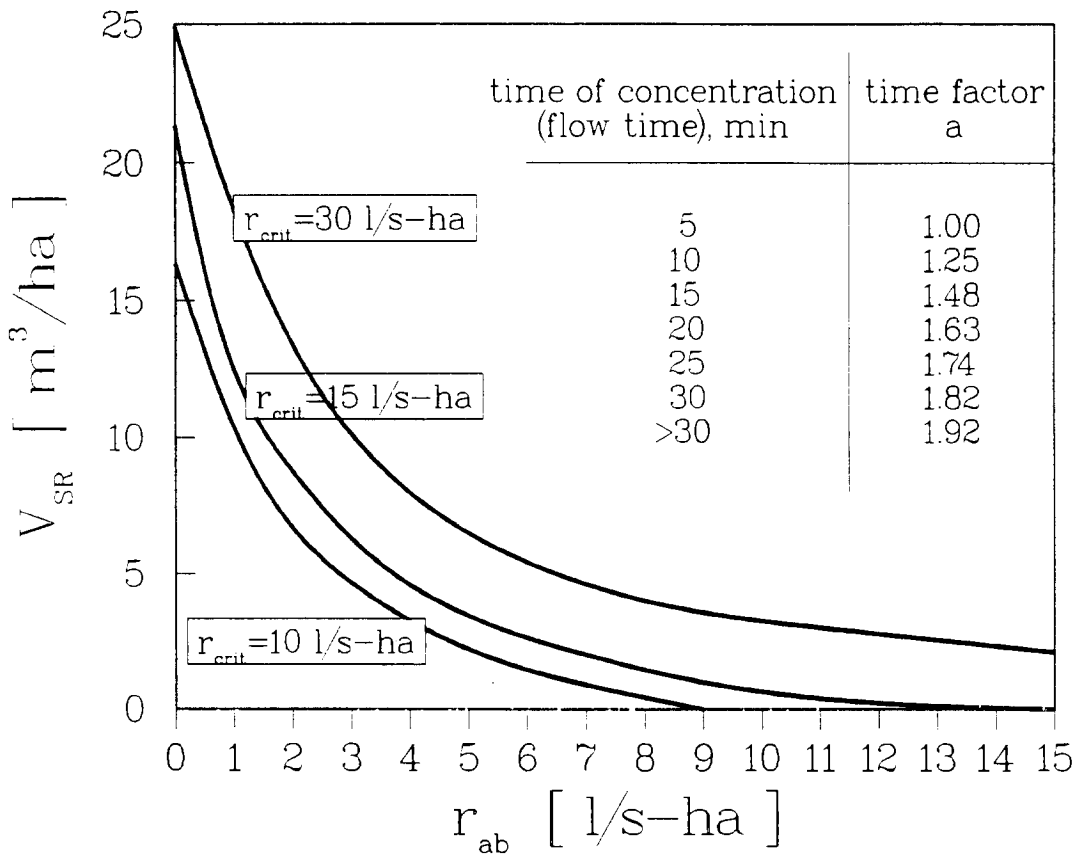


FIGURE 10.33. Dimensioning diagram for storage basins for first flush control of CSOs. (After Novotny et al., 1989.)

DC = fraction of directly connected impervious area of the drainage basin

A_{imp} = impervious area of the drainage basin

If r_{crit} is the minimum rainfall intensity at which overflow is allowed in l/sec-ha (to convert from mm/hr to l/sec-ha, divide by 0.36), then the specific detention volume V_{SR} can be obtained from Figure 10.33. The volume of the first flush retention basin is then obtained from

$$V = V_{SR}a(DC)A_{imp} \tag{10.12}$$

where the coefficient, a , which is related to the time of concentration, is read from the table in Figure 10.33. Dimensioning of the first flush basin is shown in Example 10.7. This design provides 10 to 30 minutes detention in the basin at the critical flow $Q_{crit} = r_{crit} \times DC \times A_{imp}$. As shown on Figure 10.24 taken from Roesner (1988), r_{crit} can be related to the frequency of overflows.

Example 10.7: First Flush In-Line or Off-Line CSO Control Storage Basin

Estimate the size of the CSO control retention basin that would capture overflow resulting from rainfalls with an intensity of up to 10 mm/hr. The drainage basin of 200 ha is 60% impervious, the directly connected impervious area is about 65% of the impervious surface, the time of concentration (overland and sewer) is 20 minutes. Downstream interceptor capacity is $0.3 \text{ m}^3/\text{sec} = 300 \text{ l/sec}$.

Solution The impervious area $A_{\text{imp}} = (60\%/100) \times 200 \text{ (ha)} = 120 \text{ ha}$. Calculate the specific allowable flow, r_{ds}

$$r_{ds} = \frac{300}{0.65 \times 120} = 3.84 \text{ l/sec-ha}$$

Calculate the critical rainfall intensity at which overflow is permissible

$$r_{\text{crit}} = 10 [\text{mm/hr}] / 0.36 [(\text{mm/hr}) / (\text{l/sec-ha})] = 27.78 \text{ l/sec-ha}$$

By interpolation in Figure 10.33 the specific volume is obtained as $V_{\text{SR}} = 8 \text{ m}^3/\text{ha}$. The time factor for the concentration of 20 minutes is $a = 1.63$. Then the basin volume becomes

$$V = V_{\text{SR}} a D C A_{\text{imp}} = 8 (\text{m}^3/\text{ha}) \times 1.63 \times 0.65 \times 120 = 1017.12 \text{ m}^3$$

Estimate the detention time at the critical rainfall rate

$$\begin{aligned} Q_{\text{crit}} &= r_{\text{crit}} \times DC \times A_{\text{imp}} = 10 \times 0.65 \times 120 = 780 \text{ l/sec} \\ &= 0.78 \text{ m}^3/\text{sec} \end{aligned}$$

$$\text{Detention time } t = V / Q_{\text{crit}} = 1017.12 / 0.78 = 1304 \text{ sec} / 60 = 21.7 \text{ min.}$$

Flow Balancing Retention Basins

The flow-balancing method (FBM) was developed in Sweden by Karl Dunkers (Field, Dunkers, and Forndran, 1990; Field, 1990). The features that make the flow-balancing method attractive are (1) this type of facility uses a portion of an existing receiving body of water, thus reducing the need for land space for siting, and (2) the materials used offer low structurally intensive and low-cost alternatives compared with more costly conventional construction on land.

Typically, an FBM detention facility is built of pontoons and curtains that form a series of interconnected rectangular tanks (Fig. 10.34). The

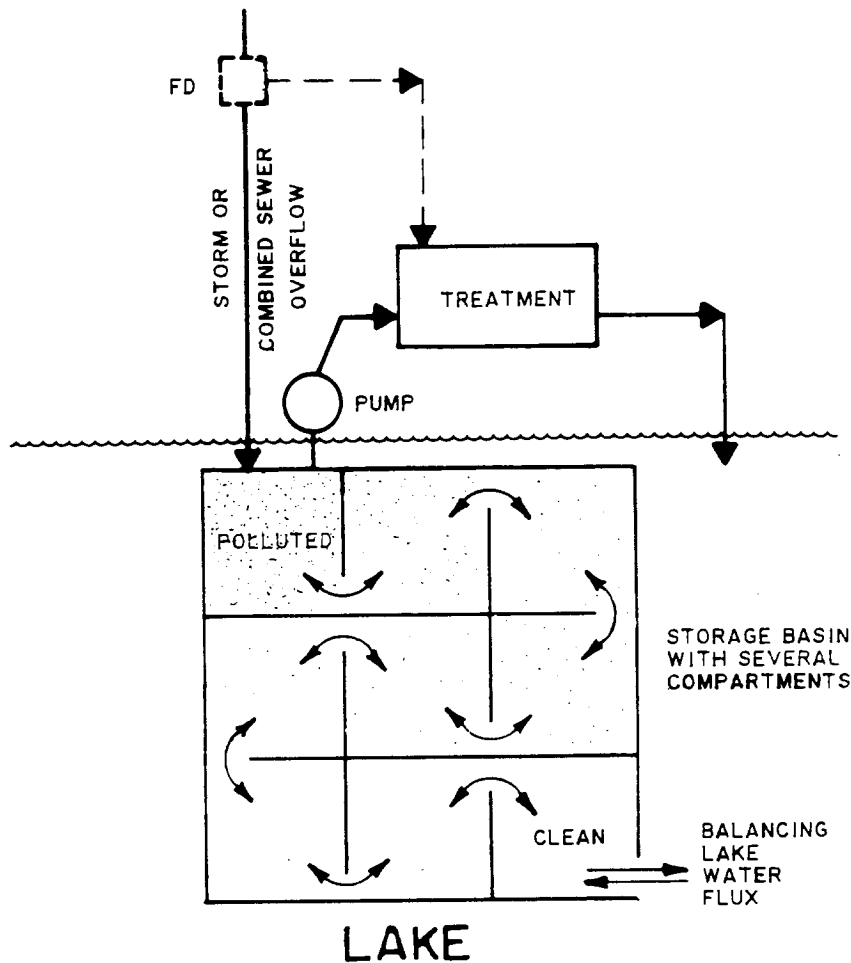


FIGURE 10.34. Dunkers' flow-balancing overflow storage basin.

arrangement of the tanks in series simulates so-called “plug flow” conditions, which minimizes mixing between the compartments and short-cutting. Typically, nine tank compartments suffice. If the facility is located in a body of water with brackish or saline waters, an interface between the less saline combined sewer overflow or storm-water will be formed in the tanks. For example, in a saline water environment, because CSO water is lighter than the saline water from the receiving body of water, the CSO will begin to fill the tank from the top.

Before the first overflow arrives after the detention facility is installed the entire volume of all tanks is filled with water from the receiving body of water (clean water). During the overflow event, polluted water from the overflow is directed into the first compartment. Polluted water will then begin to push clean water from the compartments, beginning with the first compartment, into which the overflow is directed. During storms that exceed the design storm volume, the process continues until the

entire facility is filled. Only at this point can polluted water enter the receiving body of water. For all other smaller storms, the polluted water should be contained by the FBM facility.

After the CSO or storm-water discharge event, a floating pump located in the first receiving compartment starts to pump the polluted water back into the sewer interceptor for conveyance to the treatment plant. As clean water enters the facility from the opposite side, it pushes the polluted water back toward the first compartment. After clean water refills the system, pumping stops.

Solids and floatables that accumulate in the facility must periodically be pumped out. A majority of the solids typically settle out in the first two compartments.

The same methods are used to determine the size of the facility as are used for detention facilities located on land. An experimental-demonstration FBM detention facility was built in a tributary of Jamaica Bay in Brooklyn, New York (Forndran, Field, and Dunkers, 1989; Field, Dunkers, and Forndran, 1990).

Wetlands

Oberts (1982), studied 17 urban and suburban basins in the Minneapolis-St. Paul area and found that wetlands can have a significant impact on pollution attenuation of urban watersheds. His research found a significant inverse relation between the pollutant loads and the percent of the watershed area that is wetland.

Incorporation of wetlands into a comprehensive storm-water management system achieves several objectives: reduced operation/maintenance, aesthetic buffering, development amenities, wetland preservation, and restoration, and, hence, enhanced wetland value (Livingston and Roesner, 1991). Wetlands combine both sedimentation and biological utilization effects to remove pollutants from runoff. The largest pollutant reduction can be achieved during the "active" wetland growing season, that is, May to September in northern climatic conditions. During the dormant (winter) condition, wetlands can become a source of pollution that is leached from dead vegetation. In southern (Florida) climatic conditions wetlands efficiency remains more-or-less constant throughout the year.

Wetlands can be used in combination with other treatment, such as detention ponds (Wotzka and Oberts, 1988; Schueler, Kumble, and Heraty, 1991). A more detailed discussion and presentation of the design of wetlands for quality control of runoff and in-stream water quality restoration, as well as the impact of diffuse-pollution loads on wetlands, is presented in Chapter 15.

Both natural and man-made wetlands have been used for runoff pollution control; however, natural wetlands are considered to be receiving waters and are subjected to effluent restrictions. On the other hand, man-made wetlands are considered to be treatment facilities and the standards mostly apply to the outlet from the wetland (Hey et al., 1982; Stockdale and Horner, 1987; Linker, 1989; Hammer, 1989). Therefore, currently only man-made or restored wetlands can be used for the treatment of runoff and CSOs.

However, state-of-the-art use of wetlands for urban storm-water and CSO control is still evolving, and many issues have not yet been resolved or answered by long-term research (Livingston and Roesner, 1991). For example, how long can a wetland continue to accumulate pollutants without impairing its biological functions? How long can a wetland remove pollutants effectively? What type of maintenance (harvesting, sediment banking, and others) should be planned for a given wetland area?

TREATMENT

In areas that are highly developed and have existing combined or separate sewer systems, a large-scale structural treatment method may be the only feasible way to eliminate pollutants from storm sewer discharges and CSOs. With the exception of vortex separators, use of treatment unit processes requires storage, mostly underground (tunnels). Most large-scale treatment facilities used for the control of pollution by CSOs and urban runoff are a part of the dry-weather treatment facilities that are enlarged to accommodate the stored wet-weather flows. However, the Emscher River treatment plant in the Ruhr industrial district of Germany treats the entire flow of a river that is heavily polluted by municipal and industrial point and nonpoint sources before its confluence with the Rhine River.

Wetland systems and modified dry and wet ponds provide inexpensive treatment of urban runoff, and have been used extensively and successfully. Traditional treatment methods that are used for point sources are difficult to implement for non-CSO urban nonpoint sources due to their variable and intermittent nature. This is especially true for the micro-organism-dependent (biological) treatment processes, although a report by Agnew et al. (1975) describes a relatively successful combined treatment of municipal wastewaters and stored overflows.

There are three types of treatment used for treating wastewater discharges, including those occurring during wet-weather conditions, namely, physical, physical-chemical, and biological. Some systems use a

TABLE 10.9 Efficiency of Various Storm-Water and CSO Treatment Processes

Process	Efficiency (%)				
	Suspended Solids	BOD ₅	COD	Total P	TKN ^a
<i>Physical-chemical</i>					
Sedimentation	20-60	50	34	20	
Without chemicals					38
With chemicals	68	68	45		
Vortex separation	40-60	25-60	50-60		
Screening					
Microstrainers	50-95	10-50	35	20	30
Rotary screens	20-35	1-30	15	12	10
Sand-peat filters ^b	90	90	NA	70	50
<i>Biological^c</i>					
Contact stabilization	75-95	70-90		50	50
Biodiscs	40-80	40-80	33		
Oxidation ponds	20-57	10-17		22-40	57
Aerated lagoons	92	91			
Facultative lagoons	50	50-90			

Source: After Lager et al. (1977).

^aTotal Kjeldahl nitrogen.

^bAfter Galli (1990), the peat-sand filter are similar to biological anaerobic-aerobic slow filters. They are applicable for treatment of urban runoff.

^cBiological treatment is feasible only for CSOs.

combination of two or all three types. For example, even simple systems such as wetlands provide a multitude of treatment processes to separate pollutants (see Chapter 14 for a description of pollutant separation and removal processes in wetlands and for wetland applications). The treatment efficiency of some processes is given in Table 10.9. Design parameters for treatment units are given in Novotny et al. (1989) and Field (1990). Field's summary also includes pilot and prototype performance and cost information.

Physical-chemical treatment processes have shown some promise in overcoming the shock loadings (Field, 1986). Chemical additives used to enhance separation of particles from liquid include chemical coagulants, such as lime, alum, ferric chloride, and various polyelectrolytes. During the process of chemical clarification, which consists of chemical addition, flocculation, and particle separation (clarification and/or filtration), the flocs are formed from finer particles, mainly by reduction of surface charge and/or by adsorption on surfaces provided by the added chemicals. Chemical addition is also effective for removal of phosphorus, toxic

metals, and some organic colloids. Handling and disposal of produced sludge is a costly problem.

The physical-chemical processes that have been successfully demonstrated and/or installed include fine-mesh screening, fine-mesh screening/high-rate infiltration, sedimentation, sand and peat-sand filtration, fine mesh-screening/dissolved-air flotation, and swirl separator. A pilot plan study (Innerfeld et al., 1979) using high-rate filtration of CSOs was conducted in New York.

Sand filters represent a new application of an old technology for treatment of urban storm water (Schueler, Kumble, and Heraty, 1992). These filters can be used effectively for the control of runoff from small sites, or the treatment of runoff prior to entering another storm-water management structure (for example, before a vegetative filter), or as a retrofit strategy in an urban storm-water system. The design includes both sedimentation and filtration components (Fig. 10.35). Sedimentation removes the larger particles (grit and sand), and filtration removes silt and clay-size particles. The drainage area should be less than 2 ha of impervious surface, and the sedimentation chamber volume should be about 50 m³ per hectare of connected impervious area. The sedimentation chamber is followed by a filtration chamber that has the same volume.

Since sand is inert and has little or no adsorbing capacity, a dissolved fraction of the dissolved priority pollutants is not removed (unless an organic-microbiological population develops in the filter). However the top layer of the filter, which contains the organic matter, has to be periodically removed and replaced by clean sand.

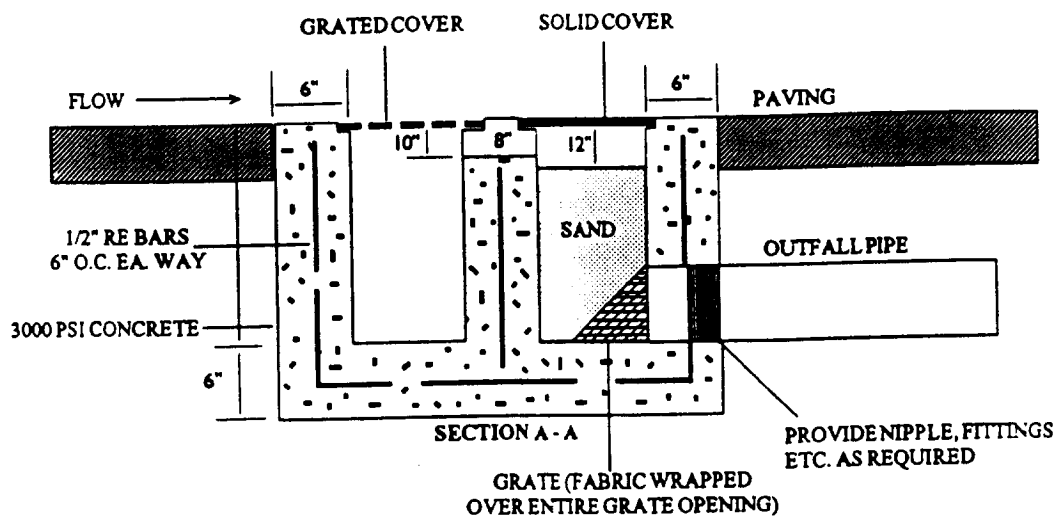


FIGURE 10.35. Sand filter used for storm-water pollution control. (From Schueler, Kumble, and Heraty, 1992.)

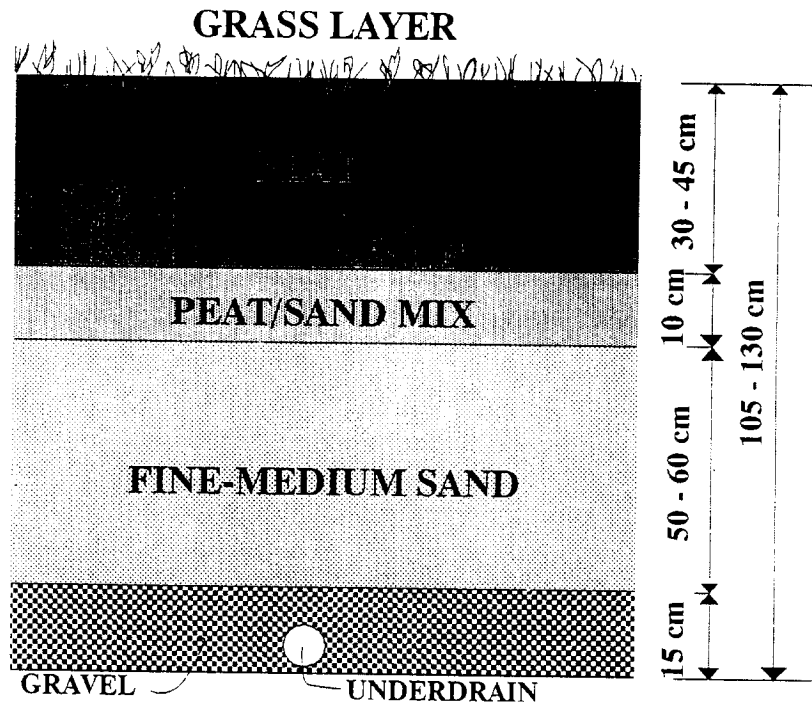


FIGURE 10.36. Peat-sand filter for storm-water treatment. (After Galli, 1990.)

Slow-rate filters, in which development of mostly anaerobic microbiological layer occurs, remove priority pollutants by microbiological action and by adsorption on organic particulates. Carlo et al. (1992) showed that a sand filter, when operated at higher loading rates, did not remove selenium, while when operated as an anaerobic slow-rate filter, its removal efficiencies for selenium were 74–97%. Selenium was retained in the organic layer of the filter.

Enhanced (peat-sand) filters utilize layers of peat, limestone, and/or top soil, and may also have a grass cover top (Fig. 10.36). Sand filters can be used in areas with thin soils, high evaporation rates, low soil infiltration, and limited space. Sand filters remove mostly particulate pollutants. To minimize clogging both sand and enhanced filters should be preceded by a solids-removing unit, such as a pond or a filter strip.

Peat-sand filters provide high phosphorus, BOD, and nitrogen removals, in addition to the removal of solids. For sizing the filters, Galli (1990) recommends that the filter area should be about 0.5% of the contributing watershed area, and the annual hydraulic loading should equal 75 m/year or less. As with most biological filtration systems, the peat-sand filter works best during the growing season, since a part of the nutrient load is taken up by grass. Also the filter should remain aerobic. The peat has a very high affinity for adsorbing and removing toxic com-

pounds (see Chapter 14 for a discussion), hence peat-containing filters are effective for removing priority pollutants.

Biological treatment processes are effective for flows, such as CSOs, where organics are concentrated. Storm-water discharges have low BOD concentrations (on the order of biologically treated effluents), and are not amenable to biological treatment. Biological treatment processes do not perform satisfactorily or not at all when highly irregular flows and quality are treated. Because it is very difficult to keep the biota alive between events, conventional activated sludge can be used for treatment of CSO only in combination with dry-weather sewage treatment (Milwaukee and Chicago) and storage for CSOs to equalize the transient loads.

Other biological treatment processes that have been tested and/or implemented for treatment to CSOs include contact stabilization, attached media units (trickling filters and rotating biological contractors), and lagoons. The peat-sand filter described earlier also includes biological removal and uptake of biodegradable organics and nutrients.

Due to the permanent water content in lagoons, it is less difficult to keep active biota alive in them between events. Many storage ponds can be, and in fact are, used as biological lagoons. Lagoons can be categorized according to type and the amount of aeration they receive. Lagoons relying mostly on atmospheric aeration and photosynthesis for their oxygen supply are called *oxidation ponds*. Aerated lagoons rely on mechanical floating or mounted aerators for their oxygen supply. These more complicated unit operations are less suitable for CSO control.

Oxidation ponds are too shallow (0.2 to 0.5 m plus additional volume for sludge storage) to allow for light penetration. Their treatment efficiency depends on temperature and climatic conditions, and detention time is between 3 days (warmer climatic conditions) and 20 days (colder zones of the United States, Canada, and central and northern Europe). Lagoons and ponds can be used in combination with pretreatment or posttreatment. Typical inexpensive units include grass filters and/or wetlands and settling ponds. Without posttreatment the effect of lagoons on organic (BOD, COD) loads may be negative due to possible high algal growths.

Disinfection

Disinfection has been used primarily to control CSOs. Disinfectant and contact times are usually long, and research has concentrated on the use of high-rate applications by static and mechanical mixing, higher disinfection dosages, and more rapid disinfectants, such as chlorine dioxide, ozone, and ultraviolet light (Field, 1986). Adding chlorine to storm water and CSOs is questionable in today's systems. Chlorine, in

addition to being toxic to bacteria and viruses, is also a strong oxidizing agent for other organics and for some reduced chemical components, such as iron, manganese, sulphides, and ammonia, that may be present in water. Since these oxidation processes must be accomplished before chlorine acts as a disinfectant, from 20 to 40 mg of Cl_2/l may be required to kill bacteria. Some chlorinated organic matter of natural (for example, humates) and man-made origin found in urban runoff and wastewater are carcinogenic and subject to regulation.

EFFICIENCY OF BEST MANAGEMENT PRACTICES FOR CONTROL OF PRIORITY POLLUTANTS

Dissolved versus Total Concentration of Priority Pollutants in Runoff

In Chapter 6 and throughout the book it was pointed out that priority pollutants can exist in the dissolved and adsorbed-precipitate phases. The proportion between these two phases is basically given by the solubility of the pollutant, which for nonpolar organic chemicals is given by the octanol partitioning coefficient, K_{ow} . A similar partitioning concept can also be applied to metals and some polar compounds, for which the partitioning coefficient would be pH dependent.

Few investigators have measured both soluble and particulate fractions of priority pollutants in urban runoff and other similar discharges. Some of the findings were presented in Chapter 8. Whipple and Hunter (1980) found that petroleum hydrocarbons in urban runoff are about 4.5% soluble. The flow-weighted total hydrocarbon concentrations measured in North Philadelphia, Pennsylvania, and Trenton, New Jersey, urban storm discharges ranged from 2.16 to 5.9 mg/l. The following hydrocarbons were detected: Di-benzo-thiopene and PAHs, such as naphthalene, anthracene, pyrenem fluoranthene, chrysene, and benzo(a)pyrene.

Herrmann and Kari (1990) measured particle-size distribution of PAHs and PCBs in urban runoff, and found that most of these contaminants can be found in particle-size fractions between 2 to 63 μm (silt size), while the clay fraction ($d < 2 \mu\text{m}$) was less contaminated. It can be suspected that the organic particulates to which these contaminants are most attracted are in these size fractions. The authors also found that the priority pollutants with high partition coefficients ($K_{ow} > 10^5 \text{l/kg}$) exhibited a more pronounced "first flush effect" than those that are more dissolved. The first flush phenomenon indicates that the load of the pollutants is limited by the "supply" on the street surface. Most of the PAHs are associated with street dust (see also Herrmann, 1984).

Morrison et al. (1984) found that in urban runoff zinc and cadmium exhibited preference for the dissolved phase, whereas lead predominated in the suspended phase. Similarly, Hewitt and Rashed (1992) found that the particulate phase ($>0.45\ \mu\text{m}$) in the contents of highway runoff in England contained more than 90% lead, 70% copper, and 50% cadmium. Data on urban runoff quality collected by Pitt and Baron (1989) showed that most of the organics and metals were associated with non-filterable (particulate) sample fractions. However, some organic pollutants and zinc had significant filterable fractions. The organic pollutants with high filterable portions included bis(2-chloroethyl) ether (19–50% filterable), 1,3-dichlorobenzene (31–100% filterable), hexachloroethane (93%), naphthalene (2–79%), anthracene (25%), benzyl butyl phthalate (33%), fluoranthene (5–100%), and pyrene (15–100%). These compounds have an octanol partitioning coefficient between 10^3 and 10^5 .

Most of the reports and manuals on best management practices report the removal efficiencies for a few conventional pollutants, such as suspended sediment. However, such information may be misleading when attempted to estimate the efficiencies of BMPs for priority pollutants. The pollutants exist in water–sediment systems either in a dissolved–dissociated state in water or adsorbed onto fine sediments and particulate organic matter. Sand particles do not adsorb most priority pollutants, and BMPs that effectively remove coarser particles only may be ineffective for some removal of priority pollutants. A swirl concentrator is an example of a (1980) control unit that removes mostly coarser particles from runoff, but may be ineffective for the removal of priority pollutants.

The proportion between the dissolved and adsorbed portions of a storm priority pollutant in water depends on its adsorption characteristics—typically its octanol partitioning number—and the concentration of fine particles—primarily particulate organic matter. Figure 10.37 shows a simplified proportion of an adsorbed fraction versus the total (particulate PAHs and dissolved) content of a priority pollutant related to its octanol partitioning characteristics and content of suspended organic particles indicated as volatile suspended solids.

Particulate organic carbon (POC) is the best surrogate parameter for estimating the removal efficiencies of nonionic toxic chemicals (DDT, PCBs, PAHs, and a large number of organic pesticides) with a higher octanol partitioning coefficient ($K_{ow} > 10^5\ \text{l/kg}$). The POC measurements are relatively simple and should be made a routine part of chemical analyses of waste loads from point and nonpoint urban sources. In the absence of the POC measurements, volatile suspended solids (VSS) parameters may be substituted. The efficiency of various abatement

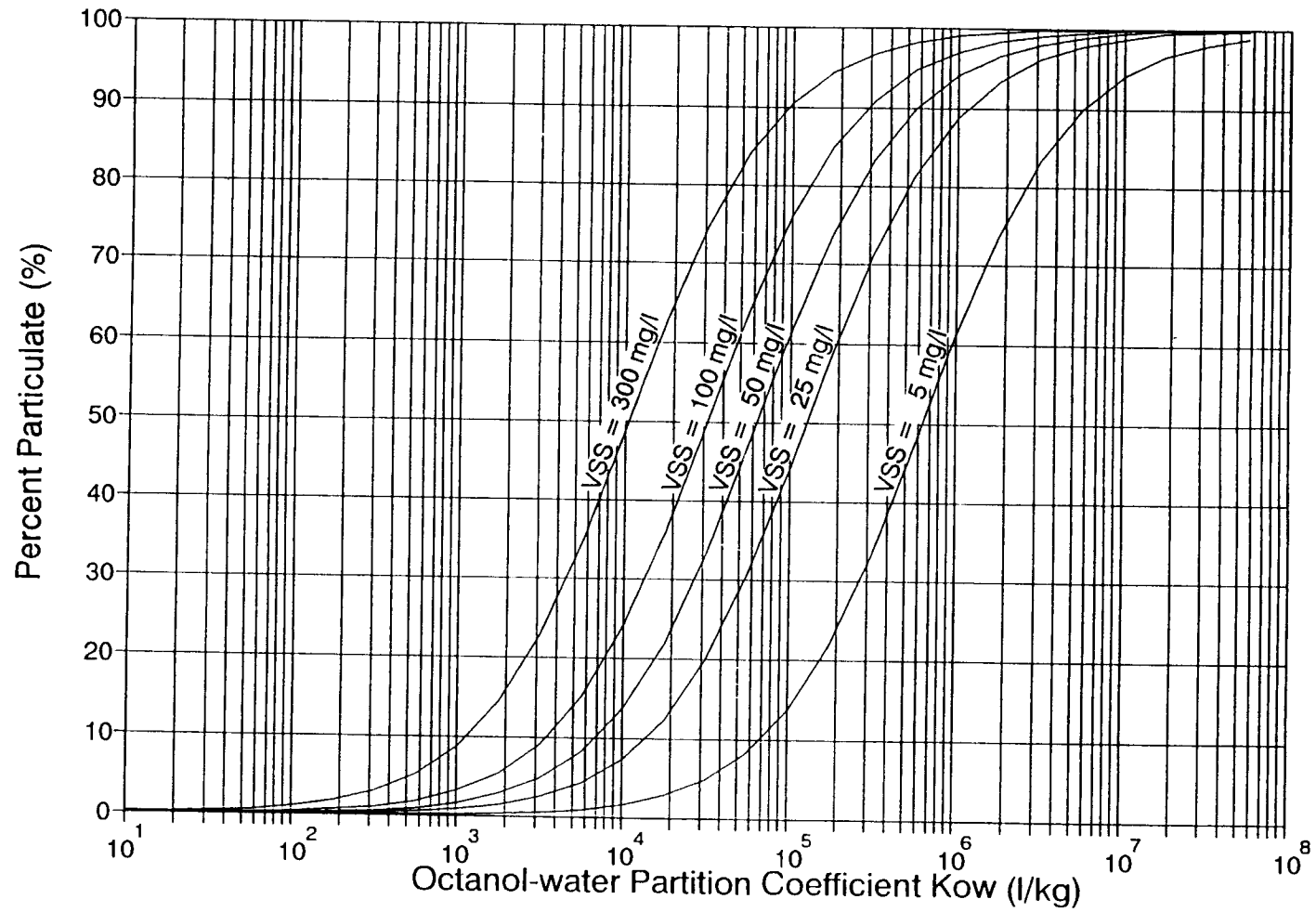


FIGURE 10.37. Relationship of dissolved and total concentrations of organic priority pollutants related to the octanol partitioning number and volatile suspended solids content of runoff.

practices for the removal of nonionic organic priority pollutants may be correlated to the removal or fate of organic particulate matter, although the fate models may not always be simple (for example, for processes that produce organic matter such as eutrophic ponds). Compounds with a low octanol partitioning cannot be removed with particulates, and the removal efficiency is related to their biodegradability and volatilization.

Toxic Metals

When toxic metals are added to water, both from natural and man-made sources, they undergo complexation with ligands. Ligands are chemical constituents, both organic and inorganic, that combine with the metals in a chemical complex. From the basic physical chemistry it is known that metals precipitate because of changes in pH, when oxidized, or when they change their chemical composition. However, the process of complexation in natural waters is more complex.

Of the metal complexing mechanisms that were described in Chapter 6, iron and manganese oxyhydrates provide the strongest adsorption sites, followed by particulate organics and clays. The adsorption reactions are strongly pH dependent, ranging from zero adsorption in low pH to 100% adsorption in higher pH. The change from no adsorption to full adsorption is fairly sharp, usually over two pH units (Salomons and Förstner, 1984). However, at pH > 7 most of metals are complexed, while at pH 5 the concentrations of free metal ions increase dramatically. The methylated compounds are still strongly polar and in sediments behave similarly to free metal ions, that is, they are adsorbed by the sediments and ligands in the sediment.

The best surrogate parameter for estimating efficiency of runoff and waste pollution-control facilities for metals would be suspended sediment without its sand fractions and particulate organic carbon. The efficiency of removal of metals can be correlated to the efficiency of removing clay and silt fractions of the sediment and POC (VSS) parameters. However, only lead appears to be strongly associated with particulates. Other toxic metals are typically found both in dissolved and particulate phases.

Removal of Priority Pollutants or Toxicity by Best Management

Practices and Treatment

The objective of best management practices (BMPs) for control of priority pollutants has to be specified. There are two criteria by which the efficiency of BMPs can be judged; one is the removal of the total mass of a pollutant or pollutants, and the other is the removal of toxicity.

Typically, storm-water permits rely on two sets of standards. One involves numerical limits on concentrations of priority pollutants; the second requires testing for the toxicity of the runoff. The methods used for toxicity testing must conform to EPA-recommended guidelines for conducting static acute toxicity tests (see Chapter 12 for a discussion on toxicity).

The EPA has also published guidelines pertaining to the performance of toxicity reduction evaluation (TREs) for discharges from municipal and industrial wastewater treatment plants. It is certain that most of the major elements of TREs are also applicable to BMPs for storm-water discharges (Collins, Roller, and Walton, 1992), although very little information is available on the performance of various BMPs regarding toxicity reduction. According to these guidelines the efficiency of BMPs is tested using the cumulative acute toxicity measurements.

The partitioning theory of priority pollutants specifies that a majority of the priority pollutants in an aqueous environment exist as either dissolved or adsorbed precipitates and are associated with particulates. Numerous studies have proved that only dissolved pollutants are bio-available, and therefore, toxic. This is true of both metals, where the divalent ion Me^{++} has been identified as a toxic compound, and for organics. In a simplified way, Equation (6.8) provides the key to understanding the relation of toxicity removal to total pollutant removal in a BMP process designed primarily to remove solids. This is explained in the following example.

Example 10.8: Removal of Toxicity by Best Management Practices

Suppose that storm-water runoff contains 1 mg/l (1000 μ g/l) of a priority pollutant that has the octanol partitioning coefficient $K_{ow} = 10^5$ l/kg. The runoff also contains 300 mg/l (0.3 g/l) of suspended solids, of which 10% are organic carbon— $f_{OC} = 0.1$. A BMP facility provides 80% removal of total solids and 65% removal of OC.

Solution Calculate total and toxic (dissolved) concentrations of the priority pollutant before and after treatment.

BMP Influent From Equation (6.20a)

$$K_{OC} = 0.63K_{ow}/1000 = 0.63 \times 10^5/1000 = 63 \text{ l/g}$$

The partitioning coefficient is then (Eq. (6.19))

$$\Pi = K_{OC} \times (OC)/100 = 63 \times 10/100 = 6.3$$

The dissolved (toxic) priority pollutant concentration is then calculated from Equation (6.8) as

$$C_{d-in} = \frac{C_{Tin}}{1 + \Pi m_{ss}} = \frac{1000}{1 + 6.3 \times 0.3} = 340 \mu\text{g/l}$$

The adsorbed (nontoxic) fraction of the pollutant in the influent is then $1000 - 340 = 660 \mu\text{g/l}$.

BMP Effluent The BMP facility removed 85% of solids and 65% of OC. Hence the effluent solids concentrations is $(1 - 0.85) \times 300 = 45 \text{ mg/l}$, and the OC concentration is $(1 - 0.65) \times 0.1 \times 300 = 10.5 \text{ mg/l}$ and the organic carbon fraction $f_{OC} = 10.5/45 = 0.23$. Since the particulate fraction of the pollutants is associated with organic carbon, the same removal rate (65%) applies to the removal of adsorbed pollutant. The dissolved fraction is not removed by the facility. Hence after the treatment the total concentration of the pollutant in the effluent is

$$C_{T-ef} = (1 - 0.65) \times 660 + 340 = 571 \mu\text{g/l}$$

The facility removed $100 \times (1000 - 571)/1000 = 43\%$ of the total mass of the contaminant.

A new equilibrium is reached in the effluent between the dissolved (toxic) and adsorbed (unavailable) fractions of the contaminant. The new partition coefficient is

$$\Pi = K_{OC} \times f_{OC} = 63 \times 0.23 = 14.51/\text{g}$$

and the dissolved concentration is

$$C_{d-ef} = \frac{C_{T-ef}}{1 + \Pi m_{ss-ef}} = \frac{571}{1 + 14.5 \times 0.045} = 345 \mu\text{g/l}$$

which indicates that no reduction of toxicity was achieved by the facility.

This example illustrates the dilemma planners of BMPs for storm-water toxicity reduction are facing. Apparently, toxicity cannot be reduced by BMPs relying on settling only, so other mechanisms such as biodegradation, volatilization and adsorption on additional organic matter must be considered.

Conclusions on the Efficiencies of BMPs

A detailed evaluation of the efficiencies of BMPs for removing priority pollutants was prepared by Novotny et al. (1992), and a summary paper was published by Scholze, Novotny, and Schonter (1993). The U.S. EPA Risk Reduction Engineering Laboratory (RREL) in Cincinnati, Ohio, has compiled an extensive computerized "Treatability Data Base" of many chemicals, including all priority pollutants. The information contained in the data base was retrieved from the literature. The data base also considers a large number of treatment processes, such as lagoons (both aerobic and anaerobic), suspended and attached growth reactors (both aerobic and anaerobic), physical chemical coagulation-flocculation processes, dissolved air flotation, ion exchanges, filtration and ultrafiltration, reverse osmosis, and electrodialysis. Altogether 35 different treatment unit processes are considered. However, only a few can be considered as BMPs for the control of runoff pollution.

The following conclusions are pertinent when BMPs are proposed for control of priority pollutants.

1. Priority pollutants are a category of pollutants that in many aspects differ from such traditional pollutants as biodegradable organic matter and nutrients. Hence not all of the BMPs designed for the removal of traditional pollutants may be effective for the removal of priority pollutants.
2. Although a large number of pollutants have been classified as *priority*, the pollutants themselves and their treatability and treatment-removal process selection may be characterized by a few parameters, including octanol partitioning number for polar organics, biodegradability, and volatility.
3. Because most of the priority pollutants are effectively immobilized by organic matter, the most effective removal processes include organic matter in some form. The organic matter is provided in several effective forms, such as vegetation and its residues, peat and other wetland vegetation and its residues, or particulate organic solids contained in runoff.
4. BMPs designed to remove solids, such as settling ponds, sand filters, and microstrainers, can remove the mass of priority pollutants associated with particulates, but are not effective for removing toxicity that is attributed to the dissolved components of the priority pollutants.
5. Most effective BMPs are pond enhanced with wetlands, wetlands themselves, water hyacinth ponds, overland flow systems, and sand-

peat filters. Typically these BMPs should provide 80% to 99% removal of priority pollutants.

6. Some priority pollutants, such as toxic metals and several organics, are best immobilized in an anaerobic environment by binding to sulfides and other complexing ligands, or are best biodegraded in an anoxic or anaerobic environment. Wetlands, sand-peat filters, and facultative lagoons provide such a reducing environment.
7. Priority pollutants with high volatility can be removed in systems that provide a high degree of aeration and exposure to the atmosphere. Such systems include aerated ponds (followed by another unit to remove the remaining dissolved and particulate fractions of the pollutant), and overland (sheet flow) flow systems.
8. BMPs with moderate removal efficiencies include most ponds, grassed swales, sand filters, porous pavements, and microstrainers. Since such systems primarily remove solids, their efficiency for removal of pollutants with low octanol partitioning and toxicity reduction may be diminished.
9. BMPs using infiltration may be moderately effective for the removal of pollutants with a higher octanol partitioning number and for some metals, but these methods are unsuitable for the control of pollutants with a low octanol partitioning number and for some metals (for example, zinc) that have a high dissolved fraction in the runoff.
10. Street sweeping and similar surface control of pollutants may be ineffective, although they may be required for aesthetic enhancement and for maintaining the cleanliness of the area.
11. Due to the multiplicity of priority pollutants present in runoff, there is no single BMP that would be effective for all priority pollutants. Typically a combination of units is required. Proper and effective combinations of unit processes are still being researched.

REAL-TIME CONTROL OF CSOS AND POLLUTION LOAD TRADE-OFFS

An urban sewerage system, including all of its components (sewers, storage, main and satellite treatment, and disposal) is a time-variable-dynamic system. Input into the system—rainfall, sewage and industrial wastewater flows, infiltration—and the treatment processes themselves are highly variable, with ranges of parameters often over one order of magnitude. Process upsets, malfunctions, monitoring errors, labor force behaviors, and management communications are additional factors contributing to the variability of the processes and parameters.

Traditionally however, these systems have been designed and operated

using time-invariant (fixed), rather simple models and rules and criteria. For example, as specified previously, a combined sewer system typically has the design capacity of four to eight times the dry-weather flow contributions, which is also a fixed parameter. Use of storage facilities and treatment of overflows make the system more dynamic; however, the operation can still be based on static invariant control. Instead of the hydraulic capacity of the interceptor, which determines whether the overflow is initiated, the capacity of the storage basin controls the overflows.

It is apparent that an urban sewerage system has a number of so-called "bottlenecks" or control-decision points. These bottlenecks include the interceptor capacity, storage capacities, and the capacity of the treatment plant. If the flow or pollutant load exceeds the capacity of the bottleneck, an overflow results. In the treatment process there are a number of points where a bottleneck situation may occur. The most important control point is the secondary clarifier of the biological treatment units. If the hydraulic or solids loading of the final clarifiers is greater than its cap-

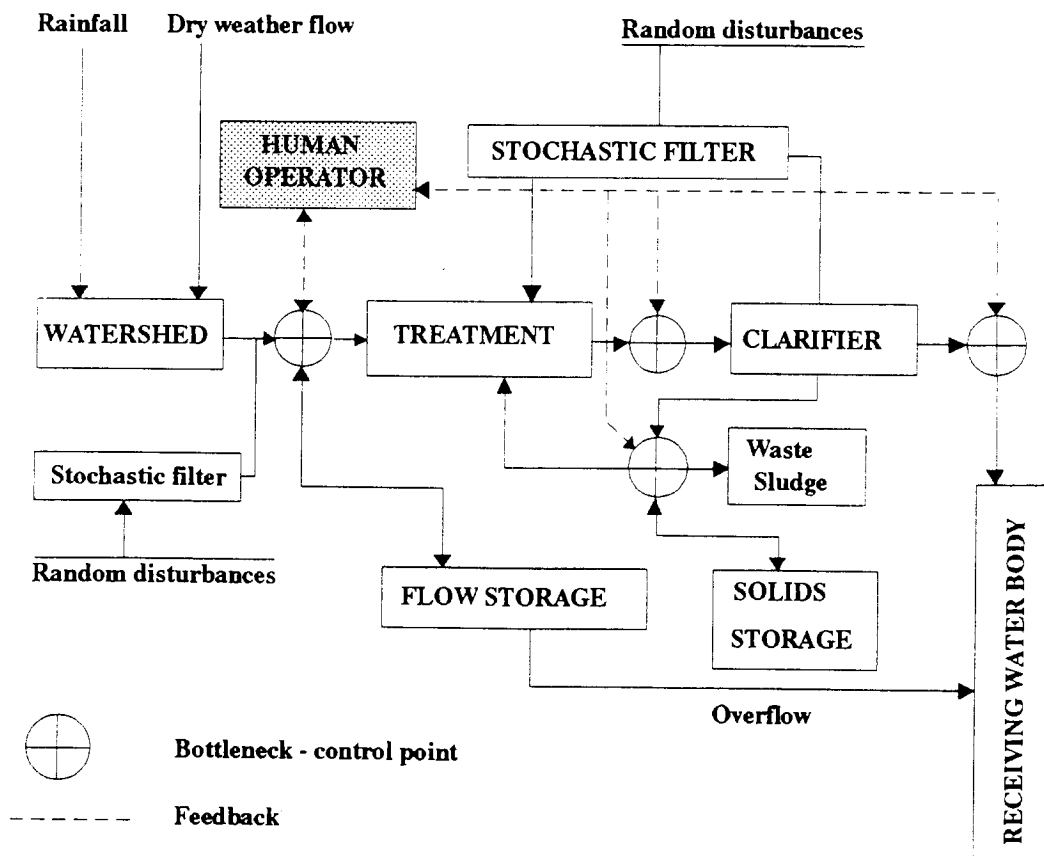


FIGURE 10.38. Schematic of an urban sewerage system with control (bottleneck) points. (After Novotny and Capodaglio, 1992.)

acity, a massive spill of solids into the effluent can occur. The effluent standards and the waste-assimilative capacity of the receiving body of water are the final bottlenecks of the system. Figure 10.38 shows a typical schematic of an urban sewerage system with the locations of possible bottleneck points.

The invariant steady-state mode of operation typical of a majority of sewerage and wastewater disposal facilities dictates that when the input to a bottleneck-control point exceeds its capacity, the excess is bypassed and discharged without treatment. The steady-state solution is to enlarge the bottleneck, which is costly. This mode of operation is inefficient and results in large quantities of untreated sewage being discharged into receiving waters on the one hand, but on the other hand, there may be unused or idle capacity in some other places throughout the system. Furthermore, funds for structural enlargement of existing capacities of sewers and treatment facilities to accommodate transient wet-weather loads may be limited. Thus a conversion of existing facilities to a dynamic operational-control scheme or a conversion of existing facilities to such control may be an economical and more efficient alternative to a massive structural enlargement of sewer and treatment facilities.

Dynamic control of CSOs implies a computerized monitoring, forecasting, and optimization system by which the drainage system, including pipes, overflows, and storage can be controlled and optimized. Most of the research has been conducted in Europe and reported in the Proceedings of the International Conferences on Urban Storm Drainage (Balmér, Malmqvist, and Sjöberg, 1984; Gujer and Krejci, 1987; Iwasa and Sueishi, 1990) or in a special International Association for Water Pollution Research and Control (IAWPRC) publication edited by Schilling (1989).

Objectives and Strategy of RTC Systems for Control of Urban Sewerage

The objective of RTC control is to provide optimization and minimization of the total pollution load of a joint conveyance and disposal of dry-weather (sewage and wastewater) and wet-weather (storm runoff) flows. However, the RTC systems have been researched and proposed mostly for the control of CSOs (Schilling, 1989). Only recently has their use been extended to consider the treatment facility in a pollution trade-off (Novotny, Capodaglio, and Feng, 1990; Durchschlag and Schilling, 1990). Note that control and minimization of overflows require treatment and storage, and the operation of the two components should be optimized in a coordinated fashion. The alternative objectives that have been pre-

sented for the control of urban sewerage systems are (Novotny and Capodaglio, 1992):

1. *Minimization of discharges of untreated overflows.* This objective was suggested in several RTC systems dealing with combined sewer overflows (CSOs). In this type of control, storage and release from storage, and overflow within the system are manipulated to minimize the volume of the untreated mixture of rain and wastewater from CSOs (Labadie, Morrow, and Chen, 1980). Several modifications of this control strategy have been proposed by German researchers (Weynard and Dohman, 1990; Kollatsch and Schilling, 1990).
2. *Stability of the treatment process and meeting the effluent standard.* This is a typical control objective and strategy of manual control for PTW. In this system the operator maintains stable MLSS and dissolved oxygen throughout the treatment process, and maintains the effluent BOD and SS below the effluent standards. The control variables in this type of RTC are the return sludge flow, excess sludge, sludge waste, and the air flow into the aeration basins. Avoidance of sludge bulking is another objective in this category.

This type of control mostly responds to the present situation occurring in the plant, unless the operator is capable of anticipating and predicting the plant overloads and upsets. Although bypasses are not allowed by law (with exceptions stipulated by the U.S. EPA), the control strategy itself does not pay much attention to what happens before treatment. The excess flow over the capacity of the treatment plant is automatically bypassed.

3. *Minimization of total pollution load.* This objective considers both the effluent discharges and bypasses—overflows from the system, and minimizes the total pollution load from the system. The system recognizes the excess capacity of the sewer system to temporarily store a part of the wet-weather flow and the full dynamic capacity of the treatment plant. Auxiliary off-line storage for excess influent flow and excess solids enhances the capability of the system to handle and treat wet-weather flows (Novotny, Capodaglio, and Feng, 1990). Since this control focuses on the treatment facility, overflows from distant CSOs within the sewer system cannot be considered. The control variables are a manipulation of the off-line storage of flow and solids, return sludge flow, and sludge waste. Nelen (1990) proposed a control objective based on the water quality impact of the overflows, which depends on the total pollution load and conditions of the receiving water body.

4. *Avoidance or minimization of "bottleneck" situations.* This control objective was formulated by Capodaglio (1990). A bottleneck occurs when:
- a. the capacity of a process unit within the system is exceeded due to
 - (1) flow exceeding the hydraulic capacity of the unit
 - (2) mass load exceeding the mass loading limit of the unit
 Exceeding these limits would lead to a drastic reduction of treatment efficiency and/or overflow. Such limitations are caused by the hydraulic capacity of the units, limited aeration capacity, settling characteristics of solids, and other factors.
 - b. effluent or stream quality limitations would be exceeded
This factor represents a failure of the total system similar to the partial failure specified previously.
 - c. capacity of residue (sludge) disposal units would be exceeded
Temporary auxiliary sludge storage may be needed to handle this bottleneck situation. Many plants employ digesters and/or sludge lagoons for the storage of sludge. Including these facilities in the RTC system enhances the dynamic efficiency of the treatment process.
5. *Control strategy by Tan et al. (1990).* A group of Australian RTC researchers specified the following for an RTC urban sewerage system:
- a. Main goals
 - (1) minimize overflows
 - (2) reduce peak energy consumption of pumping (lift) stations
 - (3) reduce treatment cost at the downstream treatment plant
 - b. Constraints
 - (1) the capacity of the treatment plant
 - (2) pumping capacity of lift stations
 - (3) in-line storage capacity

Additional alternative control objectives were presented and discussed by Béron et al. (1984), Schilling (1986, 1989), and several other authors.

Strategy

In a typical RTC system, pumps, valves, gates, etc., have to be operated to store wastewater or solids in-line or off-line before the bottleneck and route them to subsequent treatment and receiving waters. It is highly undesirable to create flooding, overflows, and/or violation of standards if the system has an unused storage capacity upstream or downstream from the bottleneck at the same time.

The time sequence of the set points of all the regulators in an RTC system is termed *control strategy*. The determination of a control strategy

can be either automatic (built-in) or manual. The strategy can be found through mathematical modeling, optimization, search, decision matrices, control scenarios, trial-and-error (heuristically), or through a self-learning expert system. This can be done either during the on-going process (online) or beforehand, after predictions (off-line). After the control strategy has been defined, it is distributed throughout the system for implementation.

Components of RTC Systems

The RTC system represents a set of models and concepts that is used for optimal and flexible control of urban sewerage systems. In an RTC system the following elements are used (Schilling, 1989):

- A (measurement) sensor that is used to monitor the on-going process, for example, water level gauge or turbidity meter
- A (corrective) regulator that manipulates the process, for example, gates, valves, and so forth
- A controller that causes the regulator to bring the process back to its desired value (set or goal point)
- A communication system that carries the measured data from the sensor to the controller and the signals of the controller back to the regulator, for example, a telemetry system

The RTC systems do not have to a priori imply automation. As a matter of fact, significant and substantial operator interface, control, and override are desirable. In either case, computers equipped with RTC software are an integral part of the system.

In the context of the RTC, the meaning of "system" is the controllable part of the urban drainage and treatment system. A real-time control can be planned and implemented if numerical simulation models (deterministic, statistical, stochastic, neural network, etc.) are available and included in the RTC system.

As pointed out in Chapter 8, the following processes have to be modeled in order to obtain a comprehensive overview of the performance of the system:

- *Input to the system*: deterministic and/or stochastic rainfall-runoff-sewer flow and quality models will provide short-term forecasts of the inputs
- *System response to the input*: transfer function models will provide pollutant loads, flows, water levels, mixed liquor solids, etc., at key bottleneck points of the system
- *Total output to the environment*: all partial outputs (CSO, bypasses, effluent) will be summarized in order to minimize in an optimal way the total output from the system

- *Optimization of the system:* optimization routines are applied to achieve the goal
- *Response of the receiving environment to the total output:* the total output after system optimization can be compared with the waste-assimilative capacity of the receiving body of water, and further restrictions may be imposed on the system if the waste-assimilative capacity is exceeded. In such case the system optimization is repeated with the modified goals

Modeling Technology

The operational models to be used in an RTC system for control of discharges and pollution loads from urban sewerage and industrial wastewater processes ought to be adaptive both in response to any changes of the input waste loads and to the variations in the system parameters. Such models should be capable of on-line installation in an operational mode of a typical medium to large wastewater disposal facility. An automated operational mode is an option, but not a requirement, nor constraint imposed on the RTC operational systems.

A new idea advanced in research and implementation of the RTC systems is that all wastewater treatment and disposal organizations routinely and periodically collect data on input waste loads, effluent quality, and various operational parameters (temperature, sludge volume index, mixed liquor solids, dissolved oxygen in the basins, and others). The period of acquisition for most operations varies from hours to days, while some variables are recorded continuously. Currently, most of the data-collecting activities are for administrative and statistical purposes, and for recordkeeping and/or are required by the discharge permit. However, the same data may also contain needed and valuable information on the parameters and interrelationships between the parameters that can assist the operator in selecting the most optimal mode of operation.

A model of the system and of the processes occurring therein is a fundamental component of the RTC system. Since daily or even hourly series of data are accepted and managed by the RTC system, the models are in the category of a time series analysis. The adaptive modeling technology available to carry out the tasks according to the stated objectives of RTC control (i.e., adaptiveness, predictiveness, and efficiency) can be divided into two categories (Novotny, Capodaglio, and Feng, 1990):

1. *Auto-regressive moving-average models (ARMA)*, introduced by Box and Jenkins, 1976). Both univariate and transfer function models and their modifications can be considered.

2. *Artificial Neural Network Models* (ANN), proposed by Hopfield (1982) and Kohonen (1972).

The fundamentals of these models were introduced in the preceding chapter.

The “traditional” nonadaptive deterministic models can also be used alone, or the most important known deterministic relationships can be incorporated into the stochastic ARMA–transfer function models.

**Existing RTC Systems for Wet-Weather–
Dry-Weather Wastewater–Urban Runoff Treatment
and Disposal Operation**

CSO control. Most of the existing RTC systems in prototype use today are for the operation of the combined sewer overflows. In the United States such systems are in operation, for example, in Milwaukee, Wisconsin, Seattle, Washington, and Lima, Ohio. The objective of these systems is to minimize the overflows from combined sewers. The municipality of Seattle has been using computers for the control of their sewerage system since 1973. Currently, the city is developing an advanced system for real-time control of pump stations and regulator control in the combined sewer system to minimize overflows. The system is called CATAD (computer-augmented treatment and disposal). It is a centralized control system that has been implemented on a large mainframe computer. Of note is the AM analysis of the economy of the application of the real time control performed for the municipality by CH2M-Hill Consultants, which estimated that incorporating the real-time software would result in savings of \$18 million annually in Seattle alone (Vitasovic, Swarner, and Speer, 1988).

The new CATAD system employs hydrologic and hydraulic routines to estimate dynamic flows at various points of the sewer system. The models are essentially deterministic. A number of CSO control RTC systems have been implemented in Europe (Durschlag and Schilling, 1990; Kolatsch and Schilling, 1990; Bennerstedt, Arnel, and Svenson, 1990; Weyand and Dohman, 1990).

Wastewater Treatment Operation

CS-FOCUS. The senior author and coworkers (Novotny, 1989; Novotny, Capodaglio and Feng, 1990) developed a computerized real-time control system with the acronym CS-FOCUS (computerized system for overflow control of urban sewerage). The overall schematic block diagram is shown on Figure 10.39.

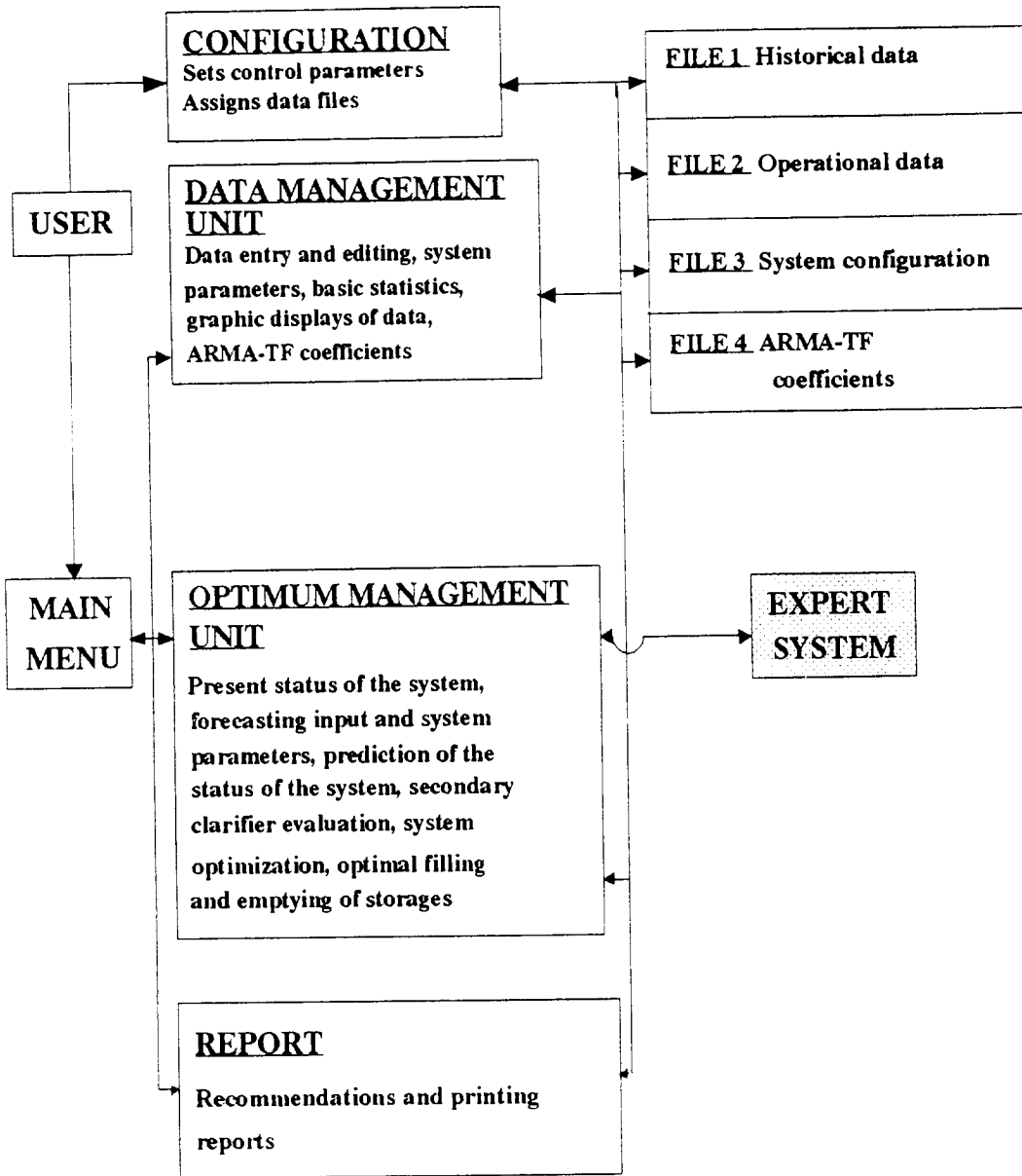


FIGURE 10.39. Real-time control (RTC) system for wet-weather overflow control (CS-FOCUS). (After Novotny, 1989, and Novotny and Capodaglio, 1992.)

Other RTC Systems. Berthouex, Lai, and Darjatmoko (1989) of the University of Wisconsin in Madison, developed a statistics-based expert system for the treatment plant operation. The system has three basic units: the data entry system, the data analysis system, and the expert advisory system. The data entry and data analysis systems are similar in their function to the data management unit of the CS-FOCUS (data entry

and basic and intermediate statistics). The expert system incorporated in the Berthouex, Lai, and Dayatmoko system is rule based. The rules were taken from domain experts. At the time of the writing of their paper there were only four rules tested for activated sludge plant operation; however, up to thirteen rules were anticipated by the creators. The system remembers the variables that are periodically tested and the limits within which they should be maintained. The computer software of the University of Wisconsin RTC system was written for a larger mainframe computer and relies on some specific software packages.

An advanced computerized system for control operation of wastewater treatment plants is being developed in Japan (Hiraoka and Tsumura, 1988; Hiraoka et al., 1990) by a team from the University in Kyoto. This approach uses a computer statistical package SACCES (Statistical Analysis and Computer Control of Environmental Systems). The statistical package provides AR-TF functional models from the observed time series of the input and output data.

The Kyoto group selected the multivariate AR models as the most appropriate models for operational control of activated sludge systems. The control process and methodology proposed by the Japanese team is very complex; nevertheless, the approach is theoretically very sound. Of note also is the hierarchical concept of treatment plant operational control suggested by the Kyoto group. An upper level control system controls long-term (seasonal) variations, an intermediate system is aimed at an overall optimization of the process, and a lower level system deals with diurnal or hourly fluctuations.

Research conducted in Australia by Tan et al. (1990) was mentioned previously. An ARMAX model (similar to the AR-TF model developed and used by the senior author and coworkers), recursively updated by Wittenmark's (1974) self-tuning predictor, has been used for predicting flows in the sewer network. Excellent predictive capability has been achieved. The strategy of the RTC system worked and led to improved economy and reduction of overflows and pumping energy requirements, and to the improved performance of the treatment plant during wet-weather flow condition.

The LOCUS system being developed by Delft University in Holland (Nelen, 1990) included treatment in the consideration of the RTC sewerage system. The assumption incorporated in the LOCUS concept is that the efficiency of most treatment units is not affected by an increase in the hydraulic load as long as the critical hydraulic capacity is not exceeded. The simplified treatment mode of clarifier control uses four parameters: undisturbed clarifier treatment efficiency; period of time with maximum load, after which sludge overflow occurs; disturbed treatment efficiency;

and the period of time it takes for the system to recover when the hydraulic load is reduced below the critical value.

Practical Applications

The RTC systems for wastewater treatment plant operations discussed herein have not yet been fully implemented in a real prototype situation. However, following the impetus by the latest amendments of the Federal Water Pollution Control Act (Clean Water Act), the Environmental Protection Agency will be implementing tougher standards in order to minimize pollution from overflows and bypasses of existing wastewater treatment facilities. As has been documented in Seattle in a real prototype situation, implementing on-line RTC can save a municipality substantial amounts of financial resources.

Considering the fact that Congress is unlikely to authorize any substantial grants for upgrading and enlarging the sewerage systems that anticipate unacceptable overflows or violations of effluent standards, applications of alternative, less costly solutions are needed. The presented computerized RTC system will provide such remedial alternatives. The operators of the treatment system will also benefit from the interpretive expert system and graphical display features that may be incorporated into the RTC system. The modern RTC systems are "smart" control systems that enable the operators to forecast and anticipate the problems caused by transient wet-weather loads and, with the aid of the system, suggest optimal solutions to what otherwise could be a crisis situation in a treatment process. Such systems also provide for recordkeeping of data, graphical displays, and, possibly, reporting required by regulating and permitting authorities.

References

- Agnew, R. W., et al. (1975). *Biological Treatment of Combined Sewer Overflow at Kenosha, Wisconsin*, EPA 670/2-75-019, National Environmental Research Center, Office of Research and Development, U.S. Environmental Protection Agency, Washington, DC, 247p.
- American Association of State Highway and Transportation Officials (1980). *Design of Sedimentation Basins*, Transportation Research Board, National Cooperative Highway Research Program Synthesis of Highway Practice, 53p.
- American Public Works Association (1969). *Water Pollution Aspects of Urban Runoff*, Res. Project No. 120, Federal Water Pollution Control Administration, Washington, DC.
- American Public Works Association (1981). *Urban Stormwater Management*, APWA, Special Rep. No. 49, 285p.
- Aronson, G. L., D. S. Watson, and W. C. Pisano (1983). Evaluation of catch-

- basin performance for urban stormwater pollution control: Project summary, EPA 600/S2-83-043, Municipal Environmental Research Laboratory, U.S. Environmental Protection Agency, Cincinnati, OH.
- Asmussen, L. E., A. W. White, Jr., E. E. Hauser, and J. M. Shendon (1977). Reduction of 2,4-D load in surface runoff down a grassed waterway, *J. Environ. Qual.* 6(2):159-162.
- Balmér, P., P. A. Malmqvist, and A. Sjöberg (1984). *Proceedings, Third International Conference on Urban Storm Drainage*, 4 vol., Chalmers University of Technology, Göteborg, Sweden, 1669p.
- Bannerman, R., K. Baun, and M. Bohn (1983). *Nationwide Urban Runoff Program, Milwaukee, Wisconsin. Evaluation of Urban Nonpoint Source Pollution Management in Milwaukee County, Wisconsin. Executive Summary*, EPA P005432-01-5, Wisconsin Dept. of Natural Resources, Madison, Environmental Protection Agency, Chicago, IL, Region V (available from NTIS), 11p.
- Bannerman, R., K. Baun, P. Hughes, and D. Graczyk (1983). *Nationwide Urban Runoff Program, Milwaukee, Wisconsin. Evaluation of Urban Nonpoint Source Pollution Management in Milwaukee County, Wisconsin*. Vol. 1. *Urban Stormwater Characteristics, Sources and Pollutant Management by Street Sweeping*. Vol. 2. *Feasibility and Application of Urban Nonpoint Source Water Pollution Abatement Measures*, EPA P005432-01-5, Wisconsin Dept. of Natural Resources, Madison, Environmental Protection Agency, Chicago, IL, Region V (available from NTIS).
- Barfield, B. J., D. T. Y Kao, and E. W. Toller (1975). *Estimation of the Sediment Filtering Action of Grasses Media*, Res. Paper No. 90, Water Research Institute, University of Kentucky, Lexington, KY.
- Bender, G. M., and W. W. Rice (1982). Effects of municipal street sweeping on urban storm runoff quality, in *Proceedings, 1982 International Symposium on Urban Hydrology, Hydraulics and Sediment Control*, University of Kentucky, Lexington, KY, pp. 217-223.
- Bennerstedt, K., V. Arnel, and G. Svenson (1990). Real time control of a combined sewer system, in *Proceedings, Fifth International Conference on Urban Drainage, Suita-Osaka*, Y. Iwasa and T. Sueishi, eds., University of Osaka, Japan, July 23-27, pp. 1255-1260.
- Béron, P., F. Brière, J. Rouselle, and J. P. Riley (1984). An evaluation of some real-time techniques for controlling combined sewer overflows, in *Proceedings, Third International Conference on Urban Storm Drainage*, Vol. 3, Chalmers University of Technology, Göteborg, Sweden, pp. 1093-1097.
- Berthouex, P. M., W. Lai, and A. Dayatmoko (1989). Statistics-based approach to wastewater treatment plant operations, *J. Environ. Eng. Div. ASCE* 115(3): 650-671.
- Box, G. E., and G. M. Jenkins (1976). *Time Series Analysis: Forecasting and Control*, Holden-Day, Oakland, CA.
- Brombach, H. (1987). Liquid-solid separation on vortex-storm-overflows, in *Topics in Urban Storm Water Quality, Planning and Management*, W. Gujer and V. Krejci, eds., *Proceedings, Fourth International Conference on Urban*

- Storm Drainage*, IAWPRC-IAHR, École Polytechnique Fédérale, Lausanne, Switzerland, pp. 103–108.
- Brombach, H. (1989) Combined-sewer overflow control in West Germany—history, practice, and experience, in *Design of Urban Runoff Quality Controls*, L. A. Roesner, B. Urbonas, and M. B. Sonnen, eds., American Society of Civil Engineers, New York, NY, pp. 359–374.
- Capodaglio, A.G. (1990). Identification of a Stochastic Modeling Procedure for Wastewater Treatment Operation and Control, Ph.D. thesis, Department of Civil Engineering, Marquette University, Milwaukee, WI.
- Carlo, P. L., L. P. Owens, G. P. Hanna, Jr., and K. E. Longley (1992). The removal of selenium from water by slow sand filtration, *Water Sci. Technol.* **26**(9–11):2137–2140.
- Chen, C. (1975). Design of sediment retention basins, in *Proceedings, National Symposium on Urban Hydraulics and Sediment Control*. University of Kentucky, Lexington, KY, July 28–31.
- Clark, D. E., and W. C. Cobbins (1963). *Removal Effectiveness of Simulated Dry Fallout from Paved Areas by Motorized and Vacuumized Street Sweepers*, U.S. Naval Radiological Defense Laboratory, San Francisco, CA.
- Collins, M. A., K. Roller, and G. Walton (1992). Managing toxic pollutants in stormwater runoff, *Water Environ. Technol.* **4**(5):60–64.
- Dally, L. K., and D. P. Lettenmaier (1984). Urban storm drainage detention facility for runoff and water quality control, *Proceedings, Third International Conference on Urban Storm Drainage*, Vol. 1, Chalmers University of Technology, Göteborg, Sweden, pp. 47–56.
- DeGroot, W., ed. (1982). *Proceedings, Conference on Stormwater Detention Facilities: Planning, Design, Operation and Maintenance*, American Society of Civil Engineers, New York, 431p.
- DiToro, D. M., and M. J. Small (1979). Stormwater interception and storage, *J. Environ. Eng. Div. ASCE* **105**(EE1):43–54.
- Dorman, M. E., et al. (1988). *Retention, Detention, and Overland Flow for Pollutant Removal from Highway Stormwater Runoff: Interim Guidelines for Management Measures*, FHWA/RD-87/056, U.S. Department of Transportation, Federal Highway Administration, McLean, VA.
- Driscoll, E. D. (1988). Long term performance of water quality ponds, in *Design of Urban Runoff Quality Controls*, Proceedings, Eng. Foundation Conference, L. A. Roesner, et al., eds., American Society of Civil Engineers, New York, pp. 145–162.
- Durchschlag, A., and W. Schilling (1990). The total pollution discharge as a design criterion for combined sewer systems, in *Proceedings, Fifth International Conference on Urban Drainage, Suita-Osaka*, Y. Iwasa and T. Sueishi, eds., University of Osaka, Japan, July 23–27, pp. 1077–1082.
- Ellis, J. B. (1985). Design of urban detention basins for water quality control, in *Proceedings, Non-point Pollution Abatement Symposium*, V. Novotny, ed., Marquette University, Milwaukee, WI.
- Ferrara, R. A., and A. Hildick-Smith (1982). A modeling approach for storm

- water quantity and quality control via detention basins, *Water Resour. Bull.* **18**(6):975-982.
- Field, R. (1984). The USEPA Office of Research and Development's view of combined sewer overflow control, in *Proceedings, Third International Conference on Urban Storm Drainage*, Vol. 4, P. Balmér, P. A. Malmqvist, and A. Sjöberg, eds., Chalmers University of Technology, Göteborg, Sweden, pp. 1233-1356.
- Field, R. (1985a). Urban runoff: Pollution sources, control, and treatment, *Water Resour. Bull.* **21**(2):197-206.
- Field, R. (1985b). Management and control of pollution by urban runoff, in *Proceedings, Non-point Pollution Abatement Symposium*, V. Novotny, ed., Marquette University, Milwaukee, WI.
- Field, R. (1986). Urban stormwater runoff quality management: Low structurally intensive measures and treatment, in *Urban Runoff Pollution*, H. C. Torno, J. Marsalek, and M. Desbordes, eds., New York, pp. 677-699.
- Field, R. (1990). Combined sewer overflows: Control and treatment, in *Control and Treatment of Combined-Sewer Overflows*, P. E. Moffa, ed., Van Nostrand Reinhold, New York, pp. 119-191.
- Field, R., K. Dunkers, and A. Forndran (1990). Demonstration of in-receiving water storage of combined sewer overflows in a marine/estuarine environment by the Floe Balance Method, in *Proceedings, Fifth International Conference on Urban Storm Drainage, Suita-Osaka*, Y. Iwasa and T. Sueishi, eds., University of Osaka, Japan, July 23-27, pp. 759-764.
- Florida Concrete Products Association (1989). *Pervious Pavement Manual*, FCPA, Tallahassee, FL.
- Forndran, A., R. Field, and K. Dunkers (1989). *Demonstration of the Flow Balancing Method for Combined Sewer Overflow Abatement in Estuarine Waters*, Water Pollution Control Federation 62nd Annual Conference, San Francisco, CA, 30p.
- Fujita, S. (1984). Experimental sewer system for reduction of urban stormwater runoff, in *Proceedings, Third International Conference on Urban Storm Drainage*, P. Balmér, P. A. Malmqvist, and A. Sjöberg, eds., Chalmers University of Technology, Göteborg, Sweden, pp. 1211-1220.
- Fujita, S., and T. Koyama (1990). Pollution abatement and the experimental sewer system, *Proceedings, Fifth International Conference on Urban Storm Drainage*, IAWPRC-IAHR, Suita-Osaka, University of Osaka, Japan, pp. 799-904.
- Galli, J. (1990). *Peat-sand Filters: A Proposed Stormwater Management Practice for Urban Areas*, Department of Environmental Programs, Metropolitan Washington Council of Governments, Washington, DC.
- Galli, J. (1992). *Preliminary Analysis of the Performance and Longevity of Urban BMPs Installed in Prince George County, Maryland*, prepared for Department of Environmental Resources, Prince George's County, MD.
- Goforth, G. F., E. V. Diniz, and J. B. Rauhut (1984). Stormwater hydrological characteristics of porous and conventional paving systems: Project summary,

- EPA 600/S2-83-106, Municipal Environmental Research Laboratory, U.S. Environmental Protection Agency, Cincinnati, OH.
- Goldman, S. J., K. Jackson, and T. A. Bursztynsky (1986). *Erosion and Sediment Control Handbook*, McGraw-Hill, New York.
- Graham, P. H. (1978). *Report to Congress on Control of Combined Sewer Overflow in the United States*, EPA 430/9-78-006, Facility Requirements Division, Office of Water Program Operations, U.S. Environmental Protection Agency, Washington, DC.
- Griffin, D. M., Jr., C. W. Randall, and T. J. Grizzard (1980). Efficient design of stormwater holding basins used for water quality protection, *Water Res.* **14**:1549-1554.
- Grizzard, T. L. et al. (1986). Effectiveness of extended detention ponds, in *Urban Runoff Quality*, American Society of Civil Engineers, New York.
- Gujer, W., and V. Krejci, eds. (1987). Topics in urban storm water quality, planning and management, in *Proceedings, Fourth International Conference on Urban Storm Drainage*, IAHR-IAWPRC, Lausanne, Switzerland.
- Haak, A., and G. Oberts (1983), *Surface Water Management: Management Practices Evaluation*, Metropolitan Council of the Twin Cities Area, St. Paul, MN, 215p.
- Hammer, D. A. (1989). *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural*, in *Proceedings, First International Conference on Constructed Wetlands for Wastewater Treatment*, Chattanooga, TN, June 13-17, 1988, Lewis Publ., Chelsea, MI.
- Harrington, B. W. (1989). Design and construction of infiltration trenches, in *Design of Urban Runoff Quality Controls*, L. A. Roesner, B. Urbonas, and M. B. Sonnen, eds., American Society of Civil Engineers, New York, pp. 290-304.
- Hartigan, J. P. (1989). Basis for design of wet detention basin BMP's, in *Design of Urban Runoff Quality Controls*, L. A. Roesner, B. Urbonas, and M. B. Sonnen, eds., American Society of Civil Engineers, New York, pp. 122-143.
- Herrmann, R. (1984). 3,4-benzopyrene washoff efficiency in different urban environment, in *Proceedings, Third International Conference on Urban Drainage*, P. Balmér, P. A. Malmqvist, and A. Sjöberg, eds., Chalmers Technical University, Göteborg, Sweden, pp. 1001-1008.
- Herrmann, R., and F. G. Kari (1990). Grain size dependent transport of non-polar trace pollutants (PAH, PCB) by suspended solids during urban runoff storm runoff, in *Proceedings, Fifth International Conference on Urban Storm Drainage, Suita-Osaka*, Y. Iwasa and T. Sueishi, eds., University of Osaka, Japan, July 23-27, pp. 499-503.
- Hewitt, C. N., and M. B. Rashed (1992). Removal rates of selected pollutants in the runoff waters from a major rural highway, *Water Res.* **26**(3):311-319.
- Hey, D. L., J. M. Stockdale, D. Kropp, and G. Wilhelm (1982). *Creation of Wetland Habitats in Northeastern Illinois*, Illinois Department of Energy and Natural Resources Document No. 82/09, May 1982, 117p.
- Hiraoka, M., and K. Tsumura (1988). System identification and control of ac-

- tivated sludge process by use of a statistical model, *Water Sci. Tech.* **21**:1161–1173.
- Hiraoka, M., K. Tsumura, I. Fujita, and T. Kanaya (1990). System Identification of Activated Sludge Process by Use of Autoregressive Model, in *Advances in Water Pollution Control*, Proceedings, Fifth IAWPRC Workshop on Instrumentation, Control and Automation of Water and Wastewater Treatment and Transport Systems, R. Briggs, ed., Pergamon Press, Oxford, pp. 121–128.
- Hopfield, J. J. (1982). Neural networks and physical systems with emergent collective computational abilities, *Proc. Nat. Acad. Sci.* **79**:2554–2558.
- Innerfeld, H., A. Forndran, D. D. Ruggiero, and T. J. Hartman (1979). Dual process high-rate filtration of raw sanitary sewage and combined sewer overflows, EPA 600/2-79-015, Municipal Environmental Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, 105p.
- Iwasa, Y., and T. Sueishi, eds. (1990). *Proceedings, Fifth International Conference on Urban Storm Drainage, Suita-Osaka*, University of Osaka, Japan, July 23–27.
- Jackson, T. J., and R. M. Ragan (1974). Hydrology of porous pavement parking lots, *J. Hydraul. Div. ASCE* **12**:1739–1752.
- Kao, D. T. Y., B. J. Barfield, and A. E. Lyons (1975). On site sediment filtration using grass strips, in *Proceedings, National Symposium on Urban Hydrology and Sedimentation Control*, University of Kentucky, Lexington, KY.
- Kohonen, T. (1972). Correlation matrix memories, *IEE Trans. Comput.* **C-21**: 353–359.
- Kollatsch, D., and W. Schilling (1990). Control strategy of sanitary sewage detention tanks to reduce combined sewer overflow pollution loads, in *Proceedings, Fifth International Conference on Urban Drainage, Suita-Osaka*, Y. Iwasa and T. Sueishi, eds., University of Osaka, Japan, July 23–27, pp. 1365–1370.
- Krejci V. (1988) New strategies in urban drainage and stormwater pollution control in Switzerland, in *Proceedings of the Symposium on Nonpoint Pollution: 1988—Policy, Economy, Management and Appropriate Technology*, V. Novotny, ed., AWRA, Bethesda, MD, pp. 203–213.
- Krejci, V., and E. Baer (1990). Screening structures for combined sewer overflow treatment in Switzerland, in *Proceedings, Fifth International Conference on Urban Storm Drainage, Suita-Osaka*, Y. Iwasa and T. Sueishi, eds., University of Osaka, Japan, July 23–27, pp. 927–932.
- Krejci, V., L. Dauber, B. Novak, and W. Gujer (1987). Contribution of different sources to pollutant loads in combined sewers, in *Topics in Urban Storm Water Quality, Planning and Management*, W. Gujer and V. Krejci, eds., Proceedings, Fourth International Conference on Urban Storm Drainage, IAWPRC-IAHR, École Polytechnique Fédérale, Lausanne, Switzerland, pp. 34–39.
- Kuo, C. Y., and W. Y. Ni (1984). Pollutant trap efficiency in a detention basin, in *Proceedings, Third International Conference on Urban Storm Drainage*, Vol. 1,

- P. Balmér, P. A. Malmqvist, and A. Sjöberg, eds., Chalmers University of Technology, Göteborg, Sweden, pp. 21-28.
- Kuo, C. Y., and J. L. Zhu (1989). Design of a diversion system to manage the first flush, *Water Resour. Bull.* **25**(3):517-525.
- Labadie, J. W., D. M. Morrow, and Y. H. Chen (1980). Optimal control of unsteady combined sewer flow, *J. Water Resour. Planning Manage. ASCE*, **106**(WR1):205-223.
- Lager, J. A., and W. G. Smith (1974). *Urban Stormwater Management and Technology: An Assessment*, EPA 670/2-74-040, National Environmental Research Center, Office of Research and Development, U.S. Environmental Protection Agency, 447p.
- Lager, J. A., W. G. Smith, W. G. Lynard, R. M. Finn, and E. J. Finnemore (1977). *Urban Stormwater Management and Technology: Update and Users' Guide*, EPA 600/8-77-014, Municipal Environmental Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, 313p.
- Lessard, P., and M. B. Beck (1991). Dynamic simulation of storm tanks, *Water Res.* **25**(4):375-391.
- Linker, L. C. (1989). Creation of wetlands for the improvement of water quality: A proposal for the joint use of highway right-of-way, in *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural*, D. A. Hammer, ed., Lewis Publ., Chelsea, MI, pp. 695-701.
- Livingston, E. H., and L. A. Roesner (1991). Stormwater management practices for water quality enhancement, in *Manual of Practice Urban Stormwater Management*, American Society of Civil Engineers, New York.
- Loganathan, V. G., J. Delleur, and R. I. Segarra (1985). Planning detention storage for stormwater management, *J. Water Resour. Planning Manage. ASCE* **111**(4):382-398.
- McCuen, R. H. (1980). Water quality trap efficiency of storm water management basins, *Water Resour. Bull.* **16**(1):15-21.
- Maryland Water Resources Administration (1984). *Standards and Specifications for Infiltration Practices*, MWRA, Annapolis, MD.
- Maryland Water Resources Administration (1986). *Minimum Water Quality Objectives and Planning Guidelines for Infiltration Practices*, MWRA, Annapolis, MD.
- Metropolitan Washington Council of Governments (1983). *Nationwide Urban Runoff Program, Washington Metropolitan Area Urban Runoff Demonstration Project: Final Report, 1979-1982*, Metropolitan Washington Council of Governments, Washington, DC; Northern Virginia Planning District Commission, Annandale, VA; Water Planning Div., Environmental Protection Agency, Washington, DC, 120p. (available from NTIS).
- Minnesota Pollution Control Agency (1989). *Protecting Water Quality in Urban Areas*, Division of Water Quality, St. Paul, MN.
- Moffa, P. E., ed. (1990). *Control and Treatment of Combined-Sewer Overflows*, Van Nostrand Reinhold, New York.
- Morrison, G. M. P., D. M. Revitt, J. B. Ellis, G. Svensson, and P. Balmér

- (1984). The physico-chemical speciation of zinc, cadmium, lead and copper in urban stormwater, in *Proceedings, Third International Conference on Urban Storm Drainage*, P. Balmér, P. A. Malmqvist, and A. Sjöberg, eds., Chalmers University of Technology, Göteborg, Sweden, pp. 989–1000.
- Mulhern, P. F., and T. D. Steele (1988). Water-quality ponds—are they the answer? in *Design of Urban Runoff Quality Controls*, Proceedings, Engineering Foundation Conference, L. A. Roesner, B. Urbonas, and B. Sonneu, eds., American Society of Civil Engineers, New York, pp. 203–213.
- Nelen, A. J. M. (1990). Control strategies based on water quality aspects, in *Proceedings, Fifth International Conference on Urban Drainage, Suita-Osaka*, Y. Iwasa and T. Sueishi, eds., University of Osaka, Japan, July 23–27, pp. 1311–1316.
- Nieswand, G. H., et al. (1990). Buffer strips to protect water supply reservoirs: A model and recommendations, *Water Resour. Bull.* **26**(6):959–966.
- Nix, S. J., and T. K. Tsay (1988). Alternative strategies for stormwater detention, *Water Resour. Bull.* **24**(3):609–614.
- Nix, S. J., J. P. Heaney, and W. C. Huber (1988). Suspended solids removal in detention basins, *J. Environ. Eng. ASCE* **114**(6):1331–1343.
- Novak, B. (1983). Schutz des Vorfluters vor Schmutzstoffen aus Regenentlastungen, in *Jahresbericht des EAWAG*, Dübendorf/Zurich, Switzerland.
- Novotny, V. (1984) Efficiency of low cost practices for controlling pollution by urban runoff, in *Proceedings, Third International Conference on Urban Storm Drainage*, P. Balmér, P. A. Malmqvist, and A. Sjöberg, eds., Chalmers University of Technology, Göteborg, Sweden, pp. 1241–1250.
- Novotny, V. (1989). *CS-FOCUS-Computerized System for Overflow Control of Urban Sewerage: Concepts, Theory and System Description*, AquaNova International, Mequon, WI.
- Novotny, V., and A. Capodaglio (1992). Strategy of stochastic Real-Time Control of wastewater treatment plants, *ISA Trans.* **32**(1):73–86.
- Novotny, V., and G. Chesters (1981). *Handbook of Nonpoint Pollution: Sources and Management*, Van Nostrand Reinhold, New York.
- Novotny, V., and G. Chesters (1989). Delivery of sediment and pollutants from nonpoint sources: A water quality perspective, *J. Soil Water Conserv.* **44**(6): 568–576.
- Novotny, V., and G. Kincaid (1982). Acidity of urban precipitation and its buffering during overland flow, in *Urban Stormwater Quality, Management and Planning*, B. C. Yen, ed., Water Resources Publ., Littleton, CO, pp. 1–9.
- Novotny, V., A. Capodaglio, and X. Feng (1990). Stochastic Real Time Control of wastewater treatment plant operation, in *Advances in Water Pollution Control*, Proceedings, Fifth IAWPRC Workshop on Instrumentation, Control and Automation of Water and Wastewater Treatment and Transport Systems, R. Briggs, ed., Pergamon Press, Oxford, pp. 539–544.
- Novotny, V., R. Schonter, and L. Novotny (1992). *Efficiency of Best Management Practices for Control of Pollution by Priority Pollutants with Emphasis on Army Operations*, US Army Corps of Engineers Civil Eng. Res. Lab., Proj. No. DACA88-92-M0594, Champaign, IL.

- Novotny, V., H. M. Sung, R. Bannerman, and K. Baum (1985). Estimating nonpoint pollution from small urban watersheds, *J. WPCF* 57(4):339–348.
- Novotny, V., K. R. Imhoff, M. Othoff, and P. A. Krenkel (1989). *Karl Imhoff's Handbook of Urban Drainage and Wastewater Disposal*, Wiley Interscience, New York.
- Oakland, P. H. (1983). An evaluation of urban stormwater pollution removal through grassed swale treatment, in *Proceedings, International Symposium on Urban Hydrology, Hydraulics and Sediment Control*, University of Kentucky, Lexington, KY.
- Oberts, G. (1982). Impact of wetlands on nonpoint source pollution, in *Proceedings, International Symposium on Urban Hydrology, Hydraulics and Sediment Control*, University of Kentucky, Lexington, KY.
- Ormsbee, L. E. (1984). Systematic planning of dual purpose detention basins, University of Kentucky, Office of Engineering Services, (*Bulletin*) UKY BU 135:143–151.
- Pisano, W. C. (1989). Swirl concentrators revisited: The American experience and new German technology, in *Design of Urban Runoff Quality Controls*, L. A. Roesner, B. Urbonas, and M. B. Sonnen, eds., American Society of Civil Engineers, New York, pp. 390–402.
- Pisano, W. C. (1990). Recent United States experience with designs of German vortex solids separators for CSO control, in *Proceedings, Fifth International Conference on Urban Storm Drainage*, IAWPRC-IAHR, Suita-Osaka, University of Osaka, Japan, pp. 933–938.
- Pisano, W. C. (1991). *Design of CSO Treatment Facilities*, paper presented at Combined Sewer Overflow Conference, Technical University of Nova Scotia Centre for Water Resources, May 10, 1991.
- Pisano, W. C., D. J. Connick, and G. L. Aronson (1985). *In-system Storage Controls for Reduction of Combined Sewer Overflow, Saginaw, Michigan*, EPA 905/2-85-001-A, Great Lakes National Program Office, U.S. Environmental Protection Agency, Chicago, IL.
- Pitt, R. (1979). *Demonstration of Nonpoint Pollution Abatement through Improved Street Cleaning Practices*, EPA 600/2-79/161, U.S. Environmental Protection Agency, Cincinnati, OH.
- Pitt, R., and P. Baron (1989). *Assessment of Urban and Industrial Stormwater Runoff Toxicity and the Evaluation/Development of Treatment for Runoff Toxicity Abatement—Phase I*. Storm and Combined Sewer Pollution Program, U.S. Environmental Protection Agency, Edison, NJ.
- Pitt, R., and G. Shawley (1981). *San Francisco Bay Area National Urban Runoff Project. Demonstration of Non-point Source Pollution Management on Castro Valley Creek: Executive Summary* (final rept.), Water Planning Div., Environmental Protection Agency, Washington, DC (available from NTIS).
- Pitt, R., R. Field, M. Lalor, and G. Driscoll (1990a). Analysis of cross-connections and storm drainage, in *Proceedings, Urban Stormwater Enhancement—Source Control, and Combined Sewer Technology*, Davos, Switzerland, Eng. Foundation, American Society of Civil Engineers, New York.
- Pitt, R., M. Lalor, N. Miller, and G. Driscoll (1990b). *Assessment of Non-*

- Stormwater Discharges into Separate Storm Drainage Networks—Phase I: A Manual of Practice*, Storm and Combined Sewer Section, U.S. Environmental Protection Agency, Edison, NJ.
- Ports, M. A. (1975). *Urban Sediment Control Design Criteria and Procedures*, Technical Paper No. 75-2567, American Society of Agricultural Engineers, 28p.
- Potts, R. R., and J. L. Bai (1989). Establishing variable width buffer zones based upon site characteristics and development type, in *Proceedings, Symposium on Water Laws and Management*, American Water Resources Association, Bethesda, MD.
- Pravoshinsky, N. A. (1975). Basic principles for determining regulating structures to prevent rain sewers receivers from contamination, *Prog. Water Tech.* 7(2): 301–307.
- Ree, W. O., and V. J. Palmer (1949). *Flow of Water in Channels Protected by Vegetative Lining*, U.S. Soil Conservation Serv. Bull. No. 967, Washington, DC.
- Roesner, L. A. (1988). Aesthetic implementation of nonpoint source controls, in *Proceedings, Symposium on Nonpoint Pollution: 1988—Policy, Economy, Management, and Appropriate Technology*, V. Novotny, ed., American Water Resources Association, Bethesda, MD, pp. 213–223.
- Roesner, L. A., E. H. Burges, and J. A. Aldrich (1991). *The Hydrology of Urban Runoff Quality Management*, paper presented at WPCF-ASCE Conference on Urban Runoff, New Orleans, LA, May 21–22.
- Roesner, L. A., B. Urbonas, and B. Sonnen, eds. (1989). *Design of Urban Runoff Quality Controls*, Proceedings, Engineering Foundation Conference, American Society of Civil Engineers, New York.
- Salomons, W., and U. Förstner (1984). *Metals in the Hydrocycle*, Springer Verlag, Berlin, New York.
- Sartor, J. D., and G. B. Boyd (1972). *Water Pollution Aspects of Street Surface Contamination*, EPA R2-72-081, U.S. Environmental Protection Agency, Washington, DC.
- Sartor, J. D., G. B. Boyd, and F. J. Agardy (1974). Water pollution aspects of street surface contamination, *J. WPCF* 46:458–463.
- Schilling, W. (1986). Urban runoff quality management by Real-Time Control, in *Urban Runoff Pollution*, H. C. Torno, J. Marsalek, and M. Desbordes, eds., Springer Verlag, New York, pp. 765–817.
- Schilling, W. (1989). *Real-Time Control of Urban Drainage Systems. The State-of-the-Art*, Sci. Tech. Rep. No. 2, IAWPRC, London.
- Scholze, R., V. Novotny, and R. Schonter (1993). *Efficiency of Best Management Practices for Control of Pollution by Priority Pollutants*, Paper presented at the First IAWQ Conf. on Diffuse Pollution—Chicago, IL, Sept. 1993, *Water Sci. Technol.* 28(3–5):215–224.
- Schueler, T., and M. Helfrich (1988). Design of extended detention wet pond systems, in *Design of Urban Runoff Quality Controls*, L. A. Roesner, B. Urbonas, and B. Sonnen, eds., Proceedings, Engineering Foundation Conference, American Society of Civil Engineers, New York, pp. 180–200.

- Schueler, T. R., P. A. Kumble, and M. A. Heraty (1992). *A Current Assessment of Urban Best Management Practices. Techniques for Reducing Non-point Source Pollution in the Coastal Zone*, Technical Guidance Manual, Metropolitan Washington Council of Governments, Office of Wetlands, Oceans, and Watersheds, U.S. Environmental Protection Agency, Washington, DC.
- Stahre, P., and B. Urbonas (1989). Swedish approach to infiltration and percolation design, in *Design of Urban Runoff Quality Controls*, L. A. Roesner, B. Urbonas, and M. B. Sonnen, eds., American Society of Civil Engineers, New York, pp. 307–322.
- Stahre, P., and B. Urbonas (1990). *Stormwater Detention For Drainage, Water Quality, and CSO Management*, Prentice Hall, Englewood Cliffs, NJ.
- Stockdale, E. C., and R. R. Horner (1987). Prospects for wetlands use in stormwater management, *Proceedings, Fifth Symposium on Coastal and Ocean Management*, Vol. 4, American Society of Civil Engineers, pp. 3701–3714.
- Striegl, R. G. (1987). Suspended sediment and metals removal from urban runoff by a small lake, *Water Resour. Bull.* **23**(6):985–996.
- Sullivan, R. H. (1968). Problems of combined sewer facilities and overflows, *J. WPCF* **41**(1):113–121.
- Sullivan, R. H., et al. (1982). *Design Manual: Swirl and Helical Bend Pollution Control Devices*, EPA 600/8-82-013, Storm and Combined Sewer Section, U.S. Environmental Protection Agency, Edison, NJ.
- Tan, P. C., K. P. Dabke, R. G. Mein, and C. S. Berger (1990). Real-Time Control of sewer networks, in *Proceedings, Fifth International Conference on Urban Drainage, Suita-Osaka*, Y. Iwasa and T. Sueishi, eds., University of Osaka, Japan, July 23–27, pp. 1231–1237.
- Terstriep, M. L., M. G. Bender, and D. C. Noel (1982). *Nationwide Urban Runoff Project, Champaign, Illinois: Evaluation of the Effectiveness of Municipal Street Sweeping in the Control of Urban Storm Runoff Pollution*, Final report, Illinois State Water Survey Div., Champaign; Illinois State Environmental Protection Agency, Springfield; Environmental Protection Agency, Chicago, IL, Region V, 246p. (available from NTIS).
- Thelen, E., W. C. Grover, A. J. Hoiberg, and T. I. Haigh (1972). *Investigation of Porous Pavements for Urban Runoff Control*, Office of Research and Monitoring, U.S. Environmental Protection Agency, Washington, DC, 142p.
- Toller, E. W., B. J. Barfield, C. T. Haan, and D. T. Y. Kao (1977). Suspended sediment filtration capacity of simulated vegetation, *Trans. ASAE* **19**(5): 678–682.
- Torno, H. C., J. Marsalek, and M. Desbordes, eds. (1986). *Urban Runoff Pollution*, Springer Verlag, Heidelberg, New York.
- Urbonas, B. (1984). Summary of findings by ASCE Task Committee on Detention Basins, in *Proceedings, Third International Conference on Urban Storm Drainage*, Vol. 2, Chalmers University of Technology, Göteborg, Sweden; pp. 733–742.
- U.S. Environmental Protection Agency (1976). *Erosion and Sediment Control, Surface Mining in Eastern U.S.A.*, EPA 625/3-76/006, U.S. EPA, Washington, DC.

- U.S. Environmental Protection Agency (1977). *Nonpoint Source Control Guidance Hydrologic Modifications*, Water Planning Division, Nonpoint Sources Branch, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1983). *Results of the Nationwide Urban Runoff Program*, Water Planning Division, U.S. EPA, Washington, DC.
- Urbonas, B., and L. A. Roesner, eds. (1986). *Urban Runoff Quality—Impact and Quality Enhancement Technology*, *Urban Runoff Quality Enhancement Technology*, Proceedings, Engineering Foundation Conference, ASCE, New York, 477p.
- Urbonas, B., and W. P. Ruzzo (1986). Standardization of detention pond design for phosphorus removal, in *Urban Runoff Pollution*, H. C. Torno, J. Marsalek, and M. Desbordes, eds., New York, pp. 739–760.
- Vitale, A. M., and P. M. Spray (1975). *Total Urban Water Pollution Loads: The Impact of Storm Water*, U.S. Council on Environmental Quality, Washington, DC.
- Vitasovic, Z., R. Swarner, and E. Speer (1988). *Real Time Control for CSO Reduction*, paper presented at WPCF Annual Conference, Dallas, TX.
- Walesh, S. (1989). *Urban Surface Water Management*, Wiley-Interscience, New York, 518p.
- Walker, W. W. (1987). Phosphorus removal by urban runoff detention basins, in *Lake and Reservoir Management*, Vol. III, North American Lake Management Society, Washington, DC, pp. 314–326.
- Wanielista, M. F., Y. A. Yousef, H. H. Harper, and C. L. Cassagnol (1982). Detention with effluent filtration for stormwater management, in *Urban Stormwater Quality, Management and Planning*, B. C. Yen, ed., Water Resources Publ., Littleton, CO, pp. 314–330.
- Water Pollution Control Federation (1990). *Manual of Practice for the Design and Construction of Urban Stormwater Management Systems*, WPCF, Alexandria, VA.
- Weyand, M., and M. Dohman (1990). Experience and results of Real-Time Control within a combined sewers system, in *Proceedings, Fifth International Conference on Urban Drainage, Suita-Osaka*, Y. Iwasa and T. Sueishi, eds., University of Osaka, Japan, July 23–27, pp. 1237–1248.
- Whipple, W., Jr., et al. (1983). *Stormwater Management in Urbanizing Areas*, Prentice Hall, Englewood Cliffs, NJ.
- Whipple, W., Jr., and J. V. Hunter (1980). *Settleability of Urban Runoff Pollution*, PB80-182017, National Technical Information Service, Springfield, VA.
- Wiesner, P. E., A. M. Kassem, and P. W. Cheung (1982). Parks against storms, in *Urban Stormwater Quality, Management and Planning*, B. C. Yen, ed., Water Resources Publ., Littleton, CO, pp. 322–330.
- Wilson, L. G. (1967). Sediment removal from flood water, *Trans. ASAE* 10(1): 35–37.
- Wittenmark, B. (1974). A self-tuning predictor, *IEEE Trans. Autom. Control* A-19:848–851.
- Woodard, S. E. (1989). The Effectiveness of Buffer Strips to Protect Water

- Quality, Master of Science thesis, Department of Civil Eng., University of Maine, Orono, ME.
- Wotzka, P., and G. Oberts (1988). The water quality performance of a detention basin-wetland treatment system in an urban area, in *Nonpoint Pollution: 1988-Policy, Economy, Management, and Appropriate Technology Symposium Proceedings*, American Water Resources Association, Bethesda, MD, pp. 237-247.
- Yen, B. C. (1982). *Urban Stormwater Quality, Management and Planning*, Water Resources Publ., Littleton, CO.

Agricultural Issues

Soil and water are critical components of the resource base upon which agriculture depends. To move towards sustainability, agriculture and natural resource management interests must recognize that they are equal partners in the effort. The challenge is to adapt and extend our knowledge about soil and water to develop economically productive, culturally appropriate, and environmentally sound agricultural systems. A flexible, ongoing process is necessary to set research priorities to support inherently dynamic agricultural systems.

Committee on International Soil and Water Research
and Development, National Research Council

American agriculture is a large and complex industry. The extent of land devoted to agricultural production, the unusual pricing system in place, the intensity and efficiency of modern farming, ranching, and other animal production systems all contribute to the complexity and diversity of the system. Agricultural production has a great potential to adversely affect the environment, most particularly from nonpoint source runoff, hazardous waste disposal, habitat destruction, and in localized areas, nuisance odors. In other developed countries agriculture can occupy a

Susan Alexander was the primary writer of this chapter. A literature review by Smolen, Jennings, and Huffman (1990) was also a major source of materials.

similar niche in a nation's economy and social structure depending upon the extent of arable land and population pressures. In developing and undeveloped countries agriculture operates at more of a subsistence level status. Subsistence farming practices have an even greater potential for causing environmental damage than modern methods used in developed countries due to the high erosion rates associated with slash and burn or other types of subsistence farming. Additionally, many small or poor nations do not have the regulatory framework in place to prohibit the use of highly toxic or persistent pesticides, nor do farmers in these countries have access to the educational materials and programs offered in developed countries.

This chapter examines how current agricultural management techniques contribute to the diffuse-pollution problem; explores which best management practices (BMPs) are most effective in protecting water quality; explains some of the many programs or methods state and federal governments use to get BMPs installed; discusses a few of the reasons that agricultural nonpoint source (NPS) controls have not been more widely adopted; and recommends or suggests some method that might speed up the installation of agricultural BMPs. For the most part, it focuses on the agricultural systems in place in the United States, because the data base is most extensive.

The following manuals may be studied for more detailed discussions of the management of agricultural and silvicultural pollution:

- Stewart et al. (1975): Control of Water Pollution from Cropland
- Loehr et al. (1979): Best Management Practices for Agriculture and Silviculture
- U.S. EPA (1992c): Managing Nonpoint Source Pollution—Final Report to the Congress
- U.S. Forest Service (1989): Best Management Practices Guide for Integrated Resource Planning on National Forest Lands in the Southwestern Region
- Humenik et al. (1982): Best Management Practices for Agricultural Nonpoint Source Control—Animal Waste
- Thompson et al. (1989): Poison Runoff: A Guide to State and Local Control of Nonpoint Source Water Pollution
- U.S. EPA (1993): Guidance Specifying Management Measures for Sources of Nonpoint Pollution in Coastal Waters

Some concepts to look for in this chapter that will provide a more thorough understanding of agricultural issues include:

- Agricultural nonpoint source abatement is implemented mostly through voluntary programs.
- BMPs may not have a direct measurable benefit to the land or the water immediately adjacent to the land to which they are applied (benefits and impacts are off-site).
- Agricultural producers (in general) have a particular outlook on their profession.
- Agricultural producers “buy at retail and sell at wholesale” (they do not set the prices), thus a large network of commodity programs has been established in most developed countries).
- Variability in BMP effectiveness can be extreme.
- Diffuse-pollution control on agricultural lands can turn into a private lands rights issue, quickly becoming an emotional and political “hot potato” that no one wants to touch.
- Most of the technology to control agricultural nonpoint pollution is currently available.

EXTENT OF AMERICAN AGRICULTURE

According to the USDA Economic Research Service (USDA, National Agricultural Statistics Service, 1992a–1992g), sales of agricultural commodities account for 16% of the gross national product of the United States. Agricultural land is generally divided into the following categories for inventory purposes:

Dryland cropland

Irrigated cropland

Pasture land

Range land

Forest land

Confined animal feeding operations

Specialty areas (such as aquaculture, orchard crops, and wildlife land)

These divisions are useful to water quality planning efforts and nonpoint source pollution control since each type of land area generally has a somewhat distinct set of pollutants of most concern associated with that land use, and because most current BMP reference guides are divided by these categories rather than by pollutant.

Row and field crops, such as corn or hay, are generally produced on privately owned land in the United States, but beef cattle, sheep and goats, and timber are often produced on publicly held lands that are leased to private individuals for production. To be effective in controlling

TABLE 11.1 Extent of American Agriculture Production for Selected Commodities in 1991

Commodity	Amount	Cash Receipts or Value of Product (\$)	Acres Planted
Corn	7,474,480,000 bu	18,063,205,000	75,951,000
Cotton	17,541,500 bales ^a	5,322,413,000	14,143,000
Grain sorghum	579,490,000 bu	1,347,366,000	11,014,000
Soybeans	1,985,564,000 bu	11,078,422,000	59,060,000
Small grains ^b	2,697,486,000 bu	7,117,902,000	89,172,000
Rice	154,457,000 cwt	1,166,077,000	2,851,000
Vegetables ^c	—	5,277,606	—
Fruits and nuts	—	9,574,134,000	—
Floricultural crops ^d	—	2,802,510,000	—
Hay	153,485,000 tons	9,800,766,000	62,575,000
Beef cattle	100,110,000 head	39,632,086,000	—
Sheep	10,850,000 head	399,097,000	—
Hogs	57,684,000 head	11,064,101,000	—
Broilers ^e	6,138,350,000 birds	8,385,284,000	—
Turkeys	285,000,000 birds	2,344,016,000	—
Eggs	68,958,000,000 eggs	3,886,810,000	—
Spent layers	197,518,000 birds	67,548,000	—
Catfish ^f	390,870,000 lb	346,638,970	—
Dairy cows	9,990,000 head produced 148,526,000 pounds of milk	18,075,191,000 milk and cream	—

Sources: From USDA NASS (1992a-g).

^a One bale of cotton = 500 lb.

^b Small grains = wheat, rye, barley, oats.

^c Not all vegetable crops reported upon.

^d Estimate, consists of flowering plants and cut flowers.

^e Excludes states that produced less than 500,000 broilers.

^f Four states only reporting, estimate.

water pollution from agriculture it is important to understand the issues and control options of agricultural production on both types of land.

Table 11.1 summarizes the extent of American agriculture production of selected food and fiber crops and their associated cash receipts or value during 1991.

Agricultural Philosophy and Perspective

The Family Farm. The average farm size in North America is about 200 ha in developed European countries, such as Italy, it averages about 10 ha. Although many farms are operated by a single family, that family often forms a corporation to protect its financial assets. A growing num-

ber of farms and ranches are owned by absentee landlords and operated on a cash-lease or crop-share agreement. About 30% of the agricultural land in the United States is owned by an absentee landlord, although much of this land may be farmed by a neighbor. An undetermined number of acres are owned by large corporations that may or may not specialize in agricultural products. Insurance companies, banks, chemical companies, petroleum conglomerations, etc., often hold agricultural land for the tax benefits it provides (American Farm Bureau Federation, 1989). Agricultural land is taxed at a lower rate than land zoned for development. Use of land for agricultural production at a net loss, when held in conjunction with other interests, can also provide tax advantages to the land owner (Levi, 1978). The small 15–40 ha family farm typical of the 1940s has basically disappeared due to economic pressures and population migration. It is very difficult for the small farmer or rancher to produce the volume of product required to be competitive and to pay for the capital investment farming requires. Regardless of the love of farming or the commitment to produce cheap food and fiber, agricultural producers must be good businesspeople. They must make their land-use decisions based on the economic conditions under which they operate (Brade and Uchtmann, 1985). This economic reality must also be recognized by persons wishing to control nonpoint source pollution from agricultural lands.

Commodity Programs and Price Supports: Short-Term Economic Gain versus Long-Term Environmental Stability. In the United States most farmers operate as independent small businesses, buying production inputs locally (and at retail cost), but selling products on the world market at wholesale prices. The profit margin for most of the commodity crops grown in the United States has steadily declined since the 1940s. Agricultural producers have compensated by increasing production. Farm policies of the USDA encouraged this trend in the 1970s and tried to cushion its effects in the 1980s and 1990s. Technological advances in equipment, hybrid varieties of crops and livestock, fertilizers, and pest control have supported improved efficiency and increased production (Tankersley, 1981). For example, during the 1920s wheat sold for \$3.10 per bushel and the tractor used on that farm cost \$6500; however, in the 1990s while wheat still sells for \$3.10 per bushel, tractor prices have risen to \$72,000 (Andrielenas and Eichers, 1982).

A combination of factors, including fluctuating land values, low market prices for commodity crops, high input costs, and a high debt ratio, have caused most producers to operate on an annual credit basis. Many utilize systems like the Commodity Credit Corporation (CCC—a federally managed corporate commodity brokerage and lending firm) to obtain

annual operating loans to purchase feed, seed, fertilizer, gasoline, and other annual inputs for the upcoming year, using as collateral their anticipated crop yield. This dependence on "upfront" financing limits farmer flexibility since many lending institutions will only make loans on certain high-value/commodity crops. Thus most producers manage and plan for short-term economic returns and specialize in one or two continuously grown crops or livestock. Little cash is available and little incentive provided for conservation work. Long-term planning and management for resource protection do not provide the short-term profits most large specialized producers need to stay solvent. Diversified operations that produce a variety of grain, cash, grass, animal, and specialty crops are often not as dependent upon the short-term cash flow from one particular crop or animal, and thus have more ability to operate in a resource-conserving manner. This diversification provides an economic cushion in case other crops fail or prices are low (Hosapple, 1992).

To compensate for the low sales price (in relation to the high input costs) for most commodity crops, the surplus of most crops produced each year by American farmers, and to provide consumers with somewhat stable low prices for food and fiber products and farmers with a more competitive edge in world agricultural markets (which are even more heavily subsidized than those in the United States), the USDA has developed a variety of price-support and other commodity or subsidy programs. These programs attempt to balance the crop to be planted with expected market needs and assist producers with deficiency payments that pay the difference between average sales price and average production price (U.S. General Accounting Office, 1990).

For many years a conflict existed within the USDA between conservation programs (designed to protect the resource base) and commodity programs (designed to stabilize farm income, etc.). In many cases commodity programs provided much more financial incentive for farms to overproduce and utilize land inappropriately than conservation programs provided incentives to protect the resource base (Sampson, 1981). In recent years these programs have been made more flexible, more careful of their environmental effects, and have decreased the actual cash or in kind (commodity) payments provided to each farmer. Recent farm legislation has also placed limits on total revenues collected. The newer environmental "consciousness" of recent American agricultural policy can be seen in the Food, Agriculture, Conservation and Trade Act of 1990, commonly referred to as the Farm Bill, which contain "cross compliance" provisions that make eligibility for participation in commodity programs dependent upon use of sound conservation practices (U.S. General Accounting Office, 1990).

Role of the European Community and Other World Market Systems

For national security and other reasons every country wishes to be food self-sufficient. In some European countries, Japan, and other countries where land available for agriculture is limited, subsidies and market price guarantees are used extensively to ensure national production of important food crops that are not possible to produce profitably. These heavily subsidized crops, combined with restrictive trade barriers and tariffs, make international crop and livestock marketing difficult (Harrington, Krupnich, and Peskin, 1985).

Agricultural Land-Use and Private Property Rights in the United States

There is a long tradition of personal freedom associated with private property land-use decisions in the United States. In the past, since there was plenty of land still available, land-use decisions that degraded soil and water were not considered by many to be a problem. Once one farm was depleted, settlers move on to more fertile, less worn out land. Thomas Jefferson wrote and spoke extensively on this trend at the end of the 1700s, saying that land stewardship was essential for maintenance of private property rights. The U.S. Constitution provides that the federal government cannot "take" land from an individual without just compensation. A number of court cases (mostly involving wetland issues) have been filed claiming that land-use control regulations (in those cases denial of dredge and fill permits for wetland conversion to shopping complexes) constitute a "taking" without just compensation since the economic value of the land for development far exceeds its agricultural or wetlands value. Americans in general are extremely resistant to being told what to do on "their" property. In Germany where land is at a premium and population densities high, citizens routinely recycle garbage and abide by garbage bulk limits for the public benefit without making it a private land right issue. More than any other factor, the concept of balance in private property rights with the good of society is at the heart of any agricultural diffuse-pollution dispute.

Private Use of Public Lands by Ranching

In the western United States a large percentage of the land is owned or controlled by the federal government. These lands are generally part of the National Forest System, the Bureau of Land Management System,

or a part of a public works project such as a Bureau of Reclamation irrigation and water supply reservoir and canal system. Under the multiple-use, sustained-yield, and greatest benefit management mandates of many federal land management agencies these lands are harvested for timber and leased for cattle grazing. A large percentage of these rangelands are in poor to fair condition, their natural productivity in decline, and their contribution to poor water quality significant (New Mexico Environmental Improvement Division, 1989). The rental rates for this property are usually below the comparable rates for leasing of private lands. Stocking rates allowed on this range have traditionally been more than the resource could support; however, in recent years livestock numbers have come more into balance. The traditional practices of continuous yearlong grazing in one undivided pasture continues to be a barrier to water quality protection. Ranchers on public lands are reluctant to install costly structural BMPs on property that they do not own, yet they have also resisted low-cost management BMPs (U.S. Forest Service, 1989).

Sustainable Agricultural Practice

As the epigraph for this chapter indicates, farm operators must realize that the production process must be in a harmony with the natural resources (land and soil, water), must not overburden them, and provide for sustainable agricultural production of future generations and not destroy the resources. Agriculture that destroys forests, drains wetlands, overuses fertilizers, and causes excessive soil loss has been and still is practiced in some parts of the world.

While based on concepts of stewardship, biodiversity, and mixed farming systems, the concept of sustainable agriculture should not advocate return to the past, but rather it focuses on an integrated approach that embraces biotechnology, engineering, and system studies for individual farming operations also using ecoregional concepts and knowledge of diversity. Sustainable agriculture employs a multiplicity of approaches, such as hybrids, sustainable pest and weed management, and soil fertility practices (Francis, Flora, and King, 1992; Committee on International Soil and Water Research and Development, 1991). Because agricultural diffuse pollution is primarily caused by the unwise use or overuse of lands, monocultural farming, and excessive use of fertilizers and herbicides, it is expected that sustainable agricultural practices should bring about a significant reduction if not even outright elimination of diffuse pollution.

AGRICULTURE AND ITS EFFECT ON THE ENVIRONMENT

Sources of Pollution

As pointed out in Chapter 1 diffuse pollution is a landscape phenomenon that is strongly affected by land-use activities. Chapter 1 also introduced the types of land uses in a dynamic concept and how they are related to the generation of diffuse pollution. Many factors affect pollution loads from agricultural operations. Among those are the types of land use, type of crops or animals, crop rotation, soils on which crop is grown, climatic conditions, farming technology, and irrigation and drainage. Proximity of polluting agricultural operations, such as feedlots and barnyards, to watercourses (Fig. 11.1) is one of the major causes of agricultural pollution. On the other hand, pollutant loads are reduced if buffer strips and measures that prevent cattle and cattle waste from entering the watercourses are installed.

Ritter (1988) reviewed the literature on the topic and compared loads



FIGURE 11.1. Proximity of polluting agricultural operations to a watercourse is a cause of agricultural pollution. (Photo: University of Wisconsin.)

TABLE 11.2 Comparative Magnitude of Diffuse Sources from Agriculture

Source	Total N		Total P	
	ml/l	kg/ha-yr	mg/l	kg/ha-yr
Precipitation (U.S.) ^a	0.7-1.3	5.6-10	—	0.05-0.1
Precipitation (Minn.) ^a	—	—	0.01	0.1
Precipitation (Ohio) ^a	2.0-2.8	12.8	—	—
Precipitation (Del.) ^a	—	45	—	1.5
Forest (Minn.) ^a	—	—	0.04-1.2	0.1
Forest (Ohio) ^a	0.05-0.9	2.1	0.01-0.01	0.4
Silviculture (Va.) ^a	1.1-1.8	2.7	0.01-0.2	0.3
Upland native prairie (Minn.) ^a	—	1	—	—
Grassland—rotate graze (Okla.) ^a	1.5-1.6	1.5	0.06-0.8	0.9
Grassland—continuous graze (Tex.) ^a	0.6	—	1.3	3.2
Grassland—44 kg N/ha (N.C.) ^a	—	8.4	—	—
Farmland (Ohio) ^a	0.09-3.1	5.1	0.02	0.06
Agricultural (Va.) ^a	1.1-1.8	2.7	0.02-0.3	0.3
Poorly drained coastal plain (Va.) ^a	1.7-2.3	1.6	0.4-0.7	0.2
Well-drained coastal plain (Va.) ^a	1.5-4.1	4.9	9-40	0.9
Corn—240 kg N/ha (Ga.) ^a	0.2-0.4	—	0.1-0.2	—
Feedlot runoff (Great Plains) ^a	3000-18,000	100-1600	47-300	10-620
Land applied dairy manure (Minn.) ^a	13-62	2.8-3.7	7.5-8.9	0.5-0.6
Land applied dairy manure (Wis.) ^a	—	2.8-8	1.8-4.9	0.1-17
Land applied dairy manure (S.C.) ^a	10-12	—	—	8.2-14
Irrigation return flow ^b	—	5-30	—	1-4
Subsurface tile drainage ^b	—	5-20	—	3-10

^a Compiled by Ritter (1988).

^b Compiled by Loehr (1974).

and concentrations from a variety of sources. Table 11.2 shows that nitrogen and phosphorus contamination may vary over a wide scale.

The major pollutants associated with agriculture include sediment, nutrients (especially N and P), pesticides and other toxins, bacteria or pathogens, and salts or salinity. Different types of agricultural land use are more likely to contribute certain pollutants than others. Runoff and subsurface water from agricultural and silvicultural operations is a source of the following pollution:

1. Dryland cropland: most often contains sediment, adsorbed nutrients, and pesticides
2. Irrigated cropland: irrigation return flow most often contains sediment, both adsorbed and dissolved nutrients and pesticides, traces of certain metals, salts and sometimes bacteria, viruses, and other microorganisms

3. Pasture land: contains bacteria, nutrients, sediment, and sometimes pesticides
4. Range land: can contain sediment, bacteria, nutrients, and occasionally metals or pesticides
5. Forest land: most often contains sediment, organic materials, and adsorbed nutrients due to logging operations
6. Confined animal feeding operations: contain bacteria, viruses, and other microorganisms; both dissolved and adsorbed nutrients, sediment, organic material, salts, metals
7. Specialty areas
 - a. aquaculture: most often contain dissolved nutrients, bacteria, and other pathogens
 - b. orchards or nurseries: contain nutrients (generally dissolved), pesticides, salts, bacteria, organic material, and trace metals
8. Wildlife land: can contribute bacteria and nutrients if wildlife populations become unbalanced

Just as significant as pollutants in the water column are the physical changes that occur in or adjacent to riparian areas as a result of a variety of agricultural land-use activities. These changes are responsible for much of the nonattainment of water quality standards and associated designated beneficial uses reported by state water quality agencies in their Nonpoint Source Assessments Reports and biennial Water Quality Inventories (U.S. EPA, 1990; U.S. EPA, 1992a). As states increase the use of biologic criteria in their water quality standards programs, these physical effects on biological communities will become more widely reported and used as measurements of maintenance of the biologic integrity of the nation's waters (U.S. EPA, 1990).

Some specific factors and activities that affect the diffuse loadings of pollutants from agricultural and silvicultural areas are:

- Livestock permitted uncontrolled access to riparian areas cause shear and sloughing of stream-bank soils and eliminate stream-bank vegetation, which results in changes to the surface and subsurface hydrology and morphology of the stream area, subsequent changes in stream flow, increases in water temperature, and reductions in dissolved oxygen content (Platts 1990; U.S. EPA, 1992a, 1992b; New Mexico Environmental Improvement Division, 1989).
- Crop production systems that plow fields right up to the edge of the streambank destabilize the banks causing them to collapse or for gully headcuts to form. The material from these caved-in streambanks and active gullies smothers remaining riparian vegetation and silts in the

adjacent stream channel. Aquatic organism species composition and richness changes as a result of loss of cover, food source, and changes in flow, temperature, and dissolved oxygen (DO) (Robillard et al., 1982; U.S. EPA, 1992a).

- Draining low-lying areas (generally wetlands) for crop production has had a profound effect on the physical state of the area. Loss of riparian vegetation through clearing, loss of a natural high water table from drainage, and loss of the flood buffering capacity of these areas results in a stream system that is more flashy when wet and more drought prone when dry. Loss of consistent stream base flow is one of the most detrimental changes affecting the reproduction habits of most aquatic organisms (U.S. EPA, 1992a; Platts, 1990).

Crop Production

On croplands pollutants arise from surface runoff (erosion of topsoil particles and irrigation return flows), interflow (mostly tile drainage and leachate of excess irrigation), and ground-water base flow (high nitrate content due to overfertilization). Often the reduction of pollution in one component may cause an increase in another component. Figures 11.2 and 11.3 show a typical example of the water quality consequences of diffuse pollution from farm runoff.

Erosion and soil loss by surface runoff and nitrate leaching into ground water are considered predominant sources of pollution from cropland. The disturbing activity associated with tillage increases the erosion potential of croplands. Conversely, increased hydrologic surface storage and permeability of tilled fields reduce hydrologic activity, which sometimes balances the increased erosion potential.

Alberts, Schuman, and Burwell (1978) and McDowell et al. (1989) reported that of the nutrient (N and P) losses (reported in Table 11.2), about 90% was associated with soil loss. Alberts et al. also noted that the losses of nutrients from croplands represent a relatively small portion of the applied fertilizer; nevertheless, the concentrations in the runoff almost always exceeds the criteria for preventing accelerated eutrophication of receiving bodies of water.

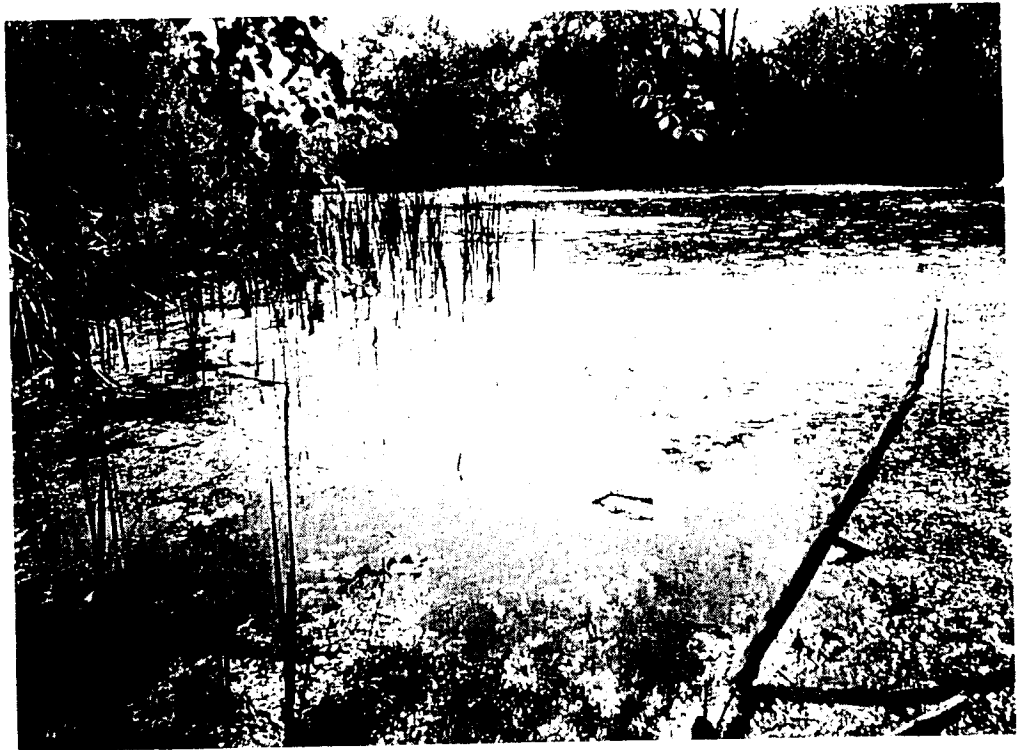
Dudley and Karr (1979) noted that bacterial contamination from agricultural drainage in the Black Creek Watershed of Indiana exceeded accepted standards for recreation. Organic pollutants and fecal contaminants originate from manure application, unconfined livestock, and septic tank drainage fields. Applying manure to frozen fields is a practice that is most damaging to surface-water bodies.

The contribution of pesticides of diffuse pollution is also of great concern. Atrazine and small amounts of other pesticides have been



FIGURE 11.2. Algal infestation of a drainage canal caused by agricultural drainage in the watershed of the Lagoon of Venice in Italy. (Photo: G. Bendoricchio.)

FIGURE 11.3. Poor water quality of a sluggish stream in an agricultural watershed. (Photo: University of Wisconsin.)



detected in drinking water supplies in, among other places, Tennessee (Klaine et al., 1988) and in the Po River Valley in Italy (Capodaglio, 1988). Measurements by Klaine et al. have shown that just over 1% of the total atrazine applied to a small agricultural watershed was lost with runoff; however, concentration in runoff reached as high as 250 $\mu\text{g}/\text{l}$.

Pasture- and Rangeland

Pasture- and rangeland account for the largest proportion of total land use in the United States and includes about 40% of all nonfederal land. Range- and pastureland is used directly for livestock production. The grazing practices include continuous and seasonal or rotational grazing. *Rangeland* typically refers to land covered with native or adapted grasses or shrub vegetation used for large-scale, but low-density animal operations in arid or semiarid western states. In the past, rangeland livestock grazing had adverse impact on fragile watersheds of the west and southwest United States.

Smolen, Jennings, and Huffman (1990) have pointed out that pastureland becomes a source of diffuse pollution when proper erosion control practices are not in place or when grazing livestock is allowed to approach or enter surface waters (Fig. 11.4). They also reported that sediment yield from pastures in the state of Washington was minimized when vegetation cover remained greater than 50%, regardless of animal trampling disturbances. Owens et al. (1989) investigated nutrient and sediment losses from unimproved pastures, that is, pastures not treated with fertilizer, weed control, or selected forage species, and found insignificant differences in nutrient losses from grazed as compared to ungrazed areas, although sediment losses were greater in grazed than in ungrazed areas.

From these findings it becomes apparent that overgrazing and permitting livestock to approach and enter watercourses are the major pollution activities on pastures and rangelands. If such activities are controlled, pollution from pastures and rangelands may be minimal.

Woodlands (Silviculture)

Undisturbed forests or woodlands represent the best protection of lands from sediment and pollutant losses. Woodlands and forests have low hydrologic activity due to high surface storage in leaves (interception), ground, mulch, and terrain roughness. Even lowland forests with a high ground-water table (containing wooded wetlands) absorb large amounts of precipitation and actively retain water and contaminants.

Uncontrolled logging operations (clear-cutting), however, disturb the forest's resistance to erosion (Fig. 11.5). Observations and records indicate that almost all sediment reaching waterways from forestlands originate from construction of logging roads (Haupt, 1959; Beschta, 1978)



FIGURE 11.4. Pollution of bodies of water located in rangeland and pastures is due to the practice of allowing cattle to enter the water body. (Photo: University of Wisconsin.)



FIGURE 11.5. The highest pollutant loads from silviculture (forestry) are caused by clear-cutting and road building.

and from clear-cuts (Beschta, 1978; Cheng, 1988). The chief sources are roads that disturb the natural drainage channels.

The potential for substantial water quality impact from increased aluminum concentrations resulting from whole tree harvesting was demonstrated by Lawrence and Driscoll (1988). The aluminum content of runoff is greatly increased when deforestation is caused by acidic deposition. (See Chapters 1 and 6 for a discussion of the so-called "chemical time bomb," which is an excessive loss of soil aluminum and its consequences caused by acidic precipitation.) Streams draining lowland forests with wetlands may have elevated organic and nutrient levels caused by leaching from soils. Also the dissolved oxygen may be depressed due to intensive decay processes taking place in the wetland portions of the forests. Despite these effects, pollutant loads from woodlands are near background levels.

Intensive Animal Production

Feedlots and barnyards can be the most intensive land uses in rural areas. With the advent of improved feeding methods and handling of ensiled materials, cattle are no longer put out to pasture but are held in relatively small areas. An obvious potential problem exists in feedlots, barnyards, and exercise areas where herds are confined on a year-round or seasonal basis (Fig. 11.6).

Commonly, feedlots themselves (beef, dairy, swine, sheep, and poultry) are classified as point sources, requiring in the United States a permit for discharge of pollution. Indirectly, however, the solid waste disposal methods and methods of management cause this subcategory to have a major diffuse-pollution impact. The majority of feedlot wastes reaching surface waters are transported by surface runoff. Loehr (1972) has pointed out that the quantity of runoff and the hydrologic activity of feedlot areas depend on the degree of perviousness of the lot and the permeability of the soils (which is greatly reduced by cattle trampling and high compacted organic content), antecedent soil moisture conditions, the number of cattle on the lot, surface storage characteristics, and rainfall intensity. In fact, permeability and infiltration characteristics in many feedlots are only remotely related to the native soil. The high organic content of the soil surface crust in a feedlot protects against erosion, and consequently sediment yields from feedlots are lower. Loehr (1972) conducted one of the first analyses of agricultural runoff pollution potential, reporting feedlot runoff concentrations of 1000 to 12,000 mg BOD₅/l, 2400 to 38,000 mg COD/l, 6 to 800 mg/l of organic N, and 4 to 15 mg/l of P. Runoff from feedlots is turbid and represents a high nutrient and organic shock loading to receiving waters.

Land application of feedlot waste has been traditionally a source of



FIGURE 11.6. Hog and beef feeding lot adjacent to a body of water. Bacterial contamination in the pond was 10 times the maximum for safe swimming and recreation use. (Photo: USDA, Soil Conservation Service.)

fertilizer as well as means of disposal. However, rates must be controlled to ensure stand persistence if perennial crops are grown as well as to protect water quality (Burns et al., 1987). In the follow-up study, Westerman et al. (1987) concluded that pollution hazards from disposal of manure exists for any rate, especially when rainfall occurs soon after application.

BEST MANAGEMENT PRACTICES: THEIR IMPLEMENTATION AND EFFECTIVENESS

Best management practices (BMPs) are methods and practices or combinations of practices for preventing or reducing nonpoint source pollution to a *level compatible with water quality goals*. Effective BMPs must also be economically and technically feasible (see Chapter 1 for a definition

of best management practices). BMPs can be classed as structural, vegetative, or management, and each class is somewhat more effective in controlling certain types of diffuse pollution than others. As discussed previously throughout this book diffuse water pollution occurs after a pollutant becomes available, detached, and is transported to the body of water. When selecting BMPs it is important that the pollutants and the forms in which they are transported is known. It is also important to realize that BMPs are not necessarily synonymous with resource management systems or conservation treatments. Many practices listed in the technical handbooks of land resource agencies were designed to protect soil or grass resources or to provide economic stability for farmers by improving productivity. For example, while conservation tillage is extremely effective in erosion control, it is not very effective in removing or controlling soluble pollutants from runoff or leached water (USDA, Soil Conservation Service 1988; Robillard et al., 1982; Haith and Loehr, 1979).

BMPs can be selected in basically two ways: to control a known or suspected type of pollution (for example, phosphorus or bacteria) from a particular source (runoff from a corn field or a dairy feedlot), or to prevent pollution from a category of land-use activity (such as agriculture row crop farming or containerized nursery irrigation return flow). (Robillard et al., 1982; Alexander, 1992). When selecting BMPs to solve a water quality problem the following process can be used (USDA, Soil Conservation Service 1988; USDA, Soil Conservation Service, 1980; Gordon and Hansen, 1989; Brach, 1990);

- Identify the water quality problem (e.g., annual summer algal bloom in lake)
- Identify the pollutants contributing to the problem and their probable sources (e.g., nutrients from septic systems adjoining the lake and runoff from a nearby horse pasture)
- Determine how each pollutant is delivered to the water (e.g., soluble nutrients from septic tank drain fields rise to the surface when the systems are overloaded and are carried to the lake by overland flow during rain storms or snow melt)
- Set a reasonable water quality goal for the resource and determine the level of treatment needed to meet that goal
- Evaluate feasible BMPs for water quality effectiveness, effect on ground water, economic feasibility, and suitability of the practice to the site

When selecting BMPs to prevent a problem a more technology-based approach can be followed. Such an approach divides agricultural land into

use categories such as irrigated cropland and rangeland, and specifies a minimum level of treatment necessary to protect the resource base. This type of planning has been used for years by the USDA in developing conservation plans for soil conservation. Technical guidance for the new Coastal Zone Management Act employs this basic "minimum treatment level" approach for diffuse-pollution prevention. Technically and economically achievable performance standards for each type of agricultural land use (and its associated pollutants) are used in this approach, which does not require that a cause-and-effect linkage between site-specific land-use activities and site-specific water quality problems be established. This approach is based on the rationale that neither the money nor the time is available to establish the complex monitoring programs that would be needed for such site-specific problem and source identification (U.S. EPA, 1993).

The following section discusses some individual BMPs and their effectiveness. In a fashion similar to urban practices, agricultural BMPs can be divided into source controls, hydrologic modifications, reduction of delivery, and storage and "treatment." Efficiencies of various practices have been summarized by Smolen, Jennings, and Huffman (1990); Brach (1990); U.S. EPA (1992a, 1992b) and others. Table 11.3 lists the factors for the selection of BMPs.

Cropping Practices: Conservation Tillage, Cover Crops, and Crop Rotation (Source Control Measures)

Definitions. These three types of cropping systems stress maintenance of vegetative cover during critical times, and their primary objective is to reduce erosion and hence soil loss. Conservation tillage is any tillage method that leaves at least 30% of the soil surface covered with crop residue after planting. The soil is only tilled to the extent needed to prepare a seedbed. Cover crops are close-growing grasses, legumes, or small grain crops that cover the soil during the critical erosion period for the area. Crop rotation is a system of periodically changing the crops grown on a particular field. The most effective crop rotations for water quality protection involve at least two years of grass or legumes in a four-year rotation.

No-till planting (Fig. 11.7) has been considered the most effective erosion-control practice applicable to agricultural lands. Planting is accomplished by placing seeds in the soil without tillage and retaining previous plant residues. The previous planting can be killed and weed control accomplished by the use of chemical herbicides. The plant and root residues provide the necessary surface protection. The magnitude of

TABLE 11.3 Selecting BMPs by Pollutant: Rules of Thumb

Pollutant	Methods of Control	Structural	Vegetative	Management
Sediment (TSS, cobble embeddedness, turbidity)	1. Control erosion on land and streambank	Terraces; diversions; grade stabilization structures; streambank protection and stabilization	Cover crops and rotations; conservation tillage; critical area planting	Contour farming; riparian area protection; proper grazing use and range management
	2. Route runoff through BMPs that capture sediment	Sediment basins	Filter strip; grassed waterway; stripcropping; field borders	
	3. Dispose of sediment properly			Beneficial use of sediment—wetland enhancement
Nutrients: N, P (nuisance algae, low dissolved oxygen, odor)	1. Minimize sources	Animal waste system (lagoon, storage area); fences (livestock exclusion); diversions; terraces	Range management; crop rotations	Range and pasture management; proper stocking rate; waste composting; nutrient management
	2(a). Uptake all that is applied to the land or contain and recycle/reuse (dissolved form control—commercial nutrients)	Terrace; tailwater pit; runoff retention pond; wetland development	Cover crop; strip cropping; riparian buffer zone; change crop or grass species to one that is more nutrient demanding	Recycle/reuse irrigation return flow and runoff water; nutrient management; irrigation water management
	2(b). Contain animal waste, process and land apply, or export to a different watershed (dissolved form control—animal waste)	Diversion; pit/pond/lagoon; compost facility	See 2(a)	Lagoon pump out; proper irrigation management

R0023667

	3. Minimize soil erosion and sediment delivery (adsorbed form control)	Terrace; diversion; stream-bank protection and stabilization; sediment pond; critical area treatment	Conservation tillage; filter strip; riparian buffer zone; cover crop	Nutrient management
	4. Intercept, treat runoff before it reaches the water (suspended form control)	See 1-3; water treatment (filtration or flocculation) for high-value crops	Riparian buffer zone	See 1-3
Pathogens (bacteria, viruses, etc.)	Minimize source	Fences		Animal waste management, especially proper application rate and timing
	Minimize movement so bacteria dies	Animal waste storage; detention pond	Filter strips; riparian buffer zones	Proper site selection for animal feeding facility; proper application rate of waste
	Treat water	Waste treatment lagoon; filtration	Artificial wetland/rock reed microbial filter	Recycle and reuse
Metals	Control soil sources		Crop/plant selection	Avoid adding materials containing trace metals
	Control added sources	Tailwater pit; reuse/recycle system	Crop selection	Irrigation water management; integrated pest control
	Treat water	Filtration	Artificial wetland/rock reed microbial filter system	
Salts/salinity	Limit availability			Drip irrigation
	Control loss	Evaporation basins; tailwater recovery pits; ditch lining; replace ditches with pipe	Crop selection; saline wetland buffer; land-use conversion	Irrigation water management

TABLE 11.3 (Continued)

Pollutant	Methods of Control	Structural	Vegetative	Management
Pesticides and other toxins	Minimize sources		Plant variety/crop selection	IPM; change planting dates; proper container disposal
	Minimize movement and discharge	Terrace; sediment control basin; retention pond with water reuse/recycle system	Buffer zone; conservation tillage; filter strips (adsorbed control only); wetland enhancement	Irrigation water management; IPM
	Treat discharge water	Carbon filter system (high-value crops)	Rock-reed microbial filter system/artificial wetland	
Physical habitat alteration	Minimize disturbance within 100 feet of water	Road and turnrow realignment; fencing/livestock water crossing facility	Buffer strips; riparian buffer zones	Proper grazing management, including limiting livestock access
	Control erosion on land Maintain or restore natural riparian area vegetation and hydrology	See sediment BMPs Streambank stabilization; channel integrity repair	Wetland enhancement	Proper grazing use and range management; limit livestock access

Sources: U.S. EPA (1993); Brach (1990); Alexander (1993a); USDA, Soil Conservation Service (1988).

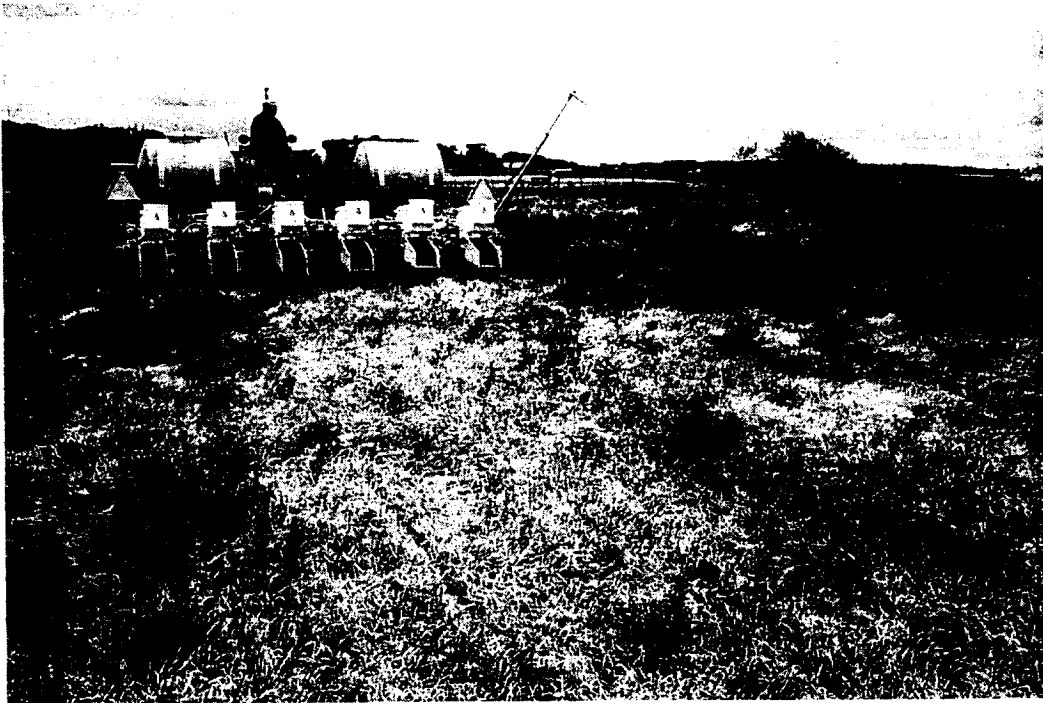


FIGURE 11.7. No-till planting. Seeds are placed in the soil and vegetation residues are left on the ground to minimize erosion. Weeds are killed by herbicide application. (Photo: University of Wisconsin.)



FIGURE 11.8. Strip-cropping. (Photo: USDA, Soil Conservation Service.)

TABLE 11.4 Effect of the No-Till Practice on the Magnitude of the Cover Factor (C) in the USLE

Tilling Practice	C Factor Production Level	
	High	Moderate
<i>No-till</i>		
Corn after sod ^a	0.017	0.053
Corn after corn	0.18	0.18
Soybean after corn	0.18	0.22
Grain	0.11	0.18
Corn after small grain ^a	0.062	0.14
<i>Conservation tillage^b</i>		
Corn—chisel plowing	0.19	0.26
Corn—fall chisel—spring disk	0.24	0.30
Corn—strip till—row zones	0.16	0.24

Source: Adapted from Stewart et al. (1975).

^aRemoved by chemical treatment.

^bResidue not removed.

the cover factor (C) of the universal soil loss equation (see Chapter 5) for no-till and conservation tillage practices are given in Table 11.4. However, a greater use of herbicides and sometimes lower yields are experienced on some soils.

Contour plowing is a widely recommended supporting practice in which plowing and crop rows follow field contours across the slope. It provides excellent erosion control for moderate rainstorms. The effects of contouring on the erosion-control practice factor (P) of the universal soil loss equation are given in Chapter 5.

Strip-cropping (Fig. 11.8) alternates contour-plowed strips of row crops and close-grown crops.

How the practices work to control pollution. Conservation tillage controls erosion and sediment by decreasing soil detachment, which also helps control the loss of adsorbed pesticides and nutrients. Cover crops also reduce erosion and sedimentation by decreasing detachment. The growing vegetation also retards movement of any detached particles and their adsorbed pollutants. When legumes are used as a cover crop or as part of a crop rotation, they provide a nitrogen source for the subsequent crop, thus minimizing the addition of commercial fertilizers. The cover crops also use nutrients that are otherwise lost through leaching during fallow periods. Crop rotations reduce erosion and associated adsorbed pollutant loads by improving soil structure, which decreases detachment,

but they also reduce the need for pesticide and nutrient applications since insect reproductive cycles are disrupted by changing host crops and nutrient needs more correctly balanced for a moderate level of production.

Effectiveness. Conservation tillage has been found to be highly effective in sediment reduction but of very little effect in controlling soluble nutrients and pesticides. Sediment reductions of 30–90%, total phosphorus reductions of 35–85%, and total nitrogen reductions of 50–80% have been found; however pollution potential could increase (and reduction efficiencies decline) if fertilizers are not soil incorporated in a CT system. Cover crops have been found to be 40–60% effective in reducing sediment, 30–50% in removing total phosphorus. The effectiveness of a crop rotation system is difficult to measure. Often the best estimate is an average annual load reduction over the entire rotation, so a higher reduction in one year coupled with a low reduction in another gives a moderate level for the entire rotation. Some rotations have been estimated to reduce nitrogen 50% and phosphorus 30% annually.

Berg et al. (1988) reported in a 10-year study that no-till management controlled erosion within acceptable limits on highly erodible graze-out wheat land. Conservation of nutrients in soil water was shown to be another benefit from no-tillage systems (Stinner et al., 1988). Parkin and Meisinger (1989) reported that no-till did not significantly affect subsurface biological activity and denitrification.

Strip-cropping systems can provide up to a 75% load reduction in sediment and up to a 50% load reduction in total phosphorus.

Integrated Pest Management

Definition. Integrated pest management (IPM) is a combination of practices to control crop pests (insects, weeds, diseases) while minimizing pollution. It uses traditional practices such as selection of resistant crop varieties and crop rotations, modified planting dates along with sophisticated pesticide application management, which includes pest scouting, minimal application rates, most judicious timing, and selection of the least toxic and least persistent chemical.

How the practice works to control pollution. IPM works mainly by decreasing the amount of pesticide or crop-protection chemical available and by selecting the least toxic, least mobile, and/or least persistent material, it decreases the potential for pollution.

Effectiveness. Complete studies on the effectiveness of IPM show varying results depending on the exact type of pesticide, soil, moisture, and crop conditions. Some estimates are extremely high and others low.

Nutrient Management

Definition. Nutrient management is a series of practices designed to decrease the availability of excess nutrients through improvements in timing, application rates, and location selection for fertilizer placement. It also involves fertilizer-type selection, crop-variety selection, and conservation cropping systems (such as rotations, cover crops, and strip-cropping). Nutrient management is currently based on the limiting nutrient concept (that fertilizer application rate should be based upon the nutrient most needed by the plant for optimum growth—most often nitrogen). New research is showing that in some cases, phosphorus, thought to be relatively immobile and strongly sorbed to the soil can become available and contribute to water quality problems. When animal waste is used as a fertilizer in areas with phosphorus water quality problems (nitrogen to phosphorus ratios cannot be manipulated with animal waste), the EPA recommends that phosphorus be considered as the limiting nutrient.

How the practice works to control pollution. Nutrient management most effectively limits the availability of nutrients through a lowered or more precise application rate.

Effectiveness. Nitrogen and phosphorus loss can be reduced by 20–90%. This practice is especially effective in controlling the soluble phases of these nutrients.

Terraces and Diversions

Definition. A *terrace* is an earthen embankment, channel, or a combination of ridge and channel constructed across a slope to intercept runoff. Terraces reduce the slope effect on erosion by dividing the fields into segments with lesser or even near-horizontal slopes (Fig. 11.9). This practice therefore reduces the slope-length factor of the USLE. Terraces can be level or graded. Level terraces store or pond water for infiltration or outlet through an underground conduit. Graded terraces divert water to a suitable outlet. *Diversions* are a form of graded terrace, but are defined as a channel constructed across the slope with a supporting ridge on the lower side.

How the practice works to control pollution. Terraces change the effective slope of the land, which decreases runoff velocity. Soil particles and adsorbed pollutants are thus not transported from the field. Terraces that retain all water applied to the field are also effective in reducing the loss of dissolved pollutants. Graded terraces or those with outlets allow runoff to leave the field and therefore do not control dissolved or suspended pollutants well. Diversions transport excess water from areas needing protection to sites suitable for disposal. They can prevent water from flowing across areas where pollutants are available, such as highly ero-



FIGURE 11.9. Parallel terraces. (Photo: USDA, Soil Conservation Service.)

dible soils, feedlots, or areas where fertilizers or pesticides are mixed, stored, or applied at high rates. Diversions must be provided with a stable outlet, such as a grassed waterway or buffer zone.

Effectiveness. Level terraces can remove up to 95% of the sediment, up to 90% of its associated adsorbed nutrients, and between 30% and 70% of dissolved nutrients. Diversions can reduce sediment movement by 30–60% and adsorbed nutrients by 20–45%.

Example 11.1: Effect of Conservation Tillage

A corn-growing field on a 10% slope, plowed up and down, had estimated erosion losses of soil, organic matter, and P of 20 tonnes/ha-yr, 1.0 tonne/ha-yr, and 30 kg/ha-yr, respectively. Estimate the effect of: (1) No-till planting, (2) sod-based rotation, and (3) terracing on the erosion losses.

Solution Most of the loss (80%) occurred during the critical erosion period (i.e., a period of 2 months after spring plowing). The overland flow length (downslope) is 200 m.

For reference practice of conventional plowing up and down the slope, the average cover factor of USLE for the critical period is $C = 0.8$ and

afterward $C = 0.15$ (see Table 5.4 for appropriate magnitudes). The erosion-control factor $P = 1.0$. In the no-till practice, crop residues and root systems from the previous harvest remain on the soil as a protective cover. This can reduce significantly erosion during the critical period. If the preceding crop was small grains, the C factor would approximate 1.0. Hence, the effect of this practice on the soil and pollutant loss is

$$\begin{aligned} \text{Percent loss reduction} &= \left[1 - \frac{\text{weighted } C \text{ with no-till}}{\text{weighted } C \text{ without no-till}} \times 100 \right] \\ &= \left[1 - \frac{0.8 \times 0.1 + 0.2 \times 0.1}{0.8 \times 0.8 + 0.2 \times 0.15} \right] 100 = 87.8\% \end{aligned}$$

If all other variables remain the same (a crude and often unrealistic assumption), the soil loss with no-till practice should be

$$A_s = (1 - 0.878) \times 20 = 2.44 \text{ tonnes/ha-yr}$$

and similarly, organic matter and P losses become

$$A_{OM} = (1 - 0.878) \times 1.00 = 0.122 \text{ tonnes/ha-yr}$$

and

$$A_P = (1 - 0.878) \times 35 = 4.27 \text{ kg/ha-yr}$$

With sod-based rotation, sod is planted once during the rotation cycle. For the purpose of this example, assume that the rotation cycle is 2 years and that the sod is plowed during the spring season by conventional plowing methods. The C factor for sod during the off-year is $C = 0.01$. In the following year when corn is planted, the sod residues may reduce C to a value of 0.6. Average soil loss reduction over the 2-year cycle is

$$\begin{aligned} \text{Percent loss reduction} &= \left[\frac{1 - (0.01)/(0.8 \times 0.8 + 0.2 \times 0.15) + 1}{2} \right. \\ &\quad \left. - \frac{(0.8 \times 0.6 + 0.2 \times 0.15)/(0.8 \times 0.8 + 0.2 \times 0.15)}{2} \right] \times 100 \\ &= 61.1\% \end{aligned}$$

If slopes are reduced by terracing, the slope-length factor (LS) is decreased. If terraces are built 1 meter high on a 10% slope, the overland flow length is reduced to 10m. From Figure 5.14, the original LS factor

for a 200 m overland flow distance and 10% slope is 3.7. Having terraces 10-m wide with an average slope of 1% reduces the LS factor to 0.1. Hence the soil and pollution loss reduction becomes

$$\begin{aligned} \text{Percent loss reduction} &= \left[1 - \frac{(\text{LS}) \text{ factor after terracing}}{(\text{LS}) \text{ factor before terracing}} \right] \times 100 \\ &= \left[1 - \frac{0.1}{3.5} \right] 100 = 97.1\% \end{aligned}$$

Although the cover factor or slope-length factor seem to be the predominant factor for the situation discussed in this example, other factors should be included in a more exact analysis. For example, reduction of the slope by terracing increases surface storage (Fig. 3.3); no-till practices, on the other hand, reduce surface storage and permeability. For a more complex analysis, use of mathematical models is the best approach to evaluate the combined effects of several factors. Such models are described in Chapter 9.

Critical Area Treatment (Grade-Stabilization Structure and Critical Area Planting—Pollutant Delivery Control)

Definition. A grade-stabilization structure is used to control the grade and gully-head cutting in natural or artificial channels. It can provide an outlet for other conservation practices. Structures can be constructed of metal, wood, rocks, concrete, or earth. The area surrounding the structure must be suitably stabilized. Critical-area planting involves planting suitable vegetation (such as grass, trees, or shrubs) on highly unstable sites such as gullies, denuded streambanks, dams, and embankments where vegetation is difficult to establish by the usual planting methods. Intensive methods, such as use of erosion blankets, tiedowns, gabions, mulches, hydromulching, increased seeding rate, and hand planting, are often needed.

How the practice works to control pollution. Grade-stabilization structures reduce water velocity, thus preventing additional sediment detachment and decreasing the transport capacity of the water. They also often act as traps, reducing additional sediment transport. Sediment adsorbed nutrients are thus controlled much more effectively than dissolved pollutants. Planting vegetation on other critical areas works in the same manner by slowing velocity and decreasing detachment and transport.

Effectiveness. Grade-stabilization structures can result in a 5–75% reduction in suspended solids leaving the unstable area, but data on the area below the structures are unavailable. Data on the vegetation methods are

also lacking, but soil-erosion estimates using soil-loss models could be used as a rough approximation of effectiveness.

Sediment Basins, Water and Sediment-Control Basins, and Detention—Retention Ponds

Definition. All earthen embankments are generally designed to trap and store sediment or other pollutants. Sediment basins are used to preserve the storage capacity of downstream reservoirs, canals, etc., and often serve an entire small watershed (Fig. 11.10). Water- and sediment-control basins are used to reduce on-site water, control gully erosion, and protect downstream water quality. Detention ponds are used to collect runoff from agricultural fields for storage and pollution control.

How the practice works to control pollution. These structures stop water flow, allowing heavier particulate to settle out. Because the sediment basins and water- and sediment-control basin structures are usually drained and the water released downstream or across land, soluble pollutants are not controlled. Detention ponds are not drained and often support aquatic vegetation. Available phosphorus and nitrogen can be taken up by these plants, and it is believed that nonpersistent pesticides will have time to degrade. Design methods for sediment detention basins were presented in Chapter 10.



FIGURE 11.10. Retention pond—sedimentation basin. (Photo: S. Alexander.)

Effectiveness. Sediment basins can remove 40–87% of the incoming sediment, up to 30% of the adsorbed nitrogen, and 40% of the total phosphorus. Effectiveness of detention–retention ponds, especially those with aquatic vegetation, is generally higher. As is pointed out in Chapter 13, sedimentation basins are not effective for the removal of toxicity or dissolved toxic compounds.

Animal Waste Storage and Treatment Facilities and Methods (Lagoons, Storage Bins and Ponds, Composters), Also Known as Animal Waste Management System

Definitions. A combination of practices for storing manure and other wastes, controlling runoff from feedlots, managing removed waste and runoff, and disposing of animal carcasses, or other waste. Waste storage ponds and structures hold waste until it can be land applied without causing a water-pollution problem. Waste treatment lagoons biologically treat liquid waste to reduce the nutrient and BOD content. Composters are structures to biologically degrade dry waste or dead animals (especially poultry carcasses). Storage areas, lagoons, and composters all must be emptied and their contents disposed of properly. Management BMPs are then applicable. The waste can be land applied at the proper rate, time, and in the proper way, or removed from the farm for proper use or disposal elsewhere.

Generally, animal wastes are classified as point sources and require a permit in many states. However, pollution is generated by runoff from the contaminated premises and typically requires runoff control BMPs in addition to treatment and/or safe disposal.

How the practices work to control pollution. Runoff-control systems for feedlots and manure storage are governed by two basic principles. First, all clean water originating outside the feedlot or storage area should be diverted so that it does not come in contact with the contaminated soil. Second, the water originating inside the feedlot should be disposed of in a way that minimizes its pollution potential. To meet these objectives, four components should be considered in the design of runoff pollution-control systems: clear water diversion, runoff collection, runoff containment, and controlled disposal (Fig. 11.11).

Clear water diversion systems include terraces, which direct water from upland watersheds away from the feedlot or barnyard, and gutters with downspouts on buildings, which prevent roof drainage from entering the area. When a natural waterway (creek) crosses a feedlot (Fig. 11.11), it may be necessary to pass it through a culvert or relocate the feedlot.

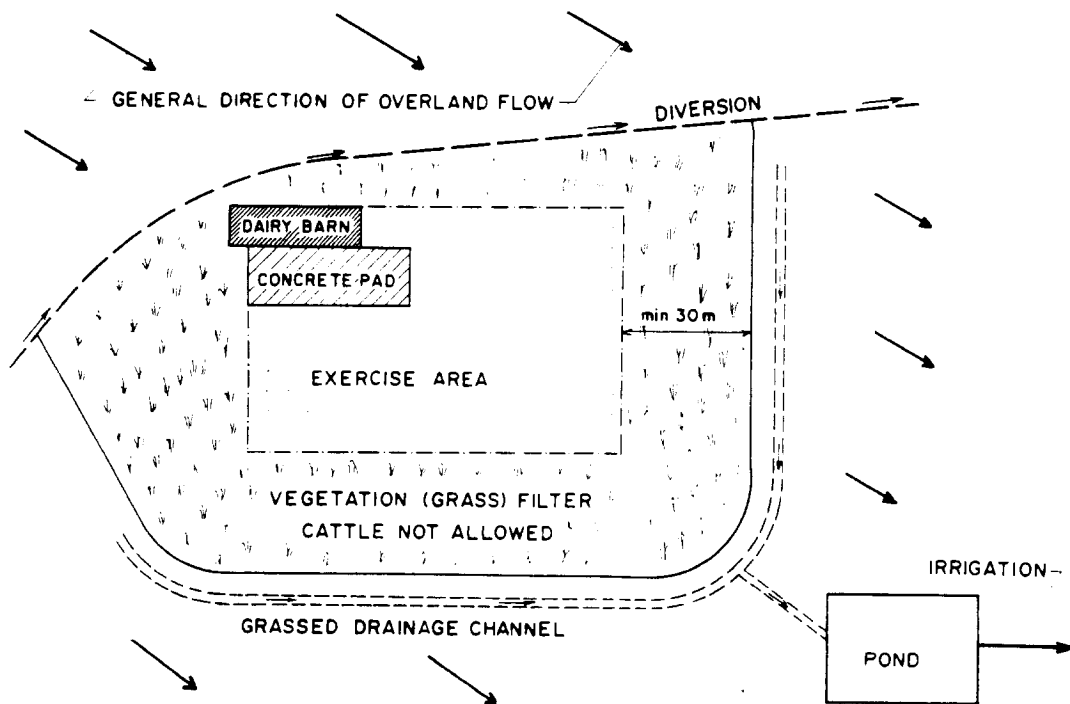


FIGURE 11.11. Best management practices for containment of feedlot runoff.



FIGURE 11.12. Large two-stage lagoon for containment of dairy animal waste. The first stage provides solids separation. (Photo: S. Alexander.)

Collecting runoff with a system of curbs, gutters, or terraces prevents it from directly entering defined watercourses and concentrates polluted runoff for treatment before disposal. Runoff can be contained for short periods of time in settling channels or basins, vegetative filters, or infiltration areas. Holding ponds (Fig. 11.12) provide long-term storage and they also act as facultative (aerobic-anaerobic) lagoons (see Chapter 10) and provide sedimentation and biological treatment (Loehr, 1972).

Often waste storage ponds or structures contain all pollutants, thus allowing none to enter the receiving bodies of water (of course, they must be pumped out later and disposed of). With waste treatment lagoons the detention time allows bacteria to die off and nitrogen to volatilize, and the biological process within the lagoon decrease the oxygen demand of the effluent. Composting of solid wastes from feedlots allows some slight mineralization of nitrogen. The heat and enzymatic activity generated during the composting process usually kills most bacteria. Composting is mainly used to reduce the volume and odor of the waste and improve its handling and management. Proper management is crucial to the effectiveness of each component of the system because it limits the availability of nutrients and pathogens and the detachment potential of the waste pollutants.

Effectiveness. Animal waste systems (all practices combined) have been estimated to reduce pollutant loadings as follows (averages): bacteria 74%, sediment 64%, total nitrogen 62%, total phosphorus 21%. Individual practices in the system vary widely. For example, a waste treatment lagoon (and proper land application) reduces bacteria by 65–100%, dissolved phosphorus 10–69%, and total nitrogen 32–91%. Most BMP recommendations suggest applying manure to satisfy crop nitrogen needs. This practice often results in excess phosphorus being applied.

Livestock Exclusion (Fences)

Definition. Excluding livestock from areas where grazing or trampling will cause erosion of streambanks and lowering of water quality by livestock activity in or adjacent to the water. This is generally accomplished by permanent or temporary fences or herding.

How the practice works to control pollution. By keeping animals out of streams, deposition (availability) of fecal material (nutrient and bacterial source) will be reduced, turbidity from in-stream trampling will be eliminated (particles will not be detached), and streambanks will not be denuded (sediment will not detach), but will instead provide a filter for runoff (transport and delivery of sediment and adsorbed materials will be

reduced), and if the riparian area is large enough, uptake some of the nutrient load (availability of dissolved pollutants will be decreased).

Effectiveness. Reductions of 50–90% of suspended solids and total phosphorus have been reported.

Filter Strips and Field Borders

Definitions. Filter strips and field borders utilize strips of closely growing vegetation, such as sod or bunch grasses or small grain crops, with the primary purpose of water quality protection. Filter strips are generally placed between the agricultural land being used (for example, cropland fields, animal waste application areas, or forests) and the body of water to be protected (Fig. 11.13). They are designed to remove sediment and other pollutants from sheet runoff. Field borders normally consist of perennial vegetation planted at the edge of fields to control erosion, regardless of their proximity to water. In contrast, strip-cropping, discussed earlier, involves planting straight or contoured strips of grass or closely growing crops in alternating bands between strips of row crops, and its primary purpose is to reduce soil losses.

How the practices work to control pollution. Both practices interfere with the transport of sediment, slowing water velocity, and allowing the material and any adsorbed pollutants to drop out. If filter strips are wide enough (detention time is adequate) and planted with appropriate



FIGURE 11.13. Vegetated buffer strips between plowed field and drainage ways. (Photo: University of Wisconsin.)

vegetation (high-nutrient-demanding species), they can uptake some nutrients, thus decreasing their availability. Field borders are effective in preventing detachment of soil particles in the areas covered by the border, but have little effect on controlling erosion or pollutant detachment or transport from the fields they surround. Designs and design considerations for grass filter strips were discussed in Chapter 10.

Effectiveness. Filter strips are very effective in removing sediment and sediment-bound nitrogen (about 35–90%), but much less effective in removing phosphorus, fine sediment, and soluble nutrients, such as nitrate (14%) or orthophosphate (5–50%). During the active growing season some filter strips have not been effective in removing phosphorus. Field borders are most effective in sediment removal and erosion control. Pollutant reductions for borders alone have not been fully evaluated. The most effective systems rely upon perennial grasses. New research appears to indicate that deep-rooted native bunch grasses in strip-cropping systems are also effective in “recovering” leached nitrogen that has escaped below the roots of conventional row crops.

Wetland Rehabilitation and Development, and the Beneficial Use of Sediment

Definitions. Rehabilitation and development involve restoring, rehabilitating, or enhancing existing wetlands to function as self-sustaining ecosystems that process, remove, transfer, and store pollutants. Beneficial use of sediment is a newer technology designed to utilize clean sediment recovered from dredging operations to mitigate damage to wetlands in other places by enhancing or rebuilding degraded wetlands. Wetland management and the types of constructed and restored wetlands are presented in Chapter 14.

How the practices work to control pollution. The low slopes of wetlands serve to slow the velocity of water and allow sediment to precipitate out (decrease transport capacity of water), the long retention time promotes bacterial die-off, and the wetland vegetation uptakes and uses nitrogen (decreasing availability) and provides a source of carbon for soil microbial action. This carbon fuels the immobilization of some phosphorus and nitrogen by microbial action and the transference of nitrogen by denitrification. Utilization of dredged materials to mitigate damage to degraded wetlands solves two environmental problems: (1) it provides a safe and effective disposal site for materials that previously have been dumped in off-shore ocean disposal sites or inland in poorly functioning sediment containment pits, and (2) improves wetlands that have been depleted of

their needed annual sediment replenishment/nourishment by changes in hydrology.

Effectiveness. Wetlands are 80–90% effective in sediment removal, 40–80% effective in nitrogen removal (depending upon location of riparian vegetation in relation to the systems hydrology; those wetlands located in the upper reaches of the watershed appear to be more effective), and 10–70% effective in phosphorus removal, also depending upon location. See Chapter 14 for a more detailed discussion.

Riparian Buffer Zones

Definition. Vegetated area along a body of water containing a complex assemblage of organisms and their environment, typically part of a riparian system (Fig. 11.14). It can be a complete ecosystem or function as an ecotone between aquatic and terrestrial ecosystems, but maintains a distinct set of vegetational and soil characteristics. The area is maintained in its native vegetation state at widths sufficient for pollution-control functions.

How the practice works to control pollution. The grasses and low vegetation in the zone filters both surface and subsurface flow, while the roots of taller vegetation take up and transform pollutants and nutrients from shallow ground water.

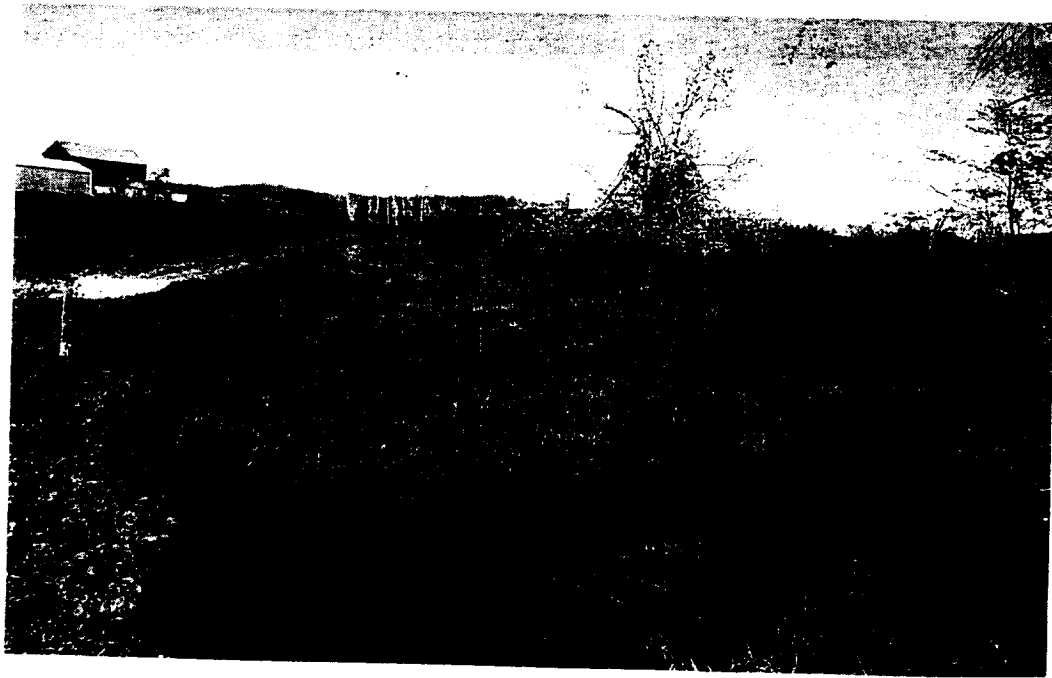


FIGURE 11.14. Riparian buffer zone separates farm operations from a watercourse. (Photo: University of Wisconsin.)

Effectiveness. Sediment removal efficiencies of 80–90%, total phosphorus removal efficiencies of 50–75%, and total nitrogen removal efficiencies of 80–90% have been measured. Pesticide removal has been documented but not adequately quantified.

Irrigation Water Management

Definition. Irrigation water management (IWM) is a combination of practices that control irrigation water to prevent pollution and reduce water loss. It also applies to chemigation application of chemicals (pesticides or fertilizers) through the irrigation system. IWM includes proper scheduling, efficient application, efficient transport systems, utilization and reuse of tailwater and runoff, and management of drainage water.

How the practice works to control pollution. Seepage-control practices (lining ditches with concrete or converting to pipe conveyances) greatly reduces leaching losses and decreases the availability of salts in canals for application (and runoff) from fields. Tailwater pits (which are pits, usually lined, to catch rainwater runoff or the water that runs out of the furrows at the end of a field) retain any pollutants and keep them from entering any bodies of water. Improvements in management include the efficiency of water application and prevention of excess runoff that carries with it dissolved nutrients and pesticides.

Effectiveness. IWM is particularly effective in reducing nitrogen and pesticide loadings to ground water and salt loading to surface waters. Tailwater pits are about 50% effective in sediment removal and can be highly effective in nutrient and pesticide pollution control as long as the collected water is reused and not discharged.

Stream-bank Stabilization

Definition. Structural or vegetative methods to protect the streambank from erosion. Riprap, concrete, wood, or rock gabions can be used, but vegetative stabilization is the most effective for pollution control as long as it is capable of withstanding the hydrology. See Chapter 15 for more information on stream-bank stabilization techniques.

How the practice works to control pollution. Eroding streambanks not only contribute sediment to the water and streambed but also cause increased water temperatures (due to the loss of the riparian vegetation that shades the stream). Loss of floodwater storage or hydrologic assimilative capacity to regulate flow is also an effect. Stabilization methods work to eliminate the detachment of soil particles and to restore one part of the hydrology of the system.

Effectiveness. Effective stabilization can reduce sediment loading by 90%, but is highly dependent upon the type of vegetation used and the stability and width of the reclaimed area. When combined with riparian area restoration, removal rates of 50–70% for phosphorus and 80–90% for nitrogen have been found. When planted in conjunction with a grass filter strip, load reductions of 80% TSS, 60–90% total nitrogen, and 30–90% total phosphorus are possible.

Rock–Reed Microbial Filters

Definition. A long shallow hydroponic plant/rock filter system that treats polluted waste and waste water. Combines horizontal and vertical flow of water through the filter, which is filled with aquatic and semiaquatic plants, microorganisms, and provides a high surface area of support media, such as rocks or crushed stone (see Chapter 14 for a description of submerged bed systems; also, see Jones, 1990).

How the practice works to control pollution. Symbiotic relationship between the microbes on the plant roots and the surface media synergistically effect the degradation of pollutants. Rocks and roots also provide filters and plants further contribute by translocating oxygen and by adsorbing metabolites from the microbes. Thus each system uses the other's waste in a cyclic and sustainable system. If the filters are used for removal of heavy metals and certain other toxins, harvesting of plant materials may be periodically necessary. These materials must be degraded (composted) and the residue tested before proper disposal.

Effectiveness. Biological oxygen demand has been decreased from 110 mg/l to 10 mg/l in a 24-hour retention time, and toxic organic compounds from 9 mg/l to 0.05 mg/l in a 24-hour retention time. The systems have been used for toxic heavy metals and on some radioactive elements, but efficiency figures are not available. The systems provide complete disinfection.

Range and Pasture Management

Definition. Systems of practices to protect the vegetative cover on improved pasture and native rangelands. It includes such practices as seeding or reseeding, brush management (mechanical, chemical, physical, or biological), proper stocking rates and proper grazing use, and deferred rotational systems. Rangeland is land generally in native grass and managed as range. Pastures are normally seeded with an improved grass variety and managed with agronomic practices (liming, fertilization, irrigation).

How the practices work to control pollution. Keeping land permanently covered in high-quality (closely spaced) vegetation decreases soil loss to negligible amounts (detachment and transport are interrupted), and sediment and adsorbed pollutants are not lost from the land surface. Since rangeland is not fertilized or irrigated, it can represent a low input system if properly managed.

Effectiveness. If correctly managed, range and pastures function like filters to reduce the load of nitrogen, phosphorus, and sediment up to 80%.

ROLES, RESPONSIBILITIES, AND PROGRAMS OF FEDERAL, STATE, AND LOCAL AGENCIES AND GROUPS

Within the United States there are more than 31 federal agencies, and numerous state and local agencies with programs that can be used for the control of diffuse pollution from agriculture (U.S. General Accounting Office, 1990). It is important to understand the responsibilities and opportunities available from each of these agencies. Effective control of diffuse pollution from agriculture requires that these programs be coordinated, nonduplicative, and focused on the goal of water quality protection. Table 11.5 lists programs by various agencies. The large number of agencies with overlapping responsibilities indicate fragmentation of efforts and often a lack of coordination.

BALANCING AGRICULTURAL PRODUCTION WITH ENVIRONMENTAL PROTECTION

Maintaining an adequate and available supply of food through domestic production is a goal of almost every country. Such a goal helps ensure national self-sufficiency and provides some degree of internal stability. Maintaining a clean and safe environment through proper resource management is also becoming a priority for most countries. It is possible to protect the environment while maintaining and sometimes enhancing agricultural production. Most consumers want plentiful, inexpensive, and safe food and fiber, as well as clean water and air, all without making personal sacrifices in time, money, or quality of life. This is not possible. What is possible is a series of changes in values, economics, and methods of agricultural production and pollution control. Needed are a modification in the attitudes of citizens, agricultural producers, and legislators, a strengthening of the pollution-control laws and programs, advances in agriculture production technologies, and changes to agricultural programs

(Text continues on page 723.)

TABLE 11.5 Summary of Programs in the United States that Could Be Used for Control of Diffuse Pollution from Agriculture

Agency and Program	Program Descriptions and Agency Responsibilities	Resources Available and Possible Roles
U.S. Environmental Protection Agency (EPA)	Administers educational and regulatory programs designed to protect the environment (prevent and control pollution). Provides environmental assessments, water quality monitoring, regulations and regulatory oversight, education, planning, technical assistance, grants and loans for pollution control. <i>Works mainly with state, federal, regional, and local agencies on pollution-control efforts</i>	Staff, information and data, laboratories and research facilities, grants and loans for pollution control, educational materials, monitoring equipment. <i>Offices located in 10 regional centers and Washington, D.C.</i>
EPA—permits	NPDES permits for confined animal feeding operations, enforcement for noncompliance	Staff for technical assistance with modeling and permit drafting, site inspections and compliance monitoring. Funds for special studies or projects
EPA—pesticides	Regulation of pesticide labeling and registration, which includes application rates, allowable crops and pests, environmental and human health cautions, disposal procedures. Licensing of restricted use pesticide applicators	Staff for review of research results, assistance with strategic planning, education and training, oversight of enforcement procedures of states. Funds for special projects and studies
EPA—water quality	Overall water quality planning and management through the following programs: <ol style="list-style-type: none"> 1. Nonpoint Source Control: program that oversees and approves state development of water quality assessments and implantation of management programs designed to control NPS. Directs funds to high-priority watershed or projects 2. Clean Lakes: program provides funds to restore or enhance publicly owned lakes 	Staff for technical assistance to state and local agencies, review and approval of state programs, research and special studies. Grants to states for most water quality protection activities, educational materials, and programs. Funds for special studies or projects

R0023687

3. Water Quality Standards: program provides technical assistance in developing numeric, narrative, and biological limits (standards) to protect water quality and its use
4. Coastal programs: a number of different programs and initiatives designed to assess coastal resources and study ways to protect coastal waters, including the National Estuary program. Administers the new CZMA

EPA—ground water

Administers the Sole Source Aquifer Protection program, provides technical and programmatic assistance to state wellhead protection programs

Staff for technical assistance, funds for special studies

EPA—wetlands

Oversight of the Corps of Engineers on wetlands dredge and fill permits, takes enforcement actions for illegal wetlands filling, technical support for wetlands delineations

Staff for oversight and enforcement activities, monitoring of wetland status, health and trends, and funds for special studies, educational materials, and programs, data

EPA—monitoring and surveillance

Environmental assessment, data analysis, oversight of state monitoring programs, special studies and agency research, EPA lab and Office of Research and Development coordination

Staff for technical assistance to states and citizens on monitoring programs and projects, special studies and data analysis upon request, water quality monitoring at select locations

EPA—drinking water

Regulates public drinking water supplies and suppliers, special studies on human health and risk, develops drinking water criteria and MCLs (maximum contaminant levels). Administers a special program that allows watershed treatment work to be done to decrease pollution loads to drinking water supplies if installation of the BMPs is cheaper than the water treatment method needed

Staff for technical assistance in setting drinking water standards, special studies, oversight and compliance monitoring of public water supplies and suppliers

TABLE 11.5 (Continued)

Agency and Program	Program Descriptions and Agency Responsibilities	Resources Available and Possible Roles
EPA—Office Research and Development (ORD)	Conducts basic and applied research to support EPA mission, including biological and physical studies on fate and transport of environmental contaminants and ecosystems at large	Reports, data, maps, monitoring equipment, study, and demonstration sites, staff for technical assistance in interpreting research results. <i>Laboratories and research stations located throughout the country</i>
U.S. Department of Agriculture (USDA)	Stabilize and support the efficient production, marketing, and distribution of food and fiber. In addition to commodity and public welfare programs, administers a number of conservation programs designed to assist private and federal land owners or managers in natural resource conservation and multiple-use management. <i>Works mainly with private individuals on improving resource management</i>	Staff, technical assistance, information and data, educational materials, cost-share funds, engineering equipment. <i>Unless otherwise indicated each agency has field offices located in almost every county or parish, state offices in each state, and a Washington, D.C. office</i>
USDA—Multiple agency administration of 1985 and 1990 “Farm Bill” programs: <ol style="list-style-type: none"> 1. Conservation Reserve Program, 2. Wetlands Reserve Program, 3. Sustainable Agricultural Research and Education Program, 4. Conservation cross-compliance (sodbuster and swampbuster) 5. Water Quality Incentives Program 	<ol style="list-style-type: none"> 1. Program to conserve/protect highly erodible or other environmentally sensitive land from production by putting it in permanent vegetative cover through 10-year easements and annual rental payments 2. Program available only in pilot states to return drained wetlands to wetland status and protect existing wetlands. Uses same easement/payment method as CRP 3. A practical research and education and grant program to promote lower input methods of farming 	<p>In most cases responsibilities within these programs are divided between departments of USDA as follows:</p> <p>SCS: technical assistance in planning, design, and implementation of BMPs</p> <p>ASCS: Administrative oversight of program and cost-share funding disbursement</p> <p>CES: Education and information about the variety of conservation and economic choices available</p> <p>CSRS: Research, data, and the results of</p>

R0023689

	<ol style="list-style-type: none"> 4. A quasi-regulatory program that denies subsidy payments to farmers who plow out highly erodible land or drain wetlands 5. A watershed treatment program designed to improve or protect soil and water resources in watersheds impacted or threatened by NPS pollution 	demonstration field trials of new technologies
USDA—Soil Conservation Service (SCS)	Technical assistance on the planning, site specific design and installation and management of soil and range conservation, animal waste, and water quality management systems and special land and water resource assessments and inventories. Cost-share funds for installation of BMPs on private lands are available from some of the programs listed below.	Staff and equipment in field offices for technical assistance including engineering designs, survey work, and planning for water resource protection
USDA, SCS—Small Watershed Program (PL-566)	Evaluation and treatment of small agricultural watersheds with multiple resources to protect; includes land and natural resource inventories and assessments, basinwide planning and targeting of resources, technical assistance, and educational programs	Staff for technical assistance to landowners and decision-makers in the basin, funds for demonstration projects, reconnaissance, and intensive inventories of resources
USDA, SCS—Great Plains Conservation Program (GPCP)	Intensive conservation treatment for individual farms located within the Great Plains ecoregion through long-term agreements (3–10 year contract) with farmers	Technical assistance, cost-share funds up to 75% of the average cost of selected high-priority conservation practices
SDA, SCS—Resource Conservation and Development Program (RC&D)	Voluntary program to promote the economic development and to intensify resource protection in priority areas through the use of public participation in RC&D councils	Planning assistance for small communities for communitywide resource protection
USDA, SCS—Natural Resource Assessment programs: Soil Survey, Natural Resources Inventory, River Basin Studies	Various programs to map and assess the condition of natural resources (generally soil, water, vegetation, and wildlife) and conservation treatments	Maps, reports, data information, statistical analysis

TABLE 11.5 (Continued)

Agency and Program	Program Descriptions and Agency Responsibilities	Resources Available and Possible Roles
USDA—Agricultural Stabilization and Conservation Service (ASCS)	Provides administrative oversight and cost-sharing for approved conservation practices from ASCS and other USDA administered programs. Tracks crop production and other statistics. Distributes crop subsidy and deficiency payments	Maps, conservation practice status information, cost-share funds
USDA, ASCS—Agricultural Conservation Program (ACP)	Cost-sharing on an annual basis for a number of soil-conserving, production efficiency improving, and water quality practices	Funds for cost-share, generally limited to \$3500 per farm per year
USDA, ASCS—Emergency Conservation Program (ECP)	Cost-sharing on an annual basis to replace conservation treatments (mainly structural) that were destroyed in areas designated as disaster areas due to an act of nature	Funds for cost-share of high-priority conservation practices
USDA, ASCS—Water Bank Program	Designed to improve and restore wetland areas through financial compensation for 10-year easements on private property	Funds for easement compensation on eligible lands in participating states
USDA, ASCS—Colorado River Salinity Control Program (CRSCP)	Financial assistance for on-farm projects that seek to control salinity levels delivered to the basin, primarily irrigation water management	Funds, reports, data on level of conservation treatment, demonstration sites, funds for cost-share, monitoring, and education
USDA, ASCS—Forestry Incentives Program (FIP)	Cost-share to revegetate and improve timber stands on private lands	Cost-share funds
USDA, Cooperative Extension Service (CES)	Educational programs and information to aid individuals in the selection, operation, and maintenance of the most beneficial conservation treatments. Economic analysis and data for each farm or ranch. Provides technical assistance in integrated pest management. Programs generally carried out in cooperation with state land grant universities	Staff for educational programs and technical assistance, personalized economic analysis, and to coordinate small-scale demonstrations on local farms. Educational materials

R0023691

USDA, Cooperative State Research Service (CSRS)

Applied research, usually at state experiment stations on agricultural production and soil and water conservation generally using demonstration plots. Conducts the Sustainable Agriculture Research and Education program (SARE). Many projects in cooperation with state land grant universities

Reports, data, equipment. Occasionally funds for joint/special projects outside the normal research agenda. Grants for Agriculture in Concert with the Environment (ACE) program

USDA—Forest Service (USFS)

Management of national forests and grasslands for sustained production and multiple use. *Works with individuals, industries and other agencies*

Staff, maps, reports, equipment for construction and monitoring, educational materials, occasionally funds for special projects. *Field offices located in each national forest, regional offices located in 9 areas, Washington, D.C. office*

USDA, USFS—Permit Program

Oversight of timber sales and harvest contracts, grazing leases, minerals development on USFS property. Provides technical assistance to permittee on proper resource use

Staff for technical assistance and compliance monitoring

USDA, USFS—Air and Watershed Programs

Overall environmental planning and technical support for forest management decisions. Special studies and watershed demonstration projects in certain areas

Funds for special studies and watershed demonstration projects. Natural resource inventories and reports, water quality/habitat monitoring, environmental analysis of resource trends and conditions

USDA, USFS—Forest Stewardship Initiative

Technical assistance and cost-share to private holdings or lands adjacent to national forest lands for installing BMPs

Funds and technical assistance to individuals

USDA—Farmers Home Administration (FmHA)

Loans and loan guarantees to eligible producers for operating expenses, land purchase, and conservation measures

Funds and loans for property improvement and conservation treatment installation and water-conservation practices

USDA—Agricultural Research Service (ARS)

Basic and applied research on agricultural production and conservation measures, including fertilizers, pesticides, and BMP effectiveness

Reports, BMP effectiveness and environmental fate and transport data, demonstration sites, occasionally funds for joint-sponsored projects. *Research stations located throughout each state; most specialize in particular types of investigations*

TABLE 11.5 (Continued)

Agency and Program	Program Descriptions and Agency Responsibilities	Resources Available and Possible Roles
U.S. Department of the Interior (USDOI)	Oversight, management or monitoring of national natural resources, including land, water, and wildlife.	Staff, maps, reports, demonstration sites, educational materials, monitoring equipment. <i>Offices located in regional centers, field offices in each management area, Washington, D.C. office</i>
USDOI—Geological Survey (USGS)	Long-term baseline monitoring of water resources (quantity, flow, and quality), hydrologic and geologic investigations and data, special intensive short-term studies	Maps, data, and information on hydrology and water quality status and trends. Staff for technical assistance in designing a monitoring plan
USDOI—Fish and Wildlife Service (USF & WLS)	Oversight and regulation of the nation's wildlife resources. Management of national wildlife reserves, enforcement of federal game and fish laws, cooperative administration of national wetlands program with COE and EPA. Cooperative projects to enhance wildlife habitat, special studies, especially fisheries investigations	Staff for enforcement of Endangered Species Act and other laws on public and private agricultural land, research reports and data on habitat, populations and management of wildlife. Funds for cooperative projects. Educational materials, teacher training, curricula, and maps
USDOI—Bureau of Land Management (BLM)	Administration and management of federal lands. Oversight of grazing leases, mineral exploration, and extraction bids and leases on BLM lands. Technical assistance to permittees on BLM land in proper resource use. Oversight of recreational users of BLM land	Staff for environmental analysis and trend evaluation on BLM land, technical assistance, and oversight. Funds for special studies and cost-share for permittees for certain conservation practices (generally grazing/range management). Funds for range improvement, riparian area management, and recreational area development projects. Maps

USDOl—Bureau of Indian Affairs
(BIA)

Technical assistance to tribes on tribal lands, mainly for social services. Some assistance for conservation work and educational programs. Natural resource inventories and monitoring of ground and surface water

Maps, natural resource inventories of Indian and tribal lands. Funds for special projects. Staff for technical assistance to tribes

USDOl—Bureau of Reclamation

Administers, constructs, and oversees water supply facilities in western states. Regulates discharge from these facilities. Joint administration of the Colorado River Salinity Control Program with many agencies to set consistent salinity standards and manage public and private lands within the basin. New initiative to reclaim lands damaged by federal irrigation projects

Staff for oversight of projects and management of federal property and facilities, assessment of water quality around reservoirs as part of the national irrigation water quality program. Maps and reports, some data

USDOl—National Park Service

Administers and manages national parks for preservation of natural resources

Staff for oversight and administration. Funds for special studies and occasionally cooperative projects on land adjoining park boundaries

USDOl—Office of Surface Mines
(OSM)

Regulates the removal and reclamation of surface mined minerals, mostly coal on private lands

Staff for oversight and technical assistance in mining operations and reclamation efforts, for engineering studies, and for vegetative site inspections and monitoring of resources. Educational materials, data, and reports

U.S. Department of Defense—Army
Corps of Engineers (COE)

Oversees construction and operation of large flood-control and public water supply reservoirs. Conducts water quality monitoring on lakes within their jurisdiction. Regulates in-lake activities and shoreline development. Cooperatively administers the wetlands dredge and fill permit program with EPA and USF&WLS. Can enforce permit requirements for BMPs or other mitigation

Maps, special studies, water quality monitoring data. Staff and funds for improvement of existing projects. Staff for review and oversight of 404 (wetlands) permits. *Field offices located in various districts throughout states. Washington, D.C. office*

TABLE 11.5 (Continued)

Agency and Program	Program Descriptions and Agency Responsibilities	Resources Available and Possible Roles
U.S. Department of Commerce— National Oceanic and Atmospheric Administration (NOAA)	Administers programs in cooperation with states to inventory and manage coastal resources. Funds and performs basic research and assessments relating to coastal eutrophication. Maintains data base for agricultural pesticides and nutrient loadings	Funds to state coastal programs. Staff for technical assistance. Data, reports, educational materials. Occasionally funds for special demonstration projects
USDOC, NOAA—Coastal Zone Management Act programs (CZMA)	In cooperation with EPA, administers a quasi-regulatory coastal protection program that sets performance-based management measures for control and prevention of NPS pollution in coastal areas for all land-use activities	Staff for technical assistance. Funds for plan development
State Water Quality Agencies	Administers many programs (similar to EPAs) for protection of water quality in ground and surface water, including the NPDES permit program, water quality standards regulations, the NPS program, ambient statewide monitoring programs	Staff for technical assistance to local governments and individuals in BMPS application. Water quality monitoring, data, and reports. Funds for pollution-control projects, educational materials, and programs
State Natural Resource Agencies	Administer programs for wetlands and coastal protection programs	Staff for technical assistance to local governments. Monitoring of natural resource trends. Reports, data. Educational materials and programs
State Departments of Agriculture	Regulate pesticide registration and use, administers marketing and rural development programs. Sometimes issues permits for fertilizer or feedlots	Staff for oversight of applicators and other regulatory functions

State Departments of Health	Administer septic tank and public drinking water regulatory programs. Monitors water supplies. Provides technical assistance to local governments	Staff for technical assistance to local governments, monitoring, and educational programs. Data, reports, and educational materials
State Soil and Water Conservation Commissions	Administer cooperative programs with the USDA SCS to conserve soil and water resources on private lands. Provide technical assistance to individuals	Staff for technical assistance to individuals, engineering, or construction equipment, services and supplies that support BMP implementation. Some states have state cost-share funds for BMPs
State Fish and Game Agencies	Regulate the harvest of fish and wildlife resources by individuals and commercial operations. Responsible for cost recovery to state of lost fish and wildlife due to environmental contamination	Staff for enforcement of state fish and game laws and for technical assistance in wildlife and fisheries management for private individuals. Educational materials, natural resource inventory data
State Water Rights Agencies	Responsible for allocation of water rights (mostly in western states). Regulates consumptive use of water resources	Staff for permit writing and oversight. Data and reports on water flow
Local Planning and Zoning boards	Specify land-use zoning and boundary determinations, general community planning, oversight of program operation	Maps, long-range plans, inventory of local resources, special reports, budget information, staff for technical assistance
Local County Commissioners	Manage, construct, and maintain county roads and bridges, oversight and approval of county budgets for all county programs. Taxing authority	Information on county conditions, equipment for construction and maintenance, budget reports, occasionally funds for special projects
Local SWCDs Local School Boards and School Administration	Local field office of state agency. See earlier Oversee public education within jurisdictional boundaries. Can set local curricula requirements and priorities. Taxing authority, bond-issuing authority	Information on status of current educational programs, assistance in developing new initiatives
Local Municipal Utilities Districts	Oversees construction and maintenance of public works projects for water and sewer (occasionally energy). Taxing and bond-issuing authority	Information and special reports on water issues. Funds for special projects to enhance system operation and reduce costs

TABLE 11.5 (Continued)

Agency and Program	Program Descriptions and Agency Responsibilities	Resources Available and Possible Roles
Regional River Authorities	Manage and coordinate activities within their basin for flood control, water quality protection, energy development. Taxing authority	Data, reports, maps, water quality monitoring. Staff for technical assistance to local government and other agencies or groups. Funds for special projects
Regional Councils of Government	Assist in the coordination of activities of all governments within the councils area. Provides technical assistance, information, and promotes special projects of benefit to all	Staff for technical assistance to local governments, occasionally water quality monitoring, reports, and data about local conditions. Funds for special projects
Others—Commodity Groups	Various groups usually formed to improve marketing and lobbying capabilities for specific crops or livestock interests. Almost every major crop has at least one such group	Staff for data gathering and analysis, public education campaigns, technical support to growers, legislative and market analysis. Funds from members for special projects
Environmental Organizations	Various groups formed to protect, conserve, or preserve the environment in general or to address a specific issue. Lobby for environmental laws and programs as well as funding. Many perform volunteer services such as water quality monitoring, natural resource rehabilitation work, cost-share, or other funds for special projects Too numerous to list, consult a directory	Staff and volunteers for assistance with local projects, occasionally funding for cooperative work. Educational materials and programs. Reports and data on environmental conditions and trends
Social and Service Clubs	Formed for reasons other than resource protection, most do have local projects that enhance or beautify the community. Staffed with volunteers, these organizations can provide labor, supplies, and equipment on mutually beneficial projects, as well as insight into the community	Volunteers for special projects

Sources: Alexander (1993); U.S. EPA (1992b); Mass et al. (1987).

and pricing to support sustainable, low-*impact* farming systems. Most water quality professionals agree that the largest (and most difficult) change must be to attitudes (Ehorn, 1989).

It is clear, however, from a review of the water quality data as well as visual inspection of many streams, rivers, and lakes, that in the United States and elsewhere, diffuse or nonpoint source pollution from agriculture has not been adequately controlled. As more data are gathered by state and federal agencies, and as citizens become more educated about its causes and effects, and thus more active in looking for it, the extent of nonpoint source pollution reported will likely increase. Most of the technology to control diffuse pollution from agriculture is currently available, but to ensure that this technology is implemented a number of changes must occur and a number of political, institutional, and financial barriers must be overcome.

Before change will occur an individual or an institution (government, business, social group, etc.) must (1) know that a problem exists, (2) understand its significance, and (3) have some type of incentive (either internal or external) to change. Experts have noted that diffuse pollution from agriculture has not been adequately controlled because:

1. Farmers, agribusinesspeople, agricultural agencies, agricultural advocacy groups, lawmakers, and the general public have not been made aware (and many do not believe) of the problem or its extent.
 - a. The effects on the environment of improper agricultural land management are generally not evident or visible to the land user/owner since they often occur downstream or off-site.
 - b. The effects of NPS pollution are cumulative. While the "discharge"/runoff from a single site may not be great (dirty) enough to cause a use impairment, the minor contributions from many other sites add up to a larger and more significant problem.
 - c. Many agricultural producers have no historical, highly visible basis of comparison between water quality now and that of 100 years or more ago. The impression that the water has "always looked this way" is often not dispelled, even when data are presented.
 - d. Data establishing a discrete cause-effect relationship is not always clear. While adequate and defensible research may exist for some problems, the nature of NPS is so site specific that an argument can always be made that "This area is different; its soils, its hydrology and climate, its crop varieties and farming practices . . . etc." because in natural systems research this is almost always true.
 - e. When data documenting changes to water chemistry or fisheries habitat are available, often they are unclear if the effect is great

- enough to cause a use impairment now or in the future. It is especially difficult (and expensive) to obtain data sets extensive enough to establish the sources of each pollutant found, and even more difficult to show the causes (sources) of biological changes.
2. These individuals and institutions do not fully understand (or believe) the potentially serious short- and long-term effects of diffuse pollution on human and environmental health.
 - a. Inconclusive data and conflicting reports about the effect of various contaminants on human health and the inexact science of risk analysis breeds skepticism and doubt.
 - b. Long-term studies that accurately determine the risks associated with the many overlapping conditions found in biological systems research are lacking and are often cost prohibitive.
 - c. The type of contamination from diffuse pollution is insidious and often results in small changes in species composition that go unnoticed.
 - d. Fortunately there have been few major "catastrophes" from diffuse pollution, like oil spills or nuclear reactor meltdowns, but lack of such events often allows a problem to go unresolved until one does occur.
 3. The incentives currently available are not sufficient to control the problem. A number of political, institutional, and financial barriers exist that must be removed before existing incentives can work effectively, and in some cases new or additional incentives are needed. Barriers to control agricultural diffuse pollution include:
 - a. The lack of education of all sectors of society about the problem, effective controls available, and methods that can be used to get BMPs applied.
 - (1) Previous educational programs have concentrated on educating the agricultural producer, an important step but an incomplete one. Effective education for legislators, citizens, agricultural agencies, marketing associations, and all other facets affected by agriculture need water quality education targeted to their particular role/responsibility in solving the problem.
 - b. The lack of trained professionals specializing in agricultural water quality issues at local, state, regional, and national levels.
 - (1) The "lag-time" or built-in institutional inertia that exists has hampered agricultural agencies and interest groups and water quality agencies from taking a strong leadership role in diffuse pollution control and prevention. A lack of vision and grasp of

the changing reality of American politics kept some of the agricultural "institutions" (agencies and special interest and commodity groups) from taking a leadership role in diffuse-pollution research, education, and technical training and assistance. Instead these agencies and groups worked in a counterproductive fashion to "disprove" the claims of water quality agencies that agriculture was and could cause a problem in an effort to "protect" the farmer. The farmer, hearing conflicting information from many sources and trying to adjust production practices to a changing market, does not know where to turn for unbiased, factual, and useful information (Babcock, 1985).

- c. The lack of consistent regulatory "back-up" from the state or federal level to support local control efforts.
 - (1) The Clean Water Act explicitly exempts agricultural irrigation return flow from the federal NPDES system. For many years people maintained that diffuse pollution from agriculture was not regulatable. They were only partially correct. Diffuse pollution from agriculture cannot be regulated under the NPDES permit system by the federal government, but it can be regulated by state NPDES permit systems and by other state regulations. Some states do have laws that deal with diffuse pollution, most notably erosion and sediment controls from construction activities, but some states are writing laws for agriculture, too. In Wisconsin, a number of farmers were frustrated with their neighbors. Some producers had installed the needed BMPs, but others would not "comply," even when substantial incentives (a high cost-share rate) were offered. As a local initiative Wisconsin passed their "Bad Actors Law" that provides regulatory backup in the form of the state NPDES permit requirements for those farmers not voluntarily installing BMPs (U.S. EPA, 1993).
 - (2) When considering legislation to "regulate" agricultural NPS pollution at the federal level, the issues of private land-use rights and "taking" of private property arise. However, without consistent federal regulation states and local governments, which are subject to a great deal of public and political pressure, risk losing industry, jobs, or business income from their state if they pass more stringent environmental laws than their neighbors. A considerable water quality problem from dairy waste in Texas is the result of Arizona and California passing

tough pollution-control laws causing large dairies to move to a state with a more "favorable" environmental climate (Gordon and Hansen, 1989).

- d. The lack of targeting of resources to the most threatened or impacted watersheds.
 - (1) For the most part agricultural NPS control programs are voluntary and the cash costs of BMP installation, maintenance, and management are often not recoverable as cash, although they are sometimes recoverable through conserving natural resources (soil, grass, timber) for long-term sustainability of the farming system. Thus each individual land owner/user decides which, if any, BMPs will be installed on his/her property. In making this decision agricultural producers must factor the economic benefit and cost of any BMP into the farm's profit margin. For effective NPS control, those farms causing the most problem must be targeted for implementation first. In a purely voluntary program it is very difficult to overcome the economic forces to achieve this targeting. In addition, USDA programs are generally designed to "share the wealth" and provide equal opportunities for cost-share funds to all producers, regardless of the environmental priority (Duda and Johnson, 1985; Mass et al., 1987).
- e. Conflicting laws and programs of various agencies that work in a counterproductive fashion.
 - (1) Programs within many agencies often have conflicting requirements. Sometimes even within the same agency or department laws and programs can be counterproductive. For example, within the USDA many programs exist that provide farmers price supports and other financial benefits to produce certain crops at the highest yield rate regardless of the impacts of these intensive production systems on the soil, water, or energy resources, yet other programs provide other financial incentives to install conservation treatments. Producers cannot follow the guidelines for conservation programs without jeopardizing their benefits from the commodity programs (U.S. General Accounting Office, 1990).
- f. Lack of coordination between existing programs resulting in overlap or duplication of some efforts and scant attention to other needed areas.
 - (1) The laws governing natural resource management (land management) and environmental quality have not been developed or implemented in concert with one another. Agricultural laws

have traditionally been written to stabilize production and ensure that consumers have a safe, reliable source of inexpensive food and fiber. Even today economic, supply, and demand concerns dominate agricultural laws and policies. Environmental laws have concentrated on a one-problem, one-solution type of approach rather than on a more holistic way of managing natural and man-made systems. Current laws have practically mandated this approach by being narrowly focused on quick fixes where "nobody gets hurt too much" (a concession to economic realities). These laws have often turned out not to be fixes at all, but rather redirection of the problem (U.S. EPA, 1992; Ribaud, 1985).

- g. Lack of support services built into the economy that perpetuate the BMPs once the incentives have expired.
 - (1) In areas or watershed where projects to control diffuse pollution from agriculture have been successfully implemented, those with long-term lasting results had a number of private enterprises support the implementation and operation and maintenance of the recommended BMPs. These companies supplied services and equipment that individuals could not afford to own or acquire. Without these services or equipment there is a tendency to neglect BMP maintenance once the incentive (cost-share, land easement rental payment) expires. Some examples include firms specializing in animal waste lagoon pump-out (cleaning) and land application, companies that specialize in prescribed burning for brush control and range management, or professional associations such as the Association of Independent Crop Consultants, which are skilled in the use of integrated pest management techniques (American Farm Bureau Federation, 1992; Gordon and Hansen, 1989; Alexander, 1993b).
- h. The lack of funds for all facets of control programs (education, technical assistance, cost-share, regulatory aspects).
 - (1) The extent of the diffuse-pollution problem is such that there will never be enough state or federal funds available to completely control the problem in all impacted watersheds. Many policy analysts maintain that NPS pollution problems should be treated and funded no differently than point source pollution problems in which the polluter pays for the costs of control and adsorbs it as the cost of doing business. Others believe that because agricultural producers have no way of passing these costs on to consumers, and since diffuse pol-

lution is so much more difficult to pinpoint to a particular location (a single farm) than point source discharges, society should compensate farmers for their pollution-control efforts (Harrington, Krupnick, and Peskin, 1985). In most parts of the United States this second option is most widespread; however, it has not proved to be fully effective in controlling the problem.

Once some of the barriers have been lessened the addition of new incentives and the modification of existing incentives can be used much more effectively to encourage adoption of BMPs by individuals. Not all are equally effective, and some are not the politically astute choice in certain areas, but all should be considered as options. These modified or expanded incentives can include the following.

1. *Education.* Programs that target key audiences and tailor both the message and the method of presentation to that audience are most effective in eliciting a behavior change.
2. *Technical assistance.* Involves one-on-one interaction between the professional water quality person and the farmer, and includes recommendations about environmental conditions and BMPs appropriate for the specific site in question. It also includes assistance of on-site engineering or agronomic work during the installation of BMPs.
3. *Tax advantages.* Can be provided through state and local taxing authorities or by a change in the federal taxing system that rewards those producers who install BMPs.
4. *Price supports or subsidies.* Crops or agricultural products that have a distinct water quality benefit and are not currently in the commodity program could be added to encourage planting or production.
5. *Cost-share to individuals.* Direct payment to individuals for the installation of specific BMPs (for example, terraces) has been effective in many areas if the cost-share rate is high enough to compensate the farmer for the perceived and real risk associated with the use of the BMP.
6. *Cross-compliance legislation built into existing programs.* Currently in effect in the 1985 and 1990 U.S. Farm Bills. Generally a type of quasi-regulatory incentive/disincentive that conditions benefits received on meeting certain requirements or performing in a certain way.

7. *Direct purchase of lands contributing the greatest problem or of riparian corridors for mitigation.* Direct purchase of special areas for preservation has been used extensively by groups such as the Nature Conservancy. Use of community-owned greenbelts in urban areas is another variation. Costs of direct purchase are generally high, but effectiveness can also be exceptional. The practice could be used by any government or group to obtain control of a certain piece of land whose owner is not willing to install needed BMPs through the existing programs.
8. *"Oversight/site inspections" in a nonregulatory program.* Any individual, group, agency, etc., will perform better with personal attention. A powerful incentive for voluntary installation of BMPs is a site visit by the local, state, or federal regulatory agency to encourage BMP installation on the visited site.
9. *Peer pressure.* Social acceptance by ones peers can be a motivational factor for installation of BMPS by some individuals. If a community values a clean environment or the use of certain agricultural BMPs, producers in those communities are more likely to install them.
10. *Direct regulation of land-use and production activities.* Often considered to be the option/incentive of last resort, regulatory programs that are simple, direct, and easy to enforce are quite effective. Such programs can regulate the type of land use allowed (like zoning ordinances), or the kind and extent of activity allowed on that type of land (like pesticide application rates), or set performance standards for the level of environmental damage allowed from that land activity (such as retention and reuse of the first inch of runoff from the property).
11. *Consumer demand.* Demand for a particular product or for a commodity produced in a certain way (organically raised wheat or vegetables) can create a market that may pay a premium for items in high demand.

CONCLUSIONS

Control and prevention of diffuse pollution from agriculture is and will continue to be a challenge. A number of experimental programs have documented varying degrees of success. The current programs for non-point source control from agriculture are just the beginning of a long process that will require a long-term commitment of time, resources, and funds from every sector. Obtaining these commitments from agencies whose missions may not be water quality oriented, and working together with diverse groups, a great deal can be done.

References

- Alberts, E. E., G. E. Schuman, and R. E. Burwell (1978). Seasonal runoff losses of nitrogen and phosphorus from Missouri Valley loess watersheds, *J. Environ. Qual.* 7:203-208.
- Alexander, S. (1992). *Managing Containerized Nurseries for Pollution Prevention*, Water Quality Branch, U.S. Environmental Protection Agency, Dallas, TX, and Terrene Institute, Washington, DC.
- Alexander, S. (1993a). *Clean Water in Your Watershed—A Citizen's Guide to Watershed Protection*, U.S. Environmental Protection Agency, Dallas, TX (in press).
- Alexander, S. (1993b). Effective education programs change behavior. Lessons Learned from the Nonpoint Source Program, in *Proceedings, Riparian Biosystems in the Humid Southeast*, Atlanta Ga, March, 1993, National Association of Conservation Districts and U.S. Environmental Protection Agency, Washington, DC.
- American Farm Bureau Federation (1989). *Innovation and Technology Transfer—America's Farmers: Profitable Environmental Managers*, AFBF, Park Ridge, IL.
- Andridenas, P., and T. Eichers (1982). What farmers buy makes production fly, in *Food from Farm to Table—1982 Yearbook of Agriculture*, J. Hays, ed., Science and Education Administration, Washington, DC.
- Babcock, H. (1985). Compelling on-the-ground implementation of measures to control nonpoint source pollution, in *Perspectives on Nonpoint Source Pollution*, EPA 440/5-85-001, Office of Water, U.S. Environmental Protection Agency, Washington, DC.
- Berg, H., et al. (1988). Management effects on runoff, soil, and nutrient losses from highly erodible soils in the Southern Plains, *J. Soil. Water Conserv.* 43:407.
- Beschta, R. L. (1978). Long-term patterns of sediment prediction following road construction and logging in Oregon coast range, *Water Resour. Res.* 14:1011-1016.
- Brach, J. (1990). *Agriculture and Water Quality—Best Management Practices for Minnesota*, Division of Water, Minnesota Pollution Control Agency, St. Paul, MN.
- Brade, J., and D. Uchtmann (1985). Agricultural nonpoint pollution control: An assessment, *J. Soil Water Conserv.* 40(1):23-26.
- Burns, J. C., et al. (1987). Swine manure and lagoon effluent applied to a temperate forage mixture: Persistence, yield, quality, and elemental removal, *J. Environ. Qual.* 16:99.
- Capodaglio, A. (1988). Diffuse groundwater contamination by herbicides in an agricultural area of Northern Italy, in *Proceedings, Nonpoint Pollution 1988—Policy, Economy, Management and Appropriate Technology*, American Water Resources Association, Bethesda, MD, pp. 303-313.
- Cheng, J. D. (1988). Streamflow changes after clear-cut logging of a pine beetle-infested watershed in southern British Columbia, Canada, *Water Resour. Res.* 25:449.

- Committee on International Soil and Water Research and Development, National Research Council (1991). *Toward Sustainability. Soil and Water Research Priorities for Developing Countries*, National Academy Press, Washington, DC.
- Duda, A., and R. J. Johnson (1985). Cost-effective targeting of agricultural nonpoint-source pollution controls, *J. Soil. Water Conserv.* **40**(1):108–111.
- Dudley, R. R., and J. R. Karr (1979). Concentrations and sources of fecal and organic pollution in an agricultural watershed, *Water Resour. Bull.* **15**:911–923.
- Ehorn, D. A. (1989). *Offsite Assessment: A National Workshop*, EPA 440/5-89-001, Office of Water, U.S. Environmental Protection Agency, Washington, DC.
- Francis, C. A., C. B. Flora, and L. D. King (1992). *Sustainable Agriculture in Temperate Zones*, Van Nostrand Reinhold, New York.
- Gordon, D. G., and N. Hansen (1989). *Managing Nonpoint Source Pollution: An Action Plan Handbook for Puget Sound Watersheds*, Puget Sound Water Quality Authority, Seattle, WA.
- Haith, D. A., and R. C. Loehr, eds. (1979). *Effectiveness of Soil and Water Conservation Practices for Pollution Control*, EPA 600/3-79-106, Environmental Research Laboratory, U.S. Environmental Protection Agency, Athens, GA.
- Harington, W., A. J. Krupnick, and H. M. Peskin (1985). Policies for nonpoint source water pollution control, *J. Soil Water Conserv.* **40**(1):27–32.
- Haupt, H. F. (1959). Road and slope characteristics affecting sediment movement from logging roads, *J. Forestry* **61**:329–332.
- Hosapple, T. (1992). Diversification and Stability of Hosapple farms, in Growing into the 21st Century, 1992 Sustainable Agriculture Conference, National Association of Conservation Districts, Washington, DC.
- Humenik, F. A., F. A. Koehler, R. P. Mass, S. A. Dressing, J. M. Kreglow, and D. D. Johnson (1982). *Best Management Practices for Agricultural Nonpoint Source Control—Animal Waste*, NC Agricultural Extension Service, NCSU, Raleigh, NC.
- Jones, A. (1990). *Technology Designed for Outer Space an Outstanding Success on Earth—Microbial Rock Reed Filter an Emerging and Promising Low Cost Low O&M Wastewater Treatment Bio-Technology*, Sixth Annual Regional Municipal Technology Forum, Louisiana State University Ag Center, U.S. Environmental Protection Agency, Dallas, TX.
- Klaine, S. J., et al. (1988). Characterization of agricultural nonpoint pollution: Pesticide migration in a West Tennessee Watershed, *Environ. Toxicol. Chem.* **7**:609.
- Lawrence, G. B., and C. T. Driscoll (1988). Aluminum chemistry downstream of a whole-tree harvested watershed, *Environ. Sci. Technol.* **22**:1293.
- Levi, D. A. (1978). *Agricultural Law*, Department of Agricultural Economics, Texas A&M University, Lucas Brothers Publ. Columbia, MO.
- Loehr, R. C. (1972). Agricultural runoff characteristics and control, *J. Sanitary Eng. Div.* **98**:909–925.

- Loehr, R. C. (1974). Characteristics and comparative magnitude of nonpoint sources, *J. WPCF* **46**:1849–1872.
- Loehr, R. C., et al., eds. (1979). *Best Management Practices for Agriculture and Silviculture*, Ann Arbor Sci. Publ., Ann Arbor, MI.
- McDowell, L. L., et al. (1989). Nitrogen and phosphorus yields in run-off from silty soils in the Mississippi delta, U.S.A., *Agric. Ecosyst. Environ.* **25**:119.
- Mass, R. P., M. D. Smolen, C. A. Jamieson, and A. C. Weinberg (1987). *Setting Priorities: The Key to Nonpoint Source Control*, Office of Water, U.S. Environmental Protection Agency, Washington, DC.
- New Mexico Environmental Improvement Division (1989). *Priority Waterbodies for Water Quality Action*, New Mexico Environmental Improvement Division, Santa Fe, NM.
- Owens, L. B., et al. (1989). Sediment and nutrient losses from an unimproved, all-year grazed watershed, *J. Environ. Qual.* **18**:232.
- Parkin, T. B., and S. E. Meisinger (1989). Denitrification below the crop rotating zone as influenced by tillage, *J. Environ. Qual.* **18**:12.
- Platts, W. (1990). *Managing Fisheries and Wildlife on Rangelands Grazed by Livestock—A Guidance and Reference Document for Biologists*, Nevada Department of Wildlife, Reno, NV.
- Ribaudo, M. O. (1992). Options for agricultural nonpoint-source pollution control, *J. Soil Water Conserv.* **47**(1):42–46.
- Ritter, W. F. (1988). Reducing impact of nonpoint pollution from agriculture: A review, *J. Environ. Sci. Health* **25**:821.
- Robillard, P., et al. (1982). *Planning Guide for Evaluating Agricultural Nonpoint Source Water Quality Controls*, EPA 600/3-82-021, Environmental Research Laboratory, U.S. Environmental Protection Agency, Athens, GA.
- Sampson, N. (1981). *Farmland or Wasteland: A Time to Choose*, Rodale Press, Emmaus, PA.
- Smolen, M. D., G. D. Jennings, and R. L. Huffman (1990). Impact of nonpoint sources of pollution on aquatic systems: Agricultural land uses, in *Nonpoint Source Impact Assessment*, Rep. No. 90-5, CH₂M-Hill, Water Environment Research Foundation, Alexandria, VA.
- Stewart, B. A., et al. (1975). *Control of Water Pollution from Cropland*, Vol. I, EPA 600/2-75/026a, U.S. Environmental Protection Agency, Washington, DC.
- Stinner, B. R., et al. (1988). Phosphorus and cation dynamics of components and processes in conventional and no-tillage soybean ecosystem, *Agric. Ecosystems. Environ.* **20**:81.
- Tankersley, H. C. (1981). Whoa There! Let's Smarten Up on Land Use, in *Will There be Enough Food? The 1981 Yearbook of Agriculture*, USDA Science and Education Administration, Washington, DC.
- Thompson, P., et al. (1989). *Poison Runoff: A Guide to State and Local Control of Nonpoint Source Water Pollution*, National Resources Defense Council, New York.
- U.S. Department of Agriculture, National Agricultural Statistics Service (1992a). *Crop Values 1991 Summary*, Agricultural Statistics Board, Washington, DC.

- U.S. Department of Agriculture, National Agricultural Statistics Service (1992b). *Crop Production 1991 Summary*, Agricultural Statistics Board, Washington, DC.
- U.S. Department of Agriculture, National Agricultural Statistics Service (1992c). *Poultry Production and Value 1991 Summary*, Agricultural Statistics Board, Washington, DC.
- U.S. Department of Agriculture, National Agricultural Statistics Service (1992d). *Milk Production, Disposition and Income 1991 Summary*, Agricultural Statistics Board, Washington, DC.
- U.S. Department of Agriculture, National Agricultural Statistics Service (1992e). *Meat Animals Production, Disposition and Income 1991 Summary*, Agricultural Statistics Board, Washington, DC.
- U.S. Department of Agriculture, National Agricultural Statistics Service (1992f). *Floricultural Crops 1991 Summary*, Agricultural Statistics Board, Washington, DC.
- U.S. Department of Agriculture, National Agricultural Statistics Service (1992g). *Catfish Production*, Agricultural Statistics Board, Washington, DC.
- U.S. Department of Agriculture, Soil Conservation Service (1980). *Field Office Technical Guide—Section III*, USDA SCS, Washington, DC.
- U.S. Department of Agriculture, Soil Conservation Service (1988). *National Handbook of Conservation Practices*, Washington, DC.
- U.S. Environmental Protection Agency (1990). *Biological Criteria, National Program Guidance for Surface Waters*, EPA 440/5-90-004, Office of Water Regulations and Standards, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1992a). *National Water Quality Inventory—1990 Report to Congress*, EPA 503/9-92/006, Office of Water, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1992b). *Managing Nonpoint Source Pollution—Final Report to Congress on Section 319 of the Clean Water Act*, EPA 506/9-90, Office of Water, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1993). *Guidance Specifying Management Measures for Sources of Nonpoint Pollution in Coastal Waters*, EPA 840-B-92-002, Office of Water, U.S. EPA, Washington, DC.
- U.S. Forest Service (1989). *Best Management Practices Guide for Integrated Resource Planning on National Forest Lands in the Southwestern Region*, U.S. Department of Agriculture, USFS, Albuquerque, NM.
- U.S. General Accounting Office (1990). *Water Pollution—Greater EPA Leadership Needed to Reduce Nonpoint Source Pollution*, GA/RCED-91-10, Resources, Community and Economic Development Division, Washington, DC.
- Westerman, P. W., et al. (1987). Swine manure and lagoon effluent applied to a temperate forage mixture. II. Rainfall runoff and soil chemical properties, *J. Environ. Qual.* **16**:106.

Receiving Water Impacts

Problems that are harder to assess and control, such as sedimentation, nutrient enrichment, runoff from farmlands, and toxic contamination of fish tissues and sediments, are becoming more evident. . . . By some extent, it is certainly true that the more we look, the more we find.

EPA's National Water Quality Inventory: 1988 Report to Congress

ASSESSMENT OF WATER QUALITY PROBLEMS

Throughout the late 1970s and early 1980s when the results of the 208 studies were published, using the traditional point source assessment approaches, researchers and pollution abatement administrators struggled with the answers to the question "What is the water quality problem of diffuse pollution?" The following quote taken from Sonnen (1980) illustrates the dilemma of this post-208 period:

We still have not designed let alone implemented even one monitoring program based on postulated mechanisms of fundamental physics and chemistry to demonstrate in one urban area, much less all of them, whether urban runoff poses a (water) quality problem or not, and to the degree that it does, what could be done about it.

However, ten years later in 1989, the National Resources Defense Council published *Poison Runoff* (Thompson, Adler, and Landman, 1989) claiming that, based on the studies of the Resources for the Future, the

National Oceanographic and Atmospheric Administration, and the U.S. EPA, diffuse pollution (i.e., *poison runoff*) contributed to the nation's receiving waters the following proportions of pollutants:

- 90% of total nitrogen
- 90% of fecal coliform bacteria
- 70% of chemical oxygen demand
- 70% of oil
- 70% of zinc
- 66% of phosphorus
- 57% of lead
- 50% of chromium

According to *Poison Runoff*, the high levels of nitrogen in urban and agricultural runoff promotes excess algal growth in streams, lakes, and estuaries, and the toxic components also diminish the beneficial uses of bodies of water.

The 1988 U.S. EPA Report to Congress on the state of water quality of the nation's waters (U.S. EPA, 1990) provided the statistics on the impact of various sources of pollution on the nation's streams and lakes (excluding Alaska and islands). A quote from the report opens this chapter. Of the total river kilometers of U.S. streams, 70% were fully supporting the designated uses (that is, they were considered as unpolluted), 20% were partially supporting the uses of the rivers (moderately polluted) and 10% were not supporting the uses (severely polluted). Referring to the nation's inland lakes (excluding the Great Lakes and Alaska), 74% of lake area fully supported the designated uses, 17% were partially supporting, and 10% were nonsupporting. The designated uses were specified by the Clean Water Act (see Chapter 1) and generally implied attainment of water quality that would support aquatic life, contact recreation (swimming), and in some special cases, potable use of water. Tables 12.1 and 12.2 provide a summary of the assessment. The values reported in the tables are percentages of the impaired river kilometers or lake areas. Hence to obtain a percentage of the total river kilometers, the values would have to be multiplied by 0.3 (fraction of total river kilometers where uses are not being met) and by 0.26 for lakes.

The Great Lakes are the largest freshwater bodies in the world (the largest lake in the world, the Caspian Sea in the former USSR, is a salt-water body). They are so large that they should be considered freshwater seas. However, only 8% of the total shoreline kilometers in the United States were supporting the designated uses, 18% of the shoreline may only partially support the designated uses, while 73% were not supporting.

TABLE 12.1 Causes of Pollution of U.S. Surface Waters

Pollution Cause	Percent of River Kilometers Affected by Source		Percent of Lake Surface Affected by Source	
	Moderately	Severely	Moderately	Severely
Siltation	33	9	18	8
Nutrients	22	4	33	13
Pathogens	15	7	8	2
Organic enrichment	11	3	16	4
Pesticides	8	1	5.5	2
Metals	9	5	5	1
Suspended solids	5	2.5	7.5	3
Salinity	5.5	2	4	2
Habitat modification	5	2.5	12	0.5

Source: After U.S. EPA (1990).

TABLE 12.2 Adverse Water Quality (Pollution) Impact of Sources

Pollution Source	Percent of River Kilometers Affected by Source		Percent of Lake Area Affected by Source	
	Moderately	Severely	Moderately	Severely
Agriculture	40	15	27	14
Municipal (POTW)	13	5	5	2
Mining	8	3	2	0
Habitat modification	12	4	16	2
Urban storm runoff	7	3	3	0.5
Silviculture	4	0.5	1.5	0
Industrial	7	3	3	0.5
Construction	5	1.5	2	0
Land disposal of waste	4	1	4	1
Combined sewers	3	1	<0.5	0

Source: After U.S. EPA (1990).

Priority (toxic) organics are by far the most extensive cause of impairment of Great Lakes water and biota quality. Metals were also reported by the states as a source of impairment, with nutrients and organic enrichment/low dissolved oxygen also cited as contributors to the impairment of use.

Based on the 1988 EPA Report to Congress, of the U.S. coastal waters in late the 1980s, 72% were fully supporting the designated uses, 23% were partially supporting, and 6% were not supporting. Nutrients, pathogens, and organic enrichment were listed as the three major causes of impairment of the coastal shore waters. Unlike the inland waters, for

which agriculture was the major polluting land-use activity, urban point sources (POTW effluents), mining, urban storm runoff, CSOs, and land disposal of wastes were the four major sources responsible for about 90% of impairment (pollution) of coastal water, all of which are related to urban land use and urbanization.

As pointed out in Chapter 1 diffuse-pollution problems are not limited only to the United States. The North Sea and shorelines of the Adriatic and Mediterranean seas are grossly polluted by the impact of nutrient discharges, to the point that beaches are closed for swimming and fish is contaminated and unfit for human consumption. Many rivers in Europe, Latin America, and Asia are extremely polluted by waste discharges from a variety of sources. Due to inadequate sanitation in undeveloped countries, urban runoff may contain sewage and gray wastewater (domestic wastewater without fecal matter), which is still being discharged into drainage systems.

A number of researchers have tried to define the impact of diffuse pollution on receiving waters. The impact is different depending on the state of the drainage system development, degree of urbanization expressed by the percent imperviousness of the urban area, land-use distribution, hydrologic and hydraulic characteristics of the body of water, use and type of chemicals for fertilization of soil and pest control, and other factors. For example, Novotny (1985) related the water quality impact to the degree of watershed development, which was discussed previously (Chapter 1).

Myers et al. (1985) summarized the EPA's data and reports on the magnitudes of nonpoint pollution and their water quality impact. The authors emphasized that substantial water quality benefits can be achieved by targeting the resource to lands and land uses that are most polluting. Such land uses include urban and highway construction. Agricultural activities were found to be the main contributors of nonpoint pollution, followed by urban runoff.

Urban runoff has been identified as a major source of toxics, including toxic metals and petroleum hydrocarbons; however very little was known about the long-term effects of these substances on the biota of the receiving bodies of water (U.S. EPA, 1983; Torno, 1984; Niedzialkowski and Athayde, 1985). The NURP study identified receiving water quality problems associated with urban runoff. Frequently, discharges of urban runoff are the cause of streams and rivers exceeding heavy metal ambient water quality criteria for freshwater aquatic life. Copper, lead, and zinc appear to pose a significant threat to aquatic life in some urban areas of the United States. On the other hand, organic priority pollutants in urban runoff do not appear to pose a general threat to freshwater aquatic life.

Several NURP projects identified possible problems in sediments because of the buildup of priority pollutants contributed wholly or in part by urban runoff. However, there are only a few studies on this topic, and these are limited in scope, so the findings must be considered only indicative of the problem. Further studies are therefore needed, particularly regarding the long-term impact.

Nutrients in urban runoff may accelerate eutrophication problems and severely limit recreational uses, especially in lakes. However, the NURP lake studies indicated that the degree of beneficial use impairment varied widely, as did the significance of the urban component. Coliform bacteria discharges in urban runoff can have a significant negative impact on the recreational use of lakes.

Adverse effects of urban runoff in marine waters were found to be a highly specific local situation. Though estuaries and embayments were studied by NURP to a very limited extent, they were not believed to be threatened by urban runoff, though specific instances where use was impaired or denied could be of significant local or regional importance. Coliform bacteria present in urban runoff could have a direct impact on shellfish harvesting and beach closing. The significant impact of urban runoff on shellfish harvesting was identified in the Long Island, New York, NURP project. The specific and detrimental impact of diffuse pollution on Chesapeake Bay was covered in Chapter 1. It should be pointed out, however, that the NURP studies focused on urban runoff only, primarily on its surface flow component, and did not include mixed overflows from the combined sewers or subsurface flows from septic systems.

In the conclusion of their nationwide assessment, Heaney and Huber (1984) stated that:

1. Documented studies of the impact of urban runoff and combined sewer overflows on receiving waters are scarce.
2. Numerous definitions of the terms "impact" and/or "problem" exist.
3. Receiving waters that were included in the evaluations ranged from small creeks and ponds to large estuaries and oceans. No clear line of demarcation exists to distinguish the urban drainage systems from the receiving bodies of water.
4. Some evidence was gathered that linked fish kill and beach closing to discharges of urban runoff and overflows. Such evidence was weak.
5. The studies of dissolved oxygen downstream of urban areas have produced the most definitive information on the impact of urban runoff on water quality. However, the impact is primarily on dampen-

TABLE 12.3 BOD and Nitrogen Concentration in Border Points of Some Czech Rivers

River Border Crossing	Concentration (mg/l)		
	BOD ₅	NO ₃ ⁻	NH ₄ ⁺
Labe (Elbe)	6.6	19.4	2.67
Morava	9.7	17.4	0.91
Dyje (Thaya)	8.8	37.5	1.7
Odra (Oder)	9.8	19.5	7.58

Source: Data from Ministry of Environment (1991).

ing of diurnal fluctuations of dissolved oxygen concentrations and less on actual reduction of the average daily concentrations.

PLUARG (Pollution from Land Use Activities—Reference Group) activities were reviewed by the Nonpoint Source Task Force of the Water Quality Board of the International Joint Committee (IJC) (1983), which stated that the major sources of water quality degradation are loading of phosphorus and toxics, but that urban runoff pollution has only local impact and does not constitute a significant problem on a lakewide basis.

Although the information on the impact of diffuse pollution in North America is most extensive, in the world context it is not the most severe one. For, example, almost 56% of all major streams in the Czech Republic have been classified as unfit for most uses (Ministry of the Environment, 1991). Average concentration of pollutants in major rivers leaving the Czech Republic measured at the border sections were as documented in Table 12.3. The water quality of these rivers is comparable to treated sewage without nutrient removal. The rivers are also heavily contaminated by toxic organic chemicals and metals. The extremely high nitrate content of the Czech rivers is attributed primarily to excessive use of chemical fertilizers and discharges of untreated or marginally treated urban and industrial wastewater.

THE AQUATIC ECOSYSTEM

Basic Interrelationships

The goal of all our water quality protection, restoration, and pollution abatement activities is to restore, protect, and enhance the ecosystems or ecology. The ecosystems can be divided into two components, the biotic, or living, and abiotic, or nonliving. Ecosystems are considered to be in a dynamic equilibrium with their inputs and surroundings. When one of the

inputs is changed, the ecosystem will readjust to a new equilibrium. Typical aquatic ecosystems are rivers, lakes, wetlands, estuaries, or soils.

The hydrodynamical, biological, and chemical processes in aquatic systems cannot be separated and must be treated simultaneously. Studying water quality changes resulting from diffuse inputs requires multidisciplinary approaches. It is beyond the scope of this book to cover the details of these processes, and the reader is referred to monographs (Stumm, 1985; Krenkel and Novotny, 1980) for further review.

Sun energy reaching the aquatic ecosystems is the primary input. However, other energy inputs, such as organics in waste discharges, or waste heat from cooling-water discharges, can contribute significantly to the overall behavior and composition of these systems. The equilibrium behavior of the components of the ecosystem is also affected by the presence or absence of various elements, chemicals, or other compounds that stimulate the growth of biotic components, or compounds that retard the growth, damage, or kill the organisms. The former category of chemicals is called *nutrients* or *food*, the latter include *toxic* (carcinogenic or noncarcinogenic) compounds. The biotic components of aquatic ecosystems and their role can be divided into three groups of organisms.

Producers and productivity. Producers can accept inorganic materials and energy from the sun to produce new organic matter. The process is called *photosynthesis*, and the group of participating organisms are known as *autotrophic*. These include green aquatic plants attached to the bottom or vascular aquatic plants and microscopic floating planktonic organisms known as *phytoplankton*. A special group of microorganisms utilizing energy obtained from the oxidation of reduced chemical compounds (such as the oxidation of ammonia to nitrite and nitrate by *Nitrosomonas* and *Nitrobacter*) are called *chemotrophs*.

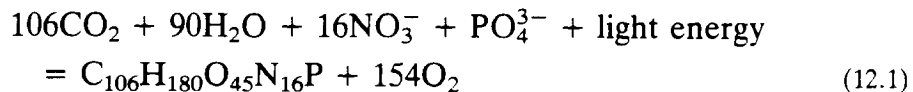
The photoautotrophs and chemotrophs use carbon dioxide or alkalinity as their sole carbon source and form new organic matter. The organisms also require nutrients, such as nitrogen, phosphorus, and trace elements. Typically in aquatic ecosystems the quantities of trace elements are in abundance; however, nitrogen, phosphorus, or sometimes alkalinity may be in a short supply. The nutrient that is in the shortest supply is called the *limiting nutrient* because its quantity affects the rate of production (growth) of organic matter by the producers. Water bodies may be carbon (alkalinity) limited if alkalinity is very low—few mg CaCO_3/l ; however, most of surface-water bodies are either phosphorus (most of the inland lakes) or nitrogen (some estuaries and rivers) limited.

To control the eutrophication process (see the subsequent section) the control should be focused on the limiting nutrient, but this should not be considered as a strictly to obey rule. For example, if nitrogen is the

limiting nutrient, it still may be more efficient to control phosphorus loads if nitrogen originates from difficult to control sources, such as nitrates in ground water. For most inland lakes phosphorus is the limiting nutrient. To determine whether nitrogen or phosphorus is limiting the productivity (eutrophication) of a particular body of water, one can plot the nitrogen concentrations versus the consequent phosphorus concentrations on an arithmetic plot. The straight line of the best fit will intercept either the nitrogen or phosphorus abscissa. The nutrient that at low concentrations is exhausted first (zero or negative intercept on the abscissa) is then the limiting nutrient.

The total nitrogen to total phosphorus ratio may also determine what algae may prevail. For example, nitrogen-fixing blue-green algae might be favored during periods of low nitrogen content ($N:P < 10:1$). The nitrogen-fixing capability of some algae is another reason why limiting phosphorus inputs is usually the only practical solution to the excess productivity of water bodies, which results in accelerated eutrophication (U.S. EPA, 1990).

In the photosynthetic (chemosynthetic) reactions organic matter and oxygen are produced. Stumm and Morgan (1962) in a classic paper proposed the following stoichiometric representation of the production of organic matter by aquatic autotrophs. Another, slightly different modification of the equation was presented by Schindler (1985) and others.



It should be pointed out that phytoplankton can utilize both ammonia (NH_4^+) and nitrate (NO_3^-) forms of nitrogen in the photosynthetic process.

This equation indicates that under abundant light and alkalinity conditions, one atom of phosphorus or 16 atoms of nitrogen (whichever nutrient is limiting) will produce 154 moles of oxygen. Less oxygen (about 106 moles of O_2) is produced if ammonia is substituted for nitrate in Equation (12.1). The productivity relation can be illustrated by the following example.

Example 12.1: Oxygen and Organic Matter Production by Photosynthesis

Lake concentration of phosphate-P was measured as 1 mg P/l. Estimate the amount of organic matter and oxygen produced by photosynthesis

stimulated by the phosphate nutrient, assuming that other nutrients (inorganic carbon and nitrogen) are not limiting.

Solution Equation (12.1) states that 1 mole (or millimole) of phosphate will approximately stimulate growth of one (pseudo-) mole (millimole) of organic matter and 154 moles (millimoles) of oxygen. The atomic weights are hydrogen, H = 1, carbon, C = 12, nitrogen, N = 14, oxygen, O = 16, and phosphorus, P = 31.

This calculation follows the following stoichiometric relationship derived from Equation (12.1):

$$\frac{\text{P concentration (mg/l)}}{[\text{P}]} = \frac{\text{organic matter production (mg/l)}}{[\text{C}_{106}\text{H}_{180}\text{O}_{45}\text{N}_{16}\text{P}]}$$

$$= \frac{\text{oxygen production (mg/l)}}{154 [\text{O}_2]}$$

Then $1 \text{ mg P/l} / [\text{P}] = 1/31 = 0.0323 \text{ m mole/l}$.

The molar weight of organic matter is

$$[\text{C}_{106}\text{H}_{180}\text{O}_{45}\text{N}_{16}\text{P}] = 106 \times 12 + 180 \times 1 + 45 \times 16 + 16 \times 14$$

$$+ 1 \times 31$$

$$= 2427$$

Hence 1 mg P/l will stimulate production of

$$0.0323 \times 2427 = 78.4 \text{ mg/l of organic matter}$$

The molar weight of produced oxygen is

$$154 [\text{O}_2] = 154 \times 2 \times 16 = 4928$$

Oxygen production is then

$$0.0323 \times 4928 = 159 \text{ mg/l}$$

Such production of oxygen would result in the supersaturation of water by oxygen, and the excess oxygen would eventually be lost by deaeration.

The producers are essential building blocks in the construction of basic primary organic matter and the only organisms that can convert inorganic materials into organic tissue. Since these organisms need sunlight, they are productive in the well-lighted upper zones of bodies of water called the *euphotic layer*, which represents the depth of effective light penetra-

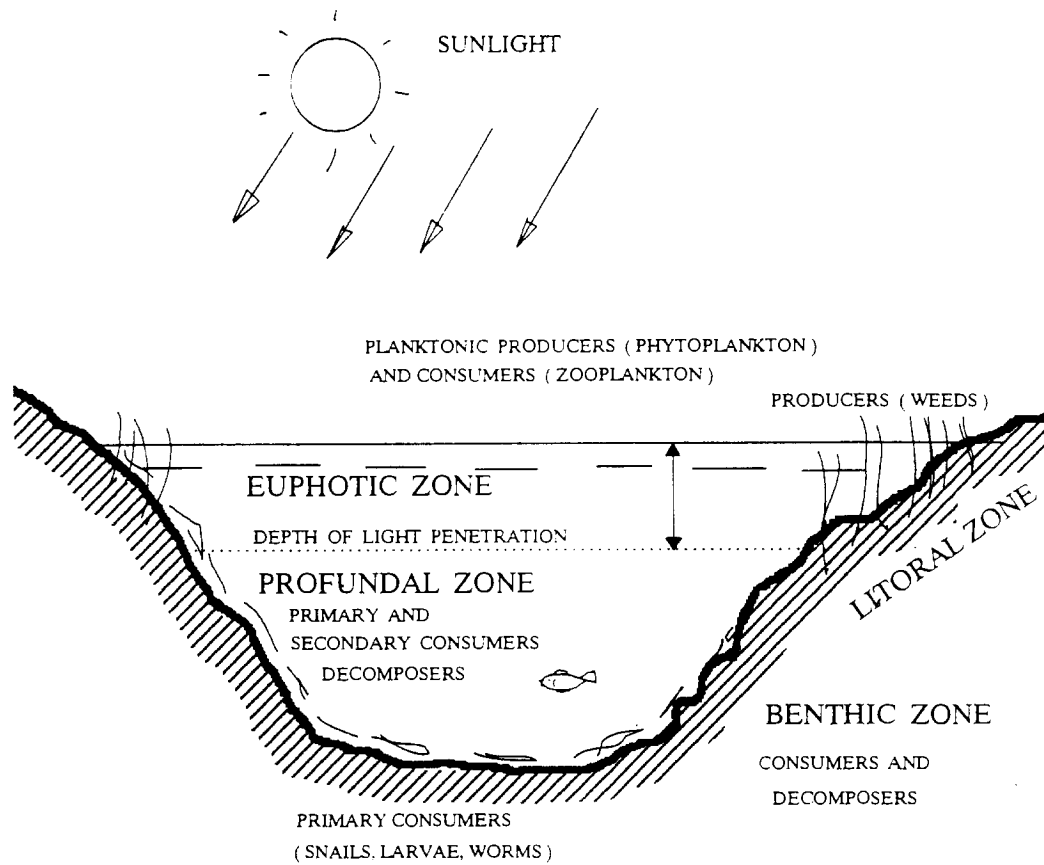


FIGURE 12.1. Biological zones in aquatic ecosystems.

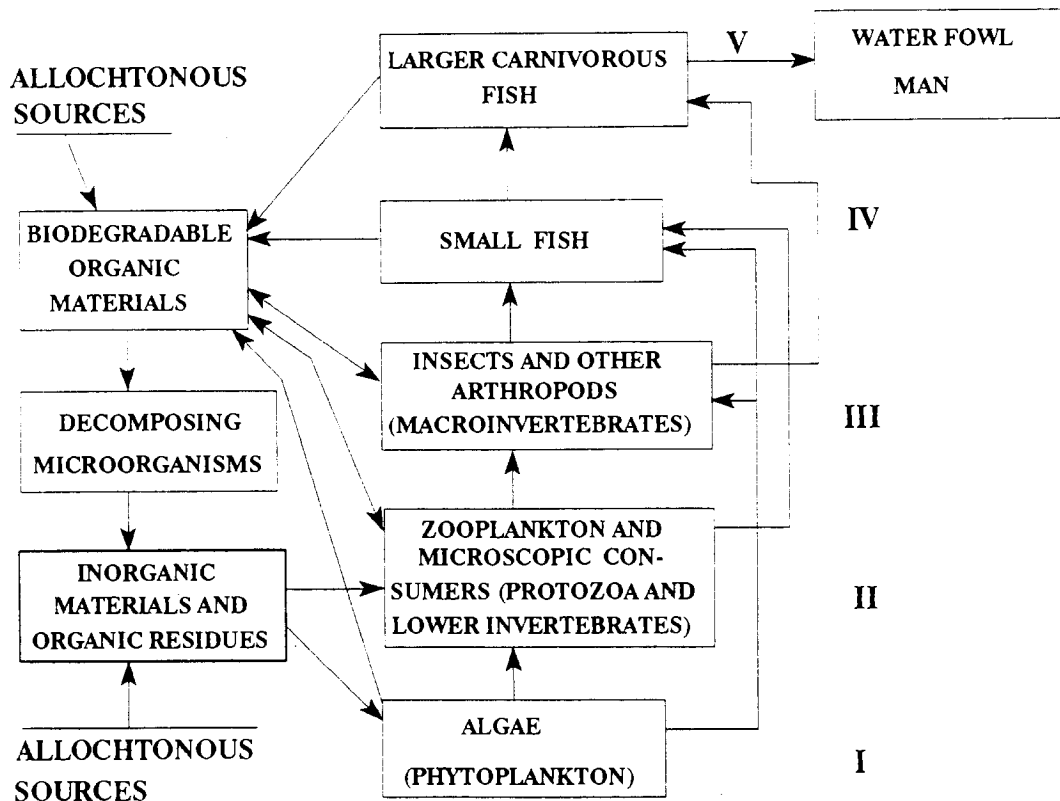


FIGURE 12.2. Food web schematic.

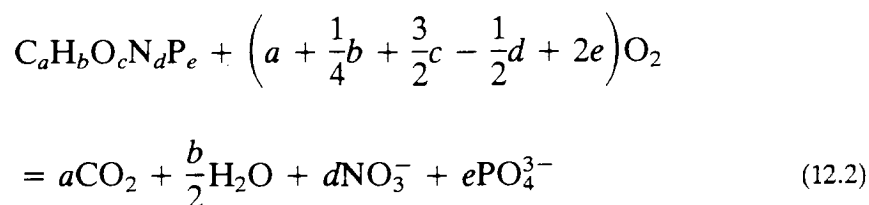
tion for photosynthesis. This depth depends on turbidity of the water (Fig. 12.1). In the absence of light energy (during night hours or in the deeper, profundal zones of bodies of water) the process is reversed and the organisms use oxygen to sustain their living process. This is called *endogenous respiration*.

Consumers and transformation of produced organic matter. Consumers are organisms that utilize organic carbon as their source of energy. Apparently these organisms depend on the producers for their nourishment. The primary consumers can use the produced algal and plant tissues and other biomass directly, while secondary consumers receive their nourishment by feeding on primary or smaller secondary consumers. Organisms relying on organic carbon as their source of energy and nourishment are called *heterotrophic*.

The producers and consumers form a *food web*. This chain has several *trophic levels*, starting with phytoplankton as level I, zooplankton feeding on algae (herbivorous zooplankton) as level II, followed by plankton-eating microorganisms (carnivorous zooplankton) as level III, and continuing to higher levels of invertebrates, insects, fish, and ending with fish- and shellfish-eating fowl and man (Fig. 12.2). Of interest to water quality is bioaccumulation and biomagnification of some compounds as they are transferred throughout the food web from the lowest trophic levels to the highest with consequent increase of body tissue concentrations of the compounds in each trophic level. The phenomenon of bioaccumulation and biomagnification is discussed in the next chapter.

Decomposers and decomposition of organic matter. Decomposers are the organisms, primarily bacteria and fungi, that can decompose organic matter to basic minerals and organic residues that may then become available again to the producers. Organic matter decomposed by the organisms originate from dead bodies and cellular tissues of all organisms and from excreta and organic byproducts of living processes. Decomposing organisms are also heterotrophic.

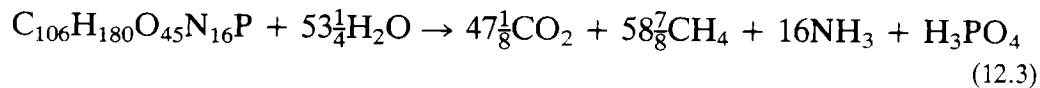
The character and intensity of the decomposition process depends on the availability of dissolved oxygen. If oxygen is in abundance, then decomposition is *aerobic*. The stoichiometric representation of the aerobic decomposition is a reverse of Equation (12.1), or



Decomposition in the water column is usually aerobic. In the absence of oxygen decomposition, it may be *anoxic* or *anaerobic*. In anoxic decomposition oxygen is supplied by the reduction of oxidized compounds, such as nitrates (NO_3^-) and sulphates (SO_4^{--}). The end products are the same as during aerobic decomposition.

In anaerobic decomposition, which is typical for sediments, organic matter is broken down by bacteria into methane, carbon dioxide, and ammonia. This process is called *diagenesis*. The anaerobic decomposition in sediments is a complex process that progresses in several stages. The intermediate products of the decomposition are fatty and other organic acids. *Methanogenesis*, that is, the formation of methane in addition to carbon dioxide and other byproducts is the final process of the anaerobic decomposition of organics in sediments. In freshwater sediments those products are formed from methyl carbon of acetate (Rudd and Taylor, 1980). Methane is released from the sediments into the overlying water by diffusion and ebullition.

Using Stumm and Morgan's representation of chemical composition of the algal organic biomass the stoichiometric balance of anaerobic decomposition can be represented as follows:



This equation is only an approximation of stoichiometric proportions and does not represent the biochemical process itself, which is far more complex and progresses in several stages (acidic and methane fermentation). However, using this approximation for aquatic organic sediments one could deduce that after the decomposition is completed the molar proportions of methane and carbon dioxide evolution are about the same and that the methane carbon is about one-half of the organic carbon load.

From sludge digestion studies it is known that not all organic carbon is converted to methane or other mineral products. The proof is in coal deposits that resulted mostly from the anaerobic decomposition of plants millions of years ago. A significant fraction of sedimentary organic matter is either inert or refractory (decays at a very slow to negligible rate). Foree, Jewell, and McCarty (1971) measured the inert-refractory fraction of decomposition algae and found that on average 50% of organic carbon, nitrogen, and phosphorus incorporated in algal biomass is refractory under aerobic conditions, and 40% of organic carbon and phosphorus and 60% of algal nitrogen is refractory under anaerobic conditions. Wide ranges of refractory-inert fraction values were measured in the experiments.

The fraction of inert organic solids also depends on the type of solids. For example, bottom sediments in the Milwaukee harbor contain organic solids from combined sewer overflow, urban runoff, and upstream algae and organic detritus. It was found that most of the inert organic matter originated from the CSOs and urban runoff, while most solids from upstream algae development were biodegradable (Anon., 1987).

Robertson (1979) and Rudd and Taylor (1980) found that for lake sediments (mostly of algal origin) the range of methane carbon release to carbon input varied from 0.36 to 0.6. They also concluded that methane production is correlated to the rate of organic carbon input to the sediments, and the rate is not strongly affected by temperature.

The dissolved methane released from the anaerobic zones is oxidized in the oxic–anoxic sediment–water interface layer. In freshwater sediments most methane oxidation is aerobic and hence exerts *sediment oxygen demand (SOD)*. In marine sediments, oxidation may be anoxic, using oxygen from sulphates and nitrates. Since methane has a relatively low solubility in water, most of it may escape into overlying water by ebullition, where part of it may dissolve and become a part of the pool of the dissolved organic carbon that may be oxidized by heterotrophic bacteria and exert an oxygen demand. This addition to the dissolved carbon pool of the water column should not be omitted. It should be noted that the present standard methods for biochemical oxygen demand

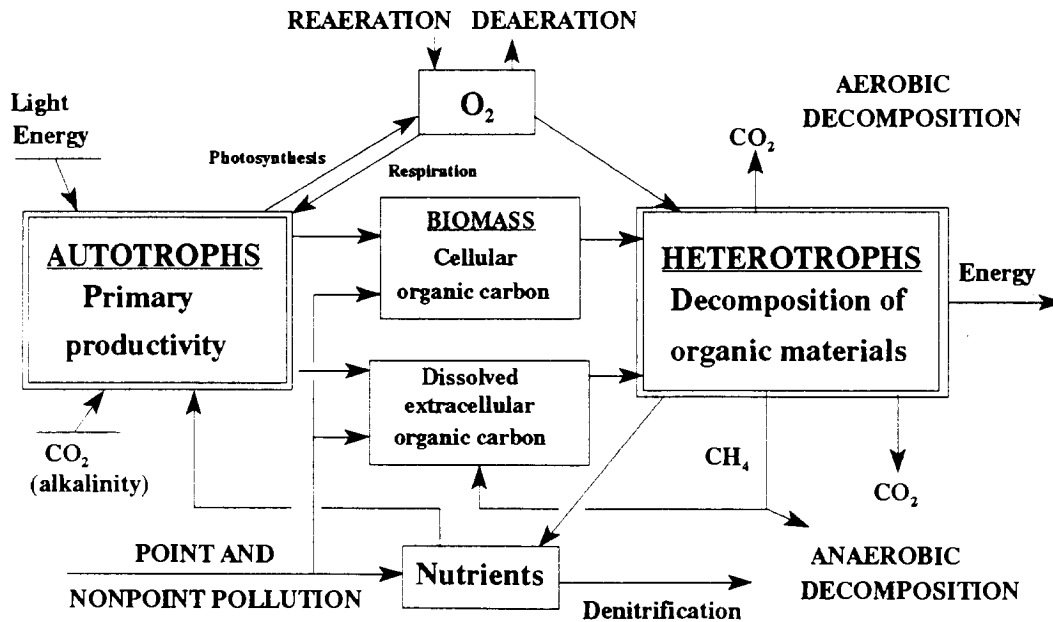


FIGURE 12.3. Carbon and nutrient cycles in water–sediment systems and their impact on the dissolved oxygen balance.

analysis require aeration of the samples, which may strip out the dissolved methane. Figure 12.3 shows the interrelationships in carbon and nutrient transport and the participating organisms in water-sediment systems. Such aquatic systems include lakes, wetlands, estuaries, and slow moving (impounded) rivers.

The relative distribution of producers, consumers, and decomposers depends on the type and amount of energy stored or entering the system. Different types of organisms will be emerging, growing, and dying at rates affected by ambient conditions. The overall growth rate for each group of organisms is a summation of their growth rate, death rate, and grazing by other organisms. If a body of water initially possesses a small number of consumers and decomposers, producers will develop, sometimes in very large numbers, depending on the optimal energy of the sun and the amount of mineral nutrients. As the overall organic matter concentration increases, consumers and decomposers develop, and when the energy inputs diminish, the decomposers liquidate the remaining biodegradable organic matter.

Several other definitions should be mentioned (Krenkel and Novotny, 1980):

The overall rate of organic matter production of a body of water in $\text{mg}/\text{m}^2\text{-day}$, or a similar unit, is called *productivity*, while *primary productivity* refers to organic matter production by producers only.

Plankton are organisms of small size that drift on the water and are subject to the action of the water. They may be defined further as *phytoplankton*, or photosynthetic planktonic organisms, and *zooplankton*, or animal plankton. As pointed out, phytoplankton represents the first trophic level, while zooplankton, which graze on phytoplankton, are the second trophic level.

Nekton are organisms of larger size, such as fish and insects, that swim freely and determine their own distribution in water, while *seston* are nonliving and living bodies of plants and animals that float or swim in the water.

Peryphyton are attached organisms, as are *sessile* organisms.

Benthos is the living and nonliving parts of the bottom sediments.

Invertebrates are a group of small multicellular organisms that are more complex than plankton. They can range in size from microinvertebrates (such as protozoa) to macroinvertebrates (such as larvae and worms). The benthic macroinvertebrates are of great importance in studying the pollution effects and impact of various remedial measures. These organisms are typically sedentary and live for an extended period of time (months to years). Hence, their occurrence and abun-

dance reflects the biochemical–ecological status of the body of water, and their presence is commonly used by limnologists for determining the health (integrity) and/or the degree of pollution of a body of water.

Mason (1991) has described in detail the effects of pollution and other water quality alterations on the biota of impacted aquatic systems. Such impacts may be reversible or irreversible.

Integrity of Surface-Water Bodies: Ecoregional Approaches

Integrity of Aquatic Ecosystems

The objective of the water pollution control policies in the United States expressed in the Clean Water Act is to *restore and maintain the chemical, physical and biological integrity of the Nation's waters* (Clean Water Act, Sec. 101(a)). When this definition is tied to ecological systems, the term *integrity* has been defined as the ability to support and maintain “a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural biota of the region” (Karr and Dudley, 1981).

Recent policy changes of the U.S. Environmental Protection Agency have recognized the fact that surface-water bodies differ among themselves and that there is a regional clustering of their ecological characteristics. Aquatic ecologists have recognized that there is a natural physical–chemical–biological variability in and among the bodies of water. Swamps and low pH bogs, for example, exhibit low dissolved oxygen levels that are unrelated to human activities (Karr et al., 1986). When metallic ores reach the land surface, higher metal concentrations can be found in surface waters. If the water quality standards that are in force do not recognize this natural variability, water quality regulators may find themselves enforcing the unenforceable. Karr et al. (1986) have stated that regulations are still being governed by point source (NPDES) permits that rely on monitoring of chemical and toxic concentrations and overlook perturbations whether they are caused by natural factors or by humans. When diffuse pollution is considered, in addition to or in a combination with point pollution, the integrity of the body of water has to be considered, and the water quality goals of abatement must be formulated in a bioregional context, including natural (background) loads.

An aquatic system is in a quasi-equilibrium with the boundary stresses on the system and its internal perturbations, which change as a result of natural and cultural factors. In Chapters 1 and 2 it was shown that

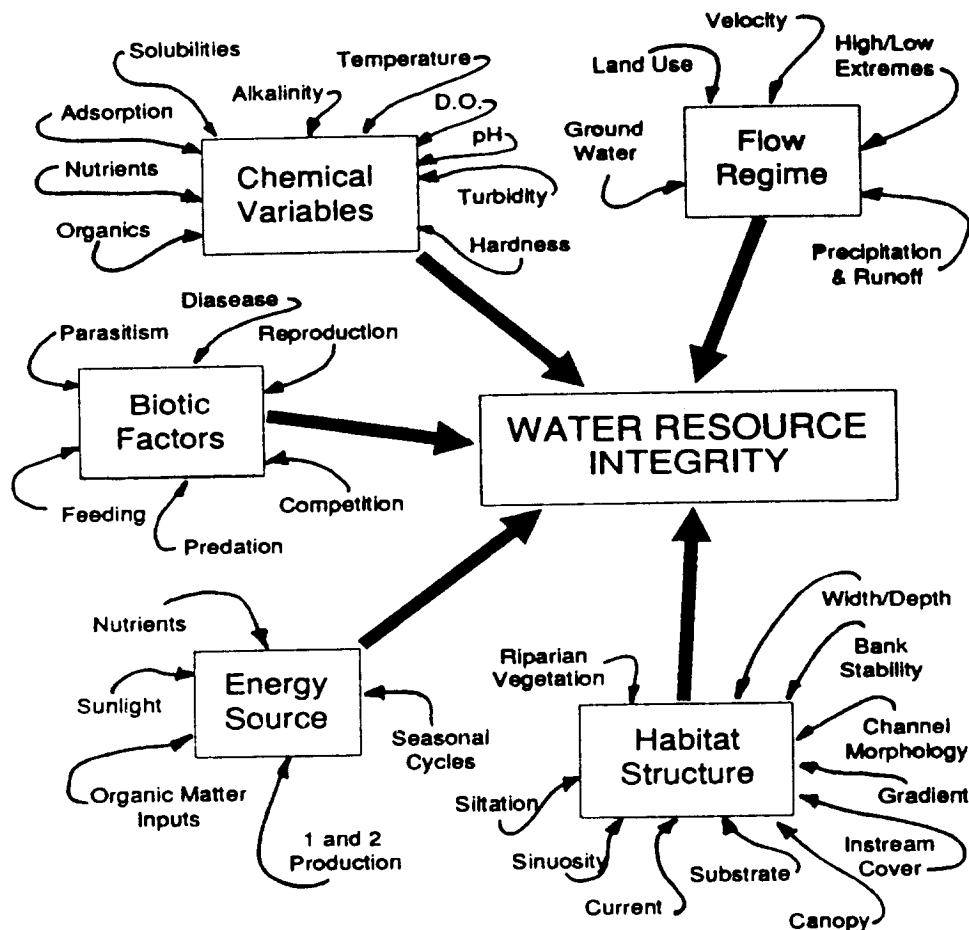


FIGURE 12.4. Five principal factors and their physical, chemical, and biological components that influence the resultant integrity of surface waters. (After Ohio EPA, 1987, modified from Karr et al., 1986.)

ecosystems have changed throughout the history of man (thousands of years) and even more so throughout geological history (millions of years). Although many environmental factors have been instrumental in the evolution of an aquatic ecosystem, Karr et al. (1986) have documented that these factors can be grouped into five major classes (Fig. 12.4). Altering any of these factors and parameters will have an impact on the ecosystem and the biota, which is its integral component. As is pointed out in Chapter 15 on system restoration, to achieve an improvement in the integrity of an aquatic ecosystem the perturbed processes and factors must be identified in all five classes. This obviously leads to an *integrated approach* to point and nonpoint pollution abatement.

Figure 12.4 also indicates why a single-factor, simplistic approach may fail in an attempt to remedy water quality and the ecological integrity of

the aquatic system. For example, take a riverine system that is stressed by point source discharges of typical wastewater and nonpoint source discharges of nitrate and phosphate. Without an abatement the river will exhibit low dissolved oxygen levels and a shift to excessive heterotrophic microbial populations (decomposers and grazing zooplankton). These microorganisms typically keep the algal population in check. Focusing on point source control may improve the dissolved oxygen problem and reduce the undesirable excessive heterotrophic microbial and zooplankton population; however, the nutrient levels in the stream may cause nuisance development of algal-dominated biota and an excessive primary production of organic matter, creating algal blooms, and hence negating the benefits of the point source cleanup effort. In spite of large expenditures for point source cleanup of biodegradable organics (BOD) such streams, besides being unsightly, may not be fit for many mandated uses, including swimming (swimmers may get a rash from enzymes produced by algae), fish (fish-kills may occur as a result of low dissolved oxygen caused by algal respiration), and drinking (the taste and color of the water may be severely impaired). In this case, the cleanup effort must include a reduction of nutrient inputs and development of riparian (stream-bordering) and in-stream habitats that are suited to enriched biota. The effects of nutrients can also be reduced by planting shade trees along the river, which restricts the light and limits the growth of algae.

Ecoregions. The work at the U.S. Environmental Protection Agency by Omernik (1987) and others (Gallant et al., 1989) resulted in a division of the conterminous United States into a number of geographical-ecological subunits that exhibit relative homogeneity in ecological systems (Fig. 12.5). These *ecoregions* are typically different from other geographical units such as physiographic units that are based strictly on land-surface forms or hydrological watershed units. The ecoregional map and ecoregional delineation are based on the premise that regional patterns of environmental factors would be reflected in regional patterns of surface quality. The factors that affect the extent of an ecoregion include, among others, land-surface forms, potential natural vegetation, soils, and land use. The boundaries between the ecoregions should not be viewed as abrupt changes, since they are basically subareas of change from one ecoregion to the next one. The homogeneity within the ecoregion is only relative. There is less variability within an ecoregion than between ecoregions.

The ecoregional concept is used for many cartographic purposes dealing with natural resources, including water quality. For example, it is assumed that attainable biological and chemical quality within the ecoregion can be approximated by measuring the physical, chemical,

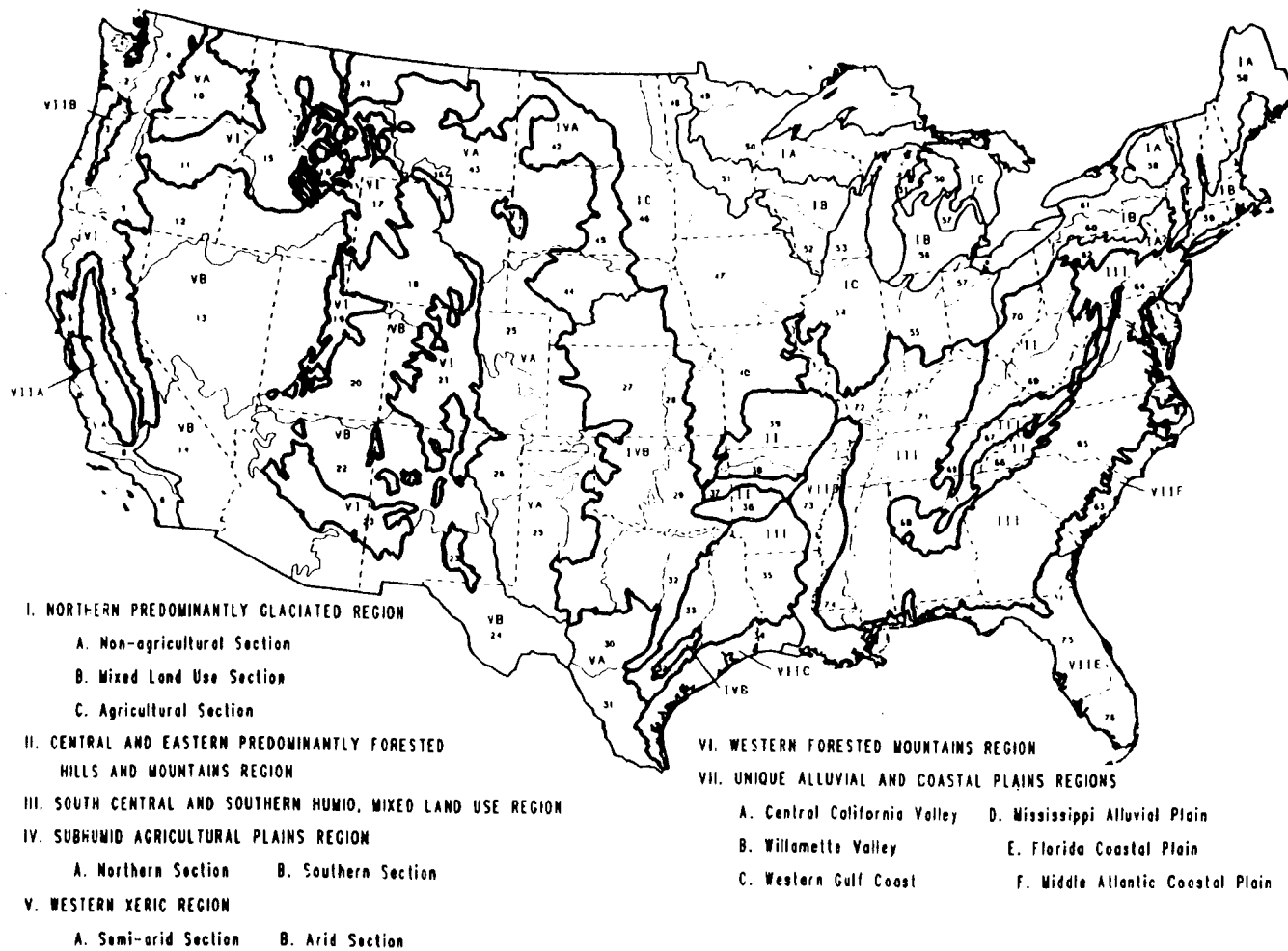


FIGURE 12.5. Ecoregions of the conterminous United States. (From Omernik, 1987.)

and biological quality of the streams draining watersheds that are representative of the natural environmental characteristics typifying the region and subject to the least possible human interference. Such streams and lakes constitute *reference bodies of water*. The degree of water quality deterioration can be ascertained by comparing the biology and chemistry of the water body of interest with the reference body of water of the same character located within the ecoregion. Preserving and protecting the undisturbed "reference reaches" should receive the highest priority by federal and state pollution-control agencies (Committee on Restoration of Aquatic Ecosystems, 1992).

Measures of Biotic Integrity

A healthy surface-water body provides a habitat to many species. However, specific species distribution exhibits geographic variability that reflects the ecosystem equilibrium with boundary inputs. The natural inputs that control the species distribution are meteorology, geography, latitude, elevation (both elevation and latitude affect some meteorological parameters), stream or lake morphology, stream habitat alterations by man, and water chemistry.

Water contamination will alter the species distribution and their abundance. Some organisms are more tolerant to pollution and, hence, they may develop in larger numbers, while the less tolerant will diminish. This is the principle behind several species diversity indices that have been used for water quality assessment (Krenkel and Novotny, 1980; Mason, 1991; Karr et al., 1986).

Use of biological indices for assessing water quality and the health (integrity) of the biota dates back almost 100 years. The earliest proposed biotic index was the saprobien system of Kolkwitz and Marson (1908). In this system indicator organism species are assigned a saprobien value ranging from 1 to 4-7 (depending on the classification) and a weighted average (depending on the abundance of the organisms), then expressing the saprobien index. The original system recognized four saprobien classes of quality related to the state of decomposition of organic matter. An *oligosaprobic* classification reflects the cleanest water quality, corresponding to unpolluted natural waters, while a *polysaprobic* characterization reflects highly polluted waters; α and β -mezosaprobic are intermediate classifications. Sládeček (1979) reviewed this system and its modifications. The saprobien system is widely used in Europe. In fact, water quality standards in most European countries have been based on and related to the four classes of the saprobien system.

Species diversity indices have been developed in the United States and are used to measure the stress of an ecosystem. It is considered that in

unpolluted systems a large number of organism species coexist, with no single species dominating. When the ecosystem becomes stressed by contamination, species sensitive to that particular stress will be eliminated, thus reducing the variety of the biota. In addition, certain species may be favored, thus increasing in numbers. This system appears logical, however, Karr et al. (1986) pointed out that combining species richness with species abundance can yield ambiguous results and may prove difficult to interpret. Species diversity indices were reviewed by Krebs (1989). A relationship between water quality and diversity indices is difficult to establish.

Two biotic integrity indices have recently been developed and introduced in the United States. These indices have the advantage of detecting impacts that do not reflect on the chemistry of the receiving body of water, such as channelization of the stream. These systems measure the response of fish or benthic macroinvertebrate communities. The first system by Karr et al. (1986) uses fish as biological monitors. The second system uses macroinvertebrates.

The *Index of Biotic Integrity* (IBI) by Karr et al. (1986) uses measurements of the distribution and abundance or absence of several fish species types listed in Table 12.4. The fish samples are obtained by electrically shocking the sampling reach (fish is not killed). The Index of Biotic Integrity is a comparative index in which the measurements in a test reach of the particular body of water of interest are compared with fish distribution in reference to unimpacted, undisturbed reaches within the same ecoregion (Karr et al., 1986; Rankin, Yoder, and Mishne, 1990). Presumably, these reference bodies of water should exhibit no or minimal human impact. Table 12.5 and Figure 12.6 show the conceptual response of fish communities to various stresses and the ranges of the IBI for the five integrity classes. IBI of less than 12 implies that no fish were found by repeated sampling.

IBI relies on multiparameters, since the fact that the ecosystems to be evaluated are complex. It requires professional judgment and appropriate sampling methodology. As with any other biological tool, IBI must be used appropriately. IBI does not replace chemical indices and evaluation of chemical water quality, but it does provide a long-term picture of impacts, including those that are undetectable by chemical methods. Ideally, IBI can be correlated to chemical water quality, as indirectly indicated in Table 12.5.

It has to be pointed out that in addition to the variation among ecoregions, the fish population and number of fish species are a function of the stream order (headwater streams with no tributaries have order 1, streams with one or more first-order tributaries have order 2, streams

TABLE 12.4 Metrics Used to Assess Fish Communities in the Midwestern United States

Category	Metric	Scoring Criteria ^a		
		5	3	1
Species richness and composition	1. Total number of fish species	Expectations for metrics 1–5 vary with stream size and region and are discussed in the text		
	2. Number and identity of darter species			
	3. Number and identity of sunfish species			
	4. Number and identity of sucker species			
	5. Number and identity of intolerant species			
	6. Proportions of individuals as green sunfish	5%	5–20%	>20%
Trophic composition	7. Proportion of individuals as omnivores ^b	<20%	20–45%	>45%
	8. Proportions of individuals as insectivorous cyprinids	<45%	45–20%	<20%
	9. Proportion of individuals as piscivores (top carnivores)	>5%	5–1%	<1%
Fish abundance and condition	10. Number of individual in sample	Expectations for metric 10 vary with stream size and other factors and are discussed in the text		
	11. Proportion of individuals as hybrids	0%	>0–1%	>1%
	12. Proportion of individuals with diseases, tumors, fin damage, and skeletal anomalies	0–2%	>2–5%	>5%

Source: After Karr et al. (1986).

^aRatings of 5, 3, and 1 are assigned to each metric according to whether its value approximates, deviates somewhat from, or deviates strongly from the value expected at a comparable undisturbed reference body of water.

^bOmnivores are defined here as fish species with diets composed of >25% plant material and >25% animal material.

TABLE 12.5 Conceptual Model of the Response of Fish Community as Portrayed by the Index of Biotic Integrity

Integrity Class Total IBI Score	Severely Degraded (Very Poor) 12-22	Degraded (Poor) 28-34	Moderate Impact (Fair) 40-44	Enrichment (Good) 48-52	Least Impacted (Excellent) 58-60
Community conditions, characteristics	No community organization—few or no species, very low numbers most tolerant only, high percentage of anomalies, diseases	Poorly organized community—few species, low numbers tolerant species only, many anomalies, fish dominated by omnivores	Reorganized community—tolerant species, intolerant in very few numbers, omnivores predominate, older age classes of top predators rare	Good community organization—good numbers of sensitive species, some intolerant, trophic structure shows some signs of stress	Highly organized community—insectivores, top carnivores, intolerants predominate, high diversity
Chemical conditions	Acutely toxic chemical conditions	Low dissolved oxygen with chronic toxicity	Low dissolved oxygen, nutrient enriched, no recurrent toxicity	Adequate dissolved oxygen, no acute/chronic toxicity effects, elevated nutrients	No effects evident, background conditions within the ecoregion, good dissolved oxygen
AND/OR Physical conditions	Total habitat loss, extremely contaminated sediments	Severe habitat degradation, severe sediment contamination	Modified stream channel, heavy siltation, canopy removal	Good habitat, no significant channel modifications	Excellent habitat, no modification evident
Example of perturbation	Toxic discharges, acid mine drainage, severe thermal stress, anoxic hypolimnetic discharges	Municipal and industrial discharges, intermittent acute impacts	Municipal sewage, combined sewers, heavy agricultural use, nonacid mine drainage, moderate thermal stresses	Minor sewage inputs, most agricultural and urban storm-water diffuse point affected areas	No or minimal human impact, no perturbation evident, reference assessment reaches

Sources: After Karr et al. (1986) and Rankin, Yoder, and Mishne (1990).

FIGURE 12.6. Total number of fish species versus stream order for least disturbed sites along the Embarras River in Illinois. (After Fausch, Karr, and Yant, 1984.)

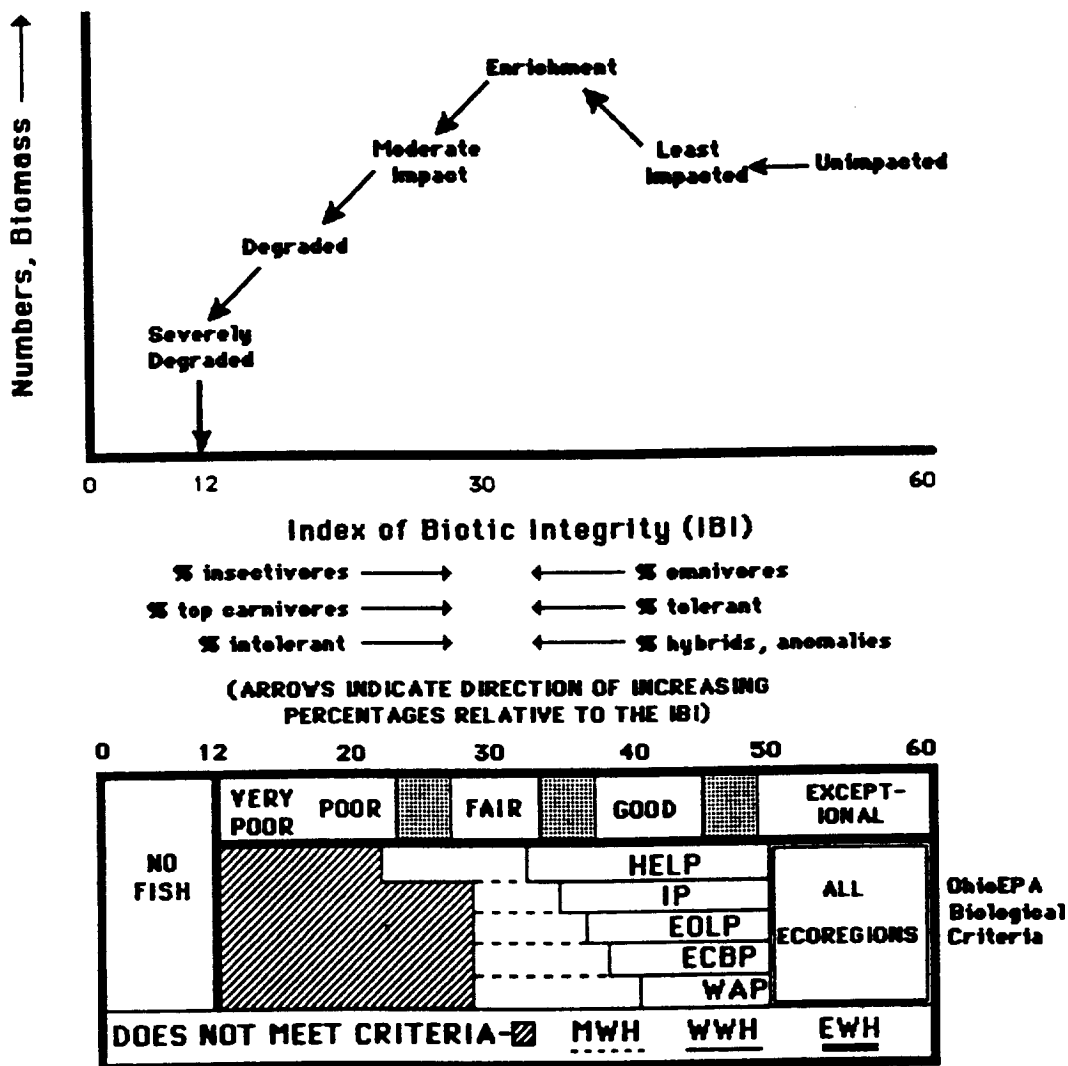
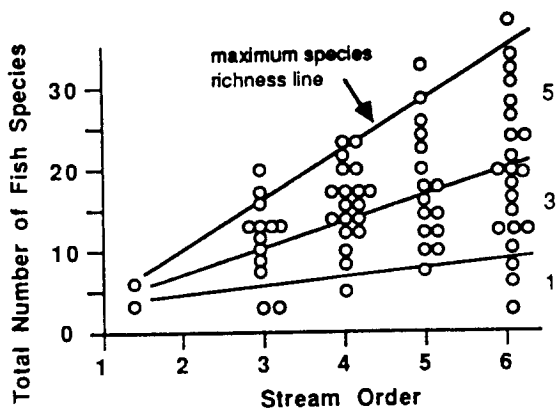


FIGURE 12.7. Biotic integrity (IBI) response to different levels of stress. (After Ohio EPA, 1987.)

with one or more second-order tributaries have order 3, and so forth) as shown in Figure 12.7. It is known that the total number of fish and the species richness vary with the stream size within the ecoregion. Unfortunately, most of the undisturbed reference reaches in most populated ecoregions are small headwater streams. Therefore, when assessing the streams a judgment must be made on the richness of the most probable species of higher order streams as indicated in Figure 12.7.

The *Invertebrate Community Index* (ICI), proposed by the Ohio Environmental Protection Agency (Ohio EPA, 1987), is a modification of the IBI index. This report contains extensive guidelines on the use of biotic indices and criteria. Use of macroinvertebrates has some advantages over fish sampling, as shown in the following list (Mason, 1991; Ohio EPA, 1987):

1. Macroinvertebrates form permanent, relatively immobile communities.
2. Microinvertebrates can be easily collected and are present in large numbers in even the smallest of streams.
3. They can be easily sampled at a low cost.
4. They occupy all stream habitats and, even within family and generic grouping, display a wide range of functional feeding preferences (i.e., predators, collectors, shredders, scrapers).
5. They are quick to react to environmental change and are incapable of escape.
6. They inhabit the middle of the aquatic food web and are a source of food for fish and other aquatic and terrestrial animals.
7. Taxonomy has developed to the point where species-level identification of many larval forms are available along with much environmental and pollution tolerance information.

It should be pointed out that the saprobien system of Kolkwitz and Marson uses extensively benthic macroinvertebrates as test organisms. Publications by Sládeček (1979) and Mason (1991) list pollution tolerance levels (saprobien numbers) for a number of benthic invertebrates. The Ohio EPA's system uses the following organisms as indicators of pollution: dipterans, mayflies, caddisflies, midges, plus other location-specific tolerant organisms, such as worms, snails, and mollusks.

The toxicity impact of various constituents on aquatic biota was incorporated into the formulation of chemical water quality criteria (U.S. EPA, 1987; also, see Chapter 13). The IBI and ICI indices also enable formulation of the ecoregional water quality criteria once the relationship between the water quality and habitat degradation and pollution loads is established. Following this concept a proper and attainable use of the

body of water can be determined by use attainability analysis, which is mandated by the Clean Water Act. This concept is introduced in Chapter 16.

Biological Criteria

Until recently the methods used to achieve water quality goals were based primarily on chemical and toxicological criteria (see the Appendix). Recent emphasis on the biological criteria in the United States (note that the biological criteria based on the saprobien concept have been used in Europe for a long time) make aquatic life protection the focus. Several states have adopted the biotic criteria in their water quality assessment and plans. At the time of writing this book Ohio's approach appeared as the most progressive and advanced (Ohio EPA, 1987).

The use of biological communities, particularly fish and macroinvertebrates, offers a holistic system approach to surface-water quality assessment and management. Aquatic organisms not only integrate a variety of environmental influences, but complete their life cycles in the body of water and by so doing are continuous monitors of environmental quality.

Biological criteria can be developed using the biosurvey/ecoregional approach, which may be outlined as follows:

1. Classification and selection of reference sites
2. Conduct water quality and biological surveys at the reference sites
3. Develop a predictive model
4. Conduct biological and chemical surveys at the test site, a habitat assessment, and a survey of existing and potential sources of contaminations
5. Apply the model to the test site (estimation of unimpacted water quality and biological composition at the test site)
6. Determination of impairment. Ascertain toxicity levels at the test site by bioassays with the test organisms obtained from reference reaches
7. Determine acceptable water quality goals and the definitions of site-specific criteria (standards) for water and sediments (site specificity will be more profound for sediments)

The reasons for establishment of site-specific criteria based on the reference water quality and habitat integrity are the shift from "end of the pipe" control to an integrated point and nonpoint source abatement, and aquatic habitat restoration.

The biotic integrity indices can provide the measure of the water quality goals. For example, water quality goals can be met if the biotic

IBI and/or ICI index is classified as "good" to "excellent." The biotic indices and water body assessments described herein were incorporated into Rapid Bioassessment Protocols by the U.S. EPA (Plafkin et al., 1989).

Nitrogen Cycle: Nitrification and Denitrification

Nitrogen transformation and cycle in aquatic bodies of water, which is affected by discharges of waste loads from point and nonpoint sources, is almost as important as the organic matter-dissolved oxygen cycle. Nitrogen transformation may also affect the dissolved oxygen balance. Nitrogen in aquatic water bodies may exist in several forms: (a) dissolved nitrogen gas (N_2); (b) organic nitrogen incorporated into proteinaceous organic matter; (c) ionized and nonionized ammonia (NH_4^+ and NH_3); (d) nitrite ion (NO_2^-); and (e) nitrate ion (NO_3^-).

The nitrogen cycle in aquatic environments is shown on Figure 12.8. The reactions take place in water under aerobic and anaerobic conditions, in sediments mostly under anaerobic conditions, and in the sediment-water interface, where the transformation may be both aerobic and anaerobic. The processes in the sediments and water are similar to nitrogen transformation in soils.

Ammonium. In aquatic environments, decomposers break down organic proteinaceous matter and, consequently, ammonia is released. This process of biological degradation and release of ammonia is called *deamination*. Deamination may occur in sediments, water, soils, and also in biological treatment processes. Three ways of producing ammonia from organically bound nitrogen can be distinguished (Painter, 1970):

1. From extracellular organic nitrogen containing compounds, chemically or biochemically (for example, from urea)
2. From living bacterial cells during endogenous respiration
3. From dead and lysed cells

Depending on the pH of water the dissolved ammonium gas and ammonia ions will exist in the equilibrium given by the following reaction



At a pH of 7 or below most of the ammonia will be ionized. At higher pH levels the proportions of deionized ammonia will increase.

Deionized ammonium is toxic to fish, while ionized ammonia is a nutrient to algae and aquatic plants and also exerts dissolved oxygen demand (see the nitrification process below). The un-ionized ammonium

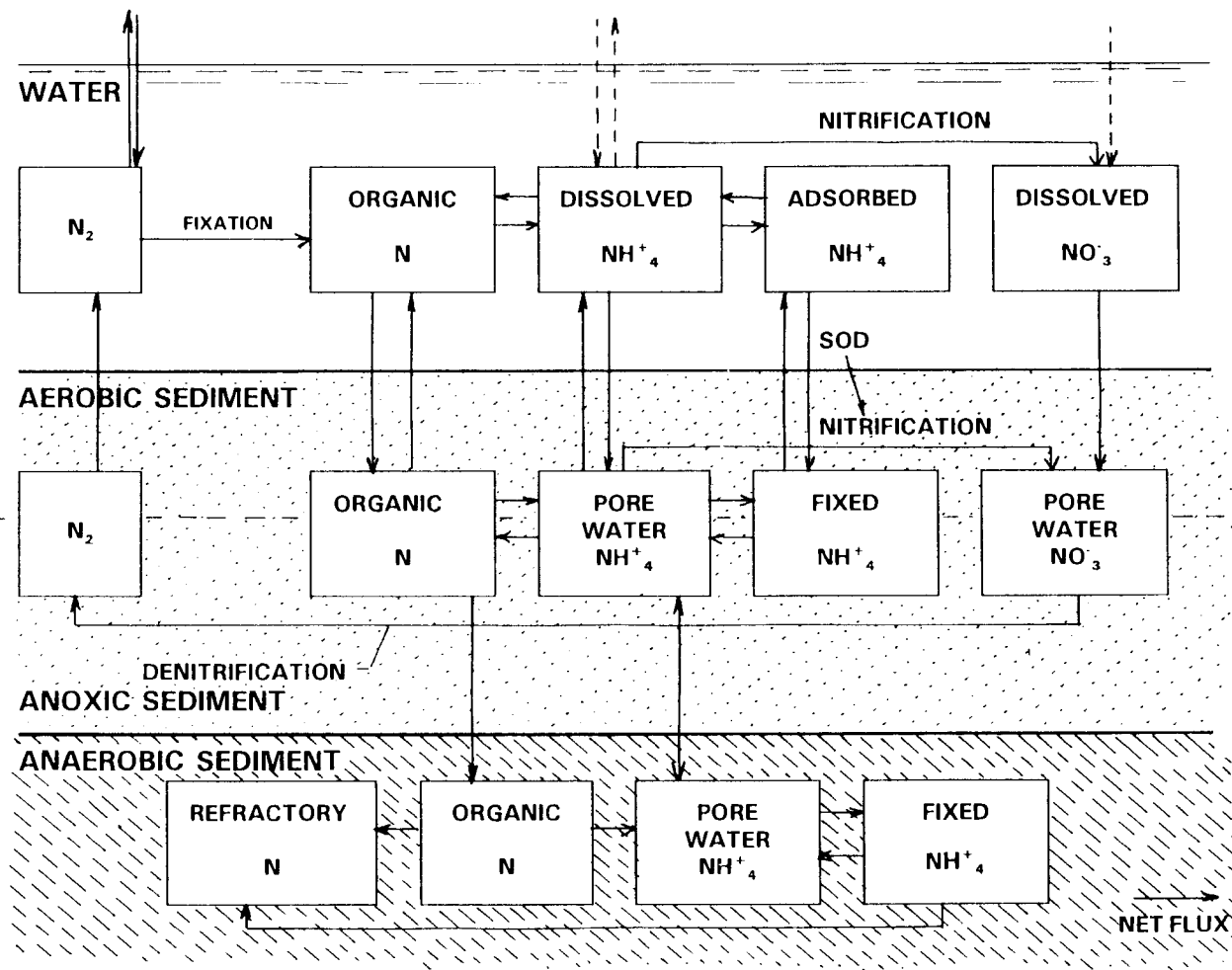
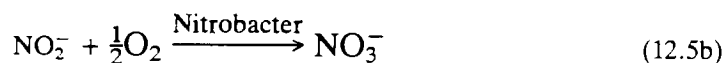
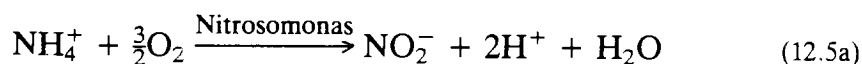


FIGURE 12.8. Nitrogen cycle in aquatic bodies.

is a gas that will mostly volatilize from water. The amount of un-ionized ammonia dissolved in water, which is toxic, depends on its solubility, which is affected by temperature and pH. Consequently, the toxicity of deionized ammonium depends on the same parameters and on the type of biological species being affected. The toxicity criteria for deionized and total ammonia concentrations are included in the Appendix.

As shown in Chapter 6, ammonia ions have an affinity for adsorption on soil particles (primarily clay and organic fractions) and by the same mechanism on sediments. Absorbed (fixed) ammonia is not available as a nutrient in the primary organic matter production process and is not toxic.

Nitrification. Nitrification is a two-stage process involving two types of chemotrophic bacteria (bacteria deriving energy from exothermic nitrification reactions). In the first step an ammonia ion is converted to nitrite by *Nitrosomonas*; in the second step nitrite is converted to nitrate by *Nitrobacter*. In simplified form the reactions may be written as



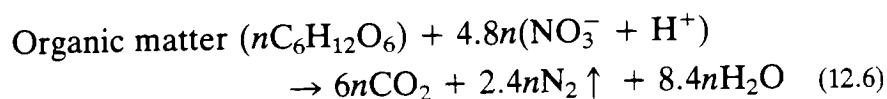
Stoichiometrically, the oxygen requirement for the overall nitrification reaction is 4.56 mg O₂/mg of NH₄⁺. However, since the reaction is basically autotrophic the bacterial growth essentially follows Equation (12.1), that is, oxygen is produced as a result of bacterial growth. Consequently, the overall oxygen requirement for nitrification is less than the stoichiometrical value. Wezernak and Gannon (1968) measured the O₂ requirement in nitrification as about 4.3 mg O₂/mg of NH₄⁺.

Several environmental factors affect the nitrification rate (Krenkel and Novotny, 1980):

1. The reaction is strictly aerobic. If the oxygen concentration is depleted below 2 mg/l, the reaction rate decreases rapidly.
2. Its optimum pH range is between 8 and 9. Below a pH of 6, the reaction essentially ceases.
3. The nitrifying organisms tend to be attached either to sediments or solid surfaces.
4. The growth rate of nitrifying organisms is much less than the growth rate of heterotrophic decomposers. Consequently, if higher concentrations of biodegradable organics are present, the heterotrophs will limit the growth of nitrifiers and nitrification will be suppressed.

5. The optimal temperature range is between 20° and 25°C. The rate decreases if the temperature is less or more than the optimum.

Denitrification. Under anoxic conditions, the nitrate–nitrogen ion becomes the electron acceptor in the essentially organic matter oxidation reaction. The reaction can be represented as (Schindler, 1985)



Other kinds of organic compounds may serve as the source of energy in this reaction. For example, dissolved methane that evolves from the anaerobic decomposition in the sediments can be used by the denitrifying organism in the interstitial layer.

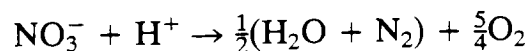
This reaction represents a loss of nitrogen from water since the produced nitrogen gas volatilizes because it is in excess of the equilibrium between the gaseous nitrogen in the air and that dissolved in water. Only a few planktonic algal organisms (for example, blue-green algae) are capable of fixing dissolved N₂-nitrogen in the photosynthetic reaction.

In the anoxic sediments, the decomposers break down the organic matter to methane and carbon dioxide (methanogenesis) and nonbiodegradable residues. Ammonia is released in this process. Conditions at the sediment–water interface can then have a major impact on the N balance of the system. Unless the water at the bottom is completely anoxic, an aerobic thin sediment upper layer will form. Nitrification should occur in the oxic sediment layer. Because the gradient of NO₃⁻ is downward to the sediment (zero nitrate concentration) and not upward, the NO₃⁻ thus formed diffuses back into the anaerobic sediment where denitrification is certain (Keeney, 1973). This is referred to as a *simultaneous nitrification–denitrification process* by which nitrogen is lost from the system. By combining Equations (12.5) and (12.6), the following simplified overall chemical balance can be obtained

Nitrification



Denitrification



Overall reaction



If these reactions proceed simultaneously three-quarters of a mole of O_2 is used per each mole of ammonia converted to nitrogen gas (1.7 g of O_2/g of $NH_4^+ - N$). This O_2 consumption will then be accounted for in the sediment oxygen demand, which is shown in the next section.

Experiments by Keeney, Schmidt, and Wilkinson (1975) have shown that about one-third of nitrogen input to a lake may be lost from the body of water by the simultaneous nitrification–denitrification process. Kadlec (1988) and others documented similar nitrogen losses for wetlands.

Not all of the ammonia released by decomposition in the sediments will be returned to water. Part of it will remain adsorbed on the sediments and part will be recycled into rooted macrophytes growing in the sediments located in the shallow euphotic (light abundant) zone, provided that the hydraulic conditions are favorable for the macrophyte growth. Simultaneous nitrification–denitrification can only occur in the sediment–water interface. Nitrification in free-flowing water is relatively rare (Tuffey, Hunter, Matulewich, 1974).

DISSOLVED OXYGEN PROBLEM

As pointed out in the first section of this chapter the waste discharges from diffuse sources embody rarely a direct threat to the dissolved oxygen regime of receiving bodies of water. The exceptions to this statement include runoff with a high concentration of biodegradable organics from concentrated animal operations, spring runoff from fields with manure spread on still frozen soils, and some instances of CSO discharges. However, indirect and secondary effects of diffuse-pollution discharges on the dissolved oxygen regime caused by nutrient enrichment and consequent sediment oxygen demand may be profound and detrimental to a receiving body of water and its uses (Novotny and Bendoricchio, 1989).

Dissolved Oxygen Standard

Water quality investigations and toxicity studies indicate that the dissolved oxygen content is the most important parameter for protecting fish and aquatic biota. The level of protection can be classified into three groups (Krenkel and Novotny, 1980). The first level of oxygen concentration would just permit the fish to live (prevention of fish-kills), the second level would permit the fish or aquatic organisms to be active to a specified degree, the third level would allow the organisms to live, grow, and reproduce in a given area. Typically, fish-kills occur when fish are exposed for a certain period of time (a few hours) to dissolved oxygen

concentrations of less than 3 mg/l. Present standards provide, however, the third level of protection.

The levels of fish tolerance to low dissolved oxygen stresses vary. Cold-water fish (salmon, trout) and biota require higher dissolved oxygen concentrations than warm-water fish and biota. Therefore the standards distinguish between cold- and warm-water biota protection. Further considerations are given to whether the body of water or a portion thereof represents a breeding area or a migratory route. Appendix B contains the values of water quality criteria for U.S. bodies of water.

For the general well-being of trout, salmon, and associated cold-water biota, the dissolved oxygen concentration should not be below 6.5 mg/l. Under extreme conditions, the concentration may range between 5 and 6 mg/l (second level of protection), provided that all other water quality parameters are within the acceptable ranges. The 4-mg/l dissolved oxygen standard for cold-water biota and 3 mg/l for warm-water biota are essentially set to prevent fish-kills, and should not be included in standards aimed at protecting the biological integrity of receiving waters. Both biodegradable organics (five-day biochemical oxygen demand) and dissolved oxygen standards are used in the Economic Community and other countries of Europe.

Dissolved Oxygen Balance of Streams

The content of biodegradable organics in waste discharges (from both point and nonpoint sources) is commonly expressed as the biochemical oxygen demand (BOD). The reader is referred to standard environmental and water quality texts for the explanation of the fundamentals of the BOD reaction and its course (for example, see Krenkel and Novotny, 1980). Other tests expressing the amount of matter that will then complement the BOD data include chemical oxygen demand (COD), total organic carbon (TOC), and volatile suspended solids (VSS).

When waste discharge containing biodegradable organics is introduced into receiving waters, decomposers begin immediately to decompose the biodegradable organic matter and then convert it, ultimately, to carbon dioxide (alkalinity), water, and mineral and organic nonbiodegradable residues. According to Equation (12.2), dissolved oxygen is used in this biochemical reaction.

The dissolved oxygen balance is an important water quality consideration for streams and estuaries. Reservoirs and lakes can also be affected; however, the autochthonous productivity of these water bodies may increase the dissolved oxygen balance therein and overwhelm the effect of allochthonous inputs of oxygen-demanding wastes. This is especially

true for deep reservoirs and lakes that stratify during the productive season. Shallow river mixed impoundments can be considered as streams.

In streams, the dissolved oxygen concentration is a response to various oxygen sinks and sources. The sinks of oxygen, that is the biochemical and biological processes that use oxygen, include:

1. Deoxygenation of biodegradable organics whereby bacteria and fungi (decomposers) utilize oxygen in the biooxidation-decomposition process.
2. Sediment oxygen demand (SOD), where oxygen is utilized by the upper layers of the bottom sediment deposits.
3. Nitrification, in which oxygen is utilized during oxidation of ammonia and organic nitrogen to nitrates.
4. Respiration by algae and aquatic vascular plants that use oxygen during night hours to sustain their living processes.

Major oxygen sources are:

1. Atmospheric reaeration, where oxygen is transported from the air into the water turbulence at the air-water interface.
2. Photosynthesis, where chlorophyll-containing organisms (producers such as algae and aquatic plants) convert CO_2 (or alkalinity of water) to organic matter with a consequent production of oxygen.

The basic concept of the dissolved oxygen balance in streams was proposed by Streeter and Phelps (1925) and was later summarized by Phelps (1944). This concept, which is primarily used for evaluation of the point source impact on water quality, is still used with some modifications. However, it will be subsequently shown that in present state-of-the-art models (WASP or QUAL) the original dissolved oxygen balance concept proposed by Streeter and Phelps constitutes only a small part of the overall water quality picture. Furthermore, the original model presumes a point discharge of waste.

The basic assumptions of more recent dissolved oxygen balance models applicable to diffuse sources are:

1. The biochemical oxygen demand (BOD) load to the stream can take place either as a point source (combined or storm sewer outfall or a major tributary) or as a distributed source.
2. The flow conditions do not change considerably during the time of analysis—an assumption that is often violated, but as shown by Meadows, Weeter, and Green (1978), no significant error will result as

long as $X \leq 0.05 Q_0/q_r$, where X is the length of the reach, Q_0 is the flow at the head of the reach, and q_r is the internal inflow per unit length of the reach.

3. Effects of nitrification and photosynthesis are negligible (an assumption that is often violated in streams where most pollution loads originate from diffuse sources).
4. Effects of dispersion, such as in tidal estuaries, is negligible.
5. The reach is uniform with no appreciable changes in cross-sectional dimensions, flow velocity, and depth.

Under these assumptions, the equation for the BOD and dissolved oxygen concentration becomes (Meadows, Weeter, and Green, 1978; Thomann and Mueller, 1987):

BOD variations

$$L = L_0 e^{-K_r(X/u)} + \frac{L_r}{K_r}(1 - e^{-K_r(X/u)}) \quad (12.7)$$

Dissolved oxygen variations

$$\begin{aligned} D = & D_0 e^{-K_a(X/u)} + \frac{K_r}{K_a - K_r}(e^{-K_d(X/u)} - e^{-K_a(X/u)})L_0 \\ & + \left[\frac{K_d}{K_d K_r}(1 - e^{-K_a(X/u)}) - \frac{K_d}{(K_a - K_r)K_r}(e^{-K_r(X/u)} - e^{-K_a(X/u)}) \right] L_r \\ & + (1 - e^{-K_a(X/u)}) \frac{S_B}{K_a H} \end{aligned} \quad (12.8)$$

where

$D = c_s - c$ is the oxygen deficit at the end of the reach (mg/l)

D_0 = initial oxygen deficit (mg/l)

c_s = oxygen saturation (mg/l)

c = oxygen concentration at the end of the reach (mg/l)

L_0 = initial ultimate BOD concentration (mg/l)

L_r = ultimate BOD concentration input due to the lateral distributed sources (mg/l-day)

K_r = overall BOD removal coefficient (day^{-1})

K_a = reaeration coefficient (day^{-1})

K_d = BOD deoxygenation coefficient (does not include BOD removal by sedimentation of particulate BOD component) (day^{-1})

X = length of the reach (m)

u = average stream flow velocity (m/day)

S_B = sediment oxygen demand (SOD) ($\text{g}/\text{m}^2\text{-day}$)

H = average depth of flow (m)

Equation (12.7) does not include the effect of nutrients and photosynthesis.

Coefficient of deoxygenation. The coefficient of deoxygenation measured in streams is a composite of three processes that may or may not occur simultaneously. These processes are deoxygenation of BOD in free-flowing water, effect of benthic slimes on BOD absorption and removal, and sedimentation of particulate organics. Hence (Novotny and Krenkel, 1975)

$$K_r = K_1 + K_3 + B \quad (12.9)$$

and

$$K_d = K_1 + B$$

where

K_1 = deoxygenation rate constant for free-flowing water approximately equaling the laboratory BOD bottle (respirometer) rate

K_3 = BOD removal by sedimentation, which does not result in oxygen demand

B = effect of absorption of BOD by benthic slimes, which results in oxygen demand

The magnitudes of the laboratory BOD removal coefficient, K_1 , have been reported in ranges of 0.1 to 0.6 day^{-1} (K_1 base e), while the overall ranges of the deoxygenation rate, K_r , can be between 0.1 and 5 day^{-1} , depending on the hydraulic and biological character of the stream and biodegradability of the organic pollution.

Atmospheric reaeration. The gas transfer theory demonstrated that stream reaeration by atmospheric oxygen is proportional to the turbulence intensity at the water surface and the ratio of the surface area to water volume. The magnitude of the reaeration coefficient has been subjected to intensive investigations ranging from mostly theoretical, such as that of O'Connor and Dobbins (1958) to experimental field investigations, such as those by Churchill, Elmore, and Buckingham (1962). The bulk of the research on stream reaeration was conducted and published 30 to 40 years ago. These studies usually resulted in a relationship of the following type

$$K_a = CU^m H^n S_e^p \quad (12.10)$$

TABLE 12.6 Summary of Coefficients for the Reaeration Formula

Investigators	C	m	n	p
O'Connor and Dobbins (1958)	3.92	0.5	-1.5	0
Churchill, Elmore, and Buckingham (1962)	5.05	0.962	-1.673	0
Owens, Edwards, and Gibbs (1964)	3.0	0.67	-1.65	0
Krenkel and Orlob (1962)	173.6	0.408	-0.66	0.408

Note: K_a is in day^{-1} ; U = stream velocity in m/sec; H = flow depth in m; S_e slope is in m/m (dimensionless).

where

U = stream velocity (typically m/sec)

H = flow depth (m)

S_e = energy slope or slope of water surface (dimensionless)

C, m, n, p = coefficients

The coefficients C, m, n, p for some most common estimating equations are given in Table 12.6.

Sediment oxygen demand (SOD). Stream bottoms are commonly covered by sediments. It is the stream morphology that determines what type of sediments will deposit on the bottom and what role, if any, the sediments will play in the transport and storage of pollutants. The sediments stored on the bottom are called *bedload*, while those in suspension are a part of *washload* (see Chapter 5 for a more detailed discussion on sediment transport).

Generally, if the stream velocity is high and the bed is composed mostly of sand and gravel, which have a very low affinity for adsorbing pollutants, most of the fine (cohesive and flocculent) sediments remain in suspension, the organic sediment content will be very low, and the bed will exhibit very low to no demand for oxygen. Consequently, for such streams the BOD removal rate due to sedimentation of particulate organics will be negligible. Fine sediments will deposit if the bottom shear stress drops below a certain threshold. As a rule of thumb, if flow velocity is below 0.3 m/sec, then organic particulates and other finer particles may settle to the bottom and become a part of the organic bottom sediments. The aerobic and anaerobic decomposition processes in the sediment layer are then responsible for the sediment oxygen demand (SOD).

The first scientific measurements of SOD by Fair, Moore, and Thomas (1941) related the oxygen demand of river sludges (primarily sewage solids) to the rate of deposition of organic sludge solids on the bottom and to the organic content of the sediments (expressed as Volatile Sol-

ids). Others related the SOD to the chemical oxygen demand (COD) of the sediment (Gardiner, Auer, and Canale, 1984). The measurements revealed that SOD is proportional to the square root of the organic content of the sediment expressed either as VS or COD.

After primary and secondary treatment was installed in most of the U.S. point wastewater discharges, the dominance of diffuse urban and rural sources became evident. Recent investigations have revealed that SOD includes oxygen demand from several separate processes (Bowman and Delfino, 1980; DiToro et al., 1990): (1) biological respiration and oxygen consumption of all living organisms residing in the upper aerobic benthic zone; (2) chemical oxidation of reduced substances in the sediments, such as reduced iron, manganese, sulfides after they come to oxygen-containing water; (3) biochemical oxidation of methane and ammonia (simultaneous nitrification–denitrification) that evolve from the lower anaerobic sediments.

Rudd and Taylor (1980) found that most of the SOD is attributed to the biochemical oxidation of evolving methane. Since methane has low solubility in water, the SOD portion due to methane oxidation by aerobic bacteria may be limited since only dissolved methane is bioavailable. The simultaneous nitrification–denitrification may be controlled by the growth rate of nitrifiers. The oxidation of methane and the simultaneous nitrification–denitrification in the interstitial layer were explained in the preceding section.

The *diagenesis model* proposed for SOD estimation by DiToro and coworkers (DiToro et al., 1990; DiToro, 1986) relates SOD to the input of particulate organic matter into the bottom sediment layer and its decomposition. In this concept an oxygen demand equivalent for reduced species (electron donors), such as organic carbon ($\text{CH}_4(\text{aq.})$), HS^- , Fe^{2+} , and NH_4^+ , is estimated. A mass-balance equation of the oxygen demand equivalents is then used to calculate their flux in the sediment–water interface, a consequence of which is the SOD. The production of soluble reduced end products in the sediments occurs via the bacterial breakdown of particulate organic matter. The most important products of the breakdown of organic matter are methane and ammonia. Inorganic reduced compounds may be significant in cases where insufficient organic matter is deposited into the sediment layer. Both methane and ammonia will exert an oxygen demand in the upper aerobic layer of the sediment in the sediment–water interface. The inorganic portion of SOD may amount to $0.3 \text{ g/m}^2\text{-day}$ (Walker and Snodgrass, 1986). Figure 12.9 shows the diagenesis concept of the SOD. The diagenesis model and extensive field data measured in the Milwaukee River (DiToro et al., 1990) led to several important conclusions on the mechanism of SOD:

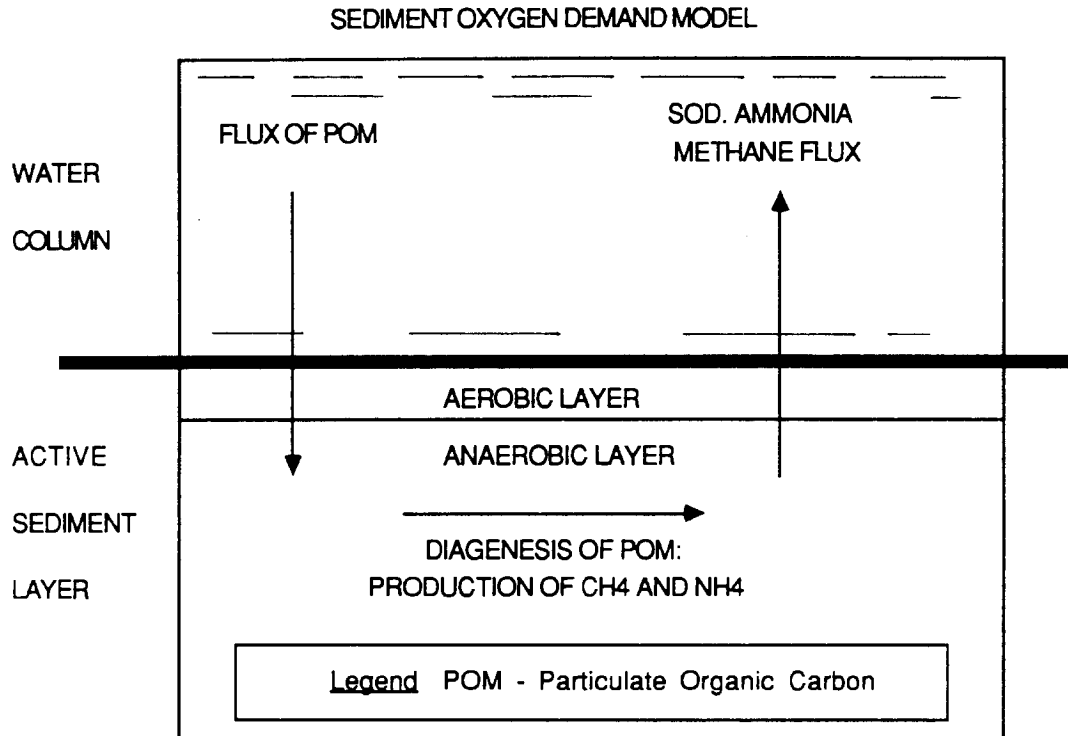


FIGURE 12.9. Diagenesis concept of sediment oxygen demand. (After DiToro et al., 1990, by permission of ASCE.)

1. Since SOD is a result of the oxidation of end products of an anaerobic diagenesis process in the sediment, it is related to the availability of oxygen in the sediment–water interface, which is related to the concentration of dissolved oxygen in water. The dissolved oxygen dependency of SOD has been observed for dissolved oxygen concentrations that are much larger than concentrations known to limit oxidation (10% of saturation).
2. SOD is related to the solubility of methane in water, which is relatively low. For this reason the dependence of SOD on the reduced carbon (methane) flux from the sediments is not linear. DiToro et al. (1990) developed a square root dependency function between SOD and carbon fluxes.

The SOD model by DiToro et al. is complex and requires calibration with field data that are not commonly measured (e.g., gas fluxes from the sediment). Typical in situ measured SOD data are given in Table 12.7.

Temperature Effects

Almost all of the reaction rates in the oxygen balance process are temperature dependent. The relationship is expressed in the following form:

TABLE 12.7 In-Situ Measured Sediment Oxygen Demand

Benthic Deposit	Sediment Oxygen Demand (g/m ² -day)	Temperature (°C)	Investigator
Rivers and estuaries			
River sludges	0.9		Oldaker et al. ^a
River muds—2 cm	3.4		McDonnel and Hall (1969)
25 cm	6.4		
Sandy bottom	0.2–1.0		Thomann (1972)
Mineral soils	0.05–0.1		
<i>Sphaerotilus</i> covered bottom 10 g dry wt/m ²	7.0		
Estuarine mud	1–2		
River Benaviken	1.44	2	Edberg and Hofsten (1973)
	0.68	10	
River Sjomosjon	0.31	0	
Jaders Bruk	0.84	9	
	1.4	2	
Lower Milwaukee River	2.6–6.7	20	Kreutsberger et al. (1980)
Potomac estuary	2.5	15	O'Connel and Wales ^a
Lakes			
Lake Erken	2.6	18	Edberg and Hofsten (1973)
	0.43	4	
Lake Ramsen	2.3	17	
Lake Michigan–Green Bay	1.6–1.9	12	Paterson, Epstain, and McEnvoy (1975)

^aQuoted in Filos and Molof (1972).

$$K_T = K_{20}\theta^{(T-20)} \quad (12.11)$$

where

K_T = reaction rate at temperature T (T in °C)

K_{20} = reaction rate at the reference temperature of 20°C

θ = thermal factor, which has the following accepted values

Deoxygenation rates K_r , K_d , K_1	$\theta = 1.047$
Reaeration rate K_a	$\theta = 1.025$
Sediment oxygen demand, S_B	$\theta = 1.05-1.06$

Temperature also affects the oxygen saturation value, which can be approximated as follows (Krenkel and Novotny, 1980):

$$c_s = 14.652 - 0.41022T + 0.007991T^2 - 0.000077774T^3 \quad (12.12)$$

Temperature conditions selected for the waste assimilative capacity evaluation should correspond to the average temperature of the warmest month of the year (e.g., August). Statistically rare temperatures in combination with statistically rare low flow would lead to excessively rare and unrealistic conditions.

Example 12.2: Dissolved Oxygen Computation*

A medium-sized city with a population of 50,000 and a city area of 20 km² is discharging its treated sewage and storm runoff by separate sewer systems into an impounded river. The length of the impounded reach is $X = 20$ km. Lateral (diffuse) flow increase is about $q_r = 0.1$ m³/sec-km, with an average BOD₅ concentration of 3 mg/l.

Calculate the dissolved oxygen concentration in the river during a storm that would follow a critical dry period. The storm resulted in a runoff event with a volume of $R = 1.5$ cm that lasted 2 hours. The measured sediment oxygen demand was 2 g/m²-day. The per capita BOD₅ loads in sewage is assumed as 54 g/cap.-day, and sewage flow is 400 l/cap.-day (Novotny et al. 1989).

Other given parameters:

Low flow (7 days duration—10 years expectancy)	$Q = 15$ m ³ /sec
Average depth in the impoundment	$H = 3$ m
Average width of the impoundment	$B = 64.33$ m
Water temperature	$T = 25^\circ\text{C}$
Deoxygenation rate (at 20°C)	$K_r = K_d = 0.25$ day ⁻¹
Upstream BOD ₅ concentration = 1.2 mg/l	
Upstream dissolved oxygen concentration $c_0 = 7.6$	

Solution From Equation (12.12) the oxygen saturation concentration at 25°C becomes $c_s = 8.4$ mg/l.

Sewage characteristics

Flow

$$q = 0.4 \text{ m}^3/\text{cap.-day}$$

$$0.4 \times 50,000 (\text{pop}) \times 1 (\text{day})/86,400 (\text{sec}) = 0.23 \text{ m}^3/\text{sec}$$

* This example is presented to illustrate the concepts under steady-state assumption. Such an assumption may not be appropriate for diffuse-pollution discharges driven by storm-water events. Dynamic models (see the section on models in this chapter) are more proper for this type of analysis.

$$\begin{aligned}\text{BOD}_5 \text{ discharge (sewage)} &= 54 \text{ g/cap.-day} \\ &= 54 \text{ (g/cap.-day)} \times 50,000 \text{ (pop)} \times 1 \text{ (day)}/86,400 \text{ (sec)} \\ &= 31.25 \text{ g/sec}\end{aligned}$$

Assuming 90% removal in the sewage treatment plant, the BOD₅ loading from sewage is

$$0.1 \times 31.25 = 3.12 \text{ g/sec}$$

Storm characteristics

Flow from storm sewers

$$\begin{aligned}Q_s &= 1.5 \text{ (cm)} \times 0.01 \text{ (m/cm)} \times 20 \text{ (km}^2\text{)} \times 10^6 \text{ (m}^2\text{/km}^2\text{)} \\ &\quad \times \frac{1}{2} \text{ (hr)} \times 1 \text{ (hr)}/3600 \text{ (sec)} \\ &= 41.67 \text{ m}^3\text{/sec}\end{aligned}$$

(Note that the flow of sewage—dry-weather flow—is insignificant when compared to the storm water—wet-weather—flow). From Table 1.4 the average BOD₅ concentration of urban storm water is 30 mg/l(g/m³). Hence the BOD load from storm water is

$$\text{BOD}_5 \text{ discharge (storm)} = 41.67 \text{ m}^3\text{/sec} \times 30 \text{ (g/m}^3\text{)} = 1250 \text{ g/sec}$$

Note again that the BOD load by treated sewage during the wet-weather period is small when compared to the BOD load by urban runoff.

Total initial river flow

$$\begin{aligned}Q_0 &= \text{river flow} + \text{storm water flow} + \text{sewage flow} \\ &= 15 + 41.67 + 0.23 = 56.9 \text{ m}^3\text{/sec}\end{aligned}$$

Terminal (at the end of the reach) flow

$$Q_e = Q_0 + X \times q_r = 56.9 + 20 \times 0.1 = 58.9 \text{ m}^3\text{/sec}$$

Average flow

$$Q_{av} = (56.9 + 58.9)/2 = 57.9 \text{ m}^3\text{/sec}$$

Average velocity

$$u_{av} = Q_{av}/(B \times H) = 57.9/(63.33 \times 3) = 0.3 \text{ m/sec} = 25.92 \text{ km/day}$$

According to Meadows, Weeter, and Green (1978), the steady-state dissolved oxygen model (Eqs. (12.7) and (12.8)) can be used when

$$X = 20 \text{ km} < 0.05 Q_0/q_r = 0.05 \times 56.9/0.1 = 28.45 \text{ km}$$

BOD input from lateral diffuse flow

$$\begin{aligned} L_r &= \frac{1.4(\text{lateral flow} \times \text{BOD}_5 \text{ conc})}{\text{cross-sectional area}} \\ &= \frac{1.4 \times 0.1(\text{m}^3/\text{km-sec}) \times 0.001(\text{km/m}) \times 86,400(\text{sec/day}) \times 3(\text{g/m}^3)}{[3 \times 64.33]\text{m}^2} \\ &= 0.19 \text{ g/m}^3\text{-day} \end{aligned}$$

Initial Conditions Initial ultimate BOD concentration downstream from the city

$$\begin{aligned} L_0 &= 1.4 \times \left[\frac{\text{sewage BOD}_5 \text{ load} + \text{storm-water BOD}_5 \text{ load}}{Q_0} \right. \\ &\quad \left. + \text{upstream BOD}_5 \right] \\ &= 1.4 \times \left[\frac{3.12 + 1250}{56.9} + 1.2 \right] = 23.22 \text{ mg/l} \end{aligned}$$

Initial oxygen deficit

$$D_0 = c_s - c_0 = 8.4 - 7.6 = 0.8 \text{ mg/l}$$

The coefficient of reaeration can be calculated by Equation (12.10) with the O'Connor/Dobbins parameters (Table 12.6)

$$K_a = 3.92U^{0.5}H^{-1.5}S_e^0 = 3.92 \times 0.3^{0.5} \times 3^{-1.5} \times 1.0 = 0.41 \text{ day}^{-1}$$

Correct K_a for 25°C

$$K_{a25^\circ\text{C}} = 0.41 \times 1.25^{(25-20)} = 0.46 \text{ day}^{-1}$$

Correct K_r for 25°C

$$K_{r25^\circ\text{C}} = 0.25 \times 1.047^{(25-20)} = 0.31 \text{ day}^{-1}$$

The dissolved oxygen deficit at the end of the 20-km-long reach is then computed by Equation (12.8). In the calculation substitute the travel time t for X/u or

$$t = 20 \text{ km}/25.92 \text{ km/day} = 0.77 \text{ days}$$

$$\begin{aligned} D &= 0.8e^{-0.46 \times 0.77} + \frac{0.31}{0.46 - 0.31}(e^{-0.31 \times 0.77} - e^{-0.46 \times 0.77})23.22 \\ &+ \frac{0.31}{0.46 \times 0.31} \left[(1 - e^{-0.46 \times 0.77}) \right. \\ &\quad \left. - \frac{0.31}{(0.46 - 0.31)0.31}(e^{-0.31 \times 0.77} - e^{-0.46 \times 0.77}) \right] 0.19 \\ &+ (1 - e^{-0.46 \times 0.77}) \frac{2}{0.46 \times 3} = 6.20 \text{ mg/l} \end{aligned}$$

At the end of the reach the dissolved oxygen concentration will be

$$c = c_s - D = 8.4 - 6.2 = 2.2 \text{ mg/l}$$

For the conditions presented in this example the dissolved oxygen standard would be violated.

Several environmental texts contain more detailed discussions of the dissolved oxygen balance model, its fundamentals, and variations (Krenkel and Novotny, 1980; Orlob, 1983; Thomann and Mueller, 1987; Velz, 1984, and others).

Oxygen Balance Model for Tidal Rivers and Estuaries

In tidal rivers and estuaries the tides induce longitudinal mixing and dispersion of the constituents. The measure of the tidal mixing is the tidal dispersion coefficient, E , which has a unit of length²/time. Development of the dissolved oxygen balance equations for estuaries is covered in a publication by O'Connor (1960). The form of the solution is similar to the classic dissolved oxygen equation (Eqs. (12.7) and (12.8)). Neglecting the lateral inflow, the dissolved oxygen balance equation becomes

$$L = L_0 \exp\left[\frac{U}{2E}(1 + \alpha_r)\right] \quad (12.13)$$

$$\begin{aligned} D &= D_0 \exp(j_a X) + \frac{K_r L_0}{K_a - K_r} \left[\exp(j_r X) - \exp(j_a X) \right] \\ &+ \frac{S_B}{K_a H} [1 - \exp(j_a X)] \quad (12.14) \end{aligned}$$

where

$$j_a = 0.5\bar{U}(1 - \alpha_a)/E$$

$$j_r = 0.5\bar{U}(1 - \alpha_r)/E$$

$$\alpha_r = \sqrt{1 + \frac{4K_r E}{\bar{U}^2}}$$

$$\alpha_a = \sqrt{1 + \frac{4K_a E}{\bar{U}^2}}$$

For a single-point input

$$L_0 = W/(Q\alpha_r)$$

Also

W = point source total BOD load (g/day)

E = tidal dispersion coefficient (m^2 /day)

Q = freshwater input in the estuary (m^3 /day)

\bar{U} = Q/A = freshwater advance velocity averaged over the tidal cycle

A = cross-sectional area of the estuary

In the preceding equation X is measured in the direction of flow toward the sea. It should be noted that if Equation (12.10) is used for estimating the reaeration coefficient K_a , it should be based on the average tidal velocity, u_t , and not on the averaged advance velocity \bar{U} .

The longitudinal tidal dispersion coefficient may be estimated from the salinity distribution in the estuary by plotting the salinity versus distance on a semilog paper (log salinity versus arithmetic distance). The slope of the line is then \bar{U}/E . Since \bar{U} is known, then $E = \bar{U}(X_2 - X_1)/\ln(s_2/s_1)$, where 1 and 2 are arbitrarily chosen points on the plot and s and X are respective corresponding salinities and distances (see Thomann and Mueller, 1987 and Krenkel and Novotny, 1980).

Effect of Nutrients on Dissolved Oxygen Balance of Streams and Estuaries

The nutrients (nitrogen and phosphorus) increase the productivity of surface-water bodies, including that of streams and estuaries. The productivity impacts the dissolved oxygen balance of streams by (a) production of the photosynthetic oxygen, (oxygen demand by respiration of algal and macrophyte biomass, and (b) oxygen demand by decomposing dead algal biomass (exhibited most commonly as SOD since the dead biomass

may settle into the bottom sediment layer). In addition, the unoxidized organic and ammoniacal nitrogen present in the water column impose the dissolved oxygen demand by nitrification.

Nitrification in the water column. As pointed out previously in this chapter, the process of nitrification in the bottom sediment may affect the sediment oxygen demand. The same process that is oxidation of ammonia to nitrite and nitrate by *Nitrosomonas* and *Nitrobacter*, may also be of significance to the oxygen balance of streams. It was also pointed out that since the nitrifying organisms associate themselves with solids and solid surface, their concentrations in free-flowing water may not be significant. Tuffey, Hunter, and Matulevich (1974) surveyed the available literature and arrived at the conclusion that a significant impact of nitrification on the oxygen balance of the streams can be only found for shallow streams with a solid bottom (rocky, boulders) where attached growths are dominant and for estuaries where sufficient detention (more than five to seven days) enables the development of sufficient densities of nitrifying organisms. Requiring some treatment plants to nitrify in the treatment process in order to reduce the ammonia toxicity of their effluents may result in enough nitrifying organisms being introduced with the effluent into the receiving water body to stimulate nitrification.

The occurrence of nitrification in a stream section is characterized by a simultaneous disappearance of ammonia and an increase of nitrate concentration. Ammonia decrease alone may not be a sufficient indication, since ammonia may be adsorbed by clay and organic particulates and settle into sediments. In addition, both ammonia and nitrates are nutrients to photosynthetic organisms (see the next section).

The nitrification process in free-flowing water and its impact on the dissolved oxygen regime is described in great detail in Krenkel and Novotny (1980). Mathematically, the nitrogenous oxygen demand (NOD) is treated in the same way as BOD. In a laboratory BOD bottle or stream, the dissolved oxygen balance model NOD equals approximately $4.33 \times \text{TKN}$ (total Kjeldahl nitrogen) = organic nitrogen + ammonia. The oxygen deficit increase due to nitrification is then

$$\Delta D (\text{nitrification}) = \frac{K_N L_{0N}}{K_a - K_N} [\exp(-K_N X/u) - \exp(-K_a X/u)] \quad (12.15)$$

where

L_{0N} = initial nitrogenous oxygen demand (NOD = $4.33 \times \text{TKN}$)

K_N = first-order nitrification rate, day^{-1}

Consequently, the NOD reduction in the stream is

$$L_N = L_{0N} \exp(-K_N X/u)$$

For estuaries and tidal streams the term $-K_N X/u$ would be replaced by $j_N X$, where

$$j_N = 0.5\bar{U}(1 - \alpha_N)/E$$

and

$$\alpha_N = \sqrt{1 + \frac{4K_N E}{\bar{U}^2}}$$

The magnitude of the nitrification reaction rate coefficient has been reported to range from 0.1 to 15.8 day⁻¹ (Ruane and Krenkel, 1975). Several authors have proposed a zero-order model as better describing the nitrification process when TKN concentrations are higher. Both models (first-order for lower TKN concentrations and zero-order for higher nitrogen concentration) may be unified by the Michaelis–Menten equation describing substrate removal by microorganisms. Temperature affects nitrification differently than other processes. The thermal factor for nitrification is around $\theta = 1.1$.

Example 12.3: Effect of Nitrification

The oxygen balance of the stream described in Example 12.2 is also affected by nitrification of 0.6 mg of TKN present in the stream. The nitrification rate was estimated as $K_N = 0.5 \text{ day}^{-1}$. Estimate the impact on the dissolved oxygen at the end of the 20-km section of the river.

Solution Recall that the average stream velocity in the section was $u = 25.92 \text{ km/day}$ and the reaeration coefficient at 25°C was $K_a = 0.46 \text{ day}^{-1}$. Calculate the initial NOD

$$L_{0N} = 4.33 \times \text{TKN} = 4.33 \times 2.6 \text{ mg/l}$$

Adjust K_N

$$K_{N25^\circ\text{C}} = 0.5 \times 1.1^{(25-20)} = 0.8 \text{ day}^{-1}$$

Then by introducing the remaining parameters in Equation (12.14) the increase of the dissolved oxygen deficit, D_N , due to nitrification becomes

$$\begin{aligned}\Delta D_N &= \frac{0.8 \times 2.6}{0.46 - 0.8} [\exp(-0.8 \times 20/25.92) - \exp(-0.46 \times 20/25.92)] \\ &= 0.99 \text{ mg/l}\end{aligned}$$

The total dissolved oxygen deficit at the end of the reach will then become

$$D = D_c + \Delta D = 6.2 + 0.99 = 7.19 \text{ mg/l}$$

and the dissolved oxygen concentration at the end of the reach is

$$c = C_s - D = 8.4 - 7.19 = 1.21 \text{ mg/l}$$

Productivity. Odum (1956) has shown that nutrients affect the productivity of streams and have an impact on the dissolved oxygen balance. Odum pointed out that the ratio of the photosynthetic organic matter production and the respiration (decay) in relation to the concentration of the organic matter itself is the key in determining whether there is a net increase in the organic matter or a net decrease. The fundamental relation of the organic matter balance was postulated by Odum (1956) as

$$I + P = E + R \quad (12.16)$$

where

I = import rate of organics from allochthonous sources and/or from upstream

P = production rate of organic matter

R = decomposition and/or respiration

E = export of organic matter downstream

Decomposition is a function of the organic matter concentration, while production depends on nutrient level and light, provided that the source of inorganic carbon (alkalinity) is sufficient and not limiting.

It was pointed out previously that based on the Stumm and Morgan relationship, 1 atom of phosphorus or 16 atoms of nitrogen will generate 154 molecules of oxygen during photosynthesis, and the same amount of oxygen will be used during decomposition in the form of BOD.

Originally, the trophic status concept was applied only to lakes; today all bodies of water can be evaluated according to their trophic status. Žáková (1989) classified stream trophic status according to the algal biomass concentration, as shown below:

<i>Trophic Degree</i>	<i>Algal Biomass (mg/l)</i>
ultraoligotrophic	<5
oligotrophic	5–50
oligomezotrophic	50–100
mezotrophic	100–200
mezoeutrophic	200–350
eutrophic	350–500
polytrophic	500–1000
hypertrophic	>1000

The trophic status terminology for lakes is explained in the next section. For streams the term *oligotrophic* would imply a stream with a low concentration of nutrients (*oligo* means *few*), while *eutrophic* would mean nutrient-enriched streams. *Ultraoligotrophic* would typically mean a stream with little biological activity, such as a stream draining a glacier or a stream affected by toxics or acid rainfall or snowmelt (an unhealthy situation).

Using the previous concepts of stream enrichment by productivity stimulated by nutrient inputs, Novotny and Bendoricchio (1989) have described the linkage of nutrients and primary productivity to organic carbon production, its deposition, and subsequent impact on oxygen resources as follows:

- Oxygen produced by photosynthesis above saturation in streams is released to the atmosphere and is lost from the nutrient-oxygen cycle.
- In flowing waters, photosynthesis and respiration do not occur at the same time and in the same place. Photosynthesis takes place on sunny days in shallow reaches and euphotic zones, while respiration takes place when light is limited (cloudy days and night hours) and in profundal zones. The fact that lack of light commonly accompanies storm runoff events may explain some of the dissolved oxygen anomalies mentioned in papers by Field and Turkeltaub (1981), and Keefer, Simons, and McQuivey (1979).

In streams where shallow productive reaches are followed by deeper sections, oxygen deficiency becomes the most troublesome (Novotny and Zheng, 1988; Novotny and Bendoricchio, 1989). Such situations include rivers entering reservoirs or bays, and estuaries or larger slower moving rivers. The algal biomass produced in shallow well-lighted reaches in which most oxygen above saturation was lost, may enter river sections that are deeper and in which only the upper layer is productive and

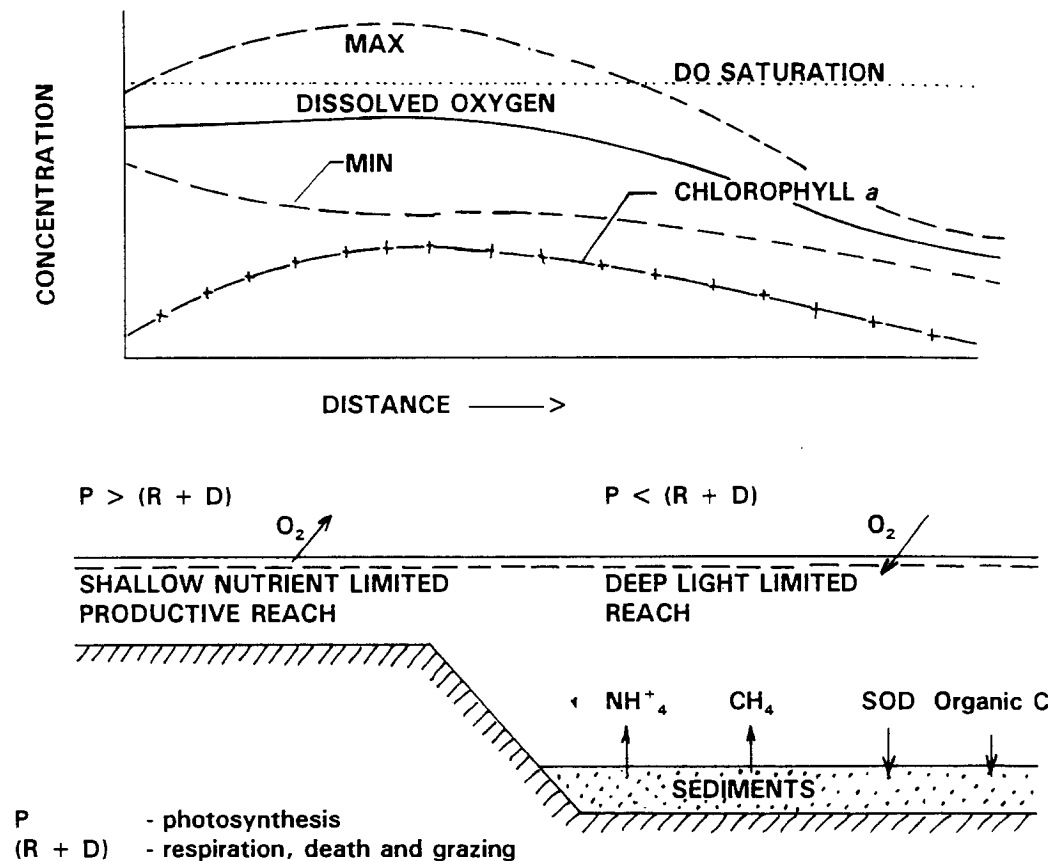


FIGURE 12.10. Concept of the impact of primary productivity on stream dissolved oxygen.

capable of producing photosynthetic oxygen while the algae in the deeper layer are respiring. Figure 12.10 illustrates this concept.

Novotny and Bendoricchio (1989) concluded that:

- Primary productivity resulting from nutrient inputs during medium and small flows represent the link between nonpoint pollution inputs and water quality deterioration. Large masses of organic matter can be produced in urban and suburban nutrient-enriched streams, which may subsequently impose dissolved oxygen demand on the receiving aquatic systems. This secondary oxygen demand caused by nutrients may be higher than the primary oxygen demand from discharged BOD in storm runoff and combined sewer overflows.
- The primary productivity process in streams and to a lesser degree in stagnant bodies of water is generally oxygen deficient because a significant portion of the oxygen produced above saturation levels may escape from the system. In most adverse cases, very little of the

photosynthetically produced oxygen may be included in the dissolved oxygen balance of the body of water.

- Control of nutrient inputs from erosion during high-flow runoff events generated by large but hydrologically rare storms, may not be effective since primary productivity in streams is most active during low flows and algae and macrophytes may actually be flushed out during the high flows. Also sediment inputs and the resulting turbidity cut down on light penetration and inactivates part of the phosphorus and ammoniacal nitrogen.

NUTRIENT PROBLEM OF LAKES: EUTROPHICATION

The problem of nutrient inputs is especially important for lakes, impoundments, and shallow estuaries (e.g., Chesapeake Bay). In these bodies of water the classic dissolved oxygen balance model may not work in evaluating the waste-assimilative capacity of these bodies of water, and the production of organic matter (primary productivity) by phytoplankton and large plants (macrophytes) may greatly exceed the allochthonous point and nonpoint BOD and NOD contributions. Oxygen levels are also affected by photosynthesis and respiration, and BOD concentrations are

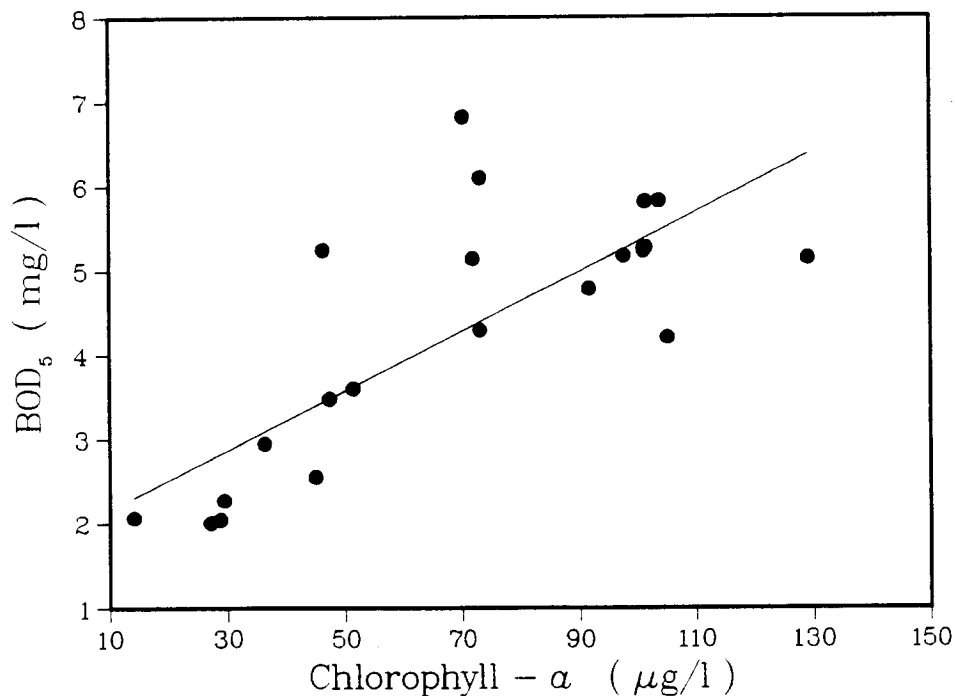


FIGURE 12.11. Relation of chlorophyll-*a* to BOD in the Milwaukee River. (Data from the Metropolitan Milwaukee Sewerage District.)

affected by the density of planktonic organisms and their residues (Fig. 12.11).

The production of organic matter in lakes, shallow estuaries, and reservoirs depends on the trophic status of the impoundment. Such bodies of water have experienced accelerated eutrophication in the last few decades, caused by among other factors, increased use of agricultural and lawn "care" chemical fertilizers. Streams and lakes that 50 years ago were reasonably clean are today infested by algae to a point that their color is rich green with little light penetration, instead of transparent water. This situation is especially troublesome for Eastern and Central Europe. An example of the Lake Balaton situation in Hungary was featured in Chapter 1, and Table 12.3 documented extremely high levels of nitrates in the rivers of the Czech Republic.

In this context the term *eutrophication* represents a process by which a body of water progresses from its origin (for example, a glacial lake) to its extinction, which is dry land. Eutrophication is not synonymous with pollution; however, pollution can accelerate the rate of eutrophication. It is a dynamic morphological process that has variable rates, which differ from year to year, season to season, and even hour to hour. It progresses in several stages. The net increase of organic matter (net productivity) that drives eutrophication takes place in a surface-water body in which organic matter production (primary productivity) nourished by nutrients and minerals exceeds its loss by respiration, decay, grazing by higher organisms, and outflow. The rate of eutrophication (productivity) depends on a number of factors, many of them uncontrollable. In today's context "Eutrophication refers to the natural and artificial addition of nutrients to bodies of water and to the effect of these added nutrients on water quality" (Rohlich, 1969). The stage of eutrophication is determined by the trophic state of the body of water. The trophic status may be related to the algal biomass or chlorophyll-*a* concentrations and other parameters, as shown in Table 12.8.

TABLE 12.8 Trophic Status of Lakes

Water Quality	Oligotrophic	Mezotrophic	Eutrophic	Source
Total P ($\mu\text{g/l}$)	<10	10–20	>20	U.S. EPA (1974)
Chlorophyll- <i>a</i> ($\mu\text{g/l}$)	<4	4–10	>10	U.S. EPA (1974)
Secchi disc depth (m)	>4	2–4	<2	U.S. EPA (1974)
Hypolimnetic oxygen (percent of saturation)	>80	10–80	<10	U.S. EPA (1974)
Phytoplankton production (g of organic C/m ² -day)	7–25	75–250	350–700	Mason (1991)

The term *oligotrophic* is used to describe a clean body of water, typically the youngest stage of lake formation. This body of water has a very low mineral content, so productivity and the algal content are low, and the water is highly transparent. Ultraoligotrophic bodies of water commonly refer to bodies of water whose productivity was eliminated, for example, by a severe drop of pH due to acid rainfall or by toxicity. Unlike the oligotrophic bodies of water whose biota may be healthy although few, ultraoligotrophic bodies of water may be unhealthy.

As the nutrient and mineral content of water is increased by runoff and/or by wastewater disposal, the photosynthetic (autotrophic) organisms (the producers) increase in number. As the productivity increases the organic matter content of the body of water becomes *mezotrophic*, and when the nutrient and organic contents are high the body of water becomes *eutrophic*. The *polytrophic* and *hypertrophic* (*hypereutrophic*) classification may be again added to describe bodies of water that are severely impacted by nutrient inputs, mostly from the excessive use of fertilizing chemicals, which in many parts of the world have caused more than an order of magnitude increases of nutrient levels of receiving bodies of water and very severe algal infestations. Chlorophyll-*a* concentrations of hypertrophic lakes may exceed 200 µg/l. For lakes and reservoirs, the final stages of their existence are silted pond, swamp, marsh, or other types of wetland. Figure 12.12 shows symptoms of eutrophication, while Figure 12.13 clearly demonstrates aesthetic impairment of water quality caused by the excessive growth of phytoplankton and aquatic weeds in a hypertrophic reservoir.

The photosynthetic process of algae and macrophytes in eutrophic water bodies can be recognized by cyclic fluctuations of the dissolved oxygen concentrations in the euphotic zone (a zone where light can penetrate). Oxygen is produced during the day by photosynthesis and consumed during the night and cloudy daytime hours by endogenous respiration. On bright sunny days during the production season, this often results in supersaturation of the euphotic zone with oxygen during afternoon hours and a significant drop of dissolved oxygen concentrations during the late night and early morning hours. Similar dissolved oxygen fluctuations are also common in eutrophic streams.

The related physical and chemical changes caused by advanced eutrophication (pH variations, oxygen fluctuations or lack of oxygen in the lower hypolimnetic layer of stratified bodies of water, waste excretions by some algae, etc.) may interfere with recreational and aesthetic uses of the body of water and may also cause a shift in fish and shellfish populations from better quality fish to rough fish. In addition, possible taste and odor problems caused by algae can make water less suitable for potable use

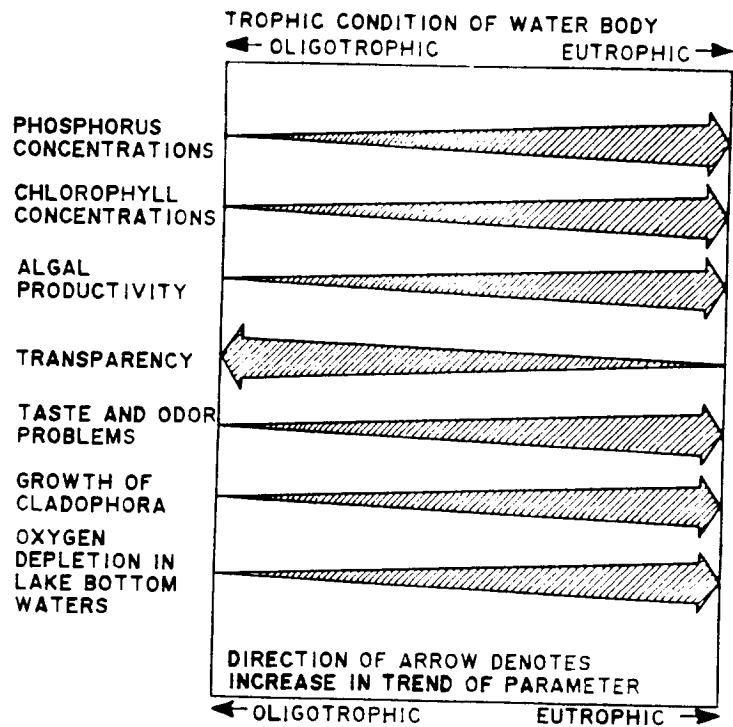


FIGURE 12.12. Symptoms of eutrophication.



FIGURE 12.13. Impact of eutrophication on a shallow reservoir. (Lake Sinissippi in Wisconsin in the 1970s; in the late 1980s the water quality of the lake was restored by the Wisconsin Department of Natural Resources.)

and contact recreation. When the concentration of phytoplankton algae exceeds a certain threshold nuisance value during the late summer period, the situation is termed *algal bloom*.

Hutchinson (1969) pointed out that the terms eutrophic, oligotrophic, mezotrophic, etc., should not only be used to describe water quality but should also be related to the drainage area and sediment and nutrient transport. Also Žáková (1989) pointed out that the trophic status of a lake or a reservoir is related to the trophic status in the tributaries; however, this relation is also affected by other factors, such as depth of the reservoir. In this context the term *available nutrients* should be used to describe eutrophication potential.

The necessity of including tributaries (watershed) in eutrophication systems caused the recategorization of lakes and reservoirs (and to a lesser degree large estuaries) into *autotrophic*, that is bodies of water that receive a major portion of the nutrients from internal sources (sediment, atmosphere), and *allotrophic*, which receive a major portion of the nutrients from external sources. *Allochthonous* nutrients are those originating from the watershed, that is, from point and nonpoint sources, while *autochthonous* nutrient sources include nutrients stored in the lake water and sediments.

Trophic indices and trophic status. Determination of the trophic status of lakes and other bodies of water based on limnological observations, taxonomy or the distribution of organisms and their productivity, and chemical water quality, is relatively difficult and requires experience and often subjective judgment. Table 12.8, relating trophic status to the concentration of organic (photosynthetic) matter, is only one of the possible judgmental classifications. A lack of a precise definition of "trophic status" makes it difficult to develop an accurate engineering tool that would enable estimation of the stage of the eutrophication process of a given body of water. However, modeling of the eutrophication process advanced in the 1980s to the point where a reasonably reliable judgment could be made on the relation of nutrients to productivity and, hence, the trophic status of a water body.

Some methods of estimating the trophic status based on few indicators have evolved and have been published and/or used for classifying the lakes. Many of the techniques were relative systems in which lakes were classified and ranked only in respect to each other or to an average water quality of a given group of lakes and not to some objective independent scale. These systems often give different weights to various parameters similar to those shown on Figure 12.12, and some of these are then included trophic status indices. The most frequently used ones are dissolved oxygen (if the water body is stratified, then dissolved oxygen in the

lower hypolimnetic layer is considered), total phosphorus, transparency by Secchi disc (the depth at which a white disc can no longer be seen by the observer above the water surface), inorganic nitrogen, and chlorophyll-*a* concentration.

Trophic Index by Carlson (1977). This index was developed for lakes that are phosphorus limited. Carlson based his index on the fact that there are intercorrelations between the transparency expressed by the Secchi disc depth, algal concentrations expressed by chlorophyll-*a*, and vernal (spring) or average annual phosphorus concentrations. The trophic status index (TSI) was then defined as

$$\text{TSI}(\text{SD}) = 10 \left(6 - \frac{\ln \text{SD}}{\ln 2} \right) \quad (12.17)$$

where SD = Secchi disc depth in meters. From the correlations between the chlorophyll-*a* concentrations, total phosphorus, and Secchi disc depth, the other two expressions for the TSI became

$$\text{TSI}(\text{Chl}) = 10 \left(6 - \frac{2.04 - 0.68 \ln \text{Chl}}{\ln 2} \right) \quad (12.18)$$

and

$$\text{TSI}(\text{TP}) = 10 \left(6 - \frac{48/\text{TP}}{\ln 2} \right) \quad (12.19)$$

where

Chl = concentration of chlorophyll-*a* ($\mu\text{g/l}$)

TP = concentration of total phosphorus ($\mu\text{g/l}$)

The method offers three indices instead of one single value. The best indicator of the trophic status varies from lake to lake and from season to season. Secchi disc values may be erroneous in lakes where turbidity is caused by factors other than algae. The Carlson index works well for north temperate lakes, but performs poorly in lakes with excessive weed problems (North American Lake Management Society, 1990).

Based on the observations of several northern lakes, most of the oligotrophic lakes had TSIs below 40, mezotrophic lakes had TSIs between 35 and 45, while most eutrophic lakes had TSIs greater than 45. Hypertrophic lakes, on the other hand, can have TSI values above 60 (Sloey and Spangler, 1978; Krenkel and Novotny, 1980).

Trophic status derived from nutrient loading of lakes. During investigations of eutrophication problems of lakes in Madison, Wisconsin, Sawyer (1947) noted that algal blooms occurred when concentrations of inorganic nitrogen (NH_4^+ , N_2^- , NO_2^-) and inorganic phosphorus exceeded respective values of 0.3-mg N/liter and 0.001-mg P/liter. It should be noted that due to the uptake of nutrients by algae, very low concentrations would be measured during the productive summer period; therefore, the critical nutrient concentrations should be evaluated during the spring (vernal) season while the spring overturn is in progress. Similarly, most of the lakes were eutrophic if the phosphorus concentration was above 20 $\mu\text{g/l}$.

Vollenweider (1975, 1976) then developed a simple input-output model phosphorus model for a completely mixed lake as follows (Thomann and Mueller, 1987):

$$V \frac{dp}{dt} = W - v_s A_s p - Qp \quad (12.20)$$

where

V = volume of the lake (l^3)

p = phosphorus (total) in the lake (M/l^3), for example, $\mu\text{g/l}$

Q = outflow (l^3/time)

A_s = lake surface area (l^2)

W = allochthonous source of phosphorus (M/time), for example, g/sec

v_s = settling rate of phosphorus, l/time

At a steady-state $dp/dt = 0$, and dividing both sides by the surface area, Equation (12.19) becomes

$$p = \frac{W'}{q_s + v_s} \quad (12.21)$$

where

q_s = hydraulic overflow rate $Q/A_s = H\rho$

ρ = $Q/V = 1/\tau_w$ = flushing rate

τ_w = detention time in the lake

The problem with using Equation (12.21) is lack of adequate understanding on the settling (removal) rates of phosphorus. Vollenweider (1976) substituted the following approximation

$$v_s = H\sqrt{\rho}$$

which led to

$$P = \frac{W'}{H\rho(1 + \sqrt{1/\rho})} \quad (12.22)$$

The relationship of annual loading of phosphorus per unit lake area and the hydraulic parameters for respective phosphorus concentrations of $10 \mu\text{g/l}$ and $20 \mu\text{g/l}$ by Vollenweider is plotted on Figure 12.14. The lake depth, H , is expressed in meters, and the flushing rate $\rho = Q/V$ in years^{-1} .

Assuming that phosphorus is the limiting nutrient, Lee, Rust, and Jones (1978) have expanded the Vollenweider relationship to include chlorophyll-*a*, Secchi disc depth, and hypolimnetic oxygen depletion, thus closing the circle between the nutrient loading, water quality, and trophic status. Figures 12.15 through 12.17 show the results of Lee, Rust, and Jones. On the abscissa, $L(P) = W'$ = annual surface phosphorus loading in $\text{mg/P/m}^2\text{-year}$, hydraulic overflow loading, q_s , is in meters/year, mean depth of the lake, H , is in meters, and the hydraulic residence time, τ_w = water body volume/annual flow, is in years.

Example 12.4: Lake Eutrophication

The annual loadings for a lake with 28 km^2 surface area and an average depth of 13 meters are given in Table 12.9. Estimate the trophic level of the lake using Vollenweider and Lee et al. concepts.

Solution Annual flushing rate

$$\rho = \frac{\text{Annual flow}}{\text{Lake volume}} = \frac{78,000,000 (\text{m}^3/\text{yr})}{28 (\text{km}^2) \times 13 (\text{m}) \times 10^6 (\text{m}^2/\text{km}^2)} = 0.21 \text{ yr}^{-1}$$

Annual phosphorus loading

$$L_p = \frac{23,300 (\text{kg/yr}) \times 10^3 (\text{g/kg})}{28 (\text{km}^2) \times 10^6 (\text{m}^2/\text{km}^2)} = 0.83 \text{ g/m}^2\text{-yr}$$

Annual nitrogen loading

$$L_N = \frac{236,000 (\text{kg/yr}) \times 10^3 (\text{g/kg})}{28 (\text{km}^2) \times 10^6 (\text{m}^2/\text{km}^2)} = 8.43 \text{ g/m}^2\text{-yr}$$

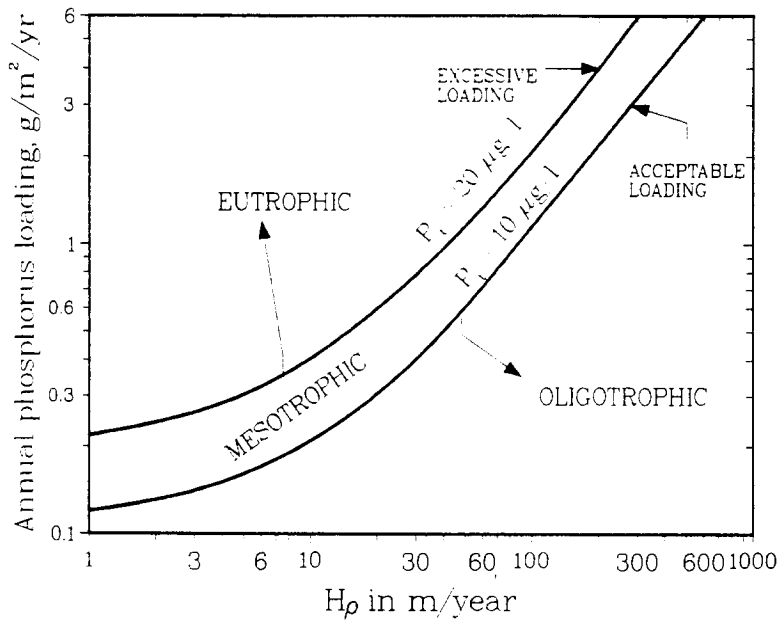


FIGURE 12.14. Relationship between nutrient loading and lake trophic conditions. (Adapted from Vollenweider [1975]. Courtesy *Schweizerische Zeitschrift für Hydrologie*.)

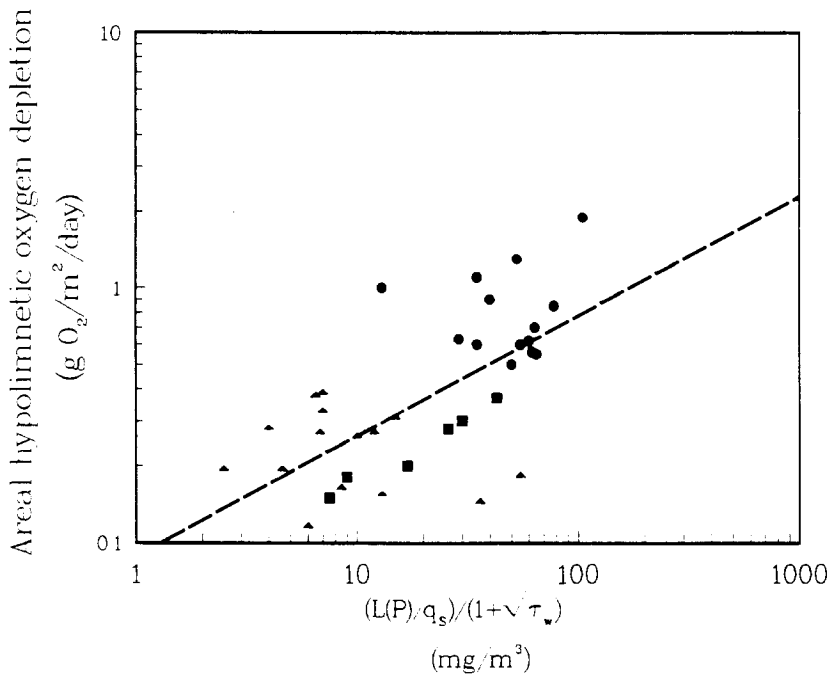


FIGURE 12.15. Phosphorus loading characteristics and hypolimnetic oxygen depletion in lakes. (Redrawn from Lee, Rust, and Jones [1978] with permission. Copyright 1978 by the American Chemical Society.)

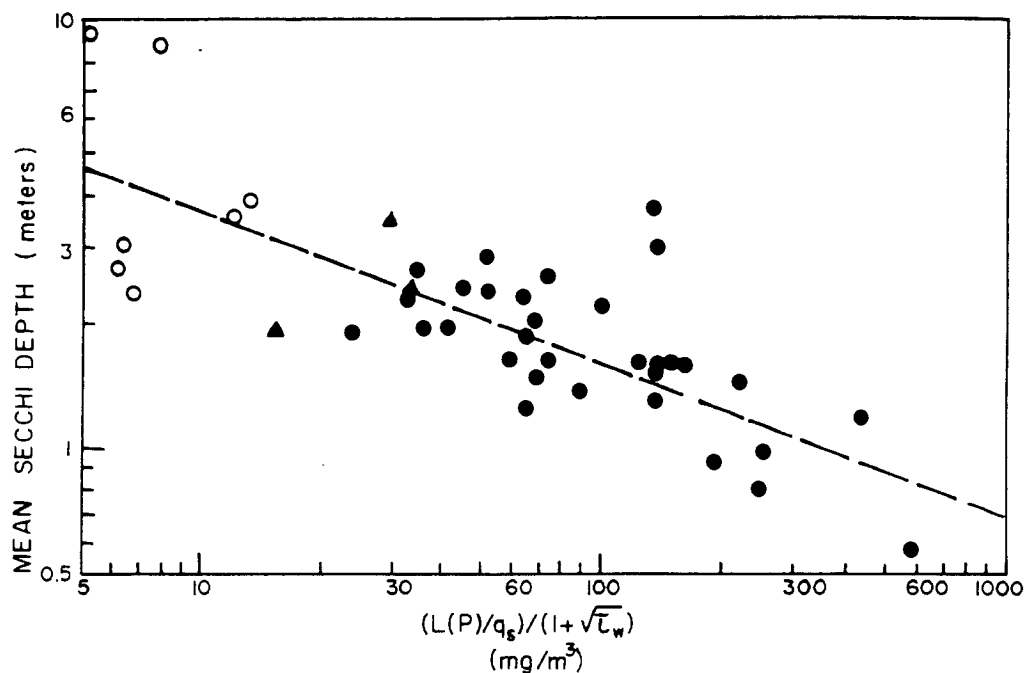


FIGURE 12.16. Phosphorus loading characteristics and mean Secchi disc depths for lakes. (Redrawn from Lee, Rust, and Jones [1978] with permission. Copyright 1978 by the American Chemical Society.)

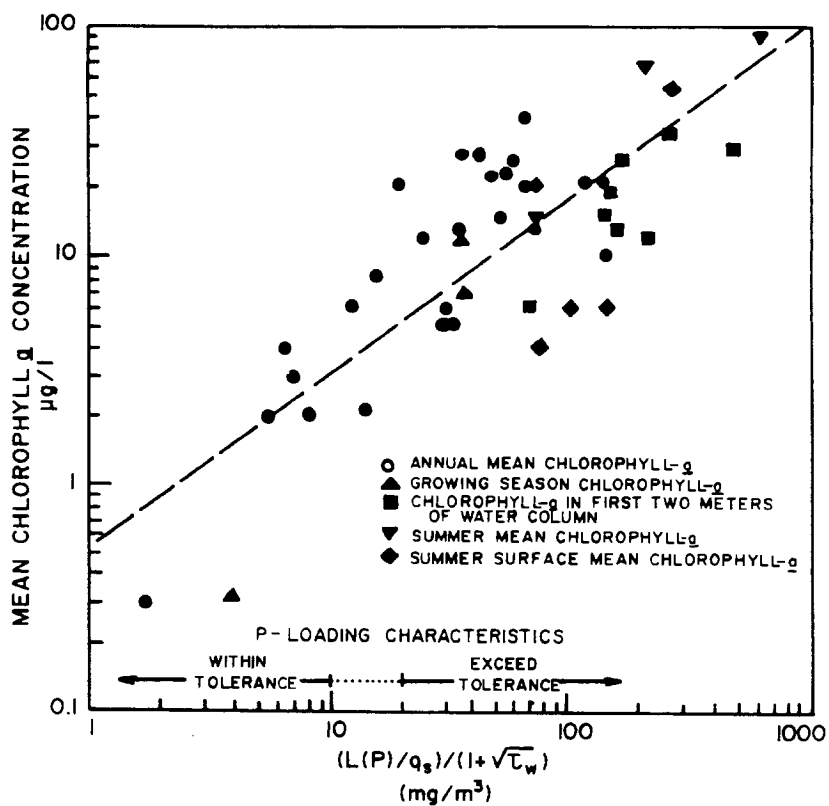


FIGURE 12.17. Phosphorus loading characteristics and mean chlorophyll-a relationship in lakes. (Redrawn from Lee, Rust, and Jones [1978] with permission. Copyright 1978 by the American Chemical Society.)

Even though the nitrogen-to-phosphorus load ratio is about 10:1 and nitrogen may be limiting, controlling phosphorus is the only practical solution, considering that more than 50% of nitrogen originates from nitrogen in ground water (suspected sources include septic tank discharges and overuse of fertilizers in agriculture).

Vollenweider concept From Figure 12.14 (Vollenweider's chart) for $H\rho = 13 \times 0.21 = 2.73$ m/yr the acceptable loading (to maintain oligotrophic conditions) is about 0.12 g P/m²-yr and the excessive loading (eutrophic conditions) is about 0.23 g P/m²-yr. Since these phosphorus loadings are exceeded, the lake is considered to be eutrophic.

Lee et al. concept Use, for example, Figure 12.16 (Secchi disc) and apply the following values.

$$q_s = H\rho = 2.73 \text{ m/yr}$$

$$\tau_w = 1/\rho = 1/0.21 = 4.78 \text{ yr}$$

$$(L_p/q_s)/(1 + \sqrt{\tau_w}) = \frac{0.83 \text{ (g/m}^2\text{-yr)} \times 10^3 \text{ (mg/g)}}{2.73 \text{ (m/yr)}} / (1 + \sqrt{4.78}) = 95.4$$

The expected Secchi disc depth would be around 2 meters. Then from Equation (12.17)

$$\text{TSI(SD)} = 10 \left(6 - \frac{\ln 2}{\ln 2} \right) = 50$$

indicating again borderline eutrophic conditions.

TABLE 12.9 Annual Loadings

Source	N (kg/yr)	P (kg/yr)	Flow (mil m ³ /yr)
Urban sewage	23,000	8,000	1.0
Urban runoff	15,000	4,000	7.0
Rural runoff	26,000	10,000	10.0
Precipitation	47,000	1,000	21.0
Ground water	125,000	300	47.0
Evaporation	—	—	-8.0
Total	236,000	23,300	78.0

Example 12.5: Maximum Nutrient Loadings to Maintain Dissolved Oxygen Standard

A lake is receiving nutrient inputs from runoff and treated municipal wastewater. Estimate the maximum loading of phosphorus to the lake that would keep the lake below eutrophication status and maintain summer dissolved oxygen concentrations in the lower hypolimnion layer above 5 mg/l.

Lake characterization

$$\text{Surface area} = 10 \text{ km}^2$$

$$\text{Average depth} = 8 \text{ m}$$

$$\text{Annual inflow including precipitation} = 30 \times 10^6 \text{ m}^3/\text{yr}$$

Annual flushing rate

$$\rho = \frac{\text{Annual flow}}{\text{Lake volume}} = \frac{30,000,000 \text{ (m}^3\text{/year)}}{10 \text{ (km}^2\text{)} \times 8 \text{ (m)} \times 10^6 \text{ (m}^2\text{/km}^2\text{)}} = 0.375 \text{ year}^{-1}$$

From Figure 12.14 (Vollenweider chart) the admissible and dangerous loadings of phosphorus for $H\rho = 8 \times 0.375 = 3.00$ are

$$\text{Dangerous P loading} = 0.28 \text{ g}/(\text{m}^2\text{-year)}$$

$$\text{Acceptable P loading} = 0.13 \text{ g}/(\text{m}^2\text{-year)}$$

If 0.20 g/(m²-year) is selected, the loading will be

$$\begin{aligned} \text{max P load} &= 0.2 \text{ [g}/(\text{m}^2\text{-year)}] \times 10 \text{ (km}^2\text{)} \times 10^6 \text{ (m}^2\text{/km}^2\text{)} \times 0.001 \text{ (kg/g)} \\ &= 2000 \text{ kg/year} = 5.48 \text{ kg of P/day} \end{aligned}$$

Solution To estimate the maximal loading to maintain the dissolved oxygen standard using the Lee et al. concept, several assumptions must be made:

1. Oxygen is used in the hypolimnion (approximately 5 meters depth from the bottom, $H_b = 5$ meters) and there is no oxygen interchange

between the upper (epilimnion) and lower (hypolimnion) layers. This is a reasonable estimate since the transfer across the epilimnion–hypolimnion boundary (so-called thermocline) is greatly reduced and is possible only by molecular diffusion (see Krenkel and Novotny (1980) for a discussion).

2. The average temperature of the hypolimnion is 15°C and the initial oxygen concentration at the beginning of the summer productive (growing) season is near saturation. The oxygen saturation, c_s , at 15°C is 10.2 mg/l.

Then the allowable drop of the dissolved oxygen during the growing season between the spring and summer overturn (assume 4-month growing period) is

$$\Delta C = c_s - \text{DO standard} = 10.2 - 5.0 = 5.2 \text{ mg/l [= g/m}^3\text{]}$$

The allowable oxygen depletion rate (4-month period) is

$$\frac{\Delta C}{\Delta t} H_b = \frac{5.2 \text{ (g/m}^3\text{)}}{122 \text{ days}} \times 5 = 0.21 \text{ g/(m}^2\text{-day)}$$

From the Lee et al. chart (Fig. 12.15) the loading parameter for the maximal rate of reduction of the hypolimnetic oxygen of 0.21 g/(m²-day) is

$$[L_p/q_s]/(1 + \sqrt{\tau_w}) = 7 \text{ mg/m}^3$$

Herein

$$q_s = H\rho = 3.00 \quad \text{and} \quad \tau_w = 1/\rho = 1/0.375 = 2.67 \text{ years}$$

Then

$$L_p = 7q_s(1 + \sqrt{\tau_w}) = 7 \times 3.00 \times (1 + \sqrt{2.67}) = 55 \text{ mg/m}^2\text{-yr}$$

The corresponding P loading is then

$$\begin{aligned} \text{max P load} &= \frac{55 \text{ [mg/(m}^2\text{-yr)]} \times 10^{-6} \text{ [kg/mg]} \times 10 \text{ [km}^2\text{]} \times 10^6 \text{ [m}^2\text{/km}^2\text{]}}{365 \text{ [days/year]}} \\ &= 1.51 \text{ kg of P/day} \end{aligned}$$

MODELING FATE OF CONVENTIONAL POLLUTANTS AND NUTRIENTS

One has to realize that water quality management and pollution abatement are typically designed for hydrologically rare events. Fifty or so years ago when point source abatement was the only objective and biodegradable pollutant discharges the only type of pollution considered, steady-state simple models of the dissolved oxygen balance were the only type of models used by planners and pollution abatement authorities. Including eutrophication and toxic components into the water quality picture and also considering diffuse sources in addition to point source pollution in an integrated manner made the simple steady-state dissolved oxygen models obsolete. Also with the advent of fast and efficient yet small desktop or lap (personal or work station) computers, far more complex and detailed modeling is now required and has become common.

To describe mathematics and the software design of present dynamic water quality models is beyond the scope of this book. The reader is referred to publications by Thomann and Mueller (1987), O'Connor (1988), Orlob (1983), or by the senior author (Krenkel and Novotny, 1980), and to the various manuals describing the published models.

A dynamic (time variable) compartmentalized water quality model must simultaneously solve three basic mathematically described physical-chemical-biological relationships:

1. *The equation of continuity or water volume conservation*, which describes the physical law, stating that a change of water mass within a compartment must be balanced by inflow (inputs) and outflow, or

$$\frac{dV_i}{dt} = Q_{in_i} - Q_{out_i} \quad (12.23)$$

where

V_i = volume of the compartment i

Q_{in_i} = sum of flows into the compartment from adjoining compartments and from allochthonous sources (waste discharges, runoff, precipitation, etc.)

Q_{out_i} = sum of all flows leaving the compartment (to downstream compartment, water withdrawn, water lost by evaporation, etc.)

For one-dimensional rectangular channel segments, the equation continuity may be modified to

$$\frac{\partial H}{\partial t} = -\frac{1}{B} \frac{\partial Q}{\partial X} \quad (12.24)$$

where

H = depth

B = width of the channel

X = distance

The water volume (mass) continuity equation predicts water heights (depths) within the system.

2. *The equation of hydraulic motion*, which is based on Newton's second law of mechanics, stating that forces acting on the body of water are balanced by acceleration of the mass within the body. The forces include internal forces (gravity, pressure, and turbulent friction) and external forces, such as wind drag. Many variations of this equation have been published in the literature, including the St. Venant equation, characteristic equations, and kinematic wave approximation.

Typically the hydrodynamic equations of motion solve the velocity and flow distribution within the system. The kinematic wave approximation, the most common algorithms in dynamic water quality models describes the propagation of a long wave through a shallow-water system while conserving both momentum (energy) and volume (mass).

3. *The pollutant mass balance equations* for each pollutant and compartment, which can be generally written as (O'Connor, 1988)

$$V_i \frac{dc_{ij}}{dt} = J_{ij} + \Sigma R_{ij} + \Sigma T_{ij} + \Sigma S_{ij} \quad (12.25)$$

where

c_{ij} = concentration of the pollutant j in compartment i

J_{ij} = transport of the pollutant j through the compartment

R = reactions within the system (biological degradation, growth, chemical modification, etc.)

T = transfer from one phase to another (for example, volatilization, biological uptake)

S = allochthonous (boundary) sinks and sources of the substance (for example, settling, atmospheric aeration, wastewater, and nonpoint source inputs).

A dynamic model solves these equations simultaneously assuming a small time interval, Δt , and a small element with a volume ΔV and dimension ΔX . In these numerical solutions, finite-difference terms $\Delta E/\Delta t$

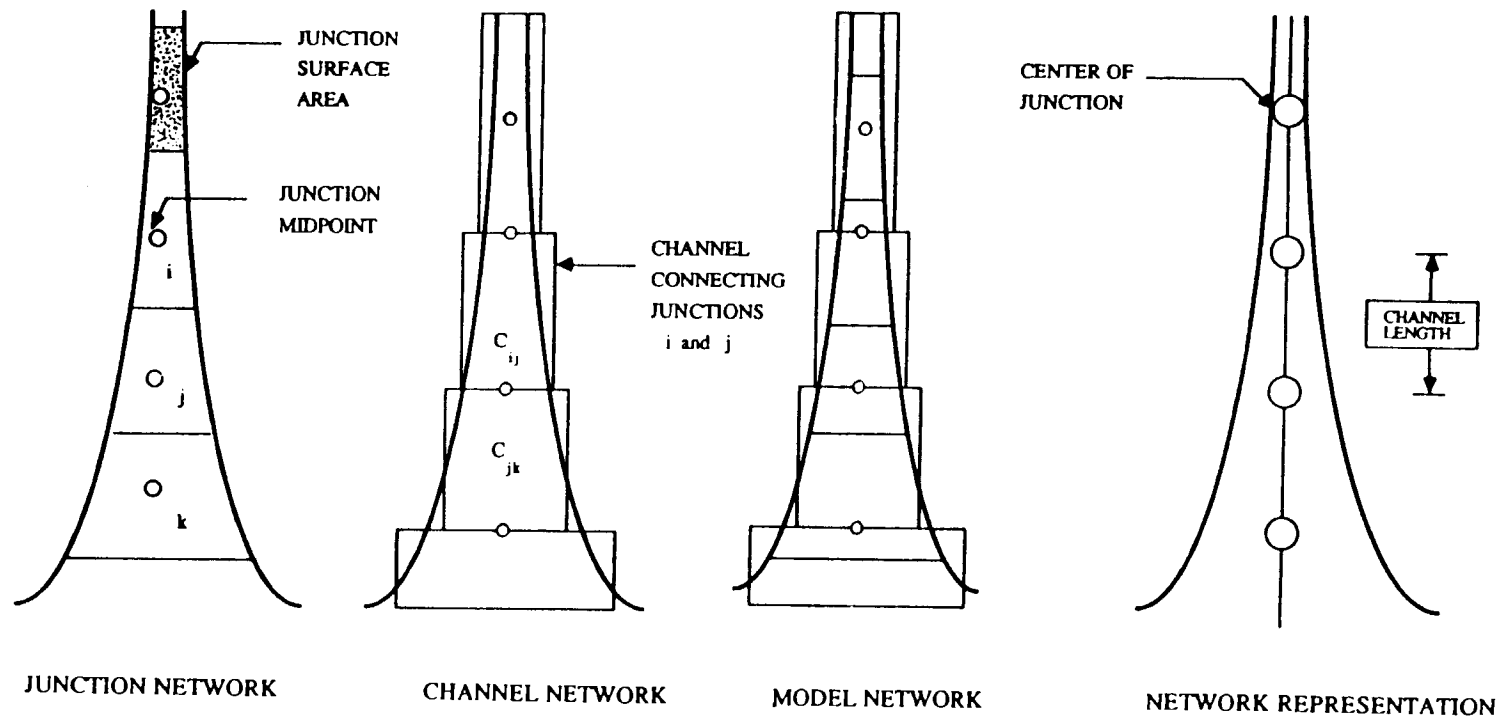


FIGURE 12.18. Link-mode representation of an aquatic system—WASP model. (After Ambrose et al., 1990.)

and $\Delta F/\Delta X$ replace equivalently the derivatives dE/dt and dF/dX , respectively. Herein, E and F are computational variables (depth, concentrations, flow, etc.).

A link-node representation has been developed for such computational schemes (Ambrose et al., 1990). In this representation (Fig. 12.18) the system is divided into a series of interconnected uniform channels (links) connected by junctions (nodes). Each junction is a volumetric unit that acts as a receptacle for the water transported through its connecting channels. In water quality simulations junctions are equivalent to segments in the water quality model, whereas channels correspond to segment interfaces. Channel flows are used to calculate mass transport between the segments in the water quality model. Junction volumes are used to calculate pollutant concentrations within each quality segment. Each segment is assumed to be completely mixed. Figure 12.19 shows that for water quality simulations, the network may be subdivided three-dimensionally, that is, laterally, vertically, as well as longitudinally and

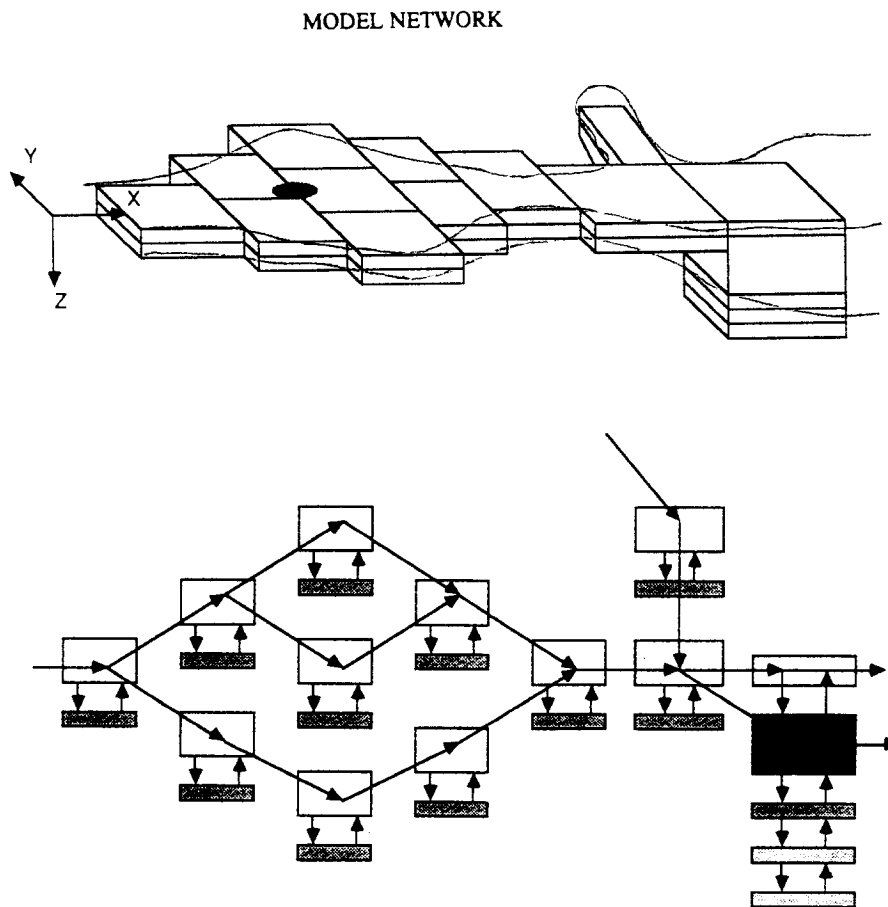


FIGURE 12.19. Layered division of the body of water system—WASP model.

benthic segments can be included along with water column segments. Segment volumes $\Delta V (= B \times H \times \Delta X)$ and simulation time Δt are directly related. As one decreases or increases, the other must do the same in order to preserve the stability of the solution.

The WASP4 Model

The preceding schematics and modeling concepts are a representation of the U.S. EPA model WASP4 (Ambrose et al., 1990). The schematics of the WASP4 model is shown on Figure 12.20. The WASP4 model consists of two separate units, the hydrodynamics submodel DYNHYD5, which simulates the hydraulics of the water body, and WASP, which simulates the fate of pollutants. The water quality model WASP then

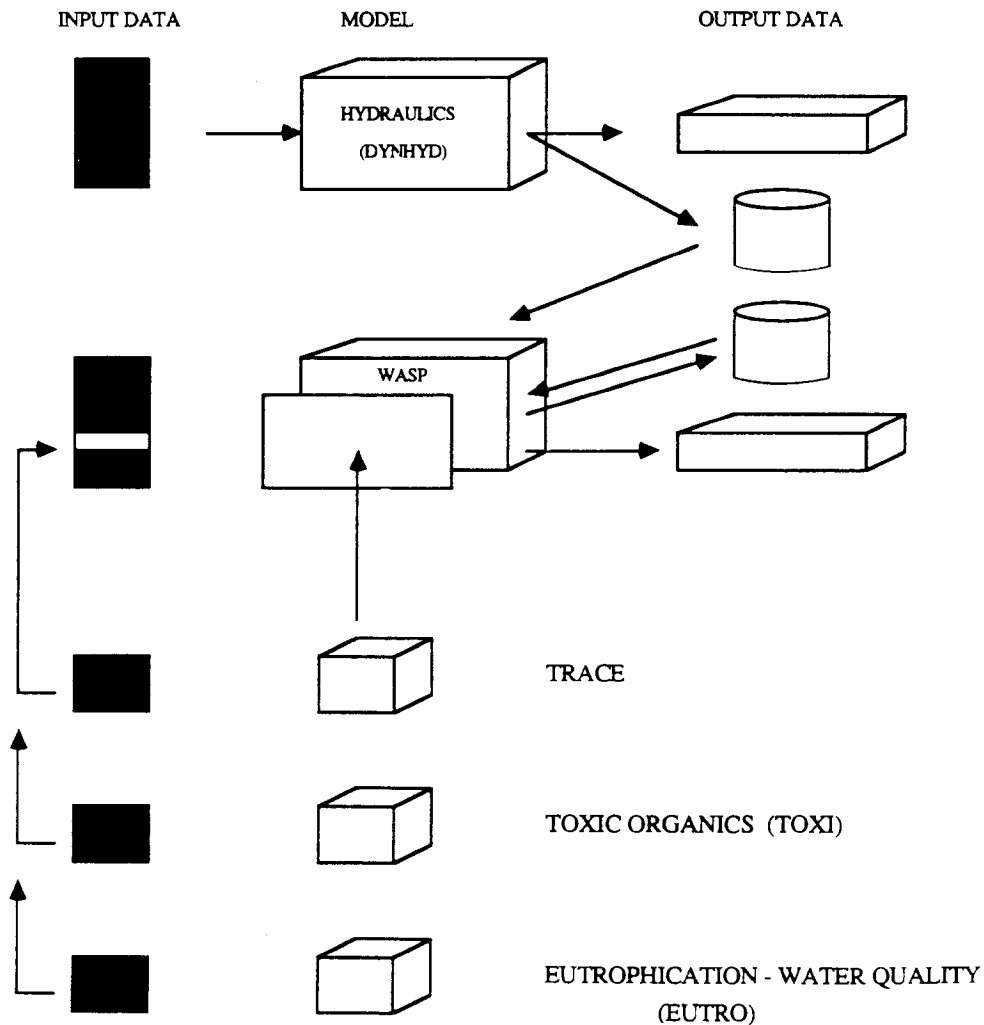


FIGURE 12.20. Block schematics of the WASP model. (From Ambrose et al., 1990.)

includes two kinetic packages: EUTRO4 simulates the dissolved oxygen, nutrients, and eutrophication, and TOXI4 simulates the fate of toxic pollutants. EUTRO4 and TOXI4 may involve both water column and sediment segment simulations. The EUTRO4 and TOXI4 submodels use the output of flows, velocities, and depths from the DYNHYD5 model. Alternatively, the hydrodynamic conditions may be specified by the user. Only concepts of the EUTRO4 submodel will be discussed in this chapter. The fate of the toxic compounds described in the TOXI4 submodel is briefly introduced in the next chapter.

Modeling eutrophication and dissolved oxygen. The separate steady-state models for dissolved oxygen, nutrients (nitrogen and phosphorus)

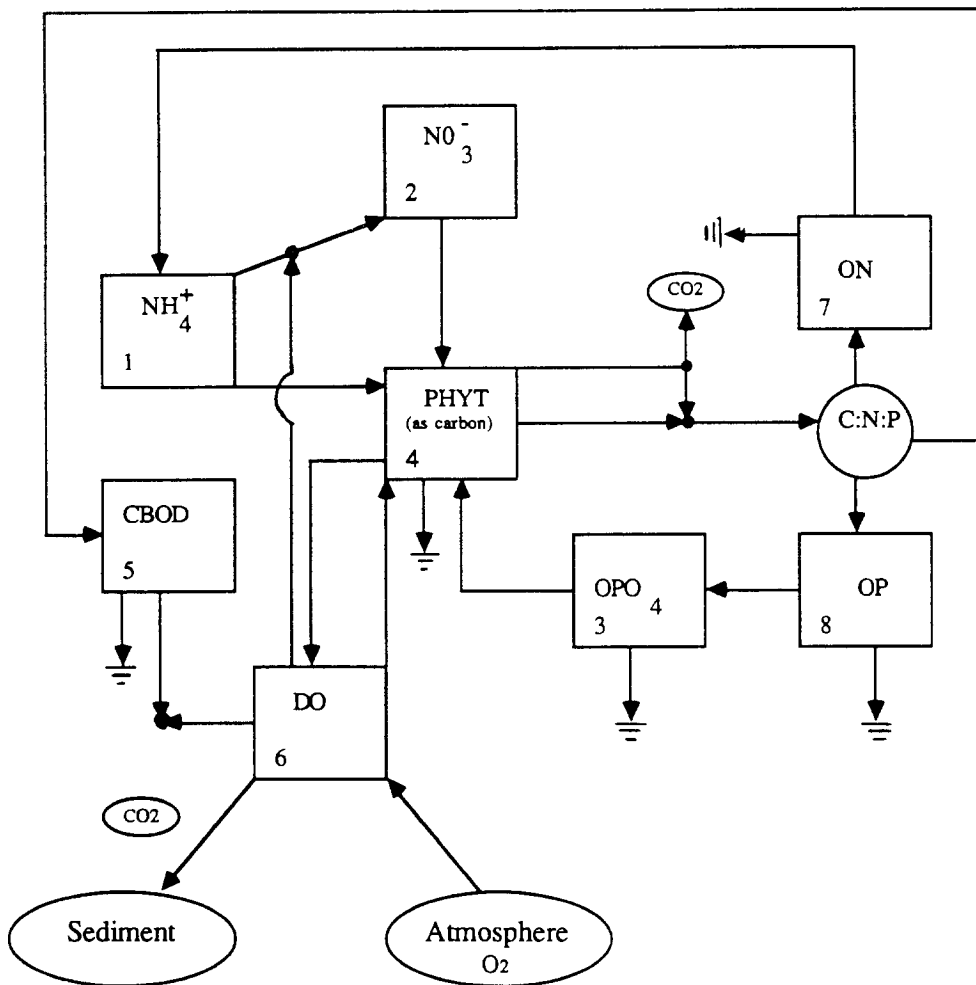


FIGURE 12.21. EUTRO4 coupled modeling concept of dissolved oxygen and eutrophication simulation. Key: 1. Ammonia—N; 2. nitrate—N; 3. orthophosphate—P; 4. phytoplankton biomass; 5. biodegradable organics (BOD); 6. dissolved oxygen; 7. organic nitrogen; 8. organic phosphorus. (From Ambrose et al., 1990.)

and eutrophication (phytoplankton or chlorophyll-*a* densities) discussed in the preceding sections are replaced in the WASP4 model by a coupled dynamic system shown on Figure 12.21. The eutrophication model of WASP is called EUTRO4 and it is a simplified version of the Potomac Eutrophication Model (PEM) (Thomann and Fitzpatrick, 1982; Thomann and Mueller, 1987).

As shown on Figure 12.21, the pollutant mass balance equation (Eq. (12.25)) in the EUTRO4 submodel is replaced by a set of coupled differential equations that includes BOD balance, ammonia, nitrite, nitrate, dissolved oxygen, phosphate, sediment, and chlorophyll-*a*, phytoplankton biomass. Six levels of modeling complexity are then identified by the creators of the model: (1) Simple BOD–DO steady-state balance (Streeter–Phelps equation), (2) modified Streeter–Phelps equation (similar to Eqs. (12.7) and (12.8)), (3) full linear dissolved oxygen balance, (4) simple eutrophication kinetics, (5) intermediate eutrophication kinetics, and (6) intermediate eutrophication kinetics with benthos.

Benthos and suspended sediment interactions may play an important role in the eutrophication process. Consider, for example, both ammonia and phosphorus, which can exist in the water environment as adsorbed on sediments (particulate form) (in this form they are not available for algal growth) and in available dissolved form.

Sediment–water interactions can be considered at two levels of complexity: in the first level the user specifies the fluxes of ammonia, phosphate, and SOD in the sediment–water interface. These inputs may be specified as either constant or time variable (seasonal). In the second level these fluxes are calculated from a modeling diagenesis process in the sediment layer.

The particulate nutrients as well as produced organic matter may settle in reaches with low velocity. Subsequently the nutrients stored in benthos can be released back into the water column. The decomposition of organic matter in benthic sediment can have a profound effect on the concentration of the dissolved oxygen and nutrients in the overlying water. The diagenesis processes thus determine the sediment oxygen demand.

The growth of phytoplankton is stimulated by the levels of available nutrients (ammonia, nitrate, and dissolved phosphates) and by light. The reaction rates are affected by temperature.

pH AND ACIDITY

The pH that expresses the molar concentration of the hydrogen ion as its negative logarithm ($\text{pH} = -\log[\text{H}^+]$) is one of the primary indicators used for evaluation of surface-water quality and the suitability for various

beneficial uses. Most aquatic biota are sensitive to pH variations. Fish-kills and reduction and change of other species result when the pH is altered outside their tolerance limits. Biochemical degradation and transformation (for example, nitrification) are also pH sensitive and diminish when the pH reaches acidic levels. Most of the aquatic species prefer a pH near neutral but can withstand a pH in the range of about 6 (7 for nitrifiers) to 8.5.

The toxicity of many compounds can also be altered if the pH is changed. The solubility of many metals as well as other compounds is affected by pH, resulting in increased toxicity in the lower pH range. As pointed out in Chapter 6, leaching of some metals (aluminum) from soils caused by elevated acidity of precipitation represents a "time bomb" and a threat to sustainability of soil resources.

A change of pH and acidity of surface waters resulting from diffuse inputs can occur mainly from two sources: (1) acid precipitation, and (2) mine acid drainage. Both sources have similar origins and have been explained in previous chapters. Acid drainage from mines is a result of mine water being in contact with sulfur-bearing minerals, while acidity of precipitation is caused by atmospheric sulfur and nitrogen oxide emissions. Oxidation and hydrolysis of these compounds in surface or atmospheric water produces sulfuric and nitric acids, which then dissociate to H^+ , SO_4^{2-} , and NO_3^- ions.

Acid rain, which is defined as rain with a pH of less than 5.6, is a result of sulfuric (SO_2) and nitrate (NO_x) emissions from urban, industrial, transportation (automobile), and electric utility fuel-burning operations that use sulfur- and nitrogen-containing fuels. The process of acid rain formation was explained in Chapter 4. The lethal and sublethal effects of acid rain or acid drainage from mines have been noticed both in North America and Europe (Scandinavia) (Almer et al., 1974; Beamish and Harvey, 1972; Likens and Borman, 1974). Undesirable "oligotrophication" (a severe loss of productivity by low pH conditions) and fish-kills are the most visible and dangerous consequences of acidification. Loss of the natural fish population due to acidic rain and snowmelt inputs in the lakes of New York's Adirondack Mountains and many other pristine lakes of North America (primarily Canada) and Scandinavia have been documented and widely publicized. The damage to the fish population of these lakes was brought about by both long-term exposure to low pH and short-term pH shocks by runoff and snowmelt events. Table 12.10 shows the biological effects of decreased pH on lakes.

Many watersheds and surface-water systems have a natural ability to neutralize the excess acidity (although it was pointed out in Chapter 6 that this buffering may be limited and may be exhausted in some places if

TABLE 12.10 Biological Impact of Acidification

pH Range	General Biological Effects
6.5 to 6.0	Small decrease in species diversity of plankton and benthic invertebrate communities but no measurable change in total community abundance or production. Some adverse effects may be noticed for highly acid-sensitive fish species (e.g., fathead minnow, striped bass).
6.0 to 5.5	Loss of sensitive species of minnows and dace; in some waters decreased reproduction of lake trout and walleye. Visual accumulation of filamentous green algae in the littoral zone. Distinct decrease in species diversity of plankton and benthic invertebrate composition, but no appreciable decrease in total community biomass or productivity.
5.5 to 5.0	Loss of several important sport fish species, including lake trout, walleye, rainbow trout, and smallmouth bass, as well as additional nongame species such as creek chub. Further increase in the extent and abundance of filamentous green algae in lake littoral areas and streams. Continued shift in the species composition and decrease in species diversity of plankton, periphyton, and benthic invertebrate communities, decrease in total abundance and biomass of benthic invertebrates and zooplankton in some waters. Inhibition of nitrification.
5.0 to 4.5	Loss of most of fish species, including most important sport fish species, such as brook trout and Atlantic salmon, few fish species able to survive and reproduce below pH 4.5. Measurable decline in the whole system rates of decomposition of some forms of organic matter, potentially resulting in decreased rates of nutrient cycling. Substantial decrease in the number of species of zooplankton and benthic invertebrates and further decline in the species richness of the phytoplankton and periphyton communities, measurable decrease in the total community biomass of zooplankton and benthic invertebrates in most waters. Reproductive failure of some acid-sensitive species of amphibians such as salamanders and some frogs.

Source: After Baker et al. (1990).

the acidic inputs remain unabated or even increase in the future). During the overland flow, rain and snowmelt dissolve calcium- and magnesium-containing rocks, leach aluminum from soils, and are enriched by mineral and organic salts, such as phosphates and humates. In urban areas, buffering is provided by the infrastructure, as is pointed out in Chapter 8. The elutriated constituents from the soils, minerals, and infrastructure often provide enough buffering capacity to maintain the pH of surface runoff within the acceptable ranges. At this time, rain acidity does not seem to have a noticeable adverse effect on larger bodies of water, such as the Great Lakes, that have elevated alkalinity, hardness, and salt content. However, it has also been pointed out that leached dissolved

aluminum (not a problem in nonacidic bodies of water) can reach toxic levels in acidic bodies of water (Baker and Schofield, 1982).

The ability of surface waters to neutralize acidic inputs depends primarily on carbonate (CO_3^{2-}) and bicarbonate (HCO_3^-) content that is expressed as *alkalinity*. Some North American and Scandinavian lakes are particularly sensitive to acidic inputs. These lakes have watersheds underlain by siliceous hard rocks, such as granite, some gneisses, quartzites, and quartz sandstones. These materials are resistant to weathering and produce waters that contain very low concentrations of neutralizing compounds (alkalinity less than $30 \text{ mg CaCO}_3/\text{l}$). When acid rain falls on such watersheds the acids are not neutralized during the overland flow, and streams and lakes become acidified. A relationship between sulphur loads in acidic rainfall and the acidity of Swedish lakes is shown in Figure 12.22.

Areas of highly siliceous bedrock are widespread on the Precambrian Fennoscandian Shield in Scandinavia, the Rockies, the Canadian Shields, New England, the Adirondack Mountains of New York, the Appala-

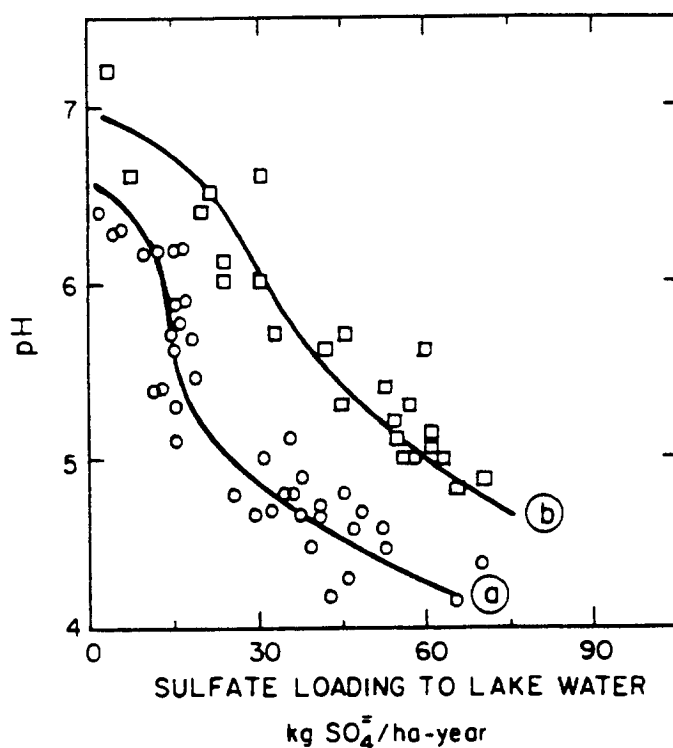


FIGURE 12.22. Sulphate loading effects on the pH of Swedish lakes. (a) Very sensitive lake systems. (b) Less sensitive lake systems. (Reprinted from Glass, Glass, and Renie [1979] with permission; copyright 1979 by the American Chemical Society. Data from National Environmental Protection Board, Jolna, Sweden.)

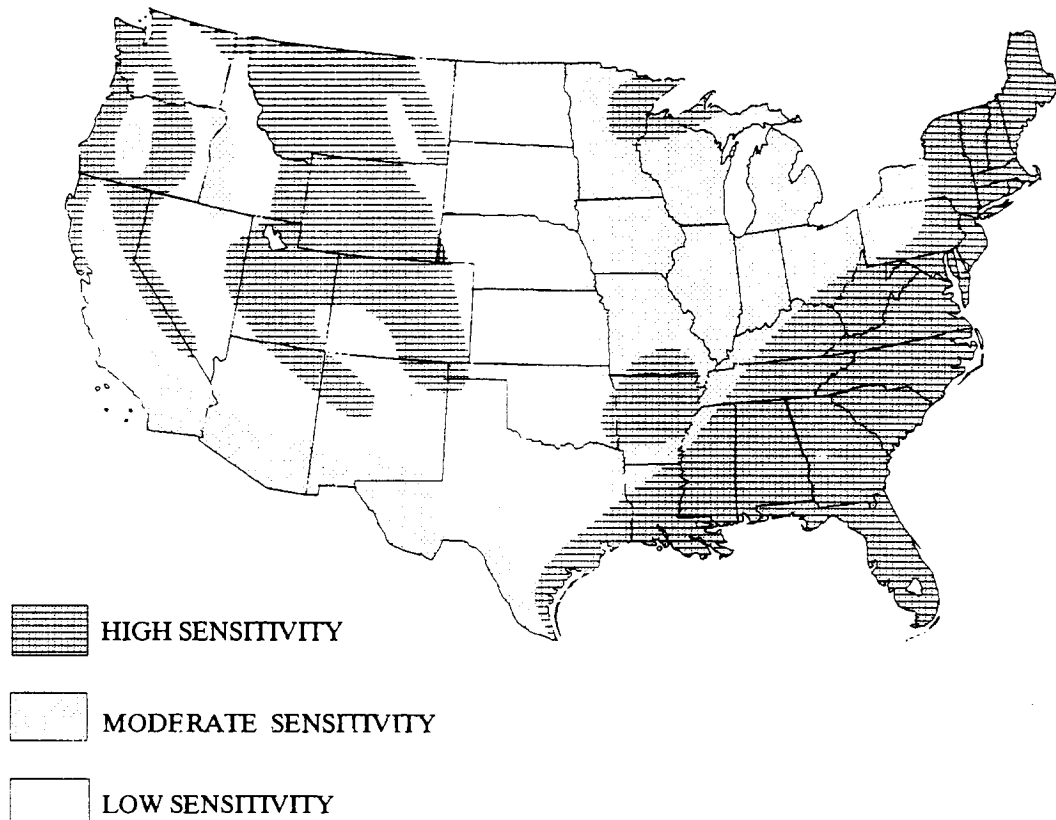


FIGURE 12.23. Areas of the conterminous United States sensitive to acid deposition based on soils, climatic conditions, geology, and types of vegetation. (Source: U.S. Environmental Protection Agency.)

chians, and smaller areas elsewhere. These areas contain lakes that are very sensitive to acid rainfall inputs (Fig. 12.23).

As reported by Likens et al. (1979) the acidification of thousands of freshwater lakes and streams in southern Norway has affected fish populations in a 33,000-km² area. As shown in Table 12.10, other adverse effects attributed to acidification must also be considered. Effects of acidification and acid deposition on surface-water bodies were also discussed by Haines (1981), Huckabee et al. (1989), Stokes (1986), and others.

BACKGROUND (NATURAL) WATER QUALITY

As defined in Chapters 1 and 7, background water quality represents the chemical and biological composition of surface waters that would result from natural causes and factors. However, some natural impacts may be transient and devastating to water quality, such as the eruption of Mount

St. Helen, which had afterward deposited an enormous amount of ash in the streams and lakes of the U.S. Northwest. Hence, for planning purposes and water quality evaluations one should consider more or less steady processes and/or common expectable events.

Approaching the natural or background water quality is also the goal of integrated water pollution abatement planning. It is impossible and legally unenforceable to eliminate all sediment from rivers, as well as it is impossible to bring BOD down to zero or oxygen to saturation values, notwithstanding the fact that such measures could have severe water quality and other consequences. Estimating the background (natural) water quality is also a key in use attainability studies since, legally, use-based water quality standards cannot be enforced if their violation is caused by natural causes (see Chapter 16).

The U.S. Geological Survey has established the National Hydrologic Benchmark Network of monitoring stations. These freshwater stations are located in watersheds that are among the least affected by the activities of man that can be found across the United States. Most of these stations are located in national parks, wilderness areas, state parks and forests, and similar areas protected from development. Water quality measured at these stations may provide the best approximation within the ecoregional context of the background (natural) levels of contaminants in U.S. surface waters.

A study by the Midwest Research Institute compiled the data of the average concentrations of various pollutants from undisturbed streams. The values from 57 stations composing the National Hydrologic Benchmark Network showed distinct regional distribution as summarized in Table 12.11. However, it should be noted that the ranges for BOD₅, nitrate, suspended solids, phosphate, and bacteria reported in the table are based on approximately only one station per state.

TABLE 12.11 Approximate Regional Natural Water Quality—Average Annual Concentrations

Parameter	U.S. Regions				
	Eastern	Midwest	Great Plains	Mountains	Pacific
Suspended sediment (mg/l)	5–10	10–50	20–100	5–20	2–5
BOD ₅ (mg/l)	1.0	1–3	2–3	1–2	1
Nitrate N (mg/l)	0.05–0.2	0.2–0.5	0.2–0.5	0.1	0.05–0.1
Total P (mg/l)	0.01–0.02	0.02–0.1	0.1–0.2	0.05	0.05–0.1
Total coliforms (MPN/ 100 ml)	100–1000	1000–2000	500–2000	100	100–500

Source: From McElroy et al. (1976).

Background levels of some toxic metals and background mineral contributions were discussed in Chapter 7. It must be remembered that there are no background levels for organic chemicals (pesticides, PAHs, halogenated hydrocarbons, volatile organic chemicals, etc.) since these substances are man-made and are alien to nature. The apparent background levels of these substances mostly originate from atmospheric fallout.

The natural levels of biodegradable organic pollution as well as sediment loads depend on the type of stream, the character of the basin, and geographical regions. There are four general types of undisturbed types of native lands (excluding mountains): *arid land (including deserts)*, *prairie*, *wetland*, and *woodland*. These four basic types have various forms, depending on the geographical location and elevation. It is obvious that the loads of constituents and background water quality in these four natural lands will vary but there are some common characteristics:

1. Streams from *arid lands* are often ephemeral and typically have very high sediment content during intense but infrequent storm events. The salinity of these streams may also be elevated. Nutrient (N and P) content is very low.
2. *Prairie streams* have elevated solids content during wet-weather flow events. Nutrient loading is low. Actually prairies represent a sink of atmospheric nutrient loads.
3. *Forested land*
 - a. *Mountain forests* generally exhibit the best water quality; also they have low mineral and almost no organic contents.
 - b. *Lowland forest* streams have higher organic content. Often these streams originate in wooded wetlands and water may contain residues of organic decomposition occurring in soils and wetlands. These streams would have measurable BOD and COD contents.
4. *Headwater wetlands*, that is, streams draining wetlands and wetland bodies of water themselves, have higher organic content and may also have low dissolved oxygen content caused by the decomposition of organic matter. Humic substances increase color and turbidity of these water bodies.

Northern wetlands have two distinct seasons. During the productive season, wetlands are sinks of nutrients and other water contaminants. During the dormant season, nutrients and contaminants may be released. Typically stream flow from northern wetlands is more acidic, contains organic acids and appreciable levels of nutrients and other contaminants, and has relatively low dissolved oxygen levels (Table 12.12). Natural

TABLE 12.12 Average Concentration of Contaminants in Runoff from Natural Minnesota Peatlands

Characteristic	Units	Peatland Type		
		Bog	Transition	Fen
Dissolved oxygen	mg/l	6.4	5.4	5.6
O ₂ saturation	%	56	50	53
pH		5.6	6.5	7.0
Alkalinity as CaCO ₃	mg/l	10	29	75
Color	mg/l	311	260	242
Aluminum	mg/l	0.55	0.25	0.25
Mercury	µg/l	6	3	5
Arsenic	µg/l	2	3	3
Selenium	µg/l	1	1	1
Total phosphorus	mg/l	0.06	0.05	0.08
Total Kjeldahl nitrogen ^a	mg/l	1.5	1.6	1.8
Nitrate—N	mg/l	0.06	0.05	0.08
Humic acid	mg/l	11	9	9
Fulvic acid	mg/l	100	104	97
Chemical Oxygen Demand	mg/l	118	104	97

Source: From Clausen and Brooks (1983).

^a Ammonia plus organic nitrogen.

wetlands can be classified as bogs or fens (see Chapter 14). Bogs are acidic and nutrient poor because they are isolated from regional ground water, while fens are nourished by regional ground water. Many wetlands are transitional (Clausen and Brooks, 1983).

It is important to note that the background (natural) contaminant loads are not related to the designated water use of the body, while the existing receiving water quality standards are. The natural water quality and constituent loads can be related to morphological, geological, and geographical characteristics, land cover, soil type, and other ecological factors. *Ecoregions* represent regions with the same natural water quality and ecological characteristics (Omernik, 1987; Gallant et al., 1989). Figure 12.5 showed the distribution of the major ecoregions of the conterminous United States. Apparently, natural (background) water quality within an ecoregion will vary less than between the ecoregions.

Generally, natural background water quality should be measured rather than estimated. In many ecoregions there are still some headwater stream sections that may be considered as unaffected or undisturbed by man's activities. As pointed out in the section of this chapter titled "The Aquatic Ecosystem," these stream sections can then be considered as

reference stream sections. Background water quality represents the limit of the waste assimilation process that polluted waters should eventually approach during recovery.

References

- Almer, B., C. Ekstrom, E. Hornstorm, and V. Miller (1974). Effect of acidification on Swedish lakes, *Ambio* 3:30-36.
- Ambrose, R. B., Jr., T. A. Wool, J. P. Connolly, and R. W. Schranz (1990). *WASP4, A Hydrodynamic and Water Quality Model—Model Theory, User's Guide, and Programmer's Guide*, revision 1, Environmental Research Laboratory, U.S. Environmental Protection Agency, Athens, GA.
- Anonymous (1987). *A Water Resources Management Plan for the Milwaukee Harbor Estuary*, Southeastern Wisconsin Regional Planning Comm., Waukesha, WI.
- Baker, J. P., and C. L. Schofield (1982). Aluminum toxicity to fish in acidic waters, *Water Air Soil Pollut.* 18:289-309.
- Baker, J. P., D. P. Bernard, S. W. Christensen, and M. J. Sale (1990). Biological effects of changes in surface water acid-base chemistry, in *National Acid Precipitation Assessment Program, Acidic Deposition: State of Science and Technology*, Vol. II, September 1990, State of Science and Technology Report 13.
- Beamish, R. J., and H. H. Harvey (1972). Acidification of the La Cloche Mountain Lakes: Resulting fish mortalities, *J. Fish Res. Board Can.* 29(8):1131-1143.
- Bowman, B. G. T., and J. J. Delfino (1980). Sediment oxygen demand techniques: A review and comparison of laboratory and in-situ systems, *Water Res.* 14(5):491-500.
- Carlson, R. E. (1977). A trophic state index for lakes, *Limnol. Oceanogr.* 22:361-369.
- Churchill, M. A., H. L. Elmore, and R. A. Buckingham (1962). The prediction of stream reaeration rate, in *Advances in Water Pollution Research*, Vol. 1, Proceedings, IAWPR International Conference, London, Pergamon Press, Oxford.
- Clausen, J. C., and K. N. Brooks (1983). Quality of runoff from Minnesota peatlands. 1. A characterization, *Wat. Resour. Bull.* 19(5):763-767.
- Committee on Restoration of Aquatic Ecosystems (1992). *Restoration of Aquatic Ecosystems*, National Academy Press, Washington, DC.
- DiToro, D. M. (1986). A diagenetic oxygen equivalent model of sediment oxygen demand, in *Sediment Oxygen Demand, Processes, Modeling, and Measurement*, K. J. Hatcher, ed., University of Georgia Press, Athens, GA, pp. 171-208.
- DiToro, D. M., P. R. Paquin, K. Subburamu, and D. A. Gruber (1990). Sediment oxygen demand model: Methane and ammonia oxidation, *J. Environ. Eng. ASCE* 116(5):945-986.
- Edberg, N., and B. V. Hofsten (1973). Oxygen uptake of bottom sediments studied in situ and in the laboratory, *Water Res.* 7(9):1285-1294.
- Fair, G. M., E. W. Moore, and H. A. Thomas, Jr. (1941). The natural purification of river muds and pollutional sediments, *Sew. Works J.* 13:270-307, 756-778.

- Fausch, K. D., J. R. Karr, and P. R. Yant (1984). Regional application of an index of biotic integrity based on stream-fish communities, *Transact. Am. Fish. Soc.* **113**:39–55.
- Field, R., and R. Turkeltaub (1981). Urban runoff receiving water impact: Program overview, *J. Environ. Eng. Div. ASCE* **107**(EE1):83–100.
- Filos, J., and A. Molof (1972). Effect of benthic deposits on oxygen and nutrient economy of flowing waters, *J. WPCF* **44**(4):644–662.
- Foree, E. G., W. J. Jewell, and P. L. McCarty (1971). The extent of nitrogen and phosphorus regeneration from decomposing algae, in *Proceedings, Fifth International Conference on Water Pollution Research*, IAWPR, San Francisco, CA, Pergamon Press, Oxford.
- Gallant, A. L., et al. (1989). *Regionalization as a Tool for Managing Environmental Resources*, EPA 600/3-89/060, U.S. Environmental Protection Agency, Corvallis, OR.
- Gardinier, R. D., M. T. Auer, and R. P. Canale (1984). Sediment oxygen demand in Green Bay (Lake Michigan), in *Proceedings, 1984 Specialty Conference of ASCE*, American Society of Civil Engineers, New York, pp. 514–519.
- Glass, N. R., G. E. Glass, and P. J. Renie (1979). Effect of acid precipitation, *Environ. Sci. Technol.* **13**:1350–1355.
- Haines, T. A. (1981). Acidic precipitation and its consequences for aquatic ecosystems: A review, *Trans. Am. Fish. Soc.* **110**:669–706.
- Heaney, J. P., and W. C. Huber (1984). Nationwide assessment of urban runoff impact on receiving water quality, *Water Resour. Bull.* **20**(1):35–42.
- Huckabee, J. W., et al. (1989). An assessment of the ecological effects of acidic deposition, *Arch. Environ. Contam. Toxicol.* **18**:3–27.
- Hutchinson, G. E. (1969). Eutrophication, past and present, in *Eutrophication: Causes, Consequences, Correctives*, G. A. Rohlich, ed., National Academy of Sciences, Washington, DC.
- Kadlec, R. H. (1988). *Denitrification in Wetland Treatment Systems*, paper presented at 61st Conference of the Water Pollution Control Federation, Session 26, Dallas, TX.
- Karr, J. R., K. D. Fausch, P. L. Angermeier, P. R. Yant, and I. J. Schlosser (1986). *Assessing Biological Integrity of Running Waters. A Method and Its Rationale*, Illinois Natural History Survey, Spec. Publ. No. 5, Champaign, Ill.
- Karr, J. R., and D. R. Dudley (1981). Ecological perspectives on water quality goals, *Environ. Manage.* **5**:55–68.
- Keefer, T. N., R. K. Simons, and R. S. McQuivey (1979). Dissolved oxygen impact from urban storm runoff, EPA 600/2-79-156, U.S. Environmental Protection Agency, Cincinnati, OH.
- Keeney, D. R. (1973). The nitrogen cycle in sediment-water systems, *J. Environ. Qual.* **2**(1):15–19.
- Keeney, D. R., S. Schmidt, and C. Wilkinson (1975). *Concurrent Nitrification-Denitrification at the Sediment-Water Interface as a Mechanism for Nitrogen Losses from Lakes*, Tech. Rep. No. 75-07, Water Resources Center, University of Wisconsin, Madison, WI.
- Kolkwitz, R., and M. Marson (1908). Ökologie der pflanzlichen Saprobien, *Berl. Deutscher Botan. Ges.* **261**:505–519.

- Krebs, C. J. (1989). *Ecological Methodology*, Harper and Row, New York.
- Krenkel, P. A., and V. Novotny (1980). *Water Quality Management*, Academic Press, New York.
- Krenkel, P. A., and G. T. Orlob (1962). Turbulent diffusion and the reaeration coefficient, *J. Sanit. Eng. Div. ASCE* **88**(3):53–83.
- Kreutsberger, W. A., T. L. Meinholz, M. Harper, and J. Ibach (1980). Predicting the impact of sediments on dissolved oxygen concentrations following combined sewers overflow events in the Lower Milwaukee River, *J. WPCF* **52**: 192–201.
- Lee, G. F., W. Rust, and R. A. Jones (1978). Eutrophication of water bodies: Insight of an age-old problem, *Environ. Sci. Technol.* **12**:900–908.
- Likens, G. E., and F. H. Borman (1974). Acid rain: A serious regional environmental problem, *Science* **184**:1176–1179.
- Likens, G. E., R. F. Wright, J. N. Galloway, and T. J. Butler (1979). Acid rain, *Sci. Am.* **241**:43–51.
- McDonnell, A. J., and S. D. Hall (1969). Effect of environmental factors on benthic oxygen demand, *J. WPCF* (res. supplement) **41**:353–363.
- McElroy, A. D., S. Y. Chiu, J. W. Nebgen, A. Aleti, and F. W. Bennett (1976). *Loading Functions for Assessment of Water Pollution from Nonpoint Sources*, EPA 600/2-76/151, U.S. Environmental Protection Agency, Washington, DC.
- Mason, C. F. (1991). *Biology of Freshwater Pollution*, 2d ed., Longman Sci. & Technical, Wiley, New York.
- Meadows, M. E., D. W. Weeter, and J. M. Green (1978). Assessing nonpoint water quality for small streams, *J. Environ. Eng. Div. ASCE* **104**(EE6):1119–1133.
- Ministry of the Environment (1991). *Environment of the Czech Republic*, Publishing House EkoCentrum, Brno, Czechoslovakia.
- Myers, C. F., J. Meek, S. Tuller, and A. Weinberg (1985). Nonpoint sources of water pollution, *J. Soil Water Conserv.* **40**(1):14–18.
- Niedzialkowski, D., and D. Athayde (1985). Water quality data and urban nonpoint source pollution: The Nationwide Urban Runoff Program, in *Perspectives on Nonpoint Source Pollution*, Proceedings, Nationwide Conference, U.S. Environmental Protection Agency, Washington, DC, pp. 437–441.
- Nonpoint Source Control Task Force (1983). Nonpoint source pollution abatement in the Great Lakes basin—An overview of post-PLUARG development, International Joint Commission, Windsor, Ontario.
- North American Lake Management Society—NALMS (1990). *The Lake and Reservoir Restoration Guidance Manual*, EPA 440/4-90-006, U.S. Environmental Protection Agency, Washington, DC.
- Novotny, V. (1985). Urbanization and nonpoint pollution, in *Proceedings, Fifth World Congress on Water Resources*, International Water Resources Association, Paper No. 17, Asp. 6, Brussels, Belgium.
- Novotny, V., and G. Bendoricchio (1989). Water quality—Linking nonpoint pollution and deterioration, *Water Environ. Technol.* **1**(Nov.):400–407.
- Novotny, V., and P. A. Krenkel (1975). A waste assimilative capacity model for a shallow, turbulent stream, *Water Res.* **9**:233–241.

- Novotny, V., and S. Zheng (1988). Impact of nonpoint pollution on a Great Lakes freshwater harbor estuary, in *Proceedings, Spec. Group Seminar, Water Quality Impact of Storm Sewage Overflows on Receiving Waters*, J. B. Ellis, ed., 14th Biennial Conf., IAWPRC, Brighton, England, pp. 27–36.
- Novotny, V., K. Imhoff, M. Olthof, and P. Krenkel (1989). *Karl Imhoff's Handbook of Urban Drainage and Wastewater Disposal*, Wiley, New York.
- O'Connor, D. J. (1960). Oxygen balance of an estuary, *J. Sanit. Eng. Div. ASCE* **86**(SA3):35–55.
- O'Connor, D. J. (1988). Models of sorptive toxic substances in freshwater systems, *J. Environ. Eng. Div. ASCE* **114**(3).
- O'Connor, D. J., and W. E. Dobbins (1958). Mechanism of reaeration in natural streams, *Trans. ASCE* **123**:641–666.
- Odum, H. T. (1956). Primary productivity of flowing streams, *Limnol. Oceanogr.* **1**(2):102–117.
- Ohio Environmental Protection Agency (1987). *Biological Criteria for the Protection of Aquatic Life*, Ohio EPA, Columbus, OH.
- Omernik, J. M. (1987). Ecoregions of the conterminous United States, *Ann. Assoc. Am. Geol.* **77**(1):118–125.
- Orlob, G. T., ed. (1983). *Mathematical Modeling of Water Quality*, Wiley, New York.
- Owens, M., R. W. Edwards, and J. W. Gibbs (1964). Some reaeration studies in streams, *Int. J. Air Water Pollut.* **8**:469.
- Painter, H. A. (1970). A review of literature on inorganic nitrogen metabolism in microorganisms, *Water Res.* **4**:393–450.
- Paterson, D. J., E. Epstain, and J. McEnvoy (1975). *Water Pollution Investigation: Lower Green Bay and Lower Fox River*, EPA 600/3-77/105, U.S. Environmental Protection Agency, Washington, DC.
- Phelps, E. B. (1944). The oxygen balance, in *Stream Sanitation*, Wiley, New York.
- Plafkin, J. L., M. T. Barbour, K. D. Porter, S. K. Gross, and R. M. Hughes (1989). *Rapid Bioassessment Protocols for Use in Streams and Rivers: Benthic Macroinvertebrates and Fish*, EPA/444/4-89-001, U.S. Environmental Protection Agency, Washington, DC.
- Rankin, E. T., C. O. Yoder, and D. Mishne (1990). *Ohio Water Resources Inventory*, executive summary and Vol. 1, Ohio Environmental Protection Agency, Columbus, OH.
- Robertson, C. K. (1979). Quantitative comparison of the significance of methanogenesis in the carbon cycle of two lakes, *Arch. Hydrobiol. Ergebnis Limnol.* **12**:123–135.
- Rohlich, G. A. (1969). *Eutrophication: Causes, Consequences, Correctives*, National Academy of Science, Washington, DC, pp. 3–7.
- Ruane, R. J., and P. A. Krenkel (1975). Nitrification and other factors affecting nitrogen in the Holston River, in *Proceedings, IAWPR Conference on Nitrogen as a Water Pollutant*, Copenhagen, Denmark.
- Rudd, J. W. M., and C. D. Taylor (1980). Methane cycling in aquatic environments, in *Advances in Aquatic Microbiology*, M. R. Droop and H. W. Jannach,

- eds., Vol. 2, Academic Press, New York, pp. 77–150.
- Sawyer, C. H. (1947). Fertilization of lakes by agricultural and urban drainage, *New Engl. Water Works Assoc.* **61**:109–127.
- Schindler, D. W. (1985). The coupling of elemental cycles by organisms: Evidence from whole lake chemical perturbations, in *Chemical Processes in Lakes*, W. Stumm, ed., Wiley, New York, pp. 225–250.
- Sládeček, V. (1979). Continental systems for the assessment of river water quality, in *Biological Indicators of Water Quality*, A. James and L. Evison, eds., Wiley, Chichester, England, New York, 3.1–3.32.
- Sloey, W. E., and F. L. Spangler (1978). Trophic status of the Winnibego pool lakes, in *Proceedings, Second Annual Conference, Wisconsin Section, AWRA, WRC-Univ. of Wisconsin, Madison*.
- Sonnen, M. B. (1980). Urban runoff quality: Information needs, *J. Tech. Counc. ASCE* **106**(TC1):29–41.
- Stokes, P. (1986). Ecological effects of acidification on primary producers in aquatic ecosystems, *Water Air Soil Pollut* **30**:421–424.
- Streeter, H. W., and E. B. Phelps (1925). *A Study of the Pollution and Natural Purification of the Ohio River*, Public Health Bull. 146, U.S. Public Health Service, Washington, DC.
- Stumm, W., ed. (1985). *Chemical Processes in Lakes*, Wiley, New York.
- Stumm, W., and J. J. Morgan (1962). Stream pollution by algal nutrients, in *Transactions, Twelfth Annual Conference of Sanitary Engineers*, University of Kansas.
- Thomann, R. V. (1972). *Systems Analysis and Water Quality Management*, McGraw-Hill, New York.
- Thomann, R. V., and J. J. Fitzpatrick (1982). *Calibration and Verification of a Mathematical Model of the Eutrophication of the Potomac Estuary*, prepared for Department of Environmental Services, Government of District of Columbia, Washington, DC.
- Thomann, R. V., and J. A. Mueller (1987). *Principles of Surface Water Quality Modeling and Control*, Harper & Row, New York.
- Thompson, P., R. Adler, and J. Landman (1989). *Poison Runoff*, National Resource Defense Council, Washington, DC.
- Torno, H. C. (1984). The nationwide urban runoff program, in *Proceedings, Third International Conference on Urban Storm Drainage*, Chalmers University of Technology, Göteborg, Sweden.
- Tuffey, T. J., J. V. Hunter, and V. A. Matulewich (1974). Zones of nitrification, *Water Resour. Bull.* **10**(3):555–564.
- U.S. Environmental Protection Agency (1974). *The Relationships of Phosphorus and Nitrogen to the Trophic State of Northeast and North-central Lakes and Reservoirs*, National Eutrophication Survey Working Paper No. 23, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1983). *Results of the Nationwide Urban Runoff Program*, Vol. 1. *Final Report*, WH-554, Water Planning Division, U.S. EPA, Washington, DC.

- U.S. Environmental Protection Agency (1987). *Quality Criteria for Water 1986*, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1990). *National Water Quality Inventory—1988 Report to Congress*, EPA 440-4-90-003, Office of Water, U.S. EPA, Washington, DC.
- Velz, C. J. (1984). *Applied Stream Sanitation*, 2d ed., Wiley, New York.
- Vollenweider, R. A. (1975). Input-output models with special reference to the phosphorus loading concept in limnology, *Schweiz. Z. Hydrol.* **37**:53–83.
- Vollenweider, R. A. (1976). Advances in defining critical loading levels for phosphorus in lake eutrophication, *Mem. Ist. Ital. Hydrobiol.* **33**:53–83.
- Walker, R. R., and W. J. Snodgrass (1986). Model for sediment oxygen demand in lakes, *J. Environ. Eng. ASCE* **112**(1):25–43.
- Wezernak, C. T., and J. J. Gannon (1968). Evaluation of nitrification in streams, *J. Sanit. Eng. Div. ASCE* **94**(SA5):883–895.
- Žáková, Z. (1989). Phytoplankton of reservoirs in relation to the trophic potential of inflow water, *Arch. Hydrobiol. Beih. Ergebn. Limnol.* **33**:373–376.

Toxic Pollution and Its Impact on Receiving Waters

The presence of toxic substances, such as organic and inorganic chemicals, heavy metals and radionuclides, has become a major environmental problem in recent years. The substances are present in varying degrees in all phases of the environment, air, water and land. They are transferred between and among these media, undergo transformation within each and accumulate in viable and nonviable constituents.

Donald J. O'Connor, Manhattan College, Bronx, NY

TOXICITY CONCEPTS

Surface waters are the primary recipients of waste materials. As long as the consequences of waste discharges are not injurious to the uses of the body of water or otherwise do not impair its integrity, then the body of water is not polluted. The uses that can be adversely affected by toxic pollution include, for example, water supply, fishing, irrigation, navigation, or recreation. The injury of the integrity of the body of water implies a harm done to the aquatic ecosystem (biota within the system including fish, insects, planktonic microorganisms, and aquatic plants). In order to protect the uses or the aquatic ecosystem from injury the contaminant levels must be kept below a certain "safe" threshold. However, almost any compound, including even kitchen salt (NaCl) has a toxic threshold which, if exceeded, will cause harm to the aquatic ecosystem and/or man (Krenkel and Novotny, 1980). But the same example of toxicity of sodium chloride clearly shows that the toxicity threshold varies among the species. Freshwater organisms would die in

salt or even brackish environment and, vice versa, saltwater organisms would die if salinity is reduced. Diversion of Mississippi River flood waters into brackish Louisiana coastal bodies of water during high floods results in die-off of shell fish accustomed to the higher salt concentrations of these waters. Hence, in some cases organisms can adapt to higher concentrations of some potentially toxic compounds.

The presence and exposure of organisms to the toxic levels of contaminants can have effects such as fish-kills and disease and contamination of fish flesh to levels that make fish unfit for human consumption. In addition toxic materials can damage or stop the biological processes occurring in the aquatic ecosystems, including long-term inhibition of growth, reproduction, and migration of organisms, and adverse effects on the rate of degradation of biodegradable contaminants.

The definition of toxicity in the Clean Water Act (Section 502(13)) is very specific:

The term "toxic pollutants" means those pollutants, or combinations of pollutants, including disease-causing agents, which after discharge and upon exposure, ingestion, inhalation or assimilation into any organism, either directly from the environment or indirectly by ingestion through food chains, will, on the basis of information available to the administrator, cause death, disease, behavioral abnormalities, cancer, genetic mutation, physiological malfunctions (including malfunctions in reproduction) or physical deformations, in such organisms or their offspring.

This definition of toxicity encompasses essentially all the possible adverse effects of pollutants on all types of organisms (Weber, 1981).

The concept of "biological integrity" was defined in Chapter 12. The biological integrity is related to the habitat of an unimpaired body of water. It also includes such impacts as habitat alteration by channel modification that do not result from toxic discharges or contamination. Typically "toxicity" is determined by bioassays in which test organisms are exposed in laboratory or in-situ (field) conditions to various doses or concentrations of the pollutant, while "biological integrity" is determined by in-situ examination of the biological community itself and comparing it to an unimpacted (reference) biotic composition within the same ecoregion.

Even though almost all water quality constituents can become toxic at high enough levels, there is a definite number of compounds that are either toxic at relatively low levels or at levels that may result from waste discharges. These potentially toxic compounds have been designated by the Environmental Protection Agency as *priority pollutants*. Some of the priority pollutants are *carcinogenic*, that is, they can increase the risk of

cancer to the human population and/or to fish. The list of priority pollutants is continuously under review and periodic updating and is growing. At the time of writing this book it contained 129 pollutants, of which 13 were toxic elements (metals) and the remaining were mostly organic chemicals. The priority pollutants are listed in the Appendix B.

In Europe, the European Community has listed the most dangerous toxic substances on the so-called "Black List" and less dangerous pollutants on the "Gray List" (Mason, 1991).

Toxicity is generally considered a man-made (cultural) phenomenon resulting from discharges of contaminants into surface waters. However, sometimes toxicity may be caused by natural phenomena. Hydrogen sulfide or hydroxylamine, both known byproducts of natural processes taking place in natural waters, can be toxic to aquatic organisms. The periodic table includes over 90 elements from hydrogen to transuraniums, and all but 20 can be characterized as metals. As many as 59 of these elemental metals can be classified as "heavy metals" and potentially toxic. However, only 17 of these metals are considered both very toxic and available in places at concentrations that exceed toxicity levels. Of these 17 toxic metals (Table 13.1), nine are being mobilized into the environment by man at rates greatly exceeding those of natural geological processes (Chapman, 1978; Weiss et al., 1975).

Figure 13.1 shows the effect of toxic discharges from point and diffuse sources on human health and the integrity of aquatic life. Almost all toxic organic compounds are man-made and have been introduced since the

TABLE 13.1 Toxic Metals of Particular Environmental Concern

Very Toxic and Readily Accessible	Man-Induced Mobilization Higher than Natural Rate
Co, Bi	Ag ^a
Ni, Cu	Cd
Zn, Sn ^a	Cu
Se, ^a Te ^a	Hg ^a
Pd, ^a Ag ^a	Ni
Cd, Pt ^a	Pb ^a
Ag, ^a Hg ^a	Sb
Tl, ^a Pb ^a	Sn ^a
Sb	Zn

From Chapman (1978).

^aMetal alkyls stable in aqueous environment, reported to be biomethylated in sediments.

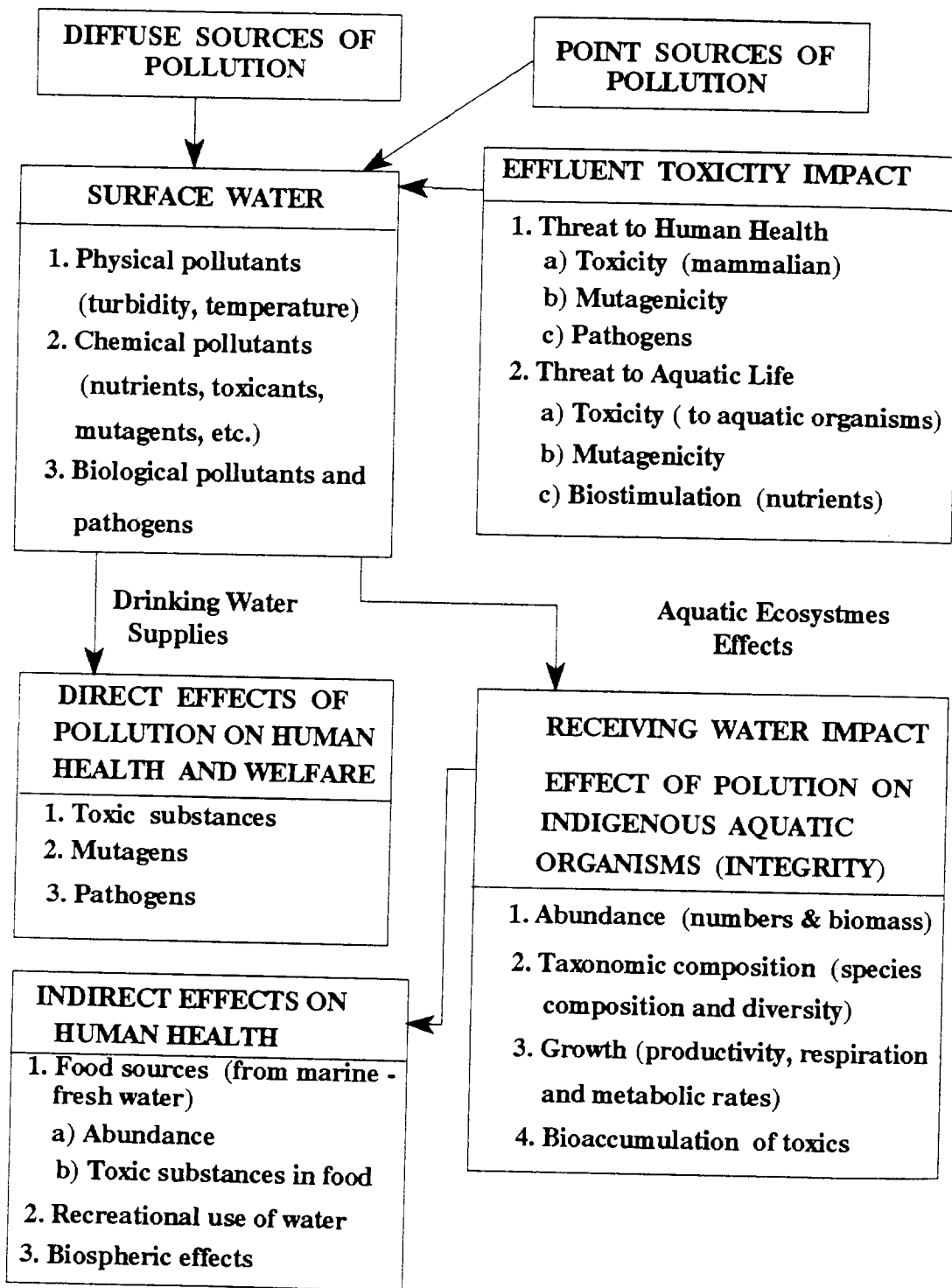


FIGURE 13.1. Impact of toxic emissions on human health and integrity of aquatic biota. (After Weber, 1981.)

1900s. With one or two exceptions, there are no background (natural) levels of these contaminants.

The sources of toxic pollutants are so numerous that an attempt to list them may omit some of them. Sources of diffuse nature have been listed in the preceding chapters. As pointed out in Chapter 7, toxic contamination from mining metals has been occurring for 2000 years. But it was the introduction of toxic organic chemicals during and after World War II that made the distribution of toxics globally widespread and threatening to global and regional ecological systems. Without control the widespread use of toxic compounds by man, including the use of household chemicals, pesticide and fertilizer use in agriculture, industrial emissions, and disposal of waste, may ultimately be threatening to the existence of life on this planet. On the other hand, life in today's society is impossible without the use of chemicals. Many economists claim that high yields of agricultural crops needed to feed an increasing population throughout the world can only be sustained with the use of chemicals. A sustainable use of chemicals that would not pose a harm to bodies of water and to terrestrial ecosystems must emerge in a relatively short time.

The toxic pollutants affecting receiving bodies of water include (Mason, 1991):

1. Metals, such as cadmium, copper, chromium, lead, mercury, nickel, and zinc, that arise from industrial operations, ores, mining, deteriorating infrastructures, traffic, agricultural use, and others sources.
2. Organic compounds, such as pesticides, PCBs, solvents, petroleum hydrocarbons, surfactants, organometallic compounds, phenols, formaldehyde, biochemical methylation of metals in aquatic sediments, etc.
3. Dissolved gases, such as chlorine and ammonium.
4. Anions, such as cyanides, fluorides, sulfides, and sulphates.
5. Acids and alkalis.

TOXICITY AND ITS MEASUREMENT

Toxicity may be defined as an alteration or impairment of the normal functions of organisms caused by an exposure to or ingestion of a compound or mixture of compounds. The effects of contaminants on living organisms are generally evaluated using the basic concepts that were originally formulated by Sprague (1969).

1. *Acute toxicity*. The exposure of organisms to a compound or a mixture of compounds will result in a crisis, usually in a short time during or following the exposure.

2. *Chronic toxicity*. The exposure will have a sublethal damaging impact on the organisms occurring over a longer period of time up to the entire life cycle.
3. *Lethal toxicity*. Exposure will result in death of the organism.
4. *Sublethal toxicity*. Exposure is damaging to the organism, but it will not result in death.
5. *Cumulative toxicity*. The effects on the organisms are brought about, or increase in strength, by successive exposure.

There are two considerations of the toxic levels of contaminants in aquatic water bodies. The first consideration is to protect human health; the second is to protect the well-being of aquatic life. The acute toxicity concept has been applied primarily to aquatic organisms and protection of aquatic biota. Toxicity levels (criteria) for the protection of human health are related to "acceptable" risk to the population of contracting an additional case of cancer (for carcinogenic toxicants) and/or a debilitating disease. These accepted risks are extremely low and below risk levels related to others of man's activities, such as driving a car, flying on a commercial jet plane, participating in sports, or walking on a street.

Toxicity Tests

The toxicity levels of compounds for aquatic organisms have been established by scientists in toxicity bioassay tests. Ideally the test organisms should include representatives from four groups, that is, microorganisms, plants, invertebrates, and fish. At least eight different families of species specified by states are required for a complete test of toxic effects. The test organisms should be amenable to captivity, accurately identified, relatively uniform in size and healthy, and acclimated to laboratory conditions.

The test organisms are placed in containers with various dilutions of the toxicants plus one container with test water only. The number of organisms surviving and/or unaffected (for example, by observing the mobility or respiration rate of the organisms) after the specified time periods (24, 48, 72, 96 hours) is recorded. The data are then used for establishing a functional dose (concentration)-response relationship. It is obvious that in the test the organisms will not respond uniformly to the dose of the toxic compounds because of different sensitivities to the compound. Therefore the most important parameter of interest in the toxicity bioassay test is the dose or concentration at which, after the specified test period, 50% of the test organisms survive or their life functions are not affected by the dose. A dose is defined as the amount

of the toxic compound ingested by the test organism divided by the body mass of the organism (for example, mg/kg). In the test of aquatic organisms the toxicity is related to concentration (mg of contaminant/volume of water, i.e., mg/l or $\mu\text{g/l}$) rather than to dose because of the inherent difficulty in establishing a dose.

The *lethal dose or concentration (LD or LC)* implies that an exposure of the test organism has resulted in death. The 50% survival dose or concentration value then represents so-called LD_{50} (dose) or LC_{50} (concentration), and it is a representative of the acute toxicity of the compound or of the waste containing a mixture of potentially toxic substances. The time of exposure is important in toxicity studies as well as in toxicity risk assessment studies of toxic discharges into receiving bodies of water. The $\text{LC}_{50}(48)$ is the concentration of a toxic material at which 50% of the test organisms died after 48 hours of exposure.

The *effective dose or concentration (ED or EC)* is a term used when other than lethal effects are considered such as impact on reproduction or respiratory stresses. The terms ED_{50} or EC_{50} are then equivalently used to describe such adverse effects in 50% of the test organisms within the prescribed test period.

The *chronic criteria* are not based on a dose-response relationship obtained in an bioassay, but rather on an observed long-term impact of the contaminants on the life functions of organisms. The endpoint of the chronic toxicity test is the *no observed effect concentration (NOEC)* and the *low observed effect concentration (LOEC)*. The NOEC is the highest concentration of toxicant to which the test organisms are exposed that causes no observable effects. The effects measured may include a decrease in reproduction and growth. The LOEC is the lowest concentration of toxicant to which the test organisms are exposed that causes an observed effect.

Review of Toxicity Tests

Currently, the water quality based pollution controls specify either chemically specific water quality limitations (such as those included in Appendix B) or whole body toxicity testing. The whole body toxicity test (conducted both on the effluent and the receiving water) appropriately indicates the effect of the wastes on the aquatic life. Toxicity tests are classified based on the method of adding test solutions and the duration of the test (APHA, 1989):

1. *Static test.* In this test, the test organism remains in the same water for the entire duration of the test. The tested (effluent or receiving body) and dilution waters are mixed in a chamber to the desired concen-

tration. The test organism is then placed in the chamber containing the diluted tested water. The static nature of the test may cause erroneous results under certain conditions. For example, a high BOD content may cause a depletion of dissolved oxygen in the mixture, which will result in the death of the organisms. Also, some toxic compounds may degrade or volatilize during the test, thus distorting the toxicity effect on the test organisms.

2. *Recirculation test.* In this test the mixture in which organisms are residing is pumped through an apparatus, such as a filter, to maintain water quality but not to reduce the concentration of the test material. This type is not routinely used because it is expensive and the results may be distorted.
3. *Renewal test.* The renewal test is similar to the static test except the test solutions and control water are periodically renewed and the test organisms are transferred to chambers with freshly prepared mixtures of test and control water or by replacing test mixtures in the original chambers.
4. *Flowthrough test.* The test solutions and control water flow into and out of the chambers in which the test organisms are maintained. The flow may be intermittent or continuous. Stock solutions of the test material can be continuously mixed with the dilution water in different proportions. Flowthrough tests are desirable for high BOD samples and for those containing volatile or unstable substances. Organisms with high metabolic rates are difficult to maintain in standing water, whereas flowthrough tests provide well-oxygenated test solutions, stable concentrations, and continuous removal of metabolic wastes.

Several factors should be considered in making the choice of toxicity test system. On-line continuous flowthrough testing can sample and measure "peaks" of toxicity as they may occur during the test period. This may be of concern when testing runoff from industrial sites that exhibits flushes of toxics. If the discharge toxicity is highly variable and continuously discharges (industrial effluents) either a flowthrough or a renewal test would be appropriate (U.S. EPA, 1991). However, the sample should be a composite collected over the period of discharge. If the effluent is not considered variable, such as a discharge from a pond with a long detention time, then a static or renewal test using a grab or 24-hour composite sample would be an appropriate test system. Flowthrough tests are more costly and require a complex delivery system that is not typically available for measuring the toxicity of diffuse sources.

Duration of the test determines what type of toxicity is been investi-

gated. *Acute toxicity tests* are short-duration tests, typically representing a small fraction of the lifetime of the organisms. The concentrations are higher and the impact on the organisms is severe, usually death. The test is usually completed in less than 4 days (96 hours). In *chronic toxicity tests*, organisms are exposed to lower concentrations, preferably for the entire reproductive life of the organism. In *subchronic tests*, the exposure of the organisms lasts for a time period that is less than a complete reproductive life cycle, but longer than that for acute toxicity testing. Testing involves the exposure of organisms during sensitive life cycle periods, such as early stages of development, critical life stages, embryo-larval, or frog egg tests (Rand and Petrocelli, 1985). Typically, a 7-day period is assumed unless specified otherwise.

Test Organisms

The species used in characterizing the toxicity of an effluent or runoff will depend on the requirements of the regulatory agencies. The U.S. Environmental Protection Agency (1991) recommends as a minimum three species (for example, a vertebrate-fish, an invertebrate, and a plant) should be tested; however, the EPA recommends against selecting “*a most sensitive species*” for toxicity testing. Species that have been widely used in toxicity tests and are acceptable test organisms in freshwater are listed below (after U.S. EPA, 1985):

Vertebrates

1. Cold Water
 - a. Brook trout (*Salvelinus fontinalis*)
 - b. Coho salmon (*Oncorhynchus kisutch*)
 - c. Rainbow trout (*Salmo gairdneri*)
2. Warm Water
 - a. Bluegill (*Lepomis macrochirus*)
 - b. Channel catfish (*Ictalurus punctatus*)
 - c. Fathead minnow (*Pimephales promelas*)

Invertebrates

1. Cold Water
 - a. Stoneflies (*Pteronarcys* spp.)
 - b. Crayfish (*Pacifastacus leniusculus*)
 - c. Mayflies (*Baetis* spp. or *Ephemerella* spp.)
2. Warm Water
 - a. Amphipods (*Hyaella* spp., *Gammarus lacustris* or *G. fasciatus* or *G. pseudolimnaeus*)
 - b. Cladocera (*Daphnia magna* or *D. pulex* or *Ceriodaphnia* spp.)

- c. Crayfish (*Orconectes* spp. or *Cambarus* spp. or *Procambarus* spp.)
- d. Mayflies (*Hexagenia limbata* or *H. bilineata*)
- e. Midges (*Chironomus* spp.)

Table 13.2 lists the duration and observable endpoints for several test species in short-duration chronic toxicity tests.

Conducting toxicity tests using three species quarterly for 1 year is recommended by the EPA to adequately assess the variability of toxicity in waste discharges. Analysis of species sensitivity ranges found in the national water quality criteria indicates that if tests are conducted on three particular species (*Daphnia magna*, *Pimephales promelas*, and *Lepomis macrochirus*) the most sensitive of the three will have an LC₅₀ within one order of magnitude of the most sensitive of all species.

TABLE 13.2 Short-Term Chronic Toxicity Methods

Species/Common Name	Test Duration	Test Endpoints
<i>Freshwater species</i>		
Cladoceran <i>Ceriodaphnia dubia</i>	Approximately 7 days (until 60% of control have 3 broods)	Survival, reproduction
Fathead minnow <i>Pimephales promelas</i>	7 days	Larval growth, survival
	9 days	Embryo-larval survival, percent hatch, percent abnormality
Freshwater algae <i>Selenastrum capricornutum</i>	4 days	Growth
<i>Marine/estuarine species</i>		
Sea urchin <i>Arbacia punctulata</i>	1.5 hours	Fertilization
Red macroalgae <i>Champia parvula</i>	7-9 days	Cystocarp production (fertilization)
Mysid <i>Mysidopsis bahia</i>	7 days	Growth, survival, fecundity
Sheepshead minnow <i>Caprinodon variegatus</i>	7 days	Larval growth, survival
	7-9 days	Embryo-larval survival, percent hatch, percent abnormality
Inland silverside <i>Menidia beryllina</i>	7 days	Larval growth, survival

Source: After U.S. EPA (1991).

Functional relationships. The simplest way to find a functional toxicity relationship is by plotting the percent survival versus logarithms of the dose or concentration of the contaminant in the container). Other methods involve fitting the data to a mathematical function.

Figure 13.2 shows a graphical representation of a toxicity bioassay test. Besides the graphical plotting the dose (concentration) relationship has been analyzed using several mathematical models. The most popular method, which also appears to be the most theoretically sound, is the *probabilistic* or *probit method*. A second widely applied method is the *log-logistic model*.

In the *probit method* the variation in the organisms' sensitivity to the toxic compound is represented by a probability density function. In the *log-logistic model* the fraction of the population that responds to a certain dose of the compound is represented by an empirical function that can be linearized and plotted on semilog paper or the parameters of the equation can be determined by a linear regression. Computer

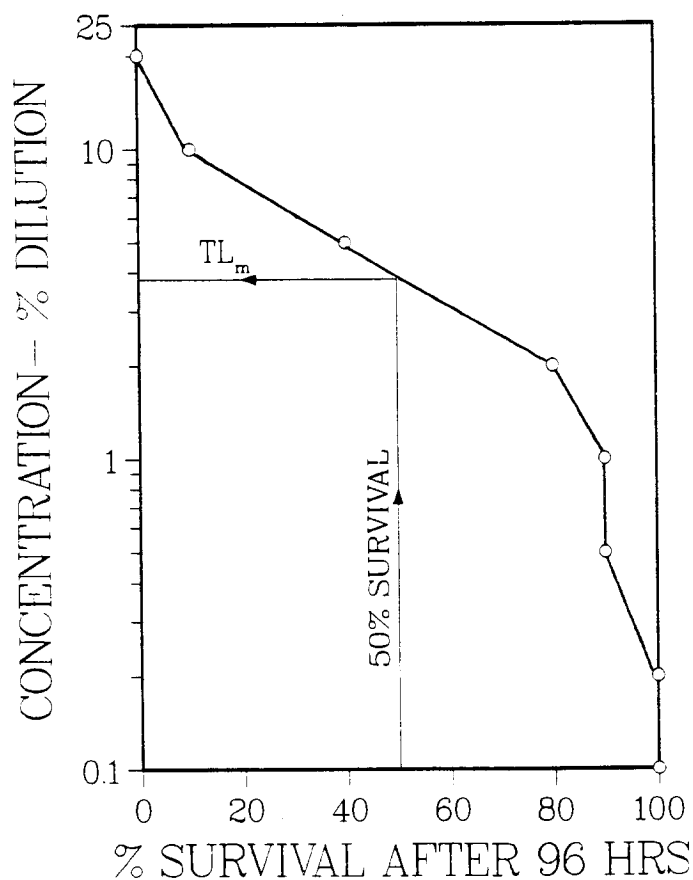


FIGURE 13.2. A functional relationship of toxicity of a compound vs. mortality in a bioassay test.

programs are also available for determining LC_{50} (LD_{50}), which were described by Sprague (1973), Rosiello, Essigmann, and Wogar (1977), and by Altshuler (1981).

Effect of bioavailability. It was shown in Chapter 6 that some components in the soil environment are strongly tied to the soil particles or precipitate as solids. Such compounds include potentially toxic compounds, such as most of organic chemicals, toxic metals, ammonia, and nontoxic phosphates. The same is true in the aquatic environment when solids are present, such as benthic layer or in flows with higher content of suspended solids.

Potentially toxic compounds may affect organisms via two pathways. The first pathway involves adsorption or the bonding of the compound by the organic matter of the organism. This essentially affects the cell tissues inside the body of the organism. For higher organisms the adsorption and bonding differ between the various organs and are higher for lipids (fats) and the liver of fish and other aquatic organisms. Only dissolved and dissociated (ionized) toxic compounds are available for such bonding to cell tissues. The second route is by ingestion of contaminated food or sediments. Contaminated sediments may pass through the digestive system of the organism and become available due to changed chemistry inside the digestive tract. Figure 13.3 outlines the processes involved in the toxic interactions of various forms of metals. It has been observed that in the presence of solids aquatic organisms and plants respond differently to toxic exposure expressed as total concentration. The ingestion route is applicable only for higher trophic level organisms.

Essentially for lower aquatic organisms and for fish the adsorbed fraction is biologically unavailable. Hence the toxicity data for sediments and solids flows may not be correlated to the total concentration of the toxic compound, but the concentration-response curve could be correlated to the dissolved fraction concentration in the water or interstitial (pore) water of sediments. For this reason a new unit has been introduced for high solids media (DiToro et al., 1991), that is

$$\text{Sediment toxic unit (STU)} = \frac{\text{pore water concentration}}{\text{water only } LC_{50}}$$

It is presumed that benthic organisms residing in sediments with different total concentrations of the toxicants, but with the same STU would exhibit the same toxic response. An STU of one would occur when the pore water concentration of the toxic compound equals LC_{50} , even though the total concentration of the toxicant in the sediment may be much higher than LC_{50} .

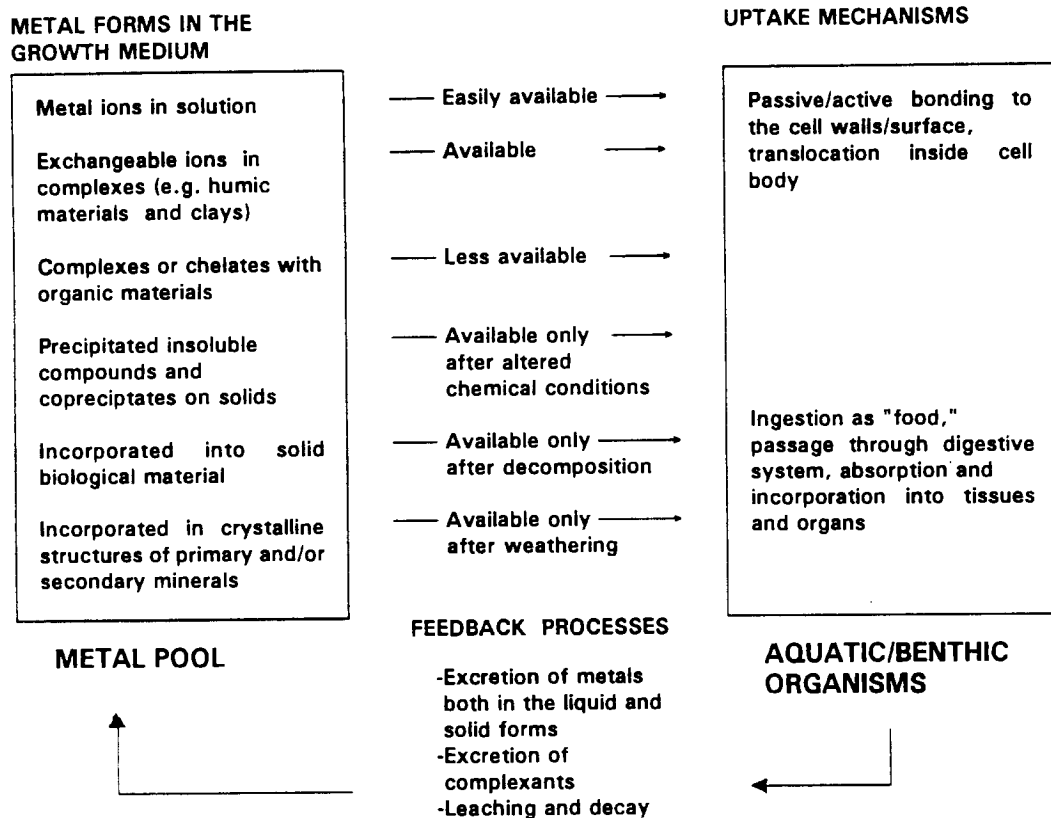


FIGURE 13.3. Biotic uptake processes for toxic metals. (After Salomons and Förstner, 1984.)

As elucidated previously (Chapter 6) the relation between the dissolved (pore) water concentration of an adsorbable compound and the adsorbed (precipitated) fraction of the pollutant can be expressed by an isothermic equation. For most organic toxic chemicals a linear partition isotherm is used (see Chapter 6 for a more detailed discussion of partitioning of toxic chemicals on sediments), or (Eq. (6.6))

$$r = \Pi c_d$$

where

c_d = dissolved (pore water) concentration of the chemical, $\mu\text{g/l}$

r = solid phase (adsorbed) chemical concentration, $\mu\text{g/g}$

Π = partition coefficient, l/g

The total concentration of the pollutant is then the sum of the dissolved (c_d) and particulate concentrations (c_p). Hence, if $c_p = m_{ss} \times r$, where m_{ss} is the concentration of solids in g/l and θ is the water content (θ = porosity for sediments and $\theta = 1$ for water), then (Eq. (6.7))

$$c_T = \theta c_d + c_p = c_d + m_{ss} \times r = c_d(\theta + \Pi m_{ss})$$

The relation between the dissolved (pore water) concentration of the pollutant and the total concentration in the presence of appreciable solids load is then simply (Eq. (6.8))

$$c_d = \frac{1}{\theta + \Pi m_{ss}} c_T$$

The coefficient for nonionic organic chemicals is related to the organic carbon related partition coefficient K_{oc} such as $\Pi \times m_{ss} = K_{oc} \times m_{oc}$, where m_{oc} is the organic carbon concentration of the sediment or suspended solids. On the other hand the magnitudes of the K_{oc} can be related to the octanol partition coefficient K_{ow} , as shown in Chapter 6. The magnitude of the octanol partition coefficient for some chemicals is given in Appendix C.

Example 13.1: Bioavailability of a Toxic Compound

Calculate the bioavailable fraction of a potentially toxic compound whose partition coefficient $\Pi = 101/g$ (related to the total solids concentration) in water ($m_{ss} = 50 \text{ mg/l} = 0.05 \text{ g/l}$ and $\theta = 1$), and in the sediment ($m_{ss} = 150,000 \text{ mg/l} = 150 \text{ g/l}$ and $\theta = 0.9$).

In water

$$\frac{c_d}{c_T} = \frac{1}{1 + 10 \times 0.05} = 0.666$$

or two-thirds of the total concentration is dissolved and may be considered bioavailable and toxic.

In sediment

$$\frac{c_d}{c_T} = \frac{1}{0.9 + 10 \times 150} = 0.00066$$

Less than 0.1% of the compound in the sediment is bioavailable, therefore its toxicity is far less than that in water. On the other hand, sediments are long-term sinks of toxic compounds and the total concentration of the toxicant after prolonged contact with the contaminated overlying water can become much higher than in the water column.

The bioavailability effect explains why some potentially heavily contaminated sites with organic carbon sediments (such as some wetlands) can still support relatively viable biota.

Bioaccumulation and Biomagnification

In the 1960s when the full impact of DDT use was realized, biologists noted a large discrepancy between the DDT levels in organisms of different trophic levels. The higher the trophic level of the organism, the higher the bodily concentration of DDT and some other compounds (Table 13.3). It is interesting to note that the concentration of DDT was $4.14/0.003 = 1380$ times magnified throughout the trophic levels. If water concentrations had been reported in this work, the differences in the bodily concentrations and those of DDT in water would have most likely been several orders of magnitude, that is, the DDT concentrations in water could have been below the detection limit.

At the onset of the discussion it is necessary to define the terms *biomagnification*, *bioaccumulation*, and *bioconcentration*. Both bioaccumulation and bioconcentration are caused by an imbalance in the organism body between the intake of the toxic compound in water and/or food and euration (excretion), resulting in a progressive increase of the bodily content of the toxic compound. The intake of the compound is due to water transfer across the gills, surface sorption, and ingestion of contaminated food. Depuration of the compound from the body is due to desorption, metabolism, excretion, and growth.

Biomagnification means that the concentration of the contaminant is increasing with the trophic level of the organism, as pointed out in Table 13.3, while *bioaccumulation* can be independent of the trophic level. Not all bioaccumulating toxic pollutants will also exhibit biomagnification, although, on the other hand, biomagnification cannot take place without bioaccumulation. The food web (trophical levels) of organisms was

TABLE 13.3 Concentration of DDT in a Marine Environment

Species	Trophic Level	DDT (mg/kg)
Oar weed	1	0.003
Sea urchin	2	0.05
Lobster	3	0.024
Shag liver	4	2.8
Cormorant liver	5	4.14

Source: After Robinson et al. (1967). Reprinted with permission; copyright 1967 by Macmillan Magazines Limited.

presented in the preceding chapter and shown on Figure 12.2. A *bioconcentration factor* is then defined as the ration of the concentration of the toxic pollutant in the bodily tissue of the organism to that in water. A *bioaccumulation factor* is defined as involving contaminant uptake from both water and food. A *biomagnification factor* is a ratio of the bodily concentration of a compound to that in lower trophic level organisms.

Simple mass balance will explain the mechanism in a single aquatic first trophic level organism (such as phytoplanktonic alga) that is being exposed to a compound in water (Thomann, 1991a, 1991b; Thomann and Mueller, 1987)

$$\frac{dwv}{dt} = k_uwc - Kvw \quad (13.1)$$

where

v = body concentration of the chemical, $\mu\text{g/g}$ weight of the organism

k_u = uptake sorption and/or transfer rate, $\text{l/day-g}(\text{weight})$

w = body weight of the organism, $\text{g}(\text{weight})$

c = bioavailable water concentration of the compound, $\mu\text{g/l}$

K = desorption and excretion rate, day^{-1}

t = time in days

The equation, which at this moment ignores intake of the contaminant in food, basically states that if the bodily intake given by $k_u c$ is greater than the bodily loss by depuration expressed by Kvw , then the bodily concentration of the compound will increase. The product $v w$ is the total bodily burden of the compound. To obtain the rate of the bodily concentration, the deferential on the left side of the equation can be expanded as

$$\frac{d(wv)}{dt} = v \frac{dw}{dt} + w \frac{dv}{dt} \quad (13.2)$$

Letting $(dw/dt)/w = G$, where G (day^{-1}) is the net specific growth of the organism, and $K' = K + G$, then Equation (13.1) becomes

$$\frac{dv}{dt} = k_u c - K'v \quad (13.3)$$

At a steady state (no net increase of the whole body mass burden of the contaminant)

$$v = \frac{k_u c}{K'} \quad (13.4)$$

A bioconcentration factor (BCF) or N_w is defined as the ratio of the bodily concentration of the compound to its water (ambient) concentration at a steady state, when the intake of the toxicant is by biosorption only, or

$$N_w = \frac{v}{c} = \frac{k_u}{K'} \quad (13.5)$$

The units for the bioconcentration factor N_w are $(\mu\text{g/g})/(\mu\text{g/l}) = \text{l/g}$.

Thomann (1991a, 1991b) has pointed out that for lower trophic level aquatic organisms the sorption rates are generally much faster than the uptake and excretion rates of higher levels of the food chain. Then an instantaneous equilibrium may be assumed, and from Equation (13.5)

$$v = N_w c \quad (13.6)$$

For species that are above the first trophic level, uptake of toxicant due to ingestion of contaminated food must be considered. This uptake is a function of the contaminant concentration in the food (prey), rate of consumption of the food, and the degree to which the ingested toxicant in the food is actually accumulated by the organism (Connolly and Thomann, 1991b). Adding the food intake into Equation (13.1) yields

$$\frac{dv_2}{dt} = k_u c + \alpha G v_f - K' v_2 \quad (13.7)$$

where

α = the fraction of ingested toxicant that is assimilated

v_f = toxicant concentration in the food (prey), $\mu\text{g/g}$ (weight)

v_2 = bodily concentration of the toxicant

If Equation (13.7) represents the bioaccumulation of a second trophic level organism, then the concentration of the toxicant in the food will be $v_f = N_w c$. Identical equations can be written for higher than second trophic levels. The steady-state solution for a simple food web is then (Thomann, 1991b)

$$v_1 = N_{w1} c \quad (13.8a)$$

and

$$v_j = N_{wj}c + \frac{\alpha_{j,j-1}C_{j,j-1}}{K'_j}v_{j-1} \quad \text{for } j > 1 \quad (13.8b)$$

where

$C_{j,j-1}$ = specific food intake rate of the organism, g(weight) prey per g(weight) predator per day

N_{wj} = bioconcentration factor due to biosorption only, l/g(weight)

In terms of the bioaccumulation factor $N_j = v_j/c$, Equation (13.8) becomes by successive substitutions for v_{j-1}

$$\begin{aligned} N_1 &= N_{w1} \\ N_2 &= N_{w2} + f_{21}N_{w1} \\ N_3 &= N_{w3} + f_{32}N_{w2} + f_{32}f_{21}N_{w1} \\ N_4 &= N_{w4} + f_{43}N_{w3} + f_{43}f_{32}f_{21}N_{w1} \end{aligned} \quad (13.9)$$

(and so forth if higher trophic levels are considered), where

$$f_{j,j-1} = \frac{\alpha_{j,j-1}C_{j,j-1}}{K'_j}$$

The classic growth equation for organisms is

$$G = aC - r \quad (13.10)$$

where

a = food assimilation efficiency

r = respiration rate, day⁻¹

Then

$$C_{j,j-1} = (G_j + r_j)/a_{j,j-1} \quad (13.11)$$

The bioconcentration factor, $BCF = N_{wj}$, is due to biosorption of the toxic compound only, primarily through the gills or through the lipoprotein membranes of organisms. The bioaccumulation factor, $BAF = N_j = v_j/c$, includes both biosorption and intake from the food.

The bioconcentration factor N_w (and the bioaccumulation factor N) has a dimension of liter per gram of bodily weight. Since many toxic chemicals tend to accumulate primarily in the lipid tissue of the organisms, the bioaccumulation factor may be normalized by the weight fraction of the lipid, such as $N'_w[l/g(\text{lipid})] = N_w[l/g(w)] \times [g(w)/g(\text{lipid})]$. For phytoplankton the bioaccumulation coefficient may be normalized by the organic carbon (approximately 50% of dry weight) content. It was then found that the lipid-normalized bioaccumulation factor $BAF = N'_j$ (and also the bioconcentration factor $BAC = N'_{wj}$) may be correlated to the octanol partition coefficient for the chemical, K_{ow} (see Chapter 6 for an explanation of partitioning of organic chemicals between water and organic solids and the Appendix for the magnitudes of K_{ow}). The correlation is different between the trophic-level organisms due to the different effects of growth rate and food ingestion on the accumulation of components with widely varying biosorption capabilities. The relation of N_j to K_{ow} is shown on Figure 13.4. For the first trophic level (phytoplankton), the normalized (organic carbon) bioconcentration factor $N'_1 = N'_{w1} = K_{ow}$ for all nonpolar organic toxic chemicals; for higher trophic levels, the

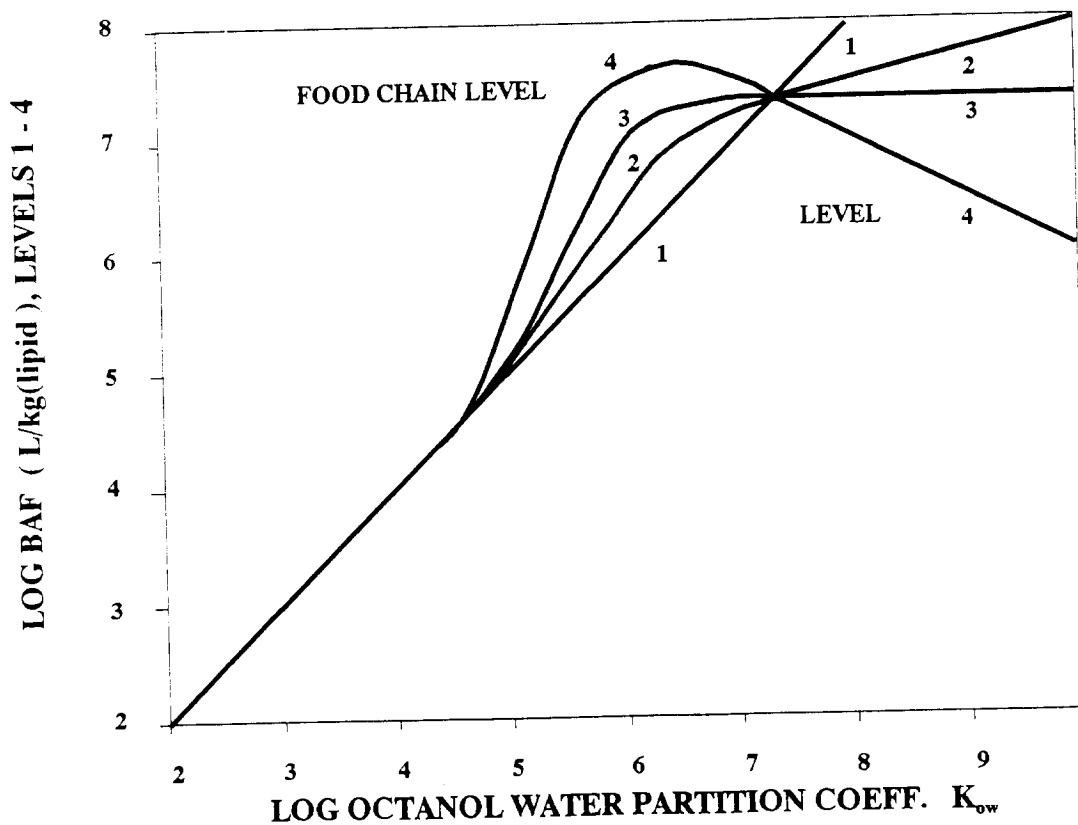


FIGURE 13.4. Relation of the lipid (organic content) normalized bioaccumulation factor, N_j , to the octanol partitioning coefficient, K_{ow} . (After Thomann, 1991a.)

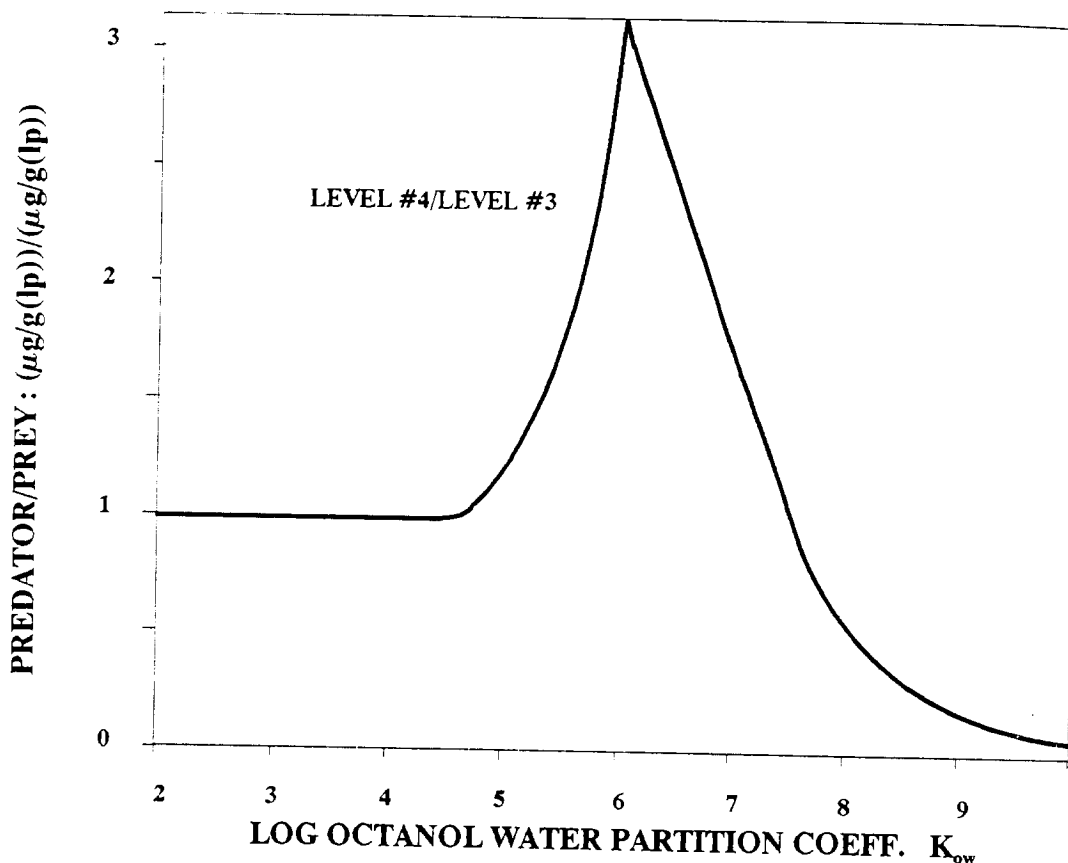


FIGURE 13.5. Predator-prey bodily concentration ration. (After Thomann, 1991a.)

equality $N'_{wj} = K_{ow}$ holds for chemicals with $K_{ow} < 5$. Figure 13.5 shows the predator-prey bodily concentration ratio (biomagnification factor), which can also be related to the biosorption characteristic of the chemical, the octanol partitioning coefficient.

Table 13.4 shows the ratio of BAF to the BCF as a function of the trophic level of an aquatic organism, and the $\log K_{ow}$. The BAF/BCF ratio ranges from 1 to 100, with the highest ratios applying to organisms in higher trophic levels, and to chemicals with $\log K_{ow}$ close to 6.5. For chemicals with $\log K_{ow}$ greater than 7 (l/kg), there is some uncertainty regarding the degree of bioaccumulation.

Trophic level-four organisms are typically the most desirable species for sport fishing, and therefore the multipliers corresponding to this trophic level should be used in setting water quality standards. In rare situations where only lower trophic-level organisms are found (for example, oyster beds) the BAF/BCF factors for lower trophic levels can be used. BAF/BCF factors are very difficult to measure accurately, the process is very expensive, and the results may be problematic and subject to uncertainty (U.S. EPA, 1991).

The steady-state assumption in this concept is crude and neglects the

TABLE 13.4 Estimated Food Chain Multipliers FM = BAF/BCF

log K_{ow}	Trophic Levels		
	2	3	4
3.5	1.0	1.0	1.0
4.0	1.1	1.0	1.0
4.2	1.1	1.1	1.1
4.4	1.2	1.1	1.1
4.6	1.2	1.3	1.3
4.8	1.4	1.5	1.5
5.0	1.6	2.1	2.6
5.1	1.7	2.5	3.2
5.2	1.9	3.0	4.3
5.3	2.2	3.7	5.8
5.4	2.4	4.6	8.0
5.5	2.8	5.9	11
5.6	3.3	7.5	16
5.7	3.9	9.8	23
5.8	4.6	13	33
5.9	5.6	17	47
6.0	6.8	21	67
6.1	8.2	25	75
6.2	10	29	84
6.3	13	34	92
6.4	15	39	98
6.5	19	45	100
>6.5	19.2 ^a	45 ^a	100 ^a

Source: After U.S. EPA (1991).

^aThese values are conservative best estimates.

growth and life span of the organisms. The steady state during the duration of an experiment and/or the life span of organisms can only be reached if the duration of the experiment and life span of the organisms are long enough to reach equilibrium. For some chemicals with very strong biosorption (high K_{ow}) and/or shorter life span, the steady state may not be reached at higher trophic levels. Consequently, a time-variable model should be used.

The state of the art of modeling bioaccumulation in the food chain of aquatic organisms has evolved to the point where reasonably accurate (order of magnitude) simulations are possible. The EPA model WASP includes the food chain bioaccumulation model (at the time of writing this book the food chain model was not yet incorporated into the public domain version of the WASP model). The reader is referred to pertinent EPA publications on WASP and to the publications of the Manhattan

College, Bronx, New York, team that developed the food chain bioaccumulation models (Connolly and Winfield, 1984).

Example 13.2: Biomagnification Calculation

Estimate the PCB concentration in Lake Michigan invertebrates (*Mysis*—trophic level 2), which have a life span of about 1 year. A constant dissolved PCB concentration (note that adsorbed PCBs on sediments is not bioavailable) is 1 ng/l ($= 10^{-3}$ $\mu\text{g/l}$). The following additional information is also given (adapted from Connolly and Thomann, 1991b):

The octanol partitioning coefficient for PCB's: $K_{ow} = 10^{6.5}$ l/kg = 3000 l/g

Fraction of dry weight of phytoplankton: 0.1

Organic carbon/dry weight ratio: 0.5

Fraction of dry weight of invertebrates: 0.2

PCB Concentration in the Phytoplankton ($N_1 = N_w = K_{ow}$)

$$\begin{aligned} v_1 &= N_1 \times c = 3000 \text{ [l/g(g of org. C)]} * 0.001 \text{ [}\mu\text{g/l]} \\ &= 3.0 \text{ }\mu\text{g/g(org. C)} \times 0.5 \text{ [g(org. C)/g(dry weight)]} \\ &\quad \times 0.1 \text{ [g(dry)/g(total weight)]} \\ &= 0.15 \text{ (}\mu\text{g/g)} = 0.15 \text{ mg/kg} \end{aligned}$$

Since 1 liter of phytoplankton (total) biomass weighs approximately 1 kg (weight), the concentration of PCBs in the phytoplankton is 15,000 times greater than that in water.

PCB Concentration in the Invertebrates From Figure 13.4 for $\log K_{ow} = 6.5$ and trophic level 2, $\log N'_2 = 7.0$ or $N_2 = 1.0 \times 10^7$ l/kg = 10^4 l/g (lipid or carbon). By a calculation similar to that for the phytoplankton

$$v_2 = 10,000 \times 0.001 \times 0.5 \times 0.2 = 1.0 \text{ (}\mu\text{g/g)} = 1.0 \text{ mg/kg}$$

Hence the steady-state PCB concentration in *Mysis* is about 10^6 times greater than that of PCBs in water.

WATER QUALITY AND SEDIMENT CRITERIA AND STANDARDS FOR TOXICITY

Water Quality Criteria and Standards

The legal difference between criteria and standards has already been explained in Chapter 1. The criteria from which standards were adopted

by the states have developed primarily from the toxicity bioassay conducted on different organisms and from studies of the effects of toxic compounds on man. The federal water quality criteria adopted in the United States specify maximum exposure concentrations that will provide protection of aquatic life and human health. Generally, however, the water quality criteria describe the quality of water that will support a particular use of the water body. These criteria may be then used by the states as a basis of enforceable water quality standards. The water quality standards apply to all waters of the United States, including most of the natural wetlands. The latest edition of the federal water quality criteria was published in 1987 (U.S. EPA, 1987a), followed by publication of specific criteria in several issues of the *Federal Register*.

The purpose of the following discussion is (1) to provide the rationale behind the federal water quality criteria and extrapolate the logic of how they apply to sediments, (2) provide guidance to the development of site-specific criteria. The water (sediment) quality standards are paramount to integrated pollution abatement, since it is expected that most of the effluent and waste load limitations based on the National Pollution Discharge Elimination System will be related more to the quality of the receiving water and will be more stringent than those based on the required waste elimination technology. Furthermore, the same logic and methodology may be applied to the assessment of the risks of contaminated sediments and water to aquatic biota and man.

The most important uses of surface-water bodies is to provide water for human consumption and contact recreation, and for aquatic life protection and propagation. For the protection of aquatic life a two-value criterion has been established to account for acute and chronic toxicity of waste compounds. The human health criterion specifies the risk incurred with exposure to the toxic compounds at the criterion concentration. The latter (human health) criterion is associated with the increased risk of contracting a debilitating disease, such as cancer.

Aquatic Life Protection Criteria

The water quality standards regulation in the United States allows the individual states to develop numerical criteria of their own or modify the EPA's recommended criteria to account for site specificity or other scientifically defensible factors (U.S. EPA, 1991). The criteria may be based on chemically specific numeric values for the priority pollutants or on the whole effluent toxicity (the term effluent applies to point discharges regardless of whether these are of diffuse or traditional point origin).

Chemical-Based Numerical Criteria

The LC_{50} or EC_{50} acute toxicity values and chronic toxicity observations determined for various key aquatic organisms have been used as a basis for the development of the single-chemical numerical water quality criteria. The development of site-specific toxicity criteria based on a complete toxicity bioassay performed on at least three species of test organisms follows the following procedures (U.S. EPA, 1991; Connolly and Thomann, 1991a).

Acute toxicity criterion or criterion of maximum concentration (CMC). Using acute toxicity data (LC_{50} or EC_{50}) with at least three different families of species (specified by state guidelines) calculate:

1. Geometric mean of all LC_{50} (EC_{50}) values for each species, yielding "species mean value (SMAV)"
2. Geometric mean of all SMAV values within a genus, "genus mean acute value (GMAV)"
3. Assuming that GMAV values are log-normally distributed, determine the 5% GMAV (that is, the concentration exceeded by 95% of the GMAV values) yielding the "final acute value (FAV)"
4. The acute toxicity criterion (CMC) is then $CMC = \alpha \times FAV$, where the α multiplier corrects the FAV values derived from the 50% lethality value LC_{50} rather than from a threshold-lethal (zero mortality) effective concentration (U.S. EPA, 1991). The recommended value for this procedure is $\alpha = 0.5$ (Connolly and Thomann, 1991a).

Chronic toxicity criterion or criterion continuous concentration (CCC). The chronic toxicity criteria also evolved from toxicity tests. Two values are determined in the test as follows: (1) a lower chronic toxicity limit is the highest tested concentration that did not cause an "unacceptable" amount of adverse effects, and (2) an upper chronic toxicity limit is the lowest tested concentration that did not cause an "unacceptable" amount of an adverse effect. The effects are related to the long-term viability of the species and include loss of reproductive capability, reduction in mobility, change in feeding, and reduction of metabolic rates.

To establish CCC the following procedure is outlined:

1. Compute chronic values from chronic toxicity data. A chronic value is the geometric mean of the lower and upper chronic limits. The geometric mean of the two values is thus called the *chronic value*.
2. If sufficient chronic values are available (at least three species are required), then they are analyzed in the same way as outlined in the procedure given earlier for CMC.

3. In the absence of a large enough chronic toxicity data base, ratios of LC_{50} or LD_{50} to available chronic toxicity data (ACR) are then used to establish the chronic toxicity criteria. The ACR is the ratio of acute LC_{50} or EC_{50} to the chronic value. ACR values are needed from at least three families including one fish, one invertebrate, and one acutely sensitive species.
4. A final ACR (FACR) is calculated from species geometric mean ACR values. This procedure depends on whether the ACR is a function of the SMAV defined previously. The minimum allowable ACR is 2.

Bioaccumulation, additivity, antagonism, synergism, and persistence of the chemical may be important in determining the magnitude of the standard.

Traditional "point source" application of receiving water standards was related to extremely rare low flows, such as the 10 years expectancy-7 days exceedence low flow (Q_{7-10}). Such low-flow concepts are of little use in storm-water management or in water-quality-based controls of discharges of priority pollutants. Statements such as "the standard should not be exceeded at all times" are also inappropriate, since concentrations represent statistical time series for which only infinitesimally large values would have a 100% statistical probability of not being exceeded. Thus a standard (or a criterion) for a harmful substance must have three components (U.S. EPA, 1991):

- Magnitude: The amount of a pollutant (or pollutant parameter, such as toxicity), expressed as concentration, is allowable.
- Duration: The period of time (averaging period) over which the in-stream concentration is averaged for comparison with criteria concentrations. The specification limits the duration of concentration above the criteria.
- Frequency: How often criteria can be exceeded.

The permissible frequency of exceedence of toxicity criteria based on the July 29, 1985, issue of the *Federal Register* is:

- *Acute toxicity criteria*: One hour average concentration (essentially a grab sample), not to be exceeded more than once in three years on an average
- *Chronic toxicity criteria*: Four day average concentration, not to be exceeded more than once in three years on the average.

Since most of the water quality constituent concentrations from a sufficiently long record follow a log-normal distribution, the acute toxicity criterion (standard) would be violated if the 99.9 percentile of *maximum*

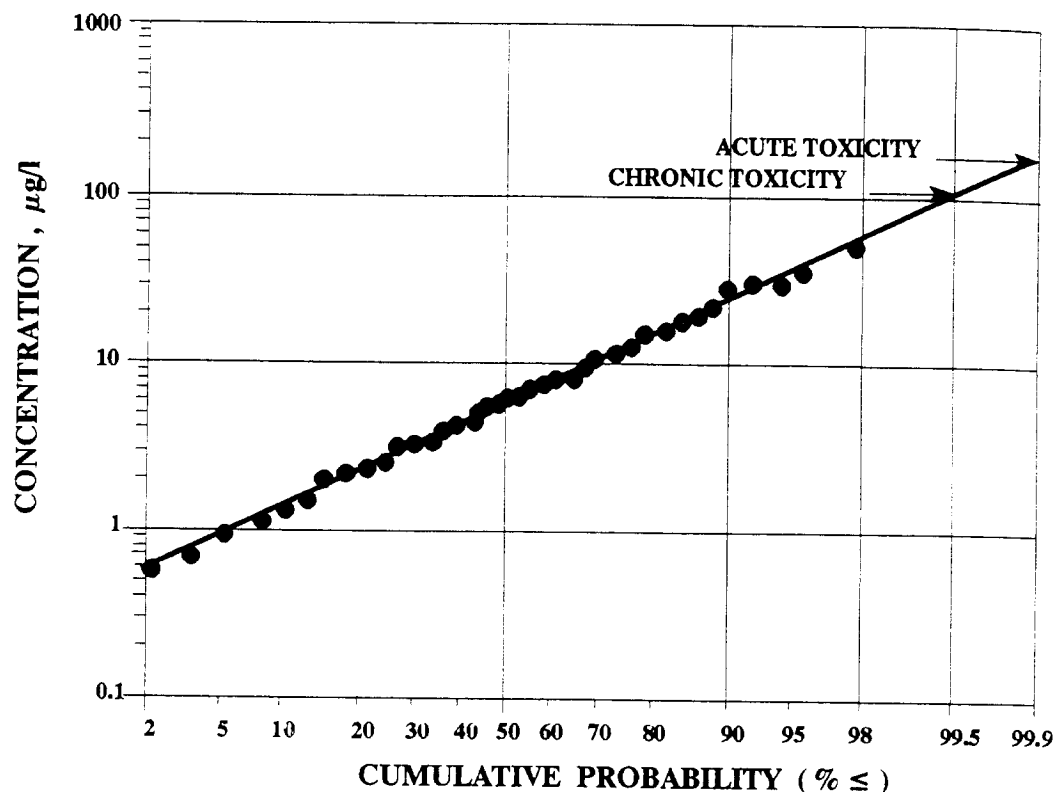


FIGURE 13.6. Probability concept of criteria for acute and chronic toxicity.

daily concentrations, arranged in ascending order of magnitude on the log-cumulative probability plot, exceeded the criterion (Fig. 13.6). Similarly, the chronic toxicity criterion would correspond to the 99.6 percentile log-cumulative probability characteristic of *average daily concentrations*. A one hour average value for acute toxicity would imply grab samples taken randomly. For chronic toxicity, composite samples (over a 24 hour period) are more appropriate. For planning studies and modeling testing for chronic toxicity violations, both in water and sediments, resulting from known and anticipated point and nonpoint discharges will suffice. The three year recurrence was derived from observations of the length of recovery of ecosystems after a toxic spill.

Appendix B presents the water quality criteria for recognized pollutants and other water quality parameters adapted as standards in the United States. It should be pointed out that the values in Appendix B were derived from toxicity bioassays performed in laboratory conditions, with organisms residing mostly in water with no or minimum sediment. Thus the criteria would logically correspond to bioavailable concentrations of the toxicant. Legal interpretation of the toxicity standards are not

clear as to whether total (adsorbed + dissolved-bioavailable) or bioavailable fractions should be considered.

Whole Effluent Toxicity Criteria

Whole effluent toxicity is the total toxic effect of an effluent measured directly by a toxicity test. In such measurements the toxicity is expressed in toxicity units, both chronic and acute.

The *acute toxic unit* (TU_a) is the reciprocal of the effluent concentration in the dilution water (in percent) that causes 50% of the organisms to die by the end of the acute exposure period, or

$$TU_a = 100/LC_{50}$$

The *chronic toxic unit* (TU_c) is the reciprocal of the effluent concentration (in percent in dilution water) that causes no observable effects on the test organisms by the end of the chronic exposure period, or

$$TU_c = 100/NOEC$$

For example, a waste discharge with an acute toxicity of an LC_{50} in 5% effluent (5% wastewater in 95% dilution) is discharge containing 20 TU_a . The toxic units enable specifying water quality criteria based upon the whole toxicity. For comparative purposes, a waste discharge that contains 20 TU_c is twice as toxic as a discharge containing 10 TU_c .

The whole effluent (body of water) test is conducted as delineated before. At least three different species are required for the test. To protect aquatic biota the EPA recommends that the criterion maximum concentration (CMC) be set to 0.3 TU_a for the most sensitive of at least three test species. The selection of the test species is not critical (see the list on page 825 and Table 13.2 for selection) provided that species from ecologically diverse taxa are used (i.e., a fish, an invertebrate, and a plant). The factor of 0.3 is again used to adjust the typical LC_{50} (50% mortality), LC_0 (virtually no mortality). For chronic protection, the CCC should be set at 1.0 toxic units (TU_c).

Human Health Protection Criteria

Human health criteria are based on the average daily dosage of a potential toxicant. Man can receive toxic dosages originating in surface (ground) water via two pathways: (1) by drinking contaminated water, and (2) by eating contaminated fish and shellfish. All criteria are chemical specific and contain only a single expression of allowable magnitude: a criterion

concentration that protects humans against long-term chronic health effects during an average lifetime period, which was set as 70 years.

The daily intake of a toxicant is

$$I = \phi_f v + \omega c \quad (13.12)$$

where

I = daily intake of the toxic compound by man, mg/day

ϕ_f = average daily consumption of fish, kg

v = fish chemical concentration, mg/kg

ω = average daily water intake, l

c = the water concentration, mg/l

The EPA-recommended values of daily average fish consumption are given below (compiled by U.S. EPA, 1991):

<i>Intake</i>	<i>Explanation</i>
6.5 g/day	Represents an estimate of average consumption of fish and shellfish from estuarine and freshwater by the entire U.S. population. This is an average of both consumers and nonconsumers of fish (most states).
20 g/day	Represents an estimate of the average consumption of fish and shellfish from marine, estuarine, and freshwater by the U.S. population. This level also includes consumers and nonconsumers of fish (Arizona, Illinois, Louisiana, and Wisconsin).
165 g/day	Represents consumption of fish and shellfish by the 99.9 percentile of the U.S. population consuming the most fish and seafood.
180 g/day	This is a reasonable "worst case" based on the assumption that some individuals would consume fish at a rate equal to the combined consumption of red meat.

Using a bioaccumulation factor N as

$$N_{\text{lipid}} = \frac{v}{c}$$

The bioaccumulation factor is normalized by the lipid content of the fish, hence the unit of N_{lipid} is liters/kilogram of lipid. Then the daily intake is

$$I = [\phi_f N + \omega]c \quad (13.13)$$

where $N = N_{\text{lipid}} \times (\% \text{ lipid in fish})/100$.

At a low dose level the incremental risk (for example, of contracting cancer) is proportional to the dose of the toxic compound, or

$$R = qI \quad (13.14)$$

where

R = incremental risk over the background (no contamination) risk of contracting a disease

q = risk-dosage coefficient obtained from toxicity studies

Then for an average person weighing W kg, the maximum daily intake of the toxicant is

$$I_{\text{max}} = \frac{W * R}{q} \quad (13.15)$$

Substituting Equation (13.15) into Equation (13.13) the maximum water concentration (criterion) that will provide human health protection at risk R becomes

$$C_{\text{max}} = \frac{WR}{q(\phi_f N + \omega)} \quad (13.16)$$

There has been considerable controversy among environmentalists and scientists over the acceptable risk level for human health protection. By accepting a very small risk factor $R = 10^{-6}$, one still de facto accepts the fact that probably one person in a million will get a cancer or a debilitating disease because of water contamination. Again those not familiar with the statistical characteristics of natural time series may advocate no risk, which would imply an absolute never exceeding zero concentration (an impossible task). For a comparison the following activities pose about the same risk of death or debilitating injury to man (1 death increase in a population of one million): riding 16 km on a bike or driving 500 km in a car or flying 1600 km by a commercial jet plane or travelling 6 minutes by a canoe or drinking one-half liter of wine (J. P. Connolly, 1991, pers. comm.).

Example 13.3: Toxicity Criterion Estimate for PCBs

Estimate a water quality criterion for PCBs that would provide human health protection at an acceptable risk level of $R = 10^{-6}$ (an increase of

one cancer per one million population over the background level). The following information has been obtained:

Bioaccumulation factor for PCBs: $N_{\text{lipid}} = 3 \times 10^6 \text{ l/kg (lipid)}$

Average body weight of a person: $W = 70 \text{ kg}$

Average daily fish consumption: $\phi_f = 20 \text{ g} = 0.02 \text{ kg}$

Average lipid content of fish: 5%

Average daily water intake: $\omega = 2 \text{ l}$

Risk-dose coefficient of $q = 3 \times 10^{-4}$

Solution Calculate the biomagnification factor

$$N = N_{\text{lipid}} \times (\% \text{ lipid})/100 = 3 \times 10^6 \times 3/100 = 9 \times 10^4 \text{ l/kg}$$

Then from Equation (13.6)

$$c_{\text{max}} = \frac{70 \times 0.000001}{0.0003 \times (0.02 \times 9 \times 10^4 + 2)} = 1.3 \times 10^{-4} \text{ mg/l}$$

Example 13.4: Water Quality Standard Based on Maximal Body Intake

From epidemiologic studies it was found that a daily intake of a carcinogenic compound has the following impact on the test human population:

Daily Dosage (mg)	Number of Additional Cancers/Population
10	5/100,000 (5×10^{-5})
50	2/10,000 (2×10^{-4})
100	5/10,000 (5×10^{-4})

The compound has $\log K_{ow} = 6$.

Estimate the water quality criterion for a compound that would protect the population at the risk level of 1 additional cancer per 1,000,000.

Solution Assume

Bioaccumulation factor for carnivorous fish (trophic level 4) from Figure 13.3 for

$$K_{ow} = 10^6 \quad N_{\text{lipid}} = 3 \times 10^6 \text{ l/kg,}$$

Average body weight of a person: $W = 70 \text{ kg}$

Average daily fish consumption: $\phi_f = 20 \text{ g} = 0.02 \text{ kg}$

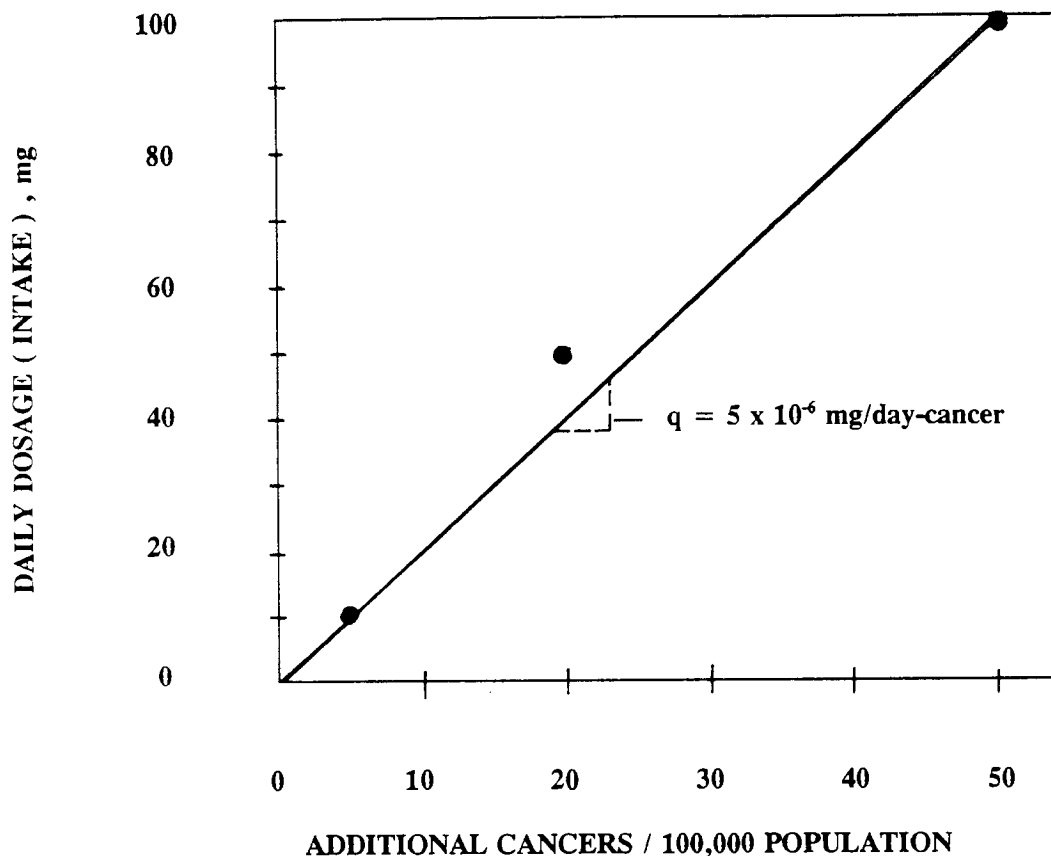


FIGURE 13.7. A plot of cancer deaths increases vs. bodily toxic compound daily dosage above the ambient dose for Example 13.4.

Average lipid concentration of fish: 4%
 Average daily water intake: $\omega = 21$

By plotting the dosage–cancer increase numbers in Figure 13.7 the risk–dose coefficient is approximately $q = 5 \times 10^{-6}$ mg/day. The corrected bioaccumulation factor is $N = N_{\text{lipid}} \times (\text{lipid content of fish}) = 3 \times 10^6 \times 4/100 = 1.2 \times 10^5$.

Repeating the calculation similarly to the previous example for $R = 10^{-6}$, the human health protection criterion is

$$c_{\max} = \frac{70 \cdot 0.000001}{5 \times 10^{-6} (0.02 \times 1.2 \times 10^5 + 2)} = 0.0113 \text{ mg/l}$$

It may be interesting to note that for these cases most of the compound intake and risk of cancer are due to eating contaminated fish.

The U.S. Environmental Protection Agency recommends using the most current risk information when updating or generating criteria. For this purpose the agency has developed the Integrated Risk Information System (IRIS), which is an electronic on-line data base that provides chemical exposure and estimated human health effects (U.S. EPA, 1987b). The right side of Table B.3 in Appendix B contains the human health protection water quality criteria.

Sediment Water Quality Criteria

The role of sediments and their impact on toxicity of many priority pollutants must be considered. It has already been pointed out that suspended solids and their organic, and to a lesser degree clay fractions, have an affinity to adsorb constituents. Adsorbed and/or precipitated pollutants may not be bioavailable, especially to lower trophic-level organisms of the food web. Example 13.1 has shown that the concentrations of a toxic compounds in the pore water of sediments may be much smaller than those measured in the overlying water. This is due to partitioning between dissolved (bioavailable) and adsorbed-precipitated (nonbioavailable) fractions of the compound.

The importance of the bioavailability and partitioning phenomena in sediments can be overlooked. This will impact directly on the LC_{50} of organisms residing in the sediment. The observations have shown that the concentration-response curve for the biological effect of a constituent could not be correlated to the total concentration of the chemical in the sediment, but to a certain parameter that reflects on the bioavailability of the compound in the sediment. It was shown that this parameter was the dissolved pore water concentration of the compound (DiToro et al., 1991). This parameter is then expressed by the sediment toxic unit (STU). The response curves of the toxicity test and the LC_{50} for various compounds and organisms correlated surprisingly well to the STUs of the sediments, as shown in Figure 13.8. Furthermore, it was shown that the benthic organisms have about the same sensitivity to toxic compounds as the water column organisms. This apparent equality between the sensitivities of benthic and aquatic water column organisms enables one to use water toxicity criteria to also protect the organisms residing in the benthos (DiToro et al., 1991).

Nonionic organic toxic chemicals. Since the adsorptivity of the non-ionic organic chemicals is related to the organic carbon particulate fraction of the sediment, the sediment organic carbon is the primary route of exposure of the organisms. Recall that the relation between the sediment chemical concentration (microgram of chemical/gram of sediment) and the pore water concentration (in micrograms/liter) is $r = \Pi c_d$, also $\Pi \approx$

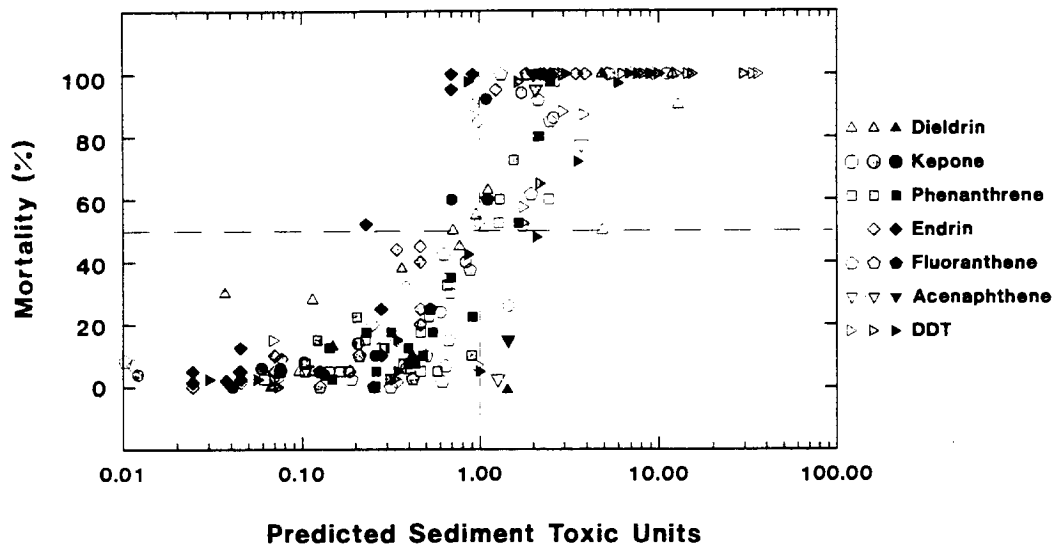


FIGURE 13.8. Correlation of the mortality of aquatic benthic organisms vs. calculated sediment toxic units. (After DiToro et al., 1991a, reprinted with permission, copyright 1991, SETAC.)

$(m_{oc}/m_{ss}) \times K_{ow}$. Then substituting the sediment quality criterion (SQC) for r and the water quality criterion (WQC) for c_d , the relationship between the sediment toxicity and water quality criteria is the same simple partition equation (DiToro et al., 1991)

$$SQC = \Pi WQC \approx f_{oc} K_{oc} WQC \tag{13.17}$$

where $f_{oc} = m_{oc}/m_{ss}$ is the fraction of organic carbon in the sediment.

DiToro et al. (1991) proposed the use of chronic water toxicity criteria for WQC in order to protect benthic biota, thus implying that compliance with the chronic toxicity criterion will also satisfy the acute toxicity criterion. This concept was accepted by the U.S. EPA. The estimated sediment toxicity unit is then approximately

$$STU = \frac{c_s(f_{oc} \times K_{oc})}{\text{water only WQC}} \tag{13.18}$$

where c_s is the concentration of the chemical in the sediment expressed in micrograms per gram of dry weight of the sediment. The limit of applicability of the preceding relationships is the organic carbon fraction, $f_{oc} > 0.1\%$. Below $f_{oc} = 0.1\%$ other factors that influence partitioning, such as particle size, pH, and the electrical charge of the minerals in the sediment, influence the equilibria between the sorbed and dissolved fractions of the chemicals.

Observations also indicated that in addition to the particulate carbon in the sediments, chemicals may be made nonbioavailable by dissolved organic carbon, such as that contained in humic compounds (McCarty and Jimenez, 1985; Landrum et al., 1985, 1987; DiToro et al., 1991). The presence of dissolved organic carbon in the pore water may distort the partitioning between the adsorbed and dissolved toxic compounds, and the apparent partitioning coefficient $K'_p = c_s/c_d$ may not agree with the expected partitioning coefficient $f_{oc} \times K_{oc}$ (DiToro et al., 1991). The differences may amount to an order of magnitude. However, such differences between observed and estimated data are common and should be expected in modeling toxicity relationships. The reliability of the estimates then coincides with the risk in the formulation of the toxicity limits.

Example 13.5: Relation between Sediment and Water Column Toxicities

Estimate maximal acceptable concentration of lindane in an organic sediment. The following information is given:

Octanol partition coefficient of lindane (Appendix C): $K_{ow} = 10^{3.7} = 50001/\text{kg}$

Organic content of the sediment: $f_{oc} = 0.1$ (10%)

Chronic toxicity criterion (Appendix B): $\text{WQC} = 0.08 \mu\text{g/l}$

Solution From Equation (6.20a) the carbon partition coefficient $K_{oc} = 0.63 \times 5000 = 31501/\text{kg} = 3.151/\text{g}$. The sediment toxicity criterion is then

$$\text{SQC} \approx f_{oc} K_{oc} \text{WQC} = 0.10 \times 3.15 \times 0.08 = 0.025 \mu\text{g/g of dry weight}$$

Example 13.6: Sediment Toxicity Estimation

Chemical analysis by gas chromatograph–mass spectrometer apparatus yielded the sediment concentration of methoxychlor as $0.2 \mu\text{g/g}$ (= mg/kg) of sediment. The sediment organic content is 10% on a dry weight basis. Make a judgment as to whether the sediment contamination is toxic to the benthic biota. Given are:

Methoxychlor sediment concentration: $c_s = 0.2 \mu\text{g/g}$ of methoxychlor

Octanol partitioning coefficient for methoxychlor (Appendix C): $K_{ow} = 10^{4.78} = 60,0001/\text{kg}$

Chronic water quality criterion (Appendix B): $\text{WQC} = 0.03 \mu\text{g/l}$

Sediment organic carbon fraction: $f_{oc} = 0.1$

$$K_{oc} = 0.45 \times 60,000^{0.989} = 23,9221/\text{kg} = 23.921/\text{g}$$

The organic carbon partitioning coefficient is

$$K_{oc} = 0.63 \times K_{ow} = 60,0001/\text{kg} = 601/\text{g}$$

The toxicity in sediment toxicity units (STU) is (Eq. (13.18))

$$\text{STU} = \frac{c_s/f_{oc} \times K_{oc}}{\text{WQC}} = \frac{0.2/(0.1 \times 60)}{0.03} = 1.1$$

Since $\text{STU} \approx 1.0$ one could judge the sediment as mildly toxic (based on Fig. 13.6) to the benthic biota.

Example 13.8: Volumetric Concentration Toxicity Limit for Sediments

Using the information from the previous Example, estimate the toxicity limit for lindane in the sediment in micrograms/liter of wet sediment. The following information is given:

Porosity: $p = 50\%$

Volatile (organic) fraction of the sediment: $VC = 10\%$

Specific gravity of organic fraction: 1.1

Specific gravity of mineral fraction: 2.4

Estimated chronic SQC for lindane (Example 13.5): $0.025 \mu\text{g/g}$ of dry weight

$$\begin{aligned} \text{Volume (ml/g)} &= (\text{dry weight in grams}) \\ &\times [(1 - VC)/2.4 + VC/1.1]/(1 - \text{porosity}) \\ &= 1 \text{ g} \times [(1 - 0.1)/2.4 + 0.1/1.1]/(1 - 0.5) = 0.93 \text{ ml/g} \end{aligned}$$

Hence

$$\begin{aligned} \text{SQC}(\mu\text{g/ml} = \text{mg/l}) &= 0.025(\mu\text{g/g})/0.93(\text{ml/g}) \\ &= 0.027 \mu\text{g/ml} \\ &[= \text{mg/l of sediment}] \end{aligned}$$

Note that the value of SQC just given is about 1000 times greater than the corresponding water column criterion from which it was derived.

MODELING THE FATE OF TOXIC COMPOUNDS IN RECEIVING WATER BODIES

Basic Equations and Fundamentals

The fundamentals of modeling the fate of toxic compounds and sediment-water interactions were explained in Chapter 6. The water quality modeling concepts were presented in the preceding chapter. The mass balance continuity equation for toxic substances was introduced by O'Connor (1988) in the preceding chapter as

$$V \frac{dc_i}{dt} = J_i + \Sigma R_i + \Sigma T_i + \Sigma W \quad (13.19)$$

in which

c_i = concentration of the chemical in compartment, i

J = transport through the system

R = reactions within the system

T = transfer from one phase to another

W = inputs

A chemical in water can exist either dissolved or as a particulate (adsorbed). As pointed out previously, only a dissolved fraction is considered bioavailable, and hence, toxic. Expanding Equation (13.19) for dissolved and adsorbed fractions, substituting appropriate reaction formulas for transformation and decomposition, and dividing both sides of the equation by V , yields (O'Connor, 1988)

$$\frac{dc_d}{dt} = \frac{W_d}{V} - \frac{c_d}{t_0} - (K_1 + K_c + K_a)c_d + K_2c_p \quad (13.20a)$$

$$\frac{dc_p}{dt} = \frac{W_p}{V} - \frac{c_p}{t_0} + K_1c_d - (K_2 + K_p + K_s)c_p \quad (13.20b)$$

where

C_d = dissolved fraction concentration of the compound

c_p = adsorbed fraction concentration

V = volume of the computational segment (water column or sediment)

t_0 = detention time within the segment = V/Q

Q = flow through the segment

K_1 = adsorption rate coefficient

K_2	= desorption rate coefficient
K_c	= decay rate coefficient of the dissolved fraction of the compound
K_a	= transfer (volatilization) rate coefficient of the dissolved fraction
K_p	= decay coefficient of the particulate (adsorbed) fraction
$K_s = v_s/H$	= sediment transfer (settling or resuspension) rate coefficient
W_c, W_p	= mass input of the dissolved and adsorbed compound fractions
v_s	= settling (resuspension) velocity of the particulates

In addition to the water continuity equation (Eq. (13.24)), the sediment mass-balance continuity equation must also be considered, or

$$\frac{dm}{dt} = \frac{m_i}{t_0} - m \left[\frac{1}{t_0} + K_s \right] \quad (13.21)$$

where

m	= concentration of suspended solids in the segment
$m_i = W_s/Q$	= average concentration of suspended solids in the input
W_s	= mass input of suspended solids into the segment

The process of sedimentation and resuspension of cohesive sediments was explained in Chapter 5.

To simplify Equations (13.20) the following assumptions were made:

1. An instantaneous adsorption-desorption is reached. From Equation (6.6) $c_d = \Pi c_p/m$ (because c_p and m are typically expressed in milligrams or micrograms per liter, r in micrograms/gram, respectively). This eliminates the adsorption-desorption rate coefficients K_1 and K_2 .
2. From Equations (6.7) and (6.8) the dissolved and particulate fractions of the compound related to the total concentration, c_T , are

$$f_d = \frac{c_d}{c_T} = \frac{1}{\theta + \Pi m}$$

and

$$f_p = \frac{c_p}{c_T} = \frac{\Pi m}{\theta + \Pi m}$$

where θ = dimensionless water content (θ = porosity for sediments, for water $\theta = 1$). Note that for bottom sediments, $\Pi m \gg \theta$, hence, $f_{pb} \approx 1$ and $f_{ab} \approx 1/(\Pi_b m_b)$, where the subscript b denotes bottom conditions.

Adding Equations (13.20a) and (13.20b) and considering the preceding simplifications results in

$$\frac{dc_T}{dt} = \frac{W_T}{V} - \frac{c_T}{t_0} - f_d(t)[K_c + K_a]c_T - f_p(t)[K_p + K_s]c_T \quad (13.22)$$

The magnitudes of the constants and rate coefficients for modeling the fate of toxic compounds were compiled by Schnoor et al. (1987). The processes involved in modeling the fate of toxic compounds when solids are present were explained in Chapter 6.

The toxic chemical model. The toxic chemical submodel TOXI4 included in the WASP model (Ambrose et al., 1991) is a dynamic compartmental model of the transport and fate of toxic chemicals. Its concepts and components were introduced in Chapter 12. This model considers several physical-chemical processes, and can handle up to three chemicals and up to three types of particulate matter per each simulation. The chemical can be independent or linked by reaction yields (for example, the DDT to DDE transformation). Each chemical can be considered in up to five forms, including the natural form and ionic species, and each of them can exist in five phases (dissolved, sorbed to organic carbon, and sorbed to three species of solids). The equilibrium between the solids adsorbed and dissolved phases is described by the partition equilibrium equation (Eq. (6.6)).

The transfer and transformation reactions and processes that the toxic chemicals can undergo in aquatic environments include sorption, volatilization, ionization, photolysis, hydrolysis, biodegradation, and chemical oxidation. WASP (TOXI4) describes sorption and ionization as equilibrium reactions, while the remaining reactions are described as concentration change rate equations.

TOXI4 can be used to simulate toxic metals and organic chemicals. However, it should be pointed out that metal modeling requires site-specific judicious selection of certain key parameters (Ambrose et al., 1991). As with EUTRO4, TOXI4 can be implemented at different levels of complexity that take into consideration increasing sophistication and detail in the description of solids behavior, and kinetic and equilibrium reactions.

Modeling Strategies for the Fate of Toxic Chemicals in Aquatic Systems

Typically, the waste-assimilative capacity of a receiving water body for organic pollutants discharged from point sources is determined using calibrated and verified steady-state models (or dynamic models executed as steady state) applied to an extreme low-flow condition, such as the 7-day duration-10-year recurrence interval flow characteristic (so-called Q_{7-10}).

However, the formulation of toxic standards does not allow a simple application of steady-state models. As pointed out in the section titled "Water Quality and Sediment Criteria and Standards for Toxicity" of this chapter, toxic criteria are related to the probability (frequency) of the exceedance of the concentrations, which for acute toxicity allows a one-hour average concentration of a priority pollutant to exceed the criterion once in 3 years, while for chronic toxicity the 4-day average concentration can exceed the criterion once in 3 years (on average). Hence the modeling requires some statistical considerations.

The EPA (1991) guidance manual states that the hydrologically based low-flow, 7-day-exceedance-10-year-recurrence-interval-flow Q_{7-10} is similar to the biologically based 4-day exceedance-3-year-recurrence flow. Therefore in steady-state studies of a traditional point source discharge, Q_{7-10} may be used for chronic toxicity impact calculations using the CCC criterion. Using the Q_{7-10} for acute, CMC estimations would lead to an excessive number of water quality criteria exceedances. Using hydrologically based low-flow characteristics for storm-water-based discharges is inappropriate and makes no sense.

The toxic concentrations in runoff leaving a source area (the event mean concentrations (EMCs)) or in a point discharge are statistically distributed. The statistical characteristics of EMCs in urban runoff were presented in Chapter 8. It was shown that EMCs are log-normally distributed with a coefficient of variation (CV) ($= \log \text{ standard deviation} / \log \text{ mean}$) of around 1. There is no significant correlation between the magnitude of runoff and other parameters, and EMC magnitude for most toxic parameters.

In lieu of a single worst case steady-state modeling three dynamic techniques were recommended by the U.S. Environmental Protection Agency (U.S. EPA, 1991; Ambrose et al., 1988): (a) Monte Carlo simulation, (b) log-normal probability modeling, and (c) dynamic continuous modeling.

The in-stream (lake, estuary) response of the concentration of a toxic

compound can be schematically represented by an input–output transformation concept, or

$$c = \mathbf{F}(\mathbf{p})\mathbf{L}_i \quad (13.23)$$

where

- c = in-stream concentration response to a toxic input
 \mathbf{L} = toxic load vector from point and nonpoint sources (including sediment)
 $\mathbf{F}(\mathbf{p})$ = transfer function or a model for the receiving water system
 \mathbf{p} = parameters–coefficients of the model
 i = source subscript

The form of transfer function may range from a simple dilution formula for a soluble conservative compound to a complex dynamic model such as WASP. In this sense, Equation (13.23) is a schematic representation and does not imply a simple multiplication of \mathbf{L} by $\mathbf{F}(\mathbf{p})$. Both \mathbf{L} (load) and \mathbf{F} (transfer function) can be considered to be deterministic or stochastic (probabilistic).

In the *Monte Carlo methodology* (Marr and Canale, 1988) a simplified steady-state toxic compound–sediment fate model may be used. The inputs (loads) and system parameters of the model are statistically analyzed and their characteristics (mean, coefficient, and variation and probability distribution) are obtained.

The *log-normal probability modeling* (DiToro, 1979; Mancini, 1983) calculates the mean and standard deviation of in-stream concentration

TABLE 13.5 Toxicants Fate and Transport Models

Model	Environment	Time Domain	Spatial Domain	Chemical
DYNTOX	River	Dynamic	Far-field 1-dimensional	Organic metal
EXAMS-II	Lake, river, estuary	Steady-state quasi-dynamic	Far-field 3-dimensional	Organic
WASP4	Lake, river, estuary	Steady-state dynamic	Far-field 3-dimensional	Organic metal
HSPF	River	Dynamic	Far-field 1-dimensional	Organic metal
SARAH-2	River	Steady-state	Treatment plant, near field	Organic
MINTEQA2	Lake, river, estuary	Steady-state	—	Metal

Source: After U.S. EPA (1991).

response of a toxic compound from the mean and standard deviation of the inputs (runoff volume and EMC) and stream flows assuming log-normal distribution.

Table 13.5 presents a summary of available models for modeling the fate and transport of toxic chemicals. These models are mostly public domain models, available from the U.S. EPA laboratory in Athens, Georgia. User's training seminars are periodically organized by the EPA. Most water quality models were developed with an emphasis on the dynamics in the water column and water column concentrations. Two models listed in Table 13.5 (EXAMS-II and WASP4) are capable of simulating water-column-sediment interactions, including resuspension, settling, and diffusion. However, additional work needs to be completed on the mechanisms of sediment-water-column exchange before the models can be validated for predictive applications involving sediments (U.S. EPA, 1991).

References

- Altshuler, B. (1981). Modeling of dose-response relationships, *Environ. Health Perspect.* **42**:23-27.
- Ambrose, R. B., et al. (1988). Toxic wastes: Waste allocation simulation models, *J. WPCF* **60**(9):1646-1655.
- Ambrose, R. B., T. A. Wood, S. L. Martin, J. P. Connolly, and R. W. Schanz (1991). *WASP5.x, a Hydrodynamic and Water Quality Model—Model Theory, User's Manual, and Programmer's Guide*, U.S. Environmental Protection Agency, Athens, GA.
- APHA (1989). *Standard Methods for Examination of Water and Wastewater*, 17th ed., American Public Health Association, Washington, DC.
- Chapman, G. (1978). Toxicological considerations of heavy metals in the aquatic environment, in *Toxic Materials in the Aquatic Environment*, Oregon State University, Water Resources Institute, Corvallis, OR.
- Connolly, J. P., and R. V. Thomann (1991a). Toxicity, in *Contaminated Sediments: Models for Criteria, Exposure and Remediation*, Continuing Education Seminar notes, Manhattan College, Bronx, NY.
- Connolly, J. P., and R. V. Thomann (1991b). Food chain, in *Contaminated Sediments: Models for Criteria, Exposure and Remediation*, Continuing Education Seminar notes, Manhattan College, Bronx, NY.
- Connolly, J. P., and R. P. Winfield (1984). *WASTOX, a Framework for Modeling the Fate of Toxic Chemicals in Aquatic Environments. Part 2: Food Chain*, Environmental Engineering Program, Manhattan College, Bronx, NY.
- DiToro, D. M. (1979). *Statistics of Receiving Water Response to Runoff*, paper presented at National Conference on Urban Stormwater and Combined Sewer Overflow Impact on Receiving Water Bodies, University of Central Florida, Orlando, FL.
- DiToro, D. M., et al. (1991). Technical basis for establishing sediment quality

- criteria for nonionic organic chemicals using equilibrium partitioning, *Environ. Toxicol. Chem.* **10**(12):1541-1583.
- Krenkel, P. A., and V. Novotny (1980). *Water Quality Management*, Academic Press, New York.
- Landrum, P. F., S. R. Nihart, B. J. Eadie, and L. R. Herche (1987). Reduction in bioavailability of organic contaminants to the amphipod *Pontoporeia hoyi* by dissolved organic matter of sediment interstitial waters, *Environ. Toxicol. Chem.* **6**:11-20.
- Landrum, P. F., M. D. Reinhold, S. R. Nihart, and B. J. Eadie (1985). Predicting the bioavailability of organic xenobiotics to *Pontoporeia hoyi* in the presence of humic and fulvic materials and natural dissolved organic matter, *Environ. Toxicol. Chem.* **4**:459-467.
- McCarty, J. F., and B. D. Jimenez (1985). Reduction in bioavailability to bluegills of polycyclic aromatic hydrocarbons bound to dissolved humic material, *Environ. Toxicol. Chem.* **4**:511-521.
- Mancini, J. L. (1983). Development of methods to define water quality effects of urban runoff, EPA 600/2-83-125, U.S. Environmental Protection Agency, Cincinnati, OH (NTIS Publ. 2584-122928, Springfield, VA).
- Marr, J. K., and R. P. Canale (1988). Load allocation for toxics using Monte Carlo techniques, *J. WPCF* **60**(5):659-666.
- Mason, C. F. (1991). *Biology of Freshwater Pollution*, 2d ed., Longman Sci. & Technical, Wiley, New York.
- O'Connor, D. J. (1988). Models of sorptive toxic substances in freshwater systems, Part I to III, *J. Environ. Eng. Div. (ASCE)* **114**(3).
- Robinson, J., et al. (1967). Organo-chlorine residues in marine organisms, *Nature* **214**:1307-1311.
- Rosiallo, A. P., J. M. Essigmann, and G. N. Wogar (1977). Rapid and accurate determination of the median lethal dose (LD₅₀) and its error on a small computer, *J. Toxicol. Environ. Health* **3**:797-809.
- Salomons, W., and U. Förstner (1984). *Metals in the Hydrosphere*, Springer Verlag, Berlin, New York.
- Schnoor, J. L., C. Sato, D. McKechnie, and D. Sahoo (1987). *Processes, Coefficients, and Models for Simulating Toxic Organics and Heavy Metals in Surface Waters*, EPA 600/3-67/015, U.S. Environmental Protection Agency, Athens, GA.
- Sprague, J. B. (1969). Review paper. Measurement of pollutant toxicity to fish. I. Bioassay methods for acute toxicity, *Water Res.* **3**:793-821.
- Sprague, J. B. (1973). The ABC's of pollutant bioassay using fish, in *Biological Methods for the Assessment of Water Quality*, J. Cairns and K. L. Dickson, eds., American Society for Testing and Materials, Philadelphia, PA.
- Thomann, R. V. (1991a). Bioconcentration and depuration by aquatic organisms, in *Contaminated Sediments: Models for Criteria, Exposure and Remediation*, Continuing Education Seminar notes, Manhattan College, Bronx, NY.
- Thomann, R. V. (1991b). Bioaccumulation model of organic chemical distribution in aquatic food chain, in *Contaminated Sediments: Models for Criteria,*

- Exposure and Remediation*, Continuing Education Seminar notes, Manhattan College, Bronx, NY.
- Thomann, R. V., and J. A. Mueller (1987). *Principles of Surface Water Quality Modeling and Control*, Harper and Row, New York.
- U.S. Environmental Protection Agency (1985). *Methods for Measuring the Acute Toxicity of Effluents to Freshwater and Marine Organisms*, EPA 600/4-85/013, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1987a). *Quality Criteria for Water 1986*, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1987b). *Integrated Risk Information System*. Vol. 1. *Supportive Documentation*, EPA 600/8-86/032a, Office of Health and Environmental Assessment, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1991). *Technical Support Documents for Water Quality-Based Toxics Control*, EPA 505/2-90-001, Office of Water, U.S. EPA, Washington, DC.
- Weber, C. I. (1981). Evaluation of the effects of effluents on aquatic life in receiving waters—An overview, in *Ecological Assessment of Effluent Impacts on Communities on Indigenous Aquatic Organisms*, ASTM STP 730, J. M. Bates and C. I. Weber, eds., American Society for Testing and Materials, Philadelphia, PA, pp. 3–13.
- Weiss, H., K. Bertine, M. Koise, and F. D. Goldberg (1975). The chemical compositions of Greenland glacier, *Geochim. Cosmochim. Acta.* **29**:1–10.

Wetlands

Wetlands are described as "the kidneys of the landscape."

W. J. Mitsch and J. G. Gosselink

Until the last century the drainage basin of the Lagoon of Venice in Italy was composed primarily of lowland marshes and wetlands transected by canals and tributary streams. Marshes and wetlands served as sinks for nutrients that promoted the growth of lush vegetation and at the same time protected the lagoons and other waterways from the symptoms of eutrophication and hypereutrophication, which are now serious problems.

The drainage work that began in the basin of the Lagoon of Venice in approximately 1880 and has continued until today, has transformed the wetlands of the basin into agricultural and urban drylands. A complex network of drainage canals with pumping stations has significantly lowered the ground-water levels throughout the basin, and large quantities of drainage and irrigation return flows rich with nutrients and residues of other agricultural and industrial chemicals are now directed toward the Lagoon. By draining the wetlands, the Lagoon was deprived of a natural buffering system for nutrients that before the beginning of the drainage works were retained therein. Increased use of fertilizers by the agricultural sector, and the switch from organic (manure) fertilizers to chemicals, are another factor contributing to the greatly increased transport of nutrients from the basin to the Lagoon.

Originally, the Lagoon itself was surrounded by tidal brackish marshes, which still remain in some parts of the Lagoon. These marshes constituted

a natural buffer that shielded the Lagoon from the influx of nutrients by runoff and shallow ground-water flows from surrounding lands. Impounding the marshes and forming the fish ponds (*valli da pesca*) limited this buffering capacity.

Loss of wetlands is not limited only to the watershed of the Lagoon of Venice. Until recently in the United States, as well as throughout the world, wetlands were considered a source of disease (malaria) and an obstacle to man's use of land resources for growth, agriculture, and economic development. Early "wetland management programs" in the United States—both governmental and private efforts—concentrated on drainage, filling, and conversions to agriculture and urban uses. The "success" of these early programs has been documented by the U.S. Fish and Wild Life Service (a government agency under the U.S. Department of Interior), which estimated that more than one-half of the approximately one million square kilometers of wetlands in the lower 48 states were lost between the arrival of the first settlers and the present time. The rate of loss of wetlands in the years before 1980 was about 1000 to 2500 km² per year. In Illinois alone, over 40,000 km² of wetlands, representing about 27% of the total state area, were drained and converted to agriculture and urban lands since the mid-1800s.

Today, wetlands are viewed in a very different light. Numerous studies have found that wetlands are essential for healthy hydrology and the ecology of watersheds. They prevent floods, cleanse waters, protect shoreline, and recharge ground-water aquifers. They retain nutrients and other pollutants, which are then incorporated in the wetland biomass. For example, studies of urban and rural watersheds near Minneapolis, Minnesota, by Oberts (1982) documented that nutrient export from watersheds with a significant wetland area (10% to 20% of the watershed area) were much lower than the export from watersheds where the wetlands were drained. Wetlands are also ecological assets because they provide habitat for waterfowl, animals, and vegetation. Control of mosquitoes can be achieved by planting mosquito-eating fish (*Gambusia affinis*) or by fish management.

The Congress, the federal government, and many states have realized the very high ecological and hydrological values of wetlands and enacted wetland protection and rehabilitation acts. For example, in Florida developers must now reestablish 2 hectares of artificial wetland for each hectare of natural wetland lost due to development. Draining of wetlands is now prohibited in many states and efforts are on the way to reestablish formerly drained wetlands. Through the efforts of the Wetland Foundation in Chicago programs aimed at wetland restoration are promoted. An example of such programs is the Des Plaines experimental wetland located

north of Chicago where scientists study wetland restoration, their incorporation into the landscape, habitat creation and enhancement, and diffuse-pollution control (Hey et al., 1989). Restoration of riparian wetlands and the construction of new wetlands play an important role in diffuse-pollution abatement.

DEFINITIONS AND TYPES OF WETLANDS

There are a variety of lands and bodies of water that are called wetlands. Wetland definitions include the following features (Mitsch and Gosselink, 1986):

1. Wetlands are distinguished by the presence of water.
2. Wetlands often have unique soils that differ from adjacent uplands.
3. Wetlands support vegetation adapted to the wet conditions (hydrophytes), and consequently are characterized by the absence of flooding intolerant vegetation.

Other less distinct features have also been pointed out, such as:

1. The depth of standing water in wetlands may vary throughout the year or be absent for some time.
2. Wetlands are often at the margin (boundary) between deep water and terrestrial uplands, and are affected by both systems.
3. Wetlands vary in size from small—one to a few hectares of prairie wetlands—to large expansive wetlands of several thousands of square kilometers (example, the Everglades in Florida).

In general terms, wetlands are defined (Cowardin et al., 1979) as lands where saturation with water is the dominant factor determining the nature of soil development and the types of plant and animal communities living on the surface.

Due to the existence types of various of wetlands, a precise definition for wetlands that would satisfy all is not possible (Mitsch and Gosselink, 1986). The most comprehensive definition for wetlands was advanced by the U.S. Fish and Wildlife Service:

Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. Wetlands must have one or more of the following attributes: (1) at least periodically, the land supports predominantly hydrophytes, (2) the substrate is predominantly undrained hydric soils, or (3) the substrate is nonsoil

(organic matter) with water or covered by shallow water at some time during the growing season each year.

Since 1975 the U.S. Fish and Wildlife Service has been conducting an inventory of wetlands throughout the United States. U.S. wetland terminology recognizes a variety of types and terms that are applied to wetlands (Mitsch and Gosselink, 1986). The most important types of wetlands are as follows.

Coastal or Marine Wetlands

The marine system extends from the outer edge of the continental shelf to the high water or spring tides. *Marine subtidal* includes areas that are continuously submerged, while *marine intertidal* includes areas in which the substrate (soil and organic bottom matter) is alternately exposed and flooded by tides.

The estuarine system consists of deepwater tidal habitats that are semienclosed by land but have open, partially obstructed, or sporadic access to the open sea or ocean and in which sea water is at least occasionally diluted by freshwater runoff from the land. *Estuarine subtidal*



FIGURE 14.1. Tidal marsh inside the Lagoon of Venice in Italy. (Photo: V. Novotny.)

is that portion that is continuously submerged (considered deepwater habitat), while *estuarine intertidal* is the portion that is by turns exposed and flooded by tides. The marshes within the tidal portion of the Lagoon of Venice could be categorized as estuarine intertidal or subtidal wetlands (Fig. 14.1).

Depending on salinity and type of vegetation, marine wetlands can be categorized as:

Tidal salt marsh. Contains primarily salt or brackish water.

Tidal freshwater marsh. More distant from the coast; experiences some tidal effects, but mostly contains freshwater.

Mangrove wetlands. Saltwater marshes in tropical and subtropical areas (Florida). Salt-tolerant trees and brush usually dominate these wetlands.

Inland Wetlands

The *palustrine wetlands* include all nontidal wetlands excluding those adjacent to or that are a part of river channels and littoral zones of larger



FIGURE 14.2. Forested wetland: Huricon marsh in Wisconsin. (Photo: A. Capodaglio.)

and distinct lakes and other impoundments. Most definitions divide the palustrine wetlands into the following groups:

Forested and scrub land. Wetlands dominated by the presence of woody vegetation (Fig. 14.2).

Emergent. Wetlands with primary erect, rooted herbaceous plants typically found in wet environments and other palustrine areas.

Nonvegetated wetlands. Small inland bodies of water and wetlands dominated by aquatic beds.

An arbitrary depth of 2 meters has been used by the U.S. Fish and Wildlife Service to distinguish between wetland and deepwater systems.

Riverine wetlands (Fig. 14.3) adjoin deepwater stream channels, while *lacustrine wetlands* (Fig. 14.4) are littoral wetlands situated in topographic depressions or surrounding lakes or impounded rivers (reservoirs). The surface area of the lake or impoundment should be more than 8 hectares (20 acres), or be deeper than 2 meters, or have an active wave-formed or bedrock shoreline feature. The term *riparian wetlands* is commonly used for lacustrine and riverine wetlands.

The following terms are commonly associated with nontidal wetlands:

Swamp. Wetland dominated by trees and shrubs.

Marsh. Frequently or continually inundated wetland characterized by emergent vegetation adapted to saturated soil conditions. In European terminology, a marsh does not accumulate peat.

Bog. A peat-accumulating wetland that has no significant inflows or outflows and supports acidophilic mosses.

Fen. A peat-accumulating wetland that receives some drainage. Fens and bogs are generically called peatlands.

Slough. A swamp or shallow lake system, or a slowly flowing swamp or marsh.

Other Wetlands

Rice fields, some disposal of wastewater on land, and artificial wetlands represent most of the remaining land surfaces that can be categorized as wetlands. An artificial wetland is a man-made transformation of previously dry upland into a wetland, and is accomplished by one or more of the following measures:

1. Disconnecting and abandoning previously installed drainage.
2. Permanent or periodic flooding of the land with water from nearby bodies of water.



FIGURE 14.3. Meandering stream and wetland in northern Wisconsin.
(Photo: University of Wisconsin.)



FIGURE 14.4. Lacustrine riparian wetland in northern Wisconsin.
(Photo: University of Wisconsin.)

3. Diverting urban storm water and/or treated or partially treated effluents onto the land, which creates saturated soil conditions (similar to land-disposal practices for wastewater).
4. Grading and excavating top soil layers to the ground-water table.

Las Vegas Wash, a wetland located between the City of Las Vegas and Lake Mead in Nevada, was unintentionally created by increased effluent discharges from a rapidly developing metropolis into a previously ephemeral stream and surrounding land in desert climatic conditions. Generally, however, urbanization results in loss of wetland.

Creating an artificial wetland is an engineering and ecological endeavor that must be planned and executed properly to avoid development of unwanted species (for example, obnoxious algal mats in shallow standing waters or nuisance growths of water hyacinths in subtropical and tropical areas). *Ecological engineering* is a new branch of environmental engineering that deals with protection and restoration of ecological systems, including wetlands (Mitsch and Jørgensen, 1989). To ensure that proper vegetation will develop, the bottom surface of the constructed wetland should be covered by a mulch brought from an existing nearby wetland, and the wetland should be seeded with proper wetland vegetation. Typical wetland vegetation is shown in Table 14.1. All plant species reported in the table and shown in Figure 14.5 are common throughout the world. However, Mitsch (1990) pointed out that the list of possible wetland plants is very large and that an almost infinite number of combinations of plants exists. It should also be pointed out that water hyacinth and duckweed are suitable in aquaculture ponds. However, their use in wetlands is questionable and even undesirable.

Creation of wetlands and their use for water-pollution control is now

TABLE 14.1 Typical Emergent Wetland Plants

Common Name	Scientific Name	Temperature Range (°C)		Maximum Salinity Tolerance (g/l)	Effective pH Range
		Desirable	Seed Germination		
Cattail	<i>Typha</i> spp.	10-30	12-24	30	4-10
Common reed	<i>Phragmites communis</i>	12-23	10-30	45	2-6
Rush	<i>Juncus</i> spp.	16-26		20	5-7.5
Bulrush	<i>Scirpus</i> spp.	16-27		20	4-9
Sedge	<i>Carex</i> spp.	14-32			5-7.5

Source: After U.S. EPA (1988).

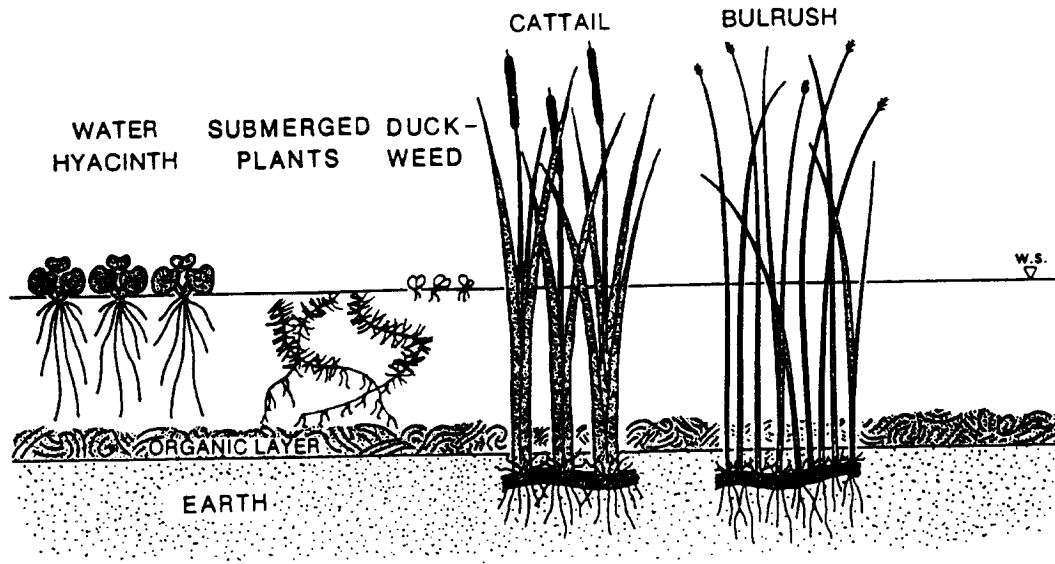


FIGURE 14.5. Common aquatic plants.

enhanced by recent efforts to take polluting agricultural lands out of production. In the United States such lands are commonly converted to so-called "buffer strips" or high slope woodland. Creation of riparian wetlands in combination with overland-flow buffer strips could provide an improved water-pollution control benefit. Large projects to restore riverine wetland systems are being carried out on the Des Plaines River in Lake County, Illinois (Hey, 1988), the Milwaukee River in Wisconsin (Gayán and D'Antuono, 1988), and several other rivers in the United States. A large number of wetland restoration projects have been carried out in Florida, which enacted the most far-reaching wetland protection and restoration laws (Fisk, 1989). In southeastern Wisconsin, a 36-ha (90-A) man-made wetland has been created to protect and enhance the water quality of the Delavan Lake (810 ha). The wetland was constructed as a critical component of a major restoration project and is very important to the long-term management of the lake.

WETLAND FUNCTION

In contrast to early views that regarded wetlands as a nuisance, beneficial uses of wetlands are numerous and should be considered in any wetland protection, enhancement, use, and creation endeavor. The most important uses and ecological functions of wetlands are as follows.

- *Flood storage and conveyance.* Riverine wetlands and adjacent flood-plain form natural corridors in which floodwater is stored and held

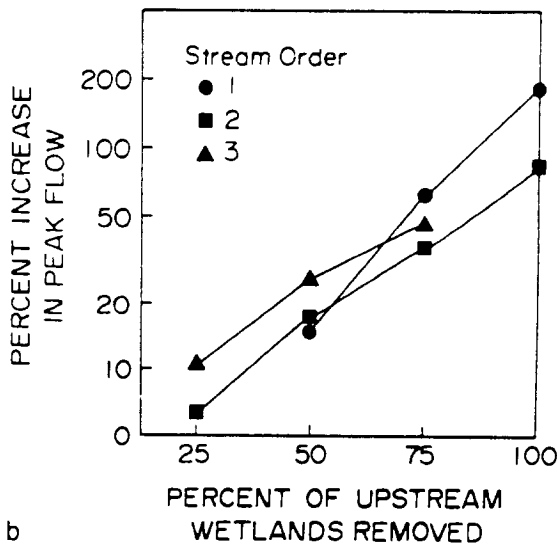
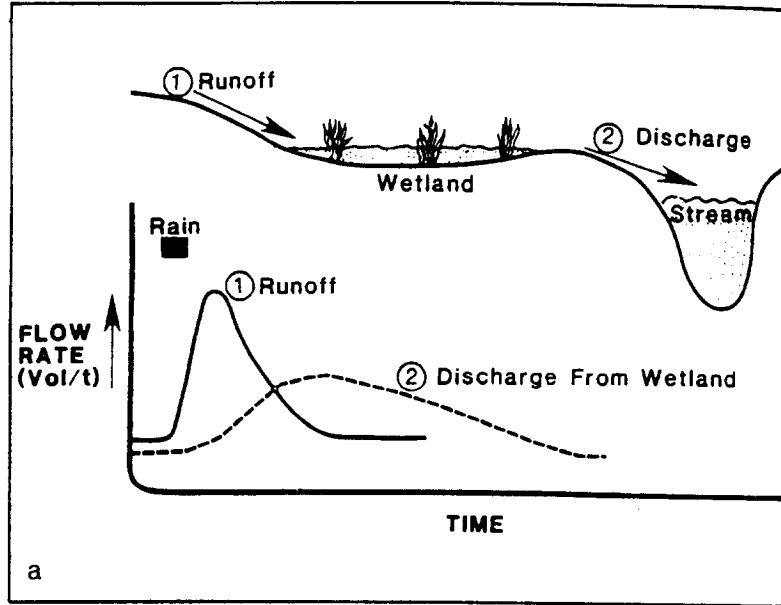


FIGURE 14.6. (a) Effect of wetlands on stream flow. (After Mitsch and Gosselink, 1986.) (b) Impact of wetland drainage on peak flow and flooding. (After Ogawa and Male, 1983.)

back. Palustrine wetlands that receive surface runoff from a surrounding watershed will attenuate the runoff peaks (Fig. 14.6).

- *Stream flow modification.* Wetlands have an impact on other flows as well. Their tendency to retain water is most evident during the summer, fall, and winter periods, while during spring when wetlands are commonly flooded they tend to release more flow than the surrounding upland.

- *Erosion reduction and sediment control.* Riverine and lacustrine wetland vegetation slows runoff and flood flow, thus mitigating shoreline erosion. The vegetation in all types of wetlands slows down the flow, thus enhancing sedimentation. Vegetation stems also filter sediments from water, and the roots of the vegetation then bind and stabilize the deposited silt.
- *Ground-water recharge/discharge.* Wetlands and ground water are interconnected; however, the connection is poor since the very existence of the wetland usually implies highly impervious substrate soils. Possible connections are shown on Figure 14.7, taken from Novitzki (1979). If shallow ground water is discharging into a wetland, it can bring nutrients (primarily nitrate nitrogen) and minerals, which are then used by the wetland vegetation. The ground-water zone can then be recharged with water from the wetland with less nitrate contamination. If a wetland is recharging (the flow is from the wetland toward ground water), it is a natural barrier preventing some mobile pollutants from entering the ground-water zones.
- *Pollution control.* The foregoing discussion on pollution prevention of ground-water resources and sediment control can be extended to other bodies of water with which wetlands interact. Wetland vegetation and microorganisms residing on the stems and roots of vegetation and in sediments can filter, absorb, and decompose suspended and dissolved organic matter, and convert organic and ammoniacal nitrogen to nitrate (nitrification) with subsequent denitrification of nitrate to nitrogen gas. Some phosphorus can be used by vegetation and/or adsorbed onto sediments. The accumulated residual organic matter can also immobilize toxic pollutants and essentially bury them without harm to the biota. Extensive research is being conducted on the use of natural and man-made wetlands as tertiary treatment facilities for municipal and industrial wastewater and for storm-water runoff pollution control.
- *Wildlife habitats.* The land–water interface, including upland buffer zones, is among the richest wildlife habitats in the world. This is because of the abundance of water, nutrients, and shelter provided by the wetlands and their vegetation. Many endangered species of animals rely on wetlands for their habitats. For example, in Illinois 40% of the state's threatened and endangered species depend on the remaining wetlands, which now represent less than 3% of the state's area. Many subtropical species in Florida and Louisiana and many other areas would vanish with the loss of wetlands.
- *Recreation and enjoyment.* Over twenty million sport and commercial fishers nationwide use wetlands, along with millions of hunters. Millions more just enjoy watching wetland birds, wildlife, and plants.

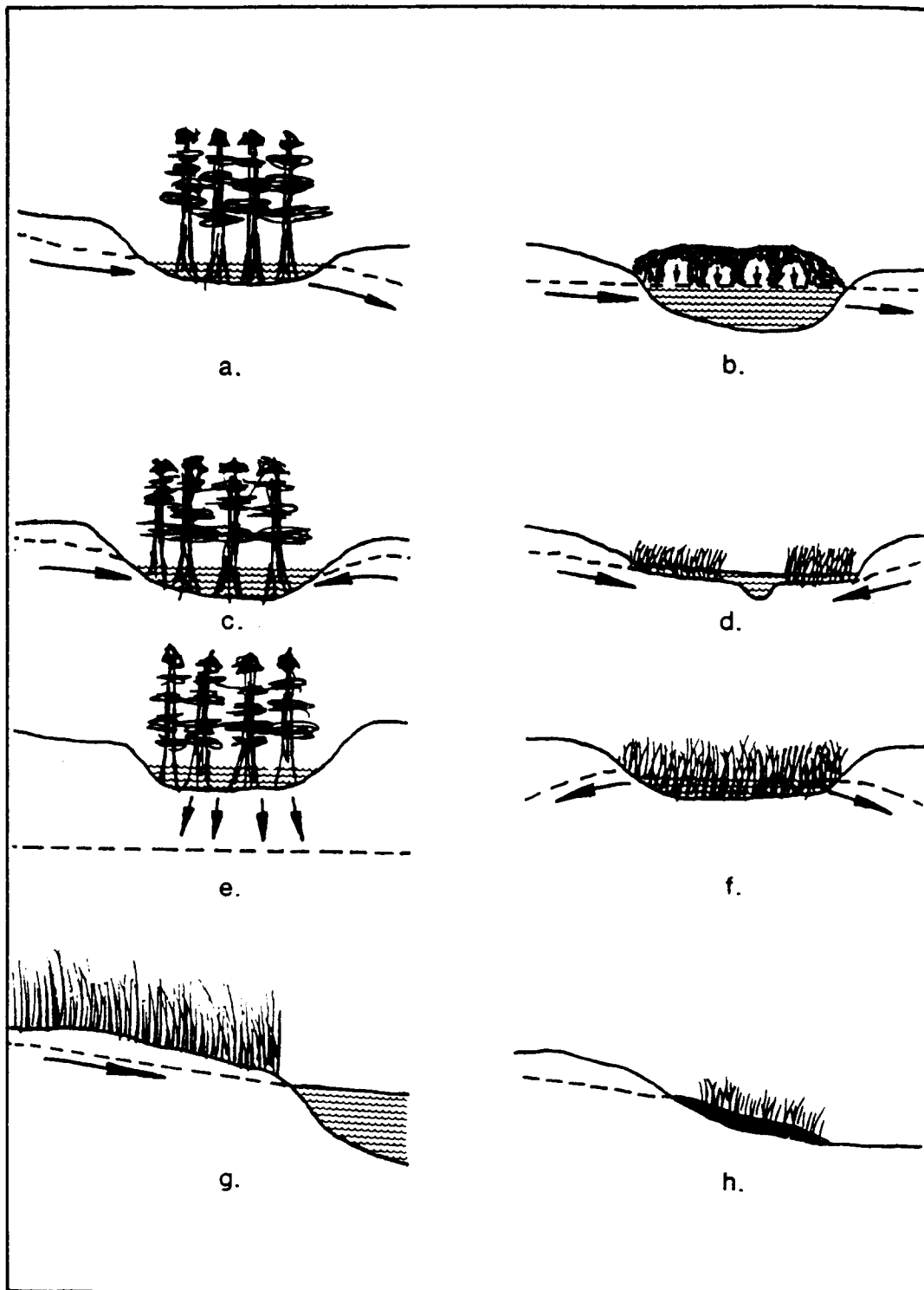


FIGURE 14.7. Wetland-ground-water interactions. (a) Both inflows and outflows of ground water through the marsh. (b) Underflow of raised bog. (c) Swamp as ground-water depression wetland. (d) Marsh as ground-water depression wetland. (e) Perched swamp or surface-water depression wetland. (f) Marsh as ground-water source (recharging). (g) Ground-water flow through saltwater marsh or riparian wetland. (h) Ground-water seep wetland or ground-water slope wetland. (After Novitzki, 1979.)

Use of Wetlands for Water Quality Control

A large amount of the recent scientific and technical literature focused on the application of wetlands—natural or artificial—for water-pollution control (for example, Reed, Middlebrooks, and Crites, 1988; Hammer, 1989a). An article in *Civil Engineering* (Dawson, 1989) pointed out the benefits of using wetlands for treatment or posttreatment of municipal wastewater in several U.S. locations. A treatise on ecological considerations and mitigation of wetlands that are used for water quality control and disposal of wastewater has been published by Godfrey et al. (1985).

Following Reed, Middlebrooks, and Crites (1988) and Hammer (1989a), the major uses of wetlands in combination with the disposal of wastewater effluents and storm-water runoff are:

- Disposal of treatment effluents into natural wetlands.
- Use of natural wetlands for further wastewater renovation.
- Use of effluents or partially treated wastewater for the enhancement, restoration, or creation of wetlands.
- Use of constructed (artificial) wetlands as a wastewater treatment process.
- Use of natural or man-made wetland for storm-water management and treatment.
- Potential use of constructed wetlands for immobilization of toxics and their permanent safe burial.

In the United States, natural wetlands are considered as any other receiving surface-water body; hence, they are protected from excessive discharges of pollution and any discharge requires a permit. This generally precludes discharges of untreated or partially treated (with a primary treatment only) wastewater or polluted storm water (runoff) into natural wetlands. Most states (except those that enacted special wetland standards) make no distinction between a wetland with standing water and adjacent surface water. One may expect that similar legislation is in place or will be enacted in other developed countries. In most cases it would mean that the discharges of wastewater into wetland systems must receive treatment.

The use of natural wetlands is beneficial and applicable for nutrient control and removal, for effluent polishing, for removal of nutrients and other pollutants from urban storm-water runoff, for in-stream waste-assimilative capacity enhancement (for example, diverting a part of the flow of a polluted stream onto an adjacent riparian wetland), and as buffers between agricultural areas and surface-water bodies. However, as is true for any surface-water body, the ability of wetlands to receive and

assimilate pollution is not limitless, and those who plan to discharge pollution into natural wetlands as well as the regulatory agencies must make sure that the waste-assimilative capacity of the wetland into which the discharge is planned is not exceeded. The waste-assimilative capacity of wetlands can be estimated by modeling (Mitsch, Straskraba, and Jørgensen, 1989). Since the hydraulics of wetland systems is the same as for surface-water bodies with a significant sediment component, the U.S. EPA's water quality model WASP may be used, with or without modifications, for wetland modeling.

On the other hand, artificial (constructed) wetlands (Figs. 14.8 to 14.10) are generally not subjected to surface-water quality standards and can be conveniently and inexpensively used for effluent treatment, polishing, nutrient removal, and storm-water management and treatment. Constructing a wetland in a place where there is none now would bypass in most cases the regulations restricting discharges into surface-water bodies and only the outflow from the wetland would have to meet the effluent standards (which as was stated previously can be effluent or stream limited). Unlike for natural wetlands, where both influent into the wetlands and the outflow are regulated and must comply with environmental standards, only outflow from artificial wetlands is commonly

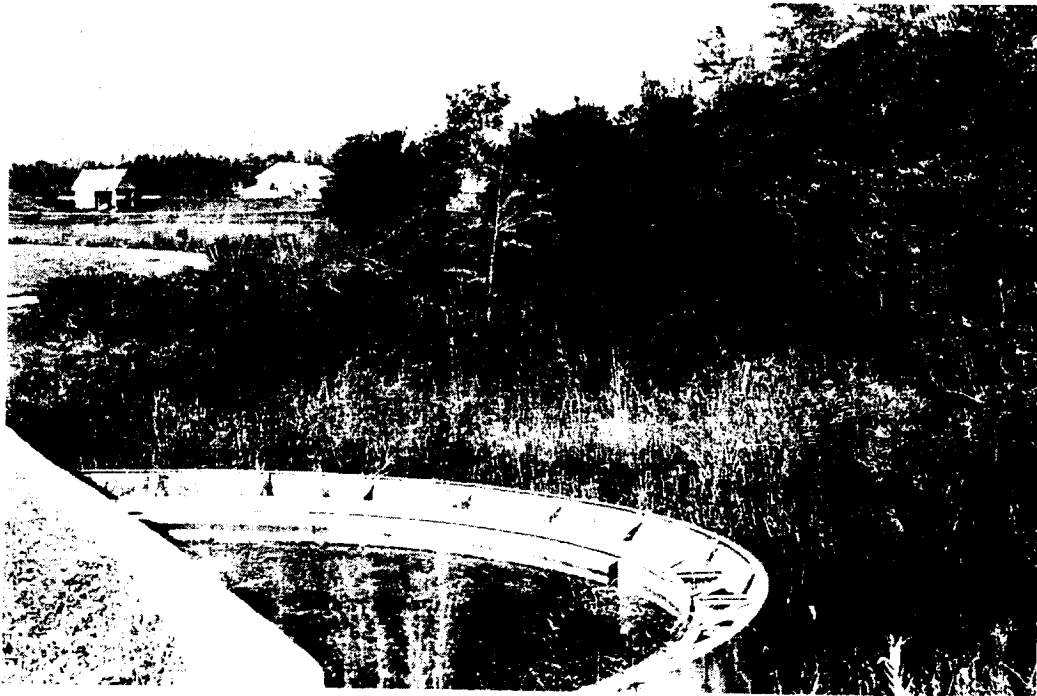


FIGURE 14.8. Wetland restoration project near Tampa, Florida. Wetlands are incorporated into a new residential development. (Photo: V. Novotny.)



FIGURE 14.9. Restored wetland used for storm-water treatment in the Minneapolis-St. Paul area. (Photo: G. Oberts.)

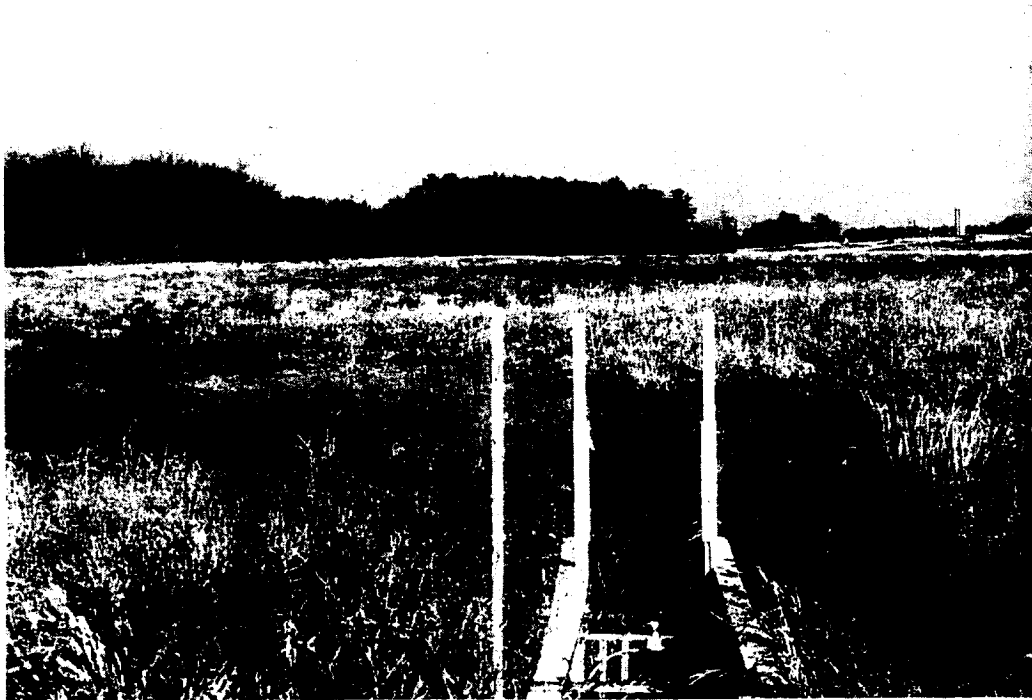


FIGURE 14.10. Riparian wetland restoration, Des Plains River, Lake County, Illinois. (Photo: V. Novotny.)

regulated. Also water quality control performance can be significantly improved since the vegetation and wetland management of the artificial wetland system can be optimized for the most efficient removal of pollutants.

Organic Matter and Nutrient Mass Balance in Wetlands

Natural and well-constructed man-made wetlands contain an abundance of diverse vegetative and bacterial populations that are very effective in removing a variety of pollutants, including decomposition and immobilization of toxics. Both emerged and submerged plants and algae are usually present. In addition to the biological and biochemical absorption and degradation of pollutants, physical processes also attenuate pollution. The processes that contribute to the removal of pollutants by wetlands can be grouped as follows.

1. Physical processes
 - a. Sedimentation
 - b. Filtration
2. Physical-chemical processes
 - a. Adsorption of pollutants on plants and soil and organic substrates
3. Biochemical processes
 - a. Aerobic biochemical degradation of organic matter by bacteria in water, attached to plants, stems, in the top layer of sediments, and in aerobic pockets near roots and rhizomes of plants
 - b. Nitrification by nitrifying organisms primarily residing on plant stems, in the top layer of humic sediments and on the roots and rhizomes of plants
 - c. Denitrification by bacteria residing in anaerobic water and sediments
 - d. Anaerobic decomposition of organic matter in sediments and anaerobic water
 - e. Uptake of nutrients and some pollutants by the plants and their incorporation in the plant biomass

The richness and variety of the processes is the primary asset of wetland treatment. Due to very low velocities of water and dense vegetation, sedimentation and filtration are very effective in removing suspended solids and the pollutants attached on them. Bacteria attached to plant stems and in the upper part of sediments are the major factor for BOD

removal. In a sense, wetlands in their function of BOD removal resemble a trickling filter rather than a free-standing body of water.

Kadlec and Alvord (1989) pointed out that there are three major processes that participate in the removal of waterborne pollutants entering the wetland: *biomass increases*, *burial* and *gasification*. The nutrients that enter the wetland are incorporated into the biomass, primarily vegetation and algae (autotrophic biota); however, some nutrients as well as biodegradable organics are incorporated into the biomass of heterotrophic organisms. The production of new biomass due to the nutrient and organic input is a temporary but often long-term sink for assimilated pollutants.

Formation of new organic soils and peat that contain organics and nutrients represent a more permanent sink. Note that coal deposits were formed millions of years ago by the deposition of organic matter in wetlands. Assimilated nitrogen, carbon, and sulphur after biological transformation can be released into the atmosphere and permanently lost from the system. Organic solids and other chemicals can adsorb and/or immobilize and even decompose toxic materials. If toxic loads are below the assimilative capacity of the wetland the constructed wetlands could safely dispose of the toxics and bury the end products. Unfortunately, the state-of-the-art knowledge on assimilation of toxics by wetlands is still not advanced enough to make any definite conclusions and recommendations.

Production, Retention, and Removal of Biodegradable Organic Matter

The concepts of organic productivity and the roles of nutrients and oxygen in aquatic bodies of water and underlying sediments were explained in Chapter 12. Processes occurring in soils were the subject of Chapter 6. The same concepts are applicable to wetlands since wetlands are considered a transition between land and bodies of water.

Both tidal marshes and inland wetlands are the most productive ecosystems in the world. Up to 2 tonnes per hectare of organic matter can be produced annually by these systems; however, typical values of organic matter production by primary productivity in tidal marshes are between 50 and 300 kg/ha-year, while productivity of inland wetlands is somewhat higher, ranging from 90 to 600 kg/ha-year. This is higher than the productivity of intensively cultivated agricultural fields (Mitsch and Gosselink, 1986). In this context, an allochthonous (= from sources outside the wetland, such as runoff and wastewater discharges) influx of organics may not be significant in comparison with the productivity of the wetland. Table 14.2 presents the compiled data on the organic productivity

TABLE 14.2 Production of Organic Matter (Net Primary Productivity) in Natural Wetlands

Wetland Type	Net Primary Productivity (g/m ² -year) (Range)	Total Biomass (kg/m ³)	Nitrogen Loading (g N/m ² -yr)	Plant Nitrogen Uptake (g/m ² -yr)
Northern bog	500 (153-1943)	53	0.8	9
Inland fresh marsh	1980 (1070-2860)	46	22	48
Tidal fresh marsh	1370 (780-2100)	46	75	54
Salt marsh	1950 (330-3700)	46	30-100	25
Swamp forest	870 (390-1780)	52	900	14
Riparian forest	1040 (750-1370)	37	10,000	17
Mangrove	1500 (0-4700)	60	30	24

Source: After Mitsch and Gosselink (1986).

of various types of wetlands. These data are important when considering the assimilative capacity of wetlands for various pollutants.

The removal of organic matter is by sedimentation, filtration, and biological absorption by microorganisms residing in water, attached to vegetation, and living in sediments. Figure 14.11 shows the processes in water and sediment columns. The biodegradation can be both aerobic and anaerobic, with aerobic degradation taking place primarily in the water column and in the upper part of the sediments, and anaerobic degradation occurring in sediments and in water if the oxygen supply is exhausted. However, it has been pointed out by Reed, Middlebrooks, and Crites (1988) that some rooted plants (for example, the water hyacinth) can transfer oxygen to their root zone and rhizomes and in this way create small pockets for aerobic degradation and nitrification (Fig. 14.12).

About half of the organic matter is retained and decomposed in tidal marshes and the second half is transported to the downstream sea or estuary system (Mitsch and Gosselink, 1986). The export of organic matter from inland wetlands varies greatly and depends on the type of the wetland, its hydrology, and many other factors.

Immobilization of Toxic Compounds by the Organic Substrate of Wetlands

Organic matter production in wetlands may also be a key to their use in disposing (sink) of low quantities of toxic compounds. Most toxic com-

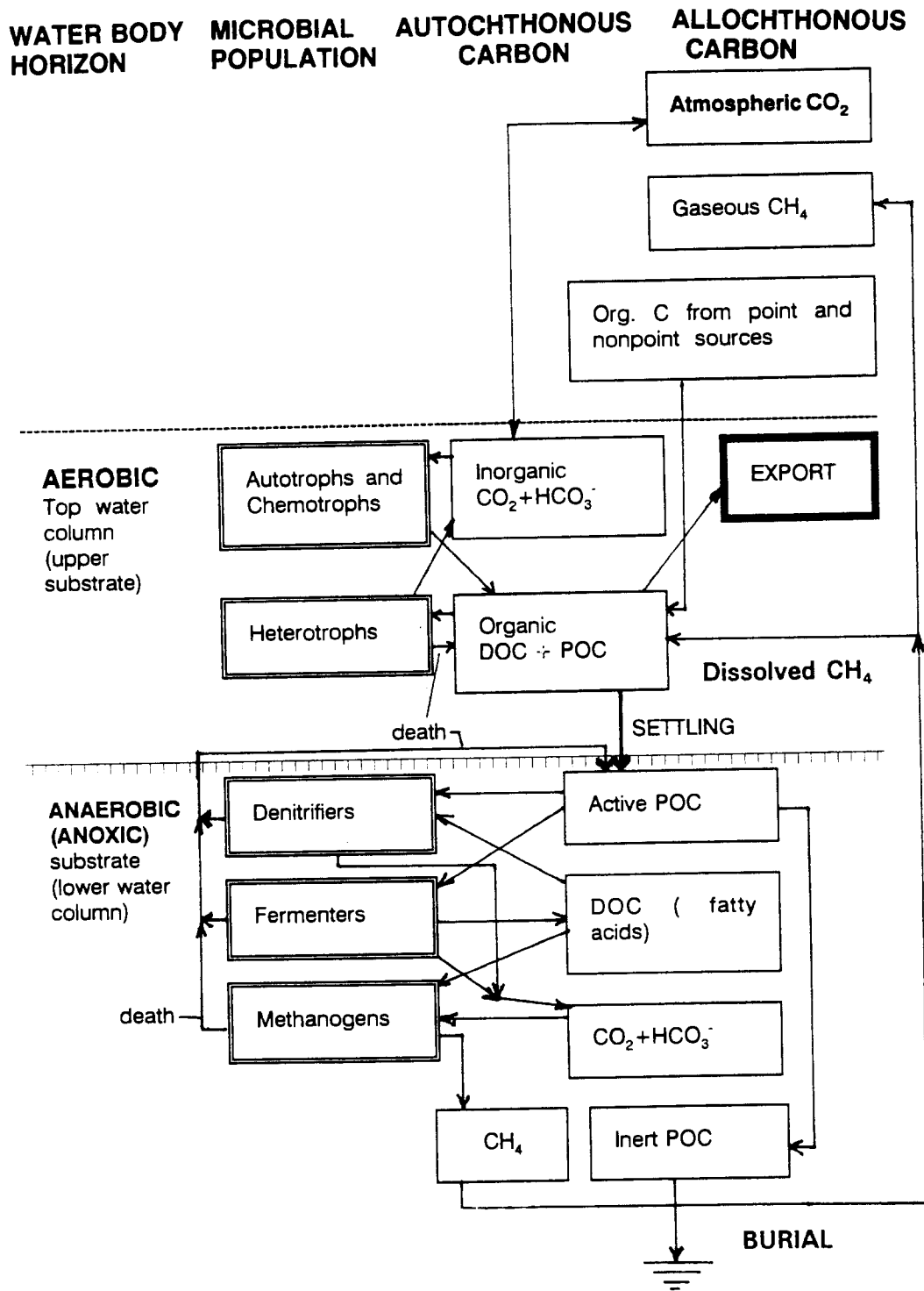


FIGURE 14.11. Water-sediment column processes. Key: POC-particulate organic carbon; DOC-dissolved organic carbon.

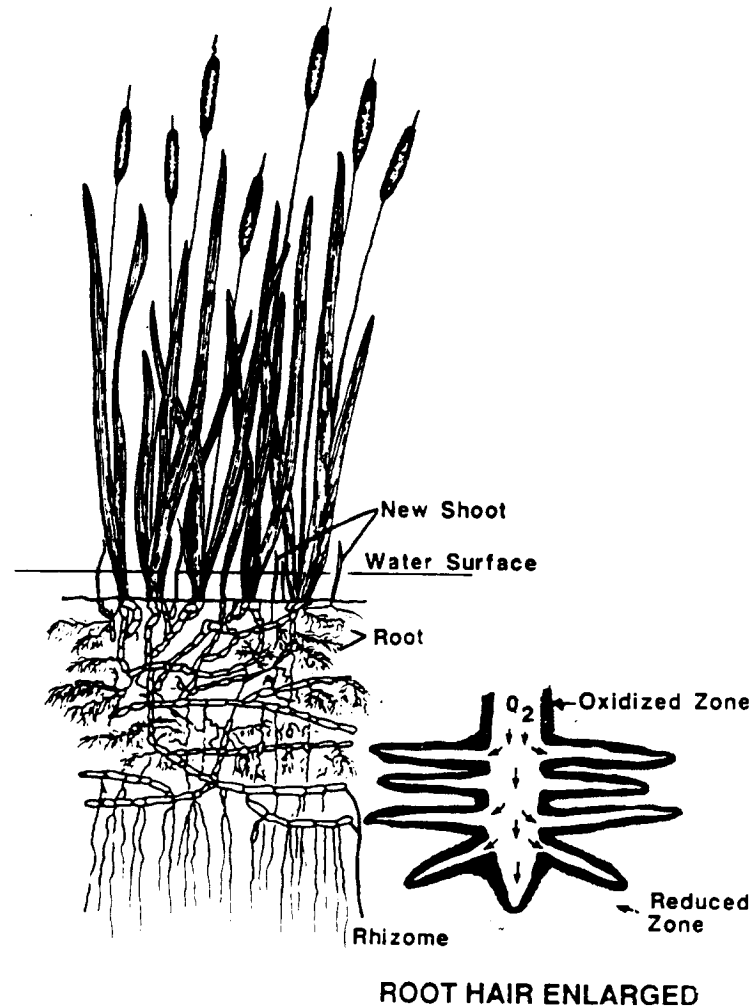


FIGURE 14.12. Transport of oxygen by roots and rhizomes in anaerobic wetland substrate. (After Hammer and Bastian, 1989, with permission from Lewis Publishers, a subsidiary of CRC Press, Boca Raton, Florida.)

pounds, such as toxic metals and organic chemicals, have a strong affinity for immobilization by particulate and colloidal organic matter (toxic metals and organics) and by sulfide ligands (toxic metals) generated by anaerobic decomposition of organic matter, both in sediments and wetland substrates. Complexed toxic compounds are unavailable to aquatic biota, including plankton, plants, and animal (invertebrate and higher) forms. Complexed and adsorbed toxic compounds can also be filtered and/or settled out in wetlands. Quantitatively, this function of wetlands is generally unknown. Some concepts and calculations will be presented in the next section of this chapter.

The assimilative capacity of wetlands for toxic compounds is related to the primary productivity of the wetland that is stimulated by the nutrient inputs and recycling.

Nutrient Transformation and Removal

The best use of wetlands for water quality control, in addition to sediment and particulates filtering, is for nutrient removal. The high production of organic matter by wetlands requires proportionate sources of nutrients that may come from autochthonous (inside the ecosystem) or from allochthonous (outside the boundary of the ecosystem) sources. Autochthonous sources imply nutrient recycling within the wetland whereby nutrients (N and P) are taken up by plants and released back after death and decomposition. Allochthonous sources include the influx of nutrients from the tributaries and from the atmosphere. Whether a wetland becomes an effective sink of nutrients depends on the buildup of inert residual organic matter and on the permanent burial rate in the sediment of the produced organic matter (organic forms of nutrients).

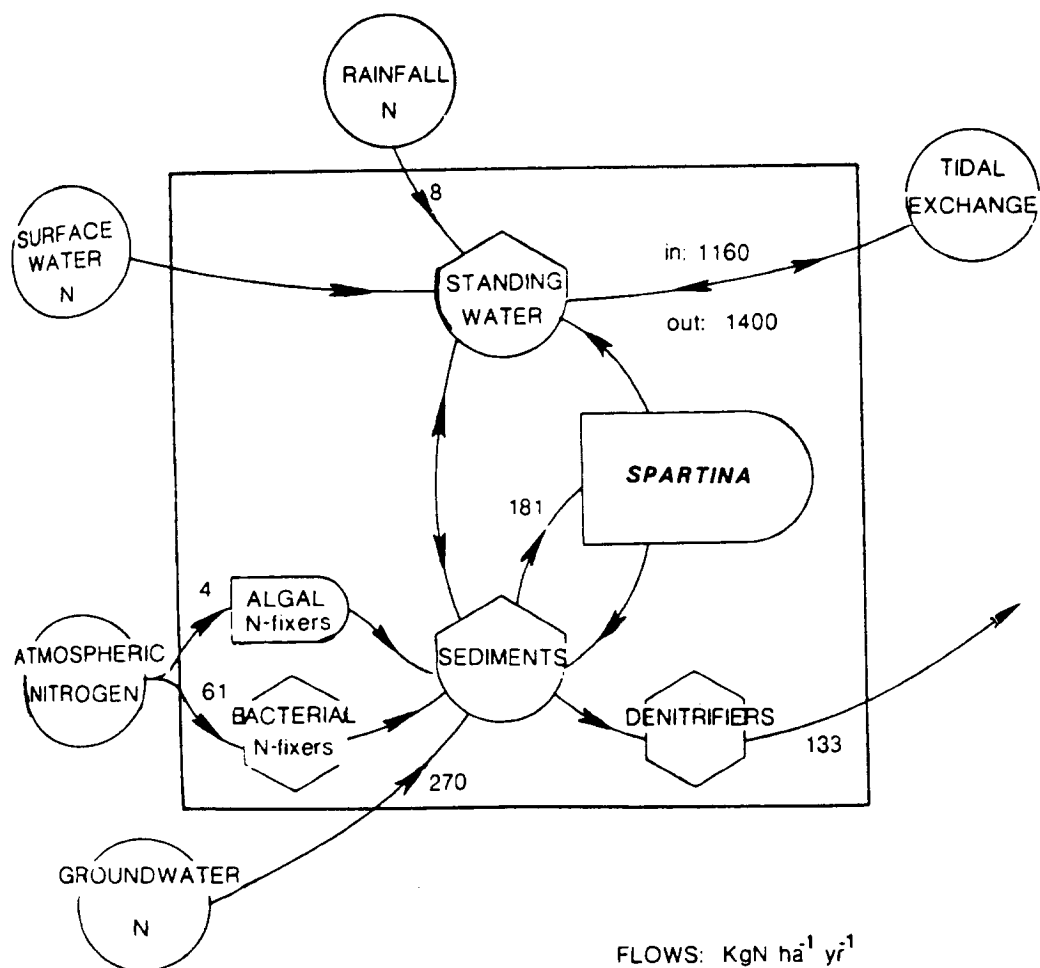


FIGURE 14.13. Nutrient balance of the Great Sippewissett salt marsh in Massachusetts. (After Mitsch and Gosselink, 1986.)

their permanent adsorption and immobilization by the clay and organic components of the sediment (ammonia and phosphates), and on the gasification rate of nitrate-nitrogen by denitrification.

Kadlec and Alvord (1989) have shown that a northern wetland (located in Michigan) responds dramatically to elevated nutrient inputs from allochthonous sources; in this case, from biologically treated wastewater. Increases in nitrogen and phosphorus caused an expansion of the biomass and a buildup of several millimeters of new organic soil from the organic residues of plants grown in the wetland, which then buried significant quantities of nutrients. Nitrification was minimal during growing season because of the lack of oxygen in the sediments and bottom layers. Denitrification of nitrate was almost complete and the nitrate was normally exhausted in the wetland during the growing season.

A nutrient budget for a saltwater marsh located in Massachusetts is shown on Figure 14.13. Denitrification in saltwater marshes can be high and the marshes can become nutrient sinks, especially when high nutrient waters pass through them (Valiela, Teal, and Sass, 1973). Many coastal wetlands are nitrogen limited.

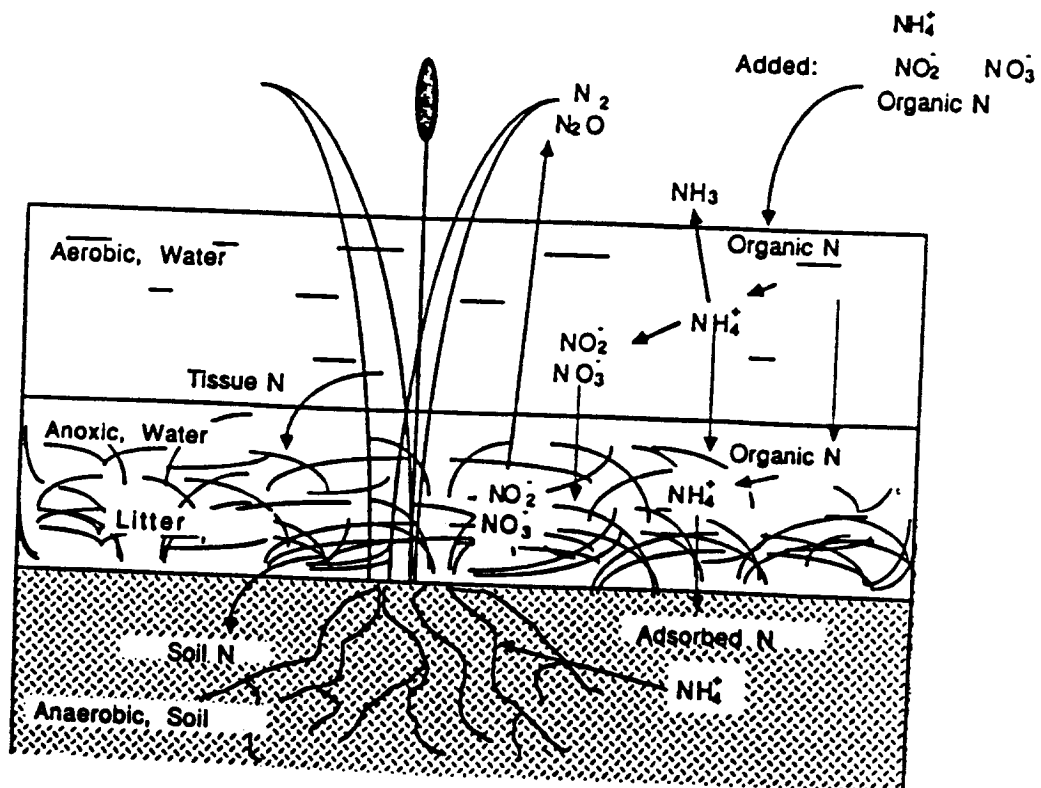


FIGURE 14.14. Routes and transformation of nitrogen in a palustrine wetland receiving wastewater input. (After Kadlec, 1988.)

Figure 14.14 shows routes and transformations of nitrogen in a palustrine wetland used for wastewater treatment. Due to the heavier organic load and high productivity of the wetland, the bottom water layer is mostly anaerobic. Thus, the entire substrate is anaerobic with the exception of possible small aerobic pockets around the roots of plants. In another situation, especially on bright sunny days during photosynthetic oxygen production, the water column becomes aerobic and the oxygen supply creates a thin aerobic top sediment layer that allows the nitrification process.

Mitsch and Gosselink (1986) compiled data on the performances of natural wetlands for the removal of nutrients. The performance or treatment efficiency of wetlands is greatly affected by the geographical locations. Northern wetlands located in colder climatic zones have two distinct seasons: the growing season and the dormant season. During the growing season the nutrients are effectively removed by the plants and by simultaneous nitrification–denitrification. However, during the dormant season, nutrients are released and the wetland may become a source of

TABLE 14.3 Examples of Natural Wetlands Receiving Wastewater Inputs and Nutrients Removal Efficiencies

Type of Wetland	Location	Loading (Pop./ha)	Substrate	Nutrient removal (percent)	
				Total N	Total P
<i>Northern peatland</i>					
Bog	Wisconsin	30	O	98	78
<i>Nontidal freshwater marsh</i>					
Cattail marsh	Wisconsin	17	O	80	88
Lacustrine marsh	Ontario	—	—	38	24
Deepwater marsh	Florida	99	O	—	97
Lacustrine marsh	Hungary	—	—	95	—
Riverine swamp	South Carolina	—	O	—	50
<i>Tidal freshwater marsh</i>					
Deepwater marsh	Louisiana	—	O	51	53
Complex marsh	New Jersey	198	I	40	0
<i>Tidal salt marsh</i>					
Brackish marsh	Chesapeake Bay	—	O/I	0	1.5 ^a
Salt marsh	Georgia	Sludge	O/I	50	—
Salt marsh	Massachusetts	Sludge	O/I	85	—

Source: Compiled by Mitsch and Gosselink (1986).

Note: O: organic substrate; I: inorganic substrate; (—): information not given.

^aLoad given in g/m²-year.

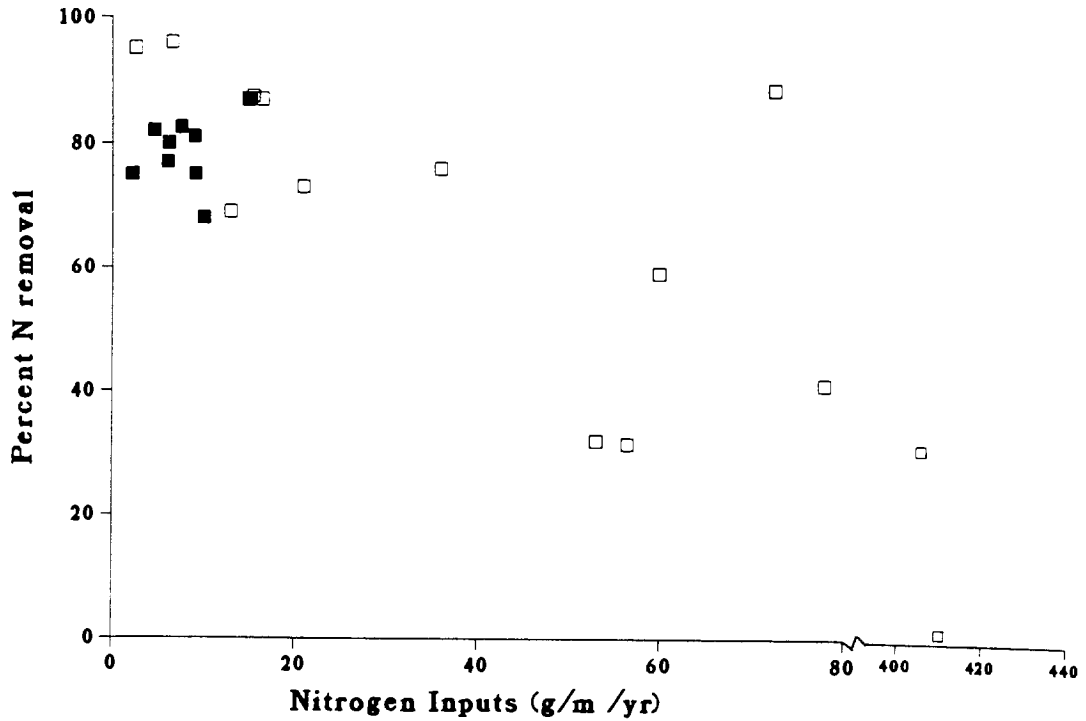


FIGURE 14.15. Nitrogen removal efficiency by wetlands as a function of nitrogen loading rate. (Based on data from Mitsch and Gosselink, 1986; Richardson and Nichols, 1985; and Knight, Windchester, and Highman, 1984.)

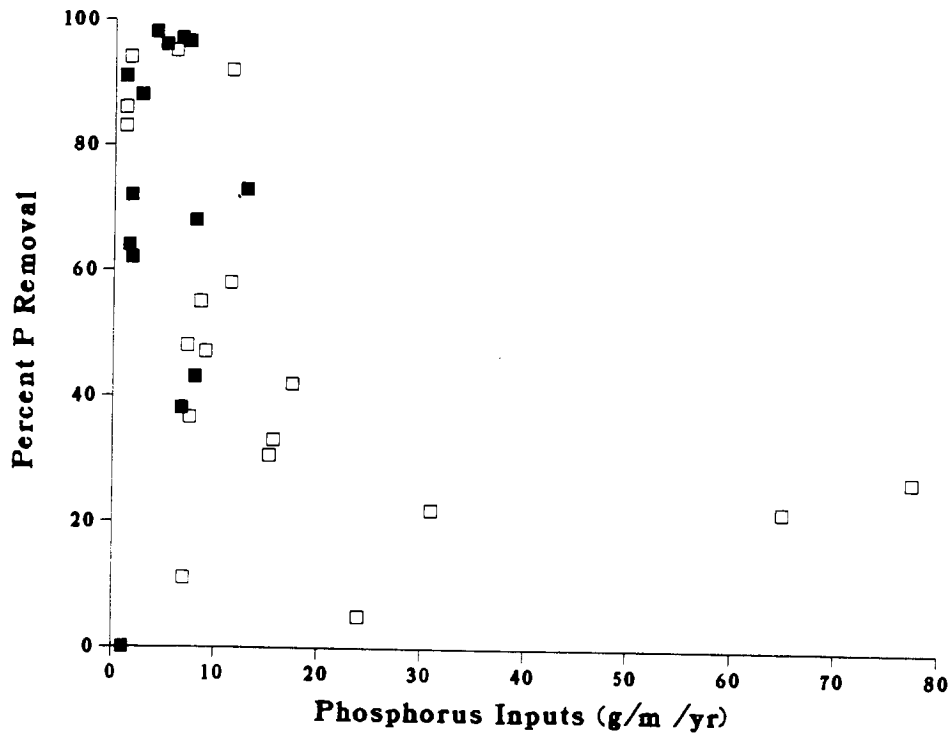


FIGURE 14.16. Phosphorus removal efficiency by wetlands as a function of phosphorus loading rate. (Based on data from Mitsch and Gosselink, 1986; Richardson and Nichols, 1985; and Knight, Windchester, and Highman, 1984.)

the nutrients. The ranges of removal efficiencies compiled by Mitsch and Gosselink (1986), shown in Table 14.3 and in Figures 14.15 and 14.16 (related to the nutrient loading), were very wide, both for phosphorus and nitrogen.

From the foregoing discussion and from literature findings the following rules have become apparent:

1. The wetland removal efficiency for nitrogen is high if the nitrogen is in a nitrate (oxidized) form. In this case, both nitrogen uptake by plants with subsequent burial of dead plants and algae, and denitrification are responsible for nitrogen removal. In some cases, 98% to 99% nitrogen removal efficiencies have been measured in such systems. Nitrate nitrogen is not adsorbed by soil organic particulates. Generally, wetlands require a start-up period for full development of denitrification (Kadlec, 1988). Maintaining aerobic pools in the wetland enhances nitrification but suppresses denitrification.

Stengel and Schulz-Hock (1989) have shown that denitrification in constructed wetlands was possible throughout the entire year in the climatic conditions of central Europe (similar to the northeastern United States).

2. If nitrogen arrives in the wetland in an ammoniacal or organic form, the primary removal mechanisms are by ammonification of the organic nitrogen and by the uptake of the ammonia by plants and heterotrophic microorganisms. Burial of the organic nitrogen and phosphorus with the produced organic matter and adsorption of ammonia and phosphate onto the soil (clay) and organic particulates then represent the primary removal mechanisms for the nutrients (both nitrogen and phosphorus). Unless specifically managed to enhance nitrification (for example, by alternate draining and flooding of the wetland), it may be difficult to nitrify the ammoniacal nitrogen; hence, gasification of the nitrogen by denitrification may be minimal or greatly reduced. Expected total nitrogen removal could be around 40% to 60% on an annual basis, depending on the geographical location (see 3 below).

As stated before and shown on Figure 14.12, limited nitrification can occur in wetlands even when sediment and the sediment-water interfacial area are anaerobic. Under these circumstances, photosynthetic oxygen is supplied by plant roots and rhizomes that penetrate into the anaerobic zones and provide a limited aerobic environment for nitrification and nitrifying microorganisms. In such cases the depth of the anaerobic zone

and the type of plant can make a significant difference (Reed, Middlebrooks, and Crites, 1988; Hammer and Bastian, 1989).

3. If the wetland is located in northern climatic conditions, part of the nutrients accumulated in the substrate is released and the wetland may become a source of the nutrients (both nitrogen and phosphorus) during the dormant season.
4. Removal of phosphate is by assimilation into new organic matter and by adsorption on clay, iron, aluminum, and organic particles with subsequent burial. Phosphorus removal in many wetlands is not effective due to the limited opportunity of phosphates to interact with soils and other adsorbing media. If the pH is low, as it is in many peat wetlands (bogs and fens), the phosphate may precipitate. In this sense the mechanism of phosphate removal and retention by wetlands is similar to that of saturated soils.

From lake sediment studies it is known (Mortimer, 1971) that phosphate is released from the settled organic matter and sediments under anaerobic conditions in the sediments and sediment-water interface. If the sediment-water interface is aerobic, the release is to a great degree blocked. The efficiency of wetlands to remove phosphorus is generally lower than for nitrogen.

5. Tidal and marine marshes are less effective in their nutrient removal efficiencies than the freshwater wetlands. This has been reported to be due to salt stresses on the microorganisms (Mitsch and Gosselink, 1986). Tidel brackish marshes with a high variability of salt concentrations may exhibit the lowest removal efficiencies. For example, brackish marshes around Chesapeake Bay showed no nitrogen removal capability (Bender and Correl, 1974).
6. Management of wetlands for water quality control and pollution removal is very important, as are the type of vegetation and water depths. Precise guidelines for the control of these factors have not been developed and are now only emerging. With the exception of one wetland located in Michigan (Kadlec and Alvord, 1989) no long-term experience (ten years or more) has been gathered for wetland management for water-pollution control.

The following list contains typical design loadings and expected performance of natural wetlands receiving secondary effluents (after Reed, Middlebrooks, and Crites, 1988).

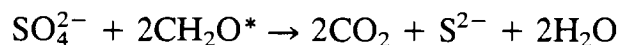
<i>Pretreatment needs</i>	<i>Secondary treatment</i>
Climatic conditions	Warm
Detention time, days	10
Depth, m	0.2–1
Hydraulic loading, m ³ /ha-day	100
Expected effluent quality	
BOD ₅ , mg/l	5–10
TKN, mg/l	5–10

Due to variations in storm-water characteristics and poor understanding of the wetland processes that remove pollutants under highly variable conditions, treatment efficiency predictions of a wetland storm-water system are not possible (Livingston, 1989).

Retention of Sulfur

Few studies are available that describe the retention and transformation of sulfur in wetlands, yet, from the water quality management viewpoint, sulfur may play a very significant role in the process of immobilization and detoxification of toxic metals. Sulfur transformation in wetlands is biologically mediated (Faulkner and Richardson, 1989) and is related to and governed by redox and pH interactions. Major transformations (shown on Fig. 14.17) are reduction of sulfate (SO₄⁻) to sulfide (S⁻) compounds in an anoxic environment, aerobic sulfate and elemental sulfur oxidation, and mineralization of organic sulfur to inorganic sulfate.

Sulfate reduction is accomplished by heterotrophic bacteria that use sulfate as an electron acceptor instead of oxygen. It should be pointed out that in the absence of oxygen bacteria preferentially use nitrate over sulfate; nevertheless, in wetlands an anoxic substrate environment, both processes can take place almost simultaneously. One mole of sulfate is reduced in the anoxic environment to sulfide for every two moles of organic carbon oxidized, as shown in the following reaction (Hedin, Hammock, and Hyman, 1989; DiToro et al., 1990)



where CH₂O* represents a simple organic molecule such as acetate. Depending on the chemical environment, hydrogen sulfide is released as gas, ionizes to HS⁻ and S⁻, or precipitates as a polysulfide, elemental sulfur, or iron sulfides. It is the metal sulfide form that immobilizes and detoxifies the toxic metals. The precipitation of iron sulfide is

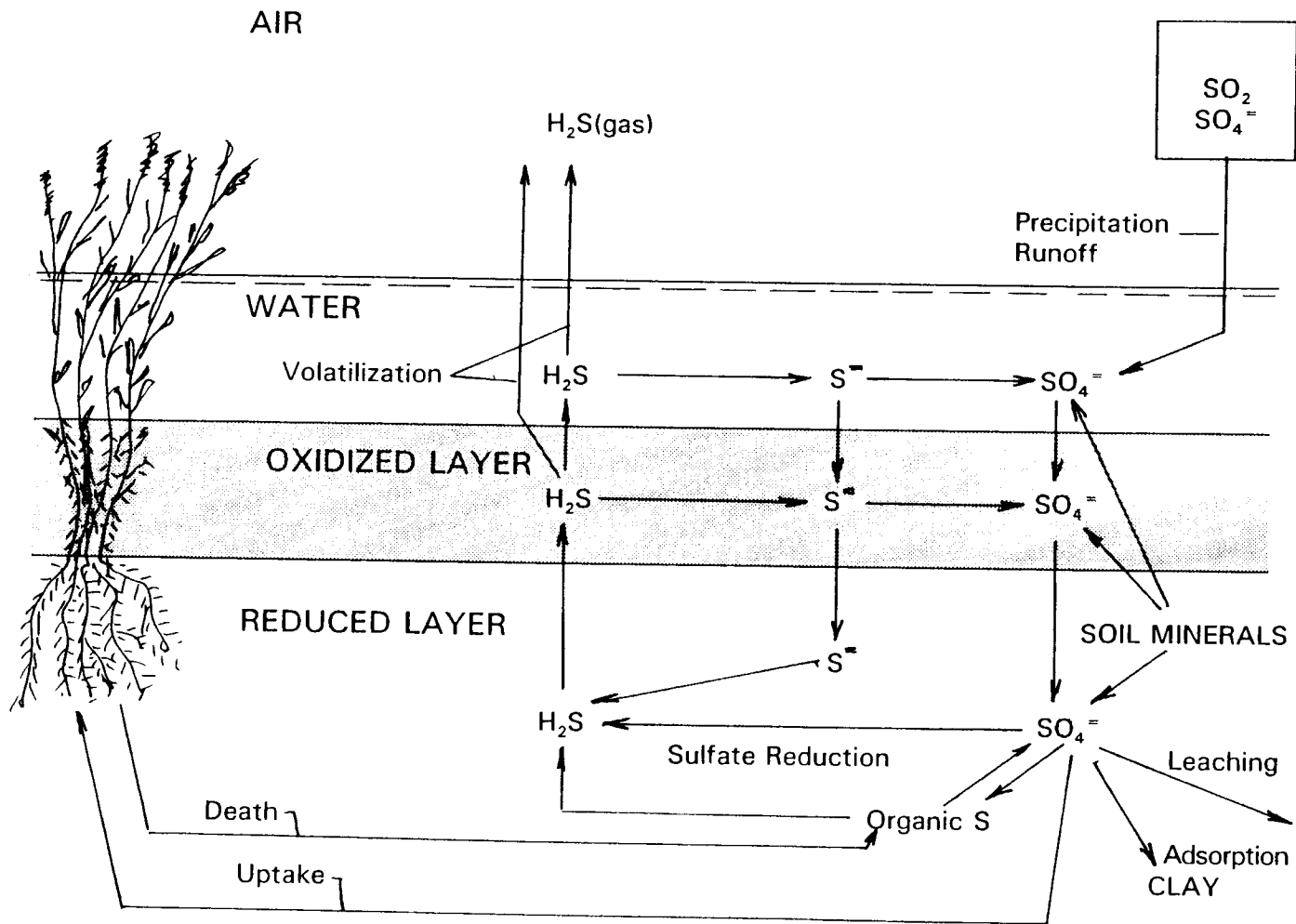
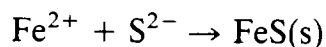


FIGURE 14.17. Sulphur transformation in wetlands. (After Mitsch and Gosselink, 1986.)

R0023860



The measure of this immobilization capability is the acid volatile sulfide (AVS) test (DiToro et al., 1990, 1991b). The iron sulfide precipitate is the major component of the AVS (DiToro et al., 1990). As was reported in the preceding chapter for each mole of AVS present in the substrate one mole of toxic metal is immobilized (complexed) and, therefore, detoxified.

Wetlands function as effective sulfur sinks. Sulfur inputs originate from the atmosphere (acid rainfall) or from water inputs and wastewater effluents, such as acid mine drainage, and from natural sources. Up to 87% sulfur removals have been reported; however, if the wetland substrate and overlying waters become aerobic subsequent oxidation of reduced S to SO_x^{2-} can release sulfur back to the water, from where the sulfur can be flushed from the system.

Measured sulfur removals in natural wetlands ranged from 20% to 90% at sulfur loading rates between 20 and 50 g of S/m²-year (Faulkner and Richardson, 1989). Long-term sulfur removals and retention by wetlands requires strongly reduced conditions to enhance SO_4^{--} reduction. Wetlands constructed to remove and store S (for example, those used for acid mine drainage disposal and/or from disposal and treatment of toxic metals containing wastewater and runoff) should be designed to promote such conditions.

Metal sulfide complexes (including potentially toxic metals) can be destroyed by acidic and aerobic conditions. Making the substrate aerobic will promote carbonic acid formation in pore water, which will reduce the pH, and concurrent oxidation of sulfides will reduce AVS to such levels that sulfides become ineffective for metal complexation. Under anaerobic conditions only very large acid inputs, such as those typical for overloaded constructed wetlands treating acid mine drainage may volatilize the sulfide-metal complexes. The most common cause of sulfide destruction is oxidation. When sulfides are oxidized and acid volatile sulfide concentrations become low, as they would be in fully aerobic substrate, then other properties of the substrate, such as organic matter and clays, will control the toxic metal activity (toxicity).

CONSTRUCTED WETLANDS

Constructed wetlands are man-made systems that imitate the functions of the natural systems; however, they provide a better chance for management and control. They can include swamps (wet regions dominated by trees, shrubs, and other woody vegetation) or bogs (low-nutrient, acidic

waters dominated by mosses). Most common constructed wetlands are marshes, which are shallow water bodies dominated by emergent herbaceous vegetation, such as cattails (*Typha*), bulrushes (*Scirpus*), rushes (*Juncus*), and reeds (*Phragmites*). As pointed out by Hammer (1989b), marshes can be adapted to a tremendous variety of soils, climatic conditions, and to wide fluctuations of water quality and hydrological conditions.

Unlike the overland-flow wastewater treatment systems where vegetation is not essential (for example, in rapid infiltration systems briefly introduced in Chapter 6), vegetation in wetland systems is the predominant purification medium. The principal function of vegetation in created (and natural) wetland systems is to provide an additional environment for microbial populations, in addition to slowing down the flow and filtering the solids. The plants provide a substantial amount of reactive surfaces for microbes. As pointed out plants also increase the amount of aerobic microbial environments in the substrate due to a unique ability of wetland plants to transfer air to their root and rhizomes.

Due to the previously mentioned limitation imposed on natural wetlands by regulations and by their function, constructed wetlands are more amenable to be used as a treatment facility and can be operated and managed more efficiently. Essentially, constructed wetlands retain the positive characteristics of natural wetlands and, with proper management and some maintenance, can minimize the drawbacks. Constructed wetlands can be and have been used for the following water-management and pollution-control tasks:

- Disposal of treated or partially treated municipal effluents (Kadlec, 1988; Kadlec and Alvord, 1989; U.S. EPA, 1988; Mitsch, 1990).
- Wastewater polishing and renovation (Reed, Middlebrooks, and Crites, 1988; U.S. EPA, 1988).
- Urban and suburban storm-water management and treatment (Palmer and Hunt, 1989; Esry and Cairns, 1989; Livingston, 1989; McArthur, 1989; Mitsch, 1990).
- River water quality restoration and nonpoint pollution control (Hey, 1988; Hey et al., 1989).
- Mine acid drainage disposal (Reed, Middlebrooks, and Crites, 1988; Brodie et al., 1989a).
- Disposal of some industrial wastes, such as ash pond seepage (Brodie et al., 1989b), refinery wastewater (Lichfield and Schutz, 1989), and pulp mill effluents (Thut, 1989).
- Flood control (Mitsch, 1990).
- Detoxification of wastewaters and runoff containing toxic metals and adsorbable organic chemicals.

Unlike natural wetlands that are limited by location, soils, and other factors, constructed wetlands can be built almost anywhere, including lands that would have limited use. Conversion of marginal agricultural lands, or lands that were taken out of production under the farm conservation programs, to riparian wetlands is feasible and an attractive alternative for control of pollution from diffuse (nonpoint) sources. They also provide superior performance and flexibility.

Types of Constructed Wetlands

The EPA (U.S. EPA, 1988) and Water Pollution Control Federation (WPCF) (1990) manuals and other treatises on the design of constructed wetlands (Reed, Middlebrooks, and Crites, 1988; Hammer, 1989a; Kusler and Kentula, 1989; Mitsch, 1990) recognize two fundamental types of constructed wetlands.

Free water surface (FWS) system with emergent plants: A FWS system (Fig. 14.18) typically consists of basins or channels with a natural or constructed subsurface barrier of clay or impervious geotechnical material (lining) to prevent seepage. The basins are then filled with soils to support emergent vegetation. Water flows slowly over the soil surface. Figure 14.19 shows an outline of a typical storm-water pond-wetland system recommended by the Maryland Department of Natural Resources for urban storm-water quality control (Livingston, 1989; Mitsch, 1990).

If the soil is brought from an existing wetland, wetland vegetation will emerge without seeding; however, seeding and planting of vegetation are a part of the construction process. Mitsch (1990) referred to the former type of constructed wetland as a *self-designed wetland*, and to the latter as a *designer wetland*. To develop a wetland that is ultimately a low maintenance one, the natural successional process needs to be allowed to proceed. Often this may mean some initial period of invasion by undesirable species, but if proper hydrologic and nutrient loads are maintained, this invasion is usually temporary.

Subsurface flows systems (SFS) with emergent vegetation: An SFS wetland, also known as vegetated submerged bed (VSB), consists of a trench or bed underlaid with an impermeable layer of clay or synthetic material. As shown in Figure 14.20, the system is built with a slight inclination of 1% to 3% between the inlet and outlet. The media used include crushed stone or rock fill, gravel, and different soils. The water flows through the medium and is purified by filtration, absorption by microorganisms and adsorption onto soils and organic matter, and by the roots of the plants. The subsurface medium is generally anaerobic; however, as specified previously, plants can convey by their roots and rhizomes some oxygen into the subsurface layer, thus creating therein small aerobic pockets.

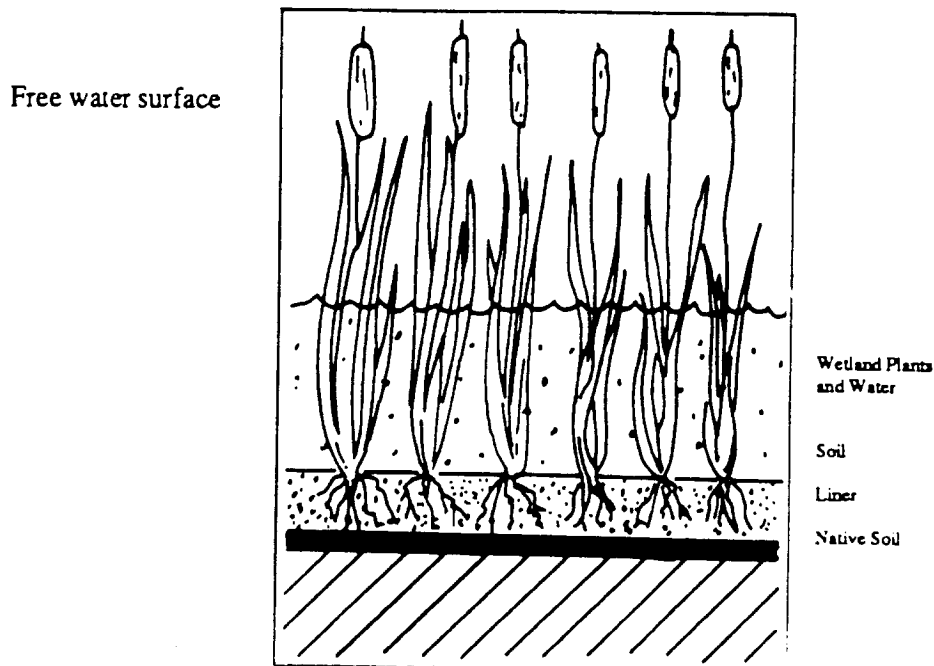


FIGURE 14.18. Free water surface wetland schematics. Water level is above the surface; vegetation is rooted and emergent above the water surface; water flow is primarily above ground; vegetation may be planted or allowed to colonize voluntarily.

As pointed out by Reed, Middlebrooks, and Crites (1988), both wetland-type treatment systems are “attached-growth” biological reactors, comparable in function to trickling filters and rotating biological contactors. Hence, their performance depends on the detention time of the waste in the system, the loading rates and the condition of the biota that develops within the system, and on the oxygen availability. If denitrification is an objective, alternate aerobic–anaerobic conditions should be promoted within the system and the anaerobic–denitrification zones should have an ample supply of organic carbon, which can be provided by primary productivity and from organic residues of plants grown in the wetland.

Subsurface-flow (SFS) constructed wetlands have been used extensively in Europe under the name *root-zone method*, *hydrobotanical system*, *soil-filter trench*, *biological macrophytic* and *marsh beds* (Reed, Middlebrooks, and Crites, 1988) or *vegetated submerged bed* (VSB) systems (Water Pollution Control Federation, 1990). The root-zone method was developed in Germany. The reed-bed system (Cooper and Hobson, 1989) used in the United Kingdom evolved from the German system. The performances of some constructed wetland systems in the United States and Canada are shown in Table 14.4.

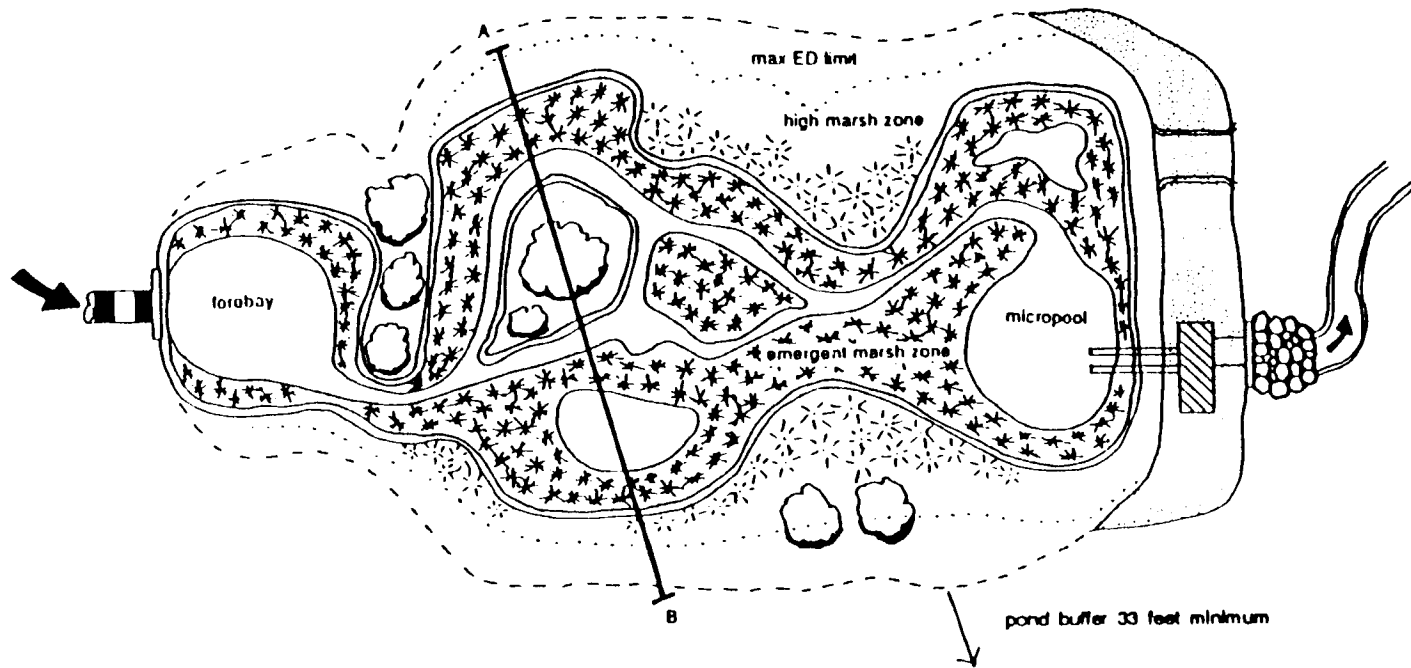


FIGURE 14.19. Maryland pond-wetland system. (After Schueler, 1992.)

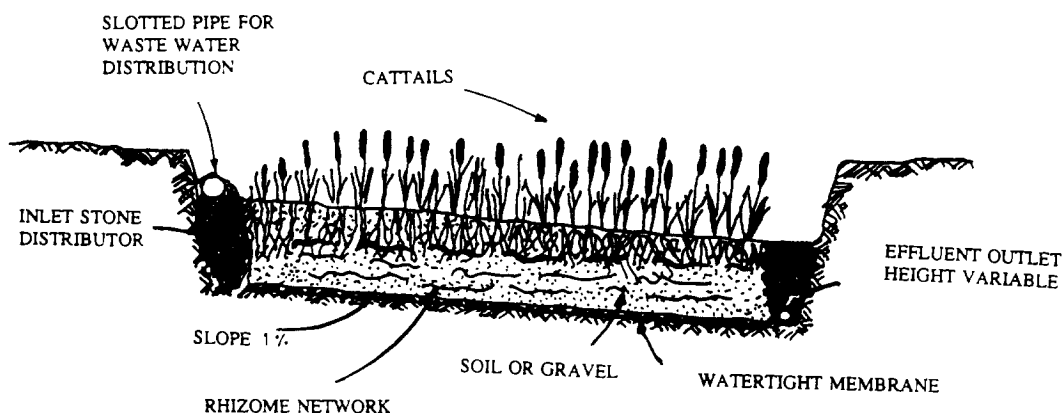


FIGURE 14.20. Submerged flow system (SFS). (After U.S. EPA, 1988.)

Nitrogen and phosphorus removal efficiencies related to loading rates were published recently in a comprehensive treatise by Hammer (1989a). The removal efficiencies related to nutrient loading rates are shown on Figures 14.15 and 14.16.

Design Considerations and Parameters for Constructed Wetlands

Mitsch and Jørgensen (1989) pointed out that ecological engineering insists on taking advantage of our ever-increasing knowledge of ecology and its principles (for example, succession, energy flow, self-design) in order to design a system that will be as close to the natural features of the landscape as possible and that will require a minimum amount of maintenance. This means resisting the ever-present temptation to overengineer.

The basic principles of ecological engineering of wetland restoration

TABLE 14.4 Performance of Pilot-Scale Constructed Wetlands

Location	Wetland Type	Effluent Concentration (mg/l)					
		BOD	SS	NH ₄ ⁺	NO ₃ ⁻	Total N	Total P
Listowel, Ontario, Canada	Open water, channel	10	8	6	0.2	8.9	0.6
Arcata, California	Open water, channel	<20	>8	<10	0.7	11.6	6.1
Santee, California	Gravel-filled channels	<30	<8	<5	<0.2	—	—
Vermontville, Michigan	Seepage basin wetland	—	—	2	1.2	6.2	2.1

Source: From Read, Middlebrooks, and Crites (1988).

and creation were outlined by Mitsch (1990), Mitsch and Jørgensen (1989), and Odum (1989) as follows.

1. Design the system for minimum maintenance. The system of plants, animals, microbes, substrate, and water flows should be developed for self-maintenance and self-design (Mitsch and Jørgensen, 1989; Odum, 1989).
2. Design a system that utilizes natural energies, such as the potential energy of streams.
3. Consider the landscape when developing a system. The best sites are locations where wetlands previously existed or where nearby wetlands still exist. Do not overengineer wetland designs with structures, unnatural shapes of basins, uniform depths, and regular morphology. Ecological engineering promotes mimicking nature.
4. Design a system as an ecotone. This means including a buffer strip around the site, but also means that the wetlands site itself needs to be viewed as a buffer between the upland and the aquatic system to be protected.
5. Consider the surrounding lands and future land-use changes. For example, in agricultural zones planned idling of land for soil-erosion control may obviate the need for a wetland to control pollution by nonpoint runoff.
6. Hydrologic conditions are paramount. Without water for at least a part of the growing season, a wetland is impossible. A detailed surface and ground-water study is therefore necessary.
7. Give the system time. Wetlands are not created overnight.
8. The soils should be surveyed. Highly permeable soils do not support viable wetland systems.
9. The chemical composition of feed waters, including ground-water discharge, can be significant to wetland productivity and/or bioaccumulation of toxic materials.
10. Riparian wetlands present a particular problem because their flooding also causes scouring, sediment shifts, erosion, and deposition. Convex sides of river channels may be preferable to concave sides because of the higher erosive forces on the latter.

Other factors to be considered during design include site accessibility (site access should be controlled to avoid vandalism), land ownership, land prices and water-use rights (if water is to be diverted from the stream onto the wetland), and wildlife and fishery considerations (water fowl migratory routes, fish spawning areas). Few design parameters and limited experience are available for constructed wetlands used for diffuse-

TABLE 14.5 Wetland Design Parameters

Design Considerations	Constructed		
	Free Water Surface (FWS)	Submerged Bed Systems (SFS)	Natural
Minimum size requirement ha/1000 m ³ /d	2-4	1.2-1.7	5-10
Hydraulic loading, ^a cm/day	2.5-5	5.8-8.3	1-2
Maximum water depth, cm	50	Water level below ground surface	50; depends on native vegetation
Bed depth, cm	NA	30-90	NA
Minimum aspect ratio	2:1	NA	1:4
Minimum hydraulic residence time, days	5-10	5-10	14
Minimum pretreatment	Primary; secondary is optional	Primary	Primary; secondary; nitrification; TP reduction
Configuration	Multiple cells in parallel and series	Multiple beds in parallel	Multiple discharge sites
Distribution	Swale, perforated pipe	Inlet zone (>0.5 m wide) of large gravel	Swale, perforated pipe
Maximum loading, kg/ha-day			
BOD ₅ ^b	100-110	80-120	4
Suspended solids	up to 150		
TKN	10-60	10-60 up to 15	3
Phosphorus	?		0.3-0.4
Additional considerations	Mosquito control with mosquitofish; remove vegetation	Allow flooding capability for weed control	Natural hydroperiod should be >50%; no vegetation harvest

Sources: After Water Pollution Control Federation (1990; copyright Water Environment Federation) and Mitsch (1990).

^aHydraulic loading is a reciprocal of the minimum-size requirement after conversion.

^bApplicable only to wastewater treatment systems.

pollution control. The data in Table 14.5 contain typical design parameters for constructed wetlands treating wastewater effluents.

Hydrology

Important hydrologic factors and parameters in the design of constructed (restored) wetlands include hydroperiod, hydraulic loading rate (hydro-

logic size requirement), hydraulic residence time, infiltration capacity, and the overall balance (Water Pollution Control Federation, 1990; Mitsch, 1990). The hydrologic factors that affect these parameters are dependent on climate and the seasonal patterns of stream flow and runoff (both surface and subsurface) feeding the wetland. D'Avanzo (1989) and Mitsch (1990) have pointed out that is improper hydrology that leads to most failures of created wetland systems. On the other hand, if hydrology is favorable, chemical and biological conditions will also have a favorable and healthy response. Hydrology is less self-correcting and forgiving than the ecology of the wetland (plants and animals), which may self-correct initial mistakes (Mitsch, 1990).

Hydroperiod is defined as the depth of water over time. This parameter is important mostly for natural wetlands. Typical hydroperiods are shown on Figure 14.21. In addition to depth the key component of the hydroperiod factor is the duration of flooding. One has to realize that soil chemistry during flooding turns rapidly anaerobic and that each plant has different specific capabilities to adapt and survive in low-oxygen soils. Wetlands with variable depths have the most potential for developing a diversity of plant and animal species. Table 14.6 summarizes hydroperiod tolerance ranges for a variety of wetland plants and species. This information can be used in checking appropriate native wetland habitats for general suitability and for the design of constructed systems (Water Pollution Control Federation, 1990).

Alternate flooding and aeration of solids promotes nitrification-denitrification. Deepwater areas, devoid of emerging vegetation, offer habitats for fish (for example, *Gambusia affinis*, the mosquito eating fish). Water levels can be controlled by inflow and outflow structures, including weirs and feed pumps.

Several states have developed guidelines for bottom profiles of wetlands to provide a variety of depths (remember the third rule of ecological engineering specified previously). For example, Florida regulations on wetlands require a littoral shelf with gently sloping sides of 6:1 or flatter extending out from a point 60–77 cm below the water surface. They further recommend less than 70% of open water surface. A mean depth of 1–3 meters is recommended for permanent pools. Maryland's regulations for wetlands used for storm-water treatment specify that 75% of the wetland should have a depth under 30 cm (50% less than or equal to 15 cm and 25% 15–20 cm; 25% should have a depth ranging from 60 cm to 100 cm).

Infiltration capacity and soil texture. As pointed out previously, highly permeable soils are not suitable for wetland creation. As a matter of fact, most natural wetlands resulted because of poor soil permeability conditions. Consequently, permeability (infiltration rates) must be kept

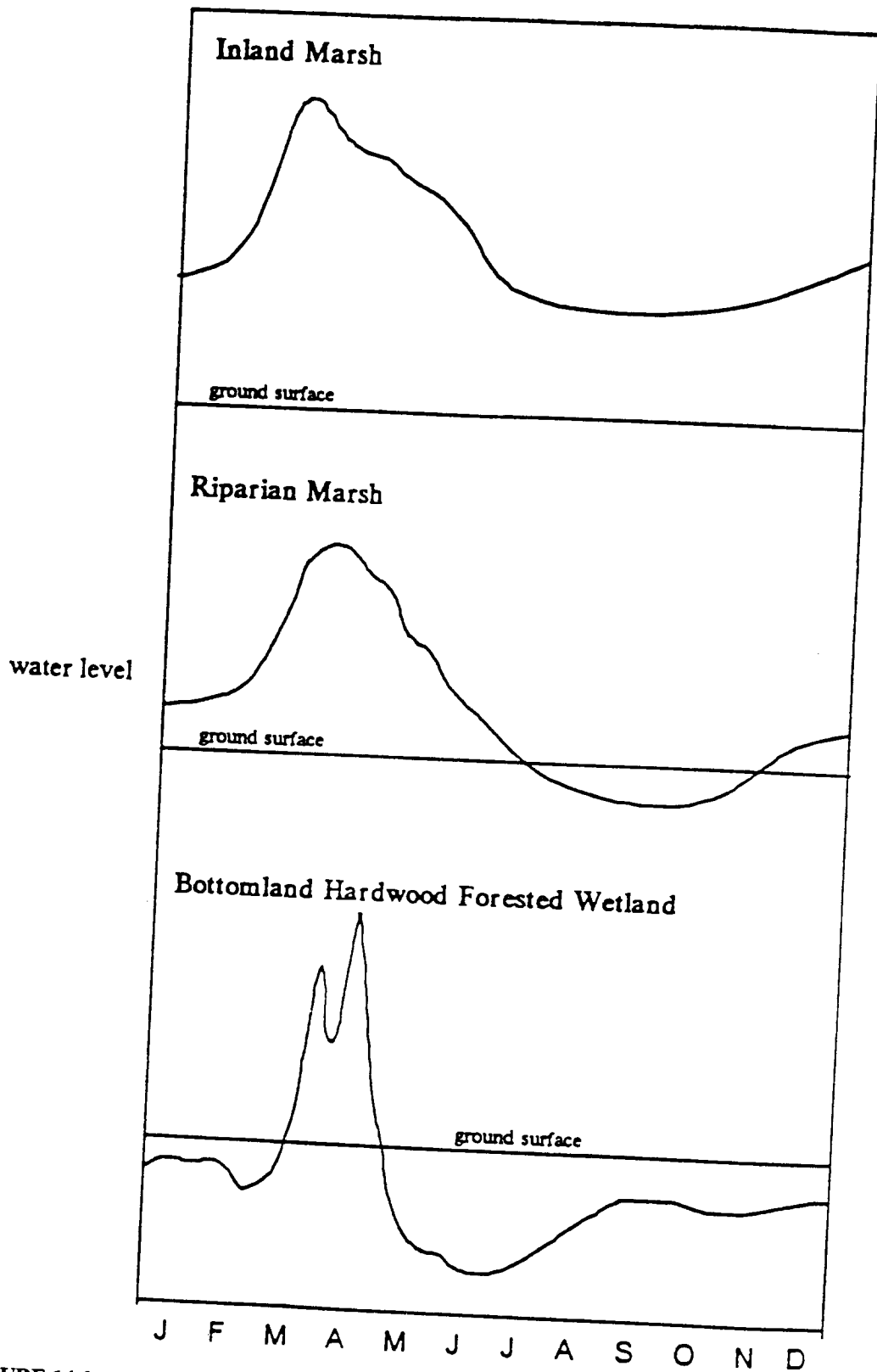


FIGURE 14.21. Typical hydroperiods for inland marsh, riparian marsh, and bottomland hardwood forested wetlands. (After Mitsch, 1990.)

TABLE 14.6 General Hydroperiod Tolerance Ranges for Selected Wetland Plant Communities

Wetland Type	Typical Species	Average Water Depth (m)	Average Hydroperiod (%)
Floating deep	Hyacinths, pennywort		
Floating rooted aquatic	Water lily, water dock, water shield	0.5-2	70-100
Submerged aquatic	Hydrills, egeria, water millfoil, sagittaria, naiad	0.5-3.0	80-100
Emergent marsh	Cattails, pickerelweed, sawgrass, bulrush, sedge, maidencane	0.1-1	40-100
Floodplain	Red maple, black gum, cabbage palm, pond cypress, oaks, pines, bald cypress, ash	0.2-0.3	10-50
Swamp forest	Bald cypress, ash, black gum, tupelo, gum, red maple	0.3-1	50-80
Cypress dome	Pond cypress, red maple, black gum, Dahoon holly	0.1-0.3	50-75
Wet prairie	St. Johns wort iris, sagittaria	0.1-0.2	20-50
Oak, palm, hammock	Oak, cabbage palm	<0.1	5-10

Source: From Water Pollution Control Federation (1990), copyright Water Environment Federation.

below a certain threshold value, which obviously may vary depending on local site-specific and geographic conditions. Measured infiltration rates for natural wetlands range from zero for a fen in Michigan (Haag, 1979) to about 0.5 cm/day (Water Pollution Control Federation, 1990).

The maximum substrate (soil) infiltration rate for wetlands should be around 1 mm/hr (Water Pollution Control Federation, 1990). This rate essentially implies a substrate or subsoil rich in clay. High clay content of the active substrate may limit root and rhizome penetration and may be impermeable for plant roots. Therefore the use of local coarser texture (loam) soils underlain by an impermeable clay layer is often the best design (Mitsch, 1990).

Hydraulic loading rates. Table 14.5 lists the suggested hydraulic loading rates (HLR) that are commonly given as the depth of water per area per time, or

$$HLR = Q/A \tag{14.1}$$

An inverse of HRL is the area requirement per unit flow. Most of what is known about the hydraulic loading rates has been gathered from observations of wetlands receiving wastewater. Mitsch (1990) pointed out

that the hydraulic loading rates used for wetlands treating wastewater would be too low for riparian wetlands used for runoff and stream quality control. The maximum hydroperiod tolerance range of the wetland plant community must also be considered when a suitable HRT is being determined.

Retention time. Table 14.5 also provides the optimum retention time of water in the wetland. The retention time for free water surface (FWS) systems can be calculated from a simple formula that considers the water volume of the wetland and average flow, or

$$\text{HRT} = pV/Q \quad (14.2)$$

where

HRT = hydraulic residence time, days

$V = W * H * L$ = active volume of the wetland, m^3

p = porosity or (water volume)/(total volume) ratio

$p = 0.9$ to 1 for free water surface wetlands depending on the growth density of vegetation, and

p = void fraction of the substrate for subsurface flow systems (SFS)

W = width of the system, m

H = average depth of the system, m

L = length of the wetland (m) = $W * \text{AR}$

AR = aspect ratio (length/width)

Q = average flow, m^3/day

A = surface area, m^2

Alternatively, the hydraulic residence time for SFS systems may be calculated from the Darcy's law as

$$\text{HRT} = p \frac{V}{Q} = \frac{L}{K_p S} \quad (14.3)$$

where

L = length of the bed in meters

K_p = hydraulic (saturated) permeability of the wetland, m/day

S = bed slope, m/m

Normal design bed depths for SFS systems are between 0.6 and 0.9 meters (Water Pollution Control Federation, 1990). For time-variant intermittent diffuse sources, the HRT concept makes sense only if the hydraulic load is average over a longer time period (season or year). A

simple relationship between the hydraulic loading rate (HLR), porosity, p , depth, H , and hydraulic residence time is $HLR = p \times H/HRT$.

System Configuration

The length–width ratio (also called the *aspect ratio* (AR)) for wetlands treating wastewater should be at least 4:1 to 6:1 (Water Pollution Control Federation, 1990). However, Mitsch (1990) pointed out that more elongated wetland cells (with an aspect ratio of at least 10:1) are needed in riparian settings and/or when receiving larger flows such as storm-water runoff.

In constructing a wetland for treatment purposes, the wetlands are arranged in cells with or without ponds. Cells can be parallel so that alternate drawdown management can be practiced to enhance nitrification, oxidation, mosquito control, and to maintain optimum hydroperiods for a variety of plants. The ponds remove pollutants and assist in the operation of the system. In an SFS system the ponds can interrupt short circuiting and reestablish uniform flow through the cells. They can also help in mosquito control by providing a living environment for *Gambusia*, which can migrate from the pond into the shallow wetlands.

Generally, the components of a wetland system include a treatment or pretreatment unit (note that it is neither advisable nor feasible to discharge raw untreated wastes into the wetland system), the wetland cells, and ponds. Pretreatment units may not be needed if wetlands are used for storm-water management. In such cases, natural (swale) drainage can provide enough filtering and pretreatment of storm water.

Possible design configurations of constructed wetlands were included in a paper by Steiner and Freeman (1989) and are shown on Figure 14.22. Mitsch (1990) presented nine actual design of wetlands located throughout the United States. Figures 14.23 to 14.27 show possible design alternatives of constructed or restored riparian wetlands.

Constituent Loadings

For wetlands treating wastewater the organic load expressed as kilograms of BOD_5 /ha-day (or a similar unit such as lb/acre/day or g/m^2 -day) is of major concern. Wetlands are extremely efficient at assimilating the BOD and also suspended solids loads. However, the organic load from allochthonous sources may be small when compared to the primary productivity (autochthonous organic matter production) of the wetland itself. For wetlands treating nonpoint pollution the organic loading is of less importance, and the parameters of concern are suspended solids, nutrients, and toxic components.

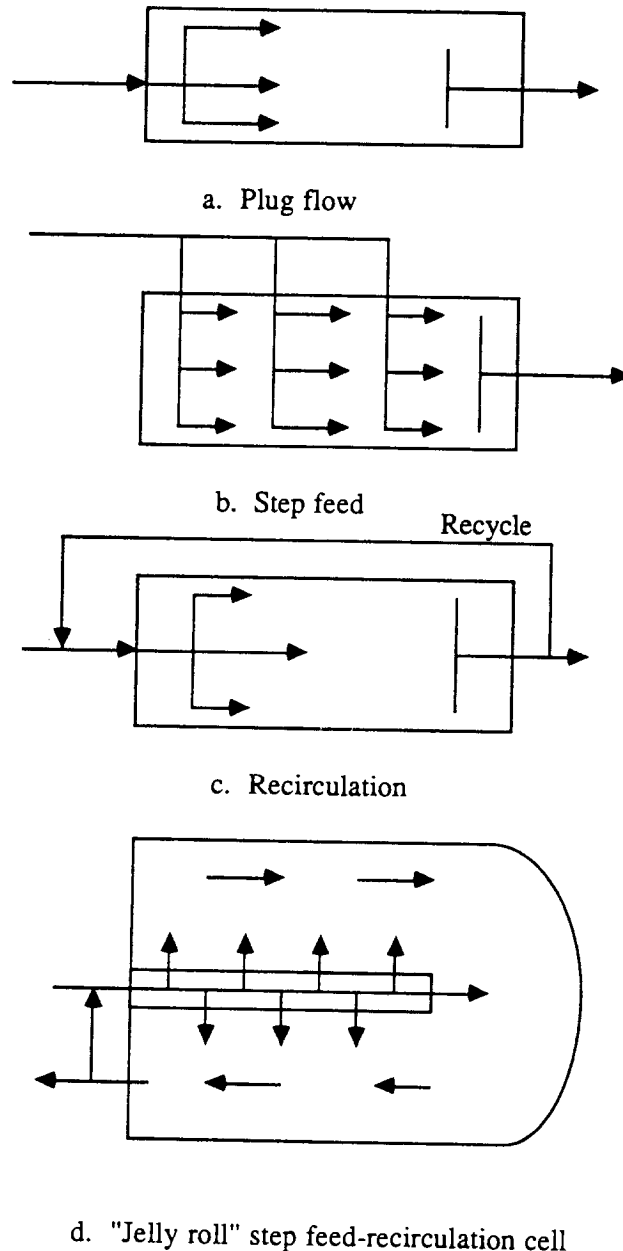


FIGURE 14.22. Basic flow patterns and configurations for constructed wetlands. (After Steiner and Freeman, 1989, with permission from Lewis Publishers, a subsidiary of CRC Press, Boca Raton, Florida.)

Suspended solids. Very high SS removals have been noticed in constructed and natural wetlands. The removal efficiency for suspended solids is inversely proportional to influent SS concentrations and the surface area of the wetland and the detention time, which is consistent. As pointed out previously solids are removed by settling, filtration, and adsorption. Of

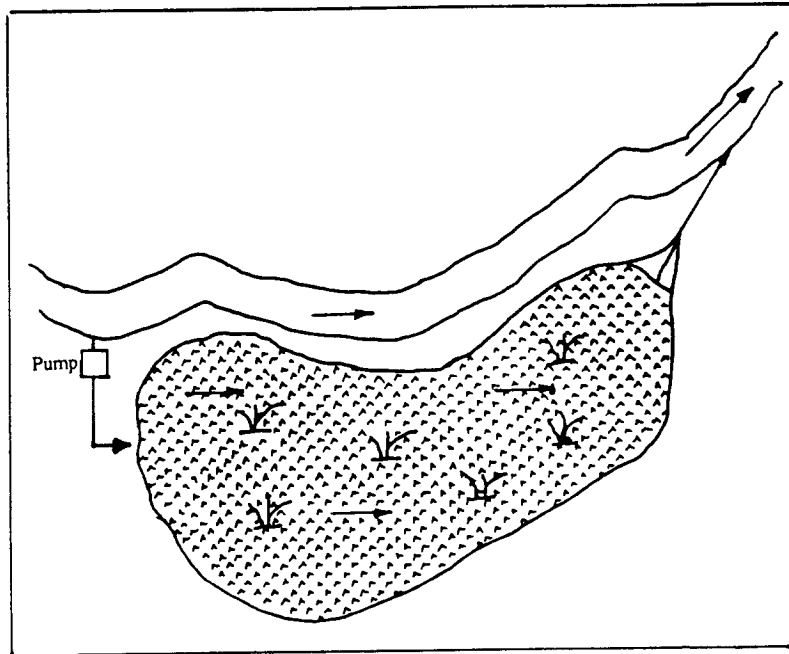


FIGURE 14.23. Riparian wetland fed by pump. (After Mitsch, 1990.)

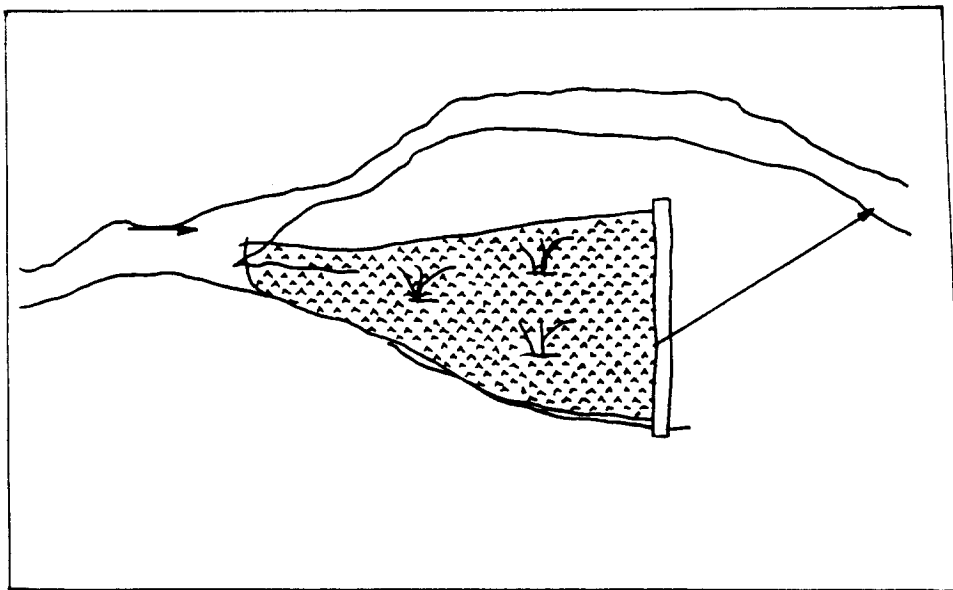


FIGURE 14.24. Wetland built by incorporating diversion. (After Mitsch, 1990.)

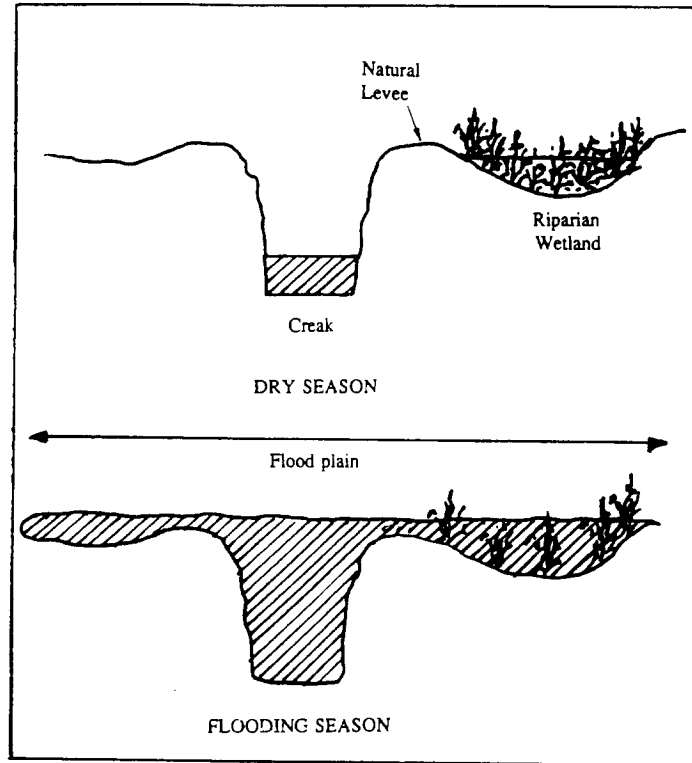


FIGURE 14.25. Riparian wetland design fed by natural flooding over the stream banks. (After Mitsch, 1990.)

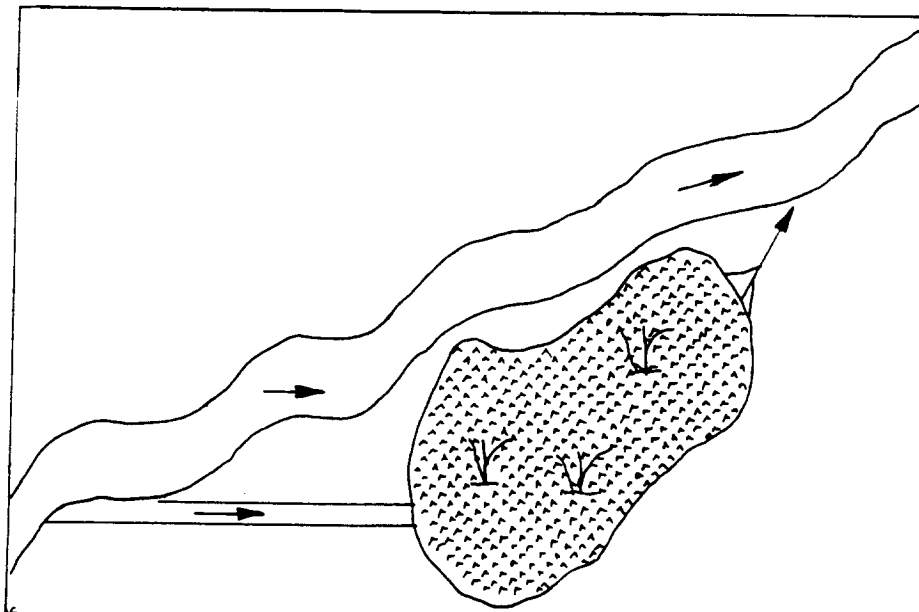


FIGURE 14.26. Riparian wetland design fed by gravity flow. (After Mitsch, 1990.)

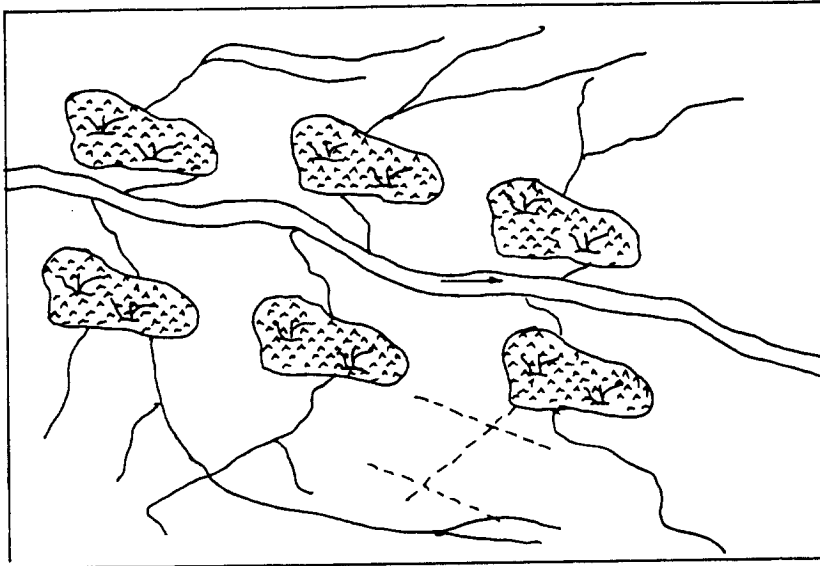


FIGURE 14.27. Schematics of small wetlands intercepting small polluted tributaries of a stream. (After Mitsch, 1990.)

concern are extremely high loads that could clog up the ecosystem by clay and silt particles. The critical HRT for achieving TSS removal efficiencies above 70% is about 5 days.

The operational experience gathered at the Des Plaines, Illinois, experimental wetland shows that while the influent suspended solids were primarily composed of soil particles in concentrations of tens to hundreds at milligrams/liter, effluent solids primarily included organic solids produced by the wetland, with effluent concentrations of only a few milligrams/liter. The influent is typically turbid, while the effluent color is from the humic substances of the wetland. The effluent character is comparable to native prairie streams.

Biodegradable organics. If the constructed wetland is used for the treatment of wastewater, or if the diffuse input contains appreciable concentrations of biodegradable organics (such as flows from concentrated animal operations), BOD removals may be considered. For FWS and SFS wetlands BOD removal may be described by a plug flow equation of the type

$$\frac{C_e}{C_o} = \exp(-k_T \text{HRT}) \tag{14.4}$$

where

C_e = effluent BOD₅ (mg/l or mass load in kg/day)

C_o = influent BOD₅ (mg/l or kg/day)

k_T = temperature-dependent first-order BOD removal coefficient, day⁻¹

The temperature-dependent BOD removal coefficient (equivalent in concept to the deoxygenation coefficient in streams; see the preceding chapter) is commonly reported at a reference temperature of 20°C. Equation (12.11) with the thermal factor $\theta = 1.06$, may be used to adjust the coefficient for other design temperature (Water Pollution Control Federation, 1990). Most of the available data on BOD removals are for municipal wastewater and may not be applicable to diffuse loads from agricultural or urban sources.

Nutrients. Both nitrogen and phosphorus are necessary nutrients for wetland biota growth and are retained by it. The process of nitrogen and phosphorus assimilation was discussed previously. It was also pointed out that nitrogen assimilation depends on the form of the nitrogen in the input with the nitrate nitrogen resulting in the highest removal efficiencies.

TKN (total Kjeldahl nitrogen = organic N plus $\text{NH}_4^+ - \text{N}$) removal is correlated to loading. Loads as high as 10 kg/ha-day still provide removal efficiencies between 75% and 95%. With loading rates within the 10 to 80 kg/ha-day range, TKN removal rates are highly dependent on HRT and decrease significantly at design HRTs of less than 5 days. At HRTs of 5 days or more nitrate nitrogen in the inflow is almost completely assimilated by denitrification anaerobic processes occurring in the wetlands.

Based on a regression analysis of available data from the operation of natural and constructed wetlands, the Water Pollution Control Federation (1990) manual reported the following equation relating effluent and influent total nitrogen (TKN = organic + ammoniacal) concentrations:

$$C_e = 0.193 \times C_o + 1.55 \text{ HLR} - 1.75 \quad (14.5)$$

The correlation coefficient for this relationship is $r^2 = 0.79$.

Phosphorus removals are not as high as those for nitrogen. A compilation of wetland systems contained in the Water Pollution Control Federation (1990) manual revealed phosphorus removal efficiencies between 30% and 50% for most of the created systems studied. Total phosphorus removal efficiency increases with higher input concentrations, form of the phosphorus (particulate adsorbed onto the sediment vs. dissolved), phosphorus partitioning between the dissolved and suspended forms (which is related to the concentration of suspended solids) and the subsequent burial of the adsorbed phosphates with the sediment and organic matter in the substrate, oxic-anoxic conditions in the water-substrate interface, and on the hydraulic retention time. In colder climatic conditions, nutrients are released during the dormant season. Nitrogen and phosphorus removals for natural wetlands were shown on Figures 14.15 and 14.16.

Example 14.1: Free Water Surface Wetland Design

Design a free water surface (FWS) wetland for treatment of agricultural drainage that has average total nitrogen concentrations of 15 mg/l. Most of the nitrogen is either organic or ammoniacal. The desired effluent concentration is about 5 mg/l (66% reduction). The design drainage flow is about $0.1 \text{ m}^3/\text{sec} = 8640 \text{ m}^3/\text{day}$. Select a shallow active depth of water in the wetland (0.4 meters) to provide for nitrification–denitrification.

Solution From Equation (14.5) the hydraulic loading rate is

$$\begin{aligned} \text{HLR} &= \frac{C_e - 0.193C_o + 1.75}{1.55} = \frac{5 - 0.193 \times 15 + 1.75}{1.55} \\ &= 2.48 \text{ cm/day} = 0.025 \text{ m/day} \end{aligned}$$

The corresponding surface is then

$$A = Q/\text{HLR} = (8640 [\text{m}^3/\text{day}]) / (0.025 [\text{m}/\text{day}]) = 345,600 \text{ m}^2 = 35 \text{ ha}$$

As a rule of thumb, the Water Pollution Control Federation manual recommends that to attain total nitrogen removals of 50% or greater, the designer should use a minimum constructed area of wetland of about 4 ha/1000 m^3/day of flow.

Assuming the average depth of 0.4 meter, the hydraulic residence time in the wetland is $\text{HRT} = p \times H/\text{HLR} = 0.9 \times 0.4/0.025 = 14.4$ days. This detention time will provide for nitrification–denitrification nitrogen removal processes in the wetland (HRT should be greater than 7 days).

With a selected aspect ratio (based on the terrain configuration) of 4:1, the width of the wetland is $W = \sqrt{A^2/(1 + 4)} = \sqrt{350,000/5} = 264$ m and the length of the wetland is $350,000/264 = 1326$ m. At least four cells are recommended in two parallel series. The first cell may be shallower FWS (nitrification), the second deeper FWS or SBS (denitrification).

Note: This design is based on the assumption that most of the nitrogen is in ammoniacal and organic forms. If nitrate nitrogen is the dominating compound, the wetland area and volume may be smaller, as shown in the following example.

Example 14.2: Wetland Design for Nitrate-N Removal

Experimental data indicate that 95% removal may be achieved if the loading rate of nitrate is about 10 kg/day or less, while if the loading

exceeds 80 kg/ha-day the wetland nitrate removal efficiency is marginal. By interpolation, at between 10 and 80 kg of nitrate N/ha-day, 66% removal may be achieved (very approximately) using a loading rate of about 33 kg/ha-day. Select 25 kg/ha-day for safety reasons.

Solution The daily nitrate load is then

$$\begin{aligned}\text{Load} &= Q \times C = 8640 \text{ m}^3/\text{day} \times 15 \text{ mg/l (g/m}^3) \times 0.001 \text{ kg/g} \\ &= 129.6 \text{ kg/day}\end{aligned}$$

and the surface area becomes

$$A = 129.6 [\text{kg/day}] / 25 [\text{kg/ha-day}] = 5.2 \text{ ha} - \text{select } 6 \text{ ha} = 60,000 \text{ m}^2$$

The hydraulic loading is then

$$\text{HLR} = Q/A = 8640 [\text{m}^3/\text{day}] / 60,000 [\text{m}^2] = 0.14 \text{ m/day}$$

which is high (Table 14.5). Select an HLR of 8 cm/day = 0.08 m/day. Then the adjusted surface area is

$$A_{\text{adjusted}} = Q/\text{HLR} = 8640/0.08 = 108,000 \text{ m}^2 = 10.8 \text{ ha}$$

Since the nitrogen is in nitrate form, the dominant component removal process should be anoxic (anaerobic) to assure denitrification. In this case, either deeper FWS ($H \approx 0.9 \text{ m}$) or saturated SFS systems should be selected, and the hydraulic retention time should be above 5 days.

The hydraulic residence time requirement gives the estimate of the depth of the bed. Select a minimal HRT of 5 days. Then the required volume of the bed is

$$V = Q \times \text{HRT}/p = 8640 \times 5/0.5 = 86,400 \text{ m}^3$$

and the required depth of the bed becomes

$$H = V/A = 86,400/108,000 = 0.8 \text{ m}$$

If the hydraulic conductivity of the bed is known, one can also estimate the velocity in the bed. A judgment may then be made as to whether the flow will be fully submerged (in the bed) or whether ponding will occur. Say that the hydraulic conductivity of the bed is 10 m/day and the slope of the bed is 5%. Then the velocity in the bed is (from Darcy's equation)

$$v = K_p \times S = 10[\text{m/day}] \times 0.05 = 0.5 \text{ m/day}$$

This velocity is too low to accommodate the flow solely in the bed. The designer would therefore have to design the wetland as a combination of the FWS and SFS systems.

Metals and Toxics

Due to very high concentrations of organic matter in the substrate, wetlands could conceivably have a higher assimilative capacity for toxics and metals. As with aquatic sediment conditions, toxicity, adsorption (immobilization), and the burial of toxic compounds depend on the substrate characteristics and adsorption-desorption and the precipitation processes occurring in the substrate. DiToro (1989) and DiToro et al. (1990) pointed out that the toxicity of metals and organics correlates to their activity, which is more-or-less related to the dissolved and ionic concentration of the constituent in the pore water of the substrate.

In the reduced environment of the wetland substrate, sulfide precipitation of metals is the predominant process of inactivation of toxic metals, while the organic particulate matter content of the substrate controls immobilization of nonionic organic compounds (PCBs, DDT, PAHs, many pesticides) and organic mercury (DiToro et al., 1990). The assimilative capacity for organic chemicals is also related to the chemical's biodegradability and volatility. However, models for estimating the assimilative capacity of wetlands for toxic compounds are not available and the estimates are crude, at best. One may speculate, however, that if constructed wetlands are to be used for the disposal of small quantities of toxic compounds, the preferable design would be SFS, which provides higher concentrations of organic matter in the bed as compared to concentrations of particulate organic matter in the water in FWS systems.

Mass Balance of Toxic Compounds in a Submerged Bed Wetland

The capability of wetlands to safely assimilate toxic materials is related to the fate of the toxic materials in the wetland system. The fundamental concepts were explained in the preceding two chapters. The primary productivity of the wetland autotrophic biomass provides organic solids that may effectively immobilize the toxic compounds and make them biologically unavailable. It was also pointed out that in sediment the toxicity of a compound should be related to the pore water concentration of the compound and not to the total concentration of the compound in the sediment volume. The same postulate applies even more to wetlands.

Several investigations have shown that wetland substrates may appear heavily contaminated by toxics, and yet the wetland biota appears either unaffected or only marginally impacted. This apparent paradox will be explained by the partitioning of the pollutants between the dissolved (toxic) and immobilized components in the wetland substrate. In contrast to lake and river sediments, wetland substrate is richer in particulate carbon and generally is anaerobic, both of which situations contribute to the reduction of the compound's toxicity to the wetland biota.

These concepts can be illustrated in an example of a simple, single-basin submerged bed system. For multiple basins in a series the same considerations will apply for the first basin receiving the full load. For modeling the fate of toxics in a more complex free surface system, one can use the EPA model WASP described in the preceding chapter.

The following assumptions are made in the subsequent calculation:

1. There is a steady buildup of organic matter in the wetland due to the inputs of the allochthonous (influent) and autochthonous (primary productivity) organic matter.
2. Mass-balance considerations are made over a relatively long time period. Diurnal and seasonal variations are not considered.
3. A significant portion of the organic input is nonrefractory (inert) and constitutes the inert organic carbon buildup in the wetland. According to the information provided by Foree, Jewell, and McCarty (1971) this inert fraction ranges between 40% to 60% in a mostly anaerobic environment.
4. The primary immobilizing (adsorbing) component is particulate inert organic carbon.
5. The wetland substrate is compacted and relatively uniform.

Following O'Connor's (1991) concepts the mass balance of the inert organic solids and toxics in the SFS wetland becomes:

A. Balance of Particulate Organics

$$\begin{aligned} \frac{dM_b}{dt} &= \frac{d(V_b m_b)}{dt} = Q_{in} POM_{in} + NPP \times A_b - K_b V_b m_b - Q_{out} POM_{out} \\ &= \phi_r Q_{in} POM_{in} + NPP \times A_b - K_b V_b m_b \end{aligned} \quad (14.6)$$

where

M_b = total mass balance of organic particulate carbon in the wetland, kg

- V_b = volume of the wetland bed (substrate), m^3
 m_b = concentration of the particulate organic matter in the substrate, kg/m^3
 POM_{in} = particulate organic matter concentration of the influent, $g/l = kg/m^3$
 NPP = net primary productivity of the particulate organic carbon by the wetland, $kg/m^2\text{-day}$
 A_b = surface area of the wetland, m^2
 Q_{in} = influent flow, m^3/day
 Q_{out} = effluent flow, m^3/day
 POM_{out} = effluent concentration of the particulate organic carbon, g/l
 K_b = decay rate of the organic carbon in the wetland, day^{-1}
 ϕ_r = removal rate of organic solids in the lagoon, dimensionless

In order to protect biota in the wetland the pore water concentration of the toxic should be related to inert organic matter, especially in cases when the toxic compound is conservative. This is because the decomposition process of the deposited particulate organic matter in the wetland is only qualitatively known. Hence let m_{oc} be the inert (nonbiodegradable) carbon fraction of the organic particulate matter. Also the effluent POC concentration is negligible when compared to the sum of the influent organic solids load and the net primary productivity in the wetland. Then Equation (14.6) becomes

$$\frac{dV_b m_{oc}}{dt} = f_{NRC}(\phi_r Q_{in} POM_{in} + NPP \times A_b) \quad (14.7)$$

Here $f_{NRC} = m_{oc} m_b$ is a fraction of the particulate organics, which is inert carbon, and m_{oc} is the concentration of inert organic carbon in the substrate in kg/m^3 . Expand the left side differential to

$$\frac{dV_b m_{oc}}{dt} = V_b \frac{dm_{oc}}{dt} + m_{oc} \frac{dV_b}{dt} \quad (14.8)$$

Since an assumption was made that the concentration of the inert organic matter remained constant over time, that is, $dm_{oc}/dt = 0$, Equation (14.7) can be modified to express the increase in the wetland volume due to annual deposition of inert particulate organic matter

$$\frac{dV_b}{dt} = \frac{f_{NRC}}{m_{oc}} (\phi_r Q_{in} POC_{in} + NPP \times A_b) \quad (14.9)$$

The preceding equation represents a uniform linear increase of the volume of the wetland. This relationship will be recalled in the further development of the allowable load of a toxic compound to the wetland.

B. Toxic Chemical Balance in the Wetland

$$\begin{aligned} \frac{dM_{Ch}}{dt} &= \frac{dV_b C_T}{dt} \\ &= W - K_a A_b C_d - K_d C_d V_b p - Q_{out}(C_d - POM_{out} r) \end{aligned} \quad (14.10)$$

where

- M_{Ch} = total mass of the chemical in the wetland, mg
- C_T = concentration of the total chemical in the wetland, $\mu\text{g/l} = \text{mg/m}^3$
- W = total load of the chemical, mg/day
- K_a = rate of volatilization of the chemical, m/day
- C_d = concentration of the dissolved chemical in pore water, $\mu\text{g/l}$
- K_d = decay rate of the chemical in the wetland by photolysis, chemical hydrolysis, and biochemical degradation, day^{-1}
- p = porosity (pore volume) of the wetland, dimensionless

The total mass of the chemical is a composite of the chemical dissolved in the pore water and that adsorbed on or complexed by the sediment, or

$$M_{Ch} = V_b C_T = p V_b C_d + (1 - p) V_b \rho_s r \quad (14.11)$$

where

- ρ_s = is the dry weight density of the sediment, kg/m^3
- r = adsorbed or complexed particulate chemical, $\text{mg/kg} = \mu\text{g/g}$

Introducing the partition relationship between the dissolved and particulate phase concentrations ($r = \Pi C_d$) presented in the preceding chapter and dividing both sides by V_b , one obtains

$$C_T = p C_d + (1 - p) \rho_s \Pi C_d \quad (14.12)$$

It was also shown in Chapters 6 and 13 that the partition coefficient for a chemical may be related to the octanol partition coefficient, K_{ow} , for the chemical and the particulate organic carbon fraction of the sediment. Then $\Pi \approx f_{NRC} \times K_{ow}$, where in this particular case f_{NRC} is the inert particulate organic carbon fraction of the wetland substrate (sediment). Recall also that the unit for Π and K_{ow} is $\text{l/g} = \text{m}^3/\text{kg}$. If K_{ow} is reported

in liters/kilogram, divide the reported value by 1000. The relationship between the total concentration of the toxicant commonly measured in micrograms/liter of the total volume of the wetland substrate and the concentration of the dissolved toxicant in pore water is then

$$\frac{C_d}{C_T} = \frac{1}{p + \rho_s \Pi (1 - p)} = \frac{1}{p + \rho_s f_{NRC} K_{ow} (1 - p)} = \frac{1}{p + m_{oc} K_{ow}} \quad (14.13)$$

This equation, similar to Equation (6.8) in Chapter 6, is very important. In a fashion similar to the consideration of sediment toxicity discussed in Chapter 6 one can substitute the aquatic toxicity standard for C_d and obtain the maximum total concentration limit (C_T) of the chemical in the substrate of the wetland.

Equation (14.13) can also be introduced into Equation (14.10) to relate the load, W , to the pore water concentration. Then

$$(p + m_{oc} K_{ow}) \frac{dV_b C_d}{dt} = W - K_a A_b C_d - K_d C_d V_b p - Q_{out} C_d (1 + POM_{out} f_{NRC} K_{ow}) \quad (14.14)$$

Expand the differential on the left side to

$$\frac{dV_b C_d}{dt} = V_b \frac{dC_d}{dt} + C_d \frac{dV_b}{dt}$$

and drop the first term on the right side because C_d should be kept constant (meaning it should not rise over a period of one year or longer). Also Equation (14.9) is substituted for an increase in the wetland volume, dV_b/dt . Then the dissolved, steady-state concentration of the toxicant in the wetland substrate is

$$C_d = \frac{W}{\{(p + m_{oc} K_{ow})(f_{NRC}/m_{oc})(\phi_r Q_{in} POM_{in} + NPP \times A_b)\} + \{K_a A_b + K_d V_b p + Q_{out}(1 + POM_{out} f_{NRC} K_{ow})\}} \quad (14.15)$$

For a wetland, $m_{oc} \times K_{ow} \gg p$. Also assume that in humid regions $Q_{in} \approx Q_{out}$, hence, $POM_{out} = (1 - \phi_r)POM_{in}$. Then by dropping p and by dividing the numerator and denominator of the right side of Equation (14.15) by A_b , the relation between the dissolved (bioavailable) concentration of the toxicant, and the specific load of the toxic compound, W' (mg/m₂-day) becomes

$$C_d \approx \frac{W'}{K_{ow} \times f_{NRC}(HLR \times POM_{in} + NPP) + (K_a + K_d H p + HLR)} \quad (14.16)$$

where

HLR = hydraulic loading rate, m/day

H = average depth of the substrate, m

Note that the first part of the right side of the denominator reflects the immobilization of the toxic compound by the input and buildup of the inert organic carbon, and the second part represents the loss of the toxicant by volatilization, decay, and dissolved loss in the effluent.

For a conservative highly adsorbable (complexed) compound

$$C_d = \frac{W'}{K_{ow} f_{NRC}(HLR \times POM_{in} + NPP)} \quad (14.17)$$

Example 14.3: Maximal Permissible Concentration of Lindane in SFS Wetland

Estimate the maximum volumetric concentration of lindane that would protect biota. The aquatic water quality criterion for lindane is $WQC = 0.08 \mu\text{g/l}$, the octanol partitioning coefficient is $K_{ow} = 51/\text{g}$, and the organic particulate concentration (measured, for example, as volatile suspended solids) in the wetland substrate is 1000 kg/m^3 ($= \text{g/l}$), of which 50% is inert ($f_{NRC} = 0.5$). The porosity of the substrate is 40% ($p = 0.4$).

Solution Estimate inert organic carbon concentration

$$m_{oc} = 0.5 \times 1000 [\text{kg/m}^3] = 500 \text{ kg/m}^3$$

From Equation (14.9) the maximal total concentration limit is

$$C_T = WQC(p + m_{oc}K_{ow}) = 0.08(0.4 + 500 \times 5) = 200 \mu\text{g/l}$$

Example 14.4: Maximal Load and Influent Concentrations of Toxic Compounds

The runoff entering the wetland contains the pesticide lindane and PCBs. O'Connor (1991) and Schnoor and O'Connor (1991) provided the following absorption and degradation characteristics for the chemicals in aquatic environments:

Lindane

Decay rate of the chemical: $K_d \approx 0.06 \text{ day}^{-1}$

Volatilization rate: $K_a \sim 0.00001 \text{ m/day}$

Octanol partition coefficient: $K_{ow} \sim 50001/\text{kg} = 51/\text{g}$

PCBs

Decay rate: $K_d \approx 0$

Volatilization: $K_a \approx 0.13 \text{ m/day}$

Octanol partition coefficient: $K_{ow} \approx 10^6/\text{kg} = 10001/\text{g}$

The aquatic toxicity water quality criterion for lindane is $\text{WQC} = 0.08 \mu\text{g/liter}$ and that for PCBs is $\text{WQC} = 0.14 \mu\text{g/l}$.

Estimate the maximal load of lindane and PCBs per square meter of the wetland that has the following parameters:

Depth of the bed: $H = 0.5 \text{ m}$

Porosity: 35% $p = 0.35$

Net primary productivity (Table 14.2): $\text{NPP} = 2 \text{ kg/m}^2\text{year} = 0.0055 \text{ kg/m}^2\text{day}$

Hydraulic loading: $\text{HLR} = 0.02 \text{ m}^3/\text{m}^2\text{-day}$

Influent volatile suspended solids: $\text{POM}_{\text{inf}} = 50 \text{ mg/l} = 0.05 \text{ g/l}$

Fraction of inert carbon versus organic solids: $f_{\text{NRC}} = 0.3$

Lindane The maximal load is calculated by Equation (14.16) as

$$\begin{aligned} \max W' &= \text{WQC}[K_{ow} \times f_{\text{NRC}}(\text{HLR} \times \text{POM}_{\text{in}} + \text{NPP}) \\ &\quad + (K_2 + K_d H p + \text{HLR})] \\ &= 0.08[5 \times 0.3 \times (0.02 \times 0.05 + 0.0055) \\ &\quad + (0.0001 + 0.006 \times 0.5 \times 0.4 + 0.02)] \\ &= 0.0025 \text{ mg/m}^2\text{-day} \end{aligned}$$

Since $W' = C_{\text{in}} \times Q_{\text{in}}/A_b = C_{\text{in}} \times \text{HLR}$, the maximal influent concentration limit becomes

$$\begin{aligned} C_{\text{in}} &= \max W'/\text{HLR} = 0.0025 [\text{mg/m}^2\text{-day}]/0.02 [\text{m}^3/\text{m}^2\text{-day}] \\ &= 0.125 \text{ mg/m}^3 (= \mu\text{g/l}) \end{aligned}$$

The wetland would provide about $100(0.125 - 0.08)/0.125 = 36\%$ removal of lindane. Note that because of relatively low K_{ow} , K_d , and K_a parameters lindane will not be effectively removed and stored in the wetland.

PCBs

$$\begin{aligned}\max W' &= 0.014[1000 \times 0.3(0.02 \times 0.05 + 0.0055) + (0.13 + 0 + 0.02)] \\ &= 0.029 \text{ mg/m}^2\text{-day}\end{aligned}$$

and the allowable influent concentration limit of PCBs is then

$$C_{in} = 0.029/0.02 = 1.47 \mu\text{g/l}$$

The wetland would provide $100(1.47 - 0.014)/1.47 = 99\%$ environmentally safe removal of the PCB load.

For the safe burial of toxic metals in the wetland or in the riverine and lacustrine sediments, the decay and volatilization rates are zero. Alternatively, for SFS wetlands or anaerobic sediments, the complexation of metals can be related to the availability of reduced sulfides, hence, sulfur compounds balance must be performed. If the sulfur input in the influent (mostly as sulphates), expressed in moles per time, is substantially greater than the input moles of toxic metals per time and the wetland or sediment system is mostly anaerobic, most of the metal input will be complexed and made nonbioavailable by the sulfides formed in the system.

Other Considerations

The advantages of constructed wetlands include low construction cost—essentially for grading, mulching, and planting wetland vegetation, with few structural components—and very low operation—maintenance costs for monitoring water levels, flows, the quality and vitality of vegetation, and grounds maintenance. Based on the experience of the Tennessee Valley Authority, Hammer (1989b) does not recommend plant harvesting except in special applications involving the water hyacinth or duckweed ponds incorporated within the wetland system (see the subsequent section on aquaculture). Plant harvesting will substantially increase operation and maintenance costs. Furthermore harvesting would eliminate the valuable organic matter needed for immobilization of toxics.

As said throughout this chapter wetland operation is related to the treatment objective. If nitrification and organic load removal is the objective, a substantial portion of the wetland system should be aerobic. Shallow free water systems are most suited for this purpose. Denitrification requires anoxic conditions. Also immobilization and detoxification of metals are best accomplished in an anaerobic environment. If an SFS wetland is heavily loaded by metals, clogging from substantial deposits of

precipitated metals may occur (Watson et al., 1989). Only very limited and short-term (less than three years) data on the removal of metals and other toxics by wetlands were reported in the literature at the time of the writing of this book.

Disadvantages of constructed wetland systems include the land requirement and a long start-up period. Current design recommendations specify 1.3 to 5.5 hectares per 1000 m³/day of treated wastewater (Table 14.5), depending on pretreatment and the design discharge limits (Hammer, 1989b). The start-up period before full removal efficiencies are achieved may extend over several growing seasons. However, considering that a typical structural installation of a conventional treatment plant may also take several years, this drawback may not be important.

As pointed out before, improperly designed and operated wetland systems can also cause a pest problem (mosquitoes and rodents) that may be a nuisance to the surrounding population. Both can be simply and easily avoided with proper design and operation.

In conclusion, constructed and, to a lesser degree, natural wetlands provide an effective control of pollution from various sources, including control of nutrients and sediments from diffuse sources. These are the most natural pollution control facilities, requiring in most cases no chemicals or energy other than light. Yet, their efficiency is comparable to expensive structural treatment plants. Constructed wetlands can be installed anywhere, including the lands that were formerly used for agricultural production, especially those that represent former drained wetlands. Such systems are now considered as a part of ecological engineering and remediation approaches to pollution problems (Mitsch and Jørgensen, 1989).

Mosquitoes should not be a problem for subsurface (SFS) systems. For free water systems *Gambusia* can be used, provided that some water pools (ponds) remain aerobic and fish can migrate from the pools into the other parts of the wetland. Nichols (1988) described a method developed by Hruby, an ecologist, who proposed a wetland-ditch combination. The ditches create narrow reservoirs in which water levels are manipulated to permit a fish population to enter marshes to spawn. The newly hatched fish larvae then feed on the mosquito larvae. Up to 97% mosquito control can be achieved.

AQUATIC PLANT SYSTEMS

Aquatic plant systems are shallow ponds with floating or submerged aquatic plants. As pointed out in the preceding section, ponds are commonly a part of natural wetlands system and are recommended for in-

clusion in constructed systems. Generally, after some time most ponds will develop some kind of aquatic vegetation. Typical types of aquatic vegetation were shown in Figure 14.5. In the context of aquatic plant systems, consider ponds into which aquatic vegetation was introduced or planted specifically for water-pollution abatement. These systems include two types based on the dominant plant types. The first type includes floating plants, and the second employs submerged plants.

Floating Plant Systems

The *water hyacinth* (*Eichhornia crassipes* (Fig. 14.5)) has been the most common floating plant used in aquatic treatment systems. It is a nuisance plant that can develop in large quantities and densities in subtropical regions and may completely cover a body of water with a dense layer (Fig. 14.28).

The water hyacinth has been used extensively for the treatment and posttreatment of wastewater. The major characteristics of water hyacinths that makes them attractive for the removal of pollutants are their extensive root system, which provides a medium for bacteria, and a rapid growth rate. However, these plants are temperature sensitive and will die rapidly during winter freezing conditions. Such systems may not be suitable for



FIGURE 14.28. Water hyacinth pond.

year-round applications in most of Europe, with the exception of the most southern parts (such as Sicily), and a majority of the United States, with the exception of Florida and the southern portions of California, Texas, and Arizona. However, Žáková and Veber (1990) documented that in the colder climatic conditions of central Europe, water hyacinth plants can be put into greenhouses during the winter and replanted in the spring. The best winter survival was achieved when plants were winterized in peat. For year-round operation, water hyacinth treatment lagoons were installed in Austin, Texas in a covered greenhouse.

Duckweed (Fig. 14.5) are small green freshwater plants (*Lemna* sp., *Spirodela* sp., *Wolffia* spp., and several others) that range in size from a millimeter to a few millimeters. Duckweed has a very fast growth rate, believed to be faster than that of water hyacinths. Effluents from duckweed lagoons (Fig. 14.29) should exceed the performance of conventional facultative treatment lagoons for BOD, suspended solids, and nitrogen removal.

Plant uptake (about 25% of the removal), ammonia volatilization, and nitrification–denitrification are the primary mechanisms for nitrogen removal in duckweed lagoons (Reed, Middlebrooks, and Crites, 1988). Overall nitrogen removal follows a first-order kinetics described by the following equation



FIGURE 14.29. Lemna pond. (Courtesy Lemna, Inc.)

$$\frac{N}{N_0} = e^{-kt} \quad (14.18)$$

where

N_t = total nitrogen in system effluent, mg/l

N_0 = total nitrogen in the influent, mg/l

k = a rate constant dependent on temperature and plant density

t = time

Reed, Middlebrooks, and Crites (1988) and Zirschky and Reed (1988) emphasized that frequent harvesting of plants was needed to sustain high levels of nitrogen removal. Due to the fact that the water environment of the lagoon is mostly anaerobic and the duckweed cultures do not have the capability of transferring oxygen because their roots are very small and do not extend into the water, nitrification is limited or does not occur. Therefore if the input of nitrogen is in an ammoniacal form, the denitrification process may not have an adequate supply of nitrate-nitrogen and can be suppressed. Due to the fact that water in the duckweed ponds is anaerobic, the BOD removal is about the same as that in anaerobic facultative lagoons. Anaerobic effluent should be aerated (for example, on cascades).

Phosphorus removal in duckweed lagoons, just as in wetland systems, is not very high, and if removal of P is the object, the use of precipitating chemicals, such as alum or ferric chloride, is recommended (Reed, Middlebrooks, and Crites, 1988).

The use of duckweed for pollution abatement is not as well developed as is the use of water hyacinths. Mosquitoes should not be a problem as long as a thick mat of duckweed plants is maintained on the surface. Mosquito larvae cannot survive in the anaerobic environment created in the lagoon. The mat of duckweed also prevents algae from developing. Unharvested dead and dying plants can create a significant BOD and suspended solids load on the lagoon and release the nutrients accumulated in their tissues. For this reason, as stated before, the duckweed crop should be periodically harvested to ensure rapid regrowth and effective nutrient uptake. Special harvesting equipment is commercially available.

The harvested plant biomass has a higher protein content than other crops used for feeding farm animals. It also contains about 1.5% of phosphorus and about 6% of nitrogen (based on dry weight). The harvested plants can be used directly as feed if transportation is not an issue, or the biomass can be dried and/or composted. In the context of sustainable environmental management, research directed toward converting the harvested plants from the aquaculture lagoons and wetlands should be

initiated and the organic fertilizer derived from the biomass should be substituted for chemical fertilizers. The benefit of this substitution would be less ground-water contamination by nitrates and subsequently less nutrient input to surface-water bodies. This may involve the cooperation of the fertilizer manufacturing industries.

Submerged Plants

The potential for using submerged plants is limited due to their tendency to be shaded by algae and their sensitivity to anaerobic conditions (U.S. EPA, 1988). Also clogging of roots and stems by filamentous organisms can severely limit their functioning. The best results for BOD, suspended solids, and nitrogen removal were obtained in an experimental study (Eighmy and Bishop, 1985) with a common water weed (*Elodea* sp.).

In summary, the water hyacinth aquaculture plants may not be suitable for most of North American and European climatic zones. Duckweed lagoons can be used in colder zones and will provide effective removal of organics and nitrogen. A wetland-aquaculture lagoon combination can be considered, but should be investigated in a pilot study. In contrast to water hyacinth lagoons, duckweed ponds and lagoons may not provide enough nitrification because the environment in the lagoon may be mostly anaerobic. This may also cause an odor problem. These problems can be mitigated and alleviated.

References

- Bender, M. E., and D. L. Correl (1974). *The Use of Wetlands as Tertiary Treatment Systems*, National Science Foundation report NSF-RAE-74-033, Washington, DC.
- Brodie, G. A., D. A. Hammer, and D. A. Tomijanivich (1989a). Treatment of acid drainage with constructed wetland at the Tennessee Valley Authority 950 Coal Mine, in *Constructed Wetlands for Wastewater Treatment*, D. A. Hammer, ed., Lewis Publ., Chelsea, MI, pp. 201-210.
- Brodie, G. A., D. A. Hammer, and D. A. Tomijanivich (1989b). Constructed wetlands for treatment of ash pond seepage, in *Constructed Wetlands for Wastewater Treatment*, D. A. Hammer, ed., Lewis Publ., Chelsea, MI, pp. 211-220.
- Cooper, P. F., and J. A. Hobson (1989). Sewage treatment by Reed Bed System: The present situation in the United Kingdom, in *Constructed Wetlands for Wastewater Treatment*, D. A. Hammer, ed., Lewis Publ., Chelsea, MI, pp. 153-172.
- Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe (1979). *Classification of Wetlands and Deepwater Habitats in the United States*, U.S. Fish and Wildlife Service pub. FWS/OBS-79/31, Washington, DC.

- D'Avanzo, C. (1989). Long-term evaluation of wetland creation project, in *Wetland Creation and Restoration: The Status of the Science*, Vol. I, J. A. Kunsler and M. E. Kentula, eds., EPA 600/3-89-038a, U.S. Environmental Protection Agency, Corvallis, OR, pp. 75-84.
- Dawson, B. (1989). High hopes for cattails, *Civil Eng.*, no. 4, pp. 48-50.
- DiToro, D. M., et al. (1989). *Briefing Report to the EPA Science Advisory Board on the Equilibrium Partitioning Approach to the Sediment Quality Criteria*, EPA 440/5-89.002, Office of Water and Regulations and Standards Division, U.S. Environmental Protection Agency, Washington, DC.
- DiToro, D. M., et al. (1990). Toxicity of cadmium in sediments: The role of acid volatile sulfide, *Environ. Toxicol. Chem.* 9:1487-1502.
- DiToro, D. M., et al. (1991). Technical basis for establishing sediment quality criteria for non-ionic organic chemicals using equilibrium partitioning, *Environ. Toxicol. Chem.* 10(12):1541-1583.
- Eighmy, T. T., and P. L. Bishop (1985). *Preliminary Evaluation of Submerged Aquatic Macrophytes in: A Pilot-Scale Aquatic Treatment System*, Department of Civil Engineering, University of New Hampshire, Durham, NH.
- Esry, D. H., and D. J. Cairns (1989). Overview of the Lake Jackson restoration project with artificially created wetlands for treatment of urban runoff, in *Wetlands: Concerns and Successes*, D. W. Fisk, ed., American Water Resources Association, Bethesda, MD, pp. 247-257.
- Faulkner, S. P., and C. J. Richardson (1989). Physical and chemical characteristics of freshwater wetland soils, in *Constructed Wetlands for Wastewater Treatment*, D. A. Hammer, ed., Lewis Publ., Chelsea, MI, pp. 41-72.
- Fisk, D. W. (1989). *Wetlands: Concerns and Successes*, American Water Resources Association, Bethesda, MD.
- Foree, E. G., W. J. Jewell, and P. L. McCarty (1971). The extent of nitrogen and phosphorus regeneration from decomposing algae, in *Proceedings, International Conference on Water Pollution Research*, IAWPR, San Francisco, CA, Pergamon Press, Oxford.
- Gayan, S. L., and J. D'Antuono (1988). An integrated experience—The Milwaukee River East-West Branch Watershed Resource Management Plan, in *Proceedings, Nonpoint Pollution: 1988—Policy, Economy, Management, and Appropriate Technology*, V. Novotny, ed., American Water Resources Association, Bethesda, MD, pp. 165-172.
- Godfrey, P. J., E. R. Kaynor, S. Pelczarski, and J. Benforado (1985). *Ecological Considerations in Wetlands Treatment of Municipal Wastewaters*, Van Nostrand Reinhold, New York.
- Haag, R. D. (1979). *The Hydrology of the Houghton Wetland*, University of Michigan Wetlands Ecosystem Research Group, Ann Arbor, MI.
- Hammer, D. A. (1989a). *Constructed Wetlands for Wastewater Treatment*, Lewis Publ., Chelsea, MI.
- Hammer, D. A. (1989b). *Constructed Wetlands for Wastewater Treatment—An Overview of Emerging Technologies*, paper presented at Urban Stream Corridor and Stormwater Management Symposium, March 14-16, Colorado Springs, CO.

- Hammer, D. A., and R. K. Bastian (1989). Wetland ecosystems: Natural water purifiers? in *Constructed Wetlands for Wastewater Treatment*, D. A. Hammer, ed., Lewis Publ., Chelsea, MI, pp. 5–20.
- Hedin, R. S., R. Hammack, and D. Hyman (1989). Potential importance of sulfate reduction processes in wetlands constructed to treat mine drainage, in *Constructed Wetlands for Wastewater Treatment*, D. A. Hammer, ed., Lewis Publ., Chelsea, MI, pp. 508–514.
- Hey, D. L. (1988). Wetlands: A future nonpoint pollution control technology, in *Proceedings, Nonpoint Pollution: 1988—Policy, Economy, Management, and Appropriate Technology*, V. Novotny, ed., American Water Resources Association, Bethesda, MD, pp. 225–236.
- Hey, D. L., M. G. Cardamore, J. H. Sather, and W. J. Mitsch (1989). Restoration of riverine wetlands: The Des Plaines River wetland demonstration project, in *Ecological Engineering. An Introduction to Ecotechnology*, W. J. Mitsch and S. E. Jørgensen, eds., Wiley, New York, pp. 159–184.
- Kadlec, R. H. (1988). *Denitrification in Wetland Treatment Systems*, paper presented at the 61st Conference of the Water Pollution Control Federation, Session 26, Dallas, TX.
- Kadlec, R. H., and H. Alvord, Jr. (1989). Mechanisms of water quality improvement in wetland treatment system, in *Wetlands: Concerns and Successes*, D. W. Fisk, ed., American Water Resources Association, Bethesda, MD.
- Keeney, D. R., S. Schmidt, and C. Wilkinson (1975). *Concurrent Nitrification—Denitrification at the Sediment—Water Interface as a Mechanism for Nitrogen Losses from Lakes*, Tech. Rep. no. 75-07, Water Resources Center, University of Wisconsin, Madison, WI.
- Knight, R. L., B. R. Windchester, and J. C. Highman (1984). Carolina Bays—feasibility for effluent advanced treatment and disposal, *Wetlands* 4:177–204.
- Kusler, J. A., and M. E. Kentula (1989). *Wetland Creation and Restoration: The Status of the Science*, Vol. I, EPA 600/3-89-038a, Environmental Research Laboratory, U.S. Environmental Protection Agency, Corvallis, OR.
- Lichfield, D. K., and D. D. Schutz (1989). Constructed wetlands for wastewater treatment at Amoco Oil Company's Mandan, North Dakota refinery, in *Constructed Wetlands for Wastewater Treatment*, D. A. Hammer, ed., Lewis Publ., Chelsea, MI, pp. 233–238.
- Livingston, E. H. (1989). Use of wetlands for urban stormwater management, in *Constructed Wetlands for Wastewater Treatment*, D. A. Hammer, ed., Lewis Publ., Chelsea, MI, pp. 253–263.
- McArthur, B. H. (1989). The use of isolated wetlands in Florida for stormwater management, in *Wetlands: Concerns and Successes*, D. W. Fisk, ed., American Water Resources Association, Bethesda, MD, pp. 185–194.
- Mitsch, W. J. (1990). *Wetlands for the Control of Nonpoint Source Pollution: Preliminary Feasibility Study for Swan Creek Watershed of Northwestern Ohio*, Ohio Environmental Protection Agency, Columbus, OH.
- Mitsch, W. J., and J. G. Gosselink (1986). *Wetlands*, Van Nostrand Reinhold, New York.

- Mitsch, W. J., and S. E. Jørgensen, eds. (1989). *Ecological Engineering. An Introduction to Ecotechnology*, Wiley, New York.
- Mitsch, W. J., M. Straskraba, and S. E. Jørgensen (1989). *Wetland Modeling*, Elsevier, Amsterdam.
- Mortimer, C. H. (1971). Chemical interchanges between sediments and water in the Great Lakes, *Limnol. Oceanogr.* **16**(2):387-404.
- Nichols, A. B. (1988). A vital role for wetlands, *J. WPCF* **60**(7):1215-1221.
- Novitzki, R. P. (1979). Hydrologic characteristics of Wisconsin's wetlands and their influence on floods, stream flow, and sediment, in *Wetland Function and Values: The State of Our Understanding*, P. E. Greeson, J. R. Clark, and J. F. Clark, eds., American Water Resources Association, Minneapolis, MN.
- Oberts, G. L. (1982). Impact of wetlands on nonpoint source pollution, in *Proceedings, International Symposium on Urban Hydrology, Hydraulics and Sediment Control*, University of Kentucky, Lexington, KY, pp. 225-231.
- O'Connor, D. J. (1991). Basic concepts and equations, in *Contaminated Sediments, Models for Criteria, Exposure, and Remediation*, 36th Institute in Water Pollution Control, Manhattan College, Riverdale, NY.
- Odum, H. T. (1989). Ecological engineering and self-organization, in *Ecological Engineering: An Introduction to Ecotechnology*, W. J. Mitsch and S. E. Jørgensen, eds., Wiley, New York, pp. 79-101.
- Ogawa, H., and J. W. Male (1989). *The Flood Mitigation Potential of Inland Wetlands*, Water Resources Research Center Publ. No. 138, University of Massachusetts, Amherst, MA.
- Palmer, C. N., and J. D. Hunt (1989). Greenwood urban wetland: A manmade stormwater treatment facility, in *Wetlands: Concerns and Successes*, D. W. Fisk, ed., American Water Resources Association, Bethesda, MD, pp. 205-214.
- Reed, S. C., E. J. Middlebrooks, and R. W. Crites (1988). *Natural Systems for Waste Management and Treatment*, McGraw-Hill, New York.
- Richardson, G. J., and D. S. Nichols (1985). Ecological analysis of wastewater management criteria in wetland ecosystems, in *Ecological Considerations in Wetlands Treatment of Municipal Wastewaters*, P. J. Godfrey, et al., eds., Van Nostrand Reinhold, New York.
- Schnoor, J. L., and D. J. O'Connor (1991). Kinetics and transfer processes, in *Contaminated Sediments, Models for Criteria, Exposure, and Remediation*, 36th Institute in Water Pollution Control, Manhattan College, Riverdale, NY.
- Schueler, T. R., P. A. Kumble, and M. A. Heraty (1992). *A Current Assessment of Urban Best Management Practices. Techniques for Reducing Non-point Source Pollution in the Coastal Zone*, Technical Guidance Manual, Metropolitan Washington Council of Governments, Office of Wetlands, Oceans, and Watersheds, U.S. Environmental Protection Agency, Washington, DC.
- Steiner, G. R., and R. J. Freeman, Jr. (1989). Configuration and substrate design considerations for constructed wetlands for wastewater treatment, in *Constructed Wetlands for Wastewater Treatment*, D. A. Hammer, ed., Lewis Publ., Chelsea, MI, pp. 363-378.

- Stengel, E., and R. Schulz-Hock (1989). Denitrification in artificial wetlands, in *Constructed Wetlands for Wastewater Treatment*, D. A. Hammer, ed., Lewis Publ., Chelsea, MI, pp. 484–492.
- Stumm, W., and J. J. Morgan (1962). Stream pollution by algal nutrients, in *Transactions, Twelfth Annual Conference on Sanitary Engineering*, University of Kansas, Lawrence.
- Thut, R. N. (1989). Utilization of artificial marshes for treatment of pulp mill effluents, in *Constructed Wetlands for Wastewater Treatment*, D. A. Hammer, ed., Lewis Publ., Chelsea, MI, pp. 239–244.
- U.S. Environmental Protection Agency (1988). *Constructed Wetlands and Aquatic Plant Systems for Municipal Wastewater Treatment. Design Manual*, EPA 625/1-88/022, U.S. EPA, Washington, DC.
- Valiela, I., J. M. Teal, and W. Sass (1973). Nutrient retention in salt marsh plots experimentally fertilized with sewage sludge, *Estuar. Coast. Marine Sci.* 1:261–269.
- Water Pollution Control Federation (1990). *Natural Systems for Wastewater Treatment*, Manual of Practice FD-16, WPCF, Alexandria, VA.
- Watson, J. T., S. C. Reed, R. H. Kadlec, R. L. Knight, and A. E. Whitehouse (1989). Performance expectations and loading rates for constructed wetlands, in *Constructed Wetlands for Wastewater Treatment*, D. A. Hammer, ed., Lewis Publ., Chelsea, MI, pp. 319–352.
- Žáková, Z., and K. Veber (1990). *Biologické základy pěstování a využívání vodního hyacintu v ČSFR* (Biological fundamentals of use of water hyacinth in CSFR), Academia, Prague, Czech Republic.
- Zirschky, J., and S. C. Reed (1988). The use of duckweed for wastewater treatment, *J. WPCF* 60(7):1253–1258.

Management and Restoration of Streams, Lakes, and Watersheds

Accomplishing restoration means ensuring that ecosystem structure and function are recreated or repaired and that natural dynamic ecosystem processes are operating effectively again.

Committee on Restoration of Aquatic Ecosystems,
National Research Council

A large proportion of the surface waters of the United States, especially streams and lakes, have been degraded from chemical, biological, and physical habitat destruction as a result of urbanization, deforestation, overgrazing, industrialization, agricultural practices, mining, flood-control projects, channelization, reservoir and dam construction, diversions, dredging, and other land uses. On the other hand, because of improvements in water quality from control of point source discharges, restoration of the biological and physical habitat in these waters could produce large improvements in the structure and function of the biological communities beyond those gained by improving water quality alone through control of point sources. When used in conjunction with elimination or reduction of nonpoint sources in the watershed, ecological restoration can lead to significant water quality benefits. Improvements in the physical and biological habitats of surface waters also can lead to improvements in water quality by increasing the capacity of aquatic ecosystems to process contaminants.

The National Research Council (Committee on Restoration of Aquatic Ecosystems, 1992) concluded that habitat degradation is a primary factor in limiting the attainment of beneficial uses of the nation's surface waters.

The researchers also concluded that an accelerated effort toward restoration of aquatic ecosystems is needed and that failure to restore aquatic ecosystems promptly will, in many cases, result in sharply increased environmental costs later through the extinction of species or ecosystem types or permanent ecological damage.

Stream restoration projects often are labor intensive; therefore, the use of funds for such projects has the additional benefit of creating numerous jobs, potentially in urban and rural areas with high unemployment rates. This chapter describes the necessary steps for planning a restoration project and the numerous approaches to restoration and management for streams, rivers, lakes, ponds, and their watersheds.

Although there are separate sections discussing river/stream and lake/reservoir systems, an integrated approach to restoration is recommended. Proposed restoration projects should consider the major ecological interactions in a watershed. Rivers, streams, lakes, and reservoirs are interconnected parts of a watershed that also often include ground water, estuaries, and wetlands. A practical and effective approach to restoration of aquatic ecosystems would include the consideration of all significant ecological elements on a watershed scale. In this way, the cumulative impacts of an ecosystem can best be evaluated.

RESTORATION VERSUS RECLAMATION, REHABILITATION, AND MANAGEMENT

The fundamental goal of restoration is to return the ecosystem to a condition that approximates its condition prior to disturbance (Cairns, 1988; Committee on Restoration of Aquatic Ecosystems, 1992). The terms restoration, reclamation, rehabilitation are often used interchangeably, but their meanings are different. *Restoration* is the return of an ecosystem to a close approximation of its former condition. *Reclamation* is a process designed to adapt a resource to serve a new or altered use. This could mean a process to convert a native ecosystem to agricultural uses or to convert a disturbed resource, such as a mined surface area, to productive use, such as pasture land. *Rehabilitation* is a term that describes putting a severely disturbed and/or damaged resource back into good working order. It is often used to indicate improvements that are primarily of a visual nature. Hence, restoration is a holistic process achieved through manipulation of all relevant elements, including the reintroduction of plants and animals. It is the most ambitious of the processes and is not possible in all situations; and where possible, it may not be economically feasible.

Whether an aquatic ecosystem is restored, reclaimed, rehabilitated, or

preserved, it usually needs to be managed. *Management* is used here to mean the manipulation of an ecosystem to ensure the maintenance of one or more conditions or functions (Committee on Restoration of Aquatic Ecosystems, 1992). Watershed management is a term used to describe the management practices within an entire watershed that are designed to control or reduce inputs of pollution. From a technical standpoint, the watershed is the most logical scale on which to undertake restoration or management of aquatic resources. Unfortunately, political boundaries do not usually coincide with watershed boundaries and other institutional constraints can make restoration and management at the watershed scale difficult.

Restoration Project Planning

Planning a restoration project requires consideration of four basic items

1. *Project mission*. The overall general purpose, such as the restoration of a lake and perhaps an adjoining wetland.
2. *Goals*. The outcome desired, such as improving lake fisheries.
3. *Objectives*. Derived from the goals, such as the improvement in populations of specific fish species in the lake.
4. *Performance indicators*. Specific measurable quantities that are taken from each objective, such as an increase in the standing crop of fish.

Developing appropriate indicators requires selecting characteristics from a large number of possible ecological assessment criteria. These indicators are taken from certain structural, functional, and holistic criteria. The criteria listed below have been adapted from Berger (1990) and the National Research Council (Committee on Restoration of Aquatic Ecosystems, 1992).

Structural Criteria

- *Water quality*. Parameters include dissolved oxygen, temperature, pH, dissolved cations and anions, toxic contaminants, and suspended matter.
- *Soil condition*. Parameters include soil chemistry, erodibility, permeability, organic content, and soil stability.
- *Geological condition*. Includes surface and subsurface rock.
- *Hydrology*. Includes discharge, surface-flow properties, sediment load, retention times (for lakes), and ground-water flow and exchange with surface water.

- *Topography and morphology.* Surface contours, relief, configuration of site surface features, and the subsurface features.
- *Flora and fauna.* Presence and characteristics of the species, including evidence of biotic stress.
- *Waste assimilative capacity (loading capacity).* Includes nutrient availability, nutrient flux, and food web support.

Functional Criteria

- Surface and ground-water storage, recharge, and supply
- Floodwater and sediment retention
- Transport of organisms, nutrients, and sediments
- Humidification of atmosphere
- Oxygen production
- Nutrient cycling
- Biomass production, food web support, and species maintenance
- Provision for shelter for biota
- Detoxification of waste and purification of water
- Reduction of erosion and mass wastage
- Energy flow

Holistic Criteria

- *Resilience.* The ability of the ecosystem to recover from perturbations.
- *Persistence.* The ability of the ecosystem to survive or undergo natural successional processes without human management intervention.
- *Verisimilitude.* A characteristic of the restored ecosystem that reflects its similarity to a reference standard or its conditions prior to restoration.

Performance indicators can be constructed from a review of the preceding criteria to determine which are important in evaluating the particular project. For example, in the case of a project involving the improvement in water quality, criteria might include indicators such as pH, toxic contaminants, and suspended sediment.

The National Research Council (Committee on Restoration of Aquatic Ecosystems, 1992) presented a checklist to be considered before, during, and after any restoration activity (see below). Each of the questions in the checklist should be answered before proceeding to the next step in any restoration project.

Project Planning and Design

1. Has the problem requiring treatment been clearly understood and defined?

2. Is there a consensus on the restoration program's mission?
3. Have the goals and objectives been identified?
4. Has the restoration been planned with adequate scope and expertise?
5. Does the restoration management design have an annual or midcourse correction point in line with adaptive management procedures?
6. Are the performance indicators—the measurable biological, physical, and chemical attributes—directly and appropriately linked to the objectives?
7. Have adequate monitoring, surveillance, management, and maintenance programs been developed along with the project, so that monitoring costs and operational details are anticipated and monitoring results will be available to serve as input in improving restoration techniques used as the project matures?
8. Has an appropriate reference system (or systems) been selected from which to extract target values of performance indicators for comparison in conducting the project evaluation?
9. Have sufficient baseline data been collected over a suitable period of time on the project ecosystem to facilitate before-and-after treatment comparisons?
10. Have critical project procedures been tested on a small experimental scale in part of the project area to minimize the risks of failure?
11. Has the project been designed to make the restored ecosystem as self-sustaining as possible to minimize maintenance requirements?
12. Has thought been given to how long monitoring will have to be continued before the project can be declared effective?
13. Have risk and uncertainty been adequately considered in project planning?

During Restoration

1. Based on the monitoring results, are the anticipated intermediate objectives being achieved? If not, are appropriate steps being taken to correct the problem(s)?
2. Do the objectives or performance indicators need to be modified? If so, what changes may be required in the monitoring program?
3. Is the monitoring program adequate?

Post-Restoration

1. To what extent were project goals and objectives achieved?
2. How similar in structure and function is the restored ecosystem to the target ecosystem?
3. To what extent is the restored ecosystem self-sustaining, and what are the maintenance requirements?
4. If all natural ecosystem functions were not restored, have critical ecosystem functions been restored?
5. If all natural components of the ecosystem were not restored, have critical components been restored?
6. How long did the project take?

7. What lessons have been learned from this effort?
8. Have those lessons been shared with interested parties to maximize the potential for technology transfer?
9. What was the final cost, in net present value terms, of the restoration project?
10. What were the ecological, economic, and social benefits realized by the project?
11. How cost-effective was the project?
12. Would another approach to restoration have produced desirable results at lower cost?

In order to address some of the questions presented in the checklist, the following items must be considered.

- *Project duration.* Each project should have a realistic schedule to allow enough time for evaluation. The duration of the project and its accompanying monitoring should be sufficient to address the particular environmental conditions of the site. For example, regions where environmental conditions include frequent floods or drought should be monitored long enough so that these variables do not interfere with the accurate assessment of project success.
- *Project scale.* The areal scale of the project must also be sufficient to account for spatial factors and for interaction between the target system to be managed or restored and the surrounding landscape.
- *Budgetary factors.* Not all projects have adequate funding to accomplish complete restoration. When funding conditions are inadequate or uncertain, efforts should be made to prioritize activities to best meet the overall project goals and objectives.

Generally, the goals of water body restoration–rehabilitation efforts are (a) restoration of damaged aquatic ecosystems, and (b) increasing the waste-assimilative capacity for increased safe disposal of wastes.

RESTORATION TECHNIQUES FOR RIVERS AND STREAMS

Much progress has been made in the past few decades in restoring rivers and streams affected by point sources of pollution through a variety of federal, state, and local programs. Unfortunately, there is no comparable suite of programs to deal with the restoration of rivers and streams affected by diffuse sources. Yet development pressures, agriculture, and changing land use have resulted in diffuse sources becoming the major fraction of wasteload to surface waters. And unlike slow-moving lakes, rivers and streams have historically served as a convenient and

inexpensive means of waste disposal because the flow carries away or assimilates the pollutants.

The idea of stream waste-assimilative capacity enhancement was first proposed and implemented by Karl Imhoff, a pioneer of European wastewater management, in the 1920s in the industrialized Ruhr district of Germany (see Chap. 1 for a discussion of river basin management agencies (verbände) in the Ruhr area). He recognized the fact that the small rivers of the district would not be capable of safely assimilating municipal and industrial wastewater discharges, even after a full (primary plus secondary biological) treatment. The waste-assimilative capacity and water quality enhancement measures implemented therein included (Imhoff and Imhoff, 1990; Novotny et al., 1989):

1. Building stream impoundments. The objectives are (a) to enlarge the water surface, (b) to prolong the detention time, and (c) to induce deposition and retention of suspended solids and sludges. Loading of solids should be allowed in a limited amount. Deep reservoirs that stratify in summer are unsuitable for assimilation of large quantities of wastewater. When the impoundment fills with solids it must be dredged or abandoned and restored. As will be pointed later abandonment and restoration of old river impoundments to a more natural streamlike state is now being practiced in several places.
2. In-stream aeration by turbines and aerators to increase the reaeration capacity of the streams.
3. Lining of a small stream (Emscher River) that was heavily loaded by partially treated discharges of municipal and industrial wastewater. The lining increases velocity and decreases depth, resulting in a greatly increased aeration.
4. Back-pumping of the Rhine River into the Ruhr River (by the power plants on the Ruhr reservoirs) to increase low flows during times of extreme droughts.
5. Low-flow augmentation or increase of low-water flows.

The present Ruhr River basin management agency also implemented

6. Weed cutting and nutrient inactivation by chemicals in the impoundments.
7. Dredging of accumulated bottom sediments from the impoundments.

The agencies are also involved in stream restoration activities (see Chap. 1).

This section provides an overview of the various techniques available

for the restoration of rivers and streams. Several types of river and stream problems are frequently encountered, and these can be categorized in seven main areas: (1) sedimentation; (2) low dissolved oxygen levels resulting in fish kills; (3) overgrazing of riparian areas; (4) stream channelization; (5) drinking water taste, odor, color, and organics; (6) poor fishing; and (7) acidic conditions. For each of these major problem areas, several in-stream techniques have been found to be effective, long-lasting, and generally without significant negative impact when used properly. These procedures will be described with regard to their underlying ecological principles and modes of action, effectiveness (including brief case histories), potential negative impacts, and additional benefits and costs. The reader will be referred to further reports in the basic scientific literature (for example, Committee on Restoration of Aquatic Ecosystems, 1992; Krenkel and Novotny, 1980). The less-studied or less-effective procedures will also be briefly described.

Basic Assumptions

The following discussions of in-stream technique effectiveness, except where explicitly stated, always assume that loadings of nutrients, silt, and organic matter to the watercourse have already been controlled. Most instream procedures will be quickly overwhelmed by continued accumulation of these substances. The watercourse and its watershed are coupled. In-stream programs can complement watershed efforts; however, such problems as bank erosion and sedimentation may persist despite load reductions or diversion projects unless an in-stream procedure is also used.

As for restoration and management techniques that are not mentioned in this section, in nearly every case these procedures have not been described in the open scientific literature, and therefore have not had the benefit of testing, discussion, explanation, and criticism that is so vital to the development of techniques of proven effectiveness and minimal negative impact. Caution should be exercised in the use of a procedure not listed here.

Selecting an Appropriate Stream-Restoration Technique

Selecting the wrong type of action may make the aquatic ecosystem worse off than no intervention at all. In addition, to be effective the action must address key limiting factors or major problems affecting the ecosystem. Otherwise, money and effort will be expended with little to no positive

returns. For these reasons, none of the methods described here should be attempted without first consulting a professional familiar with ecosystem restoration and management. Furthermore, some of these activities require permits or are prohibited by some state regulatory authorities.

This section provides a comprehensive review of most possible stream-restoration techniques; its scope is not limited by any specific objectives or philosophy. Nevertheless, in selecting the techniques to apply in any given situation, careful thought and analysis are necessary to ensure that the program (a) addresses the cause of the problems and major use limitations to ensure an effective expenditure of funds and effort, (b) will not result in unexpected adverse effects on other water uses or ecosystem components, and (c) is consistent with public values and the long-term goals for the watercourse and its watershed.

Costs are another important consideration in selecting an approach. The cost of implementing a given technique can vary widely, depending on the area of the country, watershed size, accessibility, and other site-specific conditions and factors. Because of this variability, cost estimates are not provided here, although relative costs are discussed for some techniques. As part of the planning process, cost estimates should be obtained for the subset of options being considered. Both implementation costs and long-term operation and maintenance costs over the life of the project must be evaluated. Certain questions must be answered (see the section on project planning). How long will the treatment or beneficial effects last? How often will management actions need to be repeated? How much effort and expense will be required for routine upkeep and monitoring?

The size of the river or stream and associated watershed can have a major influence on the feasibility and effectiveness, as will the costs of various techniques. Restoration in larger rivers is more problematic than in smaller systems because of the size and complexity of the systems and often the different types and sources of problems. For rivers interconnected to other rivers, streams, and lakes, special consideration must be given to the potential for effects downstream or in adjacent waters. For example, exotic plants and animals may rapidly disperse throughout a drainage basin, becoming a regionwide nuisance.

Also, the seasonal and annual fluctuations in water quantity and the quality of flowing waters often require a longer time series of data than is available or funded in connection with a restoration project. Without an adequate time series of data, restoration projects are often hampered by their inability to determine success or failure.

Important concepts related to the management and restoration of rivers and streams were reviewed by the National Research Council

(Committee on Restoration of Aquatic Ecosystems, 1992) and are presented below:

- Flow and retention
- Openness
- Dynamism
- Patchiness
- Resistance and resilience

These attributes are integral to understanding the connection between rivers and streams and their ecosystems. Large rivers and floodplains are intimately linked and, therefore, should be restored and managed as a single ecosystem. For small streams, the stream and its riparian zone are linked. River and stream problems requiring restoration or management can be in the river or stream channels and pools, the riparian zone or floodplain, or the watershed. The types of problems that can occur in river–floodplain or stream–riparian systems are listed below (adapted from the Committee on Restoration of Aquatic Ecosystems, 1992):

Acidity
 Bank erosion
 Blockage of stream or river channel
 Braided channel
 Dissolved oxygen deficiency
 Excessive flooding
 Food scarcity for biota
 Loss of fish refugia
 Nutrient loss or excess
 Pool deficit
 Poor spawning success
 Sediment loss
 Siltation
 Species (extinct, threatened, or endangered)
 Stream cover (deficit or overgrown)
 Water quality (turbidity or chemical pollution)
 Water velocity (too high or too low)

Stream Restoration Techniques

Assuming that watershed inputs of pollution are controlled, the various approaches that can be used in river and stream restoration can be divided into structural and nonstructural techniques. Nonstructural techniques are broadly defined as any method that does not require either physical alteration of the watercourse or construction of a dam or other

structure. Structural methods range from simple biotechnical approaches, such as the use of tree trunks and branches along banks to slow water velocity (sometimes referred to as “soft” engineering techniques), to “hard” engineering approaches, such as the use of concrete or riprap for bank stabilization.

Nonstructural Techniques

Nonstructural techniques do not require physical alteration of the watercourse and include those administrative or legislative policies and procedures that limit or regulate some activity. Nonstructural methods are listed and discussed below.

<i>Techniques</i>	<i>Possible Approaches to Technique</i>
Flow regulation	Legislative Administrative
Plantings	Trees Brush Herbaceous vegetation Grass
Pollution abatement	Instream controls Riparian zone controls Watershed controls
Propagation facilities	Incubation Spawning
Land acquisition	Greenways Buffer strips Parks and other lands
Land use regulation	Instream Riparian zone Watershed
Biomanipulation	Game fish stocking Control of undesirable fish species and stunted fish populations Prey enhancements to supplement food supplies

Flow Regulation

Regulating the flow through legislative or administrative approaches is a viable nonstructural technique for restoration. Although reserving or reclaiming flow for instream uses (fish, wildlife, and recreation) is usually thought of as an important restoration technique in the arid western parts

of the United States, the approach is becoming more widely used elsewhere as well. For example, droughts in the Tennessee Valley region in the late 1980s resulted in requests to modify water withdrawal practices and potential conflicts with the health of aquatic life in the Tennessee River system.

Plantings

A common problem in developed streams and rivers is the clearing of riparian zones. Plantings in the riparian zone that do not alter the watercourse can be considered nonstructural restoration activities. The buffer zones can be gradually reforested over time through plantings of trees, brush, herbaceous vegetation, and grass. Often these plantings can be performed by volunteers, and have been very effective when local governments arrange the logistics, assemble the sites, and obtain the planting stock. Bernstein and Kumble (1992) described a successful volunteer effort in the Washington, D.C., metropolitan area. Herson (1992) presented practical guidance on riparian reforestation in an urban watershed. Important considerations in reforestation of stream riparian zones include:

- Site assessment
- Soil preparation
- Species selection
- Planting techniques
- Long-term maintenance

Pollution Abatement

Nonstructural approaches to pollution abatement include modifying practices in the stream, riparian zone, and watershed to prevent pollution, and include changing practices, such as lawn maintenance, construction phase activities, and recreation. For example, simply phasing construction sequencing to limit the amount of disturbed area at any given time will greatly reduce the downstream suspended-sediment levels. Streams that are affected by nutrient loads from lawn fertilization can be improved by changing the type of fertilizer used (such as a low-phosphorus mixture) or the frequency and timing of fertilizations. Nonstructural pollution abatement activities also include the use of fencing around streams and riparian zones. Fencing Sheep Creek in Colorado was effective in excluding livestock and humans (Stuber, 1985). Compared to unfenced areas, the fencing resulted in a narrower and steeper stream, increased streambank vegetation, and twice the estimated trout population.

Propagation Facilities

The use of facilities that propagate aquatic species through incubation and spawning is a common approach where it is desirable to maintain fishing in an area that otherwise would not have a sustainable sport fish population. Hatchery raised fish, however, may not always successfully survive in the wild.

Land Acquisition

Acquiring land near rivers and streams can protect the watercourse through the maintenance of buffer zones and prevention of potentially destructive land uses in the watershed. Examples of land-acquisition approaches include establishment of greenways, buffer strips, and parks. These can be purchased by the government or special foundations and trusts specially designed to provide such protection. The state of Wisconsin provides funds for the acquisition of lands (buffer zones and riparian wetland) along the Wisconsin and other protected rivers.

Land Use Regulation

Regulating land uses in the riparian zone and watershed through legislative or administrative approaches is a viable non-structural technique for restoration.

Bio-manipulation

Many fisheries management techniques involve the direct manipulation of the fish community and other organisms that may serve as prey for or be predators or competitors with the fish species of interest. Three types of activities include (1) game fish stocking, (2) control of undesirable fish species and stunted fish populations, and (3) prey enhancements to supplement food supplies.

Structural Techniques

Structural techniques are those that require some type of physical alteration of the watercourse, and may include amendments to existing man-made structures, such as dams and levees. Administrative or legislative policies and procedures that limit or regulate some activity are often inadequate because natural restorative processes can take decades. In these instances, structural techniques are needed to speed up natural restoration processes. These methods generally help in two main areas: (1) stabilization of streambanks to reduce erosion, and (2) improvement of aquatic habitat through the creation or improvement of certain morphological procedures. Karouna (1992) categorized stream restoration techniques into three areas: (1) bioengineering techniques, (2) bank

armoring techniques, and (3) aquatic habitat improvement methods. The specific techniques under each category are listed and described below (list adapted from Karouna, 1992):

<i>Bioengineering Techniques</i>	<i>Bank Armoring Techniques</i>	<i>Aquatic Habitat Techniques</i>
Live stakes	Gabions	Log drop structures
Live fascines	Riprap	V-notch gabions
Brush mattresses	Joint plantings	Log wedge
Branch packings	Root wads	K-dams
Brush layering		Jack dams
Vegetated geogrids		Boulder "S" dams
Live cribwalls		Boulder clusters
		Deflectors
		Constrictors
		Channel block
		Overhead or inclined logs
		Brush bundles
		Bank covers

A manual by the Metropolitan Washington Council of Governments (Kumble and Schueler, 1992) presents a large variety of alternative means of stream restoration and aquatic habitat protection:

1. *Bioengineering techniques.* Bioengineering methods use plants to mimic natural streambank stabilization in situations where the streambank has been eroded or lacks vegetation through some destructive process (Figs. 15.1–15.4). Plantings can provide more ecological benefits than erosion control through the addition of stream cover, shade, and improvement of bank soil conditions. Native plant species are recommended because they are generally better adapted to local environmental conditions.
2. *Bank armoring techniques.* Bank armoring methods use rock, wood, steel, and other conventional construction materials to stabilize streambanks (Fig. 15.5). These methods rarely provide significant ecological benefits other than erosion control unless they are combined with bioengineering techniques.
3. *Aquatic habitat improvement methods.* These techniques involve improving aquatic habitat through the installation of certain instream structures (Figs. 15.6–15.9). Disturbed streams often lack diverse morphological features. Habitat improvement structures can add these features:

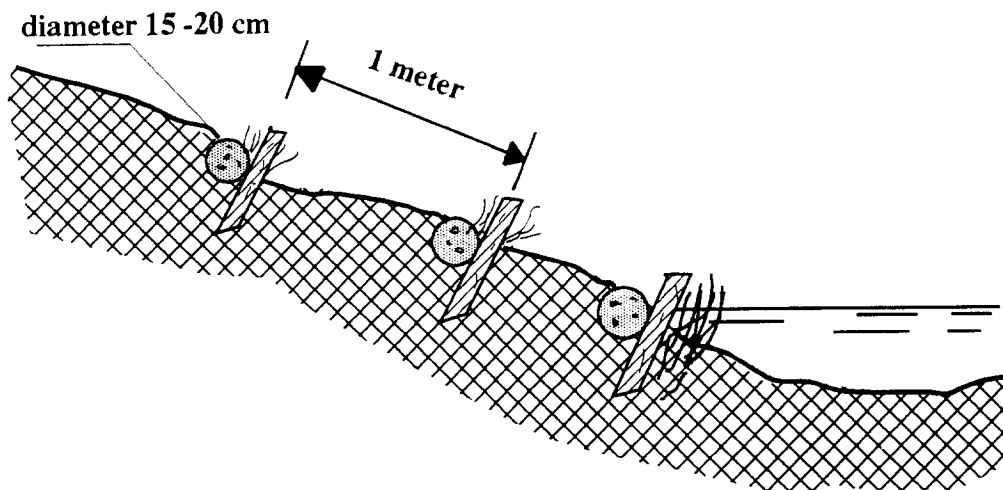


FIGURE 15.1. Live fascines for stream-bank stabilization. Live fascines are bundles of live cuttings wired together and secured into the streambank with live or dead stakes. Live fascines are used to protect banks from washout and seepage, particularly at the edge of a stream, and where water levels fluctuate moderately. (From Kumble and Schueler, 1992; drawings courtesy Metropolitan Washington Council of Governments.)

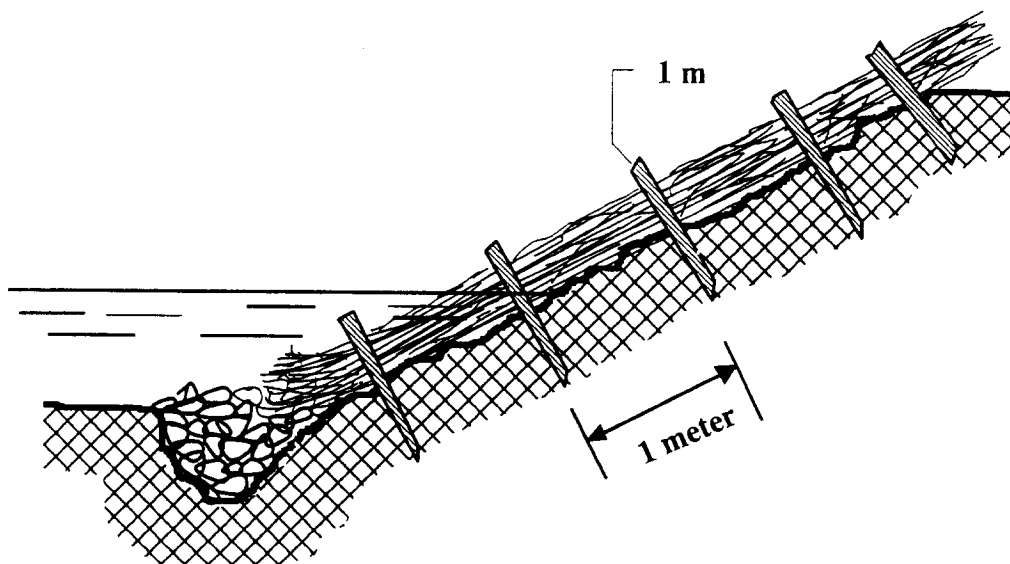


FIGURE 15.2. Brush mattresses for stream-bank stabilization. Six-foot willow switches are wired together to form a mat, which is then secured to the bank by stakes, fascines, poles, or rock fill. The toe of the slope is reinforced with a brushlayer anchored by a live fascine or riprap. The mattress should lie perpendicular to the water. (From Kumble and Schueler, 1992; drawings courtesy Metropolitan Washington Council of Governments.)

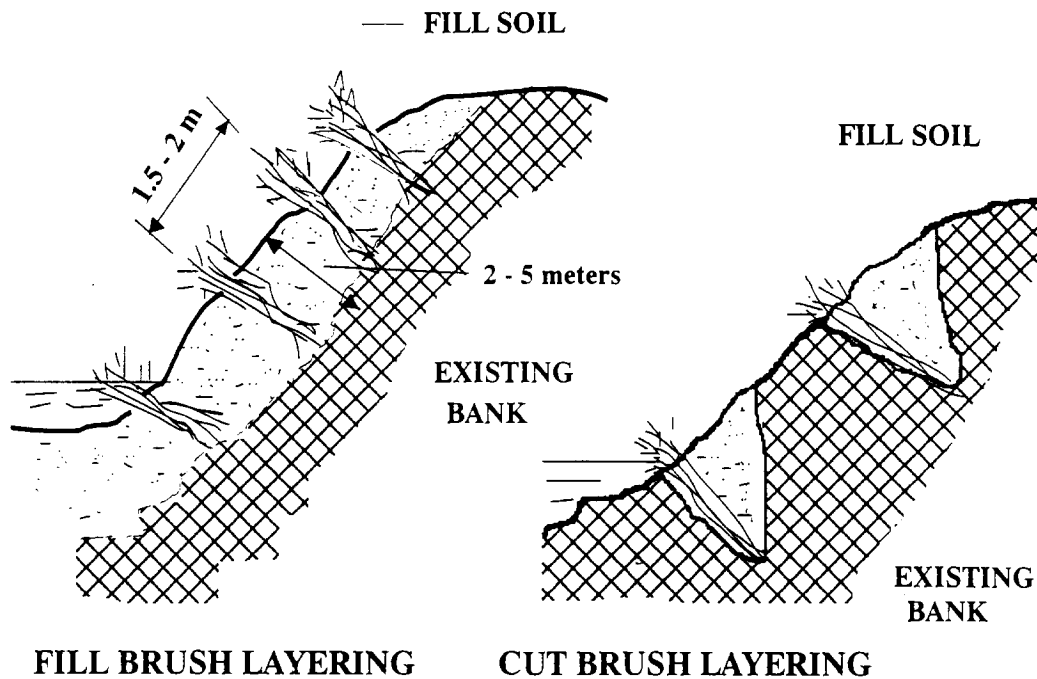


FIGURE 15.3. Brush layering for stream-bank stabilization. Live branches are placed in regular arrays on the face of a slope. The branches are oriented perpendicular to the face of the slope. On fill slopes, the plants are placed on prepared earth lifts. On cut slopes, narrow trenches are dug in the slope and plant material is placed there. Branches are 1.3–7.2 cm in diameter. A fill layer is used on heavily eroded slopes. (From Kumble and Schueler, 1992; drawings courtesy Metropolitan Washington Council of Governments.)

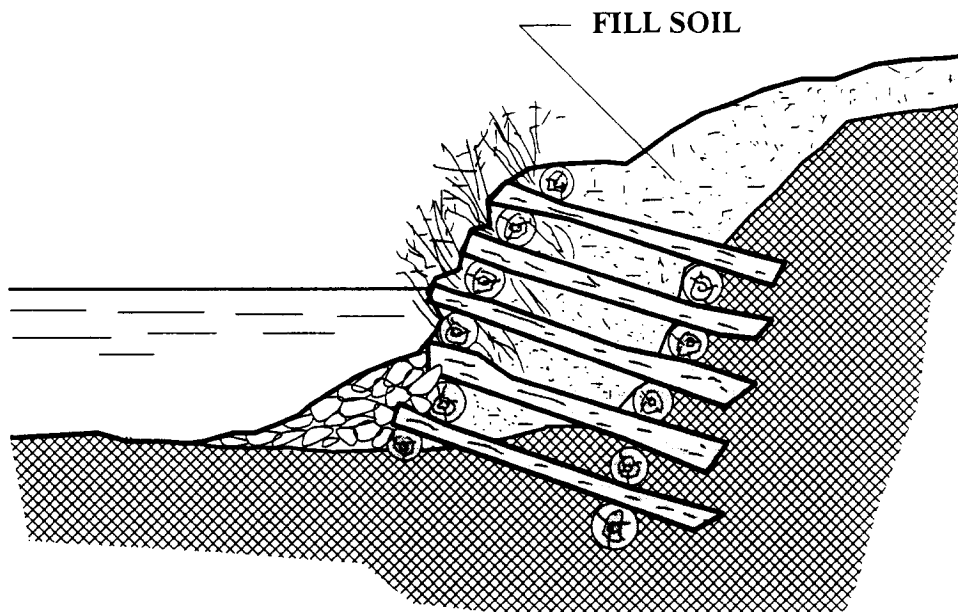


FIGURE 15.4. Live cribwall for stream-bank stabilization. The live cribwall is a rectangular framework of logs, rock, and woody cuttings. It is used to protect an eroding streambank especially at the outside bends of main channels where strong currents are present and locations where an eroding bank eventually forms a split channel. (From Kumble and Schueler, 1992; drawings courtesy Metropolitan Washington Council of Governments.)

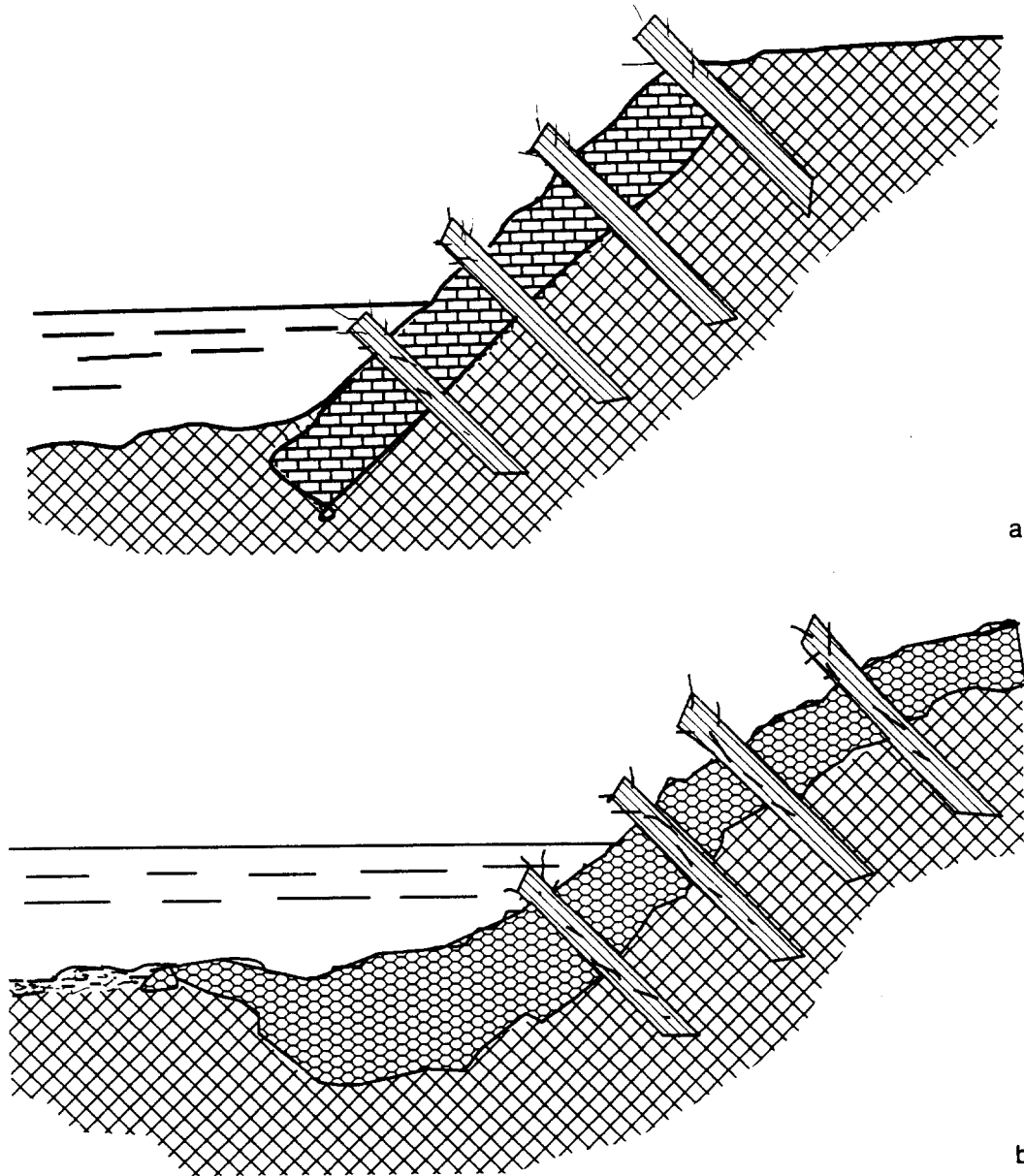


FIGURE 15.5. Joint planting—bank armoring. Willow cuttings are placed in the joints of (a) gabions or (b) riprap. (From Kumble and Schueler, 1992; drawings courtesy Metropolitan Washington Council of Governments.)

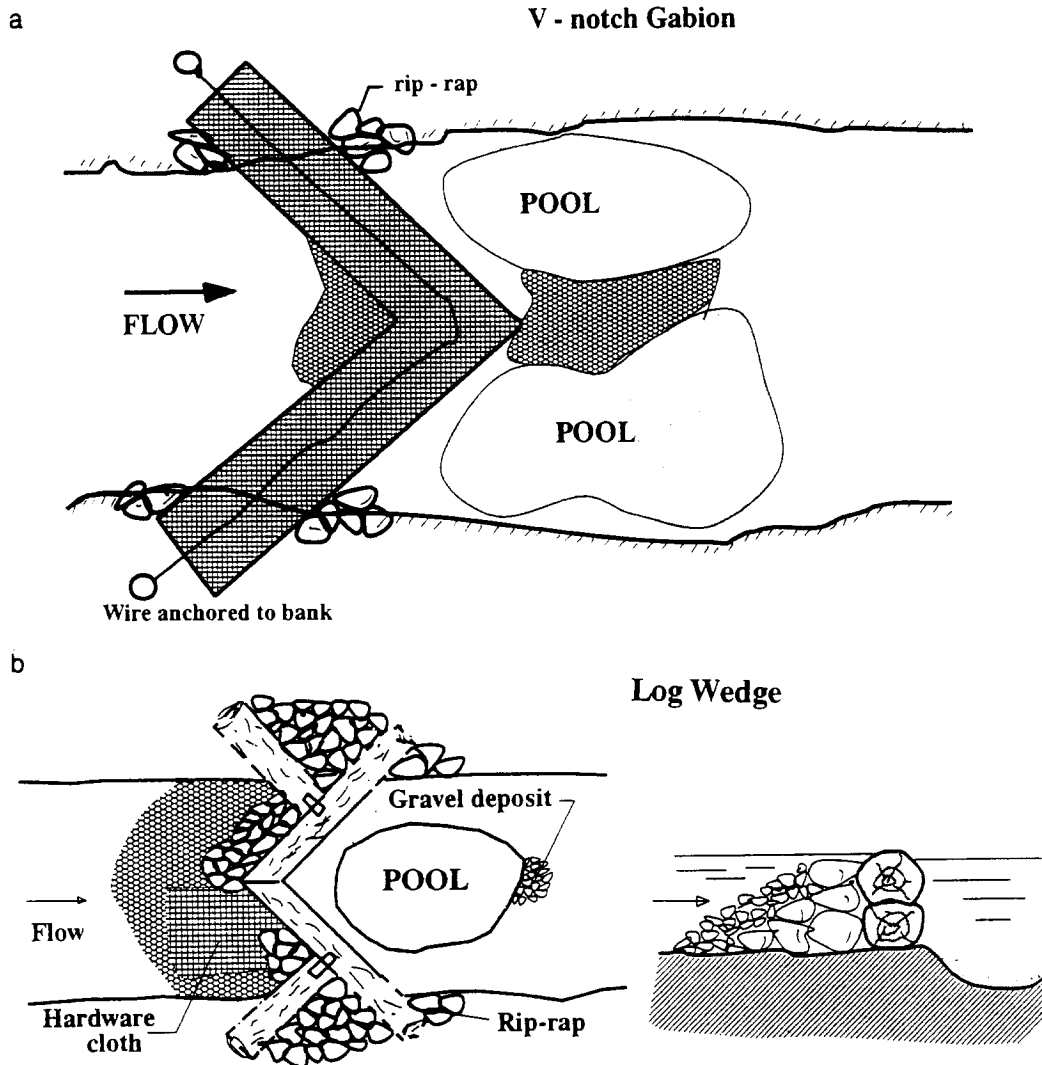


FIGURE 15.6. Aquatic habitat improvement. (a) V-Notch gabion. (b) Log wedge. These structures are designed to create and maintain large in-stream pools. They will also cause small gravel deposits to form that are used for spawning purposes. (From Kumble and Schueler, 1992; drawings courtesy Metropolitan Washington Council of Governments.)

- a. Gravel beds
- b. Structural complexity
- c. Restricted flow
- d. Riffles and pools

These features are important to the creation of spawning and rearing areas for aquatic life. The selection of a particular technique or combination of techniques depends on current habitat deficiencies, watershed conditions, and the current morphology and hydrology of the watercourse. Karouna (1992) described several general guides for the construction of habitat improvement devices:

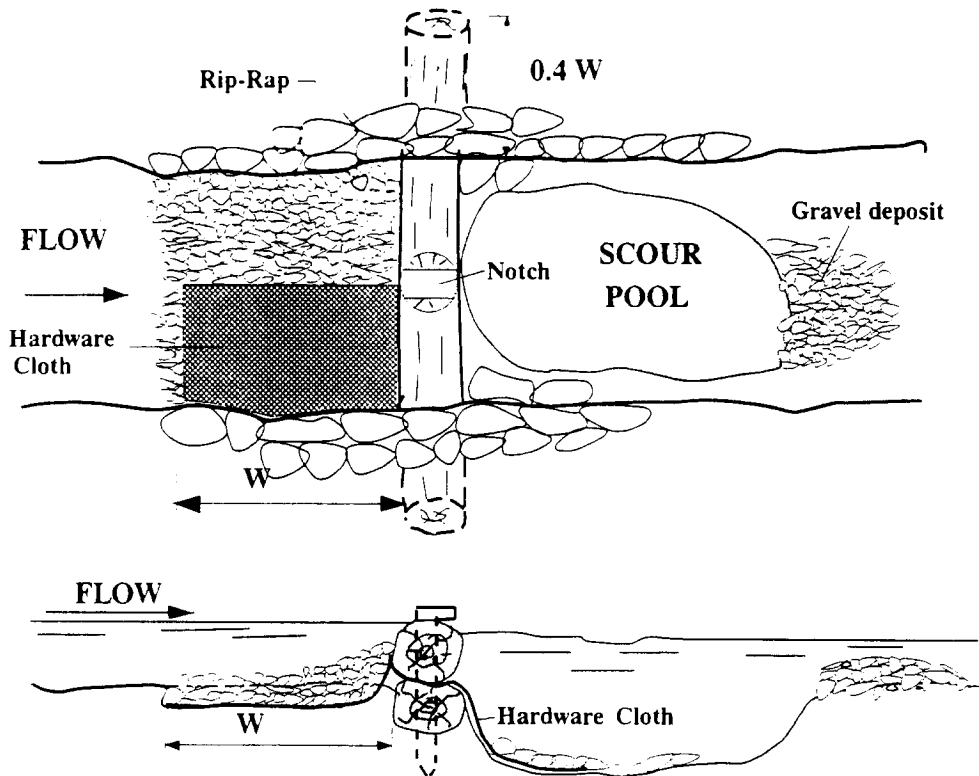


FIGURE 15.7. Aquatic habitat improvement: log-drop structures (check dams). Log-drop structures create scour pools downstream of the structure by directing water down into the stream bottom. The scour pools provide cover for fish. Gravel will deposit upstream of the log drop and also downstream of the scour pool. These deposits are often used as spawning areas. (From Kumble and Schueler, 1992; drawings courtesy Metropolitan Washington Council of Governments.)



FIGURE 15.8. Log dam for stream restoration. (Photo: H. Olem.)

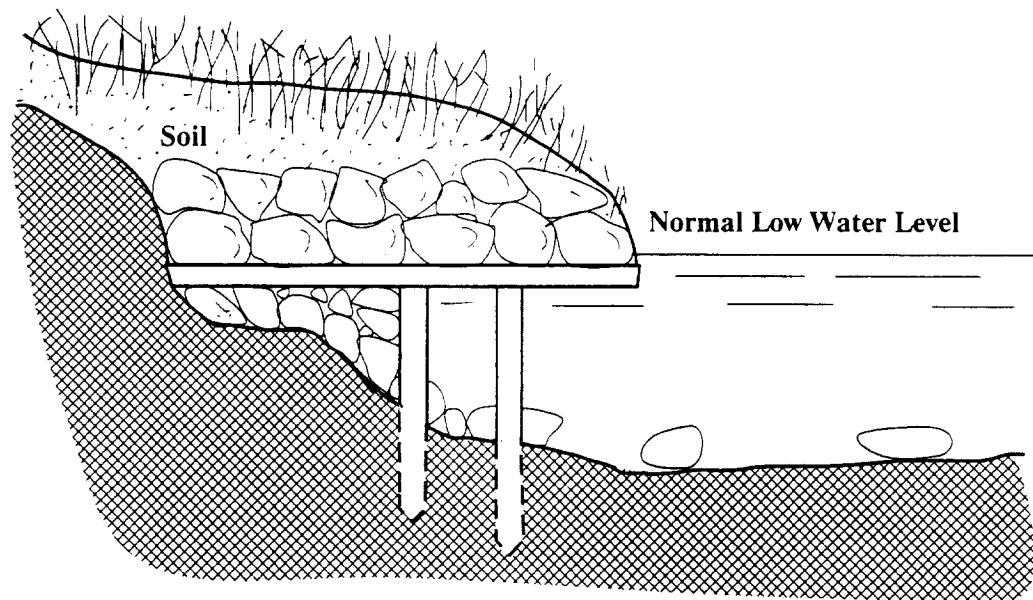


FIGURE 15.9. Aquatic habitat restoration—bank cover. These structures are installed to create an undercut bank effect, which provides cover for fish. Bank covers also serve to stabilize eroding banks. (From Kumble and Schueler, 1992; drawings courtesy Metropolitan Washington Council of Governments.)

- a. The device should be most effective during low flows, while having negligible effects during high-flow periods.
 - b. The device should be located according to habitat requirements. For example, pools form naturally at bends, while riffles form along straight sections.
 - c. The structures should not form a barrier to fish migration.
 - d. The habitat created should not adversely affect another.
 - e. Structures should not alter the stream flow of the watercourse and adversely affect unprotected streambanks.
 - f. Structures should not be built during certain inappropriate time periods. For example, periods of fish spawning and incubation should usually be avoided.
4. *Low-flow augmentation.* Low-flow augmentation is an accepted measure that provides cleaner diluting flow during times of water quality emergencies. The source of the diluting flow may include upstream reservoirs, pumping from a nearby larger body of water or from another watershed, or by recycling (by back pumping) water from cleaner and more diluted downstream flows. Interbasin transfer of water can be difficult in states that follow a riparian water rights doctrine (see Chap. 2 for the legal ramifications resulting from water rights doctrines.) The most widely known case of low-flow augmentation as a water-pollution control measure is the diversion of the

Chicago River, which originally flowed into Lake Michigan, into the Illinois River, which is a tributary of the Mississippi River. This measure, implemented in the early 1900s diverted sewage from the lake (which is a source of drinking water for the metropolis) into the Illinois River. In Milwaukee, Wisconsin, cleaner Lake Michigan water is pumped into harbor sections of the Milwaukee River (see also Chap. 1) during times of low oxygen levels in the harbor. The dissolved oxygen depression (often to anoxia) is mainly caused by algal respiration. The cause of algal biomass is upstream productive reaches stimulated by nutrients from diffuse pollution (Novotny and Bendricchio, 1988). The amount of dilution water needed can be estimated by water quality models (see Chap. 12).

5. *In-stream aeration*. In-stream aeration is a feasible measure for maintaining adequate dissolved oxygen levels for streams as well as for lakes and reservoirs. This alternative may be feasible for streams that are dystrophic and exhibit low dissolved oxygen levels, because they originate from wetlands or naturally eutrophic water bodies. In-stream aeration is accomplished by turbine aeration in power plants or by the installation of floating or submerged aerators. Also more natural cascades, spillways, and water falls provide additional oxygen (Krenkel and Novotny, 1980). In-stream aeration is simpler than aeration of stratified impoundments, which are discussed in the subsequent section, because streams are typically vertically mixed without stratification. Aeration is a temporal measure that can be used (a) when summer low oxygen levels drop even for a short time (e.g., during the night and early morning hours) below the dissolved oxygen limit for fish protection (4 to 6 mg/l, depending on the type of fish; or (b) during the winter when stream aeration is reduced by ice cover (also used for lakes and reservoirs). Aeration can be useful for streams draining wetlands that exhibit dystrophic conditions (low dissolved oxygen levels caused by high rates of decomposition of organic matter in the wetlands). The efficiency of aeration increases as the oxygen deficit increases. The Lippe River Association in the Ruhr area of Germany used aeration during the extremely dry summer of 1959 to keep an overloaded stretch of the river aerobic and odorless (Imhoff and Imhoff, 1990). Using the turbines of hydropower plants decreases the overall efficiency of the power plant. On the average 0.8–2.5 kg of O₂ can be supplied per kilowatt of power loss. Investigations into the aeration of lake Baldeney and the lower Ruhr reaches examined the efficiency and economy of floating centrifugal aerators, diffuser pipes, turbine aeration, and aeration from weirs and spillways. It was concluded that the turbine and weir aeration schemes provided the cheapest alternatives (Imhoff, 1969). The experience of

the Tennessee Valley Authority with turbine aeration was described by Davis et al. (1983).

The oxygen concentration increase at the point of aeration (the *sag point* on the dissolved oxygen longitudinal profile) is computed according to the following equation (Novotny et al., 1989)

$$\Delta C_0 = \frac{B_L D}{180 \times Q} K' \quad (15.1)$$

where

ΔC_0 = oxygen concentration increase at the point of aeration, mg/l

B_L = aeration power input in kilowatts [for weir and spillway aeration, substitute $9.81 \times \text{flow (m}^3/\text{s)} \times \text{fall height (m)}$]

D = average oxygen deficit, %

Q = flow, m³/s

K' = coefficient

The coefficient, K' , represents the oxygen yield per kilowatt-hr at 50% average oxygen deficit (mean of the upstream and downstream deficit) at the point of aeration at 20°C. When accurate data are not available, K' , can be estimated from the following:

	K' (kg O ₂ /kW-hr)
Cascades and rapids	1.5
Sharp-crested weir	0.6
Weirs and spillways	0.4
Turbine aeration	1.0
Surface aeration by floating aerators	0.5
Diffuse aeration	0.4

6. *Removal of river impoundments and stream channel restoration.* Removal of stream structural devices can be considered restoration techniques, the same as construction of new devices or modification of existing structures. For example, the Maine legislature passed a resolution in 1990 calling for the removal of the Edwards Dam on the Kennebec River by the year 2000 (Committee on Restoration of Aquatic Ecosystems, 1992). Despite some dam modifications to allow fish passage, the dam still blocks the migration of Atlantic salmon and impairs eleven other species of fish.

Many dams were built on streams for various purposes more than a hundred years ago. These were the structures that provided hydraulic head for hydropower plants, mills, and navigation. Sediment accumulated behind these dams in the last 80 or so years originated from urban and rural diffuse sources, wastewater discharges, and combined sewer overflows. In many cases this sediment is contaminated by toxic

pollutants and exhibits high sediment oxygen demand (SOD). Often the impoundment is filled with sediments and thus has essentially ceased to function.

The time may have arrived for many such installations to be removed and the river rehabilitated to a new, more natural use. The problem is what to do with the sediments that are contaminated. The North Avenue Dam in Milwaukee, Wisconsin, is an example. This small dam (about 10 meters high) was built in the second half of the nineteenth century. Its original use was to provide a navigation head for a canal between Lake Michigan and a tributary of the Mississippi River that was never built. Additional uses of the dam in the first half of the twentieth century included power production and limited recreation. After more than a hundred years of existence the dam became a water quality problem. Turbidity (Secchi disc depth) in the impoundment decreased to 0.25 meters. The dissolved oxygen in the flow typically drops by 2 mg/l while the flow passes through the impoundment.

In 1990 the dam was open for repair and left open. In a few months a new channel was eroded in 10 meters of clayey-organic sediments (Figs. 15.10 and 15.11) and the exposed mudflats were experimentally revegetated. Although the accumulated sediments have a very high content of toxic metals (lead, zinc, and others), PAHs, and PCBs, the new river bed sediments are more sandy and far less toxic than the accumulated sediments. A study has been conducted to determine the remediation of the sediments (the study was not completed by the time of the writing of this book). The options are (1) protect and stabilize the new channel and cap the exposed mudflats with a cleaner fill, followed by revegetation; (2) protect and stabilize the new channel and convert the mudflats to a riparian wetland (wetlands have higher waste assimilative capacity for toxic compounds; see Chap. 14); (3) dredge the contaminated mudflats and dispose of the dredge spoils in a safe manner (most likely in an engineered disposal site located in the harbor or the lake) and establish a new channel; (4) fence off the contaminated mudflats. The Wisconsin Department of Natural Resources has identified a number of other candidate river impoundments that have ceased their original function and now pose a water quality problem.

RESTORATION TECHNIQUES FOR LAKES AND RESERVOIRS

This section provides an overview of the various techniques available for the restoration of lakes and reservoirs. Much of the information in this



FIGURE 15.10. North Avenue impoundment on the Milwaukee River immediately after drawdown with exposed mudflats. (Photo: W. Wawrzyn, Wisconsin Department of Natural Resources.)



FIGURE 15.11. North Avenue impoundment after drawdown. The mudflats revegetated. (Photo: W. Wawrzyn, Wisconsin Department of Natural Resources.)

section is taken from Cooke and Olem (1990) and Baker et al. (1993). Six types of lake or reservoir problems are frequently encountered by lake users. These are (1) nuisance algae; (2) excessive shallowness; (3) excessive rooted plants ("weeds" or macrophytes) and their attached algae mats; (4) drinking water taste, odor, color, and organics; (5) poor fishing; and (6) acidic conditions. For each of these major problem areas, several in-lake techniques have been found to be effective, long-lasting, and generally without significant negative impact when used properly. These procedures are described in detail in Olem and Flock (1990), with regard to their underlying ecological principles and modes of action, effectiveness (including brief case histories), potential negative impacts, and additional benefits and costs. The most promising techniques are summarized here. The less well-studied or less-effective procedures are also briefly described.

Basic Assumptions

The effectiveness of each in-lake technique described in Cooke and Olem (1990), except where explicitly stated, always assumes that loadings of nutrients, silt, and organic matter to the lake have already been controlled. Most inlake procedures will be quickly overwhelmed by continued accumulation of these substances. The lake and watershed are coupled. In-lake programs can complement watershed efforts; however, such problems as algae, turbidity, and sedimentation often remain despite load reductions unless an in-lake procedure is also used.

As for restoration and management techniques that are not mentioned in this section, in nearly every case these procedures have not been described in the open scientific literature and therefore have not had the benefit of testing, discussion, explanation, and criticism needed to develop techniques of proven effectiveness and minimal negative impact. Caution should be exercised in the use of a procedure not listed here.

Selecting an Appropriate Lake-Restoration Technique

Selecting the wrong type of action may make the lake ecosystem worse off than no intervention at all. In addition, to be effective, the action must address key limiting factors or major problems affecting the lake ecosystem. Otherwise, money and effort will be expended with little to no positive returns. For these reasons, none of the methods described here should be attempted without first consulting a professional familiar with ecosystem restoration and management. Furthermore, some of these

activities require permits or are prohibited by some state regulatory authorities.

A number of the techniques described are suitable only in limited or certain circumstances. For example, in many regions of the country, fertilizing a lake to increase fish productivity would be considered highly undesirable and not a viable option because other lake uses and water quality would be adversely affected. This section provides a comprehensive review of most possible approaches; its scope is not limited by any specific objectives or philosophy. Nevertheless, in selecting the techniques to apply in any given lake, careful thought and analysis are necessary to ensure that the program (a) addresses the cause of the lake's problems and major use limitations to ensure an effective expenditure of funds and effort, (b) will not result in unexpected adverse effects on other lake uses or ecosystem components, and (c) is consistent with public values and the long-term goals for the lake and its watershed.

Costs are another important consideration in selecting an approach. The cost of implementing a given technique can vary widely, depending on the area of the country, lake size, accessibility, and other lake-specific conditions and factors.

The size of the lake, watershed, and associated drainage system can have a major influence on the feasibility and effectiveness, as well as the cost of various techniques. In-lake treatments (such as alum additions to reduce internal nutrient loads) may have relatively little effect in lakes with short hydraulic residence times and can be very costly in large lakes. The larger and more diverse a lake's watershed, the more difficult it may be to control nonpoint source inputs of nutrients and sediment. For lakes interconnected to other lakes, rivers, and streams, special consideration must be given to the potential for effects downstream or in adjacent waters. For example, exotic plants and animals may rapidly disperse throughout a drainage basin, becoming a regionwide nuisance. If undesirable fish species, such as carp, also occur in adjacent, connected waters, efforts to reduce or eliminate the species in just one lake are likely to fail.

In many cases, restoration, management, and protection begin on the land, in the surrounding watershed. Olem and Flock (1990) include a detailed discussion on watershed management, including the use of best management practices to reduce erosion and the export of sediments, nutrients, and toxic contaminants to receiving waters. Best management practices have been developed for agricultural, silvicultural, urban, and construction activities (U.S. EPA, 1987). The effectiveness, cost, and chance of negative side effects associated with selected watershed best management practices are summarized in Table 15.1.

TABLE 15.1 Summary of the Effectiveness, Costs, and Chance of Negative Side Effects Associated with Selected Watershed Best Management Practices

	Effectiveness				Cost	Chance of Negative Effects
	Sediment	Nitrogen	Phosphorus	Runoff		
<i>Agriculture</i>						
Conservation tillage	G-E	P	F-E	G-E	F-G	F-G
Contour farming	F-G	U	F	F-G	G	P
Contour stripcropping	G	U	F-G	G-E	G	P
Range and pasture management	G	U	U	G	G	P
Crop rotation	G	F-G	F-G	G	F-G	P
Terraces	G-E	U	U	F	F-G	F
Animal waste management	N/A	G-E	G-E	N/A	P	F
<i>Urban</i>						
Porous pavement	F-G	F-G	F-G	G-E	P-G	F
Street cleaning	P	P	P	P	P	U
<i>Silviculture</i>						
Ground cover maintenance	G	G	G	G	G	P
Road and skid trail management	G	U	U	U	P	F
<i>Construction</i>						
Nonvegetative soil stabilization	E	P	P	P-G	F-G	F
Surface roughening	G	U	U	G	F	P
<i>Multicategory</i>						
Streamside management zones	G-E	G-E	G-E	G-E	G	F
Grassed waterways	G-E	U	P-G	F-G	F-G	P
Interception or diversion practices	F-G	F-G	F-G	P	P-F	P
Streambank stabilization	G-E	P	P	G-E	F-G	F
Detention/sedimentation basins	G	U	U	P	P-G	F

Note: E = Excellent; F = Fair; G = Good; P = Poor; U = Unknown.

Control of Toxic Contaminants

Toxic contaminants in lakes can include metals, pesticides, and oils, in agricultural, industrial, and urban runoff. Elevated levels of these substances can degrade water quality and impact aquatic organisms. These pollutants may bioaccumulate in fish tissues, limiting the suitability of fish for human consumption. Possible corrective actions include:

- Eliminating the source of the contaminants, by applying best management practices
- Dredging and removing contaminated lake sediments
- Isolating contaminants concentrated in the bottom sediments from the overlying water column, by covering the sediments with a relatively impermeable layer, such as bentonite (a form of clay) or a plastic liner
- On-site water treatments, for example, diluting the contaminated water with "clean" water pumped in from other sources or withdrawing and treating the lake water (e.g., by chemical precipitation/coagulation and filtration), and then returning the treated water to the lake
- The addition of chemicals, such as alum (aluminum sulfate), to the lake, which may accelerate the precipitation of toxic substance(s) out of the water column into the bottom sediments
- Deep-water aeration for contaminants (e.g., some metals and ammonia) that precipitate or become nontoxic in the presence of dissolved oxygen
- Controlling changes in water level, if the exposure and suspension of contaminated sediments tend to increase the solubility and mobilization of the toxic substance
- Biomanipulation, if the potential for human exposure to bioaccumulated toxics can be reduced by altering the food chain or targeting fish species for fisheries management (e.g., avoiding game fish, such as lake trout, that are top predators and have high levels of body fat).

Relatively few field tests have been conducted to evaluate the long-term effectiveness of most of these techniques. In addition, a number of them can have potentially serious negative side effects (e.g., dredging operations may resuspend toxic contaminants and actually increase bioavailability). Not infrequently, no action is the most environmentally sound and cost-effective approach, allowing natural processes, such as sedimentation, to gradually reduce the concentration and availability of toxic substances after the source of contaminants has been eliminated.

Reducing Nutrient Loads and Nutrient Availability

Eutrophication can decrease dissolved oxygen levels, increase water temperatures, decrease water clarity, and alter the types and abundance

of phytoplankton, zooplankton, and other organisms available as fish prey. Increased nutrient loadings generally increase lake productivity, in many cases including the total production of fish. However, the accompanying changes in physical and chemical habitat often result in an undesirable shift in the types of fish species able to survive and flourish in these waters. As an indirect effect of high nutrient loads, lakes may be unsuitable for game species of particular interest to local anglers.

Olem and Flock (1990) describe methods for evaluating and controlling problems with lake eutrophication. Two basic approaches for reducing nutrient inputs and availability are described: (1) managing the watershed, and (2) various in-lake treatments. "Managing the watershed" may involve changes in land use or land-use practices in the watershed, in particular the application of best management practices (see Table 15.1). The relative merits and effectiveness of each depend on the relative importance of nutrient sources to the lake.

Watershed management reduces external nutrient loadings, while in-lake treatment procedures eliminate internal nutrient sources or reduce nutrient availability. In general, in-lake treatments will only be effective over the long-term if accompanied or preceded by efforts to reduce external nutrient loads. In conjunction with a plan for improved watershed management, in-lake treatments may serve to accelerate the process of lake recovery.

Many of the in-lake treatments just noted for reducing toxics are also effective for reducing nutrients. The following types of in-lake treatments summarized in Baker et al. (1993) have been demonstrated to effectively reduce nutrient availability in at least some circumstances:

Chemical treatments for phosphorus. Aluminium salts, such as aluminum sulfate (alum) and sodium aluminate, have a strong affinity to adsorb and absorb inorganic phosphorus and remove phosphorus-containing particulate matter from the water column as part of the "floc," or loose precipitate, that forms. The result, after the floc settles, is not only a reduction in phosphorus availability, but also, generally, a substantial increase in water clarity. Adverse effects may occur, however, if the dosage is too high. Especially in softer, more acidic waters, excessive inputs of aluminum salts can decrease lake pH and result in concentrations of dissolved aluminum in the water column that are toxic to fish and other biota. Example applications of alum treatments to reduce nutrient levels in lakes include Horseshoe and Snake Lakes, Wisconsin (Peterson et al., 1973; Garrison and Knauer, 1984), Medical Lake, Washington (Gasperino et al., 1980; Soltero et al., 1981), Annabessacok Lake, Maine (Dominie, 1980), and Dollar and West Twin Lakes, Ohio (Cooke et al., 1982).

Sediment removal. Sediments with high concentrations of phosphorus or nitrogen can serve as an internal nutrient source. If nutrients are concentrated within the upper sediment layers, dredging and the removal of these nutrient-rich sediments may substantially reduce nutrient recycling and availability (Olem and Flock, 1990). At the same time, however, dredging operations can cause extensive damage to the benthic community, which may be an important food source for fish, and may disturb fish spawning habitats, if not carefully designed and implemented. Example applications include Lake Trummen, Sweden (Andersson, Berggren, and Hambrin, 1975; Bengtsson et al., 1975; Cronberg, Gelin, and Larsson, 1975) and Lilly Lake, Wisconsin (Dunst et al., 1984).

Dilution and flushing. For small lakes, it may be possible to reduce nutrient concentrations in the water column by adding sufficient amounts of low-nutrient waters from other sources, thereby diluting and flushing the high-nutrient water out of the lake. Applications of this technique are limited, however, by the general absence of suitable alternative water supplies. Examples include Moses Lake, Washington (Welch and Patmont, 1980; Perkins, 1983).

Aeration. Phosphorus remobilization from lake sediments is generally higher in anaerobic waters (with no dissolved oxygen) than in well-aerated waters (because the oxygenated forms of iron and manganese in natural waters form an insoluble precipitate with phosphorus). Thus, aeration techniques that increase oxygen levels in deeper waters may decrease nutrient recycling and availability. Increases in dissolved oxygen also have a direct positive effect on fish. Specific aeration methods are discussed later in this chapter.

Sediment oxidation. Oxidation of the lake's sediments may also reduce the remobilization of phosphorus into the water column. Rather than add air, however, a solid, such as calcium nitrate, is added to the sediments as an oxidizing agent. The procedure is termed RIPLOX, after its originator (Ripl, 1976), and is still experimental. Example applications include Lake Lillesjon, Sweden (Ripl and Lindmark, 1978) and Long Lake, Minnesota (Willenbring, Miller, and Weidenbacher, 1984).

Water withdrawals from the hypolimnion. The colder, deeper layers of a thermally stratified lake or reservoir generally have higher nutrient concentrations than waters in the epilimnion or metalimnion. Selectively withdrawing these nutrient-rich waters, using a siphon or deep-water outlet in a dam, can decrease the quantity of nutrients stored and recycled in the waterbody. However, hypolimnetic withdrawals may also trigger thermal instability and lake turnover. In addition,

the discharge of these nutrient-rich, often anaerobic waters from the hypolimnion of eutrophic lakes can cause adverse effects in downstream receiving waters. Example applications include Lake Wononscopomuc, Connecticut (Kortmann et al., 1983) and Lake Mauensee in Switzerland (Gachter, 1976; Cooke et al., 1993).

All of these in-lake treatments require significant expenditures on equipment, chemicals, and labor. Case studies, as well as additional information on the methods, costs, and potential negative side effects of each approach, can be found in Olem and Flock (1990) and Cooke et al. (1993).

Increasing Dissolved Oxygen Levels

Low levels of dissolved oxygen may occur in lakes as a result of natural conditions, as well as cultural eutrophication. The lowest concentrations tend to occur in the deeper waters of the hypolimnion during thermal stratification in late summer, during long periods of snow and ice cover in winter, or in dense macrophyte beds at night or following long periods of cloud cover.

One important option to consider for lakes with problems with low dissolved oxygen is to manage the fisheries for species able to tolerate relatively low levels of oxygen, or that do not inhabit areas of the lakes (such as the hypolimnion) that experience oxygen depletion. For example, many salmonid species require both relatively high levels of dissolved oxygen and, because of their intolerance of warmer water temperatures, must reside in the cooler waters of the hypolimnion during summer. Therefore, sustaining a fisheries for salmonids may require a costly and continuing effort to aerate the lake's hypolimnion during some or all of the summer. Alternatively, if the problems with oxygen depletion are moderate, the lake may be able to support a cool-water or warm-water fisheries without extensive restoration efforts.

Problems with low dissolved oxygen can also be alleviated by one or more of the following:

- Decreasing the quantity of organic matter decomposed in the lake (the major process consuming oxygen) by (a) limiting the export of organic materials from the watershed to the lake, in particular excessive exports associated with human activities, such as runoff from feedlots or direct discharges of sewage wastewaters, (b) dredging to remove organic-rich sediments, and (c) decreasing in-lake productivity, by reducing nutrient loads and nutrient availability

- Increasing photosynthesis (an oxygen-generating process), especially during times and in locales (deeper waters of the hypolimnion) subject to oxygen depletion, primarily by increasing light penetration
- Destratifying the lake (artificial circulation), bringing low oxygen waters in the hypolimnion in contact with the lake surface and the well-oxygenated waters of the epilimnion
- Direct aeration

The approaches and optimal design criteria for lake aeration projects vary somewhat for systems installed to alleviate problems with winterkill (oxygen depletion during winter) as opposed to low levels of dissolved oxygen in the hypolimnion during summer. During winter, the goal is not to aerate the entire body of water, but instead to create an oxygen-rich refuge area for fish, generally near the lake surface. Major design concerns include problems with equipment ice-up and the need, for safety reasons, to minimize the loss or weakening of the lake's ice cover. Hypolimnetic aeration systems must deal with the more difficult problem of aerating waters at greater depths. Where the objective is to establish or maintain a cold-water fishery, hypolimnetic aeration must be achieved without disturbing the lake's thermal stratification. Otherwise, low levels of dissolved oxygen can be avoided by preventing thermal stratification, that is, through artificial circulation of the water column. In both cases, during both winter and summer, maximum reliability at minimal cost is an important design feature.

Pump and Baffle Aeration System

One type of winter aeration system is the pump and baffle. Oxygen-poor water is extracted from a nearshore area of the lake, pumped to the top of a chute located on shore, and then allowed to cascade over a set of baffles (constructed of wooden boards). The turbulence created as the water passes over the baffles helps to reaerate the water. The re-oxygenated water is then returned to a different part of the lake, away from the intake area, creating a zone of oxygen-rich water.

Generally, approximately 10% of the lake's volume should be aerated (S. McComas, 1993, pers. comm.). Thus, for a lake with a 40-ha area with an average depth of 1.8 m (volume 72,000 m³), the objective would be to aerate 7000 m³ of water. A typical rig uses a 7.5-kW (10-hp) motor with a 150-mm or 200-mm (6- or 8-in.) pump, that delivers between 10.0 and 190 m³/sec (1600 and 3000 gallons per minute or 2.3 to 4.3 million gallons per day). Thus, a suitable refuge area for the example lake could be created within five to ten days. For larger lakes, multiple rigs may be needed and, as a result, the pump and baffle technique may be too costly.

The velocity of water discharged into the lake must be neither too low nor too high. At low velocities, the amount of water released would be too small to produce an oxygen-rich zone of sufficient size. At high velocities, the reoxygenated water discharged will mix too thoroughly with the oxygen-poor lake water, elevating the entire lake's oxygen level only slightly (and too little to significantly improve fish survival), rather than creating a smaller refuge area with higher oxygen levels. The level of oxygen that must be maintained depends on the fish species in the lake.

Freeze-up can be a major problem with pump and baffle systems, especially during cold winter days. Freeze-up can occur at the water intake, on the chute, or at the water discharge. For example, the chute may become top heavy, due to ice buildup, and fall over. Thus, the system must be checked daily to ensure proper operation.

Pump and baffle systems have several major advantages relative to other aeration techniques. In particular, when properly operated, only a small area of the lake's ice cover is opened. Open areas and thin ice are safety hazards, for which the operator of an aeration system is liable. All of the major pieces of equipment are on shore. In addition, the chute can be mounted on a trailer and moved from one lake to another or to different areas of the lake as needed. Generally, to prevent winterkill, aeration will be required for about two months, depending on winter conditions. By monitoring dissolved oxygen levels in the lake, the system can be operated only during those times and winters when needed.

Pump and baffle systems have been built by lake associations, or may be purchased as a unit from a number of manufacturers.

Artificial Circulation

Artificial circulation eliminates thermal stratification, or prevents its formation, either by mechanical pumping or through the injection of compressed air from a pipe or ceramic diffuser at the lake's bottom (Fig. 15.12). The rising column of bubbles, if sufficiently powered, will produce lakewide mixing. As a result, the conditions that create hypolimnetic oxygen depletion (isolation of the deeper waters from the atmosphere, with little to no primary production in these deeper, darker waters) are eliminated.

The most common cause of failure in artificial circulation projects is the lack of sufficient air flow to produce satisfactory mixing. On average, about $1.1 \text{ m}^3/\text{min}$ of air flow is required per hectare of lake surface to adequately mix the lake and elevate levels of dissolved oxygen (see Lorenzen and Fast, 1977). In general, it is easier and more effective to apply the mixing energy early enough to prevent stratification, rather than attempting to turnover an already stratified lake (Burns, 1988).

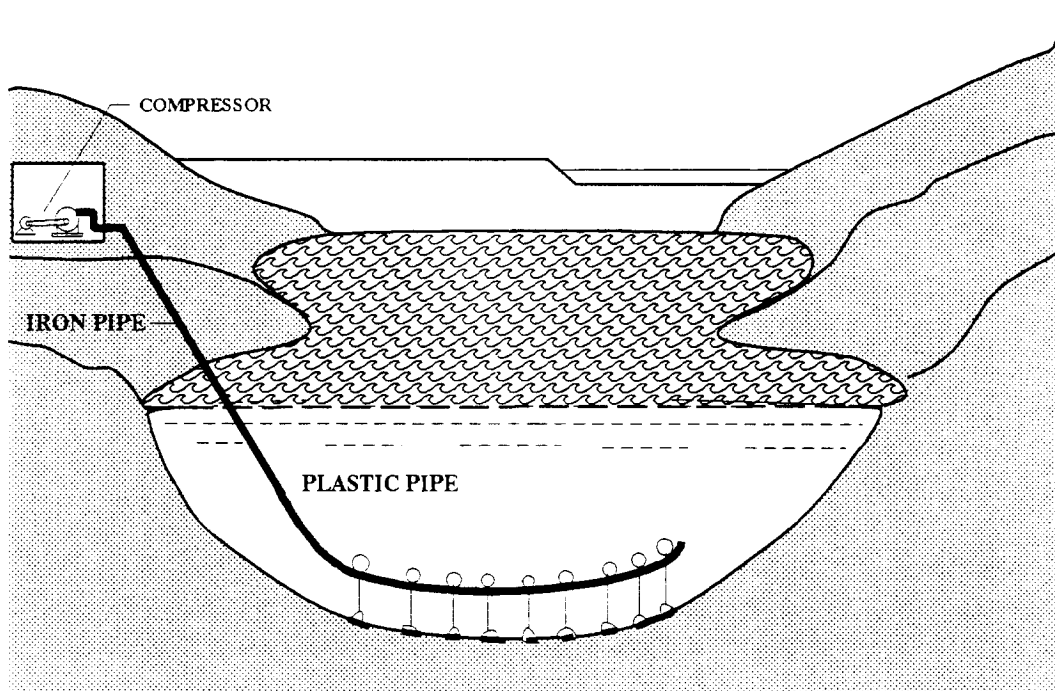


FIGURE 15.12. Schematic of lake (reservoir) destratification by aeration.

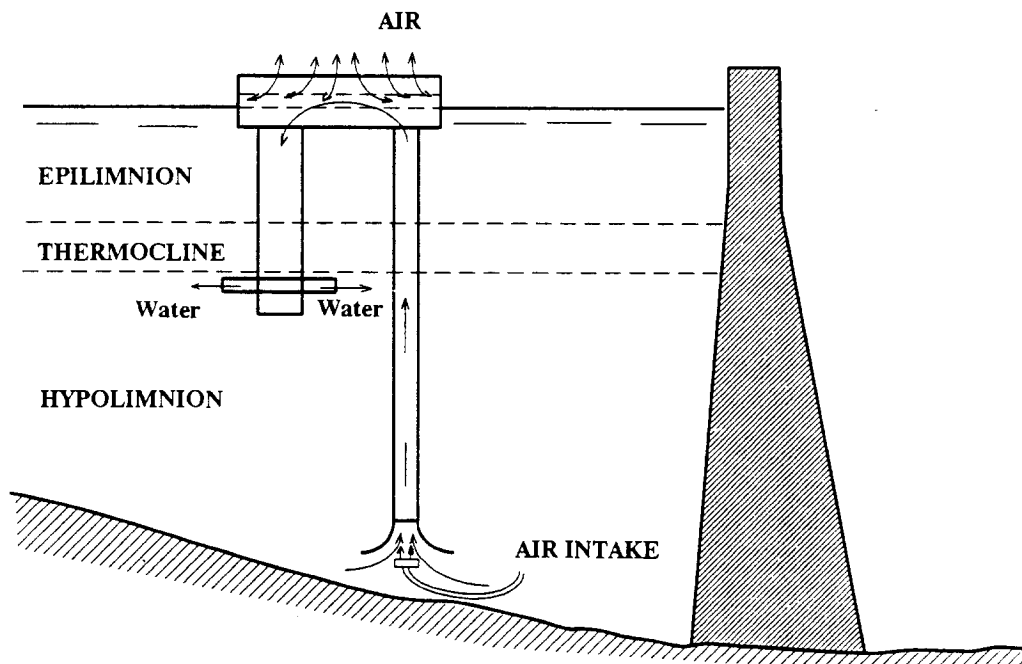


FIGURE 15.13. Hypolimnetic aeration.

Artificial circulation is one of the most commonly used lake restoration techniques (Cooke et al., 1993). Examples of its utility for improving fisheries yields include Parvin Reservoir, Colorado (Lackey, 1972) and Corbett Lake, British Columbia (Halsey, 1968). The technique is best used in lakes that are not nutrient limited; nutrient concentrations are often higher in the hypolimnion and, as a result, mixing can stimulate increased algal growth. In addition, artificial circulation is not a viable option for cold-water fish species, which use the hypolimnion as a thermal refuge during summer.

Hypolimnetic Aeration

Hypolimnetic aerators can be used to increase oxygen levels in the hypolimnion without disturbing the lake's thermal stratification. An airlift device is used to bring cold hypolimnetic water to the surface. The water is aerated by contact with the atmosphere; gases such as methane, hydrogen sulfide, and carbon dioxide, which accumulate under anaerobic conditions, are lost; and then the water is returned to the hypolimnion (Fig. 15.13).

Hypolimnetic aerators require a large hypolimnion to work properly and are generally ineffective in shallow lakes and reservoirs. Costs depend on the amount of compressed air needed, which is a function, in turn, of the area of the hypolimnion, the rate of oxygen consumption in the lake, and the degree of thermal stratification (Kortmann, 1989). Example applications include Waccabuc Lake, New York (Fast, Dorr, and Rosen, 1975), Larson and Mirror Lakes, Wisconsin (Smith, Knauer, and Wirth, 1975), and Tory Lake, Ontario (Taggart and McQueen, 1981).

Oxygen Injection

Recent studies have shown that it is often more cost-effective and practical to inject pure oxygen into the hypolimnion, as opposed to air injections or aerating the hypolimnion via air-lift systems (Aquatic Systems Engineering, 1990). At Richard B. Russell Reservoir, Georgia, dissolved oxygen levels in the hypolimnion have been increased from <3 mg/l to >9 mg/l, with an oxygen transfer efficiency of about 75% (Gallagher and Mauldin, 1987; Mauldin et al., 1988). Liquid oxygen is stored in tanks on-site and connected to several supply heads submerged and anchored in the reservoir. Flexible membrane diffusers mounted on the supply heads are used to maximize absorption efficiency and minimize maintenance requirements. Flexible membrane systems should last 2–6

years (or 10 years or more if operated ≤ 6 months per year); the compressor and distribution system should last substantially longer (estimated 30-year life).

Snow Removal to Increase Light Penetration

Snow removal from the lake surface, to increase light penetration and photosynthesis (oxygen generation) under the ice, is a low-tech, low-cost alternative to aerators that may be sufficient to prevent winterkill in lakes with marginal levels of dissolved oxygen (S. McComas, 1993, pers. comm.). Snow is a much more effective absorber of light than is ice. While 85% of the available light will penetrate 12.5 cm of clear ice, 12.5 cm of snow over 7.5 cm of ice will block out almost all light. Even thin layers of snow can greatly decrease light penetration, decreasing primary productivity, and thus leading to oxygen depletion and winterkill.

Using volunteers with snowplows, snow removal can be completed at relatively low cost. Alternating strips of snow removal are recommended, rather than clearing the entire area and stockpiling the accumulated snow in a single location. Thirty percent or more of the lake surface should be cleared. In general, snow removal is more effective at preventing winterkill in shallow lakes with abundant rooted macrophytes than in deep lakes with phytoplankton as the dominant primary producer (S. McComas, 1993, pers. comm.).

Additional information on lake aeration systems can be found in Olem and Flock (1990), Cooke et al. (1993), and Lorenzen and Fast (1977). The most cost-effective approach for increasing dissolved oxygen levels depends on the size (area and depth) of the lake, nature and causes of the problem, and fisheries management objectives.

Liming Acidic Lakes

Waters may be naturally acidic, for example, in regions with naturally acidic soils and large inputs of organic acids, or acidified as a result of acid mine drainage, acidic deposition, or other anthropogenic sources of acids. Extensive research, using a variety of neutralizing agents as well as application techniques, has been conducted in recent years to refine and test methods for neutralizing acidic lake waters. There are five basic approaches to treating acidic lakes:

- *Limestone addition to the lake surface.* Small limestone particles, limestone powder, or a limestone slurry are dispersed via boat, plane, or helicopter over the lake. (Or, during winter, the limestone can be spread on the ice by truck, entering the lake in the spring as the ice melts.) Direct addition of limestone to the lake surface is the

most commonly employed method to date for decreasing lake acidity. Because limestone is used for agricultural liming, it is usually available at low cost. However, the cost of the limestone dispersal can be significant, particularly for remote lakes without road access. In addition, repeated applications are needed; lakes with short water-retention times may need to be treated annually.

- *Injection of base materials into the lake sediment.* Limestone, hydrated lime, or sodium carbonate can be injected into the lake sediments, resulting in a gradual decrease in lake acidity. This technique is largely experimental, however, and limited to small, shallow lakes with soft organic sediments and road access for transport of the application equipment. The treatment may last substantially longer than surface applications, but at the same time the lake's benthic community is disturbed, turbidity may increase, and the costs are higher.
- *Mechanical stream doser.* Lake acidity may be decreased by neutralizing the acidic waters in upstream tributaries. Mechanical dosers are automated devices that release dry powder or slurred limestone directly into the stream, with the quantity of material added controlled by monitors of stream flow or stream chemistry. The treatment is continuous, expensive, and generally not recommended for lakes unless all other alternatives have been ruled out.
- *Limestone addition to the watershed.* Limestone is spread on all or parts of the lake's watershed, decreasing the acidity of runoff and shallow ground-water flow into the lake. Although the costs of one application are higher, the overall costs may be lower than for limestone applications to surface waters, because the effects are much more long-lasting. Watershed liming may be especially appropriate for lakes with short retention times (less than six months).
- *Pumping of alkaline ground water.* Where abundant supplies of alkaline ground water are available, these waters may be discharged directly into lakes or lake tributaries, decreasing acidity. Applications of this method have been limited, however, and the costs and effectiveness of this approach are not well known.

Several books have been published recently on methods for liming lakes and streams, including Olem (1991), Olem et al. (1991), and Brocksen, Marcus, and Olem (1992).

Spawning Habitat Management

Fish production in some waters may be limited by the availability of suitable sites for natural reproduction, or the poor quality of available sites, resulting in relatively low reproductive success. The types of hab-

itats required for spawning and the factors that influence spawning and reproductive success vary greatly among species. Thus, the methods employed for spawning-site management are also, to some degree, species-specific. The types of activities fall within three broad categories: (1) efforts to protect existing spawning habitat, (2) spawning habitat improvements, and (3) the construction of new spawning sites.

The first task required to either protect or improve spawning habitat is to locate existing sites in the lake or associated streams used by target fish species. Some species spawn in dispersed areas throughout the lake, while others concentrate activities within fairly localized sites used consistently year after year. For fish species that spawn in shallow waters, finding spawning sites can be relatively easy. Potential spawning areas should be visited regularly during the spawning season by trained observers looking for spawning adults, spawning nests, or egg masses. For species that spawn in deeper waters or that provide little noticeable evidence of spawning activity (e.g., that do not build nests or lay their eggs in visible masses), finding spawning areas will be much more difficult and time-consuming, requiring SCUBA, for underwater observations, or sampling gear to detect the presence of eggs in the bottom sediments or water column.

Protection of Spawning Habitat

Many fish species spawn in relatively shallow nearshore areas, subject to disturbance from swimmers, boat traffic, and runoff from the adjacent watershed. It may be desirable to limit construction, land uses, or fertilizer applications in watersheds adjacent to critical spawning areas (including tributary streams used for spawning), or to divert and treat runoff that is high in suspended solids (to prevent excessive siltation that would decrease egg and larval survival).

Fish species that guard their spawning nests, such as largemouth and smallmouth bass and most sunfish species, are particularly susceptible to disturbances from swimming and boating. If repeatedly disturbed, males may eventually desert their nests, resulting in poor survival of eggs and fry. During spawning season, important spawning sites can be identified with buoys, and boat traffic and swimming restricted from these areas. Waves rebounding off retaining walls may also drive off bass and sunfish males guarding nests. Shoreline structures that better absorb wave energy, such as riprap or vegetation, are preferable near important spawning areas.

Spawning Habitat Improvements

Siltation is a major cause of degraded spawning habitat. Some species, such as bass, crappies, and bluegill, sweep the nest area off with their tails

before spawning, and thus moderate siltation is not a major concern. Others, however, such as walleye, do not; the build up of silt in walleye spawning areas can significantly decrease egg survival and may limit or prevent spawning activity.

Silt can be removed from spawning areas by using a water pump to blow the silt and algal growth off the rocks. For example, a 7.5-cm (3-in.) pump can generate enough force to remove silt from the rock face or turn cobble-sized rocks over to expose a new face. If mounted on a pontoon or raft, several spawning sites can be cleaned in a half-day (S. McComas, 1993, pers. comm.).

Brook trout have fairly restrictive spawning requirements and, as a result, the availability of suitable spawning areas is often an important factor limiting brook trout natural reproduction and productivity. Brook trout prefer to spawn in areas with upwelling, well-oxygenated ground water, in the littoral zone, tributary streams, or the lake outlet. Plants and sediments can accumulate on these underwater springs, obstructing ground-water flow and preventing brook trout spawning. The locations of underwater springs can be confirmed by inserting a small-diameter pipe (e.g., a 5-cm (2-in.) PVC-type pipe) into the lake bottom. If upwelling ground water is present, the water level in the pipe will rise above the lake level. Small-scale dredging techniques can then be used to remove the blanket of material obstructing the flow. Carline (1980) provides examples of the success of this technique for spawning-site improvements in Wisconsin lakes.

Finally, some fish species, including muskellunge and northern pike, spawn in flooded marshes and other heavily vegetated areas in bays or river floodplains. These heavily vegetated flooded areas also serve as important nursery areas for the young fish of many species. Therefore, in lakes and reservoirs with controlled water levels, reproductive success can be increased by raising water levels during the spring to coincide with fish spawning and the occurrence of early life stages. The role of water-level management for managing fisheries in lakes and reservoirs is discussed further later in this chapter.

Aquatic Plant Management

In lakes and reservoirs where thick beds of macrophytes cover a high proportion of the lake bottom, an aquatic plant control program may be needed to improve yields of large, predatory game fish and increase the growth rates of panfish species, such as bluegill and white and black crappie. For uses such as swimming and boating, minimizing macrophyte beds are desired. For fisheries management, on the other hand, moderate

growths of aquatic plants enhance the fisheries. The complete elimination of macrophyte beds may be as harmful to fisheries as are excessive plant growths.

The objective of aquatic plant management is therefore to provide the appropriate amount of aquatic plants, taking into account the effects of macrophytes on fish communities, other lake uses (e.g., swimming and boating), nutrient cycles, and aesthetics. Macrophytes and terrestrial vegetation also help to stabilize the lake bed and shoreline, reducing problems with lakeshore erosion and high turbidity.

Excessive plant growths, as a result of eutrophication or the inadvertent introduction of an exotic macrophyte species, are a common lake problem. Olem and Flock (1990) provide a thorough discussion of various approaches to controlling nuisance plant growths:

Sediment removal and sediment tilling. Lakes can be dredged to remove sediments and deepen the lake, so that less of the lake bottom receives adequate light for macrophyte growth. The maximum depth at which macrophytes are able to grow depends on water transparency and the plant species. Hydrilla, a nuisance exotic plant in southern waters, can grow at lower light intensities than native plants (Canfield et al. 1985), making control through lake deepening a difficult task. Reductions in nutrient loads, to control eutrophication, can increase lake transparency, increasing the depth at which macrophytes can grow and countering the effectiveness of dredging to reduce macrophyte growth. Sediment removal and tilling [e.g., rototilling using cultivation equipment (Newroth and Soar, 1986)] can also be used to disturb the lake bottom, tearing out plant roots for short-term macrophyte control. Both dredging and tilling can have negative side effects, including destruction of the benthic community and an increase in the turbidity and siltation.

Water-level drawdown. In lakes where water levels can be controlled, lake levels can be lowered to expose macrophytes in the littoral zone to prolonged drying and/or freezing. Some species of plants are permanently damaged by these conditions, killing the entire plant, including roots and seeds, with exposures of 2–4 weeks. Other plant species are unaffected or even increase. Water-level management to control macrophyte growths and for other purposes is discussed further in the next section.

Shading and sediment covers. Covers can be placed on the water or sediment surface, as a physical barrier to plant growth or to block light (Engel, 1984). Sediment covers, made of polypropylene, fiberglass, or other similar material, can effectively prevent growths in small areas,

such as near docks and swimming areas, but are generally too expensive to install over large areas. Applications of silt, sand, clay, or gravel have also been used, but plants eventually root in them. Shading, to reduce growth rates, can be provided by floating sheets of polyethylene (Mayhew and Runkel, 1962) or by planting evergreen trees along the lakeshore (Engel, 1989).

Introduction of grass carp. Grass carp is an exotic fish species that feeds on macrophytes. Grass carp do not consume aquatic plant species equally readily. Generally, they avoid alligatorweed, water hyacinth, cattails, spatterdock, and water lily. The fish prefer plant species that include elodea, pondweeds (*Potamogeton* spp.), and hydrilla. Low stocking densities can produce selective grazing on the preferred plant species, while other less preferred species, including milfoil, may even increase. Overstocking, on the other hand, will eliminate the weeds. Biological control has the objective of achieving long-term control of plants without introducing expensive machinery or toxic chemicals. However, the use of grass carp for aquatic plant management is only allowed in certain states.

Introduction of insects that infest macrophytes. Several exotic insect species have been imported to the United States and approved by the U.S. Department of Agriculture for use in macrophyte control. Each insect species grows and feeds on only select target plant species. In particular, insects have been used in southern waters to aid in the control of alligatorweed and water hyacinth (Sanders and Theriot, 1986; Haag, 1986). Because insect populations tend to grow more slowly than the plants, insects work best when used in conjunction with another plant control technique (e.g., harvesting or herbicides). No significant negative side effects from insect infestations have been documented (Olem and Flock, 1990).

Mechanical harvesting. Mechanical harvesters, constructed on low-draft barges, can be used to cut and remove rooted plants (Fig. 15.14) and floating water hyacinths (Fig. 15.15). Cutting rates range from 0.1 to 0.3 ha/hr, depending on machine size; Cooke et al. (1993) provide a listing of commercially available plant cutters. Harvesters can effectively clear an area of vegetation, although the benefits are only temporary. Rates of plant regrowth can be rapid (within weeks), but can be slowed if the cutter blade is lowered into the upper sediment layer (Conyers and Cooke, 1983). Cut plants are removed from the lake, eliminating an internal source of nutrients and organics with potential long-term benefits. However, some plant species, such as milfoil (Nicholson, 1981), may be fragmented and dispersed, and actually increase in abundance after harvesting operations. Also, small fish



FIGURE 15.14. Small aquatic weed harvester.

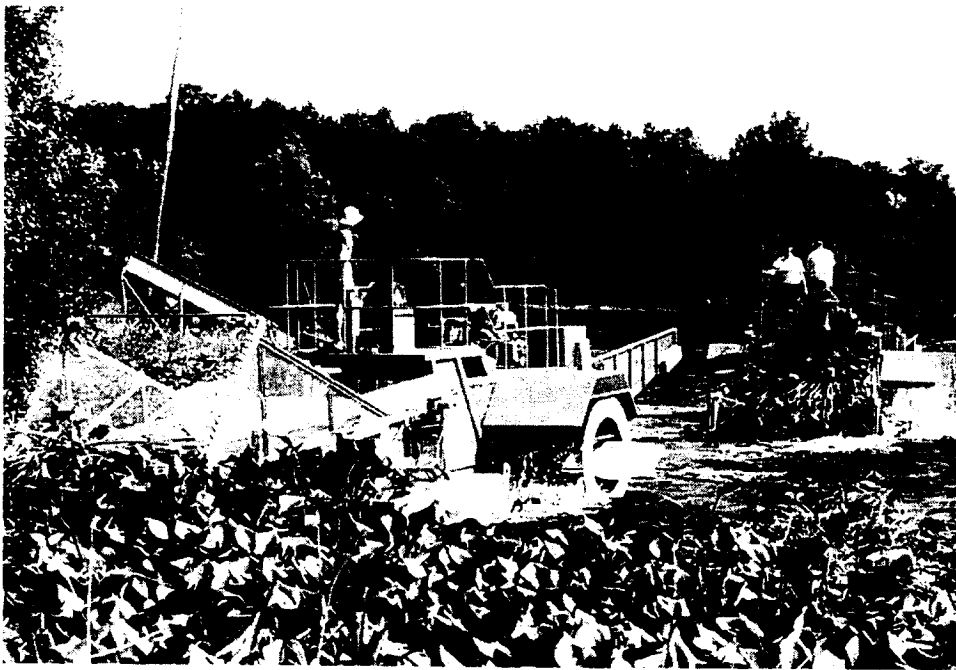


FIGURE 15.15. Water hyacinth harvesting.

can be caught and killed by mechanical harvesters. Thus, harvesting operations should precede spawning periods and/or avoid important spawning and nursery areas.

Herbicides. Herbicides used to kill aquatic plants include Diquat, endothall, 2,4-D, glyphosate, and fluridone (Olem and Flock, 1990). Although herbicide treatments can rapidly reduce macrophyte growths, the benefits are short-term and the potential for negative side effects is high. Plants are left in the lake to die and decompose, releasing plant nutrients and, in some cases, causing oxygen depletion and algal blooms. Plants generally regrow after several weeks or months, or may be replaced by other, more tolerant macrophyte species. Most chemicals currently approved are toxic to aquatic organisms and humans only at relatively high doses. Little information is available, however, on the long-term ecological consequences of herbicide use. Herbicide applicators must be licensed, have adequate insurance, wear protective gear, use only EPA-approved chemicals, and follow label directions exactly. Generally, because herbicides do not remove nutrients or organics from the lake or address the cause of the aquatic plant problem, herbicides should be used only where other techniques are unacceptable or ineffective. Westerdahl and Getsinger (1988) provide guidelines for herbicide use and application.

The relative effectiveness, costs, and potential for negative side effects for each of the preceding techniques is summarized in Table 15.2. In some lakes, especially newly impounded reservoirs and acidic bodies of water, the problem may be too few rather than too many macrophytes. In

TABLE 15.2 Comparison of Lake Restoration and Management Techniques for Control of Nuisance Aquatic Weeds

Treatment One Application	Short-Term Effectiveness	Long-Term Effectiveness	Cost	Chance of Negative Effects
Sediment removal	E	E	P	F
Drawdown	G	F	E	F
Sediment covers	E	F	P	L
Grass carp	P	E	E	F
Insects	P	G	E	L
Harvesting	E	F	F	F
Herbicides	E	P	F	H

Source: From Olem and Flock (1990).

Note: E = Excellent; F = Fair; G = Good; P = Poor; H = High; L = Low, based on the consensus judgment of 12 lake restoration experts (Olem and Flock 1990).

new impoundments, it may be desirable to introduce (transplant) suitable plants, such as sago pondweed, water celery, and lily pads, or to raise water levels to flood vegetated areas. In lakes with fluctuating water levels, terrestrial plants, such as winter wheat or ryegrass, can be planted in the exposed lake bed during periods of water drawdown. These plants provide both erosion control during drawdown and fish cover when the lake is reflooded, with no risk of developing nuisance levels or excessive plant growths. Howells (1986) and Evans (1989) provide guidelines for establishing plants in water-fluctuation zones and for managing plant communities in southern reservoirs.

Water-Level Management

Controlled changes in water level, at the right time, place, and magnitude, can be beneficial and provide an important management tool for maintaining and improving a lake ecosystem and fisheries. Baker et al. (1993) describe the use of water-level management for a number of purposes:

- Water-level drawdowns for vegetation control
- Water-level drawdowns for overcrowded, slow-growing fish populations
- Water-level drawdowns to enhance fish cover
- High-water levels in the spring to provide increased spawning and nursery areas
- Water-level drawdowns during spawning to reduce the reproductive success of undesirable fish species

In lakes with a well-maintained outlet structure and drawdown capability, the dollar costs of a water level-management program are minimal and any or all of the preceding actions can be useful for fisheries management. However, potential adverse side effects include (a) impacts on other lake uses, such as swimming and boating, (b) damage to shore banks, shorelines, and shoreline retaining walls, (c) reduced numbers and diversity of benthic invertebrates in the littoral zone, which are important prey items for some fish species, and (d) an increased likelihood of winterkill during winter drawdowns. Drawdowns should be conducted only in lakes with a steady water inflow, sufficient to refill the lake when needed. Ploskey (1982) presents additional information on the benefits and design of water-level management programs.

Reservoir Construction

Physical features that can be altered very little once a lake is constructed can drastically affect the fish community and the cost of maintaining

quality fishing. Factors such as watershed area, watershed usage, and the erodibility of soils in the watershed will impact lakewater quality. Lake volume, mean and maximum depth, watershed topography, shoreline development, and basin slope all influence the degree of thermal stratification and other lake characteristics that indirectly affect the suitability of the lake habitat for fish. For example, Hill (1986) identified three primary physical characteristics associated with the quality of fishing in small reservoirs in Iowa: (1) the mean basin slope, (2) the watershed-to-lake-area ratio, and (3) an adjusted siltation index based on the watershed-to-lake-area ratio, soil erosion rates for soil types in the watershed, and the proportion of the watershed farmed using approved soil-conservation practices. All of these physical features can be determined prior to lake construction and should influence site selection, lake design, and construction methods.

Baker et al. (1993) describe a number of important factors to consider in the design and construction of small lakes and reservoirs, relevant specifically to fisheries management:

- The watershed should be well vegetated to prevent excessive siltation, and free of pesticides and other pollutants. Runoff from row crops, livestock operations, and industrial sites (or from areas where erosion cannot be controlled) should be treated or diverted around the lake.
- The optimal watershed size (and watershed-to-lake-area ratio) depends on the lake volume, rate of rainfall, topography of the watershed, and land uses in the watershed. In Georgia, the recommended watershed size for a small, one-acre lake is about ten acres of pasture or 25 acres of forested watershed (Georgia Department of Natural Resources, 1988). The state of Illinois recommends 9 hectares of watershed per surface acre of water as a rule of thumb (Illinois Department of Conservation, n.d.).
- The amount of timber and brush that should be cleared from the lake basin before filling depends on the lake's size and planned uses. In small lakes (<1.4 ha), all trees, stumps, and brush should be removed to produce the maximum harvestable fish biomass. In larger reservoirs, on the other hand, it is often desirable to leave small blocks of well-placed timber and brush near the shoreline in coves and embayments to provide fish cover. Timber should be cleared, however, in the area of the dam, in recreational and boating areas, and in the deeper parts of the reservoir, where it is unlikely to improve fishing and may aggravate problems with oxygen depletion as the organic matter left behind gradually decomposes. Too much standing timber and brush can provide excessive fish cover, and should be avoided.
- If it is necessary to supplement natural spawning areas and fish cover,

spawning sites and artificial reefs can be constructed more easily before the lake fills than afterward.

- Extensive shallow areas will encourage the growth of aquatic macrophytes, an undesirable feature in most small lakes designed for fishing and fish production. Recommended slope ratios along the shoreline are generally 2:1 or 3:1. Also, sand blankets, gravel beds, or fiberglass mats can be used to inhibit rooted plant growth in selected areas.
- Water depths should be sufficient to minimize problems with low levels of dissolved oxygen in the winter and late summer. The state of Illinois, for example, recommends that lakes be at least 0.45 ha in area and 3 to 3.6 m deep in 25% of the basin for successful fish management (Illinois Department of Conservation, 1989).
- The dam and water overflow should be constructed to allow easy control of water levels, using a drain pipe or gate valve. In most cases, a standpipe overflow is recommended that draws water from deeper areas of the lake.
- Upstream migration of undesirable fish and, to a lesser degree, the downstream migration of game fish can be hindered by constructing an emergency spillway with a substantial vertical fall (at least 1.2 m) and sufficiently wide so that water depths over the spillway never exceed 5 cm.

Additional guidelines on the design and construction of small lakes and reservoirs are available from most state fisheries agencies (e.g., Oklahoma Department of Wildlife Conservation, 1984; Illinois Department of Conservation, n.d.) and the U.S. Department of Agriculture, Soil Conservation Service (1982).

Bio-manipulation

Many fisheries management techniques involve the direct manipulation of the fish community and other organisms that may serve as prey for or be predators or competitors of the fish species of interest. Three types of activities include: (1) game fish stocking, (2) control of undesirable fish species and stunted fish populations, and (3) prey enhancements to supplement food supplies.

INTEGRATED AQUATIC RESTORATION: CASE STUDIES

Aquatic ecosystem restoration projects should be undertaken using an integrated approach. This type of approach attempts to consider the major ecological actions in a watershed and respond holistically to

cumulative ecological impacts. Fragmentation of the ecological restoration and management efforts has been commonplace in government and industry. There are many barriers to integrated aquatic ecosystem restoration. Watershed boundaries often overlap with political boundaries. Even within watersheds, there is often administration of ecological concerns by different agencies.

Fortunately, over the last 20 years there has been an increase in the formation of agencies with jurisdictions at the watershed or ecoregion level. For example, the International Joint Commission established joint U.S. and Canadian goals for the Great Lakes. Throughout the United States, river basin and interstate commissions have been created to deal with environmental issues at the watershed level. These regional planning agencies include such organizations as the Interstate Commission on the Potomac River Basin, the Ohio River Sanitation Commission, and the Tahoe Regional Planning Agency.

This section presents two case studies of aquatic restoration efforts where there was a concerted effort to integrate restoration and promote the philosophy of watershed protection. These include the Merrimack River Watershed Protection Project and the Canaan Valley, West Virginia, Project (U.S. EPA, 1993). The integrated plans to restore the Milwaukee River in Wisconsin, the Emscher River in Germany, Lake Balaton in Hungary, and the Lagoon of Venice in Italy were discussed in Chapter 1.

Merrimack River, Massachusetts and New Hampshire

The Merrimack River watershed covers 13,000 km³ in parts of Massachusetts and New Hampshire. More than 300,000 people rely on the river for drinking water. The river also provides water for industrial and agricultural uses, and serves to assimilate waste and generate electricity. Many people use the river and its shores for relaxation and recreation.

Wastewater discharges, toxic contaminants, urban runoff, increased water withdrawal, and wetlands loss are the primary threats to long-term water quality and ecological integrity in the river. Project participants include several federal agencies (EPA, USDA, Department of the Interior, Army Corps of Engineers); state governments (Massachusetts and New Hampshire); regional planning agencies; local governments, industries, and utilities; agricultural, environmental, and recreational organizations; and universities. The participants are resolving both water quality and quantity issues by developing data-management systems and striving to balance competing needs within the watershed. A few projects underway aim to:

- Provide decisionmakers with information on the extent and condition of wetlands to protect the most valuable areas
- Help light industries (such as auto repair shops, dry cleaners, or photofinishers) understand what steps they can take to prevent pollution
- Provide decisionmakers with information about potential contamination of water supplies, helping them to focus regulatory activities (such as inspections and permitting) on preventing pollution and planning for emergency response if spills occur

Canaan Valley, West Virginia

The 15,000-ha Canaan Valley in West Virginia, designated as a National Natural Landmark in 1975, comprises fragile wetlands areas containing a unique and irreplaceable boreal ecosystem. The Blackwater River, originating in the valley's southern end, is an important source of drinking water and the largest stream network in the state with a self-sustaining brown trout population.

The valley and its resources attract a wide spectrum of interests. For example, a power company proposes flooding 3150 ha of the valley; real estate developers plan to increase the number of vacation homes, golf courses, ski slopes, and condominiums; a major off-road vehicle race, called the Blackwater 100, is held in the valley annually; and natural resource conservationists strive to protect rare plants and wildlife habitat, including wetlands.

In 1990, federal, state, and local participants formed the Canaan Valley Task Force to resolve a variety of issues, ensuring long-term environmental protection while allowing reasonable, sustainable economic growth. Early accomplishments include:

- A study of the impact of off-road vehicles
- A study of the economic impact of the proposed Canaan Valley National Wildlife Refuge
- Suspension of certain nationwide general permits for discharges of dredged or fill material in wetlands in the valley
- Advanced identification of wetlands
- Establishment of a wetlands surveillance program
- Implementation of a public outreach program

CONCLUSIONS

Many methods are available for management and restoration. The best approach for any given situation will vary, depending on the target species, nature of the problem, management goals and objectives, and

the characteristics of the body of water. There are no single solutions, although experience has demonstrated that some methods tend to work better in certain situations and for some problems than others.

Unnecessary actions or overmanagement are as undesirable as no activity at all. Monitoring should be conducted routinely to continually assess the effectiveness of restoration and management actions, and the plan altered appropriately. The goal should be to select and apply the approach(es) that will achieve the desired goals for the lowest cost and effort and also ensure the long-term sustainability of the ecosystem.

An integrated approach to restoration is recommended. Proposed restoration projects should consider the major ecological interactions in a watershed. Rivers, streams, lakes, and reservoirs are interconnected parts of a watershed that also often include ground water, estuaries, and wetlands. A practical and effective approach to restoration of aquatic ecosystems would include the consideration of all significant ecological elements on a watershed scale. In this way, the cumulative impacts of an ecosystem can best be evaluated.

References

- Andersson, G., H. Berggren, and S. Hambrin (1975). Lake Trummen restoration project. III. Zooplankton macrobenthos and fish, *Verh. Int. Ver. Limnol.* 19:1097.
- Aquatic Systems Engineering (1990). *Assessment and Guide for Meeting Dissolved Oxygen Water Quality Standards in Hydroelectric Plant Discharges*, EPRI GS-7001, Electric Power Research Institute, Palo Alto, CA.
- Baker, J. P., H. Olem, C. S. Creager, and B. P. Parkhurst (1993). *Fish and Fisheries Management in Lakes and Reservoirs*, EPA-841-R-93-002, U.S. Environmental Protection Agency, Washington, DC.
- Bengtsson, L., S. Fleischer, G. Lindmark, and W. Ripl (1975). Lake Trummen restoration project. I. Water and sediment chemistry, *Verh. Int. Ver. Limnol.* 19:1080-1087.
- Berger, J. J. (1990). Evaluating Ecological Protection and Restoration Projects. A Holistic Approach to the Assessment of Complex Multi-attribute Resource Management Problems, Ph.D. dissertation, University of California, Davis.
- Bernstein, G., and P. Kumble (1992). Anacostia watershed restoration small habitat improvement program: Checklist and design matrix, in *Watershed Restoration Sourcebook: Collected Papers Presented at the Conference; Restoring Our Home River: Water Quality and Habitat in the Anacostia*, November 6 and 7, 1991, College Park, MD, P. Kumble and T. Schueler, eds., Publication No. 92701, Metropolitan Washington Council of Governments, Washington, DC.
- Brocksen, R., M. Marcus, and H. Olem (1992). *Practical Guide to Managing Acidic Surface Waters and Their Fisheries*, Lewis Publ., Chelsea, MI.
- Burns, F. L. (1988). *Aeration of Lakes and Reservoirs in Australia*, presented at North American Lake Management Society annual meeting, St. Louis, MO.

- Cairns, R. F. (1988). Increasing diversity by restoring damaged ecosystems, in *Biodiversity*, E. O. Wilson, ed., National Academy Press, Washington, DC, pp. 333-343.
- Canfield, D. E., Jr., K. A. Langeland, S. B. Linda, and W. T. Haller (1985). Relations between water transparency and maximum depth of macrophyte colonization in lakes, *J. Aquat. Plant Manage.* **23**:25-28.
- Carline, R. F. (1980). Features of successful spawning site development for brook trout in Wisconsin ponds, *Trans. Am. Fish. Soc.* **109**:453-457.
- Committee on Restoration of Aquatic Ecosystems (1992). *Restoration of Aquatic Ecosystems*, National Research Council, National Academy Press, Washington, DC.
- Conyers, D. L., and G. D. Cooke (1983). A comparison of the costs of harvesting and herbicides and their effectiveness in nutrient removal and control of macrophyte biomass, in *Lake Restoration, Protection, and Management*, EPA 440/5-83-001, U.S. Environmental Protection Agency, Washington, DC, pp. 317-321.
- Cooke, G. D., and H. Olem (1990). Lake and reservoir restoration and management techniques, in *The Lake and Reservoir Guidance Manual*, 2d ed. EPA 440/4-90-006, North American Lake Management Society for U.S. Environmental Protection Agency, Washington, DC, pp. 117-159.
- Cooke, G. D., R. T. Heath, R. H. Kennedy, and M. R. McComas (1982). Change in lake trophic state and internal phosphorus release after aluminum sulfate application. *Water Resour. Bull.* **18**:699-705.
- Cooke, G. D., E. B. Welch, S. A. Peterson, and P. R. Newroth (1993). *Restoration and Management of Lakes and Reservoirs*, 2d ed., Lewis Publ., Boca Raton, FL.
- Cronberg, G., C. Gelin, and K. Larsson (1975). Lake Trummen restoration project. II. Bacteria, phytoplankton, and phytoplankton productivity, *Verh. Int. Ver. Limnol.* **19**:1088-1096.
- Davis, J. L., et al., eds. (1983). *Proceedings, International Conference on Water Power 1983*, University of Tennessee, Knoxville, TN, pp. 1326-1335.
- Dominie, J. R. (1980). Hypolimnetic aluminum treatment of softwater Annabesacook Lake, in *Restoration of Lakes and Inland Waters*, EPA 440/5-81-010, U.S. Environmental Protection Agency, Washington, DC, pp. 417-423.
- Dunst, R. C., J. G. Vennie, R. B. Corey, and A. E. Peterson (1984). *Effect of Dredging Lilly Lake, Wisconsin*, EPA 600/3-84-097, EPA Cooperative Agreement No. R804875, Environmental Research Laboratory, U.S. Environmental Protection Agency, Corvallis, OR.
- Engel, S. (1984). Evaluating stationary blankets and removable screens for macrophyte control in lakes, *J. Aquat. Plant Manage.* **22**:43-48.
- Engel, S. (1989). Lake use planning in local efforts to manage lakes, in *Enhancing States' Lake Management Programs*, North American Lake Management Society, Washington, DC, pp. 101-105.
- Evans, J. W. (1989). *Objectives and Guidelines for Aquatic Plant Management in Georgia*, Game and Fish Division, Georgia Department of Natural Resources, Fort Valley, GA.

- Everhart, W. H., and W. D. Youngs (1981). *Principles of Fishery Science*, Cornell University Press, Ithaca, NY.
- Fast, A. W., V. A. Dorr, and R. J. Rosen (1975). A submerged hypolimnion aerator, *Water Resour. Res.* 11:287-293.
- Gächter, R. (1976). Die Tiefenwasserableitung, ein Weg zur Sanierung von Seen, *Schweiz. Z. Hydrol.* 38:1-28.
- Gallagher, J. W., and G. V. Mauldin (1987). Oxygenation of releases from Richard B. Russell Dam, in *Proceedings, CE Workshop on Reservoir Releases*, U.S. Army Corps of Engineers, Vicksburg, MS, pp. 121-124.
- Garrison, P. J., and D. R. Knauer (1984). Long-term evaluation of three alum treated lakes, in *Lake and Reservoir Management*, EPA 440/5-84-001, U.S. Environmental Protection Agency, Washington, DC, pp. 513-517.
- Gasperino, A. F., M. A. Beckwith, G. R. Keizur, R. A. Soltero, D. G. Nichols, and J. M. Mires (1980). Medical Lake improvement project: Success story, in *Restoration of Lakes and Inland Waters*, EPA 440/5-81-010, U.S. Environmental Protection Agency, Washington, DC, pp. 424-428.
- Georgia Department of Natural Resources (1988). *Management of Georgia Fish Ponds*, Game and Fish Division, GDNA, Atlanta, GA.
- Haag, K. H. (1986). Effective control of waterhyacinth using *Neochetina* and limited herbicide application, *J. Aquat. Plant Manage.* 24:70-75.
- Halsey, T. G. (1968). Autumnal and overwinter limnology of three small eutrophic lakes with particular reference to experimental circulation and trout mortality, *J. Fish. Res. Board Can.* 25:81-99.
- Herson, L. (1992). Riparian restoration: A guide to reforestation planning, in *Watershed Restoration Sourcebook: Collected Papers Presented at the Conference; "Restoring Our Home River: Water Quality and Habitat in the Anacostia."* November 6 and 7, 1991, College Park, MD, P. Kumble and T. Schueler, eds., Publication No. 92701, Metropolitan Washington Council of Governments, Washington, DC, pp. 217-231.
- Hill, K. R. (1986). Classification of Iowa lakes and their fish standing stock, *Lake Reserv. Manage.* 2:105-109.
- Howells, R. G. (1986). *Guide to Techniques for Establishing Woody and Herbaceous Vegetation in the Fluctuation Zones of Texas Reservoirs*, Federal Aid in Sport Fish Restoration Project F-31-R-12, Texas Parks and Wildlife Department, Austin, TX.
- Illinois Department of Conservation (n.d.). *Small Lakes and Ponds: Their Construction and Care*, Fishery Bulletin No. 3, Division of Fisheries, IDC, Springfield, IL.
- Illinois Department of Conservation (1989). *Management of Small Lakes and Ponds in Illinois*, 2d ed., Division of Fisheries, IDC, Springfield, IL.
- Imhoff, K. R. (1969). *Advances in Water Pollution Research*, Pa. II-7, Pergamon Press, Oxford.
- Imhoff, K., and K. R. Imhoff (1990). *Taschenbuch der Stattenwässerung*, Oldebourg Verlag, Munich.
- Karouna, N. (1992). Techniques for restoring urban streams, in *Watershed Restoration Sourcebook: Collected Papers Presented at the Conference; "Restoring*

- Our Home River: Water Quality and Habitat in the Anacostia.* November 6 and 7, 1991, College Park, MD, P. Kumble and T. Schueler, eds., Publication No. 92701, Metropolitan Washington Council of Governments, Washington, DC, pp. 103–122.
- Kortmann, R. W. (1989). Aeration: Technologies and sizing methods, *Lake Line* 9:6–7, 18–19.
- Kortmann, R. W., E. Davis, C. R. Frink, and D. D. Henry (1983). Hypolimnetic withdrawal: Restoration of Lake Wonoscopomuc, Connecticut, in *Lake Restoration, Protection and Management*, EPA 440/5-83-001, U.S. Environmental Protection Agency, Washington, DC, pp. 46–55.
- Krenkel, P. A., and V. Novotny (1980). *Water Quality Management*, Academic Press, New York.
- Kumble, P., and T. Schueler, eds. (1992). *Watershed Restoration Sourcebook: Collected Papers Presented at the Conference; "Restoring Our Home River: Water Quality and Habitat in the Anacostia."* November 6 and 7, 1991, College Park, MD. Publication No. 92701, Metropolitan Washington Council of Governments, Washington, DC.
- Lackey, R. T. (1972). Response of physical and chemical parameters to eliminating thermal stratification in a reservoir, *Water Resour. Bull.* 8:589–599.
- Lorenzen, M. W., and A. W. Fast (1977). *A Guide to Aeration/Circulation Techniques for Lake Management*, EPA 600/3-77-004, U.S. Environmental Protection Agency, Washington, DC.
- Mauldin, G., R. Miller, J. Gallagher, and R. E. Speece (1988). Injecting an oxygen fix, *Civ. Eng.* 58(Mar.):54–56.
- Mayhew, J. K., and S. T. Runkel (1962). The control of nuisance aquatic vegetation with black polyethylene plastic, *Proc. Iowa Acad. Sci.* 69:302–307.
- Newroth, P. R., and R. J. Soar (1986). Eurasian watermilfoil management using newly developed technologies, *Lake Reserv. Manage.* 2:252–257.
- Nicholson, S. A. (1981). Changes in submersed macrophytes in Chautauqua Lake, 1937–1975, *Freshw. Biol.* 11:523–530.
- Novotny, V., and G. Bendoricchio (1989). Water quality—Linking nonpoint pollution and deterioration, *Water Environ. Technol.* 1(Nov.):400–408.
- Novotny, V., K. R. Imhoff, M. Olthof, and P. A. Krenkel (1989). *Karl Imhoff's Handbook of Urban Drainage and Wastewater Disposal*, Wiley Interscience, New York.
- Oklahoma Department of Wildlife Conservation (1984). *Pond Management in Oklahoma*, Fish Division, ODNC, Oklahoma City, OK.
- Olem, H. (1991). *Liming Acidic Surface Waters*, Lewis Publ. Chelsea, MI.
- Olem, H., and G. Flock, eds. (1990). *The Lake and Reservoir Restoration Guidance Manual*, 2d ed. EPA 440/4-90-006, North American Lake Management Society for U.S. Environmental Protection Agency, Washington, DC.
- Olem, H., R. K. Schreiber, R. W. Brocksten, and D. B. Porcella (1991). *International Lake and Watershed Liming Practices*, Terrene Institute, Washington, DC.
- Perkins, M. A. (1983). *Limnological Characteristics of Green Lake: Phase I*

- Restoration Analysis*, Department of Civil Engineering, University of Washington, Seattle, WA.
- Peterson, J. O., J. T. Wall, T. L. Wirth, and S. M. Born (1973). *Eutrophication Control: Nutrient Inactivation by Chemical Precipitation at Horseshoe Lake, Wisconsin*, Tech. Bull. 62, Wisconsin Department of Natural Resources, Madison, WI.
- Ploskey, G. R. (1982). *Effects of Water-level Changes on Reservoir Ecosystems, Fish, and Fishing, with Guidance on Reservoir Operations to Enhance Fisheries*, Tech. Rep. 81, U.S. Fish and Wildlife Service, Washington, DC.
- Ripl, W. (1976). Biochemical oxidation of polluted lake sediment with nitrate—A new restoration method, *Ambio* 5:132–135.
- Ripl, W., and G. Lindmark (1978). Ecosystem control by nitrogen metabolism in sediment, *Vatten* 34:135–144.
- Sanders, D. R., Sr., and E. A. Theriot (1986). *Large-scale Operations Management Test (LSOMT) of Insects and Pathogens for Control of Waterhyacinth in Louisiana. II. Results for 1982–83*, Tech. Rep. A-85-1, U.S. Army Corps of Engineers, Vicksburg, MS.
- Smith, S. A., D. R. Knauer, and T. L. Wirth (1975). *Aeration as a Lake Management Technique*, Tech. Bull. No. 87, Wisconsin Department of Natural Resources, Madison, WI.
- Soltero, R. A., D. G. Nichols, A. F. Gasperino, and M. A. Beckwith (1981). Lake restoration: Medical Lake, Washington, *J. Freshw. Ecol.* 1:155–165.
- Stuber, R. J. (1985). Trout habitat, abundance, and fishing opportunities in fenced vs. unfenced riparian habitat along Sheep Creek, Colorado, in *Riparian Ecosystems and Their Management: Reconciling Conflicting Uses*, Gen. Tech. Rep. RM120, U.S. Forest Service, Ft. Collins, CO, pp. 310–314.
- Taggart, C. T., and D. J. McQueen (1981). Hypolimnetic aeration of a small eutrophic kettle lake: Physical and chemical changes, *Arch. Hydrobiol.* 91: 150–180.
- USDA Soil Conservation Service (1982). *Ponds—Planning, Design, and Construction*, Agric. Handb. No. 590, U.S. Government Printing Office, Washington, DC.
- U.S. Environmental Protection Agency (1987). *Guide to Nonpoint Source Pollution Control*, Office of Water, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1993). *The Watershed Approach. Annual Report 1992*, EPA-840-S-93-001, Office of Water, U.S. EPA, Washington, DC.
- Welch, E. B., and C. R. Patmont (1980). Lake restoration by dilution: Moses Lake, Washington, *Water Res.* 14:1317–1325.
- Westerdahl, H. E., and K. D. Getsinger, eds. (1988). *Aquatic Plant Identification and Herbicide Use Guide*, Vol. I and II, Aquat. Plant Control Res. Progr. Tech. Rep. A-89-9, U.S. Army Corps of Engineers, Vicksburg, MS.
- Willenbring, P. R., M. S. Miller, and W. D. Weidenbacher (1984). Reducing sediment phosphorus release rates in Long Lake through the use of calcium nitrate, in *Lake and Reservoir Management*, EPA 440/5-84-001, U.S. Environmental Protection Agency, Washington, DC, pp. 118–121.

Integrated Planning and Control of Diffuse Pollution—Watershed Management

The environment functions as an integrated whole and each part is to some degree dependent on the other. Recognition of this inter-relatedness would improve our ability to constrain and reduce pollution.

United Kingdom, Department of the Environment,
Integrated Pollution Control: A Consultation Paper, July 1988

Until recently, emphasis by environmental engineers and decisionmakers has been directed toward the treatment of traditional point source pollution (in this sense, classification of point source pollution included mostly discharges from municipal and industrial wastewater sources, without considering wet-weather inputs). Billions of dollars were spent in the United States on the point source cleanup mandated by the earlier versions of the Clean Water Act (1972 and 1977). Similar policies were in place in several industrialized countries of Europe. As a result of these policies, marked improvements of water quality of some bodies of water were noticed. The River Thames in London, which for decades exhibited anoxic faulty conditions during warm summer periods, is alive again and can support a viable recreational fishery. The dissolved oxygen levels of the Potomac River and estuary near Washington, D.C., have risen from zero to about half-saturation during the summer. The successes and failures of the environmental efforts implemented in the United States under the Clean Water Act have been documented by Wolman (1988), who noted that focusing on the point source abatement has only maintained more-or-less a status quo in the majority of water quality monitoring stations throughout the United States. It has to be pointed out,

however, that most of the monitoring stations operated by the U.S. Geological Survey may not be located in places where the most profound water quality changes have occurred. Hence the recent experience of emphasis on point source abatement indicates that focusing on one type of pollution (municipal and industrial wastewater discharges) may not be efficient and that an integrated approach that would address both point and nonpoint (diffuse) sources is needed.

The elimination or reduction of pollution sources may require excessive expenses. Furthermore, many diffuse sources cannot be regulated (see Chapter 1) and enforcement of control is not feasible. Therefore, the integrated solutions should also address the magnitude of the waste-assimilative capacity and its enhancement wherever possible (primarily in limited water quality situations). For example, in Milwaukee, Wisconsin, after spending over \$2 billion on point source cleanup and sewer rehabilitation, the receiving waters of the lower Milwaukee River and estuary still remain unacceptably polluted due to diffuse pollution from upstream rural sources and urban runoff from the Milwaukee metropolitan area. Focusing on point source abatement in the Boston harbor, Puget Sound, and other coastal watersheds is not sufficient. A water quality plan for the Milwaukee harbor (lower Milwaukee River) suggested in-stream measures (dilution, in-stream aeration) that would cost only about 0.2% of the price of the point source cleanup and would bring the water quality within the standards for conventional pollutants (dissolved oxygen, bacteria, etc.) prescribed for this body of water. The dilution water is conveyed into the Milwaukee harbor from Lake Michigan by so-called flushing tunnels that were built in 1888 and have become an engineering landmark. The dilution remedy is not a universal solution. As a matter of fact, it is quite unusual and not available in many places. However, other solutions for enhancing the waste-assimilative capacity of receiving bodies of water, such as reestablishing riparian wetlands, may be feasible.

As was pointed out in the preceding chapter, in the 1920s Karl Imhoff was the first prominent environmental engineer and planner who recognized that a water-pollution problem cannot be resolved by point source cleanup only. In the Ruhr area of Germany several very small rivers carry the entire burden of water supply and wastewater disposal from one of the most industrialized regions of the world, with a resident population of about 8 million. The integrated approach used both innovative institutional arrangement (see, for example, Kneese and Bower, 1968, for a discussion) by creating watershedwide agencies (authorities) with great powers to control pollution, and in-stream measures for enhancing the waste-assimilative capacity, such as (Imhoff and Imhoff, 1990):

- Dilution (low-flow augmentation) by increasing flows from upstream reservoirs during critical water quality periods and by back-pumping Rhine River water into the small receiving streams (tributaries) of the Rhine
- Building river reservoirs to act as polishers during posttreatment
- In-stream aeration by power plant turbines and floating aerators
- Dredging accumulated sediment to reduce sediment oxygen demand and improve water quality
- Cutting weeds in reservoirs affected by higher nutrient inputs

Today, promising in-stream water quality enhancement methods include restoration of riparian (water bordering) wetlands, installation of buffer strips, polishing lagoons and ponds, and even fish management (Mitsch and Jørgensen, 1989; Novotny and Chesters, 1981). The concepts of integrated pollution control are covered in a publication by Haigh and Irwin (1990).

POLLUTION LOAD TRADE-OFFS AND THE PERMIT PROCESS

To the receiving body of water whose integrity is impaired by pollution discharges it does not matter whether the pollution originates from point or nonpoint sources. In either or both cases damage is done to the uses of the body of water, including its use for fish and wildlife protection and propagation and/or for recreation, water supply, or other beneficial and designed uses. For example, if water quality deterioration is caused by discharges of phosphorus, all sources of phosphorus must be considered. In addition, the forms of phosphorus and their availability to stimulate the photosynthetic algal and plant growth are secondary factors to be considered.

In the planning-permit preparation phase the waste-assimilative capacity of the receiving bodies of water is generally determined by theoretically and mathematically sound models that have been calibrated and verified by extensive field data. The use of models is necessary because the waste-assimilative capacity is either determined for hydrological conditions that are statistically rare, or are defined in terms of probabilities. Furthermore the pollution inputs are projected into the future. Modeling is also used for assessing the possibilities of the in-stream and off-stream waste-assimilative capacity enhancement, such as the impact on water quality of riparian wetlands and buffer strips.

In the United States, legislatures generally attempt to treat polluters

uniformly. Unfortunately most of the legislative control regulations contain added exceptions for special interests, such as the exclusions from the NPDES permits of some agricultural and industrial sources, and diffuse-pollution sources from smaller urban areas. Under the uniform control scenario of the NPDES permit system, if the waste-assimilative capacity of the receiving body of water is exceeded, the allowable pollution load is distributed by the regulating agency uniformly among the polluters (those covered by the permit requirement). The sources of diffuse pollution from urban and nonurban areas are well known and have been described in great detail throughout this book.

The planning studies that will lead to the allocation of pollutant loads among the sources must consider whether the pollutant mass load or pollutant concentration is the cause of the problem. Pollutant mass loading from all sources can be a problem in bodies of water undergoing accelerated eutrophication. In this case, nutrient loads (in kilograms per year or season) from all sources, *regardless of whether they are point or nonpoint*, must be restricted. In a *trade-off approach* the regulatory agency will issue the permits based on the estimated maximal pollutant load that will not impair the integrity of the receiving body of water. That implies nonviolation of water quality standards. Even the present water quality criteria (standards in some states) for nitrogen and phosphorus are based on seasonal limits rather than short-term exposures.

The maximal long-term load of some toxic components should be considered if these toxic components are incorporated into the sediment or tissues of biota in the affected bodies of water (bioaccumulation). At the present time, fish tissue standards are in place, but sediment standards are being implemented. For example, PCB concentrations in water of affected water bodies are commonly undetectable even though sediment and fish tissue concentrations are unacceptable. In these and similar cases, the long-term loads from all sources (including point and nonpoint and atmospheric, dry- and wet-weather sources alike) and not short-term effluent concentrations must be restricted. These standards should be based on long-term chronic toxicity and appropriate safety factors. The allowable long-term (for example, annual or seasonal) loads can only be determined by relatively complex dynamic long-term simulation effects or other modeling methodologies described in Chapters 12 and 13. At present, the state of the art of available models is still evolving, and application of the models is very expensive, requiring extensive field data and calibration and verification efforts.

The short-term chemical-based concentration standards are typically based on lethal toxicity obtained from toxicity tests lasting from two to four days. These standards are applied to water, and the loads are

then determined from mass-balance models. Due to the fact that only exposure to the maximal concentration during a relatively short period is considered, the models are more simple, more reliable. Typically, steady-state receiving water quality models suffice for dry-weather point sources, supplemented with event-oriented loading models or estimates for diffuse wet-weather sources.

WATER USES AND USE ATTAINABILITY

In the United States, water quality standards define water quality goals that are related to the designated use of the body of water. The law requires states to adopt water quality standards to protect public health or welfare, enhance the quality of water, and serve the purposes of the Clean Water Act. "Serve the purposes of the act" means that water quality standards should

- Include provisions for restoring and maintaining the chemical, physical, and biological integrity of state waters
- Wherever attainable, provide water quality for the protection and propagation of fish, shellfish, wildlife, and recreation in and on the water
- Consider the use and value of state waters for public water supplies, propagation of fish and wildlife, recreation, agricultural and industrial purposes, and navigation.

When states designate uses that will support aquatic life propagation, contact recreation and human health (drinking water supply) considerations must be given to whether such uses can be attained. If the state does not designate uses that would comply with the goals of the Clean Water Act, a use attainability analysis must be performed.

A *designated use* is a term that is specified in water quality standards for a body of water or a segment of a body of water. Typical uses include public water supply, propagation of fish and wildlife, recreational purposes, agricultural, industrial, and navigational. The EPA does not recognize waste transport and assimilation as an acceptable water use (U.S. EPA, 1988).

According to the Clean Water Act, the desirable water uses that must be protected include public water supply, recreation, and propagation of fish and wildlife. Each state develops its own use classification system based on the generic uses cited in the Clean Water Act. The states may differentiate and subcategorize the types of uses that are to be protected,

such as cold-water or warm-water fisheries, or specific species that are to be protected, such as trout, salmon, or bass. States may also designate special uses to protect sensitive or valuable aquatic life or habitat.

An existing use is a use that was achieved on a body of water on or after November 28, 1975. It can be modified or changed only if new important uses of the body of water have been added that require more stringent criteria. *Attainable uses* are those uses (based on the state's system of water-use classifications) that can be achieved when effluent limits under the Clean Water Act, Sections 301(b)(1)(A) and (B) and Section 306, are implemented for point source discharges and when cost-effective and reasonable best management practices are implemented for nonpoint sources.

The Clean Water Act (Section 305(b)) requires states to submit a water quality assessment report (U.S. EPA, 1991a), which should contain an evaluation of the extent of support of designated uses in their surface-water bodies. The 1990 summary of use attainment is presented in Table 16.1. The so-called 305(b) process is "the principal means by which EPA, Congress, and the public evaluate water quality, the progress made in maintaining and restoring water quality and the extent to which the problem remains" (U.S. EPA, 1991a). The law requires that the States submit their water quality assessment reports to the EPA administrator every two years.

Statutory water uses. The specific water uses of a body of water are

TABLE 16.1 Support of Designated Uses in U.S. Waters in 1990

	Rivers (×1000 km)	Lakes (×1,000 ha)	Estuaries (km ²)	Great Lakes Shoreline (km)	Coastal Shoreline (km)
U.S. total	2573	15,944	91,162	8200	30,874
Assessed	1041 (64%)	7483 (47%)	69,135 (75%)	7723 (94%)	6758 (22%)
Fully support designated use	655 (23%)	3308 (21%)	38,860 (42%)	137 (2%)	6074 (89%)
Threatened	69 (2%)	1175 (7%)	7905 (9%)	111 (2%)	78 (1%)
Partially support designated use	216 (7%)	1405 (9%)	17,024 (18%)	2276 (29%)	466 (7%)
Do not support designated use	100 (3%)	1603 (10%)	5345 (6%)	5290 (68%)	187 (3%)

determined by the states. The statutory body of water uses specified in the Section 305(b) reports include:

- Fish consumption (commercial fishing)
- Shell fishing (commercial)
- Aquatic life support
- Swimming and other contact recreation
- Secondary water contact (boating)
- Drinking water supply
- Agriculture (irrigation)
- State defined uses

Types of bodies of water. The guidelines for the Section 305(b) reports (U.S. EPA, 1991a) recognize the following types of bodies of water: streams, lakes, estuaries, coastal waters, wetlands. As shown in Table 16.1 the classification of the attainability of water uses of U.S. surface-water bodies categorizes the surface-water bodies as

Use attained

- fully supporting—the designated use is attained
- fully supporting but threatened

Use not attained

- partially supporting
- not supporting

The *outstanding national resource waters (ONRW)* are high-quality or ecologically sensitive unique waters, such as those within the jurisdiction of national and state parks and wildlife refuges. The primary intent of designating the bodies of water ONRW is to protect or attain the highest quality and/or unique waters. The ONRW category also protects waters of ecological significance for which water quality expressed by some traditional parameters may not be high (for example, dissolved oxygen in wetlands and some productive estuaries). The states establish the ONRW criteria in order to protect the characteristics that prompted a body of water to be designated ONRW. In many cases, the ONRW represent the ecoregional reference reaches for establishing goals of water quality pollution abatement. This represents another reason why a national policy must be developed and implemented that would protect the ONRW (Committee on Restoration of Aquatic Ecosystems, 1992).

Use Attainability

If the body of water has been classified as partially supporting or not supporting the designated use, the states or designated agencies perform use attainability analysis (UAA) to determine the proper use of the body of water. Based on the UAA it is possible to modify or change nonexistent designated water use, if attaining the use is not possible due to the existence of one or more of the following factors (U.S. EPA, 1988):

1. Naturally occurring contaminant concentrations that prevent the attainment of the use
2. Natural, intermittent, or low flow or water levels that prevent the attainment of the use
3. Human-caused conditions or sources of pollution that prevent the attainment of the use
4. Dams, diversions, or other hydrologic modifications that preclude attainment of the use
5. Physical conditions associated with the natural features of the body of water, unrelated to quality, which impede attainment of aquatic life protection uses
6. More stringent controls than those required by Sections 301(b) and 306 of the Clean Water Act would be needed to attain the use and implementation if such controls would result in substantial and wide-spread adverse social and economic impact

There are basically two reasons why UAA is needed for water bodies where use is not attained. One reason is to determine what levels of quality are possible to attain by implementation of various feasible point and nonpoint source abatement measures. For water quality abatement of streams and estuaries, EPA regulations and guidelines recognize only the source-control measures. However, in addition to the pollution source controls, water-body restoration and waste-assimilative capacity enhancement measures should also be considered (Committee on Restoration of Aquatic Ecosystems, 1992). These could include a variety of measures, such as in-stream aeration, dredging of contaminated sediments, low-flow augmentation, and restoration of riparian wetlands and buffer strips with tall shoreline vegetation. These measures could be made a part of an overall ecosystem restoration effort for the body of water. Ecosystem restoration and waste-assimilative capacity enhancement concepts for lakes have been considered and recommended by the U.S. EPA (NALMS, 1990).

The second purpose is to determine the most desirable use. This concept involves a study of the socioeconomic impact of the attainment of a specified use. The states are required to periodically review their water quality standards and revise them if appropriate. Concurrently, the UAA analysis is to be performed for bodies of water for which the designated use is not attained. The UAA is a multifaceted assessment of the physical, chemical, biological, and economic factors that affect the attainment of a use. The methodology of the UAA is described in pertinent manuals (U.S. EPA, 1983; Novotny et al., 1994). The UAA generally answers the following questions about the conditions of the water body:

1. What is the existing use to be protected?
2. What is the extent to which pollution (as opposed to physical factors) contribute to the impairment of the use?
3. What is the level of point source control required to restore or enhance the use?
4. What is the level of nonpoint source control required to restore or enhance the use?

Two other questions can be added:

5. What are the needed stream-restoration (waste-assimilative capacity enhancement) measures that would alter the adverse physical conditions of the receiving body of water that is impacting the aquatic habitat as well as water quality?
6. What is the optimal use of the body of water that would not impose a widespread adverse socioeconomic impact on the population involved and society as a whole?

A UAA consists of a survey and assessment of the body of water, a wasteload allocation (total maximum daily load process), and socioeconomic analyses (Fig. 16.1). It is a comprehensive process of evaluation, in its ecoregional context, of a body of water.

Once a designated use is attained it must be maintained. The Clean Water Act and ensuing antidegradation regulations do not allow a change to a lower quality use. However, states may modify nonexisting designated uses when it can be demonstrated through a UAA that attaining the higher designated use is not feasible.

In the integrated approach to water quality based pollution control there is another avenue open for the control of pollution discharges. Chemical-specific criteria are scientifically based numerical limits developed from laboratory bioassays with a safety factor. The EPA (1991b)

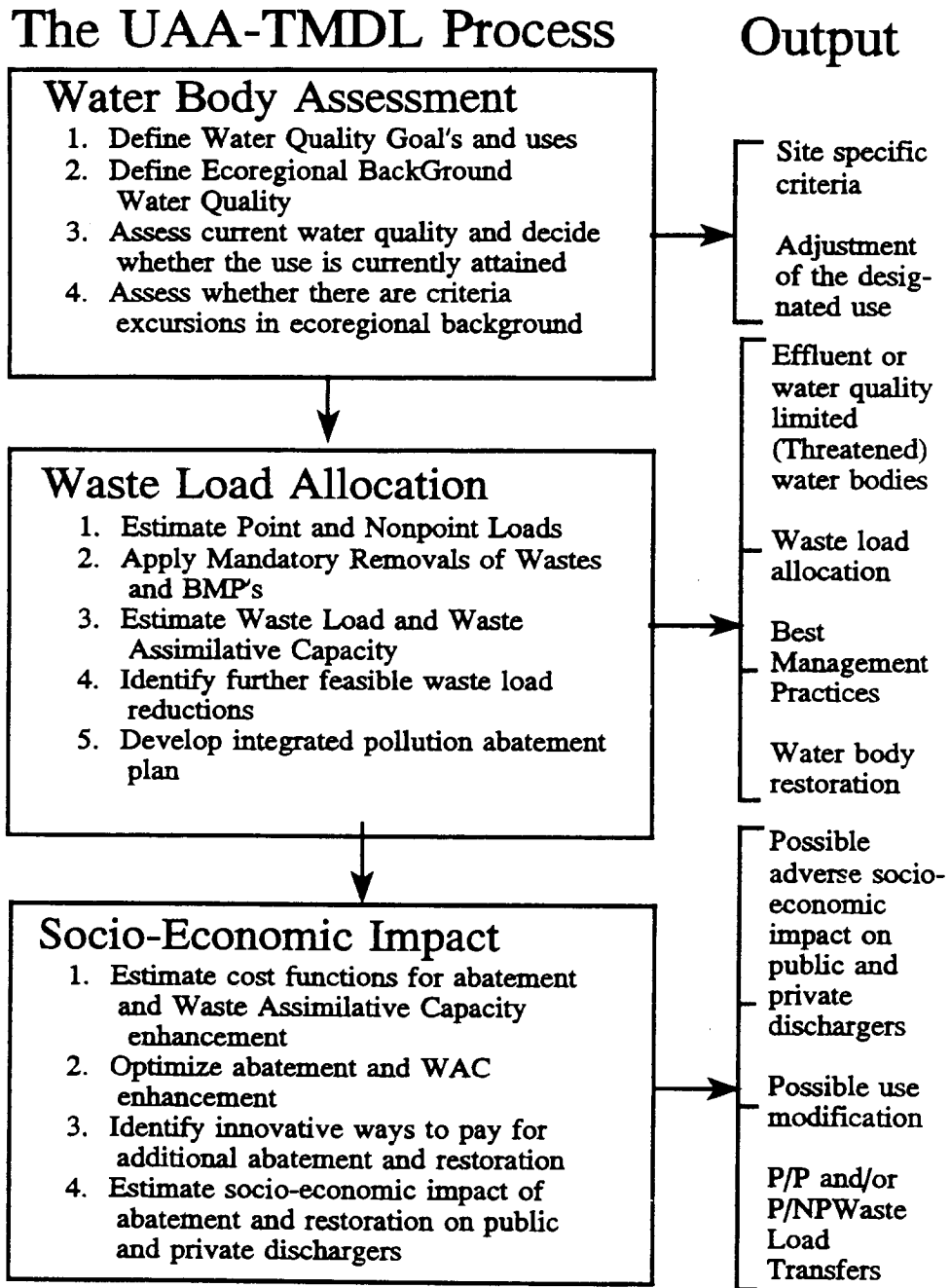


FIGURE 16.1. Three components of the water quality planning process. (Adapted from Novotny et al., 1994.)

has documented by experimental stream bioassay studies that such criteria provide nearly full protection of most aquatic species. Yet these criteria and standards are designed to protect most but not all of the species most of the time. Furthermore, natural conditions and irreversible modifications of the habitat may have caused a violation of a nationwide chemical standard, yet no impairment of integrity is noticed. In this case, site-specific criteria that could be related to the biological integrity of the receiving body of water could be adopted.

Waste-Assimilative Capacity: Loading Capacity

As previously defined the objective of water quality planning and evaluation efforts is to achieve and/or preserve the physical, chemical, and biological integrity of the receiving bodies of water. This is typically interpreted as maintaining water quality that would comply with the ambient water quality standards specified for the designated use of the body of water. Consequently, wastewater discharge from any source that does not result in a violation of a standard or will not impair the biological integrity of the body of water may be considered as noninjurious by the regulatory agencies (although in a court of law compliance with standards is not a defense against a liability if damage has occurred). The quantity of waste and contaminant loads that can be discharged into the environment without damage or use impairment of a receiving body of water, atmosphere, or land, minus the natural load is then called a *waste-assimilative capacity (WAC)*.

Cairns and Orvas (1989) defined the WAC as that range of concentration of a substance or a mixture of substances that will cause no deleterious effects upon the receiving ecosystems. The WAC is another important concept in our understanding of the process. If a waste discharge load into a receiving body of water is below the WAC, it does not violate the water quality standards and could be considered an acceptable waste load. The WAC is an economical asset that should always be considered, although it is not recognized by the EPA as a beneficial use of a body of water.

The EPA's Water Quality Planning and Management Regulation expressed in a guidance manual (U.S. EPA, 1991c) defines *loading capacity (LC)* as the greatest amount of loading a water can receive without violating water quality standards. Apparently, the loading capacity, as defined by the EPA, is almost synonymous with the waste-assimilative capacity defined herein; however, LC only refers to violations of the established standards, while the broader WAC definition involves the integrity of the receiving body of water. Typically, the WAC (or LC)

of surface-water bodies may be higher for decomposable organic matter, but it is very low to nil for some toxic chemicals that bioaccumulate in tissues of aquatic organisms and become injurious to animals and man using them as food. These differences are reflected in the magnitude of the standards.

According to these definitions, a waste constituent becomes a pollutant if it is discharged into the environment in quantities that are injurious to or impair beneficial uses of the environmental resources. Many waste constituents in small quantities are not injurious, and some of them may even be beneficial in low quantities and become injurious (toxic) only in quantities that exceed the WAC. For example, some metals that in high concentrations are known to be toxic, such as zinc, are necessary nutrients to aquatic life in smaller trace quantities.

Water quality management has become increasingly more complicated. The introduction of *priority pollutants* into the framework of water quality abatement planning and biological, physical, and chemical integrity concepts for the receiving bodies of water is causing planners and pollution control authorities to determine the required pollution controls using water quality based waste load allocations rather than technology-based effluent limitations. For this reason the EPA has been promoting and enforcing the *total maximum daily load (TMDL)* process, which establishes the allowable loadings for a body of water based on its WAC, and thereby provides the basis for the states to establish water quality based controls (U.S. EPA, 1991b, 1991c). According to the EPA's guidelines, the TMDL is the sum of *allowable loads* that include loads from both point (WLA) and nonpoint (LA_{NP}) sources as well as the natural background loads (BL). Hence

$$TMDL \leq LC \quad (16.1)$$

and

$$TMDL = WLA + LA_{NP} + BL$$

Note: Previously in many water quality control plans and studies for waste load allocation, the terms TMDL and WLA (allowable point source loads) were incorrectly used interchangeably instead of considering both LA and WLA as components of TMDL.

Margin of Safety

The margin of safety (MOS) is normally incorporated into the conservative assumptions used to develop the TMDL that is subsequently ap-

proved by regulatory agencies. If the safety margin is required to be larger than that incorporated in the assumptions of TMDL, additional MOS is added as a separate component, or

$$\text{TMDL} = \text{WLA} + \text{LA}_{\text{NP}} + \text{BL} + \text{MOS} \quad (16.2)$$

The WAC (LC) of most receiving bodies of water is not fixed, but can be enhanced (increased) by various management measures, including stream and lake aeration, low-flow augmentation, sediment dredging, nutrient inactivation, restoration and implementation of riparian wetlands, and planting shade trees. Returning a previously lined and straightened or impounded stream to its more natural state typically increases its capacity to receive and safely assimilate pollutants.

Waste-assimilative capacity enhancement should be a part of an overall ecosystem restoration effort, whereby the entire ecosystem and its boundary stresses are considered (Fig. 12.4). A simple aeration or waste dilution may cause a particular water quality standard to be met, but may not restore the ecosystem's integrity. Both stream restoration and pollution input reduction must be considered and their efficiency weighed against the overall ecosystem restoration goal, of which water quality is perhaps the most important component. For example, restoring a stream to a more natural state by removing unnecessary impoundments and stream linings and by providing shade trees along the stream may have a better water quality impact than an excessive reduction of nutrient levels in the effluents.

FINANCING DIFFUSE-POLLUTION ABATEMENT PROGRAMS: WHO SHOULD PAY?

Although Congress has made CSOs and a part of urban and agricultural storm-water management mandatory, no significant financial resources have been allocated for future projects; however, some CSO abatement is still receiving federal grants authorized by the previous amendments of the Clean Water Act (1972 and 1977). The General Accounting Office (U.S. General Accounting Office, 1990) noted that for nonpoint source abatement "*a key contributing factor to these resource constraints is that available funds are overwhelmingly oriented toward point sources (traditional) rather than nonpoint sources. . . .*" "*EPA has not allocated the amounts required to meet the most basic elements of its nonpoint source pollution agenda. . . .*" In addition, the EPA has not requested and appropriated funding for Section 319 programs as authorized by

Congress. As pointed out in the U.S. General Accounting Office (1990) study, out of \$400 million authorized by Congress between 1988 and 1991 only \$22 millions were requested by the EPA. It is quite apparent that under present budgetary constraints and EPA policies the bulk of funding for urban diffuse-source abatement must come from state and local sources.

Some states (for example, West Virginia and New Hampshire) are tying the implementation of nonpoint pollution programs to the availability of federal funding. Other states (Wisconsin, New Jersey) have established or plan to establish their own state funds for implementation of NP programs. Typically, states in their Section 319 reports expected that the bulk of the programs would be funded by public financing and cost sharing.

Generally, there are three fundamental means of recovering the costs of pollution abatement.

1. Placing the cost burden on polluters alone ("polluters pay principle").
2. Putting the economic burden of pollution abatement on the beneficiaries of improved water quality. Those individuals benefiting will then provide grants and subsidies to the polluters to implement the technology necessary to reduce or eliminate pollutant loads. This is called the "benefits received" approach.
3. A combination of the two systems.

Polluters Pay

Polluters bear the chief responsibility for abatement and its direct cost. This is the most equitable payment plan. It has been primarily used to control pollution from industrial sources and has been implemented through the NPDES permit system. As pointed out in the previous section, this system requires the reclassification of diffuse sources as point sources. The payments are clear in the case of industrial polluters, who pay for pollution from their revenues. The form of payment of municipal polluters is not as clear since several types of payments are common. These include property taxes, user fees, income and sales taxes, and general revenues from other sources.

The difference between taxes and user fees is that taxes are compulsory, while charges are optional. The use of user fees to finance storm-water and CSO management implies that the service is private goods; however, storm-water management in urban areas historically has been regarded as public goods and has generally been financed by property

taxes. Without regulation it usually has been considered a low-priority service, so funding has been inadequate (Lindsay, 1988).

Regulations

Under the "polluter pays" scenario, polluters are usually required to meet specific abatement standards, subject to penalties if falling short. This is again the fundamental enforcement principle incorporated into the NPDES permit system by the states. Many have questioned why regulation and enforcement are needed in pollution abatement, specifically in nonpoint pollution programs where past (unsuccessful) practice was to rely on voluntary participation.

In the absence of regulation, pollution abatement must rely on other means to persuade polluters to install abatement devices. These means were listed by Novotny (1988), and include moral persuasion, court litigation, and financial grants. Economists have demonstrated that market forces do not work in pollution abatement (market failure) because of the external character of pollution discharges (Solow, 1978; Gaffney, 1988). An economic externality is an economic activity or action of an individual or group that economically impairs others without their consent. A technological external diseconomy (Kneese and Bower, 1968) is a situation where a particular action produces economical results on an independent entity. Diffuse pollution, both from urban and nonurban sources, is a classic example of external diseconomics (Novotny, 1988): pollutants from diffuse sources are transferred downstream where they impair downstream uses of bodies of water, yet the sufferers of downstream pollution have no economic mechanism to recover the damages from the polluters. Those responsible for the pollution of the Boston Harbor and adjacent coastal areas, or the Chesapeake Bay and other coastal bodies of water, damage the fishing and recreation industries (users), but, without regulation and its enforcement, there is no market mechanism to recover the cost of these damages. An externality problem can cross state and even international boundaries. The water quality problems of the Chesapeake Bay are caused by pollution discharges from several states, yet those who suffer the consequences of the pollution (excluding, of course, transient vacationers) live in Virginia and Maryland. A well-publicized case of coastal pollution in the San Diego, California, area¹ documents international externality problems.

¹ San Diego Bay is being polluted by waste discharges from both San Diego and Tijuana, Mexico.

Fiscal Federalism

In general terms, the U.S. federal system allocates the cost of government programs in proportion to benefits received. Where local benefits predominate, mainly local funds are used; where state or national benefits are significant, costs should be borne by state and federal governments (Braden, 1988a). Diffuse-pollution control programs should reflect these attributes of federalism. Litter control and street sweeping are a local problem, and its control is paid for locally by either taxes or user fees or both. Federalism and the recent reluctance of the federal government to commit adequate funds for diffuse-pollution abatement puts the financial burden on states and local funding. However, the federal government must play a role in financing and/or regulating and enforcing diffuse-pollution programs where interstate externality problems would impair efficient abatement. Such is the case of coastal pollution.

Uniformity

The U.S. Constitution precludes the arbitrary application of laws. In the Clean Water Act this led to (effluent) standards that are uniformly applied to all polluters, regardless of the capability of the receiving water bodies to assimilate pollution and/or the extent of damage or absence of damage imposed on downstream users. This scheme, if applied to diffuse-pollution abatement, most likely would not lead to a least cost solution that would maximize water quality improvement. Furthermore, facing the huge cost of the abatement, the uniform approach could result in a situation where pollution of some waters would remain unacceptable, while unnecessary funds would be spent in places where they would not be needed. Using uniform standards mostly implies the "polluter pays" principle, where the cost of pollution abatement is borne by the polluter and the abatement is enforced by a discharge permit.

"Targeting" is used for nonuniform abatement (Braden, 1988a) in water quality limited watersheds. Targeting diffuse-pollution abatement for inland and coastal watersheds that have a diffuse-pollution problem will result in a more efficient use of funds and will bring more water quality improvement benefits for less money. Wisconsin's or North Dakota's "Priority Watershed Program" is an example of targeting specific watersheds that have a diffuse-pollution problem. State funds in the form of grants and cost sharing (up to 70%) are then allocated for abatement in those targeted watersheds. The polluter pays principle and its enforcement are difficult to implement when targeting is used, although the mechanism for enforcement exists in some states if the

polluters fail to implement diffuse-pollution abatement. For example, the “bad actors”² legal doctrine was proposed in Wisconsin to provide further incentives to rural polluters to voluntarily implement nonpoint pollution-control practices. After sign-up and a grace period, it was proposed that the Wisconsin Department of Natural Resources (DNR) would be able to mandate participation of critical nonparticipants in the program whose nonpoint pollution was causing the DNR to fail to meet its water quality goals. Those bad actors would have to participate at a reduced subsidy (from 75% to 25%) from the state. Similar provisions for agricultural nonpoint sources are incorporated in the Food Security Act.

Targeting is also used when, for example, specific water quality improvement and waste-assimilative capacity enhancement projects are proposed, such as wetland restoration and in-stream water quality control measures. These projects require funding that should be recovered from both the beneficiaries and polluters. The funds may be recovered from the polluters, for example, in the form of fines for excessive pollution, from user charges for storm-water disposal, from fees paid by developers for their erosion-control plans, from a part of the gasoline tax, or from motor vehicle license fees. The use of stream standards and a definition of water quality limited “impaired” bodies of water, similar to point source abatement, may also be used to enforce the polluter pays principle for the abatement of diffuse pollution in targeted watersheds. The distribution of allowable waste loads based on the waste-assimilative capacity of the body of water in question can be made initially in a uniform fashion by a regulatory agency (state or the EPA). Then the polluters can be allowed to negotiate the discharge loads among themselves as long as the waste-assimilative capacity of the body of water is not exceeded. This is the principle behind the *transferable discharge permits* or *bubble approaches* (Braden, 1988b; Tietenberg, 1985).

Benefits Received Approach

In this approach the beneficiaries of improved water quality either tax themselves or pay user fees. The revenues are then allocated for pollution abatement. This can be accomplished in several ways. The beneficiaries can provide grants and subsidies to polluters to install abatements or the funds are used for mitigation of adverse water quality effects (e.g., weed control or pollution diversion).

This concept presumes that the sufferers of adverse water quality are

²In 1992 the “bad actor” provision was included in the Wisconsin Nonpoint Pollution Control Legislation, but was vetoed by the governor.

willing to pay for the water quality improvement benefits. The willingness to pay is not the same as the damage (cost) of pollution, but it is related to it. Nor should the willingness to pay be equated with the ability to pay because of the very important premise that those asked to pay for the benefits of improved water quality either suffer or can potentially suffer from adverse water quality. Thus, users of beaches or recreational fishers are willing to pay for improved water quality and, consequently, for diffuse-pollution abatement.

The problem with *willingness to pay* arises when the beneficiaries and polluters are not a part of the same group and there is a distance between the pollution source and the location where the benefits will be received. Suburban owners of septic systems (including those that are failing) located far from a body of water are generally unwilling to connect to a sewer system if they do not receive water quality benefits, while owners of water shoreline properties are willing to tax themselves in return for improved water quality. Therefore, the polluter pays principle is most equitable when polluters and beneficiaries of improved water quality come from the same population group, for example, a large urban area on a body of water that is in or reasonably close to it. The Boston harbor, located within metropolitan Boston, is such an example. The construction industry and its clients (future homeowners) living in the area affected by construction pollution represent another example where the polluters pay principle should be strictly adhered to.

Considering the magnitude of the diffuse-pollution problem, both payment schemes may be considered. The *polluter pays* funding approach should be applied to most local problems, using regulation and standards. Such standards should apply uniformly to all sources of urban pollution, category by category. These standards can be in the form of *design (performance) standards* that establish that certain pollution-abatement technology is to be applied uniformly to all sources within the source category (e.g., construction erosion and sediment control ordinances, mandatory street sweeping, mandatory collection and treatment of a specified portion of the average surface runoff, or limiting the number of CSOs) or *effluent standards* that limit or prohibit certain polluting substances in effluents, CSOs, runoff, and subsurface flows.

In "targeted watersheds," that is, in areas with a serious and extended water quality problem and with the important premise that beneficiaries are separated from the polluters, a two-level approach may be used (Novotny, 1988). In the first level, the polluters pay principle, using mandatory pollution control, should be enforced by a NPDES-type permit system.

The NPDES permit program currently being implemented for the

control of urban stormwater is not uniform nor is it equitable (see Chapter 2 for a discussion of equity and other imperatives of pollution control). It also contains many exclusions such as small urban areas, interurban freeways and highways, military facilities and training grounds, suburban zones with septic tanks, small construction sites, and other urban and nonurban sources. Such exclusions could preclude an effective diffuse-population program. If these controls do not suffice, a combination of the benefits received and polluter pays approaches could be used to achieve the water quality goals. The benefits received approach is then used to secure additional necessary funding for abatement. This can be accomplished by a combination of grants from state and federal funds, from taxes on received benefits related to the improved water quality, and user fees, such as money collected for fishing licenses, boat launching fees, and the fees for the use of public beaches. Taxing riparian land owners who receive disproportionately large benefits from the use of bodies of water is a viable alternative.

These funds can then be allocated to the targeted areas. The polluter pays approach can be incorporated into the discharge permit in the targeted areas. This permit should be related to the waste-assimilative capacity of the body of water or coastal area in question. The polluters (urban areas, industries, storm-water management utilities) should then be allowed to renegotiate the discharge levels among themselves according to the transferable discharge permits scheme (Braden, 1988b; Tietenberg, 1985). The equity of proportioning the additional cost of abatement in the targeted areas between the polluters and users (beneficiaries) must be incorporated in the plans of abatement.

Many authors (Gaffney, 1988; Braden, 1988b; Novotny, 1988) warn against the excessive reliance on subsidies in diffuse-pollution abatement programs. Although subsidies are politically attractive to lawmakers, it is a well-established fact that in the absence of regulation and enforcement, polluters will do nothing unless and until they receive a full subsidy for the cost of abatement. Furthermore, allocating the subsidies to only certain types of abatement may lead to inefficient solutions. Braden (1988b) also pointed out that subsidies actually encourage pollution activities by shielding them from their true cost to society. He concluded that subsidies are too costly and too prone to perversion. On the other hand, since taxing riparian and other beneficiaries of improved water quality is a viable alternative for raising revenues, partial cost sharing and incentives may be considered. The bulk of the funding raised from the beneficiaries and from fines for pollution, however, should be used for actions such as ecological restoration and the protection of coastal wetlands, and not as grants to polluters.

The first level of implementation of diffuse pollution abatement does not require a great deal of reliance on water quality standards; however, it is paramount that wet-weather water quality standards be considered when the second-level approach is used. Adequate wet-weather water quality standards are needed in situations when mandatory uniform abatement does not result in the satisfactory water quality improvement of local bodies of water, because the level of abatement must then be related to the waste-assimilative capacity of the body of water in question, which is determined (primarily by water quality modeling) by using the accepted and enforced water quality standards.

The "Bubble Approach" and the Bubble Dimension

In a "classic" watershed planning approach the benefits of the project are weighted against its cost, while other objectives of the development are also considered, such as the resource conservation and preservation and well-being of people affected by the development (see Maas et al., 1970 for a most authoritative discussion on the objectives and benefit-cost analysis of water resource development projects). In the water-pollution control area, the objectives were given by Congress in the Clean Water Act (1972 Amendments) and are well known and followed. The major objectives of the Clean Water Act are fish and aquatic wildlife protection and preservation and the suitability of receiving bodies of water for contact recreation. These objectives were translated into water quality criteria, which were then implemented by the states as standards. Furthermore, the quality of point sources must comply with the equity, irreversible impact technology-based standards.

The use of dual standards resulted in two planning scenarios with interchangeable objectives and constraints. In the first scenario, the waste-assimilative capacity of the receiving body of water is greater than the combined pollution loads from all sources after the effluent standards for point sources are implemented. This leads to the so-called "effluent controlled" or "nontargeted" planning scenario. The objective in this scenario is to minimize the cost at each individual source, with the avoidance of violation of the effluent standard as a constraint. Point source effluent discharge permits are then based on the technology-based effluent standards. Under the present regulations, incentives for nonpoint source control are minimal and the abatement approaches rely mostly on voluntary approaches. This violates the "principle of uniformity," which states that all polluters should be treated equally.

In the second scenario, the waste-assimilative capacity is less than the combined pollution load *from all sources*, resulting in a violation of the

water quality standards. This is the so-called “stream (receiving body of water) -limited” or “targeted” situation, whereby effluent discharge permits are based on the waste-assimilative capacity of the receiving body of water. The objective of water quality abatement is thus to meet the stream water quality standards at minimum cost or under the economic and fiscal constraints.

Hence, the waste-assimilative capacity of the receiving body of water is the most important constraint that determines the maximum allowable pollution load that cannot be exceeded. This is the maximum size of the “bubble,” as it is referred to in the jargon of pollution-abatement literature. If the pollution load exceeds the waste-assimilative capacity, the receiving body of water is damaged and the downstream water uses are impaired—the bubble has burst. However, the WAC may be increased by instream WAC enhancement measures. Hence, the “bubble” is flexible and can expand. Under the “effluent-limited” situation, however, the primary objective of the abatement efforts is minimization of cost. Hence the size of the “bubble” is then related to the budgetary constraints of the agencies responsible for financing the abatement.

The policy options that can be used to achieve the water quality goal (i.e., to attain the designated use) include pollution taxes, water quality standards, effluent standards, tradeable discharge permits, process design standards, liability for damages, and abatement subsidies. Incentives can be both positive and negative, that is, rewarding abatement or penalizing pollution. No single option is available to achieve the goal, hence the use attainability analysis must consider the best combination of policy options. It should be pointed out that the United States has made comparatively little use of fiscal incentives for pollution control (Braden, 1993).

An important source of inefficiency in pollution control occurs in the waste loads allocation process where high-cost abaters are forced to meet the same standards as low-cost abaters, which can lead to wasteful abatement. It is true that using differential treatment for polluters, based on the cost of abatement, is causing legal problems and may be difficult or impossible to implement. However, using the so-called *transferrable discharge permits* to transfer the cost from high-cost to low-cost polluters is feasible. Here the polluters negotiate among themselves the waste allocation subject to the constraint of the limited loading capacity of the receiving body of water. It appears that “pollution trading” may be on the verge of becoming a major tool for the EPA and states to attack the nation’s biggest source of water pollution—the diffuse-source pollution loads (*Clean Water Report*, June 9, 1992). Hence, both point/point and point/nonpoint source abatement trading is feasible.

Under a proposal considered by the U.S. EPA in the early 1990s, a point/point source (P/P) trading program would allow a discharger to avoid a part of treatment or treatment upgrade by paying or arranging for at least an equal reduction in discharges from other facilities releasing in the same body of water. All facilities would still have to comply with the technology-based (equity) effluent limitations. Hence, these trading concepts are applicable to water quality limited bodies of water.

Targeting water quality limited streams implies the selection and application of pollution-control policies that promote economic efficiency. There are three aspects of targeting. First, the cost of administering policies should be balanced against the benefits of policy refinement. Second, targeting enables differentiation between the sources and may promote the use of transferrable discharge permits. Third, it allows differentiation in space and time, that is, the bodies of water that require abatement most and/or are most desirable will be abated ahead of some other less desirable bodies of water. The third aspect is dictated by budgetary and other economic constraints imposed on pollution abatement and by the fact that there is a tremendous variability in physical and ecological characteristics among the nation's surface and coastal waters. Consequently, different systems respond in different ways to pollution stresses.

The Committee on Restoration of Aquatic Ecosystems of the National Academy of Engineering (1992) recommends a national body of water restoration effort that would include four elements:

1. National restoration goals and assessment strategies for each ecoregion.
2. Principles for priority setting and decision making.
3. Policy and program redesign for federal and state agencies to emphasize restoration.
4. Innovation in financing and use of land and water markets.

In light of existing and anticipated budgetary constraints, innovative ways to finance restoration efforts are necessary.

INSTITUTIONAL ISSUES

As pointed out previously, water quality is a response to various sources of potential contaminants, including natural and cultural sources. The cultural sources are then divided into point and nonpoint. Equity and irreversible impact criteria (technology-based effluent and performance standards) must be formulated and enforced in an integrated fashion both on point and nonpoint sources. Excluding certain sources would violate

the equity criterion. Third, the receiving water standards and criteria determine the maximum permissible load $TMDL = WQS \times Q$, where WQS is the standard and Q is the flow. The standards are related to the designated use of the body of water. In some cases, the natural load, BL, may be greater than TMDL. If this is the case, then either the designated use is changed or no waste discharges should be permitted. This may happen if the body of water in question is designated as an outstanding natural resource or the water is used for water supply.

In most cases, however, there may be an excess load between the TMDL and the background load (BL). This excess permissible load then constitutes the waste-assimilative capacity WAC or $WAC = TMDL - BL = F^{-1}(WQS)$. The F^{-1} is a functional relationship (model) that relates the water quality to allochthonous waste discharges. If the total load from cultural sources (TDL) is less than WAC, then the body of water is not water quality limited and cannot be considered a targeted system. The waste-discharge permits can be issued based on equity and irreversible impact standards. Such permits can be issued by state or even local (county) land-use and water quality agencies and could be based on design and/or performance criteria. Tying these permits to water quality and requiring excessive monitoring from individual dischargers would be wasteful and could present an administrative nightmare.

If, however, the total load (TDL) is greater than WAC, then the receiving body of water is limited and should be considered as a potentially *targeted system*. A targeted watershed with a major diffuse pollution problem can be declared "impaired." Waste-discharge permits must then be related to the waste-assimilative capacity of the body of water in question (for example, an estuarine system). The management options in targeted (impaired) watersheds are:

1. Enforce more stringent effluent and performance standards
2. Improve WAC (e.g., by dredging in-place sediments, low-flow augmentation)
3. Change receiving water standards by proving by the use attainability analysis that the standards cannot be attained

Based on these options, it is evident three types of institutions are needed.

The first category of institution is *regulators*. These institutions are already in place in the form of the U.S. Environmental Protection Agency, to execute federal policies, and state pollution-control agencies (Departments of Environmental Protection, Natural Resources, etc.), for intrastate pollution control. These institutions carry out the legislative policy mandates, specify standards and criteria, provide oversight, arbi-

tration, research, provide and distribute grants when federal and state interests in water quality remediation are involved, among other activities. In the integrated pollution-abatement scenario these agencies decide whether a body of water is a targeted (impaired) system or a national outstanding resource. They also designate the uses of the bodies of water, specify the standards and execute use attainability studies. They provide oversight for water management and pollution abatement agencies. In nontargeted watersheds where permits are based on effluent (point sources) design-performance (diffuse sources standards), the agencies may delegate their permit-issuing authority to local (county) agencies or establish their own permit-issuance branch. They can also provide arbitration in disputes between pollution-abatement and water quality management institutions. This agency *should not manage the water resources or pollution abatement*. Financing the agencies should be derived from state or federal sources, and should not use funds obtained from polluters or users.

The second category of institution includes the *waste dischargers*. Present municipal sewerage agencies and industrial polluters are covered by this category. These agencies are responsible for urban, CSO, and point source abatement. As an alternative to municipal sewerage districts, which in most cases are responsible for urban point and diffuse-pollution abatement and which utilize taxes for financing, many communities have recently created storm-water utilities financed by user fees (Lindsay, 1988; American Society of Civil Engineers, 1985). Users are people and organizations that generate storm-water runoff that is discharged into the utility drainage system. In rural areas, drainage and irrigation districts and utilities, in addition to individual farm operators, are examples of potential organization of waste discharges.

Financing for these agencies and their pollution-abatement efforts should be mostly based on the *polluter pays principle* and regulated by a permit. For example, fees can be based on estimates of the amount of runoff or drainage (irrigation return) flow that leaves a user's property. Rate factors can be correlated with land use. With the utility approach, users are differentiated from beneficiaries, who are people protected from the damage of runoff or who receive other benefits, such as improved water quality. For wastewater treatment the charges are typically based on the measured or estimated flow and pollutant load. In some states, existing water management organizations (Florida) are now assuming the role of a storm-water management utility (Viessman, 1988).

In nontargeted watersheds, as an alternative approach to issuing individual permits for each pollution source, the pollution-abatement agency can ask the regulating institution for an umbrella technology-

based permit for each category of pollution source within its jurisdiction. The pollution-abatement agency will then issue permits and/or monitor compliance. In nontargeted watersheds the two agencies (regulators and pollution abatement) are the only institutions needed.

In targeted watersheds, a new concept of water quality management agencies should be established. This regional watershedwide agency would be responsible for planning the actions, managing discharges and their control, managing water quality, and raising revenues for carrying out the operation and capital investments for waste-assimilative capacity enhancement measures, such as restoration and creation of riparian wetlands or the management of buffer strips.

The decision whether a watershed is a targeted (impaired) or nontargeted is based on a scientific-engineering study that would estimate the total maximum daily load (TMDL) and waste-assimilative capacity, use attainability analysis, and on the public surveys of use impairment and use attainability. Such studies should be performed by the regulatory agency or by planning agencies (consultants) designated by the regulatory agency (e.g., the EPA for interstate and coastal water bodies and state regulatory agencies for interstate watersheds). Funding authorization for these studies can be derived from Section 319 or 208 of the Clean Water Act. Unlike previous Section 208 studies, if the body of water is declared a targeted or impaired body of water, the planning activity must be followed by an implementation that includes the establishment of watershedwide agencies responsible for the implementation.

Diffuse-pollution problems are ubiquitous, their dimension and origin not well established, often fitting watershed boundaries rather than political ones (Viessman, 1988). By analyzing the problems of diffuse pollution in a regional rather than a local context, it is possible to identify more efficient options that might not be recognized or available otherwise. For example, successes with the regional approach to diffuse pollution management in the Washington, D.C., area demonstrated that gains can be substantial (Viessman and Welty, 1985). Other examples of successful regional water and water quality management organizations include the Florida Water Management Districts,³ the Water Pollution

³The Florida Water Resources Act of 1972 provided for the creation of water management districts. These regional management agencies have broad powers relative to the allocation and use of Florida's waters. In the early days the districts were largely concerned with water quantity issues, water supply, flood control, and drainage. With the passage of time, however, they have moved solidly into the water quality management area. Furthermore, they have taken on an environmentally slanted orientation.

For example, the core mission of the South Florida Water Management District is:

Management Associations in the Ruhr industrial district of Germany (Kneese and Bower, 1968), the Nebraska Natural Resources Districts and British Water Authorities.

These regional organization, which have the broad authority to raise revenues and implement management, would be more efficient than fragmented local and state organizations, which in many cases have limited authority to regulate, but do not have the authority to conduct in-stream water quality management (e.g., wetland restoration or low-flow augmentation) or collect user fees from the beneficiaries of improved water quality. Also discharge permit trading (the bubble approach to pollution discharge permitting) can only be accomplished if a regional watershedwide authority is a "referee" in the trading process and the same authority will make sure that the final discharge loads agreed upon by the polluters do not exceed the waste-assimilative capacity. Subsequently, the agency will continually reissue the permits.

The management of diffuse pollution also requires interagency cooperation and integrated approaches. Programs currently or potentially related to diffuse pollution in coastal areas are carried out by the U.S. Environmental Protection Agency, NOAA, U.S. Department of Agriculture, and many state and local agencies. The Florida Water Management Organizations again provide a good example of how a regional umbrella watershed management organization can cooperate and coordinate these multiagency efforts. Organizational arrangements and responsibilities of the regional water quality management organizations should be incorporated in the planning process (for example, Section 319 planning).

The core mission of the South Florida Water Management District is to manage water and related resources for the benefit of the public and in keeping with the needs of the region for the purpose of providing:

- Environmental protection and Enhancement
- Water Supply
- Flood Protection
- Water quality protection

This is being implemented through the coordination of operations, planning, public involvement, regulation, and construction. Inherent in the mission is the responsibility to assist the public and government officials in growth management by identifying water resource impact of land-use decisions and by advising on options for reducing adverse impacts and protecting water resources.

The water management districts have dealt with a variety of nonpoint pollution problems and in doing so have employed a number of nonconventional approaches, including Save Our River programs, Lake Okeechobee restoration, and the Lake Apopka land swamp proposal (Viessmann, 1988).

Water quality management today is heavily influenced by regulatory measures, all of which come from several directions (the EPA, Department of Interior programs and authorizations, Department of Agriculture, states, etc.). Viessman (1988) correctly pointed out that water management guided by narrowly devised regulatory measures, that is, by regulatory actions that only call for "planning," but do not provide for implementation (such as the planning activities of Section 208 of the Clean Water Act), are destined to be mediocre at best and destructive at worst.

Strategy of Abatement

Generally, mitigation of diffuse pollution from urban and nonurban areas relies on structural and nonstructural measures, or the so-called best management practices. These are well known and have been published in numerous EPA and state manuals, as well as in several chapters of this book. The strategy, however, means selecting the most optimal alternatives under the given legislative and fiscal constraints.

As pointed out in many articles (for example, see Novotny [1988] for a summary), as well as earlier in this chapter, abatement requires regulation and enforcement. As we have also seen, under the current legislative means of enforcement, only those diffuse sources that have been legally classified as "point sources" have a reasonable chance for clean-up. Furthermore, additional mitigation and land protection can be achieved using present or near-future wetland protection laws. This still leaves a large number of sources for which chances of abatement are minimal.

It is evident that the land-disturbing activities (construction, wetland drainage, stream channelization, etc.) during land-use transition are responsible for the highest loads of pollutants. Hence the strategy of mitigation should first focus on their control. However, enforcement is not available or is insufficient.

In established urban lands without significant construction, the NURP study established that there is not much difference between the strength (EMCs) of urban runoff from the major land uses (see Chapter 8). However, since the degree of imperviousness affects the runoff volume, the highest loads can be expected from highly impervious urban lands. Therein, structural facilities, such as detention-retention, followed by treatment, seem the most appropriate. For more pervious areas, typically for low- to medium-intensity residential areas with storm sewers, less structurally intensive measures, such as buffer strips along the streams; floodplain management; retention, storage, and sedimentation in sub-

urban ponds and lakes; and public education to reduce or eliminate (by switching to xeriscape) the use of fertilizers and pesticides on suburban lawns, are appropriate.

In suburban zones that are served by septic tanks and do not have sewers, controls aimed at the reduction of ground-water protection from nitrate contaminations and surface pollution by surfacing septic tank effluents, should be strengthened. Since construction activities yield the highest number of sediments, effective construction erosion-control regulations must be implemented on a national scale.

The best and most efficient results in controlling pollution by urban diffuse sources is achieved if the management is incorporated into the development plans. However, the recent report to the Congress by the General Accounting Office (U.S. GAO, 1990) pointed out that the EPA regulations aimed at the control of pollution by urban runoff are not sufficient. Many areas of the country experiencing the most rapid growth are not included, and by delaying the implementation of storm-water control programs, the U.S. Environmental Protection Agency "is losing a valuable opportunity to prevent stormwater problems rather than rely on expensive structural controls after development has occurred."

In agricultural areas, the programs mandated by the Food Security Act (the Farm Bill) should concentrate on lands and practices that have the most adverse water quality impact, with soil and water conservation as a constraint. Reliance on purely voluntary approaches is ineffective and typically results in very low participation of polluters in the programs.

References

- American Society of Civil Engineers (1985). Street users fees, *Civ. Eng.*, p. 43.
- Braden, J. B. (1988a). Financing nonpoint pollution abatement—the peculiar case of agriculture, in *Political, Institutional and Fiscal Alternatives for Nonpoint Pollution Abatement Programs*, V. Novotny, ed., Marquette University Press, Milwaukee, WI.
- Braden, J. B. (1988b). Nonpoint pollution policies and politics: The role of economic incentives, in *Proceedings, Nonpoint Pollution 1988—Policy, Economy, Management and Appropriate Technology*, V. Novotny, ed., American Water Resources Association, Bethesda, MD, pp. 57–65.
- Braden, J. B. (1993). Categorization of water bodies—targeting, in *Use Attainability Analysis Manual*, prepared for Water Environment Research Foundation, Alexandria, VA, AquaNova Internat., Mequon, WI.
- Cairns, J., Jr., and D. Orvas (1989). Ecological consequence assessment: Predicting effects of hazardous substances upon aquatic ecosystems, in *Ecological Engineering: An Introduction to Ecotechnology*, W. Mitsch and S. E. Jørgensen, eds., Wiley, New York, pp. 409–442.
- Committee on Restoration of Aquatic Ecosystems (1992). *Restoration of Aquatic*

- Ecosystems—Science, Technology, and Public Policy*, National Academy Press, Washington, DC.
- Gaffney, M. (1988). Nonpoint pollution: Tractable solutions to intractable problems, in *Political, Institutional and Fiscal Alternatives for Nonpoint Pollution Abatement Programs*, V. Novotny, ed., Marquette University Press, Milwaukee, WI.
- Haigh, N., and F. Irwin (1990). *Integrated Pollution Control in Europe and North America*, The Conservation Foundation, Washington, DC.
- Imhoff, K., and K. R. Imhoff (1990). *Taschenbuch der Stadtenwasserung*, Oldenbourg Verlag, Munich.
- Kneese, A. V., and B. T. Bower (1968). *Managing water quality: Economics, Technology, Institutions*, Johns Hopkins University Press, Baltimore, MD.
- Lindsay, G. (1988). Equity and efficiency aspects of alternate stormwater management financing systems, in *Proceedings, Nonpoint Pollution 1988—Policy, Economy, Management and Appropriate Technology*, American Water Resources Association, Bethesda, MD, pp. 87–96.
- Maas, A., et al. (1970). *Design of Water Resources Systems*, Harvard University Press, Cambridge, MA.
- Mitsch, W., and S. E. Jorgensen, eds. (1989). *Ecological Engineering: An Introduction to Ecotechnology*, Wiley, New York.
- North American Lake Management Society (1990). *Lake and Reservoir Restoration Guidance Manual*, EPA 440/4-90-006, U.S. Environmental Protection Agency, Washington, DC.
- Novotny, V. (1988). Diffuse (nonpoint) pollution—a political, institutional, and fiscal problem, *J. WPCF* 60(8):1404–1413.
- Novotny, V., and G. Chesters (1981). *Handbook of Nonpoint Pollution: Sources and Management*, Van Nostrand Reinhold, New York.
- Novotny, V., J. Braden, D. White, H. Jones, and R. Schonter (1993). *Integrated Approach to Water Quality Abatement—Role of the Use Attainability Analysis*, paper presented at Surface Water Quality Symposium, Water Environment Federation Annual Conference, Anaheim, CA, October, 1993.
- Novotny, V., et al. (1994). *Use Attainability Analysis—A Key Component of Integrated Water Quality Planning Efforts*, Water Environment Research Foundation, Alexandria, VA.
- Solow, R. M. (1978). The economist's approach to pollution and its control, in *Pollution Taxes, Effluent Charges and Other Alternatives for Pollution Control*, U.S. Congress Rep. No. 95-5, Washington, DC, pp. 137–142.
- Tietenberg, T. H. (1985). *Emissions Trading*, Resources for the Future, Washington, DC.
- U.S. Environmental Protection Agency (1983). *Technical Support Document for Waterbody Surveys and Assessments for Conducting Use Attainability Analysis*, Vol. I–III, Office of Water Regulations and Standards, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1988). *Introduction to Water Quality Standards*, Office of Water, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1991a). *Guidelines for the Preparation of*

- the 1992 State Water Quality Assessments*, 305(b) reports, Office of Wetlands, Oceans and Watersheds, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1991b). *Technical Support Document for Water Quality-Based Toxics Control*, EPA 605/2-90-001, Office of Water, U.S. EPA, Washington, DC.
- U.S. Environmental Protection Agency (1991c). *Guidance for Water Quality-Based Decisions: The TMDL Process*, EPA 440/4-91-001, Assessment and Watershed Protection Division, U.S. EPA, Washington, DC.
- U.S. General Accounting Office (1990). *Water Pollution—Greater EPA Leadership Needed to Reduce Nonpoint Source Pollution*, report to Congress, GAO/RCED-91-10, Washington, DC.
- Viessman, W. (1988). Regional options for managing and regulating nonpoint pollution, in *Political, Institutional and Fiscal Alternatives for Nonpoint Pollution Abatement Programs*, V. Novotny, ed., Marquette University Press, Milwaukee, WI.
- Viessman, W., and C. Welty (1985). *Water Management: Technology and Institutions*, Harper and Row, New York.
- Wolman, M. G. (1988). Changing national water quality policies, *J. WPCF* **60**(10):1774–1781.

Epilogue

People can be attracted by new ways of ordering their lives, as well as driven by the recognition of what will happen if they do not change.

Herman Daly and John Cobb (quoted in
State of the World 1990, L. R. Brown)

We began this book with the gloom of threatening environmental problems caused by some forms of diffuse pollution. We would like to end with a more optimistic closure.

In 1992/1993 a two-year experiment was conducted in Arizona. In the experiment eight scientists lived in a hermetically sealed environmental structure called *Biosphere II*. Supposedly the world outside of this structure is Biosphere I. During the experiment living processes generated waste; however, the waste and energy in it were recycled back in the ecological cycle. Essentially, Biosphere II did not generate a substantial net increase of waste.

It was not a precisely scientific experiment and some criticized it as a commercialized, publicity project. Nevertheless, this experiment has brought us to the third millennium. If man is to survive on this planet, a sustainable ecology is a must. Figures 1 and 2 show the contrast between the ecological cycles before the year 2000 and those expected to occur after the year 2000. They show change from the traditional energy-demanding and waste-producing living process to a sustainable low-waste, less energy-demanding ecology. Ecological engineering introduced in the Prologue and Chapter 1 will be a key in achieving this goal.

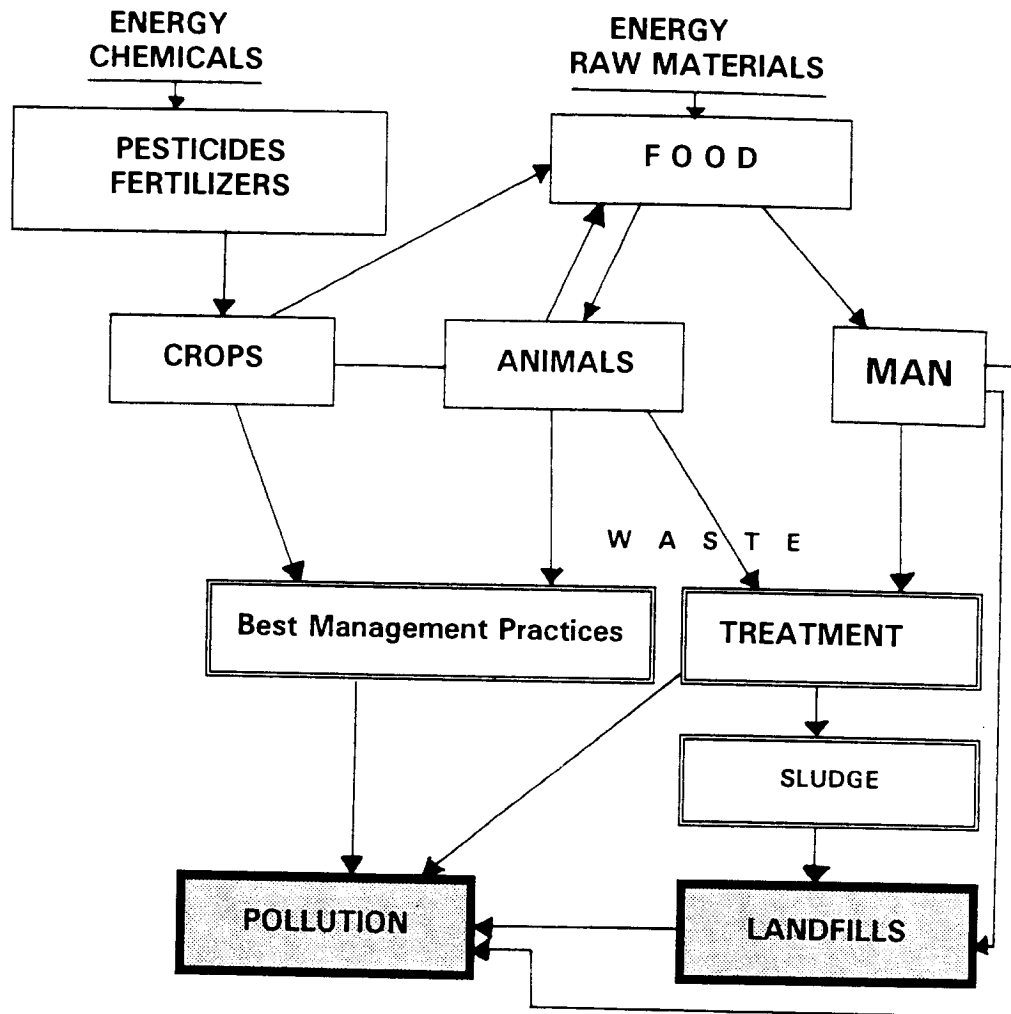


FIGURE 1. Ecological cycles of intensive mass flows of energy, raw materials, and waste, producing high levels of pollution.

Hence, the environmental policies of the third millennium will be different from those of the second millennium, and especially of the second half of the twentieth century. The fact that diffuse pollution is now both global as well as local requires broad international cooperation. With the adoption of the *sustainable development imperative* by the OECD nations, the overall approach to environmental policy is changing. Attention is now being focused on ensuring that all dimensions of environmental problems are considered, including pollution control, resource management, and broad quality of life; on developing more effective institutional arrangements for the formulation and implementation of environmental policies; on promoting technological changes

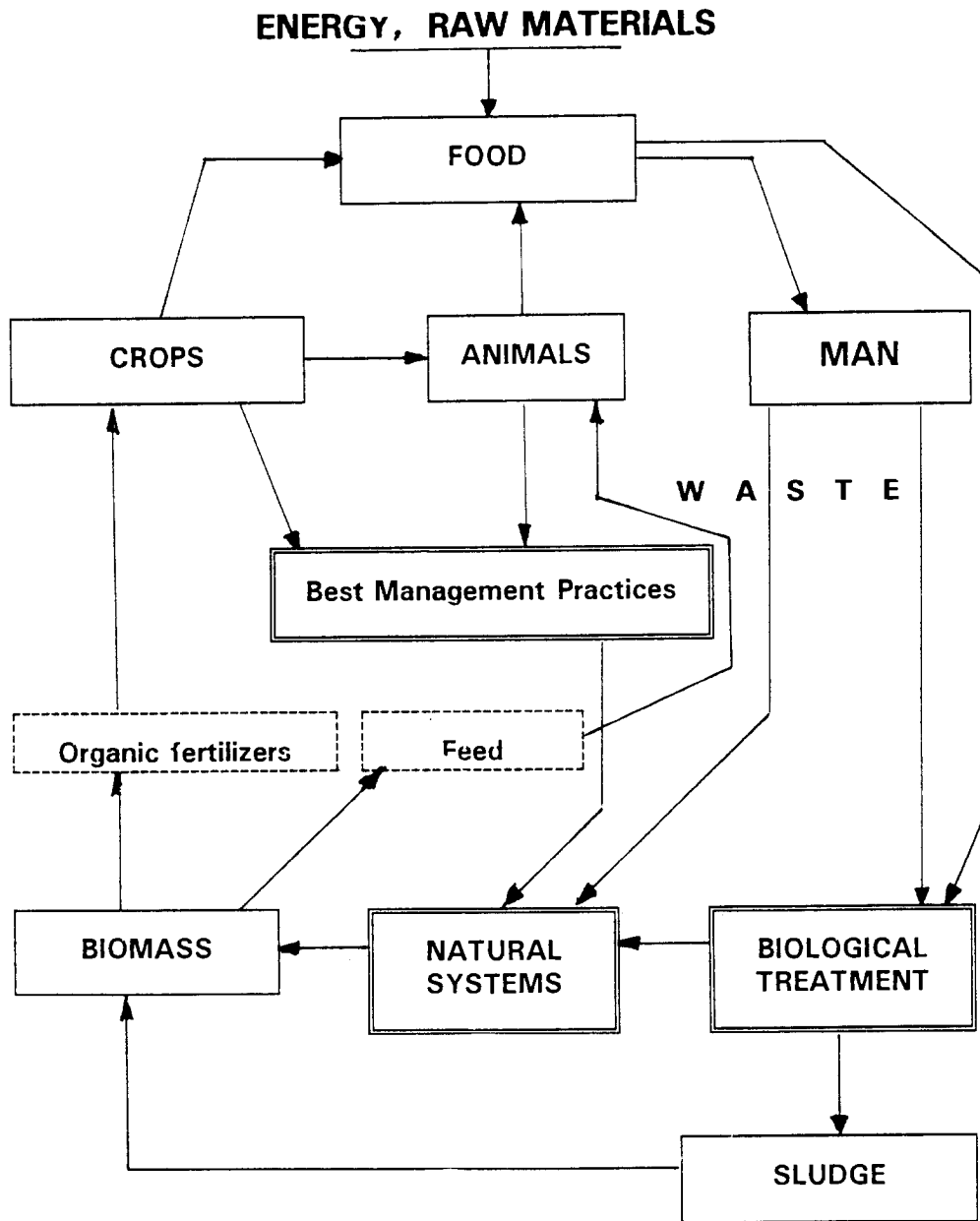


FIGURE 2. Ecological cycle of mass flows of energy, raw materials, and byproducts in a closed-loop sustainable mode with minimum pollution.

toward “clean, green” growth; on using economic instruments to provide market signals for environmental protection that better reflect relative scarcities; on streamlining the regulatory instruments for greater efficiency and cost effectiveness; on modifying production/consumption patterns to maintain a stock of scarce resources and reduce pollution; and on analyzing linkages between the environment and the economy and de-

veloping environmental indicators for measuring environmental performance. Today's and future approaches also emphasize (OECD, 1991):

- Broader objectives for environmental policy based upon the concept of sustainable development, and focus upon resource conservation as well as pollution control
- More effective institutional arrangements for the formulation and implementation of environmental policy
- More use of economic instruments to provide appropriate market signals for environmental protection; and improvements to the efficiency and cost effectiveness of regulatory instruments
- Wider use of anticipatory approaches to the formulation and implementation of environmental policy
- Development of more integrated approaches to environmental controls, both within the environmental sector and between the environmental sector and other sectors of the economy.

As pointed out in the *State of the World 1990* (Brown, 1990), building a more environmentally stable (sustainable) world society requires a vision of it. If the acid rain producing fossil-fuel power plants are to be replaced, then with what? If forests are not to be cleared for intensive agriculture, then where will the food be coming from? If a throwaway culture leads to pollution, then what should replace it? All these activities produce diffuse pollution.

A sustainable society is the one that satisfies its needs without jeopardizing the prospects of future generations. It is becoming increasingly apparent that the future economy cannot be driven by fossil-fueled power plants, heating, and vehicular traffic that can have catastrophic climatic consequences. It has been estimated that to stabilize the climate the per capita fossil-fuel consumption must be cut to about one-eighth of what it was in the late 1980s. If this is achieved, it obviously will also reduce in about the same proportion associated acid rainfall and toxic (mostly diffuse) emissions. Most likely solar, geothermal, and to a lesser degree (safe) nuclear energy sources will replace fossil fuel. Electric automobiles and people movers have already been introduced and successfully tested. Electric trains have long been the dominant means of transportation in many advanced countries (however, note that the production of electricity for electric trains and cars causes acid rain and other forms of diffuse pollution).

Reuse and recycling will replace throwaway waste. Biological and natural treatment systems provide the link between the waste production and reusable biomass. Biomass harvested from buffer strips and lands put into the conservation reserve (wood and hay) can be reused. Similarly,

organic fertilizers and soil conditioners can be produced from the biomass that is harvested from the wetlands and aquaculture systems used for assimilating the waste. Organic fertilizers are less likely to pollute ground-water aquifers and, with proper management, their use will not threaten surface-water resources. In some other ecological waste-disposal systems the produced organic biomass may immobilize and help to decompose residual toxic chemicals that will still be used.

In the third millennium, land could be used in accordance with the rules and laws of sustainability. There should be no loss of wetlands. As a matter of fact, lowlands surrounding bodies of water that have been drained will be restored to their wetland status. High slope eroding lands will be reforested or terraced. Agriculture will rely more on nitrogen-fixing plants and organic fertilizers and less on chemicals.

Farmers could be planting less monoculture crops and will be relying more on crop rotation and soil conservation to maintain soil moisture and to reduce soil loss and pesticide and fertilizer application rates. It is already apparent that farming in arid and semiarid areas, which relies heavily on mining ground-water aquifers, will cease in a few decades. New, less water-demanding crops will replace the thirsty and polluting crops in arid areas. This will result in less discharge of heavily polluted irrigation return flows.

In urban areas, diffuse-pollution-generating urban sprawl should be controlled, and suburbanites in developed countries will consider xeriscape rather than planting environmentally damaging lawns. Diffuse-pollution abatement will be incorporated into the landscape of urban environments, along with the retardance and water-conservation drainage.

As pointed out in this book, the solution of the problem of diffuse pollution is closely tied to political and economic factors. The new political order now emerging will spend less on military and more on environment, less on heavy polluting industries, oil refineries, and vehicular traffic and more on solar energy, recycling, and environmental protection. The *State of the World 1990* concluded that "as the transition to a more environmentally benign economy progresses, sustainability will gradually reclipse growth oriented economy as the focus of economic policy making." As far as the environmental engineering aspects of this change is concerned, the present book has tried to open the door.

References

- Brown, L. R. (1990). *State of the World 1990*, W.W. Norton, New York.
- Organization for Economic Co-operation and Development (1991). *The State of the Environment*, OECD, Paris.

Appendixes

APPENDIX A

CONVERSION FACTORS

U.S. Customary to S.I. (Metric)			
U.S. Customary Units	Multiply by to obtain	S.I.	
		Symbol	Name
acre	0.405	ha	hectare ^a
acre-ft	1233.5	m ³	cubic meter ^b
acre-in	102.79	m ³	
foot (ft)	0.3048	m	meter ^c
ft/s (fps)	0.3048	m/s	meter per second
ft ² (sqft)	0.0920	m ²	square meter ^d
ft ³ (cuft)	0.0283	m ³	cubic meter
ft ³ /s (cfs)	0.0283	m ³ /s	cubic meters per second
°F	0.555 (°F-32)	°C	degrees Celsius
gallon-U.S. (gal)	3.785	l	liter
gal/acre	9.353	l/ha	liter per hectare
gal/ft ²	40.743	l/m ²	liters per square meter
gal/ft ²	0.0407	m ³ /m ² = m	meter
gal/ft ² -day	4.72 × 10 ⁻⁷	m/s	meter per second
gal/ft-day	1.438 × 10 ⁻⁷	m ² /s	square meter per second
hp	0.746	kW	kilowatts
inch (in.)	2.54	cm	centimeter
in. ²	6.452	cm ²	square centimeter
in. ³	16.39	cm ³	cubic centimeters
in. ³	0.0164	l	liter
pound (lb)	0.454	kg	kilograms ^e
lb	454	g	grams

Continued

U.S. Customary to S.I. (Metric)			
U.S. Customary Units	Multiply by to obtain	S.I.	
		Symbol	Name
lb/acre	1.121	kg/ha	kilograms per hectare
lb/ft ³	16,042	g/m ³	grams per cubic meter ^f
lb/in. ² (psi)	0.0703	kg/cm ²	kilograms per square cm ^g
lb/Mgal	0.120	mg/l	milligrams per liter
lb/mi	0.282	g/m	grams per meter
mile (mi)	1.609	km	kilometer
mile/hour (mph)	0.447	m/s	meter per second
million gallons (Mgal)	3785	m ³	cubic meters
million gallons per day (Mgd)	0.0438	m ³ /s	cubic meters per second
mi ² (sqmi)	2.59	km ²	square kilometer
parts per billion (ppb) in water	1.0	µg/l	micrograms per liter
parts per million (ppm) in water	1.0	mg/l	milligrams per liter
ton (short)	0.907	t	tonne
tons/acre	2240	kg/ha	kilograms per hectare
tons/sqmi	3.503	kg/ha	kilograms per hectare
yard (yd)	0.914	m	meter
yd ³	0.765	m ³	cubic meter

^a 1 ha = 10,000 m² = 0.01 km²

^e 1 kg = 1000 g = 0.001 tonne

^b 1 m³ = 1000 l

1 g = 1000 mg = 10⁶ µg = 10⁹ ng

^c 1 m = 100 cm = 0.001 km = 1000 mm

^f 1 g/m³ = 1 mg/l

^d 1 m² = 10,000 cm² = 10⁻⁶ km²

^g 1 kg/cm² = 0.968 atm = 0.981 bars

APPENDIX B

U.S. EPA Water Quality Criteria for Dissolved Oxygen Concentrations

	Cold-Water Biota		Warm-Water Biota	
	Early Life Stages ^{a,b}	Other Life Stages	Early Life Stages ^b	Other Life Stages
30-day mean	NA ^c	6.5	NA	5.5
7-day mean	9.5 (6.5)	NA	6.0	NA
7-day mean minimum	NA	5.0	NA	4.0
1-day minimum ^d	8.0 (5.0)	4.0	5.0	3.0

^a Recommended water column concentrations to achieve the required intergravel dissolved oxygen concentrations shown in parentheses. The 3-mg/l difference is discussed in the criteria document. The figures in parentheses apply to species that have early life stages exposed directly to the water column.

^b Includes all embryonic and larval stages and all juvenile forms to 30 days following hatching.

^c NA—not applicable.

^d All minima should be considered as instantaneous concentrations to be achieved at all times. Further restrictions apply for highly manipulative discharges.

Criteria for Ammonia

Acute Toxicity Criterion (1 hr average, once in 3 yr exceedence)

$$\text{Criterion (in mg/l)} = \frac{0.52}{2 * FT * FPH}$$

where $FT = 10^{0.03(20 - TCAP)}$ for $TCAP \leq T \leq 30$
and

$$FT = 10^{0.03(20 - T)} \quad \text{for } 0 \leq T \leq TCAP$$

$TCAP = 20^\circ\text{C}$ where salmonids or other sensitive coldwater species are present and 25°C
where salmonids and other sensitive coldwater species are absent

$T =$ water temperature, $^\circ\text{C}$

Chronic Toxicity Criterion (4 day average, once in 3 yr exceedence)

$$\text{Criterion (in mg/l)} = \frac{0.8}{\text{Ratio} * FT * FPH}$$

where FT and FPH are calculated using the same formula as given above. $TCAP$ is determined as follows:

$TCAP = 15^\circ\text{C}$ where salmonids or other sensitive coldwater species are present and 20°C
where salmonids and other coldwater species are absent

$$\text{Ratio} = 16 \quad \text{for } 7.7 \leq \text{pH} \leq 9$$

and

$$\text{Ratio} = \{(24)[10^{(7.7 - \text{pH})}]\} / [1 + 10^{(7.4 - \text{pH})}] \quad \text{for } 6.5 \leq \text{pH} \leq 7.7$$

Source: From *Ambient Water Quality Criteria for Ammonia—1984*, U.S. EPA 440/5-85-001, Washington, DC.

U.S. EPA Selected Water Quality Criteria in Micrograms per Liter for Priority Pollutants

Chemical	Aquatic Life				Human Health 10^{-6} Risk		Priority Pollutant/ Carcinogenic
	Freshwater		Marine		Water and Fish Ingestion	Fish Ingestion Only	
	Acute ^a	Chronic ^a	Acute ^a	Chronic ^a			
Aldrin	3		1.3		7.4×10^{-5}	7.9×10^{-5}	Y/Y
Alkalinity		>20,000					N/N
Antimony	9000	1600			146	45,000	Y/N
Arsenic					0.0022	0.0175	Y/Y
Arsenic (penta)	850	48	2319	13			Y/Y
Arsenic (tri)	360	190	69	36			Y/Y
Asbestos					30,000 ^b		Y/Y
Barium					1000		N/N
Benzene	5300		5100	700	0.66	40	Y/Y
Cadmium	c	c	43	9.3	10		Y/N
Carbon tetrachloride	35,200		50,000		0.4	6.94	Y/Y
Chlordane	2.4	0.0043	0.09	0.004	0.00046	0.00048	Y/Y
Chlorinated benzenes	250	50	160	129			Y/Y
Chloroform	28,900	1240			0.19	15.7	Y/Y
Chlorophenoxy herbicides 2,4,5 - T					10		N/N
Chromium (hexavalent)	16	11	1100	50	50		Y/N
Chromium (trivalent)	c	c	10,300		170,000	3,433,000	N/N
Copper	c	c	2.9	2.9			Y/N
Cyanide	22	5.2	1	1	200		Y/N
DDT	1.1	0.001	0.13	0.001	0.000024	0.000024	Y/Y
DDE (DDT metabolite)	1050		14				Y/Y
Demeton		0.1		0.1			Y/N
Di-2-ethylhexylphthalate					15,000	15,000	Y/N
Dibutylphthalate					35,000	154,000	Y/N
Dichlorobenzenes	1120	763	1970		400	2600	Y/N

R0023989

Dichlorobenzidine					0.01	0.02	Y/Y
Dichloroethane 1,2	118,000	20,000	113,000		0.94	243	Y/Y
Dichloroethylene	11,600		224,000		0.33	1.85	Y/Y
Dichlophenol 2,4	2020	365			3090		N/N
Dichloropropane	23,000	5700	10,300	3040'			Y/N
Dichloropropene	6080	244	790		87	14,100	Y/N
Dieldrin	2.5	0.0019	0.71	0.0019	0.000071	0.000076	Y/Y
Diethylphthalate					350,000	1,800,000	Y/N
Dimethylphenol 2,4	2120						Y/N
Dimethylphthalate					313,000	2,900,000	Y/N
Dinitrotoluene	330	230	590	370	70	14,300	N/Y
Dioxin (2,3,7,8-TCDD)	0.01	0.00001			1.3×10^{-8}	1.3×10^{-8}	Y/Y
Diphenylhydrazine 1,2	270				0.0423	0.56	Y/N
Dinitro- <i>o</i> -cresol 2,4					13.4	765	Y/N
Dinitrotoluene 2,4					0.11	9.1	Y/N
Endosulfan	0.22	0.056	0.034	0.0087	74	159	Y/N
Endrin	0.18	0.0023	0.037	0.0023	1		Y/N
Ethylbenzene	32,000		430		1400	3280	Y/N
Fluoranthene	3980		40	16	42	54	Y/N
Guthion		0.01		0.01			N/N
Haloethers	360	122					Y/N
Halomethanes	11,000		12,000	6400	0.19	15.7	Y/Y
Heptachlor	0.52	0.0038	0.053	0.0036	0.00028	0.00029	Y/Y
Hexachlorobenzene					0.00072	0.00074	Y/N
Hexachlorobutadiene	90	9.3	32		0.45	50	Y/Y
Hexachlorocyclohexane (Lindane)	2	0.08	0.16				Y/Y

1021

R0023990

U.S. EPA Selected Water Quality Criteria in Micrograms per Liter for Priority Pollutants (continued)

Chemical	Aquatic Life				Human Health 10 ⁻⁶ Risk		Priority Pollutant/ Carcinogenic
	Freshwater		Marine		Water and Fish Ingestion	Fish Ingestion Only	
	Acute ^a	Chronic ^a	Acute ^a	Chronic ^a			
Hexachlorocyclohexane (technical)					0.012	0.041	Y/Y
Hexachlorocyclohexane (α)					0.0092	0.031	Y/Y
Hexachlorocyclohexane (β)					0.016	0.055	Y/Y
Hexachlorocyclohexane (γ)							
Hexachlorocyclopentadiene	7	5.2	7		206		Y/N
Hexachloroethanes	980	540	940		1.9	8.74	N/Y
Iron		1000			300		N/N
Isophorone	117,000		12,900		5200	5,200,000	Y/N
Lead	c	c	140	5.6	50		Y/N
Malathion		0.1		0.1			N/N
Manganese					50	100	N/N
Mercury	2.4	0.0012	2.1	0.025	0.144	0.146	Y/N
Methoxychlor		0.03		0.03	100		N/N
Mirex		0.001		0.001			N/N
Monochlorobenzene					488		Y/N
Napthalene	2300	620	2350				Y/N
Nickel	c	c	75	8.3	13.4	100	Y/N
Nitrates					10,000		N/N
Nitrobenzene	27,000		6680		19,800		Y/N
Nitrophenols	230	150	4850		13.4	7.65	Y/N
Nitrosamines	5850		3,300,000				Y/Y
Nitrosobutylamine N					0.0064	0.587	Y/Y
Nitrosodiethylamine N					0.008	1.24	Y/Y

R0023991

Nitrosodipentylamine <i>N</i>					4.9	15.1	Y/Y
Nitrosopyrrolidine <i>N</i>					0.16	91.9	Y/Y
Parathion	0.065	0.013					N/N
PCB's	2	0.014	10	0.03	0.000079	0.000079	Y/Y
Pentachlorinated Ethanes	7240	1100	390	281			N/N
Pentachlorobenzene					74	85	N/N
Pentachlorophenol	20	13	13	7.9	1010		N/N
Phenol	10,200	2560	5800		3500		Y/N
Phthalate esters	940	3	2944	3.4			Y/N
Polynuclear aromatic hydrocarbons			300		0.0028	0.031	Y/Y
Selenium	260	35	410	54	10		Y/N
Silver	c	0.12	2.3		50		Y/N
Sulfide-hydrogen sulfide		2		2			N/N
Tetrachlorinated ethanes	9320						Y/N
Tetrachlorobenzene 1,2,4,5					38	48	Y/N
Tetrachloroethane 1,1,2,2		2400	9020		0.17	10.7	Y/Y
Tetrachloroethylene	5280	840	10,200	450	0.8	8.85	Y/Y
Tetrachlorophenol 2,3,5,6				440			Y/N
Thallium	1400	40	2130		13	48	Y/N
Toluene	17,500		6300	5000	14,300	424,000	Y/N
Toxaphene	0.73	0.0002	0.21	0.0002	0.00071	0.00073	Y/Y
Trichlorinated ethanes	18,000				0.6	41.8	Y/Y
Trichloroethane 1,1,1			31,200		18,400	1,030,000	Y/N

U.S. EPA Selected Water Quality Criteria in Micrograms per Liter for Priority Pollutants (*continued*)

Chemical	Aquatic Life				Human Health 10 ⁻⁶ Risk		Priority Pollutant/ Carcinogenic
	Freshwater		Marine		Water and Fish Ingestion	Fish Ingestion Only	
	Acute ^a	Chronic ^a	Acute ^a	Chronic ^a			
Trichloroethane 1,1,2		9400			0.6	41.8	Y/Y
Trichloroethylene	45,000	21,900	2000		2.7	80.7	Y/Y
Trichlorophenol 2,4,5					2600		N/N
Trichlorophenol 2,4,6		970			1.2	3.6	Y/Y
Vinyl Chloride					2	525	Y/Y
Zinc	^c	^c	95	86			Y/N

Sources: U.S. Environmental Protection Agency (1987); *Quality Criteria for Water 1986*; and *Federal Register*.

^aPermissible exceedence: acute toxicity criteria—1-hr average concentration, not to be exceeded more than once in three years on the average; chronic toxicity criteria—4-day average concentration, not to be exceeded more than once in three years on the average.

^bExpressed in fibres/liter.

^cMetals toxicity related to hardness by the relationship $AT \text{ or } CT (\mu\text{g/l}) = \exp\{\alpha \ln[\text{hardness}(\text{mg CaCO}_3/\text{l}) + \beta]\}$

Metal	Acute Toxicity AT ^b		Chronic Toxicity CT ^b	
	α	β	α	β
Cadmium	1.128	-3.828	0.7852	-3.490
Chromium (trivalent)	0.819	+3.688	0.8190	+1.561
Copper	0.9422	-1.464	0.8545	-1.465
Lead	1.266	-1.416	1.266	-4.661
Nickel	0.8460	+3.3612	0.8460	+1.1645
Zinc	0.8473	+0.8604	0.8473	+0.7614

R0023993

Toxicity Characteristics for Contaminated Sediments in the TCLP Test

Priority Constituent	Maximum Concentration (mg/l)
Arsenic	5.0
Barium	100.0
Benzene	0.5
Cadmium	1.0
Carbon tetrachloride	0.5
Chlordane	0.03
Chlorobenzene	100.0
Chloroform	6.0
Chromium	5.0
Cresols	200.0
2,4-D	10.0
1,4-Dichlorobenzene	7.5
1,2-Dichloromethane	0.5
1,1-Dichloroethylene	0.7
2,4-Dinitrotoluene	0.1
Endrin	0.02
Heptachlor (and its epoxide)	0.008
Hexachlobenzene	0.1
Hexachloro-1,3-butadiene	0.5
Hexachloroethane	3.0
Lead	5.0
Lindane	0.4
Mercury	0.2
Methoxychlor	10.0
Methyl ethyl ketone	200.0
Nitrobenzene	2.0
Pentachlorophenol	100.0
Pyridine	5.0
Selenium	1.0
Silver	5.0
Tetrachloroethylene	0.7
Toxaphene	0.5
Trichloroethylene	0.5
2,3,5-Trichlorophenol	400.0
2,4,6-Trichlorophenol	2.0
2,4,5-TP (Silvex)	1.0
Vinyl chloride	0.2

Source: U.S. EPA, *Federal Register* (March 29, 1990).

APPENDIX C

Environmental Characteristics of Priority Pollutants

Priority Pollutant	Solubility (mg/l @°C)	Vapor Pressure (mm Hg @°C)	Henry's Constant (atm·m ³ /M [@25°C])	log <i>K</i> _{ow} (l/kg)	Biochemical Decay Coefficient	Parameters Affecting <i>K</i> _p
<i>Halogenated Aliphatic Hydrocarbons</i>						
Bromoform	3,200 @30	5.6 @25	5.32 E-4	2.30	1	V LB
Carbon tetrachloride	800 @20	113 @25	2.93 E-2	2.73	2	V LB
Chlorodibromomethane	NA	15 @10.5	7.83 E-4	2.09	0	V LB
Chloroethane	5,740 @20	2,660 @25	1.11 E-2	1.43	NA	V LB
Chloroform	9,300 @25	160 @20	3.39 E-3	1.97	2	V
Dichlorobromomethane	4,000 @25 ^a	50 @20	2.12 E-3	1.88	0	V LB
Hexachlorobutadiene	2 @20	0.15 @20	0.0256	4.78	NA	V S
Hexachlorocyclopentadien	1.8 @25	0.08 @25	0.016	3.99	2	V LS
Hexachloroethane	50 @22	0.40 @25	9.85 E-3	4.14	2	S ^b
Methyl bromide (bromomethane)	1.75 E4 @20	1,420 @20	NA	1.1	NA	V LB
Methyl chloride (chloromethane)	6,360 @20	3,800 @20	8.82 E-3	0.91	NA	V
Methylene chloride	1.67 E4 @25	429 @25	3.19 E-3	1.25	2	V LB
Tetrachloroethylene	150 @25	19 @25	2.87 E-2	2.53	1	V B
Trichloroethylene	1,100 @25	77 @25	1.17 E-2	2.53	1	V B
Vinyl chloride (chloroethene)	1.1 @25	2,580 @20	2.78 E-2	0.60	NA	V
1,1,1-Trichloroethane	4,400 @25	100 @20	4.08 E-3	2.47	1	V LB
1,1,2,2-Tetrachloroethane	2,900 @20	4 @25	3.8 E-4	2.39	0	V LB
1,1,2-Trichloroethane	4,500 @20	25 @25	7.42 E-4 ^a	2.17	1	V LB
1,1-Dichloroethane	5,500 @20	234 @25	5.45 E-3	1.79	1	V LB
1,1-Dichloroethylene	210 @25	591 @25	1.49 E-2	1.48	2	V LB
1,2-Dichloroethane	8,690 @20	79 @25	1.10 E-3	1.45	1	V LB

R0023995

1,2-Dichloropropane	2,700 @20	42 @20	2.82 E-3	2.28	1	V LB
1,2- <i>trans</i> -Dichloroethylene	600 @20	331 @25	5.32 E-3	1.48	1	V LB
1,3-Dichloropropylene	2,700 @25	43 @25	3.5 E-3	1.98	1	V LB
<i>Phenolic Compounds</i>						
<i>P</i> -Chloro- <i>m</i> -cresol	2.5 E4 @25 ^a	0.11 @20	NA	1.96	NA	^c
Pentachlorophenol	14 @20	1.1 E-4 @20	2.8 E-6	5.01	1	S B
Phenol	8 E4 @25	0.35 @25	1.3 E-6	1.46	2	LV HB
2,4,6-Trichlorophenol	800 @25	1 @76.5	4 E-6	3.69	2	B
2,4-Dichlorophenol	4,500 @25	1 @25	2.8 E-6	2.75	2	HB
2,4-Dimethylphenol	590 @25 ^a	98 @104	1.7 E-6	2.42	2	^d
2,4-Dinitrophenol	5,600 @18	1.49 E-5 @25 ^a	6.45 E-10	1.54	2	B
2-Chlorophenol	2.85 E-4 @20	40 @82	1.03 E-5	2.15	NA	B
2-Nitrophenol	2,100 @20	1 @49.3	7.56 E-6 ^a	1.79	2	B
4,6-Dinitro- <i>o</i> -Cresol	130 @20	1 E-4 @25	1.4 E-6	2.85	0	B
4-Nitrophenol	16,000 @25	2.2 @146	2.5 E-6 ^a	1.91	2	B
<i>Polycyclic Aromatic Hydrocarbons</i>						
Acenaphthene	3.42 @25	10 @131	2.41 E-4	3.92	2	LS HB
Acenaphthylene	3.93 @25	0.029 ^a	1.14 E-4	4.07	2	S HB
Anthracene	1.29 @25	1.7 E-5 ^a	8.6 E-5	4.45	1	V S HB
Benzo (a) anthracene	0.010 @24	2.2 E-8 @20 ^a	1 E-6 ^a	5.61	0.0528	LV S HB
Benzo (a) pyrene	0.0038 @25	5 E-9 @20	4.9 E-7	5.98	0	LV S B
Benzo (b) fluoranthene	0.014 @25 ^a	5 E-7 @20 ^a	1.22 E-5 ^a	6.57	NA	LV S HB
Benzo (ghi) perylene	2.6 E-4 @25 ^a	1 E-10 @20 ^a	1.44 E-7 ^a	7.23	0.1310 ^a	LV HS B
Benzo (k) fluoranthene	4.3 E-3 @25 ^a	9.59 E-11 @20	1.22 E-5 ^a	6.84	0.1111 ^a	LV HS HB
Chrysene	0.006 @25	6.3 E-9 @25	1.05 E-6	5.61	1	LV S HB
Dibenzo (a,h) anthracene	5 E-4 @25	1 E-10 @20 ^a	7.3 E-8	5.97	0.1310 ^a	LV S HB
Fluoranthene	0.265 @25	5.0 E-6 @25	6.5 E-6	5.33	2	LV S HB
Fluorene	1.9 @25	10 @146	1.17 E-4	4.18	1	S HB
Indeno (1,2,3-cd) pyrene	5.3 E-4 @25 ^a	1.1 E-10 @20 ^a	6.95 E-8	7.66	NA	LV HS B
Naphthalene	30 @25	0.082 @25	4.83 E-4	3.37	2	HB ^c
Phenanthrene	0.816 @21	1 @118.2	3.93 E-5	4.46	2	LV S HB
Pyrene	0.16 @26	2.5 @200	5.1 E-6	5.18	2	LV S B

Environmental Characteristics of Priority Pollutants (*continued*)

Priority Pollutant	Solubility (mg/l @°C)	Vapor Pressure (mm Hg @°C)	Henry's Constant (atm-m ³ /M [@25°C])	log K _{ow} (l/kg)	Biochemical Decay Coefficient	Parameters Affecting K _p
<i>Halogenated Ethers</i>						
Bis-(chloromethyl) ether	2.2 E4 @25	30 @22	2.1 E-4	-0.38	NA	—
Bis-(2-chloroethyl) ether	1.02 E4	0.71 @20	1.3 E-5	1.58	2	LB ^c
Bis-(2-chloroethoxy) methane	8.1 E-4 @25	<0.1 @20	2.8 E-7 ^a	1.26	0	f
Bis-(2-chloroisopropyl) ether	1.7 E3 ^a	0.85 @20 ^a	1.1 E-4 ^a	2.10	2	LB ^e
2-Chloroethyl vinyl ether	6,000 @25	26.75 @20	2.5 E-7 ^a	1.28	2	V ^f
4-Bromophenyl phenyl ether	4.8 @25 ^a	1.5 E-3 @20 ^a	1.0 E-4 ^a	4.28	0	LS LB ^c
4-Chlorophenyl phenyl ether	3.3 @25	0.0027 @25	2.19 E-4	4.08	0	LS LB ^e
<i>Phthalate Esters</i>						
Bis (2-ethylhexyl) phthalate	0.4 @25	1.2 @200	3 E-7	5.3	1	LV S B
Butyl benzyl phthalate	2.9	8.6 E-6 @20	8.3 E-6 ^a	4.78	2	LV B
Diethyl phthalate	896 @25 ^a	0.05 @70	1.2 E-6	2.96	2	LV B
Dimethyl phthalate	5,000 @20	<0.01 @20	2.15 E-6	1.87	2	LV B
Di-n-butyl phthalate	400 @25	0.1 @115	2.8 E-7	5.2	2	LV S B
Di-n-octyl phthalate	3 @25	<0.2 @150	1.7 E-5	9.2	1	LV HS B
<i>Monocyclic Aromatics</i>						
Benzene	1,780 @20	95 @25	5.55 E-3	2.13	2	V LB
Chlorobenzene	488 @25	11.8 @25	3.93 E-3	2.84	2	V LB
Ethylbenzene	152 @20	10 @25.9	6.44 E-3	3.15	2	V ^g
Hexachlorobenzene	0.11 @24	1 @114.4	1.70 E-3	5.47	0	LV S LB

R0023997

Nitrobenzene	1,900 @25	0.15 @20	2.38 E-5	1.85	2	LB
Toluene	515 @25	28 @25	5.92 E-3	2.69	2	V ^g
1,2,4-Trichlorobenzene	19 @22	1 @38.4	1.42 E-3	3.98	1	V LS LB
1,2-Dichlorobenzene	145 @25	1.5 @25	1.94 E-3	3.38	1	V LB
1,3-Dichlorobenzene	123 @25	1.0 @12.1	2.63 E-3	3.38	1	V LB
1,4-Dichlorobenzene	79 @25	1.8 @30	2.72 E-3	3.39	1	V LB
2,4-Dinitrotoluene	270 @22	0.0013 @59	4.5 E-6	2.01	1	LB
2,6-Dinitrotoluene	270 @22	6 @150	7.9 E-6 ^a	2.05	1	LB
<i>Pesticides</i>						
Acrolein	2.08 E-5 @20	210 @20	5.66 E-5	-0.09	2	LV B
Aldrin	0.017 @25	2.3 E-5 @20	4.96 E-4	5.11	0	V S HB
BHC-alpha	1.63 @25	0.06 @40	NA	3.81	0	B ^h
BHC-beta	0.70 @25	0.17 @40	NA	3.8	0	B ^h
BHC-delta	21.3 @25	0.02 @20	NA	4.14	0	LS B ^h
BHC-gamma	7 @20	9.4 E-6 @20	4.93 E-7	3.24	2	B ^h
Endosulfan-alpha	0.32 @22	1 E-5 @25	1.0 E-5	3.55	0	V B
Endosulfan-beta	0.33 @22	1 E-5 @25	1.9 E-5	3.62	0	V B
Chlordane	0.0156 @25 ^a	1 E-5 @25	4.79 E-5	2.78	0	V LB
Dieldrin	0.186 @20	1.8 E-7 @25	5.84 E-5	4.09	0	LS
Endosulfan sulfate	0.22	1 E-5 @25 ^a	2.6 E-5	3.66	0	B ⁱ

Environmental Characteristics of Priority Pollutants (*continued*)

Priority Pollutant	Solubility (mg/l @°C)	Vapor Pressure (mm Hg @°C)	Henry's Constant (atm·m ³ /M [@25°C])	log K _{ow} (l/kg)	Biochemical Decay Coefficient	Parameters Affecting K _p
Endrin	0.26 @25	2 E-7 @25	4 E-7	5.6	0	S B ⁱ
Endrin aldehyde	50 @25 ^a	2.0 E-7 @25 ^a	2 E-9	3.15	NA	B ⁱ
Heptachlor	0.056 @25	3 E-4 @25	1.48 E-3	4.41	0	V LS LB
Heptachlor epoxide	0.35 @25	3 E-4 @25 ^a	3.16 E-5	2.65	0	LB ^e
Isophorone	1.2 E-4	0.38 @20	5.75 E-6 ^a	1.7	2	f
<i>p,p'</i> -DDD	0.16 @24	1 E-7 @30	2.2 E-8	5.99	0	V S LB
<i>p,p'</i> -DDE	0.040 @20	6.5 E-6 @20	2.2 E-5	5.69	0	V S LB
<i>p,p'</i> -DDT	0.0031 @25	1.5 E-7 @20	3.89 E-5	6.19	0	V HS LB
Toxaphene	0.5 → 3 @25	0.2 → 0.4 @25	4.89 E-3	3.3	0.0161 ^a	B ^j
2,3,7,8-Tetrachlorodibenzo- <i>p</i> -dioxin (TCDD)	1.93 E-5	7.4 E-10 @25	2.1 E-3	6.64	NA	HS LB
<i>PCBs and Related Compounds</i>						
PCB-1016	0.049 @24	4 E-4 @25	1.35 E-2	4.38	0	V S B ^k

PCB-1221	0.59 @24	6.7 E-3 @25	2.28 E-4	4.09	2	V S B ^k
PCB-1232	1.45	4.06 E-4 @25	NA	>4.54	2	V S B ^k
PCB-1242	0.10 @24	4.06 E-4 @25	3.43 E-4	4.11	0	V S B ^k
PCB-1248	0.054	4.94 E-4 @25	4.4 E-4	5.75	0	V S B ^k
PCB-1254	0.057 @24	7.71 E-5 @25	8.37 E-3	6.03	0	V HS B ^k
PCB-1260	0.080 @24	4.05 E-5 @25	3.36 E-4	>6.11	0	V HS B ^k
2-Chloronaphthalene	6.74 @25	0.017 @20	5.4 E-4 ^a	4.12	2	V S B ^k

Metals and Cyanide

Antimony	IN	1 @886	NA	NA		V S B ^l
Arsenic	IN	1 @372	NA	NA		V S B ^m
Beryllium	IN	NA	NA	NA		LV
Cadmium	IN	1 @393	NA	NA		S
Chromium	IN	1 @1616	NA	NA		LS
Copper	IN	1 @1628	NA	NA		S LB ⁿ
Lead	IN	1 @970	NA	NA		S LB ^o
Mercury	0.056 @25	1.2 E-3 @20	1.14 E-2	NA		V S B ^p
Nickel	IN	1 @1800	NA	NA		HS ^q
Selenium	NA	NA	NA	NA		V S B ^r

Environmental Characteristics of Priority Pollutants (continued)

Priority Pollutant	Solubility (mg/l @°C)	Vapor Pressure (mm Hg @°C)	Henry's Constant (atm-m ³ /M [@25°C])	log K _{ow} (l/kg)	Biochemical Decay Coefficient	Parameters Affecting K _p
Silver	IN	1 @1310	NA	NA		S ^s
Thallium	IN	1 @825	NA	NA		S ^r
Zinc	IN	1 @487	NA	NA		S ^u
<i>Miscellaneous</i>						
Acrylonitrile	7.35 E4 @20	137 @30	8.8 E-5	-0.92	2	V
Asbestos	NA	NA	NA	NA		S
Benzidene	400 @12	5 E-4 ^a	3 E-7 ^a	1.81	NA	6 ^f
Cyanide	NA	NA	NA	NA		HV LS HB ^v
N-Nitrosodi-n-propylamine	9900 @25	NA	NA	1.31	0	LB
N-Nitrosodimethylamine	MISCIBLE	8 @25	NA	-0.47	0.0451 ^a	LB
N-Nitrosodidiphenylamine	35	0.1	3.13	2.79	2	B
1,2-Diphenylhydrazine	221	2.6 E-5 @25 ^a	3.4 E-9	3.03	1	f
3,3'-Dichlorobenzidene	3.1 @25	1 E-5 @22 ^a	8 E-7 ^a	3.51	NA	u

Key to Rate Summary: 0 = No significant degradation rate. 1 = 0.05 day⁻¹ < k_B < 0.5 day⁻¹; use 0.05 day⁻¹. 2 = k_B > 0.5 day⁻¹; use 0.5 day⁻¹

Notes: Unless noted, data (except K_p data) obtained from U.S. EPA Risk Reduction Engineering Laboratory (RREL), Cincinnati, Ohio, Treatability Database. K_p data obtained from Michael A. Callahan et al., *Water-related Environmental Fate of 129 Priority Pollutants*. EPA-440/4-79-029, 1980. NA = Data not available; IN = Insoluble; LV = little/somewhat volatile; V = moderately volatile; HV = highly volatile; LS = little/somewhat sorbed; S = moderately sorbed; HS = highly sorbed; LB = little/slowly biodegradable; B = moderately biodegradable; HB = highly biodegradable.

^aData obtained from U.S. EPA, *Processes, Coefficients, and Models for Simulating Toxic Organics and Heavy Metals in Surface Waters*. EPA/600/3-87/015, June 1987.

^bNo specific data on V nor B found in the literature.

^cBiodegradation demonstrated in soil samples and acclimated sewage sludge, but uncertain in ambient surface waters.

^dB = information inconclusive.

^eV = unknown.

^fB = unknown.

^gS and B = importance not determined.

^hV = unknown—information contradictory; B = varies with environment.

ⁱV and S = no information.

^jB = only in anaerobic sediments.

^kV = depressed by presence of organic solids; S = strongly adsorbed by solids, especially with high organic content; B = only important for those with fewer than 4 chlorines per molecule.

^lBiomethylation may occur.

^mMetabolized by a number of organisms to organic arsenicals having increased mobility.

ⁿSome copper complexes may be metabolized. Organic ligands are important in sorption and complexation processes.

^oBiomethylation in sediments can remobilize lead.

^pSorption strongest onto organic materials. Can be metabolized by bacteria to methyl and dimethyl forms that are quite mobile.

^qMost mobile of heavy metals.

^rVolatilization via biomethylation or formation of H_2Se . Hydrous metals strongly sorb Se. Metabolism may result in methylation with subsequent volatilization.

^sStrongly sorbed by hydrous Mn and Fe oxides, clay minerals, and organics.

^tAdsorbed to clay minerals and hydrous metal oxides.

^uStrong affinity for hydrous metal oxides, clays, and organic matter. Sorption increases with pH.

^vAt pH ≤ 10 , most free cyanide is HCN, which is quite volatile.

Index

- accumulation function, 464–466
- acid and methane fermentation in
sediments, 746
- acid-base reactions, 408
- acidic surface waters, 204–208
- acidification of soil, 313
- acidity, 41, 802–806
 - rain, natural, 202
 - of urban precipitation, 480–481
- acid mine drainage, 40, 208, 803
- acid rain, 102, 185, 202, 578, 803–804
 - areas sensitive to acidic precipitation, 806
 - biological impact on receiving water bodies, 804
 - buffering in urban areas, 481
 - effect on drinking water, 211–215
 - natural, 202
 - sulfuric acid formation, 210
 - sulphate loading of lakes, 805
- acid volatile sulfides (AVS), 357–358, 360, 889
- adsorbing compounds, for metals in soils and sediments, 355
- adsorption-desorption reactions, 410
- adsorption
 - of pollutants
 - incorporation in models, 530
 - in soils, 342–345, 352–357
 - in wetlands, 882
 - of viruses by soils, 378
- advection, in groundwater zones, 398–399
- aeration for water quality improvement, 947–948
 - in-lake aeration, 957–962
- aerobic decomposition, 745–746, 878
- aerobic processes, 745
- aggradation, 243
 - floodplain, 293
- agricultural and silvicultural pollution, 574–690
 - sources, 681–689
- agricultural chemicals, 10
- agricultural conservation programs, 678
- agricultural diffuse pollution models, 540–546
- agricultural land use, 679
- Agricultural Nonpoint Source Pollution Model (AGNPS), 540–541, 546
- agricultural pollution, 6–7, 9, 29–34
- agricultural production, 676
- Agricultural Research Service (ARS), 541
- Agricultural Runoff Model (ARM), 536, 541
- agriculture
 - average farm size, 676
 - commodity programs, 677–679
 - extent, 674–680
 - intensive, 29
 - management, 674–690, 712–715
 - private land use, 77, 679–680, 725
 - sustainable, 3, 680
- air pollution sources, 188
- air turbulence, 463–465
- aldrin, 6
 - in soils, 340
 - in urban runoff, 480
- algal biomass, 781, 784
- algal bloom, 53, 787
- algal respiration, 747, 766

- algicides, 340
- alkalinity, 805
- aliphatic hydrocarbons, 344, 390
- alum, addition to lakes, 955-956
- aluminum toxicity, due to acidic inputs, 805
- American Public Works Association, 447
- ammonium, 760, 778
 - adsorption, 369-371, 762
 - ammonification, 362, 368
 - fixation, 364, 762
 - release from sediments, 295
 - unionized, 41
 - volatilization, 209, 364, 371, 919
- animal waste management, 9, 48, 703-705
- anoxic (anaerobic) decomposition, 746, 878
- antecedent soil moisture conditions. *See* SCS Runoff Curve model
- anthracene, 642
- antidegradation regulation and principle, 989
- apatites, 334
- appropriation water rights, 83
- aquaculture, 683
- aquatic ecosystem, 740-764
 - biotic components, 741-742
 - integrity of, 749-759
- aquatic habitat, 4-5, 12
 - destruction, 5, 927-928
 - erosion effects on, 250
 - management, 963-965
 - protection, 90
 - restoration, 927-929, 940-946
- aquatic life protection criteria, 839-843
- aquatic plants
 - management, 965-970
 - systems, 917-921
- aquatic weed control, 969
- aquifer, 172, 177, 413
 - artesian (confined), 174
 - composition, 392-394
 - mining, 178
 - recharge
 - by infiltration from basins and trenches, 600
 - by land application of wastewater, 423
 - by using pervious pavement, 590
 - by wetlands, 175, 871
 - hydrological, 174-177
- representation, 173
- safe yield, 178
- Areal, Nonpoint Source Watershed Environmental Response Simulation (ANSWERS) model, 540, 546
- Areawide Stormwater Model (ABMAC), 532, 546
- arid lands, 26, 32
 - water quality, 808
- ARMA models, 166-170, 546-550, 655
- AROCHLOR. *See* polychlorinated biphenyls
- aromatic hydrocarbons, 345
- arsenic
 - arsenides-mineral sources, 412
 - in natural base flow, 407
 - in soils, 359
 - in surficial materials, 319
 - in urban runoff, 42, 477-479
 - source minerals, 412
- artificial wetlands, 866, 889-917. *See also* constructed wetlands
- asbestos, 41
 - in urban runoff, 477-479
 - traffic emission, 456
- Aswan high dam, 14
- atmosphere-surface water body integrations, 198-202
- atmospheric contamination, by chemicals from soils, 341
- atmospheric deposition, 7, 10, 444, 451-454, 469, 808
 - control, 578
 - dry, 28-29, 453, 578
 - mercury, 220
 - monitoring, 559
 - pesticides, 227-231
 - sources, 187
 - wet, 28, 453, 578
- atmospheric emissions, 28, 189, 195
- atmospheric nitrogen, impact on surface waters, 216
- atmospheric particulates, deposition rate of, 452
- atmospheric pollution, by toxic compounds, 217-231
- atmospheric reaeration, 766, 768
 - coefficient, 768-769
- atrazine, 230, 340, 684
- attainable water use, 986
- automatic samplers, 561
- availability factor for washoff of pollutants, 472
- background (natural) water quality, 13-14, 32, 806-810

- bank armoring techniques, 940
- base flow
 - hydrologic definition, 103–105, 172
 - of urban streams 441
 - quality (natural), 406–416
- bedload, 245, 246, 294
- benthic macroinvertebrates, 758
- benthos, 744, 748
- benzene, in urban runoff, 478–479
- benzoperylene, in urban runoff, 480
- Best Available Technology
 - Economically Achievable (BATEA), 81, 90
- Best Management Practices
 - agricultural, 689–711
 - definition, 18
 - efficiency of agricultural BMPs, 697–711
 - hydrological, 109
 - reliability and longevity, 575
 - selection for agriculture, 692–693
 - urban, 573–575
- bioaccumulation, 4, 16, 325, 745, 831–838
- bioaccumulation factor, 832–835
 - lipid normalized, 835–836
- bioavailability of toxic compounds, 325, 828–829
- biochemical oxygen demand (BOD), 21
 - in CSOs, 499
 - in dissolved balance of surface waters, 765–768
 - in feedlot runoff, 688
 - in landfill leachate, 430
 - in septic tank effluents, 418
 - in solids from urban areas, 448
 - NURP studies, 492–493
 - removal in ponds, 627
 - removal in wetlands, 905–906
 - traffic emission, 456
- bioengineering, 939–943
- biological criteria, 759
- biological degradation, 330–332
- biological treatment, 641
- biomagnification, 745, 831–838
- biomagnification factor, 832–835
 - food chain multiplier, 837
 - mass balance model, 832–834
 - predator–prey concentration ratio, 836
- biomanipulation, 972
- biotic indices, 753–759
 - index of biotic integrity, 754–758
 - invertebrate community index, 758
 - rapid bioassessment protocol, 759
 - saprobien index, 753–754
- biotic integrity of water bodies,
 - measures, 753–759, 818
- biotransformation, 330
- black list of toxic pollutants, 819
- buffering, of acid rain in urban areas, 481
- buffer strips, 52, 94, 602
- buffer zones, 601–602
- buildup and washoff, 463–469, 470–473, 519–520, 522, 536–537
- Bureau of Land Management, 679
- cadmium
 - in natural base flow, 407
 - in precipitation, 453–454
 - in soils, 359
 - in solids from urban areas, 448
 - in surficial materials, 319
 - in urban runoff, 42, 477–479
 - source minerals, 412
- Canaan Valley, 974
- capillary soil water zone, 172
- carbonate rocks, 394
- carbon/nitrogen ratio, effect on nitrogen
 - in soils, 369
- catch basins, 602, 612–613
- cesium-137 use for soil loss estimates, 284–286
- channel erosion, 460
 - control, 603–604
- channel stabilization, 603–611
- Chebotarev, Ignaovich, Souline
 - sequence of groundwater quality, 411
- check dams, 607
- chemical-based numerical criteria, 840–843
- chemical oxygen demand (COD), 21
 - in CSOs, 499
 - in feedlot runoff, 688
 - in landfill leachate, 430
 - in septic tank effluents, 418
 - in solids from urban areas, 448
 - NURP studies, 492–493
 - removal in ponds, 627
 - surface waters impairment, 736
 - traffic emission, 456
- chemical time bomb, 313, 803
- chemicals application control, 588
- Chemicals, Runoff and Erosion from
 - Agricultural Management Systems (CREAMS) model, 268–269, 540–541, 546
- Chesapeake Bay, 46–48, 783, 886
 - pesticide input from atmosphere, 230

- Chicago River, diversion of, 947
- chlordan, 6
in urban runoff, 478–480
- chlorinated hydrocarbons, from land
wastewater disposal systems, 422
- chloroform, in urban runoff, 478–479
- chlorophyll-a
in lakes, 784–790
relation to BOD, 783
- chromium
in natural base flow, 407
in precipitation, 453–454
in soils, 360
in surficial materials, 319
in urban runoff, 42, 477–479
source minerals, 412
surface waters impairment, 736
traffic emission, 456
- chrysene, in urban runoff, 478–479
- clay, 395, 396
dispersion index, 290
clear-cutting, 686–687
- coastal wetlands, 864–865. *See also* tidal
marshes
- Coastal Zone Management Act, 91
- coefficients
atmospheric reaeration, 768–769
deoxygenation, 768
dispersion, 403
Manning's, 154–155
partitioning, 339, 356–357, 359, 361,
404, 829–830
octanol, 326, 342–345, 404–405,
642–647, 830, 835, 850, 912
permeability (hydraulic conductivity),
396
removal, for street pollutants, 463–
464
runoff, 440–441
for Rational Formula, 146–148
tidal dispersion, 776–777
urban washoff, 472
variation, 487
- collection system control, 602–616
- coliforms, 8
in CSOs, 36, 476, 499
in soils, 376
in treatment plant effluent, 36
in urban runoff, 36, 476
surface water impairment, 736
- combined sewer overflows (CSO), 3,
37–40, 50, 440, 495–500
bottlenecks in CSO control, 650
characterization of quality, 36, 496–
500
chemical oxygen demand in, 499
coliforms in, 36, 476, 499
control of pollution, 89–90, 577–602,
612–613, 627–635
control strategy, 651–654
critical rainfall intensity for initiation
of, 39, 498–499
dry period effects on loads, 482
first flush in, 499, 629
frequency and duration, 496–498
microorganisms in, 476, 497–499
nitrogen and phosphorus in, 36, 499
quality control basins, sizing, 618–619
sewer flushing, 613
storage facilities, 618
suspended solids, 36
- combined sewers, 7, 496
- Commerce Clause of U.S. Constitution,
77, 84, 88
- Commodity Credit Corporation, 677
- complexation reactions, 410
- computer aided design (CAD), 512, 526
- confined animal operations, 29, 34, 683
pollutant loads, 688
management, 703–705
- Conservation Reserve programs, 93
- conservation tillage, 9, 94, 691–697
- constructed wetlands, 866, 889–917
basic flow patterns, 902
components, 901
configuration, 901
design parameters, 894–909
loading, 901–902
pollutant removals, 902–907
problems, 917
retention, time, 899–900
subsurface flow, 891–893
types, 891–894
- construction site erosion, 29, 248, 483
USLE estimates, 262, 264
- consumers (organisms), 745
- continuous models, 513, 517, 532–536,
562
- copper
from NURP studies, 492–493
in landfill leachate, 427
in natural base flow, 407
in precipitation, 453
in soils, 360
in surficial materials, 319
in urban runoff, 42, 477–489
source minerals, 412
traffic emission, 456
- coprecipitation, 328
- corrosive waters, 212

- cost sharing, 728
- cribwall, for stream bank stabilization, 942
- criteria and standards, 14–16, 839–848
 - acceptable risk level for human health protection, 845
 - acute toxicity, 823
 - based on equity imperative, 80–81
 - biological, 759
 - chronic toxicity, 823
 - effluent, 96, 81
 - groundwater quality, 391–392
 - human health protection, 843–848
 - performance, 96
 - permissible exceedence, 841
 - sediment, 848–852
 - site specific, 759
 - whole effluent toxicity, 843
- cumulative probability distribution, 485–486, 489
- curb-side pollutant loads, 461–470
- curb
 - height, effect on accumulation of street solids, 467–468
 - length density, relation to imperviousness, 446
 - loading of pollutants 448
- cyanides, 41
 - in street salt, 460
 - in urban runoff, 42, 477
- Darcy's law, 395
- data sources and retrieval, 557–558
- DDT
 - in atmospheric deposition, 229
 - in soil environment, 340
 - in urban runoff, 480
 - levels in organisms, 831
 - residence in atmosphere, 232
- deaeration, 743
- deamination, 760
- decomposers, 745
- deep well injection of wastewater, 388
- deforestation, 10–11, 23, 26–28, 246–247
- degradation, 243–244, 293
- deicing
 - chemicals, 459, 469
 - operations, 126, 459, 469
 - salts, 588
 - impact on groundwater resources, 388–389
- delivery ratio, 239, 270–271, 521, 544, 577, 603
 - for lead, 361
 - in streams, 293–294
 - overland flow effects, 274–275
- denitrification, 367–368, 421, 542, 760–764, 871, 882
- denudation, 238
- deoxygenation coefficient, 768
- deposits (geological), 392–394
- designated use, 16, 985–987
- design storm, 129–131
- detention–retention facilities, 616–637
 - ponds, 621
- deterministic models, 509–547
- diazinon, in urban runoff, 480
- di-benzo-thiopene, 642
- dieldrin, 340, 478–480
- diffuse pollution, 18
 - abatement
 - strategy of, 1007–1008
 - targeting, 996–998, 1002
 - programs
 - financing, 993–1002
 - voluntary, 7, 48, 726
 - urban
 - magnitude of, 443–445
 - winters, 459–460
- diffuse sources, definition, 21
- discharge permits, marketable or transferable, 75, 999, 1001
- disinfection, 641
- dispersion
 - coefficient of, 403
 - of groundwater contaminants, 398–403
- dissolution–precipitation equilibria, in groundwater, 409
- dissolved oxygen
 - balance of estuaries, 776–777
 - balance of streams, 747, 765–776
 - dystrophic conditions, 947
 - effect of nutrients, 777
 - in Milwaukee River, 51
 - fluctuations in eutrophic water bodies, 785
 - problem, 764–783
 - standard, 764–765
 - Streeter–Phelps models, 766–769, 802
 - WASP model, 802
- drainage, 54
 - and irrigation districts, 1004
 - natural (swale), 35
 - systems, 576, 577
- dry weather flow, 3, 166
- dry (french) wells, 593, 601
- duckweed, 868–869, 919–920

- dustfall, 454, 469
dystrophic conditions, 947
- Earth Day, 9
- ecological balance, 23
ecological engineering, 868
ecological system, 22–23
ecologic cycle, 1013
economic externality, 995
economic incentives and instruments,
75, 78, 724, 993–1002, 1014
 benefits-received approach, 997–1000
 polluter pays principle, 994–996, 1004
ecoregions, 751–753, 809, 987
ecosystem, 740–741
 management, 928–929
effluent-limited water bodies, 441, 1000
Emscher River, 60–62, 933
enrichment ratio, 246, 286–292
 for urban snowmelt, 474
 pesticides, 349
 soil organic matter, 322
 soil phosphorus in runoff, 338
enrichment to delivery relation, 289,
290, 291, 292
environmental abatement objectives, 74
environmental corridors, 601–602. *See*
 also buffer zones
environmental degradation, 11
environmental laws, 72, 85–96, 725
 judicial, 96
 state, 95
 statutory, 86–96
environmental policies, 74–80, 94
 compromises, 76
 imperatives, 77–80
environmental political parties, 76
ephemeral gullies, 545
ephemeral streams, 104, 175, 387, 441
equation of continuity, 796
equation of motion, 797
equilibrium curb loads, 466
equity imperative, 78, 80–81
erodibility, 239
erosion, 237–244
 agricultural, 248
 construction site erosion, 262
 control, 8, 94, 686
 critical shear stress, 299–300
 definition, 238–244
 deforestation, 26
 effect of land use on, 248
 effect of vegetation, 243
 gully, 241
 highway, 248
 in urban areas, 442, 444, 460
 land use impact, 247
 of cohesive sediments, 299–304
 of topsoil, 684
 rill and interrill, 240
 silvicultural, 248
 streambank, 241, 249
 strip mines, 249
 topographical factors, 253
 upland, 153
 urban, 248
 wind, 243
erosion blankets, 701
estuaries, longitudinal mixing and
 dispersion in, 776
ethers, 643
euphotic layer, 743–744
eutrophication, 30, 53, 361, 741
 lake and reservoir, 783–796
 modeling, 798–802
 symptoms of, 785–786
evaporation and evapotranspiration,
120–126
 pan, 121
 potential, 124
Evaporation Index method, 124–125
event mean concentration (EMC), 488,
855
 mean values for pollutant load
 estimates, 493
 of urban runoff, 466, 477, 484–485
event-oriented models, 513, 517, 531–
536, 562
expected value, definition, 487
expert systems, 535
externality, definition, 73
- fascines, for stream bank stabilization,
941
fecal coliform die-off in soils, 376–377
fecal coliforms/fecal streptococci ratio,
477
feedlots. *See* confined animal operations
fertilizers, 32
 application
 impact on groundwater, 388
 in urban areas, 442
 organic, 322
 rates, 312, 361–362
 management, 9
Fick's law, 330
field capacity soil moisture, 111
filters, 639, 710
filter strips, 589, 593–598, 603, 706–707
 critical distance, 593–595
 designs, 594–598
 model of trap efficiency, 595–598

- use for pretreatment, 600
- vegetation, 589, 593–598
- first flush
 - control, 629, 642
 - effect, 482, 484
 - in urban snowmelt, 474
 - in CSOs, 499, 629
 - retention, 632–635
 - sediment flow, 283
- fish community assessment method, 755.
 - See also* biotic indices
- fish consumption, 844
- fish kills, 58, 764, 803, 818, 958
- fish ponds, 55
- flexible membrane diffusers, for oxygen injection, 961
- flood control, 144, 869
- flooding potential, 441
- Florida Water Management Districts, 1005–1006
- flow
 - components, 102
 - dividers, 496–497, 629
 - for waste assimilative capacity determination, 855
 - measurement and devices, 559–560
 - sediment-carrying capacity of, 276
- flow-weighted composite sample, 484, 560–562
- fluidsep (vortex) separator, 614
- fluoranthene, in urban runoff, 480
- fluorocarbons, 10
- food chain multiplier, 837
- Food, Agriculture, Conservation and Trade Act, 678, 1008
- Food Security Act, 93
- food web, 744
- forest land, 683
- forests, water quality, 808
- frequency and probability distribution functions, 485–486
- Freundlich adsorption isotherm, 327, 370
- fugitive dust emission, 453
- fugitive particulate losses from streets, 464
- functional dose (concentration), response in toxicity tests, 822, 827–828
- fungicides, 340
- gabions, 611, 701, 789
- galena, 412. *See also* lead minerals
- Gaussian normal distribution, 486
- Gelhar–Wilson groundwater model, 433–435
- geographical information systems (GIS), 507, 512, 526–528
- grab samples, 560
- grass carp, 967
- grassed waterways, 603–611
 - design velocities, 607
 - Manning's roughness factor, 609
 - vegetative cover, 608
- grass strips. *See* filter strips
- gravity soil water storage, 111
- gray list of toxic pollutants, 819
- Great Lakes, 4, 8
 - pesticide inputs, 230
 - sources of PCBs, 352
 - studies, 6
 - Water Quality Agreement, 7
- Green–Ampt infiltration equation, 542
- Green function, in stochastic models, 549
- greenhouse effect, 10, 26
- groundwater
 - cleanup, 390–391
 - contamination, 32, 388–390, 392
 - control of landfill leachate, 431–432
 - discharge, 387, 394, 406
 - dispersion, 398–403
 - dissolution–precipitation equilibria in, 409
 - flow, retardation of pollutants in, 404–405
 - hardness, 409
 - hydrological balance, 177–178
 - mining of, 103
 - models, 179, 432–435
 - movement, 171, 392–406, 433
 - natural, 406–416
 - water quality, processes, 408–411
 - pollution, 104, 312
 - by soil nitrogen, 362
 - by land wastewater disposal systems, 422
 - by leaching of chemicals from soils, 341
 - by solid-waste disposal sites, 429–432
 - sources, 389, 415–432
 - quality
 - models, 432–435
 - natural (background), sources of, 411–415
 - recharge, 387, 394
 - retardation of pollutants, 404–405
 - safe yield, 387, 394
 - soil water zones, 172–173
 - systems, 171–179
 - zones, fate of chemicals in, 392

- Groundwater Loading Effects of
Agricultural Management
Systems (GLEAMS) model, 542,
546
- halogenated aliphatics
 in groundwater, 419
 in urban runoff, 478–479
- hazardous lands, 522
- hazardous waste, disposal sites, 430
- helical bend regulator/concentrator, 615
- Henry's constant, 329
- heptachlor, 340
- herbicides, for weed control of
 impoundments, 969
- highway drainage model, 535, 539
- human cancer criteria, urban runoff
 violations of, 480
- human health protection criteria, 843–
 848
 acceptable risk, 845
- humic acids, 407, 410
- hydraulic conductivity, 109–110, 396–
 397
 coefficient, 396
 of frozen soils, 120
 of pervious pavements, 590
- hydrogen sulfide, 887
- hydrograph, 163–165, 284
- hydrologic activity, 35
- hydrologically active areas, 142, 525
- hydrological models, 165, 527–546
 agricultural (rural) diffuse pollution,
 540–546
 calibration and verification, 142, 535,
 554–556
 classification, 510
 components, 529
 continuous, 513
 deterministic, 509–546
 distributed parameter, 509–512, 540
 erosion and soil loss, 252–270
 event oriented, 513, 517
 lumped parameter, 508–512, 540–
 541
 reliability and usefulness, 513–516
 SCS runoff curve model, 521
 selection, 516–519
 single reservoir, 513
 use of models, 515–516
 watershed models, 526
- hydrological procedures, 507
- hydrologic cycle, 101–103
- Hydrologic Simulation Program—
 FORTRAN (HSP-F) model,
 536–539, 546
- hydromulching, 586, 701
- hydroseeding, 586–587
- hypolimnetic aeration, 960–961
- hypolimnetic oxygen depletion, 791,
 958
- Illinois Urban Drainage Area Simulator
 (ILLUDAS), 533
- impaired water bodies, 1002–1003
- impervious area, directly connected,
 141–142, 464, 592–593
- imperviousness, 440
- impervious urban surfaces, pollutant
 loads, 519–521
- impoundments
 harvesting of weeds in, 967–968
 water-level drawdown, of, 966
- Indian water rights, 85
- infiltration, 109–121, 589, 598–602
 control, 119
 estimating, vegetation factor for,
 114
 Green-Ampt equation, 118
 Holtan formula, 113–115
 Horton formula, 113
 into frozen soils, 119–120
 of rooftop runoff, 601
 on site, 63
 Philip's model, 116–118
 problems with appropriation water
 rights, 85
 rates, 110–120
 wetlands, 897
- infiltration basins, 593, 600
- infiltration devices, 598–602
 design parameters, 601
 into sewers, 632
 maintenance, 601
 rapid infiltration systems, 421–422
 vegetation factor, 114
- infiltration trench, 599–600
- inland wetlands, 865–869
- institutions, 9, 982–983, 1002–1008
- in-stream aeration, 947–948
- in-stream water quality enhancement,
 983
- integrated aquatic restoration, 972–974
- integrated pest management, in
 agriculture, 697
- Integrated Risk Information System
 (IRIS), 848
- integrated water quality management,
 16, 79, 750, 807, 982, 989, 1004
- intensity–duration models for rainfall,
 130–131
- interbasin water transfer, 83

- interflow, 171, 317
- invertebrates, 748
- ion-exchange reactions, 410
- irreversible impact doctrine, 78
- irrigated cropland, 9, 34, 424-425, 682
- irrigation return flow, 34, 53, 85, 682, 684, 709, 1015
 - impact on groundwater resources, 388-389, 424-429
- irrigation water management, 709
- isopluvial map, 130

- karst, 394
- kinematic wave model, 152, 156, 536

- lacustrine wetlands, 866
- Lagoon of Venice, 52-57, 861
- lagoons, 641, 702-705
- Lake Balaton, 57-59, 784
- Lake Erie, 4
- lake (impoundment) restoration, 949-972
 - addition of chemicals, 954-955
 - aeration, 956-962
 - artificial circulation, 959-960
 - control of toxic contamination, 954-955
 - deep water aeration, 954
 - destratification, 959-961
 - dilution and flushing, 956
 - exotic plant and animal control, 952
 - in-lake treatment, 952
 - liming acidic lakes, 962-963
 - sediment removal and control, 956
 - spawning habitat management, 963-965
 - techniques, 951-954
 - water level management, 870
 - watershed management, 952-953
 - water withdrawal from hypolimnion, 956
- lakes
 - chemical treatment of, 955-956
 - injection of liquid oxygen into, 961
 - with low dissolved oxygen, 957
- land application
 - of sludge, 423
 - of water and wastewater
 - impact on groundwater, 419-424
 - problems and restrictions, 422-423
- land cover factor (*C*), 586
- land drainage, 6
- landfills, 429-432
 - impact on groundwater resources, 388-389, 429-432
 - leachate management, 430, 431-432
- land use, 25-41
 - effects
 - on erosion and sediment loads, 248-249
 - on event mean concentrations, 491
 - on urban diffuse loads, 439-451
- Langmuir adsorption isotherm, 327, 335, 339
- leaching
 - requirement and ratio, 427-428
 - septic tanks, 418, 444
- lead
 - emissions, global, 218
 - from traffic, 218, 456
 - health effects, 220
 - in atmospheric deposition, 217-220
 - in CSOs, 36, 499
 - in groundwater, 213, 407
 - in landfill leachate, 427
 - in natural base flow, 407
 - in NURP studies, 491-493
 - in potable water, 213
 - in precipitation, 453-454
 - in soils, 360
 - in solids from urban areas, 448
 - in surficial materials, 319
 - in urban runoff, 36-37, 42, 477-479
 - in treatment plant effluents, 36
 - surface waters impairment, 736
- lead minerals, 412
- legal doctrines, 77-80, 96
- ligands, 326, 353-357, 645, 880
- limestone, 394-396
 - addition to lakes, 962
 - addition to watersheds, 963
- liming acidic lakes, 962-963
- lindane, 41
 - in soils, 340
 - in urban runoff, 478-480
- linear isotherm, 328
- linear particle accumulation process, 463, 469
- linear particle removal concept, 471-472
- linear watershed models, 150
- link-node representation, for water quality models, 798-799
- litter control programs, 578-579
- loading function and models, 317-319, 373, 508, 522-524
- log-drop structures, for stream restoration, 945
- log-normal probability distribution, 484-487, 491, 563
- Love Canal incident, 389
- low-flow augmentation, 946

- malaria control, 5
- malathion, in urban runoff, 480
- mangrove wetland, 865
- manhole, modification for increased infiltration, 599
- Manning's formula, 278, 608–609
- Manning's coefficient, 154–155
 - for grassed waterways, 609
 - for ripraps, 612
- median, definition, 487
- melting point, 129
 - of saline snowmelt, 473
- mercury
 - agriculture, 221
 - atmospheric deposition, 220
 - environmental effects, 226
 - exposure limits, 223–225
 - human exposure, 221–226
 - in drinking water, 224
 - in soils, 361
 - intake limits, 224
 - lowest observed effects on humans, 226
 - methyl mercury, 223
 - occupational exposure, 224
 - pathways in human uptake, 222
 - poisoning, Minamata, 76, 358
 - residence time, 221
 - soil contamination, 227
 - sources, 220
 - toxic effects, 226
- Merrimack River, 973
- metals. *See also* toxic metals
 - activity in soils and sediments, 352–353
 - chemical equilibria in soils and sediments, 356
 - complexation, 353
 - coprecipitation, 353
 - interaction between solid and dissolved species, 354
 - methylation, 358–359, 361
- methane evolution, from sediments, 736, 748
 - effect on BOD, 738
- methanogenesis, in sediments, 736
- methylene chloride, in urban runoff, 478–479
- Michaelis–Menton equation, 331, 779
- microorganisms. *See also* pathogenic microorganisms
 - autotrophic, 330
 - chemotropic, 364
 - from agriculture, 684
 - in CSOs, 476, 497–499
 - in nitrogen cycle, 362–364, 366–367
 - in septic tank effluent, 418
 - in soils, 374–378
 - in street refuse, 448
 - in urban runoff 476
- Milwaukee River, 50–52, 974, 982
- Minamata mercury poisoning, 76, 358
- mineral loads, in base flow, 413
- minerals, dissolution of, 410
- mining nonpoint pollution, 40
- mixing layer, 323
- Model Enhanced Unit Loads (MEUL), 522–523
- modeling, black box concepts, 50
- Model of Urban Runoff and Sewer Flow (MOURSEF), 532–533, 539
- models, 507–546. *See also* hydrological models
 - accuracy and reliability of, 513–516
 - agricultural, 540–546
 - application of, 515–516
 - calibration and verification, 535, 554–556
 - continuous, 513, 517, 532–536, 562
 - conventional pollutants and nutrients, 796–802
 - deterministic, 509–547
 - EPA computerized loading procedures, 321
 - event-oriented, 513, 517, 531–536, 562
 - groundwater, 432–435
 - highway drainage, 535, 539
 - infiltration, 113–118
 - kinematic wave, 152, 156, 536
 - linear watershed, 150
 - neural network, 550, 656
 - rainfall, intensity–duration, 130–131
 - regional regression, 524–525
 - screening, 522–528
 - selection of, 516–519
 - statistical routines and screening, 509, 517–528
 - steady state, 796
 - stochastic, 166, 540–550
 - surface and groundwater, 507–546
 - toxic chemicals, 852–857, 910–914
 - toxicity, 827
 - types, 508–513
 - urban runoff, 471, 531–540
 - urban washoff, 472
 - water quality, 766
 - link-node representation, 798–799
- models, other
 - Areal Nonpoint Source Watershed Environmental Response Simulation, 540, 546

- Areawide Stormwater, 532, 546
- ARMA, 166–170, 546–550, 655
- Chemicals, Runoff and Erosion from Agricultural Management Systems, 268–269, 540–541, 546
- Gelhar–Wilson groundwater, 433–435
- Model of Urban Runoff and Sewer Flow, 532–533, 539
- Monte Carlo, 547, 855–856
- Negev erosion, 269
- Nonpoint Simulator, 536
- Pesticide Root Zone, 543, 546
- Pesticide Runoff Simulator, 542
- Real Time Control, of sewerage, 655–656
- SCS Runoff Curve, 134–141, 158–163, 521, 532, 541–543
- Simulator for Water Resources in Rural Basins (SWRRB), 542, 546
- Stanford Watershed, 107, 127, 536
- Storage-Treatment-Overflow Runoff, 532, 539
- STORM, 463
- Stormwater Management (SWMM), 463, 533–534, 539
- Streeter–Phelps, 766–769, 802
- TVA base flow quality, 413–417
- TVA-HYSIM, 532
- Unit Hydrograph, 148–156
- USGS Distribute Routing Rainfall Runoff, 535
- Vollenweider's for lakes, 789–793
- WASP, 798–802, 854–855
- WASSP-QUAL sewer flow, 533, 538–539
- monitoring and data acquisition, 556–564
 - flow, 556–560
 - quality, 560–562
- monitoring station, 558–561
- Monte Carlo models, 547, 855–856
- mosquito control, 5, 862, 897, 917
- Mount St. Helen, eruption of, 807
- mulching and protective covers, 586–587, 701

- naphthalene, 642
- National Estuary Program, 88
- National Hydrologic Benchmark Water Quality Monitoring Network, 807
- National Nonpoint Pollution Control Program, 10
- National Oceanic and Atmospheric Administration (NOAA), 91, 121, 556

- National Pollution Discharge Elimination System (NPDES), 37, 87–90, 92, 839, 994, 1005
- Nationwide Urban Runoff Project (NURP), 7, 41, 444, 446, 451, 476–480, 484, 491–495, 557, 574, 585, 738–739
- natural flow, 103
- Negev erosion model, 269
- nekton, 748
- neural network models, 550, 656
- nickel
 - in natural base flow, 407
 - in precipitation, 453–454
 - in roof runoff, 480
 - in surficial materials, 319
 - in urban runoff, 42, 477–479
 - source minerals, 412
 - traffic emission, 456
- Nile River, 14
- nitrate, 418, 427
 - natural content of groundwater, 412–413
 - pollution, 31–32, 362
 - contamination of groundwater, 388, 412–413
 - in Long Island, NY, 418
 - of Central Europe, 388
- nitrification, 363, 542, 741, 760–764, 766, 778–779
 - clean-up of nitrate pollution of groundwater, 391
 - in aquaculture ponds, 919
 - in land application systems, 421
 - in soils, 365–367
 - reaction rate, 779
 - simultaneous with denitrification in sediment-water interface, 763–764, 770
 - wetlands, 871, 882
- nitrogen, 21
 - enrichment in overland flow, 291
 - fixation, 362–363
 - forms in waters, 760–762
 - from agriculture, 682
 - immobilization, 369
 - in atmospheric deposition, 215–217
 - in CSOs, 36, 499
 - in feedlot runoff, 34, 688
 - in landfill leachate, 427
 - in precipitation, 453
 - in soils, 361–374
 - in solids from urban areas, 448
 - in urban runoff, 36
 - limiting nutrient, 742
 - loads to Lagoon of Venice, 56

- nitrogen (*cont.*)
 NURP studies, 491–493
 removal in land disposal systems, 421
 removal in ponds, 627
 removal in wetlands, 882–885
 selection of BMPs for control, 692–693
 sources, 361–362
 surface waters impairment, 736
 traffic emission, 456
 nitrogen cycle, 362–363, 761
 nitrogen oxide emissions, 189
 nitrogen to phosphorus ratio, 742, 792
Nitrosomonas and *Nitrobacter*, 741, 762, 778
 nonpoint pollution loads, suburban, 483
 Nonpoint Simulator Model (NPS), 536
 nonpoint sources, definition, 20–21
 North Avenue Dam, Milwaukee, 949–950
 NPDES Stormwater Permit Program, 92, 483, 563
 nutrient control, 59
 agricultural management, 698
 in lakes, 954–956
 transformation and removal in wetlands, 881–887, 906–908
 nutrient loading of lakes, 789–793
 nutrient loads, 56, 61
 nutrient losses, 30
 nutrients
 allochthonous and autochthonous, 787
 Chesapeake Bay, 46
 effect on dissolved oxygen balance, 777
 impact on surface waters, 738–739
 limiting, 741
 octanol partitioning coefficient, 326, 342–345, 404–405, 642–647, 830, 835, 850, 912
 Office of Underground Storage tanks (U.S. EPA), 419
 oil, surface water impairment, 736
 on-site analyses and monitoring, 560–561
 organic chemicals
 control by wetlands, 909–917
 in soils, 339–352
 mobility in soils and sediments, 341–347
 volatile, emissions, 189
 organic matter, 316
 production
 in surface water bodies, 742
 in wetlands, 876–878
 transformation of, 745
 Organization for Economic Cooperation and Development (OECD), 79
 Outstanding National Resource Waters (ONRW), 987, 1003–1004
 overflow frequency to unit storage
 volume relationship, 619–620
 overflow rate, 624, 789
 overland flow
 routing, 144–171, 536
 sediment transport capacity of, 275–280
 treatment and disposal systems, 421
 oxidation ponds, 641
 oxygen. *See also* dissolved oxygen
 depletion in lakes, 957
 injection to lakes, 961
 sinks and sources in surface waters, 766
 supersaturation by photosynthetic action, 743
 ozone depletion, 10
 Pareto optimality, 73
 Parshall flume, 559
 particulate organic carbon, 643–644
 partitioning, of chemicals and metals, 325–332, 829–831, 910–914
 partitioning coefficient, 326, 339, 342–345, 356–357, 359, 361, 404, 829–830, 912. *See also* octanol partitioning coefficient
 pasture, 29–32, 663, 686
 management, 710
 pathogenic microorganisms, 8
 control in agriculture, 693
 in CSOs, 497–499
 in soils, 375–378
 in urban runoff and CSOs, 476–477
 pavement conditions, impact on
 pollutant loads, 456
 pentachlorophenol, in urban runoff, 478–479
 perched water table, 397
 perennial streams, 104, 175, 387
 permeability, 109–110, 554
 coefficient of (hydraulic conductivity), 396
 of animal feedlots, 688
 of geological materials, 395–396
 relation to soil texture, 110
 pervious (porous) pavements, 589–592
 benefits, 590
 connected, decreasing, 592
 construction cost, 592

- drainage requirements, 590
- groundwater contamination potential, 592
- hydraulic conductivity, 590
- installation, 591
- load bearing strength, 590
- longevity, 592
- urban, control of, 586–589
- periphyton, 748
- Pesticide Root Zone Model (PRZM), 543, 546
- Pesticide Runoff Simulator (PRS) model, 542
- pesticides, 41, 345–347
 - adsorption, 326–330
 - application rate, 312
 - atmospheric deposition, 227–231
 - chemical breakdown in soils, 348
 - concentration in precipitation, 229
 - contamination of groundwater, 388–390
 - control in agriculture, 694
 - cycle, 226
 - enrichment in overland flow, 291
 - from agriculture, 32–33
 - inputs to Chesapeake Bay, 230
 - inputs to Great Lakes, 230
 - inputs to wetlands, 230
 - losses from soils, 348
 - microbial metabolism, 348
 - persistence in soils, 347
 - photodecomposition, 346
 - residence in atmosphere, 231
 - source of pollution by urban runoff, 478
 - wet deposition, 227
 - washoff, 227
- pH
 - effect on ammonia volatilization, 371
 - effect on nitrification, 365
 - effect on precipitation of metals, 357
 - of landfill leachate, 427
 - of rainfall
 - in North America, 203
 - worldwide, 205–206
 - of urban snowmelt, 474
- phenanthrene, in urban runoff, 478–479
- phosphorus, 21
 - control, 50
 - lake restoration, 955–956
 - removal in ponds, 627
 - removal in wetlands, 886
 - enrichment in overland flow, 291
 - from agriculture, 682
 - in CSOs, 36, 499
 - in feedlot runoff, 688
 - in soils, 334–339
 - in street salt, 460
 - in urban runoff, 36
 - limiting nutrient, 742
 - mobility in soil, 336
 - natural base flow, 407
 - NURP studies, 491–493
 - release from sediments, 295
 - removal in aquaculture systems, 920
 - selection of agricultural BMPs, 692–693
 - solubility in soil water, 336
 - surface water impairment, 736
 - traffic emission, 456
- photosynthesis, 741–745
- phthalate esters, in urban runoff, 478–479
- phytoplankton, 741–745
- piezometric pressure, 173
- plankton, 748
- plant available soil water capacity, 111
- plant systems, floating, 918–921
- plant uptake of chemicals and nutrients, 332
- point sources, 34
 - definition, 18–22
 - control, 87
 - impact on stream ecology, 927
- pollutants. *See also* adsorption, of pollutants; nitrate pollution; priority pollutants
 - accumulation, near curb, 447, 461–470
 - analyses, 562
 - atmospheric transport of, global, 196–200
 - black list of, 819
 - carcinogenic, 4, 41–42, 345, 818–819
 - conservative, 317
 - curb loading of, 448
 - delivery of, 145, 371, 602–616
 - gray list of, 819
 - half-life of, 334
 - highway loads of, 457, 479
 - in precipitation, 452
 - in soils
 - mobility of, 324
 - pathways of, 317–318
 - loads from urban areas, 442–443
 - mass balance equation, 797, 802, 852, 919–911
 - overland transport of, 317
 - removal of, 627, 638
 - by precipitation (washoff), 470–473
 - by snowmelt in urban areas, 473–476

- pollutants (*cont.*)
 sorption of, in solids and sediments, 332–333
 unit loads of, 507, 519–522
 washoff, availability factor, 472
 wet deposition of, 200–217
 winter accumulation of, 469–470
 polluter pays principle, 79, 994–996, 1004. *See also* economic incentives and instruments
 pollution. *See also* groundwater pollution; toxic pollution
 agricultural, 6–7, 9, 29–34
 abatement, uniformity in, 996
 air, sources, 188
 atmospheric, by toxic compounds, 217–231. *See also* atmospheric pollution
 atmospheric transport of, global, 196–200
 definition, 11
 diffuse, 18. *See also* diffuse pollution
 economic and social causes of, 70–74
 in urban areas, annual variations, 470
 loads, 39, 64, 80, 482–483
 manure, hazard from disposal of, 689
 market impacts on, 75–76
 on impervious surfaces, 461–470
 prevention, 80
 removal efficiencies in models, 525
 sediment, 23, 250, 294, 479
 Pollution from Land Use Activities Reference Group (PLUARG), 740
 polychlorinated biphenyls (PCB), 4, 37, 43–46
 biodegradability in soils and sediments, 352
 distribution, 44–45, 642
 properties, 44–45
 fish contamination, 46
 polycyclic aromatic hydrocarbons (PAHs), 37, 41, 586, 642
 in atmospheric deposition, 229
 in soils, 340–341
 in urban runoff, 478–479
 ponds, 620–629
 agricultural, 702
 dual use for quality and flood control, 629
 enhanced high efficiency wet pond system, 623
 hydraulic detention time, 628
 pond-wetland system for urban runoff pollution control, 893
 potency factors, 318
 removal efficiency, 622, 627
 sizing and dimensions, 628
 potency factors, 449, 521–523
 Potomac Eutrophication Model. *See* WASP water quality model
 prairies, 29
 streams, water quality, 808
 precipitation
 composition of, 200–217
 pollutant content, 452
 reactions in soils and sediments, 325–332
 runoff relationship, 102–106
 priority pollutants, 8, 41–46
 in urban runoff, 477–480
 removal by Best Management Practices, 642–647
 Priority Watershed Program, 996
 probability of exceedence, 485–486
 probability paper, 486–487
 producers, 741
 productivity, 741, 748, 780–785, 878–879
 profundal zone, 744
 pump and baffle aeration of lakes, 958–959
 pyrene, 642–643
 in urban runoff, 478–480

 radar, use in Real Time Control of urban sewerage, 553
 rain gauges, 559
 rainfall, excess, 103, 129–142
 estimation, 521
 from impervious areas, 141–142
 from pervious areas, 131–141
 hydrologically active areas, 142, 525
 rainfall intensity
 critical for initiation of CSO, 498–499
 design storm, 129–131
 duration curves, 130–131
 rainfall runoff transformation process, 106–107
 excess rain, 129–142
 initial subtraction, 108
 overland routing, 144–171
 peak runoff by SCS method, 160
 schematics, 105
 range land, 29, 32, 683, 686
 management, 710
 rapid infiltration wastewater disposal system, 421–422
 rational formula, 145–148, 532

- Real Time Control (RTC) of urban sewerage, 552–554, 649–659
 bottleneck control, 650
 components of, 654–655
 existing systems, 656–658
 models, 655–656
 strategy, 651–654
 receiving water criteria, 80
 redox reactions, 410
 reference water bodies, 753–756, 810, 987
 regional regression models, 524–525
 regulated waters, definition, 89
 regulation, 72–79, 729, 995
 wetland, 897
 regulators and concentrators, 613–616
 removal coefficient, for street pollutants, 463–464
 reservoir construction parameters, 970–972
 riparian buffer zones, 84, 94, 708, 869
 riparian owners, 84
 riparian water rights doctrine, 82–84, 947
 riparian wetlands, 84, 869, 904
 ripraps, 611, 709
 river impoundments, removal of, 948–949
 riverine wetlands, 866
 river restoration, 61
 roadside swales, 603–607
 rock-reed microbial filter, 710
 roof runoff contamination, 480
 runoff, 476. *See also* urban runoff
 and pollutant routing, 530
 enrichment of
 by clays, 286–292
 by nitrogen, 371
 by soil organic matter, 322
 by soil pollutants, 317
 variations, 291
 surface, 129–142
 runoff coefficient, 440–441
 for Rational Formula, 146–148
 runoff pollution, generation process in urban areas, 461–476
 Rural Clean Water Program (RCWP), 9
- Salmonella*, in urban runoff, 476
 salt, 427, 459
 samples, 560–563
 sand, sandstone, 394–396
 sand filter, 639
 sanitary landfills, leaching, 483
 saprobien biotic index, 753
- Sartor et al. equation for washoff, 472–473
 screening, 616, 639
 models, 522–528
 SCS hydrologic soil groups, 109
 SCS Runoff Curve model, 134–141, 158–163, 521, 532, 541–543
 SCS soil maps, 109, 315
 SCS soil texture classification, 289
 SCS unit hydrograph, effluent of imperviousness, 160–161
 secchi disc, 788
 section 208 studies, 87, 522, 557
 section 319 plans, 88, 92
 sediment barriers and silt fences, 611–612
 sediment control basins, 620
 design, 624–626
 dredging and removal of contaminated sediments, 91, 954, 966
 ideal settling basin, 625
 sediment delivery, 239, 249, 270–286, 521, 544, 577, 603
 effect of drainage, 282–283
 estimated by cesium, 137, 284–286
 factors affecting delivery, 272–282
 overland flow effects, 274–275
 sediment flow, discharge relationship, rating curve for, 252, 282
 sediment loads, from urban areas, 449
 sediment oxygen demand (SOD), 747, 764, 766, 769–772
 in situ measured, 772
 remediation, 948–949
 sediment particle sizes, 145
 sediments
 acid and methane formation in, 746
 bedload and washload, 769
 cohesive, 245–246, 275
 affinity to adsorb pollutants, 295
 aggregation, 297
 physical rates of transport, 295–304
 contaminated, dredging and removal of, 53, 954
 control in agriculture, 692
 criteria, 848–852
 deposition of, 293–294
 diagenesis and methanogenesis, 746, 770–771
 effects on toxicity and bioavailability, 251
 inert or refractory fraction, 746
 loss from feedlots, 34
 methane evolution from, 736, 748
 noncohesive, 245, 275

- sediments (*cont.*)
 - organic-rich, 295
 - pickup and transport by runoff, 471
 - pollution, 23, 250, 294, 479
 - rating curve, 251
 - sorption of pollutant, 245
- sediment toxic unit, 828, 849
- sediment transport
 - capacity of overland flow, 275–280
 - cohesive sediments, 295–304
 - critical tractive force for, 278
 - hysteric loops in, 284
 - in streams, 292–304
 - overland, 270–292
 - Shields diagram for, 278
 - storage of transported sediment in streams, 293
- sediment traps, 606
- sediment-water interface
 - carbon and nutrient cycles in, 747
 - processes, 747, 764, 802, 879
- sediment yield, 239, 247, 251–253
- sedimentary rocks, 394
- sedimentation, 251, 301–304
- selenium, 94, 319
- septic systems, leaking into storm sewers, 483
- septic tanks, 415–418
 - failures, 35
 - impact on groundwater resources, 388–389, 415–418
 - sources of nitrogen pollution, 365
- settling, 297, 302
- settling velocity, 297, 299
 - of individual particles, 595, 624
 - particles associated with urban runoff, 627
- phosphorus in lakes, 789
- sewerage agencies, 1004
- sewers. *See also* storm sewers
 - combined, 37. *See also* combined sewer overflows
 - flushing of, 613
 - separated, 37, 62
 - solids, 444, 481–483
 - control, 579
 - effect on CSO loads, 499
- Shakiji River, 62–64
- sheet and rill erosion, 460
- Shields diagram for sediment transport, 278
- Shirako River, 62–64
- silviculture, 686–688
- simazine, 230
- Simulator for Water Resources in Rural Basins (SWRRB) model, 542, 546
- slow rate land application systems of waste, 419–420
- snow enrichment by pollution, 126
- snowmelt, 126–129
 - degree-day formula, 126
 - enrichment of, 291, 474–476
 - runoff 126, 460
 - winter rates, 473
- snow removal, 469
 - to increase light penetration into lakes, 962
- Soil Conservation Service (SCS), 93, 521, 558
- soil loss, 6
 - measured by cesium-137, 284–286
 - urban, 35
- soil map, 109, 315
- soil particles, specific surface of, 287
- soil profile, 314–316
- soils
 - acidification, 313
 - chemical stabilization of, 586
 - classification, 109–110
 - conservation, 8–9
 - contamination, 4, 313, 317–319, 323–334
 - detachment of organic matter, 322
 - moisture characteristics, 111
 - moisture distribution during infiltration, 112
 - permeability, 396
 - porosity, 395
 - sorptivity, 117–118
 - water storage, 111
- soil texture, 109, 253
- soil water zone, 172–173
- spawning habitat management in lakes, 963–965
- surface runoff, 129–142
- standard deviation, definition, 487
- standards, 14–16. *See also* criteria and standards
 - technology-based, 81
- Stanford Watershed Model, 107, 127, 536
- statistical routines and screening models, 509, 517–528
- statutory water use, 986
- steady-state models, 796
- stochastic (ARMA) models, 540–550
- Stokes law for settling, 297, 595, 624
- storage, 575. *See also* surface storage; underground storage tanks
 - depression, 106, 442
 - hydrological, 106, 589
 - interception, 107

- storage basins. *See also* ponds; tunnels
 - agricultural, 702–703
 - dimensioning, 632–635
 - flow balancing, 634–636
 - for CSO control, 39, 61, 629–636
 - in combination with treatment, 637–638
- Storage-Treatment-Overflow-Runoff Model (STORM), 532, 539
- STORET data management system, 555
- STORM model, 463
- storm sewers, 442
 - control of pollution by runoff, 577–602
 - historic outlook, 2, 3
 - nonstormwater discharges into, 483
- stormwater
 - management
 - for quality control, 575
 - utilities, 1004
 - permits, 90
 - reuse, 62
- Stormwater Management Model (SWMM), 463, 533–534, 539
 - calibration, 555
 - data needs, 535
- straw mulch application, 586
- stream bank stabilization, 709, 939–943
- stream restoration, 932–949
 - biomanipulation, 939
 - flow regulation, 937–938
 - land acquisition and regulation, 939
 - nonstructural, 937–939
 - plantings, 938
 - pollution abatement, 938
 - propagation facilities, 939
 - techniques, 936
- streams
 - effluent dominated, 104
 - enrichment by productivity of organic matter, 781
 - ephemeral, 387
 - order, 754, 757
 - perennial, 104, 175, 387
- Streeter–Phelps equation, 766–769, 802
- street flushing, 579
- street refuse
 - deposition, 454
 - particle size distribution, 457–458
- street salt additives and composition, 460
- street solids
 - accumulation, 462, 466
 - urban, vegetative input, 454–455
- street sweepers, sweeping, 463, 578–586
 - efficiency, 581–582, 585
 - particle removal by, 582–583
 - types, 579–582
 - water quality benefits, 584–585
- submerged plant systems, 921
- subsidies, 73, 999
- sulfides
 - immobilization of metals, 889. *See also* acid volatile sulfides
 - minerals containing, 410
 - sulfur, in wetlands, 887–889
 - sulfur dioxide emissions, 189
 - sulphate, 805, 886
- Superfund sites. *See* hazardous waste, disposal sites
- surface and groundwater models, 507–546
 - surface runoff, definition, 103
 - surface storage, 554, 592
 - surface waters, contamination, 312
 - suspended solids, 36
 - removal, 627, 902
 - sustainable agriculture, 3, 680
 - sustainable development, 24, 1012–1014
- swirl-flow regulator, solids concentrator, 613–616
- taxes, 994
- TCLP toxicity analysis, 432
- telemetered monitoring and monitors, 559
- temperature effects
 - on dissolved oxygen balance, 771–773
 - on nitrification, 365
- terraces, 698–699
- Thames River, 3
- thermal factor, 772, 779, 906
- thermal stratification, 961
- tidal dispersion coefficient, 776–777
- tidal marshes, 864–865, 882
- tidal water exchange, 56
- time of concentration, 145, 153–156
- toluene, in urban runoff, 478–479
- Total Maximal Daily Load, 989–993, 1005
- toxic compounds, 6
- toxic chemicals
 - sources, 478
 - model, 852–857
- toxic dissolved gases, 821
- toxicity
 - acute, 821–823
 - acute toxicity unit, 843
 - chemically specific tests, 823–825
 - chronic, 822–823
 - chronic toxicity unit, 843

- toxicity (*cont.*)
 - concepts, 817-821
 - criteria, 840-843
 - frequency (permissible) of
 - exceedence of, 841
 - cumulative, 822
 - definition, 818
 - functional dose (concentration)
 - response relationship, 822, 827
 - lethal, 822
 - measurement, 821-838
 - natural, 819
 - of urban runoff, 477
 - reduction by Best Management Practices, 642-647
 - sediment toxic unit, 828
 - TCLP analysis, 432
 - test organisms, 825
 - tests, 822-831
 - whole body toxicity testing, 823-825, 843
- toxic loads, Chesapeake Bay, 47
- toxic metals, 41, 645, 819, 821
 - atmospheric, 186-195
 - biotic uptake of, 829
 - in septic tank effluents, 418
 - in soils and sediments, 319-320, 352-362
 - interaction between solid and dissolved species, 354
 - in urban runoff, 477-479
 - in wetlands, 909
- toxic organic chemicals, 821
- toxic pollution, 6, 21, 41-46
 - in the atmosphere, 217-231
 - biotic uptake, 829
 - control, 52, 642-649
 - global, 821
 - impact on human health, 820
 - impact on integrity of water bodies, 820
 - partitioning of toxic chemicals, 829-831
 - subsurface disposal, 388
- traffic
 - contribution to diffuse pollution, 456
 - emissions, 188
 - lead, 218
 - exhaust emissions, 455-456
 - impact on urban pollutant loads, 455-459
 - source of urban runoff pollution, 4, 478
- transfer function, 549-551
- treatment plant bypasses, 89
- trophic degree, 781
- trophic indices for lakes, 787-788
- trophic levels, 744-745
- trophic status, 780-781, 784-789
- tunnels (underground), 39, 50, 629-631, 637
- TVA base flow quality model, 413-417
- TVA-HYSIM model, 532
- unconfined animal operation, 29
- underground formations, 395, 397, 398
- underground storage tanks, 418-419, 483
- unit hydrograph, 148-156, 163-171, 541, 549-551
 - convolution integral, 149
 - deconvolution, 163-165
 - direct estimation, 165
 - effect of urbanization, 155
 - instantaneous, 150
 - in stochastic models, 166
 - SCS method, 158-161
 - statistically estimated, 163-171
 - synthetic, 151-158
- unit loads
 - definition, 446
 - from impervious surfaces, 520
 - from pervious urban areas, 521-522
 - from urban areas, 444-451
 - in screening procedures, 522
 - Model Enhanced Unit Loads (MEUL), 522-523
 - of pollutants, 507, 519-522
- Universal Soil Loss Equation (USLE), 254-270, 521, 528, 530, 542-543, 555, 586, 696
 - cropping management factor, 261-263
 - erosion control practice factor, 263
 - modifications, 268-270
 - rainfall energy factor, 254-257
 - reliability, 263-265
 - slope length factor, 259-260
 - soil erodibility factor, 257-259
- urban drainage, 576
- urban erosion 460
- urbanization, 34-39
 - hydrological impacts of, 104
- urban land uses, 443-444
- urban runoff. *See also* runoff
 - control and management, 575-602
 - control of pervious areas, 586-589
 - effects of runoff volume on event
 - mean concentration, 491
 - filtration of, 639-640
 - historic, 1
 - loads to Lagoon of Venice, 56
 - models, 471, 531-540

- pesticides, 478
- pollutant strength, 36
- pollution, 7-8, 491-495, 738-740
- pollution carrying, frequency of, 442
- quality, 491-495
- quality from street flushing, 579
- source control measures, 577-602
- sources of pollution, 445, 451-461
- statistical quality characteristics of, 484-495
- storage strategies for control, 619
- surface waters pollution from, 738
- toxicity, 477
- treatment of, 637-642
- use of wetland control, 891-893
- urban sediment sources, 250
- urban sewerage system, 2, 650
- urban stormwater runoff, 439
- urban washoff coefficient, 472
- urea, hydrolysis of, 368
- use attainability, 16, 807, 988-994
- Use Attainability Analysis (UAA) 988-994, 1001, 1005
- user fees, 994
- U.S. Army Corps of Engineers diffuse pollution control programs, 93, 719
- U.S. Department of Agriculture, 675
 - agricultural pollution management programs, 714-717
- U.S. Department of Interior, diffuse pollution programs, 718-719
- U.S. Environmental Protection Agency (U.S. EPA), 558, 818
 - agricultural pollution management programs, 712-715
- U.S. Fish and Wildlife Service, 5, 862-863
- U.S. Geological Survey (USGS), 556
- USGS Distribute Routing Rainfall Runoff Model (DR3 QUAL), 535
- vadose-aerated zone, 172-174, 392
- variation, coefficient of, 487
- vehicle pollutant emissions, 456, 463
- velocity, design for grassed waterways and swales, 607
- velocity sensors, 559
- Venice canals, 54, 55
- vinyl chloride, 586
- viruses, in soils, 377
- volatilization, 328-330
- Vollenweider's model, for lake eutrophication, 789-793
- vortex flow/solids separator, 614-615
- washload, 243, 249, 294-295
- washoff
 - model (urban), 472
 - of pollutants from impervious surfaces, 439, 472, 520
- washout functions (atmospheric), 200
- WASP water quality model, 798-802, 854-855
- WASSP-QUAL sewer flow model, 533, 538-539
- waste assimilative capacity, 16, 85, 982-984, 990-993, 1000-1003, 1005
 - definition and determination, 16
 - enhancement, 933, 982-983
 - for toxic chemicals, 855
 - of soils, 313
 - of wetlands, 874, 880, 909-917
- waste distribution, 3
- wastewater treatment, 4
 - by land application, 419-424
 - by wetlands, 871-873
 - reuse, 419
- water
 - bicarbonate content, 408-409
 - conservation, 589
- water bodies
 - biotic integrity, measures, 753-759, 818
 - fate of toxic chemicals in, 852-857
 - impaired, 1002-1003
 - management of, 929-1003
 - reference, 753-756, 810, 897
 - restoration, 16, 927-972, 993
 - goals, 928-929
 - lakes and reservoirs, 949-972
 - planning, 929
 - rivers and streams, 932-949
 - structural criteria, 929-932
 - types of, 987
 - uses, 5, 985-993
 - support of designated uses, 736-738
- waterborne epidemics, 2
- Water Erosion Prediction Project (WEPP) Hillslope Profile Model, 269, 543-546
- water hyacinth, 868, 918, 968
- water pollution
 - causes, 737
 - definition, 11-18
 - of coastal waters, 738
- Water Pollution Control Act
 - Amendments. *See* Clean Water Act
- water quality
 - abatement, 5

- water quality (*cont.*)
- approaches, 81
 - background (natural), 13–14, 32, 806–810
 - control, detention volume for, 617
 - criteria and standards, 838–848, 984
 - acute or chronic toxicity, 840
 - chemical-based numerical, 840–843
 - sediment, 848–852
 - ecoregional approach to, 413
 - goals, policy options, 69, 74, 1001
 - impact of diffuse sources, 737
 - improvement
 - aeration for, 947–948
 - socioeconomic impact of, 989
 - limited water bodies, 1001
 - management
 - institutions, 3, 1002–1007
 - integrated, 16, 79, 750, 807, 982, 989, 1004
 - programs, 47–48, 51–52, 57, 59, 88
 - models, 83, 766, 796–802
 - of urban runoff, 492–493
 - planning process, 81, 990
 - margin of safety, 992
 - problems, 735–740
- watersheds
- hydrologic, modification of, 144, 589–601
 - hydrological response, 440
 - hydrology, 102–106
 - management, for lake restoration, 952–955
 - response to rainfall, 153
 - transition, 25–41
- weathering, chemical, 239
- weirs, flow measuring, 559
- wet deposition of pollutants, 200–217
- Wetland Foundation, 862
- wetland–groundwater interaction, 872
- wetlands, 483
- coastal, 864–865
 - constructed. *See* constructed wetlands
 - definition, 863
 - design parameters, 894–909
 - drainage, 5–6, 28, 55, 93, 317
 - filling, 93
 - free water surface, 896–892, 907
 - function, 869–873
 - hydraulic loading of, 899–901
 - hydrology, 896–901
 - hydroperiod of, 897–899
 - inland, 865–869
 - lacustrine, 866
 - loss, historical, 862, 868
 - management programs, 5–6
 - mangrove, 865
 - metals and toxics control, 862
 - nutrient transformation, 881–887
 - organic matter production, 876–887
 - pollutant removal, 92, 876, 905–909
 - protection, 52, 92–94
 - regulation, 897
 - restoration, 869, 874–875, 894–909
 - retention time, 899–900
 - riparian, 84, 869, 904
 - streams, water quality of, 808
 - submerged bed, vegetated, 892
 - sulfur retention and transformation, 887–889
 - types, 863–889
 - use for agricultural runoff control, 707
 - use for stormwater control, 636–637, 871
 - use for water quality control, 873–876, 881
 - vegetation, 868–869
- wet weather flow, 37
- wildlife habitat, 871
- Wisconsin Priority Watershed Programs, 48–52
- xeriscape, 460, 588–589, 1015
- Yalin sediment movement equation, 276–280, 470, 542
- Yellow River, 14
- zinc
- in landfill leachate, 427
 - in natural base flow, 407
 - in NURP studies, 491–493
 - in precipitation, 453–454
 - in roof runoff, 480
 - in soils, 361
 - in urban runoff, 42, 477–479
 - source minerals, 412
 - surface waters impairment, 736
 - traffic emission, 456
- zones
- cohesive sediment transport, 301–302
 - groundwater flow, 171–179
 - groundwater quality, 411

face migration, fertilizer runoff
arms, storm water from streets, and
forms of diffuse pollution are now
nized as having serious environ-
l consequences. In more
0 percent of estimated
in which water quality
are not achieved, diffuse
ion is cited as the main
This guide surveys the
d problems and chal-
presented by this form
ution, and offers a wide
of feasible, cost-effective
ons for dealing with

*er Quality: Prevention, Identification,
nagement of Diffuse Pollution* covers
s of diffuse pollution ranging from
s and sites to health and legal con-
nces. It provides readers with valu-
scussions of
res of diffuse pollution
l definitions
hydrologic nature of diffuse loads,
the processes and land uses
buting to these loads
management practices that empha-
removal of priority pollutants
of wetlands
fate of pollutants in receiving wa-
restoration of reservoirs, streams,
s, and watersheds that have been
aged

ensive data are provided on non-
s well as toxic pollutants, including
oxic compounds. The authors pro-
an integrated approach to water
y control and explain advanced pol-
abatement technologies, such as the
wetlands. Sections on repercus-
to the economy include discussions
/benefit analyses of abatement pro-
and the payments that support the
ams.

oughout, real-world examples illus-
practical applications of various
ol, abatement, and remediation

strategies. Readers discover how to use
modeling techniques to pinpoint the ef-
fects of diffuse pollution, as well as how
to plan environmental protection and re-
covery. Other areas investigat-
ed are erosion, soil pollution,
industrial diffuse pollution
management, and the use of
aquacultures in wetland man-
agement.

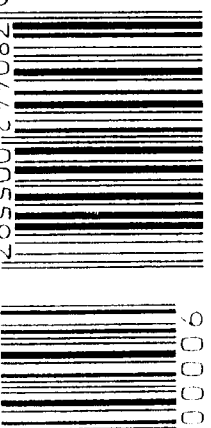
The book offers solutions to
a serious pollution problem—
making it required reading for
environmental and agricultur-
al engineers, planners, regula-
tory professionals, and reme-
diation specialists. It will also be a useful
source for graduate students in environ-
mental engineering and science.

is professor of
Environmental and Water Resources. En-
gineering, chairman of the Environmental
and Water Resources Committee at Mar-
quette University, Milwaukee, Wisconsin,
and president of AquaNova International,
Ltd., an environmental consulting firm.
Dr. Novotny has chaired and organized
several important conferences in the field
of pollution control. He is also a chairman
of the international specialists' group
within the International Association on
Water Quality that deals with diffuse pol-
lution. He has written or co-written nu-
merous books and papers, including
*Handbook of Nonpoint Pollution: Sources and
Management*, published by Van Nostrand
Reinhold. He holds a Ph.D. in Environ-
mental and Water Resources Engineering
from Vanderbilt University, Nashville,
Tennessee.

is director of the Cen-
ter for Watershed Protection and presi-
dent of Olem Associates. He earned a
Ph.D. in Civil Engineering and an M.Sc. in
Environmental Pollution Control from
Pennsylvania State University. He is the
author of *Liming Acidic Surface Waters* and
numerous articles and reports.

Steve Gottlieb ©1993

ISBN 0-442-00559-8
90000



9 780442 005597

th Avenue, New York, NY 10003

R0024023

BEWARE THE SEDIMENT SCARE

JONATHAN JONES

Under the Resource Conservation and Recovery Act, sediments that accumulate in storm water may be classified as hazardous waste. Here's what to do to limit the risk.

Your municipal or industrial storm-water facility meets best management practices (BMPs) as outlined by EPA, and is designed to meet requirements under the Clean Water Act and the accompanying National Pollutant Discharge Elimination System (NPDES). So it's protected from hazardous-waste sanctions, right? Maybe not. Because of the breadth and complexity of the Resource Conservation and Recovery Act (RCRA), a number of unusual and perhaps unintended regulatory consequences can arise. For instance, retention ponds, infiltration basins, buried concrete vaults, created wetlands and other facilities not intended to contain or manage hazardous wastes could be characterized or identified as hazardous-waste—treatment, —storage and —disposal facilities.

Under current RCRA regulations, the "mixture" and "derived-from" rules may cause sediments with extremely low concentrations (barely above the detection limit) of certain constituents to be classified as hazardous waste. The problem has been accentuated by EPA's "contained-in" policy, which dictates that any environmental media (such as water, soil and sediment) containing any amount of listed hazardous waste must be handled entirely as a hazardous waste.

In spite of aggressive source-control efforts at municipal and industrial settings, pollutants listed under RCRA (such as from vehicle/equipment maintenance and yard care) will inevitably find their way into BMP facilities. Designers and facility owners must recognize this and plan for it. There are several protective measures to consider:

- Take reasonable steps to reduce the likelihood that listed chemicals under RCRA will come into contact with precipitation or storm water. Educate workers and the public about the problem. Tell them to cover the hazardous materials or place them indoors. The owner of a 3 acre vehicle-

maintenance facility, for example, which receives no runoff from adjoining properties, should be able to implement rigorous source controls with little difficulty. All work on vehicles can occur indoors. Floor drains and sumps can be connected to pretreatment facilities that discharge into the sanitary sewer. Parking areas can be separated from storm-water conveyances by vegetated "filter strips."

- Take inventory of hazardous chemicals on a regular basis and keep track of their use. Encourage the use of replacement compounds that are not listed by RCRA. Formulate, implement and regularly monitor an aggressive spill-control program.

- Consider installing storm-water-pretreatment facilities to localize the problem. For example, try installing a sedimentation basin immediately upstream from a storm-water wetland. Even with a short retention time, a significant fraction of the total sediment load can be removed from the storm water prior to its discharge into the wetland. This not only localizes sediments and the pollutants that adsorb them, but also benefits the receiving water (stream or wetland) and results in decreased long-term maintenance costs.

- Adopt measures to reduce the quantity of sediment that will enter the BMP over time. The smaller the accumulated sediment load, the less frequently it has to be removed. This is of vital importance because RCRA, in most instances, is relevant only when sediments actually need to be removed. Rigorous erosion and sediment-control practices and judicious use of sand during the winter months can make a big difference.

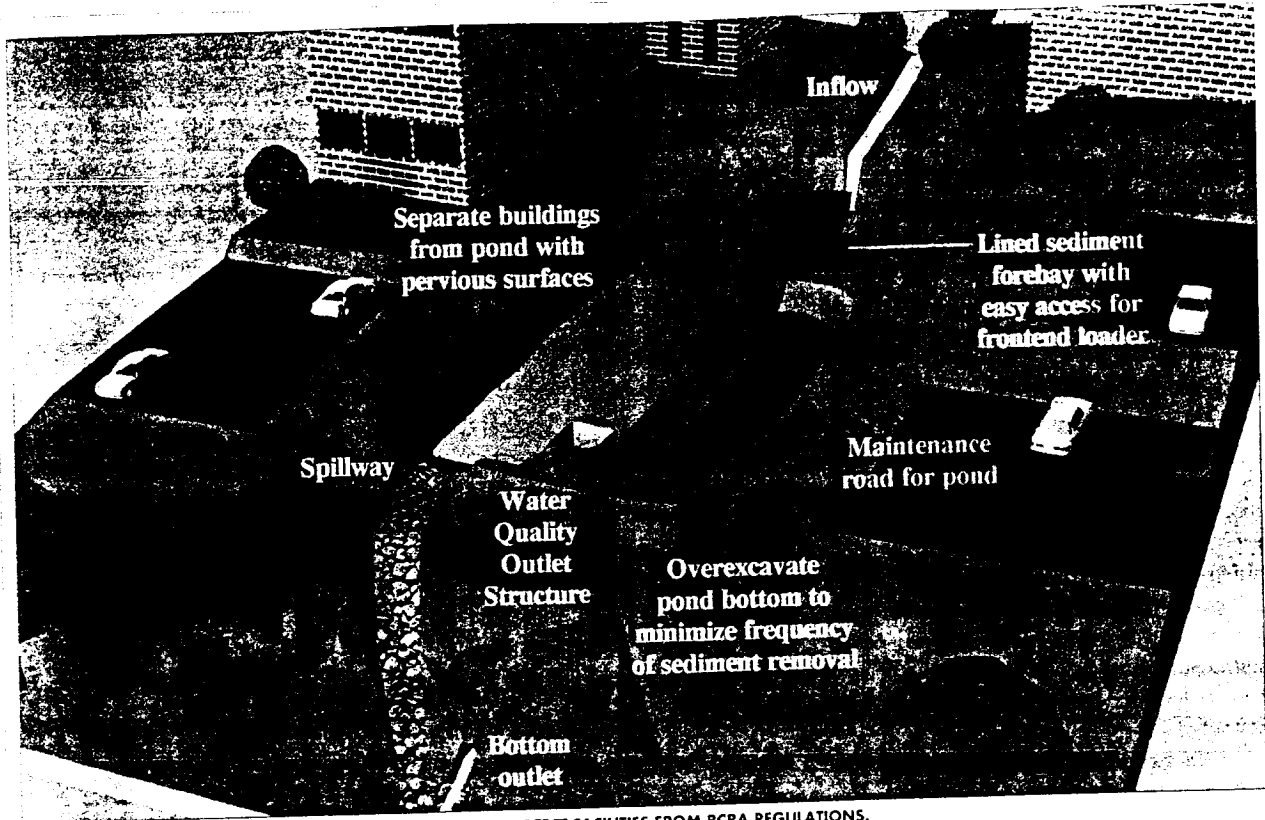
- Evaluate BMP facility design characteristics and implement techniques that will enhance the likelihood of pollutant loss within the BMP. The incorporation of fountains into retention ponds, for example, enhances the volatilization of many volatile and semi-

volatile organic compounds listed by RCRA. However, a permit is required to treat an RCRA hazardous waste, and "treatment" is defined as any method, technique or process—including neutralization—designed to change the chemical, physical, or biological character or composition of a hazardous waste.

- Ensure that facility designers discuss potential problems with regulators and their clients prior to an assignment. Professional liability can be reduced and the owner and engineer can agree, in advance, on mitigation measures. Discussions with regulators are equally important. Designers should cite regulations that force the implementation of storm-water BMPs; delineate the full range of functions that the BMP will fulfill (flood hazard reduction, storm-water quality management and so on); describe structural and nonstructural measures to be taken to reduce the probability that hazardous chemicals will enter the BMP; and suggest a regulatory agreement, in advance, that will address the issue in a reasonable, cost-effective manner if and when the sediments must be removed.

- Design an oversized sediment storage volume in the BMP. Instead of assuming that the BMP should undergo sediment removal once every five to 10 years, assume that sediment removal should occur no more frequently than once every 20–30 years. This will be impractical and excessively expensive in certain instances, but feasible in situations where additional sediment storage can be achieved economically. Recognize that in most instances RCRA actions will not be triggered until sediments need to be removed from the BMP. So every reasonable step should be taken to avoid sediment removal.

- Construct facilities, where practical, that are "off-line" rather than "on-stream." This is certainly not a panacea, but at least the facility sediments will not be located in "wa-



SEDIMENT FOREBAYS AND OTHER DESIGN TECHNIQUES CAN PROTECT FACILITIES FROM RCRA REGULATIONS.

ters of the state or the United States," which could be a complicating factor.

AFTER THE FACT

Sediments in existing BMPs may already contain RCRA-listed chemicals. If that's the case, consider the following steps:

- Conduct additional sampling to confirm the presence of the contaminants. Sample at multiple locations to assure adequate horizontal and vertical coverage. Critically analyze the data collected and determine whether additional actions are required.
- If the drainage area is small, thoroughly search the area for the source of the chemicals that have been detected in the sediments. Once sources are found, check their waste characteristics against the RCRA definition of hazardous waste (40 CFR Part 261) to ensure the wastes are, in fact, RCRA-regulated wastes. A thorough search of this kind is obviously feasible only in smaller drainage basins.
- If the search indicates that RCRA-regulated wastes have mixed with the storm-water facility sediments, take immediate action to ensure that the chemical will not continue to be discharged into the storm-water drainage system. Simply because a chemical or constituent regulated by RCRA is detected in BMP sediments, however, does not

necessarily render the sediment a hazardous waste. If, for example, a spent halogenated solvent listed as hazardous waste is detected in detention pond sediments, those sediments would be hazardous waste under the mixture rule only if the sources of the spent solvent contained more than 10% of that solvent by volume. If no product containing more than 10% of the listed solvent is found, or if contact with precipitation/runoff (including via spills) is unlikely, the pond sediments would not be classified as hazardous waste. The distinction between having one or more discernable point sources of hazardous waste (with suitable characteristics) vs. no clearly defined sources with suitable characteristics is a critical one.

- If a distinct source is not found, convey the information to the regulators. RCRA regulators will be less inclined to control sediments contaminated by diffused sources.
- Note that in many cases, a risk assessment will demonstrate that no practical risk is posed to the public or other life forms by constituent levels within the sediments. This is important information for the regulators because they can exercise some discretion in evaluating regulatory options. If volatile or semivolatile compounds are in the sediments, the risk assessment should

account for the sediment dredging and disposal process, which will cause substantial volatilization and reduce the risk.

- Evaluate modifications to the storm-water facility that may help you avoid excavation of the contaminated sediments. Raising a detention pond outlet structure, for example, increases sediment storage space. Any actions of this kind need to be carefully evaluated, however, because they can have implications related to dam safety, flood attenuation and water-quality performance.

RCRA is undergoing reauthorization and there is a possibility that the updated law may exempt storm-water BMP facilities from RCRA coverage. Early indications, however, are that most RCRA changes will focus only on hazardous-waste definitions. It is prudent to be aware of the potential problem and take steps to limit the risks to BMPs that you design or operate. At times, it may be necessary to confer with qualified legal counsel.

Jonathan Jones, P.E., vice president of Wright Water Engineers, Inc., Denver, coauthored this article with Scot Anderson, senior attorney, ARCO Coal Co., Denver; John Fognani, partner, Gibson, Dunn & Crutcher, Denver; and F. Robert McGregor, P.E., president, Waste Engineering, Inc., Denver.

CT
HP
XS
EH
PT
WY
BR
BY

LANDSCAPE ARCHITECTS AND URBAN EROSION CONTROL

Libbe S. Halevy

Where better to address urban runoff than at a project's front end, where landscape architects can control more than runoff.

All too often, landscape architects are the last to be called in to work on urban projects, allowing them little input in siting, master planning, or other tasks for which they have been trained.

Doug Wagner works as the only landscape architect on staff at Precedent Development, a residential subdivision developer in Indianapolis, IN. He's currently involved with Windemere, a 1,100-home site development northeast of Indianapolis in Fisher. "Windemere mixes first home buyers with those looking to buy \$400,000 custom homes," he explains. "Also there's a commercial area with an elaborate community park, two tennis courts, a basketball court, an extensive playground area, and an Olympic-sized swimming pool." Two hundred home sites are currently occupied, another 200 are being actively marketed, and 700 are in the planning and building stages.

In developing an erosion control plan for the site, Wagner works directly with the civil engineers. "They have to submit other plans to municipalities," he says, "so it makes sense to have them design the erosion control plan along with everything else." Wagner looks at what the engineers come up with and makes changes or recommendations to fit company needs. "We need to make certain their plan fits the pattern of what we do because when you submit your erosion control package to the state, you sign that you're going to implement, monitor, and maintain it. That's a long-term commitment for us."

According to Wagner, municipalities are becoming more aware that siltation from developments goes into storm lines. Local governments have started to require developers to flush the dirt and vacuum it out to keep the storm lines open, Wagner says. "It's more cost-effective to do a good job of preventing erosion from the outset rather than coming back and cleaning out the lines before you can get final municipal acceptance."

For Windemere, Precedent Development was turning farmland into home sites and needed to institute a total erosion control plan. Most of the subdivision is flat with a few wooded areas. "Every lot we develop has to have positive drainage," says Wagner. "Typically, water drains from the pad area where the home is built to the side yard, which then drains toward the rear yard of most homes. We place storm inlets in the back of these yards to accept storm runoff."

Once the water gets into the storm inlets, it runs underground through concrete pipes to one of several 0.2 ha to 0.8 ha (0.5-

"It's more cost-effective to do a good job of preventing erosion from the outset rather than coming back and cleaning out the lines"

ac. to 2-ac.) retention ponds. "Retention ponds are required so you have a mechanism for controlling storm runoff. You are not allowed to release stormwater from your site faster than it was being released before you developed it. Because of the streets, roofs, sidewalks, and drives, water runs off your site much quicker than when the land was being farmed, so you have to have a mechanism for holding the water back and releasing it at a calculated rate. That's what the retention ponds do."

While the minimum required depth of any retention pond is 24 m (8 ft.), typically Precedent digs its ponds 3 m to 5.5 m (10 ft. to 18 ft.) deep. "We like to overdig ours so that if there is any silt leaving the lot and being caught in our retention pond, it's not actually leaving the site. But, ultimately, your target is to stop it from getting into the

pond. You want to keep the dirt on-site."

Ultimately water is released from the ponds into legal drains, streams, or rivers. "It's put back into nature, allowed to go back into the watershed that it was meant to run to. It just happens at a slower rate than it would, given the amount of construction that has taken place."

To keep siltation from happening, Precedent is required to put in seeding material from the edge of the curb to the building. "You try to do this as soon as you can after construction," Wagner says. "Usually you wait until after the utilities are installed because otherwise they'll tear it back up." After gas and phone lines are in, Wagner has each lot seeded 4.6 m to 7.6 m (15 ft. to 25 ft.) from the curb into the right-of-way. "That keeps the soil from being washed directly into the street," Wagner explains. "We also seed rear-yard swales that have direct access to backyard inlets."

One of the biggest erosion problems for Precedent has been dirt tracked out onto streets during construction. To control this, Precedent scrapes the streets with front-end loaders and uses its own street cleaners to sweep the streets. "If you don't," Wagner admits, "the municipality gets after you. But we also need to do it because a cleaner site helps with retail marketing of the home."

Other erosion control measures include woven-fabric erosion control fences along the back of curbs to filter solid material before it gets into the drains, and straw-bale dams around backyard inlets. Many of these measures are required by the Indiana Department of Natural Resources. "We are governed by our state," Wagner says. "Indiana has a mechanism for controlling erosion off-site, and its guidelines must be followed. We like to think we follow those guidelines and do more."

Wagner believes that the worst oversight on any residential-subdivision development is failure to maintain erosion control plans. "You need to come back and replace straw bales, and check the beehive

swarm inlets and erosion control fences that protect curb inlets. If you don't, you risk losing valuable topsoil and incurring fines or delays from the local municipality regulators until the problem is solved."

Frequently the landscape architect's creative impulse becomes stifled by a lack of money. "Most people's appetites are bigger than their pocketbooks," says A. Devitt Day, currently chief building official for Hines County, MS. While still in private practice, Day worked on the grounds behind the old state capitol building in Jackson.

"The building sits on a hill 12 m (40 ft.) above the Illinois Central railroad tracks, with a 15% gradient," Day explains. The state wanted the old gravel parking lot on the hill replaced with a hard-surface parking lot and a bus turnaround/loading area, with cement stairs leading all the way down the slope. Erosion already had begun eating away at the slope, and though the state capitol was built on pylons and was not in immediate danger, stabilization of the slope carried a high priority.

"In order to grade the area and get the parking lot on that slope, we had to use just about every erosion control technique we knew to stabilize the slope until we could get substantial ground cover to hold

it in place," Day explains. Work proceeded between intermittent showers, requiring soil-stabilization efforts throughout construction. "We get better than 137 cm (54 in.) of rain a year here," Day states, "and if you don't consider the rain, you lose your soil." He laid down burlap matting held in

"When you submit your erosion control package to the state, you sign that you're going to implement, monitor, and maintain it"

place with uncoated metal pins, materials that could be left in place to biodegrade after the project was completed.

"But in hard rains you still get dirty water coming across the mat, proof that your soil is washing away." Hay bales lined up at the base of the graded area work as filters to hold back the soil. "When the site dries out, we haven't lost any soil and we haven't put any

sediment on anyone else's property."

After construction Day seeds and mulches separately "because we get a better distribution of the seed that way." For mulch, he uses hay mixed with a slight slurry of asphalt, which is then blown onto the slope under several hundred pounds of pressure. "The asphalt makes the hay stick together, and it becomes an additional mat over the seeds," Day explains. "It will hold long enough for the seeds to germinate, at which point the roots begin to hold the slopes."

Urban erosion not only happens in cemented-over cities. Some of it takes place in formerly rural or country areas when urban dwellers, bent on "getting away," take urban problems along with them.

Bob Wentz Park at Windy Point, 80 km (50 mi.) west of Austin, TX has been a popular windsurfing area for more than a decade. But the actual Windy Point launching site for windsurfers was battered by overuse. Traditionally, people pulled their cars up to the edge of the water, trampled grasses, and wore away the topsoil in pursuit of their sport. Unwittingly they were destroying the environment they came to enjoy.

In 1991, Diane Winterowd of Winterowd Associates in Austin, was asked to partici-

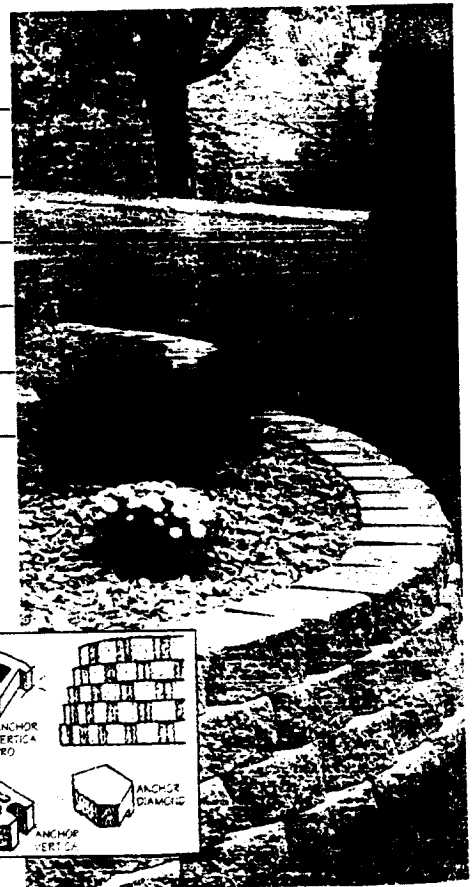
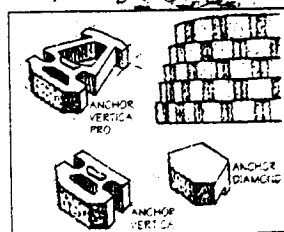
The Building Blocks Of A Beautiful Wall.

No Pins. No Mortar. No Problem.

The Anchor Wall pinless, mortarless retaining wall system makes a beautiful difference in your design. With maximum design flexibility and no height limitations, our walls accent the surroundings with warm earthtones and natural textures in maintenance-free, environmentally friendly units. Every segmental block is molded with an integral rear lip or top locator to automatically guide each course, ensuring proper alignment and set back. The best choice for hard to accommodate areas, Anchor Wall is the right fit for your project. You can't beat our system for design adaptability, easy installation and lasting performance. For the Anchor Wall dealer nearest you, call 1-800-473-4452.



ANCHOR WALL SYSTEMS, INC.
6101 Baker Road, Suite 201, Minneapolis, MN 55345-5973 1-800-473-4452



Circle #13 on Reader Service Card

R0024027

JULY/AUGUST 1995

pate in a biorevetment program at Windy Point. "By the time we got started," she says, "ninety-nine percent of the land had been denuded. All we had left was exposed stone, rock, and gravel with a few scruffy old trees near the top." The point itself is not steep, more like a gently sloping rise that manages to stick out of the water. Reconstruction began with three layers of circular terraces a few feet high that went around the point. Winterowd says, "That way we would be able to fill behind them with about 0.6 m (2 ft.) of soil gradually tapering off to the next terrace level, bringing soil back and at the same time creating a way to keep it from sliding back into the lake." The boulders and soil used in the reconstruction came from other construction locations at Windy Point and from an off-site project in similar hilly country.

The new topsoil was seeded with a variety of native grasses and what Winterowd calls "good riparian kinds of trees, such as sycamores, pecan, bald cypress, and Texas ash." As important as the revegetation to the project's success were new restrictions on public access to the site. "People are no longer allowed to drive their vehicles all the way out onto the point," she explains. "We created an area for them to unload their gear, but then they have to park in a nearby lot." The park also imposed a limit to the number of people allowed into the facility.

Winterowd is currently involved in another park project, one that is serving as a bioengineering pilot program for Austin. "A stream running through two parks (Bartholomew and Givens) in Austin has been eroding the banks badly," Winterowd states. "We've lost land and trees to it. The banks are being undercut to less stable gravel layers. As a result, it sloughs off very easily. In other areas there's debris clogging the creek and backing it up, which forces the creek to cut a route for the overflow. This secondary channel may isolate as much as an acre of land from the rest of the park."

Using a bioengineering approach has confronted Winterowd with special problems. "Bioengineering relies a lot on the use of willow and dogwood as pioneer species," she states, "but only one variety of willow and one variety of dogwood are native to the Austin area. Species used successfully in the East and Northeast do not exist in Austin. And while the city receives an average of 86 cm (34 in.) of rainfall a year, it also has long dry periods." Anxious to see if bioengineering techniques are able to cope successfully with the climate, the city still is using the Bartholomew/Givens restoration as a pilot program.

Another problem is that work can only be done on 152 m (500 lineal ft.) at a time.

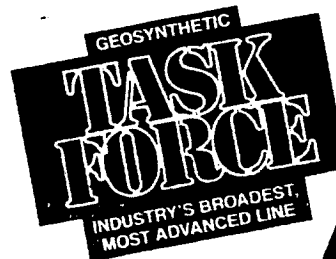
"It's a permitting situation," Winterowd explains. "For the city, 152 m (500 ft.) is considered a maintenance operation. If we exceed that, it would be considered 'new construction' and we would have to go through the city's whole permitting process. It wouldn't change anything we're doing, but it would slow down the process. This way we get a general permit to do repairs and maintenance on the property, requiring some small level of construction, without having to go through the submittals, staff reviews, and resubmissions."

Even the most creative landscape archi-

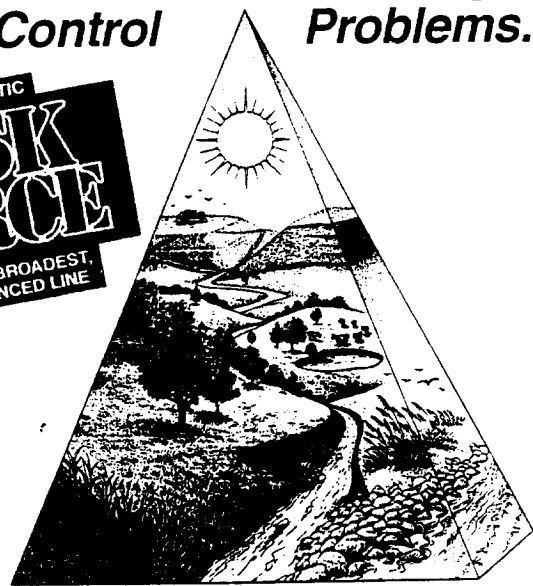
tects are called upon to do what Bill Young labels "cookie-cutter work." Young, owner of Restoration Partners in West Palm Beach, FL, decries the lack of imagination usually required by the job. "It's all dictated by standards and regulations," he says. "Frequently developers are out to do the bare minimum, and you find yourself stuck with the situation."

Young worked on the installation of street trees in the Staten Island [NY] Hillside Preservation District, a project tightly controlled by pre-existing statutes and regulations. "The City Planning Department was

The Ultimate Solution for Tough Erosion Control Problems.



NEW



PYRAMAT™ Erosion Matrix

Specify PYRAMAT for:

- ▲ Landfill Caps, Slopes & Diversion Structures
- ▲ Dams, Dike & Levee Protection
- ▲ Bank & Shoreline Stabilization
- ▲ High Flow Channels
- ▲ Geosynthetic Reinforced Earth Structures
- ▲ Slope Facings
- ▲ Veneer Cover Soil Stabilization

PYRAMAT is a tough 3-dimensional woven geotextile with unique up and down cavities. These cells interlock with soil to provide dimensional stability even under the most demanding erosion conditions.



SYNTHETIC INDUSTRIES
Construction Products Division
"Smart Solutions in Synthetics"™
4019 Industry Drive, Chattanooga, TN 37416, USA
615-899-0444 Fax 615-899-7619

For more information on PYRAMAT, call 800-FIX SOIL

Circle #17 on Reader Service Card

looking to preserve trees both for aesthetic and erosion control purposes, which was a noble idea," he says. But the city had a rigid formula for determining how many trees were "owed" in each location.

"Every tree you saved on a site was worth points determined by its size. For every tree cut down, you owed an equivalent amount of caliper-inches in on-site and street trees. The City Planning Department provided a very limited list of acceptable trees. Your only option was to select which ones were to be installed—a task that did nothing to stimulate innovative design."

Tales from the North Woods

"Not all big-city projects get smothered in institutionalized details," Young concedes. "A good example is the Central Park renewal project," the comprehensive program undertaken by New York City's Central Park Conservancy (CPC), a non-profit organization formed in 1980 to restore and maintain this vital urban wildland. Working in close coordination with the city's Department of Parks and Recreation (DPR), CPC has attracted public involvement, raised significant funds, and overseen the renovation of dozens of sites throughout the park. In 1984, when efforts to restore

the park's woodland areas were mired down in a sea of endless debate, Andropogon Associates, an outside landscape architect firm from Philadelphia, PA, was called in to develop a workable restoration and management plan.

"You can't have 32 million feet pounding the trails without turning the soil to concrete," points out Andropogon's principal, Leslie Sauer, whose task was to provide direction. "We realized we had to tap into the insight, skills, and enthusiasm of the key players—users and caretakers alike—and rely on them to find and implement solutions." Thus, she cast herself in the role of facilitator rather than author of subsequent efforts... an inspired tack as it turned out. "Involvement is crucial," she says. "You want the users doing the restoration, but it's not just a case of getting groups to work together. We wanted to develop a new constituency armed with real scientific information about what's at stake and what the options are."

Faced with such factors as the success of exotic species, the effects of air pollution, and the continuous compaction by traffic of all sorts, all parties could agree that the park would need far more than a gardening approach of simply replanting

native species. Rather, they wanted to restore the conditions under which native species would once more reproduce naturally on their own. "We experimented with control plots before taking any wholesale action on the ground," Sauer says. Accurate records have been maintained to assess the success of their efforts. Even more important, in Sauer's mind, was recognizing the need to substantially upgrade the care of the landscape and to include training for staff and volunteers in the program.

"Controlling erosion is prerequisite to stabilization," Sauer says. "Volunteers are schooled in the use of various techniques including decompacting surface soil, the use of organic erosion blankets, and refilling ravines and rutted trails." But stewardship is more than that. Sauer believes that an active and enthusiastic corps of volunteers is crucial, not just as a productive labor force, but as a conduit to various user communities and the public at large. People who are involved, they have found, are able to pass along their hands-on knowledge to others, in effect multiplying the impact of their program's efforts. EC

Guest author Libbe S. HaLevy contributes frequently to environmental journals.

GABIONS & MATTRESSES

Sound Solutions For Your Engineering Needs



OUR STRENGTH IS OUR FLEXIBILITY

- STRONG • FLEXIBLE
- PERMEABLE • COST EFFECTIVE
- ENVIRONMENTALLY SOUND
- AESTHETICALLY PLEASING
- VERSATILE APPLICATIONS

For More Information or Technical Assistance...

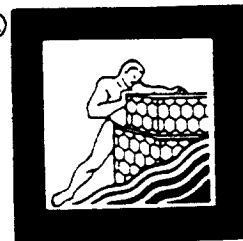
CALL

MACCAFERRI GABIONS INC.

10303 Governor Lane Blvd.
Williamsport, MD 21795-9602
TEL: (301) 223-6910
FAX: (301) 223-6134

3650 Seaport Blvd.
West Sacramento, CA 95691
TEL: (916) 371-5805
FAX: (916) 371-0764

®



Circle #18 on Reader Service Card

LANDSCAPE ARCHITECTS AND URBAN EROSION CONTROL

Libbe S. HaLevy

Where better to address urban runoff than at a project's front end, where landscape architects can control more than runoff.

Too often, landscape architects are the last to be called in to work on urban projects, allowing them little input in siting, master planning, or other tasks for which they have been trained.

Doug Wagner works as the only landscape architect on staff at Precedent Development, a residential subdivision developer in Indianapolis, IN. He's currently involved with Windemere, a 1,100-home site development northeast of Indianapolis in Fisher. "Windemere mixes first home buyers with those looking to buy \$400,000 custom homes," he explains. "Also there's a commercial area with an elaborate community park, two tennis courts, a basketball court, an extensive playground area, and an Olympic-sized swimming pool." Two hundred home sites are currently occupied, another 200 are being actively marketed, and 700 are in the planning and building stages.

In developing an erosion control plan for the site, Wagner works directly with the civil engineers. "They have to submit other plans to municipalities," he says, "so it makes sense to have them design the erosion control plan along with everything else." Wagner looks at what the engineers come up with and makes changes or recommendations to fit company needs. "We need to make certain their plan fits the pattern of what we do because when you submit your erosion control package to the state, you sign that you're going to implement, monitor, and maintain it. That's a long-term commitment for us."

According to Wagner, municipalities are becoming more aware that siltation from developments goes into storm lines. Local governments have started to require developers to flush the dirt and vacuum it out to keep the storm lines open, Wagner says. "It's more cost-effective to do a good job of preventing erosion from the outset rather than coming back and cleaning out the lines before you can get final municipal acceptance."

For Windemere, Precedent Development was turning farmland into home sites and needed to institute a total erosion control plan. Most of the subdivision is flat with a few wooded areas. "Every lot we develop has to have positive drainage," says Wagner. "Typically, water drains from the pad area where the home is built to the side yard, which then drains toward the rear yard of most homes. We place storm inlets in the back of these yards to accept storm runoff."

Once the water gets into the storm inlets, it runs underground through concrete pipes to one of several 0.2 ha to 0.8 ha (0.5-

"It's more cost-effective to do a good job of preventing erosion from the outset rather than coming back and cleaning out the lines"

ac. to 2-ac.) retention ponds. "Retention ponds are required so you have a mechanism for controlling storm runoff. You are not allowed to release stormwater from your site faster than it was being released before you developed it. Because of the streets, roofs, sidewalks, and drives, water runs off your site much quicker than when the land was being farmed, so you have to have a mechanism for holding the water back and releasing it at a calculated rate. That's what the retention ponds do."

While the minimum required depth of any retention pond is 24 m (8 ft.), typically Precedent digs its ponds 3 m to 5.5 m (10 ft. to 18 ft.) deep. "We like to overdig ours so that if there is any silt leaving the lot and being caught in our retention pond, it's not actually leaving the site. But, ultimately, your target is to stop it from getting into the

pond. You want to keep the dirt on-site."

Ultimately water is released from the ponds into legal drains, streams, or rivers. "It's put back into nature, allowed to go back into the watershed that it was meant to run to. It just happens at a slower rate than it would, given the amount of construction that has taken place."

To keep siltation from happening, Precedent is required to put in seeding material from the edge of the curb to the building. "You try to do this as soon as you can after construction," Wagner says. "Usually you wait until after the utilities are installed because otherwise they'll tear it back up." After gas and phone lines are in, Wagner has each lot seeded 4.6 m to 7.6 m (15 ft. to 25 ft.) from the curb into the right-of-way. "That keeps the soil from being washed directly into the street," Wagner explains. "We also seed rear-yard swales that have direct access to backyard inlets."

One of the biggest erosion problems for Precedent has been dirt tracked out onto streets during construction. To control this, Precedent scrapes the streets with front-end loaders and uses its own street cleaners to sweep the streets. "If you don't," Wagner admits, "the municipality gets after you. But we also need to do it because a cleaner site helps with retail marketing of the home."

Other erosion control measures include woven-fabric erosion control fences along the back of curbs to filter solid material before it gets into the drains, and straw-bale dams around backyard inlets. Many of these measures are required by the Indiana Department of Natural Resources. "We are governed by our state," Wagner says. "Indiana has a mechanism for controlling erosion off-site, and its guidelines must be followed. We like to think we follow those guidelines and do more."

Wagner believes that the worst oversight on any residential-subdivision development is failure to maintain erosion control plans. "You need to come back and replace straw bales, and check the beehive

swarm inlets and erosion control fences that protect curb inlets. If you don't, you risk losing valuable topsoil and incurring fines or delays from the local municipality regulators until the problem is solved."

Frequently the landscape architect's creative impulse becomes stifled by a lack of money. "Most people's appetites are bigger than their pocketbooks," says A. Devitt Day, currently chief building official for Hines County, MS. While still in private practice, Day worked on the grounds behind the old state capitol building in Jackson.

"The building sits on a hill 12 m (40 ft.) above the Illinois Central railroad tracks, with a 15% gradient," Day explains. The state wanted the old gravel parking lot on the hill replaced with a hard-surface parking lot and a bus turnaround/loading area, with cement stairs leading all the way down the slope. Erosion already had begun eating away at the slope, and though the state capitol was built on pylons and was not in immediate danger, stabilization of the slope carried a high priority.

"In order to grade the area and get the parking lot on that slope, we had to use just about every erosion control technique we knew to stabilize the slope until we could get substantial ground cover to hold

it in place," Day explains. Work proceeded between intermittent showers, requiring soil-stabilization efforts throughout construction. "We get better than 137 cm (54 in.) of rain a year here," Day states, "and if you don't consider the rain, you lose your soil." He laid down burlap matting held in

"When you submit your erosion control package to the state, you sign that you're going to implement, monitor, and maintain it"

place with uncoated metal pins, materials that could be left in place to biodegrade after the project was completed.

"But in hard rains you still get dirty water coming across the mat, proof that your soil is washing away." Hay bales lined up at the base of the graded area work as filters to hold back the soil. "When the site dries out, we haven't lost any soil and we haven't put any

sediment on anyone else's property.

After construction Day seeds and mulches separately "because we get a better distribution of the seed that way." For mulch, he uses hay mixed with a slight slurry of asphalt, which is then blown onto the slope under several hundred pounds of pressure. "The asphalt makes the hay stick together and it becomes an additional mat over the seeds," Day explains. "It will hold long enough for the seeds to germinate, at which point the roots begin to hold the slopes."

Urban erosion not only happens in cemented-over cities. Some of it takes place in formerly rural or country areas when urban dwellers, bent on "getting away," take urban problems along with them.

Bob Wentz Park at Windy Point, 80 km (50 mi.) west of Austin, TX has been a popular windsurfing area for more than a decade. But the actual Windy Point launching site for windsurfers was battered by overuse. Traditionally, people pulled their cars up to the edge of the water, trampled grasses and wore away the topsoil in pursuit of their sport. Unwittingly they were destroying the environment they came to enjoy.

In 1991, Diane Winterowd of Winterowd Associates in Austin, was asked to partici-

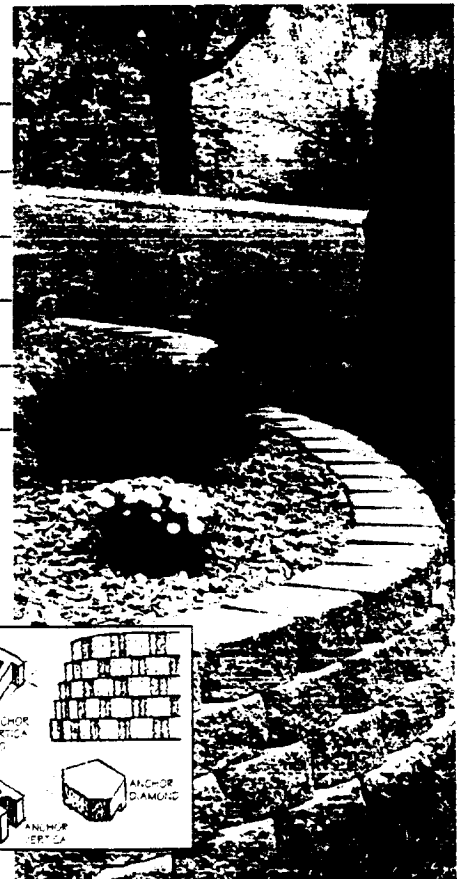
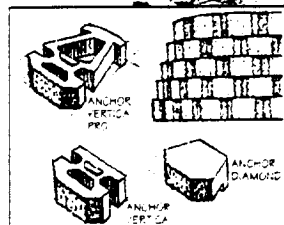
The Building Blocks Of A Beautiful Wall.

No Pins. No Mortar. No Problem.

The Anchor Wall pinless, mortarless retaining wall system makes a beautiful difference in your design. With maximum design flexibility and no height limitations, our walls accent the surroundings with warm earthtones and natural textures in maintenance-free, environmentally friendly units. Every segmental block is molded with an integral rear lip or top locator to automatically guide each course, ensuring proper alignment and set back. The best choice for hard to accommodate areas, Anchor Wall is the right fit for your project. You can't beat our system for design adaptability, easy installation and lasting performance. For the Anchor Wall dealer nearest you, call 1-800-473-4452.



ANCHOR WALL SYSTEMS, INC.
6101 Baker Road, Suite 201, Minnetonka, MN 55345-5973 1-800-473-4452



Circle #13 on Reader Service Card

pate in a bioevetment program at Windy Point. "By the time we got started," she says, "ninety-nine percent of the land had been denuded. All we had left was exposed stone, rock, and gravel with a few scruffy old trees near the top." The point itself is not steep, more like a gently sloping rise that manages to stick out of the water. Reconstruction began with three layers of circular terraces a few feet high that went around the point. Winterowd says, "That way we would be able to fill behind them with about 0.6 m (2 ft.) of soil gradually tapering off to the next terrace level, bringing soil back and at the same time creating a way to keep it from sliding back into the lake." The boulders and soil used in the reconstruction came from other construction locations at Windy Point and from an off-site project in similar hilly country.

The new topsoil was seeded with a variety of native grasses and what Winterowd calls "good riparian kinds or trees, such as sycamores, pecan, bald cypress, and Texas ash." As important as the revegetation to the project's success were new restrictions on public access to the site. "People are no longer allowed to drive their vehicles all the way out onto the point," she explains. "We created an area for them to unload their gear, but then they have to park in a nearby lot." The park also imposed a limit to the number of people allowed into the facility.

Winterowd is currently involved in another park project, one that is serving as a bioengineering pilot program for Austin. "A stream running through two parks (Bartholomew and Givens) in Austin has been eroding the banks badly," Winterowd states. "We've lost land and trees to it. The banks are being undercut to less stable gravel layers. As a result, it sloughs off very easily. In other areas there's debris clogging the creek and backing it up, which forces the creek to cut a route for the overflow. This secondary channel may isolate as much as an acre of land from the rest of the park."

Using a bioengineering approach has confronted Winterowd with special problems. "Bioengineering relies a lot on the use of willow and dogwood as pioneer species," she states, "but only one variety of willow and one variety of dogwood are native to the Austin area. Species used successfully in the East and Northeast do not exist in Austin. And while the city receives an average of 86 cm (34 in.) of rainfall a year, it also has long dry periods." Anxious to see if bioengineering techniques are able to cope successfully with the climate, the city still is using the Bartholomew/Givens restoration as a pilot program.

Another problem is that work can only be done on 152 m (500 lineal ft.) at a time.

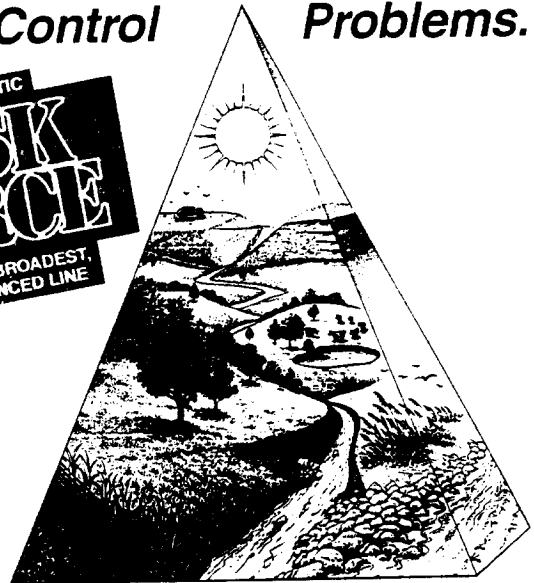
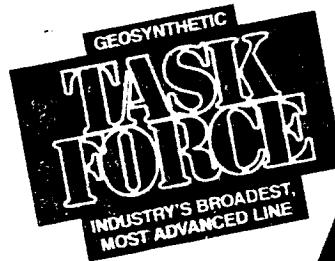
"It's a permitting situation," Winterowd explains. "For the city, 152 m (500 ft.) is considered a maintenance operation. If we exceed that, it would be considered 'new construction' and we would have to go through the city's whole permitting process. It wouldn't change anything we're doing, but it would slow down the process. This way we get a general permit to do repairs and maintenance on the property, requiring some small level of construction, without having to go through the submittals, staff reviews, and resubmissions."

Even the most creative landscape archi-

tects are called upon to do what Bill Young labels "cookie-cutter work." Young, owner of Restoration Partners in West Palm Beach, FL, decries the lack of imagination usually required by the job. "It's all dictated by standards and regulations," he says. "Frequently developers are out to do the bare minimum, and you find yourself stuck with the situation."

Young worked on the installation of street trees in the Staten Island [NY] Hillside Preservation District, a project tightly controlled by pre-existing statutes and regulations. "The City Planning Department was

The Ultimate Solution for Tough Erosion Control Problems.



NEW

PYRAMAT™ Erosion Matrix

Specify PYRAMAT for:

- ▲ Landfill Caps, Slopes & Diversion Structures
- ▲ Dams, Dike & Levee Protection
- ▲ Bank & Shoreline Stabilization
- ▲ High Flow Channels
- ▲ Geosynthetic Reinforced Earth Structures
- ▲ Slope Facings
- ▲ Veneer Cover Soil Stabilization

PYRAMAT is a tough 3-dimensional woven geotextile with unique up and down cavities. These cells interlock with soil to provide dimensional stability even under the most demanding erosion conditions.



SYNTHETIC INDUSTRIES

Construction Products Division
"Smart Solutions in Synthetics"™

4019 Industry Drive, Chattanooga, TN 37416, USA
615-899-0444 Fax 615-899-7619

For more information on PYRAMAT, call 800-FIX SOIL

Circle #17 on Reader Service Card

looking to preserve trees both for aesthetic and erosion control purposes, which was a noble idea," he says. But the city had a rigid formula for determining how many trees were "owed" in each location.

"Every tree you saved on a site was worth points determined by its size. For every tree cut down, you owed an equivalent amount of caliper-inches in on-site and street trees. The City Planning Department provided a very limited list of acceptable trees. Your only option was to select which ones were to be installed—a task that did nothing to stimulate innovative design."

Tales from the North Woods

"Not all big-city projects get smothered in institutionalized details," Young concedes. "A good example is the Central Park renewal project," the comprehensive program undertaken by New York City's Central Park Conservancy (CPC), a non-profit organization formed in 1980 to restore and maintain this vital urban wildland. Working in close coordination with the city's Department of Parks and Recreation (DPR), CPC has attracted public involvement, raised significant funds, and overseen the renovation of dozens of sites throughout the park. In 1984, when efforts to restore

the park's woodland areas were mired down in a sea of endless debate, Andropogon Associates, an outside landscape architect firm from Philadelphia, PA, was called in to develop a workable restoration and management plan.

"You can't have 32 million feet pound-ing the trails without turning the soil to concrete," points out Andropogon's principal, Leslie Sauer, whose task was to provide direction. "We realized we had to tap into the insight, skills, and enthusiasm of the key players—users and caretakers alike—and rely on them to find and implement solutions." Thus, she cast herself in the role of facilitator rather than author of subsequent efforts...an inspired tack as it turned out. "Involvement is crucial," she says. "You want the users doing the restoration, but it's not just a case of getting groups to work together. We wanted to develop a new constituency armed with real scientific information about what's at stake and what the options are."

Faced with such factors as the success of exotic species, the effects of air pollution, and the continuous compaction by traffic of all sorts, all parties could agree that the park would need far more than a gardening approach of simply replanting

native species. Rather, they wanted to restore the conditions under which native species would once more reproduce naturally on their own. "We experimented with control plots before taking any wholesale action on the ground," Sauer says. Accurate records have been maintained to assess the success of their efforts. Even more important, in Sauer's mind, was recognizing the need to substantially upgrade the care of the landscape and to include training for staff and volunteers in the program.

"Controlling erosion is prerequisite to stabilization," Sauer says. "Volunteers are schooled in the use of various techniques including decompacting surface soil, the use of organic erosion blankets, and refilling ravines and rutted trails." But stewardship is more than that. Sauer believes that an active and enthusiastic corps of volunteers is crucial, not just as a productive labor force, but as a conduit to various user communities and the public at large. People who are involved, they have found, are able to pass along their hands-on knowledge to others, in effect multiplying the impact of their program's efforts. EC

Guest author Libbe S. HaLevy contributes frequently to environmental journals.

GABIONS & MATTRESSES

Sound Solutions For Your Engineering Needs



- STRONG • FLEXIBLE
- PERMEABLE • COST EFFECTIVE
- ENVIRONMENTALLY SOUND
- AESTHETICALLY PLEASING
- VERSATILE APPLICATIONS

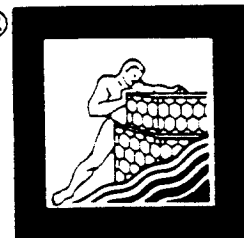
For More Information or Technical Assistance...

CALL

MACCAFERRI GABIONS INC.

10303 Governor Lane Blvd.
Williamsport, MD 21795-9602
TEL: (301) 223-6910
FAX: (301) 223-6134

3650 Seaport Blvd.
West Sacramento, CA 95691
TEL: (916) 371-5805
FAX: (916) 371-0764



OUR STRENGTH IS OUR FLEXIBILITY

Circle #18 on Reader Service Card

POTENTIAL FOR CHEMICAL TRANSPORT BENEATH A STORM-RUNOFF RECHARGE (RETENTION) BASIN FOR AN INDUSTRIAL CATCHMENT IN FRESNO, CALIFORNIA

By Roy A. Schroeder

U.S. Geological Survey
Water-Resources Investigations Report 93-4140

Prepared in cooperation with the
FRESNO METROPOLITAN FLOOD CONTROL DISTRICT

6417-17

Sacramento, California
1995

R0024034

**U.S. DEPARTMENT OF THE INTERIOR
BRUCE BABBITT, Secretary**

**U.S. GEOLOGICAL SURVEY
Gordon P. Eaton, Director**



Any use of trade, product, or firm names in this publication is for descriptive purposes only and does not imply endorsement by the U.S. Government.

For sale by the
U.S. Geological Survey
Earth Science Information Center
Open-File Reports Section
Box 25286, MS 517
Denver Federal Center
Denver, CO 80225

For additional information write to:
District Chief
U.S. Geological Survey
Federal Building, Room W-2233
2800 Cottage Way
Sacramento, CA 95825

R0024035

CONTENTS

Abstract	1
Introduction	1
Urban hydrology studies	1
Purpose and scope	2
Acknowledgments	2
Description of the study area	2
Geology, hydrology, and ground-water quality near the study area	4
Field and analytical methods	6
Site selection, well installation, and soil coring	6
Sampling of sediment from the recharge basin	9
Chemical analysis of water, soil, and sediment	10
Quality of recharge-basin water and ground water	13
Isotopic evidence for recharge	13
Major ions and trace elements	16
Organic priority pollutants and pesticides	17
Chemical concentrations in sediment and soil	17
Inorganic elements	17
Organic chemicals	24
Assessment of anthropogenic inputs	25
Laboratory elutriation experiments	26
Transport and loading of contaminated sediment	31
Comparison of basin-sediment inventories and storm-runoff loads	32
Lead-210 constraints on origin and loading of contaminated sediment	34
Summary and conclusions	35
References cited	36

FIGURES

1. Map showing location of study area 3
2. Diagram showing recharge basin and location of test holes/monitor wells and sediment-sampling sites in the recharge basin 5
- 3,4. Graphs showing:
 3. Relation between hydrogen and oxygen stable-isotope ratios in recharge-basin water and nearby ground water 16
 4. Enrichment ratios for selected constituents in subbottom sediment at site 5 in the recharge basin 24

TABLES

1. Historical ground-water-quality data from well 14S/20E-24K1 near the study site 6
2. Particle-size distribution in soil cores from two test holes adjacent to the recharge basin 7
3. Geologic log for test hole 1 adjacent to the recharge basin 8
4. Sand, silt, and clay content of soil cores from two test holes adjacent to the recharge basin 9
5. Particle-size distribution in sediment beneath the recharge basin 11
6. Concentrations of dissolved inorganic and total-recoverable organic constituents in water from the recharge basin and from two nearby monitor wells 14
7. Concentrations of elements and radioisotopes in soil cores from test hole 1 adjacent to the recharge basin 18

8. Concentrations of chemical constituents and radioisotopes in surface (0-1 cm) sediment from six sites in the recharge basin 19
9. Concentrations of chemical constituents and radioisotopes in sediment and substrata at site 5 in the recharge basin 21
10. Concentrations of extractable priority pollutants, organochlorine compounds, and organophosphorus insecticides in sediment from depths of 0-1 and 8-16 cm at site 5 in the recharge basin 23
11. Surface-enrichment ratios for selected elements and radioisotopes in sediment at site 5 in the recharge basin 26
12. Chemical concentrations in water from the recharge basin equilibrated (mixed for 0, 1, 2, and 4 days) with surface (0-1 centimeter) sediment from near site 5 in the recharge basin 27
13. Chemical concentrations in water from the recharge basin equilibrated (mixed for 4 days) with aquifer soil from 10.9-11.3 meters in test hole 1, in deionized water equilibrated (mixed for 4 days) with surface (0-1 centimeter) sediment from near site 5 in the recharge basin, and in a deionized-water blank 28
14. Concentrations of elements in recharge-basin sediment and in aquifer soil used for the laboratory elutriation tests 29
15. Concentrations of chemical constituents in recharge-basin sediment and aquifer soil after the laboratory elutriation tests 31
16. Percent extractable for selected elements in recharge-basin sediment and aquifer soil by digestion with hydrogen peroxide plus mild acid 31
17. Annual loads to the industrial recharge basin determined from runoff monitoring and sediment inventories 32

CONVERSION FACTORS, VERTICAL DATUM, ABBREVIATIONS, AND WATER-QUALITY INFORMATION

Conversion Factors

Multiply	By	To obtain
centimeter (cm)	0.3937	inch
cubic centimeter (cm ³)	0.06102	cubic inch
centimeter per day (cm/d)	0.3937	inch per day
square centimeter (cm ²)	0.155	square inch
gram (g)	0.03527	ounce (avoirdupois)
gram per cubic centimeter (g/cm ³)	0.57808	ounce per cubic inch
hectare (ha)	2.471	acre
kilogram (kg)	2.205	pound (avoirdupois)
kilogram per year (kg/yr)	2.205	pound per year
kilometer (km)	0.6214	mile
square kilometer (km ²)	0.3861	square mile
meter (m)	3.281	foot
meter per kilometer (m/km)	5.2800	foot per mile
meter per year (m/yr)	3.281	foot per year
millimeter (mm)	0.03937	inch
square millimeter (mm ²)	0.00155	square inch

Temperature is given in degrees Celsius (°C), which can be converted to degrees Fahrenheit (°F) by the following equation:

$$^{\circ}\text{F}=1.8(^{\circ}\text{C})+32.$$

Vertical Datum

Sea Level: In this report, "sea level" refers to the National Geodetic Vertical Datum of 1929—a geodetic datum derived from a general adjustment of the first-order level nets of the United States and Canada, formerly called Sea Level Datum of 1929.

Abbreviations

°C/min	degrees Celsius per minute
dpm	disintegrations per minute (radioisotope activity)
GC-FID	gas chromatography with flame-ionization detection
GC-MS	gas chromatography with mass spectrometric identification
keV	thousand electron volt
M	molar
meq/L	milliequivalents per liter
mBq	millibecquerels
MWL	global meteoric water line
NURP	Nationwide Urban Runoff Program (established in 1978 by U.S. Environmental Protection Agency)
permil	parts per thousand
pCi/L	picocuries per liter
TU	tritium unit (1 tritium atom per 10 ¹⁸ hydrogen-1 atoms [approximately equal to 3.2 picocuries per liter])
µm	micrometer (3.937 × 10 ⁻⁵ in.)

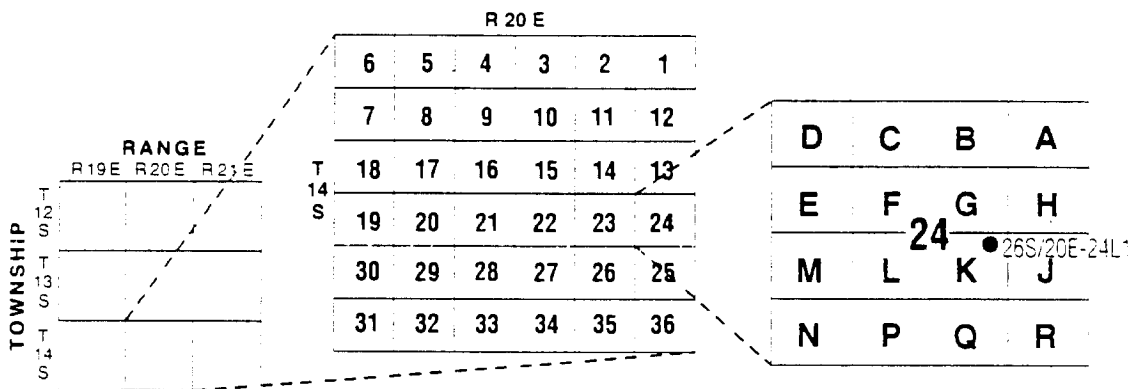
Water-Quality Information

Electrical conductivity is expressed as specific conductance, in microsiemens per centimeter at 25°C (µS/cm).

Trace-element and pesticide concentrations in water samples are given in micrograms per liter (µg/L). One thousand micrograms per liter is equivalent to 1 milligram per liter (mg/L). Micrograms per liter is approximately equivalent to "parts per billion" (ppb). Trace-element concentrations in sediment are given in micrograms per gram (µg/g). Micrograms per gram is equivalent to "parts per million" (ppm). Pesticide concentrations in sediment are given in micrograms per kilogram (µg/kg). One thousand micrograms per kilogram is equivalent to 1 microgram per gram (µg/g). Micrograms per kilogram is equivalent to "parts per billion" (ppb).

Well-Numbering System

Wells are identified and numbered according to their location in the rectangular system for the subdivision of public lands. Identification consists of the township number, north or south; the range number, east or west; and the section number. Each section is divided into sixteen 40-acre tracts lettered consecutively (except I and O), beginning with "A" in the northeast corner of the section and progressing in a sinusoidal manner to "R" in the southeast corner. Within the 40-acre tract, wells are sequentially numbered in the order they are inventoried. The final letter refers to the base line and meridian. In California, there are three base lines and meridians: Humboldt (H), Mount Diablo (M), and San Bernardino (S). All wells in the study area are referenced to the Mount Diablo base line and meridian (M). Well numbers consist of 15 characters and follow the format 014S020E24L001M. In this report, well numbers are abbreviated and written 14S/20E-24L1. Wells in the same township and range are referred to only by their section designation, 24L1. The following diagram shows how the number for well 14S/20E-24L1 is derived.



POTENTIAL FOR CHEMICAL TRANSPORT BENEATH A STORM-RUNOFF RECHARGE (RETENTION) BASIN FOR AN INDUSTRIAL CATCHMENT IN FRESNO, CALIFORNIA

By Roy A. Schroeder

Abstract

A wide variety of chemicals from urban runoff were found at elevated concentrations in sediment that accumulated in a storm-runoff recharge basin in an industrial part of the city of Fresno. The chemicals include as many as 20 inorganic elements and about the same number of organic compounds, primarily organochlorine pesticides and polycyclic aromatic hydrocarbons. Most of these contaminants were found to be sorbed to the upper 4 centimeters of sediment, which also is the maximum depth to which atmospheric lead-210 penetrated. None of the contaminants were detected above background concentrations in the sediment at depths greater than 16 centimeters. In shallow sediment, zinc is the inorganic element that was most enriched; its concentration was 38 times higher in surface sediment (0–1 centimeter) than in deeper strata (below 16 centimeters). Similarly, organic carbon enrichment was nearly 1,000 times higher in the surface sediment.

Although batch-elutriation experiments demonstrated the potential for leaching of contaminants attached to sediments, a sharp decrease in concentrations with increasing sediment depth, and the extremely low level of contaminants in two monitor wells adjacent to the basin, confirmed the absence of contaminant transport to the water table. Continued long-term protection for ground water is afforded by an approximately 8-meter-thick unsaturated zone beneath the basin. Zinc, on the basis of its hundredfold-higher concentration in water from the recharge basin than in ground water, is indicated as the most sensitive surrogate for monitoring possible ground-water degradation by inorganic cations.

Results from this study were compared with previously published results from comprehensive monitoring of urban runoff to the basin during 1981–83. The comparison revealed remarkably similar patterns of chemical contamination, although estimates of sediment quantity delivered to the basin differed by a factor of 1.8. The higher estimate of sediment quantity delivered obtained by this study likely reflects imprecision in the determination of sediment depths and an unusual open texture of the thin veneer that accumulated during the 4-year period the basin had been in existence. A technique using lead-210 is illustrated that could, with adequate data collection, provide estimates of sediment and contaminant quantity delivered to recharge basins without the numerous analyses and sample-collection effort necessary to monitor runoff.

INTRODUCTION

URBAN HYDROLOGY STUDIES

In response to concerns that urban runoff may be a significant nonpoint source of contaminants, the U.S. Environmental Protection Agency (EPA) established the Nationwide Urban Runoff Program (NURP) in 1978 to characterize the quality of urban runoff and to determine the extent to which urban runoff might contribute to impairment of water quality in receiving water bodies (U.S. Environmental Protection Agency, 1983). Studies were done at 28 urban areas throughout the Nation, and the U.S. Geological Survey (USGS) participated in 11 of the studies (Driver and others, 1985). At two study areas—in Fresno, California (Oltmann and Shulters, 1989), and Long Island, New York (Ku and Simmons, 1986)—urban stormwater runoff is routed to constructed retention/recharge basins where it then percolates through the soil and recharges the aquifer.

In the studies at Fresno and Long Island, it was concluded that properties of the soil beneath the basins are highly effective in retaining urban runoff contaminants, although the ability of the soil to continue to retain contaminants was said to be unknown (U.S. Environmental Protection Agency, 1983).

Nightingale (1975; 1987a,b,c; 1989) showed that concentrations of several trace elements and synthetic organic compounds from urban runoff were elevated above background levels in near-surface sediments in recharge basins in residential and commercial parts of the city of Fresno, but that the concentrations decreased rapidly with increasing sediment depth and also generally were very low in interstitial water from the unsaturated zone and in ground water beneath the basins. The study reported here extends these previous investigations to a recharge basin in an industrial catchment where the Fresno NURP study found most contaminant concentrations to be several times higher than in runoff from residential and commercial catchments (Oltmann and Shulters, 1989). This study thereby completes the evaluation of these recharge basins in each of the major urban land-use categories (industrial, commercial, and residential). This investigation included analyses of numerous synthetic organic compounds, a broad suite of more than 40 elements, and some stable and radiogenic isotopes. This study was done by the USGS in cooperation with the Fresno Metropolitan Flood Control District.

PURPOSE AND SCOPE

This report presents the results of a study during 1986-87 whose purpose was to determine whether stormwater runoff to a recharge basin in an industrial part of the city of Fresno has caused, or has the potential to cause, degradation of the ground-water quality. The study included three approaches:

1. Comparison of chemical concentrations in water from the recharge basin to concentrations in water from an aquifer beneath the recharge basin,
2. Determination of chemical-concentration profiles in sediment beneath the recharge basin, and
3. Laboratory elutriation tests in which water was mixed with contaminated basin sediment or clean aquifer soil and reanalyzed.

This study builds on previous results of the Fresno NURP study for which contaminants in urban runoff were monitored during 1981-83. Results of the NURP study were published by Brown and Caldwell Consulting Engineers (1984), Fresno Metropolitan Flood Control District (1987), Nightingale (1987a,b,c; 1989), Nightingale and others (1985), Oltmann and Shulters (1989), and Oltmann and others (1987).

ACKNOWLEDGMENTS

The author thanks Harry Nightingale of the U.S. Department of Agriculture in Fresno and Doug Harrison and Gerald Lakeman of the Fresno Metropolitan Flood Control District for helpful suggestions before and during the study. Removal of water from the recharge basin by the Fresno Metropolitan Flood Control District prior to sampling greatly facilitated the collection of sediment samples.

DESCRIPTION OF THE STUDY AREA

The city of Fresno is about 260 km southeast of San Francisco (about 180 km east of the Pacific coastline at Monterey Bay), in the San Joaquin Valley and between the San Joaquin River to the north and the Kings River to the south (fig. 1). Average annual rainfall in Fresno is about 25 cm, nearly all of which falls between October and April (Page and LeBlanc, 1969). Topographic relief is low with an average gradient of only about 1.6 m/km. Because of the flat terrain and the lack of adequate water courses through the city, almost 100 stormwater retention/recharge basins (hereinafter referred to as "recharge basins") have been constructed to dispose of surface runoff and to recharge an aquifer that has been designated as a sole-source aquifer by the U.S. Environmental Protection Agency. The average size of the basins is 4 to 6 ha, and the average size of the urban catchments served by the basins is about 2.6 km².

The industrial-catchment basin that was the subject of this study is in the southeast part of the Fresno metropolitan area (fig. 1). The area of the catchment is 113 ha, of which 34.2 percent is idle or vacant and 65.8 percent is used for industrial purposes. Impervious surface covers 52.5 percent of the catchment (Oltmann and Shulters, 1989). Industries present in the catchment include cotton mills, food processing, petroleum operations, chemical storage, carpet manufacturing, and light trucking (Brown and Caldwell Consulting Engineers, 1984).

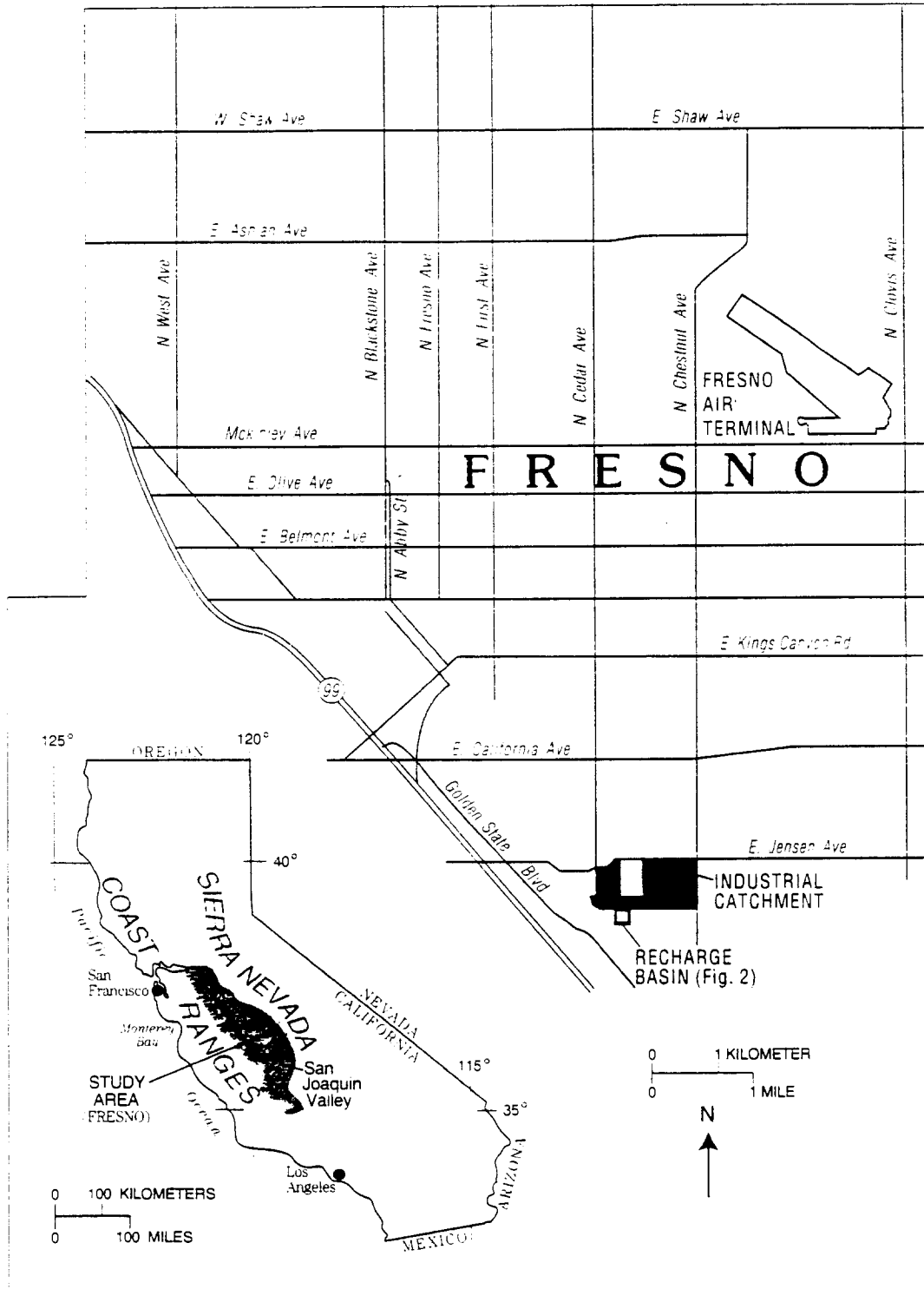


Figure 1. Location of study area.

Runoff is routed to a basin (approximately one-half hectare) on the south side of the catchment. A detailed diagram of the basin and its immediate surroundings as they existed at the time of this study (1986) is shown in figure 2. Prior to this study, the basin was last dredged (scraped) in 1982, when it was deepened and rechannelized to its present form to facilitate sample collection for the NURP study (G.E. Lakeman, Fresno Metropolitan Flood Control District, oral commun., 1992). Although excavation of the basin exposed a highly permeable coarse sand, the very fine texture of muds that soon accumulate on the basin's floor may considerably reduce the effective area through which percolation takes place by restricting it to the sloping sides of the basin at high-water stages. Over a period of time, the structure increasingly functions as an evaporation (retention) pond as opposed to a recharge basin and must be scraped to again expose a more permeable bottom.

Vegetation on the rim of the basin is sparse and the water itself contains algae but no macrophytes. The area surrounding the basin is disked periodically to maintain a generally barren environment so as not to attract wildlife. The basin normally contains water to a depth of a meter or more, even during the dry summer months; however, it was pumped dry in July 1986 to permit subbottom sampling on August 4, 1986.

GEOLOGY, HYDROLOGY, AND GROUND-WATER QUALITY NEAR THE STUDY AREA

A general description of the geology, hydrology, and ground-water quality at the study area is inferred from Page and LeBlanc's (1969) summary for Fresno and its surrounding area. Their summary, in turn, was based on data and interpretations resulting from several regional studies in the Central Valley of California (Lippincott, 1902; Mendenhall and others, 1916; Davis and Poland, 1957; Davis and others, 1959, 1964; Olmsted and Davis, 1961; Meade, 1967; and Croft and Gordon, 1968).

The study area is in the Kings River high alluvial fan geomorphic unit. The water-bearing unconsolidated deposits of this unit (and of the adjacent units)

are underlain by a basement complex of consolidated rocks of pre-Tertiary age. Depth to bedrock at the study area is about 1,200 m. The unconsolidated fine-grained continental deposits of Tertiary and Quaternary age, derived from weathering of crystalline rocks in the Sierra Nevada to the east, are overlain by coarser oxidized granitic alluvium of Quaternary age. Thickness of the alluvium is about 150 m at the study area, where it is capped by a veneer (a few meters thick, or less) of heavily reworked fine-grained soil.

Prior to development of the ground-water resources, the direction of ground-water movement in the Fresno area was toward the southwest; however, pumping has created a permanent cone of depression beneath the city of Fresno that now deflects ground-water movement toward the north in the vicinity of the study area (Mitten, 1984). A hydraulic gradient of 2 m/km toward the west-northwest is inferred at the study site from water-level contours for 1963 and regional maps that show similar conditions prevailing in 1975 (Page, 1975). Although ground water is unconfined on a regional scale, local areas of confinement or partial confinement may exist beneath intercalated clay and silt lenses.

A trend toward higher dissolved-solids concentration, and dominance of sodium and chloride ions, with increasing depth probably exists throughout the Fresno area, particularly in the older, fine-grained continental deposits. The base of the freshwater (defined as water in which ionic concentration is less than 2,000 mg/L in NaCl equivalents) is estimated to be 400 m below sea level (about 500 m below land surface) at the study area. However, nearly all wells in the area penetrate only the younger alluvium, in which the alkaline earths (calcium or magnesium) and bicarbonate are the predominant ions. Ground water from a 65-m-deep (perforated to only 51 m) agricultural production well, 14S/20E-24K1, about 0.4 km east of the study area probably is representative of regional ground-water quality; data from a 1957 analysis of water from this well are reproduced in table 1. Predominant ions are calcium, magnesium, and bicarbonate, and the dissolved-solids concentration of 226 mg/L is equivalent to a specific conductance of about 350 μ S/cm (microsiemens per centimeter).

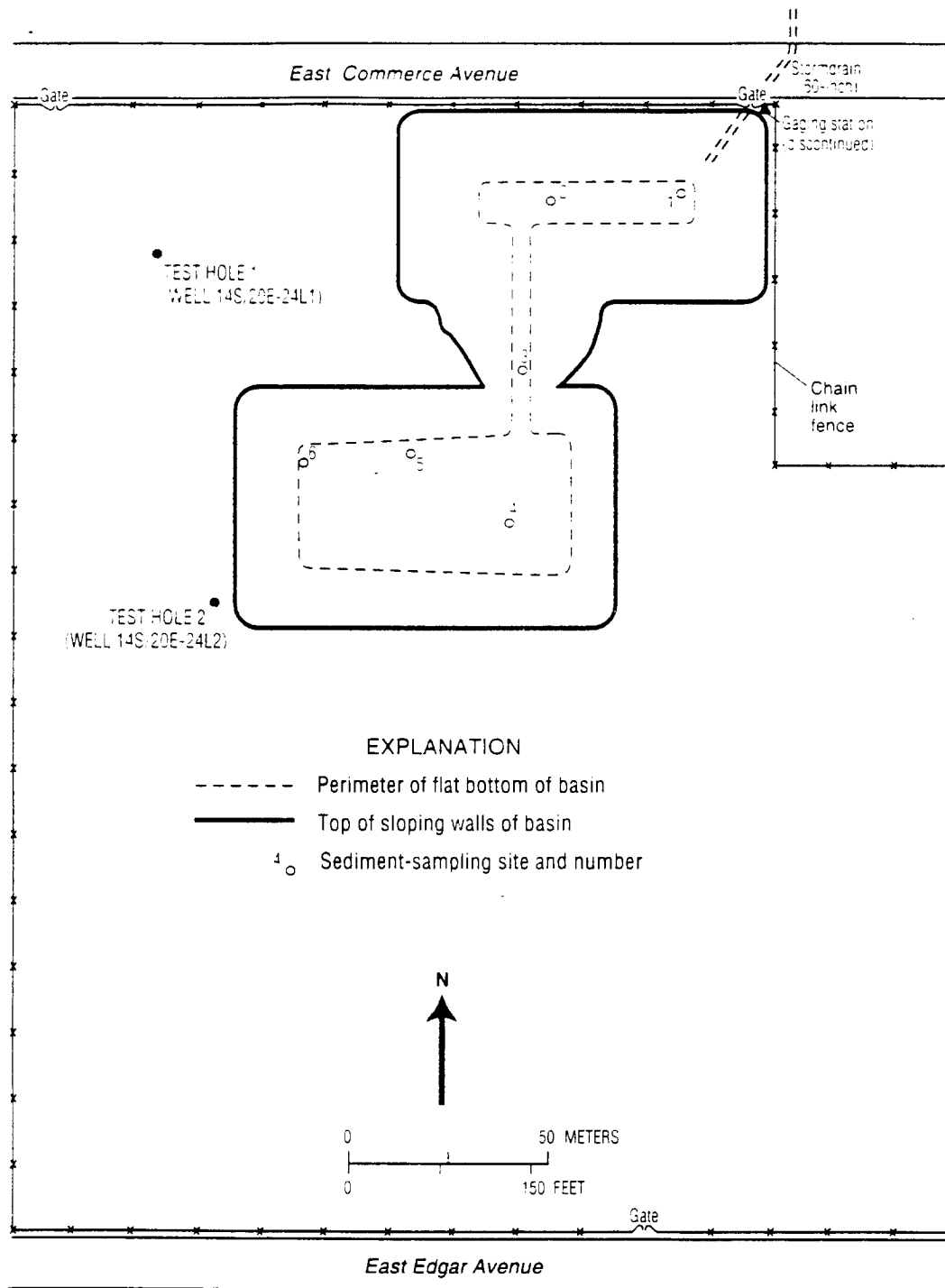


Figure 2. Recharge basin and location of test holes/monitor wells and sediment-sampling sites in the recharge basin.

Table 1. Historical ground-water-quality data from well 14S/20E-24K1 near the study site

[Sample collected September 6, 1957, by the California Department of Public Health (Page and LeBlanc, 1969, tables 14 and 20). mg/L, milligram per liter; meq/L, milliequivalent per liter; --, no data]

Constituent	Concentration	
	(mg/L)	(meq/L)
Calcium (Ca)	23	1.15
Magnesium (Mg)	12	.99
Sodium (Na)	18	.78
Potassium (K)	4	.10
Iron (Fe)	.08	--
Bicarbonate (HCO ₃)	140	2.29
Sulfate (SO ₄)	7.2	.15
Chloride (Cl)	14	.39
Fluoride (F)	.1	.01
Nitrate (NO ₃)	14	.23
Dissolved solids	226	--

pH=8.2 (standard units)

FIELD AND ANALYTICAL METHODS

SITE SELECTION, WELL INSTALLATION, AND SOIL CORING

Two test holes (fig. 2) were drilled: at latitude 36°41'54"N, longitude 119°44'57"W; and at 36°41'51"N, 119°44'56"W (about 100 m south of the first test hole). The holes were drilled with a 6-inch-diameter (15 cm) hollow-stem auger, and 2-inch-diameter (5 cm) PVC monitor wells, 14S/20E-24L1 and 14S/20E-24L2, were installed in the holes.

As auger flights were pulled up and removed following insertion of the well casing in the hollow stem, the annulus between the walls of the test hole and the casing quickly was filled with a slurry of native material from the saturated zone. During this procedure the casing sank, exposing a much longer perforated interval (6 m) to the saturated zone than was intended (1–2 m). The wet, cohesionless sand (Durnford and others, 1991) below the water table rose several meters above the water table in the annulus. The remaining annulus above this sand was backfilled with a layer of bentonite pellets, then with drill cuttings from the unsaturated zone, and finally with a 1.5-m cement seal to land surface. Altitude of land surface is 86.5 m at well 14S/20E-24L1 (test hole 1) and 85.5 m at well 14S/20E-24L2 (test hole 2). Both monitor wells are perforated (slotted) from

12 to 18 m below land surface. Because the casing sank during installation, the top of the perforated interval was only about 0.6 m above the water table at well 14S/20E-24L1 and was actually about 0.3 m below the water table at well 14S/20E-24L2 when samples for water-quality analysis were collected during summer and autumn 1986.

Both monitor wells are adjacent to the recharge basin (fig. 2) and, on the basis of regional ground-water-flow patterns discussed in the previous section, the first well installed (14S/20E-24L1) is believed to be downgradient of the basin. Irregular topography at this location precluded drilling the first test hole closer than about 50 m from the recharge basin. Flatter terrain to the south permitted drilling of the second test hole within a few meters of the edge of the recharge basin, thereby increasing the probability of intercepting any small ground-water mound should one extend only a short distance beyond the basin area. The second well also decreases the chance that a contaminant plume would be missed if the local hydraulic gradient were toward the southwest (the direction of the regional gradient prior to pumping in Fresno) for any reason, including pumping from any wells near the study area itself (see Mitten and Guay, 1986, for a summary of municipal pumpage and location of wells). Although a monitor well upgradient of the basin would have provided information about the local hydraulic gradient and confirmation that water from the basin is reaching the water table and producing a ground-water mound, this study did not have sufficient resources to install a third well. Installation of lysimeters beneath the recharge basin itself, as in a previous study, would have been especially valuable but was precluded by concern that they would provide a pathway for enhanced transport of contaminants through the unsaturated zone.

Soil cores were recovered from selected depths during drilling of the test holes by lowering a 2-ft (0.6 m), split-spoon steel corer through the hollow drill stem. The depths selected for coring in test hole 1 were based on field observations of change in texture as determined from drill cuttings brought to the surface by the auger and from the drill operator's perception of changes in drilling characteristics. Depths for coring in test hole 2 were chosen to confirm the presence of low-permeability zones found in test hole 1. Particle-size distribution in the core soils was determined using sieves for the sand-size fraction and hydrometers for the fine fraction using methods described by Guy (1969). The data are given in table 2.

Table 2. Particle-size distribution in soil cores from two test holes adjacent to the recharge basin

[m, meter; --, no data]

Depth interval (m)	Percent finer than size, in millimeters, indicated											
	Gravel	Coarse sand		Medium sand	Fine sand			Silt			Clay	
	4	2	1	0.5	0.25	0.125	0.062	0.031	0.016	0.008	0.004	0.002
Test Hole 1 (Monitor Well 14S/20E-24L1)												
1.2-1.8	95	94	81	79	59	46	35	26	18	10	6	5
3.1-3.7	--	100	98	74	31	16	10	7	5	3	2	1
4.9-5.5	--	100	95	52	22	12	9	6	4	3	2	2
5.8-6.2	--	--	100	99	98	95	80	50	25	9	8	7
6.2-6.4	--	--	100	85	46	30	21	14	9	6	5	4
6.4-6.7	--	100	98	83	72	60	36	26	19	13	8	8
6.7-7.2	--	--	100	99	92	72	34	20	12	5	4	4
7.2-7.3	--	--	--	100	99	95	87	73	51	35	20	12
9.1-9.8	--	--	100	99	92	59	13	6	5	3	3	3
10.7-10.9	--	--	100	99	95	90	82	73	56	36	19	12
10.9-11.3	--	--	100	96	83	67	48	39	32	25	20	16
11.6-12.2	--	100	99	93	71	53	38	28	21	17	15	12
13.4-14.0	--	--	100	89	70	53	37	29	23	16	12	9
15.5-15.8	--	--	100	92	74	61	51	42	33	23	16	13
16.5-17.1	--	100	99	76	36	21	14	10	6	4	3	2
Test Hole 2 (Monitor Well 14S/20E-24L2)												
6.4-6.5	--	--	100	98	93	86	71	51	31	14	9	9
9.7-9.8	--	--	100	98	93	85	65	50	42	34	22	12
10.1-10.4	--	100	99	91	70	51	35	25	20	15	12	9
10.4-10.7	--	--	100	95	79	66	58	49	40	30	21	16
14.4-14.6	--	100	99	95	84	78	72	64	50	30	13	7

Particle-size data were used in conjunction with field observations to compile the lithologic logs given in table 3. Depths were recorded with a precision of 0.03 m during drilling; however, they were measured relative to land surface that was highly irregular because soil excavated to form the recharge basin had been deposited near the test holes. Therefore, accu-

racy of the logs relative to a fixed benchmark probably is no better than 0.3 m.

Although changes in color and texture commonly are gradual and subtle, three distinct zones of silty clay were found and correlated between the test holes: at 7.2-7.3 m, 10.7-10.9 m, and 15.5-15.8 m in test

Table 3. Geologic log for test hole 1 adjacent to the recharge basin

[Rock-color terms are from Goddard and others, 1948. m, meter; mm, millimeter]

Depth (m)	Thickness (m)	Description
0.6	0.6	Fine sand, silt, and clay in about equal proportions, moderate-yellowish-brown (10 YR 5/4). Surface soil heavily reworked and mixed with soil excavated in construction of recharge pond.
2.5	1.9	Sand, fine to coarse, silty, moderate-yellowish-brown (10 YR 5/4) tending to dark-yellowish-orange (10 YR 6/6). Numerous small concrete chips decreasing in abundance with depth, thin (1 mm) calcareous stringers and bands.
5.8	3.3	Sand, fine to coarse, pale-yellowish-brown (10 YR 6/2) with some moderate-yellowish-brown (10 YR 5/4) marbling. Biotite, quartz, and mica grains abundant. Measured 10-percent silt and 90-percent sand. Becoming finer below 5.6 m.
6.2	.4	Silt, fine sand, pale-yellowish-brown (10 YR 6/2) with numerous small (1-3 mm) light-brown (5 YR 5/6) blebs. Measured 70-percent silt.
6.4	.2	Sand, fine to medium, silt, pale-yellowish-brown (10 YR 6/2). Measured 15-percent silt.
7.2	.8	Sand, fine, silty, moderate-yellowish-brown (10 YR 5/4). Measured 30-percent silt.
7.3	.1	Silt, clayey, moderate-yellowish-brown (10 YR 5/4) tending to dark-yellowish-brown (10 YR 4/2). Measured 70-percent silt and 20-percent clay.
10.7	3.4	Fine sand, some silt, color fluctuating between moderate-yellowish-brown (10 YR 5/4) and light-brown (5 YR 5/6) throughout interval. Measured 80-percent fine sand and 10-percent silt.
10.9	.2	Silt, clayey, pale-yellowish-brown (10 YR 6/2) to dark-yellowish-brown (10 YR 4/2). Measured 60-percent silt and 20-percent clay.
13.0	2.1	Sand, fine to medium, with silt and clay, poorly sorted, light-brown (5 YR 5/6) blebs at top of interval blending entirely into moderate-brown (5 YR 4/4) matrix below 11.6 m. Clay and silt decreasing and fine sand increasing slightly with depth. Measured 15-percent clay and 20- to 30-percent silt. Bottom of interval in saturated zone.
15.5	2.5	Sand, fine to coarse, with silt and clay, (slightly coarser than interval above), changing gradually from moderate-yellowish-brown (10 YR 5/4) at top of interval to light-brown (5 YR 5/6) at bottom of interval.
15.9	.4	Sand, silty with greyish clay blebs, moderate-yellowish-brown (10 YR 5/4). Measured 35-percent silt and 15-percent clay.
18.3	2.4	Sand, fine to coarse, becoming coarser with increasing depth, moderate-yellowish-brown (10 YR 5/4) tending to light-brown (5 YR 5/6). Biotite, quartz, and mica grains abundant (similar to 2.5-5.8 m interval). Measured 10-percent silt.

¹Finer, less-permeable zones visually identified during coring.

hole 1; and at 6.4-6.5 m, 9.7-9.8 m, and 14.4-14.6 m in test hole 2 (table 4). Difference in the depth below land surface of these three zones matches closely the difference in land-surface altitude at the test holes. The water table was between the middle and deepest of these zones in each test hole.

SAMPLING OF SEDIMENT FROM THE RECHARGE BASIN

To facilitate sampling of accumulated sediment in the recharge basin, all water was pumped from the basin in mid-July 1986, and the exposed sediment was allowed to dry for about 3 weeks prior to sampling. As the fine-grained sediment covering the floor of the basin dried, shrinkage cracks formed thin polygonal plates. The plates were about 1 cm thick (although actual thickness is somewhat imprecise because the bottom surface of the plates was irregular) and represent the most recently deposited sediment that accumulated in the recharge basin.

Removal of the plates (the 0- to 1-cm depth samples) exposed an unusual geometric surface that bore a striking resemblance to a honeycomb, the individual cells of which were several millimeters across and deep. The sediment was strongly reduced, as indicated by its black color, except for a thin veneer of brown (oxidized) clay rimming the edge of each cell wall. It is suggested that the cells were formed by escape of gas bubbles as the mud was drying. Gas bubbles, accompanied by a small oil sheen, commonly were seen breaking the water's surface when the basin was flooded. This gas is presumed to be hydrogen sulfide, carbon dioxide, methane, and (or) nitrogen generated by microbial activity within the sediments. After the basin's water had been removed, the underlying sediment was exposed to air entering through cracks at the sediment surface, and oxidation produced the thin brown rind on the cells.

The 1-cm-thick surface-sediment plates were removed for analysis from six sites on the basin floor. The basin floor slopes gently downward from the

Table 4. Sand, silt, and clay content of soil cores from two test holes adjacent to the recharge basin

[Contents, in weight-percent, calculated from data in table 2. m, meter]

Depth interval (m)	Coarse sand	Medium sand	Fine sand	Silt	Clay
Test Hole 1 (Monitor Well 14S/20E-24L1)					
1.2-1.8	15	20	24	29	6
3.1-3.7	26	43	21	8	2
4.9-5.5	48	30	13	7	2
5.8-6.2	1	1	18	72	8
6.2-6.4	15	39	25	16	5
6.4-6.7	17	11	36	28	8
6.7-7.2	1	7	58	30	4
¹ 7.2-7.3	0	1	12	67	20
9.1-9.8	1	7	79	10	3
¹ 10.7-10.9	1	4	13	63	19
10.9-11.3	4	13	35	28	20
11.6-12.2	7	22	33	23	15
13.4-14.0	11	19	33	25	12
¹ 15.5-15.8	8	18	23	35	16
16.5-17.1	24	40	22	11	3
Test Hole 2 (Monitor Well 14S/20E-24L2)					
¹ 6.4-6.5	2	5	22	62	9
¹ 9.7-9.8	2	5	28	43	22
10.1-10.4	9	21	35	23	12
10.4-10.7	5	16	21	37	21
¹ 14.4-14.6	5	11	12	59	13

¹Finer, less-permeable zones visually identified during coring.

northeast corner, where the storm runoff enters the basin, toward the south (fig. 2), therefore, a spatial gradation of increasing fines toward the south was anticipated. The particle-size data from the six sites that are presented in table 5 provide some evidence for such a trend, although the difference in percentage of fine fraction is small. Clay-plus-silt content ranges from 67 percent at sediment-sampling site 2 to 77 percent at site 4; however, clay contents are about 10 percent greater at sites 5 and 6 in comparison with the other four sites. In addition to the 0-1 cm (surface-sediment plate) samples, two successive thin layers, estimated to each be only about 1 mm thick, were scaped from the surface near site 5. Both were found to contain approximately equal proportions of clay and silt and virtually no sand (table 5). Accordingly, the small differences in the percentage of fine fraction between surface samples at the six sites could simply be due to varying proportions from the very thin layer of fine sediment at the surface (top few millimeters) and the coarser sediment immediately below the surface (to a depth of 1 cm).

In addition to surface samples from the five sites, sediment and native soil to a depth of 1.5 m were collected at site 5 in the recharge basin. This site, which seemed to be the deepest part of the basin, was 4.9 m below the top of the cement seal at well 14S/20E-24L1 (test hole 1). Successive layers of sediment were exposed and collected at site 5 by excavating a rectangular pit with a small hand trowel. The pit was "telescoped" inward in a staircase arrangement to eliminate sloughing of overlying sediment as successive layers were uncovered. Excavation continued in this manner to a depth of 64 cm; the deepest sample, from 120 to 150 cm, then was obtained with a manually operated 2-inch-diameter (5 cm) steel auger. Although the color of sediment samples lightened considerably with increasing depth, even the deepest sample had a grayish-tan color indicative of an anaerobic environment.

The particle-size-distribution data (table 5) for the subbottom-deposit samples collected at site 5 show the existence of a shallow zone, from the sediment surface to a depth of 8 cm, in which silt and clay predominate, and a deeper zone, below 16 cm, in which sand predominates (with little silt and clay present). The interval between 8 and 16 cm is a transition zone between the overlying sediment derived from outside the basin, presumably brought in by runoff from the catchment, and the underlying natural alluvium referred to throughout this report as "soils" or "substrata."

On the basis of the difference in altitude between test hole 1 and basin sediment-sampling site 5, one would expect the lithology of the basin deposits immediately below 16 cm to be similar to that of the test hole between 4.9 and 5.5 m. However, minor differences in particle-size distribution and in element concentrations suggest that the basin deposits below 16 cm more closely resemble those from 5.5 to 5.8 m in test hole 1 (for which chemical data exist but particle-size data do not). Specifically, the basin deposits below 16 cm consist predominantly of a fine-to-medium sand, whereas the soil from 4.9 to 5.5 m in test hole 1 consists predominantly of a coarse-to-medium sand (compare data in tables 2 and 5). In addition, comparison of selected element (Fe, Mg, Mn, Th, Ti, and V) concentrations (data given in tables 7 and 9 and discussed in later sections of this report) shows that basin deposits below 16 cm are closer in chemical composition to soils between 5.5 and 5.8 m than to soils between 4.9 and 5.5 m in test hole 1. It should be noted, however, that chemical differences between the two soil samples are small and that the element arsenic presents a pattern opposite to that of the aforementioned six elements.

CHEMICAL ANALYSIS OF WATER, SOIL, AND SEDIMENT

A Teflon bailer with Teflon ball-check valve was used to collect ground-water samples from the two monitor wells installed for this study. Aliquots for analysis of inorganic constituents and nutrients were filtered through a membrane with 0.45- μ m pores. Aliquots for analysis of gross organic indicators (such as methylene blue active detergents, tannin plus lignin, and oil and grease) and specific organic compounds were not filtered. Samples for analysis of organic compounds were stored at temperatures near 0°C and analyzed within a few days of their collection. Samples for analysis of cations and trace elements were stabilized by addition of nitric acid to pH approximately 2 and nutrients were preserved by addition of mercuric chloride and storage at temperatures near 0°C. Samples for trace-element analysis in the low-concentration (sub-ppb) range were acidified with ultrapure nitric acid and stored in Teflon bottles.

Grab samples of water from the recharge basin, and samples from the laboratory elutriation study (discussed in detail later in the "Laboratory Elutriation Experiments" section) were processed and analyzed in a manner similar to that used for the ground-water samples, except that collection and sample-preservation techniques for low-level (sub-ppb) trace-

Table 5. Particle-size distribution in sediment beneath the recharge basin

[cm, centimeter; ~, interval is approximate. --, no data]

Site No.	Percent finer than size, in millimeters, indicated											
	Gravel		Coarse sand		Medium sand	Fine sand		Silt			Clay	
	4	2	1	0.5	0.25	0.125	0.062	0.031	0.016	0.008	0.004	0.002
	Surficial (0-1 cm) Sediment from Recharge Basin											
1	--	--	100	97	92	84	70	51	35	21	19	17
2	--	--	100	98	92	84	67	49	30	16	14	13
3	--	--	100	97	86	79	70	65	46	26	16	14
4	--	--	--	--	100	90	77	68	53	33	19	11
5	--	--	--	100	95	80	71	68	58	44	27	17
6	--	--	--	100	87	75	69	66	58	42	25	14
	Subbottom Sediment from Site 5 in Recharge Basin											
Depth interval (cm)												
~0-1	--	--	--	--	100	99	98	96	88	71	50	34
~1-2	--	--	--	100	99	99	97	91	81	68	51	35
0-1	--	--	--	100	95	80	71	68	58	44	27	17
1-2	--	--	--	100	84	72	65	61	51	37	20	15
2-4	--	--	--	100	94	87	70	56	39	21	13	13
4-8	--	--	--	100	93	87	80	72	48	25	15	12
8-16	--	100	99	97	89	80	53	40	20	7	5	5
16-32	--	--	100	96	75	42	9	3	1	1	1	1
32-64	--	--	100	96	60	29	7	3	1	1	1	1
120-150	--	--	100	96	69	23	5	2	1	1	1	1

element samples described above were used for only some of the samples (readily recognized in data tables 6, 12, and 13 by the reporting of significant digits to the tenths microgram-per-liter (sub-ppb) level for several trace elements). Solutions from the elutriation test were centrifuged rather than filtered.

Aqueous chemical constituents were analyzed by the USGS National Water Quality Laboratory in Arvada, Colorado, using methods described by Wershaw and others (1987) and Fishman and Friedman (1989). Tritium initially was analyzed at the University of California, San Diego, and these analyses later were confirmed at the USGS Isotope Laboratory in Reston, Virginia, using a liquid-scintillation counting method. Stable isotopes of hydrogen and oxygen also were analyzed in the Reston laboratory.

Sediment and soil samples for determination of organic constituents were stored in glass jars at temperatures near 0°C until their analysis a few days after collection, or they were frozen for later use in the laboratory elutriation tests. Organic constituents and nutrients were analyzed by the USGS National Water Quality Laboratory using methods described by Wershaw and others (1987) and Fishman and Friedman (1989). Groups of organic compounds that were analyzed include the organochlorines, organophosphorous insecticides, chlorophenoxy-acid herbicides, and extractable priority pollutants. In addition to these specific compounds, extractable nonspecific organic compounds were analyzed by gas chromatography with flame ionization detection (GC-FID).

The GC-FID scans provide only a general indication of organic contaminant concentrations because total concentrations determined by GC-FID amount to only about 1 percent of total organic carbon (analyzed by combustion) in near-surface sediments from the basin. For GC-FID analysis, a 25-g (approximate dry weight) subsample of sediment was extracted with a methanol plus methylene chloride solvent mixture for 12 hours using continuous Soxhlet extraction. The extract then was concentrated using a Kuderna-Danish apparatus. Surrogate compounds were added prior to extraction to check recovery, and perdeuterionaphthalene ($C_{10}D_8$) was added as an internal standard to the extract just prior to GC-FID analysis. Chromatographic separation was done using a 25-m by 0.21-mm SE-54 fused silica capillary column held initially at 50°C for 5 minutes then increased to 300°C at 6°C/min. Concentrations were calculated relative to the response of the internal standard (as " $C_{10}D_8$ equivalents").

For the gas chromatography-electron impact mass spectrometric (GC-MS) analysis of two sediment samples (0-1 cm and 8-16 cm depths) from site 5, approximately 25 g (dry weight) of sediment was processed and chromatographed in the same way as for GC-FID analysis. However, the chromatography column was coupled directly to a mass spectrometer, which was set to analyze from 45-450 atomic mass units with a scan time of 0.25 second. Compound identifications were based on comparison of the gas chromatographic peak retention time and mass spectrum with similar data for standard compounds. In addition to priority pollutants, a computerized library search for nontarget compounds was done. Although many compounds were detected, especially in the surface sample, only a few could even be identified by the compound class to which they belong.

Analysis of soil and sediment samples for element concentrations was done in the USGS Analytical Facility in Denver using methods (Severson and others, 1987) identical to those also used for a multisite reconnaissance of irrigation drainage effects that was done concurrently with this study (see, also, Schroeder and others, 1988, and Arbogast, 1990, for details on analytical methods). The samples were prepared for analysis by drying at about 30°C and grinding to pass an 80-mesh (180 μ m) sieve. Virtually complete mineral digestion was effected with a combination of strong acids. Inductively coupled argon-plasma atomic emission spectrometry (ICP) was used to analyze a broad suite of about 40 elements. Arsenic and selenium were analyzed using hydride-generation atomic absorption. Mercury was analyzed using flameless cold-vapor atomic absorption. Boron was analyzed on a hot-water extract. Uranium and thorium were analyzed using delayed-neutron activation.

Lead-210 (^{210}Pb) and radium-226 (^{226}Ra) were analyzed in 13 sediment samples from the recharge basin and in 2 soil samples from test hole 1. J.K. Cochran, at the State University of New York, Stony Brook, analyzed these samples using a non-destructive gamma-counting technique. About 90 to 150 g dried sediment (or soil) was sealed in 100 cm^3 aluminum cans and stored for 2 weeks to allow gaseous radon-222 (^{222}Rn) to reach decay equilibrium with ^{226}Ra . Gamma emissions were counted on a 1,000 mm^2 , 10-mm-thick, Canberra Electronics planar germanium detector. The 46.5 keV (thousand electron volt) ^{210}Pb peak and the 352 keV ^{214}Pb peak were used to measure ^{210}Pb and ^{226}Ra activity, respectively. Counting time was about 24 hours. Detector efficiencies were determined using laboratory as well as National Bureau of Standards sediment standards.

Corrections for sample self-absorption of gamma rays were made by measuring the attenuation of ^{210}Pb and ^{226}Ra sources placed on the sample cans. Concentrations are reported in disintegrations per minute per gram dry sediment (dpm/g) with precision based on counting statistics.

QUALITY OF RECHARGE-BASIN WATER AND GROUND WATER

The most direct way of determining whether recharge from the study basin has impaired ground-water quality is to measure chemical concentrations in the basin sediment and concentrations in nearby ground water. Nightingale (1975, 1987a,b,c, and 1989) found that several trace elements and synthetic organic compounds were strongly sorbed by soils beneath five recharge basins in residential and commercial parts of the city of Fresno and that their concentrations were low in soil moisture from the unsaturated zone and in ground water immediately below a basin. Accordingly, although contaminant levels in sediment were expected (and found) to be higher at the recharge basin in the commercial catchment that was the subject of this study, little or no effect on ground-water quality was anticipated.

An important consideration in evaluating the potential for ground-water contamination by urban-runoff chemical constituents is establishing whether runoff in the basin actually reaches the water table below the basin in significant amounts. Measurement of soil-moisture content in the unsaturated zone and installation of several monitor wells to determine the local hydraulic gradient would provide direct evidence of recharge but were beyond the resources available for this study. Nevertheless, water-budget calculations indicate that the potential for substantial recharge exists. On the basis of mean annual rainfall of 26 cm, measured rainfall-runoff coefficient, soil-water capacity, and physical dimensions of the catchment and recharge basin (Oltmann and Schulters, 1989), mean annual runoff should be sufficient to saturate soil above the water table for a distance of about 100 m beyond the basin's boundaries. Of course, this simplified calculation assumes symmetrical spreading beneath the recharge basin. On the basis of saturated hydraulic conductivities calculated from measured soil characteristics, Nightingale (1989) concluded that subsurface silt and clay strata (lenses) would cause lateral movement of water beneath a Fresno recharge basin that had a measured infiltration rate of 16 cm/d (almost 60 m/yr). If the soil deposits in the recharge area above the lenses are assumed to be isotropic and

homogeneous and the recharge rate is constant, then the radius of the area infiltrated must vary inversely with the square root of vertical hydraulic conductivity (Freeze and Cherry, 1979, p. 16). On the basis of this relation and the inferred variation in hydraulic conductivities (Nightingale, 1989), the water would be expected to spread laterally a distance of about three times the basin's diameter. The likelihood that recharge reaches the water table directly beneath and near (as opposed to some great distance from) the basin is inferred on the basis of arguments given above, along with the absence of significant lithologic discontinuities at the test holes.

Additional evidence that ground water at the water table has mixed with a substantial volume of water from the recharge basin was obtained from analysis of tritium and the stable isotopes of hydrogen and oxygen.

ISOTOPIC EVIDENCE FOR RECHARGE

Tritium, the radioactive isotope of hydrogen having a half-life of 12.43 years, is produced naturally by cosmic-ray interactions with the upper atmosphere. Atmospheric detonation of thermonuclear weapons produced large quantities of tritium with the result that tritium concentrations in precipitation began increasing above natural levels about 1950, reaching a peak that was 10^2 - 10^3 times natural levels in the mid-1960's, and then gradually declining following the international treaty that banned atmospheric testing by the major nuclear powers. Tritium in precipitation today remains several times higher than natural levels. It is this bomb-produced tritium transient that can be used to distinguish post-1950 water from older water.

On the basis of nationwide maps of tritium deposition (Michel, 1989), it is estimated that average annual tritium concentration in precipitation at the city of Fresno was slightly greater than 10 TU (tritium units) at the time of this study. (One tritium unit is one tritium atom per 10^{18} hydrogen-1 atoms and equals approximately 3.2 picocuries per liter.) On the basis of physical properties of the aquifer (Williamson and others, 1989, and Hugh Mitten, USGS, written commun., 1986), it is concluded that regional ground water in the city of Fresno that originates as recharge in the Sierra Nevada foothills is unlikely to contain any tritium. The measured tritium concentration of 10.2 TU in the recharge basin (table 6) is close to the estimated tritium concentration in local precipitation. It is presumed that the

Table 6. Concentrations of dissolved inorganic and total-recoverable organic constituents in water from the recharge basin and from two nearby monitor wells

[Water from recharge basin collected January 29, 1986, except sample for analysis of isotopes, mercury, and bromide collected July 11, 1986. Monitor wells sampled August 4, 1986. MBAS, methylene blue active surfactants; PCBs, polychlorinated biphenyls; $\mu\text{S}/\text{cm}$, microsiemen per centimeter at 25° Celsius; °C, degrees Celsius; mg/L, milligram per liter; $\mu\text{g}/\text{L}$, microgram per liter; permil, parts per thousand; --, no data; <, less than indicated reporting limit]

Constituent	Concentration, in indicated units		
	Recharge basin	Well 14S/20E-24L1	Well 14S/20E-24L2
Major Constituents			
Temperature (°C)	--	17	17
Specific conductance ($\mu\text{S}/\text{cm}$)	360	420	379
pH (standard units)	6.9	8.1	8.1
Calcium (mg/L)	13	22	34
Magnesium (mg/L)	5.3	11	16
Sodium (mg/L)	14	53	15
Potassium (mg/L)	18	3.2	3.4
Chloride (mg/L)	23	10	7.4
Sulfate (mg/L)	32	24	26
Fluoride (mg/L)	.1	.1	.1
Silica (mg/L as SiO_2)	5.4	49	49
Alkalinity (mg/L as CaCO_3)	13	171	--
Nutrients			
Nitrite + nitrate (mg/L as N)	4.0	4.0	4.8
Ammonium (mg/L as N)	7.0	.02	.02
Kjeldahl nitrogen (mg/L as N)	13	.6	.7
Orthophosphate (mg/L as P)	4.3	.04	.06
Organic carbon (mg/L as C)	--	3.9	3.9
Trace Elements ($\mu\text{g}/\text{L}$)			
Arsenic (As)	3	3	2
Barium (Ba)	--	100	100
Boron (B)	90	80	110
Bromine (Br)	<10	<10	<10
Cadmium (Cd)	1	.73	1.1
Chromium (Cr)	<1	.50	.35
Cobalt (Co)	3	1.0	1.5
Copper (Cu)	7	3.5	4.5
Iron (Fe)	210	16	48
Lead (Pb)	2	<.08	<.08
Manganese (Mn)	120	210	78
Mercury (Hg)	.1	<.1	<.1
Molybdenum (Mo)	10	15	5
Nickel (Ni)	4	6.8	11
Selenium (Se)	<1	<1	<1
Silver (Ag)	<1	<.03	<.03
Strontium (Sr)	170	170	220
Vanadium (V)	--	19	15
Zinc (Zn)	130	.5	.4
Isotopes			
δD (permil)	-16.0	-51.5	-49.5
$\delta^{18}\text{O}$ (permil)	1.19	-7.20	-7.05
^3H (tritium units)	10.2±.8	6.4±.7	5.8±.6

Table 6. Concentrations of dissolved inorganic and total-recoverable organic constituents in water from the recharge basin and from two nearby monitor wells--*Continued*

Constituent	Concentration, in indicated units		
	Recharge basin	Well 14S/20E-24L1	Well 14S/20E-24L2
Broad Organic Indicators			
Tannin and lignin (mg/L as tannic acid)	2.8	<1	<1
Phenols (µg/L as phenol)	5	3	--
MBAS detergents (mg/L)	.29	.11	.08
Synthetic Organic Compounds (µg/L)			
1,2-Dichloroethane	<3	<.2	.2
1,1,1-Trichloroethane	<3	.2	.2
Dichlorodifluoromethane	<3	.2	<.2
Lindane	.01	<.001	<.001
Endosulfan	<.01	.002	<.001
Malathion	.01	<.01	<.01
Parathion	.01	<.01	<.01
Diazinon	<.01	.01	<.01
Purgeable priority pollutants	<3	<.2	<.2
Base/neutral-extractable priority pollutants	(¹)	--	--
Acid-extractable priority pollutants	(¹)	(¹)	(¹)
Organochlorine insecticides + PCB's	<.01	<.001	<.001
Organophosphorus insecticides	<.01	<.01	<.01

¹Value is less than reporting limit, which is 5 µg/L for most extractable organic priority pollutants.

basin contains a mixture of ground water used by industries in the catchment area and not discharged to a municipal treatment plant in addition to precipitation runoff, and that it is the precipitation runoff that was the principal source of tritium in the sample from the recharge basin collected January 29, 1986.

Tritium concentrations of about 6 TU in both monitor wells (table 6) seem to indicate significant recharge from the basin. However, the evidence is compromised by the recent finding of tritium in many wells throughout the San Joaquin Valley, including 46 TU in well 14S/20E-34G1, a 124-foot well about 3 km southwest of the study site (Shelton and Miller, 1991). Recharge by extensive irrigation with surface water likely is the source of tritium in ground water throughout the San Joaquin Valley; in the city of Fresno, artificial recharge also is a potential source of tritium.

Stable-isotope ratios of hydrogen (²H/¹H or D/H) and oxygen (¹⁸O/¹⁶O) also can be used to infer

recharge and possibly to place constraints on the timing of such recharge from the basin. Both isotope ratios are expressed here as permil in the commonly used δ (delta) notation relative to the Vienna-Standard Mean Ocean Water (V-SMOW) standard. The general relation between hydrogen- and oxygen-isotope ratios in precipitation is shown by the global meteoric-water line (MWL) in figure 3 (Craig, 1961). This line has a slope of 8, and the isotopic composition of waters that have undergone evaporation typically fall to the right of this line; that is, they are enriched in ¹⁸O relative to deuterium. This deviation from the MWL, as well as relatively high values of δD and δ¹⁸O, indicates that water in the recharge basin at the time of sampling for stable isotopes on July 11, 1986, was highly evaporated.

The hydrogen-isotope ratio of precipitation in the city of Fresno can be inferred, from regional-scale data along an east-west traverse between the Pacific Coast and Nevada, to be about -80 permil (Ingraham and Taylor, 1991). Ground water from most wells in

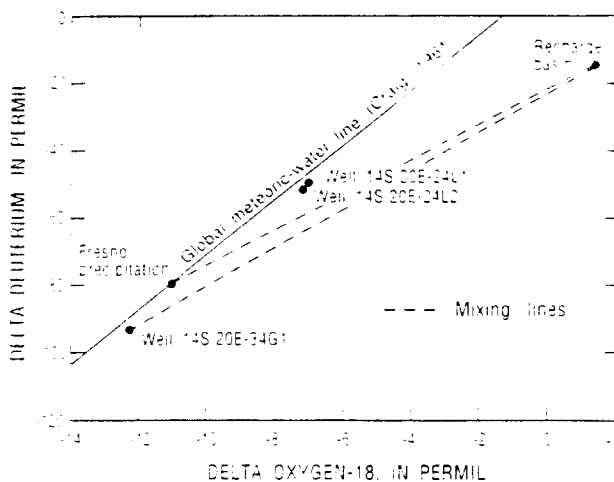


Figure 3. Relation between hydrogen and oxygen stable-isotope ratios in recharge-basin water and nearby ground water.

the vicinity of the city of Fresno has a δD slightly less than -90 permil. Even the isotopic composition of water from well 14S/20E-34G1 (alluded to in the earlier discussion on tritium), at $\delta D = -91.5$ permil and $\delta^{18}O = -12.3$ permil, does not differ greatly from the composition expected for ground water free of local recharge by surface water or irrigation—apparently because the local recharge has not undergone substantial evaporation. As can be seen in figure 3 (dashed lines), there is no way the isotopic composition measured in the two monitor wells can be obtained by mixing regional ground water or local precipitation with recharge-basin water in varying proportions. On the basis of this departure from a simple two-member mixing model, it is concluded that one or more of the following conditions are necessary to explain the isotopic composition found in the monitor wells:

1. The δD of precipitation may have been higher (less negative) at the time of this study than the inferred estimate of -80 permil.
2. The D and ^{18}O contents of regional ground water at the study site may be higher than represented by well 14S/20E-34G1. D and ^{18}O contents are reported to be higher in some wells several kilometers from the study area (Shelton and Miller, 1991).
3. Recharge in the basin might be occurring during periods when the basin's water has undergone less evaporation, such as during the winter; as a result, 1H and ^{16}O concentrations

ordinarily would be considerably higher (2H and ^{18}O would be lower) than during summer when the basin was sampled. It is reasonable that recharge should be much greater following winter storms when the basin was observed by the author to be fuller than during the dry summer season when it contains less water.

Although the relative importance of each of the above explanations cannot be evaluated without frequent monitoring over a period of time, the fact that δD and $\delta^{18}O$ values in the monitor wells generally are less negative than in other wells in the Fresno area argues strongly that recharge from the basin is reaching the water table locally.

The stable-isotope data, together with the tritium data, suggest that basin recharge is comparable in importance at the study site to regional ground-water flow from the Sierra Nevada foothills, so that any storm-runoff contaminants not effectively removed by the soil beneath the basin should be easily detectable by means of shallow monitor wells such as those that were installed for this study. Nevertheless, the sparse isotope data presented do not support quantitatively apportioning ground water at the monitor wells between regional flow and basin recharge. Apart from the limitations discussed above, seasonal and annual isotope variations in precipitation are likely to be quite large. For example, mean annual tritium concentration in precipitation during 1980-81 is nearly double that of the near-record wet period of 1982-83 (Michel, 1989). Also, mean annual δD in precipitation was found to vary as much as 40 permil between 1981 and 1989 at some sites in a monitoring network that included 32 locations in southeastern California and adjacent parts of Nevada and Arizona (Friedman and others, 1992).

MAJOR IONS AND TRACE ELEMENTS

Comparison between water samples from the recharge basin and the monitor wells (table 6), and comparison of both with regional ground-water composition listed in table 1, reveals little difference in such broad measures of water quality as dissolved-solids concentration and specific conductance. For chloride, which is an ideal unreactive tracer (as are isotopes), differences are larger but still too small to be useful. Although differences are large for silica and bicarbonate (alkalinity), precipitation and dissolution reactions make both unsuitable as tracers. Of the major ions, potassium (K) is the most useful indicator of whether cationic contaminants in the re-

charge basin reach the water table. Aqueous potassium concentrations are about 5 times higher in the recharge basin than in either regional ground water (table 1) or the nearby monitor wells (table 6)—indicating that there is no discernible transport of this cation to the water table. Profiles of dissolved- and exchangeable-cation concentrations beneath a recharge basin in the city of Fresno show that removal of dissolved potassium in percolating recharge water occurs through ion-exchange reactions with clay minerals in the soil (Nightingale, 1989).

Zinc is potentially an even more sensitive surrogate than is potassium, as the concentration of zinc is more than 100 times higher in the recharge basin than in the monitor wells (table 6). Motor vehicles, and especially motor oil, have been implicated as the major source of zinc in the urban environment (Wigington and others, 1983). The reason for such a large enrichment (comparative concentration ratio) for aqueous zinc in the recharge basin is unknown; however, it may be related partly to its role as a biological micronutrient and partly to its greater solubility than other heavy metals with large anthropogenic inputs (Harter, 1983; Wigington and others, 1983). (These same reasons can be used to explain the high concentrations of potassium.) In any case, because the available data show that trace metals in urban runoff have not affected ground-water quality at this site, monitoring for zinc would be very effective in demonstrating continued absence of impairment by trace-metal cations.

ORGANIC PRIORITY POLLUTANTS AND PESTICIDES

Water from the recharge basin and from monitor wells was analyzed for three gross organic indicators: tannin and lignin, total phenols, and methylene blue active surfactant (MBAS) detergents. All were reported present in the recharge basin and some also were detected in the monitor wells (table 6). However, the baseline for colorimetric detection of these analytes in dissimilar waters is likely to be variable and is sufficiently uncertain that their reported detection at very low levels in the monitor-well samples is not conclusive evidence for their actual presence (that is, the levels detected likely represent "false positives").

Several groups of synthetic organic chemicals also were analyzed: purgeable and extractable priority pollutants, organochlorine compounds, and organophosphorous insecticides. Only a few were found and, because their reported aqueous concentrations were at

the laboratory reporting limit, their presence should not be inferred without further sampling.

Organic (and trace-metal) concentrations generally are much higher in sediments than in water and, for this reason, the sediment (soils) data presented in the next section of this report provide a much more sensitive indicator of organic contaminant migration than do the aqueous-phase data.

CHEMICAL CONCENTRATIONS IN SEDIMENT AND SOIL

Chemical analysis of sediment beneath the recharge basin and of soil from test holes provides this study with the strongest evidence that sorption by the solid phase is a very effective mechanism for removal of inorganic and organic contaminants, and that only a very small fraction of the available capacity for removal is used after a few years. Results of these analyses are presented in tables 7 to 10, and in table 14 (results reported in table 14 are discussed later, in the "Laboratory Elutriation Experiments" section). The results given in these tables indicate marked attenuation in concentration of numerous elements and synthetic organic chemicals within the top few centimeters beneath the recharge basin.

INORGANIC ELEMENTS

The determination of "background levels" is more difficult for inorganic elements than for synthetic organic chemicals; whereas the former occur naturally in variable concentrations, the latter are completely absent in soil and sediment known to be free of human influence.

Comparisons of trace-element data show that natural variability far exceeds any differences resulting from sampling and analytical imprecision. The range in concentration for 43 elements analyzed in five soil cores from test hole 1 exceeds 10 percent for all 32 of the elements whose concentrations exceed reporting limits (table 7). This contrasts markedly with the nearly identical concentrations measured for sediment samples collected immediately adjacent to one another from depths of 0 to 1 cm (table 8) and 120 to 150 cm (tables 8 and 9) at site 5 in the recharge basin. For the highly contaminated surficial samples, measurable concentrations are identical for 20 elements and differ by more than 10 percent for only 4 elements (molybdenum, potassium, thorium, and ytterbium). For the background-level

Table 7. Concentrations of elements and radioisotopes in soil cores from test hole 1 adjacent to the recharge basin

[Element concentrations in micrograms per gram dry weight and radioisotope concentrations in disintegrations per minute per gram dry weight. --, no data; <, less than indicated reporting limit]

Element/isotope	Symbol	Concentration				
		Depth (meters below land surface)				
		4.9-5.5	5.5-5.8	5.8-6.2	6.2-6.4	10.9-11.3
Aluminum	Al	69,000	73,000	82,000	70,000	89,000
Arsenic	As	1.2	4.5	9.5	3.9	4.2
Barium	Ba	1,000	900	830	820	510
Beryllium	Be	1	1	2	1	<1
Bismuth	Bi	<10	<10	<10	<10	<10
Boron	B	<0.4	<.4	<.4	<.4	1.9
Cadmium	Cd	<2	<2	<2	<2	<2
Calcium	Ca	17,000	18,000	23,000	19,000	24,000
Cerium	Ce	35	50	63	54	37
Chromium	Cr	16	20	26	29	140
Cobalt	Co	4	7	9	6	19
Copper	Cu	4	8	12	6	26
Europium	Eu	<2	<2	<2	<2	<2
Gallium	Ga	12	14	16	13	16
Gold	Au	<8	<8	<8	<8	<8
Holmium	Ho	<4	<4	<4	<4	<4
Iron	Fe	13,000	23,000	25,000	23,000	38,000
Lanthanum	La	21	29	36	31	19
Lead	Pb	24	19	23	16	14
Lithium	Li	11	17	24	14	30
Magnesium	Mg	3,400	5,700	8,200	5,200	15,000
Manganese	Mn	230	400	560	390	590
Mercury	Hg	<.02	<.02	<.02	<.02	<.02
Molybdenum	Mo	<2	<2	<2	<2	<2
Neodymium	Nd	14	20	25	22	18
Nickel	Ni	7	10	12	10	140
Niobium	Nb	5	9	11	8	5
Phosphorus	P	300	400	800	300	300
Potassium	K	30,000	29,000	28,000	27,000	18,000
Scandium	Sc	3	5	8	4	14
Selenium	Se	<.2	<.2	<.2	<.2	<.2
Silver	Ag	<2	<2	<2	<2	<2
Sodium	Na	22,000	23,000	26,000	23,000	19,000
Strontium	Sr	360	350	380	350	280
Tantalum	Ta	<40	<40	<40	<40	<40
Thorium	Th	7.74	13.9	19.3	12.8	4.7
Tin	Sn	<10	<10	<10	<10	<10
Titanium	Ti	1,300	2,100	3,000	2,100	3,100
Uranium	U	1.57	3.10	3.52	2.53	2.24
Vanadium	V	29	54	63	56	91
Ytterbium	Yb	<1	1	2	2	2
Yttrium	Y	9	13	18	14	17
Zinc	Zn	22	38	53	33	69
Lead-210	²¹⁰ Pb	2.01±.24	--	--	2.95±.32	--
Radium-226	²²⁶ Ra	1.43±.05	--	--	2.70±.21	--
Lead-210 excess	²¹⁰ Pb _{ex}	.58±.25	--	--	.25±.38	--

Table 8. Concentrations of chemical constituents and radioisotopes in surface (0-1 cm) sediment from six sites in the recharge basin

[Chemical concentrations in micrograms per gram dry weight; radioisotope concentrations in disintegrations per minute per gram dry weight. GC-FID, gas chromatography with flame-ionization detection. cm, centimeter; --, no data; <, less than indicated reporting limit]

Constituent	Symbol (where applicable)	Concentration at site:						
		1	2	3	4	5 ¹	5 ²	6
Aluminum	Al	59.000	66.000	70.000	71.000	70.000	70.000	72.000
Arsenic	As	5.3	17	6.9	9.0	18	17	16
Barium	Ba	580	650	680	680	640	680	680
Beryllium	Be	1	1	1	1	1	1	<1
Bismuth	Bi	<10	<10	<10	<10	<10	<10	<10
Boron	B	7.9	5.6	5.4	3.6	3.9	3.9	4.3
Cadmium	Cd	<2	<2	<2	<2	<2	<2	<2
Calcium	Ca	28.000	27.000	22.000	19.000	17.000	17.000	17.000
Cerium	Ce	42	46	52	53	54	53	54
Chromium	Cr	44	59	61	71	83	84	86
Cobalt	Co	10	10	13	14	17	17	17
Copper	Cu	63	74	64	97	140	140	130
Europium	Eu	<2	<2	<2	<2	<2	<2	<2
Gallium	Ga	12	13	15	16	16	16	17
Gold	Au	<8	<8	<8	<8	<8	<8	<8
Holmium	Ho	<4	<4	<4	<4	<4	<4	<4
Iron	Fe	22.000	23.000	30.000	34.000	38.000	39.000	40.000
Lanthanum	La	24	26	30	31	31	31	32
Lead	Pb	91	92	110	130	170	170	160
Lithium	Li	20	24	33	37	45	45	46
Magnesium	Mg	10.000	11.000	12.000	12.000	14.000	14.000	14.000
Manganese	Mn	490	490	600	560	640	650	650
Mercury	Hg	.07	.06	.10	.11	.22	.23	.19
Molybdenum	Mo	2	<2	3	3	6	5	6
Neodymium	Nd	17	17	21	21	20	20	20
Nickel	Ni	31	33	43	51	66	68	68
Niobium	Nb	8	7	8	7	7	7	8
Phosphorus	P	1,500	1,200	1,500	1,800	2,200	2,300	2,400
Potassium	K	19,000	21,000	22,000	22,000	18,000	21,000	22,000
Scandium	Sc	6	7	9	9	10	10	10
Selenium	Se	.4	.6	.5	.7	1.0	1.0	.9
Silver	Ag	<2	<2	<2	<2	<2	<2	<2
Sodium	Na	20,000	20,000	18,000	16,000	13,000	13,000	13,000
Strontium	Sr	410	370	340	290	270	270	270
Tantalum	Ta	<40	<40	<40	<40	<40	<40	<40
Thorium	Th	12.7	12.3	19.7	21.4	20.6	30.1	25.1
Tin	Sn	<10	<10	<10	<10	<10	<10	<10
Titanium	Ti	2,300	2,500	2,900	3,000	2,900	3,100	3,200
Uranium	U	5.08	4.64	4.78	5.70	11.6	11.3	10.0
Vanadium	V	59	64	75	83	94	94	97
Ytterbium	Yb	1	1	2	2	1	2	2
Yttrium	Y	13	13	14	14	14	14	14
Zinc	Zn	590	490	510	750	1,200	1,200	1,200
Nitrite+nitrate (as N)		10	23	11	7	8	--	3
Ammonium (as N)		1,800	1,400	830	1,400	1,100	--	990
Phosphorus (as P)		2,000	1,700	1,800	1,900	87	--	2,800
Organic carbon (as C)		170.000	95.000	100.000	130.000	160.000	--	150.000

Footnote at end of table.

Table 8. Concentrations of chemical constituents and radioisotopes in surface (0-1 cm) sediment from six sites in the recharge basin—Continued

Constituent	Symbol (where applicable)	Concentration at site:						
		1	2	3	4	5 ¹	5 ¹	6
Inorganic carbon (as C)		200	100	100	100	200	--	300
Oil and grease GC/FID (as C ₁₀ D ₈ equivalents)		24,000	13,000	17,000	16,000	22,000	--	20,000
Aldrin		<.0005	<.001	<.001	<.001	<.0005	--	<.0005
Lindane		.001	<.0011	<.0011	.0011	.0011	--	<.0005
Chlordane		.13	.14	.64	.20	.21	--	.18
DDD		.064	.005	.060	.005	.035	--	.032
DDE		.065	.043	.070	.071	.040	--	.077
DDT		.0050	.0041	<.0011	.012	<.0005	--	<.0005
Dieldrin		.0021	.0041	.0051	.0061	<.0005	--	.0015
Endosulfan		--	--	<.001	--	--	--	<.0005
Endrin		<.0005	<.0011	<.0011	<.0011	<.0005	--	<.0005
Heptachlor		.0025	.0031	.0031	.0031	.0021	--	.0031
Heptachlor epoxide		<.0005	<.0011	<.0001	<.0011	<.0051	--	<.0005
PCB		.031	.021	.033	.030	.039	--	.040
PCN		<.005	<.010	<.010	<.010	<.005	--	<.005
Lead-210	²¹⁰ Pb	5.18±.17	5.40±.18	6.92±.21	8.81±.29	10.1±.3	--	10.9±.3
Radium-226	²²⁶ Ra	2.70±.10	3.03±.11	3.33±.12	4.71±.18	5.10±.15	--	5.15±.19
Lead-210 excess	²¹⁰ Pb _{xs}	2.48±.20	2.37±.21	3.59±.24	4.09±.34	5.00±.34	--	5.75±.36

¹Two adjacent samples at site 5 were analyzed.

samples from the deepest interval, measurable concentrations are identical for 13 elements and differ by more than 10 percent for only 3 elements (nickel, niobium, and thorium).

Such small differences between duplicate sediment analyses provides strong evidence that the much larger site-to-site differences in surface sediments from the recharge basin (table 8) are real. Many of the trace elements whose concentrations are highest at the surface have concentrations at sites 5 and 6 that are about double those at sites 1 and 2. This trend of increasing concentration toward the south end of the recharge basin and away from the discharge pipe is consistent with a similar trend of increasing percentage of clay (table 5) to which these contaminants could be expected to adsorb.

Although the sediment at site 5 shows a marked decrease in many chemical concentrations with increasing depth (table 9 and fig. 4), the rather large variability in natural abundances and sharp change in lithology (from fine to coarse sediment) several centimeters below the recharge basin makes the assignment of background values somewhat uncertain. This variability in background concentrations is

especially evident from the chemical differences between soil cores at five different depths (table 7), all of which must represent uncontaminated natural conditions. The concentrations of several elements are highest in the core interval from 10.9 to 11.3 m, which likely is related to the increased proportion of clay and silt in this sample relative to the other four samples (table 4). Radium-226 provides similar evidence of natural variability that probably implies differences in mineral assemblage. Concentrations of this naturally occurring radioisotope differ substantially between two of the soil cores (table 7), between the six surface-sediment sites in the recharge basin (table 8), and at different subbottom depths from site 5 beneath the recharge basin. On the basis of the particle-size distributions and selected chemical concentrations, as was noted in the section "Sampling of Sediments from the Recharge Basin," it was concluded that recharge-basin sediment below 16 cm is chemically most similar to soil between 5.5 and 5.8 m in test hole 1. However, the fact that chemical concentrations for a number of elements vary by a factor of 2 or more in uncontaminated soil from various depths in test hole 1 indicates that "background" for fine-grained basin deposits above 16 cm may be quite different from "background" for coarse-

Table 9. Concentrations of chemical constituents and radioisotopes in sediment and substrata at site 5 in the recharge basin

[Chemical concentrations in micrograms per gram dry weight; isotope concentrations in disintegrations per minute per gram dry weight. Data from 0-1 cm also are given in table 8. cm, centimeter; ~, approximate value; --, no data; <, less than indicated reporting limit]

Constituent	Symbol (where applicable)	Depth interval below recharge basin (cm)									
		0-1 ¹	0-1 ¹	1-2	2-4	4-8	8-16	16-32	32-64	120-150 ¹	120-150 ¹
Aluminum	Al	70,000	70,000	79,000	82,000	86,000	82,000	75,000	76,000	74,000	73,000
Arsenic	As	18	17	13	9.5	3.7	1.2	1.0	1.2	1.1	1.2
Barium	Ba	640	680	740	800	900	840	800	790	770	780
Beryllium	Be	1	1	1	1	2	2	1	1	1	1
Bismuth	Bi	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
Boron	B	3.9	3.9	2.4	1.4	<.4	<.4	<.4	<.4	<.4	.4
Cadmium	Cd	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2
Calcium	Ca	17,000	17,000	17,000	21,000	24,000	23,000	21,000	21,000	21,000	20,000
Cerium	Ce	54	53	63	67	66	58	59	52	56	55
Chromium	Cr	83	84	87	54	24	26	21	21	23	22
Cobalt	Co	17	17	17	13	10	8	6	6	6	6
Copper	Cu	140	140	120	52	13	10	6	5	6	6
Europium	Eu	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2
Gallium	Ga	16	16	18	18	17	16	14	14	13	14
Gold	Au	<8	<8	<8	<8	<8	<8	<8	<8	<8	<8
Holmium	Ho	<4	<4	<4	<4	<.4	<.4	<.4	<.4	<.4	<.4
Iron	Fe	38,000	39,000	44,000	34,000	22,000	19,000	19,000	18,000	24,000	23,000
Lanthanum	La	31	31	36	39	39	33	36	29	32	31
Lead	Pb	170	170	160	88	22	19	19	18	19	18
Lithium	Li	45	45	53	39	27	21	15	15	13	14
Magnesium	Mg	14,000	14,000	15,000	12,000	9,800	7,500	5,500	5,400	5,000	5,100
Manganese	Mn	640	650	700	620	550	450	370	350	370	370
Mercury	Hg	.22	.23	.22	.13	<.02	<.02	<.02	<.02	<.02	<.02
Molybdenum	Mo	6	5	4	<2	<2	<2	<2	<2	<2	<2
Neodymium	Nd	20	20	25	26	27	23	24	21	23	23
Nickel	Ni	66	68	70	39	12	11	10	10	9	8
Niobium	Nb	7	7	7	10	12	9	8	8	10	9
Phosphorus	P	2,200	2,300	2,100	1,200	1,000	700	500	500	300	300
Potassium	K	18,000	21,000	24,000	26,000	31,000	27,000	27,000	28,000	27,000	27,000
Scandium	Sc	10	10	11	10	9	7	5	5	5	5
Selenium	Se	1.0	1.0	<1.0	.8	<.2	<.2	<.2	<.2	<.2	<.2
Silver	Ag	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2
Sodium	Na	13,000	13,000	13,000	21,000	27,000	27,000	25,000	25,000	25,000	25,000
Strontium	Sr	270	270	280	340	410	400	380	380	370	370
Tantalum	Ta	<40	<40	<40	<40	<40	<40	<40	<40	<40	<40
Thorium	Th	20.6	30.1	29.9	23.6	25.9	15.2	15.6	12.8	13.2	17.6

Footnote at end of table.

Table 9. Concentrations of chemical constituents and radioisotopes in sediment and substrata at site 5 in the recharge basin--
Continued

Constituent	Symbol (where applicable)	Depth interval below recharge basin (cm)									
		0-1 ¹	0-1 ¹	1-2	2-4	4-8	8-16	16-32	32-64	120-150 ¹	120-150 ¹
Tin	Sn	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
Titanium	Ti	2,900	3,100	3,500	3,500	3,300	2,800	2,200	2,100	2,200	2,200
Uranium	U	11.6	11.3	8.73	4.81	3.11	3.32	2.98	3.09	3.20	
Vanadium	V	94	94	100	84	59	51	50	47	58	56
Ytterbium	Yb	1	2	2	2	2	2	2	2	2	2
Yttrium	Y	14	14	15	17	18	16	15	14	15	15
Zinc	Zn	1,200	1,200	760	390	71	49	31	33	33	30
Nitrite+nitrate (as N)		8	--	<2	<2	<2	3	2	3	9	
Ammonium (as N)		1,100	--	960	560	230	97	32	170	1.2	--
Organic nitrogen (as N)		--	--	--	--	--	43	--	--	--	--
Phosphorus (as P)		87	--	2,600	990	1,300	730	690	760	470	--
Organic carbon (as C)		160,000	--	150,000	50,000	800	~600	~200	~900	~100	--
Inorganic carbon (as C)		200	--	300	200	100	<100	<100	<100	<100	--
Oil and grease		22,000	--	24,000	11,000	1,000	<1,000	<1,000	<1,000	<1,000	--
GC/FID (as C ₁₀ D ₈ equivalents)		1,500	--	600	380	4.2	1.7	2.9	1.2	5.7	--
Aldrin		<.0005	--	<.0011	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	--
Lindane		.0011	--	.0071	.0032	.0003	<.0001	<.0001	<.0001	<.0001	--
Chlordane		.21	--	.35	.20	.001	.001	<.001	<.001	<.001	--
DDD		.035	--	.055	.067	<.0001	--	<.0001	<.0001	<.0001	--
DDE		.040	--	.072	.048	.0004	.0001	<.0001	<.0001	<.0001	--
DDT		<.0005	--	.10	.0021	.0001	<.0001	<.0001	<.0001	<.0001	--
Dieldrin		<.0005	--	.055	<.0001	.0001	<.0001	<.0001	<.0001	<.0001	--
Endosulfan		--	--	<.001	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	--
Endrin		<.0005	--	.085	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	--
Heptachlor		.0021	--	.0021	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	--
Heptachlor epoxide		<.0051	--	<.0011	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	--
PCB		.039	--	.06	.04	<.001	<.001	<.001	<.001	<.001	--
PCN		<.005	--	<.01	<.001	<.001	<.001	<.001	<.001	<.001	--
Lead-210	²¹⁰ Pb	10.1±.3	--	9.19±.23	6.68±.19	2.42±.11	2.17±.09	2.57±.10	2.57±.10	2.71±.11	--
Radium-226	²²⁶ Ra	5.10±.15	--	3.54±.11	3.92±.11	2.48±.09	2.53±.08	2.58±.08	2.27±.07	2.29±.07	--
Lead-210 excess	²¹⁰ Pb _{ex}	5.00±.34	--	5.65±.25	2.76±.22	-.06±.14	-.36±.12	-.01±.13	.30±.12	.41±.13	--

¹Analysis of two samples collected next to one another.

Table 10. Concentrations of extractable priority pollutants, organochlorine compounds, and organophosphorus insecticides in sediment from depths of 0-1 and 8-16 cm at site 5 in the recharge basin

[Concentrations in micrograms per gram dry weight, cm. centimeter; <, less than indicated reporting limit]

Organic constituent	Concentration		Organic constituent	Concentration	
	0-1 cm	8-16 cm		0-1 cm	8-16 cm
Extractable Priority Pollutants					
Acenaphthylene	<0.2	<0.2	2,6-Dinitrotoluene	<0.2	<0.2
Acenaphthene	<.2	<.2	4-Bromophenyl phenyl ether	<.2	<.2
Anthracene	.46	<.2	Chlorophenyl phenyl ether	<.2	<.2
Benzo (a) anthracene	3.9	<.2	4-Nitrophenol	<1.2	<.6
Benzo (b) anthracene	9.2	<.4	4,6-Dinitro-o-cresol	<.6	<.6
Benzo (k) fluoranthene	6.4	<.4	Phenol	<.2	<.2
Benzo (g,h,i) perylene	11	<.4	Pentachlorophenol	<.6	<.6
Benzo (a) pyrene	6.5	<.4	Organochlorine Compounds		
bis (2-Chloroethyl) ether	<.2	<.2	Aldrin	<0.0005	<0.0001
bis (2-Ethylhexyl) phthalate	6.2	<.2	Lindane	.0011	<.0001
Chrysene	9.7	<.4	Chlordane	.21	.001
Diethylphthalate	<.2	<.2	DDD	.035	--
Dimethylphthalate	<.2	<.2	DDE	.040	.0001
di-n-Butyl phthalate	<.2	<.2	DDT	<.0005	<.0001
Fluoranthene	9.4	<.2	Dieldrin	<.0005	<.0001
Fluorene	<.2	<.2	Endosulfan	--	<.0001
Hexachlorobenzene	<.2	<.2	Endrin	<.0005	<.0001
Hexachlorobutadiene	<.2	<.2	Ethion	<.0005	<.0001
Hexachlorocyclopentadiene	<.2	<.2	Toxaphene	<.05	<.01
Hexachloroethane	<.2	<.2	Heptachlor	.0021	<.0001
Indeno (1,2,3,ed) pyrene	9.5	<.2	Heptachlor epoxide	.0051	<.0002
Isophorone	<.2	<.2	Methoxychlor	<.025	<.0001
n-Nitrosodi-n-propylamine	<.2	<.2	Mirex	<.0005	<.0001
n-Nitrosodiphenylamine	<1.5	<.2	Perthane	<.005	<.001
n-Nitrosodimethylamine	<.2	<.2	PCB	.039	<.001
Naphthalene	.15	<.2	PCN	<.005	<.001
Nitrobenzene	<.2	<.2	Organophosphorous Insecticides		
p-Chloro-m-cresol	<.6	<.6	Malathion	<0.0005	<0.0001
Phenanthrene	2.7	<.2	Parathion	<.0005	<.0001
Pyrene	15	<.2	Diazinon	<.0005	<.0001
1,2-Dichlorobenzene	<.2	<.2	Methyl parathion	<.0005	<.0001
1,2,4-Trichlorobenzene	<.2	<.2	Methyl trithion	<.0005	<.0001
1,2,5,6-Dibenzanthracene	<.4	<.4	Trithion	<.0005	<.0001
1,3-Dichlorobenzene	<.2	<.2	Chlorophenoxy Acid Herbicides		
1,4-Dichlorobenzene	<.2	<.2	2,4-D	<0.0001	<0.0001
2-Chloronaphthalene	<.2	<.2	Dicamba	<.0001	<.0001
2-Chlorophenol	<.2	<.2	2,4-DP	<.2	<.2
2-Nitrophenol	<.2	<.2	Pichloram	<.0001	<.0001
Di-n-octylphthalate	<.4	<.4	Silvex	<.0001	<.0001
2,4-Dichlorophenol	<.2	<.2	2,4,5-T	<.0001	<.0001
2,4-Dinitrotoluene	<.2	<.2			
2,4-Dinitrophenol	<.6	<.6			
2,4,6-Trichlorophenol	<.6	<.6			

grained basin deposits below 16 cm. Sampling of near-surface soil in the vicinity of the study basin that could have established more suitable background levels for the shallower basin sediment was not done. Extensive tilling and grading in the area indicated that

nearby soil would not have been suitable for direct comparison because reworking is likely to have mixed contaminated (by atmospheric deposition) and uncontaminated soil, and soil of differing texture, in variable proportions.

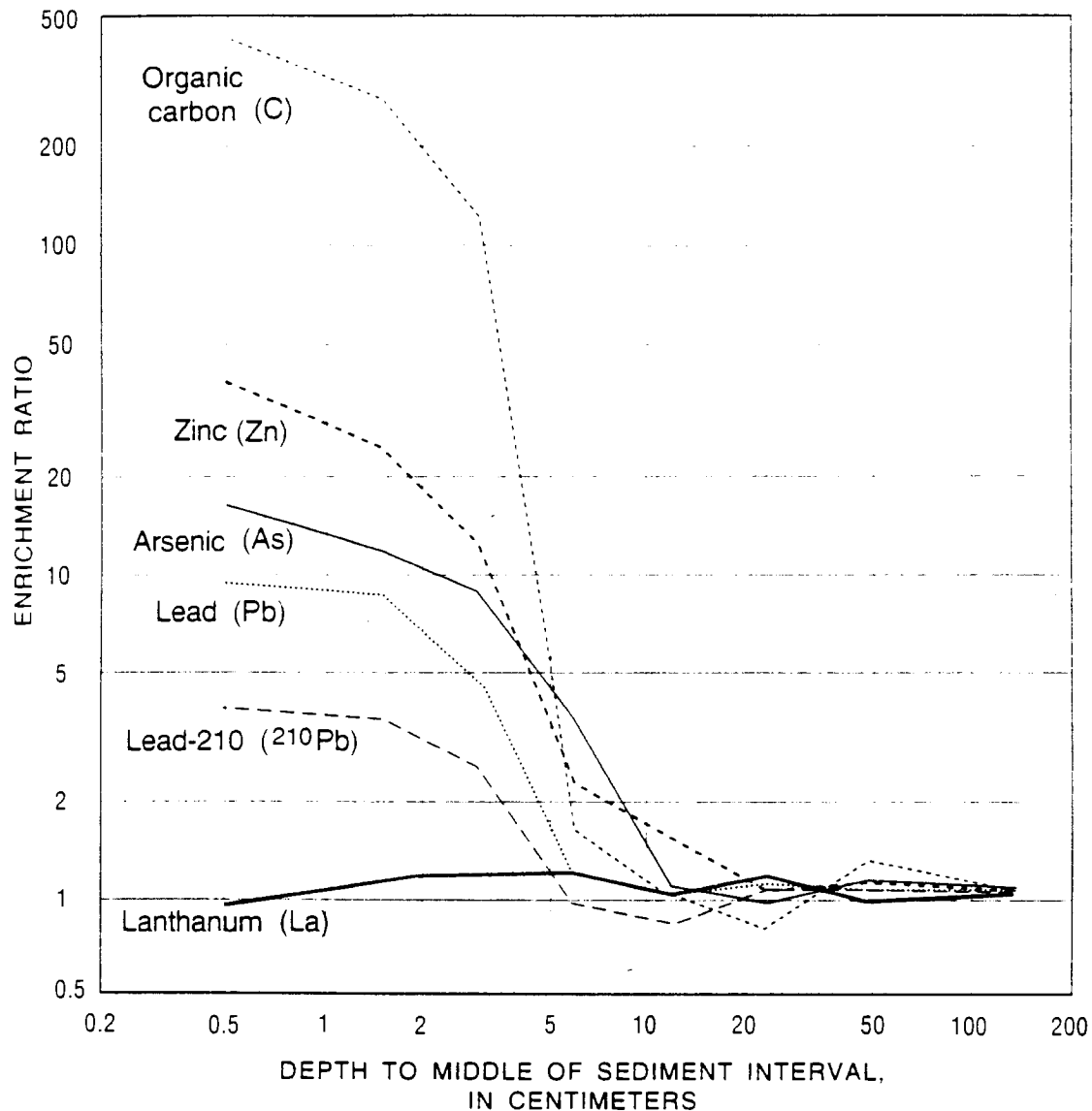


Figure 4. Enrichment ratios for selected constituents in subbottom sediment at site 5 in the recharge basin.

ORGANIC CHEMICALS

The results for organic carbon, oil and grease, GC-FID organic carbon, and organochlorine compounds in near-surface (0-1 cm) sediments at the six recharge-basin sites are given in table 8. Table 9 gives results for the same suite of organic constituents with increasing subbottom depth at site 5. Results for analysis of extractable priority pollutants, organochlorine compounds, organophosphorus insecticides, and chlorophenoxy-acid herbicides at depths of 0 to 1 cm and 8 to 16 cm at site 5 are given in table 10. The pattern of results for organic chemicals generally

is similar to that shown by inorganic elements, although there exists considerably less uniformity in either the areal trend (table 8) or the profile (table 9) for the organic chemical constituents. No pesticides were detected below 16 cm at site 5, and only two, chlordane and DDE, were identified (at their reporting limits of 0.001 $\mu\text{g/g}$ and 0.0001 $\mu\text{g/g}$, respectively) in sediment from the 8- to 16-cm interval.

The only pesticides that were found in basin sediment are the long-lived organochlorine compounds. It was anticipated that some of the more widely used organophosphorus insecticides, on the

basis of their abundance in rainfall and runoff (Oltmann and Shulters, 1989), also might be present in sediment, but none were detected (table 10). Similarly, their absence in sediment from five recharge basins in residential and commercial catchments during the Fresno NURP study was attributed to probable degradation (Nightingale, 1987c). A large number of phosphorus-containing organic compounds were reported during the gas-chromatographic analyses for this study, but they could not be specifically identified because of poor resolution and many interferences (M.P. Schroeder, USGS, oral commun., 1987). High organic content and the anaerobic nature of the sediment—which made cleanup of the sample extracts difficult and may have interfered with detection of organophosphorus insecticides—is the cause of the variable analytical reporting limits for the organochlorine compounds that are listed in the tables.

Chlordane was the organochlorine compound reported in highest concentration, a finding similar to that obtained during monitoring for the NURP study (Oltmann and Shulters, 1989). However, one of the more puzzling results obtained by this study is the finding of DDT itself as opposed to only its metabolites, DDD and DDE, which are more commonly present in environmental samples today. The highest DDT concentration of 0.10 $\mu\text{g/g}$ was found not in surface sediment but rather in sediment from a depth of 1 to 2 cm at site 5. Use of DDT was banned in the United States about 10 years prior to the Fresno NURP study and the operation of this recharge basin in its present configuration. Confirmation of DDT's presence is strong, as shown by its detection in several of the sediment samples analyzed (tables 8 and 9). Furthermore, the Fresno NURP study found that DDT concentrations consistently exceeded DDD and DDE concentrations in street sweepings from residential, commercial, and industrial catchments (Oltmann and Shulters, 1989).

Of 50 extractable organic priority pollutants analyzed, 13—including 12 polycyclic aromatic hydrocarbons and 1 phthalate ester—were detected in surficial sediment from site 5 (table 10). None were detected in sediment from the 8- to 16-cm interval. Total concentration of all the identified organic compounds in the surficial interval is slightly less than 100 $\mu\text{g/g}$, nearly all of which consists of the polycyclic aromatic hydrocarbons listed in table 10. Comparison with data given in table 9 shows that this represents only about 5 percent of the organic carbon reported by the GC-FID analysis, and less than 0.1 percent of total organic carbon. Although two addi-

tional polycyclic aromatic hydrocarbons (dimethyl-substituted naphthalenes) and at least 20 compounds belonging to the broad classes of alkanes and alkenes (or cycloalkanes) were identified by GC-FID analysis, virtually all the organic matter in the basin sediment remains uncharacterized. It seems likely that much of it may be generated by biological activity in the overlying water column; however, the organic nitrogen data that might have supported this assertion with the observation of low carbon/nitrogen ratios are lacking because all sediment samples but one listed in table 9 were accidentally destroyed in the laboratory during analysis.

ASSESSMENT OF ANTHROPOGENIC INPUTS

Chemical profiles that show a decrease in concentration with increasing sediment depth not only are strong evidence for sorptive removal of contaminants but also can be used, with some assumptions, to quantify the relative importance of anthropogenic (human-related) and natural (background) sources. The concept of enrichment ratios, defined herein as the concentration in a particular sediment interval divided by the mean concentration for all intervals below a depth of 16 cm (considered background concentration), is useful for this discussion. Profiles of enrichment ratios for selected constituents are shown in figure 4. If the surficial (0 to 1 cm) sediment is presumed to represent the combined inputs of current anthropogenic and natural sources, then decreases below this depth represent removal of contaminants, and possibly dilution by mixing with deeper soil that is free of anthropogenic influence. As expected, for rare-earth elements such as lanthanum, which has very few industrial uses, the concentration profile shows no change and represents only the natural abundance of this element. Two other rare earths that are present above reporting limits, cerium and neodymium, also show no change in concentration with depth (table 9).

Surface-enrichment ratios for all elements were calculated and are reported to the nearest whole number in table 11 for those values that differ from 1. These values include about half the elements analyzed. For those elements not listed in table 11, either there is no anthropogenic source or (1) concentrations are uniformly below detection, (2) natural levels are so high that they mask any anthropogenic input, or (3) variability in mineralogy or lithology exerts the predominant influence on concentrations. Just as an enrichment ratio of 1 does not preclude an anthropogenic component—conversely, an

Table 11. Surface-enrichment ratios for selected elements and radioisotopes in sediment at site 5 in the recharge basin

[Surface-enrichment ratio is concentration of 0-1 centimeter interval divided by mean concentration of all intervals below 16 centimeter. >, greater than indicated value]

Constituent	Symbol	Enrichment ratio
Arsenic	As	16
Boron	B	>9
Chromium	Cr	5
Cobalt	Co	3
Copper	Cu	25
Iron	Fe	2
Lead	Pb	9
Lithium	Li	3
Magnesium	Mg	3
Manganese	Mn	2
Mercury	Hg	>11
Molybdenum	Mo	>2
Nickel	Ni	7
Phosphorus	P	5
Scandium	Sc	2
Selenium	Se	>5
Thorium	Th	2
Uranium	U	4
Vanadium	V	2
Zinc	Zn	38
Lead-210	²¹⁰ Pb	4
Radium-226	²²⁶ Ra	2

enrichment ratio greater than 1 is not necessarily conclusive proof of anthropogenic contamination, especially when these ratios are low. Such results could be an artifact of natural variability, as already discussed, or the results could be unique to the geochemical behavior of individual elements.

Despite these limitations on interpretation, it is clear from some of the very large enrichment ratios listed in table 11 that anthropogenic fluxes to the recharge basin far exceed natural fluxes for several trace metals that are in widespread industrial use. Results for three elements at this recharge basin (copper=25, lead=9, and zinc=38) can be compared with the enrichment ratios for these same elements (Cu=6, Pb=15, and Zn=5) in sediment from Irondequoit Bay, New York (Schroeder, 1985), which also was one of the NURP study sites (Kappel and others, 1986).

The results for four elements (As, Cu, Pb, and Ni) from this study also can be compared with those from a previous investigation at five other recharge basins in the city of Fresno, the oldest of which has been in use for about 20 years. The five basins are in

predominantly residential catchments that include some commercial land use and minimal industry. Surface-enrichment ratios, calculated by comparing published data for depths of 0-2 and 90-120 cm in the five basins (Nightingale, 1987b), range from 1 to 18 for arsenic, 3 to 13 for copper, 60 to 400 for lead, and 2 to 4 for nickel. Values for this study (table 11) are arsenic = 16, copper = 25, lead = 9, and nickel = 7. Enrichments are greater for the trace metals copper and nickel and much less for lead in the industrial catchment in comparison with the residential/commercial catchments, a pattern which is similar to that found during direct monitoring of runoff in the NURP study (Oltmann and Shulters, 1989).

Surface-enrichment ratios can be calculated similarly for organic priority pollutants and organochlorine compounds; however, the calculations yield only minimum estimates of 10^2 and 10^3 , respectively, because these compounds are below detection levels in the deeper sediment. Organic carbon, analyzed by combustion and by GC-FID (fig. 4 and table 9), also yields surface-enrichment ratios of almost 10^3 ; for organic carbon, detectable concentrations exist below 16 cm for both methods of analysis.

The data presented and the foregoing discussion indicate that a wide variety of trace elements and synthetic organic chemicals from the industrial catchment are removed within the top 16 cm of sediment in the recharge basin. Given the existence of nearly 8 m of soil between the basin and the water table and such effective removal of contaminants for the 4 years of operation in the basin's current configuration (since last scraped), local ground-water quality should be afforded protection during continued operation of the basin for a very long time. A similar inference can be made on the basis of results of studies at five other recharge basins in the city of Fresno (Nightingale, 1987a,b,c, and 1989)—although it was suggested by Nightingale (1989) that periodic monitoring of organic contaminants in soil would be advisable because of their known presence in storm runoff.

LABORATORY ELUTRIATION EXPERIMENTS

The potential for movement of urban-runoff contaminants through the unsaturated zone can be evaluated by means of laboratory experiments that simulate processes of sorption from the aqueous onto the solid phase and desorption from the solid into the aqueous phase. Both types of experiments were done for this study. As conducted, the experiments were intended to provide only a qualitative rather than a quantitative assessment. They were adapted from

Table 12. Chemical concentrations in water from the recharge basin equilibrated (mixed for 0, 1, 2, and 4 days) with surface (0-1 centimeter) sediment from near site 5 in the recharge basin

[Water from recharge basin collected April 29, 1986, and stored frozen until beginning of laboratory experiment 4 months later. $\mu\text{S}/\text{cm}$, microsiemens per centimeter at 25° Celsius; mg/L, milligrams per liter; $\mu\text{g}/\text{L}$, micrograms per liter; --, no data; <, less than indicated reporting limit]

Constituent	Equilibration time			
	0 day	1 day	2 days	4 days
Specific conductance ($\mu\text{S}/\text{cm}$)	346	638	660	696
pH (standard units)	6.2	7.7	6.9	7.3
Calcium (mg/L)	8.0	29	25	29
Magnesium (mg/L)	5.0	--	--	13
Sodium (mg/L)	16	22	21	22
Potassium (mg/L)	26	--	--	30
Chloride (mg/L)	21	23	22	24
Sulfate (mg/L)	33	120	120	110
Silica (mg/L as SiO_2)	5.9	11	14	16
Alkalinity (mg/L as CaCO_3)	7	8	8	--
Ammonium (mg/L as N)	11	26	25	29
Kjeldahl nitrogen (mg/L as N)	15	48	42	41
Nitrate + nitrite (mg/L as N)	<.1	<.1	.2	<.1
Orthophosphate (mg/L as P)	5.0	2.7	3.5	4.2
Organic carbon (mg/L as C)	24	--	--	66
Arsenic ($\mu\text{g}/\text{L}$)	3	--	--	11
Boron ($\mu\text{g}/\text{L}$)	80	--	--	620
Cadmium ($\mu\text{g}/\text{L}$)	1.4	--	--	.15
Chromium ($\mu\text{g}/\text{L}$)	2.2	--	--	2.1
Cobalt ($\mu\text{g}/\text{L}$)	.5	--	--	1.5
Copper ($\mu\text{g}/\text{L}$)	10.0	--	--	3.1
Iron ($\mu\text{g}/\text{L}$)	140	6,300	4,300	1,000
Lead ($\mu\text{g}/\text{L}$)	1.4	--	--	2.7
Manganese ($\mu\text{g}/\text{L}$)	40	370	330	290
Mercury ($\mu\text{g}/\text{L}$)	.1	--	--	<.1
Molybdenum ($\mu\text{g}/\text{L}$)	2	--	--	9
Nickel ($\mu\text{g}/\text{L}$)	5.7	--	--	10
Zinc ($\mu\text{g}/\text{L}$)	30	540	290	110
Tannin & lignin (mg/L as tannic acid)	4.3	--	--	9.0
Phenols ($\mu\text{g}/\text{L}$ as phenol)	2	--	--	18
MBAS detergents (mg/L)	.09	--	--	.27

methods commonly used to determine sorption isotherms (Harter, 1983) and from elutriation tests used to determine short-term water-quality conditions associated with dredging operations (Fuhrer and Rinella, 1983; Palermo, 1986). In one set of experiments, samples of highly contaminated surface sediment from the recharge basin were mixed with water from the recharge pond to test the ability of the sediments to desorb contaminants into the water with which it is in contact. In another experiment, water from the recharge basin was mixed with clean soil from the unsaturated zone deep beneath the recharge basin to examine the potential for removal of aqueous contaminants by the soil.

Water for the experiment was collected from the recharge basin on April 29, 1986, and stored frozen until its use 4 months later. It generally was similar in composition to water collected from the basin on January 29, 1986, as indicated by comparison of chemical concentrations in tables 6 and 12 (in table 12, see values for zero-day equilibration time). The most obvious difference between the two samples is the absence of nitrate (plus nitrite, which likely is negligible) in the water used for the laboratory experiment. Although this difference may be a consequence of seasonal variations, it seems more likely that it is the result of biological degradation because the water used for the laboratory experiment

Table 13. Chemical concentrations in water from the recharge basin equilibrated (mixed for 4 days) with aquifer soil from 10.9-11.3 meters in test hole 1, in deionized water equilibrated (mixed for 4 days) with surface (0-1 centimeter) sediment from near site 5 in the recharge basin, and in a deionized-water blank

[$\mu\text{S}/\text{cm}$, microsiemens per centimeter at 25° Celsius; mg/L, milligrams per liter; $\mu\text{g}/\text{L}$, micrograms per liter; --, no data; <, less than indicated reporting limit. MBAS, methylene blue active surfactants.]

Constituent	Basin water equilibrated with aquifer soil	Deionized water equilibrated with basin sediment	Deionized-water blank
Specific conductance ($\mu\text{S}/\text{cm}$)	548	197	2
pH (standard units)	7.8	7.1	6
Calcium (mg/L)	35	10	<.1
Magnesium (mg/L)	18	5.0	<.1
Sodium (mg/L)	29	4.0	.4
Potassium (mg/L)	14	7.8	.3
Chloride (mg/L)	27	3.5	<.1
Sulfate (mg/L)	130	11	.2
Silica (mg/L as SiO_2)	38	9.9	<.1
Alkalinity (mg/L as CaCO_3)	139	82	2
Nitrite - nitrate (mg/L as N)	<.1	<.1	<.1
Ammonium (mg/L as N)	13	12	.03
Kjeldahl nitrogen (mg/L as N)	20	20	.7
Orthophosphate (mg/L as P)	2.1	3.9	.041
Organic carbon (mg/L as C)	39	16	--
Arsenic ($\mu\text{g}/\text{L}$)	35	12	<.1
Boron ($\mu\text{g}/\text{L}$)	320	60	10
Cadmium ($\mu\text{g}/\text{L}$)	.66	.4	.02
Chromium ($\mu\text{g}/\text{L}$)	1.2	2.2	.06
Cobalt ($\mu\text{g}/\text{L}$)	10	1.0	<.20
Copper ($\mu\text{g}/\text{L}$)	4.5	4.5	--
Iron ($\mu\text{g}/\text{L}$)	150	850	10
Lead ($\mu\text{g}/\text{L}$)	.30	3.4	.20
Manganese ($\mu\text{g}/\text{L}$)	890	140	<10
Mercury ($\mu\text{g}/\text{L}$)	--	<.1	<.1
Molybdenum ($\mu\text{g}/\text{L}$)	18	5.0	<.1
Nickel ($\mu\text{g}/\text{L}$)	58	8.6	<.5
Zinc ($\mu\text{g}/\text{L}$)	30	50	<10
Tannin & lignin (mg/L as tannic acid)	4.1	7.4	--
Phenols ($\mu\text{g}/\text{L}$ as phenol)	6	4	--
MBAS detergents (mg/L)	.09	.36	--

was neither preserved with the mercuric chloride biocide nor filtered prior to its storage for later use.

One experiment was carried out by using a magnetic stirrer to continuously mix recharge-basin water and recharge-basin surficial sediment, in an approximately 10-to-1 gravimetric ratio, for as long as 4 days. The sediment-water slurry was contained in a covered, cylindrical glass chromatography jar and was continuously bubbled with nitrogen gas to maintain an oxygen-free environment throughout the experiment. Aliquots for chemical analysis were removed after 1, 2, and 4 days. The aliquots (also the initial, or zero-day, water) were centrifuged for 1 hour at 5,000 times

gravity, but they were not filtered. Although the solutions appeared visually to be free of suspended solids, the possible presence of colloidal material cannot be excluded. The increase in specific conductance from 346 to 638 $\mu\text{S}/\text{cm}$ after 1 day, and a smaller increase to only 696 $\mu\text{S}/\text{cm}$ after 4 days (table 13) shows that dissolution is initially rapid and then slows.

Calcium and sulfate were the main contributors to the increase in specific conductance. Despite a very large increase in dissolved (supernatant after centrifugation) iron and manganese during the experiment, the increase in all other cationic trace-metal concen-

trations, with the exception of zinc, was comparatively modest. Most chemical concentrations changed little between days 1 and 4. Iron is an obvious exception—its concentration decreased from 6.3 to 1.0 mg/L during this period. Failure of the needle valve that regulated release of nitrogen gas several hours before termination of the experiment may have allowed entrance of some oxygen to the chamber, with the consequent precipitation of iron.

Increases were measured in tannin and lignin, total phenols, and MBAS detergents during the 4-day experiment. However, given the potential for analytical artifacts in the colorimetric methods used for these broad-indicator constituents (Wershaw and others, 1987), their precise concentrations and detection at these low concentrations is uncertain. Nightingale (1989) reported the movement of phenols in soil moisture beneath one recharge basin in the city of Fresno that had very coarse substrata. The solubility and negative polarity of phenols make them one of the few relatively mobile classes of compounds analyzed for in this study. However, specific phenols were not detected in sediment samples from the recharge basin (table 10) and, therefore, their presence and downward transport remain uncertain.

Surficial sediment for the laboratory experiment was taken from a location near site 5 in the recharge basin and was collected in a slightly different manner than was used for all other samples of surficial sediment described elsewhere in this report. Instead of removing the entire 1-cm-thick polygonal plate, the sample was obtained by scraping the sediment surface with a Teflon-coated spatula at several locations over a wide area near site 5. Chemical composition of this composite sample (table 14) is remarkably similar to composition of the duplicate samples from 0 to 1 cm at site 5 (tables 8 and 9). For 35 detectable elements, the average (unsigned) difference in concentration is 5.4 percent between the duplicate samples and only slightly larger at 7.7 percent between the composite sample and the mean value of the duplicate samples.

In addition to the equilibration between surface sediment and water from the recharge basin, the sediment also was mixed with deionized water for about 4 days as a further check for desorption of contaminants into the aqueous phase. The results given in table 13 (results for a deionized-water blank also are shown in the table) lead to the conclusion already stated that evidence exists for possible transfer of some chemical constituents from sediment to the aqueous phase. Wigington and others (1983) also reported the potential for leaching of four inorganic

Table 14. Concentrations of elements in recharge-basin sediment and in aquifer soil used for the laboratory elutriation tests

[Concentrations in micrograms per gram dry weight. Sediment is from top 1 centimeter near site 5; soil data, also given in table 7, are from depth of 10.9-11.3 meters in test hole 1. <, less than indicated reporting limit]

Element	Symbol	Concentration	
		Basin sediment	Aquifer soil
Aluminum	Al	74.000	89.000
Arsenic	As	18	4.2
Barium	Ba	710	510
Beryllium	Be	1	<1
Bismuth	Bi	<10	<10
Boron	B	2.6	1.9
Cadmium	Cd	2	<2
Calcium	Ca	16.000	24.000
Cerium	Ce	57	37
Chromium	Cr	87	140
Cobalt	Co	17	19
Copper	Cu	140	26
Europium	Eu	<2	<2
Gallium	Ga	17	16
Gold	Au	<8	<8
Holmium	Ho	<4	<4
Iron	Fe	38.000	38.000
Lanthanum	La	33	19
Lead	Pb	170	14
Lithium	Li	46	30
Magnesium	Mg	14.000	15.000
Manganese	Mn	630	590
Mercury	Hg	.27	<.02
Molybdenum	Mo	4	<2
Neodymium	Nd	21	18
Nickel	Ni	66	140
Niobium	Nb	7	5
Phosphorus	P	2.000	300
Potassium	K	22.000	18.000
Scandium	Sc	10	14
Selenium	Se	.8	<.2
Silver	Ag	<2	<2
Sodium	Na	14.000	19.000
Strontium	Sr	270	280
Tantalum	Ta	<40	<40
Thorium	Th	29.0	4.7
Tin	Sn	<10	<10
Titanium	Ti	3.200	3.100
Uranium	U	11.4	2.24
Vanadium	V	97	91
Ytterbium	Yb	2	2
Yttrium	Y	15	17
Zinc	Zn	1.200	69

urban contaminants (Cd, Cu, Pb, and Zn) from sediment in a recharge basin in Washington, D.C. They found that as much as 2 percent of these

elements was removed by a 0.01M $\text{Ca}(\text{NO}_3)_2$ solution, of the four elements tested, zinc was the most readily solubilized. Because solid-phase concentrations typically exceed aqueous concentrations by three orders of magnitude or more, the amount leached during the experiment represents a very large input to the aqueous phase. However, their experimental procedure likely overstates the potential for leaching in the environment because $\text{Ca}(\text{NO}_3)_2$ is expected to hydrolyze to form a weakly acidic solution.

For the laboratory experiment in which water from the recharge basin was mixed with natural uncontaminated soil, the core from a depth of 10.9-11.3 m in test hole 1 was selected. Soil from this depth was chosen because its generally higher silt-plus-clay content—in comparison with other, sandier, core samples (table 4)—would be expected to favor increased sorption. Results from this experiment to check for removal of aqueous contaminants by clean soil were inconclusive. An increase in mineral dissolution, as indicated by the increase in specific conductance from 346 (table 12) to 548 $\mu\text{S}/\text{cm}$ (table 13), was measured. The major ions that showed the largest increase are calcium, sulfate, and bicarbonate (alkalinity).

Concentrations of many minor constituents, including tannin and lignin and MBAS detergents, changed little. Some constituents, such as phenols and arsenic, actually show substantial increases. The increase from 2 to 6 $\mu\text{g}/\text{L}$ for phenols is, if anything, another indication of the uncertainty in the colorimetric test for phenols at this low level. And the increase in arsenic from 3 to 35 $\mu\text{g}/\text{L}$ indicates the potential for leaching of selected trace elements from even uncontaminated soil.

In summary, the batch-elutriation experiments done for this study demonstrate the potential for leaching of urban-runoff contaminants as water in recharge basins infiltrates the basin's sediment. However, the results of similar experiments to test the ability of uncontaminated substrata to remove those contaminants were inconclusive—in part, because of the significant natural abundances of some inorganic elements; because of limitations on the colorimetric methods used to analyze broad organic indicators; and because of the possible presence of colloidal material

in the centrifuged elutriate itself. Theoretical considerations based on the physical and chemical properties of contaminants in a circumneutral (or slightly alkaline) environment and monitoring of soil moisture and ground-water chemistry provide more reliable support for evaluating transport of urban-runoff contaminants to the water table than do the results of these elutriation experiments. The observation that concentration gradients show a sharp decrease in sediment beneath the recharge basin, and the likelihood that water-infiltration rates are sufficiently slow for the soil moisture to, in effect, undergo a series of "batch-removal" processes as it percolates downward through the soil, ensures the effective removal of all urban runoff contaminants, with the possible exception of those having negative charge (anions) or negative polarity.

Perhaps the most important finding obtained from these laboratory experiments comes from analyses of the sediment and soil before (table 14) and after (table 15) the elutriation tests. The data in table 14 were obtained from complete mineral digestion with strong acids, whereas those in table 15 are based on oxidation of organic matter with H_2O_2 followed by only partial digestion with 0.3M HCL (Fishman and Friedman, 1989). The mild-acid digestion is intended to remove only surface layers while leaving the aluminosilicate mineral matrix itself largely intact. Among the alkali- and alkaline-earth elements, the mild-acid treatment removed virtually no calcium, magnesium, and sodium but did leach a substantial proportion of the total potassium. Percent recoveries from the contaminated basin sediment at 0 to 1 cm near site 5 and from clean aquifer soil at 10.9-11.3 m in test hole 1 can be obtained by dividing concentrations for partial digestion (table 15) by concentrations for complete digestion (table 14). Results of this calculation for several elements are summarized in table 16. These results show that a number of elements having high enrichment ratios (the contaminants in urban runoff) are almost entirely recoverable by the mild treatment and therefore are presumed to be present in surface coatings on mineral grains and in organic matter (Horowitz and Elrick, 1987). Percent recoveries for mild-acid digestion of the uncontaminated aquifer soil are somewhat less, but they also indicate that a substantial proportion of selected trace elements is likely associated with

Table 15. Concentrations of chemical constituents in recharge-basin sediment and aquifer soil after the laboratory elutriation tests

[Element concentrations, in micrograms per gram dry weight, are based on analysis of mild-acid leachate from sediment and soil. Pesticide concentrations are in micrograms per kilogram dry weight.]

Constituent	Basin sediment equilibrated with deionized water	Aquifer soil equilibrated with basin water	Constituent	Basin sediment equilibrated with deionized water	Aquifer soil equilibrated with basin water
Calcium	<10	<10	Organochlorine Compounds		
Magnesium	10	<10	Aldrin	<0.1	<0.1
Sodium	<10	<10	Lindane	51	<.1
Potassium	8,400	3,300	Chlordane	<1	1
Aluminum	22,000	8,200	DDD	17	.8
Arsenic	15	4	DDE	50	.6
Boron	20	<10	DDT	.6	<.1
Cadmium	5	<1	Dieldrin	5.3	<.1
Chromium	40	20	Endosulfan	10	<.1
Cobalt	<50	<50	Endrin	<.1	<.1
Copper	180	17	Ethion	<.1	<.1
Iron	36,000	13,000	Toxaphene	<10	<10
Lead	200	<10	Heptachlor	52	<.1
Manganese	580	260	Heptachlor epoxide	8.1	<.1
Mercury	.24	.02	Methoxychlor	<.1	<.1
Molybdenum	3.8	<1	PCN	<1	<1
Nickel	50	60	PCB	26	<1
Zinc	1,600	35	Organophosphorus Insecticides		
Nitrite + nitrate (as N)	6,000	3,000	Malathion	<0.1	<0.1
Ammonium (as N)	380	160	Parathion	<.1	<.1
Kjeldahl nitrogen (as N)	13,000	460	Diazinon	<.1	<.1
Phosphorus (as P)	2,300	1,000	Methyl parathion	<.1	<.1
Organic carbon (as C)	140,000	3,000	Mirex	<.1	<.1
Inorganic carbon (as C)	100	100	Ethyl trithion	<.1	<.1
Oil and grease	20	<1	Methyl trithion	<.1	<.1
			Perthane	<1	<1

Table 16. Percent extractable for selected elements in recharge-basin sediment and aquifer soil by digestion with hydrogen peroxide plus mild acid

[Calculated from data in tables 14 and 15. Basin sediment is from top 1 centimeter near site 5, and aquifer soil is from 10.9–11.3 meters in test hole 1. >, greater than; <, less than; --, indeterminate value]

Element	Percent extractable	
	Basin sediment	Aquifer soil
Aluminum (Al)	30	9
Arsenic (As)	83	>100
Chromium (Cr)	46	14
Copper (Cu)	>100	69
Iron (Fe)	95	34
Lead (Pb)	>100	<71
Manganese (Mn)	92	44
Mercury (Hg)	89	--
Nickel (Ni)	76	43
Potassium (K)	38	18
Zinc (Zn)	>100	51

hydrous ferromanganese oxides present either as coatings on mineral grains or disseminated throughout the uncontaminated soil.

TRANSPORT AND LOADING OF CONTAMINATED SEDIMENT

When results of this study are compared with results from the NURP study, some differences emerge that raise questions about how contaminants in the urban environment are delivered to the recharge basin and about the quantity of contaminants actually present in the basin. Although these are important questions, it is emphasized that they do not change this study's principal conclusion that urban contaminants are not reaching the ground water, nor do they affect sediment-disposal considerations (following periodic scraping of the basin's floor) that are based on compliance with sediment-quality criteria or standards.

Table 17. Annual loads to the industrial recharge basin determined from runoff monitoring and sediment inventories

[Annual loads in kilograms per year. Inventory loads are converted to values based on mild-acid, partial digestion (see text) for direct comparison with runoff loads, adjusted to mean annual rainfall in Fresno, obtained by Olman and Shulters (1989)]

Element	Annual load, determined from:		Inventory/runoff ratio
	Runoff	Inventory	
Aluminum (Al)	340	1,300	3.7
Arsenic (As)	.93	.57	.6
Organic carbon (as C)	11,000	5,700	.5
Chromium (Cr)	1.1	1.8	1.6
Copper (Cu)	3.5	5.1	1.5
Iron (Fe)	570	2,000	3.5
Lead (Pb)	4.5	7.0	1.6
Manganese (Mn)	19	33	1.8
Mercury (Hg)	.013	.0088	.7
Nickel (Ni)	1.3	2.3	1.8
Zinc (Zn)	26	38	1.4
Sediment	30,000	56,000	1.8

COMPARISON OF BASIN-SEDIMENT INVENTORIES AND STORM-RUNOFF LOADS

Storm-runoff loads to the industrial basin, calculated from average-annual constituent loads estimated by Olmann and Shulters (1989, table 25) on the basis of monitoring for the NURP study during the 1981-82 and 1982-83 wet seasons, are given in table 17. Quantities of constituents found in this study to have accumulated in the recharge-basin's sediment during operation in its present configuration from 1982 to 1986 also are given in the table. In order to compare annual loads estimated by means of the two methods, it is necessary to adjust the sediment inventories for differences in recovery between the mild-acid digestion used during the NURP study to analyze runoff samples and the complete mineral digestion used in this study to analyze basin sediment. This was done by multiplying total-constituent amounts measured in the sediment by recovery percentages from the "Basin sediment" column in table 16, thereby providing a comparison that is based on element recoveries from the partial-digestion analytical method. [Alternatively, an equivalent comparison could be based on loads determined from complete-digestion methods by dividing the runoff-load values of Olmann and Shulters (1989) reproduced in table 17 by the recovery percentages.] Complete recovery (100 percent) was used in the calculation for organic carbon, copper, lead, and zinc. The concentration

profiles at site 5, given in table 9, were used to calculate constituent quantities in the basin sediment. Quantities are calculated by multiplying concentrations, basin area, sediment dry-bulk density, and subbottom depth. The error that is propagated by uncertainty in each of these parameters is discussed below.

The uncertainty in element concentrations that results from analytical imprecision is small: an average of only about 5 percent on the basis of analysis of duplicate samples. The error that results from spatial variability in concentration is much greater: as much as a factor of about 2 (100 percent) for many of the urban contaminants in surface sediment (0-1 cm) at six locations in the recharge basin (table 8). However, the effect of such large spatial variability is deceptive because nearly all of the basin floor's area is in the south end, where the variability obtained by comparing surficial concentrations at sites 5 and 6 is only about 5 percent (table 8), and by comparing concentrations at site 5 with those from a broad area near site 5 (table 13) is only about 8 percent. The area of the basin floor that contains contaminated sediment is difficult to estimate because the sediment reaches an indeterminate distance up the sloping sides of the basin. The basin-floor area was measured in the field to be about 3.72×10^7 cm² (40,000 ft² or about 1 acre) with a possible error of about 25 percent.

Sediment dry-bulk density (ρ in equation 1 later) was not measured in this study, but a value of 1.5 g/cm^3 was chosen because it was measured by Nightingale (1989) for a sediment with similar grain-size distribution from another recharge basin in the city of Fresno. True density of well-homogenized sediment samples is likely to differ little from this estimate, although minor fluctuations with depth can be expected. A far greater error arises from the unusual open texture of sediment in the top few centimeters referred to in the "Sampling of Sediment from the Recharge Basin" section of this report.

These irregularities in sediment texture, and thinning of the contaminated zone along the sideslopes and channel in the northern part of the basin, create a rather large uncertainty in precise assignment of subbottom depths as well as some ambiguity in choosing the maximum depth to be used in calculating sediment inventories. Four centimeters was chosen as the maximum depth to use because sediment below this depth contained no excess (atmospheric) lead-210 ($^{210}\text{Pb}_{\text{ex}}$).

When each of the various sources of error is considered, the accuracy of calculated annual-contaminant inventories in the sediment is thought to be no better than a factor of 2 or 3. As noted, most of the imprecision is attributable to uncertainty in the determination of depth and bulk density related to the unusual sediment texture. Guay and Smith (1988) also recognized the large error in simulating a relation between quantity and quality of storm runoff in the industrial catchment and therefore developed simulations for residential and commercial catchments only. Failure of their model for the industrial catchment may be caused by discharge from large point sources unrelated to rainfall, in contrast to residential and commercial catchments, where nonpoint (as numerous small-point) contributions predominate. Median errors using data from individual storms for their simulation in a commercial catchment were 11 percent for lead and 54 percent for suspended sediment. However, even these rather large errors in estimating urban-runoff loads are much less than differences between the results obtained using the two methods described in this report. The differences for selected constituents are expressed as a ratio between the sediment-inventory and runoff-monitoring load and are given in table 17. Possible reasons for the differences include: (1) errors in one, or both, of the methods used to estimate annual loads; (2) real differences between the period of storm monitoring during 1981-83 and the period of sediment

accumulation during 1982-86; and (3) substantial delivery of sediment containing urban contaminants by some mechanism in addition to storm runoff, such as overland flow and erosion on the perimeter of the basin or atmospheric transport (dryfall).

Contaminated sediment contained in the recharge basin exceeds the amount delivered as suspended sediment in storm runoff by a factor of 1.8 (table 17). The difference is only slightly less for several elements (Cr, Cu, Pb, Mn, Ni, and Zn), possibly reflecting minor dilution with sediment (soils or substrata) containing little or no urban contaminants. The close similarity in loading patterns for these elements and for sediment is especially surprising given the very different procedures and assumptions used to obtain the two estimates. These results suggest that a single explanation may exist for the differences, such as a systematic error in the estimated quantity of sediment. Furthermore, there are simple explanations to reconcile inventory-runoff differences that are either much greater (aluminum and iron) or much less (arsenic, organic carbon, and mercury) than 1.8 for five other elements.

The amount of recoverable aluminum and iron contained in basin sediment relative to runoff apparently is overestimated because the digestion procedure produced uncharacteristically high recoveries for the strongly anaerobic sediments from the basin in comparison with the much lower recoveries of both aluminum and iron for clean aquifer soil (table 16). Possible explanations for those elements that appear to be depleted in the basin sediment (arsenic, organic carbon, and mercury) are much more varied.

The subbottom concentration profile from site 5 (table 9 and fig. 4) clearly shows that some arsenic has migrated beyond the 4-cm depth. The amount of arsenic that migrated was not included in the inventory calculation, which was cut off at 4 cm to maintain internal consistency among all elements. The greater mobility for arsenic, in comparison with many other trace elements that exist as cations, may be attributed to its possible existence as the more soluble oxyanion.

Mercury concentrations are so low that analytical errors and loss during storage and processing of the sediment samples must be considered; however, natural loss by volatilization of the element or its methylated form within the basin (Compeau and Bartha, 1985) also is possible.

Because the concentration of dissolved organic carbon in soil water (and ground water) generally is low and sorption of organic carbon to solid-particle surfaces typically is high, transport of organic carbon with downward-percolating recharge is likely to be a minor mechanism for its removal. The main removal mechanism for organic carbon is believed to be biological degradation within the subbottom sediment; the data in table 17 indicate that at least one-half the organic carbon delivered in runoff must be decomposed. In fact, the fraction of organic carbon actually mineralized (degraded) must be even greater than one-half because not only allochthonous carbon in runoff but also autochthonous carbon produced biologically within the pond itself is delivered to the subbottom sediment. The basin receives an average annual load of 730 kg total nitrogen. Dissolved phosphorus also was noted to be especially high in runoff from the industrial catchment in comparison with runoff from the residential and commercial catchments (Oltmann and Shulters, 1989). It is this rich supply of nutrients which results in the deep green color (from algae) of the pond itself and in the existence of strongly anaerobic sediment beneath the basin, and which provides an environment that supports extensive degradation of organic carbon.

LEAD-210 CONSTRAINTS ON ORIGIN AND LOADING OF CONTAMINATED SEDIMENT

Lead-210 occurs naturally as a product of the uranium-238 decay series and has a half-life of 22 years. Lead-210 has been used widely for dating lacustrine and coastal sediments that have accumulated during the last 100 years (Faure, 1977). For this study, the primary purpose in obtaining ^{210}Pb data was to ascertain the maximum depth at which inorganic elements from urban runoff occur in basin sediment. This radioisotope is especially well suited to this purpose because (1) its chemical and physical properties are expected to be similar to those of other inorganic cations, especially nonradiogenic lead, and thus it would be expected to transport similarly and (2) that portion of the sedimentary ^{210}Pb derived from atmospheric fallout, the $^{210}\text{Pb}_{\text{xs}}$ (excess lead-210), will be completely absent in the much older substrata that lie beneath the basin's recently deposited sediment if no downward transport of the isotope has occurred. Lead-210 also may provide useful insight into the origin (mechanism of delivery) and loading of urban-contaminated sediment to recharge basins.

Atmospheric ^{210}Pb is produced when ^{226}Ra in the Earth's crust decays to ^{222}Rn , an inert gas that effuses into the atmosphere, where it subsequently decays through a series of very-short-lived daughters to ^{210}Pb and falls back to the Earth's surface. Because secular (long-term) equilibrium is presumed to exist between sedimentary (or soil) ^{226}Ra and its decay products, the $^{210}\text{Pb}_{\text{xs}}$ activity can be calculated as total ^{210}Pb activity minus ^{226}Ra activity (see Graustein and Turekian, 1986, for a more complete discussion). For this study, activities are given in dpm (disintegrations per minute), which can be converted to other units of activity commonly used in the literature with the relation: 1 pCi (picocurie) = 2.2 dpm = 37 mBq (millibecquerels). Crustal variations in concentration of the parent isotope from which ^{210}Pb is derived and rapid removal of ^{210}Pb from the atmosphere in comparison with global circulation times causes regional variations in the flux of atmospheric ^{210}Pb . On the basis of measurements in soil and precipitation, Moore and Poet (1976) reported values of 0.53 to 1.0 (dpm/cm²)/yr for oceanic fluxes, 0.3 to 2.0 (dpm/cm²)/yr for continental fluxes in the north temperate zone, and 1.6 (dpm/cm²)/yr from surface-soil profiles in Colorado, Texas, and Kansas. Graustein and Turekian (1986) reported values ranging from 0.6 to 1.2 from undisturbed soil in 19 locations throughout the central and eastern United States. Published data from several locations on the west coast yield a range of about 0.2 to about 0.7 (dpm/cm²)/yr, with latitude, longitude, and precipitation quantity influencing the flux (Fuller and Hammond, 1983)—and a value of about 6 (dpm/cm²)/yr in Emerald Lake (calculated from data in Holmes and others, 1989, fig. 2). Although Emerald Lake is only 110 km east-southeast of Fresno in an alpine watershed on the west slope of the Sierra Nevada, its high ratio of watershed to lake area (almost 50) and the steep slopes of this cirque lake are likely to result in an anomalously high atmospheric-flux estimate. Although data are not available for the city of Fresno, proximity to the Pacific Ocean suggests that the ^{210}Pb flux should be near the low end of the range of continental values and likely no greater than about 1 (dpm/cm²)/yr. Data from undisturbed soil or from atmospheric precipitation (wet and dry deposition) in the Fresno area would be needed to determine the local flux more accurately.

Annual deposition of $^{210}\text{Pb}_{\text{xs}}$ in a sediment profile can be calculated from:

$$^{210}\text{Pb}_{\text{xs}} \text{ flux} = \frac{\sum_i [^{210}\text{Pb}_{\text{xs}}]_i L_i \rho}{1 - e^{-\lambda t}} \quad (1)$$

where λ is the decay constant (0.0311 yr^{-1}), ρ is dry-bulk density (1.5 g/cm^3), t is 4 years, and the other terms are defined in the following paragraph. The above formulation yields the same result as an alternative method chosen to illustrate more clearly the effect of errors in basin area and sediment mass on the result. The method also is analogous to that used for calculating element inventories in table 17.

Annual deposition of $^{210}\text{Pb}_{\text{xs}}$ to the basin also can be calculated by substituting data (in table 9) from site 5, where a complete activity profile exists, into the equation:

$$^{210}\text{Pb}_{\text{xs}} \text{ flux} = \frac{M}{t \cdot A \cdot L} \sum_i [^{210}\text{Pb}_{\text{xs}}]_i L_i \quad (2)$$

where

- $[^{210}\text{Pb}_{\text{xs}}]$ is the activity in sediment interval i ,
- L_i is the length of sediment interval i
- L is the sediment depth (4 cm) to which $^{210}\text{Pb}_{\text{xs}}$ is present,
- M is mass of sediment in the basin,
- t is length of time (4 years) the basin has been operated since last scraped, and
- A is area of the basin ($3.72 \times 10^7 \text{ cm}^2$).

Using the estimated sediment-accumulation rate (M/t) of 56,000 kg/yr given in table 17, and adjusting upward for the small amount of decay that took place during 4 years, yields a $^{210}\text{Pb}_{\text{xs}}$ deposition rate of 6.4 $(\text{dpm/cm}^2)/\text{yr}$. This value is markedly higher than the expected atmospheric flux of about 1 $(\text{dpm/cm}^2)/\text{yr}$. Fluxes determined from sediment data in surface-water bodies often are higher than atmospheric deposition rates and the difference is attributed to sediment "focusing"—the preferential scavenging on fine-grained material at the location of the cored sediment. For example, in four cores from the NURP study site in Irondequoit Bay, New York, fluxes were found to range from about 1 to 3 $(\text{dpm/cm}^2)/\text{yr}$ (Schroeder, 1985). The spatial trend shown by surficial activities that are given in table 8 provides evidence that focusing occurs in the recharge basin.

However, the equation above hints at a simpler explanation that reconciles the difference—a decrease in the sediment mass by a factor of 2 and an increase in the effective basin area by a factor of 3.

Monitoring of suspended-sediment concentration in runoff yielded the lower estimate (in relation to inventory estimate) for average annual sediment delivered to the recharge basin, and because total rainfall in Fresno during 1982-86 was only a little above average, annual sediment deliveries during this period also should be near the annual average. Erosion from the sideslopes and perimeter to the floor of the recharge basin plausibly could increase the effective area of deposition by a factor of 3 (compare total basin area to bottom area in figure 2). Such sediment may have been deposited initially as dryfall, whose contribution must be small in comparison with that of runoff, or the pattern and level of contamination in the dryfall and overland flow on the sideslopes must be similar to that in runoff in order to maintain the similarity in pattern shown by several different elements that is indicated in table 17.

The discussion above illustrates how, with relatively few measurements in comparison with much more intensive monitoring of runoff, reasonable estimates can be obtained for the quantity of contaminated sediment in an urban-runoff recharge basin. More detailed studies are needed to refine and test the validity of the method. Such studies might preferably be done in turfed basins that have been operated for a much longer time than was the subject basin in this study, thereby minimizing possible effects of erosion from the sides of the basin and providing a much thicker sediment veneer in which depths and bulk density could be measured more accurately.

SUMMARY AND CONCLUSIONS

Previous monitoring of storm-runoff quantity and quality, during 1981-83, in Fresno revealed elevated concentrations of urban contaminants in residential, commercial, and industrial catchments. The monitoring was accompanied by studies that confirmed the presence of these contaminants within the top few centimeters of sediment beneath recharge basins constructed to dispose of runoff from residential and commercial catchments. This study extends those previous studies to a 0.4-ha industrial-catchment basin where some urban contaminants were expected to be

present at even higher concentrations than in residential- and commercial-catchment basins. This study also shows that a wide variety of inorganic and organic contaminants generated in the 1.13-km² industrial catchment are removed by sorption within the top 4 cm of sediment in the recharge basin, and that the contaminants have not degraded ground-water quality beneath the basin.

The absence of local ground-water-quality impairment is documented by comparing chemical concentrations in two water-table monitoring wells to aqueous concentrations in the recharge basin. Shallow ground water near the basin contains a significant proportion of partly evaporated runoff from the basin as shown by measured delta-deuterium values of about -50 permil in both monitor wells in comparison with reported values of about -90 permil in regional ground water and -16 permil in the recharge basin itself. Because aqueous zinc concentration is about 100 times higher in the recharge basin than in the monitor wells, this element is an especially sensitive surrogate for assessing potential ground-water degradation by inorganic cations. Organic contaminants either were absent or were detected at such low concentrations (near reporting limits) that evidence for their presence in the monitor wells is questionable.

The range in chemical concentration of many urban contaminants in surface (0-1 cm) sediment from six locations on the floor of the recharge basin was a factor of about 2. Concentrations were lowest in channels at the north end of the basin nearest the pipe from which runoff discharges, and highest at the south end in the broad, topographically lowest part of the basin. The proportion of fine sediment also is highest at the south end, as would be expected. Profiles at a representative location in the south end of the basin show that concentrations of nearly all contaminants decrease to background levels at a depth of 4 cm, which also was the maximum depth to which atmospheric ²¹⁰Pb (excess ²¹⁰Pb) penetrates; no contaminants generated in the urban environment were detected below a depth of 16 cm.

Surface-enrichment ratios, defined as the ratio of concentrations in the top centimeter to concentrations in substrata below 16 cm, provide a quantitative means to compare anthropogenic (human related) and natural sources. Accordingly, a ratio of 2 is consistent with equal inputs from human and natural sources. Twenty inorganic elements were found to have enrichment ratios of 2 or greater. The value for zinc, which had the highest ratio, was 38. Values for several other elements were: arsenic=16, chromium=5,

copper=25, lead=9, mercury>11, nickel=7, phosphorus=5, and organic carbon=10³.

Enrichment ratios cannot be calculated for the approximately 20 organic compounds identified in sediment because the compounds are absent at depth. Nearly all the organic compounds detected in sediment are either organochlorine pesticides or polycyclic aromatic hydrocarbons. Among compounds in these two groups, chlordane and pyrene show the highest concentrations. Curiously, concentrations of the banned pesticide DDT exceed those of its possible metabolites DDD and DDE, a finding also noted in previous studies of urban runoff in the city of Fresno.

Estimates of sediment quantity and element load (total recoverable by mild-acid digestion) loads obtained in this study from a single core in the recharge basin exceed previously published estimates that were based solely on storm-runoff monitoring by a factor of 1.8. Possible explanations for this study's higher estimates are imprecision in the determination of sediment depths, the unusually open texture of the sediment veneer that had accumulated during the 4-year period the basin was operating, and thinning of the sediment layer in parts of the basin.

The flux of atmospheric ²¹⁰Pb, calculated to be 6.4 (disintegrations per minute/cm²)/yr using data from the only core obtained in the basin, is much higher than the maximum expected atmospheric flux of about 1 (disintegrations per minute/cm²)/yr. This difference can be reconciled if 1982-86 sediment contained in the basin is reduced by half and if the effective area of deposition is increased to 3 times larger than the approximately 0.4-ha floor of the basin. Erosion from the perimeter and sideslopes of the basin could provide the additional area required. This discussion illustrates how acquisition of ²¹⁰Pb data can be used to check estimated loads of sediment and urban contaminants delivered to recharge basins without the numerous analyses and intensive sample collection necessary to monitor runoff itself. However, additional data collection is needed to confirm and refine the method.

References Cited

- Arbogast, B.F., ed., 1990, Quality assurance manual for the Branch of Geochemistry: U.S. Geological Survey Open-File Report 90-668, 183 p.
- Brown and Caldwell Consulting Engineers, 1984, Fresno nationwide urban runoff program project: Final report prepared for the Fresno Metropolitan Flood Control District, 9 chapters, 3 appendixes, 165 p.

- Compeau, G.C., and Bartha, R., 1985. Sulfate-reducing bacteria: principal methylators of mercury in anoxic estuarine sediments: *Applied Environmental Microbiology*, v. 50, p. 498-502.
- Craig, H., 1961. Isotopic variations in meteoric waters: *Science*, v. 133, p. 1702-1703.
- Croft, M.G., and Gordon, G.V., 1968. Geology, hydrology, and quality of water in the Hanford-Visalia area. San Joaquin Valley, California: U.S. Geological Survey Open-File Report, 170 p.
- Davis, G.H., and Poland, J.F., 1957. Ground-water conditions in the Mendota-Huron area, Fresno and Kings Counties, California: U.S. Geological Survey Water-Supply Paper 1360-G, p. 409-588.
- Davis, G.H., Green, J.H., Olmsted, F.H., and Brown, D.W., 1959. Ground-water conditions and storage capacity in the San Joaquin Valley, California: U.S. Geological Survey Water-Supply Paper 1469, 287 p.
- Davis, G.H., Lofgren, B.E., and Mack, Seymour, 1964. Use of ground-water reservoirs for storage of surface water in the San Joaquin Valley, California: U.S. Geological Survey Water-Supply Paper 1618, 125 p.
- Driver, N.E., Mustard, M.H., Rhinesmith, R.B., and Middleburg, R.F., 1985. U.S. Geological Survey urban-stormwater data base for 22 metropolitan areas throughout the United States: U.S. Geological Survey Open-File Report 85-337, 219 p.
- Durnford, D.S., Brookman, J.A., Bilica, J.A., and Milligan, J.D., 1991. LNAPL distribution in a cohesionless soil: a field investigation and cryogenic sampler: *Ground Water Monitoring Review*, v. 11, no. 3, p. 115-122.
- Faure, Gunter, 1977. Principles of isotope geology: chap. 16. The U-series disequilibrium methods of dating: New York, Wiley, p. 283-304.
- Fishman, M.J., and Friedman, L.C., eds, 1989. Methods for determination of inorganic substances in water and fluvial sediments: U.S. Geological Survey Techniques of Water-Resources Investigations, Book 6, Chapter A1, 545 p.
- Freeze, R.A., and Cherry, J.A., 1979. *Groundwater*: Englewood Cliffs, N.J., Prentice-Hall, 604 p.
- Fresno Metropolitan Flood Control District, 1987. Fresno Nationwide Urban Runoff Program project: Final Report. Fresno, California, 164 p.
- Friedman, Irving, Smith, G.I., Gleason, J.D., Warden, Augusta, and Harris, J.M., 1992. Stable isotope composition of waters in southeastern California: Part I. Modern precipitation: *Journal of Geophysical Research*, v. 97, p. 5795-5812.
- Fuhrer, G.F., and Rinella, F.A., 1983. Analyses of elutriates, naive water, and bottom material in selected rivers and estuaries in western Oregon and Washington: U.S. Geological Survey Open-File Report 82-922, 147 p.
- Fuller, Christopher, and Hammond, D.E., 1983. The fallout rate of Pb-210 on the western coast of the United States: *Geophysical Research Letters*, v. 10, p. 1164-1167.
- Goddard, E.N., Trask, F.D., De Ford, R.K., Rove, O.N., Singewald, J.T., Jr., and Overback, R.M., 1948. Rock-color chart: Geological Society of America, reprinted 1984.
- Graustein, W.C., and Turekian, K.K., 1986. ^{210}Pb and ^{137}Cs in air and soils measure the rate and vertical profile of aerosol scavenging: *Journal of Geophysical Research*, v. 91, p. 14,355-14,366.
- Guay, J.R., and Smith, P.E., 1988. Simulation of quantity and quality of storm runoff for urban catchments in Fresno, California: U.S. Geological Survey Water-Resources Investigations Report 88-4125, 76 p.
- Guy, H.P., 1969. Laboratory theory and methods for sediment analysis: U.S. Geological Survey Techniques of Water-Resources Investigations, Book 5, Chapter C1, 58 p.
- Harter, R.D., 1983. Effect of soil pH on adsorption of lead, copper, zinc, and nickel: *Soil Science Society of America Journal*, v. 47, p. 47-51.
- Holmes, R.W., Whiting, M.C., and Stoddard, J.L., 1989. Changes in diatom-inferred pH and acid neutralizing capacity in a dilute, high-elevation, Sierra Nevada lake since A.D. 1825: *Freshwater Biology*, v. 21, p. 295-310.
- Horowitz, A.J. and Elrick, K.A., 1987. The relation of stream sediment surface area, grain size and composition to trace element chemistry: *Applied Geochemistry*, v. 2, p. 437-451.
- Ingraham, N.L., and Taylor, B.E., 1991. Light stable isotope systematics of large-scale hydrologic regimes in California and Nevada: *Water Resources Research*, v. 27, p. 77-90.
- Kappel, W.M., Yager, R.M., and Zarniello, P.J., 1986. Quantity and quality of urban storm runoff in the Irondequoit Creek basin near Rochester, New York. Part 2. Quality of storm runoff and atmospheric deposition, rainfall-runoff-quality modeling, and potential of wetlands for sediment and nutrient retention: U.S. Geological Survey Water-Resources Investigations Report 85-4113, 93 p.
- Ku, H.F.H., and Simmons, D.L., 1986. Effect of urban stormwater runoff on ground water beneath recharge basins on Long Island, New York: U.S. Geological Survey Water-Resources Investigations Report 85-4088, 67 p.
- Lippincott, J.B., 1902. Storage of water on Kings River, California: U.S. Geological Survey Water-Supply and Irrigation Paper 58, 101 p.
- Meade, R.H., 1967. Petrology of sediments underlying areas of land subsidence in central California: U.S. Geological Survey Professional Paper 497-C, 83 p.
- Mendenhall, W.C., Dale, R.B., and Stabler, Herman, 1916. Ground water in San Joaquin Valley, California: U.S. Geological Survey Water-Supply Paper 398, 310 p.
- Michel, R.L., 1989. Tritium deposition in the continental United States, 1953-83: U.S. Geological Survey Water-Resources Investigations Report 89-4072, 46 p.

- Mitten, H.T., 1984, Ground water in the Fresno area. California--preliminary report: U.S. Geological Survey Water-Resources Investigations Report 83-4246, 15 p.
- Mitten, H.T., and Guay, J.R., 1986, A summary of municipal pumpage for the Fresno area, California, 1931-80: U.S. Geological Survey Open-File Report 86-492, 21 p. and plate.
- Moore, H.E., and Poet, S.E., 1976, ^{210}Pb fluxes determined from ^{210}Pb and ^{226}Ra soil profiles: *Journal of Geophysical Research*, v. 81, p. 1056-1058.
- Nightingale, H.I., 1975, Lead, zinc, and copper in soils of urban storm-runoff retention basins, *American Water Works Association Journal*, v. 67, p. 443-446.
- _____, 1987a, Water quality beneath urban runoff water management basins: *Water Resources Bulletin*, v. 23, p. 197-205.
- _____, 1987b, Accumulation of As, Ni, Cu, and Pb in retention and recharge basin soils from urban runoff: *Water Resources Bulletin*, v. 23, p. 663-672.
- _____, 1987c, Organic pollutants in soils of retention/recharge basins receiving urban runoff water: *Soil Science*, v. 144, p. 373-382.
- _____, 1989, Urban storm runoff effects on exchangeable cations and percolating water constituents below recharge basins: *Soil Science*, v. 148, p. 39-45.
- Nightingale, H.I., Harrison, D., and Salo, J.E., 1985, An evaluation technique for ground-water quality beneath urban runoff retention and percolation basins: *Ground Water Monitoring Review*, v. 5, no. 1, p. 43-50.
- Olmsted, F.H., and Davis, G.H., 1961, Geologic features and ground-water storage capacity of the Sacramento Valley, California: U.S. Geological Survey Water-Supply Paper 1497, 241 p.
- Oltmann, R.N., and Shulters, M.V., 1989, Rainfall and runoff quantity and quality characteristics of four urban land-use catchments in Fresno, California, October 1981-April 1983: U.S. Geological Survey Water-Supply Paper 2335, 114 p.
- Oltmann, R.N., Guay, J.R., and Shay, J.M., 1987, Rainfall and runoff quantity and quality data collected at four urban land-use catchments in Fresno, California, October 1981-April 1983: U.S. Geological Survey Open-File Report 84-718, 139 p.
- Page, R.W., 1975, Ground-water reconnaissance in the Fresno northeast area, Fresno County, California: U.S. Geological Survey Open-File Report 75-315, 34 p.
- Page, R.W., and LeBlanc, R.A., 1969, Geology, hydrology, and water quality in the Fresno area, California: U.S. Geological Survey Open-File Report, 70 p., 21 plates.
- Palermo, M.R., 1986, Interim guidance for predicting the quality of effluent discharge from confined dredge material disposal areas: U.S. Army Corps of Engineers Miscellaneous Paper D-86-1, 36 p.
- Schroeder, R.A., 1985, Sediment accumulation rates in Irondequoit Bay, New York based on lead-210 and cesium-137 geochronology: *Northeastern Environmental Science*, v. 4, p. 23-29.
- Schroeder, R.A., Palawski, D.U., and Skorupa, J.P., 1988, Reconnaissance investigation of water quality, bottom sediment, and biota associated with irrigation drainage in the Tulare lake bed area, southern San Joaquin Valley, California: U.S. Geological Survey Water-Resources Investigations Report 88-4001, 86 p.
- Severson, R.C., Wilson, S.A., and McNeal, J.M., 1987, Analyses of bottom material collected at nine areas in the western United States for the DOI irrigation drainage task group: U.S. Geological Survey Open-File Report 87-490, 24 p.
- Shelton, L.R., and Miller, L.K., 1991, Water-quality data, San Joaquin Valley, California, April 1987 to September 1988: U.S. Geological Survey Open-File Report 91-74, 189 p.
- U.S. Environmental Protection Agency, 1983, Results of the nationwide urban runoff program: U.S. Environmental Protection Agency, Water Planning Division, Washington D.C. 20460, v. 1, final report, National Technical Information Service (NTIS) Accession Number PB84-18552, 9 chapters.
- Wershaw, R.L., Fishman, M.C., Grabbe, R.R., and Lowe, L.E., eds, 1987, Methods for the determination of organic substances in water and fluvial sediments: U.S. Geological Survey Techniques of Water-Resources Investigations, Book 5, Chapter A3, 30 p.
- Wigington, P.J., Jr., Randall, C.W., and Grizzard, T.J., 1983, Accumulation of selected trace metals in soils of urban runoff retention basins, *Water-Resources Bulletin*, v. 19, p. 709-718.
- Williamson, A.K., Prudic, D.E., and Swain, L.A., 1989, Ground-water flow in the Central Valley, California: U.S. Geological Survey Professional Paper 1401-D, p. D1-D127.

Water Quality of Storm Runoff and Comparison of Procedures for Estimating Storm-Runoff Loads, Volume, Event-Mean Concentrations, and the Mean Load for a Storm for Selected Properties and Constituents for Colorado Springs, Southeastern Colorado, 1992

by Paul von Guerard and William B. Weiss

U.S. GEOLOGICAL SURVEY

Water-Resources Investigations Report 94-4194

Prepared in cooperation with the
COLORADO SPRINGS CITY ENGINEERING DIVISION

Denver, Colorado
1995



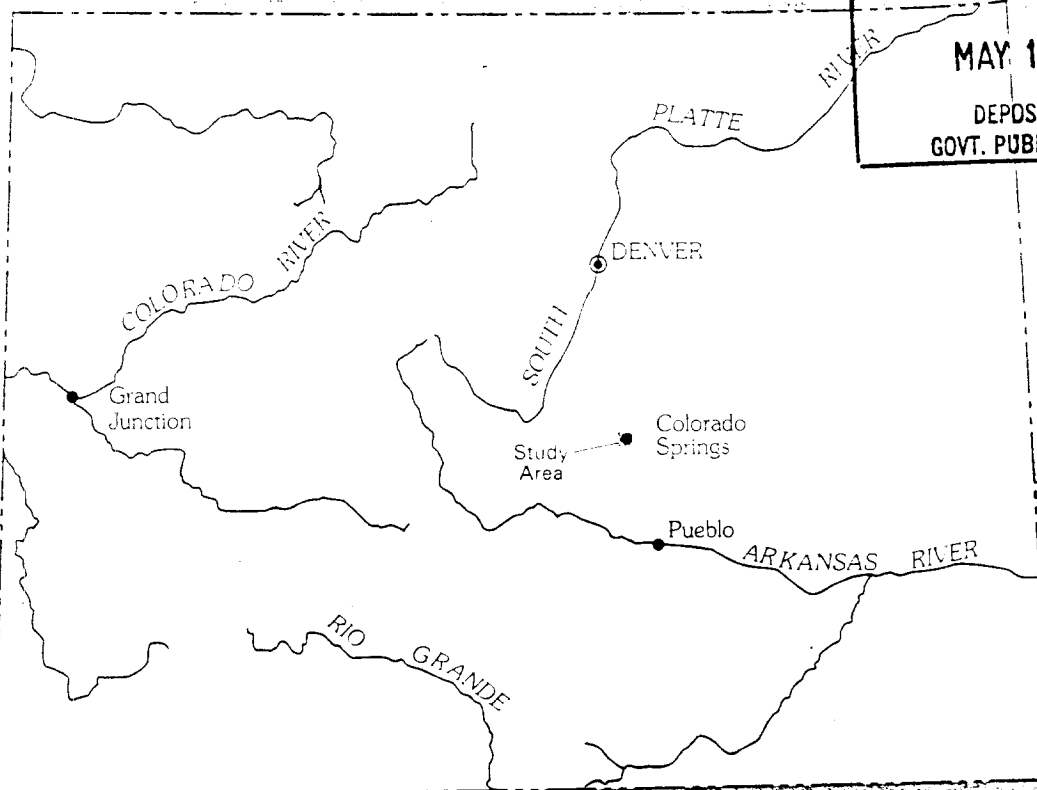
R0024078

cmr 5/11/95

WATER QUALITY OF STORM RUNOFF AND COMPARISON OF PROCEDURES FOR ESTIMATING STORM-RUNOFF LOADS, VOLUME, EVENT-MEAN CONCENTRATIONS, AND THE MEAN LOAD FOR A STORM FOR SELECTED PROPERTIES AND CONSTITUENTS FOR COLORADO SPRINGS, SOUTHEASTERN COLORADO, 1992

AJUL 9423

U.S. GEOLOGICAL SURVEY



LIBRARY
UNIVERSITY OF CALIFORNIA
SANTA BARBARA

MAY 10 1995

DEPOSITORY
GOVT. PUBLICATIONS

Water-Resources Investigations Report 94-4194

Prepared in cooperation with the
COLORADO SPRINGS CITY ENGINEERING DIVISION



GB
701
239
no. 94-4194

R0024079

Water Quality of Storm Runoff and Comparison of Procedures for Estimating Storm-Runoff Loads, Volume, Event-Mean Concentrations, and the Mean Load for a Storm for Selected Properties and Constituents for Colorado Springs, Southeastern Colorado, 1992

by Paul von Guerard and William B. Weiss

U.S. GEOLOGICAL SURVEY

Water-Resources Investigations Report 94-4194

Prepared in cooperation with the
COLORADO SPRINGS CITY ENGINEERING DIVISION

Denver, Colorado
1995



R0024080

Cont. 2/11/95

U.S. DEPARTMENT OF THE INTERIOR

BRUCE BABBITT, Secretary

U.S. GEOLOGICAL SURVEY

Gordon P. Eaton, Director

The use of trade, product, industry, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

For additional information write to:

District Chief
U.S. Geological Survey
Box 25046, MS 415
Denver Federal Center
Denver, CO 80225

Copies of this report can be purchased from:

U.S. Geological Survey
Earth Science Information Center
Open-File Reports Section
Box 25286, MS 517
Denver Federal Center
Denver, CO 80225

R0024081

CONTENTS

Abstract	1
Introduction	2
Purpose and scope	2
Description of study area and sampling sites	4
Acknowledgments.....	5
Data-collection techniques, water-quality-sampling procedures, and quality-assurance procedures	5
Precipitation and flow data-collection techniques	5
Water-quality-sampling procedures	6
Quality-assurance procedures	7
Water quality of storm runoff	7
Procedures for estimating single-storm-runoff loads, volume, and event-mean concentrations	10
Single-storm local-regression models	14
Single-storm regional-regression models	17
Comparison of observed and estimated single-storm-runoff loads, volume, and event-mean concentrations	17
Procedures for adjustment of estimates from single-storm regional-regression models using local data	20
Estimates of a mean load for a storm	29
Comparison of procedures for estimating storm-runoff loads, volumes, event-mean concentrations, and the mean load for a storm.....	32
Summary	33
References cited	35
Supplemental data	37

FIGURES

1. Map showing location of study area and storm-runoff-sampling sites	3
2. Diagram showing flowchart for selection of model adjustment procedures	24
3. Graph showing observed total-phosphorus, estimated total-phosphorus, and adjusted estimates of total-phosphorus loads for storm-runoff load at site 2	26
4. Graph showing observed total ammonia plus organic nitrogen as nitrogen, estimated total ammonia plus organic nitrogen as nitrogen, and adjusted estimates of total ammonia plus organic nitrogen as nitrogen for storm-runoff event-mean concentrations at site 2.....	27

TABLES

1. Description of and selected land-use data for storm-runoff-sampling sites in Colorado Springs	4
2. Summary statistics for selected constituents in storm runoff-sampling sites in Colorado Springs	8
3. Summary of detections of organic compounds in storm runoff-sampling sites in Colorado Springs	11
4. Summary of single-storm local-regression models for storm-runoff loads and volume for storm runoff-sampling sites in Colorado Springs.....	15
5. Summary of single-storm local-regression models for storm-runoff event-mean concentrations-sampling sites in Colorado Springs	16
6. Ranges of values of each explanatory variable used in single-storm local- and regional-regression models	18
7. Root-mean-square error from comparison of observed storm-runoff loads and volumes and estimated storm-runoff loads and volumes from single-storm regional-regression models	20
8. Summary of residual values for observed minus estimated values of storm-runoff loads, volume, and event-mean concentrations.....	21
9. Effect of model-adjustment procedures on root-mean-square error and standard error of estimate for storm-runoff loads and volume	25
10. Effect of model-adjustment procedures on standard error of estimate for storm-runoff event-mean concentrations.....	25
11. Summary of adjusted models for storm-runoff loads and volume.....	28

12. Summary of adjusted models for storm-runoff event-mean concentrations	28
13. Ranges of values of explanatory variables used in development of regression models for mean load for a storm	29
14. Comparison of computed mean load for a storm based on daily mean water discharge and land-use characteristics estimated mean load for a storm, and average of samples collected during rainfall runoff.....	30
15. Summary of r^2 values and standard error of estimate for regression models used to estimate mean load, in pounds, of a storm based on daily mean water discharge and land-use characteristics.....	31
16. Precipitation and runoff characteristics for samples collected at storm-runoff sites in Colorado Springs	38
17. Values for instantaneous water discharge, pH, specific conductance, water temperature, fecal coliform, fecal streptococci, residual chlorine, cyanide, oil and grease, and phenols for storm-runoff sites in Colorado Springs	39
18. Selected volatile-organic compounds for storm-runoff sites in Colorado Springs	41
19. Chemical oxygen demand, biochemical oxygen demand, specific conductance, alkalinity, dissolved solids, suspended solids, major ions, nutrients and total-recoverable metals for storm-runoff sites Colorado Springs	52
20. Selected acid-base/neutral organic compounds for storm-runoff sites in Colorado Springs	57
21. Selected pesticide compounds for storm-runoff sampling sites in Colorado Springs	64

CONVERSION FACTORS AND RELATED INFORMATION

Multiply	By	To obtain
cubic foot (ft ³)	0.028317	cubic meter
cubic foot per second (ft ³ /s)	0.028317	cubic meter per second
foot (ft)	0.3048	meter
gallon (gal)	3.785	liter
inch (in.)	25.4	millimeter
mile (mi)	1.609	kilometer
pound (lb)	0.454	kilogram
quart (qt)	0.9464	liter
square mile (mi ²)	2.590	square kilometer

Degree Fahrenheit (°F) can be converted to degree Celsius (°C) by using the following equation:
 $^{\circ}\text{C} = 5/9 (^{\circ}\text{F} - 32)$.

Degree Celsius (°C) can be converted to degree Fahrenheit (°F) by using the following equation:
 $^{\circ}\text{F} = 9/5 (^{\circ}\text{C} + 32)$.

The following terms and abbreviations also are used in this report:

- colonies per 100 milliliters (col/100 mL)
- microsiemens per centimeter at 25 degrees Celsius ($\mu\text{S}/\text{cm}$)
- milligram per liter (mg/L)
- microgram per liter ($\mu\text{g}/\text{L}$)

Water Quality of Storm Runoff and Comparison of Procedures for Estimating Storm-Runoff Loads, Volume, Event-Mean Concentrations, and the Mean Load for a Storm for Selected Properties and Constituents for Colorado Springs, Southeastern Colorado, 1992

By Paul von Guerard and William B. Weiss

Abstract

The U.S. Environmental Protection Agency requires that municipalities that have a population of 100,000 or greater obtain National Pollutant Discharge Elimination System permits to characterize the quality of their storm runoff. In 1992, the U.S. Geological Survey, in cooperation with the Colorado Springs City Engineering Division, began a study to characterize the water quality of storm runoff and to evaluate procedures for the estimation of storm-runoff loads, volume and event-mean concentrations for selected properties and constituents.

Precipitation, streamflow, and water-quality data were collected during 1992 at five sites in Colorado Springs. Thirty-five samples were collected, seven at each of the five sites. At each site, three samples were collected for permitting purposes; two of the samples were collected during rainfall runoff, and one sample was collected during snowmelt runoff. Four additional samples were collected at each site to obtain a large enough sample size to estimate storm-runoff loads, volume, and event-mean concentrations for selected properties and constituents using linear-regression procedures developed using data from the Nationwide Urban Runoff Program (NURP). Storm-water samples were analyzed for as many as 186 properties and constituents. The constituents measured include total-recoverable metals, volatile-organic compounds, acid-base/neutral organic compounds, and pesticides.

Storm runoff sampled had large concentrations of chemical oxygen demand and 5-day biochemical oxygen demand. Chemical oxygen

demand ranged from 100 to 830 milligrams per liter, and 5-day biochemical oxygen demand ranged from 14 to 260 milligrams per liter. Total-organic carbon concentrations ranged from 18 to 240 milligrams per liter. The total-recoverable metals lead and zinc had the largest concentrations of the total-recoverable metals analyzed. Concentrations of lead ranged from 23 to 350 micrograms per liter, and concentrations of zinc ranged from 110 to 1,400 micrograms per liter.

The data for 30 storms representing rainfall runoff from 5 drainage basins were used to develop single-storm local-regression models. The response variables, storm-runoff loads, volume, and event-mean concentrations were modeled using explanatory variables for climatic, physical, and land-use characteristics. The r^2 for models that use ordinary least-squares regression ranged from 0.57 to 0.86 for storm-runoff loads and volume and from 0.25 to 0.63 for storm-runoff event-mean concentrations. Except for cadmium, standard errors of estimate ranged from 43 to 115 percent for storm-runoff loads and volume and from 35 to 66 percent for storm-runoff event-mean concentrations. Eleven of the 30 concentrations collected during rainfall runoff for total-recoverable cadmium were censored (less than) concentrations. Ordinary least-squares regression should not be used with censored data; however, censored data can be included with uncensored data using tobit regression. Standard errors of estimate for storm-runoff load and event-mean concentration for total-recoverable cadmium, computed using tobit regression, are 247 and 171 percent.

Estimates from single-storm regional-regression models, developed from the Nationwide Urban Runoff Program data base, were compared with observed storm-runoff loads, volume, and event-mean concentrations determined from samples collected in the study area. Single-storm regional-regression models tended to overestimate storm-runoff loads, volume, and event-mean concentrations. Therefore, single-storm local- and regional-regression models were combined using model-adjustment procedures to take advantage of the strengths of both models while minimizing the deficiencies of each model.

Procedures were used to develop single-storm regression equations that were adjusted using local data and estimates from single-storm regional-regression equations. Single-storm regression models developed using model-adjustment procedures had standard errors of estimate smaller than the standard errors of estimate for the regional-regression equations. Reduction of standard error in percent ranged from -1,980 to -10.

Regression models that had been developed from the Nationwide Urban Runoff Program data base for estimating the mean load for a storm were evaluated. Mean load for a storm was estimated for selected constituents. Ninety-percent confidence intervals were computed for each mean load estimate. Estimated mean load for a storm was compared to mean load of a storm that was computed based on daily mean water discharge and land-use characteristics and was compared to the mean load from six samples collected during rainfall runoff. Generally, mean load for a storm, computed based on daily mean water discharge and land-use characteristics and on mean load from samples collected during rainfall runoff, was near or within the 90-percent confidence intervals for estimates of mean load for a storm.

INTRODUCTION

Urbanization usually increases the impervious area of a watershed, which increases storm-runoff rates and, subsequently, total volume of storm runoff. Associated with storm runoff are properties and constituents that can cause the degradation of water quality locally and in receiving waters downstream. Because of concerns about the effects of urban runoff on water quality, the Water Quality Act of 1987 contains provisions that specifically address storm-runoff discharges. The U.S. Environmental Protection Agency, under section 319

of the Water Quality Act of 1987, requires that States "assess the nature and extent of nonpoint sources of pollution." Section 402(p) of the same act requires that municipalities that have a population of 100,000 or greater obtain National Pollutant Discharge Elimination System (NPDES) permits to improve the quality of storm runoff.

Final rules published by the U.S. Environmental Protection Agency (1990) require that municipalities prepare permit applications to include, among other information, the following:

1. Characterization of the quantity and quality of storm runoff for three or more major storms at selected storm-water-discharge sites that represent different combinations of commercial, industrial, and residential land uses.
2. Estimates of annual-pollutant loads and event-mean concentrations for selected constituents for the cumulative discharges of storm-runoff discharge points in the study area.

In 1992, the U.S. Geological Survey, in cooperation with the Colorado Springs City Engineering Division, began a study to characterize the water quality of storm runoff in Colorado Springs and to compare techniques for the estimation of storm-runoff loads, volume, event-mean concentrations, and mean load for a storm for selected properties and constituents.

Purpose and Scope

This report presents water-quality data collected during 1992 to characterize the water quality of storm runoff in Colorado Springs. These data were collected to help meet the requirements of the NPDES permitting process. Precipitation, streamflow, and water-quality data were collected during 1992 at five sites in Colorado Springs (fig. 1, table 1). This report presents procedures for estimating storm-runoff loads, volume, and event-mean concentrations for selected properties and constituents at unmonitored sites in Colorado Springs and to make a comparison of several procedures for estimating storm-runoff loads, volume, and event-mean concentrations. In addition, the report presents estimates of a mean load for a storm.

Thirty-five samples were collected, seven at each of the five sites. At each site, three samples were collected for NPDES permitting purposes; two of these samples were collected during rainfall runoff, and one sample was collected during snowmelt runoff. Four additional samples were collected at each site to obtain a large enough sample size to estimate storm-runoff

2 **Water Quality of Storm Runoff and Comparison of Procedures for Estimating Storm-Runoff Loads, Volume, Event-Mean Concentrations, and the Mean Load for a Storm for Selected Properties and Constituents for Colorado Springs, Southeastern Colorado, 1992**

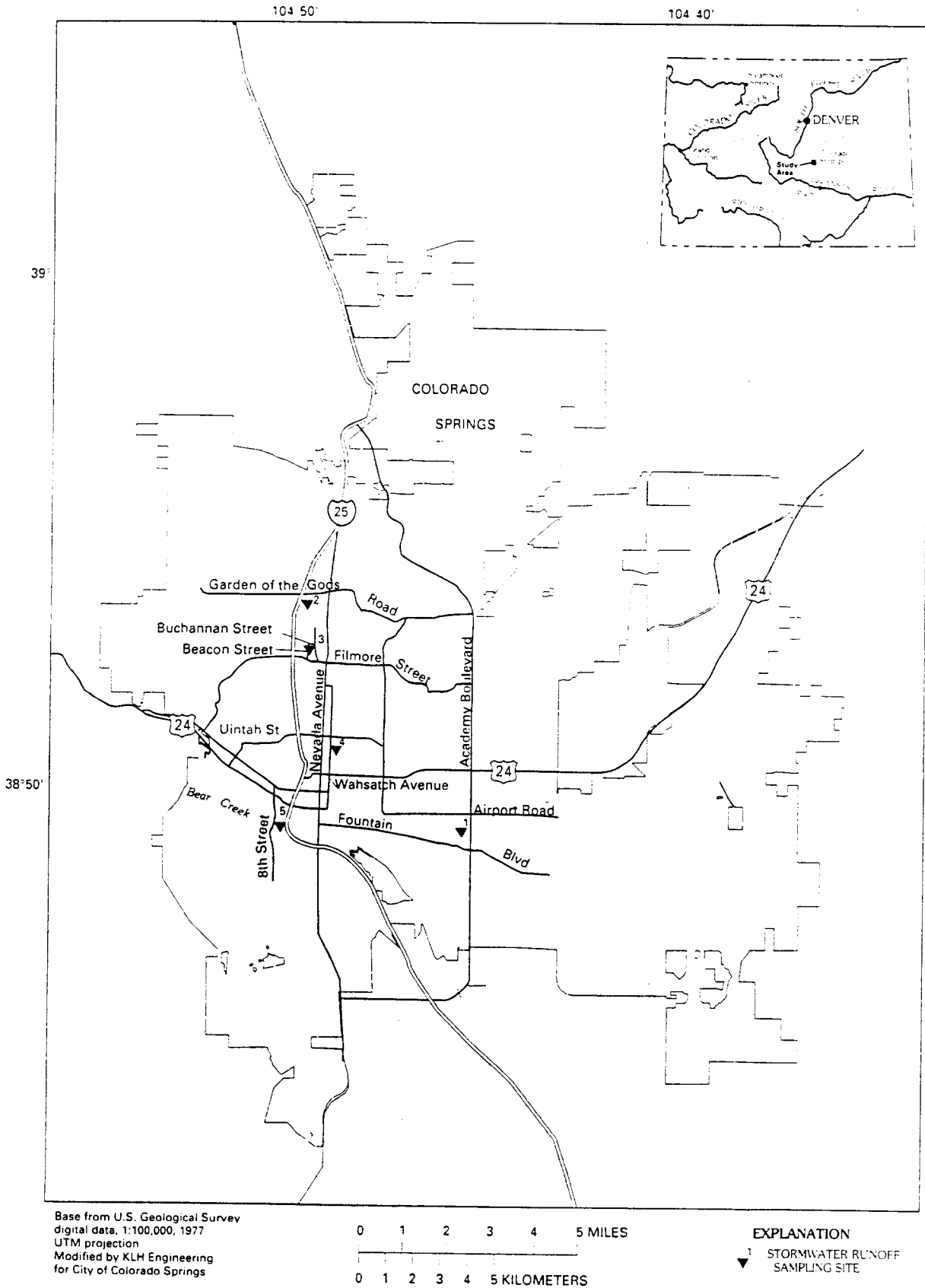


Figure 1. Location of study area and storm-runoff-sampling sites.

Table 1. Description of and selected land-use data for storm-runoff-sampling sites in Colorado Springs

Site number (fig. 1)	Site name	Latitude and longitude	Drainage-basin area (square miles)	Impervious area (percent)	Land use	Percentage of total drainage basin
1	Sixteenth Hole, Valley-Hi Golf Course	38°49'18" 104°45'42"	0.125	58.1	Commercial	61.1
					undeveloped	23.0
					residential	15.9
2	Chestnut Street at Douglas Creek	38°53'47" 104°50'06"	.165	37.5	Industrial	54.7
					undeveloped	35.9
					residential	8.5
					commercial	0.9
3	Beacon Street at Buchanan Street	38°52'40" 104°49'36"	.173	55.9	Industrial	79.5
					commercial	17.8
					residential	2.7
4	Wahsatch Street at Cross Lane	38°51'18" 104°48'58"	.327	34.2	Residential	79.4
					commercial	9.3
					public	8.3
					undeveloped	3.0
5	Walmart at Eighth Street	38°49'35" 104°50'15"	.049	40.1	Undeveloped	43.0
					commercial	39.9
					industrial	10.3
					residential	6.8

loads, volume, and event-mean concentrations for selected properties and constituents using linear-regression procedures developed using data from the Nationwide Urban Runoff Program (NURP). Storm-water samples were analyzed for as many as 186 properties and constituents (tables 16–21 in the "Supplemental Data" section at the back of this report). Some of the properties and constituents measured include pH, specific conductance, water temperature, chemical oxygen demand, biochemical oxygen demand, bacteria, dissolved and suspended solids, major ions, nutrients, residual chlorine, total-recoverable metals, oil and grease, phenols, volatile-organic compounds, acid-base/neutral organic compounds, and pesticides.

Computed, hereinafter called observed, values for storm-runoff loads, volume, and event-mean concentrations are compared by using root-mean-square error with the results from regional-regression models developed from data collected for the NURP (Ellis and others, 1984; Driver and Tasker, 1990). Procedures are presented for using local data to adjust estimates from single-storm regional-regression models.

Description of Study Area and Sampling Sites

The study area, the city of Colorado Springs, is located in and along the eastern slope of the southern

Rocky Mountains (fig. 1). The climate of the study area is semiarid. Annual precipitation for 1948–87 at the Colorado Springs airport ranged from 8.6 to 25.4 in. Mean annual precipitation at the airport was 15.2 in. Convective thunderstorms contribute most of the rainfall during May through September. Thunderstorms occur an average of 70 days each year (U.S. Geological Survey, 1970). Mean air temperatures are about 29°F in January and 70°F in July (Hansen and others, 1978).

Soils in the study area tend to be sandy, moderately deep to deep, and well drained to excessively well drained. The study area is underlain mainly by sandstone and shale and by alluvial and windlain deposits. The landform dominating the study area is the Colorado Piedmont. A more detailed description of the study area is discussed in von Guerard (1989). Another important aspect of the study area is the rate of growth associated with Colorado Springs. Population increased from 45,472 in 1950 to 281,140 in 1990. Additionally, total area of Colorado Springs increased from 9.4 mi² in 1950 to 181.4 mi² in 1990 (Christine Lytle, Colorado Springs City Engineering Division, written commun., 1992).

Each of the sampling sites, except for site 4, is located near the outfall of a drainage basin. Drainage-basin areas range from 0.049 to 0.327 mi², and a predominant land use can be attributed to each basin (table 1). However, all sites have a mixture of land

4 Water Quality of Storm Runoff and Comparison of Procedures for Estimating Storm-Runoff Loads, Volume, Event-Mean Concentrations, and the Mean Load for a Storm for Selected Properties and Constituents for Colorado Springs, Southeastern Colorado, 1992

uses, which is typical of Colorado Springs. For the purposes of this report, a drainage basin is defined by the local municipal storm-sewer network and is not necessarily delineated by the topography of the drainage basin. Selected descriptive data for the five drainage basins are listed in table 1. Except for site 4, all sites discharge into another part of the Colorado Springs storm-sewer system. Site 4 discharges directly into Monument Creek.

Site 1 is in southeastern Colorado Springs (fig. 1). The sampling site is in a 60-in. reinforced concrete pipe (RCP) and is accessed by a manhole. The manhole is located about 200 ft upstream from Spring Creek along the southern boundary fence of the Valley-Hi Golf Course and is south of the sixteenth hole. The sampling site is directly upstream from an 18-in. side drain entering the RCP from the south. Site 1 has a drainage-basin area of 0.125 mi², of which 58.1 percent is impervious area (table 1). Predominant land use in the drainage basin is commercial (table 1) and includes retail stores and two automobile dealerships that have repair facilities.

Site 2 is in northwest Colorado Springs (fig. 1). The sampling site is in a 72-in. RCP accessed by a manhole. The manhole is located about 100 ft southeast of the intersection of Garden of the Gods Road and Chestnut Street and is about 200 ft upstream from Douglas Creek. Site 2 has a drainage-basin area of 0.165 mi², of which 37.5 percent is impervious area (table 1). Predominant land use is industrial and includes tool and machine forging, computer software, heating and air-conditioning manufacturing, and metallurgy companies.

Site 3 is located in north-central Colorado Springs (fig. 1). The sampling site is in a 48-in. RCP and is accessed by a manhole. The manhole is located on the northwest corner of Beacon Street at Buchanan Street and is about 400 ft east of Monument Creek. Site 3 has a drainage-basin area of 0.173 mi², of which 55.9 percent is impervious area. Land use predominantly is industrial (table 1) and includes auto repair, machining, manufacturing, food-processing, welding, computer software, metal-fabrication, and paper-distribution companies.

Site 4 is in central Colorado Springs (fig. 1). The sampling site is in a 66-in. RCP accessed by a manhole. The manhole is about 0.7 mi upstream from Shooks Run and 75 ft from the southeast corner of Wahsatch Street and Cross Lane. Site 4 has a drainage-basin area of 0.327 mi², of which 34.2 percent is impervious area. Land use primarily is low-density residential (table 1) but includes some commercial businesses.

Site 5 is in southwest Colorado Springs (fig. 1). The sampling site is in a 42-in. RCP and is accessed by a manhole. The manhole is located about 30 ft upstream from a drainage channel leading to Bear Creek and is about 300 ft east of the southeast corner of a retail store. The sampling site is directly upstream from an 18-in. side drain entering the RCP from the northwest. Site 5 has a drainage-basin area of 0.049 mi² of which 40.1 percent is impervious area (table 1). Land use in the drainage basin predominantly is undeveloped (43 percent); however, commercial land use composes 39.9 percent of the drainage-basin area (table 1). The undeveloped area is in the upper part of the drainage basin and did not contribute runoff during the events sampled; therefore, runoff sampled is considered to be representative of commercial land use. Commercial land use includes two automobile dealerships that have repair facilities, a gas station, and several retail stores.

Acknowledgments

The authors appreciate the support and cooperation provided by Christine Lytle, Leonard Miller, and Bruce Thorson of the city of Colorado Springs. The authors thank Daniel Bunting, El Paso County Regional Floodplain Administrator, for providing precipitation duration data. The authors also thank Seth Bacon, Evan Howe, and Dennis Young, student employees of the U.S. Geological Survey, for their untiring efforts in collecting and processing storm-water samples.

DATA-COLLECTION TECHNIQUES, WATER-QUALITY-SAMPLING PROCEDURES, AND QUALITY-ASSURANCE PROCEDURES

Collection of storm-runoff data in the urban environment and for the purposes of NPDES requires specialized techniques and procedures. The following is a description of the data-collection techniques and sampling procedures used for this study.

Precipitation and Flow Data-Collection Techniques

Two to four precipitation storage gages were installed in each drainage basin. These gages were inspected at least daily from May 25 to August 16, 1992, and periodically, thereafter, until the completion

of the snowmelt sampling. Precipitation for each drainage basin was area weighted using Thiessen polygons (Chow, 1964). In addition to providing total precipitation for each storm sampled, data collected at precipitation storage gages were used to determine precipitation conditions for 6 to 72 hours prior to the collection of a sample. To meet NPDES requirements for sampled storms, precipitation could not exceed 0.10 in. during the 72 hours preceding the collection of samples. For the four additional samples collected for selected properties and constituents, precipitation could not exceed 0.05 in. during the 6 hours preceding sample collection.

The hydraulics of flow in storm-sewer systems is extremely complex, and when coupled with the safety and logistical problems associated with accessing storm sewers, the complex hydraulics make the accurate measurement of storm-water discharge difficult. Palmer-Bowles flumes were installed at each site to measure flow in the storm sewers (Kilpatrick and others, 1985). The Palmer-Bowles flume causes flow in the RCP to be subcritical at the flume approach and forces flow through critical depth in the flume throat. Kilpatrick and others (1985) developed calibration curves for the Palmer-Bowles flumes. These calibration curves are within ± 10 percent of measured flow. Depth of flow in the RCP was measured using a gas-purge conoflow pressure-regulating system and a pressure transducer. Flow depth was recorded using dataloggers.

Water-Quality-Sampling Procedures

Thirty samples were collected during rainfall runoff, and five samples were collected during snowmelt runoff. Prior to sample collection, all sample-collection bottles were washed using a nonphosphate detergent and were rinsed using tap water, 1-percent hydrochloric acid solution, and pesticide-grade methanol. Initially, and after each storm was sampled, pumping-sampler-intake lines were cleaned, using the procedure just described; in addition, the lines were given a final rinse of organic-free water. Glass mason jars used for the collection of samples for bacteria analysis were sterilized using an autoclave.

Storm-runoff samples were obtained by manually collecting grab samples and by using automatic-pumping samplers. Grab samples were collected for pH, bacteria, residual chlorine, total-recoverable cyanide, oil and grease, phenols, and volatile-organic compounds. Water temperature was measured from a grab sample immediately after collection. Grab samples were collected as depth-integrated point sam-

ples and were collected using Teflon USDH-81 samplers (Federal Inter-Agency Sedimentation Project, written commun., 1992) that were equipped with 1-qt glass jars. At all sites and at all flow depths, flow in the RCP was turbulent and well mixed.

Composite samples were collected for chemical oxygen demand, biochemical oxygen demand, specific conductance, alkalinity, dissolved and suspended solids, major ions, nutrients, total-recoverable metals, acid-base/neutral organic compounds, and pesticides. Composite samples were first collected discretely using automatic-pumping samplers equipped with Teflon intake lines and four 1-gal glass bottles. The samplers were activated by the datalogger when a predetermined flow in the RCP was exceeded. After the sampler was activated, samples were collected at intervals of 5 to 30 minutes, depending on the flow in the RCP. Samples were collected until the water level in the RCP dropped below the sampler orifice. After the bottles were filled, they were capped with Teflon-lined lids, put on ice, and transported to a field laboratory for flow-weight compositing. Flow-weighted aliquots were split from the sample into a stainless-steel Teflon-lined churn, using a Teflon cone-splitter that was equipped with Teflon tubing. The aliquot needed from each discrete sample used for flow weighting was determined using an arithmetic weighting formula:

$$S_v = (QSAMP \cdot TVSR) / TQSC \quad (1)$$

where

S_v = the aliquot from a particular discrete sample;

$QSAMP$ = instantaneous flow when the particular discrete sample was collected;

$TVSR$ = the total volume of flow-weighted sample needed for processing; and

$TQSC$ = is the sum of instantaneous flows for all discrete samples from which aliquots will be drawn.

After compositing, samples were shipped to the U.S. Geological Survey National Water-Quality Laboratory in Arvada, Colorado, for analysis.

The concentration of the flow-weighted composite sample is used to represent the storm-runoff event-mean concentration. Storm-runoff load, in pounds, was computed by multiplying the event-mean concentration by the volume of storm runoff for the storm sampled and by a unit conversion constant.

Quality-Assurance Procedures

Trip blanks and field-equipment blanks were collected and analyzed for all properties and constituents to evaluate potential field contamination. Trip blanks are sample bottles filled with water devoid of any organic or inorganic constituents. Field and laboratory spikes were used to evaluate recovery and potential loss of concentration of organic compounds. Field and laboratory spikes are sample water spiked with a constituent of a known concentration.

Trip blanks were collected to evaluate any possible contamination occurring during transport of the sample from the field to the analytical laboratory. Values and concentrations in trip blanks were almost equal to or less than the analytical detection limit for all properties and constituents. Values and concentrations in field-equipment blanks were almost equal to or less than the analytical detection limits for every property and constituent except for chemical oxygen demand and total-organic carbon indicating there was little or no field contamination.

Field and laboratory spikes for organic compounds were done to evaluate potential analytical recoveries and possible degradation of constituents from the time of collection to when samples were analyzed. Average percent recoveries for volatile organic compounds were 43 percent for field spikes and 67 percent for lab spikes. The percent recoveries of less than 100 percent for the spikes indicate some loss of constituent concentration between sample collection and analysis. Average percent recoveries for acid-base/neutral organic compounds were 80 percent for field spikes and 97 percent for laboratory spikes. For pesticide compounds, average percent recoveries were 80 percent for field spikes and 82 percent for laboratory spikes. Generally, recovery of constituents was less than 100 percent, especially for volatile organic compounds. However, percent recovery was greater than 100 percent for certain constituents. This large percent recovery can be accounted for by the possible matrix effects on certain spiked concentrations and by the precision of the analytical technique used in the analysis (Mary Olsen, U.S. Geological Survey, oral commun., 1993).

WATER QUALITY OF STORM RUNOFF

Storm-runoff water-quality data were collected at sites that represent commercial (sites 1 and 5), industrial (sites 2 and 3), and residential (site 4) land uses (table 1). The water-quality properties and constituents collected can be separated into the following major categories:

1. Properties—pH, specific conductance, temperature, chemical oxygen demand, and biochemical oxygen demand,
2. Bacteria—fecal coliform and fecal streptococci,
3. Dissolved and suspended solids and major ions,
4. Nutrients—nitrogen and phosphorus,
5. Total-recoverable metals,
6. Total-organic carbon,
7. Organic compounds—volatile, acid-base/neutral, and pesticides.

Three samples were collected at each site for NPDES permitting purposes. Four additional samples were collected for selected constituents that also were collected for NPDES purposes. The additional four samples were collected to provide a large enough sample size (seven samples per site) to estimate storm-runoff loads and event-mean concentrations for selected constituents using linear-regression procedures developed using data from the NURP. These data are summarized in table 2.

Median concentrations of chemical and 5-day biochemical oxygen demand were highest for site 4 (table 2); the median concentration for chemical oxygen demand for site 4 was 330 mg/L and for 5-day biochemical oxygen demand was 86 mg/L. Storm runoff generally had a neutral pH. The median value of pH for commercial sites was 7.5, for industrial sites was 7.3, and for the residential site was 7.4. Specific conductance of storm runoff was largest for samples collected during snowmelt runoff. Median specific conductance for all samples collected during snowmelt runoff was 385 $\mu\text{S}/\text{cm}$; during rainfall runoff, the median specific conductance was 104 $\mu\text{S}/\text{cm}$. Water temperatures ranged from 0.0 (during snowmelt runoff) to 24.5°C (table 17).

The maximum counts of fecal coliform and fecal streptococci were measured at sites 1 and 4 (table 17). Median counts for all samples at sites 1 and 4 were 4,900 and 17,000 col/100 mL. However, largest median counts for bacteria were in samples from site 4 at 4,900 col/100 mL for fecal coliform and 20,500 col/100 mL for fecal streptococci.

Dissolved-solids concentrations ranged from 34 to 4,240 mg/L (table 19) for all land uses, and the range of concentrations was similar for all sites. The largest concentrations of dissolved solids were measured from snowmelt samples. The largest concentration of suspended solids, 1,400 mg/L, was measured for site 5 (table 19). The largest median concentration of suspended solids was 826 mg/L at site 5.

Table 2. Summary statistics for selected constituents in storm runoff for storm-runoff-sampling sites in Colorado Springs

[mg/L, milligrams per liter; µg/L, micrograms per liter; -, not available due to censored data]

Summary statistic	Constituent													
	Oxygen demand, chemical (high level) (mg/L)	Oxygen demand, biochemical, 5-day (mg/L)	Solids, dissolved (mg/L)	Solids, suspended (mg/L)	Nitrogen, ammonia, total as nitrogen (mg/L)	Nitrogen, ammonia plus organic, total, as nitrogen (mg/L)	Nitrogen, nitrite, total, as nitrogen (mg/L)	Nitrogen, nitrate plus nitrite, total, as nitrogen (mg/L)	Phosphorus, total, as phosphorus (mg/L)	Phosphorus, dissolved, as phosphorus (mg/L)	Cadmium, total recoverable (µg/L)	Copper, total recoverable (µg/L)	Lead, total recoverable (µg/L)	Zinc, total recoverable (µg/L)
<u>Sixteenth Hole, Valley-HI Golf Course (site 1)</u>														
Number of samples	7	5	7	7	7	7	7	7	7	7	7	7	7	7
Minimum	100	19	63	121	0.28	0.9	0.02	0.40	0.12	0.08	<1	9	23	140
Maximum	310	53	202	524	0.98	2.4	.11	1.00	.45	.28	1	17	170	300
Mean	200	34	108	284	0.60	1.8	.05	.73	.28	.13	-	13	81	204
Median	180	29	93	274	0.67	1.7	.04	.80	.29	.12	<1	12	64	190
<u>Chestnut Street at Douglas Creek (site 2)</u>														
Number of samples	7	6	7	7	7	7	7	7	7	6	7	7	7	7
Minimum	100	14	34	198	0.23	.90	.03	.45	.09	.06	<1	12	47	230
Maximum	420	80	256	1280	1.10	2.4	.08	1.10	.72	.21	2	99	350	1400
Mean	240	36	94	595	0.49	1.4	.04	.80	.24	.11	-	39	189	730
Median	230	31	63	464	0.43	1.1	.04	.87	.17	.09	1	19	150	570
<u>Beacon Street at Buchanan Street (site 3)</u>														
Number of samples	7	7	7	7	7	7	7	7	7	7	7	7	7	7
Minimum	120	26	54	101	0.48	1.6	.03	.66	.16	.10	<1	12	23	150
Maximum	300	74	168	340	1.50	4.0	.07	1.40	.58	.51	2	24	97	400
Mean	210	51	100	220	1.01	2.9	.05	.95	.36	.23	-	16	63	270
Median	200	51	87	220	.99	2.9	.04	1.00	.33	.18	2	14	55	280

Table 2. Summary statistics for selected constituents in storm runoff for storm-runoff-sampling sites in Colorado Springs--Continued

Summary statistic	Constituent													
	Oxygen demand, chemical (high level) (mg/L)	Oxygen demand, biochemical, 5-day (mg/L)	Solids, dissolved (mg/L)	Solids, suspended (mg/L)	Nitrogen, ammonia, total as nitrogen (mg/L)	Nitrogen, ammonia plus organic, total, as nitrogen (mg/L)	Nitrogen, nitrite, total, as nitrogen (mg/L)	Nitrogen, nitrate plus nitrite, total, as nitrogen (mg/L)	Phosphorus, total, as phosphorus (mg/L)	Phosphorus, dissolved, as phosphorus (mg/L)	Cadmium, total recoverable (µg/L)	Copper, total recoverable (µg/L)	Lead, total recoverable (µg/L)	Zinc, total recoverable (µg/L)
<u>Wahsatch Street at Cross Lane (site 4)</u>														
Number of samples	7	5	6	7	7	7	7	7	7	5	7	7	7	7
Minimum	190	29	74	116	0.16	1.6	.03	.36	.22	.11	<1	8	32	110
Maximum	500	140	908	848	1.00	5.3	.08	.88	1.20	.38	2	22	130	310
Mean	310	84	229	472	0.49	3.8	.05	.59	.75	.26	-	13	89	200
Median	330	86	100	512	0.39	3.8	.05	.47	.72	.27	1	12	110	220
<u>Walmart at Eighth Street (site 5)</u>														
Number of samples	7	7	6	7	7	7	7	7	7	6	7	7	7	7
Minimum	160	29	67	388	.41	1.2	.03	.69	.21	.09	1	15	85	210
Maximum	830	260	4240	1400	3.90	7.4	.31	1.80	1.00	.34	21	70	350	730
Mean	340	94	790	846	1.30	3.3	.09	1.20	.60	.19	5	28	220	440
Median	260	67	110	826	1.00	2.3	.06	1.20	.59	.17	2	18	200	340

Samples were collected to characterize concentrations of nutrients (nitrogen and phosphorous) in storm runoff. Sites 3 and 5 had the largest median concentrations for ammonia as nitrogen and nitrate plus nitrite as nitrogen, 1.0 and 1.2 mg/L. Site 4 had the largest median concentrations for total ammonia plus organic nitrogen as nitrogen, and total and dissolved phosphorus at 3.8, 0.72, and 0.27 mg/L.

Generally, concentrations of total-recoverable metals were similar for all sites (table 19). Concentrations of total-recoverable lead were largest for sites 2 and 5 and had a median concentration of 180 µg/L. Concentrations of total-recoverable copper, nickel, and zinc were largest for sites 2 and 5 and had median concentrations of 18.0, 16.0, and 500 µg/L.

The largest median concentration of total-organic carbon was 100 mg/L at site 4. The median concentration of total-organic carbon for sites 1, 2, 3, and 5 was 52 mg/L.

Each sample collected for volatile-organic compounds (VOC) was analyzed for 61 constituents. The largest number of VOC's detected were 21 at site 4 and 31 at site 5. The number of VOC's detected, the number of samples collected, and the number of times each VOC was detected are listed for each site in table 3. Generally, volatile organic compounds were detected more often in samples collected during snowmelt runoff than in samples collected during rainfall runoff. A possible explanation for the higher number of detections of volatile organic compounds in snowmelt-runoff samples is that these samples were collected at a lower temperature than were the samples collected during rainfall runoff. Volatile organic compounds volatilize at a slower rate at the lower temperatures during snowmelt runoff. The volatile organic compounds detected generally were associated with gasoline and other petroleum products.

Each sample collected for acid-base/neutral organic compounds was analyzed for 57 constituents. The largest number of acid-base/neutral organic compounds detected was 21 at sites 1 and 3 and 26 at site 2. The number of acid-base/neutral compounds detected, the number of samples collected, and the number of times each compound was detected is listed in table 3. Except for chlordane in the sample collected at site 4 on June 12, concentrations of pesticides were less than the analytical detection limits for all samples.

PROCEDURES FOR ESTIMATING SINGLE-STORM-RUNOFF LOADS, VOLUME, AND EVENT-MEAN CONCENTRATIONS

The NPDES permitting process requires the estimation of total annual pollutant loads and event-mean concentrations for 12 properties and constituents. The

12 properties and constituents are chemical oxygen demand (COD), 5-day biochemical oxygen demand (BOD), dissolved solids (DS), suspended solids (SS), total nitrogen (TN), total ammonia plus organic nitrogen as nitrogen (TKN), total phosphorus (TP), dissolved phosphorus (DP), total-recoverable cadmium (CD), total-recoverable copper (CU), total-recoverable lead (PB), and total-recoverable zinc (ZN). In a simplistic assessment, seasonal or annual storm-runoff loads, volume, and event-mean concentrations could be estimated using mean concentrations of properties and constituents from the set of sampled storms. However, better estimates of single-storm-runoff loads and event-mean concentrations can result by using multiple-regression analysis to relate these response variables to climatic, physical, and land-use characteristics (Driver and Tasker, 1990). These regression models could be used with precipitation data and physical and land-use information to estimate single-storm-runoff loads, volume, and event-mean concentrations for individual storms at ungaged storm-runoff sites.

The form of the regression equation used for estimating single-storm-runoff loads, volume, and event-mean concentrations is a linear function of the logarithmic-transformed variables:

$$\log Y = \log B_0 + B_1 \log X_1 + B_2 \log X_2 + \dots + B_N \log X_N \quad (2)$$

Taking the antilogs, the equation becomes:

$$Y = B_0 X_1^{B_1} X_2^{B_2} \dots X_N^{B_N}, \quad (3)$$

where

- Y = estimated storm-runoff load, volume, or event-mean concentration (response variable);
- B₀, B₁, B₂, B_N = regression coefficients;
- X₁, X₂, ..., X_N = climatic, physical, or land-use variables (explanatory variables); and
- N = number of climatic, physical, and land-use variables in the regression model.

A transformation bias is produced when logarithms of the estimated mean response (log of the response variable) is retransformed (equation 3). This transformation bias usually results in the underestimation of the estimated mean response. However, the major part of this transformation bias may be eliminated by multiplying the estimated mean response by a correction factor (Duan, 1983):

$$BCF = \frac{1}{n} \sum_{i=1}^n 10^{e_i} \quad (4)$$

10 Water Quality of Storm Runoff and Comparison of Procedures for Estimating Storm-Runoff Loads, Volume, Event-Mean Concentrations, and the Mean Load for a Storm for Selected Properties and Constituents for Colorado Springs, Southeastern Colorado, 1992

Table 3. Summary of detections of organic compounds in storm runoff for storm-runoff-sampling sites in Colorado Springs

[All compounds in micrograms per liter]

Site	Constituent	Number of samples	Number of detections
VOLATILE-ORGANIC COMPOUNDS			
Sixteenth Hole, Valley-Hi Golf Course (site 1)	Benzene, total	10	1
	Ethyl-benzene, total	10	1
	Naphthalene, total	10	3
	Toluene, total	10	2
	Xylene, water, whole, total recoverable	10	3
	1,2,4-trimethyl benzene, water, whole, recoverable	10	2
	1,3,5-trimethyl benzene, water, whole, recoverable	10	1
	ACID-BASE/NEUTRAL ORGANIC COMPOUNDS		
Sixteenth Hole, Valley-Hi Golf Course (site 1)	Benzogh-i-perylenel, 12-benzo perylene, total	3	1
	Benzo-b-flouranthene, total	3	2
	Bis (2-ethylhexyl) phthalate, total	3	3
	Chrysene, total	3	2
	Di-n-butyl phthalat, total	3	3
	Fluoranthene, total	3	3
	Indeno (1,2,3-cd) pyrene, total	3	1
	Phenanthrene, total	3	3
	Pyrene, total	3	3
	VOLATILE-ORGANIC COMPOUNDS		
Chestnut Street at Douglas Creek (site 2)	Naphthalene, total	7	3
	Toluene, total	7	3
	Xylene, water, whole, total recoverable	7	4
	1,2,4-trimethyl benzene, water, whole, recoverable	7	2
ACID-BASE/NEUTRAL ORGANIC COMPOUNDS			
Chestnut Street at Douglas Creek (site 2)	Anthracene, total	3	2
	Benzo-a-anthracenel, 2-benzanthracene, total	3	2
	Benzo-a-pyrene, total	3	2
	Benzogh-i-perylenel, 12-benzo perylene, total	3	1
	Benzo-b-fluoranthene, total	3	2
	Benzo-k-fluoranthene, total	3	2
	Bis (2-ethylhexyl) phthalate, total	3	3
	Chrysene, total	3	2
	Fluoranthene, total	3	3
	Indeno (1,2,3-cd) pyrene, total	3	2
	Phenanthrene, total	3	2
	Pyrene, total	3	3

Table 3. Summary of detections of organic compounds in storm runoff for storm-runoff-sampling sites in Colorado Springs
 --Continued

Site	Constituent	Number of samples	Number of detections	
VOLATILE-ORGANIC COMPOUNDS				
Beacon Street at Buchanan Street (site 3)	Dichlorobromomethane, total	8	1	
	Naphthalene, total	8	6	
	Toluene, total	8	1	
	Xylene, water, whole, total recoverable	8	2	
	1,2,4-trimethyl benzene, water, whole, recoverable	8	1	
	1,3,5-trimethyl benzene, water, whole, recoverable	8	1	
	ACID-BASE/NEUTRAL COMPOUNDS			
Beacon Street at Buchanan Street (site 3)	Anthracene, total	3	1	
	Benzo-a-anthracene, 2-benzanthracene, total	3	1	
	Benzo-a-pyrene, total	3	2	
	Benzogh-i-perylene, 12-benzo perylene, total	3	1	
	Benzo-b-fluoranthene, total	3	2	
	Benzo-k-fluoranthene, total	3	2	
	Bis (2-ethylhexyl) phthalate, total	3	3	
	Chrysene, total	3	2	
	Fluoranthene, total	3	2	
	Indeno (1,2,3-cd) pyrene, total	3	1	
	Phenanthrene, total	3	2	
	Pyrene, total	3	2	
	VOLATILE-ORGANIC COMPOUNDS			
	Wahsatch Street at Cross Lane (site 4)	Benzene, total	8	1
Ethyl-benzene, total		8	1	
Naphthalene, total		8	4	
P-isopropyl toluene, water, whole, recoverable		8	1	
Toluene, total		8	6	
Xylene, water, whole, total recoverable		8	4	
1,2,4-trimethyl benzene, water, whole, recoverable		8	3	
1,3,5-trimethyl benzene, water, whole, recoverable		8	1	
ACID-BASE/NEUTRAL ORGANIC COMPOUNDS				
Wahsatch Street at Cross Lane (site 4)	Benzo-a-pyrene, total	3	1	
	Benzo-b-fluoranthene, total	3	1	
	Benzo-k-fluoranthene, total	3	1	
	Bis (2-ethylhexyl) phthalate, total	3	3	
	Chrysene, total	3	1	
	Fluoranthene, total	3	2	
	Phenanthrene, total	3	2	
	Pyrene, total	3	2	
R0024095				

Table 3. Summary of detections of organic compounds in storm runoff for storm-runoff-sampling sites in Colorado Springs

Site	Constituent	Number of samples	Number of detections
PESTICIDE COMPOUNDS			
Wahsatch Street at Cross Lane (site 4) --Continued	Chlordane, total	3	1
VOLATILE-ORGANIC COMPOUNDS			
Walmart at Eighth Street (site 5)	Benzene, total	9	1
	Chloroform, total	9	5
	Cis-1,2-dichloroethene, water, total	9	1
	Ethyl benzene, total	9	1
	N-butyl benzene, water, whole, recoverable	9	1
	N-propyl benzene, water, whole, recoverable	9	1
	Naphthalene, total	9	6
	Toluene, total	9	5
	Xylene, water, whole, total recoverable	9	4
	1,1,1-trichloroethane, total	9	2
	1,2,4-trimethyl benzene, water, whole, recoverable	9	3
	1,3,5-trimethyl benzene, water, whole, recoverable	9	1
ACID-BASE/NEUTRAL ORGANIC COMPOUNDS			
	Anthracene, total	3	1
	Bis (2-ethylhexyl) phthalate, total	3	3
	Di-n-octyl phthalate, total	3	1
	Di-n-butyl phthalate, total	3	2
	Fluoranthene, total	3	3
	N-butyl benzyl phthalate, total	3	2
	Phenanthrene, total	3	2
	Pyrene, total	3	3

R0024096

where

BCF = the bias correction factor,

n = the number of observations in the data set,
and

e_i = least-squares residual for observation i from
the calibration data set, in log units.

Single-Storm Local-Regression Models

Using data collected for 30 rainfall-runoff storms, single-storm local-regression models were developed for the Colorado Springs area for estimating storm-runoff loads for the 12 NPDES properties and constituents and for estimating storm-runoff volume and event-mean concentrations for the 12 NPDES properties and constituents. The data for 30 storms from 5 drainage basins (6 storms in each drainage basin) are listed in tables 16 and 19. Data collected for snowmelt samples (the November and December samples) were not available at the time of this analysis and, thus, were not included in the development of the local-regression models. Also, techniques for estimating storm-runoff loads, volumes, and event-mean concentrations were developed using data for rainfall-runoff conditions (Ellis and others, 1984; Driver and Tasker, 1990), and the snowmelt samples represent different hydrologic processes and need to be considered separately. The models were developed using ordinary least-squares regression, except for total-recoverable cadmium (CD). The CD data set had 11 of the 30 analyses reported as less than (censored) values. Ordinary least-squares regression should not be used with censored data. However, censored data can be included with uncensored data using tobit regression, which is similar to least-squares regression, in which the parameter estimates are fit using maximum-likelihood estimation (Helsel and Hirsch, 1992). Except for two, all CD concentrations were 3 $\mu\text{g/L}$ or less (table 19); however, the CD concentration of 21 $\mu\text{g/L}$ was not included in this analysis because it is not considered representative of storm runoff in the study area. Storm-runoff loads, volume, and event-mean concentrations (response variables) were modeled using the following climatic, physical, and land-use characteristics (explanatory variables):

1. Total rainfall (TRN), in inches,
2. Total contributing drainage-basin area (DA), in
3. Impervious area (IA), as a percent of total contributing drainage-basin area,

4. Industrial land use (LUI), as a percent of total contributing drainage-basin area,
5. Commercial land use (LUC), as a percent of total contributing drainage-basin area,
6. Residential land use (LUR), as a percent of total contributing drainage-basin area,
7. Nonurban land use (LUN), as a percent of total contributing drainage-basin area,
8. Period (in days) preceding collection of a sample having less than 0.10 in. of precipitation (DD).

The RSQUARE procedure (Statistical Analysis System Institute, Inc., 1990) was used to determine which combination of explanatory variables composed the most suitable regression model. The RSQUARE procedure performs all possible linear regressions for all possible combinations of explanatory variables and determines the subsets of explanatory variables that have the largest r^2 value (Statistical Analysis System Institute, Inc., 1990). For the models to have a hydrologic and physiographic basis, only subset regression models including the explanatory variables TRN and DA were evaluated. The most suitable regression model was selected on the basis of the statistical significance of explanatory variables in the regression, the values of r^2 , and checked using other model selection criteria (Statistical Analysis System Institute, Inc., 1990). An r^2 value is the proportion of the total variation of the response variable that is explained by the explanatory variables. For certain properties and constituents, all of the possible regressions included explanatory variables that were not significant at the 5-percent confidence level. Therefore, some regression models that were selected as the most suitable included explanatory variables that were not significant at the 5-percent confidence level (tables 4 and 5). However, inclusion of these variables in the models improved the computed r^2 and were considered useful predictors of the dependent variables. For event-mean concentrations for BOD, the local-regression model was not significant at the 5-percent confidence level (table 5). For evaluating all possible regression models (RSQUARE procedure) for storm-runoff loads and for event-mean concentrations for CD, only uncensored data were used (19 of the 30 values for CD were uncensored).

Plots of residual (observed values minus estimated values) compared to estimated values were analyzed to evaluate the constant variance (homoscedasticity) of the residuals. Residual plots for all of the most suitable models indicate that the variance of the residuals generally is constant throughout the entire

14 Water Quality of Storm Runoff and Comparison of Procedures for Estimating Storm-Runoff Loads, Volume, Event-Mean Concentrations, and the Mean Load for a Storm for Selected Properties and Constituents for Colorado Springs, Southeastern Colorado, 1992

Table 4. Summary of single-storm local-regression models for storm-runoff loads and volume for storm-runoff-sampling sites in Colorado Springs

[Bo, the regression coefficient that is the intercept in the regression model; B₁ and B_n, regression coefficients; TRN, total rainfall; DA, total contributing drainage-basin area; IA, impervious area; LUI, industrial land use; LUC, commercial land use; LUR, residential land use; LUN, nonurban land use; DD, period preceding collection of a sample having less than 0.10 inch of precipitation; BCF, bias correction factor; r², the coefficient of determination; COD, chemical oxygen demand, in pounds; BOD, 5-day biochemical oxygen demand, in pounds; DS, dissolved solids, in pounds; SS, suspended solids, in pounds; TN, total nitrogen, in pounds; TKN, total ammonia plus organic nitrogen as nitrogen, in pounds; TP, total phosphorus, in pounds; DP, dissolved phosphorus, in pounds; CD, total-recoverable cadmium, in pounds; CU, total-recoverable copper, in pounds; PB, total-recoverable lead, in pounds; ZN, total-recoverable zinc, in pounds; RUN, storm-runoff volume, in cubic feet; asterisk (*) indicates the explanatory variable is not significant at the 5 percent level; dashes indicate that the variable is not included in the model; ln, natural log; NA, not applicable; the form of regression equation is:

$$Y = B_0 X_1^{B_1} \dots X_n^{B_n}(\text{BCF})$$

Response variable	Bo	Explanatory variables								BCF	r ²	Standard error of estimate	
		TRN (inches)	DA (square miles)	IA + 1 (percent)	LUI + 1 (percent)	LUC + 1 (percent)	LUR + 1 (percent)	LUN + 1 (percent)	DD (days)			Percent	Log
BOD	1,000	.64	0.66	--	-0.30	--	--	--	--	1.14	.78	63	.259
DS	188,799	.70	1.51	--	-0.66	--	-1.30	.38	--	1.10	.78	54	.218
SS	1,812	1.36	.17*	--	--	--	--	.38	--	1.23	.65	88	.330
TN	0.48	.89	.05*	2.15	--	--	.55	--	--	1.07	.81	59	.191
TKN	.0001	.69	9.34	--	2.77	4.64	--	3.51	--	1.10	.78	52	.214
TP	5.98	1.00	.24*	--	-0.33	--	--	--	--	1.14	.80	58	.235
DP	1.89*	.86	.39*	--	-0.19	--	--	--	--	1.22	.60	79	.303
CD ¹	-5.09	1.46	-0.87	.10	--	--	--	--	--	1.14	NA	247	.608
CU	.00003	1.06	7.56	--	2.32	3.59	--	2.99	--	1.13	.76	61	.244
PB	750	1.36	-0.78	--	-0.95	-1.15	-1.16*	--	--	1.29	.66	101	.365
ZN	359	1.01	3.20	--	--	.58	-1.70	1.30	--	1.27	.57	115	.398
RUN	32.5	.98	7.86	--	2.24	3.80	--	2.92	--	1.07	.86	43	.178

¹equation form is $y = e^{B_0} + e^{\ln \text{TRN}^{B_1}} + e^{\ln \text{DA}^{B_2}} + e^{\ln(\text{IA}+1)^{B_3}} \text{BCF}$.

Table 5. Summary of single-storm local-regression models for storm-runoff event-mean concentrations for storm-runoff-sampling sites in Colorado Springs

[Bo, the regression coefficient that is the intercept in the regression model; B₁ and B_n, regression coefficients; TRN, total rainfall; DA, total contributing drainage-basin area; IA, impervious area; LUI, industrial land use; LUC, commercial land use; LUR, residential land use; LUN, nonurban land use; DD, period preceding collection of a sample having less than 0.10 inch of precipitation; BCF, bias correction factor; r², the coefficient of determination; COD, chemical oxygen demand, in milligrams per liter (mg/L); BOD, 5-day biochemical oxygen demand, in mg/L; DS, dissolved solids, in mg/L; SS, suspended solids, in mg/L; TN, total nitrogen, in mg/L; TKN, total ammonia plus organic nitrogen as nitrogen, in mg/L; TP, total phosphorus, in mg/L; DP, dissolved phosphorus, in mg/L; CD, total-recoverable cadmium, in micrograms per liter (µg/L); CU, total-recoverable copper, in µg/L; PB, total-recoverable lead, in µg/L; ZN, total-recoverable zinc, in µg/L; asterisk (*) indicates the explanatory variable is not significant at the 5 percent level; dashes indicate that the variable is not included in the model; ln, natural log; NA, not applicable; the form of regression equation is:

$$Y = B_0 X_1^{B_1} \dots X_n^{B_n} (\text{BCF})$$

Response variable	Bo	Explanatory variables								BCF	r ²	Standard error of estimate	
		TRN (Inches)	DA (square miles)	IA + 1 (percent)	LUI + 1 (percent)	LUC + 1 (percent)	LUR + 1 (percent)	LUN + 1 (percent)	DD (days)			Percent	Log
BOD ¹	115,432	-.35*	-6.64	--	-2.24	-3.52	--	-2.59	--	1.14	.43	65	.258
DS	35.5	-.37	.04*	--	--	--	--	0.19	--	1.05	.31	35	.148
SS	242,384	.36*	-0.57	-1.80	--	--	--	--	--	1.14	.44	62	.248
TN	.88*	-.16*	-.36	--	--	--	0.35	-.24	--	1.04	.42	32	.136
TKN	.06	-.28	-.12*	.49*	--	--	--	--	--	1.05	.38	35	.149
TP	.0055	.02*	-1.98	--	--	-.34	-1.27	-.72	--	1.06	.56	55	.224
DP	.0046	-.22*	-1.41	--	--	-.25	.93	-.60	.28	1.06	.63	38	.161
CD ²	-.097*	.26*	-.22	--	--	-.148	--	--	--	1.05	NA	171	.508
CU	15	.02	-.43	--	--	-.30	--	--	.17	--	.50	44	.181
PB	39,346	.57	-.73	-1.65	--	--	--	--	--	1.14	.49	66	.250
ZN	347	.12*	-.65	--	--	-.43	--	--	--	1.08	.60	44	.183

¹regression model significant at the 20 percent level.

²equation form is $y = e^{B_0} + e^{\ln \text{TRN}^{B_1}} + e^{\ln \text{DA}^{B_2}} + e^{\ln \text{LUC}+1^{B_3}} \text{BCF}$.

range of prediction. Because hydrologic data usually are skewed when using parametric statistical techniques, such as RSQUARE, data need to be transformed to minimize the heteroscedasticity of residuals and to linearize the x,y relation. Response and explanatory variables used in the RSQUARE procedure were log transformed (base 10).

The local-regression models for storm-runoff loads, volume, and event-mean concentrations and the corresponding BCF, r^2 values, and the standard error of estimate are listed in tables 4 and 5. The standard error of estimate is an estimate of the standard deviation about the regression. The smaller the standard error of estimate, the more precise will be the predictions (Driver and Tasker, 1990). The standard error of estimate, in percent, was calculated for all the local-regression models using the following equation (Driver and Tasker, 1990):

$$SE = 100 \{ [e^{(mse \times 5.302)} - 1] \}^{0.5} \quad (5)$$

where

SE = the standard error of estimate, in percent;

mse = the mean square error, in log (base 10) units; and

5.302 = the square of the conversion of log base-10 values to natural logs.

The values of r^2 that use ordinary least-squares regression ranged from 0.57 to 0.86 for storm-runoff loads and volume (table 4) and from 0.25 to 0.63 for storm-runoff event-mean concentrations (table 5). Except for CD, standard errors of estimate range from 43 to 115 percent for storm-runoff loads and volume and 32 to 66 percent for storm-runoff event-mean concentrations (tables 4 and 5). Standard errors of estimate for storm-runoff load and event-mean concentration for CD were 247 (table 4) and 171 percent (table 5). The accuracy of the load, volume, and concentration models cannot be compared on the basis of standard error of estimate because the units of the response variable for each model are different (Hoos and Sisolak, 1993).

The explanatory variables generally had signs (positive or negative) that were hydrologically logical. However, occasionally, the signs on individual explanatory variables seem to be counter intuitive. Driver and Tasker (1990) list the following explanations for why the signs of some regression coefficients (explanatory variables) may be counter intuitive:

1. Significant cross-correlation between explanatory variables causes multicollinearity problems in the local-regression models, however, this is accounted for in the RSQUARE procedure.

2. The process involving the effect of the explanatory variables on the water-quality constituent is not well understood.
3. The explanatory variable is a surrogate for another variable.
4. The apparent significance of an explanatory variable may be due to chance and, therefore, the relation may be spurious.

Use of the local-regression models listed in tables 4 and 5 need to be limited to the ranges of climatic, physical, and land-use (explanatory) variables listed in table 6. If values outside these ranges are used in the local-regression models, the standard errors may be considerably larger than the values reported in tables 4 and 5. As the local-regression models are applied to drainage-basin areas and to storms larger than the average drainage-basin area or storm volume of the observation sites, the accuracy of estimates of storm-runoff loads, volume, and event-mean concentrations decreases.

Single-Storm Regional-Regression Models

Procedures for estimating single-storm runoff loads, volume, and event-mean concentrations were developed by Ellis and others (1984) and Driver and Tasker (1990) for 11 of the 12 properties and constituents required for the permitting process. Regional-regression equations for BOD were not developed. Linear-regression equations were developed from data collected by the NURP. Equations developed by Ellis and others (1984) were developed using NURP data collected in the Denver metropolitan area. The Driver and Tasker (1990) equations were developed from the NURP data base and include sets of equations for three geographically distinct regions delineated by mean annual rainfall. The Colorado Springs area is included in Region 1. Comparison of estimates from these regional-regression models with observed storm-runoff loads, volume, and event-mean concentrations for samples collected in the study area will be useful in selecting the most appropriate method for estimating single-storm runoff loads, volume, and event-mean concentrations.

Comparison of Observed and Estimated Single-Storm-Runoff Loads, Volume, and Event-Mean Concentrations

Storm-runoff loads, volume, and event-mean concentrations estimated from single-storm regional-

Table 6. Ranges of values of each explanatory values of each explanatory variable used in single-storm local- and regional-regression models

[TRN, total rainfall, in inches; DA, total contributing drainage area, in square miles; IA, impervious area, in percent; DRN, duration of rainfall, in minutes; COD, chemical oxygen demand; DS, dissolved solids; SS, suspended solids; TN, total nitrogen; TKN, total ammonia plus organic nitrogen as nitrogen; TP, total phosphorus; DP, dissolved phosphorus; CD, total-recoverable cadmium; CU, total-recoverable copper; PB, total-recoverable lead; ZN, total-recoverable zinc; RUN, volume of runoff in cubic feet; dashes, no data available]

Response variable	Explanatory variable	Minimum	Maximum	Mean	Median
Local-regression model for storm-runoff-sampling sites in Colorado Springs					
(¹)	TRN	0.05	0.41	0.17	0.14
(¹)	DA	.049	.327	.17	.16
(¹)	IA	34.2	58.1	45.2	40.1
SS	DRN	7.00	39.3	62.1	35.0
Driver and Tasker (1990) regression model					
COD	TRN	.02	1.99	.36	.26
COD	DA	.05	17.50	1.18	.12
COD	IA	--	--	--	--
DS	TRN	.02	1.23	.36	.28
DS	DA	.01	80.5	4.92	.12
DS	IA	11	98.9	60.6	57
SS	TRN	.03	1.99	.39	.29
SS	DA	.05	17.50	1.45	.12
SS	IA	--	--	--	--
SS	DRN	10	2,220	358	231
TN	TRN	.03	1.99	.41	.29
TN	DA	.01	80.5	6.37	.11
TN	IA	--	--	--	--
TKN	TRN	.03	1.99	.37	.28
TKN	DA	.05	80.5	4.79	.12
TKN	IA	--	--	--	--
TP	TRN	0.03	1.99	0.38	0.28
TP	IA	--	--	--	--
DP	TRN	.03	1.99	.39	.28
DP	DA	.01	4.00	.50	.11
DP	IA	--	--	--	--
CD	TRN	.03	.93	.26	.22
CD	DA	0.01	3.03	0.36	0.12
CD	IA	--	--	--	--
CU	TRN	.02	1.99	.37	.27
CU	DA	.01	4.00	.55	.12
CU	IA	--	--	--	--
PB	TRN	.02	1.99	.39	.28
PB	DA	.004	4.00	.47	.11
PB	IA	--	--	--	--
ZN	TRN	.02	1.99	.39	.28
ZN	DA	.01	4.00	.53	.12
ZN	IA	--	--	--	--

R0024101

Table 6. Ranges of values of each explanatory values of each explanatory variable used in single-storm local- and regional-regression models—Continued

Response variable	Explanatory variable	Minimum	Maximum	Mean	Median
RUN	TRN	.02	1.99	.36	.26
RUN	DA	.004	80.5	2.93	.11
RUN	IA	0	98.9	56.7	57
Small basins [Ellis and others (1984)] regression model					
(²)	TRN	.03	1.99	.35	—
(²)	DA	.09	.63	.20	.12
(²)	IA	.60	91	36.7	38
Small and large basins [Ellis and others (1984)] regression model					
(²)	TRN	—	—	—	—
(²)	DA	0.09	24.7	6.1	0.20
(²)	IA	.6	91	31.8	24

¹Includes storm-runoff load and event-mean concentration for chemical oxygen demand, dissolved solids, suspended solids, total nitrogen, total ammonia plus organic nitrogen as nitrogen, total phosphorus, dissolved phosphorus, total-recoverable cadmium, total-recoverable copper, total-recoverable lead, total-recoverable zinc, and volume of runoff for RUN.

²Includes storm-runoff load for chemical oxygen demand, suspended solids, total nitrogen, total phosphorus, total-recoverable lead, total-recoverable zinc, and volume of runoff for RUN.

regression models were compared to observed values from data collected at five sites in the study area in 1992 (fig. 1, table 1). The response variables estimated by Driver and Tasker (1990) are storm-runoff loads for COD, SS, DS, TN, TKN, TP, DP, CD, CU, PB, and ZN; storm-runoff volume, and event-mean concentrations for COD, SS, DS, TN, TKN, TP, DP, CD, CU, PB, and ZN. The response variables estimated by Ellis and others (1984) are storm-runoff loads for COD, SS, TN, TP, PB, and ZN and storm-runoff volume. Driver and Tasker (1990) developed two sets of single-storm regional-regression models for storm-runoff loads and volume. The first set of models was based on a stepwise regression analysis of 13 explanatory variables including TRN, DA, IA, land-use, and regional climatic variables. The second set of models was based on three explanatory variables—TRN, DA, and IA. The single-storm regional-regression models for storm-runoff event-mean concentrations are based on stepwise regression analysis of the same 13 explanatory variables used to develop models for storm-runoff load and volume. Single-storm regional-regression models developed by Ellis and others (1984) for storm-runoff loads and volume were based on three explanatory variables—TRN, DA, and IA.

Comparisons of observed and estimated storm-runoff loads, volume, and event-mean concentrations were made using the root-mean-square error (RMSE) of estimate from the equation:

$$RMSE = \left[\frac{1}{n} \sum_{i=1}^n (\log O_i - \log E_i)^2 \right]^{\frac{1}{2}} \quad (6)$$

where

RMSE = the root-mean-square error in log units (base 10);

O_i = *i*th observed value;

E_i = *i*th estimated value from the regional-regression model; and

n = the number of observations in the data set.

Generally, when compared to observed values, the 3-variable models that were developed by Driver and Tasker (1990) for estimating storm-runoff loads had the smallest RMSE (table 7). For estimates of storm-runoff volume, when compared to observed values, the multivariate model developed by Driver and Tasker (1990) had the smaller RMSE (table 7).

An evaluation of residuals from a comparison of observed and estimated storm-runoff loads, volume, and event-mean concentrations can be used to determine the direction of bias of estimated values compared to observed values. Compared to observed values, most regional-regression models tended to overestimate (negative sign in table 8) storm-runoff loads, volume, and event-mean concentrations. However, the direction of bias was not consistent for all properties and constituents and runoff volume (table 8).

Table 7. Root-mean-square error from comparison of observed storm-runoff loads and volumes and estimated storm-runoff loads and volumes from single-storm regional-regression models

[COD, chemical oxygen demand, in pounds; DS, dissolved solids, in pounds; SS, suspended solids, in pounds; TN, total nitrogen, in pounds; TKN, total ammonia plus organic nitrogen as nitrogen, in pounds; TP, total phosphorus, in pounds; DP, dissolved phosphorus, in pounds; CD, total-recoverable cadmium, in pounds; CU, total-recoverable copper, in pounds; PB, total-recoverable lead, in pounds; ZN, total-recoverable zinc, in pounds; RUN, volume, in cubic feet; dashes indicate not available]

Response variable	Root-mean-square error in log units			
	Regional-regression models from Driver-Tasker (1990)		Regional-regression models from Ellis and others (1984)	
	3-variable model ¹	Multivariate model ²	Small drainage basins ³	Small and large drainage basins ⁴
COD	0.353	0.738	0.421	0.384
DS	.396	.423	—	—
SS	.463	3.70	.490	.549
TN	.256	.669	.336	.279
TKN	.960	1.11	—	—
TP	.598	1.13	.461	.469
DP	.637	1.12	—	—
CD ⁵	.409	.626	—	—
CU	.666	1.15	—	—
PB	.668	.708	.782	.583
ZN	.421	.494	.517	.495
RUN	—	.332	.375	.354

¹Equations from table 3 in Driver and Tasker (1990).

²Equations from table 1 in Driver and Tasker (1990).

³Equations from table 19 in Ellis and others (1984).

⁴Equations from table 20 in Ellis and others (1984).

⁵Root-mean-square error computed without censored data.

R0024103

Differences between the observed and estimated storm-runoff loads, volume, and event-mean concentrations can be explained by the following:

1. Hydrologic conditions controlling the detection of properties and constituents specific to Colorado Springs are not explained by the Driver and Tasker (1990) models.
2. Data collected for certain properties and constituents for the NURP studies might not be representative of the Colorado Springs area.
3. Regional-regression models were developed using a larger range of drainage-basin areas than the drainage-basin areas used for this study (tables 1 and 6). Therefore, regional-regression models might be biased and might be overestimating storm-runoff loads, volume, and event-mean concentrations for smaller drainage basins.

Procedures for Adjustment of Estimates from Single-Storm Regional-Regression Models Using Local Data

When compared to observed data, single-storm regional-regression models tended to overestimate storm-runoff loads, volume, and event-mean concentrations. As a result, single-storm local-regression models would be the preferred method for estimating storm-runoff loads, volume, and event-mean concentrations because the single-storm local-regression models were developed using local data based on the climatic, physical, and land-use characteristics of the Colorado Springs area. However, only a small number of observations (30--snowmelt samples not included) were available for the development of the single-storm local-regression models, and the use of the single-storm local-regression models need to be limited to estimates within the ranges of the explanatory variables used to develop the model (table 6). Single-storm regional-regression models are based on a large number of observations (65 to 348), and the explanatory variables have a wider range than the explanatory vari-

Table 8. Summary of residual values for observed minus estimated values of storm-runoff loads, volume, and event-mean concentrations

[DT1, regional-regression model from table 1 in Driver and Tasker (1990); DT3, regional-regression model from table 3 in Driver and Tasker (1990); EI19, regional-regression model from table 19 in Ellis and others (1984); EI20, regional regression model from table 20 in Ellis and others (1984); DT5, regional-regression model from table 5 in Driver and Tasker (1990); load, storm-runoff loads, in pounds; mg/L, storm-runoff event-mean concentration, in milligrams per liter; µg/L, storm-runoff event-mean concentration, in micrograms per liter; dashes indicate not data available; negative number in the table means the estimated value from the regional regression model is greater than the observed value; positive number in the table means the estimated value from the regional-regression model is less than the observed value]

Response variable	Regression model	Residual values				
		Minimum	Maximum	75th percentile	Median	Mean
Chemical oxygen demand (COD)	DT1 load	-2,525	1,179	-1.4	-175	-347
	DT3 load	-349	1,188	39	-18	16
	EI19 load	-455	1,138	44	6	-13
	EI20 load	-323	12,155	60	11	20
	DT5 mg/L	-1,275	265	-28	-433	-443
Dissolved solids (DS)	DT1 load	-85	226	48	12.4	33
	DT3 load	-124	206	36	-0.05	18
	DT5 mg/L	-126	49	-7.4	-62	-48
Suspended solids (SS)	DT1 load	27	2,130	756	310	448
	DT3 load	-1,021	1,064	148	-31	13
	EI19 load	-678	1,396	372	36	159
	EI20 load	-1,019	1,144	326	22	87
	DT5 mg/L	-489	702	193	-82	-34
Total nitrogen (TN)	DT1 load	0.43	79	8.5	4.0	4.7
	DT3 load	.39	7.09	2.8	2.1	1.8
	EI19 load	.44	12.3	3.9	2.6	2.2
	EI20 load	.55	9.52	2.8	1.9	1.9
	DT5 mg/L	.71	22	12	5.4	4.4
Total ammonia plus organic nitrogen as nitrogen (TKN)	DT1 load	-42	1.5	-1.4	-3.8	-7.6
	DT3 load	-16	-72	-1.7	-4.0	-5.0
	DT5 mg/L	-34	-1.2	-2.3	-10	-12
Total phosphorus (TP)	DT1 load	-22	.95	-0.12	-.60	-3.1
	DT3 load	-1.9	.22	-.13	-.47	-0.60
	EI19 load	.04	2.6	.91	.63	.72
	EI20 load	.04	2.7	.92	.65	.74
	DT5 mg/L	-8.3	.63	-.21	-2.4	-3.1
Dissolved phosphorus (DP)	DT1 load	-9.6	.31	-.06	-.24	-1.5
	DT3 load	-.95	.06	-.12	-.31	-.33
	DT5 mg/L	-4.6	.13	-.02	-.75	-1.6

Table 8. Summary of residual values for observed minus estimated values of storm-runoff loads, volume, and event-mean concentrations--Continued

Response variable	Regression model	Residual values				
		Minimum	Maximum	75th percentile	Median	Mean
Total-recoverable cadmium (CD) ¹	DT1 load	-0.02	0.001	-0.0002	-0.0006	-0.002
	DT3 load	-.002	.002	.0003	-.00006	-.0003
	DT5 µg/L	-2.8	.71	-.07	-.69	-.82
Total-recoverable copper (CU)	DT1 load	-.77	-.02	-.06	-.09	-.15
	DT3 load	-.13	.02	-.01	-.03	-.03
	DT5 µg/l	-168	-30	-46	-77	-82
Total-recoverable lead (PB)	DT1 load	-.66	.47	.11	-.06	-.08
	DT3 load	-.62	.43	-.002	-.09	-.12
	EI19 load	-1.1	.36	.03	-.11	-.21
	EI20 load	-.35	.63	.07	-.02	-.003
	DT5 µg/L	-315	200	18	-124	-90
Total-recoverable zinc (ZN)	DT1 load	-.64	1.1	.19	-.009	.02
	DT 3 load	-.69	.95	.14	-.007	.03
	EI19 load	-.87	1.0	.14	-.009	-.04
	EI20 load	-.76	1.0	.15	-.006	-.01
	DT5 µg/L	-651	869	-133	-288	-238
Volume of runoff (RUN)	DT1 cubic feet	-31,592	51,359	4,483	445	1,439
	EI19 cubic feet	-48,818	35,965	2,417	138	-4,928
	EL20 cubic feet	-32,342	47,411	4,794	964	1,028

¹Computed using uncensored data.

ables used in the single-storm local-regression models (table 6). It would be useful if single-storm local- and regional-regression models could be combined to take advantage of the strengths of both regression models while minimizing the respective deficiencies of the regression models.

Hoos (1991) presented a procedure to adjust single-storm regional-regression models using local data. Hoos and Sisolak (1993) evaluated different model-adjustment procedures (MAP's) and established criteria for selecting the appropriate MAP. The MAP is in the form of a regression analysis. Local data are used as the calibration data set. In one MAP (MAP-R-P), log-transformed local (observed) data (response variables) are regressed against the log-transformed estimates from the single-storm regional-regression models (explanatory variables). The resulting equations are the adjusted regression models used to predict storm-runoff loads, volume, or event-mean concentrations at an unmonitored site. Another form of MAP (MAP-W) is simply the weighting of log-transformed estimates from

local- and regional-regression models. The equations for the two MAPs for adjusting the regional-regression equations are Hoos and Sisolak, (1993):

1. MAP-R-P calibration equation:

$$\log O_i = \log B_o + B_1 \log R_{Ei} \quad (7)$$

where

- O_i = the observed value of storm-runoff load, volume, or event-mean concentration at site i ;
- B_o and B_1 = coefficients fitted from a simple linear-regression analysis of the calibration data set (observed data); and
- R_{Ei} = the regional estimate; estimated value of storm-runoff load, runoff volume, or event-mean concentration from the unadjusted single-storm regional-regression model at site i .

The adjusted regional-regression model (from the detransformation of eq. 7) is then:

$$AR_{Ei} = B_o R_{Ei}^{B1} \text{ (BCF)} \quad (8)$$

where

AR_{Ei} = the adjusted single-storm regional-regression estimate.

2. MAP-W calibration equation:

$$\log O_i = \{ (J \times \log RE) [(1-J) \times \log LOC] \} \quad (9)$$

where

$$J = \{ (SE_{loc}^2) / [(SE_{loc}^2) + (SE_{reg}^2)] \} \quad (10)$$

where

SE_{loc} = the standard error of estimate, in log units for the single-storm local-regression model;

SE_{reg} = the standard error of estimate, in log units as reported in Driver and Tasker (1990) for the single-storm regional-regression model; and

LOC = estimated value from the single-storm local-regression equation.

The weighted single-storm regional-regression model (the detransformation of eq. 10) then is:

$$WE = \{ (RE^J) \times (LOC^{(1-J)}) \} \times BCF, \quad (11)$$

where

WE = adjusted (weighted) single-storm regional-regression estimate.

Selection of the appropriate MAP needs to be made based on whether or not observed and estimated data are correlated and on if there is a consistent bias between the local-data (observed) and estimated-data pairs (fig. 2, step 3). Correlation between observed and estimated data was evaluated by analyzing the significance of Spearman's rho (Iman and Conover, 1983), and bias was determined using the signed-rank test on the paired data (Iman and Conover, 1983). If the null hypothesis (a significant correlation does not exist between observed and estimated values or a consistent bias does not exist between observed and estimated values) for either test is not rejected at a selected level of significance, then correlation between observed data and explanatory variables is determined by examining correlation coefficients, r^2 (fig. 2, step 4) (Hoos and Sisolak, 1993).

After evaluating Spearman's rho and the signed-rank test for the Colorado Springs data set, the appropriate MAPs were selected for adjusting the regional-regression models using local data based on the flow chart in figure 2. The RMSE was large for all comparisons between observed values and values estimated using single-storm regional-regression equations (tables 7, 9, and 10). Because observed values and estimated values from single-storm regional-regression equations were highly correlated and had a consistent direction of bias, MAP-R-P (fig. 2, steps 2 and 3) was selected for adjusting storm-runoff-load equations for TN, TKN, TP, DP, CU, and PB and for adjusting the storm-runoff event-mean concentration equations for TP, CU, and PB. Observed and estimated values for the remaining storm-runoff loads and event-mean concentrations of the remaining constituents were not highly correlated or did not have a consistent direction of bias, or both; however, the remaining observed values were significantly correlated with some explanatory variables. Therefore, MAP-W (fig. 2, step 3) was selected for adjusting storm-runoff-load equations for COD, DS, SS, CD, and ZN; for adjusting equations for estimating volume of runoff; and for adjusting event-mean concentration equations for COD, DS, SS, TN, TKN, DP, CD, and ZN.

When compared to observed storm-runoff loads and volume, the three-variable single-storm regional-regression models for storm-runoff loads and the multivariate single-storm regional-regression models for storm-runoff volume that were developed by Driver and Tasker (1990) had the smallest RMSE of all of the single-storm regional-regression models tested. These models and the 13-variable single-storm regional-regression models for event-mean concentration were adjusted using MAPs.

The MAP's decreased model error in estimating storm-runoff loads, volume, and event-mean concentrations, except for the equation for event-mean concentration for CD. Reduction of error, in percent, ranged from -1,980 to -10 percent (based on data in tables 9 and 10). The effect of MAP's on estimated storm-runoff loads and event-mean concentrations can be illustrated by plotting observed values, estimates from regional-regression models, and estimates from regional-regression models adjusted using MAPs. Two examples from site 2 are presented, one for each MAP—MAP-R-P (TP) and MAP-W (TKN) (figs. 3 and 4). In both cases, the estimates of storm-runoff load and event-mean concentration obtained using MAP-R-P (fig. 3) and MAP-W (fig. 4) were closer to the observed value than the estimate from the regional-regression equation. Adjusted models, developed using MAP-R-P and MAP W, for estimating storm-runoff loads, volume, and event-mean concentrations are listed in tables 11 and 12.

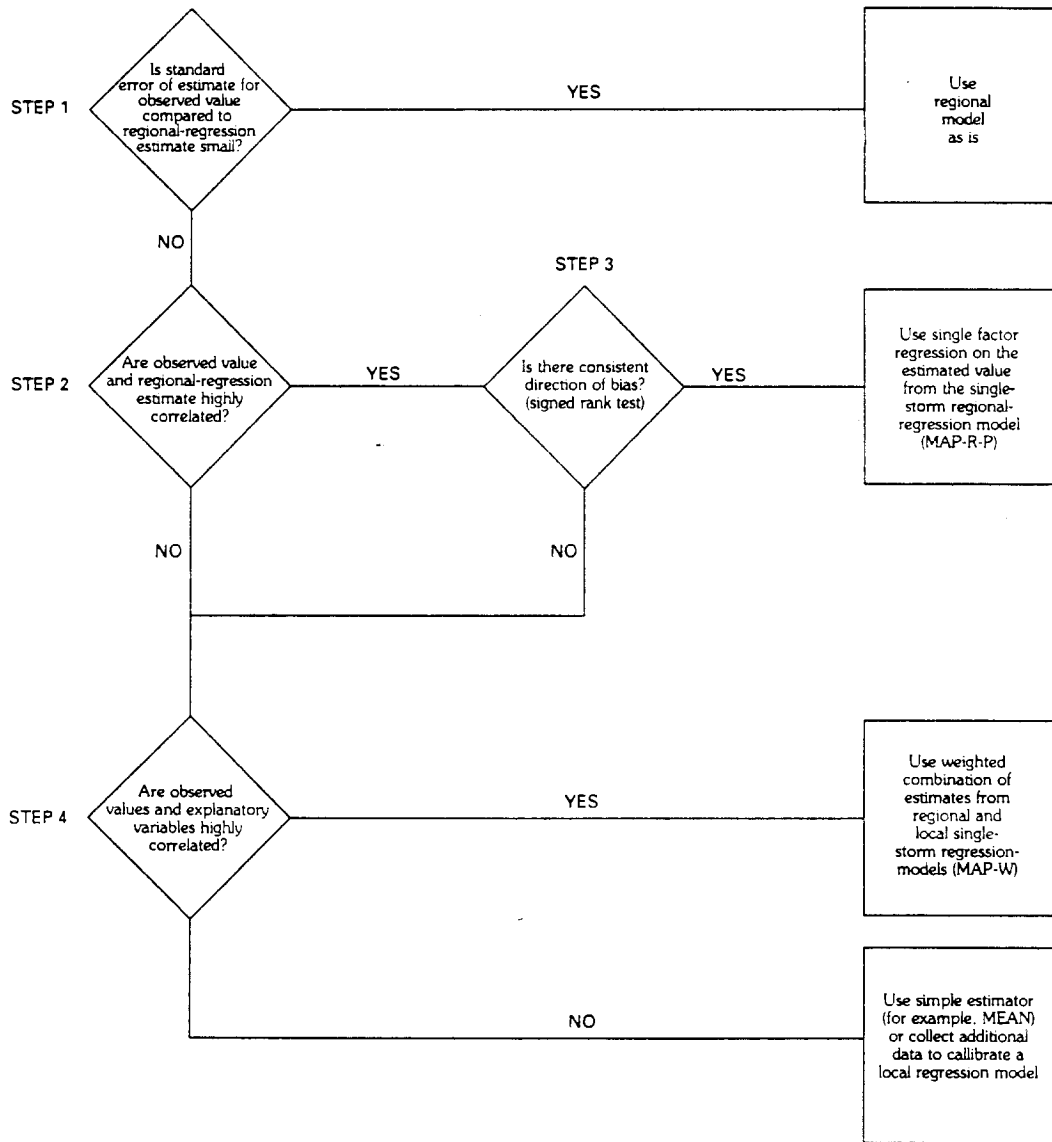


Figure 2. Flowchart for selection of model adjustment procedures (modified from Hoos and Sisolak, 1993).

R0024107

Table 9. Effect of model-adjustment procedures on root-mean-square error and standard error of estimate for storm-runoff loads and volume

[MAP-R-P, observed data regressed against the regional estimate; MAP-W, the weighted combination of local-regression estimate and regional-regression estimate; COD, chemical oxygen demand; DS, dissolved solids; SS, suspended solids; TN, total nitrogen; TKN, total ammonia plus organic nitrogen as nitrogen; TP, total phosphorus; DP, dissolved phosphorus; CD, total-recoverable cadmium; CU, total-recoverable copper; PB, total-recoverable lead; ZN, total-recoverable zinc; RUN, volume; dashes indicate no data]

Response variable	Root-mean-square error				Standard error of estimate			
	Regional-regression model from Driver-Tasker (1990)				MAP-R-P		MAP-W	
	3-variable models ¹		13-variable models ²		(percent)	(log units)	(percent)	(log units)
	(percent)	(log units)	(percent)	(log units)				
COD	97	0.353	--	--	--	--	65	0.258
DS	114	.396	--	--	--	--	59	.237
SS	145	.463	--	--	--	--	102	.366
TN	68	.256	--	--	19	0.283	--	--
TKN	1,147	.960	--	--	106	.377	--	--
TP	238	.598	--	--	109	.384	--	--
DP	276	.637	--	--	95	.348	--	--
CD ³	119	.409	--	--	--	--	81	.308
CU	308	.666	--	--	116	.402	--	--
PB	311	.668	--	--	191	.538	--	--
ZN	125	.421	--	--	--	--	115	.398
RUN	--	--	89	.332	--	--	29	.124

¹Equations from table 3 in Driver and Tasker (1990).

²Equations from table 1 in Driver and Tasker (1990).

³Computed without censored data.

Table 10. Effect of model-adjustment procedures on standard error of estimate for storm-runoff event-mean concentrations

[MAP-R-P, local data regressed against the regional estimate; MAP-W, the weighted combination of local-regression estimate and regional-regression estimate; COD, chemical oxygen demand; DS, dissolved solids; SS, suspended solids; TN, total nitrogen; TKN, total ammonia plus organic nitrogen as nitrogen; TP, total phosphorus; DP, dissolved phosphorus; CD, total-recoverable cadmium; CU, total-recoverable copper; PB, total-recoverable lead; ZN, total-recoverable zinc; dashes indicate not applicable]

Response variable	Root-mean-square error		Standard error of estimate			
	Regional-regression model from Driver and Tasker (1990)		MAP-R-P		MAP-W	
	(percent)	(log units)	(percent)	(log units)	(percent)	(log units)
COD	201	0.553	--	--	41	0.172
DS	69	.272	--	--	45	.188
SS	74	.286	--	--	64	.253
TN	282	.643	--	--	40	.168
TKN	2,078	1.07	--	--	98	.356
TP	1,398	.998	73	0.285	--	--
DP	1,220	.972	--	--	176	.516
CD ²	86	.322	--	--	238	.598
CU	459	.764	54	.219	--	--
PB	186	.531	70	.273	--	--
ZN	121	.413	--	--	44	.183

¹Equations from table 5 in Driver and Tasker (1990).

²Computed without censored data.

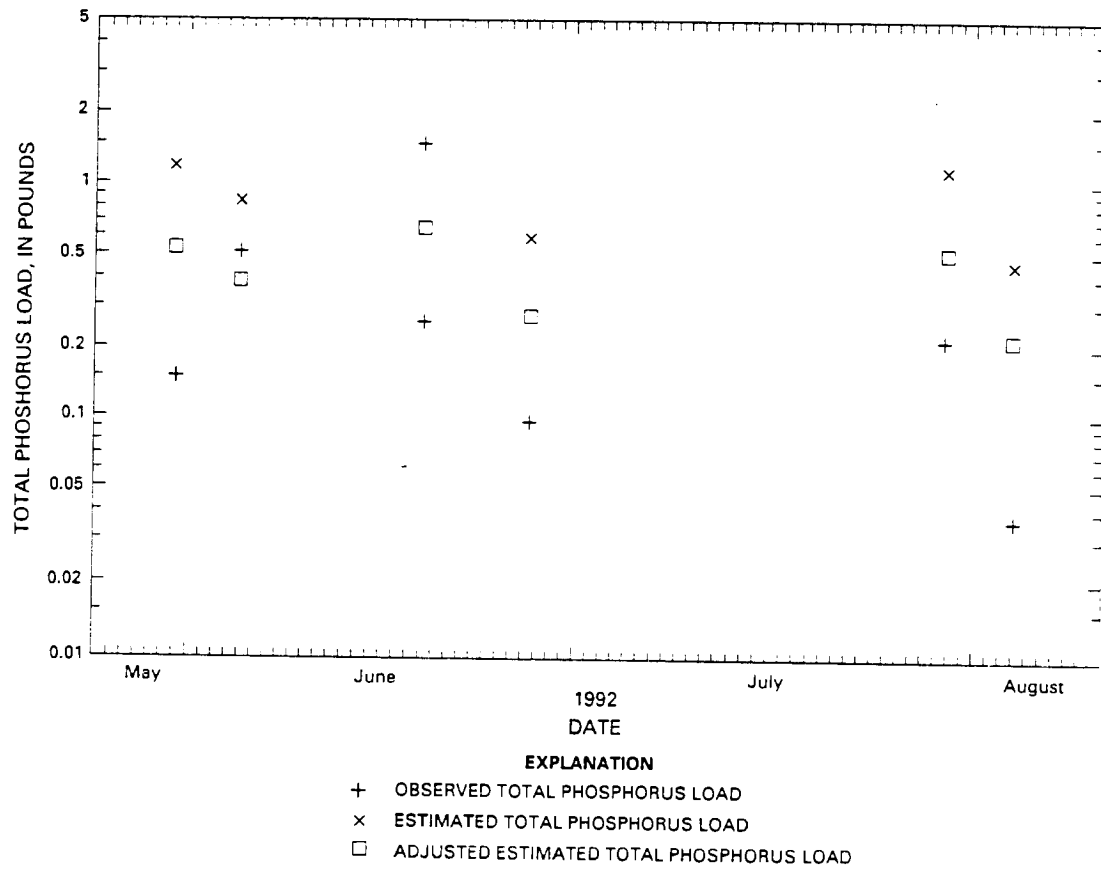
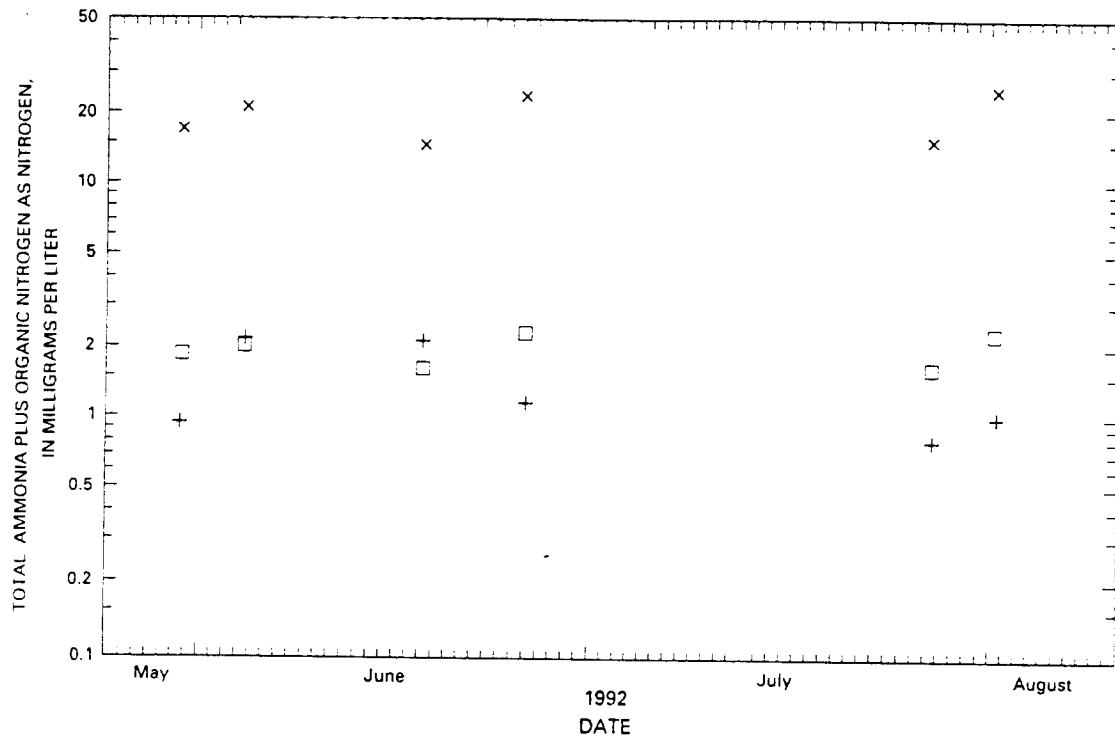


Figure 3. Observed total-phosphorus, estimated total-phosphorus, and adjusted estimates of total-phosphorus loads for storm-runoff load at site 2.

R0024109



- EXPLANATION**
- + OBSERVED TOTAL AMMONIA PLUS ORGANIC NITROGEN AS NITROGEN
 - x ESTIMATED TOTAL AMMONIA PLUS ORGANIC NITROGEN AS NITROGEN
 - ADJUSTED ESTIMATES TOTAL AMMONIA PLUS ORGANIC NITROGEN AS NITROGEN

Figure 4. Observed total ammonia plus organic nitrogen as nitrogen, estimated total ammonia plus organic nitrogen as nitrogen, and adjusted estimates of total ammonia plus organic nitrogen as nitrogen for storm-runoff event-mean concentrations at site 2.

Table 11. Summary of adjusted models for storm-runoff loads and volume

[MAP-R-P, regression of observed data against regional-regression; MAP-W, weighted combination of local-regression estimate and regional-regression estimate; Bo, B1, coefficients fitted from a simple linear regression analysis of the calibration data set (local data base); BCF, the bias correction factor; J, weighting factor; COD, chemical oxygen demand loads; DS, dissolved-solids load; SS, suspended-solids load; TN, total nitrogen load; TKN, total ammonia plus organic nitrogen load; TP, total phosphorus load; DP, dissolved phosphorus load; CD, total-recoverable cadmium load; CU, total-recoverable copper load; PB, total-recoverable lead load; ZN, total-recoverable zinc load; RUN, storm-runoff volume; dashes indicate not applicable; y, response variable; RE, estimate from regional regression; LOC, estimate from local regression]

Response variable	Model-adjustment procedure					
	MAP-R-P ¹			MAP-W ²		
	Bo	B1	BCF	J	1-J	BCF
COD	--	--	--	0.27	0.73	1.13
DS	--	--	--	.35	.65	1.14
SS	--	--	--	.23	.77	1.24
TN	0.82	0.71	1.23	--	--	--
TKN	.23	.61	1.40	--	--	--
TP	.33	.91	1.35	--	--	--
DP	.24	.82	1.32	--	--	--
CD	--	--	--	.85	.15	1.14
CU	.14	.77	1.42	--	--	--
PB	.20	.69	1.80	--	--	--
ZN	--	--	--	.42	.58	1.18
RUN	--	--	--	.24	.76	1.04

¹Form of equation is $y = BoR_{reg}^{B1}BCF$.

²Form of equation is $y = (REG^J)(LOC^{(1-J)})BCF$.

Table 12. Summary of adjusted models for storm-runoff event-mean concentrations

[MAP-R-P, regression of local data against regional estimate; MAP-W, weighted combination of local-regression estimate and regional-regression estimate; Bo, B1, coefficients fitted from a simple linear regression analysis of the calibration data set (local data base); BCF, bias correction factor; J, weighting factor; COD, chemical oxygen demand event-mean concentration; DS, dissolved-solids event-mean concentration; SS, suspended-solids event-mean concentration; TN, total nitrogen event-mean concentration; TKN, total ammonia plus organic nitrogen event-mean concentration; TP, total phosphorus event-mean concentration; DP, dissolved phosphorus event-mean concentration; CD, total-recoverable cadmium event-mean concentration; CU, total-recoverable copper event-mean concentration; PB, total-recoverable lead event-mean concentration; ZN, total-recoverable zinc event-mean concentration; dashes indicate not applicable; y, response variable; RE, estimate from regional model; LOC, estimate from local model]

Response variable	Model-adjustment procedure					
	MAP-R-P ¹			MAP-W ²		
	Bo	B1	BCP	J	1-J	BCF
COD	--	--	--	0.33	0.67	1.08
DS	--	--	--	.18	.82	1.05
SS	--	--	--	.24	.76	1.14
TN	--	--	--	.33	.67	1.07
TKN	--	--	--	.28	.72	1.06
TP	0.48	-0.26	1.2	--	--	--
DP	--	--	--	.22	.78	1.10
CD	--	--	--	.78	.22	1.05
CU	1.52	.54	1.16	--	--	--
PB	886,692	-1.69	1.18	--	--	--
ZN	--	--	--	.25	.75	1.08

¹Form of equation is $y = Bo \times REG^{B1} BCF$.

²Form of equation is $y = (RE^J)(LOC^{(1-J)})BCF$.

R0024111

ESTIMATES OF A MEAN LOAD FOR A STORM

In addition to developing single-storm regional-regression models for storm-runoff loads, volume, and event-mean concentrations, Driver and Tasker (1990) developed regression models for estimating the mean load for a storm, hereafter called mean load. Mean load is the estimate of mean load for a particular drainage basin. With the estimate of mean load, seasonal or annual loads for a particular drainage basin can be estimated by multiplying the mean load by the average number of storms for the season or year. Regression models for estimating mean load were based on drainage-basin area, percent of impervious area, mean annual rainfall, mean minimum January temperature, and a variable (dummy variable) indicating whether commercial and industrial land uses exceeded or did not exceed 75 percent of the drainage-basin area (Driver and Tasker, 1990). Regression models for estimating mean load were developed for COD, DS, SS, TN, TKN, TP, DP, CU, PB, and ZN. These regression models were developed from the NURP data base and are based on rain storms. The range of explanatory variables used in the regression models for estimating mean load are listed in table 13. In general, use of the models to estimate mean load at sites that have characteristics much beyond the range of values listed in

table 13 need to be avoided (Driver and Tasker, 1990). Using the Driver and Tasker (1990) models, mean loads were estimated for COD, DS, SS, TN, TKN, TP, DP, CU, PB, and ZN for sites 1 through 5. A 90-percent confidence interval was computed for each mean load of a storm estimated using the models from Driver and Tasker (1990) (table 14). For example, there is a 90-percent confidence level that the true mean load for TP for all storms at site 1 lies between 0.06 and 0.87 lb (table 14). Confidence intervals were not computed for single-storm regression models because matrix information was unavailable.

Estimated mean loads from the Driver and Tasker (1990) models (MLDT) were compared to mean loads estimated for 1992, based on daily mean water discharge and land-use characteristics, hereafter referred to as MLDWD, and on loads that are the mean storm-runoff load of the six storms sampled at each of the five sites in 1992 (table 14).

Linear regression was used for estimating mean loads based on daily mean water discharge. The most suitable regression models for estimating mean load from daily mean water discharge were selected using the procedures described in the section "Single-Storm Local-Regression Models." Values of r^2 ranged from 0.59 to 0.84, and standard error of estimate, in percent, for the regression models, ranged from 45 to 93 (table 15).

Table 13. Ranges of values of explanatory variables used in development of regression models for mean load for a storm (modified from Driver and Tasker, 1990)

[DA, total contributing drainage area; IA, impervious area; MAR, mean annual rainfall; MJT, mean minimum January temperature; COD, chemical oxygen demand; DS, dissolved solids; SS, suspended solids; TN, total nitrogen; TKN, total ammonia plus organic nitrogen as nitrogen; TP, total phosphorus; DP, dissolved phosphorus; CU, total-recoverable copper; PB, total-recoverable lead; ZN, total-recoverable zinc]

Response variable (mean seasonal or mean annual load)	Explanatory variables							
	DA (square miles)		IA (percent)		MAR (Inches)		MJT (degrees Fahrenheit)	
	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum
COD	0.019	0.707	4	100	8.38	62.00	3.2	58.7
DS	.020	.450	19	99	10.24	37.61	11.4	35.8
SS	.019	.707	4	100	8.38	49.38	3.2	50.1
TN	.019	.830	4	100	11.83	62.00	3.2	58.7
TKN	.019	.707	4	100	8.38	62.00	3.2	58.7
TP	.019	.830	4	100	8.38	62.00	3.2	58.7
DP	.020	.707	4	99	8.38	62.00	3.2	58.7
CU	.014	.830	6	99	8.38	46.18	10.8	35.8
PB	.019	.830	4	100	8.38	62.00	15.3	58.7
ZN	.019	.830	13	100	8.38	62.00	3.2	58.7

Table 14. Comparison of computed mean load for a storm based on daily mean water discharge and land-use characteristics, estimated mean load for a storm, and average of samples collected during rainfall runoff

[MLDWD, mean load estimated based on daily mean water discharge and land-use characteristics; MLDT, estimated mean load, from Driver and Tasker (1990) models; CI, 90-percent confidence interval; n, number of storms occurring as rain, January through December 1992; COD, chemical oxygen demand; DS, dissolved solids; SS, suspended solids; TN, total nitrogen; TKN, total ammonia plus organic nitrogen as nitrogen; TP, total phosphorus; DP, dissolved phosphorus; CU, total-recoverable copper; PB, total-recoverable lead; ZN, total-recoverable zinc]

Response variable (as load)	Mean load of a storm (pounds)				Mean of six storms in 1992 (pounds)
	MLDWD	MLDT Regression models in Driver and Tasker (1990)			
		Estimated	Lower CI	Upper CI	
Sixteenth Hole, Valley-Hi Golf Course					
Site 1 n=54					
COD	756	172	41	430	492
DS	232	294	58	947	201
SS	927	168	18.2	666	716
TN	4.3	6.63	1.24	19.5	3.7
TKN	2.4	2.16	0.43	6.68	2.3
TP	0.73	0.31	.06	0.87	0.71
DP	.28	.20	.03	.60	.24
CU	.03	.12	.02	.38	.03
PB	.54	.28	.05	.84	.27
ZN	.57	.42	.09	1.11	.49
Chestnut Street at Douglas Creek					
Site 2 n=48					
COD	523	172	41	430	217
DS	77	400	77	1,330	60
SS	1,349	203	22.1	809	603
TN	1.9	6.04	1.15	17.6	1.3
TKN	.90	1.94	.38	6.11	.76
TP	.33	.40	.08	1.12	.21
DP	.14	.23	.04	.72	.10
CU	.07	.15	.02	.47	.04
PB	.36	.25	.04	.77	.24
ZN	1.42	.39	.08	1.02	.76
Beacon Street at Buchanan Street					
Site 3 n=50					
COD	312	223	53	560	92
DS	71	424	81	1,410	37
SS	433	210	22.9	837	107
TN	2.0	2.9	0.49	9.7	1.2
TKN	0.80	0.97	.16	3.66	0.40
TP	.26	.42	.09	1.17	.16
DP	.17	.24	.04	.74	.12
CU	.02	.16	.03	.49	.007
PB	.19	.35	.06	1.07	.03
ZN	.24	.55	.12	1.43	.12

R0024113

Table 14. Comparison of computed mean load for a storm based on daily mean water discharge and land-use characteristics, estimated mean load for a storm, and average of samples collected during rainfall runoff—Continued

Response variable (as load)	Mean load of a storm (pounds)				Mean of six storms in 1992 (pounds)
	MLDWD	MLDT Regression models in Driver and Tasker (1990)			
		Estimated	Lower CI	Upper CI	
Wahsatch Street at Cross Lane					
Site 4 n=50					
COD	465	357	83.3	906	378
DS	117	1,060	173	4,140	140
SS	929	372	39.7	1,510	747
TN	4.7	10.8	2.01	31.9	5.2
TKN	.74	3.42	.66	10.9	.76
TP	1.22	.88	.18	2.5	1.20
DP	.36	.40	.06	1.32	.46
CU	.02	.30	.05	.95	.02
PB	.31	.54	.09	1.67	.14
ZN	.31	.80	.17	2.13	.31
Walmart at Eighth Street					
Site 5 n=51					
COD	339	76	18	190	58
DS	45	136	27	432	26
SS	407	103	11.1	416	237
TN	0.89	3.12	0.58	9.22	0.64
TKN	.65	1.02	.19	3.26	.27
TP	.20	.16	.03	0.46	.15
DP	.09	.13	.02	.40	.05
CU	.01	.07	.01	.22	.005
PB	.12	.12	.02	.36	.06
ZN	.20	.17	.04	.45	.11

Table 15. Summary of r^2 values and standard error of estimate for regression models used to estimate mean load, in pounds, of a storm based on daily mean water discharge and land-use characteristics

[r^2 , is the coefficient of determination]

Response variable	r^2	Standard error of estimate	
		Percent	Log units
Chemical oxygen demand	0.75	57	0.229
Dissolved solids	.82	45	.188
Suspended solids	.68	83	.313
Total nitrogen	.81	47	.195
Total ammonia plus organic nitrogen	.78	47	.193
Total phosphorus	.75	66	.262
Dissolved phosphorus	.59	81	.308
Total-recoverable copper	.84	49	.200
Total-recoverable lead	.65	93	.344
Total-recoverable zinc	.81	55	.225

R0024114

Records of daily mean water discharge were collected at all sites from about June 1 to December 6, 1992. For periods of missing record, daily mean water discharge was estimated using the following equation:

$$DMWD = 19,941 TRN^{1.08} LUI^{-1.26} LUC^{-0.98} LUR^{-1.78} \quad (12)$$

where

DMWD = daily mean water discharge, in cubic feet per second;

TRN = total rainfall, in inches;

LUI = industrial land use, in percent;

LUC = commercial land use, in percent; and

LUR = residential land use in percent.

For this equation, the value of r^2 was 0.77, and the standard error of estimate was 78 percent.

Generally, MLDWD were within the 90-percent confidence intervals of MLDT (table 14). However, estimates of COD, DS, SS, and CU, at selected sites were not within the 90-percent confidence intervals from Driver and Tasker (1990) mean load equations. At sites 1, 2, and 5, estimates of MLDWD for COD exceeded the upper 90-percent confidence interval (table 14); and at sites 1 and 2, MLDWD for SS exceeded the upper 90-percent confidence limit (table 14). At sites 3 and 4, estimates of MLDWD for DS and CU were less than the 90-percent confidence interval, and at site 2, estimates of mean load based on daily mean water discharge for ZN exceeded the upper 90-percent confidence interval (table 14).

At all sites, estimates of MLDWD for COD and SS were larger than MLDT (table 14). At all sites, estimates of MLDWD for DS, TN, and CU were less than MLDT (table 14). Estimates of MLDWD for TKN, TP, DP, PB, and ZN based on daily mean water discharge when compared to MLDT had no consistent direction of bias (table 14).

Differences between the two types of estimates of mean load may be explained by the following:

1. Hydrologic conditions controlling the occurrence of properties and constituents, especially COD and SS, that are specific to Colorado Springs are not accounted for in the Driver and Tasker (1990) models.
2. Data collected for certain properties and constituents for the NURP studies may not be representative of the Colorado Springs area.

Generally, the mean load of six storms at each site was within the 90-percent confidence interval of the Driver and Tasker (1990) mean load equations (table 14). The mean load of the six storms exceeded

the upper 90-percent confidence interval for COD and SS at site 1 (table 14). The mean load of the six storms was less than the lower 90-percent confidence interval for DS at sites 2, 3, 4, and 5; for CU at sites 3, 4, and 5; and for PB at site 3 (table 14).

The mean load of the six storms at each site compared well with the estimates derived using the Driver and Tasker (1990) mean load equations. However, these mean loads represent only 6 storms, whereas the Driver and Tasker (1990) models were developed using between 200 and 1,000 storms that represent drainage basins having a wider range of drainage-basin area and percent impervious area (table 6). Therefore, the Driver and Tasker (1990) mean load equations might provide a better estimate of annual and seasonal loads for engaged drainage basins in Colorado Springs.

COMPARISON OF PROCEDURES FOR ESTIMATING STORM-RUNOFF LOADS, VOLUMES, EVENT-MEAN CONCENTRATIONS, AND THE MEAN LOAD FOR A STORM

Various procedures for estimating storm-runoff loads, volume, and event-mean concentrations have been discussed in this report. The following is a more concise comparison of the value and limitations of these procedures. The procedures discussed include:

1. Single-storm local-regression models for storm-runoff loads, volume, and event-mean concentration (tables 4 and 5).
2. Single-storm regional-regression models for storm-runoff loads, volume, and event-mean concentration (Ellis and others, 1984; Driver and Tasker, 1990).
3. Adjustment of single-storm regional-regression models for storm-runoff loads, volume, and event-mean concentration using local data (tables 11 and 12) (Hoos and Sisolak, 1993).
4. Estimates of mean load (table 14).

The use of single-storm local-regression models needs to be limited to the ranges of explanatory variables (table 6) used to develop the model. If values outside these ranges are used in the single-storm local-regression models, the standard errors may be considerably larger than the values reported in tables 4 and 5. As the single-storm local-regression models are applied to drainage-basin areas and to storm volumes larger than the values from the observation sites, the accuracy of estimates of storm-runoff loads, volume, and event-mean concentrations decreases.

Single-storm regional-regression models were developed using explanatory variables that have a wider range than the single-storm local-regression models (table 6). When compared to observed data, single-storm regional-regression models tended to overestimate storm-runoff loads, volumes, and event-mean concentrations (table 8). Model adjustment procedures (MAPs), which use local data to decrease the model error, were applied to selected single-storm regional-regression models. The MAPs decreased model error in estimating storm-runoff loads, volume, and event-mean concentrations—model error decreased from -1,980 to -10 percent (based on data in tables 9 and 10).

A prediction of annual or seasonal storm-runoff loads, volume, and event-mean concentration at an unmonitored site can be obtained by applying the single-storm models described to a series of storms and producing a synthetic record of storm loads and volume. Values of storm characteristics used as explanatory variables listed in table 6 and in Ellis and others (1984) and Driver and Tasker (1990) may be determined for a series of storms from the long-term rainfall record for a station near an unmonitored site. The synthesized record of storm loads may be reduced to an estimate of mean annual or mean seasonal load by summing loads from each storm, then dividing by the number of years in the period of the synthetic record.

The mean load estimated for individual sites for selected constituents generally compared well to mean load estimated based on daily mean water discharge and land-use characteristics and to the mean load of six storms, for each site, sampled in 1992 (table 14). The use of the mean load procedure should be limited to the range of values of variables used to develop the models (table 13). However, the mean load procedure can be applied to larger drainage basins by dividing the drainage basin into segments that fall into the range of drainage-basin areas used to develop the mean load model and computing the mean load for each drainage-basin segment. The mean load for the drainage basin would be the sum of the loads computed for each drainage-basin segment. Annual or seasonal loads could be computed by multiplying the estimated mean load by the average, or total for a specific year, number of storms for a drainage basin.

SUMMARY

The U.S. Environmental Protection Agency requires that municipalities that have a population of 100,000 or greater obtain National Pollutant Discharge Elimination System permits to control the quality of storm runoff. In 1992, the U.S. Geological Survey, in cooperation with the Colorado Springs City Engineer-

ing Division, began a study to characterize the water quality of storm runoff and to compare techniques for the estimation of storm-runoff loads, volume, and event-mean concentrations for selected properties and constituents.

Precipitation, streamflow, and water-quality data were collected during 1992 at five sites in Colorado Springs. Thirty-five samples were collected, seven at each of the five sites. At each site, three samples were collected for National Pollutant Discharge Elimination System permitting purposes; two of the samples were collected during rainfall runoff, and one sample was collected during snowmelt runoff. Four additional samples were collected at each site to obtain a large enough sample size to estimate storm-runoff loads, volume, and event-mean concentrations for selected properties and constituents using linear-regression procedures developed using data from the Nationwide Urban Runoff Program (NURP). Storm-water samples were analyzed for as many as 186 properties and constituents. Some of the properties and constituents measured include pH, specific conductance, water temperature, chemical oxygen demand, biochemical oxygen demand, bacteria, dissolved and suspended solids, major ions, nutrients, residual chlorine, total-recoverable metals, oil and grease, phenols, volatile-organic compounds, acid-base/neutral organic compounds and pesticides.

Storm runoff sampled had large concentrations of chemical oxygen demand and 5-day biochemical oxygen demand. Chemical oxygen demand ranged from 100 to 830 mg/L, and 5-day biochemical oxygen demand ranged from 14 to 260 mg/L. Total-organic carbon concentrations ranged from 18 to 240 mg/L. The total-recoverable metals lead and zinc had the largest concentrations of the total-recoverable metals analyzed. Concentrations of lead ranged from 23 to 350 $\mu\text{g/L}$, and concentrations of zinc ranged from 110 to 1,400 $\mu\text{g/L}$.

Single-storm local-regression models for estimating storm-runoff loads, volume, and event-mean concentrations were developed. Results from these models and observed values for storm-runoff loads, volume, and event-mean concentrations are compared with the results from regional-regression models developed from the Nationwide Urban Runoff Program for the purposes of determining which regression models provide the best estimates of storm-runoff loads, volume, and event-mean concentrations.

Single-storm local-regression models were developed for estimating storm-runoff loads and event-mean concentrations for the 12 National Pollutant Discharge Elimination System properties and constituents and for estimating storm-runoff volume. The data for

30 storms representing rainfall runoff from 5 drainage basins were used in this analysis. Except for total-recoverable cadmium, the models were developed using ordinary least-squares regression. Because some cadmium concentrations were censored (less than values), tobit regression, which is similar to ordinary least-squares regression, was used to estimate storm-runoff load and event-mean concentration for total-recoverable cadmium. The response variables, which are storm-runoff loads, volume, and event-mean concentrations, were modeled using climatic, physical, and land-use characteristics.

The values of r^2 for models that use ordinary least-squares regression ranged from 0.57 to 0.86 for storm-runoff loads and volume and from 0.25 to 0.63 for storm-runoff event-mean concentrations. Standard errors of estimate ranged from 43 to 115 percent for storm-runoff loads and volume and from 32 to 66 percent for storm-runoff event-mean concentrations. Standard errors of estimate for storm-runoff load and event-mean concentration for total-recoverable cadmium were 247 and 171 percent.

Single-storm linear-regression models for estimating storm-runoff loads, volume, and event-mean concentrations were developed for 11 of the 12 properties and constituents required for the National Pollutant Discharge Elimination System permitting process. Regional-regression equations for BOD were not developed. Linear-regression equations were developed from data collected by the NURP. Equations were developed from the NURP data base and include sets of equations for three geographically distinct regions delineated by mean annual rainfall. The Colorado Springs area is included in Region I. Estimates from these regression models were compared with observed storm-runoff loads, volume, and event-mean concentrations from samples collected in the study area.

Single-storm regional-regression models tended to overestimate storm-runoff loads, volume, and event-mean concentrations observed at the five Colorado Springs sites. Because regression models developed using local data are based on the climatic, physical, and land-use characteristics of the Colorado Springs area, single-storm local-regression models would be the preferred method for estimating storm-runoff loads, volume, and event-mean concentrations. As a result, single-storm local-regression models would be the preferred method for estimating storm-runoff loads, volume, and event-mean concentrations because the single-storm local-regression models were developed using local data based on the climatic, physical, and land-use characteristics of the Colorado Springs area. However, only a small number of observations (30) were available for the development of the single-storm

local-regression models, and the use of the single-storm local-regression models needs to be limited to estimations within the ranges of the explanatory variables used to develop the model. Although the single-storm regional-regression models tended to overestimate storm-runoff loads, volume, and event-mean concentrations, these single-storm regional-regression models are based on a large number of observations (65 to 348), and the explanatory variables have a wider range than the explanatory variables used in the single-storm local-regression models. Single-storm local- and regional-regression models were combined using model-adjustment procedures to take advantage of the strengths of both models while minimizing the respective deficiencies of the models.

When compared to observed storm-runoff loads and volume, the adjusted three-variable single-storm regional-regression models for storm-runoff loads and the multivariate regional-regression model for volume of runoff had the smallest root-mean-squared error of all of the single-storm regional-regression models tested. These models and the single-storm regional-regression models for event-mean concentration were adjusted using model-adjustment procedures.

Except for the equation for event-mean concentration of total-recoverable cadmium, all model-adjustment procedures decreased the error for models estimating storm-runoff loads, volume, and event-mean concentrations. Reduction of standard error, in percent, ranged from -1,980 to -10 percent.

In addition to developing single-storm regional-regression models for storm-runoff loads, volume, and event-mean concentrations, regression models for estimating mean load for a storm were developed for ten of the National Pollutant Discharge Elimination System properties and constituents. With the estimate of the mean load, seasonal or annual loads can be estimated by multiplying the mean load by the mean number of storms for the season or year. These regression models were developed from the NURP data base and are based on rain storms.

Mean loads were estimated for ten of the National Pollutant Discharge Elimination System properties and constituents for sites 1 through 5. Estimated mean loads from the regional-regression equations were compared to mean loads estimated for 1992, based on daily mean water discharge, and on loads that are the mean storm-runoff load of the six storms sampled at each of the five sites in 1992.

Except for selected estimates of chemical oxygen demand, dissolved solids, suspended solids, and total-recoverable copper, mean loads based on daily mean water discharge were within the 90-percent confidence interval of results from the mean load equa-

34 Water Quality of Storm Runoff and Comparison of Procedures for Estimating Storm-Runoff Loads, Volume, Event-Mean Concentrations, and the Mean Load for a Storm for Selected Properties and Constituents for Colorado Springs, Southeastern Colorado, 1992

tions. At sites 1, 2, and 5, estimates of mean load based on daily mean water discharge for chemical oxygen demand, and at sites 1 and 2, estimates of mean load based on daily mean water discharge for suspended solids exceeded the upper 90-percent confidence intervals. At sites 3 and 4, estimates of mean load based on daily mean water discharge for dissolved solids and total-recoverable copper were less than the 90-percent confidence interval.

At all sites, estimates of mean load based on daily mean water discharge for chemical oxygen demand and suspended solids were larger than mean loads estimated by the mean load equations. At all sites, estimates of mean load based on daily mean water discharge for dissolved solids, total nitrogen, and total-recoverable copper were smaller than mean loads estimated by the equations. Estimates of mean load for total ammonia plus organic nitrogen as nitrogen, total phosphorus, dissolved phosphorus, total-recoverable lead, and total-recoverable zinc based on daily mean water discharge when compared to mean loads estimated by the mean load equations had no consistent direction of bias. Generally, the mean load of six storms was within the 90-percent confidence interval of the mean load equations. The mean load of the six storms for each site exceeded the upper 90-percent confidence interval for chemical oxygen demand and suspended solids at site 1. The mean load of the six storms was less than the lower 90-percent confidence interval for dissolved solids at sites 2, 3, 4, and 5; for total-recoverable copper at sites 3, 4, and 5; and for total-recoverable lead at site 3.

The mean load of the six storms at each site compared well with the estimates derived using the mean load equations. However, these mean loads represent only 6 storms, whereas the equations were developed using between 200 and 1,000 storms that represent drainage basins having a wider range of drainage-basin area and percent impervious area. Therefore, the mean load equations might provide a better estimate of mean loads, and hence annual and seasonal load, for ungaged drainage basins in Colorado Springs.

REFERENCES CITED

- Chow, Ven Te, 1964, *Handbook of applied hydrology; a compendium of water-resources technology*: New York, McGraw-Hill, variously paged.
- Driver, N.E., and Tasker, G.D., 1990, *Techniques for estimation of storm-runoff loads, volumes, and selected constituent concentrations in urban watersheds in the United States*: U.S. Geological Survey Water-Supply Paper 2363, 44 p.
- Duan, Naihua, 1983, Smearing estimate—A nonparametric retransformation method: *Journal of the American Statistical Association*, v. 78, no. 383, p. 605–610.
- Ellis, S.R., Doerfer, J.T., Mustard, M.H., Blakely, S.R., and Gibbs, J.W., 1984, *Analysis of urban storm-runoff data and the effects on the South Platte River, Denver metropolitan area, Colorado*: U.S. Geological Survey Water-Resources Investigations Report 84-4159, 66 p.
- Hansen, W.R., Chronic, John, and Matelock, John, 1978, *Climatology of the Front Range urban corridor and vicinity, Colorado* (reprint 1979): U.S. Geological Survey Professional Paper 1019, 59 p. (Reprinted in 1979)
- Helsel, D.R., and Hirsch, R.M., 1992, *Statistical methods in water resources*: New York, Elsevier, 522 p.
- Hoos, A.B., 1991, Storm-runoff quality and its relation to land use in the Nashville metropolitan area, Tennessee, in Jennings, M.E., ed., *Symposium proceedings on urban hydrology*: Bethesda, Md., American Water Resources Association, Technical Publication Series, TPS-91-4, p. 33.
- Hoos, A.E., and Sisolak, J.K., 1993, *Procedures for adjusting regional-regression models of urban-runoff quality using local data*: U.S. Geological Survey Open-File Report 93-39, 39 p.
- Iman, R.L., and Conover, W.J., 1983, *A modern approach to statistics*: New York, John Wiley, 497 p.
- Kilpatrick, F.A., Kaehrle, W.R., Hardee, Jack, Cordes, E.H., and Landers, M.N., 1985, *Development and testing of highway storm-sewer flow measurement and recording system*: U.S. Geological Survey Water-Resources Investigations Report 85-4111, 98 p.
- Statistical Analysis System Institute, Inc., 1990, *SAS/STAT user's guide—Statistics* (Version 6, 4th ed.): Cary, N.C., SAS Institute, Inc., 1661 p.
- U.S. Environmental Protection Agency, 1990, *National pollutant discharge elimination system permit, permit application regulations for storm water discharges; final rule*: *Federal Register*, v. 55, no. 222, Code of Federal Regulations, title 40, v. 55, no. 222, parts 122, 123, 124, p. 47990–48091.
- U.S. Geological Survey, 1970, *The national atlas of the United States of America*, 335 p.
- von Guerard, Paul, 1989, *Suspended sediment and sediment-source areas in the Fountain Creek drainage basin upstream from Widefield, southeastern Colorado*: U.S. Geological Survey Water-Resources Investigations Report 88-4136, 36 p.

SUPPLEMENTAL DATA

Table 16. Precipitation and runoff characteristics for samples collected at storm-runoff sites in Colorado Springs
 [--, not available]

Date	Total precipitation (inches)	Runoff (cubic feet)	Duration of runoff (minutes)	Peak flow (cubic feet per second)	Period that has less than 0.10 inch of precipitation (days)	Duration of precipitation (minutes)
384918104454201 - Sixteenth Hole, Valley-Hi Golf Course (Site 1)						
06-03-92	0.21	46,100	155	37.1	2.0	110
06-23-92	.07	26,800	170	8.57	2.0	101
06-26-92	.13	14,900	125	16.8	.5	50
07-02-92	.08	16,400	240	7.19	3.0	--
¹ 07-25-92	.33	78,500	325	22.3	9.0	127
¹ 08-10-92	.20	45,900	395	20.5	7.0	240
¹ 11-21-92	.78	34,600	560	2.99	8.0	--
385347104500601 - Chestnut Street at Douglas Creek (Site 2)						
¹ 05-31-92	.22	20,500	545	6.06	3.0	100
06-05-92	.15	11,400	115	12.1	5.0	15
¹ 06-19-92	.29	16,400	395	15.2	14.0	118
06-27-92	.10	8,330	70	8.11	.5	25
07-29-92	.23	23,200	195	20.5	5.0	10
08-03-92	.08	7,920	115	4.92	5.0	39
¹ 11-21-92	.25	10,200	620	0.87	8.0	--
385240104493601 - Beacon Street at Buchanan Street (Site 3)						
¹ 06-05-92	.12	3,470	125	2.78	5.0	15
¹ 06-19-92	.29	15,800	355	4.92	8.0	115
06-23-92	.14	5,770	210	1.24	4.0	120
06-27-92	.08	3,960	205	2.59	2.0	--
08-02-92	.06	3,450	165	2.14	5.0	15
08-03-92	.14	7,030	265	3.72	1.0	13
¹ 11-11-92	.24	3,620	515	0.50	10.0	--
385118104485801 - Wahsatch Street at Cross Lane (Site 4)						
05-26-92	.30	10,400	195	3.19	1.0	120
06-03-92	.25	9,400	160	10.2	3.0	15
¹ 06-12-92	.10	6,030	150	3.82	3.0	15
06-26-92	.32	41,400	285	15.4	1.0	45
¹ 07-25-92	.41	48,900	285	13.9	8.5	95
07-29-92	.19	22,600	175	13.9	4.0	10
¹ 11-21-92	.27	6,670	545	0.72	8.0	--
384935104501501 - Walmart at Eighth Street (Site 5)						
06-10-92	.10	6,890	115	8.58	1.0	10
06-26-92	.05	1,560	60	2.87	3.0	35
¹ 07-17-92	.15	5,990	205	9.59	9.0	7
¹ 08-03-92	.10	2,990	70	5.35	6.0	20
08-04-92	.05	1,440	70	1.92	0.8	35
08-05-92	.17	6,400	105	5.09	1.1	35
¹ 12-06-92	.15	345	210	0.10	15.0	--

¹Samples required for storm-water permit.

38 Water Quality of Storm Runoff and Comparison of Procedures for Estimating Storm-Runoff Loads, Volume, Event-Mean Concentrations, and the Mean Load for a Storm for Selected Properties and Constituents for Colorado Springs, Southeastern Colorado, 1992

Table 17. Values for instantaneous water discharge, pH, specific conductance, water temperature, fecal coliform, fecal streptococci, residual chlorine, cyanide, oil & grease, and phenols for storm-runoff sites in Colorado Springs

(ft³/s, cubic feet per second; μS/cm, microsiemens per centimeter at 25 degrees Celsius, deg C, degrees Celsius; cols./100 mL, colonies per 100 milliliters; K, nonideal count; mg/L, milligrams per liter; μg/L, micrograms per liter; <, less than; dashes indicate no data)

Date	Time	Instantaneous water discharge (ft ³ /s)	pH (standard units)	Specific conductance (μS/cm)	Temperature, water (deg C)	Fecal coliform (cols./100 mL)	Fecal streptococci (cols./100 mL)	Total residual chlorine (mg/L)	Total cyanide (mg/L)	Total recoverable oil and grease (mg/L)	Gross phenols (μg/L)
384918104454201 - Sixteenth Hole, Valley-III Golf Course (Site 1)											
07-25-92	1424	7.0	7.3	--	19.0	--	--	0	<0.01	2	13
07-25-92	1442	20	8.7	--	18.5	--	--	0	<.01	5	8
07-25-92	1553	2.2	7.5	--	19.5	--	--	0	<.01	<1	9
08-10-92	0910	6.2	6.7	--	19.0	1,300	K22,000	0	<.01	3	8
08-10-92	0955	.31	7.5	--	17.0	K223,000	53,000	0	<.01	<1	9
08-10-92	1050	.18	7.8	--	15.0	60,000	K103,000	0	<.01	<1	8
08-24-92	0825	.40	--	--	--	1,900	9,100	--	--	--	--
11-21-92	0935	.40	7.2	380	3.5	K175	2,980	0	--	--	--
11-21-92	1311	2.55	8.1	240	4.5	--	--	0	<.01	10	11
11-21-92	1540	.51	7.7	260	4.0	K75	1,070	0	--	--	--
385347104500601 - Chestnut Street at Douglas Creek (Site 2)											
05-31-92	1330	.43	7.3	385	19.0	840	9,300	0	<.01	4	4
05-31-92	1430	1.71	6.7	68	15.5	K873	K28,000	0	<.01	<1	4
05-31-92	1535	3.10	7.8	52	13.5	K1,150	4,800	0	<.01	4	5
06-19-92	1731	15.2	6.5	--	22.5	21,000	K12,000	0	<0.01	6	6
06-19-92	1830	.74	7.4	--	20.0	23,000	K13,000	0	<.01	5	6
11-21-92	0920	.04	--	--	.5	K100	1,950	0	--	--	--
11-21-92	1050	.21	7.9	390	2.0	--	--	0	<.01	2	14
11-21-92	1220	.61	8.3	750	2.0	K67	K1,400	0	--	--	--
11-21-92	1345	.74	8.3	450	2.0	--	--	0	--	--	--
11-21-92	1635	.07	8.0	290	3.5	K50	K1,070	0	--	--	--
385240104493601 - Beacon Street at Buchanan Street (Site 3)											
06-05-92	1300	1.0	6.7	92	20.5	480	27,000	--	<.01	5	10
06-05-92	1310	.50	6.7	92	20.5	1,375	7,500	--	<.01	4	9
06-19-92	1740	3.1	6.3	135	22.0	2,900	5,600	0	<.01	4	7
06-19-92	1800	4.3	7.2	76	19.5	3,500	K14,000	0	<.01	6	7
06-19-92	1835	.88	7.2	71	20.5	K2,100	K15,000	0	<.01	5	7
11-11-92	0805	.15	--	--	5.0	K20,000	59,000	0	--	--	--
11-11-92	1020	.43	7.9	--	4.0	--	--	0	<.01	7	11
11-11-92	1245	.09	7.9	--	8.0	K31,000	K36,600	--	--	--	--

SUPPLEMENTAL DATA 39

R0024121

Table 17. Values for instantaneous water discharge, pH, specific conductance, water temperature, fecal coliform, fecal streptococci, residual chlorine, cyanide, oil and grease, and phenols for storm-runoff sites in Colorado Springs--Continued

Date	Time	Instantaneous water discharge (ft ³ /s)	pH (standard units)	Specific conductance (µS/cm)	Temperature, water (deg C)	Fecal coliform (cols./100 mL)	Fecal streptococci (cols./100 mL)	Total residual chlorine (mg/L)	Total cyanide (mg/L)	Total recoverable oil and grease (mg/L)	Gross phenols (µg/L)
385118104485801 - Wahsatch Street at Cross Lane (Site 4)											
06-12-92	1507	3.4	7.6	--	23.0	5,400	43,000	0	<.01	8	12
06-12-92	1540	1.3	6.7	--	24.5	K11,000	K100,000	0	<.01	1	8
07-25-92	1410	5.6	7.1	68	19.0	--	--	0	<.01	10	10
07-25-92	1430	10.7	7.0	91	20.0	--	--	0	<.01	5	11
07-25-92	1610	1.3	7.4	84	20.5	--	--	--	<.01	<1	6
08-16-92	2050	3.6	--	--	--	3,070	24,000	--	--	--	--
11-21-92	1005	.01	--	--	5.5	K600	2,950	0	--	--	--
11-21-92	1230	.12	7.7	--	5.0	K131,000	1,070	0	<.01	6	11
11-21-92	1415	.68	6.9	--	1.0	--	--	0	--	--	--
11-21-92	1700	.09	7.4	--	2.0	4,900	K17,000	0	--	--	--
11-21-92	1730	.05	7.4	--	2.0	--	--	0	--	--	--
384935104501501 - Walmart at Eighth Street (Site 5)											
07-17-92	0255	4.7	6.5	87	15.5	--	--	0	<.01	3	2
07-17-92	0325	.14	7.1	116	15.5	--	--	0	<.01	1	3
07-17-92	0415	.08	7.8	146	15.0	--	--	0	<.01	<1	2
08-03-92	1719	4.72	6.4	93	22.0	286	31,000	0	<10	6	10
08-03-92	1733	.68	6.8	121	22.5	143	33,000	0	<10	7	8
08-03-92	1750	.06	7.2	174	22.0	675	32,000	0	<10	3	10
08-16-92	2030	--	--	--	--	K7,000	12,200	--	--	--	--
12-06-92	1120	.01	7.2	5850	0.0	K225	K667	0	.02	6	26
12-06-93	1240	.04	--	--	.0	--	--	0	--	--	--
12-06-92	1400	.01	7.8	6060	4.0	K275	2,000	--	--	--	--

Table 18. Selected volatile-organic compounds for storm-runoff sites in Colorado Springs

[All constituents in micrograms per liter; <, less than]

Date	Time	Acrolein, total	Acrylonitrile, total	Benzene, total	Bromobenzene, water, whole, total	Bromoform, total	Carbon tetrachloride, total	Chlorobenzene, total	Chlorodibromomethane, total	Chloroethane, total	Chloroform, total
384918104454201 - Sixteenth Hole, Valley-Hi Golf Course (Site 1)											
07-25-92	1424	<20	<20	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
07-25-92	1442	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	1500	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	1553	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-10-92	0910	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-10-92	0955	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-10-92	1050	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	0935	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	1311	<20	<20	.4	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	1540	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2

Date	Methylchloride, total	Cis-1,2-dichloroethene, water, total	Cis-1,3-dichloropropene, total	Di-bromochloropropane, water, whole, total recoverable	Di-bromomethane, water, whole, total recoverable	Di-chlorobromomethane, total	Di-chlorodifluoromethane, total	Ethylbenzene, total	Hexachlorobutadiene, total	Iso-propylbenzene, water, whole, recoverable
07-25-92	<0.2	<0.2	<0.2	<1.0	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
07-25-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
08-10-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
08-10-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
08-10-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	.3	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2

Table 18. Selected volatile organic compounds for storm-runoff sites in Colorado Springs--Continued

Date	Methyl-bromide, total	Methyl-ene chloride, total	N-butyl-benzene, water, recoverable total	N-propyl-benzene, water, recoverable total	Naphthalene, total	P-iso-propyl-toluene, water, whole, recoverable	Sec-butyl-benzene, water, whole, recoverable, total	Styrene, total	Tetra-chloro-ethylene, total	Toluene, total
384918104454201 - Sixteenth Hole, Valley-Hi Golf Course (Site 1)--Continued										
07-25-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-10-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-10-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-10-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	.5	<.2	<.2	<.2	<.2	2.2
										.7

Date	Cis-1,3-dichloro-propene, total	Trans-1,3-dichloro-propene, total	Tri-chloro-ethylene, total	Tri-chloro-fluoro-meth-ane, total	Vinyl-chloride, total	Tert-butyl-benzene, water, whole, recoverable, total	Xylene, water, whole, recoverable, total	1,1-dichloro-propene, water, whole, total	1,1-dichloro-ethane, total	1,1-dichloro-ethylene, total	1,1,1-tri-chloro-ethane, total
07-25-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	0.2	<.2	<.2	<.2	<.2
08-10-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-10-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-10-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	2.3	<.2	<.2	<.2	<.2
							.8	<.2	<.2	<.2	<.2

R0024124

Table 18. Selected volatile organic compounds for storm-runoff sites in Colorado Springs--Continued

Date	1,1,1,2-tetra-chloro-ethane, water, whole, total	1,1,2-tri-chloro-ethane, total	1,1,2,2-tetra-chloro-ethane, total	1,2-di-bromo-ethane, water, whole, total	0-chloro-benzene, 1,2-di-chloro-benzene, total	1,2-di-chloro-ethane, total	1,2-di-chloro-propane, total	0-chloro-toluene, water, whole, total	1,2-trans-di-chloro-ethylene, total	1, 2, 3-tri-chloro-benzene, water, whole recoverable total
384918104454201 - Sixteenth Hole, Valley-Hi Golf Course (Site 1)--Continued										
07-25-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-10-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-10-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-10-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2

Date	1,2,3-tri-chloro-propane, water, whole, total	1,2,4-tri-chloro-benzene, total	1,2,4-tri-methyl-benzene, water, whole, recoverable, total	1,3-di-chloro-benzene, total	1,3-di-chloro-propane, water, whole, total	1,3,5-tri-methyl-benzene, water, whole, recoverable	Para-chloro-toluene, water, whole, total	1,4-di-chloro-benzene, total	2-chloro-ethyl-vinyl-ether, total	2,2-di-chloro-propane, water, whole, total
07-25-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<1.0	<0.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
08-10-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
08-10-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
08-10-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
¹ 11-21-92	<.2	<.2	.6	<.2	<.2	.2	<.2	<.2	<1.0	<.2
¹ 11-21-92	<.2	<.2	.3	<.2	<.2	<.2	<.2	<.2	<1.0	<.2

Table 18. Selected volatile organic compounds for storm-runoff sites in Colorado Springs--Continued

Date	Time	Acrolein, total	Acrylonitrile, total	Benzene, total	Bromobenzene, water, whole, total	Bromoform, total	Carbon tetrachloride, total	Chlorobenzene, total	Chloro dibromomethane, total	Chloroethane, total	Chloroform, total
385347104500601 - Chestnut Street at Douglas Creek (Site 2)											
05-31-92	1330	<20	<20	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
05-31-92	1535	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	1731	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	1830	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	1050	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	1345	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	1635	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2

Date	Methylchloride, total	Cis-1,2-dichloroethene, water, total	Cis-1,3-dichloropropene, total	Di-bromochloropropane, water, whole, total-recoverable	Di-bromomethane, water, whole, recoverable	Di-chlorobromomethane, total	Di-chlorodifluoromethane, total	Ethylbenzene, total	Hexachlorobutadiene, total	Iso-propylbenzene, water, whole, recoverable
05-31-92	<0.2	<0.2	<0.2	<1.0	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
05-31-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2

Date	Methylbromide, total	Methylene chloride, total	N-butylbenzene, water, whole, recoverable	N-propylbenzene, water, whole, recoverable	Naphthalene, total	P-Isopropyltoluene, water, whole, recoverable	Sec-butylbenzene, water, whole, recoverable	Styrene, total	Tetrachloroethylene, total	Toluene, total
05-31-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
05-31-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	.4
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	.4	<.2	<.2	<.2	<.2	.2
¹ 11-21-92	<.2	<.2	<.2	<.2	.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	.2	<.2	<.2	<.2	<.2	.2

R0024126

44 Water Quality of Storm Runoff and Comparison of Procedures for Estimating Storm-Runoff Loads, Volume, Event-Mean Concentrations, and the Mean Load for a Storm for Selected Properties and Constituents for Colorado Springs, Southeastern Colorado, 1992

Table 18. Selected volatile organic compounds for storm-runoff sites in Colorado Springs--Continued

Date	Cis-1,3-dichloropropane, total	Trans-1,3-dichloropropane, total	Trichloroethylene, total	Trichlorofluoromethane, total	Vinylchloride, total	Tert-butylbenzene, water, whole, recoverable	Xylene, water, whole, total recoverable	1,1-dichloropropane, water, whole, total	1,1-dichloroethylene, total	1,1-dichloroethylene, total	1,1,1-trichloroethane, total
385347104500601 - Chestnut Street at Douglas Creek (Site 2)--Continued											
05-31-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
05-31-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	.4	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	.7	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	.3	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	.2	<.2	<.2	<.2	<.2

Date	1,1,1,2-tetrachloroethane, water, whole, total	1,1,2-trichloroethane, total	1,1,2,2-tetrachloroethane, total	1,2-dibromoethane, water, whole, total	1,2-dichlorobenzene, total	1,2-dichloroethane, total	1,2-dichloropropane, total	0-chlorotoluene, water, whole, total	1,2-trans dichloroethylene, total	1,2,3-trichlorobenzene, water, whole, recoverable
05-31-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
05-31-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2

Date	1,2,3-trichloropropane, water, whole, total	1,2,4-trichlorobenzene, total	1,2,4-trimethylbenzene, water, whole, recoverable	1,3-dichlorobenzene, total	1,3-dichloropropane, water, whole, total	1,3,5-trimethylbenzene, water, whole, recoverable	Para-chlorotoluene, water, whole, total	1,4-dichlorobenzene, total	2-chloroethylvinyl ether, total	2,2-dichloropropane, water, whole, total
05-31-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<1.0	<0.2
05-31-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
¹ 11-21-92	<.2	<.2	.4	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
¹ 11-21-92	<.2	<.2	.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2

Table 18. Selected volatile organic compounds for storm-runoff sites in Colorado Springs—Continued

Date	Time	Acrolein, total	Acrylonitrile, total	Benzene, total	Bromobenzene, water, whole, total	Bromoform, total	Carbon tetrachloride, total	Chlorobenzene, total	Chlorodibromomethane, total	Chloroethane, total	Chloroform, total
385240104493601 - Beacon Street at Buchanan Street (Site 3)											
06-05-92	1300	<20	<20	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
06-05-92	1310	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	1740	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	1800	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	1835	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-11-92	0805	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-11-92	1020	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-11-92	1245	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2

Date	Methylchloride, total	Cis-1,2-dichloroethene, water, total	Cis-1,3-dichloropropene, total	Di-bromochloropropane, water, whole, total-recoverable	Di-bromomethane, water, whole, recoverable	Di-chlorobromomethane, total	Di-chlorodifluoromethane, total	Ethylbenzene, total	Hexachlorobutadiene, total	Iso-propylbenzene, water, whole recoverable
06-05-92	<0.2	<0.2	<0.2	<1.0	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
06-05-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-11-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-11-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-11-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2

Date	Methylbromide, total	Methylenechloride, total	N-butylbenzene, water, whole, recoverable	N-propylbenzene, water, whole, recoverable	Naphthalene, total	P-isopropyltoluene, water, whole, recoverable	Sec-butylbenzene, water, whole, recoverable	Styrene, total	Tetrachloroethylene, total	Toluene, total
06-05-92	<0.2	<0.2	<0.2	<0.2	1.0	<0.2	<0.2	<0.2	<0.2	<0.2
06-05-92	<.2	<.2	<.2	<.2	1.1	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<.2	.3	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<.2	.4	<.2	<.2	<.2	<.2	<.2
¹ 11-11-92	<.2	<.2	<.2	<.2	.6	<.2	<.2	<.2	<.2	<.2
¹ 11-11-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	.2
¹ 11-11-92	<.2	<.2	<.2	<.2	.3	<.2	<.2	<.2	<.2	<.2

R0024128

46 Water Quality of Storm Runoff and Comparison of Procedures for Estimating Storm-Runoff Loads, Volume, Event-Mean Concentrations, and the Mean Load for a Storm for Selected Properties and Constituents for Colorado Springs, Southeastern Colorado, 1992

Table 18. Selected volatile organic compounds for storm-runoff sites in Colorado Springs—Continued

Date	Cis-1,3-dichloropropane, total	Trans-1,3-dichloropropene, total	Trichloroethylene, total	Trichlorofluoromethane, total	Vinylchloride, total	Tert-butylbenzene, water, whole, recoverable	Xylene, water, whole, total, recoverable	1,1-dichloropropane, water, whole, total	1,1-dichloroethane, total	1,1-dichloroethylene, total	1,1,1-trichloroethane, total
385240104493601 - Beacon Street at Buchanan Street (Site 3)—Continued											
06-05-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
06-05-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-11-92	<.2	<.2	<.2	<.2	<.2	<.2	.3	<.2	<.2	<.2	<.2
¹ 11-11-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-11-92	<.2	<.2	<.2	<.2	<.2	<.2	.5	<.2	<.2	<.2	<.2
Date	1,1,1,2-tetrachloroethane, water, whole, total	1,1,2-tetrachloroethane, total	1,1,2,2-tetrachloroethane, total	1,2 dibromoethane, water, whole, total	1,2 dichlorobenzene, total	1,2 dichloroethane, total	1,2 dichloropropane, total	O-chlorotoluene, water, whole, total	1,2-trans-dichloroethylene, total	1,2,3-trichlorobenzene, water, whole, recoverable	
06-05-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	
06-05-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	
¹ 11-11-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	
¹ 11-11-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	
¹ 11-11-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	
Date	1,2,3-trichloropropane, water, whole, total	1,2,4-trichlorobenzene, total	1,2,4-trimethylbenzene, water, whole, recoverable	1,3-dichlorobenzene, total	1,3-dichloropropane, water, whole, total	1,3,5-trimethylbenzene, water, whole, recoverable	Parachlorotoluene, water, whole, total	1,4-dichlorobenzene, total	2 chloroethylvinylether, total	2,2-dichloropropane, water, whole, total	
06-05-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<1.0	<0.2	
06-05-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2	
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2	
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2	
06-19-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2	
¹ 11-11-92	<.2	<.2	.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2	
¹ 11-11-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2	
¹ 11-11-92	<.2	<.2	<.2	<.2	<.2	.2	<.2	<.2	<1.0	<.2	

Table 18. Selected volatile organic compounds for storm-runoff sites in Colorado Springs--Continued

Date	Time	Acrolein, total	Acrylonitrile, total	Benzene, total	Bromo-benzene, water, whole, total	Bromoform, total	Carbon-tetrachloride, total	Chlorobenzene, total	Chloro-dibromomethane, total	Chloroethane, total	Chloroform, total
385118104485801 - Wahsatch Street at Cross Lane (Site 4)											
06-12-92	1507	<20	<20	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
06-12-92	1540	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	1410	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	1430	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	1610	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	1230	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	1415	<20	<20	.3	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	1730	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2

Date	Methylchloride, total	Cis-1,2-dichloroethene, water, total	Cis-1,3-dichloropropane, total	Di-bromochloropropane, water, whole, total recoverable	Di-bromomethane, water, whole, recoverable	Di-chlorobromomethane, total	Di-chloro-difluoromethane, total	Ethylbenzene, total	Hexachlorobutadiene, total	Iso-propylbenzene, water, whole, recoverable
06-12-92	<0.2	<0.2	<0.2	<1.0	<0.2	<0.2	<0.2	0.2	<0.2	<0.2
06-12-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2

Date	Methylbromide, total	Methylchloride, total	N-butylbenzene, water, whole, recoverable	N-propylbenzene, water, whole, recoverable	Naphthalene, total	P-isopropyltoluene, water, whole, recoverable	Sec-butylbenzene, water, whole, recoverable	Styrene, total	Tetra-chloroethylene, total	Toluene, total
06-12-92	<0.2	<0.2	<0.2	<0.2	0.3	0.2	<0.2	<0.2	<0.2	0.5
06-12-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	.6
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	.3
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	1.2
¹ 11-21-92	<.2	<.2	<.2	<.2	.2	<.2	<.2	<.2	<.2	1.2
¹ 11-21-92	<.2	<.2	<.2	<.2	.3	<.2	<.2	<.2	<.2	1.1
¹ 11-21-92	<.2	<.2	<.2	<.2	.4	<.2	<.2	<.2	<.2	.3

R0024130

Table 18. Selected volatile organic compounds for storm-runoff sites in Colorado Springs--Continued

Date	Cis-1,3-dichloropropene, total	Trans-1,3-dichloropropene, total	Trichloroethylene, total	Trichlorofluoromethane, total	Vinylchloride, total	Tert-butylbenzene, water, whole, recoverable	Xylene, water, whole, total recoverable	1,1-dichloropropene, water, whole, total	1,1-dichloroethane, total	1,1-dichloroethylene, total	1,1,1-trichloroethane, total
385118104485801 - Wahsatch Street at Cross Lane (Site 4)--Continued											
06-12-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	1.5	<0.2	<0.2	<0.2	<0.2
06-12-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	.3	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	.7	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	.3	<.2	<.2	<.2	<.2

1,1,1,2-tetrachloroethane, water, whole, total	1,1,2-trichloroethane, total	1,1,2,2-tetrachloroethane, total	1,2-dibromoethane, water, whole, total	1,2-dichlorobenzene, total	1,2-dichlorobenzene, total	1,2-dichloroethane, total	1,2-dichloropropane, total	O-chlorotoluene, water, whole, total	1,2-trans-dichloroethylene, total	1,2,3-trichlorobenzene, water, whole, recoverable
06-12-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
06-12-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2

Date	1,2,3-trichloropropane, water, whole, total	1,2,4-trichlorobenzene, total	1,2,4-trimethylbenzene, water, whole, recoverable	1,3-dichlorobenzene, total	1,3-dichloropropane, water, whole, total	1,3,5-trimethylbenzene, water, whole, recoverable	Para-chlorotoluene, water, whole, total	1,4-dichlorobenzene, total	2-chloroethylvinyl ether, total	2,2-dichloropropane, water, whole, total
06-12-92	<0.2	<0.2	0.9	<0.2	<0.2	0.3	<0.2	<0.2	<1.0	<0.2
06-12-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
07-25-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
¹ 11-21-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
¹ 11-21-92	<.2	<.2	.3	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
¹ 11-21-92	<.2	<.2	.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2

Table 18. Selected volatile organic compounds for storm-runoff sites in Colorado Springs—Continued

Date	Time	Acrolein, total	Acrylonitrile, total	Benzene, total	Bromobenzene, water, whole, total	Bromoform, total	Carbon tetrachloride, total	Chlorobenzene, total	Chloro dibromomethane, total	Chloroethane, total	Chloroform total
384935104501501 - Walmart at Eighth Street (Site 5)											
07-17-92	0255	<20	<20	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	
07-17-92	0325	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	2.1
07-17-92	0415	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	.9
08-03-92	1719	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	2.0
08-03-92	1733	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-03-92	1750	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 12-06-92	1120	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 12-06-92	1240	<20	<20	.3	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 12-06-92	1400	<20	<20	<.2	<.2	<.2	<.2	<.2	<.2	<.2	.7
											1.1

Date	Methylchloride, total	Cis-1,2-dichloroethene, water, total	Cis-1,3-dichloropropene, total	Di-bromochloropropane, water, whole, total recoverable	Di-bromomethane, water, whole, recoverable	Di-chlorobromomethane, total	Di-chlorodifluoromethane, total	Ethylbenzene, total	Hexachlorobutadiene, total	Iso-propylbenzene, water, whole, recoverable
07-17-92	<0.2	0.2	<0.2	<1.0	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
07-17-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
07-17-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
08-03-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
08-03-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
08-03-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
¹ 12-06-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
¹ 12-06-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
¹ 12-06-92	<.2	<.2	<.2	<1.0	<.2	<.2	<.2	<.2	<.2	<.2
								.3	<.2	<.2

Date	Methylbromide, total	Methylene chloride, total	N-butylbenzene, water, whole, recoverable	N-propylbenzene, water, whole, recoverable	Naphthalene, total	P-iso-propyltoluene, water, whole, recoverable	Sec-butylbenzene, water, whole, recoverable	Styrene, total	Tetrachloroethylene, total	Toluene, total
07-17-92	<0.2	<0.2	<0.2	<0.2	0.2	<0.2	<0.2	<0.2	<0.2	0.2
07-17-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-17-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-03-92	<.2	<.2	<.2	<.2	.2	<.2	<.2	<.2	<.2	<.2
08-03-92	<.2	<.2	<.2	<.2	.2	<.2	<.2	<.2	<.2	<.2
08-03-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	.3
¹ 12-06-92	<.2	<.2	<.2	<.2	4.7	<.2	<.2	<.2	<.2	<.2
¹ 12-06-92	<.2	<.2	.4	.3	1.6	<.2	<.2	<.2	<.2	.2
¹ 12-06-92	<.2	<.2	<.2	<.2	1.1	<.2	<.2	<.2	<.2	1.7
									<.2	.2

R0024132

Table 18. Selected volatile organic compounds for storm-runoff sites in Colorado Springs—Continued

Date	Cis-1,3-dichloropropene, total	Trans-1,3-dichloropropene, total	Tri-chloroethylene, total	Tri-chloro-fluoromethane, total	Vinyl-chloride, total	Tert-butylbenzene, water, whole, recoverable	Xylene, water, whole, total recoverable	1,1-dichloropropene, water, whole, total	1,1-dichloroethane, total	1,2-dichloroethylene, total	1,1,1-trichloroethane, total
384935104501501 - Walmart at Eighth Street (Site 5)—Continued											
07-17-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
07-17-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	.7
07-17-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	.3
08-03-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-03-92	<.2	<.2	<.2	<.2	<.2	<.2	.2	<.2	<.2	<.2	<.2
08-03-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 12-06-92	<.2	<.2	<.2	<.2	<.2	<.2	.4	<.2	<.2	<.2	<.2
¹ 12-06-92	<.2	<.2	<.2	<.2	<.2	<.2	4.4	<.2	<.2	<.2	<.2
¹ 12-06-92	<.2	<.2	<.2	<.2	<.2	<.2	.4	<.2	<.2	<.2	<.2

Date	1,1,1,2-tetrachloroethane, water, whole, total	1,1,2-trichloroethane, total	1,1,2,2-tetrachloroethane, total	1,2-dibromoethane, water, whole, total	1,2-dichlorobenzene, total	1,2-dichloroethane, total	1,2-dichloropropane, total	o-chlorotoluene, water, whole, total	1,2-trans-dichloroethylene, total	1,2,3-trichlorobenzene, water, whole, recoverable
07-17-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
07-17-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
07-17-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-03-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-03-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
08-03-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 12-06-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 12-06-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2
¹ 12-06-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2

Date	1,2,3-trichloropropane, water, whole, total	1,2,4-trichlorobenzene, total	1,2,4-trimethylbenzene, water, whole, recoverable	1,3-dichlorobenzene, total	1,3-dichloropropane, water, whole, total	1,3,5-trimethylbenzene, water, whole, recoverable	Para-chlorotoluene, water, whole, total	1,4-dichlorobenzene, total	2-chloroethylvinylether, total	2,2-dichloropropane, water, whole, total
07-17-92	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<1.0	<0.2
07-17-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
07-17-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
08-03-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
08-03-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
08-03-92	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
¹ 12-06-92	<.2	<.2	.3	<.2	<.2	<.2	<.2	<.2	<1.0	<.2
¹ 12-06-92	<.2	<.2	2.8	<.2	<.2	.9	<.2	<.2	<1.0	<.2
¹ 12-06-92	<.2	<.2	.3	<.2	<.2	<.2	<.2	<.2	<1.0	<.2

¹Snowmelt-runoff sample.

Table 19. Chemical oxygen demand, 5-day biochemical oxygen demand, specific conductance, alkalinity, dissolved solids, suspended solids, major ions, nutrients, and total-recoverable metals for storm-runoff sites in Colorado Springs

[mg/L, milligrams per liter; μ S/cm, microsiemens per centimeter at 25 degrees Celsius; μ g/L, micrograms per liter; lab, laboratory; dashes indicate no data; <, less than]

Date	Time	Oxygen demand, chemical, high level (mg/L)	Oxygen demand, biochemical, 5-day (mg/L)	Specific conductance (μ S/cm)	Alkalinity, lab (mg/L)	Solids, dissolved (mg/L)	Solids, suspended (mg/L)	Calcium, dissolved (mg/L)	Magnesium, dissolved (mg/L)	Potassium, dissolved (mg/L)	Sodium, dissolved (mg/L)
384918104454201 - Sixteenth Hole, Valley-Hi Golf Course (Site 1)											
06-03-92	1410	100	--	114	--	63	321	--	--	--	--
06-23-92	2030	170	53	147	--	107	121	--	--	--	--
06-26-92	1453	140	24	107	--	71	372	--	--	--	--
07-02-92	1459	210	47	--	--	202	136	--	--	--	--
07-25-92	1423	310	29	105	37	75	242	9.9	2.4	1.2	5.2
08-10-92	0840	180	19	133	53	93	524	13	2.9	1.9	5.4
11-21-92	0930	270	--	256	52	143	274	14	3.6	1.8	25

Date	Chloride, dissolved (mg/L)	Sulfate, dissolved (mg/L)	Nitrogen, ammonia, total, as nitrogen (mg/L)	Nitrogen, ammonia plus organic, total, as nitrogen (mg/L)	Nitrogen nitrite, total, as nitrogen (mg/L)	Nitrogen, nitrate plus nitrite, total, as nitrogen (mg/L)	Phosphorus, total, as phosphorus (mg/L)	Phosphorus, dissolved, as phosphorus (mg/L)	Antimony, total-recoverable (μ g/L)	Arsenic, total (μ g/L)	Beryllium, total-recoverable (μ g/L)
06-03-92	--	--	0.98	1.7	0.03	0.80	0.14	0.08	--	--	--
06-23-92	--	--	.88	2.2	.08	1.00	.29	.15	--	--	--
06-26-92	--	--	.28	1.6	.03	.52	.33	.08	--	--	--
07-02-92	--	--	.73	2.4	.11	.97	.36	.28	--	--	--
07-25-92	1.0	21	.28	1.5	.02	.40	.29	.12	<10	2	<10
08-10-92	.1	2.2	.67	2.1	.04	.98	.45	.14	<10	3	<10
11-21-92	32	27	.41	.9	.07	.46	.12	.08	<10	1	<10

Date	Cadmium, total-recoverable (μ g/L)	Chromium, total-recoverable (μ g/L)	Copper, total-recoverable (μ g/L)	Lead, total-recoverable (μ g/L)	Mercury, total-recoverable (μ g/L)	Nickel, total-recoverable (μ g/L)	Selenium, total (μ g/L)	Silver, total-recoverable (μ g/L)	Thallium, total (μ g/L)	Zinc, total-recoverable (μ g/L)	Carbon, organic, total (mg/L)
06-03-92	<1	--	10	140	--	--	--	--	--	180	--
06-23-92	<1	--	9	23	--	--	--	--	--	140	--
06-26-92	<1	--	15	60	--	--	--	--	--	190	--
07-02-92	<1	--	13	30	--	--	--	--	--	190	--
07-25-92	<1	43	12	170	0.1	5	<1	<1	<5	180	32
08-10-92	1	20	12	77	<1	11	1	<1	<5	300	36
11-21-92	1	14	17	64	<1	9	<2	<1	<5	250	29

R0024134

Table 19. Chemical oxygen demand, 5-day biochemical oxygen demand, specific conductance, alkalinity, dissolved solids, suspended solids, major ions, nutrients, and total-recoverable metals for storm-runoff sites in Colorado Springs--Continued

Date	Time	Oxygen demand, chemical (high level) (mg/L)	Oxygen demand, biochemical, 5 day (mg/l)	Specific conductance (µS/cm)	Alkalinity, lab (mg/L)	Solids, dissolved (mg/L)	Solids, suspended (mg/L)	Calcium, dissolved (mg/L)	Magnesium, dissolved (mg/L)	Potassium, dissolved (mg/L)	Sodium, dissolved (mg/L)
38534710450061 - Chestnut Street at Douglas Creek (Site 2)											
05-31-92	1255	230	14	385	80	68	396	--	--	--	--
06-05-92	1230	420	39	63	--	63	1280	--	--	--	--
06-19-92	1724	280	80	--	65	121	764	14	1.1	3.1	3.3
06-27-92	1705	310	--	70	--	55	198	--	--	--	--
07-29-92	1350	150	34	85	--	34	832	--	--	--	--
08-03-92	2050	100	28	77	--	60	234	--	--	--	--
11-21-92	0910	190	22	497	57	256	464	13	0.95	1.8	77

Date	Chloride, dissolved (mg/L)	Sulfate, dissolved (mg/L)	Nitrogen, ammonia, total, as nitrogen (mg/L)	Nitrogen, ammonia plus organic, total, as nitrogen (mg/L)	Nitrogen nitrite, total, as nitrogen (mg/L)	Nitrogen, nitrate plus nitrite, total, as nitrogen (mg/L)	Phosphorus, total, as phosphorus (mg/L)	Phosphorus, dissolved, as phosphorus (mg/L)	Antimony, total-recoverable (mg/L)	Arsenic, total (mg/L)	Beryllium, total recoverable (mg/L)
05-31-92	2.9	6.4	0.43	1.0	0.04	0.93	0.12	--	<10	5	<10
06-05-92	--	--	.24	2.4	.03	.61	.72	0.12	--	--	--
06-19-92	2.1	6.6	1.10	2.3	.03	1.10	.25	.21	<10	8	<10
06-27-92	--	--	.49	1.3	.06	1.10	.20	.07	--	--	--
07-29-92	--	--	.23	.90	.03	.57	.16	.10	--	--	--
08-03-92	--	--	.56	1.1	.04	.87	.09	.08	--	--	--
11-21-92	130	4.2	.36	1.1	.08	.45	.17	.06	<10	<1	<10

Date	Cadmium, total-recoverable (µg/L)	Chromium, total-recoverable (µg/L)	Copper, total-recoverable (µg/L)	Lead, total-recoverable (µg/L)	Mercury, total-recoverable (µg/L)	Nickel, total-recoverable (µg/L)	Selenium, total (µg/L)	Silver, total-recoverable (µg/L)	Thallium, total (µg/L)	Zinc, total-recoverable (µg/L)	Carbon, organic, total (mg/L)
05-31-92	1	27	19	150	<0.1	8	<2	<1	<10	700	48
06-05-92	2	--	99	350	--	--	--	--	--	1400	--
06-19-92	2	51	74	290	.1	22	<2	<1	<10	1400	52
06-27-92	<1	--	12	47	--	--	--	--	--	290	--
07-29-92	2	--	37	290	--	--	--	--	--	550	--
08-03-92	<1	--	18	86	--	--	--	--	--	570	--
11-21-92	1	28	17	110	<.1	8	<2	<1	<10	230	32

Table 19. Chemical oxygen demand, 5-day biochemical oxygen demand, specific conductance, alkalinity, dissolved solids, suspended solids, major ions, nutrients, and total-recoverable metals for storm-runoff sites in Colorado Springs--Continued

Date	Time	Oxygen demand, chemical (high level) (mg/L)	Oxygen demand, biochemical, 5 day (mg/l)	Specific conductance (µS/cm)	Alkalinity, lab (mg/L)	Solids, dissolved (mg/L)	Solids, suspended (mg/L)	Calcium, dissolved (mg/L)	Magnesium, dissolved (mg/L)	Potassium, dissolved (mg/L)	Sodium, dissolved (mg/L)
385240104493601 - Beacon Street at Buchanan Street (Site 3)											
06-05-92	1240	170	33	92	40	74	330	13	0.86	2.2	2.9
06-19-92	1735	300	66	101	44	103	340	13	.91	3.1	2.9
06-23-92	2044	200	42	110	--	87	101	--	--	--	--
06-27-92	1712	160	51	99	--	77	162	--	--	--	--
08-02-92	1845	270	74	133	--	168	220	--	--	--	--
08-03-92	2055	120	26	59	--	54	272	--	--	--	--
11-11-92	0759	260	62	238	41	137	116	13	1.0	2.6	25

Date	Chloride, dissolved (mg/L)	Sulfate, dissolved (mg/L)	Nitrogen, ammonia, total, as nitrogen (mg/L)	Nitrogen, ammonia plus organic, total, as nitrogen (mg/L)	Nitrogen nitrite, total, as nitrogen (mg/L)	Nitrogen, nitrate plus nitrite, total, as nitrogen (mg/L)	Phosphorus, total, as phosphorus (mg/L)	Phosphorus, dissolved, as phosphorus (mg/L)	Antimony, total-recoverable (mg/L)	Arsenic, total (mg/L)	Beryllium, total recoverable (mg/L)
06-05-92	3.4	8.5	0.76	2.9	.04	0.84	0.30	0.13	<20	1	<10
06-19-92	2.9	6.4	1.30	3.5	.04	1.00	.58	.51	<10	<1	<10
06-23-92	--	--	.99	2.7	.07	1.10	.27	.17	--	--	--
06-27-92	--	--	1.50	3.7	.06	1.00	.46	.20	--	--	--
08-02-92	--	--	1.50	4.0	.04	1.40	.44	.30	--	--	--
08-03-92	--	--	.48	1.6	.03	.66	.16	.10	--	--	--
11-11-92	42	5.3	.57	1.8	.07	.68	.33	.18	<10	1	<10

Date	Cadmium, total-recoverable (µg/L)	Chromium, total-recoverable (µg/L)	Copper, total-recoverable (µg/L)	Lead, total-recoverable (µg/L)	Mercury, total-recoverable (µg/L)	Nickel, total-recoverable (µg/L)	Selenium, total (µg/L)	Silver, total-recoverable (µg/L)	Thallium, total (µg/L)	Zinc, total-recoverable (µg/L)	Carbon, organic, total (mg/L)
06-05-92	2	18	17	85	<0.1	12	<2	<1	<5	280	68
06-19-92	2	27	24	83	<.1	17	<2	1	<10	350	83
06-23-92	<1	--	13	23	--	--	--	--	--	160	--
06-27-92	<1	--	13	42	--	--	--	--	--	150	--
08-02-92	2	--	22	53	--	--	--	--	--	400	--
08-03-92	2	--	12	97	--	--	--	--	--	340	--
11-11-92	1	8	14	55	<.1	6	<2	<1	<10	200	52

54 Water Quality of Storm Runoff and Comparison of Procedures for Estimating Storm-Runoff Loads, Volume, Event-Mean Concentrations, and the Mean Load for a Storm for Selected Properties and Constituents for Colorado Springs, Southeastern Colorado, 1992

Table 19. Chemical oxygen demand, 5-day biochemical oxygen demand, specific conductance, alkalinity, dissolved solids, suspended solids, major ions, nutrients, and total-recoverable metals for storm-runoff sites in Colorado Springs--Continued

Date	Time	Oxygen demand, chemical (high level) (mg/L)	Oxygen demand, biochemical, 5 day (mg/l)	Specific conductance (µS/cm)	Alkalinity, lab (mg/L)	Solids, dissolved (mg/L)	Solids, suspended (mg/L)	Calcium, dissolved (mg/L)	Magnesium, dissolved (mg/L)	Potassium, dissolved (mg/L)	Sodium, dissolved (mg/L)
385118104485801 - Wahsatch Street at Cross Lane (Site 4)											
05-26-92	1945	250	--	--	--	--	116	--	--	--	--
06-03-92	1405	230	--	84	--	74	754	--	--	--	--
06-12-92	1504	500	86	--	84	132	660	20	1.3	5.6	3.3
06-26-92	1619	360	72	84	--	60	848	--	--	--	--
07-25-92	1408	330	140	82	54	106	266	11	1.1	5.3	2.2
07-29-92	1401	340	92	110	--	93	512	--	--	--	--
11-21-92	1010	190	29	1740	40	908	148	20	1.6	3.7	300

Date	Chloride, dissolved (mg/L)	Sulfate, dissolved (mg/L)	Nitrogen, ammonia, total, as nitrogen (mg/L)	Nitrogen, ammonia plus organic, total, as nitrogen (mg/L)	Nitrogen nitrite, total, as nitrogen (mg/L)	Nitrogen, nitrate plus nitrite, total, as nitrogen (mg/L)	Phosphorus, total, as phosphorus (mg/L)	Phosphorus, dissolved, as phosphorus (mg/L)	Antimony, total-recoverable (mg/L)	Arsenic, total (mg/L)	Beryllium, total recoverable (mg/L)
05-26-92	--	--	1.00	5.0	0.05	0.38	1.20	--	--	--	--
06-03-92	--	--	.53	4.9	.06	.88	.47	0.26	--	--	--
06-12-92	3.0	6.5	.72	5.3	.07	.83	1.10	.38	<10	5	<10
06-26-92	--	--	.27	3.8	.04	.47	.95	.28	--	--	--
07-25-92	2.1	5.6	.16	3.1	.03	.37	.72	.27	<10	3	<10
07-29-92	--	--	.35	2.8	.04	.83	.60	--	--	--	--
11-21-92	470	4.5	.39	1.6	.08	.36	.22	.11	<10	<1	<10

Date	Cadmium, total-recoverable (µg/L)	Chromium, total-recoverable (µg/L)	Copper, total-recoverable (µg/L)	Lead, total-recoverable (µg/L)	Mercury, total-recoverable (µg/L)	Nickel, total-recoverable (µg/L)	Selenium, total (µg/L)	Silver, total-recoverable (µg/L)	Thallium, total (µg/L)	Zinc, total-recoverable (µg/L)	Carbon, organic, total (mg/L)
05-26-92	<1	--	9	41	--	--	--	--	--	110	--
06-03-92	1	--	12	120	--	--	--	--	--	220	--
06-12-92	2	18	22	130	<0.1	11	<2	<1	<10	300	100
06-26-92	1	--	15	130	--	--	--	--	--	310	--
07-25-92	<1	8	11	57	<.1	4	<1	<1	<5	140	100
07-29-92	1	--	17	110	--	--	--	--	--	240	--
11-21-92	<1	7	8	32	<.1	4	<2	<1	<10	110	18

Table 19. Chemical oxygen demand, 5-day biochemical oxygen demand, specific conductance, alkalinity, dissolved solids, suspended solids, major ions, nutrients, and total-recoverable metals for storm-runoff sites in Colorado Springs--Continued

Date	Time	Oxygen demand, chemical (high level) (mg/L)	Oxygen demand, biochemical, 5 day (mg/l)	Specific conductance (µS/cm)	Alkalinity, lab (mg/L)	Solids, dissolved (mg/L)	Solids, suspended (mg/L)	Calcium, dissolved (mg/L)	Magnesium, dissolved (mg/L)	Potassium, dissolved (mg/L)	Sodium, dissolved (mg/L)
384935104501501 - Walmart at Eighth Street (Site 5)											
06-10-92	1625	160	67	--	--	93	1400	--	--	--	--
06-26-92	1712	310	260	132	--	127	388	--	--	--	--
07-17-92	0250	200	35	100	51	70	662	10	1.1	2.2	6.2
08-03-92	1717	390	38	122	64	138	872	14	1.4	2.7	4.9
08-04-92	1729	260	71	142	--	--	516	--	--	--	--
08-05-92	1700	200	29	91	--	67	826	--	--	--	--
12-06-92	1115	830	160	7010	219	4240	1260	59	16	3.2	1300

Date	Chloride, dissolved (mg/L)	Sulfate, dissolved (mg/L)	Nitrogen, ammonia, total, as nitrogen (mg/L)	Nitrogen, ammonia plus organic, total, as nitrogen (mg/L)	Nitrogen nitrite, total, as nitrogen (mg/L)	Nitrogen, nitrate plus nitrite, total, as nitrogen (mg/L)	Phosphorus, total, as phosphorus (mg/L)	Phosphorus, dissolved, as phosphorus (mg/L)	Antimony, total-recoverable (mg/L)	Arsenic, total (mg/L)	Beryllium, total recoverable (mg/L)
06-10-92	--	--	0.58	1.2	0.06	1.0	0.21	0.14	--	--	--
06-26-92	--	--	.52	1.8	.10	.71	.41	.34	--	--	--
07-17-92	6.9	9.2	.41	2.2	.03	.69	.59	.19	<10	5	<10
08-03-92	6.7	15	1.00	2.3	.04	1.20	.35	.24	<10	7	<10
08-04-92	--	--	1.70	4.2	.08	1.80	.63	--	--	--	--
08-05-92	--	--	1.00	3.8	.04	1.20	1.00	.15	--	--	--
12-06-92	2000	190	3.90	7.4	.31	1.60	1.00	.09	<10	13	<10

Date	Cadmium, total-recoverable (µg/L)	Chromium, total-recoverable (µg/L)	Copper, total-recoverable (µg/L)	Lead, total-recoverable (µg/L)	Mercury, total-recoverable (µg/L)	Nickel, total-recoverable (µg/L)	Selenium, total (µg/L)	Silver, total-recoverable (µg/L)	Thallium, total (µg/L)	Zinc, total-recoverable (µg/L)	Carbon, organic, total (mg/L)
06-10-92	3	--	18	340	--	--	--	--	--	500	--
06-26-92	1	--	18	85	--	--	--	--	--	210	--
07-17-92	2	35	15	200	0.1	16	<2	<1	<5	330	69
08-03-92	21	52	26	300	.2	21	2	<1	<5	630	77
08-04-92	1	--	17	120	--	--	--	--	--	340	--
08-05-92	2	--	30	180	--	--	--	--	--	330	--
12-06-92	4	88	70	350	.1	33	<2	<1	<5	730	240

Table 20. Selected acid-base/neutral organic compounds for storm-runoff site Colorado Springs

[All concentrations in micrograms per liter; <, less than]

Date	Time	Ace-naph-thylene, total	Ace-naph-thene, total	Anthra-cene, total	Benzi-dine, total	Benzo-a-an-thra-cenel, 2-benz-anthra-cene, total	Benzo-a-py-rene, total	Ben-zogh-l-peryl-enel, 12-benzo-peryl-ene, total	Benzo-b-fluor-anthene, total	Benzo-k-fluor-anthene, total	Bis (2-ethyl-hexyl) phthal-ate, total	Bis 2-chloro-ethyl-ether, total	Bis (2-chloro-ethoxy) meth-ane, total
384918104454201 - Sixteenth Hole, Valley-III Golf Course (Site 1)													
07-25-92	1423	<5.0	<5.0	<5.0	<40.0	<10.0	<10.0	<10.0	10.0	<10.0	14.0	<5.0	<5.0
08-10-92	0840	<5.0	<5.0	<5.0	<40.0	<10.0	<10.0	11.0	11.0	<10.0	11.0	<5.0	<5.0
11-21-91	0930	<5.0	<5.0	<5.0	<40.0	<10.0	<10.0	<10.0	<10.0	<10.0	30.0	<5.0	<5.0

Date	Bis (2-chloro-iso-propyl) ether, total	Chrysene, total	Di-ethyl-phthalate, total	Di-methyl-phthalate, total	Di-n-octyl-phthalate, total	Di-n-butyl-phthalate, total	Fluor-anthene, total	Fluorene, total	Hexa-chloro-benzene, total	Hexa-chloro-buta-diene, total
07-25-92	<5.0	11.0	<5.0	<5.0	<10.0	5.0	21.0	<5.0	<5.0	<5.0
08-10-92	<5.0	12.0	<5.0	<5.0	<10.0	6.0	25.0	<5.0	<5.0	<5.0
11-21-92	<5.0	<10.0	<5.0	<5.0	<10.0	15.0	11.0	<5.0	<5.0	<5.0

Date	Hexa-chloro-ethane, total	Indeno (1,2,3-cd) pyrene, total	Iso-phorone, total	Naph-thalene, total	N-butyl benzyl phthal-ate, total	Nitro-benzene, total	N-nitro-sodi-n-propyl-amine, total	N-nitro-sodi-methyl-amine, total	N-nitro-sodi-phenyl-amine, total	Para-chloro-meta-cresol, total	Penta-chloro-phenol, total	Phenan-threne, total
07-25-92	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	14.0
08-10-92	<5.0	13.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	14.0
11-21-92	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	7.0

Table 20. Selected acid-base/neutral organic compounds for storm-runoff sites in Colorado Springs--Continued

Date	Phenol, total	Pyrene, total	Benzene-o-chloro-water, unfiltered, recoverable	1,2-diphenylhydrazine, water, total-recoverable	Benzene, 1,2,4-trichlorobenzene, unfiltered, recoverable	Benzene, 1,2-dichloro-water, unfiltered, recoverable	Benzene, 1,3-dichloro-water, unfiltered, recoverable	Benzene, 1,4-dichloro-water, unfiltered, recoverable	1,2,5,6-dibenzanthracene, total	2-chloronaphthalene, total	2-chlorophenol, total	2-nitrophenol, total	
384918104454201 - Sixteenth Hole, Valley-Hi Golf Course (Site 1)--Continued													
07-25-92	--	17.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0	
08-10-92	--	19.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0	
11-21-92	<5.0	8.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0	
Date	2,4-dichlorophenol, total	2,4-dimethylphenol, total	2,4-dinitrophenol, total	2,4,6-trichlorophenol, total	2,4-dinitrotoluene, total	2,6-dinitrotoluene, total	3,3-dichlorobenzidine, total	4-bromophenylphenol, ether, total	4-chlorophenylphenol, ether, total	4-nitrophenol, total	4,6-dinitro-ortho-cresol, total		
07-25-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0		
08-10-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0		
11-21-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0		
Date	Time	Acenaphthylene, total	Acenaphthene, total	Anthracene, total	Benzo-dine, total	Benzo-a-anthracene, 2-benzanthracene, total	Benzo-a-pyrene, total	Benzo-g-h-perylene, 12-benzoperylene, total	Benzo-b-fluoranthene, total	Benzo-k-fluoranthene, total	Bis (2-ethylhexyl) phthalate, total	Bis 2-chloro-ethyl-ether, total	Bis (2-chloroethoxy) methane, total
385347104500601 - Chestnut Street at Douglas Creek (Site 2)													
05-31-92	1255	<5.0	<5.0	6.0	<40.0	18.0	22.0	<10.0	31.0	25.0	9.0	<5.0	<5.0
06-19-92	1724	<5.0	<5.0	10.0	<40.0	37.0	46.0	14.0	73.0	90.0	24.0	<5.0	<5.0
11-21-92	0910	<5.0	<5.0	<5.0	<40.0	<10.0	<10.0	<10.0	<10.0	<10.0	24.0	<5.0	<5.0

58 Water Quality of Storm Runoff and Comparison of Procedures for Estimating Storm-Runoff Loads, Volume, Event-Mean Concentrations, and the Mean Load for a Storm for Selected Properties and Constituents for Colorado Springs, Southeastern Colorado, 1992

R0024140

Table 10. Selected acid-base/neutral organic compounds for storm-runoff sites in Colorado Springs--Continued

Date	Bis (2-chloro-iso-propyl) ether, total	Chrysene, total	Di-ethyl-phthalate, total	Di-methyl-phthalate, total	Di-n-octyl-phthalate, total	Di-n-butyl-phthalate, total	Fluor-anthene, total	Fluorene, total	Hexa-chloro-benzene, total	Hexa-chloro-buta-diene, total
385347104500601 - Chestnut Street at Douglas Creek (Site 2)--Continued										
05-31-92	<5.0	34.0	<5.0	<5.0	<10.0	<5.0	69.0	<5.0	<5.0	<5.0
06-19-92	<5.0	61.0	<5.0	<5.0	<10.0	<5.0	120	<5.0	<5.0	<5.0
11-21-92	<5.0	<10.0	<5.0	<5.0	<10.0	<5.0	8.0	<5.0	<5.0	<5.0

Date	Hexa-chloro-ethane, total	Indeno (1,2,3-cd) pyrene, total	Iso-phorone, total	Naphthalene, total	N-butyl benzyl phthalate, total	Nitro-benzene, total	N-nitro-sodi-n-propyl-amine, total	N-nitro-sodi-methyl-amine, total	N-nitro-sodi-phenyl-amine, total	Para-chloro-meta-cresol, total	Penta-chloro-phenol, total	Phenan-threne, total
05-31-92	<5.0	12.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	40.0
06-19-92	<5.0	19.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	67.0
11-21-92	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	<5.0

Date	Phenol, total	Pyrene, total	Benzene-o-chloro-water unfiltered, recoverable	1,2-di-phenyl-hydra-zine, water, total-recoverable	Benzene, 1,2,4-tri-chloro-benzene, unfiltered, recoverable	Benzene, 1,2-di-chloro-water, unfiltered, recoverable	Benzene, 1,3-di-chloro-water, unfiltered, recoverable	Benzene, 1,4-di-chloro-water, unfiltered, recoverable	1,2,5,6-di-benz-anthra-cene, total	2-chloro-naphtha-lene, total	2-chloro-phenol, total	2-nitro-phenol, total
05-31-92	--	52.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0
06-19-92	--	94.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0
11-21-92	<5.0	6.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0

Date	2,4-di-chloro-phenol, total	2,4-di-methyl-phenol, total	2,4-di-nitro-phenol, total	2,4,6-tri-chloro-phenol, total	2,4-di-nitro-toluene, total	2,6-di-nitro-toluene, total	3,3-di-chloro-benzi-dine, total	4-bromo-phenyl-phenol, ether, total	4-chloro-phenyl-phenol, ether, total	4-nitro-phenol, total	4,6-di-nitro-ortho-cresol, total
05-31-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0
06-19-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0
11-21-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0

Table 20. Selected acid-base/neutral organic compounds for storm-runoff sites in Colorado Springs--Continued

Date	Time	Ace-naph-thylene, total	Ace-naph-thene, total	Anthra-cene, total	Benzi-dine, total	Benzo-a-an-thra-cene, 2-benz-anthra-cene, total	Benzo-a-pyrene, total	Ben-zogh-l-peryl-enel, 12-benzo-peryl-ene, total	Benzo-b-fluor-anthene, total	Benzo-k-fluor-anthene, total	Bis (2-ethyl-hexyl) phthal-ate, total	Bis 2-chlo-ro-ethyl-ether, total	Bis (2-chloro-ethoxy) methane, total
385240104493601 - Beacon Street at Buchanan Street (Site 3)													
06-05-92	1240	<5.0	<5.0	5.0	<40.0	<10.0	12.0	31.0	17.0	14.0	14.0	<5.0	<5.0
06-19-92	1735	<5.0	<5.0	<5.0	<40.0	10.0	15.0	<10.0	26.0	30.0	14.0	<5.0	<5.0
11-11-92	0759	<5.0	<5.0	<5.0	<40.0	<10.0	<10.0	<10.0	<10.0	<10.0	9.0	<5.0	<5.0

Date	Bis (2-chloro-Iso-propyl) ether, total	Chrysene, total	Di-ethyl-phthalate, total	Di-methyl-phthalate, total	Di-n-octyl-phthalate, total	Di-n-butyl-phthalate, total	Fluor-anthene, total	Fluorene, total	Hexa-chloro-benzene, total	Hexa-chloro-butadiene, total
06-05-92	<5.0	22.0	<5.0	<5.0	<10.0	<5.0	45.0	<5.0	<5.0	<5.0
06-19-92	<5.0	23.0	<5.0	<5.0	<10.0	<5.0	45.0	<5.0	<5.0	<5.0
11-11-92	<5.0	<10.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0

Date	Hexa-chloro-ethane, total	Indeno (1,2,3-cd) pyrene, total	Iso-phorone, total	Naph-thalene, total	N-butyl benzyl phthalate, total	Nitro-benzene, total	N-nitro-sodi-n-propyl-amine, total	N-nitro-sodi-methyl-amine, total	N-nitro-sodi-phenyl-amine, total	Para-chloro-meta-cresol, total	Penta-chloro-phenol, total	Phenan-threne, total
06-05-92	<5.0	38.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	31.0
06-19-92	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	25.0
11-11-92	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	<5.0

Date	Phenol, total	Pyrene, total	Benzene-o-chloro-water unflit-tered, recov-erable	1,2-di-phenyl-hydra-zine, water, total, recov-erable	Benzene, 1,2,4-tri-chloro-benzene, unflit-tered, recov-erable	Benzene, 1,2-di-chloro-water, unflit-tered, recov-erable	Benzene, 1,3-di-chloro-water, unflit-tered, recov-erable	Benzene, 1,4-di-chloro-water, unflit-tered, recov-erable	1,2,5,6-di-benz-anthra-cene, total	2-chloro-naphtha-lene, total	2-chloro-phenol, total	2-nitro-phenol, total
06-05-92	--	32.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0
06-19-92	--	32.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0
11-11-92	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0

60 Water Quality of Storm Runoff and Comparison of Procedures for Estimating Storm-Runoff Loads, Volume, Event-Mean Concentrations, and the Mean Load for a Storm for Selected Properties and Constituents for Colorado Springs, Southeastern Colorado, 1992

Date	2,4-dichlorophenol, total	2,4-dimethylphenol, total	2,4-dinitrophenol, total	2,4,6-trichlorophenol, total	2,4-dinitrotoluene, total	2,6-dinitrotoluene, total	3,3-dichlorobenzidine, total	4-bromophenylphenol, ether, total	4-chlorophenylphenol, ether, total	4-nitrophenol, total	4,6-dinitro-ortho-cresol, total
385240104493601 - Beacon Street at Buchanan Street (Site 3)-Continued											
06-05-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0
06-19-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0
11-11-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0

Date	Time	Ace-naphthylene, total	Ace-naphthene, total	Anthracene, total	Benzo-dine, total	Benzo-a-anthracene, 2-benzanthracene, total	Benzo-a-pyrene, total	Ben-zogh-l-perylene, 12-benzo-perylene, total	Benzo-b-fluoranthene, total	Benzo-k-fluoranthene, total	Bis (2-ethylhexyl) phthalate, total	Bis 2-chloro-ethyl-ether, total	Bis (2-chloro-ethoxy) methane, total
385118104485801 - Wahsatch Street at Cross Lane (Site 4)													
06-12-92	1504	<5.0	<5.0	<5.0	<40.0	<10.0	11.0	<10.0	15.0	14.0	13.0	<5.0	<5.0
07-25-92	1408	<5.0	<5.0	<5.0	<40.0	<10.0	<10.0	<10.0	<10.0	<10.0	7.0	<5.0	<5.0
11-21-92	1010	<5.0	<5.0	<5.0	<40.0	<10.0	<10.0	<10.0	<10.0	<10.0	13.0	<5.0	<5.0

Date	Bis (2-chloro-isopropyl) ether, total	Chrysene, total	Di-ethyl-phthalate, total	Di-methyl-phthalate, total	Di-n-octyl-phthalate, total	Di-n-butyl-phthalate, total	Fluor-anthene, total	Fluorene, total	Hexa-chloro-benzene, total	Hexa-chloro-butadiene, total
06-12-92	<5.0	17.0	<5.0	<5.0	<10.0	<5.0	32.0	<5.0	<5.0	<5.0
07-25-92	<5.0	<10.0	<5.0	<5.0	<10.0	<5.0	11.0	<5.0	<5.0	<5.0
11-21-92	<5.0	<10.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0

Date	Hexa-chloro-ethane, total	Indeno (1,2,3-cd) pyrene, total	Iso-phorone, total	Naphthalene, total	N-butyl benzyl phthalate, total	Nitro-benzene, total	N-nitro-sodi-n-propylamine, total	N-nitro-sodi-methylamine, total	N-nitro-sodi-phenylamine, total	Para-chloro-meta-cresol, total	Penta-chloro-phenol, total	Phenan-threne, total
06-12-92	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	23.0
07-25-92	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	7.0
11-21-92	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	<5.0

Table 20. Selected acid-base/neutral organic compounds for storm-runoff sites in Colorado Springs--Continued

Date	Phenol, total	Pyrene, total	Benzene-chloro-water unfiltered, recoverable	1,2-diphenylhydrazine, water, total-recoverable	Benzene, 1,2,4-trichlorobenzene, unfiltered, recoverable	Benzene, 1,2-dichloro-water, unfiltered, recoverable	Benzene, 1,3-dichloro-water, unfiltered, recoverable	Benzene, 1,4-dichloro-water, unfiltered, recoverable	1,2,5,6-dibenzanthracene, total	2-chloronaphthalene, total	2-chlorophenol, total	2-nitrophenol, total	
385118104485801 - Wahsatch Street and Cross Lane (Site 4)--Continued													
06-12-92	--	24.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0	
07-25-92	--	9.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0	
11-21-92	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0	
Date	2,4-dichlorophenol, total	2,4-dimethylphenol, total	2,4-dinitrophenol, total	2,4,6-trichlorophenol, total	2,4-dinitrotoluene, total	2,6-dinitrotoluene, total	3,3-dichlorobenzidine, total	4-bromophenylphenol, ether, total	4-chlorophenylphenol, ether, total	4-nitrophenol, total	4,6-dinitro-ortho-cresol, total		
06-12-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0		
07-25-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0		
11-21-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0		
Date	Time	Acenaphthylene, total	Acenaphthene, total	Anthracene, total	Benzo-dine, total	Benzo-anthracene, 2-benzanthracene, total	Benzo-a-pyrene, total	Ben-zogh-l-perylene, 12-benzoperylene, total	Benzo-b-fluoranthene, total	Benzo-k-fluoranthene, total	Bis (2-ethylhexyl) phthalate, total	Bis 2-chloroethyl-ether, total	Bis (2-chloroethoxy) methane, total
384935104501501 - Walmart at Eighth Street (Site 5)													
07-17-92	0250	<5.0	<5.0	<5.0	<40.0	<10.0	<10.0	<10.0	<10.0	<10.0	18.0	<5.0	<5.0
08-03-92	1717	<5.0	<5.0	<5.0	<40.0	<10.0	<10.0	<10.0	<10.0	<10.0	41.0	<5.0	<5.0
12-06-92	1115	<5.0	<5.0	9.0	<40.0	<10.0	<10.0	<10.0	<10.0	<10.0	100.0	<5.0	<5.0

Table 20. Selected acid-base/neutral organic compounds for storm-runoff sites in Colorado Springs--Continued

Date	Bis (2-chloro-iso-propyl) ether, total	Chrysene, total	Di-ethyl-phthalate, total	Di-methyl-phthalate, total	Di-n-octyl-phthalate, total	Di-n-butyl-phthalate, total	Fluor-anthene, total	Fluorene, total	Hexa-chloro-benzene, total	Hexa-chloro-butadiene, total
384935104501501 - Walmart at Eighth Street (Site 5)--Continued										
07-17-92	<5.0	<10.0	<5.0	<5.0	<10.0	<5.0	10.0	<5.0	<5.0	<5.0
08-03-92	<5.0	<10.0	<5.0	<5.0	<10.0	7.0	14.0	<5.0	<5.0	<5.0
12-06-92	<5.0	<10.0	<5.0	<5.0	11.0	6.0	11.0	<5.0	<5.0	<5.0

Date	Hexa-chloro-ethane, total	Indeno (1,2,3-cd) pyrene, total	Iso-phorone, total	Naphthalene, total	N-butyl benzyl phthalate, total	Nitro-benzene, total	N-nitro-so-di-n-propyl-amine, total	N-nitro-sodi-methyl-amine, total	N-nitro-sodi-phenyl-amine, total	Para-chloro-meta-cresol, total	Penta-chloro-phenol, total	Phenan-threne, total
07-17-92	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	5.0
08-03-92	<5.0	<10.0	<5.0	<5.0	6.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	9.0
12-06-92	<5.0	<10.0	<5.0	<5.0	11.0	<5.0	<5.0	<5.0	<5.0	<30.0	<30.0	<5.0

Date	Phenol, total	Pyrene, total	Benzene-o-chloro-water unfiltered, recoverable	1,2-di-phenyl-hydra-zine, water, total-recoverable	Benzene, 1,2,4-tri-chloro-benzene, unfiltered, recoverable	Benzene, 1,2-di-chloro-water, unfiltered, recoverable	Benzene, 1,3-di-chloro-water, unfiltered, recoverable	Benzene, 1,4-di-chloro-water, unfiltered, recoverable	1,2,5,6-di-benz-anthra-cene, total	2-chloro-naphtha-lene, total	2-chloro-phenol, total	2-nitro-phenol, total
07-17-92	--	8.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0
08-03-92	--	12.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0
12-06-92	<5.0	10.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0

Date	2,4-di-chloro-phenol, total	2,4-di-methyl-phenol, total	2,4-di-nitro-phenol, total	2,4,6-tri-chloro-phenol, total	2,4-di-nitro-toluene, total	2,6-di-nitro-toluene, total	3,3-di-chloro-benzl-dine, total	4-bromo-phenyl-phenol, ether, total	4-chloro-phenyl-phenol, ether, total	4-nitro-phenol, total	4,6-di-nitro-ortho-creaol, total
07-17-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0
08-03-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0
12-06-92	<5.0	<5.0	<20.0	<20.0	<5.0	<5.0	<20.0	<5.0	<5.0	<30.0	<30.0

Table 21. Selected pesticide compounds for storm-runoff sites in Colorado Springs

[All concentrations in micrograms per liter, <, less than]

Date	Time	Aldrin, total	BHC Alpha, total	Aroclor, PCB, total						
				1016	1221	1232	1242	1248	1254	1260
38491810445201 - Sixteenth Hole, Valley-Hi Golf Course (Site 1)										
07-25-92	1423	<0.04	<0.03	<0.1	<1.0	<0.1	<0.1	<0.1	<0.1	<0.1
08-10-92	0840	<0.04	<0.03	<1	<1.0	<1	<1	<1	<1	<1
11-21-92	0930	<0.04	<0.03	<1	<1.0	<1	<1	<1	<1	<1

Date	Beta-benzene, hexachloride, total	Chlor-dane, cis, water, whole, total	Chlor-dane, total	Chlor-dane, trans, water, whole, total	Delta, benzene, hexachloride, total	Di-eldrin, total	Endo-sulfan-beta, total	Endo-sulfan I, Alpha, water, whole, recoverable	Endo-sulfan, sulfate, total
07-25-92	<0.03	<0.10	<0.1	<0.10	<0.09	<0.02	<0.04	<0.10	<0.60
08-10-92	<0.03	<10	<1	<10	<0.09	<0.02	<0.04	<10	<60
11-21-92	<0.03	<10	<1	<10	<0.09	<0.02	<0.04	<10	<60

Date	Endrin, aldehyde, total	Endrin water, unfiltered, recoverable	Gamma, BHC, Lindane, total	Hepta-chlor, total	Hepta-chlor-epoxide, total	Toxaphene, total	p,p' DDD, total	p,p' DDE, total	p,p' DDT, total
07-25-92	<0.20	<0.06	<0.03	<0.03	<0.80	<	<0.10	<0.04	<0.10
08-10-92	<0.20	<0.06	<0.03	<0.03	<0.80	<	<0.10	<0.04	<0.10
11-21-92	<0.20	<0.06	<0.03	<0.03	<0.80	<	<0.10	<0.04	<0.10

R0024146

Table 21. Selected pesticide compounds for storm-runoff sites in Colorado Springs--Continued

Date	Time	Aldrin, total	BHC Alpha, total	Aroclor, PCB, total						
				1016	1221	1232	1242	1248	1254	1260
385347104500601 - Chestnut Street at Douglas Creek (Site 2)										
05-31-92	1255	<0.04	<0.03	<0.1	<1.0	<0.1	<0.1	<0.1	<0.1	<0.1
06-19-92	1728	<.04	<.03	<.1	<1.0	<.1	<.1	<.1	<.1	<.1
11-21-92	0910	<.04	<.03	<.1	<1.0	<.1	<.1	<.1	<.1	<.1

Date	Beta-benzene, hexachloride, total	Chlor-dane, cis, water, whole, total	Chlor-dane, total	Chlor-dane, trans, water, whole, total	Delta, benzene, hexachloride, total	Di-eldrin, total	Endo-sulfan-beta, total	Endo-sulfan I, Alpha, water, whole, recoverable	Endo-sulfan, sulfate, total
05-31-92	<0.03	<0.10	<0.1	<0.10	<0.09	<0.02	<0.06	<0.10	<0.60
06-19-92	<.03	<.10	<.1	<.10	<.09	<.02	<.04	<.10	<.60
11-21-92	<.03	<.10	<.1	<.10	<.09	<.02	<.04	<.10	<.60

Date	Endrin, aldehyde, total	Endrin water, unfiltered, recoverable	Gamma, BHC, Lindane, total	Hepta-chlor, total	Hepta-chlor-epoxide, total	Tox-aphene, total	p,p' DDD, total	p,p' DDE, total	p,p' DDT, total
05-31-92	<0.20	<0.06	<0.03	<0.03	<0.80	<.2	<0.10	<0.04	<0.10
06-19-92	<.20	<.06	<.03	<.03	<.80	<.2	<.10	<.04	<.10
11-21-92	<.20	<.06	<.03	<.03	<.80	<.2	<.10	<.04	<.10

Table 21. Selected pesticide compounds for storm-runoff sites in Colorado Springs--Continued

Date	Time	Aldrin, total	BHC Alpha, total	Aroclor, PCB, total						
				1016	1221	1232	1242	1248	1254	1260
385240104493601 - Beacon Street at Buchanan Street (Site 3)										
06-05-92	1240	<0.04	<0.03	<0.1	<1.0	<0.1	<0.1	<0.1	<0.1	<0.1
06-19-92	1735	<.04	<.03	<.1	<1.0	<.1	<.1	<.1	<.1	<.1
11-11-92	0759	<.04	<.03	<.1	<1.0	<.1	<.1	<.1	<.1	<.1

Date	Beta-benzene, hexachloride, total	Chlor-dane, cis, water, whole, total	Chlor-dane, total	Chlor-dane, trans, water, whole, total	Delta, benzene, hexachloride, total	Di-eldrin, total	Endo-sulfan-beta, total	Endo-sulfan I, Alpha, water, whole, recoverable	Endo-sulfan, sulfate, total
06-05-92	<0.03	<0.10	<0.1	<0.10	<0.09	<0.020	<0.04	<0.10	<0.60
06-19-92	<.03	<.10	<.1	<.10	<.09	<.020	<.04	<.10	<.60
11-11-92	<.03	<.10	<.1	<.10	<.09	<.020	<.04	<.10	<.60

Date	Endrin, aldehyde, total	Endrin water, unfiltered, recoverable	Gamma, BHC, Lindane, total	Hepta-chlor, total	Hepta-chlor-epoxide, total	Tox-aphene, total	p,p' DDD, total	p,p' DDE, total	p,p' DDT, total
06-05-92	<0.20	<0.06	<0.03	<0.03	<0.80	<	<0.10	<0.04	<0.10
06-19-92	<.20	<.06	<.03	<.03	<.80	<	<.10	<.04	<.10
11-11-92	<.20	<.06	<.03	<.03	<.80	<	<.10	<.04	<.10

R0024148

Table 21. Selected pesticide compounds for storm-runoff sites in Colorado Springs--Continued

Date	Time	Aldrin, total	BHC Alpha, total	Aroclor, PCB, total						
				1016	1221	1232	1242	1248	1254	1260
385118104485801 - Wahsatch Street at Cross Lane (Site 4)										
06-12-92	1504	<0.040	<0.03	<0.1	<1.0	<0.1	<0.1	<0.1	<0.1	<0.1
07-25-92	1408	<.040	<.03	<.1	<1.0	<.1	<.1	<.1	<.1	<.1
11-21-92	1005	<.040	<.03	<.1	<1.0	<.1	<.1	<.1	<.1	<.1

Date	Beta-benzene, hexachloride, total	Chlor-dane, cis, water, whole, total	Chlor-dane, total	Chlor-dane, trans, water, whole, total	Delta, benzene, hexachloride, total	Di-eldrin, total	Endo-sulfan-beta, total	Endo-sulfan I, Alpha, water, whole, recoverable	Endo-sulfan, sulfate, total
06-12-92	<0.03	<0.10	0.2	<0.10	<0.09	<0.02	<0.04	<0.10	<0.60
07-25-92	<.03	<.10	<.1	<.10	<.09	<.02	<.04	<.10	<.60
11-21-92	<.03	<.10	<.1	<.10	<.09	<.02	<.04	<.10	<.60

Date	Endrin, aldehyde, total	Endrin water, unfiltered, recoverable	Gamma, BHC, Lindane, total	Hepta-chlor, total	Hepta-chlor-epoxide, total	Tox-aphene, total	p,p' DDD, total	p,p' DDE, total	p,p' DDT, total
06-12-92	<0.20	<0.06	<0.03	<0.03	<0.80	<	<0.10	<0.04	<0.10
07-25-92	<.20	<.06	<.03	<.03	<.80	<	<.10	<.04	<.10
11-21-92	<.20	<.06	<.03	<.03	<.80	<	<.10	<.04	<.10

Table 21. Selected pesticide compounds for storm-runoff sites in Colorado Springs--Continued

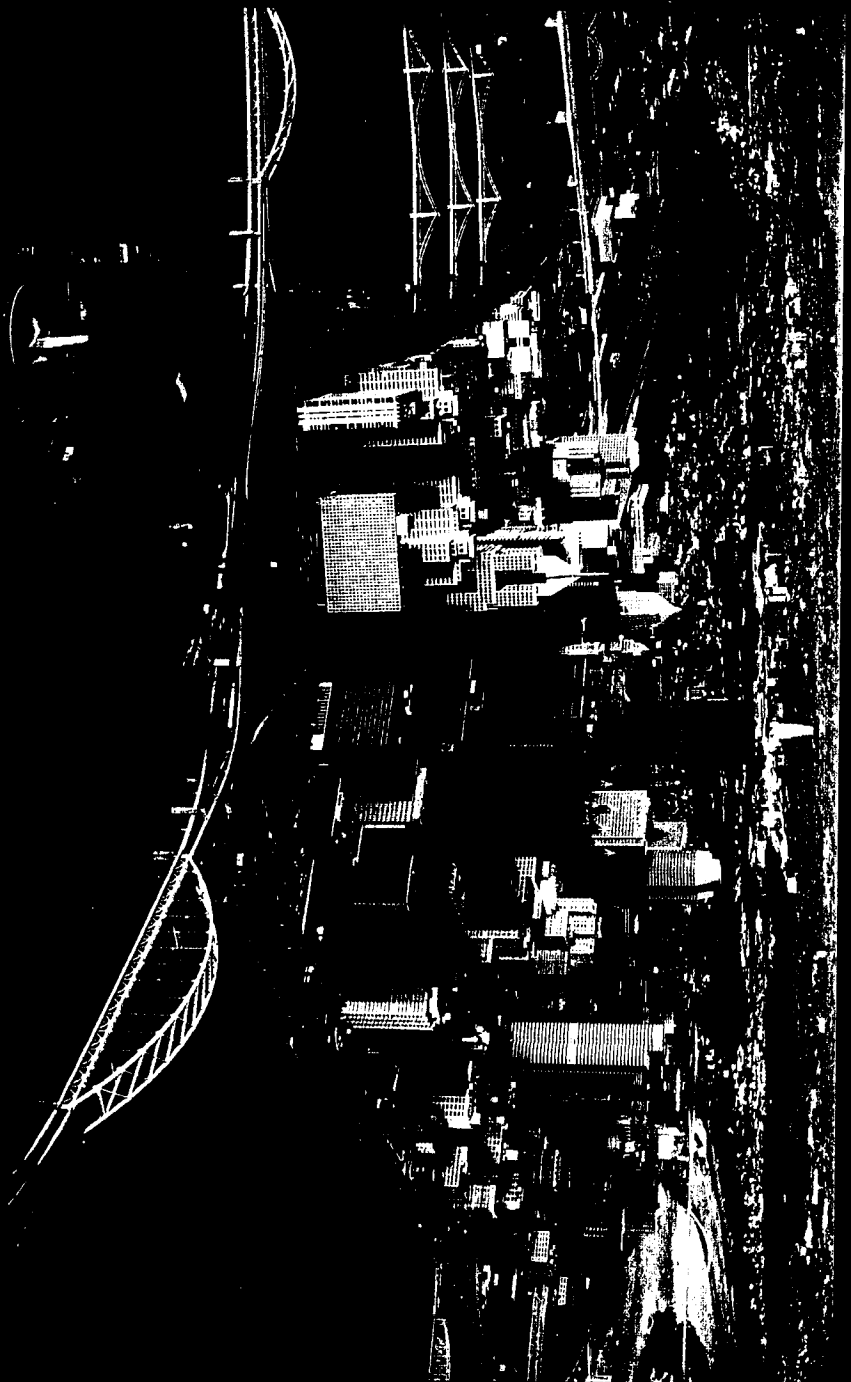
Date	Time	Aldrin, total	BHC Alpha, total	Aroclor, PCB, total						
				1016	1221	1232	1242	1248	1254	1260
384935104501501 - Walmart at Eighth Street (Site 5)										
07-17-92	0250	<0.04	<0.03	<0.1	<1.0	<0.1	<0.1	<0.1	<0.1	<0.1
08-03-92	1717	<.04	<.03	<1	<1.0	<.1	<.1	<1	<.1	<.1
12-06-92	1115	<.04	<.03	<1	<1.0	<.1	<.1	<1	<.1	<.1

Date	Beta-benzene, hexachloride, total	Chlor-dane, cis, water, whole, total	Chlor-dane, total	Chlor-dane, trans, water, whole, total	Delta, benzene, hexachloride, total	Di-eldrin, total	Endo-sulfan-beta, total	Endo-sulfan I, Alpha, water, whole, recoverable	Endo-sulfan, sulfate, total
07-17-92	<0.20	<0.060	<0.030	<0.030	<0.80	<0.02	<0.10	<0.04	<0.10
08-03-92	<.03	<.10	<.1	<.10	<.09	<.02	<.04	<.10	<.60
12-06-92	<.03	<.10	<.1	<.10	<.09	<.02	<.04	<.10	<.60

Date	Endrin, aldehyde, total	Endrin water, unfiltered, recoverable	Gamma, BHC, Lindane, total	Hepta-chlor, total	Hepta-chlor-epoxide, total	Tox-aphene, total	p,p' DDD, total	p,p' DDE, total	p,p' DDT, total
07-17-92	<0.20	<0.06	<0.03	<0.03	<0.80	<.2	<0.10	<0.04	<0.10
08-03-92	<.20	<.06	<.03	<.03	<.80	<.2	<.10	<.04	<.10
12-06-92	<.20	<.06	<.03	<.03	<.80	<.2	<.10	<.04	<.10

R0024150

Water Quality in the Allegheny and Monongahela River Basins Pennsylvania, West Virginia, New York, and Maryland, 1996–98



U.S. Department of the Interior
U.S. Geological Survey

Circular 1202

R0024151

POINTS OF CONTACT AND ADDITIONAL INFORMATION

The companion Web site for NAWQA summary reports:

<http://water.usgs.gov/nawqa/>

Allegheny-Monongahela River contact and Web site:

USGS State Representative
U.S. Geological Survey
Water Resources Division
215 Limekiln Road
New Cumberland, PA 17070
e-mail: dc_pa@usgs.gov
<http://pa.water.usgs.gov/almn/>

National NAWQA Program:

Chief, NAWQA Program
U.S. Geological Survey
Water Resources Division
12201 Sunrise Valley Drive, M.S. 413
Reston, VA 20192
<http://water.usgs.gov/nawqa/>

Other NAWQA summary reports

River Basin Assessments

Albemarle-Pamlico Drainage Basin (Circular 1157)
Apalachicola-Chattahoochee-Flint River Basin (Circular 1164)
Central Arizona Basins (Circular 1213)
Central Columbia Plateau (Circular 1144)
Central Nebraska Basins (Circular 1163)
Connecticut, Housatonic and Thames River Basins (Circular 1155)
Eastern Iowa Basins (Circular 1210)
Georgia-Florida Coastal Plain (Circular 1151)
Hudson River Basin (Circular 1165)
Kanawha-New River Basins (Circular 1204)
Lake Erie-Lake Saint Clair Drainages (Circular 1203)
Las Vegas Valley Area and the Carson and Truckee River Basins (Circular 1170)
Lower Illinois River Basin (Circular 1209)
Long Island-New Jersey Coastal Drainages (Circular 1201)
Lower Susquehanna River Basin (Circular 1168)
Mississippi Embayment (Circular 1208)
Ozark Plateaus (Circular 1158)
Potomac River Basin (Circular 1166)
Puget Sound Basin (Circular 1216)
Red River of the North Basin (Circular 1169)

Rio Grande Valley (Circular 1162)
Sacramento River Basin (Circular 1215)
San Joaquin-Tulare Basins (Circular 1159)
Santee River Basin and Coastal Drainages (Circular 1206)
South-Central Texas (Circular 1212)
South Platte River Basin (Circular 1167)
Southern Florida (Circular 1207)
Trinity River Basin (Circular 1171)
Upper Colorado River Basin (Circular 1214)
Upper Mississippi River Basin (Circular 1211)
Upper Snake River Basin (Circular 1160)
Upper Tennessee River Basin (Circular 1205)
Western Lake Michigan Drainages (Circular 1156)
White River Basin (Circular 1150)
Willamette Basin (Circular 1161)

National Assessments

The Quality of Our Nation's Waters—Nutrients and Pesticides (Circular 1225)

Front cover: Areal view of confluence of the Allegheny and Monongahela Rivers forming the Ohio River at Pittsburgh, Pennsylvania. (Photograph by Jim Schafer.)

Back cover: Left, The headwaters of the Allegheny River, Potter County, Pa. (photograph by Jim Schafer); center, Whitewater rafting on the Cheat River, West Virginia (photograph by Randy Robinson); right, agricultural fields and forest in the hills above the Allegheny River, Armstrong County, Pa. (photograph by Jim Schafer).



News Release

Address

U.S. Department of the Interior
U.S. Geological Survey

U.S. Geological Survey
1000 Church Hill Road, Suite 200
Pittsburgh, PA 15205

Release	Contact	Phone	Fax
May 29, 2001	Steve McAuley, Project Chief	(412) 490-3801	(412) 490-3828
	Robert Anderson, Biologist	(412) 490-3810	

Water quality in the Allegheny and Monongahela River Basins is generally good – with some exceptions

Stream and groundwater quality in many areas of the Allegheny and Monongahela River Basins is improving and is, in many respects, better than or comparable to what is found nationally, according to the results of a 5-year investigation by the U.S. Geological Survey (USGS).

“The quality of rivers around Pittsburgh has really improved in the last 20 years, especially for fish,” said USGS biologist and principal author, Robert Anderson. “A lot of people still think of these rivers as polluted as they were in the 1960’s, but if someone wanted to go fishing for bass or walleye, they could do very well right in downtown Pittsburgh.”

Compliance with regulations that control water discharged from active coal mines has been a factor in the improvement. In addition, government agency and citizens groups have been very active in focusing funding and energy on treatment of water from abandoned coal mines in Pennsylvania, West Virginia, New York, and Maryland.

“Coal mining remains the largest single factor affecting water quality in a large part of the Allegheny and Monongahela River Basins,” said Steve McAuley, Project Chief. “We are not seeing new sources of highly acidic discharges but the water leaving coal mines does differ from water in unmined areas.” Where treatment has not occurred, the water is generally acidic (low pH), and contains elevated concentrations of metals and sulfate. Sulfate concentrations are five times greater in streams draining mined areas than in streams draining unmined areas. In addition, the diversity and abundance of aquatic organisms remain reduced in comparison to areas where there has been no coal mining. These results are based on well and stream samples from more than 180 sites in the coal-bearing region.

The region’s water quality is influenced by more than coal mining. Past pollution still lingers in the form of PCBs, chlordane, and DDT. The use of these compounds has been prohibited for two decades but they remain in fish tissue samples and in river sediments. For example, PCB’s were detected in 43 percent of the total stream sediment and fish samples tested.

Mixtures of currently used pesticides were detected in some agricultural and urban streams and well water. Concentrations of the individual compounds were usually lower than guidelines for drinking water or protection of aquatic life. The potential risk, however, can only be partially

addressed because standards and guidelines are not available for all measured compounds, and they do not consider exposure to mixtures of chemicals.

"Some areas in the Allegheny and Monongahela River Basins appear to be of much better quality than the national average. The amount and types of life living in some forested and agricultural areas are among the most diverse in the Nation," said Anderson, "which is something to be proud of."

Copies of the 38-page, color report, "Water Quality in the Allegheny and Monongahela River Basins, 1996-98," by Robert M. Anderson, Kevin M. Beer, Theodore F. Buckwalter, Mary E. Clark, Steven D. McAuley, James I. Sams, III, and Donald R. Williams, published as USGS Circular 1202, are available free of charge from the USGS Branch of Information Services, Box 25286, Denver Federal Center, Denver, CO 80225. (303) 202-4700 (fax request to (303) 202-4693). The Circular may be viewed on the World Wide Web at <http://water.usgs.gov/nawqa>.

This assessment is part of the National Water Quality Assessment (NAWQA) Program conducted by USGS investigating water-quality conditions in more than 50 major river basins and aquifers. This program is currently releasing results on surface and ground water in 15 additional major river basins. Access the individual basin reports on the NAWQA website, in addition to other NAWQA publications and national data sets and maps.

Further detail on the effects of coal mining in the Allegheny and Monongahela River Basins can be obtained in USGS Water-Resources Investigations Report 99-4208, "Effects of Coal-Mine Drainage on Stream Water Quality in the Allegheny and Monongahela River Basins – Sulfate Transport and Trends," by James I. Sams, III, and Kevin M. Beer, and in Water-Resources Investigations Report 98-4258, "Stream Water Quality in Coal Mined Areas of the Lower Cheat River Basin, West Virginia and Pennsylvania, During Low-Flow Conditions, July 1997," by Donald R. Williams, Mary E. Clark, and Julianne Brown.

Further detail regarding pesticides and nutrients can be obtained from Water-Resources Investigations Report 00-4061, "Nutrients and Organic Compounds in Deer Creek and South Branch Plum Creek in Southwestern Pennsylvania, April 1996 through September 1998," by Donald R. Williams and Mary E. Clark. These reports are available on the web at http://pa.water.usgs.gov/pa_pubs.html. A limited supply is available at the USGS office in New Cumberland at 215 Limekiln Road, New Cumberland, Pennsylvania 17070; telephone (717) 730-6916 and the USGS office in Pittsburgh at 1000 Church Hill Road, Suite 200, Pittsburgh, Pennsylvania 15205; telephone (412) 490-3800.

As the nation's largest water, earth, and biological science and civilian mapping agency, the USGS works in cooperation with more than 2,000 organizations across the country to provide reliable, impartial, scientific information to resource managers, planners, and other customers. This information is gathered in every state by USGS scientists to minimize the loss of life and property from natural disasters, contribute to the sound conservation, economic and physical development of the Nation's natural resources, and enhance the quality of life by monitoring water, biological, energy, and mineral resources.

Water Quality in the Allegheny and Monongahela River Basins

Pennsylvania, West Virginia, New York, and Maryland, 1996–98

By Robert M. Anderson, Kevin M. Beer, Theodore F. Buckwalter, Mary E. Clark,
Steven D. McAuley, James I. Sams, III, *and* Donald R. Williams

U.S. DEPARTMENT OF THE INTERIOR
GALE A. NORTON, SECRETARY

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

2000

Free on application to the
U.S. Geological Survey
Information Services
Box 25286 Federal Center
Denver, CO 80225

Or call: 1-888-ASK-USGS

Library of Congress Cataloging-in-Publications Data

Water quality in the Allegheny and Monongahela River Basins, Pennsylvania, West Virginia, New York, and Maryland, 1996-98 / by Robert M. Anderson...[et al.],
p. cm. -- (U.S. Geological Survey Circular ; 1202)
Includes bibliographical references.
ISBN 0-607-95411-6 (alk. paper)
1. Water quality--Allegheny river Watershed (Pa. and N.Y.) 2. Water quality--Monongahela River Watershed (W. Va. and Pa.) I. Anderson, Robert M. 1961- II. Geological Survey (U.S.) III. Series.

TD223.2 .W38 2000
363.739'42'0974486--dc21

00-049451

R0024156

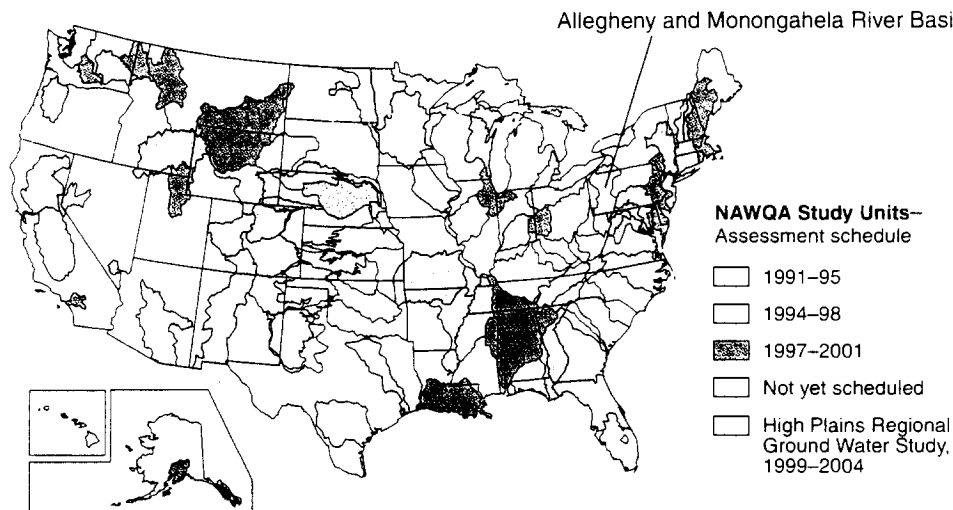
CONTENTS

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM	IV
SUMMARY OF MAJOR FINDINGS	1
Stream and river highlights	1
Ground-water highlights	2
INTRODUCTION TO THE ALLEGHENY AND MONONGAHELA RIVER BASINS	3
MAJOR FINDINGS	5
Coal mining dominates water quality	5
Sulfate is an indicator of coal-mining activity	6
Aquatic communities are affected in streams receiving large amounts of mine drainage	7
NATIONAL PERSPECTIVE—Invertebrate index status is among the best in the Nation in forested settings	8
Ground-water quality is affected near mined areas	9
Concentrations of trace elements in bed sediment exceed aquatic-life guidelines	9
NATIONAL PERSPECTIVE—Trace elements may limit aquatic life in urban streams and mined areas	10
Water quality of the large rivers of the Allegheny and Monongahela River Basins is improving	10
Persistent pesticides and PCBs were more prevalent in fish tissue than in sediment	11
Low concentrations of numerous pesticides were detected in an agricultural stream and an urban stream	12
Pesticides are at low concentrations when detected in ground water	13
NATIONAL PERSPECTIVE—Pesticides in the Allegheny and Monongahela River Basins are similar to those found nationally	15
Volatile organic compounds were detected at low concentrations in an urban stream	15
Low levels of volatile organic compounds were detected in most domestic wells sampled	16
Nitrate is common in streams and ground water	17
Radon in ground water is common but highly variable	19
REGIONAL STUDIES: Sulfate concentrations and biological communities in Appalachian coal areas indicate mining-related disturbances despite a general water-quality improvement between 1980 and 1998	20
Water-quality characteristics targeted by SMCRA improved in streams, but sulfate and metals remain high at some sites	20
Invertebrate-community impairment appears related to amount of mining	21
Sulfate and some metal concentrations were higher in ground water near surface coal mines	21
STUDY UNIT DESIGN	22
GLOSSARY	24
REFERENCES	25
APPENDIX—WATER-QUALITY DATA FROM THE ALLEGHENY AND MONONGAHELA RIVER BASINS IN A NATIONAL CONTEXT	27

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

THIS REPORT summarizes major findings about water quality in the Allegheny and Monongahela River Basins that emerged from an assessment conducted between 1996 and 1998 by the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program. Water quality is discussed in terms of local and regional issues and compared to conditions found in all 36 NAWQA study areas, called Study Units, assessed to date. Findings are also explained in the context of selected national benchmarks, such as those for drinking-water quality and the protection of aquatic organisms. The NAWQA Program was not intended to assess the quality of the Nation's drinking water, such as by monitoring water from household taps. Rather, the assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water monitoring programs. The comparisons made in this report to drinking-water standards and guidelines are only in the context of the available untreated resource. Finally, this report includes information about the status of aquatic communities and the condition of in-stream habitats as elements of a complete water-quality assessment.

Many topics covered in this report reflect the concerns of officials of State and Federal agencies, water-resource managers, and members of stakeholder groups who provided advice and input during the Allegheny and Monongahela River Basins assessment. Study-area residents who wish to know more about water quality in the areas where they live will find this report informative as well.



THE NAWQA PROGRAM seeks to improve scientific and public understanding of water quality in the Nation's major river basins and ground-water systems. Better understanding facilitates effective resource management, accurate identification of water-quality priorities, and successful development of strategies that protect and restore water quality. Guided by a nationally consistent study design and shaped by ongoing communication with local, State, and Federal agencies, NAWQA assessments support the investigation of local issues and trends while providing a firm foundation for understanding water quality at regional and national scales. The ability to integrate local and national scales of data collection and analysis is a unique feature of the USGS NAWQA Program.

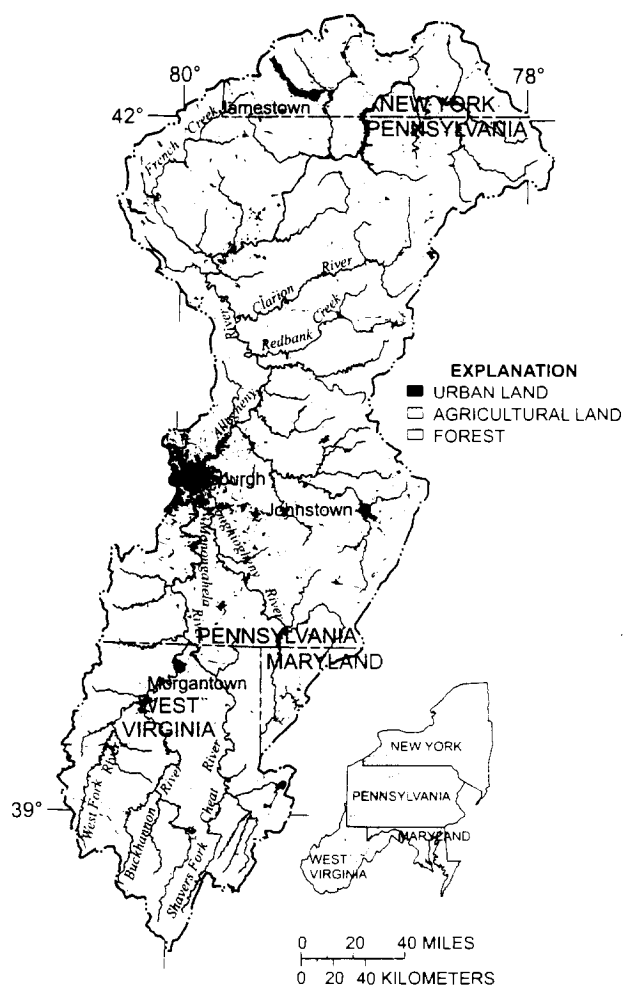
The Allegheny and Monongahela River Basins are one of 51 water-quality assessments initiated since 1991, when the U.S. Congress appropriated funds for the USGS to begin the NAWQA Program. As indicated on the map, 36 assessments have been completed, and 15 more assessments will conclude in 2001. Collectively, these assessments cover about half of the land area of the United States and include water resources that are available to more than 60 percent of the U.S. population.

SUMMARY OF MAJOR FINDINGS

Stream and River Highlights

Streams and rivers in the Allegheny and Monongahela River Basins range from those of high quality that support diverse aquatic life to those that are seriously degraded and support few aquatic species and few human uses of the water. Higher quality stream reaches are generally in the northern one-third of the study area and in mountainous areas in eastern sections. These areas are dominated by forest, low-intensity agriculture, and rural communities. Urban development and coal-mining activities through much of the basins have had a significant influence on water quality and aquatic life. Industrial activity in small and large towns has resulted in contaminated streambed sediments and contaminated fish. Acid- and/or mineral-laden mine drainage from abandoned coal mines is one of the most serious and persistent water-quality problems in the basins, limiting water use and aquatic resources.

- Sulfate concentrations were 5 times greater in streams draining mined areas than in streams draining unmined areas. Sulfate concentration is closely related to coal production in the sampled basins but not as clearly related to pH or dissolved metal concentration. (See page 6.)
- Since 1980, treatment of drainage from active and abandoned mines has generally resulted in improved water quality, with increased pH and lower metal and sulfate concentrations, but diversity and abundance of aquatic organisms remain reduced in comparison to unmined areas. (See pages 7, 20, and 21.)
- Zinc in bed sediment exceeded aquatic-life guidelines at 15 of 50 sites. (See page 9.)
- Arsenic concentrations most often exceeded aquatic-life guidelines in bed sediment in streams draining northern, once glaciated areas, and high concentrations appear to be unrelated to human activity. (See page 10.)
- Streams in forested settings are among the most diverse nationally with respect to aquatic insects among NAWQA sites sampled between 1996 and 1998. (See page 8.)
- A group of now-banned industrial chemicals, polychlorinated biphenyls (PCBs), was detected in 43 percent of sediment and fish-tissue samples. Consumption advisories are in place for several fish species because of PCB and chlordane contamination in some large river reaches. (See pages 11 and 12.)
- Some of the most degraded stream reaches have, since the early 1900s, supported few aquatic organisms. Yet, the quality of many reaches is now improving, and abundant fish and invertebrate populations include sensitive species not seen here in decades. (See page 11.)



The Allegheny and Monongahela River Basins drain 19,145 square miles of Pennsylvania, West Virginia, New York, and Maryland. About 64 percent of the study area is forested; the remainder is a patchwork of land uses. Agriculture (30 percent) is commonly low intensity pasture, dairy, and hay. Urban areas account for about 4 percent of the area, but they include many forested ridges. Coal-mining activities influence water quality in most of the study area but are not visible on this surface land-use map. (Land-use coverage is based on 1991, 1992, and 1993 land-use data.)

Major Influences on Streams and Rivers

- Surface, underground, reclaimed, and abandoned coal mines
- Impoundments and maintenance of navigation channels
- Increased urban development
- Reductions in agriculture, industrial activity, and coal production

- In sampled streams in basins dominated by urban or agricultural land, pesticides and volatile organic compounds (VOCs) were commonly detected, although generally at concentrations meeting drinking-water and aquatic-life standards and guidelines. (See pages 12–13 and 15–17.)
- Pesticide concentrations in stream water exceeded drinking-water guidelines in single samples from each of two basins, one dominated by agriculture and the other dominated by urban land use. (See pages 12 and 13.)

Ground-Water Highlights

Although not regulated, the quality of water from domestic wells—the predominant water source for residents of rural areas—meets Federal standards for drinking water for most substances analyzed in this study. Ground-water supply generally meets or exceeds expectations from wells in the highly permeable glaciofluvial deposits of the valley-fill aquifers in the northwest. Ground-water supply often meets needs but can be meager from wells tapping the water-filled fractures of the fractured-rock aquifers present throughout much of the rest of the study area.

- Compared to ground water in unmined areas of the coal-bearing rocks, water in shallow private domestic wells near reclaimed surface coal mines had higher concentrations of sulfate, iron, and manganese, even after all mining and reclamation had been completed. (See pages 9 and 21.)
- Pesticides were detected more frequently in the valley-fill aquifers of the glacial sediments than in fractured-rock aquifers. (See pages 13-15.)
- Overall, VOCs were detected at very low levels in the 95 ground-water samples analyzed. Gasoline-related compounds were detected slightly more frequently and at slightly higher concentrations in ground water near reclaimed surface coal mines than near unmined areas. (See pages 16 and 17.)
- Nitrate was detected in 62 percent of sampled wells, although only one domestic-well sample exceeded the drinking-water standard for nitrate. (See pages 17 and 18.)
- Radon was detected at levels exceeding the proposed Federal drinking-water standard of 300 pCi/L (picocuries per liter) in 56 percent of the ground-water samples. The proposed alternative standard (4,000 pCi/L) was exceeded in 2 percent of the samples. (See page 19.)

Major Influences on Ground Water

- Coal mining
- Pesticide and fertilizer application
- Widespread use of gasoline and oxygenates
- Naturally occurring concentrations of radon

Selected Indicators of Stream-Water Quality

	Small Streams				Major Rivers
	Urban	Agricultural	Forested	Mining	Mixed Land Uses
Pesticides ¹	■	■	□	--	--
Total Phosphorus ²	■	■	■	■	■
Nitrate ³	□	□	□	□	□
Trace elements ⁴	■	□	■	■	■
Organochlorines ⁵	X	X	X	X	■
Volatile organics ⁶	□	--	--	--	--

- Percentage of samples with concentrations **equal to or greater than** a health-related national guideline for drinking water, aquatic life, or water contact recreation; or above the USEPA goal for preventing excess plant growth in streams
- Percentage of samples with concentrations **less than** health-related national guideline for drinking water, aquatic life, or water-contact recreation; or below a national goal for preventing excess algal growth
- Percentage of samples with **no detection**
- Not assessed
- X Settings with less than 4 samples are not shown

Selected Indicators of Ground-Water Quality

	Domestic Supply Wells		
	Fractured-rock aquifer, mined areas	Fractured-rock aquifer	Valley-Fill aquifer
Pesticides ¹	□	--	□
Nitrate ³	□	■	■
Trace elements	■	■	--
Radon	■	■	■
Volatile organics ⁶	□	□	□

- Percentage of samples with concentrations **equal to or greater than** a health-related national guideline for drinking water
- Percentage of samples with concentrations **less than** health-related national guideline for drinking water
- Percentage of samples with **no detection**
- Not assessed

¹ Insecticides, herbicides, and pesticide breakdown products, sampled in water.

² Total phosphorus, sampled in water.

³ Nitrate (as nitrogen), sampled in water.

⁴ Arsenic, mercury, and metals, sampled in sediment.

⁵ Organochlorine compounds including DDT and PCBs, sampled in fish tissue.

⁶ Solvents, refrigerants, fumigants, and gasoline compounds, sampled in water.

INTRODUCTION TO THE ALLEGHENY AND MONONGAHELA RIVER BASINS

The Allegheny and Monongahela Rivers join at Pittsburgh, Pa., forming the Ohio River. Historically, these rivers served as a transportation corridor to the West and were of strategic military significance. The Allegheny and Monongahela River Basins were at the focus of the industrial revolution in the United States. In 1990, approximately 4.2 million people lived in the area, and although the land and water uses have changed many times, the legacy of past activities is evident. Today's stream quality reflects a blend of past and present land uses and the natural quality and quantity of the water in these basins.

Topography and Geology

The Allegheny and Monongahela River Basins (ALMN) lie almost entirely within the Appalachian Plateaus Physiographic Province. The entire study area is underlain by sedimentary rocks that have been fractured in many places by folding and faulting. These rocks carry ground water in much of ALMN and are referred to as fractured-rock aquifers. The northwestern parts of the Allegheny River Basin were glaciated. The glaciers deposited sand, gravel, silt, and clay in the valleys and eroded the hills, leaving a terrain of more consistent altitude (Becher, 1999; McAuley, 1995). The glaciofluvial and alluvial deposits overlying the sedimentary rocks are generally much more permeable and comprise the valley-fill aquifers (Risser and Madden, 1994). Glaciofluvial deposits include sediments left by water flowing within, under, or out of glaciers. In contrast, the Appalachian highlands to the east and southeast have steep, high

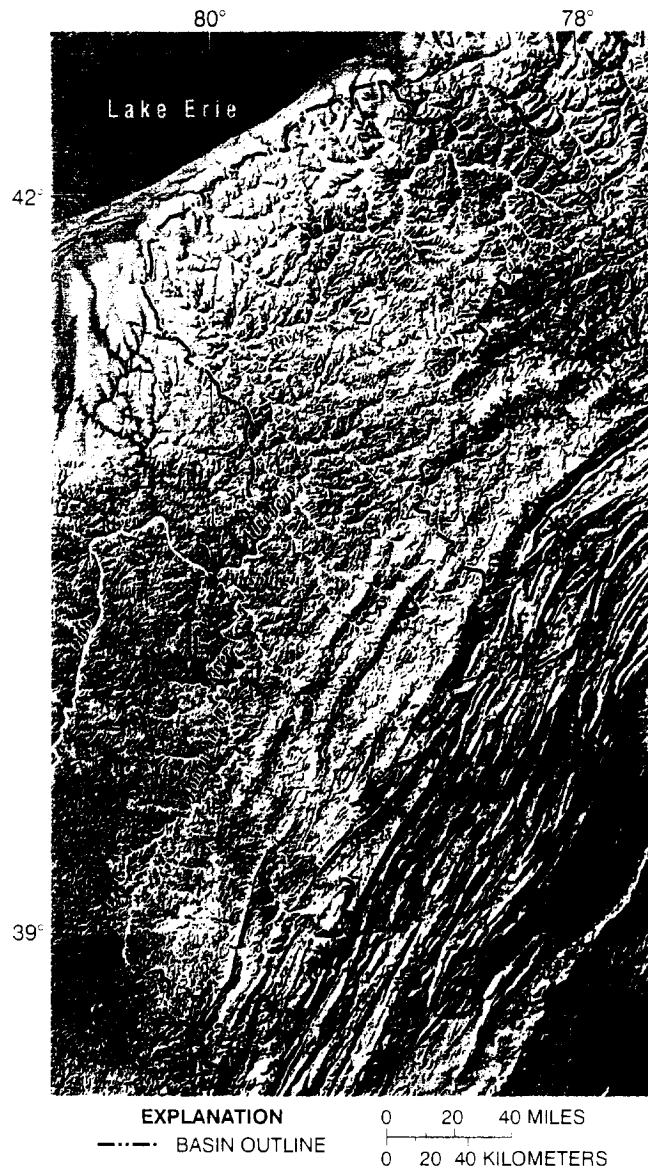


Figure 1. The Allegheny and Monongahela River Basins lie almost entirely within the Appalachian Plateaus Physiographic Province. The eastern parts of the basins are more mountainous, the west is characterized by “rolling hills,” and the northwest has relatively low relief as a result of being covered by glaciers in the last ice age. The topography affects land use, exposed geologic formations, and stream habitat—all of which, in turn, affect the quality and uses of water.

peaks, ridges, and plateaus that are deeply divided by valleys (fig. 1). Land use is limited by the rough terrain and nutrient-poor soil in much of ALMN, both of which make large agricultural fields impractical. Urban areas generally

lie along river valleys. Ridgetops are commonly forested, even in otherwise urban settings.

Ecologically, the streams of these basins present a diversity of habitats. Mountainous areas are generally dominated by streams

that are very low in nutrients and remain cold all year. These streams support trout and a few other cold-water fish species but commonly include diverse aquatic-invertebrate populations. Streams along the western side of ALMN are generally warm-water systems with a much greater diversity of fish species.

Water Use

Most water (94 percent) used in ALMN is drawn from surface-water sources. In 1995, 82 percent of water withdrawn in ALMN was for industrial uses or thermoelectric power generation. Although ground-water withdrawals are proportionally small, they are important for public supply or domestic use, especially in rural areas (fig. 2).

Reservoirs have been in place in the study area for more than 150 years for flood control (fig. 3), recreation, navigation, power generation, water quality, and water supply. Nearly all major tributaries have reservoirs constructed on them. The entire



Figure 3. One of the most devastating floods in United States history occurred in the Allegheny River Basin on May 31, 1889. A dam upstream from Johnstown, Pennsylvania, failed. Downstream, 2,209 people were killed and thousands more were injured. (Photograph used with permission from the National Park Service.)

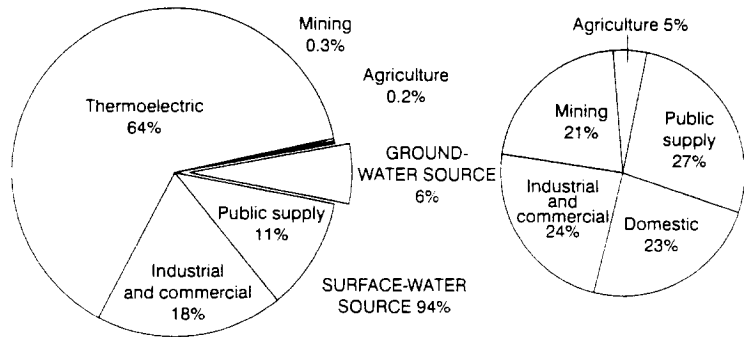


Figure 2. In 1995, water withdrawn averaged 3,284 million gallons per day. In the Pittsburgh area, nearly all water used for public supply is surface water. Ground water provides water for domestic use in most rural areas.

length of the Monongahela River and the lower 72 miles of the Allegheny River are maintained for navigation by dams. During dry periods, low streamflows are augmented by reservoir releases to dilute degraded water (Ohio River Basin Commission, 1980).

Nonconsumptive use of the water resource also is extensive in ALMN. Some streams are managed for whitewater sports, boating, or fishing, and some high-quality stream reaches are important for conservation and

management of endangered species.

Hydrologic conditions

Although streamflow roughly followed normal patterns during 1996–98, flows were substantially higher or lower than normal for short periods in response to weather extremes (fig. 4). Hence, the ALMN water-quality data set includes responses to a wide range of flows while still being largely representative of normal conditions.

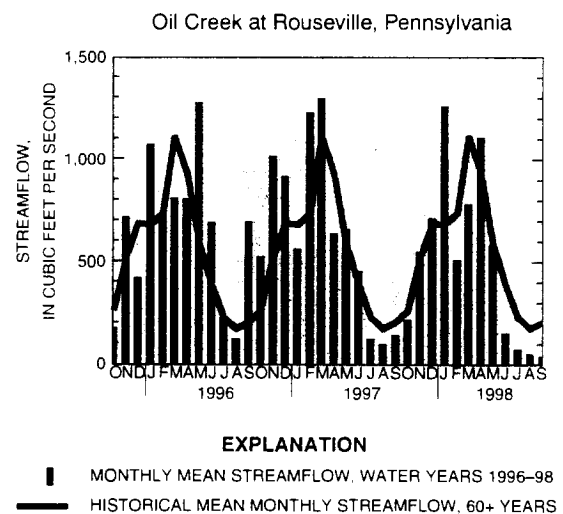


Figure 4. Streamflow was above average in 1996–97 and below average in summer 1998.

MAJOR FINDINGS

The quality of streams, rivers, and ground water reflects complex interactions of natural and human-induced conditions. Natural watershed scale factors such as climate, geography, and topography influence water chemistry and aquatic biological communities. Broad-scale land uses, as well as localized human activities combine with background conditions to influence overall water quality.

Within ALMN, the interaction of a diverse geography and an equally diverse set of land uses influence the quality of the water resource. Surface-water sampling sites were selected in a variety of land-use settings including forest, urban, agriculture, mining, and mixed land use. The study design for ground water focused on assessing the water-quality conditions of major aquifers in ALMN, with emphasis on the quality of recently recharged ground water associated with ongoing and recent human activities (see page 22) (Gilliom and others, 1995). Specific findings from particular land uses and geographic settings are presented in the rest of this part of the report.

Coal Mining Dominates Water Quality

Although not easily represented on land-use maps, mining has the greatest influence on surface and ground-water quality and aquatic habitat of any single land use in ALMN. The area of surface mined land is difficult to quantify because of revegetation: deep-mine activity leaves virtually no trace on the surface.

Coal has been mined in ALMN for more than 200 years and has been central to the economy and lifestyle of many communities. Extensive commercial coal mining

began with almost no concern for the protection of the land surface and water resources. Consequently, stream-water quality in much of ALMN was severely degraded—streams became virtually unusable and supported few aquatic species. Mine-related influences have long been recognized as among the most serious and persistent water-quality problems in Pennsylvania (Pennsylvania Department of Environmental Protection, 1996) and West Virginia (West Virginia Department of Environmental Protection, 1998), as well as throughout Appalachia, extending from New York to Alabama (Biesecker and George, 1966).

Surface and underground coal mining and coal-cleaning processes expose many elements to weathering. Pyrite and marcasite (iron disulfides also known as “fool’s gold”) are naturally occurring compounds commonly found in coal

Nearly all basins greater than 100 square miles within the coal-bearing region of ALMN have been mined at one time or another, many with several coal-extraction techniques.

and in overburden rock. Pyrite is the major source of acid mine drainage (AMD) in the Eastern United States (Rose and Cravotta, 1998). During or after mining, AMD can be formed by a series of complex geochemical and bacterial reactions that occur when pyrite is exposed to air and water (Pennsylvania Department of Environmental Protection, 1999) (fig. 5). Through these reactions, some dissolved ferrous iron will precipitate out of solution in the form of insoluble ferric hydroxide (fig. 6).

Secondary reactions of the acidic water can bring into solution other constituents in the coal and the overburden rock, such as manganese, aluminum, zinc, arsenic, bar-

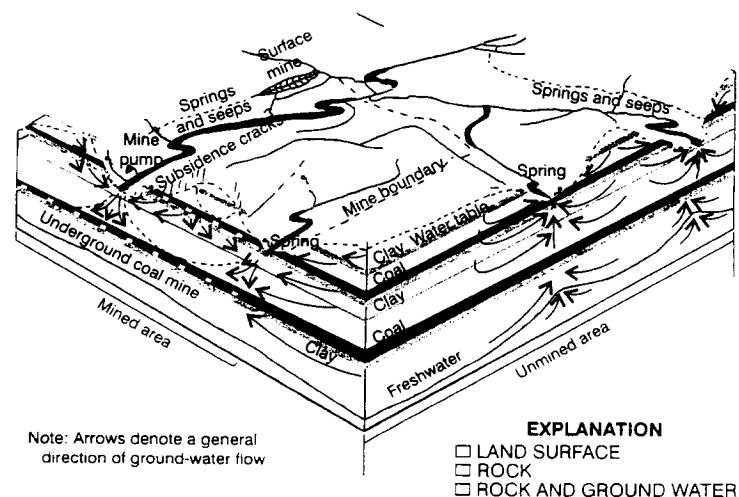


Figure 5. Coal mines disrupt existing flow patterns of ground water and surface water. Oxygen dissolved in surface water is transported to rock strata containing pyrite. Sulfuric acid is produced, which may then emerge in springs, seeps, and streams carrying large amounts of dissolved metals. (Figure adapted from Puente and Atkins, 1989.)



Figure 6. Reddish-orange iron precipitate is commonly seen in streams affected by acid mine drainage.

ium, cadmium, cobalt, copper, and silver. Some reach concentrations potentially dangerous to wildlife and may exceed drinking-water standards (table 1). These constituents are then subject to additional reactions (such as the formation of precipitates), are adsorbed onto sediments, or are taken up in the tissues of organisms (bioconcentrated).

Sulfate is an Indicator of Coal-Mining Activity

Coal-mine drainage can be acidic or alkaline and can seriously degrade both surface- and ground-water supplies. AMD, in which the acidity exceeds alkalinity, typically contains elevated concentrations of sulfate, iron, manganese, aluminum, and other dissolved material. In contrast, neutralized or alkaline mine drainage (NAMD) has an alkalinity that exceeds acidity; however, NAMD can still have elevated concentrations of sulfate, iron, manganese, and other constituents.

Dissolved trace elements are not generally reliable indicators of AMD or NAMD because they may not remain in solution. Sulfate, however, is a reliable indicator of mine drainage because sulfate is highly soluble and chemically stable at the pH levels normally found in natural waters (Hem, 1985).

The U.S. Environmental Protection Agency (USEPA) has established a Secondary Maximum Contaminant Level (SMCL) of 250 mg/L (milligrams per liter) for sulfate. SMCLs are applied to public water supplies and are nonenforceable levels for contaminants that may affect the taste, odor, or appearance of water. High sulfate concentrations in water may cause diarrhea in sensitive populations (U.S. Environmental Protection Agency, 1999a).

The amount of a constituent carried out of a stream system is called the yield. Sulfate yields were, on average, 5 times greater in stream basins where mining has occurred than in unmined basins sampled monthly in 1997–98 (fig. 7). With one exception (Stonycreek River), yields of dissolved iron and dissolved manganese were similar in mined and unmined basins.

The Stonycreek River had the highest sulfate yield of the 11 sampled streams and is considered to be highly degraded by AMD, primarily from abandoned mines.

Table 1. Regional background concentrations of constituents influenced by mine drainage were estimated by using the 90th percentile for each constituent from streams unaffected by mining

[USEPA, U.S. Environmental Protection Agency. mg/L, milligrams per liter; --, not calculated; µg/L, micrograms per liter;]

Selected mining constituents ¹	USEPA Secondary Maximum Contaminant Level	Regional background concentration
Dissolved solids	500 mg/L	--
pH	6.5–8.5	7.0
Sulfate	250 mg/L	20.8 mg/L
Iron	300 µg/L	129 µg/L
Manganese	50 µg/L	81 µg/L
Aluminum	50–200 µg/L	23 µg/L

¹Other coal-mining-related constituents include alkalinity and acidity.

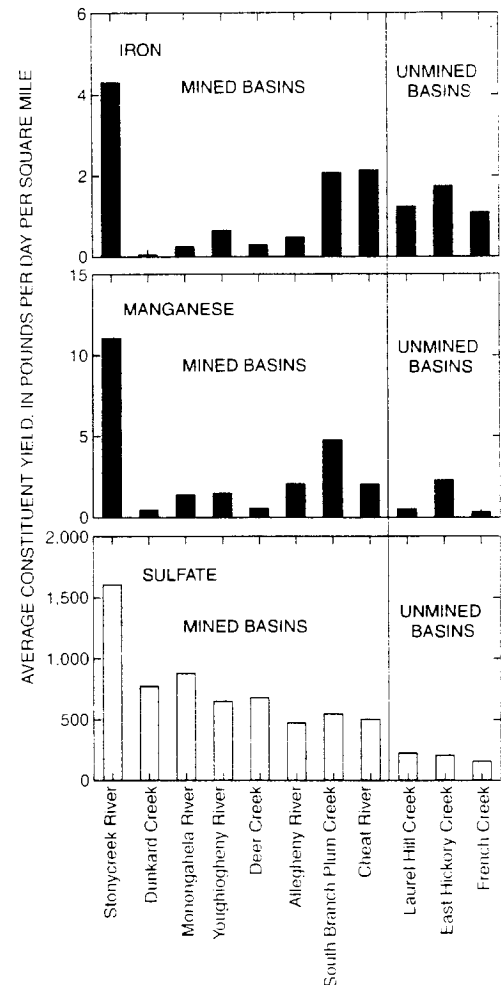


Figure 7. Sulfate is a more stable indicator of mine activity than dissolved iron or manganese.

Currently, efforts are being made to restore the water quality in this river, mainly through the construction of passive treatment systems to treat abandoned-mine discharges inventoried in 1992–95 (Williams and others, 1996). Since 1995, about \$3.5 million has been spent on mine-drainage remediation projects throughout the Stonycreek River Basin, resulting in the removal of iron, aluminum, and acidity from the Stonycreek River at a rate of 111, 133, and 1,192 tons per year, respectively (D. Seibert, Natural Resources Conservation Service, oral commun., 2000). A similar study to identify mine discharges was completed in a Monongahela River tributary, the Cheat River (Williams and others, 1999).

Aquatic Communities are Affected in Streams Receiving Large Amounts of Mine Drainage

Streams receiving mine drainage may range from supporting diverse communities of aquatic life to being lethal to many organisms, depending on a variety of factors. The ecological setting of a stream can affect the types and rates of water-quality changes in response to human influences. Ecoregions and basin size are two factors that relate to differences in aquatic communities. Ecoregions are used to group areas that are ecologically similar and can be expected to have similar aquatic communities. ALMN is divided into six ecoregions (fig. 8), five of which were included in the sampling design in ALMN. Basin size affects species diversity because larger basins tend to have a greater variety of habitats available.

The fish community was sampled at 11 sites in ALMN, 7 of which received mine drainage. A difference in fish abundance and number of fish species was evident between streams in mined basins compared to those in unmined basins. None of the streams sampled had a depressed pH (less than 6.5). In the Central Appalachian Ecoregion, at Stonycreek River, only 2 species (2 individuals) were captured, whereas in Laurel Hill Creek, a similar stream in a nearby unmined area, 16 species (384 individuals) were captured. Where basin sizes were comparable, the presence or absence of coal mining in a basin was evident in some aspects of the fish-community structure (fig. 9).

Few organisms can tolerate even brief periods of acidic or mineral- or silt-laden water. Episodic events or chronic conditions that result in concentrated AMD entering a stream are obvious and result in a nearly complete loss of aquatic species, such as in Stonycreek River. The effects are often more subtle in streams receiving NAMD, where species sensitive to sedimentation, trace-element concentration, or hydrologic changes are affected (Letterman and Mitsch, 1978). In a regional study between ALMN and the Kanawha–New River Basin, 61 sites were sampled for aquatic invertebrates (insects, worms, crustaceans, and mollusks) and water chemistry during a low-flow period in 1998. At sites where sulfate con-

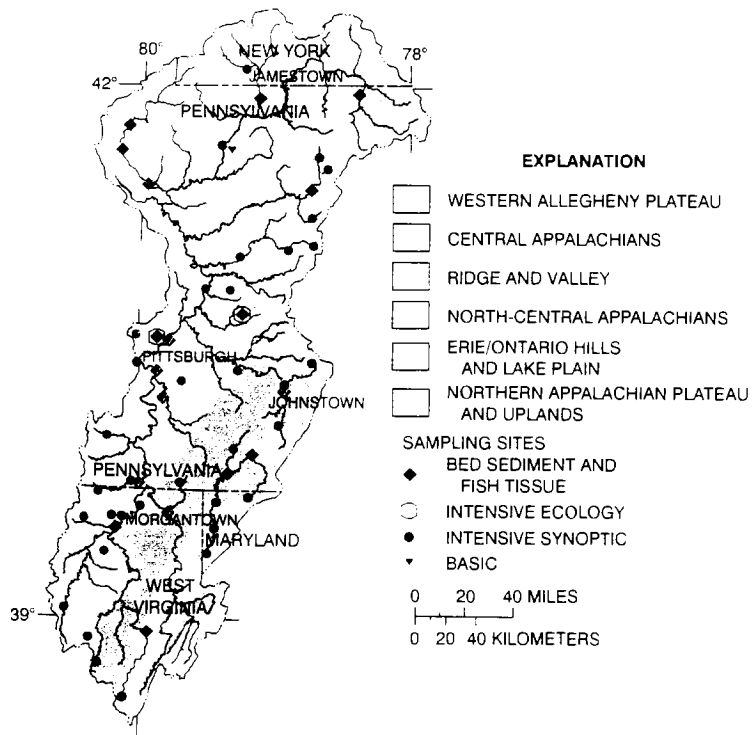


Figure 8. Assessments of the health and condition of aquatic life and habitat focused on four of the six ecoregions represented on this map. Contaminants associated with bed sediment and tissue were analyzed at 19 sites. (Ecoregions from Omernick, 1987)

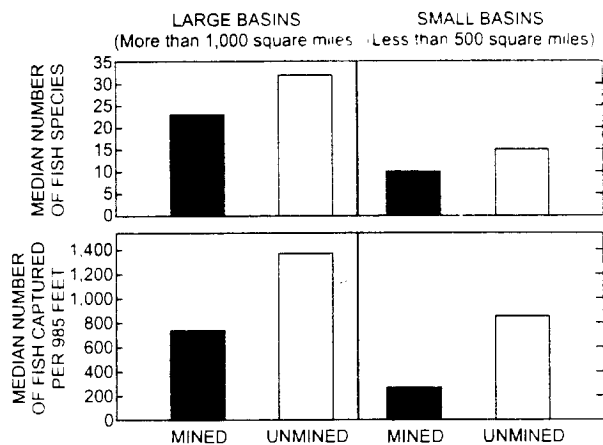


Figure 9. The number of fish species and the number of individual fish captured per 985-foot stream reach was greater at unmined sites than at mined sites of comparable size.

centrations were greater than the estimated background level. decreasing diversity was noted for three groups of sensitive insect species (mayflies, stoneflies, and caddisflies), although pH was 6.5 or greater at all these sites. (See fig. 23 on page 21.)

INVERTEBRATE INDEX STATUS IS AMONG THE BEST IN THE NATION IN FORESTED SETTINGS

Aquatic life in stream systems where human influence is minimal generally represents a more natural community than in streams strongly influenced by human activity. These sites can be used to define background (reference) conditions that are helpful in interpreting how various land uses change the types and numbers of organisms living downstream. NAWQA examines fish, invertebrate, and algal communities and uses indices based on reference sites as part of assessing water quality. For example, an invertebrate status index (T.F. Cuffney, U.S. Geological Survey, written commun., 2000) averaged 11 invertebrate-community measures (metrics) used to indicate various aspects of the life cycles of the organisms assessed. This index can be used to make relative comparisons between sites sampled by NAWQA.

An ALMN site, East Hickory Creek near Queen, Pa., whose basin is more than 95-percent forested, had the best quality (lowest invertebrate index score) nationally of 140 sites sampled between 1996 and 1998 (Appendix). In contrast, streams in either urban or coal-mine settings ranked among the highest 25 percent of those sampled.

A National Ranking of INVERTEBRATE STATUS in Streams

EXPLANATION
 INVERTEBRATE STATUS INDEX - Higher values suggest a more degraded stream site
 ● Highest 25 percent
 ⊙ Middle 50 percent
 ○ Lowest 25 percent

Ground-Water Quality is Affected Near Mined Areas

During 1996–98, 45 domestic water-supply wells were sampled in ALMN in the high-sulfur coal region of the Appalachian coal fields (Tully, 1996). Water samples were collected from 30 of the 45 wells within about 2,000 feet and hydrologically downgradient (downhill in this area) from a reclaimed surface coal mine. The additional 15 wells are in areas believed to be unmined.

Analysis of ground-water data indicates that surface coal mining continues to affect ground-water quality after all mining and reclamation has ceased. Several constituents related to mine drainage exceeded the USEPA SMCL more frequently in water sampled from wells downgradient from reclaimed surface coal mines than in well water from unmined areas.

Sulfate concentrations exceeded the USEPA SMCL for sulfate (250 mg/L) at 20 percent of domestic wells sampled in mined areas but at no wells sampled in unmined areas. Iron concentrations at wells near mined areas exceeded the SMCL (300 µg/L [micrograms per liter]) in 60 percent of the wells, compared to 20 percent of wells in unmined areas. Similarly, manganese concentrations exceeded the SMCL (50 µg/L) in 70 percent of wells from mined areas compared to 47 percent of wells in unmined areas. Finally, samples from 20 percent of the wells in mined areas exceeded the SMCL for total dissolved solids, whereas samples from only 7 percent of the wells in unmined areas exceeded the SMCL.

Concentrations of mine-related constituents, such as sulfate, iron,

manganese, and dissolved solids can exceed SMCLs for drinking water in unmined areas because of the geologic setting (mostly rocks of Pennsylvanian age that can contain high concentrations of iron and manganese). High concentrations of sulfate in ground water of the Appalachian coal fields, however, usually indicates that coal has been mined nearby or in a location hydrologically upgradient from the sample location. Current regulations do not require treatment of mine-discharge water for sulfate. Discharge water is generally regulated and treated to reduce concentrations of iron and manganese and to maintain pH in the range of 6.5 to 8.5 units.

Concentrations of Trace Elements in Bed Sediment Exceed Aquatic-Life Guidelines

Trace elements typically are present in surface-water systems in small amounts. Local geologic conditions or land-use activities can increase the concentration of some elements to levels that may impair aquatic life or limit water use. Trace elements may be dissolved in water, bound to sediments, or incorporated into the tissues of organisms, depending on the chemical properties of each element. In ALMN, several trace elements in addition to zinc and chromium were detected at high concentrations in bed sediment or tissues (Appendix).

Arsenic is a trace element that is potentially damaging to both human health and aquatic life. Increased arsenic concentrations can result from human activity, such as application of pesticides or the combustion of fossil fuel, or from natural weathering of arsenic-

bearing rock (Ferguson and Gavis, 1972). Arsenic was detected at concentrations above the estimated background concentration of 5.9 µg/g (micrograms per gram) (Canadian Council of Ministers of the Environment, 1995) at all 50 bed-sediment sites sampled between 1996 and 1998 in ALMN. The Probable Effect Level (PEL) for arsenic in bed sediment of 17 µg/g (Canadian Council of Ministers of the Environment, 1995) was exceeded at 12 of 50 sites, where concentrations ranged from 18 to 52 µg/g.

Land use did not appear to be a factor in the arsenic concentrations observed in ALMN, although atmospheric deposition cannot be ruled out. Each of the sites in ALMN where the PEL was exceeded, with the exception of Stonycreek River (a heavily mined basin), were distributed in the northern, once glaciated part of the Allegheny River Basin. Glacial action during the last ice age broke up and moved near-surface rock, exposing this rock to weathering and releasing some arsenic (Welch and others, 1988).

In contrast, concentrations of some other trace elements in ALMN appear to be related to land use. Concentrations of cadmium, copper, chromium, lead, mercury, and zinc each exceeded the PEL aquatic-life guidelines in bed-sediment samples at least once in samples from mined or mixed-land-use sites.

Concentrations of cadmium in whole-fish samples, for which no guidelines exist, are among the highest sampled by NAWQA during 1995–98. Several trace elements (such as nickel) that also have no established guidelines for either bed sediment or tissue are



TRACE ELEMENTS MAY LIMIT AQUATIC LIFE IN URBAN STREAMS AND MINED AREAS

The acidity of some mine drainage may dissolve and subsequently transport large amounts of trace elements from exposed rock. These trace elements, often found naturally in small amounts, can accumulate in streambed sediments. Trace elements, low levels of which are required by organisms, can reach toxic concentrations when concentrated in food, water, or sediments.

Aquatic-life guidelines, used as a reference level, are based on Environment Canada's guideline (Canadian Council of Ministers of the Environment, 1995) and have no regulatory force in the United States. Zinc and chromium were found at all bed-sediment sampling sites in ALMN, and at the 50 sites sampled, the aquatic-life Probable Effects Level (PEL) for zinc (315 µg/g) and chromium (90.0 µg/g) was exceeded at 15 and at 5 sites, respectively (Appendix). Eleven bed-sediment samples from ALMN had zinc concentrations among the highest 10 percent nationally of samples analyzed by NAWQA since 1991. PELs were most often exceeded in areas subjected to industrial or mining land use in ALMN.

Zinc, along with other trace elements that exceed aquatic-life guidelines, may contribute to degradation of aquatic communities in streams. Some sites in ALMN were among the most degraded sites nationally for aquatic invertebrates (Appendix).

National indicators for invertebrate status (Appendix) with zinc and chromium concentrations in bed sediment, in micrograms per gram of sediment

Stream name and location	Predominant land use	Invertebrate status	Zinc	Chromium
French Creek at Utica, Pa.	Mixed		120	58
East Hickory Creek near Queen, Pa.	Forested		190	63
South Branch Pium Creek at Five Points, Pa.	Agriculture		130	82
Deer Creek near Dorseyville, Pa.	Urban		170	88
Dunkard Creek at Shannopin, Pa.	Mining		190	88
Youghiogheny River at Sutersville, Pa.	Mixed		410	87
Stonycreek River at Ferndale, Pa.	Mining	●	700	90
Monongahela River at Braddock, Pa.	Mixed	●	510	110
Allegheny River at New Kensington, Pa.	Mixed	●	330	120
lowest 25 percent nationally (Least-degraded sites)				
middle 50 percent nationally				
● highest 25 percent nationally (Most-degraded sites)				

elevated in mixed-land-use settings (Appendix).

Water Quality of the Large Rivers of the Allegheny and Monongahela River Basins is Improving

The large rivers sampled in ALMN are important for their envi-

ronmental and esthetic qualities, as well as sources for drinking water. Sections of the upper Allegheny River are designated federally as Scenic Rivers (Pennsylvania Department of Conservation and Natural Resources, 2000). French Creek, a tributary of the Allegheny River, supports several State and

federally protected endangered species (see page 12) and is an important stream nationally for the protection of aquatic species (Masters and others, 1998). Whitewater rafting on the Youghiogheny and Cheat Rivers is a thriving recreational industry.

The water quality in a river that drains large areas integrates water potentially influenced by a broad range of natural and human factors. The industrial and resource extraction land-use history in ALMN previously resulted in poor water quality in some rivers and streams. Early in the 1900s, fish were rarely found in the lower Allegheny and Monongahela Rivers and then only during high flows, when river water was diluted by surface runoff. Crayfish also were rare, and freshwater mussels had been eliminated (Ortmann, 1909). Ortmann described lower reaches of Monongahela River tributaries, the Cheat River and Youghiogheny River, as degraded by mine drainage. As recently as the mid-1960s, fish surveys on the Monongahela River found zero to four fish species (U.S. Army Corps of Engineers, 1976).

The Allegheny River and Monongahela River sites sampled in this study have been sampled comparably under various USGS programs since the early 1970s, permitting a general comparison of water-quality conditions since that time.

A measure of the acidic and basic properties of natural waters is pH. The pH of source water is useful for determining water-treatment

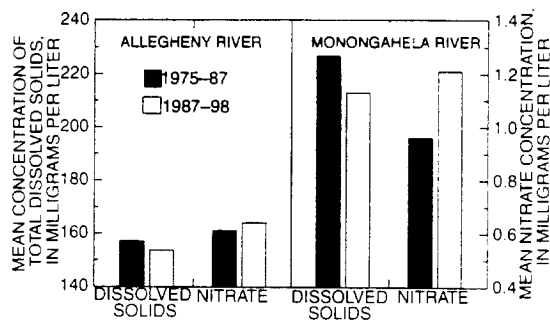


Figure 10. For the two 12-year periods examined, the Allegheny and Monongahela Rivers improved in some water-quality respects (MCL, Maximum Contaminant Level; SMCL, Secondary Maximum Contaminant Level).

options and evaluating the suitability for support of aquatic plants and animals. Natural factors, such as rock types in a basin, can affect pH, as can industrial discharges and mine drainage. As recently as the 1960s, the Monongahela River was occasionally too acidic (low pH) to support a diverse aquatic community (Finni, 1988). Since the early 1970s, the median pH at the NAWQA sampling sites increased from 7.0 in the period 1975 to 1987 to 7.4 in the period 1987 to 1998 in the Allegheny River. During the same periods, the median pH increased from 7.0 to 7.6 in the Monongahela River. Although this represents an overall increase in pH for both sites, about 1 percent of the samples collected from these two sites had pH values that were lower than the minimum aquatic-life water-quality guideline of 6.0 (Pennsylvania Department of Environmental Resources, 1984) during the period 1987 through 1998. For organisms living in these rivers, occasional periods when the pH is either too high or too low can be lethal for a particular species.

trations have decreased by 2 percent in the Allegheny River and by 6 percent in the Monongahela River. Reductions in dissolved solids in the Monongahela River have virtually eliminated the exceedences of the SMCL of 500 mg/L (fig. 10).

Elevated nitrate concentrations can result in increased plant and algal growth (U.S. Geological Survey, 1999a), which can, in turn, alter the taste of water and affect other aquatic life. Nitrate increases can be related to some of the same sources as dissolved solids, including both point-source discharges, such as industrial wastewater discharges and sewage, or nonpoint sources, including atmospheric deposition and agricultural fertilizer use (U.S. Geological Survey, 1999a). In contrast to dissolved solids, however, median nitrate concentrations have increased by 3 percent in the Allegheny River and by 25 percent in the Monongahela River. Nitrate, which contains nitrogen, can be converted to other nitrogen-containing compounds relatively easily. Total nitrogen was not routinely measured in early studies. The increase observed in

The concentration of dissolved solids in a water body can be increased as a result of industrial or municipal wastes, drainage from mines or oil fields, or drainage from agricultural land. Median concentrations of dissolved solids have decreased at the Allegheny and Monongahela River sampling sites over the last 25 years. Dis-

nitrate may be partly the result of changes in the form of nitrogen in the rivers, typically due to sewage-treatment-plant upgrades (U.S. Geological Survey, 1999a).

The general improvement in water quality described above in sections of the Allegheny and Monongahela Rivers has been accompanied by an increase in the number and species diversity of fishes. A sample of the fish community at the Monongahela River site in 1998 contained more than 1,100 individual fish representing 12 species. This included many sport fish such as smallmouth bass and sauger. Species richness was even greater in the Allegheny River, which had 21 species, again including many sport fish as well as species sensitive to pollution, such as redhorse sucker. Significantly, the silver chub, *Macrhybopsis storeriana*, a minnow that had not been seen in these rivers since the late 1800s (Cooper, 1983), was captured in both 1997 and 1998. The recovery of rare species is a further indication of the degree of improvement in water quality in these river segments.

Persistent Pesticides and PCBs were More Prevalent in Fish Tissue than in Sediment

Numerous synthetic organic compounds have been manufactured to fulfill various needs of society. These compounds have a range of stability in the environment. Some break down rapidly, whereas others can be highly stable and persistent. Some stable synthetic organic compounds are no longer in use in the United States.

Organochlorines in this group are commonly soluble in fat or can bond to particles in the water and settle out. Some bioconcentrate in

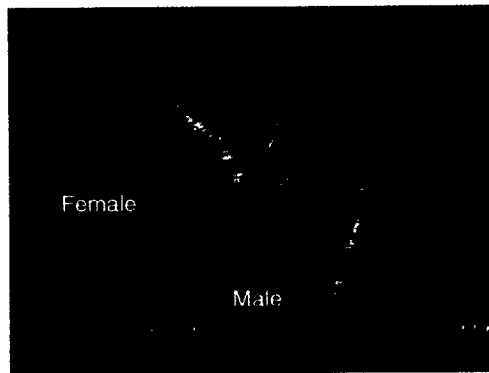
Grass-roots efforts seek to protect water quality and aquatic life

Many of the water-quality issues receiving national attention arise from landscape-scale activities known as nonpoint sources. Managing nonpoint sources often requires the cooperation of many people living in a river basin.

Watershed groups have been formed by citizens concerned with the conditions in local waterways. Nationally, as well as in ALMN, these groups are increasingly important. Many include partnerships with conservation groups and government agencies.

Many watershed organizations in ALMN are working to improve degraded streams. In contrast, several local groups are actively working to keep their resource in good condition. French Creek is an example of a stream that has maintained high water quality and whose citizens are working to maintain this resource. This creek, and its associated ground water,

supply drinking water to many homes, municipalities, and industries. French Creek also boasts the greatest number and variety of fish, invertebrates, and aquatic plants in ALMN. Many of these species are categorized as endangered by State and Federal governments. Several are found nowhere else in ALMN and at very few other places in the world (see photograph at right). These rare, sensitive species and the high overall species diversity in French Creek are evidence of the high quality of the water and stream habitat in this basin.



Northern riffleshell is a globally endangered species doing well in French Creek and sections of the Allegheny River. Two-thirds of related native North American freshwater mussels are now rare or endangered. These animals are indicators of exceptional stream quality.

fat, reaching higher concentrations in organisms than in the environment. They can accumulate in predators that eat contaminated organisms. In the tissues of animals, these compounds can have a variety of effects including toxicity, reproductive impairment, or cancer. Whole fish from 16 sites in ALMN were analyzed for 28 organochlorine compounds. Streambed sediment was analyzed for 32 compounds at these same 16 sites plus an additional 3 sites (fig. 8). At the sites where both fish and bed sediment were sampled, those compounds detected in both media were present at higher con-

centrations in fish tissue than in the sediment (Appendix).

Although use of PCBs was discontinued in the United States in the 1970s, PCBs were detected in whole-fish tissue samples at 10 of 16 sites in 1997. This mixture of compounds was detected at only 4 of 19 sites in streambed sediment. Those sites at which PCBs were below the detection level in fish tissue were in basins dominated by agriculture or forest. Fish-tissue samples from eight sites with mixed land use had total PCB concentrations above the guideline for protection of fish-eating wildlife (Newell and others, 1987).

DDT and chlordane use has been discontinued in the United States since the 1970s. DDT and its breakdown products were detected in fish at 15 of 16 sites and in sediment at 9 of 19 sites. Only at the Allegheny River at New Kensington, Pa., however, did the concentration of total DDT in fish samples exceed the guideline of 200 mg/kg (micrograms per kilogram) established to protect fish-eating wildlife. Chlordane was detected in 11 of 16 whole-fish samples and in 4 of 19 streambed-sediment samples. The guideline of 500 mg/kg for total chlordane (which also includes breakdown products) for protection of fish-eating wildlife was exceeded in fish samples only at the Monongahela River near Braddock, Pa.

Public-health advisories are in place to restrict consumption or prohibit taking of several fish species from certain sections of the Allegheny and Monongahela Rivers because of PCB and chlordane contamination (Pennsylvania Fish and Boat Commission, 1999). These compounds are relatively stable, are apparently being cycled between aquatic life and bed sediment, and may persist in ALMN for many more years.

Low Concentrations of Numerous Pesticides were Detected in an Agricultural Stream and an Urban Stream

Two basins of similar size were chosen to assess the occurrence and distribution of a broad range of pesticides under different stream-flow conditions. The Deer Creek Basin represented a predominantly residential/urban setting, and the South Branch Plum Creek Basin represented a predominantly agri-

cultural setting (Williams and Clark, 2001).

Of the 84 pesticides and pesticide metabolites (breakdown products) in this analysis, 25 were detected at least once in Deer Creek and 20 were detected at least once in South Branch Plum Creek. Some pesticides show a seasonal pattern in water samples from both streams (fig. 11).

All detectable pesticide concentrations from both streams were less than drinking-water-quality guidelines or standards (table 2). However, the maximum measured concentrations of diazinon in Deer Creek (0.097 $\mu\text{g/L}$) and South Branch Plum Creek (0.094 $\mu\text{g/L}$) exceeded the water-quality guideline to protect aquatic life of 0.08 $\mu\text{g/L}$.

Prometon is the most commonly detected herbicide in surface water and ground water in urban areas

A note on National biological status scores

Although water-quality guidelines for the protection of aquatic life were exceeded for several of the pesticides detected in ALMN, there is no indication that the concentrations have been lethal to the organisms in these streams. National invertebrate and algal scores indicate that these biological communities have not been degraded and are comparable to those at a forested site in ALMN (Appendix). The national fish status score, although indicating that the urban and agricultural setting have better quality fish communities than the forested site, places considerable emphasis on non-native fish species. The forested site is stocked with non-native trout to supplement sport fishing. Abundant non-native fish populations are an indicator of human influence and may point to habitat or water-quality degradation in other situations.

(Capel and others, 1999). It is used as a preemergent herbicide to control vegetation on bare ground around buildings and fences, along rights-of-way, and in conjunction with the application of asphalt. Prometon was detected in 90 percent of the samples collected in Deer Creek. The highest measured concentration was 0.355 $\mu\text{g/L}$ in the first of five storm samples collected on August 25–26, 1998. That concentration was more than 10 times the maximum measured concentration in 1997 but is still well below the drinking-water-quality guideline of 100 $\mu\text{g/L}$. No prometon guidelines have been established for the protection of aquatic life.

The insecticide diazinon is commonly used in homes, gardens,

parks, and commercial areas. Detections of diazinon from samples collected in Deer Creek in 1997 showed no seasonal pattern; however, five of the seven detections were in samples collected shortly after a peak in streamflow due to overland runoff.

The aquatic-life water-quality guideline for carbaryl of 0.2 $\mu\text{g/L}$ was exceeded in four of the five stormflow samples collected in Deer Creek on August 25–26, 1998.

Pesticides are at Low Concentrations when Detected in Ground Water

Ground-water samples from 58 shallow domestic wells throughout ALMN were analyzed for pes-

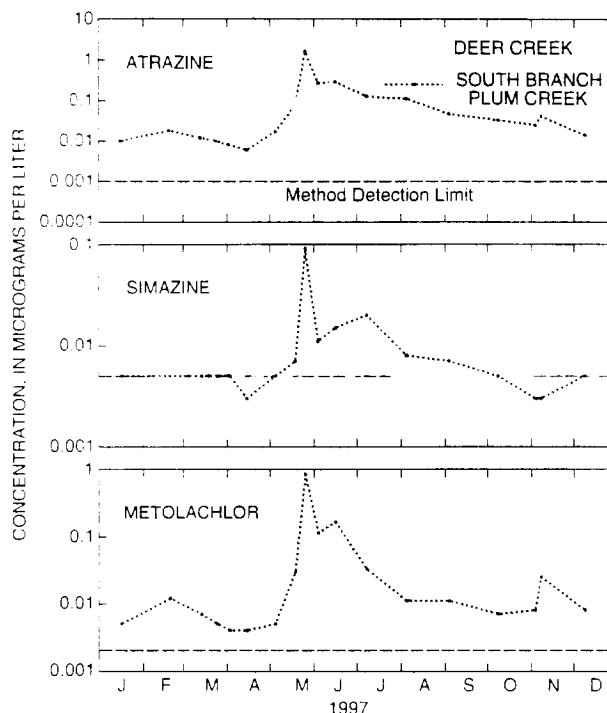


Figure 11. A distinct seasonal pattern is evident in the concentrations of atrazine, simazine, and metolachlor. The peak concentrations of these three pesticides coincide with herbicide-application periods and increased spring rainfall. (Concentrations below the method detection limit are believed to be reliable detections but with greater than average uncertainty in quantification.)

Table 2. Many pesticides are in widespread use for the control of insects (insecticides) or plants (herbicides). Pesticides may be sold under a variety of names, depending on the manufacturer (Table adapted from U.S. Geological Survey, 1999b) [$\mu\text{g/L}$, micrograms per liter; NA, not available]

Pesticide name	Trade name	Use	Drinking-water-quality guidelines or standards ($\mu\text{g/L}$)	Aquatic-life water-quality guideline ($\mu\text{g/L}$)
Atrazine	AAtrex, Atrex, Atred, Gesaprim	Herbicide	3 ¹	1.8
Diazinon	Basudin, Diazatol, Neocidol, Knox Out	Insecticide	.6	.08
Dieldrin	Panoram D-31, Octalox, Compound 497, Aldrin epoxide	Insecticide	.02	.056
Carbaryl	Carbamine, Denapon, Sevin	Insecticide	700	.2
Metolachlor	Dual, Pennant	Herbicide	70	7.8
Prometon	Pramitol, Princep, Gesagram 50, Ontracic 80	Herbicide	100	NA
Simazine	Princep, Caliber 90, Gesatop, Simazat	Herbicide	4 ¹	10

¹Drinking water-quality standard (Maximum contaminant level).

ticides. One to five pesticide compounds were detected in 34 percent of the samples. Nine different compounds were detected at concentrations ranging from less than 0.001 to 0.17 $\mu\text{g/L}$. All detections were at or below the method-detection limit. No compounds were detected above drinking water-quality guidelines or standards. The five most frequently detected compounds were the agricultural herbicides atrazine, metribuzin, and metolachlor; the insecticide diazinon; and a breakdown product of atrazine, deethylatrazine.

Of the 58 ground-water samples analyzed for pesticides, 30 samples were from wells in valley-fill aquifers and 28 samples were from fractured-rock aquifers (see page 22). Forty percent of the samples from valley-fill aquifers and 29 percent of the samples from fractured-rock aquifers contained at least one pesticide compound. Deethylatrazine was the only pesticide detected in more than 30 percent of all samples in the valley-fill aquifers. No pesticides were detected in more than 22 percent of

sampled wells in the fractured-rock aquifers.

Nine different pesticide compounds were detected in 12 samples from the valley-fill aquifers (fig. 12). The top four detected

compounds in samples from valley-fill aquifers were deethylatrazine, atrazine, metribuzin, and diazinon. Two or more pesticide compounds were detected in 20 percent of the samples in the valley-fill aquifers.

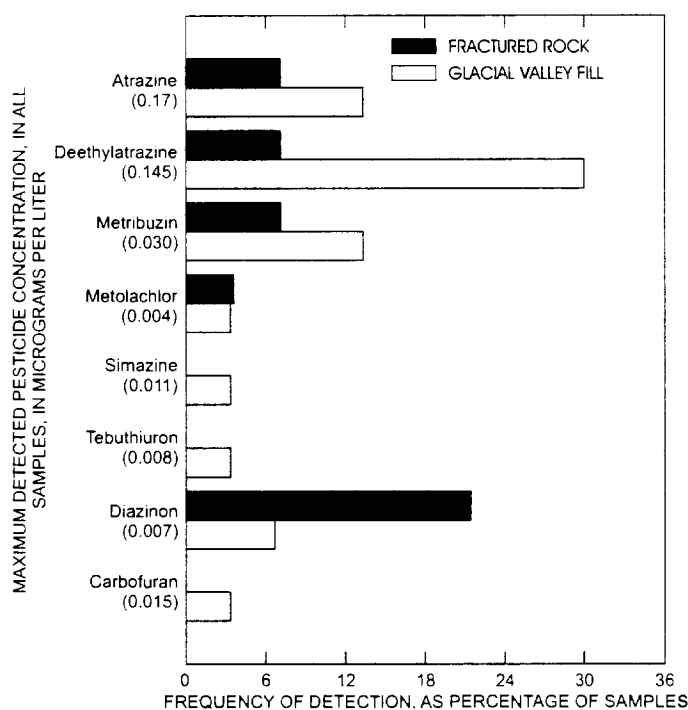
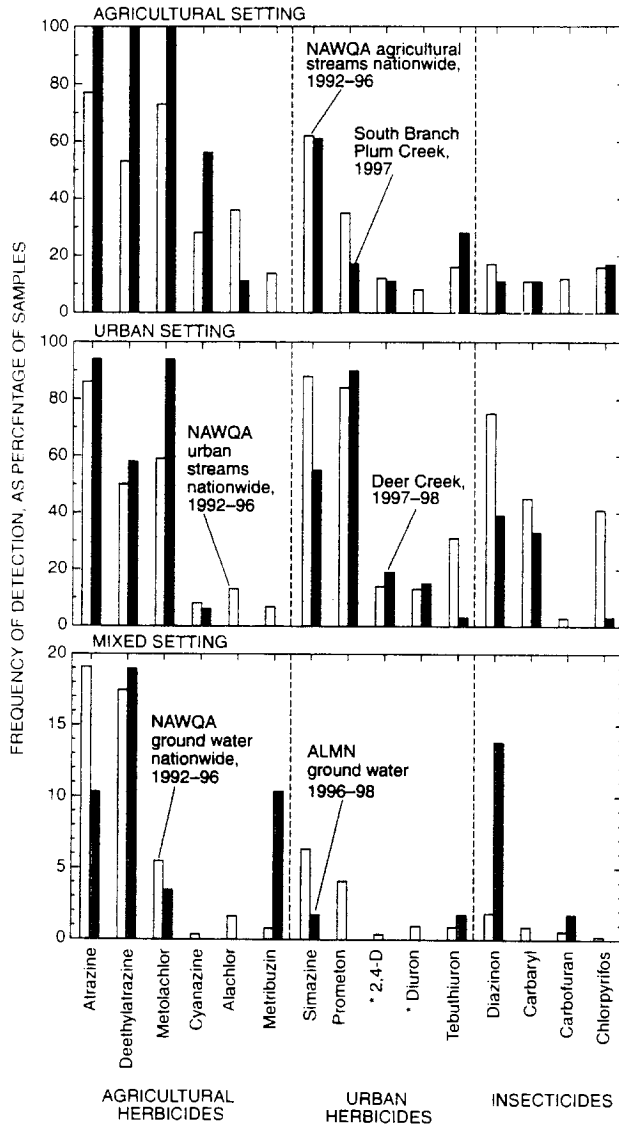


Figure 12. With exception of diazinon, pesticide-detection frequencies in ground water were higher in the valley-fill aquifers than in the fractured-rock aquifers. (Not shown above is the herbicide EPTC detected in a single sample—0.004 $\mu\text{g/L}$.)



PESTICIDES IN THE ALLEGHENY AND MONONGAHELA RIVER BASINS ARE SIMILAR TO THOSE DETECTED NATIONWIDE

The commonly detected pesticides in South Branch Plum Creek, an agricultural basin, and in Deer Creek, an urban basin, were similar to the 15 most commonly detected pesticides in streams in NAWQA studies during 1992–96. The compounds detected in ground water from wells set in both valley-fill aquifers and fractured-rock aquifers are also among those most frequently detected in mixed land-use aquifers nationwide.



*Ground-water samples were not analyzed for the herbicides 2,4-D and Diuron in the Allegheny and Monongahela River Basins.

The most frequently detected mixture of compounds was atrazine (or the metabolite deethylatrazine) and metribuzin, detected in 10 percent of samples from the valley-fill aquifers.

Five different pesticide compounds were detected in the samples from the fractured-rock aquifers (fig. 12). The four most frequently detected compounds in samples from the fractured-rock aquifers were diazinon, deethylatrazine, atrazine, and metribuzin. Two or more pesticide compounds were detected in 14 percent of the samples in the fractured-rock aquifers. The most frequently detected mixture of compounds was metribuzin and diazinon, found in 7 percent of the samples from fractured-rock aquifers.

The higher detection frequency of pesticides in the samples from the valley-fill aquifers is most likely a result of greater vulnerability to pesticide contamination due to permeability of aquifer material and contaminant availability (Lindsey and Bickford, 1999). Both aquifers have similar contaminant-availability ratings; however, the valley-fill aquifers consist of unconsolidated sediments and are more permeable than the fractured-rock aquifers.

Volatile Organic Compounds were Detected at Low Concentrations in an Urban Stream

Volatile organic compounds derived from substances commonly used in residential and urban areas, such as gasoline and cleaning solvents, were detected in 24 of the 25 samples collected from Deer Creek (Pittsburgh metropolitan area) in 1997–98. Of the 87 VOCs

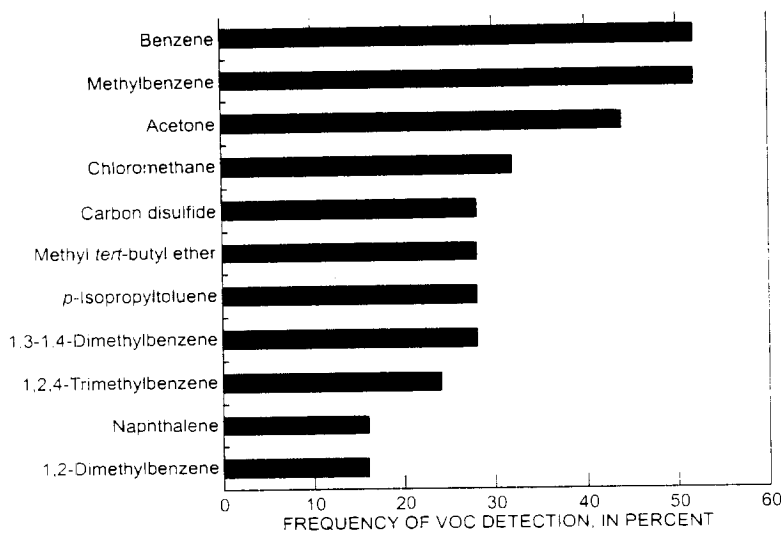


Figure 13. Of 87 VOCs analyzed for, 22 were detected in Deer Creek at Dorseyville, Pa. The 11 most frequently detected VOCs are shown.

analyzed for, 22 VOCs were detected at least once, and 55 percent of those detected were gasoline-related compounds (fig. 13). All measured concentrations of VOCs were well below drinking-water standards and guidelines.

The occurrence of benzene, methylbenzene, methyl *tert*-butyl ether (MTBE), 1,3-1,4-dimethylbenzene, and naphthalene showed

evidence of seasonality in samples collected in 1997. All five compounds were detected in samples collected in February, November, and December, but were absent in samples collected in July, August, and September (fig. 14). Water temperature is a significant factor affecting the concentration and detection of VOCs. VOCs are more likely to be stable and

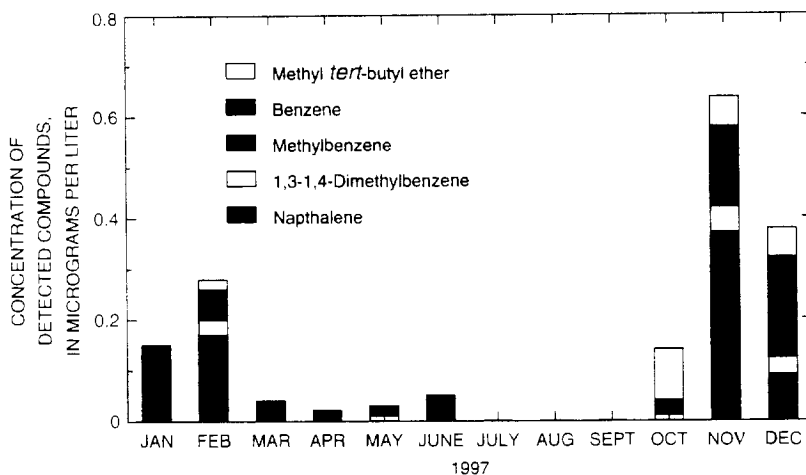


Figure 14. Volatile organic compounds were detected most often and at highest concentrations in surface-water samples in cool seasons (data from Deer Creek at Dorseyville, Pennsylvania).

detectable in cold water than in warm water. Warm temperatures tend to cause VOCs to be driven into the atmosphere. VOC concentrations in water can increase by a factor of about 3 to 7 when water temperatures decrease from 25°C (Celsius) to 5°C (Lopes and Bender, 1998).

VOCs can accumulate on impervious surfaces and can be flushed into the receiving stream during storms. Data from five storm samples collected in Deer Creek on August 25–26, 1998, showed that the maximum measured concentrations of acetone, carbon disulfide, benzene, 1,2,4-trimethylbenzene, and *p*-isopropyltoluene in a sample were collected as streamflow increased. The lowest concentrations were observed in the last samples collected as the stream receded. The concentration pattern demonstrates a flush-off effect as rains washed VOCs from the land surface to the stream (fig. 15).

Fourteen VOCs detected in a sample collected on December 10, 1997, may have resulted from a flush of accumulated VOCs from impervious surfaces in addition to a low water temperature (5.0°C). Of the 14 VOCs detected, 10 were gasoline-related compounds.

Low Levels of Volatile Organic Compounds were Detected in Most Domestic Wells Sampled

Of the 95 domestic wells throughout ALMN from which samples were collected for VOC analysis, at least one compound was detected in each of 87 samples (92 percent). A total of 28 different compounds were detected overall. Most samples (60 percent) contained two or more VOCs at detect-

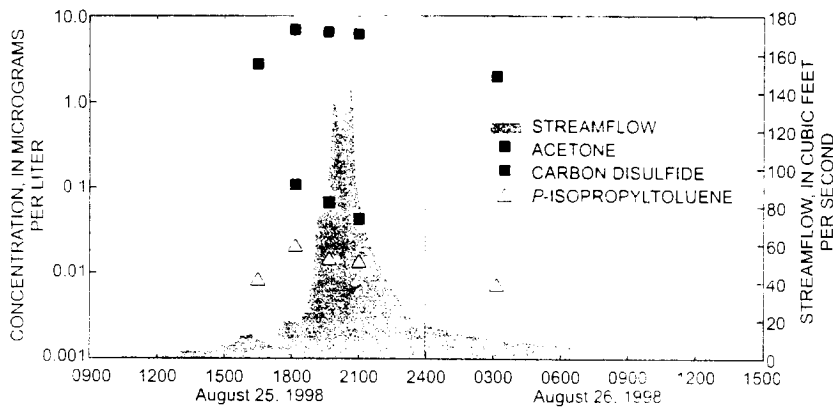


Figure 15. At Deer Creek, some VOCs are rapidly transported to streams during storms. These tend to become most concentrated before the floodwaters peak and decline before water returns to prestorm levels.

able levels, and one sample contained seven different VOCs.

All VOC detections were at low concentrations. Twelve VOCs were detected at concentrations at or above 0.1 µg/L, including the four most frequently detected compounds (fig. 16). Of the 28 detected VOCs, drinking-water standards have been established for 20. None

of the VOCs detected exceeded established drinking-water standards or guidelines.

Thirty of the water samples analyzed for VOCs were from wells downgradient from recently reclaimed surface coal mines (mined sites), and 15 of the water samples were from wells in areas underlain by coal but undisturbed

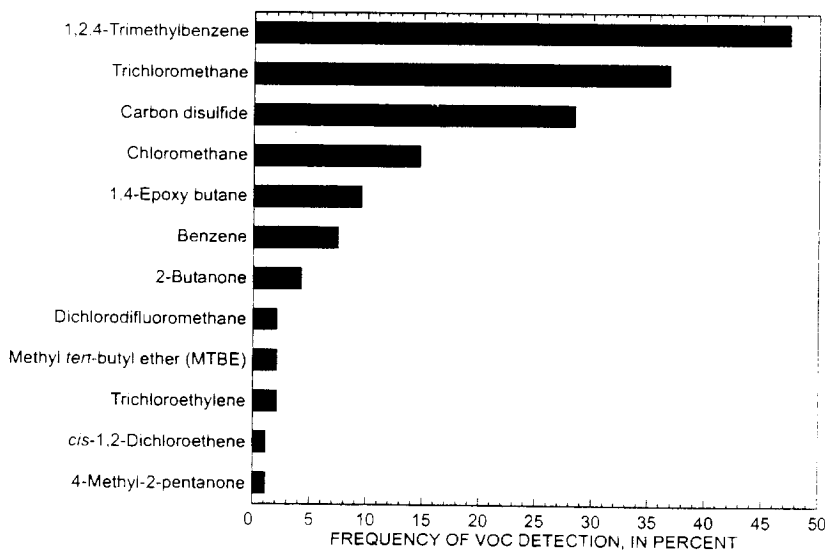


Figure 16. VOCs detected in ALMN ground water at concentrations greater than 0.1 microgram per liter.

by mining (unmined sites). Perhaps as a result of mine machinery use, fuel spills, or adjacent land use, gasoline-related compounds (1,2,4-trimethylbenzene, benzene, methylbenzene, and ethylbenzene) were detected more frequently and at higher concentrations in the samples collected from the mined sites, where these compounds were detected in 29 of 30 samples, compared to 9 detections in 15 samples from unmined sites.

Nitrate is Common in Streams and Ground Water

Nitrate is a nutrient that can affect water used either as a drinking source or as a medium for aquatic life. Nitrate is present naturally in surface water, but elevated concentrations can result in abundant algal growth and toxicity to some aquatic organisms. In well water, nitrate can be a significant health risk at high concentrations. The use of commercial and organic fertilizers and the combustion of fossil fuels has been linked to elevated nitrate concentrations in streams and shallow ground water nationwide (U.S. Geological Survey, 1999a).

In ALMN, 10 stream sites and 95 domestic wells were sampled for nitrate. Samples were collected monthly at the stream sites and once at each well during the period October 1995 through September 1998. Nitrate was detected in all surface-water samples and in 62 percent of ground-water samples. Among wells and streams, only one sample exceeded the USEPA MCL for nitrate in drinking water. The sample was collected from a domestic well in an agricultural setting. The highest median concentration of nitrate in wells and streams was in a stream



Figure 17. Livestock in South Branch Plum Creek, as in many agricultural basins, contribute nitrate to the ecosystem.

in an agricultural setting, South Branch Plum Creek (fig. 17).

The role of land use on the observed nitrate concentrations was investigated by comparison

Why is nitrate important?

Nitrate is the primary form of nitrogen dissolved in streams and ground water. Nitrate forms naturally in soil from transformations of nitrogen, nitrogen-based fertilizers, and manure.

Nitrate is the most widespread contaminant in ground water. Because most ground water eventually discharges to streams, the nitrate in ground water can pose a potential threat to surface-water quality. Surface runoff in areas where commercial fertilizers are used, as well as discharges from wastewater treatment facilities, can also contribute nitrate to streams.

Human ingestion of water with nitrate concentrations in excess of the MCL (10 mg/L as nitrogen) can lead to methemoglobinemia, or "blue-baby syndrome," a sometimes fatal blood disorder in infants.

with a national background concentration for nitrate. The background concentration was estimated from samples collected in undeveloped areas (U.S. Geological Survey, 1999).

Nitrate concentrations in surface-water samples from forested areas in ALMN were less than national background concentration. Among other land uses with potential nitrate sources, concentrations of nitrate often exceeded the background level (figs. 18 and 19).

Activities typical of agriculture and urban/residential land use can lead to an increase in nitrate concentrations. Seasonal fertilizer

applications, animal waste, and sewage are common sources of nitrate. Of the sampled streams in ALMN, 73 percent of samples from a stream draining an agricultural area exceeded background nitrate concentrations. In more populated areas (population density greater than 150 people per square mile), 54 percent of stream samples had nitrate concentrations that exceeded background concentrations.

Overall, streams in basins that integrate various land uses within ALMN had lower concentrations of nutrients than those dominated by either dense population or agri-

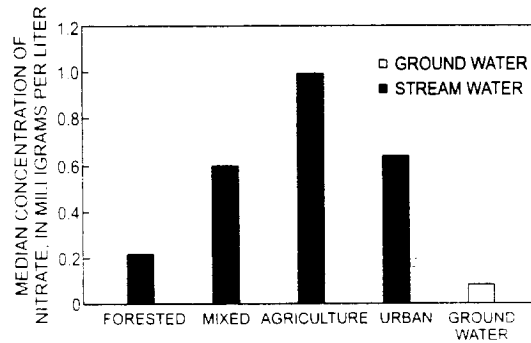


Figure 18. Median concentrations of nitrate in streams were higher than those found in ground water.

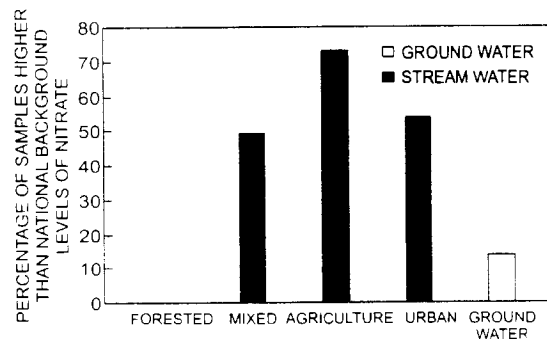


Figure 19. Streams in agricultural areas had the highest percentage of samples that exceeded national background levels for nitrate (0.6 milligrams per liter in streams and 2.0 milligrams per liter (U.S. Geological Survey, 1999a) in shallow ground water).

culture. Areas with both high population density and significant agricultural acreage exceeded background nitrate concentrations in 49 percent of stream samples.

Ground-water samples analyzed for nitrate were collected from wells in areas of mixed land use. Consequently, no agricultural-urban comparisons could be made for nitrate in ground water.

Radon in Ground Water is Common but Highly Variable

Radon is a radioactive gas that is produced naturally in rocks and soils as an intermediate product in the decay of uranium-238. Radon in ground water originates from nearby soil and rock and is a potential contributing source of radon in indoor air. Exposure to airborne radon has been identified by the U.S. Surgeon General as the second leading cause of lung cancer in the United States. About 20,000 deaths per year in the United States are attributed to airborne radon (U.S. Environmental Protection Agency, 1999b).

Radon concentrations in 56 percent of the 95 ground-water samples analyzed for radon were greater than 300 pCi/L (picocuries per liter), the USEPA proposed standard for drinking water. About 19 percent of the 95 samples exceeded 1,000 pCi/L (fig. 20). Two percent of the 95 samples exceeded the proposed Alternative Maximum Contaminant Level (AMCL) standard of 4,000 pCi/L. To comply with the AMCL, a State or local water utility must develop indoor air radon-reduction programs and reduce radon levels in drinking water to 4,000 pCi/L

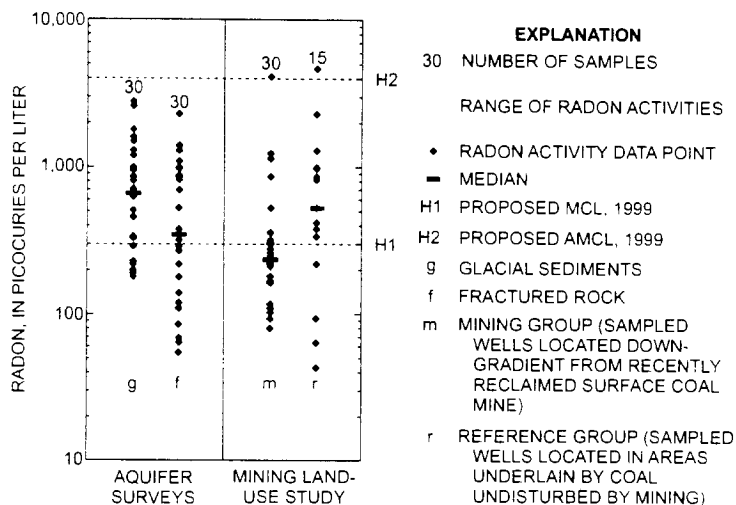


Figure 20. Radon concentration in ground water varied considerably within well groupings sampled in the Allegheny and Monongahela River Basins.

(U.S. Environmental Protection Agency, 1999b).

Large variation in radon concentration was found in ground water from the two aquifer systems sampled. Samples from wells in the valley-fill aquifers had a median radon concentration of 665 pCi/L; the median for samples from wells in the fractured-rock aquifers was 350 pCi/L. The higher radon concentrations in water of the valley-fill aquifers may be due to higher uranium content of the valley-fill deposits or may derive from the rock underlying these deposits. Samples from wells downgradient from recently re-claimed surface coal mines had a median radon concentration of 236 pCi/L. By comparison, water samples from wells in areas underlain by coal undisturbed by mining had a median radon concentration of 530 pCi/L. This difference may be due to several factors, such as (1) replacement of high-radon content overburden with lower-radon content backfill or (2) a greater release of radon directly to the air and less

entrapment in ground water after ground disturbance caused by surface mining.

Is radon a risk from your well?

The only way to be sure of radon concentration in ground water from a specific well is to have it tested. The U.S. Surgeon General recommends testing of indoor air radon levels in all homes (and apartments below the third floor). The USEPA-recommended action level for indoor air radon levels is 4 pCi/L. The USEPA recommends testing well water for radon in homes where indoor air levels of radon are high. High concentrations of radon in well water can significantly contribute to airborne levels indoors. Although few of the 95 wells that were tested in ALMN had high concentrations of radon, the results show considerable variability (fig. 20). Ground water from each well should be checked if radon is a concern. If a large part of the indoor radon is from ground-water contribution, the USEPA recommends water treatment to remove radon.

REGIONAL STUDIES: Sulfate concentrations and biological communities in Appalachian coal areas indicate mining-related disturbances despite a general water-quality improvement between 1980 and 1998

In a 1998 study to assess regional water-quality effects of coal mining (Eychaner, 1999), samples representing the Northern Appalachian coal field were collected in the Allegheny and Monongahela River Basins (ALMN), where high-sulfur coal is common and acid mine drainage was historically severe, and samples for the Central Appalachian coal field were collected in the Kanawha-New River Basin (KANA), where acid drainage is uncommon (fig. 21).

Water chemistry in 178 wadeable streams was analyzed once during low streamflow in July and August 1998. Drainage area for most streams was between 4 and 80 mi². Most (170) of these stream sites were also sampled during a 1979–81 study on the effects of coal mining (Britton

and others, 1989), before implementation of Surface Mine and Reclamation Control Act (SMCRA) Regulations began to affect regional water quality. At 61 sites, aquatic invertebrates (insects, worms, crustaceans, and mollusks) also were collected. Ground water was sampled from 58 wells near coal surface mines and 25 wells in unmined areas.

Water-Quality Characteristics Targeted by SMCRA Improved in Streams, but Sulfate and Metals Remain High at Some Sites

Median pH increased and median total iron and total manganese concentrations in streams decreased among mined basins between 1979–81 and 1998 in both coal fields, a reflection that these water-quality characteristics are regulated in mine discharges. Concentrations of sulfate, which is not regulated in mine discharges, exceeded regional background levels at sites downstream from mining (average of about 21 mg/L sulfate in

basins with no history of mining) in more than 70 percent of samples.

The highest sulfate concentrations were measured in basins with the greatest coal production. About one-fourth of all samples exceeded 250 mg/L, the USEPA Secondary Maximum Contaminant Level (SMCL) for drinking water, and all these exceedences were in mined basins (fig. 22). When coal mining ceases within a basin, sulfate concentrations gradually decrease (Sams and Beer, 2000).

Manganese, aluminum, and iron at stream sites in many mined basins also exceeded regional background concentrations (table 1). In the 1998 samples from the northern coal field, median total iron was about equal among mined and unmined basins; but in the central coal field, median total iron among mined basins was lower than among unmined basins. In both coal fields, median total manganese among mined basins was about double that among unmined basins. Exceedences of SMCLs for dissolved iron and manganese were more common in mined basins than in unmined basins, and the aluminum SMCL was exceeded in mined basins only.

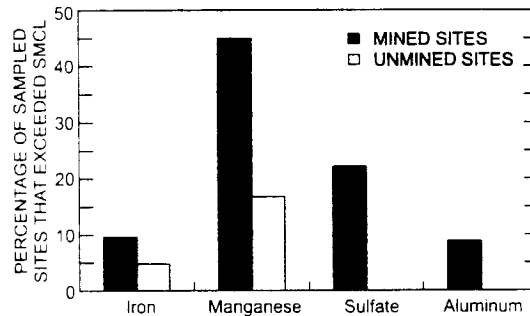


Figure 22. Stream water exceeded Secondary Maximum Contaminant Levels at mined sites more often than at unmined sites.

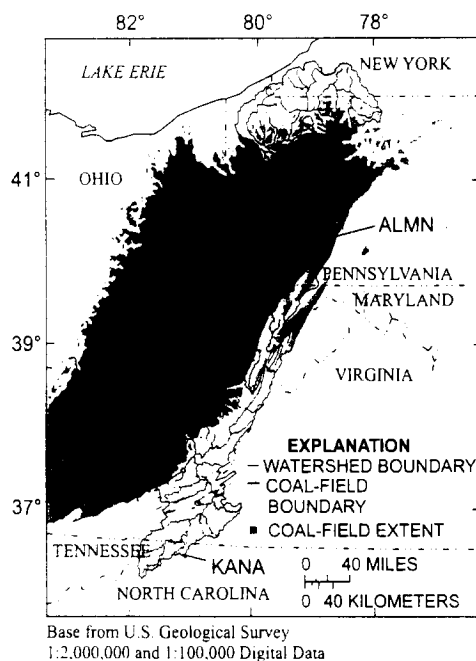


Figure 21. Coal-bearing rocks underlie 55 percent of the area sampled in the Northern and Central Appalachian bituminous coal fields. (Coal-field locations from Tully, 1996)

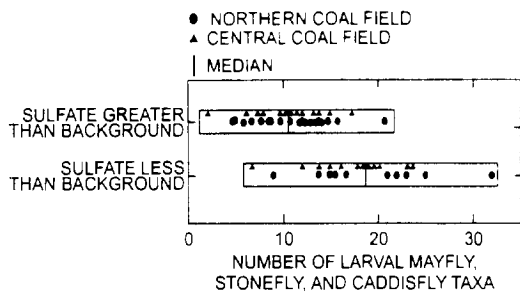


Figure 23. Sulfate concentration in stream water, an indicator of coal production in a basin, was inversely related to the number of mayfly, stonefly, and caddisfly taxa found at water-quality sampling sites.

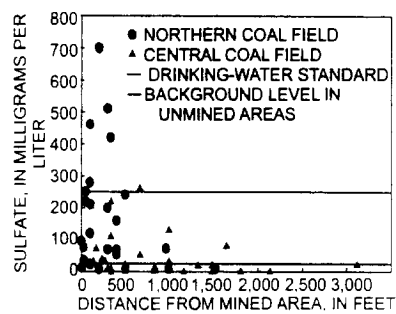


Figure 24. Sulfate concentrations in ground water generally exceeded regional background levels within about 1,000 feet from surface coal mines.

Invertebrate-Community Impairment Appears Related to Amount of Mining

Invertebrate communities tended to be more impaired in mined basins than in minimally altered basins. Pollution-tolerant species were more likely to be present at mined sites than at unmined sites, whereas pollution-sensitive taxa were few or absent in heavily mined basins. Both an increased sulfate concentration and a decline in some aquatic-insect populations was related to coal production (fig. 23). At sites where sulfate concentrations were above the estimated background level (table 1), the number of taxa of three groups of sensitive insect species (mayflies, stoneflies, and caddisflies) was reduced, although the pH was 6.5 or greater at all these sites.

At the concentrations measured, the sulfate ion is relatively non-toxic to aquatic organisms and may not represent the cause of the decline in mayflies, stoneflies, and caddisflies observed. Sulfate is, however, related to the total coal production from a basin (Sams and Beer, 2000). Invertebrate communities may also have been impaired by other large-scale landscape disturbances—for example, changes

in hydrology, siltation, or trace-metal contamination, all of which can be caused by increased coal production. The communities in basins affected by low to moderate coal production were similar to communities in basins affected by urbanization, agriculture, large construction projects, flow alterations, or wastewater effluents.

Sulfate and Some Metal Concentrations were Higher in Ground Water near Surface Coal Mines

Sulfate concentrations in ground water generally were higher than regional background concentrations in shallow domestic water-supply wells within 1,000 feet of reclaimed surface mines (fig. 24). Water from such wells in the northern coal field contained more sulfate and calcium than did wells in unmined areas in the same region, or at any of the sites in the central coal field. Iron, manganese, aluminum, magnesium, turbidity, and specific conductance also were higher than regional background concentrations within about 2,000 feet of reclaimed surface mines in both coal fields. Concentra-

tions of calcium and magnesium are higher near mined sites because these elements are components of minewater-treatment chemicals and of some of the rocks associated with coal seams. Ground water near reclaimed surface mines exceeded SMCLs for iron, manganese, sulfate, and aluminum more frequently than ground water in unmined areas (fig. 25). Iron and manganese occur naturally in native coal-bearing rocks, sometimes at high concentrations; however, nearly twice as many groundwater samples at mined sites exceeded SMCLs for iron compared to unmined sites. Wells where SMCLs for sulfate and manganese were exceeded were most commonly in the northern coal field.

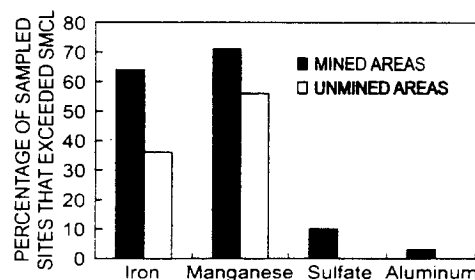


Figure 25. Ground water exceeded Secondary Maximum Contaminant Levels in mined areas more often than in unmined areas.

STUDY UNIT DESIGN

Stream Chemistry and Ecology

Surface-water assessments included water, bed sediment, and fish tissue chemistry; fish, invertebrate, and algal communities; and physical habitat. Sites were chosen across the study area for spatial coverage and distribution in the major aquatic ecological settings within the Allegheny and Monongahela River Basins (fig. 26).

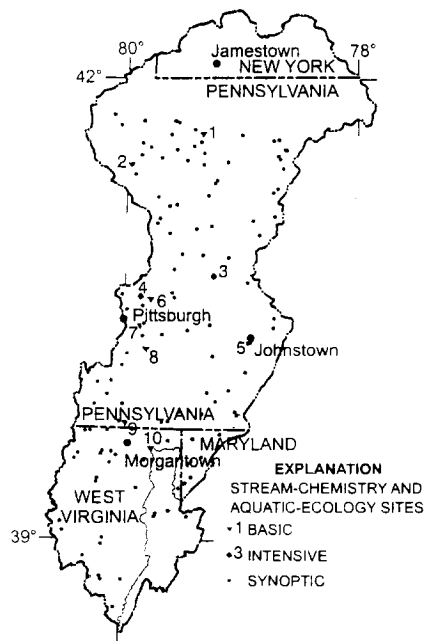


Figure 26. In addition to intensive water-quality sampling at a few sites, one-time sampling at many sites across the study area provided data related to specific land uses. The Cheat River Basin (shaded pink) was similarly sampled.

Basic and intensive sites were sampled monthly for chemistry and annually for ecological condition. One urban site and one agricultural site also were intensively sampled during storms to assess the influence of storm runoff on stream contaminant concentrations. Eighty-nine additional synoptic sites were sampled once to assess the influence of coal mining on water quality across the study area.

Ground-Water Chemistry

Two reconnaissance-type studies were done. The first focused on the fractured-rock aquifers of the coal-bearing Pittsburgh Series rocks of middle and late Pennsylvanian age. The second was set in the coarse- and fine-grained glaciofluvial deposits of the valley-fill aquifers in the northern area of the Allegheny River Basin (fig. 27).

An additional study that focused on mining land use involved sampling of wells that drew water from the fractured-rock aquifers and that

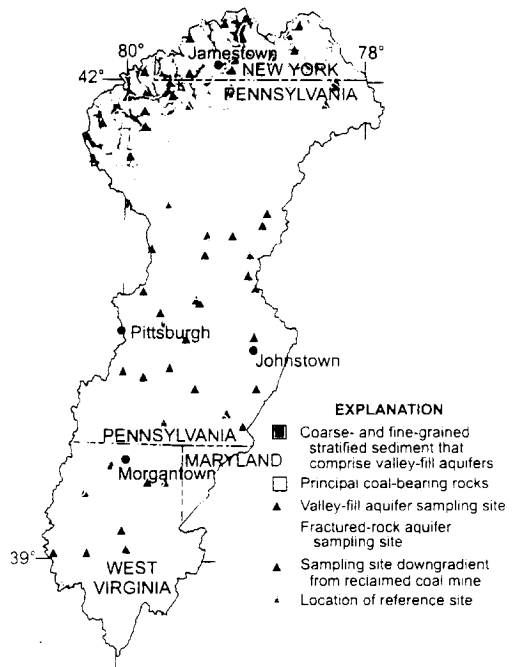


Figure 27. Ground water was sampled from two major aquifer systems, valley-fill aquifers of the northern Allegheny River Basin, and fractured-rock aquifers in the Pittsburgh Series rocks that contain the largest quantities of commercially minable bituminous coal in the ALMN.

were near surface coal mines where mining and reclamation efforts have been completed. The quality of these samples was compared to that of water from 15 wells sampled in unmined areas of the same aquifers.

Site number (fig. 26)	Site name	Site type	Basin area (square miles)	Site number (fig. 26)	Site name	Site type	Basin area (square miles)
1	East Hickory Creek near Queen, Pa.	Forested	20.3	6	Allegheny River at New Kensington, Pa.	Mixed	11,500
2	French Creek at Utica, Pa.	Mixed	1,028	7	Monongahela River at Braddock, Pa.	Mixed	7,337
3	South Branch Plum Creek at Five Points, Pa.	Agriculture	33.3	8	Youghiogheny River at Sutersville, Pa.	Mixed	1,715
4	Deer Creek near Dorseyville, Pa.	Urban	27.0	9	Dunkard Creek at Shannopin, Pa.	Mining	4,440
5	Stonycreek River at Ferndale, Pa.	Mining	451	10	Cheat River near Mt. Nebo, W. Va.	Mixed	1,132

SUMMARY OF DATA COLLECTION IN THE ALLEGHENY AND MONONGAHELA RIVER BASINS, 1996–98

Study component	What data were collected and why	Types of sites sampled	Number of sites	Sampling frequency and period
Stream Chemistry				
Basic Sites— General water chemistry	Concentrations, seasonal variation, and annual loads. Data included streamflow, field measurements, major ions, nutrients, organic carbon, suspended sediment, trace elements.	Basic Fixed Sites: Representative of common land-use mixes, as well as basin outflow sites.	8	Monthly: April 1996–Sept. 1998
Intensive sites— Pesticides and VOCs	Concentrations and seasonal variations in pesticides. Data included same constituents as above, plus 83 pesticides (dissolved) and 87 volatile organic compounds (VOCs) (only 1 site).	Basic Fixed Sites with intensive urban or agricultural land use.	2	1997, 1998
Contaminants in bed sediments	Occurrence and distribution of contaminants in bed sediment. Data include trace elements, organochlorine compounds, and volatile organic compounds.	Depositional zones of most stream sites sampled in other components of study.	19	Monthly and more frequently
Contaminants in fish tissue	Occurrence and distribution of contaminants in biota. Data included total PCBs, 30 organochlorine pesticides in whole fish, and 24 trace elements in fish livers.	Most stream sites sampled in other components of study where tissue could be collected.	17	Fish Tissue: Summer 1996 and Summer 1997 (Duplicate taxa at two sites)
Stream Ecology				
Ecological assessments	Macroinvertebrates (benthic invertebrates), fish, algae, aquatic and riparian habitat.	Basic Fixed Sites. Intensive Sites.	8 2	1996–97 (10 sites), 1998 (6 sites) One 3-reach site 1996 and 1997
Synoptic studies	Unmined basin to compare to mined basins. The same data were collected at Basic Sites.	Synoptic Site.	1	Once in 1997
Ground-Water Chemistry				
Aquifer survey— Pittsburgh Series fractured rock	Assess quality across aquifer extent. Data include field measurements, major ions, trace metals, nutrients, Pesticides, VOCs, radon, dissolved organic carbon (DOC).	Existing domestic wells chosen with a statistically random selection process. Well depth range 30 to 250 feet.	30	Once in 1996 (July–August)
Aquifer survey— Glaciofluvial deposits of the valley-fill aquifers	Assess quality across aquifer extent. Data include field measurements, major ions, nutrients, pesticides, VOCs, radon, dissolved organic carbon (DOC).	Existing domestic wells chosen with a statistically random selection process. Well depth range 30 to 250 feet.	30	Once in 1996 (September–October)
Land-use effects— Surface coal mining	Compare ground-water quality near reclaimed surface mines to that in unmined areas. Data include major ions, trace metals, nutrients, VOCs, radon, trace elements, dissolved organic carbon (DOC), chlorofluorocarbons (CFCs).	Existing domestic wells chosen with a statistically random selection process. Well depth range 30 to 250 feet. (Data from 10 fractured-rock sampling sites were re-used as reference data in the Land-use effects study.)	45	Once in 1997 (August–October)
Special Studies				
Low-flow synoptic survey of streams in the Appalachian coal fields	To assess quality of surface water relative to type and age of coal mining in the basins. Standard: Mine-drainage indicators, field measurements. Intensive: same as standard sites, plus: major ions, trace elements, macroinvertebrates, aquatic habitat.	Standard Site network.	89	Standard sites: Once in summer 1998
		Intensive Site network.	32	Intensive sites: Once in summer 1998

GLOSSARY

- Aquifer**—A water-bearing layer of soil, sand, gravel, or rock that will yield usable quantities of water to a well.
- Background concentration**—A concentration of a substance in a particular environment that is indicative of minimal influence by human (anthropogenic) sources.
- Bed sediment**—The material that temporarily is stationary in the bottom of a stream or other watercourse.
- DDT**—Dichloro-diphenyl-trichloroethane. An organochlorine insecticide no longer registered for use in the United States.
- Ground water**—In general, any water that exists beneath the land surface, but more commonly applied to water in fully saturated soils and geologic formations.
- Herbicide**—A chemical or other agent applied for the purpose of killing undesirable plants. See also Pesticide.
- Human health advisory**—Guidance provided by U.S. Environmental Protection Agency, State agencies, or scientific organizations, in the absence of regulatory limits, to describe acceptable contaminant levels in drinking water or edible fish.
- Insecticide**—A substance or mixture of substances intended to destroy or repel insects. See also Pesticides.
- Maximum contaminant level (MCL)**—Maximum permissible level of a contaminant in water that is delivered to any user of a public water system. MCLs are enforceable standards established by the U.S. Environmental Protection Agency.
- Method detection limit**—The minimum concentration of a substance that can be accurately identified and measured with present laboratory technologies.
- Micrograms per liter ($\mu\text{g/L}$)**—A unit expressing the concentration of constituents in solution as weight (micrograms) of solute per unit volume (liter) of water; equivalent to one part per billion in most stream water and ground water. One thousand micrograms per liter equals 1 milligram per liter.
- Milligrams per liter (mg/L)**—A unit expressing the concentration of chemical constituents in solution as weight (milligrams) of solute per unit volume (liter) of water; equivalent to one part per million in most stream water and ground water. One thousand micrograms per liter equals 1 mg/L .
- Organochlorine compound**—Synthetic organic compounds containing chlorine. As generally used, term refers to compounds containing mostly or exclusively carbon, hydrogen, and chlorine. Examples include organochlorine insecticides, polychlorinated biphenyls, and some solvents containing chlorine.
- Pesticide**—A chemical applied to crops, rights of way, lawns, or residences to control weeds, insects, fungi, nematodes, rodents, or other "pests."
- pH**—The logarithm of the reciprocal of the hydrogen ion concentration (activity) of a solution; a measure of the acidity (pH less than 7) or alkalinity (pH greater than 7) of a solution; a pH of 7 is neutral.
- Polychlorinated biphenyls (PCBs)**—A mixture of chlorinated derivatives of biphenyl, marketed under the trade name Aroclor with a number designating the chlorine content (such as Aroclor 1260). PCBs were used in transformers and capacitors for insulating purposes and in gas pipeline systems as a lubricant. Further sale for new use was banned by law in 1979.
- Secondary maximum contaminant level (SMCL)**—The maximum contamination level in public water systems that, in the judgment of the U.S. Environmental Protection Agency (USEPA), is required to protect the public welfare. SMCLs are secondary (nonenforceable) drinking water regulations established by the USEPA for contaminants that may adversely affect the odor or appearance of such water.
- Semivolatile organic compound (SVOC)**—Operationally defined as a group of synthetic organic compounds that are solvent-extractable and can be determined by gas chromatography/mass spectrometry. SVOCs include phenols, phthalates, and polycyclic aromatic hydrocarbons (PAHs).
- Suspended sediment**—Particles of rock, sand, soil, and organic detritus carried in suspension in the water column, in contrast to sediment that moves on or near the streambed or rests on the bottom of the stream.
- Trace element**—An element found in only minor amounts (concentrations less than 1.0 milligram per liter) in water or sediment; includes arsenic, cadmium, chromium, copper, lead, mercury, nickel, and zinc.
- Upgradient**—Of or pertaining to the place(s) from which ground water originated or traveled through before reaching a given point in an aquifer.
- Volatile organic compounds (VOCs)**—Organic chemicals that have a high vapor pressure relative to their water solubility. VOCs include components of gasoline, fuel oils, and lubricants, as well as organic solvents, fumigants, some inert ingredients in pesticides, and some byproducts of chlorine disinfection.
- Water-quality guidelines**—Specific levels of water quality which, if reached, may adversely affect human health or aquatic life. These are nonenforceable guidelines issued by a governmental agency or other institution.
- Water-quality standards**—State-adopted and U.S. Environmental Protection Agency-approved ambient standards for water bodies. Standards include the use of the water body and the water-quality criteria that must be met to protect the designated use or uses.
- Yield**—The mass of material or constituent transported by a river in a specified period of time divided by the drainage area of the river basin.

REFERENCES

- Becher, A.E., 1999, Groundwater. *in* Shultz, C.H., ed., The geology of Pennsylvania: Pennsylvania Geological Survey and Pittsburgh Geological Society. 888 p.
- Bender, D.A., Zogorski, J.S., Halde, M.J., and Rowe, B.L., 1999, Selection procedure and salient information for volatile organic compounds emphasized in the National Water-Quality Assessment Program: U.S. Geological Survey Open-File Report 99-182, 32 p.
- Biesecker, J.E., and George, J.R., 1966, Stream quality in Appalachia as related to coal-mine drainage, 1965: U.S. Geological Survey Circular 526, 27 p., 1 pl.
- Britton, L.J., Anderson, C.L., Goolsby, D.A., and Van Haveren, B.P., eds., 1989, Summary of the USGS and U.S. Bureau of Land Management National Coal Hydrology Program, 1974-84: U.S. Geological Survey Professional Paper 1464, 183 p.
- Canadian Council of Ministers of the Environment, 1995, Protocol for the derivation of Canadian sediment quality guidelines for the protection of aquatic life: Winnipeg, Manitoba, Task Group on Water Quality Guidelines report CCME EPC-98E, 38 p.
- Capel, P.D., Spexet, A.H., and Larson, S.J., 1999, Occurrence and behavior of the herbicide prometon in the hydrologic system: Environmental Science and Technology, v. 33, no. 5, p. 674-680.
- Coll, M.B., Jr., and Siwicki, R.W., 1997, Water resources data, Pennsylvania, water year 1997, v. 3, Ohio River and St. Lawrence River Basins: U.S. Geological Survey Water-Data Report PA-97-3, 339 p.
- Cooper, E.L., 1983, Fishes of Pennsylvania and the Northeastern United States: University Park, Pa., Pennsylvania State Press, 243 p.
- Eychaner, J.H., 1999, Progress of environmental studies in coal mining areas of western Pennsylvania and central West Virginia, *in* Twentieth Annual West Virginia Surface Mine Drainage Task Force Symposium, 1999, Proceedings: Morgantown, W. Va., 6 p.
- Ferguson, J.F., and Gavis, J., 1972, A review of the arsenic cycle in natural waters: Water Research, v. 6, p. 1259-1274.
- Finni, G.R., 1988, The benthic macroinvertebrates of the Monongahela River near Pittsburgh, Pennsylvania—Final report to the U.S. Fish and Wildlife Service: Mewton Corner, Mass., 48 p.
- Gilliom, R.J., Alley, W.M., and Gurtz, M.E., 1995, Design of the National Water-Quality Assessment Program—Occurrence and distribution of water-quality conditions: U.S. Geological Survey Circular 1112, 33 p.
- Hem, J.D., 1985, Study and interpretation of the chemical characteristics of natural water: U.S. Geological Survey Water-Supply Paper 2254, 263 p.
- Letterman, R.D., and Mitsch, W.J., 1978, Impact of mine drainage on a mountain stream in Pennsylvania: Environmental Pollution, v. 17, p. 53-73.
- Lindsey, B.D., and Bickford, T.M., 1999, Hydrogeologic framework and sampling design for an assessment of agricultural pesticides in ground water in Pennsylvania: U.S. Geological Survey Water-Resources Investigations Report 99-4076, 44 p.
- Lopes, T.J., and Bender, D.A., 1998, Nonpoint sources of volatile organic compounds in urban areas—Relative importance of land surfaces and air: Environmental Pollution, v. 101, p. 221-230.
- Masters, L.L., Flack, S.R., and Stein, B.A., eds., 1998, Rivers of life—Critical watersheds for protecting biodiversity: Arlington, Va., The Nature Conservancy, 71 p.
- McAuley, S.D., 1995, National Water Quality Assessment Program—The Allegheny-Monongahela River Basin: U.S. Geological Survey Fact Sheet 137-95, 2 p.
- Newell, A.J., Johnson, D.W., and Allen, L.K., 1987, Niagara River biota contamination project—Fish flesh criteria for piscivorous wildlife: New York State Department of Environmental Conservation, Division of Fish and Wildlife, Bureau of Environmental Protection Technical Report 87-3, 182 p.
- Ohio River Basin Commission, 1980, Allegheny River Basin—Regional water and land resources plan and environmental impact statement: Cincinnati, Ohio, Ohio River Basin Commission, 178 p. plus appendixes.
- Omernick, J.M., 1987, Aquatic ecoregions of the conterminous United States: accessed January 17, 2001, at URL <http://water.usgs.gov/lookup/getcover?ecoregion/>.
- Ortmann, A.E., 1909, The destruction of the fresh-water fauna in western Pennsylvania: Proceedings of the American Philosophical Society, v. 5, p. 91-110.
- Pennsylvania Department of Conservation and Natural Resources, 2000, accessed August 17, 2000, at URL <http://www.dcnr.state.pa.us/rivers/smap.htm>.
- Pennsylvania Department of Environmental Protection, 1996, Commonwealth of Pennsylvania Water Quality Assessment 305(b) Report: Pennsylvania Department of Environmental Protection, 94 p.



Diverse aquatic communities can be found in western tributaries.



More than two-thirds of the study area is hemlock and hardwood forest.

_____. 1999. The science of acid mine drainage and passive treatment: Bureau of Abandoned Mine Reclamation. 10 p.

Pennsylvania Department of Environmental Resources. 1984. Pennsylvania Code Title 25. Environmental resources—Water quality standards: Harrisburg, Pa., Bureau of Water Quality Assessment, chap. 93. 212 p.

Pennsylvania Fish and Boat Commission. 1999. accessed September 12, 2000. at URL http://www.state.pa.us/PA_Exec/Fish_Boat/fishadvi.htm.

Puente, Celso, and Atkins, J.T., 1989. Simulation of rainfall-runoff response in mined and unmined watersheds in coal area of West Virginia: U.S. Geological Survey Water-Supply Paper 2298. 48 p.

Risser, D.W., and Madden, T.M., Jr., 1994. Evaluation of methods for delineating areas that contribute water to wells completed in valley-fill aquifers in Pennsylvania: U.S. Geological Survey Open-File Report 92-635. 82 p.

Rose, A.W., and Cravotta, C.A., III, 1998. Geochemistry of coal mine drainage, *in* Coal mine drainage prediction and pollution prevention in Pennsylvania: Pennsylvania Department of Environmental Protection. 371 p.

Sams, J.I., and Beer, K.M., 2000. Effects of coal-mine drainage on stream water quality in the Allegheny and Monongahela River Basins—



Urban areas generally are within river valleys.

Sulfate transport and trends: U.S. Geological Survey Water-Resources Investigations Report 99-4208. 17 p.

Tully, John. 1996. Coal fields of the conterminous United States: U.S. Geological Survey Open-File Report 96-92. 1 sheet, scale 1:5,000,000.

U.S. Army Corps of Engineers. 1976. Monongahela River navigation projects, annual water quality report. 1976: Pittsburgh, Pa., 106 p.

U.S. Environmental Protection Agency, 1999a. Health effects from exposure to high levels of sulfate in drinking water study: U.S. Environmental Protection Agency, Office of Water and Ground Water. EPA-815-R-99-001. 25 p.

_____. 1999b. Proposed radon in drinking water rule: U.S. Environmental Protection Agency report EPA 815-F-99-006. accessed July 14, 2000. at URL <http://www.epa.gov/OGWDW/radon/fact.html>.

_____. 1996. The National Sediment Quality Survey—A report to Congress on the extent and severity of sediment contamination in surface waters of the United States: U.S. Environmental Protection Agency, Office of Science and Technology, Draft Report EPA 823-D-96-002.

U.S. Geological Survey, 1999a. The quality of our Nation's waters—Nutrients and pesticides: U.S. Geological Survey Circular 1225. 82 p.

_____. 1999b. Pesticides analyzed in NAWQA samples—Use, chemical analyses, and water-quality criteria: accessed December 28, 2000. at URL <http://cd.water.usgs.gov/pnsp/anstrat/index.htm/>.

Welch, A.H., Lico, M.S., and Hughes, J.L., 1988. Arsenic in ground water of the western United States: Ground Water, v. 26, no. 3, p. 333-347.



Farm fields, intermixed with forest, follow hill contours.

West Virginia Department of Environmental Protection. 1998. West Virginia's Water Quality Assessment 305(b) Report: West Virginia Department of Environmental Protection. 142 p.

Williams, D.R., and Clark, M.E., 2001. Nutrients and organic compounds in Deer Creek and South Branch Plum Creek in southwestern Pennsylvania. April 1996 through September 1998: U.S. Geological Survey Water-Resources Investigations Report 00-4061.

Williams, D.R., Clark, M.E., and Brown, J.B., 1999. Stream water quality in coal mined areas of the Lower Cheat River Basin, West Virginia and Pennsylvania, during low-flow conditions. July 1997: U.S. Geological Survey Water-Resources Investigations Report 98-4258. 8 p.

Williams, D.R., Sams, J.I., III, and Mulkerrin, M.E., 1996. Effects of coal-mine discharges on the quality of the Stonycreek River and its tributaries, Somerset and Cambria Counties, Pennsylvania: U.S. Geological Survey Water-Resources Investigations Report 96-4133. 95 p.



Coal mining is a difficult land use to quantify but is one that can significantly alter water quality.

APPENDIX—WATER-QUALITY DATA FROM THE ALLEGHENY AND MONONGAHELA RIVER BASINS IN A NATIONAL CONTEXT

For a complete view of Allegheny and Monongahela River Basins data and for additional information about specific benchmarks used, visit our Web site at <http://water.usgs.gov/nawqa/>. Also visit the NAWQA Data Warehouse for access to NAWQA data sets at <http://infotrek.er.usgs.gov/wdbctx/nawqa/nawqa.home>.

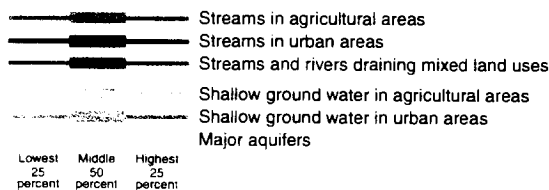
This appendix is a summary of chemical concentrations and biological indicators assessed in the Allegheny and Monongahela River Basins. Selected results for this Study Unit are graphically compared to results from as many as 36 NAWQA Study Units investigated from 1991 to 1998 and to national water-quality benchmarks for human health, aquatic life, or fish-eating wildlife. The chemical and biological indicators shown were selected on the basis of frequent detection, detection at concentrations above a national benchmark, or regulatory or scientific importance. The graphs illustrate how conditions associated with each land use sampled in the Allegheny and Monongahela River Basins compare to results from across the Nation, and how conditions compare among the several land uses. Graphs for chemicals show only detected concentrations and, thus, care must be taken to evaluate detection frequencies in addition to concentrations when comparing study-unit and national results. For example, metolachlor concentrations in the Allegheny and Monongahela River Basins urban stream sampled were similar to the national distribution, but the detection frequency was much higher (96 percent compared to 64 percent).

CHEMICALS IN WATER

Concentrations and detection frequencies, Allegheny and Monongahela River Basins, 1996–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals

- ◆ Detected concentration in Study Unit
- 66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- .. Not measured or sample size less than two
- 12 Study-unit sample size. For ground water, the number of samples is equal to the number of wells sampled

National ranges of detected concentrations, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

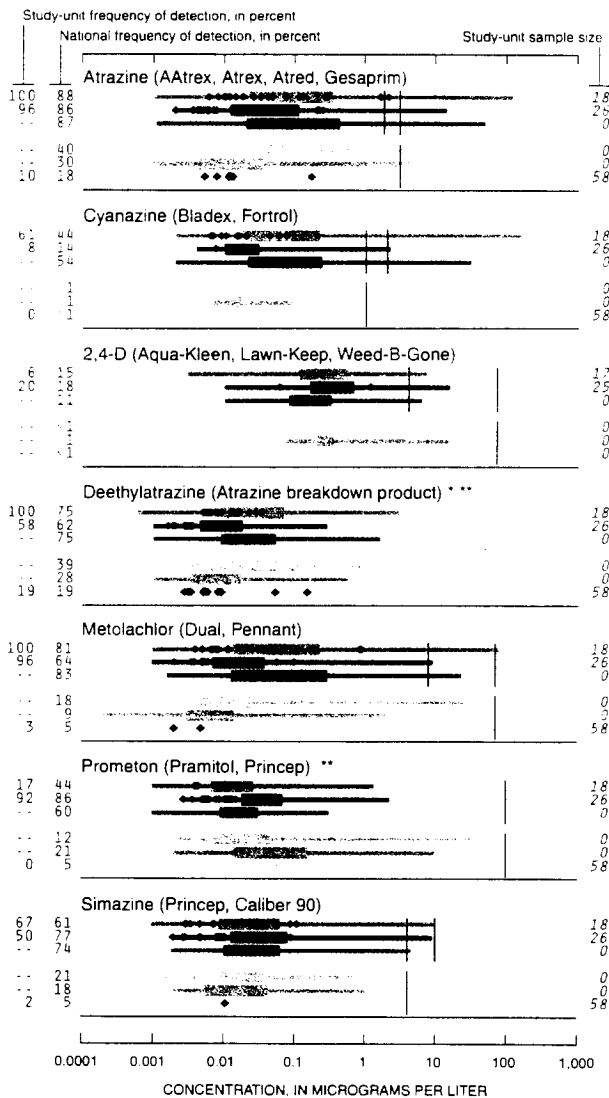


National water-quality benchmarks

National benchmarks include standards and guidelines related to drinking-water quality, criteria for protecting the health of aquatic life, and a goal for preventing stream eutrophication due to phosphorus. Sources include the U.S. Environmental Protection Agency and the Canadian Council of Ministers of the Environment

- | Drinking-water quality (applies to ground water and surface water)
- | Protection of aquatic life (applies to surface water only)
- : Prevention of eutrophication in streams not flowing directly into lakes or impoundments
- No benchmark for drinking-water quality
- .. No benchmark for protection of aquatic life

Pesticides in water—Herbicides



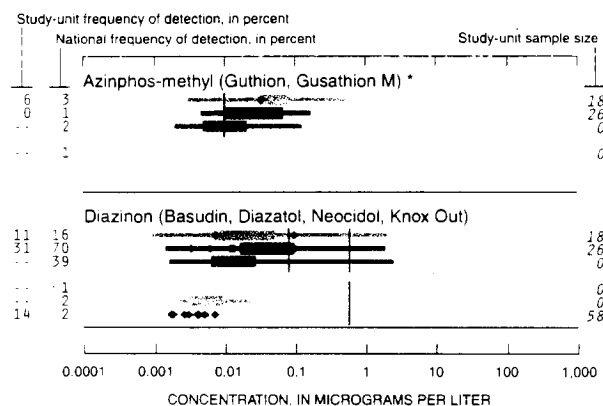
Other herbicides detected

- Acetochlor (Harness Plus, Surpass) **
- Acifluorfen (Blazer, Tackle 2S) **
- Alachlor (Lasso, Bronco, Lariat, Bullet) **
- Bentazon (Basagran, Bentazone) **
- Bromoxynil (Buctril, Brominal) *
- DCPA (Dacthal, chlorthal-dimethyl) **
- Dicamba (Banvel, Dianat, Scotts Proturf)
- Dichlorprop (2,4-DP, Seritox 50, Lentemul) **
- Diuron (Crisuron, Karmex, Diurex) **
- EPTC (Eptam, Farmarox, Alirox) **
- Fenuron (Fenulon, Fenidim) **
- MCPA (Rhomene, Rhonox, Chiptox)
- Metribuzin (Lexone, Sencor)
- Napropamide (Devrinol) **
- Neburon (Neburea, Neburyl, Noruben) **
- Pendimethalin (Pre-M, Prowl, Stomp) **
- Propachlor (Ramrod, Satecid) **
- Tebuthiuron (Spike, Tebusan)
- Terbacil (Sinbar) **

Herbicides not detected

Benfluralin (Balan, Benefin, Bonalan) * **
Bromacil (Hyvar X, Urox B, Bromax)
Butylate (Sutan +, Genate Plus, Butilate) **
Chloramben (Amiben, Amilon-WP, Vegiben) **
Clopyralid (Stinger, Lontrel, Transline) * **
2,4-DB (Butyrac, Butoxone, Embutox Plus, Embutone) * **
Dacthal mono-acid (Dacthal breakdown product) * **
2,6-Diethylaniline (Alachlor breakdown product) * **
Dinoseb (Dinosebe)
Ethalfuralin (Sonalan, Curbit) * **
Fluometuron (Flo-Met, Cotoran) **
Linuron (Lorox, Linex, Sarclex, Linurex, Afalon) *
MCPB (Thistrol) * **
Molinate (Ordram) * **
Norflurazon (E vital, Predict, Solicam, Zorial) * **
Oryzalin (Surlan, Dirimal) * **
Pebulate (Tillam, PEBC) * **
Picloram (Grazon, Tordon)
Pronamide (Kerb, Propyzamid) **
Propanil (Stam, Stampede, Wham) * **
Propham (Tuberite) **
2,4,5-T **
2,4,5-TP (Silvex, Fenoprop) **
Thiobencarb (Bolero, Saturn, Benthicarb) * **
Triallate (Far-Go, Avadex BW, Tri-allate) *
Triclopyr (Garlon, Grandstand, Redeem, Remedy) * **
Trifluralin (Treflan, Gowan, Tri-4, Trific)

Pesticides in water—Insecticides



Other insecticides detected

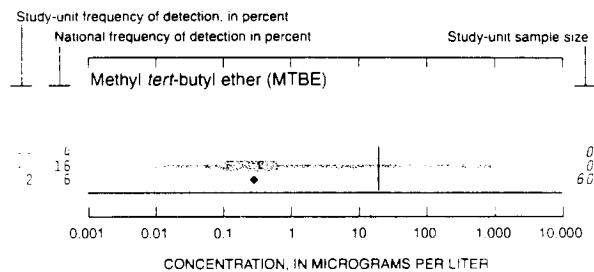
Carbaryl (Carbamine, Denapon, Sevin)
Carbofuran (Furadan, Curater, Yaltox)
Chlorpyrifos (Brodan, Dursban, Lorsban)
Fonofos (Dytonate, Capfos, Cudgel, Tycap) **

Insecticides not detected

Aldicarb (Terrik, Ambush, Pounce)
Aldicarb sulfone (Standak, aldoxycarb)
Aldicarb sulfoxide (Aldicarb breakdown product)
p,p'-DDE
Dieldrin (Panoram D-31, Octalox, Compound 497)
Disulfoton (Disyston, Di-Syston) **
Ethoprop (Mocap, Ethoprophos) * **
 α -HCH (alpha-BHC, alpha-lindane) **
 γ -HCH (Lindane, gamma-BHC)
3-Hydroxycarbofuran (Carbofuran breakdown product) * **
Malathion (Malathion)
Methiocarb (Slug-Geta, Grandslam, Mesuroil) * **
Methomyl (Lanox, Lannate, Acinate) **
Methyl parathion (Penncap-M, Folidol-M) **
Oxamyl (Vydate L, Pratt) **
Parathion (Roethyl-P, Alkron, Panthion, Phoskil) *
cis-Permethrin (Ambush, Astro, Pounce) * **
Phorate (Thimet, Granutox, Geomet, Rampart) * **
Propargite (Comite, Ornate, Ornate) * **
Propoxur (Baygon, Blattanax, Uden, Proprotax) * **
Terbufos (Conraven, Counter, Piarfox) **

Volatile organic compounds (VOCs) in ground water

These graphs represent data from 16 Study Units, sampled from 1996 to 1998



Other VOCs detected

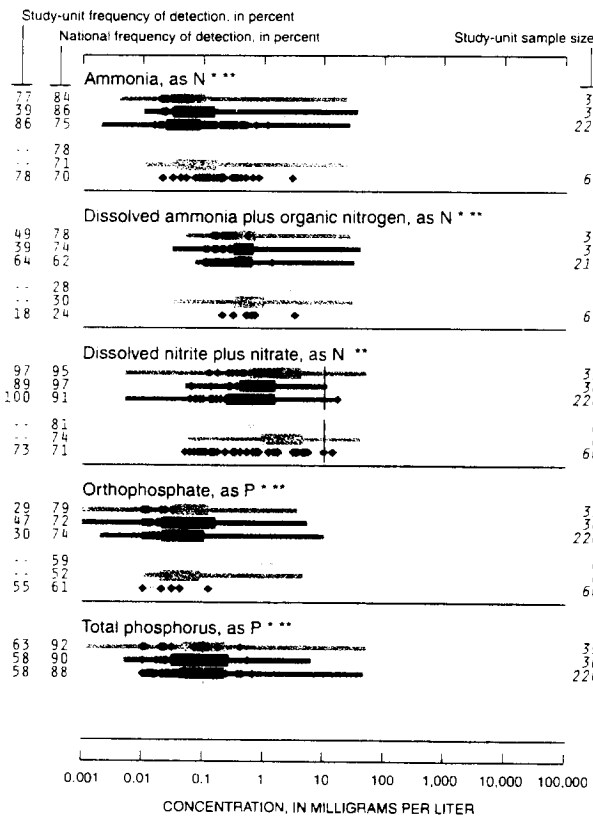
Benzene
Bromodichloromethane (Dichlorobromomethane)
2-Butanone (Methyl ethyl ketone (MEK)) *
sec-Butylbenzene *
Carbon disulfide *
Chlorobenzene (Monochlorobenzene)
Chlorodibromomethane (Dibromochloromethane)
Chloromethane (Methyl chloride)
Dichlorodifluoromethane (CFC 12, Freon 12)
cis-1,2-Dichloroethene ((Z)-1,2-Dichloroethene)
Dichloromethane (Methylene chloride)
1,2-Dimethylbenzene (*o*-Xylene)
1,4-Epoxy butane (Tetrahydrofuran, Diethylene oxide) *
Ethenylbenzene (Styrene)
Ethylbenzene (Phenylethane)
Iodomethane (Methyl iodide) *
Tetrachloroethene (Perchloroethene)
Tetrachloromethane (Carbon tetrachloride)
Tribromomethane (Bromoform)
1,1,2-Trichloro-1,2,2-trifluoroethane (Freon 113) *
1,1,1-Trichloroethane (Methylchloroform)
Trichloroethene (TCE)
Trichloromethane (Chloroform)
1,2,3-Trimethylbenzene (Hemimellitene) *
1,2,4-Trimethylbenzene (Pseudocumene) *

VOCs not detected

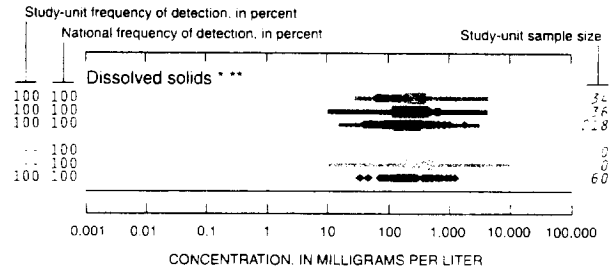
tert-Amylmethylether (*tert*-amyl methyl ether (TAME)) *
Bromobenzene (Phenyl bromide) *
Bromochloromethane (Methylene chlorobromide)
Bromoethene (Vinyl bromide) *
Bromomethane (Methyl bromide)
n-Butylbenzene (1-Phenylbutane) *
tert-Butylbenzene *
3-Chloro-1-propene (3-Chloropropene) *
1-Chloro-2-methylbenzene (*o*-Chlorotoluene)
1-Chloro-4-methylbenzene (*p*-Chlorotoluene)
Chloroethane (Ethyl chloride) *
Chloroethene (Vinyl chloride)
1,2-Dibromo-3-chloropropane (DBCP, Nemagon)
1,2-Dibromoethane (Ethylene dibromide, EDB)
Dibromomethane (Methylene dibromide) *
trans-1,4-Dichloro-2-butene ((Z)-1,4-Dichloro-2-butene) *
1,2-Dichlorobenzene (*o*-Dichlorobenzene)
1,3-Dichlorobenzene (*m*-Dichlorobenzene)
1,4-Dichlorobenzene (*p*-Dichlorobenzene)
1,2-Dichloroethane (Ethylene dichloride)
1,1-Dichloroethane (Ethylidene dichloride) *
1,1-Dichloroethene (Vinylidene chloride)
trans-1,2-Dichloroethene ((E)-1,2-Dichloroethene)
1,2-Dichloropropane (Propylene dichloride)
2,2-Dichloropropane *
1,3-Dichloropropane (Trimethylene dichloride) *
trans-1,3-Dichloropropene ((E)-1,3-Dichloropropene)
cis-1,3-Dichloropropene ((Z)-1,3-Dichloropropene)
1,1-Dichloropropene *
Diethyl ether (Ethyl ether) *
Diisopropyl ether (Diisopropylether (DIPE)) *
1,3 & 1,4-Dimethylbenzene (*m*-&*p*-Xylene)
Ethyl methacrylate *
Ethyl *tert*-butyl ether (Ethyl-*t*-butyl ether (ETBE)) *

- 1-Ethyl-2-methylbenzene (2-Ethyltoluene) *
- Hexachlorobutadiene
- 1,1,1,2,2,2-Hexachloroethane (Hexachloroethane)
- 2-Hexanone (Methyl butyl ketone (MBK)) *
- Isopropylbenzene (Cumene) *
- p-Isopropyltoluene (p-Cymene) *
- Methyl acrylonitrile *
- Methyl-2-methacrylate (Methyl methacrylate) *
- 4-Methyl-2-pentanone (Methyl isobutyl ketone (MIBK)) *
- Methyl-2-propenoate (Methyl acrylate) *
- Methylbenzene (Toluene)
- Naphthalene
- 2-Propanone (Acetone) *
- 2-Propenenitrile (Acrylonitrile)
- n-Propylbenzene (Isocumene) *
- 1,1,2,2-Tetrachloroethane *
- 1,1,1,2-Tetrachloroethane
- 1,2,3,4-Tetramethylbenzene (Phehnitene) *
- 1,2,3,5-Tetramethylbenzene (Isodurene) *
- 1,2,4-Trichlorobenzene
- 1,2,3-Trichlorobenzene *
- 1,1,2-Trichloroethane (Vinyl trichloride)
- Trichlorofluoromethane (CFC 11, Freon 11)
- 1,2,3-Trichloropropane (Allyl trichloride)
- 1,3,5-Trimethylbenzene (Mesitylene) *

Nutrients in water

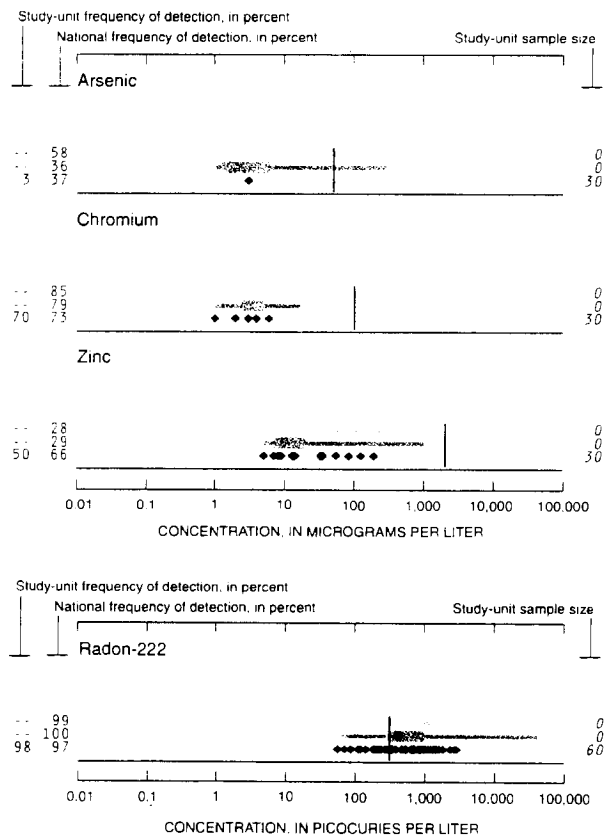


Dissolved solids in water



Trace elements in ground water

Trace-element data are only from the fractured rock aquifer survey. no trace element data were collected from the glacial sediments aquifer



Other trace elements detected

- Lead
- Selenium

Trace elements not detected

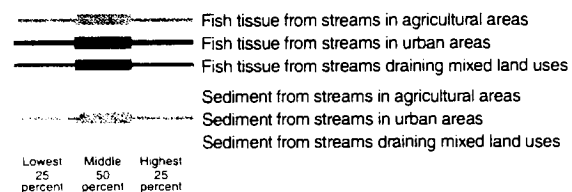
- Cadmium
- Uranium

CHEMICALS IN FISH TISSUE AND BED SEDIMENT

Concentrations and detection frequencies, Allegheny and Monongahela River Basins, 1996–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals. Study-unit frequencies of detection are based on small sample sizes; the applicable sample size is specified in each graph

- ◆ Detected concentration in Study Unit
- 66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- Not measured or sample size less than two
- 12 Study-unit sample size

National ranges of concentrations detected, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

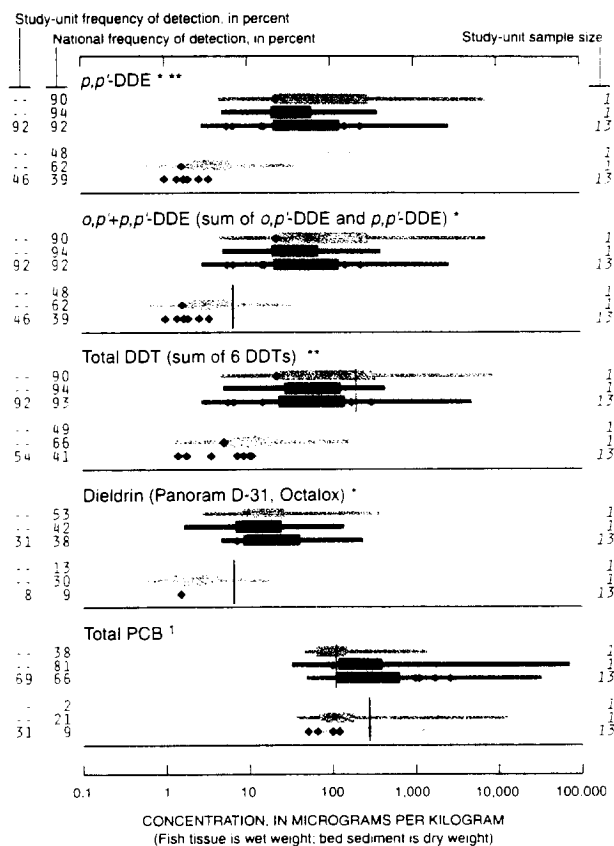
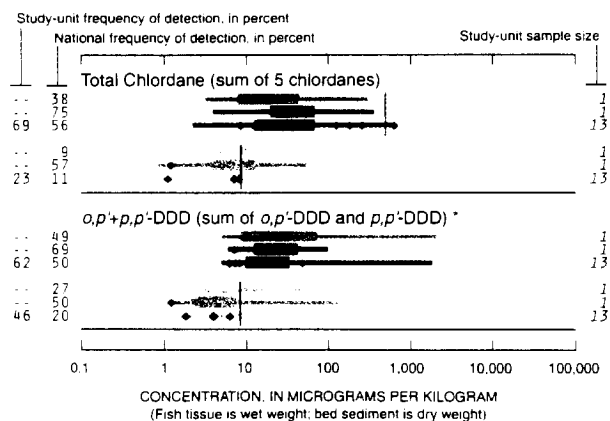


National benchmarks for fish tissue and bed sediment

National benchmarks include standards and guidelines related to criteria for protection of the health of fish-eating wildlife and aquatic organisms. Sources include the U.S. Environmental Protection Agency, other Federal and State agencies, and the Canadian Council of Ministers of the Environment

- | Protection of fish-eating wildlife (applies to fish tissue)
- | Protection of aquatic life (applies to bed sediment)
- No benchmark for protection of fish-eating wildlife
- ** No benchmark for protection of aquatic life

Organochlorines in fish tissue (whole body) and bed sediment



¹ The national detection frequencies for total PCB in sediment are biased low because about 30 percent of samples nationally had elevated detection levels compared to this Study Unit. See <http://water.usgs.gov/nawqa/> for additional information.

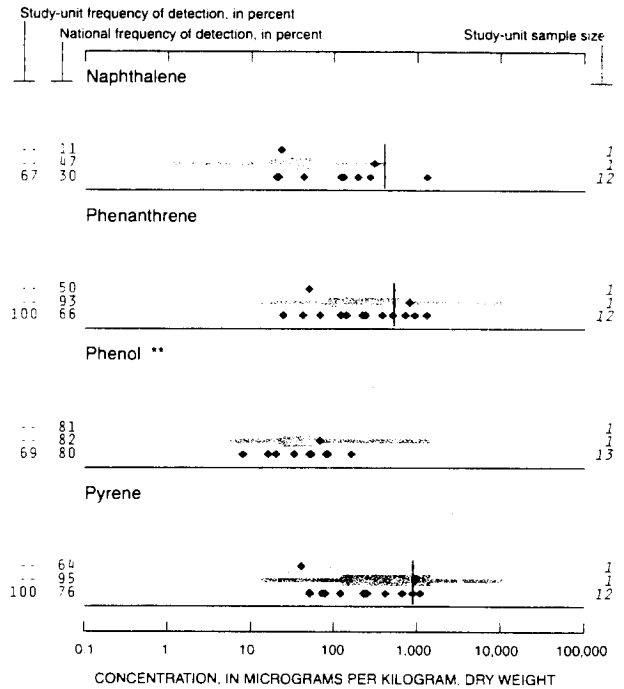
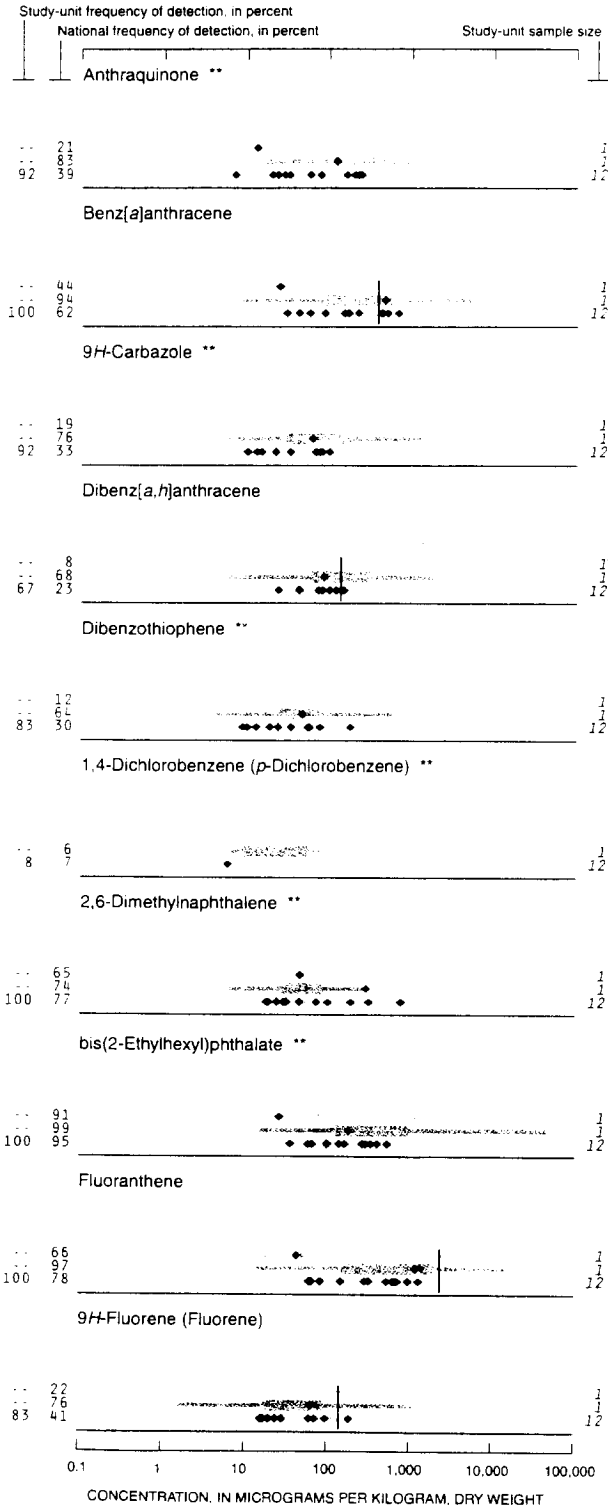
Other organochlorines detected

- o,p'+p,p'*-DDT (sum of *o,p'*-DDT and *p,p'*-DDT) *
- Dieldrin+aldrin (sum of dieldrin and aldrin) **
- Hexachlorobenzene (HCB) **
- o,p'*-Methoxychlor **
- Pentachloroanisole (PCA) * **
- cis*-Permethrin (Ambush, Astro, Pounce) * **
- trans*-Permethrin (Ambush, Astro, Pounce) * **

Organochlorines not detected

- Chloroneb (Chloronebe, Demosan) * **
- DCPA (Dacthal, chlorthal-dimethyl) * **
- Endosulfan I (alpha-Endosulfan, Thiodan) * **
- Endrin (Endrine)
- gamma-HCH (Lindane, gamma-BHC, Gammexane) *
- Total-HCH (sum of alpha-HCH, beta-HCH, gamma-HCH, and delta-HCH) **
- Heptachlor epoxide (Heptachlor breakdown product) *
- Heptachlor+heptachlor epoxide (sum of heptachlor and heptachlor epoxide) **
- Isodrin (Isodrine, Compound 711) * **
- p,p'*-Methoxychlor (Mariate, methoxychlore) * **
- Mirex (Dechlorane) **
- Toxaphene (Camphechlor, Hercules 3956) * **

**Semivolatile organic compounds (SVOCs)
in bed sediment**

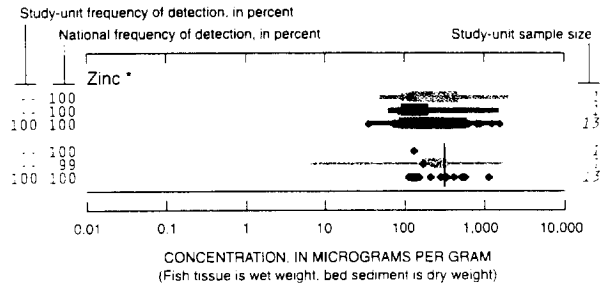


Other SVOCs detected

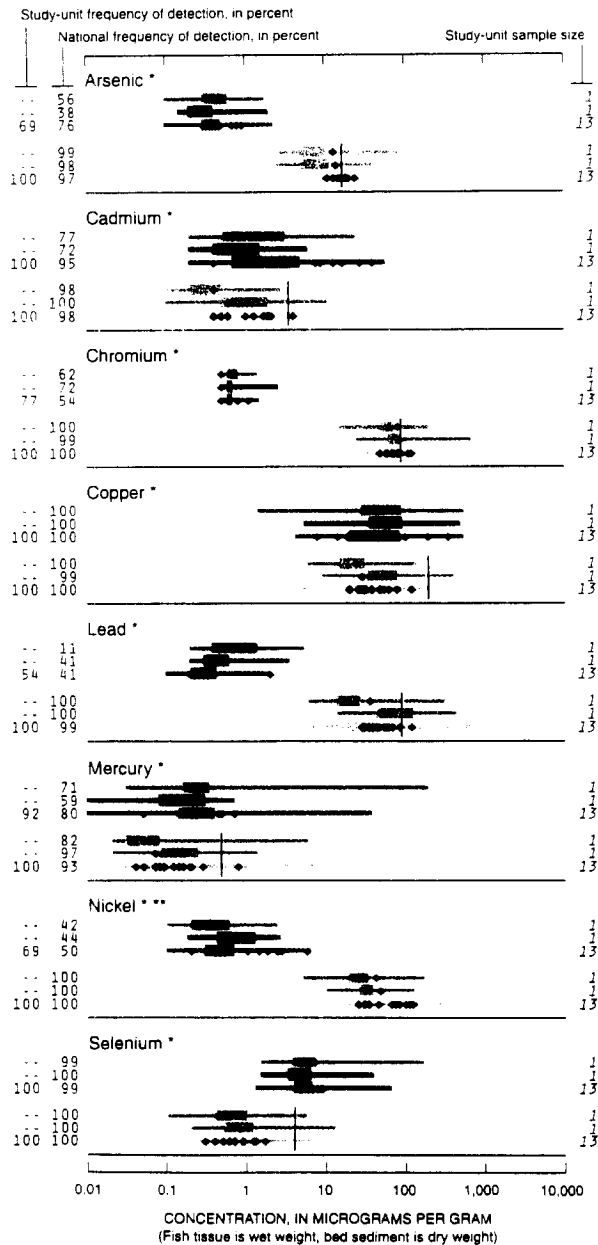
- Acenaphthene
 - Acenaphthylene
 - Acridine **
 - C8-Alkylphenol **
 - Anthracene
 - Benzo[a]pyrene
 - Benzo[b]fluoranthene **
 - Benzo[ghi]perylene **
 - Benzo[k]fluoranthene **
 - Butylbenzylphthalate **
 - Chrysene
 - p-Cresol **
 - Di-n-butylphthalate **
 - Di-n-octylphthalate **
 - Diethylphthalate **
 - 1,2-Dimethylnaphthalene **
 - 1,6-Dimethylnaphthalene **
 - 3,5-Dimethylphenol **
 - Dimethylphthalate **
 - 2-Ethylanthracene **
 - Indeno[1,2,3-cd]pyrene **
 - Isoquinoline **
 - 1-Methyl-9H-fluorene **
 - 2-Methylanthracene **
 - 4,5-Methylenepheneanthrene **
 - 1-Methylphenanthrene **
 - 1-Methylpyrene **
 - Phenanthridine **
 - Quinoline **
 - 2,3,6-Trimethylnaphthalene **
- SVOCs not detected**
- Azobenzene **
 - Benzo[c]cinnoline **
 - 2,2-Biquinoline **
 - 4-Bromophenyl-phenylether **
 - 4-Chloro-3-methylphenol **
 - bis(2-Chloroethoxy)methane **

R0024189

- 2-Chloronaphthalene **
- 2-Chlorophenol **
- 4-Chlorophenyl-phenylether **
- 1,2-Dichlorobenzene (*o*-Dichlorobenzene) **
- 1,3-Dichlorobenzene (*m*-Dichlorobenzene) **
- 2,4-Dinitrotoluene **
- Isophorone **
- Nitrobenzene **
- N*-Nitrosodi-*n*-propylamine **
- N*-Nitrosodiphenylamine **
- Pentachloronitrobenzene **
- 1,2,4-Trichlorobenzene **



Trace elements in fish tissue (livers) and bed sediment



BIOLOGICAL INDICATORS

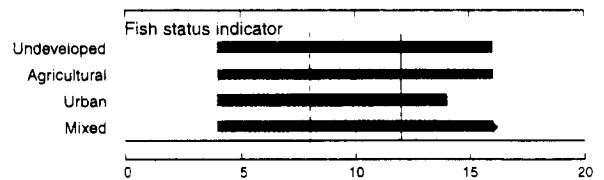
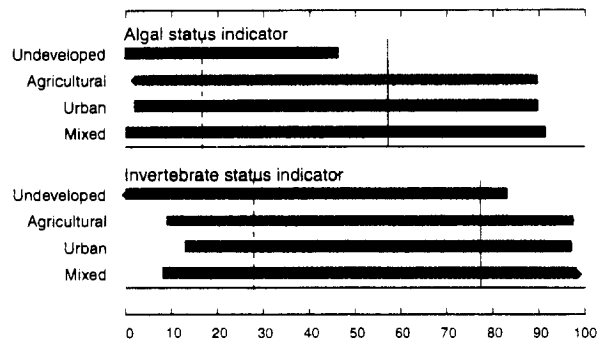
Higher national scores suggest habitat disturbance, water-quality degradation, or naturally harsh conditions. The status of algae, invertebrates (insects, worms, and clams), and fish provides a record of water-quality and stream conditions that water-chemistry indicators may not reveal. **Algal status** focuses on the changes in the percentage of certain algae in response to increasing siltation, and it often correlates with higher nutrient concentrations in some regions. **Invertebrate status** averages 11 metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. **Fish status** sums the scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that increase in association with water-quality degradation.

Biological indicator value, Allegheny and Monongahela River Basins, by land use, 1996-98

- ◆ Biological status assessed at a site

National ranges of biological indicators, in 16 NAWQA Study Units, 1994-98

- ▬ Streams in undeveloped areas
- ▬ Streams in agricultural areas
- ▬ Streams in urban areas
- ▬ Streams in mixed-land-use areas
- 75th percentile
- - - 25th percentile



A COORDINATED EFFORT

Coordination with many agencies and organizations in the Allegheny-Monongahela River Basins Study Unit was integral to the success of this water-quality assessment. We thank those who served as members of our liaison committee.

Federal Agencies

- National Park Service
- Natural Resources Conservation Service
- U.S. Army Corps of Engineers
- U.S. Department of Energy
- U.S. Environmental Protection Agency
- U.S. Fish and Wildlife Service
- U.S. Forest Service
- U.S. Geological Survey
- U.S. Office of Surface Mining, Reclamation and Enforcement

State Agencies

- Maryland Department of Natural Resources
- New York State Department of Environmental Conservation
- New York State Geological Survey
- Pennsylvania Department of Agriculture
- Pennsylvania Topographic & Geologic Survey
- Pennsylvania Department of Environmental Protection
- Pennsylvania Fish & Boat Commission
- West Virginia Department of Environmental Protection
- West Virginia Department of Natural Resources
- West Virginia Geological and Economic Survey

Local Agencies

- Allegheny County Department of Health
- Erie County Department of Public Health and Safety
- Greene County Conservation District
- Seneca Nation Health Department
- Somerset County Conservation District

Universities

- Allegheny College
- California University of Pennsylvania
- Carnegie Mellon University
- Pennsylvania State University
- University of Pittsburgh
- West Virginia University

Other public and private organizations

- Allegheny Watershed Network
- Allegheny County Sanitary Authority
- American Crop Protection Association
- French Creek Project
- Friends of the Cheat
- Jennings Environmental Education Center
- Ohio River Basin Commission
- Ohio River Valley Water Sanitation Commission
- Western Pennsylvania Coalition for Abandoned Mine Reclamation
- Western Pennsylvania Conservancy

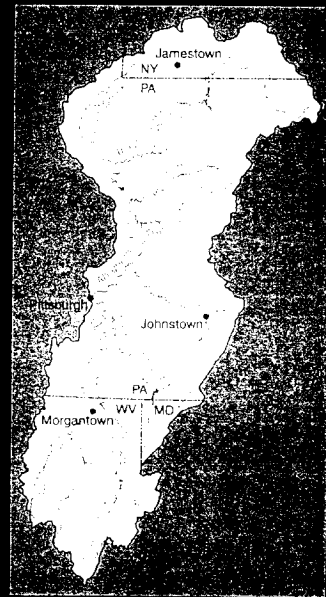
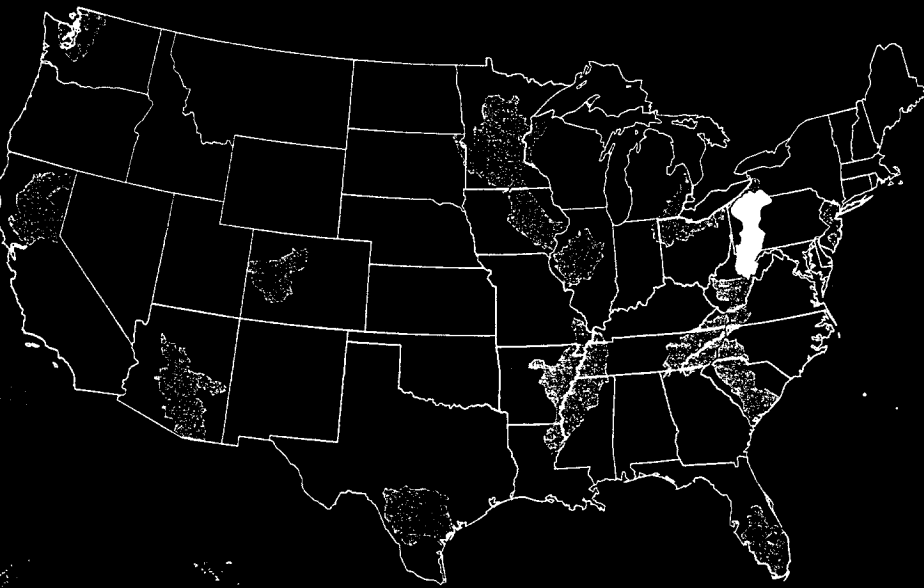
We thank the following individuals, agencies, and organizations for contributing to the success of this study

- Property owners throughout the Allegheny and Monongahela River Basins, for granting permission to access their property and to sample their wells.
- Pennsylvania Department of Environmental Protection offices in Ebensburg and Greensburg, Pa., for providing access to mine permits.
- West Virginia Department of Environmental Protection offices in Philippi and Nitro, W. Va., for providing access to mine permits.
- Randy Robinson, for providing white water rafting photographs and rafting guide services on the Cheat River.
- Dick Snyder, Pennsylvania Fish and Boat Commission, for providing access and copies of fish community and survey data.
- Jay Stauffer, Pennsylvania State University and students for assistance with electrofishing.
- Charles Durista, Pennsylvania Department of Environmental Protection, Jay Hawkins, U.S. Office of Surface Mining, and Jen Novak, Allegheny River Alliance, for providing review comments for this report.
- Technical and editorial reviewers in Lemoyne, Pa., Dr. J. Kent Crawford and Bruce Lindsey.
- The Reports Publishing Unit, Kim Otto, James Bubb, and Terriann Preston, in Lemoyne, Pa., for their outstanding efforts in bringing this report to publication.
- Project staff, including seasonal, temporary, student and volunteer staff members of the Pittsburgh office of the U.S. Geological Survey, who assisted in various aspects of this study, including Thomas Noonan, Gregory Wehner, Devon Renock, Erik Eismont, Scott Coulson, Jeffery Weitzel, Greg Hawkins, Jay Hawkins, Craig Uzelac, Douglas Chichester, Adam Locke, Rachel Mowery, Juliane Bowman Brown, and Julie Baldizar.

NAWQA

National Water-Quality Assessment (NAWQA) Program Allegheny and Monongahela Basins

Anderson and others—Water Quality in the Allegheny and Monongahela River Basins
U.S. Geological Survey Circular 1202



ISBN 0-607-95411-6



9 780607 954111

R0024192



science for a changing world

Water Quality in the Lake Erie-Lake Saint Clair Drainages

Michigan, Ohio, Indiana, New York, and Pennsylvania, 1996–98



**U.S. Department of the Interior
U.S. Geological Survey**

Circular 1203

R0024193

POINTS OF CONTACT AND ADDITIONAL INFORMATION

The companion Web site for NAWQA summary reports:

<http://water.usgs.gov/nawqa>

Lake Erie-Lake Saint Clair Drainages:

USGS State Representative
U.S. Geological Survey-Water Resources Division
6480 Doubletree Avenue
Columbus, Ohio 43229-1111
e-mail: shindall@usgs.gov
<http://oh.water.usgs.gov/nawqa/>

National NAWQA Program:

Chief, NAWQA Program
U.S. Geological Survey-Water
Resources Division
12201 Sunrise Valley Drive, M.S. 413
Reston, VA 20192
<http://water.usgs.gov/nawqa/>

Other NAWQA summary reports

River Basin Assessments

Albemarle-Pamlico Drainage Basin (Circular 1157)
Allegheny and Monongahela River Basins (Circular 1202)
Apalachicola-Chattahoochee-Flint River Basin (Circular 1164)
Central Arizona Basins (Circular 1213)
Central Columbia Plateau (Circular 1144)
Central Nebraska Basins (Circular 1163)
Connecticut, Housatonic and Thames River Basins (Circular 1155)
Eastern Iowa Basins (Circular 1210)
Georgia-Florida Coastal Plain (Circular 1151)
Hudson River Basin (Circular 1165)
Kanawha-New River Basins (Circular 1204)
Las Vegas Valley Area and the Carson and Truckee River Basins (Circular 1170)
Lower Illinois River Basin (Circular 1209)
Long Island-New Jersey Coastal Drainages (Circular 1201)
Lower Susquehanna River Basin (Circular 1168)
Mississippi Embayment (Circular 1208)
Ozark Plateaus (Circular 1158)
Potomac River Basin (Circular 1166)
Puget Sound Basin (Circular 1216)
Red River of the North Basin (Circular 1169)

Rio Grande Valley (Circular 1162)
Sacramento River Basin (Circular 1215)
San Joaquin-Tulare Basins (Circular 1159)
Santee River Basin and Coastal Drainages (Circular 1206)
South-Central Texas (Circular 1212)
South Platte River Basin (Circular 1167)
Southern Florida (Circular 1207)
Trinity River Basin (Circular 1171)
Upper Colorado River Basin (Circular 1214)
Upper Mississippi River Basin (Circular 1211)
Upper Snake River Basin (Circular 1160)
Upper Tennessee River Basin (Circular 1205)
Western Lake Michigan Drainages (Circular 1156)
White River Basin (Circular 1150)
Willamette Basin (Circular 1161)

National Assessments

The Quality of Our Nation's Waters—Nutrients and Pesticides (Circular 1225)

Front cover: Marblehead Lighthouse, a symbol of Lake Erie, located on the Marblehead Peninsula, near Sandusky, Ohio. (Photograph from Ohio Lake Erie Commission, Toledo, Ohio).

Back cover: left, waterways such as the harbor of the Ashtabula River, Ohio, are used for industry, commerce, and recreation; right, country roads and small farms are still common in some areas.

Water Quality in the Lake Erie-Lake Saint Clair Drainages

Michigan, Ohio, Indiana, New York, and Pennsylvania, 1996–98

By Donna N. Myers, Mary Ann Thomas, Jeffrey W. Frey, Stephen J. Rheaume,
and Daniel T. Button

U.S. GEOLOGICAL SURVEY CIRCULAR 1203

R0024195

U.S. DEPARTMENT OF THE INTERIOR
BRUCE BABBITT, Secretary

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

2000

Free on application to the
U.S. Geological Survey
Information Services
Box 25286 Federal Center
Denver, CO 80225

Or call: 1-888-ASK-USGS

Library of Congress Cataloging-in-Publications Data

Water quality in the Lake Erie-Lake St. Clair Drainages, Michigan, Ohio, Indiana, New York, and Pennsylvania, 1996-98 / by D.N. Myers...[et al.].
p. cm. -- (U.S. Geological Survey Circular ; 1203)
Includes bibliographical references.
ISBN 0-607-95421-3 (alk. paper)
1. Water quality--Erie, Lake, Watershed. 2. Water quality--Saint Clair, Lake, Watershed (Mich. and Ont.) I. Myers, Donna N. II. Geological Survey (U.S.) III. Series.

TD222.3 .W374 2000
363.739'42'097712--dc21

00-049458

R0024196

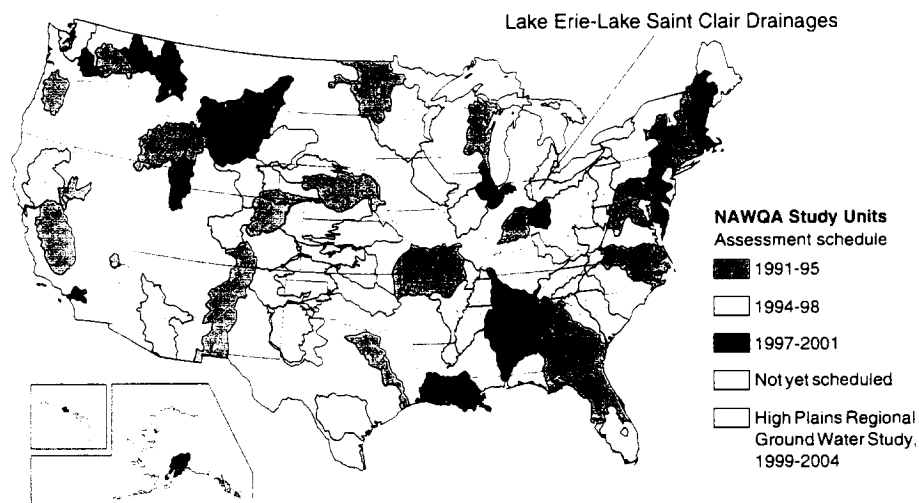
CONTENTS

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM.....	IV
SUMMARY OF MAJOR FINDINGS	1
Stream and River Highlights	1
Ground Water Highlights.....	2
INTRODUCTION TO THE LAKE ERIE-LAKE SAINT CLAIR DRAINAGES.....	3
Major water-quality issues	3
Environmental setting and hydrologic conditions.....	3
MAJOR FINDINGS	6
Pesticides were detected in every stream sample.....	6
The pesticides most frequently detected in streams were generally those most heavily applied	7
Pesticide metabolites were frequently detected.....	7
Pesticide concentrations in streams are related to land use, rainfall, runoff, and season	8
Pesticide concentrations in streams have public-health and economic consequences	8
Another consequence of heavy pesticide use is the potential effect on aquatic life	9
NATIONAL PERSPECTIVE—Concentrations of nutrients and herbicides in certain streams are among the highest in the Nation.....	10
Runoff from agricultural land to streams is an important pathway of phosphorus to Lake Erie	10
Decreases in the amounts of suspended sediment carried by rivers correspond to increases in farmers' use of conservation tillage	12
Contamination of bed sediments may be causing adverse effects on aquatic life.....	13
Land use and chemical use contribute to the contamination of sediments and fish	14
Bioaccumulative contaminants in fish may pose a risk to fish-eating wildlife	15
Land use appears to affect fish-community composition	16
Recent residential development has had a widespread effect on ground-water quality.....	18
Ground water is vulnerable to contamination in agricultural areas underlain by till	20
NATIONAL PERSPECTIVE—Nitrate and herbicides in ground water in agricultural areas	22
STUDY-UNIT DESIGN.....	23
GLOSSARY	25
REFERENCES	27
APPENDIX—WATER-QUALITY DATA FROM THE LAKE ERIE-LAKE SAINT CLAIR DRAINAGES IN A NATIONAL CONTEXT	29

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

THIS REPORT summarizes major findings about water quality in the Lake Erie-Lake Saint Clair Drainages that emerged from an assessment conducted between 1996 and 1998 by the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program. Water quality is discussed in terms of local and regional issues and compared to conditions found in all 36 NAWQA study areas, called Study Units, assessed to date. Findings are also explained in the context of selected national benchmarks, such as those for drinking-water quality and the protection of aquatic organisms. The NAWQA Program was not intended to assess the quality of the Nation's drinking water, such as by monitoring water from household taps. Rather, the assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water monitoring programs. The comparisons made in this report to drinking-water standards and guidelines are only in the context of the available untreated resource. Finally, this report includes information about the status of aquatic communities and the condition of in-stream habitats as elements of a complete water-quality assessment.

Many topics covered in this report reflect the concerns of officials of State and Federal agencies, water-resource managers, and members of stakeholder groups who provided advice and input during the Lake Erie-Lake Saint Clair Drainages assessment. Residents who wish to know more about water quality where they live will find this report informative as well.



THE NAWQA PROGRAM seeks to improve scientific and public understanding of water quality in the Nation's major river basins and ground-water systems. Better understanding facilitates effective resource management, accurate identification of water-quality priorities, and successful development of strategies that protect and restore water quality. Guided by a nationally consistent study design and shaped by ongoing communication with local, State, and Federal agencies, NAWQA assessments support the investigation of local issues and trends while providing a firm foundation for understanding water quality at regional and national scales. The ability to integrate local and national scales of data collection and analysis is a unique feature of the USGS NAWQA Program.

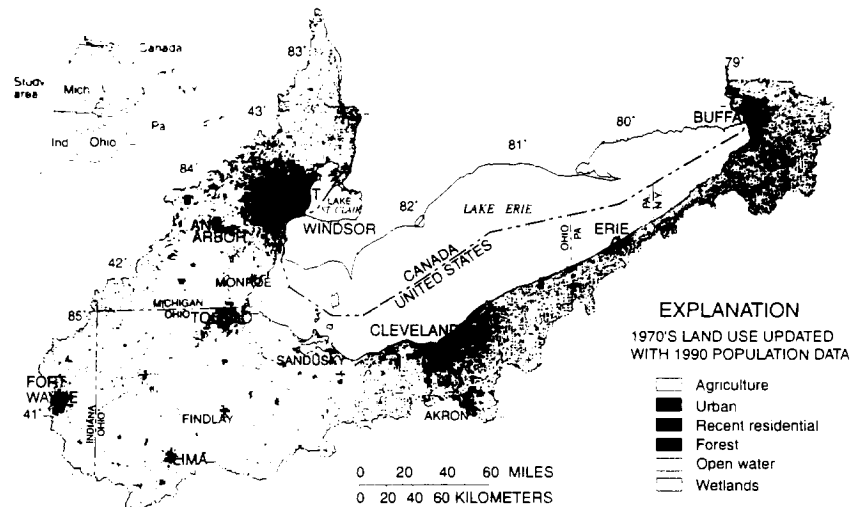
The Lake Erie-Lake Saint Clair Drainages is one of 51 water-quality assessments initiated since 1991, when the U.S. Congress appropriated funds for the USGS to begin the NAWQA Program. As indicated on the map, 36 assessments have been completed, and 15 more assessments will conclude in 2001. Collectively, these assessments cover about one-half of the land area of the United States and include water resources that are available to more than 60 percent of the U.S. population.

SUMMARY OF MAJOR FINDINGS

Stream and River Highlights

Water quality in the Lake Erie-Lake Saint Clair Drainages is greatly influenced by land use and human activities. A major pathway for contaminant transfer from the land surface to streams is storm runoff from urban and agricultural areas.

As a result of herbicides in runoff, concentrations in streams were in the top 25 percent of streams nationwide and many public-water supplies must treat stream water to reduce herbicide concentrations. As a result of nutrients in runoff, concentrations of total phosphorus and nitrate in some small streams in agricultural areas and in



Land use in the Lake Erie-Lake Saint Clair Drainages was predominantly agricultural and urban. This level of human activity has had substantial effects on water quality in the region.

Indicators of Stream Quality	Small streams		Major rivers	
	Urban	Row crop	Pasture/forest	Mixed
Pesticides ¹				
Total phosphorus ²				
Nitrate ³				
Arsenic ⁴				
Other trace elements ⁴				
Polycyclic aromatic hydrocarbons ⁵				
Polychlorinated biphenyls ⁵				

- Percentage of samples with concentrations **equal to or greater than** a health-related national guideline for drinking water, aquatic life, or above a national goal for preventing excess algal growth (^a Percentage is 1 or less and may not be clearly visible.)
- Percentage of samples with concentrations **less than** a health-related national guideline for drinking water, aquatic life, or below a national goal for preventing excess algal growth (^b Percentage is 1 or less and may not be clearly visible.)
- Percentage of samples with **no detection** (^c Percentage is 1 or less and may not be clearly visible.)

¹ Insecticides, herbicides, and pesticide metabolites, sampled in water.
² Total phosphorus, sampled in water.
³ Nitrate (as nitrogen), sampled in water.
⁴ Arsenic, cadmium, copper, lead, mercury, and zinc, sampled in sediment.
⁵ Industrial chemicals, combustion byproducts, and polychlorinated biphenyls sampled in sediment.

major rivers were in the top 25 percent of streams nationwide. Concentrations of nitrate, although elevated relative to many other streams in the Nation, were infrequently greater than the drinking-water standard of 10 milligrams per liter.

Contamination of the bed sediments of small streams and major rivers by persistent and bioaccumulative contaminants was prevalent. The highest concentrations of PCBs (polychlorinated biphenyls) and mercury were detected in streams draining highly populated urban and mixed land-use areas. Detections of contaminants in fish tissues indicate bioaccumulation; in fact, bioaccumulation of PCBs and DDT in some fish species presents a health risk to fish-eating wildlife.

- The pesticides detected most frequently were among those applied in the greatest quantities to agricultural and mixed land-use areas. The herbicides atrazine, acetochlor, cyanazine, metolachlor, and simazine were detected in 50 to 100 percent of samples. (p. 6)

- Several heavily used herbicides and insecticides were detected in spring and summer at or above a standard for drinking water or a guideline for aquatic life. Elevated pesticide concentrations in streams persisted for 4 to 6 weeks after applications in agricultural and mixed-land-use areas. (p. 8)

- Annual average concentrations of total phosphorus were greater than the U.S. Environmental Protection Agency recommended level for control of nutrient enrichment at 8 of 10 sites sampled in small streams and major rivers draining agricultural and mixed-use land. Streams draining row-crops and mixed-use land are major pathways of phosphorus to Lake Erie. (p. 10)

- Contaminants detected most often in the bed sediments of small streams and major rivers were arsenic, cadmium, copper, lead, mercury, zinc, PCBs, and PAHs (polycyclic aromatic hydrocarbons). The concentrations of arsenic, mercury, PCBs, and PAHs were equal to or greater than sediment guidelines, indicating probable adverse effects on aquatic life, in about 11 to 30 percent of samples. (p. 13)

- The most frequently detected contaminants in fish were highly persistent contaminants—DDT, chlordane, dieldrin, PCBs, and mercury. Except for mercury, use of these compounds in industry and agriculture in the United States was discontinued 15 to 25 years ago. (p. 14)

- Agricultural land use appears to be affecting fish communities in streams draining areas of row-crops. As the amount of row-crops increased relative to forested land, the number of pollution-intolerant fish species decreased. It appears that pollution-intolerant fish can live where agriculture is the primary land use when streams are protected by natural cover. (p. 17)

Major Influences on Surface-Water Quality and Aquatic Biota

- Storm runoff
- Land use and chemical releases
- Bioaccumulative and persistent contaminants

Trends in Surface-Water Quality

Suspended-sediment discharges from the Maumee River Basin decreased by 11.2 percent over the period 1970–98 and corresponded to increased use of conservation tillage to control soil erosion.

Ground Water Highlights

The glacial aquifer is the major source of drinking water in the northwestern part of the study area. In this area, ground-water quality is affected by a combination of human and natural factors. Land use determines which chemicals are used; how readily these chemicals are transported to the ground water is affected by geology.

- In residential areas underlain by sand and gravel, more than 75 percent of ground water recharged since 1953 shows evidence of human activities in the form of nitrate, chloride, or volatile organic compounds (VOCs). Probable sources are (1) septic systems containing human waste and household chemicals, (2) road salt and gasoline residue from paved surfaces, (3) waste from water softeners, or (4) lawn fertilizer. Pesticides were rarely detected. (p. 18)

- In the agricultural study area, which is underlain by till, nitrate and herbicides were detected less frequently than in

most other agricultural areas of the Nation. This observation partially supports the belief that till or tile drains protect the aquifer from contamination. Nevertheless, the ground water is still vulnerable to contamination; almost 60 percent of shallow ground water contained herbicides or elevated concentrations of nitrate. Herbicides were predominantly detected as breakdown products. (p. 20)

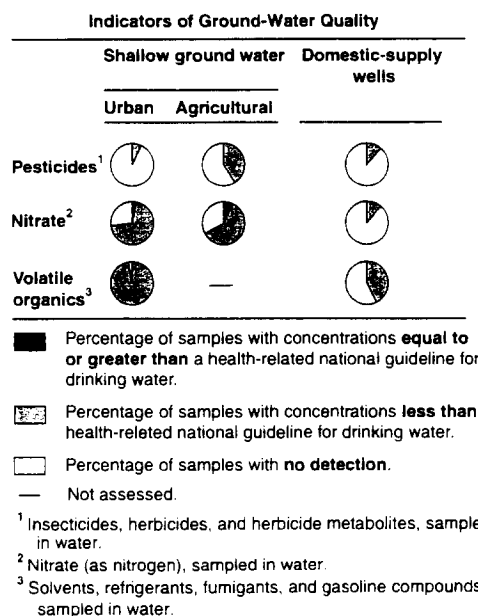
- In residential and agricultural areas, samples from domestic wells met health-related drinking-water standards. However, ground water affected by human activities was detected at depths below 25 feet, the minimum required depth for wells in Ohio and Michigan. (p. 19)

Major Influences on Ground-Water Quality

- Septic systems, roads, and lawns in residential areas
- Herbicides and fertilizer in agricultural areas
- Geology, especially deposits at land surface
- Well depth and ground-water age

Trends in Ground-Water Quality

A significant change in ground-water quality is linked to recent residential development near Detroit. Ground water recharged before 1953 (which predates suburbanization) has concentrations of chemical constituents typical of natural water. In contrast, ground water recharged after 1953 has significantly higher concentrations of constituents derived from human activities.



INTRODUCTION TO THE LAKE ERIE-LAKE SAINT CLAIR DRAINAGES

Major water-quality issues

Streams and aquifers in the Lake Erie-Lake Saint Clair Drainages study area provide water for 10.6 million people. The major water-quality issues in the study area are similar to those facing the rest of the Nation. Natural-resource managers are interested in the effects of urban and agricultural land use on the quality of surface water and ground water, specifically:

- pesticide and nutrient contamination of stream water used for public supply
- nutrient enrichment and sedimentation in streams and subsequent effects on aquatic biota
- impairment of aquatic life from bioaccumulation of organochlorine compounds and mercury
- consumption advisories for certain fish due to bioaccumulation of PCBs and mercury
- degradation of ground water used for domestic supply in areas of new residential development and areas of row-crops

Environmental setting and hydrologic conditions

Lake Erie is the 11th-largest freshwater lake in the world. About two-thirds of the Lake Erie watershed is in the United States, and is referred to as the Lake Erie-Lake Saint Clair Drainages. (For simplicity, this area will be referred to as the "basin.") This 22,300-square-mile area includes sections of Michigan, Indiana, Ohio, Pennsylvania, and New York. The land surface is gently rolling to nearly flat lying. Eight major rivers, all with drainage areas greater than 500 square miles, flow into the system.

Land use

In the Lake Erie-Lake Saint Clair Drainages, about 75 percent of the

land area is in agriculture, 11 percent is urban land, 11 percent is forest, and 3 percent is open water or wetlands. (See land-use map in summary section.)

Corn, soybeans, and wheat are grown predominantly in the western part of the basin (fig. 1). Other agricultural land uses include pasture and forage crops, grown predominantly in the eastern part of the basin (fig. 2). Orchards and vineyards, located mainly along the shores of Lake Erie and in parts of Michigan, is the least widespread type of agricultural land use.

Urban land is an important component of land use in the basin (fig. 3). Detroit and Pontiac, Mich.; Akron, Cleveland, Lima, and Toledo, Ohio; Fort Wayne, Ind.;

Erie, Pa.; and Buffalo, N.Y., were historically significant contributors to industrial America through the production of automobiles, rubber, steel, petroleum, and chemicals. These cities remain important industrial and manufacturing centers. Major urban centers rely on abundant supplies of water for shipping, electric power generation, industry, domestic consumption, and waste assimilation.

Forest and wetlands, once common throughout the basin, have been greatly reduced since the mid-1800s. The greatest percentage of forested land today is in Pennsylvania and New York. Metropolitan parks also support substantial urban forests. Wetlands lie primarily along Lake Erie and Lake Saint



Figure 1. Agriculture is a major industry in the western part of the basin, where corn and soybeans are grown. (Photograph from Natural Resources Conservation Service.)

Figure 2. Pasture and forage crops are found in the eastern part of the basin.



Figure 3. The port of Toledo, Ohio, on the Maumee River exports large amounts of grain from Midwestern farms. Toledo is an industrial center for automobile manufacturing. (Photograph from Ohio Lake Erie Commission.)

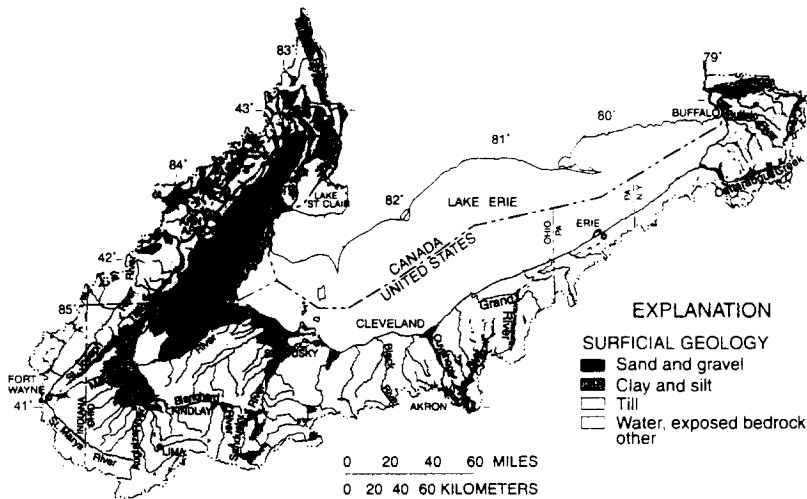


Figure 4. Glacial deposits throughout the basin influence the quality and flow of surface water and ground water.

Clair from Detroit, Mich., to Sandusky, Ohio.

Geology

A layer of glacial deposits from 20 to 200 feet thick mantles the entire basin and overlies limestone, sandstone, or shale bedrock. The glacial sediments consist of sand and gravel, till, and fine-grained sediments composed of fine sands, silts, and clay (fig. 4). Sand and gravel are present as discontinuous deposits in river valleys and in the western part of the basin. The most widespread deposit is till, a mixture

of grain sizes. Tills in the Lake Erie-Lake Saint Clair Drainages contain a high content of clay, which slows infiltration of rainfall into the ground.

Water use

More than 10.6 billion gallons of water is used each day in the basin (fig. 5). Cooling during power generation accounts

for 71 percent of the water use. Public and domestic supply account for 17 percent, and industry and mining account for 10 percent of the total use. Normal precipitation is generally adequate for agriculture, so irrigation accounts for less than 1 percent of water use.

Of the more than 1.8 billion gallons per day used for public and domestic supply, about 88 percent is from surface-water sources. Most of the major cities are near Lake Erie and Lake Saint Clair and therefore derive their water from the lakes or their connecting channels (fig. 6). Major rivers are an important source of drinking water for inland cities such as Fort

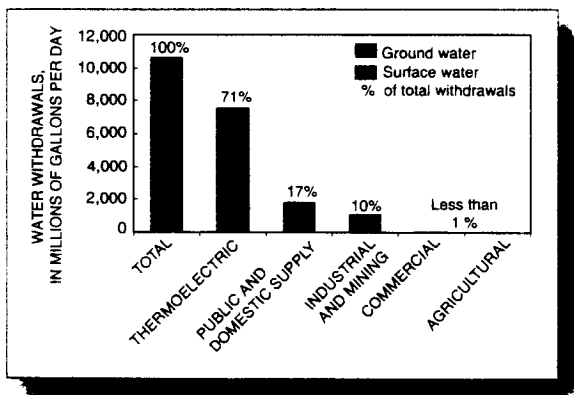


Figure 5. Water withdrawals in the basin are dominated by thermoelectric power production.

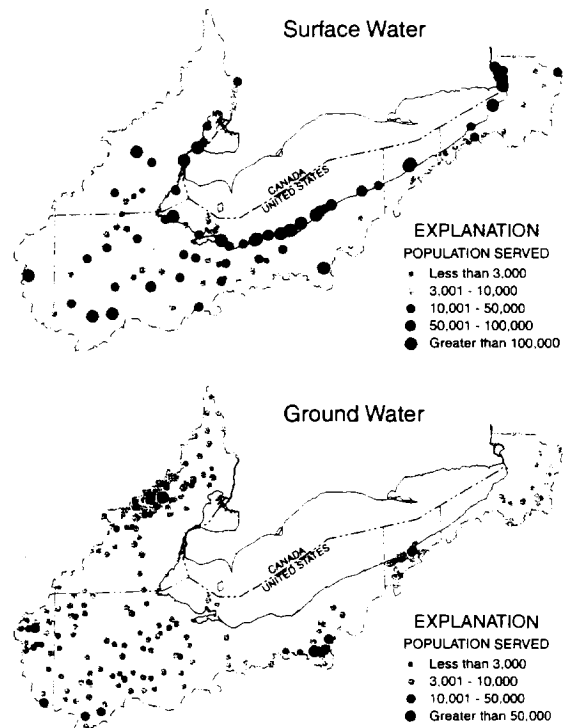


Figure 6. Many public-water supplies with surface-water sources are near Lake Erie or major rivers, whereas public-water supplies with ground-water sources generally are some distance from the lake and major rivers.

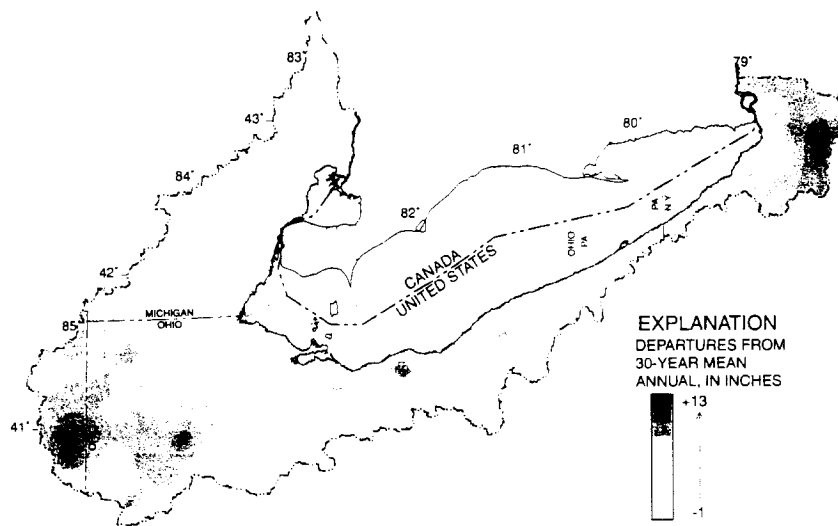


Figure 7. Departures from 30-year mean annual precipitation patterns for water years 1996–98, based on data from 133 weather stations.

Wayne, Ind., and Akron and Lima, Ohio.

About 12 percent of water for human consumption comes from ground water. The cities that use ground water as their major source of drinking water are generally inland (fig. 6). The most productive aquifers are in glacial deposits and limestone and sandstone bedrock.

Hydrologic conditions

Average annual precipitation across the basin ranges from 28 to 47 inches. Precipitation is highest in the northeast because of lake effect, whereby cool, dry air picks up moisture as it travels over warmer lake water and then produces rain or snow as it reaches land. The lowest amounts of precipitation are in the northwestern part of the basin in Michigan. During the study, annual precipitation was 4 to 5 inches above the 30-year mean annual. Some areas received 13 inches of precipitation above the mean, and a few isolated areas

received slightly less than the mean (fig. 7).

The highest streamflows are typically in February, March, and April, as a result of increased precipitation, cold temperatures and little vegetative growth. The lowest streamflows are in August, September, and October (Casey and

others, 1997). During low streamflow, ground water typically contributes most of the flow.

Mean monthly streamflows for the Maumee River at Waterville, Ohio, at the west end of the basin, exceeded the long-term averages by a factor of 2 or more for some months in 1997–98 (fig. 8). Overall, the mean annual streamflow in the Maumee River was 25 percent above average during the sampling period. For Cattaraugus Creek at Gowanda, N.Y., at the east end of the basin, mean monthly streamflow during 1996–98 closely followed the long-term average (fig. 8). In most areas of the basin, higher than average streamflows were observed in water years 1996–98.

Increased streamflow tends to carry higher concentrations of materials associated with land-surface runoff. Low streamflow conditions tend to increase concentrations of materials associated with industrial and municipal discharges or ground water.

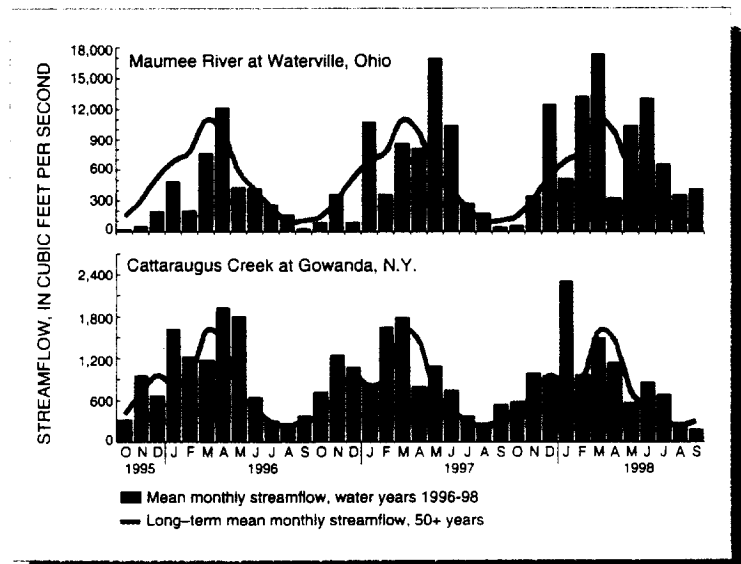


Figure 8. Mean monthly streamflows during water years 1996–98 greatly exceeded long-term averages in some months.

MAJOR FINDINGS

The quality of streams and ground water in the Lake Erie-Lake Saint Clair Drainages is affected by a complex combination of natural factors (precipitation, streamflow, geology) and human factors (land use, chemical use, population density). Specific parts of the basin were targeted for investigation because they have particular combinations of natural and human factors. (See Appendix for details.) A primary factor that was found to influence water quality was land use, and this factor is emphasized throughout the report.

Pesticides were detected in every stream sample

Between March 1996 and February 1998, 305 samples were collected from 10 streams in the basin. Every sample contained at least one pesticide, and most contained mixtures of several pesticides. Some samples contained a mixture of 18 pesticide compounds, which is among the highest number of pesticides detected in a sample nationally.

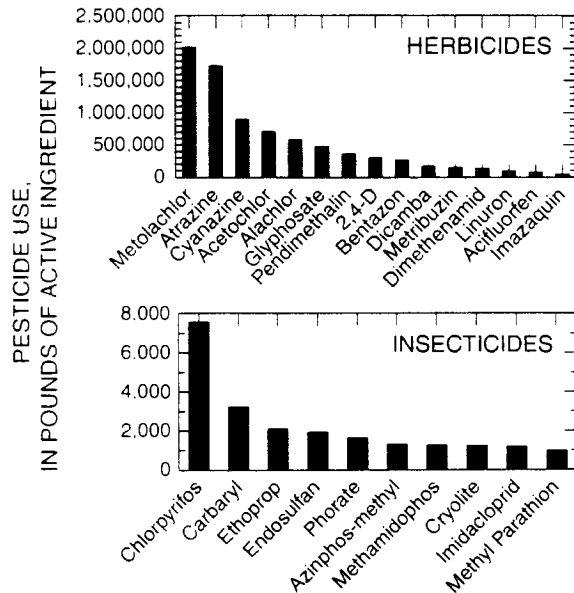


Figure 9. A total of nearly 8.3 million pounds of pesticides was applied in the agricultural areas of the Lake Erie-Lake Saint Clair Drainages during 1994-95. (Data from Brody and others, 1997.)

More pesticides were detected in large streams than small streams. Overall, 30 different pesticides were detected within the basin. The number of pesticides detected was greater in large

streams and rivers that drain areas of mixed land use (urban and agricultural) than in smaller streams that drain basins dominated by a single land use.

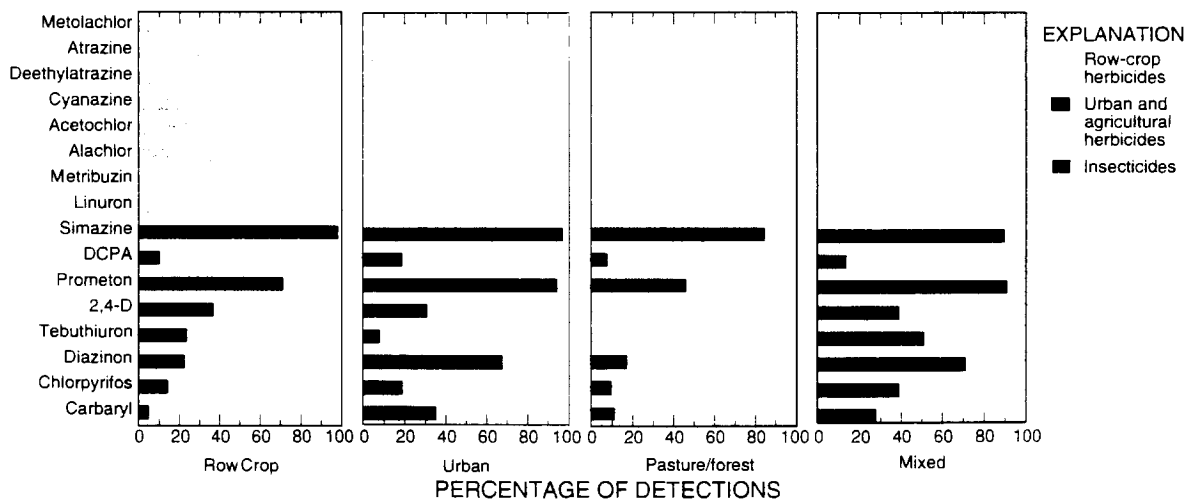


Figure 10. Metolachlor, atrazine, deethylatrazine, cyanazine, and simazine were the most frequently detected herbicides. Diazinon, chlorpyrifos, and carbaryl were the most frequently detected insecticides.

The pesticides most frequently detected in streams were generally those most heavily applied

Pesticide applications to crops in the Lake Erie-Lake Saint Clair Drainages are among the highest nationwide. The five most heavily applied agricultural pesticides—metolachlor, atrazine, cyanazine, acetochlor, and alachlor—were also among the most frequently detected (figs. 9 and 10). Atrazine was detected in every stream-water sample collected during 1996–98 (fig. 10). Metolachlor was detected in 99 percent of samples, and simazine was detected in greater than 80 percent of samples, regardless of land use (fig. 10). Two pesticides that are heavily used in the basin were not analyzed for in this study: (1) glyphosate, an herbicide used in agricultural and urban areas, and (2) endosulfan, an organochlorine insecticide, which was analyzed for in streambed sediment but not in water.

Pesticides detected more frequently in urban than in agricultural areas include the herbicide prometon and the insecticides diazinon and chlorpyrifos (figs. 10 and 11). Chlorpyrifos (Dursban), which was detected in 22 percent of the samples, was recently restricted for residential and commercial use by the U.S. Environmental Protection Agency (USEPA).

Historical-use pesticides are seldom detected in water. Aldrin and DDT are organochlorine insecticides banned in the mid-1970s. Their metabolites (dieldrin and DDE) were each detected in about 1 percent of stream samples. In contrast, total DDT was detected in

nearly 35 percent of all streambed sediment samples (fig. 19, p. 13).

Pesticide metabolites were frequently detected

Little is known about pesticide metabolites (breakdown products) on a watershed level. After application, most pesticides adsorb to soil, infiltrate to ground water, volatilize, or break down to a metabolite (Larson and others, 1999). Deethylatrazine, a metabolite of atrazine, was detected in every sample collected during 1996–98. Additional

triazine and acetanilide herbicides and their metabolites were intensively sampled for at one site, the St. Joseph River near Newville, Ind. Here, the metabolites of triazine compounds, like atrazine, showed the same seasonal pattern and relative concentrations as parent compounds. In contrast, concentrations of the acetanilide parent compounds—alachlor, acetochlor, and metolachlor—were lower than those of their metabolites for most of the year (fig. 12).

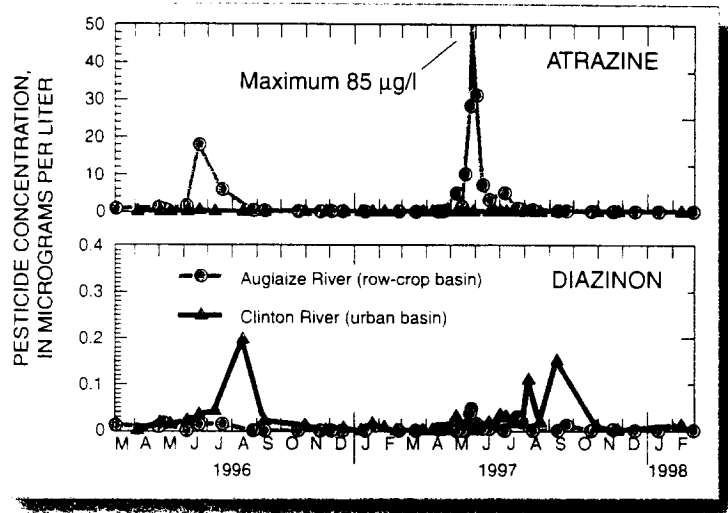


Figure 11. The concentration of atrazine was higher in streams draining row-crops, whereas the concentration of diazinon was higher in streams draining urban-land use.

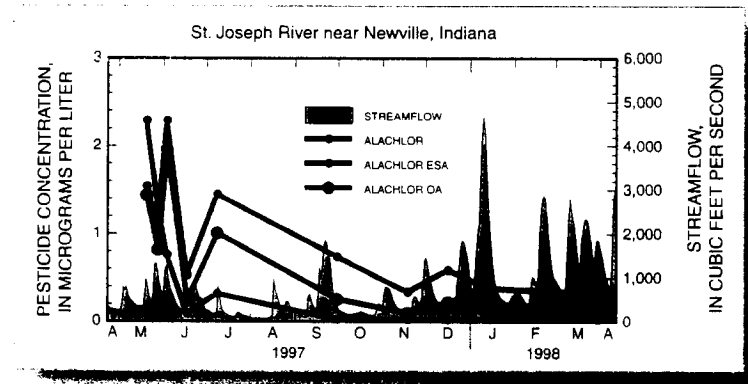


Figure 12. Metabolite concentrations for alachlor, as well as acetochlor and metolachlor (not shown), were greater than parent compounds for much of the year.

No benchmarks for metabolites have been set to protect human health or aquatic life, but some metabolites are believed to be as toxic as parent compounds (Day, 1991). Because metabolites may account for a significant amount of the total pesticides in the environment, they are considered a water-quality concern.

Pesticide concentrations in streams are related to land use, rainfall, runoff, and season

Herbicide concentrations were highest in streams draining row-crops followed by streams draining mixed-use and urban land, and were lowest in streams draining pasture/forested land. Atrazine was detected at the highest concentrations, as high as 85 µg/L (micrograms per liter), in streams draining row crops and at much lower concentrations in streams draining urban land (fig. 11).

Rainfall and runoff affected the concentrations of atrazine in the wet years of 1996–97 compared to the dry year of 1998 in the St. Joseph River near Newville, Ind. (fig. 13). In 1996, streamflow was 330 to 340 percent above average in May and June; in 1997, streamflow was 103 to 129 percent above average in May and June; and in 1998, streamflow was only 36 to 78 percent of the average. The median atrazine concentration detected in the stream was 0.40 µg/L in 1996, 0.45 µg/L in 1997, and 0.14 µg/L in 1998.

Herbicide concentrations in streams are typically highest after application and steadily decrease thereafter. Elevated concentrations of the most heavily used herbicides—metolachlor, atrazine, cyanazine, and ace-

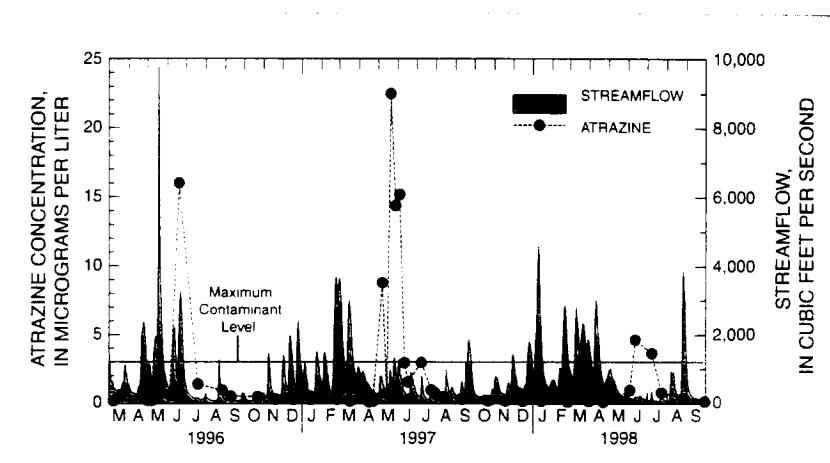


Figure 13. Pesticides such as atrazine show a seasonal trend with streamflow, as shown for the St. Joseph River near Newville, Ind. Differences in the amount and timing of precipitation affect the concentrations, as illustrated by the differences between the wet years of 1996–97 and the dry year of 1998.

tochlor—were detected for 4 to 6 weeks after rainfall and runoff in the spring and early summer (fig. 13, table 1). Pesticides used mostly in urban areas, such as prometon and diazinon, are typically applied in late summer, and this is when the highest concentrations were detected in streams (fig. 11).

Pesticide concentrations in streams have public-health and economic consequences

When concentrations of a contaminant in stream water are greater than a Maximum Contaminant Level (MCL), treatment might be needed to reduce the concentration (Box 1).

The time-weighted average annual concentrations of atrazine were not greater than the drinking-water standard, or MCL, of 3 µg/L at any of the 10

Table 1. Annual and seasonal time-weighted average concentrations of atrazine at 10 stream sites in the Lake Erie-Lake Saint Clair Drainages, 1996–98

Site and primary land use	Concentrations of atrazine	
	Annual average (µg/L)	May - July average (µg/L)
ROW-CROP LAND USE		
River Raisin, Mich.	0.04	0.09
Auglaize River, Ohio ¹	2.4	8.9
Black River, Mich.	0.91	3.0
St. Joseph River, Ind. ¹	1.6	5.5
PASTURE/FOREST LAND USE		
Cattaraugus Creek, N.Y.	0.04	0.11
Grand River, Ohio	0.53	1.3
URBAN LAND USE		
Clinton River, Mich.	0.04	0.08
MIXED LAND USE		
Cuyahoga River, Ohio	0.10	0.24
Maumee River, Ind. ¹	2.8	10.1
Maumee River, Ohio ¹	2.6	9.3

Bold indicates concentration is greater than the USEPA's aquatic-life guideline for atrazine of 1.8 micrograms per liter.

¹ Stream is used as a source of drinking water

Box 1—What concentrations of pesticides are considered safe in drinking water?

The USEPA sets drinking-water standards, or Maximum Contaminant Levels (MCLs), and health advisory–lifetime guidelines (HALs) for drinking water, to protect human health. The MCL is the maximum permissible concentration of a contaminant in water delivered to any user of public water systems. MCLs are enforceable standards and are based on an average annual concentration taken from quarterly samples of finished drinking water. HA (Health Advisories) are nonenforceable, risk-based guidelines. Health advisories indicate contaminant exposures below which no short- or long-term adverse human-health effects are expected, based on drinking a specific amount of water for a specific period of time. Risk of illness increases with exposure time and concentration. This report lists only HALs, which are based on a lifetime of 70 years. Standards and guidelines to evaluate the potential adverse effects of pesticides have limitations: (1) Few standards and guidelines have been set for pesticides; of the 88 pesticides analyzed in this study, only 14 MCLs and 38 HALs have been established. (2) Drinking-water standards are based on toxicity tests on a single pesticide and do not evaluate the additive or synergistic effects of multiple pesticides. (3) The standards and guidelines do not address possible effects of pesticides on endocrine systems of humans (Nowell and Resek, 1994; Larson and others, 1999).

stream sites sampled during 1996–98 (table 1). The values from May–July, however, were greater than the MCL at five stream sites (table 1), indicating a need for source-water treatment during that period to remove the pesticides.

Many large water-treatment facilities that withdraw surface water (other than Lake Erie) use activated-carbon filtration during the spring to meet drinking-water standards. In areas of heavy pesticide use, specialized treatment may be necessary year round. For example, the Fort Wayne Water-Filtration Plant uses activated carbon all year. The annual cost of the treatment chemicals at Fort Wayne is approximately \$210,000, about 40 percent of this amount being spent from April to July (Doug Pooler, Fort Wayne Water Filtration Plant, oral commun., 1999). Even with carbon treatment, some treated drinking-water samples tested for atrazine were in excess of the MCL

during 1996–98 (Kelleher, 1999).

Another consequence of heavy pesticide use is the potential effect on aquatic life

Of the 10 streams sampled, 8 had pesticide concentrations, in single samples, in excess of one or more aquatic-life guidelines (fig. 14) (Frey, in press). Concentrations in excess of aquatic-life guidelines were detected for the herbicides atrazine, metolachlor, cyanazine, and metribuzin, and for the insecticides chlorpyrifos, carbaryl, and diazinon. Heavily used compounds—metolachlor, atrazine, cyanazine, and metribuzin—were the compounds with concentrations

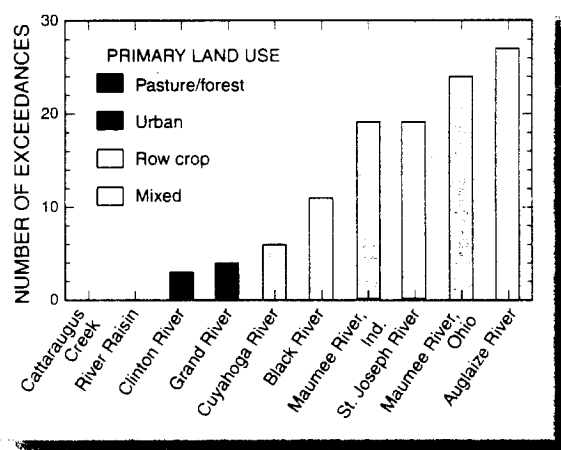


Figure 14. The greatest number of concentrations of pesticides affecting aquatic life were detected in streams draining row crops and mixed-use land.

most frequently greater than guidelines. The time-weighted average concentration of atrazine, for example, was in excess of the 1.8- $\mu\text{g/L}$ guideline at 3 of 10 sites annually and at 5 of 10 sites during May–July (table 1).

Box 2—What concentrations of pesticides are considered safe for aquatic life?

Several agencies, including the USEPA, Environment Canada, and the IJC (International Joint Commission), have set guidelines to protect aquatic life. These guidelines are designed to prevent adverse short-term (acute) and long-term (chronic) effects on aquatic life. The aquatic-life guidelines or benchmarks developed by USEPA are based on 4-day average concentrations, are intended to protect 95 percent of the aquatic species, and should not be exceeded more than once in 3 years. The Canadian and IJC aquatic-life guidelines, which are more stringent than those of the USEPA, indicate a single maximum concentration that should never be exceeded. Aquatic-life guidelines have been developed for only 18 pesticides.



Concentrations of nutrients and herbicides in certain streams are among the highest in the Nation

Because the amounts of fertilizers and herbicides applied to agricultural areas are among the highest in the Nation, concentrations of nutrients and herbicides in stream water also are comparatively high at 8 of 10 streams sampled during 1996–98. In eight streams draining agricultural and mixed-use land, average annual concentrations of total phosphorus were in excess of the 0.1-mg/L (milligram per liter) guideline recommended by USEPA to reduce stream eutrophication (fig. 15). In four streams, phosphorus concentrations were in the upper 25 percent of all streams sampled nationwide by NAWQA, as were nitrate concentrations in six streams. In two streams that serve as source waters for public supply, nitrate concentrations in a few samples were detected in excess of the drinking-water standard. Although total phosphorus concentrations appear to be relatively high, the work of other investigators has shown that concentrations of total phosphorus were 40 percent lower in 1995 than in 1976 (Baker and others, 1998). In two of the streams where nutrient concentrations were in the upper 25 percent range on a national basis, average annual concentrations of herbicides were in the upper 25 percent as well. Mixtures of pesticide compounds were found in every stream sample and a total of 30 different pesticides were detected. In one sample, 18 different pesticides were detected. This sample contained more pesticide compounds than 98.5 percent of all the samples collected at 343 sites across the Nation during 1993–98.

Runoff from agricultural land to streams is an important pathway of phosphorus to Lake Erie

In the 1960s, Lake Erie was considered “dead” from the excessive growth and decay of algae and the oxygen-depleting effects of this

decay on the bottom water of the lake (Herdendorf, 1986). By 1971, it was well known that the major sources of phosphorus to the lake were sewage, laundry detergents, and fertilizers. The Great Lakes Water-Quality Agreement (International Joint Commission, 1994) set goals for United States and Canada

to reduce phosphorus inputs to the lake. Actions taken to reduce phosphorus included a limit on the phosphorus content of detergents, a limit of 1.0 mg/L in discharges from sewage-treatment plants, and a total load limit for Lake Erie of 11,000 metric tons per year (International Joint Commission, 1994). Having achieved the point-source reductions over the period 1968–92 (Litke, 1999), the focus shifted to reducing nonpoint sources of phosphorus.

A sustained focus on reducing nonpoint sources of phosphorus to Lake Erie has continued because oxygen depletion in the bottom water in some areas occurs intermittently (Bertram, 1993; Litke, 1999). One reason may be that in years with above-average runoff, annual phosphorus loads to the lake exceed 11,000 metric tons (Dolan, 1993). The excess phosphorus in wet years is thought to come from nonpoint sources (Dolan, 1993).

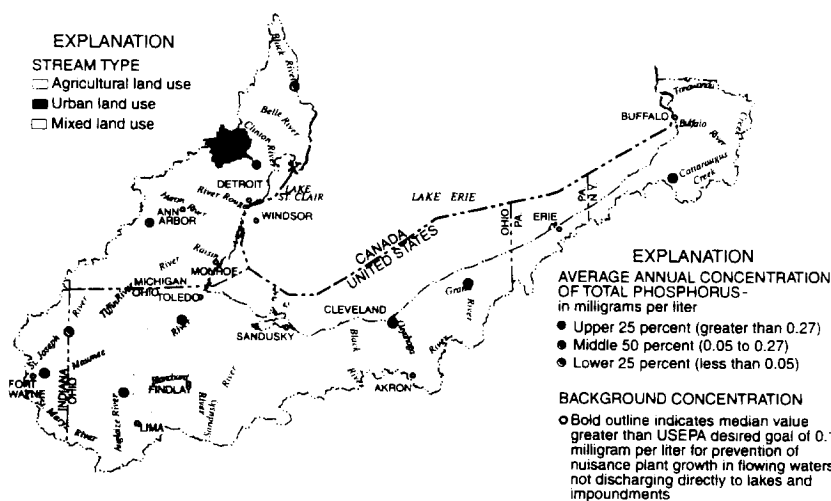


Figure 15. Median concentrations of total phosphorus at 8 of 10 sites in the Lake Erie–Lake Saint Clair Drainages were in excess of 0.1 milligram per liter, the USEPA guideline to control eutrophication in streams.

Fertilizers appear to be the major nonpoint sources of phosphorus to Lake Erie. Greater amounts of phosphorus were applied as fertilizers to each square kilometer of agricultural and mixed-use land than to urban-residential land in 1996–98 (fig. 16). Phosphorus yields, or the amount of phosphorus discharged from a stream per unit area of its basin, ranged from 8.2 to 293 kg/km² (kilograms per square kilometer). The phosphorus yield from only 1 of 10 streams sampled during 1996–98 was in the lower range of yields typical of streams flowing through undeveloped land (Clark and others, 2000). Phosphorus yields in eight streams were similar to or slightly higher than the yields typical of streams flowing through agricultural and mixed-use lands (Gibson and others, 2000).

Phosphorus concentrations were highest in the same streams where yields were highest. Therefore, streams receiving runoff from agricultural and mixed-use lands contributed greater concentrations and amounts of phosphorus to Lake Erie than did streams receiving runoff from urban-residential land

(fig. 16). A conclusion is that agricultural runoff is an important nonpoint source of phosphorus to small streams and major rivers and that major rivers, like the Maumee River, provide a direct pathway to Lake Erie. During 1996–98, the Maumee River contributed an average of 24 percent per year to the 11,000-metric-ton limit.

In the last few years, phosphorus concentrations in Lake Erie appear to be increasing above levels recommended in the Great Lakes Water-Quality Agreement (Scott Painter, Environment Canada, written commun., Oct. 2000). Continued emphasis on managing levels of phosphorus in

Lake Erie appears to be warranted (Box 3).

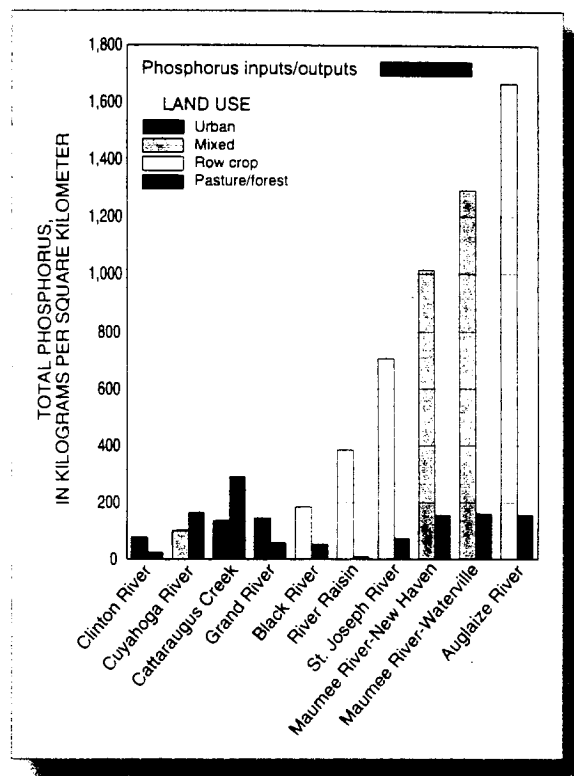


Figure 16. Unit-area phosphorus inputs to the land surface from fertilizer and manure and unit-area phosphorus outputs from 10 streams in the Lake Erie-Lake Saint Clair Drainages, 1996–98.

Box 3—Recent changes in the food web of Lake Erie—the role of phosphorus and zebra mussels

Concerns about recent declines in some important fish species and the proliferation of zebra mussels, an exotic species, have prompted a renewed interest in the role of phosphorus in Lake Erie. Zebra mussels (shown in the photograph) were introduced into Lake Saint Clair in the mid-1980s and quickly became established in Lake Erie. Because of their filter-feeding behavior, zebra mussels may be affecting fish production in Lake Erie by consuming large amounts of small invertebrates and algae, or plankton, that are the food for small fish, which in turn are the food of larger game fish like yellow perch and walleye. Recent decreases in sport- and commercial-fish harvests, proliferation of zebra mussels, and reductions in phosphorus concentrations in Lake Erie have prompted a renewed interest in how factors such as phosphorus and zebra mussels may be affecting sustainable fish harvests. Adding more phosphorus to the lake to stimulate algal productivity has been suggested, but this could result in increased eutrophication of tributary streams and a return to excessive oxygen depletion in the bottom water of the lake. Furthermore, uncertainties about the response of the lake to phosphorus additions in light of its changing food web have held back plans to apply such a solution. One thing about which everyone can agree is the remarkable ability of exotic species to cause unexpected consequences in ecosystems (U.S. Environmental Protection Agency and Environment Canada, 2000).



Photograph from Ohio Lake Erie Commission.

Decreases in the amounts of suspended sediment carried by rivers correspond to increases in farmers' use of conservation tillage

Cropland in the Maumee River Basin is the largest contributor to soil erosion and sediment in the Maumee River (U.S. Department of Agriculture, 1998), and the river is the largest tributary source of suspended sediments to Lake Erie (Myers and others, 2000). Excessive amounts of sediment discharged from the Maumee River to Lake Erie diminishes the aquatic habitats of fish and other organisms. Maintenance of navigation requires the dredging of approximately 800,000 tons of sediment each year from the Maumee River and Lake Erie at an average annual cost of about \$2.2 million. In response to these problems, the Ohio Lake Erie Commission has set a goal of reducing suspended sediment in Lake Erie tributaries by 67 percent (Ohio Lake Erie Commission, 1998), and the U.S. Army Corps of Engineers has set a



Figure 17. Conservation tillage protects the soil. (Photograph by Steven Davis, Natural Resources Conservation Service).

goal of reducing the amount of sediment dredged from the lower river and its harbor by 15 percent.

Conservation tillage (fig. 17) is a reduced-cultivation method that is being used to decrease soil erosion and thereby the amount of suspended sediment carried by the Maumee River and its tributaries. By maintaining a layer of crop residue from the past year's crop on 30 percent or more of the soil surface, conservation tillage protects the soil from the forces of water erosion. During 1993–98, about 53 percent of all crop fields in the Maumee River Basin and about 50 percent of the crop fields in northwestern Ohio were planted using conservation tillage methods (fig. 18).

Significant decreases in the amounts of suspended sediment carried by the Maumee and Auglaize Rivers over time correspond to increases in conservation tillage. Data from previous studies and new data collected at two sites in the Maumee River Basin during 1996–98 support this conclusion (Myers and others, 2000). An 11.2-percent decrease in

the amount of suspended sediment carried by the Maumee River was detected at Waterville, Ohio, near the mouth (fig. 18), and a 49.8-percent decrease was detected in the amount carried by the Auglaize River near Fort Jennings, Ohio, a tributary stream (Myers and others, 2000). During 1970–98, no trends in streamflow at these two sites were detected. If conditions leading to the downward trends remain the same, then the estimated time necessary to achieve the 15 and 67 percent reduction goals are 30.1 and 205 years, respectively, from the reference condition in 1992.

One reason for the smaller downward trend in the Maumee River compared to that found in its tributary may be related to drainage-basin size. In the large drainage basin of the Maumee River, sediment is deposited in small drainage ditches and streams. These stored sediments are available for resuspension and transport in subsequent storms. This process, as described by Trimble (1975, 1999), creates a lag in the response of rivers to conservation tillage on the land.

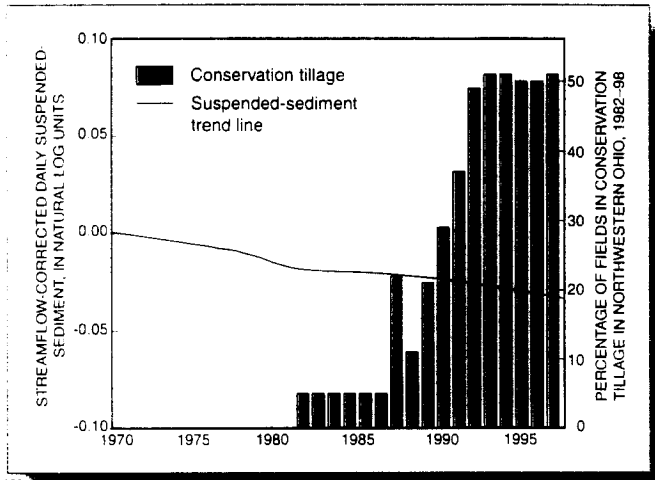


Figure 18. Increases in conservation tillage in the Maumee River Basin and elsewhere in northwestern Ohio correspond to decreases in the amount of suspended-sediment carried by the Maumee River at Waterville, Ohio.

Contamination of bed sediments may be causing adverse effects on aquatic life

Bed sediments of small streams and major rivers provide habitat for many aquatic organisms but also serve as a repository for persistent and toxic chemical contaminants that have been released into the environment or that occur naturally. Sediments, once contaminated, can provide a pathway for bioaccumulation. Activities such as dredging of sediments can release contaminants to the surrounding water.

Evidence from laboratory tests shows that contaminated bed sediments can be toxic to aquatic invertebrates (worms, clams, and insect larvae) that are of recreational, commercial, or ecological importance. Contaminated sediments also can affect the food supply required to sustain fish populations (U.S. Environmental Protection Agency, 1997).

Data collected by the U.S. Geological Survey (USGS) and other public-sector agencies at more than 800 locations from 1990 to 1997 were aggregated and analyzed to assess bed-sediment contamination. These data show that **zinc, lead, copper, arsenic, cadmium, mercury, PAHs (polycyclic aromatic hydrocarbons), PCBs (total polychlorinated biphenyls), DDT, and its breakdown products are prevalent in bed sediments, being detected in 30 to 100 percent of samples** (Rheume and others, in press) (fig. 19).

Concentrations of mercury and PCBs (figs. 20 and 21) exceeded Probable Effect Levels (PELs) in 11.8 and 22.4 percent of samples, respectively. PELs are the guide-

Box 4—How are contaminated sediments evaluated?

Effect levels are screening values used to indicate when contaminant concentrations in the bed sediments are likely to be associated with adverse effects on aquatic life. The Threshold Effect Level (TEL) is an estimate of the concentration of a contaminant in bed sediment below which adverse biological effects rarely occur and, if so, only in very sensitive species. The Probable Effect Level (PEL) is an estimate of the concentration of a contaminant in bed sediment above which adverse biological effects frequently occur. These guidelines were developed for the Great Lakes sediments and are used for national assessment in this report. For anthracene and total PAHs, no guideline was available from Smith and others (1996), so guidelines developed by Ingersoll and others (1996) for the Great Lakes were used.

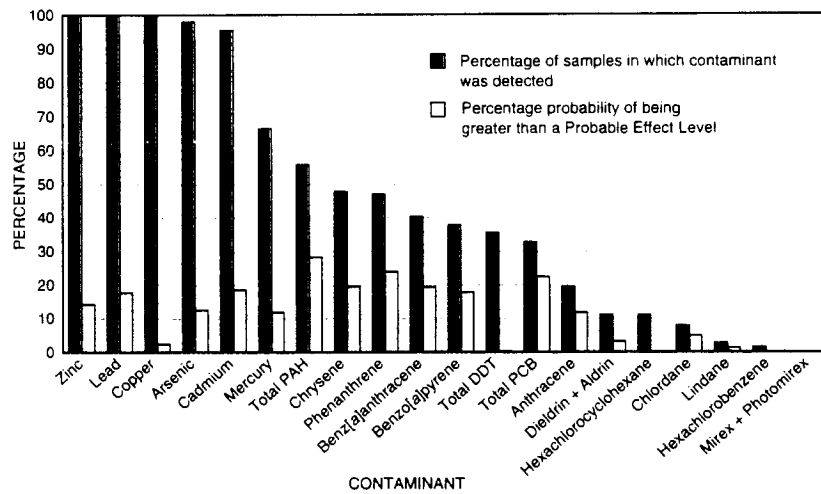


Figure 19. Percentage detection and percentage probability of sample concentrations being greater than a Probable Effect Level in bed sediments of small streams, major rivers, and lakes, Lake Erie-Lake Saint Clair Drainages, 1990–97.

lines established for the protection of aquatic life (Box 4). Contamination of bed sediments with mercury and PCBs was greatest in nearshore areas of Lake Erie, in Lake Saint Clair, and in major rivers that flow through urban areas with industry and populations over 100,000.

Concentrations of arsenic, DDT, and chlordane also exceeded PELs

in the bed sediments of some smaller streams. Samples with detection limits higher than Threshold Effect Levels (TELs) (open circles, figs. 20 and 21) were not useful for assessment purposes. Lower detection limits would have made these data more useful.

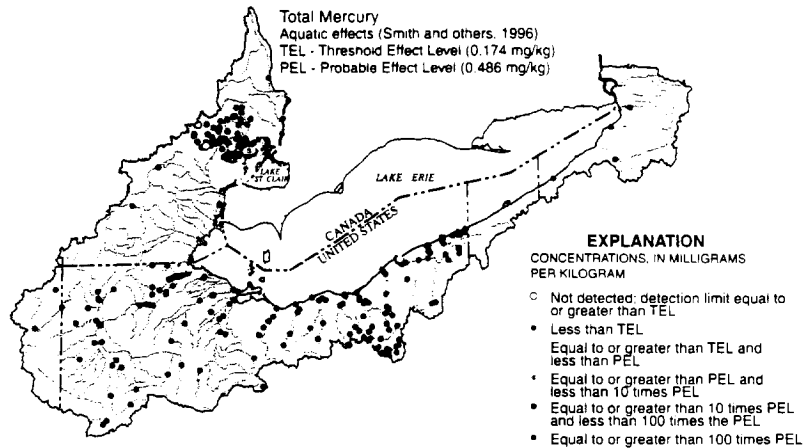


Figure 20. Concentrations of total mercury in recently deposited lakebed and streambed sediments (Data from 1990–97).

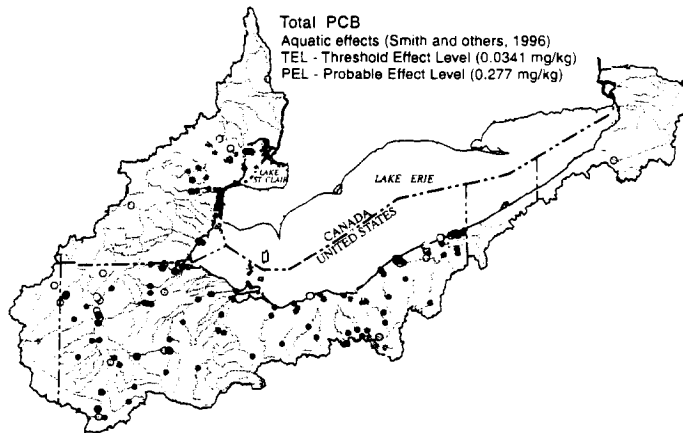


Figure 21. Concentrations of total PCB in recently deposited lakebed and streambed sediments (Data from 1990–97).

Land use and chemical use contribute to contamination of sediments and fish

Most organochlorine pesticides and PCBs have not been manufactured or used in the United States for at least 10–25 years; however, because of their chemical stability (persistence), they were detected in bed sediments and fish collected in 1996–98. Aldrin, DDT, and chlordane were the organochlorine insecticides used in the greatest quantities in the Corn Belt and Great Lakes States in the 1970s. Similarly, historical use of PCBs in industry contributed to their occur-

rence in urban and mixed-use areas.

The highest concentrations of DDT, chlordane, dieldrin, and their breakdown products were detected in bed sediments and fish collected from small streams and major rivers flowing through row-crop, urban, and mixed-use lands. The highest concentrations of PCBs, DDT, and chlordane were detected in bed sediments and fish from streams flowing through urban and mixed-use lands where historical use was greatest. Organochlorine contamination was the lowest in fish and sediments collected from streams in pasture/forest settings.

Unlike organochlorine compounds, mercury has more than 2,000 uses in industry, medicine, agriculture, and commerce. Mercury also is released to the atmosphere as a by-product of coal combustion and waste incineration (Irwin and others, 1997). In 1996–97, mercury was detected in all 15 bed-sediment samples and in all 11 fish-liver samples collected from streams, independent of land-use type. The highest concentrations of mercury, however, were detected in bed sediments and fish collected from small streams and major rivers that flow through urban and mixed-use lands. The magnitude of mercury contamination in bed sediments and fish appears to follow patterns of use in commerce and industry. Mercury was detected in several different fish species that have different feeding habits and that occupy different positions in the food web. Mercury was detected in carp, which are bottom-feeders, in northern hog suckers, which eat insects, and in rock bass, which eat smaller fish (fig. 22).



Figure 22. Total chlordane, total DDT, dieldrin, heptachlor epoxide, mercury, and total PCBs were detected in the tissues of carp, rock bass, and northern hog suckers collected in 1996–97.

Bioaccumulative contaminants in fish may pose a risk to fish-eating wildlife

Where PCBs were detected in fish, they frequently posed a threat to fish-eating wildlife because of their high concentrations. PCBs were found in fish at 8 of 11 sites, and concentrations in fish from 6 sites exceeded the New York State guideline for the protection of fish-eating wildlife (table 2). The maximum concentration of PCBs found in fish was 25 times higher than the guideline set by the New York State Department of Environmental Conservation (NYSDEC) for the protection of fish-eating wildlife (Newell and others, 1987). These fish were collected from the Cuyahoga River at Cleveland, Ohio, a major river that flows through mixed-use land.

DDT was present in fish at 10 of 11 sites but concentrations in fish from only 2 sites exceeded the NYSDEC guideline. The highest concentrations of DDT were in whole fish in streams flowing through urban and mixed-use land—the Clinton River at Sterling Heights, Mich., and the Cuyahoga River at Cleveland, Ohio. Although mercury is a concern, no wildlife consumption guidelines have been established.

Fish-consumption restrictions for humans are based on contaminant concentrations in edible fish fillets, which are typically lower than those in whole fish. The contaminants of most concern in the Lake Erie Basin for human consumption of fish are mercury, PCBs, DDT, chlordane, and dieldrin. These are the same contaminants that are a concern for wildlife consumption (fig. 23). Because of

Table 2. Summary of contaminant concentrations in fish tissue in relation to guidelines for the protection of fish-eating wildlife [$\mu\text{g}/\text{kg}$, micrograms per kilogram; NYSDEC, New York State Department of Environmental Conservation]

Compound found in fish tissue	Number of sites with detections (11 sites sampled)	Maximum detected concentration ($\mu\text{g}/\text{kg}$)	NYSDEC whole-fish guideline for the protection of fish-eating wildlife ($\mu\text{g}/\text{kg}$)	Number of sites with contamination in excess of NYSDEC criteria
Chlordane, total	6	157	500	0
DDT, total	10	450	200	2
Dieldrin	5	32.0	120	0
Heptachlor epoxide	2	10.0	100	0
PCBs, total	8	3,200	130	6

widespread human consumption advisories for fish in the Lake Erie Basin, mercury and PCBs have been identified as critical pollutants and targeted by the USEPA and

Environment Canada for cleanup and elimination (Daher, 1999).

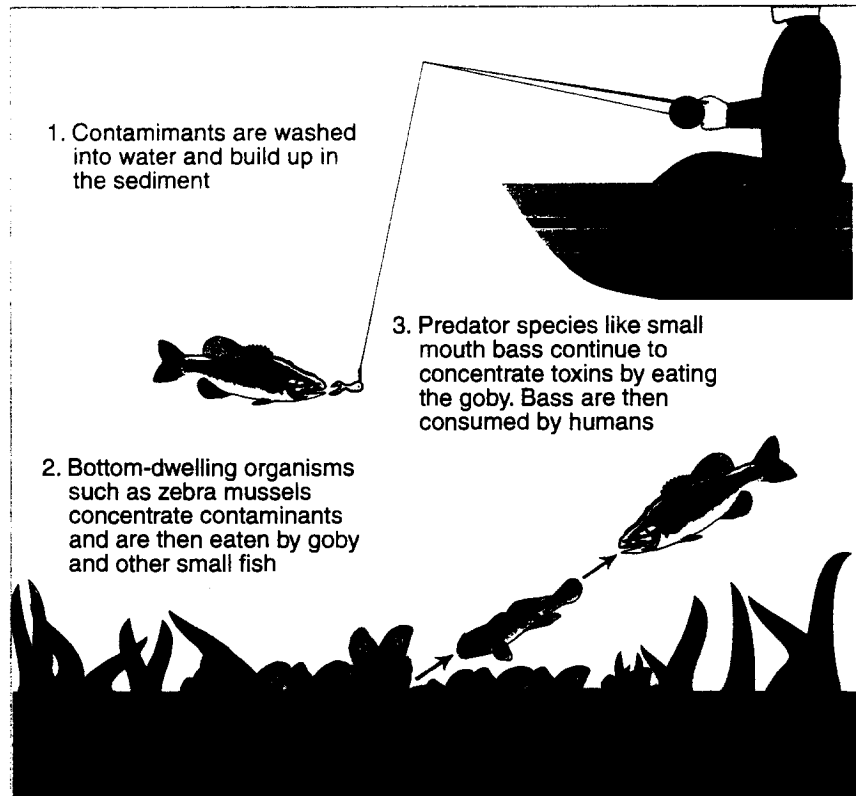


Figure 23. Contaminants can bioaccumulate upward through the food web and may affect the health of human and animal consumers of fish.

Land use appears to affect fish-community composition

Contaminants and nutrients in water and bed sediments appear to affect aquatic life, but human disturbance of physical stream habitat also is thought to be one of the most important causes of declines in certain fish species (Ohio Environmental Protection Agency, 1995). To differentiate the effects on fish of human disturbance from those of contaminants, sites considered to represent good to excellent stream habitat, as rated by the Qualitative Habitat Evaluation Index (QHEI) (Box 5) were selected for assessment of fish. Fish communities were assessed by means of the Index of Biotic Integrity (Box 5).

In 1996–98, stream habitats were rated good to excellent at 11 of 13 sites (fig. 24A), but fish communities were rated good to excellent at only 5 sites (fig. 24B). This may be an indication that the contaminants detected in water and bed sediments

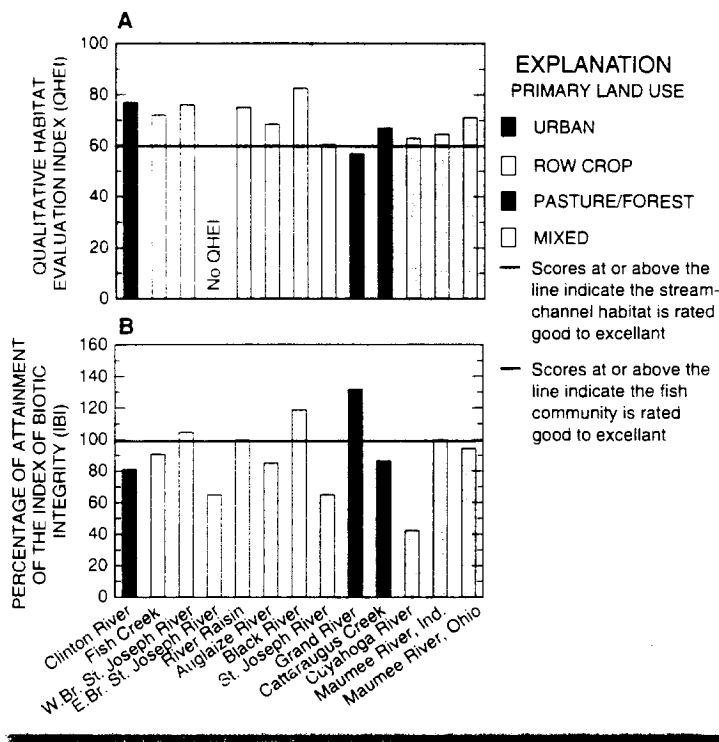


Figure 24. Although most stream sites were rated good to excellent for habitat as measured by the Qualitative Habitat Evaluation Index (A), only five sites were rated good to excellent for fish community composition as measured by the Index of Biotic Integrity (B).

Box 5—Index of Biotic Integrity and Qualitative Habitat Evaluation Index

Aquatic biological communities are sensitive indicators of stream quality. The biological condition of streams is evaluated within water-quality assessment programs by comparing the type, number, and abundance of fish species to those of streams known to be “least impacted” by human activities. The Index of Biotic Integrity (IBI), used by the Ohio Environmental Protection Agency (Ohio Environmental Protection Agency, 1989), also was used in this study to assess fish-community composition. IBI scores are derived by summing 12 individual metric scores from separate factors that describe fish communities. Examples of such metrics are the number of taxa, the number of insect-eating or omnivorous species, the number of pollution-tolerant and pollution-intolerant species, and the relative abundance of fish in each of these categories. Scores for the IBI range from 12 to 60; with ratings of poor, fair, good, and excellent quality assigned to numerical ranges. Higher scores indicate better overall fish-community diversity and abundance. Because the minimum IBI score required for a “good” rating differs somewhat from one ecological region to another, and there are five such regions in this study, each IBI score was reported as a percentage attainment of a “good” rating. A comparison of IBI scores computed independently for the same sites in the same years by the Ohio Environmental Protection Agency and USGS show that results are comparable between the two agencies as long as fish-collection methods are appropriate for the stream size (Covert, in press). A Qualitative Habitat Evaluation Index (QHEI), also developed by the Ohio Environmental Protection Agency (1989), was applied to data collected by USGS at the same stream sites where fish were assessed. The QHEI is composed of seven metrics that describe the physical habitat of streams: width, depth, pools, riffles, grain size of streambed substrates, and vegetation type and width along stream banks. Scores for the QHEI range from 0 to 100, with scores of 60 or better indicating excellent stream-channel habitat.

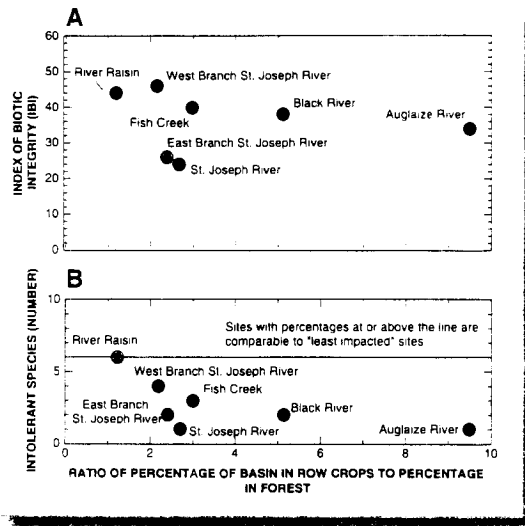


Figure 25. As the percentage of row-crop land use increased in a stream basin (A) scores for the Index of Biotic Integrity decreased and (B) the number of pollution-intolerant fish decreased.

are affecting fish-community composition.

For example, **agricultural runoff appears to be affecting fish communities in streams draining row crops.** Agricultural practices can heavily stress fish

communities because of runoff of pesticides, nutrients, and sediments that otherwise drain into streams with runoff from the land surface (Sweeney, 1992; Lowrance and others, 1997).

Clear differences can be seen in fish-community composition

among streams draining row-crops that may be related to the relative amount of row crops compared to woodlands and forests. As the amount of row-crops increased, the number of pollution-tolerant fish species in streams also increased (fig. 25B). In contrast, **more pollution-intolerant fish species were found in streams draining areas with lower percentages of row crops and higher percentages of forested land in their basins, especially when the forests were along streams.**

Although only a few sites were examined, **it appears that pollution-intolerant fish can live in streams where agriculture is the primary land use under certain conditions.** This information corroborates the findings of others on the water-quality benefits provided by undisturbed natural stream habitats and wooded riparian areas (figs. 26 and 27).

Clear differences can be seen in fish-community composition



Figure 26. High-quality stream habitat supports a variety of pollution-intolerant fish species along the West Branch of the St. Joseph River.



Figure 27. Poor-quality stream habitat supports pollution-tolerant fish along the East Branch of the St. Joseph River.

Recent residential development has had a widespread effect on ground-water quality

Two studies were done to assess ground-water quality in areas west of Detroit, where recent suburban development overlies sand-and-gravel deposits (Thomas, 2000a). Recent suburban development typically consists of homes on large residential/rural lots or in upper-middle-class subdivisions (fig. 28). Most homes have private wells, septic systems, and water softeners.

Samples of shallow and deep ground water were collected from co-located wells (fig. 29). Shallow ground water was sampled from 30 monitor wells (median depth of 25 feet). Deeper ground water was sampled from 28 domestic wells (median depth of 93 feet). In most places, clay till of varied thickness separates the shallow and deep sand-and-gravel deposits (fig. 29).

Tritium concentrations were used to distinguish ground waters recharged before 1953 (“old waters”) from those recharged after



Figure 28. Much of the recent development west of Detroit in the residential study area consists of low- to medium-density subdivisions. Most homes have private wells, septic systems, and water softeners.

1953 (“young waters”) (Plummer and others, 1993). Water in about one-third of domestic-wells was recharged before 1953, which is prior to widespread residential development in the study area. Old waters show no evidence of human activities. All nitrate concentrations were less than the estimated background value of 2 mg/L (U.S. Geological Survey, 1999), and all

chloride concentrations were less than 25 mg/L, the estimated background concentration based on a historical data set from the same area (Mozola, 1953).

In contrast, more than 75 percent of young waters show evidence of human influence:

- Seventy-six percent of young waters had chloride concentrations greater than background (fig. 30). Probable sources of elevated chloride concentrations are road salt, septic systems, and backwash from water softeners.

- Twenty-six percent of young waters contained nitrate concentrations greater than background (fig. 30). Probable sources of elevated nitrate concentrations in residential areas are septic systems, lawn fertilizer, and pet waste.

- Trace concentrations of VOCs were detected in 29 of the 30 samples from monitor wells. The most frequently detected VOCs were BTEX compounds, which are components of fuels and solvents.

Samples that contained nitrate or VOCs also contained elevated

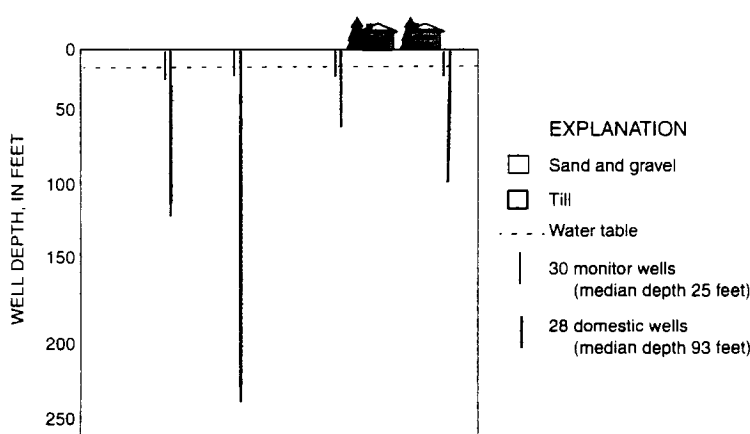


Figure 29. Ground-water samples were collected from pairs of shallow monitor wells and deeper domestic wells. Sand-and-gravel deposits are at land surface and at varied depths in the subsurface.

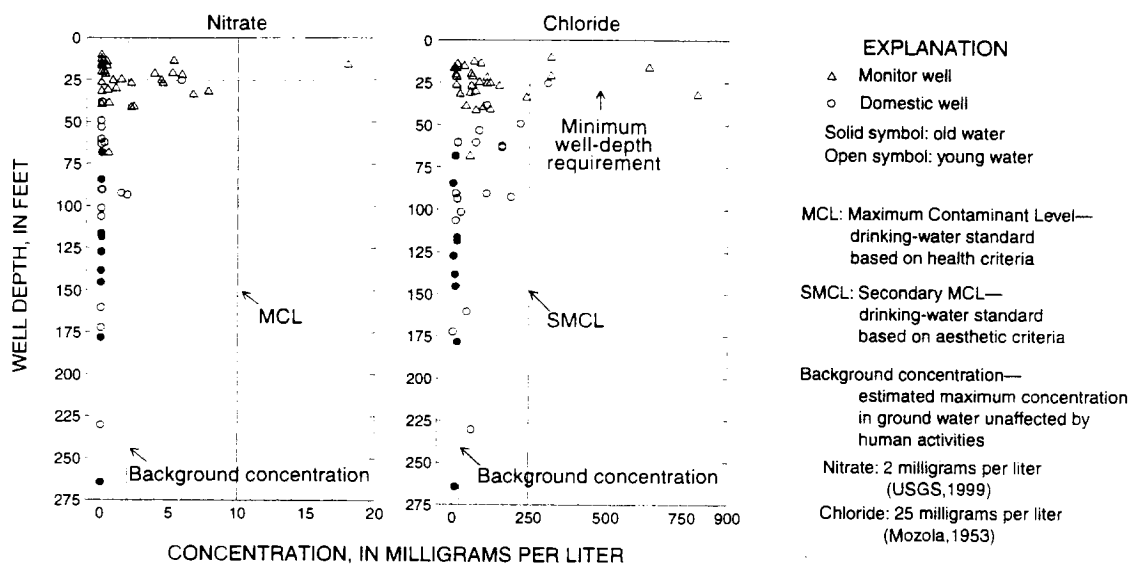


Figure 30. In the residential study area, concentrations of nitrate and chloride greater than background concentrations are the result of human activities. Elevated concentrations of chloride were detected far below 25 feet, the minimum well depth required by State regulations.

chloride concentrations. The co-occurrence of these compounds suggests that ground water is affected by a mixture of sources, such as (1) **septic systems** containing domestic waste (nitrate and chloride), water-softener backwash (chloride), and/or household solvents such as drain cleaners (VOCs), (2) infiltration of runoff from **roads** containing road salt (chloride) and fuel residue (VOCs), and/or (3) fertilizer or pet waste (nitrate) from **lawns**.

Pesticides were detected infrequently. Trace concentrations of herbicides were detected in 2 of 30 (7 percent) monitor-well samples. This value is low compared to similar studies in 25 other urban/residential areas, where the median detection frequency of herbicides in shallow ground water was 44 percent (Kolpin, 2000). In the study area, most of the pesticides commonly used on lawns and roadways were analyzed for but were not detected in the shallow ground water.

One monitor-well sample had a nitrate concentration exceeding health-related standards (fig. 30). The sample was from a well too shallow to be used as a source of drinking water, but shallow ground water can migrate to deeper parts of ground-water system or discharge to streams, lakes, and wetlands.

No health-related drinking-water standards were exceeded in samples from domestic wells; however, standards do not exist for all compounds or for mixtures of compounds. Also, all possible constituents were not tested; for example, septic systems can contain bacteria, viruses, or pharmaceuticals.

In Michigan, the minimum depth required for domestic wells is 25 feet. In the study area, the effects of human activities were detected far below this depth (fig. 30). A proposal to increase the required depth for private wells in subdivisions is under consideration by the Michigan Department of Environmental Quality.

Box 6—Chloride/bromide ratios verify that elevated chloride concentrations are due to human activities.

In parts of eastern Michigan, elevated chloride concentrations in shallow ground water have been attributed to upward movement of brine from deep bedrock formations (Mozola, 1953; Long and others, 1988). In the study area, however, Cl/Br ratios indicate that elevated chloride concentrations were not derived from brine, but from halite (rock salt) (Davis and others, 1998). Natural sources of halite are not present in near-surface bedrock or sediments. However, halite is commonly used as road salt, table salt, and as a water-softener additive. The conclusion that elevated chloride concentrations are due to recent human activities is consistent with the observations that (1) chloride concentrations are highest in young, shallow waters, and (2) all old waters have concentrations that are less than background values.



Figure 31. In the agricultural study area, the predominant crops are corn, soybeans, and small grains. (Photograph from Ohio Department of Natural Resources.)

Ground water is vulnerable to contamination in agricultural areas underlain by till

Ground-water quality was assessed in agricultural areas in the northwestern part of the Lake Erie-Lake Saint Clair Drainages (Thomas, 2000b) (fig. 31). In this

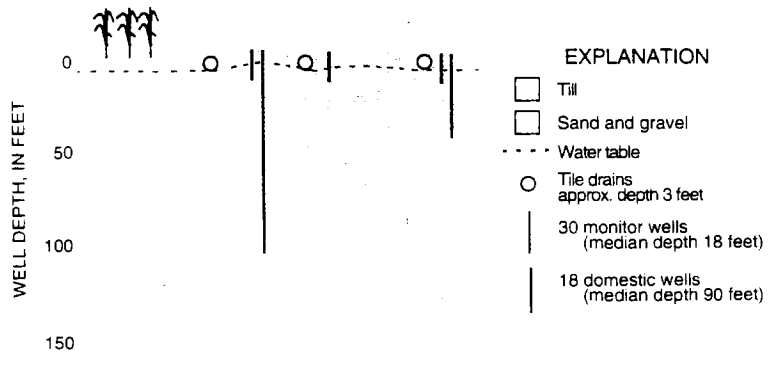


Figure 32. In the agricultural study area, clay-rich till occurs at land surface. Shallow tile drains are installed in low spots of farm fields to improve drainage.

region, clay-rich till is at the land surface, and the rate of water infiltration is generally less than in the residential study area, where sand and gravel are at the surface (compare figs. 29 and 32).

Samples of shallow ground water were collected from 30 monitor wells installed in the till at a

median depth of 18 feet. Deeper ground water, from the glacial sand-and-gravel aquifer, was collected from 18 domestic wells with a median depth of 90 feet (fig. 32).

Samples were analyzed for pesticides in the form of parent compounds (the original active ingredient) and metabolites (break-

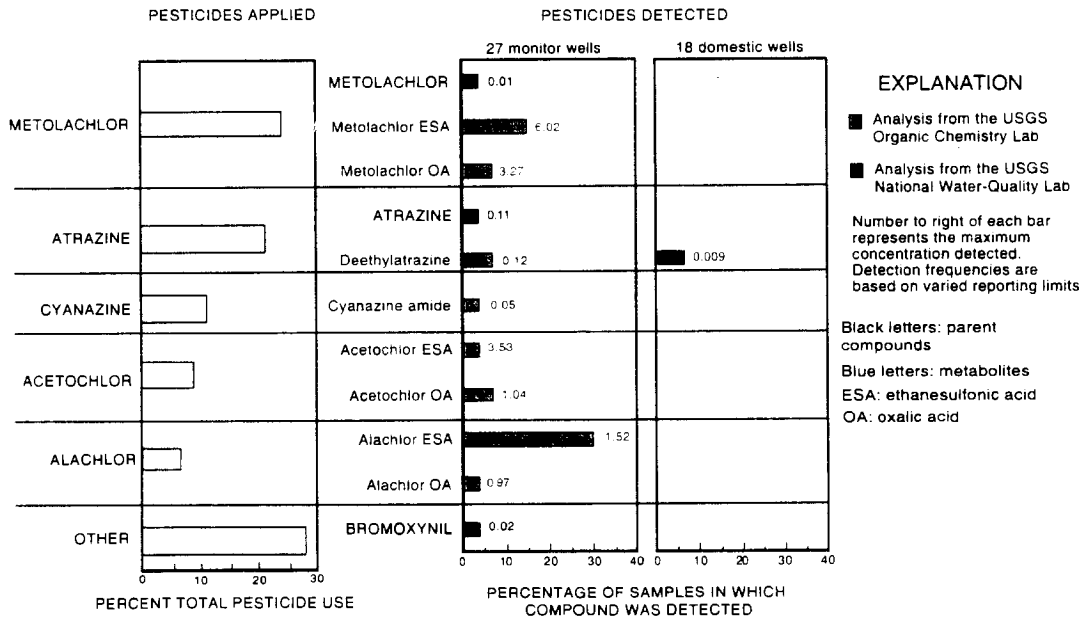


Figure 33. Pesticides detected in the shallow ground water closely correspond to those most heavily applied in the Lake Erie-Lake Saint Clair Drainages (data from Brody and others, 1997). All are herbicides used on corn and soybeans. Metabolites were detected more frequently, and at higher concentrations, than parent compounds.

down products). **Pesticides were detected in 41 percent of monitor-well samples.** The six compounds detected closely correspond to those most heavily applied in the basin—herbicides used on corn and soybeans (fig. 33). Metabolites were detected three times more frequently, and at higher concentrations, than were parent compounds. Concentrations did not exceed drinking-water standards, but standards do not exist for 9 of the 11 compounds detected. In addition, many samples contained more than one herbicide, and the health effect of mixtures is unknown.

Nitrate concentrations indicative of human influence were detected in 37 percent of monitor-well samples. Probable sources

are nitrogen fertilizers (which are heavily used in the study area), manure, or septic systems. Nitrate concentrations exceeded health-related drinking-water standards in two shallow monitor wells (fig. 34).

Pesticides and nitrate were detected much less frequently in domestic wells than in monitor wells (fig. 34). About 6 percent (1 of 18) of samples from domestic wells contained a trace concentration of a pesticide metabolite, and 6 percent contained a nitrate concentration approximately equal to the background concentration of 2 mg/L.

A common belief is that till protects the ground water from contamination. There appears to

be some truth to this idea because detections of nitrate and herbicides are low relative to other agricultural areas with similarly high application rates. At the same time, this belief is a simplification that may be misleading because ground water is still vulnerable to contamination, especially at shallow depths.

Fifty-nine percent of monitor-well samples showed evidence of human activity—either an herbicide or an elevated nitrate concentration. Moreover, 83 percent of waters from monitor wells were recharged after 1953. These observations indicate that **till does not prevent water at the land surface from moving to the shallow ground water.** In the study area,

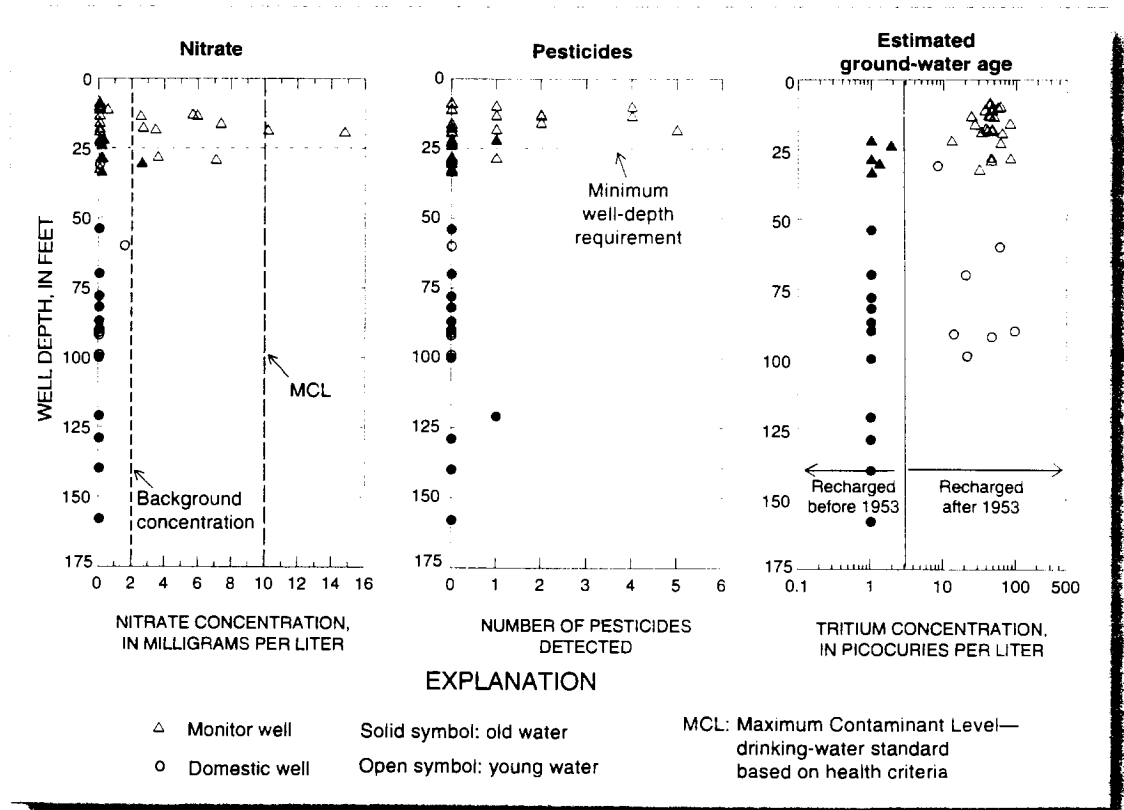


Figure 34. Detections of nitrate or pesticides are relatively infrequent at depths greater than 35 feet; however, deeper ground water is not completely isolated from the land surface because almost half of the waters between depths of 35 and 100 feet were recharged after 1953.

till contains vertical fractures and numerous stringers of sand and gravel that are potential pathways for movement of water from the surface. In areas with similar geology to that in the agricultural study area, active ground-water flow occurs to depths of 35 feet, and sometimes much deeper (Keller and others, 1991).

Similarly, till does not completely prevent water at the land surface from entering the deeper ground water. Eleven percent of domestic-well samples contained either a pesticide or elevated nitrate concentration. Moreover, almost half of the waters from domestic wells between depths of 35 to 100 feet were recharged after 1953 (fig. 34).

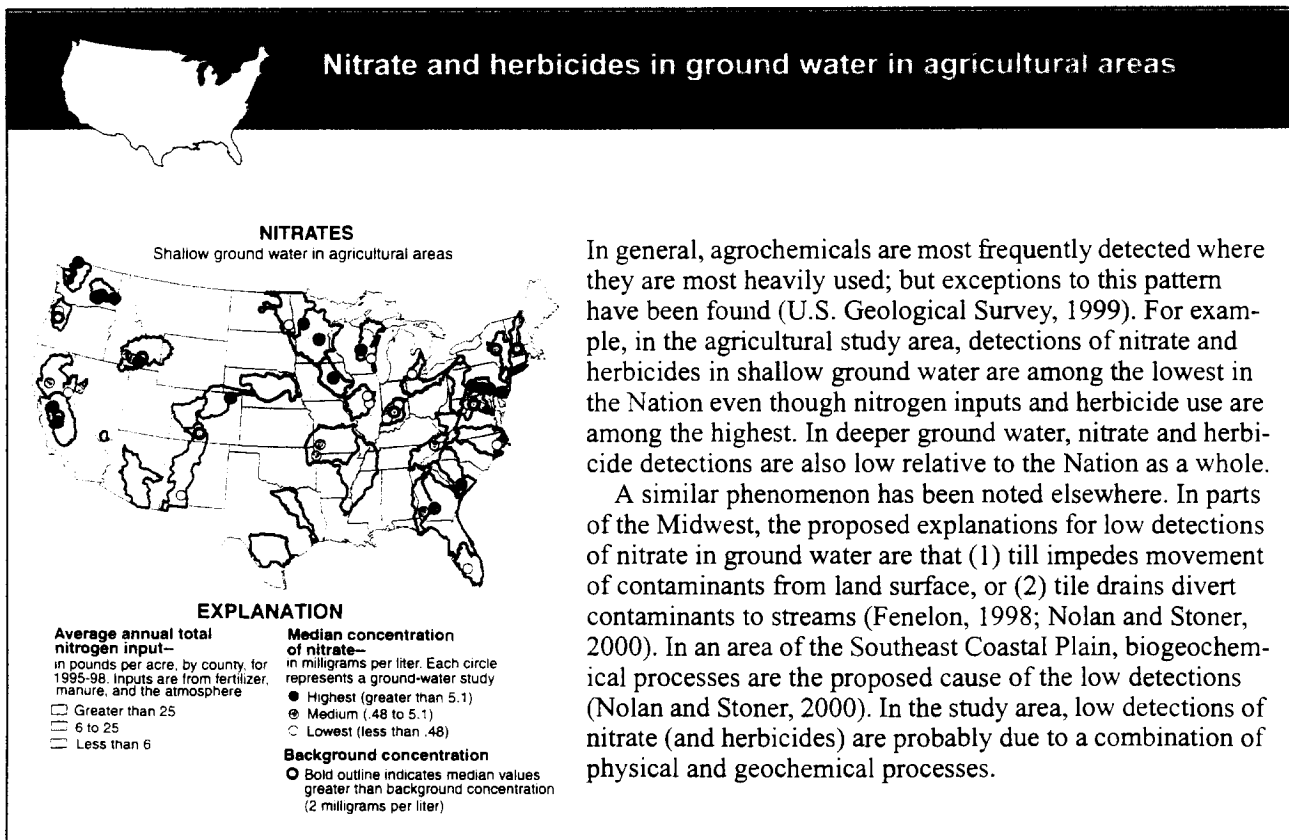
Naturally occurring **geochemical processes** may be partly responsible for the relatively low

detection frequencies of agrochemicals in the ground water. Till and till soils are generally rich in clay and organic matter and therefore are conducive to the sorption or breakdown of pesticides. Most of the pesticides detected were metabolites that had already been partially broken down (fig. 33).

Nitrate also can be transformed in the subsurface. In the presence of organic matter, low dissolved-oxygen concentrations are conducive to denitrification. Ground-water samples from 100 percent of the domestic wells and 50 percent of the monitor wells had dissolved-oxygen concentrations less than 1 mg/L. In addition, the till contains numerous fragments of black shale rich in pyrite, a mineral that can also lead to denitrification (Robertson and others, 1996).

Within the study area, the thickness and composition of the till are variable and not well defined in all locations. Therefore, the degree to which the till protects the ground water—by physical or geochemical means—is probably also variable and difficult to predict.

Low detections of agrochemicals in ground water may be partly due to tile drains, which can divert contaminants from the shallow ground water to streams (Nolan and Stoner, 2000), but tile drains do not completely prevent contaminants from reaching shallow ground water. Nitrate and pesticides were detected in monitor wells 10 to 34 feet deep, whereas tile drains are typically about 3 feet deep. Vertical fractures in the till can provide a direct pathway from the land surface to the water table.



STUDY UNIT DESIGN

Stream Chemistry

A network of 10 stream sites was sampled for various chemical constituents and physical properties from March 1996 through February 1998 (fig. 35, table 3). Sampling was done over a range of stream-flows. Sites were sampled 18–32 times per year; intensive sites were sampled more frequently than basic sites (table 3).

Assessments of contaminants in streambed sediments and fish tissue were made at the 10 fixed streams sites and at one to five

additional sites. Sediment samples were collected and sieved in the field. Whole fish were collected and analyzed for concentrations of organochlorine compounds. Fish livers were analyzed for trace-element concentrations.

Stream Ecology

Sampling was done at least once at 10 sites where water samples were collected. Multiple-year and multiple-reach sampling was done at three sites (table 3). Stream

reaches ranged from 200 to 500 meters in length.

Fish surveys involved electro-fishing and seining. Macroinvertebrates and algae were sampled from natural substrates. Habitats were assessed by identifying and mapping riparian vegetation and measuring geomorphic features of stream channels. Samples of bed and bank materials were collected and analyzed for grain size.

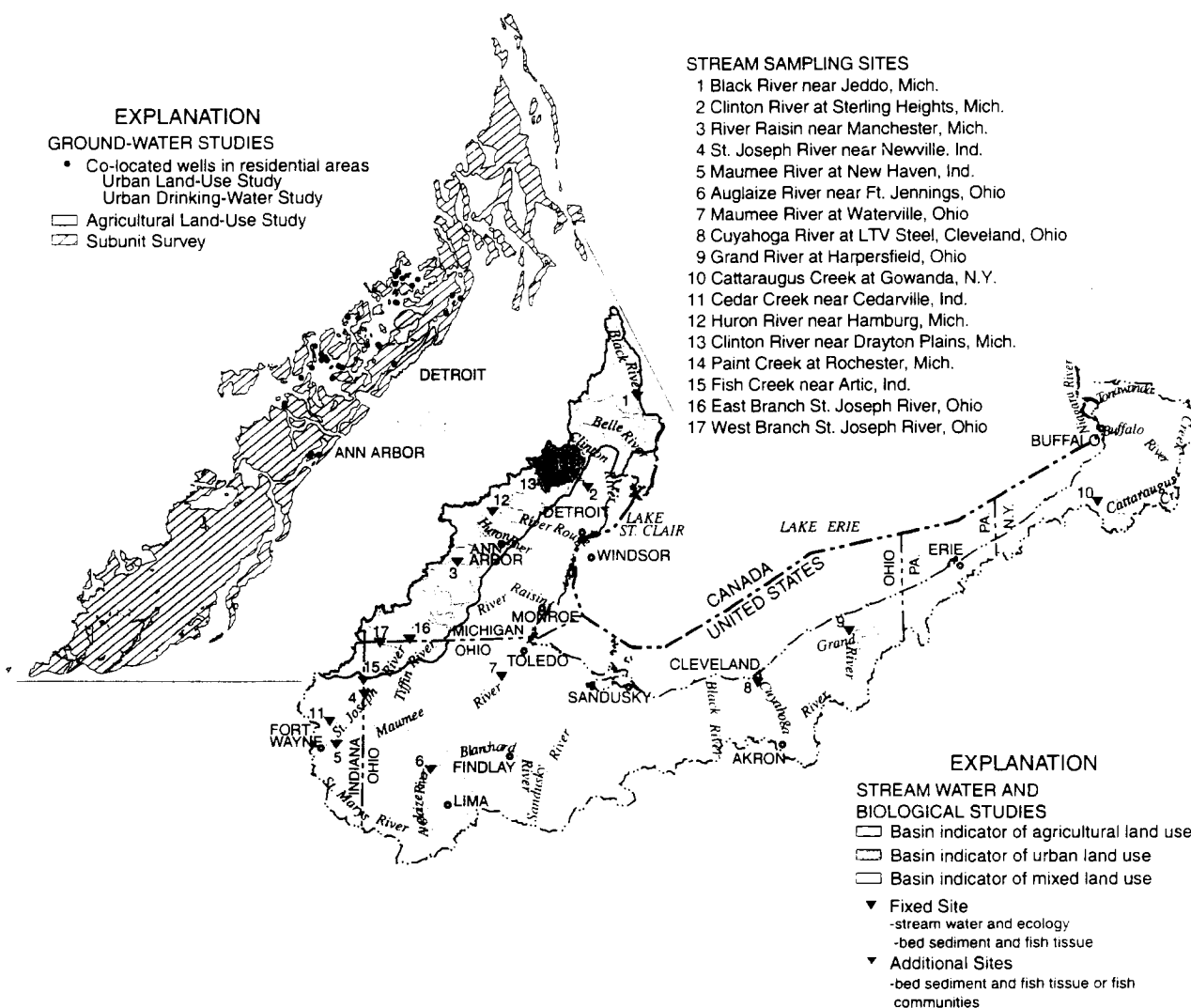


Figure 35. Stream and ground-water sampling sites in the Lake Erie-Lake Saint Clair Drainages, 1996–98.

TABLE 3. SUMMARY OF DATA COLLECTION IN THE LAKE ERIE-LAKE SAINT CLAIR DRAINAGES, 1996-98

Study component	What data were collected and why	Types of sites sampled	Number of sites	Sampling frequency and period
Stream Chemistry				
Contaminants in stream-water—basic sites	Streamflow, pH, specific conductance, temperature and concentrations of nitrogen and phosphorus compounds, major ions, organic carbon, suspended sediment, herbicides, and <i>E. coli</i> were measured to determine occurrence and distribution of contaminants and other constituents	Streams draining areas ranging in size from 132 square miles to 1,230 square miles reflecting agricultural, urban, and mixed land uses.	6	Monthly, plus storms and low flows from March 1996 to February 1998
Contaminants in stream-water—intensive sites	Physical properties and chemical and microbiological constituents mentioned above plus insecticides and VOCs.	Streams draining areas ranging from 310 square miles to 6,330 square miles reflecting agricultural, urban, and mixed land uses.	4	Weekly to monthly, plus storms and low flows from October 1996 to September 1997, bracketed by monthly plus storms from March 1996 to February 1998
Contaminants in streambed sediment	Concentrations as dry weight of trace elements, semivolatile organic compounds, organochlorine compounds, and percent organic content. Determined to assess occurrence and distribution of contaminants.	Shallow depositional zones in a 300-meter reach at all 10 sites sampled for stream-water chemistry and at 5 additional sites	15	One to three times in June–October, 1996–98
Contaminants in fish tissue	Concentrations of trace elements in fish liver and organochlorine compounds, such as PCBs and organochlorine pesticides, in fish tissue were analyzed to determine occurrence and distribution.	Resident fish such as carp, rock bass, and hog suckers at all sites sampled for stream-water chemistry and at one additional site	15	One to three times in June–October, 1996–98
Stream Ecology				
Aquatic biota at stream sites	Community composition of aquatic macroinvertebrates, algae, fish and stream habitat were surveyed to determine effects of water quality on aquatic biota.	Sites co-located with basic and intensive stream-water chemistry sites and at three additional sites	10	One to three times in June–October in 1996–98
Ground-Water Chemistry				
Subunit survey	Water level, pH, specific conductance, temperature and concentrations of nitrogen and phosphorus compounds, major ions, VOCs, organic carbon, insecticides, herbicides, herbicide degradates, radon, and tritium were measured to determine the quality of an aquifer that is an important source of drinking water.	Domestic wells in the northwestern part of the Study Unit. Eighteen of the wells are co-located with monitor wells of the agricultural land-use study.	28	Once, June–August 1998
Agricultural land-use study	Above measurements and compounds (except VOCs and radon) to determine the effects of agricultural land use on shallow ground water.	Monitor wells in agricultural areas in northwestern part of Study Unit.	30	Once, June–August 1998
Urban land-use study	Above compounds (plus VOCs and radon) to determine effects of recent residential development on shallow ground water.	Monitor wells in new residential areas near Detroit, Mich.	30	Once, September–December 1996
Special Study				
Urban drinking-water study	Same as for Subunit survey	Domestic wells co-located with urban land-use wells in new residential areas near Detroit.	28	Once, May–July 1997

Ground-Water Chemistry

Four ground-water studies were done in the northwestern part of the basin (fig. 35), where glacial deposits greater than 100 feet thick serve as the major source of drinking water.

Two of the studies were done on the far outskirts of Detroit, where recent residential developments overlie sand and gravel. An urban

land-use study was done by installing and sampling 30 shallow monitor wells (table 3). A second study involved sampling 28 domestic-supply wells, each of which was co-located with a monitor well.

The agricultural land-use study was done in areas where corn and soybean row crops overlie clay-rich till. Thirty shallow monitor wells were installed in the till and

sampled for a various constituents, including pesticide metabolites.

The subunit survey involved sampling 28 domestic wells that produce water from the glacial aquifer in areas where it is overlain by till. Of the 28 wells, 18 were co-located with monitor wells in the agricultural land-use study area (fig. 35).

GLOSSARY

- Anthropogenic** - A condition or occurrence that is the result of, or is influenced by, human activity.
- Aquifer** - A water-bearing layer of soil, sand, gravel, or rock that will yield usable quantities of water to a well.
- Background concentration** - A concentration of a substance in a particular environment that is indicative of minimal influence by human (anthropogenic) sources.
- Base flow** - Sustained, low flow in a stream; ground-water discharge is the source of base flow in most places.
- Bed sediment** - The material that temporarily is stationary in the bottom of a stream or other watercourse.
- Benthic** - Refers to plants or animals that live on the bottom of lakes, streams, or oceans.
- Bioaccumulation** - The biological sequestering of a substance at a higher concentration than that at which it occurs in the surrounding environment or medium. Also, the process whereby a substance enters organisms through the gills, epithelial tissues, dietary, or other sources.
- Biota** - Living organisms.
- Breakdown product** - A compound derived by chemical, biological, or physical action upon a pesticide. The breakdown is a natural process which may result in a more toxic or a less toxic compound and a more persistent or less persistent compound.
- BTEX** - Benzene, toluene, ethylbenzene, and xylene; volatile organic compounds found in gasoline and solvents.
- Chlordane** - Octachloro-4,7-methanotetrahydroindane. An organochlorine insecticide no longer registered for use in the U.S. Technical chlordane is a mixture in which the primary components are *cis*- and *trans*-chlordane, *cis*- and *trans*-nonachlor, and heptachlor.
- Community** - In ecology, the species that interact in a common area.
- Concentration** - The amount or mass of a substance present in a given volume or mass of sample. Usually expressed as microgram per liter (water sample) or micrograms per kilogram (sediment or tissue sample).
- Denitrification** - A process by which oxidized forms of nitrogen such as nitrate (NO_3^-) are reduced to form nitrites, nitrogen oxides, ammonia, or free nitrogen: commonly brought about by the action of denitrifying bacteria and usually resulting in the escape of nitrogen to the air.
- Detect** - To determine the presence of a compound.
- DDT** - Dichloro-diphenyl-trichloroethane. An organochlorine insecticide no longer registered for use in the United States.
- Dieldrin** - An organochlorine insecticide no longer registered for use in the United States. Also a degradation product of the insecticide aldrin.
- Discharge** - Rate of fluid flow passing a given point at a given moment in time, expressed as volume per unit of time.
- Domestic well** - A private home well, or a well that serves less than 15 households or 25 individuals.
- Endocrine system** - The collection of glands in animals that secrete hormones, which influence growth, gender and sexual maturity.
- Eutrophication** - The process by which water becomes enriched with plant nutrients, most commonly phosphorus and nitrogen.
- Ground water** - In general, any water that exists beneath the land surface, but more commonly applied to water in fully saturated soils and geologic formations.
- Herbicide** - A chemical or other agent applied for the purpose of killing undesirable plants. See also Pesticide.
- Insecticide** - A substance or mixture of substances intended to destroy or repel insects.
- Intolerant organisms** - Organisms that are not adaptable to human alterations to the environment and thus decline in numbers where human alterations occur. See also Tolerant species.
- Invertebrate** - An animal having no backbone or spinal column.
- MCL (Maximum Contaminant Level)** - Maximum permissible level of a contaminant in water that is delivered to any user of a public water system. MCLs are enforceable standards established by the U.S. Environmental Protection Agency.
- Median** - The middle or central value in a distribution of data ranked in order of magnitude. The median is also known as the 50th percentile.
- Metabolite** - See breakdown product.
- Monitor well** - A well designed for measuring water levels and testing ground-water quality.
- Nitrate** - An ion consisting of nitrogen and oxygen (NO_3^-). Nitrate is a plant nutrient and is very mobile in soils. In this report, the term "nitrate" is used as shorthand for "nitrate plus nitrite, reported as nitrogen."
- Nutrient** - Element or compound essential for animal and plant growth. Common nutrients in fertilizer include nitrogen, phosphorus, and potassium.
- Organochlorine compound** - Synthetic organic compound containing chlorine. As generally used, term refers to compounds containing mostly or exclusively carbon, hydrogen, and chlorine. Examples include organochlorine insecticides, polychlorinated biphenyls, and some solvents containing chlorine.
- Pesticide** - A chemical applied to crops, rights of way, lawns, or residences to control weeds, insects, fungi, nematodes, rodents or other "pests."
- Phosphorus** - A nutrient essential for growth that can play a key role in stimulating aquatic growth in lakes and streams.
- Polychlorinated biphenyls (PCBs)** - A mixture of chlorinated derivatives of biphenyl, marketed under the trade name Aroclor with a number designating the chlorine

content (such as Aroclor 1260). PCBs were used in transformers and capacitors for insulating purposes and in gas pipeline systems as a lubricant. Further sale for new use was banned by law in 1979.

Polycyclic aromatic hydrocarbon (PAH) - A class of organic compounds with a fused-ring aromatic structure. PAHs result from incomplete combustion of organic carbon (including wood), municipal solid waste, and fossil fuels, as well as from natural or anthropogenic introduction of uncombusted coal and oil. PAHs include benzo[a]pyrene, fluoranthene, and pyrene.

Recharge - Water that infiltrates the ground and reaches the saturated zone.

Runoff - Excess rainwater or snowmelt that is transported to streams by overland flow, tile drains, or ground water.

SMCL (Secondary maximum contaminant level) - Non-enforceable drinking-water standards established by the U.S. Environmental Protection Agency for contaminants that may adversely affect the odor or appearance of drinking water.

Sediment - Particles, derived from rocks or biological materials, that have been transported by a fluid or other natural process, suspended or settled in water.

Semivolatile organic compound (SVOC) - Operationally defined as a group of synthetic organic compounds that are solvent-extractable and can be determined by gas chromatography/mass spectrometry. SVOCs include phenols, phthalates, and polycyclic aromatic hydrocarbons (PAHs).

Streamflow - A type of channel flow, applied to that part of surface runoff in a stream whether or not it is affected by diversion or regulation.

Suspended sediment - Particles of rock, sand, soil, and organic detritus carried in suspension in the water column, in contrast to sediment that moves on or near the streambed.

Taxon (plural taxa) - Any identifiable group of taxonomically related organisms.

Tile drain - A buried perforated pipe designed to remove excess water from soils.

Till - An unsorted glacial deposit characterized by low permeability.

Triazine herbicide - A class of herbicides containing a symmetrical triazine ring (a nitrogen-heterocyclic ring composed of three nitrogens and three carbons in an alternating sequence). Examples include atrazine, propazine, and simazine.

Volatile organic compounds (VOCs) - Organic chemicals that have a high vapor pressure relative to their water solubility. VOCs include components of gasoline, fuel oils, and lubricants, as well as organic solvents, fumigants, some inert ingredients in pesticides, and some by-products of chlorine disinfection.

Water table - The point below the land surface where ground water is first encountered and below which the earth is saturated. Depth to the water table varies widely across the country.

Water year - The continuous 12-month period, October 1 through September 30, in U.S. Geological Survey reports dealing with the surface-water supply. The water year is designated by the calendar year in which it ends and which includes 9 of the 12 months. Thus, the year ending September 30, 1980, is referred to as water year 1980.

REFERENCES

- Baker, D.B., Calhoun, F.G., and Matisoff, Gerald. 1998. LEASEQ—the Lake Erie agricultural systems for environmental quality project: Heidelberg College Water-Quality Lab, Tiffin, Ohio, 2 p.
- Bertram, P.E., 1993, Total phosphorus and dissolved oxygen trends in the Central Basin of Lake Erie: *Journal of Great Lakes Research*, v. 19, p. 224–236.
- Brody, T.M., Furio, B.A., and Macarus, D.P., 1997, Agricultural pesticide use in the Great Lakes Basin—Estimates of major active ingredients applied during 1994–95 for the Lake Erie, Michigan, and Superior Basins: U.S. Environmental Protection Agency, Region 5, 15 p.
- Casey, G.D., Myers, D.N., Finnegan, D.P., and Wieczorek, M.E., 1997, National Water-Quality Assessment Program in the Lake Erie-Lake St. Clair Basin, Michigan, Indiana, Ohio, Pennsylvania, and New York—Environmental and hydrologic setting: U.S. Geological Survey Water-Resources Investigations Report 97–4256, 93 p.
- Clark, G.M., Mueller, D.K., and Mast, M.A., 2000, Nutrient concentrations and yields in undeveloped stream basins of the United States: *Journal of the American Water-Resources Association*, v. 36, no. 4, p. 849–860.
- Covett, S.A., in press, Comparison of U.S. Geological Survey and Ohio Environmental Protection Agency fish-collection methods using the Index of Biotic Integrity and Modified Index of Well Being: U.S. Geological Survey Water-Resources Investigations Report 00–4255.
- Daher, Shannon, 1999, Lake Erie Lakewide Management Plan (LaMP) status report, 1999: Burlington, Ont., Environment Canada and U.S. Environmental Protection Agency, 32 p.
- Davis, S.N., Whittemore, D.O., and Fabryka-Martin, J., 1998, Uses of chloride/bromide ratios in studies of potable water: *Ground Water*, v. 36, no. 2, p. 338–350.
- Day, K.E., 1991, Pesticide transformation products in surface waters, effects on biota, in Somasunderam, L., and Coats, J.R., eds., *Pesticide transformation products, fate, and significance in the environment*: Washington, D.C., American Chemical Society, ASC Symposium Series 459, p. 217–241.
- Dolan, D.M., 1993, Point source loadings of phosphorus to Lake Erie, 1986–1990: *Journal of Great Lakes Research*, v. 19, p. 212–223.
- Fenelon, J.M., 1998, Water quality in the White River Basin, Indiana, 1992–96: U.S. Geological Survey Circular 1150, 34 p.
- Frey, J.W., in press, Occurrence, distribution, and loading of selected pesticides in tributaries in the Lake Erie-Lake St. Clair Basin, 1996–98: U.S. Geological Survey Water-Resources Investigations Report 00–4169.
- Gibson, George, Carlson, Robert, Simpson, Jonathan, Smeltzer, Eric, Gerritson, Jeroen, Chapra, Steven, Heisarsky, Steven, Jones, Jack, and Kennedy, Robert, 2000, Nutrient criteria technical guidance manual, lakes and reservoirs: U.S. Environmental Protection Agency, Washington, D.C., EPA–822–B00–001, p. 5–14.
- Herdendorf, C.E., 1986, *Rebirth of Lake Erie*: Columbus, Ohio, Ohio Sea Grant Reprint Series OHSU–RS0051, 8 p.
- Ingersoll, C.G., Haverland, P.S., Brunson, E.L., Canfield, T.J., Dwyer, F.J., Henke, C.E., Kemble, N.E., Mount, D.R., and Fox, R.G., 1996, Calculation and evaluation of sediment effect concentrations for the amphipod *Hyalella azteca* and the midge *Chironomus riparius*: *Journal of Great Lakes Research*, v. 22, no. 3, p. 602–623.
- International Joint Commission, 1994, Revised Great Lakes Water Quality Agreement of 1978: Windsor, Ont., International Joint Commission, p. 31.
- Irwin, R.J., Van Mouwerik, M., Stevens, L., Seese, M.D., and Basham, W., 1997, *Environmental contaminants encyclopedia*: Fort Collins, Colo., National Park Service, Water-Resources Division, accessed February, 1998, at URL <http://www1.nature.nps.gov/toxic>.
- Kelleher, Todd, 1999, Ohio EPA Pesticide Special Study—May 1995 through March 1999: Ohio Environmental Protection Agency, Division of Drinking and Ground Waters, accessed November 2, 1999, at URL <http://www.epa.ohio.gov/ddagw/pestspst.html>.
- Keller, C.K., van der Kamp, G., and Cherry, J.A., 1991, Hydrogeochemistry of a clayey till, 1. Spatial variability: *Water Resources Research*, v. 27, no. 10, p. 2543–2554.
- Kolpin, D.W., 2000, Ground water network statistics, National Water-Quality Assessment Program Pesticide National Synthesis Project: U.S. Geological Survey, accessed June 13, 2000, at URL <http://wwwd-casr.wr.usgs.gov/pnsp>.
- Larson, S.J., Gilliom, R.J., and Capel, P.D., 1999, Pesticides in streams of the United States—Initial results from the National Water-Quality Assessment Program: U.S. Geological Survey Water-Resources Investigations Report 98–4222, 92 p.
- Litke, D.W., 1999, Review of phosphorus control measures in the United States and their effects on water quality: U.S. Geological Survey Water-Resources Investigations Report 99–4007, 38 p.
- Long, D.T., Wilson, T.P., Takacs, M.J., and Rezabek, D.H., 1988, Stable-isotope geochemistry of saline near-surface ground water—East-central Michigan basin: *Geological Society of America Bulletin*, v. 100, p. 1568–1577.
- Lowrance, Richard, Altier, L.S., Newbold, J.D., Schnabel, R.R., Groffman, P.M., Denver, J.M., Correll, D.L., Gilliam, J.W., Robinson, J.L., Brinsfield, R.B., Staver, K.W., Lucas, William, and Todd, A.H., 1997, Water-quality functions in riparian forest buffers in Ches-

- peake Bay watersheds: *Environmental Management*, v. 21, no. 5., p. 687-712.
- Mozola, A.J., 1953, A survey of the groundwater resources in Oakland County, Michigan: Michigan Department of Conservation, Geological Survey Division. Publication 48, pt. 2, p. 101-348.
- Myers, D.N., Metzker, K.D., and Davis, Steven, 2000. Status and trends in suspended-sediment discharges, soil erosion, and conservation tillage in the Maumee River Basin—Ohio, Michigan, and Indiana: U.S. Geological Survey Water-Resources Investigations Report 00-4091, 38 p.
- Newell, A.J., Johnson, D.W., and Allen, L.K., 1987. Niagara River biota contamination project—Fish flesh criteria for piscivorous wildlife: New York State Department of Environmental Conservation, Division of Fish and Wildlife, Bureau of Environmental Protection, Technical Report 87-3, 180 p.
- Nolan, B.T. and Stoner, J.D., 2000, Nutrients in ground waters of the conterminous United States, 1992-95: *Environmental Science & Technology*, v. 34, no. 7, p. 1156-1165.
- Nowell, L.H., and Resek, E.A., 1994, Summary of national standards and guidelines for pesticides in water, bed sediment, and aquatic organisms and their application to water-quality assessments: U.S. Geological Survey Open-File Report 94-44, 115 p.
- Ohio Lake Erie Commission, 1998, State of the lake report. Lake Erie quality index: Toledo, Ohio, Ohio Lake Erie Commission, 88 p.
- Ohio Environmental Protection Agency, 1989, Biological criteria for the protection of aquatic life—Ecological Assessment Section Division of Water Quality Planning and Assessment, volume III, standardized biological field sampling and laboratory methods for assessing fish and macroinvertebrate communities: September 1989, [variously paginated].
- _____. 1995, Ohio water-resource inventory, 1994—Summary, conclusions, and recommendations: Technical Bulletin MAS/1995-7-2, 69 p.
- Plummer, L.N., Michael, R.L., Thurman, E.M., and Glynee, P.D., 1993, Environmental tracers for age dating young ground water. in Alley, W.M., ed., 1993, Regional ground-water quality: New York, Van Nostrand Reinhold, p. 255-294.
- Rheaume, S.J., Button, D.T., Myers, D.N., and Hubbell, D.L., in press, Occurrence and distribution of contaminants of concern in surficial bed sediments of rivers and streams in the Lake Erie-Lake St. Clair Basin: U.S. Geological Survey Water-Resources Investigations Report 00-4200.
- Robertson, W.D., Russell, B.M., and Cherry, J.A., 1996, Attenuation of nitrate in aquitard sediments of southern Ontario: *Journal of Hydrology*, v. 180, p. 267-281.
- Smith, S.L., MacDonald, D.D., Keenleyside, K.A., Ingersoll, C.G., and Field, L.J., 1996, A preliminary evaluation of sediment quality assessment values for freshwater ecosystems: *Journal of Great Lakes Research*, v. 22, no. 3, p. 624-638.
- Sweeney, B.W., 1992, Streamside forests and the physical, chemical, and trophic characteristics of piedmont streams in eastern North America: *Water Science Technology*, v. 26, no. 12, p. 2653-2673.
- Thomas, M.A., 2000a, The effect of residential development on ground-water quality near Detroit, Michigan: *Journal of the American Water Resources Association*, v. 36, no. 5, p. 1023-1038.
- Thomas, M.A., 2000b, Ground-water quality and vulnerability to contamination in selected agricultural areas of northwestern Ohio, southeastern Michigan, and northeastern Indiana: U.S. Geological Survey Water-Resources Investigations Report 00-4146, 22 p.
- Trimble, S.W., 1975, Denudation studies—Can we assume stream steady state? *Science*, v. 188, p. 1207-1208.
- _____. 1999, Decreased rates of alluvial sediment storage in the Coon Creek Basin, Wisconsin, 1973-93: *Science*, v. 285, p. 1244-1246.
- U.S. Department of Agriculture, 1998, Toledo Harbor pilot project, final report: Columbus, Ohio, Natural Resources Conservation Service, 65 p.
- U.S. Environmental Protection Agency, 1997, The incidence and severity of sediment contamination in surface waters of the United States: The national sediment quality survey, v. 1, chap.1, p. 1-5.
- U.S. Environmental Protection Agency and Environment Canada, 2000, Lake Erie Lakewide Management Plan 2000, Appendixes E and F: Environment Canada, Ontario Region, Burlington, Ontario, Canada, variously paginated.
- U.S. Geological Survey, 1999, The quality of our Nation's waters—Nutrients and pesticides. U.S. Geological Survey Circular 1225, 82 p.

APPENDIX—WATER-QUALITY DATA FROM THE LAKE ERIE-LAKE SAINT CLAIR DRAINAGES IN A NATIONAL CONTEXT

For a complete view of Lake Erie-Lake Saint Clair Drainages data and for additional information about specific benchmarks used, visit our Web site at <http://water.usgs.gov/nawqa/>. Also visit the NAWQA Data Warehouse for access to NAWQA data sets at <http://infotrek.er.usgs.gov/wdbctx/nawqa/nawqa.home>.

This appendix is a summary of chemical concentrations and biological indicators assessed in the Lake Erie-Lake Saint Clair Drainages. Selected results for this Study Unit are graphically compared to results from as many as 36 NAWQA Study Units investigated from 1991 to 1998 and to national water-quality benchmarks for human health, aquatic life, or fish-eating wildlife. The chemical and biological indicators shown were selected on the basis of frequent detection, detection at concentrations above a national benchmark, or regulatory or scientific importance. The graphs illustrate how conditions associated with each land use sampled in the Lake Erie-Lake Saint Clair Drainages compare to results from across the Nation, and how conditions compare among the several land uses. Graphs for chemicals show only detected concentrations and, thus, care must be taken to evaluate detection frequencies in addition to concentrations when comparing study-unit and national results. For example, in the Lake Erie-Lake Saint Clair Drainages acetochlor concentrations in agricultural streams were similar to the national distribution, but the detection frequency was much higher (61 percent compared to 33 percent).

CHEMICALS IN WATER

Concentrations and detection frequencies, Lake Erie-Lake Saint Clair Drainages, 1996–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals

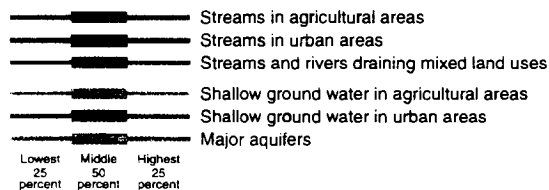
◆ Detected concentration in Study Unit

66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency

— Not measured or sample size less than two

12 Study-unit sample size. For ground water, the number of samples is equal to the number of wells sampled

National ranges of detected concentrations, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

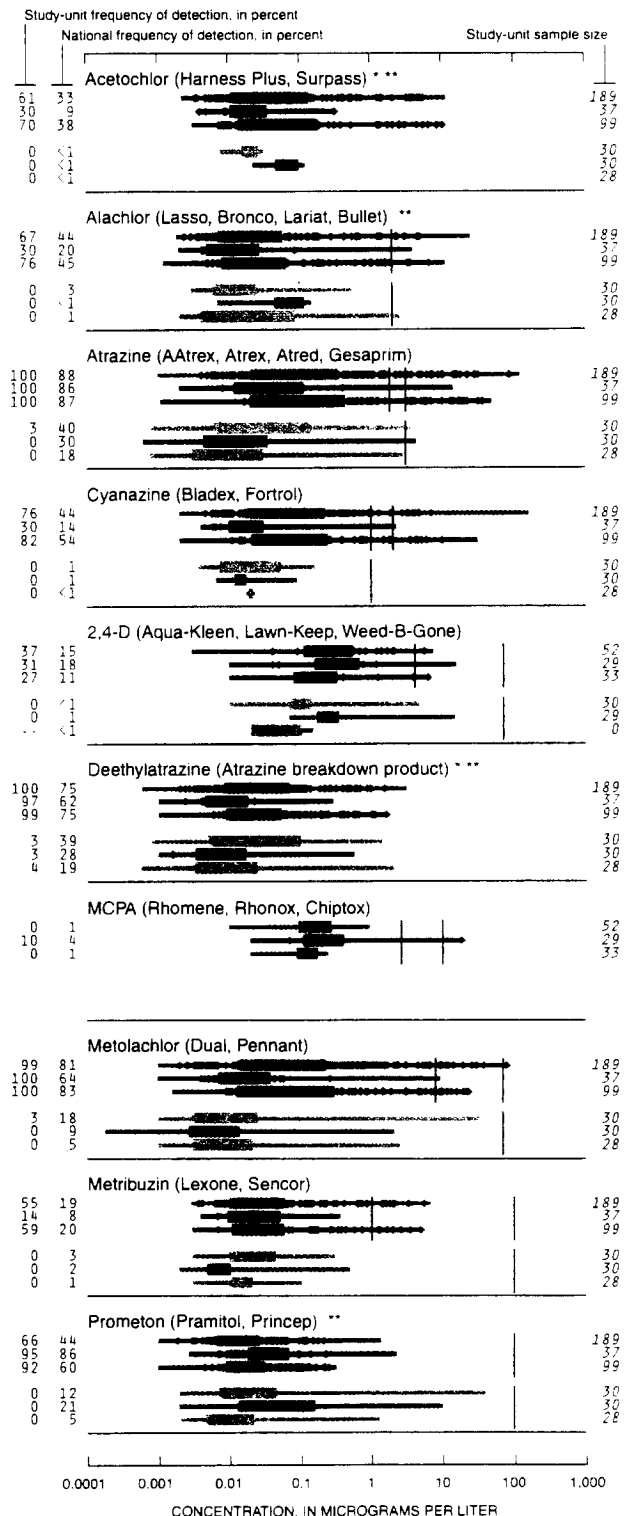


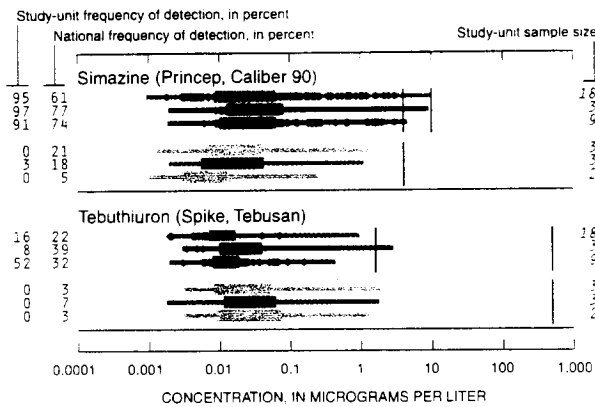
National water-quality benchmarks

National benchmarks include standards and guidelines related to drinking-water quality, criteria for protecting the health of aquatic life, and a goal for preventing stream eutrophication due to phosphorus. Sources include the U.S. Environmental Protection Agency and the Canadian Council of Ministers of the Environment

- | Drinking-water quality (applies to ground water and surface water)
- | Protection of aquatic life (applies to surface water only)
- | Prevention of eutrophication in streams not flowing directly into lakes or impoundments
- * No benchmark for drinking-water quality
- ** No benchmark for protection of aquatic life

Pesticides in water—Herbicides





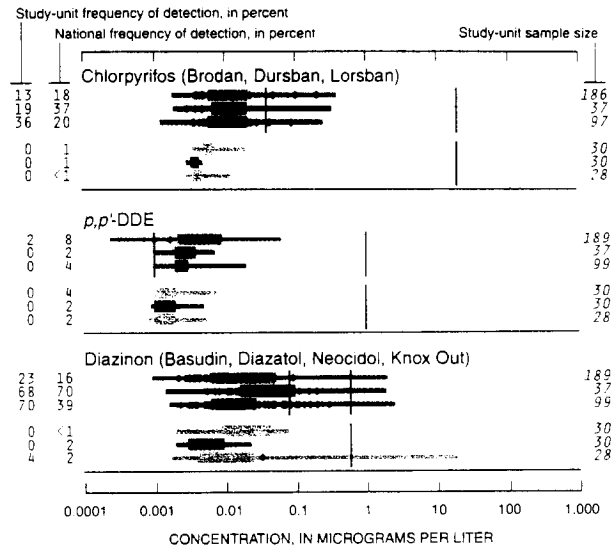
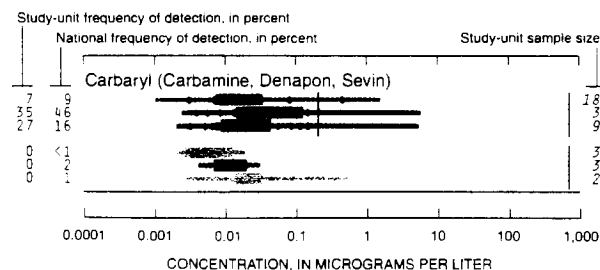
Other herbicides detected

- Acifluorfen (Blazer, Tackle 2S) **
- Benfluralin (Balan, Benefin, Bonalan) ***
- Bentazon (Basagran, Bentazone) **
- Bromoxynil (Buctril, Brominal) *
- Butylate (Sutan +, Genate Plus, Butilate) **
- DCPA (Dacthal, chlorthal-dimethyl) ***
- Dicamba (Banvel, Dianat, Scotts Proturf)
- 2,6-Diethylaniline (Alachlor breakdown product) **
- Diuron (Crisuron, Karmex, Diurex) **
- EPTC (Eptam, Farmarox, Alirox) ***
- Fenuron (Fenulon, Fenidim) **
- Linuron (Lorox, Linex, Sarclex, Linurex, Afalon) *
- MCPB (Thistrol) **
- Pendimethalin (Pre-M, Prowl, Stomp) **
- Picloram (Grazon, Tordon)
- Propachlor (Ramrod, Satecid) **
- Terbacil (Sinbar) **
- Triclopyr (Garlon, Grandstand, Redeem, Remedy) ***
- Trifluralin (Treflan, Gowan, Tri-4, Trific)

Herbicides not detected

- Bromacil (Hyvar X, Urox B, Bromax)
- Chloramben (Amiben, Amilon-WP, Vegiben) **
- Clopyralid (Stinger, Lontrel, Transline) ***
- 2,4-DB (Butyrac, Butoxone, Embutox Plus, Embutone) ***
- Dacthal mono-acid (Dacthal breakdown product) ***
- Dichlorprop (2,4-DP, Seritox 50, Lentemul) ***
- Dinoseb (Dinosebe)
- Ethalfuralin (Sonalan, Curbit) **
- Fluometuron (Flo-Met, Cotoran) **
- Molinate (Ordram) ***
- Napropamide (Devrinol) **
- Neburon (Neburea, Neburyl, Noruben) **
- Norflurazon (Evital, Predict, Solicam, Zorial) **
- Oryzalin (Surflan, Dirimal) ***
- Pebulate (Tillam, PEBC) ***
- Pronamide (Kerb, Propyzamid) **
- Propanil (Stam, Stampede, Wham) **
- Propham (Tuberite) **
- 2,4,5-T **
- 2,4,5-TP (Silvex, Fenoprop) **
- Thiobencarb (Bolero, Saturn, Benthicarb) ***

Pesticides in water—Insecticides



Other insecticides detected

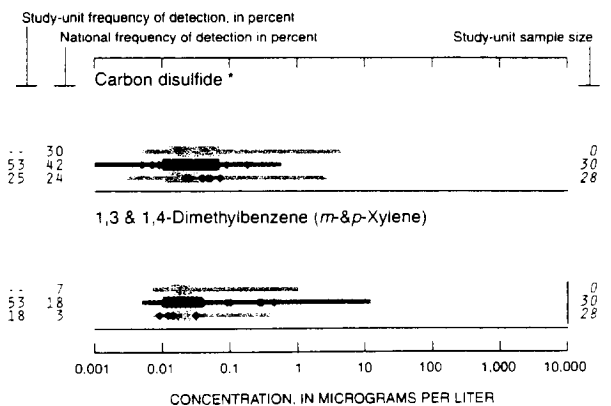
- Carbofuran (Furadan, Curaterr, Yaltox)
- Dieldrin (Panoram D-31, Octalox, Compound 497)
- Fonofos (Dyfonate, Capfos, Cudgel, Tycap) **
- alpha-HCH (alpha-BHC, alpha-lindane) **
- Malathion (Malathion)
- Methyl parathion (Penncap-M, Folidol-M) **

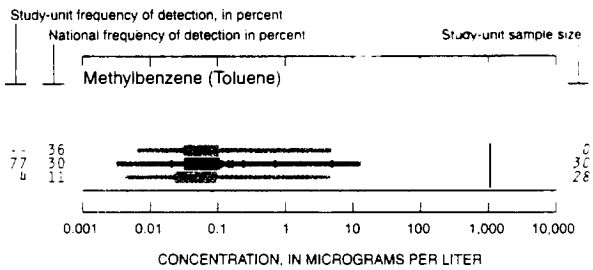
Insecticides not detected

- Aldicarb (Temik, Ambush, Pounce)
- Aldicarb sulfone (Standak, aldoxycarb)
- Aldicarb sulfoxide (Aldicarb breakdown product)
- Azinphos-methyl (Guthion, Gusathion M) *
- Disulfoton (Disyston, Di-Syston) **
- Ethoprop (Mocap, Ethoprophos) ***
- gamma-HCH (Lindane, gamma-BHC)
- 3-Hydroxycarbofuran (Carbofuran breakdown product) ***
- Methiocarb (Slug-Geta, Grandslam, Mesuro) ***
- Methomyl (Lanox, Lannate, Acinate) **
- Oxamyl (Vydate L, Pratt) **
- Parathion (Roethyl-P, Alkron, Panthion, Phoskil) *
- cis-Permethrin (Ambush, Astro, Pounce) ***
- Phorate (Thimet, Granutox, Geomet, Rampart) ***
- Propargite (Comite, Ornite, Ornamite) ***
- Propoxur (Baygon, Blattanex, Unden, Proprotox) **
- Terbufos (Contraven, Counter, Pilarfox) **

Volatile organic compounds (VOCs) in ground water

These graphs represent data from 16 Study Units, sampled from 1996 to 1998





Other VOCs detected

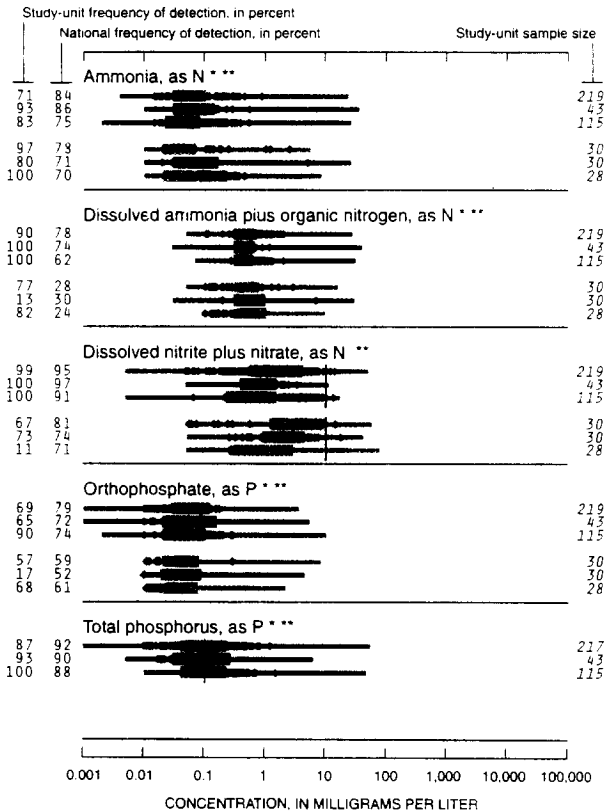
- Benzene
- Bromodichloromethane (Dichlorobromomethane)
- 2-Butanone (Methyl ethyl ketone (MEK)) *
- Chlorodibromomethane (Dibromochloromethane)
- 1,4-Dichlorobenzene (*p*-Dichlorobenzene)
- Dichlorodifluoromethane (CFC 12, Freon 12)
- Dichloromethane (Methylene chloride)
- 1,2-Dimethylbenzene (*o*-Xylene)
- 1-4-Epoxy butane (Tetrahydrofuran, Diethylene oxide) *
- Ethenylbenzene (Styrene)
- Ethylbenzene (Phenylethane)
- Iodomethane (Methyl iodide) *
- 2-Propanone (Acetone) *
- Tetrachloroethene (Perchloroethene)
- 1,1,1-Trichloroethane (Methylchloroform)
- Trichloromethane (Chloroform)
- 1,2,4-Trimethylbenzene (Pseudocumene) *

VOCs not detected

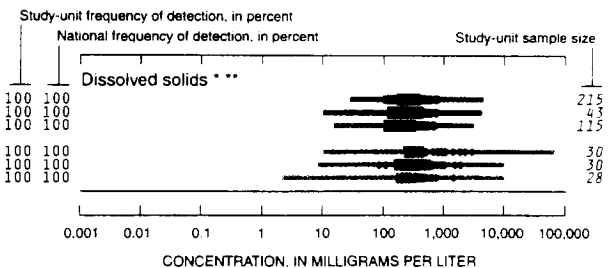
- tert*-Amyl methyl ether (*TAME*) *
- Bromobenzene (Phenyl bromide) *
- Bromochloromethane (Methylene chlorobromide)
- Bromoethene (Vinyl bromide) *
- Bromomethane (Methyl bromide)
- n*-Butylbenzene (1-Phenylbutane) *
- sec*-Butylbenzene *
- tert*-Butylbenzene *
- 3-Chloro-1-propene (3-Chloropropene) *
- 1-Chloro-2-methylbenzene (*o*-Chlorotoluene)
- 1-Chloro-4-methylbenzene (*p*-Chlorotoluene)
- Chlorobenzene (Monochlorobenzene)
- Chloroethane (Ethyl chloride) *
- Chloroethene (Vinyl chloride)
- Chloromethane (Methyl chloride)
- 1,2-Dibromo-3-chloropropane (DBCP, Nemagon)
- 1,2-Dibromoethane (Ethylene dibromide, EDB)
- Dibromomethane (Methylene dibromide) *
- trans*-1,4-Dichloro-2-butene ((*Z*)-1,4-Dichloro-2-butene) *
- 1,2-Dichlorobenzene (*o*-Dichlorobenzene)
- 1,3-Dichlorobenzene (*m*-Dichlorobenzene)
- 1,2-Dichloroethane (Ethylene dichloride)
- 1,1-Dichloroethane (Ethylidene dichloride) *
- 1,1-Dichloroethene (Vinylidene chloride)
- trans*-1,2-Dichloroethene ((*E*)-1,2-Dichloroethene)
- cis*-1,2-Dichloroethene ((*Z*)-1,2-Dichloroethene)
- 1,2-Dichloropropane (Propylene dichloride)
- 2,2-Dichloropropane *
- 1,3-Dichloropropane (Trimethylene dichloride) *
- trans*-1,3-Dichloropropane ((*E*)-1,3-Dichloropropane)
- cis*-1,3-Dichloropropane ((*Z*)-1,3-Dichloropropane)
- 1,1-Dichloropropene *
- Diethyl ether (Ethyl ether) *
- Diisopropyl ether (Diisopropylether (DIPE)) *
- Ethyl methacrylate *
- Ethyl *tert*-butyl ether (Ethyl-*t*-butyl ether (ETBE)) *
- 1-Ethyl-2-methylbenzene (2-Ethyltoluene) *
- Hexachlorobutadiene
- 1,1,1,2,2,2-Hexachloroethane (Hexachloroethane)
- 2-Hexanone (Methyl butyl ketone (MBK)) *
- Isopropylbenzene (Cumene) *
- p*-Isopropyltoluene (*p*-Cymene) *
- Methyl acrylonitrile *
- Methyl *tert*-butyl ether (MTBE)
- Methyl-2-methacrylate (Methyl methacrylate) *
- 4-Methyl-2-pentanone (Methyl isobutyl ketone (MIBK)) *

- Methyl-2-propenoate (Methyl acrylate) *
- Naphthalene
- 2-Propenenitrile (Acrylonitrile)
- n*-Propylbenzene (Isocumene) *
- 1,1,2,2-Tetrachloroethane *
- 1,1,1,2-Tetrachloroethane
- Tetrachloromethane (Carbon tetrachloride)
- 1,2,3,4-Tetramethylbenzene (Phehnitene) *
- 1,2,3,5-Tetramethylbenzene (Isodurene) *
- Tribromomethane (Bromoform)
- 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon 113) *
- 1,2,4-Trichlorobenzene
- 1,2,3-Trichlorobenzene *
- 1,1,2-Trichloroethane (Vinyl trichloride)
- Trichloroethene (TCE)
- Trichlorofluoromethane (CFC 11, Freon 11)
- 1,2,3-Trichloropropane (Allyl trichloride)
- 1,2,3-Trimethylbenzene (Hemimellitene) *
- 1,3,5-Trimethylbenzene (Mesitylene) *

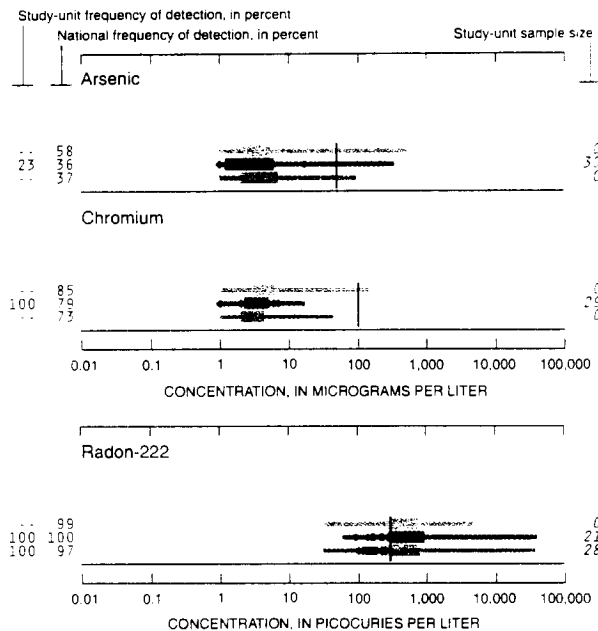
Nutrients in water



Dissolved solids in water



Trace elements in ground water



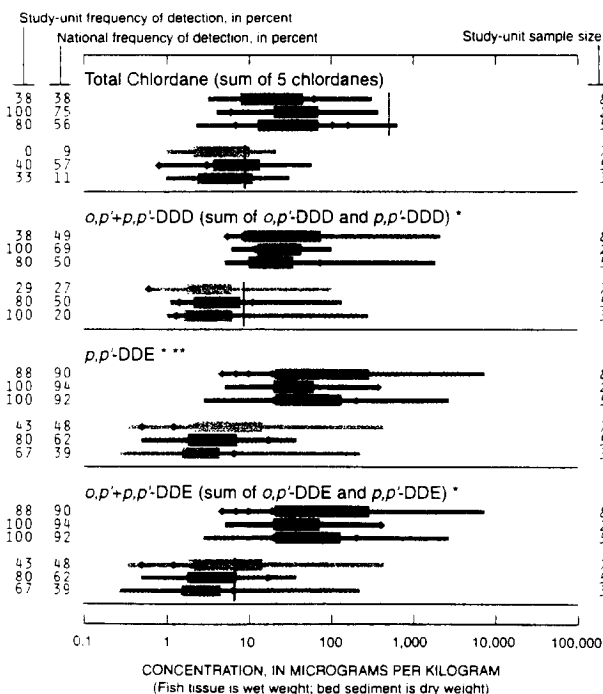
Other trace elements detected

Lead
Selenium
Uranium
Zinc

Trace elements not detected

Cadmium

Organochlorines in fish tissue (whole body) and bed sediment

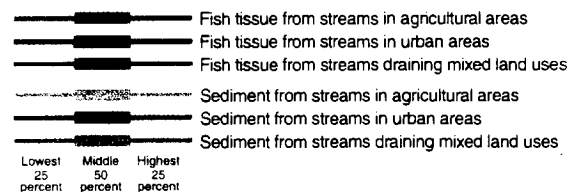


CHEMICALS IN FISH TISSUE AND BED SEDIMENT

Concentrations and detection frequencies, Lake Erie-Lake Saint Clair Drainages, 1996-98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals. Study-unit frequencies of detection are based on small sample sizes; the applicable sample size is specified in each graph

- ◆ Detected concentration in Study Unit
- 56 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- .. Not measured or sample size less than two
- 12 Study-unit sample size

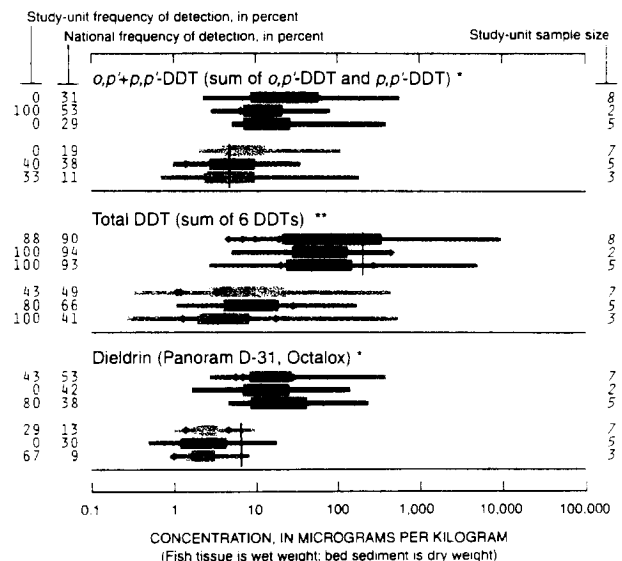
National ranges of concentrations detected, by land use, in 36 NAWQA Study Units, 1991-98—Ranges include only samples in which a chemical was detected

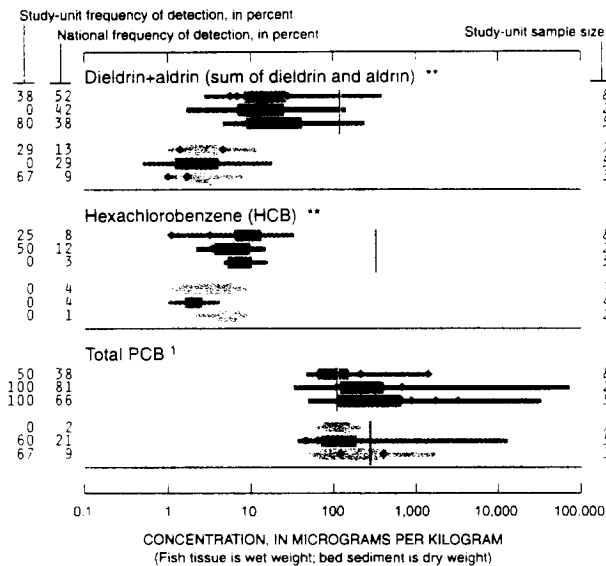


National benchmarks for fish tissue and bed sediment

National benchmarks include standards and guidelines related to criteria for protection of the health of fish-eating wildlife and aquatic organisms. Sources include the U.S. Environmental Protection Agency, other Federal and State agencies, and the Canadian Council of Ministers of the Environment

- | Protection of fish-eating wildlife (applies to fish tissue)
- | Protection of aquatic life (applies to bed sediment)
- No benchmark for protection of fish-eating wildlife
- .. No benchmark for protection of aquatic life





¹ The national detection frequencies for total PCB in sediment are biased low because about 30 percent of samples nationally had elevated detection levels compared to this Study Unit. See <http://water.usgs.gov/nawqa/> for additional information.

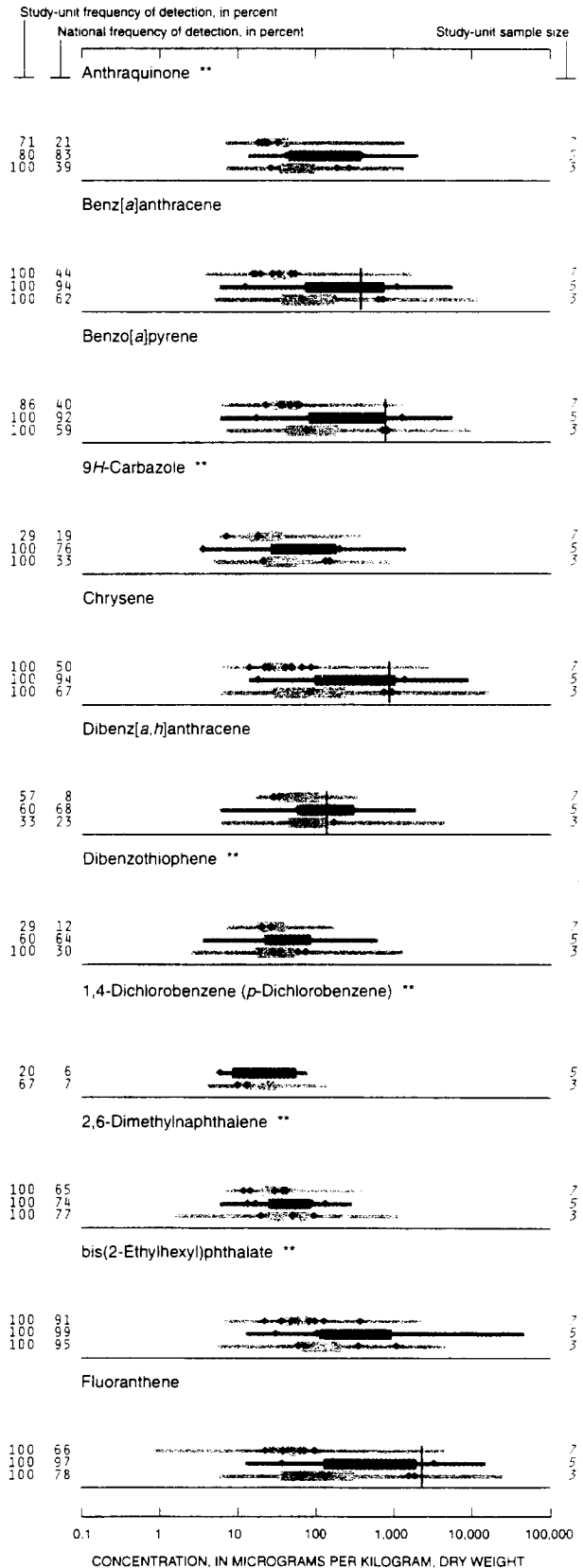
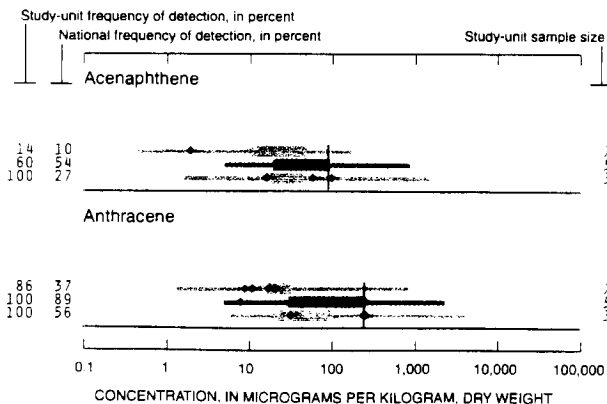
Other organochlorines detected

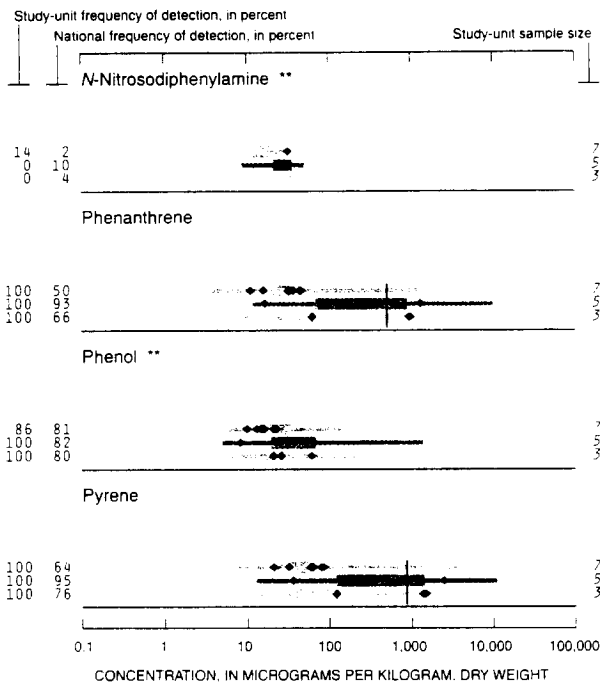
- Heptachlor epoxide (Heptachlor breakdown product) *
- Heptachlor+heptachlor epoxide (sum of heptachlor and heptachlor epoxide) **
- Pentachloroanisole (PCA) * **

Organochlorines not detected

- Chloroneb (Chloronebe, Demosan) * **
- DCPA (Dacthal, chlorthal-dimethyl) * **
- Endosulfan I (alpha-Endosulfan, Thiodan) * **
- Endrin (Endrine)
- gamma-HCH (Lindane, gamma-BHC, Gammexane) *
- Total-HCH (sum of alpha-HCH, beta-HCH, gamma-HCH, and delta-HCH) **
- Isodrin (Isodrine, Compound 711) * **
- p,p'-Methoxychlor (Marlate, methoxychlore) * **
- o,p'-Methoxychlor * **
- Mirex (Dechlorane) **
- cis-Permethrin (Ambush, Astro, Pounce) * **
- trans-Permethrin (Ambush, Astro, Pounce) * **
- Toxaphene (Camphechlor, Hercules 3956) * **

Semivolatile organic compounds (SVOCs) in bed sediment

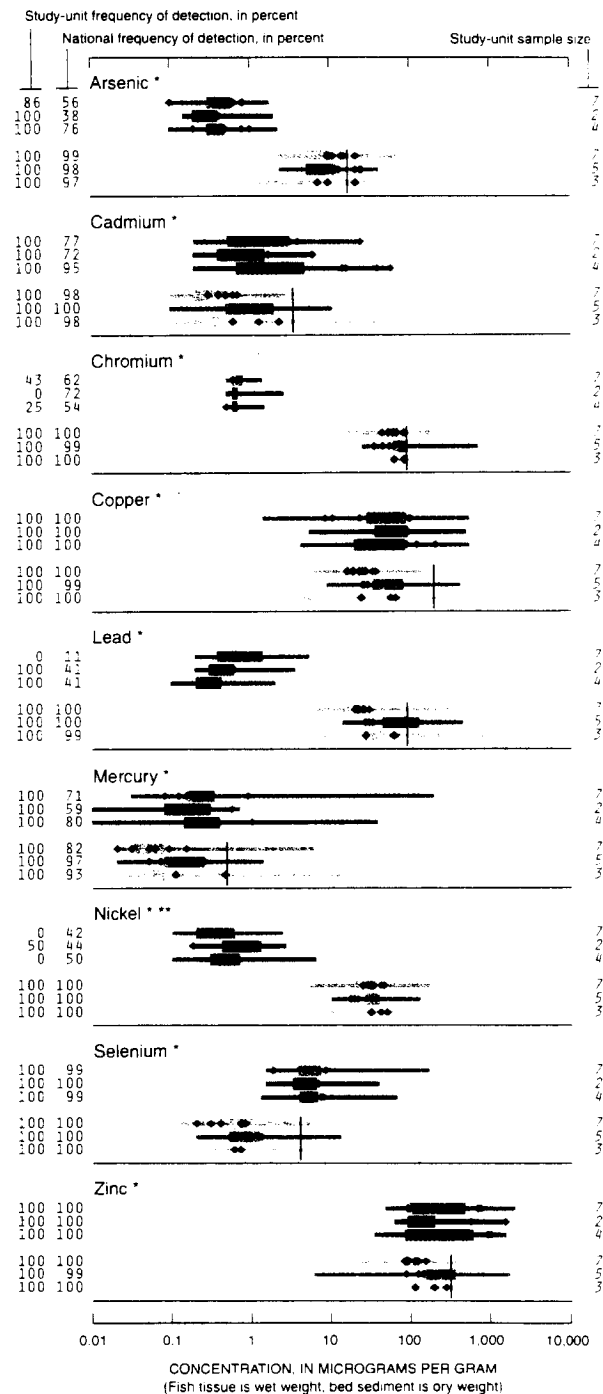




Other SVOCs detected

- Acenaphthylene
 - Acridine **
 - Benzo[b]fluoranthene **
 - Benzo[c]cinnoline **
 - Benzo[ghi]perylene **
 - Benzo[k]fluoranthene **
 - 2,2-Biquinoline **
 - Butylbenzylphthalate **
 - p*-Cresol **
 - Di-*n*-butylphthalate **
 - Di-*n*-octylphthalate **
 - Diethylphthalate **
 - 1,2-Dimethylnaphthalene **
 - 1,6-Dimethylnaphthalene **
 - 3,5-Dimethylphenol **
 - Dimethylphthalate **
 - 2-Ethylnaphthalene **
 - 9*H*-Fluorene (Fluorene)
 - Indeno[1,2,3-*cd*]pyrene **
 - Isophorone **
 - Isoquinoline **
 - 1-Methyl-9*H*-fluorene **
 - 2-Methylanthracene **
 - 4,5-Methylenephenanthrene **
 - 1-Methylphenanthrene **
 - 1-Methylpyrene **
 - Naphthalene
 - Phenanthridine **
 - 2,3,6-Trimethylnaphthalene **
- SVOCs not detected**
- C8-Alkylphenol **
 - Azobenzene **
 - 4-Bromophenyl-phenylether **
 - 4-Chloro-3-methylphenol **
 - bis(2-Chloroethoxy)methane **
 - 2-Chloronaphthalene **
 - 2-Chlorophenol **
 - 4-Chlorophenyl-phenylether **
 - 1,2-Dichlorobenzene (*o*-Dichlorobenzene) **
 - 1,3-Dichlorobenzene (*m*-Dichlorobenzene) **
 - 2,4-Dinitrotoluene **
 - Nitrobenzene **
 - N*-Nitrosodi-*n*-propylamine **
 - Pentachloronitrobenzene **
 - Quinoline **
 - 1,2,4-Trichlorobenzene **

Trace elements in fish tissue (livers) and bed sediment



BIOLOGICAL INDICATORS

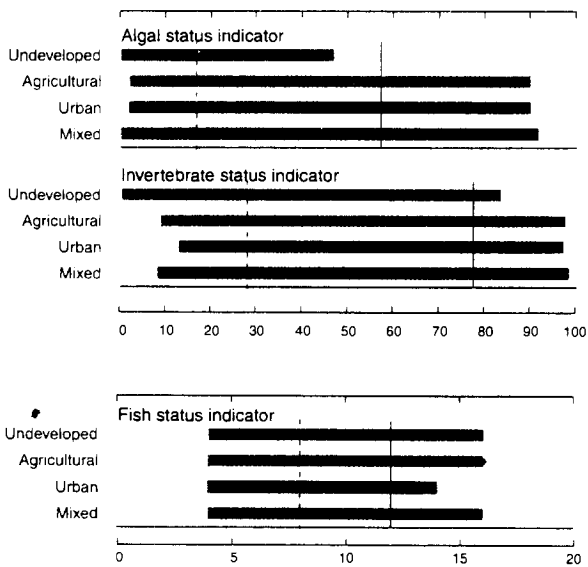
Higher national scores suggest habitat disturbance, water-quality degradation, or naturally harsh conditions. The status of algae, invertebrates (insects, worms, and clams), and fish provide a record of water-quality and stream conditions that water-chemistry indicators may not reveal. **Algal status** focuses on the changes in the percentage of certain algae in response to increasing siltation, and it often correlates with higher nutrient concentrations in some regions. **Invertebrate status** averages 11 metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. **Fish status** sums the scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that increase in association with water-quality degradation.

Biological indicator value, Lake Erie - Lake Saint Clair Drainages, by land use, 1996-98

- ◆ Biological status assessed at a site

National ranges of biological indicators, in 16 NAWQA Study Units, 1994-98

- Streams in undeveloped areas
- Streams in agricultural areas
- Streams in urban areas
- Streams in mixed-land-use areas
- 75th percentile
- - - 25th percentile



A COORDINATED EFFORT

Coordination with agencies and organizations in the Lake Erie-Lake Saint Clair Drainages was integral to the success of this water-quality assessment. We thank those who served as members of our liaison committee.

Federal Agencies

U.S. Environmental Protection Agency
U.S. Army Corps of Engineers
U.S. Department of Agriculture
Environment Canada, Great Lakes Division

State Agencies

Ohio Environmental Protection Agency
Ohio Department of Natural Resources
Ohio Lake Erie Commission
Michigan Department of Agriculture
Michigan Department of Environmental Quality
Indiana Department of Natural Resources
New York Department of Environmental Conservation

Local Agencies

City of Fort Wayne
City of Toledo
Southeast Michigan Council of Governments
Washtenaw County, Michigan, Health Department
Oakland County, Michigan, Health Department
Allen County, Indiana, Soil and Water Conservation
District

Other public and private organizations

Rouge River Program Office, Michigan
The Nature Conservancy
Great Lakes Commission
International Association of Great Lakes Researchers
Michigan State University
University of Michigan, Dearborn

We thank the following individuals for contributing to this effort.

Robert Pigg, Michigan Department of Agriculture
John Estenik, Ohio Environmental Protection Agency
Charles Staudt, Ohio Environmental Protection Agency
Terry Keiser, Ohio Northern University

Special thanks to U.S. Geological Survey employees for their contributions:

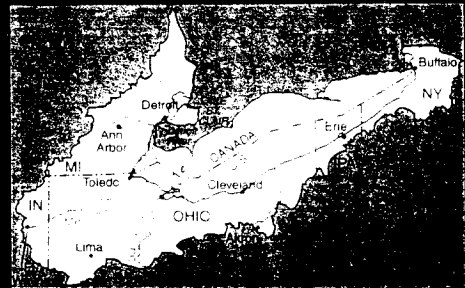
Kevin Metzker, lead technician, Erin Lynch, Bill Yost, Dennis Finnegan, Stephanie Brubeck, Alex Covert, Rick Hubbell, Craig Oberst, David Eckhardt, George Casey, and Michael Wieczorek. Richard Frehs and C. Michael Eberle for production of the report, including editing, layout, and illustrations.

Appreciation also is extended to those individuals and agencies that reviewed this report: Mark Ayers, Charles Peters, and Patrick Phillips, and others at the U.S. Geological Survey; Tom Crane, Great Lakes Commission, and Jeffrey Busch, Ohio Lake Erie Commission.

NAWQA

National Water-Quality Assessment (NAWQA) Program Lake Erie-Lake Saint Clair Drainages

Myers and others—Water Quality in the Lake Erie-Lake Saint Clair Drainages
U.S. Geological Survey Circular 1203



ISBN 0-607-95421-3



9 780607 954210

R0024235

Water Quality in the Kanawha–New River Basin

West Virginia, Virginia, and North Carolina, 1996–98



POINTS OF CONTACT AND ADDITIONAL INFORMATION

The companion Web site for NAWQA summary reports:

<http://water.usgs.gov/nawqa/>

Kanawha–New River Basin contact and Web site:

USGS State Representative
U.S. Geological Survey
Water Resources Division
11 Dunbar Street
Charleston, WV 25301
e-mail: dc_wv@usgs.gov
<http://wv.usgs.gov/nawqa/>

National NAWQA Program:

Chief, NAWQA Program
U.S. Geological Survey
Water Resources Division
12201 Sunrise Valley Drive, M.S. 413
Reston, VA 20192
<http://water.usgs.gov/nawqa/>

Other NAWQA summary reports

River Basin Assessments

Albemarle-Pamlico Drainage Basin (Circular 1157)
Allegheny and Monongahela River Basins (Circular 1202)
Apalachicola-Chattahoochee-Flint River Basin (Circular 1164)
Central Arizona Basins (Circular 1213)
Central Columbia Plateau (Circular 1144)
Central Nebraska Basins (Circular 1163)
Connecticut, Housatonic, and Thames River Basins (Circular 1155)
Eastern Iowa Basins (Circular 1210)
Georgia-Florida Coastal Plain (Circular 1151)
Hudson River Basin (Circular 1165)
Lake Erie–Lake Saint Clair Drainages (Circular 1203)
Las Vegas Valley Area and the Carson and Truckee River Basins (Circular 1170)
Lower Illinois River Basin (Circular 1209)
Long Island–New Jersey Coastal Drainages (Circular 1201)
Lower Susquehanna River Basin (Circular 1168)
Mississippi Embayment (Circular 1208)
Ozark Plateaus (Circular 1158)
Potomac River Basin (Circular 1166)
Puget Sound Basin (Circular 1216)
Red River of the North Basin (Circular 1169)

Rio Grande Valley (Circular 1162)
Sacramento River Basin (Circular 1215)
San Joaquin–Tulare Basins (Circular 1159)
Santee River Basin and Coastal Drainages (Circular 1206)
South-Central Texas (Circular 1212)
South Platte River Basin (Circular 1167)
Southern Florida (Circular 1207)
Trinity River Basin (Circular 1171)
Upper Colorado River Basin (Circular 1214)
Upper Mississippi River Basin (Circular 1211)
Upper Snake River Basin (Circular 1160)
Upper Tennessee River Basin (Circular 1205)
Western Lake Michigan Drainages (Circular 1156)
White River Basin (Circular 1150)
Willamette Basin (Circular 1161)

National Assessments

The Quality of Our Nation's Waters—Nutrients and Pesticides (Circular 1225)

Front cover: The Kanawha River at Kanawha Falls, West Virginia. (Photograph by David Fattaleh, West Virginia Division of Tourism, and used by permission.)

Back cover: Left, Electrofishing on Sewell Creek at East Rainelle, West Virginia (photograph by Edward Vincent, USGS); right, Mountaintop coal mine near Kayford, West Virginia (photograph by James H. Eychaner, USGS).

R0024237

Water Quality in the Kanawha–New River Basin West Virginia, Virginia, and North Carolina, 1996–98

By Katherine S. Paybins, Terence Messinger, James H. Eychaner, Douglas B. Chambers, *and* Mark D. Kozar

U.S. DEPARTMENT OF THE INTERIOR
GALE A. NORTON, SECRETARY

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

2000

Free on application to the
U.S. Geological Survey
Information Services
Box 25286 Federal Center
Denver, CO 80225

Or call: 1-888-ASK-USGS

Library of Congress Cataloging-in-Publications Data

Water Quality in the Kanawha-New River Basin, West Virginia, Virginia, and North Carolina, 1996-98 / by Katherine S. Paybins [et al.].
p. cm. -- (U.S. Geological Survey Circular ; 1204)
Includes bibliographical references.
ISBN 0-607-95412-4 (alk. paper)
1. Water quality--West Virginia--Kanawha River. 2. Water quality--Virginia. 3. Water quality--North Carolina. I. Paybins, Katherine S., 1966- II. Geological Survey (U.S.) III. Series.

TD224.W4 W36 2000
363.739'42'097543--dc21

00-049459

R0024239

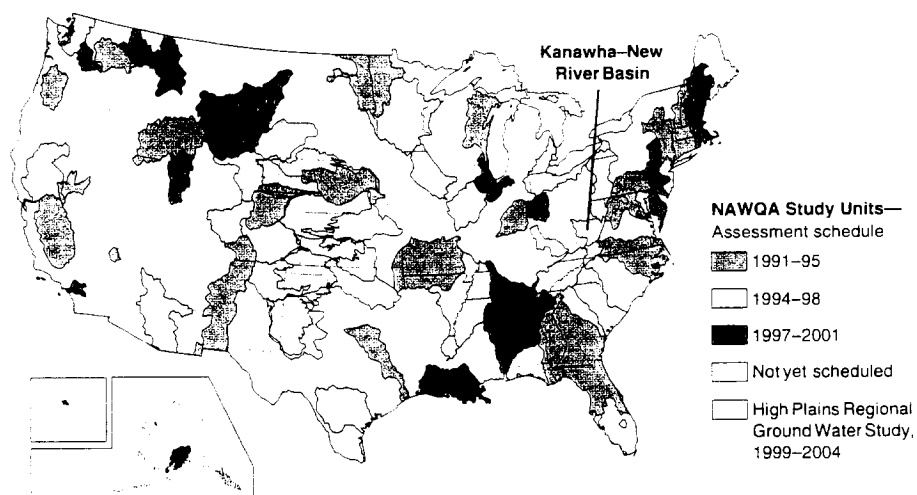
CONTENTS

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM	IV
SUMMARY OF MAJOR FINDINGS.....	1
Stream and River Highlights	1
Ground-Water Highlights.....	2
INTRODUCTION TO THE KANAWHA-NEW RIVER BASIN	3
MAJOR FINDINGS.....	5
Persistent changes in water chemistry and aquatic biology are evident in coal-mined areas.....	5
REGIONAL STUDY—Sulfate concentrations and biological communities in Appalachian coal fields indicated mining-related disturbances despite a general water-quality improvement between 1980 and 1998	9
NATIONAL PERSPECTIVE—Effects of mining on invertebrate communities were of similar magnitude as the effects caused by urban development and agriculture nationally.....	11
Some contaminants are widespread and present at potentially harmful concentrations in streambed sediment and fish tissue.....	11
Fish communities differ considerably throughout the basin, but non-native species continue to expand their range	12
High concentrations of fecal bacteria remain in streams if sources are close	14
Nutrient and organic-chemical concentrations in surface water are low in most of the basin	15
Radon concentrations and bacterial contamination are the principal ground-water-quality concerns	16
NATIONAL PERSPECTIVE—Radon concentrations in ground water were among the highest in the Nation	18
STUDY UNIT DESIGN	20
GLOSSARY	22
REFERENCES	24
APPENDIX—WATER-QUALITY DATA FROM THE KANAWHA-NEW RIVER BASIN IN A NATIONAL CONTEXT	27

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

THIS REPORT summarizes major findings about water quality in the Kanawha–New River Basin that emerged from an assessment conducted between 1996 and 1998 by the U.S. Geological Survey (USGS) National Water–Quality Assessment (NAWQA) Program. Water quality is discussed in terms of local and regional issues and compared to conditions found in all 36 NAWQA study areas, called Study Units, assessed to date. Findings also are explained in the context of selected national benchmarks, such as those for drinking-water quality and the protection of aquatic organisms. The NAWQA Program was not intended to assess the quality of the Nation’s drinking water, such as by monitoring water from household taps. Rather, NAWQA assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water monitoring programs. Comparisons made in this report to drinking-water standards and guidelines are only in the context of the available untreated resource. Finally, this report includes information about the status of aquatic communities and the condition of instream habitats as elements of a complete water-quality assessment.

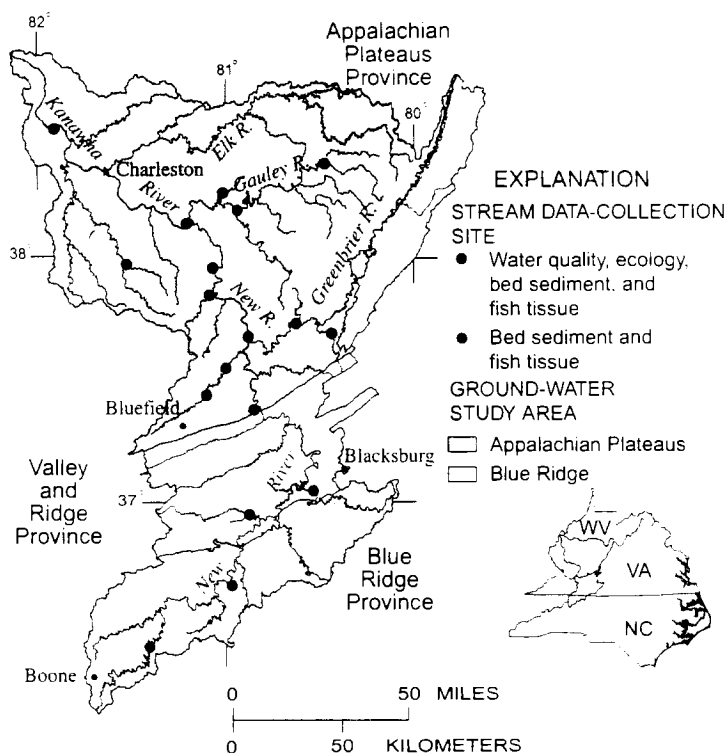
Many topics covered in this report reflect the concerns of officials of State and Federal agencies, water-resource managers, and members of stakeholder groups who provided advice and input during this water-quality assessment. Residents of West Virginia, Virginia, and North Carolina who wish to know more about water quality in the areas where they live will find this report informative as well.



THE NAWQA PROGRAM of the USGS seeks to improve scientific and public understanding of water quality in the Nation’s major river basins and ground-water systems. Better understanding facilitates effective resource management, accurate identification of water-quality priorities, and successful development of strategies that protect and restore water quality. Guided by a nationally consistent study design and shaped by ongoing communication with local, State, and Federal agencies, NAWQA assessments support the investigation of local issues and trends while providing a firm foundation for understanding water quality at regional and national scales. The ability to integrate local and national scales of data collection and analysis is a unique feature of the USGS NAWQA Program.

The Kanawha–New River Basin is one of 51 water-quality assessments initiated since 1991, when the U.S. Congress appropriated funds for the USGS to begin the NAWQA Program. As indicated on the map, 36 assessments have been completed, and 15 more assessments will conclude in 2001. Collectively, these assessments cover about one-half of the land area of the United States and include water resources that are available to more than 60 percent of the U.S. population.

SUMMARY OF MAJOR FINDINGS



The Kanawha–New River Basin is generally mountainous, forested, humid, and rural. Agriculture is concentrated in the southern half of the basin; major products are cattle and hay. Seven percent of all coal mined in the United States is produced from the Appalachian Plateaus Physiographic Province within the basin.

Stream and River Highlights

The generally low population and intensity of agriculture and urban land uses throughout the Kanawha–New River Basin are reflected in low concentrations of nutrients and pesticides in streams and rivers.

Streams in the coal region of the Appalachian Plateaus Physiographic Province generally improved between about 1980 and 1998 with respect to pH, total iron, total manganese, and sedimentation. These improvements were among the regulatory goals of the Surface Mining Control and Reclamation Act of 1977 (SMCRA). Other unregulated factors, however, show the effects of continued mining. Mine drainage in the basin is rarely acidic but has high concentrations of sulfate, which decrease slowly after mining ends. Stream-bottom sedimentation in mined basins remains greater than in undisturbed basins.

- Streams draining basins that have been mined since 1980 show increased dissolved sulfate, decreased median bed-sediment particle size, and impaired benthic-invertebrate communities compared to streams not mined since 1980. (p. 5–11)

- In all basins studied where more than 100,000 tons of coal per square mile have been mined, the stream benthic-invertebrate community is impaired in comparison to rural parts of the basin where less than 10,000 tons of coal per square mile have been mined since 1980. Some basins in which the benthic-invertebrate community is impaired, however, were not heavily mined. Benthic invertebrates are sensitive indicators of many types of disturbance and respond to impairment of either stream chemistry or physical habitat. (p. 7–8)

- Effects on stream benthic-invertebrate communities caused by coal mining were of similar magnitude to the effects caused by urban development and agriculture elsewhere in the Nation. (p. 11)

- Kanawha Falls is the upstream limit for the range of several fish species. Non-native fish continue to expand their range in tributaries of the New and Gauley Rivers. (p. 12–14)

- *Escherichia coli* (*E. coli*) bacteria concentrations exceeded the national guideline for public swimming areas in 26 percent of samples from major rivers and in 43 percent of samples from tributary streams, but no outbreak of waterborne disease was reported during 1991–98. Inadequate sewage treatment and manure management contribute to elevated *E. coli* concentrations. (p. 14–15)

- Volatile organic compounds (VOCs) continue to be detected in the Kanawha River downstream from the Charleston metropolitan area. (p. 16)

- Nickel, chromium, zinc, and certain toxic organic compounds were found in bed sediment in concentrations that could harm aquatic life. Elevated concentrations of cadmium, mercury, nickel, selenium, and zinc were measured in fish tissue at some sites. (p. 12)

Major Influences on Streams and Rivers

- Coal mining
- Improper disposal of human and animal wastes
- Past industrial activities

Selected Indicators of Stream-Water Quality

	Small Streams			Major Rivers
	Coal Mining	Agricultural	Forest	Mixed Land Uses
Pesticides ¹	—	•	—	
Nutrients ²	—	—	—	—
Bacteria ³	◐	◑	◒	◓
Trace elements ⁴	◔	—	◕	◖

◐ Percentage of samples with concentrations **greater than or equal to** health-related national guidelines for drinking water, protection of aquatic life, or contact recreation; or above a national goal for preventing excess algal growth

◑ Percentage of samples with concentrations **less than** health-related national guidelines for drinking water, protection of aquatic life, or contact recreation; or below a national goal for preventing excess algal growth

◒ Percentage of samples with **no detection** (* Detected in 1 percent or less of samples)

— Not assessed

¹ Insecticides, herbicides, and pesticide metabolites, sampled in water.
² Phosphorus and nitrogen, sampled in water.
³ *Escherichia coli* (*E. coli*) bacteria, sampled in water.
⁴ Nickel, chromium, zinc, and lead, sampled in streambed sediment.

Ground-Water Highlights

Ground water in the Appalachian Plateaus and Blue Ridge Physiographic Provinces moves mostly in a network of narrow fractures within a few hundred feet of the land surface, and drains toward the nearest stream. Wells normally tap only a few of the many local fractures. The ridgetops bound each local aquifer, which generally are affected only by local contaminant sources. In small areas of the basin where caves and solution cavities in limestone bedrock are common, wells can have high yields but are susceptible to contamination from fecal bacteria, pesticides, and other toxic chemicals.

- Radon concentrations in the Blue Ridge were among the highest in the Nation. Almost 90 percent of wells sampled there exceeded the proposed U. S. Environmental Protection Agency (USEPA) primary drinking-water standard of 300 picocuries per liter (pCi/L). One-third of these wells contained more than 4,000 pCi/L, the proposed alternate drinking-water standard. Radon is a radioactive gas that forms during the decay of natural uranium. (p. 18–19)

- Modern well construction can prevent fecal bacteria from reaching drinking water in most areas of the basin. Bacteria were frequently detected only at older wells. (p. 19)
- Potentially explosive concentrations of methane were found in water at 7 percent of wells in the coal region of the Appalachian Plateaus. (p. 17)
- Nutrients, pesticides, and VOCs were detected in low concentrations throughout the basin. In the Blue Ridge, however, water from more than 50 percent of wells contained pesticides, an indication that the ground water is vulnerable to contamination. (p. 19)
- In the Appalachian Plateaus, iron and manganese concentrations exceeded USEPA drinking-water guidelines in at least 40 percent of the wells and in about 70 percent of wells near reclaimed surface coal mines. Elevated sulfate concentration and slightly acidic water were more common at wells within 1,000 feet of reclaimed mines than elsewhere. (p. 10 and 17)

Major Influences on Ground Water

- Composition of soils and bedrock
- Improper disposal of human and animal wastes
- Current and past mining practices
- Pesticide usage and other toxic chemical releases

Selected Indicators of Ground-Water Quality

	Domestic Supply Wells		
	Appalachian Plateaus, Mining	Appalachian Plateaus	Blue Ridge
Pesticides ¹	—	—	—
Radon	◐	◑	◒
Volatile organics ²	—	—	—
Bacteria ³	◐	◑	◒
Nitrate	—	—	—

◐ Percentage of samples with concentrations **greater than or equal to** health-related national guidelines for drinking water

◑ Percentage of samples with concentrations **less than** health-related national guidelines for drinking water

◒ Percentage of samples with **no detection**

— Not assessed

¹ Insecticides, herbicides, and pesticide metabolites, sampled in water.
² Solvents, refrigerants, fumigants, gasoline, and gasoline additives, sampled in water.
³ Fecal coliform bacteria, sampled in water.

INTRODUCTION TO THE KANAWHA–NEW RIVER BASIN

Population and Human Activities

The Kanawha River and its major tributary, the New River, drain 12,223 mi² in North Carolina, Virginia, and West Virginia (Messinger and Hughes, 2000). Most of the total basin population of 870,000 (1990 data) live in rural areas, and industrial and residential areas cover less than 5 percent of the total area in the basin (fig. 1). Only about 30 percent of the population live in towns larger than 10,000 people, including the 25 percent who live in the Charleston, W. Va.,

metropolitan area. The total population has not changed substantially since the 1950s, mostly because of emigration from rural parts of the basin to urban centers in the Midwest and the South.

The only major industrial area in the basin is along the terrace of the Kanawha River, within about 20 miles of Charleston (fig. 2). Chemical industry practices that profoundly polluted the Kanawha River during the 1950s and 1960s have changed, and discharge of pollutants to streams has greatly decreased, although

bed sediment and fish remain contaminated with dioxin and other industrial chemicals (Henry, 1981; Kanetsky, 1988; West Virginia Division of Environmental Protection, 2000).

In the Kanawha–New River Basin, most coal is mined in the Appalachian Plateaus in West Virginia (McColloch, 1998). About 7 percent of the coal mined in the United States comes from the Kanawha–New River Basin (Fedorko and Blake, 1998; Messinger and Hughes, 2000). Most coal mined in the basin has a low sulfur content. Coal production has increased since passage of the Clean Air Act amendments of 1990, which mandated a reduction of sulfate emissions to decrease acid precipitation.

Physiography

The streams and rivers of the basin drain areas in three physiographic provinces: the Blue Ridge (17 percent), the Valley and Ridge (23 percent), and the Appalachian Plateaus (60 percent). In the Appalachian Plateaus, little of the land is flat, and most flat land is in the flood plains and terraces of streams.

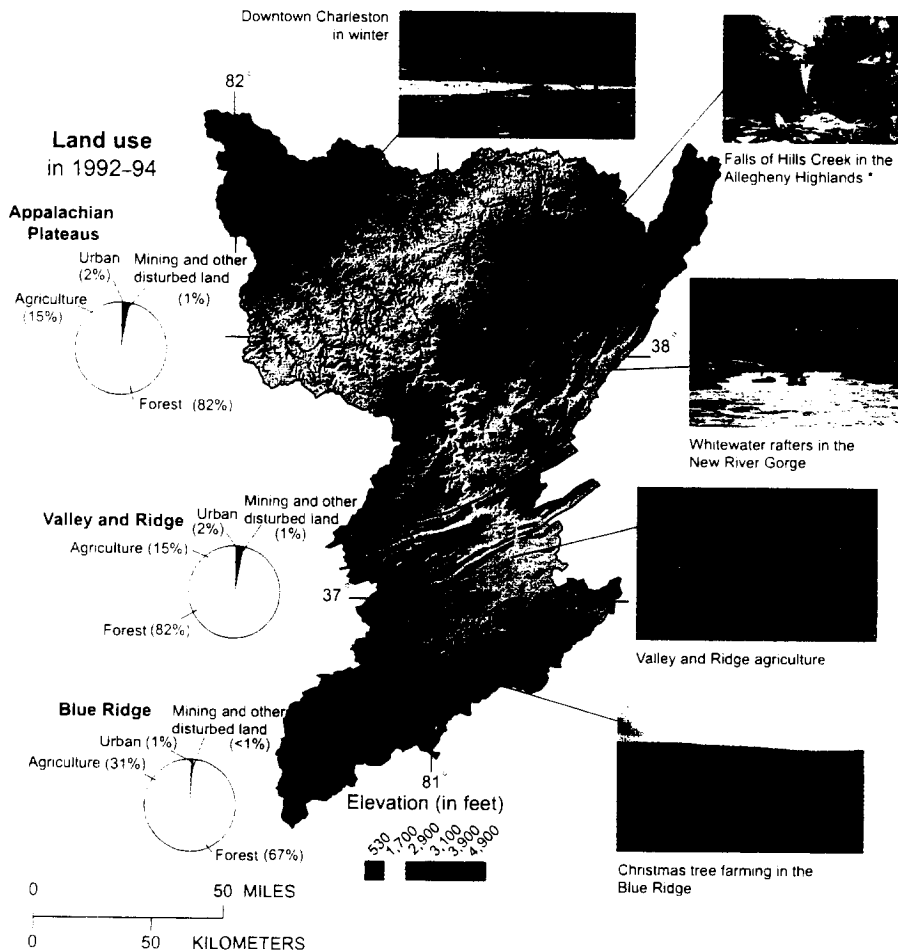


Figure 1. In the mountainous Kanawha–New River Basin, elevation ranges from over 4,000 feet in the Allegheny Highlands of the Appalachian Plateaus Province and the Blue Ridge Province to about 560 feet at the mouth of the river at Point Pleasant, W. Va. Forest accounted for 81 percent of the land cover in 1993 (Multi-Resolution Land Characteristics Interagency Consortium, 1997). Logging is a major industry throughout the basin. The entire basin was logged by the early 20th century, and no undisturbed areas remain (Clarkson, 1964). Coal mining is prevalent in the Appalachian Plateaus. The Blue Ridge Province contains proportionally more agricultural land than the Appalachian Plateaus and Valley and Ridge Provinces. Cattle, hay, and corn grown as cattle feed are the primary agricultural products (National Agriculture Statistics Service, 1999). Physiographic provinces from Fenneman, 1938.

* Photograph by Julie Archer, and used by permission.

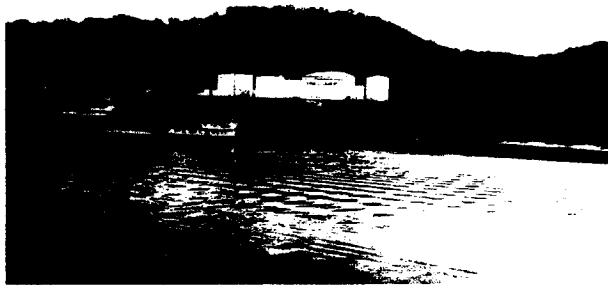


Figure 2. Coal and motor fuel commonly are transported by barge on the Kanawha River, downstream from Kanawha Falls.

The Valley and Ridge is characterized by strongly folded ridges separated by relatively flat, broad valleys. These two regions are underlain by sedimentary rocks. The Blue Ridge is characterized by igneous and metamorphic rocks that have been folded and faulted.

Water Use

In 1995, 61 percent of the basin's population depended on surface-water supplies for domestic needs (Solley and others, 1998). Thirty percent relied on domestic water wells. The remaining nine percent used public-supply water wells. In 1995, total withdrawal of water was about 1,130 Mgal/d (million gallons per day); total consumptive use was about 118 Mgal/d.

Hydrologic Conditions and Features

With some exceptions, mean streamflow during the study was within about 10 percent of long-term mean flows at most gaging stations (see records from a representative station in fig. 3). Major flooding occurred throughout the Appalachian Plateaus in January 1996, seven months before sampling began, and streamflow at several gaging stations within the Kanawha–New River Basin exceeded the 100-year flood flow (Ward and others, 1997). A thunderstorm in June 1998 caused flooding in the northwestern part of the basin where flow on a few small streams exceeded the 100-year recurrence interval (Ward and others, 1999). With the exception of these floods, no other flows exceeded the 10-year recurrence

interval. No streams in the basin were in drought conditions during the study.

Streamflow varies most through the year in the western Appalachian Plateaus, and it varies least through the year in the Blue Ridge. On average, streamflow throughout the basin is greatest in February and March and least in September through October. Maximum streamflow does not coincide with maximum precipitation because summer vegetation uses a large fraction of the precipitation.

The river system in the Kanawha–New River Basin is regulated by four major flood-control dams, three navigation dams, and several smaller dams. The two largest dams are on the Gauley River (Summersville Dam) and Elk River (Sutton Dam). The other two major dams are on the New River. The navigable reach of the Kanawha River is in backwater caused by the navigation dams. In this reach, stream depth is greater and velocity is less than in the undammed reaches of the major rivers. All pools behind dams in the basin collect sediment. Dams are also major barriers to fish movement.

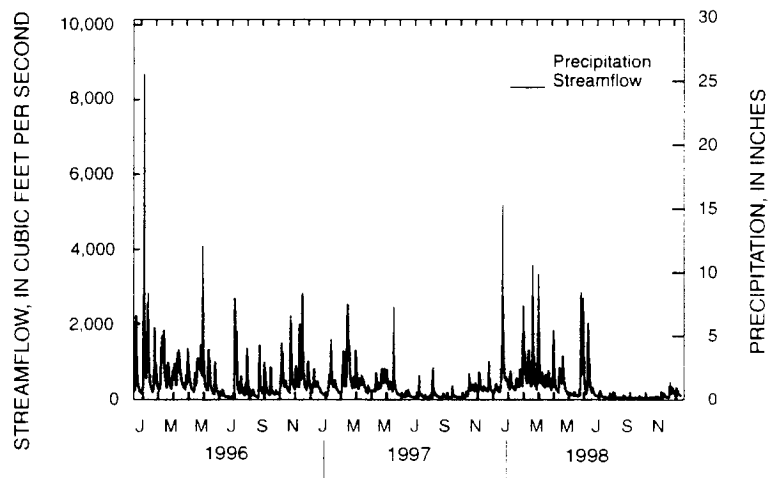


Figure 3. After a major flood in January 1996, streamflow from Williams River at Dyer, W. Va., and precipitation from Richwood, W. Va., were normal throughout the study period. The long-term average annual streamflow at Williams River at Dyer, W. Va. is 336 cubic feet per second. Long-term average precipitation at the Richwood, W. Va. location is 48 inches per year.

MAJOR FINDINGS

Persistent Changes in Water Chemistry and Aquatic Biology are Evident in Coal-Mined Areas

About 7 percent of all coal mined in the Nation comes from an area of 5,000 mi² in the Appalachian Plateaus part of the Kanawha–New River Basin. Production of the mostly low-sulfur coal nearly doubled from 1980 to 1998 as mining technology advanced, individual mines became larger, and employment decreased. Total production is about 90 million tons per year. A coal seam 1 foot thick and 1 mile square weighs about 1 million tons.

Most drainage basins within the coal region have been mined repeatedly as technology has advanced and economics have changed. Only three unmined basins greater than 10 mi² in the coal mining region were identified in this study. Among mined basins, cumulative coal production of less than 10,000 ton/mi² of coal during 1980–95 is low. Cumulative production in many basins ranged from 100,000 to 1,000,000 ton/mi².

Most water that drains from coal mines in the Kanawha–New River Basin is naturally neutral or alkaline rather than acidic. When iron pyrite in coal and adjacent rocks is exposed to air and water during mining, a series of chemical reactions produce dissolved iron and sulfuric acid (Rose and Cravotta, 1998). Natural or applied limestone, lye, or anhydrous ammonia can neutralize the acid (Skousen and others, 1998), but sulfate ions dissolved in water generally remain as evidence of the reactions. Sulfate concentrations in streams decrease slowly after mining ends (Sams and Beer, 2000).

Since 1981, Total Iron and Manganese have Decreased in Stream Basins where Coal Mining has Continued, but Sulfate has Increased

During low flow in July 1998, water samples from 57 wadeable streams (drainage area less than 1 to 128 mi²) were analyzed once. Samples were collected from streams in the region of the Appalachian Plateaus where coal has been mined. At least three analyses were available for 51 of the sites for 1979–81, before the Surface Mining Control and Reclamation Act (SMCRA) affected regional water quality (Ehlke and others, 1982). Each 1998 analysis was compared to the one earlier analysis with the closest corresponding stream-flow. Results were interpreted with respect to cumulative mining history and other land uses in each basin.

Median concentrations of total iron and total manganese were lower in 1998 than during 1979–81 in 33 basins that had been mined both before and after SMCRA, but sulfate concentration and specific conductance were higher (table 1). In 1998, median total manganese, specific conductance, sulfate, and pH were higher in 37 basins mined since 1980 than in 20 basins unmined since then; median total iron was lower in the mined basins, possibly reflecting aggressive treatment of permitted discharges.

Table 1. Medians of regulated constituents improved between 1979–81 and 1998 in 33 mined basins

[µg/L, micrograms per liter; µS/cm, microsiemens per centimeter; mg/L, milligrams per liter]

Regulated Constituents	Median value	
	1979–81	1998
pH (standard units)	7.1	7.5
Total iron (µg/L)	455	150
Total manganese (µg/L)	150	78
Unregulated Constituents		
Specific conductance (µS/cm)	360	446
Sulfate (mg/L)	91	150

At the time the SMCRA and subsequent regulations were established, acidification and subsequent increase in metal concentrations, but not sulfate concentration, were known to degrade stream quality. Regulations,

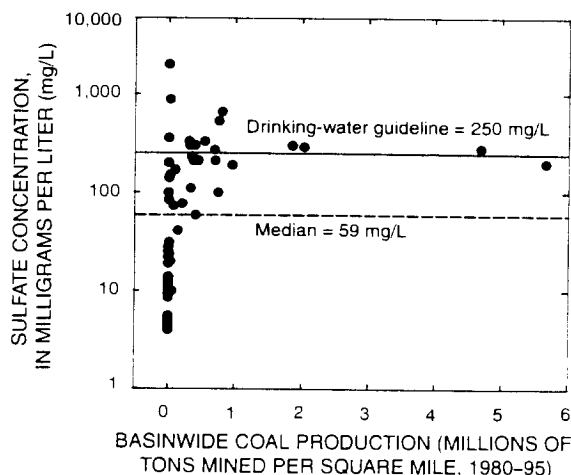


Figure 4. Sites with a low concentration of sulfate drained basins with little recent coal production. Sites with a high concentration of sulfate drained basins with a wide range of recent coal production.

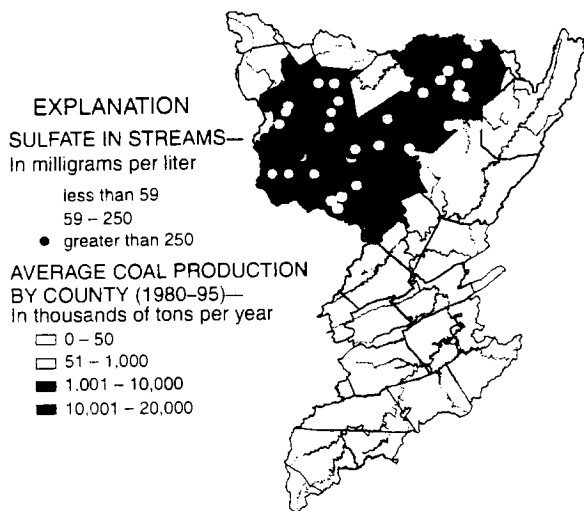


Figure 5. Sulfate concentration in wadeable streams was highest in counties with the highest coal production.

therefore, were targeted at decreasing mining-related acidification and concentrations of iron and manganese, but were not designed to decrease sulfate concentrations. Sulfate concentrations less than 59 mg/L (milligrams per liter; study median) were measured only from basins where less than 142,000 ton/mi² of coal were produced during 1980–95 (figs. 4 and 5). In contrast, manganese concentrations less than 32 µg/L (micrograms per liter; study median) were measured at several heavily mined basins (fig. 6).

Sulfate concentration in streams draining mined areas does not correlate strongly with coal production because sulfate production depends on local geology, mining practice, and possibly results from activities in addition to mining. Sulfate concentration is higher than background, however, in basins with the greatest coal production. Background sulfate concentration was less than 25 mg/L in 16 of 20 basins not mined since 1980. In contrast, sulfate concentration was greater than 250 mg/L in 8 of 15 mined basins drained by streams tributary to the Coal River. The USEPA guideline for sulfate in drinking water is 250 mg/L.

For two years, water chemistry was analyzed monthly and at high flow at two streams in heavily mined basins, and at one stream where no coal had been mined since 1980. At the mined sites, sulfate, several other ions, and specific conductance decreased as streamflow increased; at the unmined site, major-ion concentrations were low at all flows (fig. 7). Dissolved iron and manganese concentrations were virtually unrelated to flow at all three sites. At both Peters Creek near

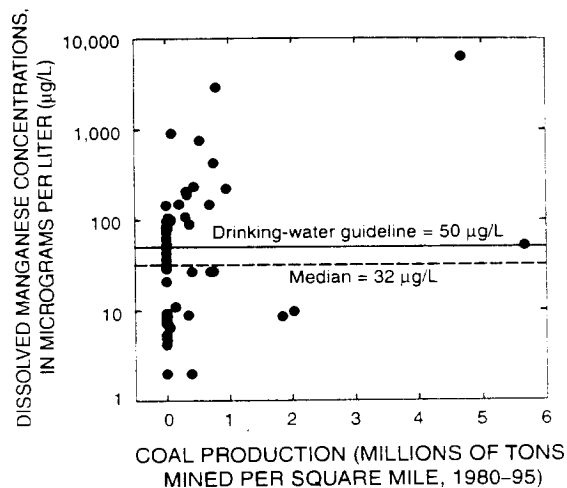


Figure 6. Concentrations of manganese in about half of the streams draining heavily mined basins were less than the study median.

Lockwood and Clear Fork at Whitesville, specific conductance was correlated with sulfate concentration, and correlations were nearly as strong between specific conductance and dissolved calcium, magnesium, sodium, and chloride. The same patterns were found in data for the sites before the implementation of the SMCRA.

Streamflow, water temperature, pH, and specific conductance were measured hourly at the two mined sites during the same two years. In the Coal River Basin at Clear Fork, sulfate concentration (estimated from the hourly specific conductance) exceeded the

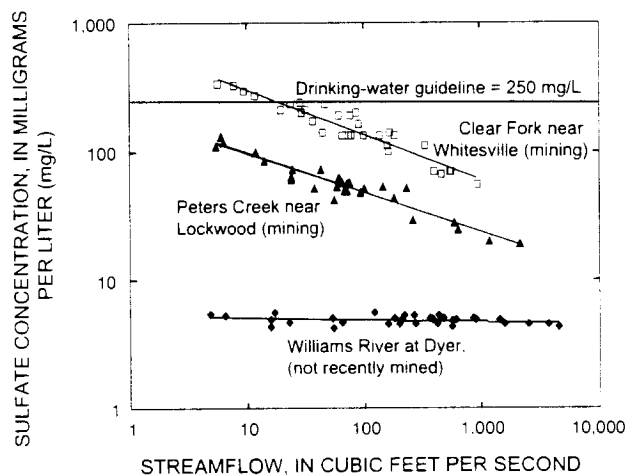


Figure 7. The concentration of sulfate, like other major ions, decreased with flow at two heavily mined sites but was consistently low at a site with no recent mining (Clear Fork $R^2 = 0.90$, Peters Cr $R^2 = 0.91$, Williams River $R^2 = 0.11$).

250-mg/L guideline about 25 percent of the time. Sulfate concentrations across a range of flow at Clear Fork were at least 10 percent greater in 1998 than in 1979–81.

Coal-mining methods in the Kanawha–New River Basin

In the Kanawha–New River Basin, half of the coal comes from underground mines and half from surface mines. Surface subsidence is expected above longwall mines, which remove about 90 percent of a coal seam, but is less common above room-and-pillar mines that may remove only 60 percent. Surface mines, both smaller contour mines and larger mountaintop mines, can remove 100 percent of a series of seams. Surface-mine operators working in steep-slope areas cannot simply replace all waste-rock material within the boundaries of the mine sites, because broken rock takes more space than consolidated rock. The excess is placed in valleys as fill material where the land is flat enough to provide a stable foundation, but the valley fills greatly affect the stream environment (U.S. Environmental Protection Agency, 2000).

Stream Benthic-Invertebrate Communities are Impaired at Mined Sites

In all streams sampled that drain areas where large quantities of coal have been mined, the benthic-invertebrate community is impaired in comparison to rural parts of the study area where little or no coal has been mined since 1980 (fig. 8). Some streams in which

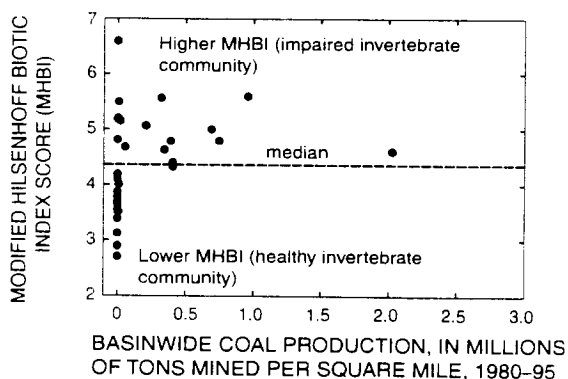


Figure 8. Only sites with little recent coal production had healthy invertebrate communities as measured by low (favorable) scores on the Modified Hilsenhoff Biotic Index, although not all impaired sites were in areas of high coal production.

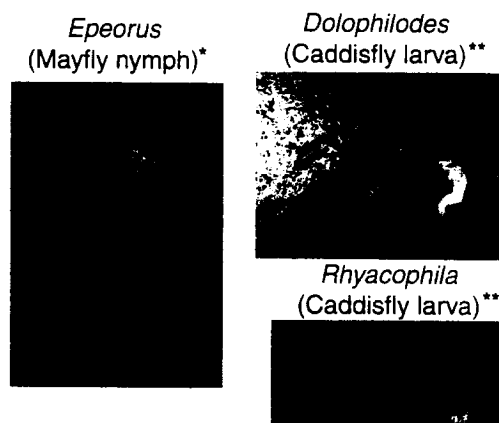


Figure 9. Invertebrates that are intolerant of fine sediment were present at unmined sites and sites with little coal production since 1980. (Photograph by * Jennifer Hiebert, University of Alberta; ** D.B. Chambers, USGS; *** Arturo Elosegi, North American Benthological Society. All photos reproduced with permission)

the community is impaired drained areas that were not heavily mined.

Invertebrate communities were sampled from riffles at 29 wadeable streams in areas of the Appalachian Plateaus where coal is or has been mined (Chambers and Messinger, 2001). The sites were separated into two groups by statistical comparison of species composition and abundance. Each group contained communities that were similar. The communities that included several insect taxa known for intolerance of fine sediment were identified as the less impaired group of sites. These taxa include *Epeorus* mayflies and *Dolophilodes* and *Rhyacophila* caddisflies (fig. 9). *Epeorus* is a genus of relatively large mayflies that cling to the bottom of large, loosely embedded rocks. Fine sediment can fill the openings in the stream bottom where they live. Caddisflies in the genus *Dolophilodes* spin finely meshed nets that can be clogged with silt. *Rhyacophila* are mobile predators typically found in clean, cool-water streams. These intolerant taxa were not present in the invertebrate communities at sites identified as poorer. In addition, scores from the MHBI (Modified Hilsenhoff Biotic Index; see glossary) and proportions of pollution-tolerant taxa from the midge family were significantly greater at the more impaired group of sites. The MHBI and other biological metrics are mathematical summaries of characteristics that change predictably in response to environmental stress. They are used to measure ecological health of a system (Karr and Chu, 1999).

Benthic invertebrates are good indicators of overall stream-water quality

Benthic invertebrates are sensitive indicators of many types of stream disturbance (Barbour and others, 1999). Because most have a life span of about a year and many remain in the same short section of stream during most of their lives, they are particularly well suited for assessments of short-term, local disturbances within a watershed. Fish, however, often move throughout a stream system, enabling them to seek refuge from such disturbances. An impaired invertebrate community is more than a disruption in the aquatic food web—it indicates that stream chemistry and (or) physical habitat are impaired. Stream-chemistry data provide useful information about the stream's quality only for the time of sampling, but benthic-invertebrate communities can show the effects of short-term disturbances that can easily be missed when stream-quality assessments rely only on chemical measurements.

Differences in land use, stream habitat, and stream chemistry between the groups of sites suggest possible causes for the different invertebrate communities. The less impaired group of sites drained basins that were unmined, or where less than 10,000 ton/mi² were mined during 1980–95. Most basins in the more impaired group of sites had been mined within the last 20 years by both surface and underground methods; most contained abandoned mines that pre-dated SMCRA and produced 100,000 to 1,000,000 ton/mi² of coal. Some of the basins in the more impaired group, however, had not been mined since 1980. Coal production during 1980–95 is not an ideal indicator of the environmental disturbance caused by coal mining, but it related better to environmental measurements than did production over a shorter interval, number of abandoned mines, or mine discharge permits (Chambers and Messinger, 2001).

At the more impaired sites, the proportion of total land area as strip mines, quarries, disturbed land, or gravel pits was significantly greater than at the less impaired sites. In addition, sulfate concentration, specific conductance, and alkalinity of stream water were all higher. Stream pH did not differ significantly between the two groups; pH is regulated in mine discharges.

Two basins that were not mined since 1980 contained valley fills similar to those constructed at large surface mines. The invertebrate community in Mill

Creek near Hopewell, W. Va., which drains an area with few relatively small fills, grouped with the less impaired sites. Davis Creek at Trace Creek, W. Va., drains several large fills at a shopping center and was in the poorer group.

Instream habitat structure also differed significantly between the two groups. Sites from the less impaired group had less sand and silt in the stream bottom. Smaller median sediment size correlated with decreased number of taxa of mayflies, stoneflies, and caddisflies (EPT taxa) and an increased (more impaired) score on the Modified Hilsenhoff Biotic Index (fig. 10; $r^2 = 0.46$ and 0.43 , respectively). Among the sites sampled, correlations between invertebrate metrics and coal production (or factors relating to coal mining) were weak, largely because some streams were impaired by other land uses. Erosion and sediment deposition in basins with active mines have decreased overall because of controls required under SMCRA, but temporal comparisons are not possible. Sedimentation in 1998 remained generally greater, however, at sites in basins with coal production since 1980 than in unmined basins.

The invertebrate-community degradation represented the cumulative effects of mining before and after SMCRA, deep mining and surface mining, mines in and out of compliance with applicable regulations, and all other nonmining disturbances in the basins. Impaired sites from this region ranked near the middle of an index that ranked NAWQA sites representing different land uses throughout the United States. (See discussion of effects on invertebrate communities nationally, p. 11). Logging and ongoing construction probably contribute to sedimentation, but their extent in each basin could not be quantified. Logging may contribute more sediment per disturbed volume of soil than mining.

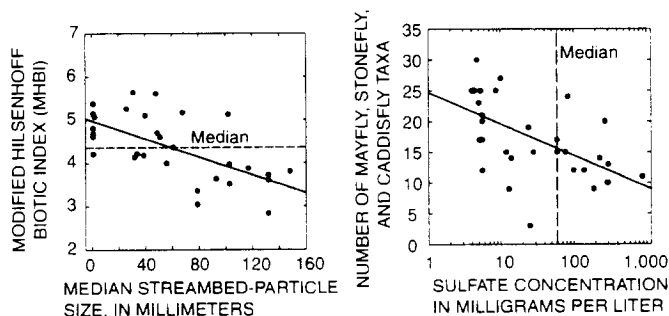


Figure 10. Invertebrate-community metrics show generally better conditions (lower MHBI) at sites with coarser streambeds and lower sulfate concentrations, although correlations are weak.

Regional study: Sulfate concentrations and biological communities in Appalachian coal fields indicated mining-related disturbances despite a general water-quality improvement between 1980 and 1998

In a 1998 study to assess regional water-quality effects of coal mining (Eychaner, 1999), samples representing the Northern Appalachian coal field were collected in the Allegheny and Monongahela River Basins (ALMN), where high-sulfur coal is common and acid mine drainage was historically severe, and samples for the Central Appalachian coal field were collected in the Kanawha–New River Basin (KANA), where acid drainage is uncommon (fig. 11).

Water chemistry in 178 Wadeable streams was analyzed once during low flow, in July and August 1998. Drainage area for most streams was between 4 and 80 mi². Most (170) of these sites were also part of a study on the effects of coal mining that was conducted during 1979–81 (Herb and others, 1981a, 1981b; 1983; Ehlike and others, 1982), before regional water quality was affected by implementation of regulations from the Surface Mining Control and Reclamation Act (SMCRA). At 61 sites, aquatic invertebrates (insects, worms, crustaceans, and mollusks) also were collected. Ground water was sampled from 58 wells near coal surface mines and 25 wells in unmined areas. Wells sampled downgradient from reclaimed surface coal mines reflect the local effects of mining.

Concentrations of Regulated Constituents Improved in Stream Base Flow From About 1980 to 1998

During low-flow conditions, sulfate in more than 70 percent of samples from streams downstream from coal mines in both coal regions exceeded the regional background concentration. Background was calculated as about 21 mg/L sulfate from data for basins with no

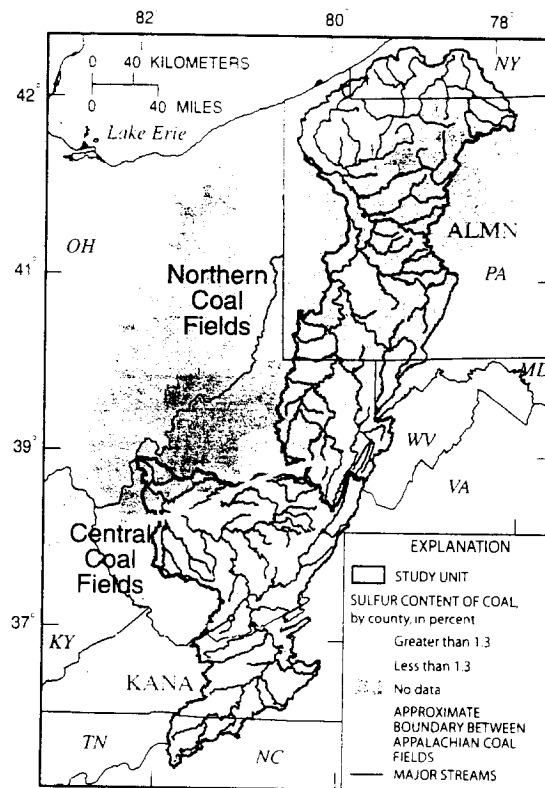


Figure 11. Coal seams in the Appalachian coal region vary in sulfur content, and the fields are identified primarily on the basis of this difference (Tully, 1996). The Kanawha–New River Basin contains mostly lower sulfur coal, while the Allegheny and Monongahela River Basins contain mostly higher sulfur coal.

history of coal mining. The highest concentrations were measured in basins with the greatest coal production. One-fourth of all samples exceeded 250 mg/L, the USEPA drinking-water guideline.

Total iron, total manganese, and total aluminum also exceeded regional background concentrations (129, 81, and 23 µg/L, respectively) in many streams in mined basins. The median concentrations of total iron in the northern coal region were about equal between mined and unmined basins, but in the central region, concentrations of median total iron among mined basins were lower than among unmined basins. In both regions, median concentrations of total manganese among mined basins were about double that among unmined basins.

Median pH increased, and median concentrations of total iron and total manganese

decreased among mined basins between 1979–81 and 1998 in both regions, reflecting that regulations restricting these constituents in mine drainage are effective. Even so, stream sites downstream from mines more commonly exceeded drinking-water guidelines for sulfate, iron, manganese, and aluminum concentrations than streams in unmined basins (fig. 12).

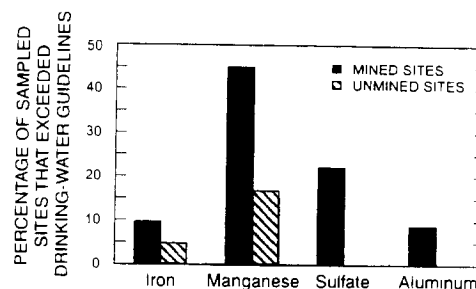


Figure 12. Stream water more often exceeded drinking-water guidelines at mined sites than at unmined sites.

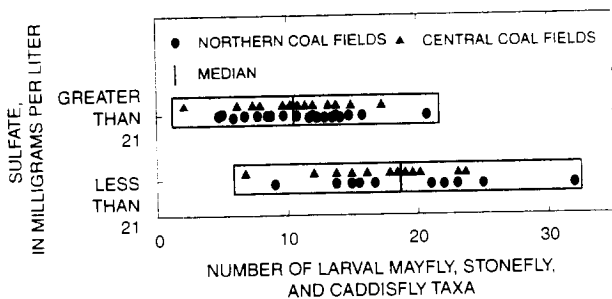


Figure 13. Sulfate concentration in stream water was inversely related to the number of mayfly, stonefly, and caddisfly taxa found at water-quality sampling sites.

Aquatic Benthic Invertebrate Communities are Impaired in Mined Basins

Aquatic invertebrate communities tended to be more impaired where there was more coal mining, when compared to basins where there was little coal mining. Pollution-tolerant species are more likely to be present at mined sites than at unmined sites, whereas pollution-sensitive taxa were fewer in number or non-existent in heavily mined basins. Increasing coal production correlated with both an increased concentration of sulfate and a decline in some aquatic insect populations (fig. 13). Of the 61 sites where aquatic invertebrates were collected, those sites with sulfate concentrations higher than the estimated background concentration had the lower diversity of three groups of sensitive insect species (mayflies, stoneflies, and caddisflies), even though the pH of the water at all sites was greater than 6.5.

At the concentrations measured, the sulfate ion is relatively non-toxic to aquatic organisms and may not represent the cause of the decline observed in mayflies and stoneflies. Sulfate concentration was, however, positively correlated with the total coal production from a basin (Sams and Beer, 2000). Other landscape disturbances associated with coal mining—changes in streamflow, siltation, or trace metal contamination—could affect the invertebrate community. Negative effects on communities caused by mining were of similar magnitude to the effects

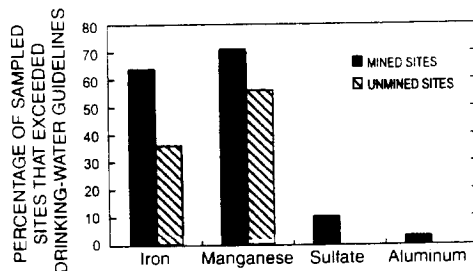


Figure 15. Ground-water samples more often exceeded drinking-water guidelines in mined areas than in unmined areas.

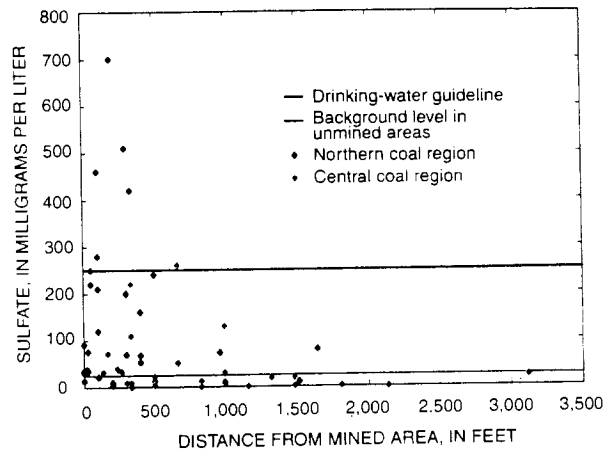


Figure 14. Sulfate concentrations in ground water are greater within 1,000 feet of reclaimed surface coal mines and in the northern coal region than at greater distance and in the central coal region.

of urban development, agriculture, large construction projects, flow alterations, or wastewater effluent.

Sulfate, Iron, and Manganese Concentrations were Elevated in Wells Near Reclaimed Surface Mines

At mined sites in both coal regions, pH was lower and sulfate concentration was greater at mined sites than at unmined sites. Sulfate concentrations in ground water were higher than background concentrations in shallow wells within 1,000 feet of reclaimed surface mines (fig. 14). Samples from wells in the northern coal region contained more sulfate than wells at unmined sites in the same region, or at any of the sites in the central coal region. Iron, manganese, and aluminum were higher than background concentrations within about 2,000 feet of reclaimed surface mines (1,800, 640, and 11 $\mu\text{g/L}$, respectively).

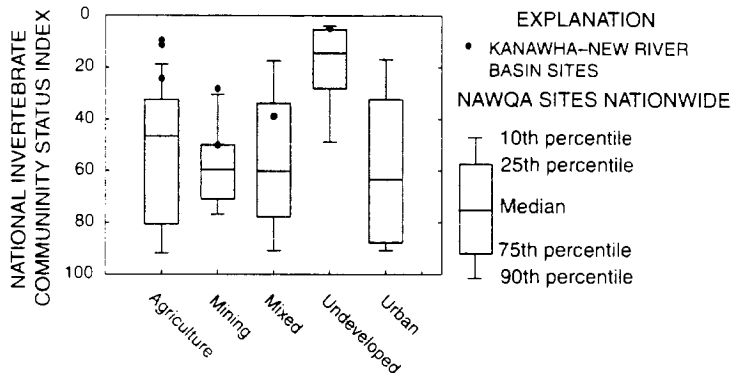
Water from most wells, except at unmined sites in the northern coal region, exceeded guidelines for iron and manganese, which make the water unpleasant to drink (fig. 15). The concentrations in both regions were higher near reclaimed mines than at unmined sites.



Effects of mining on invertebrate communities were of similar magnitude to the effects caused by urban development and agriculture nationally

Invertebrate communities at two coal mining stream sites ranked near the middle of more than 600 NAWQA sites sampled nationwide during 1991–98. These sites had index scores better than national median scores for urban sites, about the same as national median scores for agricultural sites, and worse than national median scores for undeveloped sites. The community at a forested and undeveloped site in the Appalachian Plateaus was within the best 10 percent of NAWQA sites nationally and within the best 25 percent of undeveloped sites.

Nationally, invertebrate communities at heavily agricultural sites were commonly highly impaired. In the Kanawha–New River Basin, agriculture is usually of low intensity and centers on pasturing small herds of cattle and growing cattle feed. Invertebrate communities at two agricultural sites, one in the Appalachian Plateaus and one in the Blue Ridge Physiographic Province, were within the best 10 percent of all sites nationally.



Sites in undeveloped and agricultural basins in the Kanawha–New River Basin rank among the best sites nationally in the National Invertebrate Community Status Index. More impaired sites in the Kanawha–New River Basin rank about the same or better than most sites that represent developed land uses nationally. (Low scores correspond to diverse invertebrate communities.)

Some Contaminants are Widespread and Present at Potentially Harmful Concentrations in Streambed Sediment and Fish Tissue

Ten Polycyclic Aromatic Hydrocarbons were Found in Streambed Sediments in Concentrations that may Harm Aquatic Life

Forty samples of streambed sediment from 36 sites in the Kanawha–New River Basin were analyzed for polycyclic aromatic hydrocarbons (PAHs) during 1996–98. PAHs are components of wood smoke, diesel exhaust, soot, petroleum, and coal. Their toxicity varies, and some are carcinogenic to humans and other animals. Of the 12 PAHs for which guidelines were available, 10 were detected at concentrations exceeding the Probable Effect Level (PEL; see information box on sediment-quality guidelines), and all were detected at concentrations exceeding the Threshold Effect Level (TEL).

High concentrations of PAHs were present in each physiographic setting in the basin except for the Blue Ridge, although the only high concentrations in the Valley and Ridge/Appalachian Plateaus transition zone were in basins where coal has been mined. The highest

Sediment Quality Guidelines

NAWQA's bed-sediment sampling protocol (Shelton and Capel, 1994) is designed to maximize the chance of detecting contaminants that have been transported in a stream during the previous 1–3 years. The data from this study were compared to final Canadian Sediment Quality Guidelines (SQGs) rather than the preliminary USEPA guidelines. SQGs have been issued by Environment Canada for 8 trace elements and 12 PAHs (Canadian Council of Ministers of the Environment, 1999). At concentrations below a Threshold Effect Level (TEL), contaminants are rarely expected to have a toxic effect on aquatic life. At concentrations above a Probable Effect Level (PEL), toxic effects are expected frequently. Concentrations of substances that exceed SQGs may imply, but not prove, that organisms in the streams of interest are at risk from those substances.

PAH concentrations measured in this study were in the Appalachian Plateaus. Some of the highest PAH concentrations were measured at some of the most heavily mined sites in the basin, although the correlation between coal production and streambed PAH con-

centration was weak ($r^2 = 0.52$, among 20 wadeable stream sites within the coal region). Coal samples from several commonly mined seams in West Virginia were between 20 and 85 percent PAH by mass (W.H. Orem, U.S. Geological Survey, written commun., July 2000). Coal particles are common in sediment from many streams in the coal fields. The PAHs from the coal particles, however, may not be bioavailable (Chapman and others, 1996). Unlike other NAWQA study areas, no correlation was found between most other land uses and PAH concentration.

Four Trace Elements were Present in Streambed Sediment in Concentrations That May Harm Aquatic Life

A total of 53 bed-sediment samples from 47 sites in the Kanawha–New River Basin were analyzed for trace elements during 1996–98. All eight of the trace elements for which criteria were available were found at some sites in concentrations exceeding their Threshold Effect Level (fig. 16; see information box on sediment-quality guidelines). Nickel, chromium, zinc, and lead were detected at concentrations exceeding their Probable Effect Level. Nickel concentrations exceeded the Probable Effect Level most frequently (in 47 of the 53 samples), based on the 1995 Sediment Quality Guidelines; a final SQG was not issued for nickel at the time that other SQGs were finalized.

Trace-element concentrations also were determined in livers of common carp or rock bass in 27 samples from 18 sites in 1996 and 1997. Some samples contained concentrations of arsenic, cadmium, lead, mercury, nickel, selenium, and zinc that were among the highest 25 percent of more than 900 NAWQA samples nationwide (1991–98). Concentrations of cadmium, mercury, nickel, selenium, and zinc in fish-tissue samples from the Kanawha–New River Basin ranked among the highest 10 percent of all NAWQA samples; six samples contained cadmium concentrations ranking among the highest 10 percent of all NAWQA samples, and five samples contained selenium concentrations ranking among the highest 10 percent of all NAWQA samples. One fish-tissue sample, from Kanawha River at Winfield, contained cadmium at a concentration ranking in the highest 1 percent of all samples in the

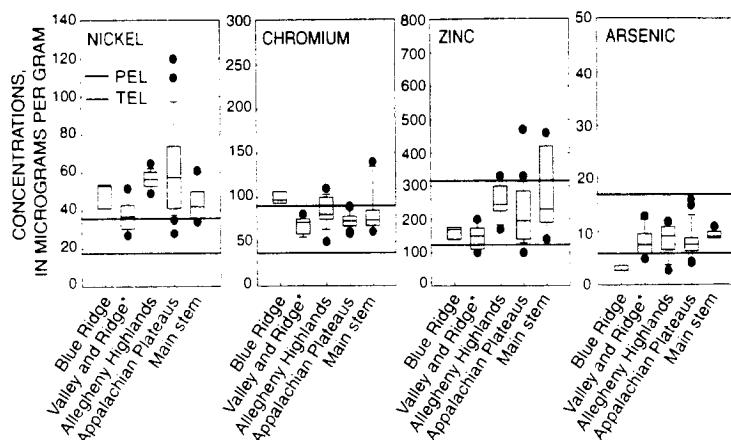


Figure 16. Some trace element concentrations in stream-bed sediment exceeded Environment Canada's effects-based criteria at several sites in the basin. Probable effects levels (PEL) are those concentrations at which harmful effects to aquatic life are thought to be likely, and were exceeded most frequently in the Allegheny Highlands and other Appalachian Plateaus streams. Threshold effects levels (TEL) were exceeded at all sites by nickel and chromium. *Valley and Ridge sites include transition zones between provinces.

Nation. Determining the human health or ecological significance of these concentrations is problematic, because tissue samples were collected from many different species and because fish-liver tissue is not normally eaten by humans.

Fish Communities Differ Considerably Throughout the Basin, but Non-native Species Continue to Expand Their Range

Fish communities in the Kanawha–New River Basin are complex and vary widely among streams of different size, physiographic setting, and land use. Individual species are distributed in patches, particularly upstream from Kanawha Falls (Jenkins and Burkhead, 1994). This patchy distribution can confound comparisons among streams (Strange, 1999). The quality of the regional fish community is generally good, although the national NAWQA fish index seems to underrate that quality because it does not consider the patchy distribution.

Non-native Fish Continue to Expand Their Range in Tributaries of the New and Gauley Rivers

Three fish species were collected for the first time at often-sampled sites in tributaries of the New and Gauley Rivers (Cincotta and others, 1999). Margined madtoms, a popular bait species, were collected for the

first time from Second Creek near the village of Second Creek. Margined madtoms are native to some parts of the New River and some of its tributaries, but they had never before been collected from the Greenbrier River Subbasin. Telescope shiners (fig. 17), natives of the Tennessee River Basin, have been collected in the New River since 1958, and they continue to expand their range. Telescope shiners were collected from another often-sampled site, Williams River at Dyer, in the Gauley River Subbasin; this was their first collection upstream from Summersville Dam, a large impoundment. Telescope shiners also were collected for the first time from two Meadow River tributaries, also in the Gauley River Subbasin. Least brook lamprey were collected for the first time from Williams River at Dyer, their second collection from the Gauley River Subbasin. Populations of all these species were well established, and the ongoing expansion of their ranges suggests that all were relatively recent bait-bucket introductions to the New River system. Two of these reaches, and all of these streams, had been thoroughly sampled in the late 1970s (Hocutt and others, 1978, 1979).



Figure 17. Example of a telescope shiner (*Notropis telescopus*), a non-native species in the Kanawha–New River Basin. (Photograph from Jenkins and Burkhead, 1994; used by permission from the Virginia Department of Game and Inland Fisheries)

Other fish collected for the first time in the basin were in tributaries of the Coal River. The new species in Coal River distribution records were from large tributaries where few or no surveys had been made since the 1930s. Mottled sculpin, bluebreast darter, river carpsucker, blacknose dace, and longnose dace all were collected for the first time from Clear Fork near Whitesville or Spruce Laurel Fork at Clothier, major tributaries to the Big or Little Coal Rivers, respectively. Several of these records represented the most upstream collections in their respective forks of the Coal River, although all had been collected from the Coal River Subbasin. These new-species records most likely represent undersampling of streams that have often been

overlooked by investigators rather than new range expansions.

In some regions of the United States, the highest proportion of non-native fish are typically present in the most impaired streams (Maret, 1997; Waite and Carpenter, 2000). In these regions, unimpaired streams are typically cold-water streams with complex physical habitat and low nutrient concentrations. In impaired streams where agricultural and urban land uses are common, stream temperature and nutrient concentrations are high and physical habitat is degraded. Many non-native fish tolerate these conditions better than many native species do, enabling the non-natives to displace the natives. No such relation was found in the Kanawha–New River Basin, where sedimentation and increased dissolved solids have impaired streams, but where temperature and nutrient concentrations have remained low (Messinger and Chambers, 2001, in press). The proportion of introduced fish in the New River system was high, even though other measures did not indicate impairment.

Fish Species Common Throughout the Ohio River Basin are Not Native Upstream from Kanawha Falls

The New River system, which fisheries biologists consider to include the Gauley River and its tributaries, supports a different collection of fish species than the downstream Kanawha River system, which is part of the larger Ohio River system (Jenkins and Burkhead, 1994). Kanawha Falls (see front cover), a 24-foot waterfall 2 miles downstream from the confluence of the New and Gauley Rivers, is the boundary between the New River and Kanawha River systems. This waterfall has been a barrier to upstream fish movement since glaciers affected streams more than 1 million years ago. The New River system lacks native species diversity, and it has unfilled ecological niches. It has only 46 native fishes and the lowest ratio of native fishes to drainage area of any river system in the Eastern United States.

The lack of native-species diversity allowed other species to develop in the New River system, which has the largest proportion of endemic species (found nowhere else in the world) in eastern North America (8 of 46). Introduced fish species have prospered in the New River system: Jenkins and Burkhead (1994) cite the New River system as having the largest number and proportion (42 of 89) of introduced freshwater species

of all major eastern and central North American drainages.

Although many species have been introduced and become naturalized throughout the 19th and 20th centuries, the New River fish fauna remain susceptible to invasion. In contrast, 118 fish species are reported from the Kanawha River system downstream from Kanawha Falls (Stauffer and others, 1995); none of these fish species are endemic to the Kanawha River system, and only 15 are considered possible, probable, or known introductions.

Fish Communities are Controlled By a Variety of Environmental Factors in the Kanawha–New River Basin

In testing the possible effects of coal mining on fish communities, results were less definitive than for benthic invertebrates (p. 8–9). No common fish metrics (Karr and Chu, 1999; Barbour and others, 1999) correlated closely with mining intensity or its surrogate, sulfate concentration. The study included sites both upstream and downstream from Kanawha Falls, and differences in many metrics between the two groups mask differences among land-use categories (Messinger and Chambers, 2001, in press). However, fish were collected at only 13 wadeable sites in the coal region, which did not represent a full gradient of mining intensity.

High Concentrations of Fecal Bacteria Remain in Streams if Sources are Close

Concentrations of *Escherichia coli* (*E. coli*) exceeded the national guideline for public swimming areas in 26 percent of samples from major rivers in the Kanawha–New River Basin and in 43 percent of samples from tributary streams (fig. 18); however, no outbreak of water-borne disease was reported from the basin during 1991–98 (Barwick and others, 2000). Bacteria concentration

in stream water varies widely, reflecting the changing balance between bacterial sources and many factors that help or hinder bacteria transport. Because of the wide variability, comparisons between streams based on only a few samples can be misleading; a few generalizations, however, can be made.

First, streams contain more bacteria if the sources are close to the stream and the sampling site. Among large rivers, median concentrations of *E. coli* were lowest in the New River Gorge at Thurmond, in a reach distant from any large city (fig. 18). Concentrations were highest in the Kanawha River downstream from the Charleston metropolitan area at Winfield. In the two tributary basins with the highest median concentrations, most homes are clustered close to the streams because the land slopes steeply elsewhere. In contrast, four tributary streams in basins with more moderate slopes, where bacteria sources are more dispersed, had median *E. coli* concentrations less than half as high. Regardless of slope, direct contamination of a stream by sewage or manure can produce extremely high concentrations, as Gillies and others (1998) observed in the Greenbrier River.

Second, bacteria concentrations exceeding guidelines are much more common when streamflow is greater than average, so streams generally contain more bacteria in winter than in summer (fig. 19). *E. coli* concentrations exceeded guidelines in less than one-third of summer samples from moderate-slope tributaries and less than one-fifth from large rivers. In the three

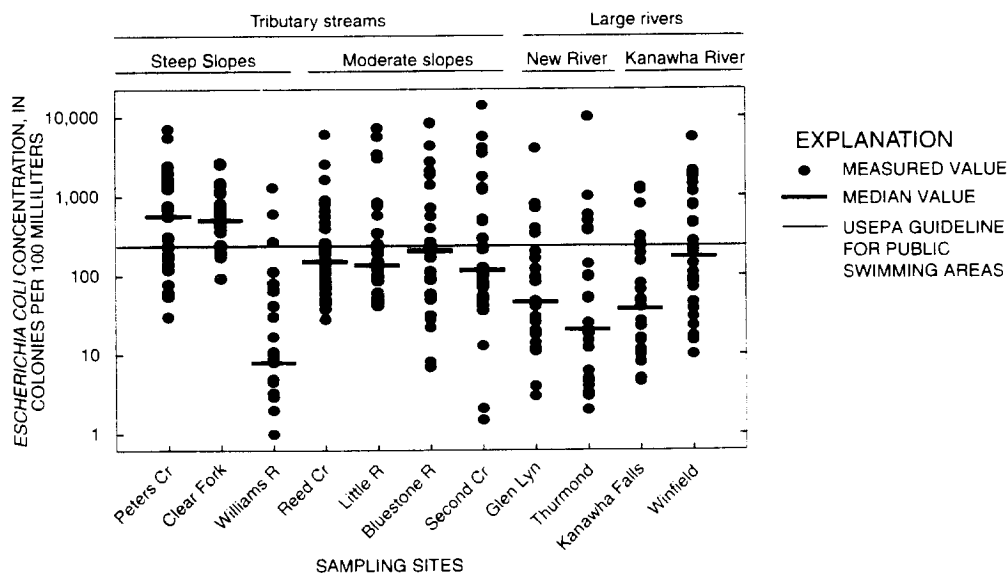


Figure 18. *E. coli* bacteria concentrations in streams vary widely.

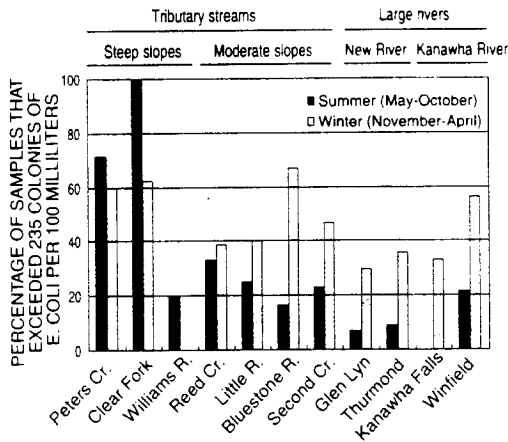


Figure 19. Guidelines for *E. coli* are exceeded more often in winter than in summer for most streams.

tributary basins with steeper slope, however, concentrations were higher in summer than winter.

Finally, streams contain more bacteria if the bacteria sources are large. Williams River, the tributary basin with the lowest median concentration of *E. coli* (fig. 18) is home to only 5 people per square mile, compared to the average of 71 people per square mile throughout the entire Kanawha–New River Basin. For twice the population density, median *E. coli* was about 300 percent higher among steep-slope tributaries. Among the moderate-slope basins, however, including the Blue-stone River Basin with 201 people per square mile, median *E. coli* was only about 10 percent higher for twice the population density. Neither the estimated number of cattle nor the percentage of agricultural land use in the tributary basins showed a relation to the median bacteria concentrations.

Facts about *E. coli*

Escherichia coli (*E. coli*) is a bacterium that grows in the intestines of people, other mammals, and birds. Most strains of *E. coli* do not cause disease, but they do indicate water contamination by feces, which could contain other disease-causing organisms. The national guideline for public swimming areas is less than 235 *E. coli* colonies per 100 milliliters of water (col/100 mL) in any single sample (U.S. Environmental Protection Agency, 1986). That level is intended to allow no more than 8 gastrointestinal illnesses per 1,000 swimmers. For waters infrequently used for full-body-contact recreation, the guideline is 576 col/100 mL.

Nutrient and Organic-Chemical Concentrations in Surface Water are Low in Most of the Basin

Nutrients were Detected at Low Concentrations in Streams of the Kanawha–New River Basin

Mean concentrations of nutrients in the Kanawha–New River Basin were at or below national background levels. Most concentrations, however, exceed those measured at a stream-water-monitoring site at Williams River, which drains mostly National forest. The highest mean nitrate concentration measured was 1.5 mg/L. Flow-weighted mean ammonia concentrations ranged from less than 0.02 to 0.04 mg/L. Mean total phosphorus concentration was less than 0.1 mg/L at nine sites; the maximum was 0.15 mg/L. Nitrate and phosphorus are typically increased by agricultural or urban land uses, and certain nutrients, such as ammonia, can accumulate from natural sources.

Differences in nutrient concentrations were found among sites because of differences in land use/land cover, and physiography. Generally, basins with more agriculture produced more mean total nitrogen than did forested basins. The lowest mean total nitrogen concentration in streams, 0.71 mg/L was that for mostly forested tributary basins in the Appalachian Plateaus produced (fig. 20). The lowest mean concentration in the basin, or background concentration, was 0.45 mg/L, at Williams River. Tributary streams with basins mostly or wholly within the Valley and Ridge Physiographic Province had the highest mean total nitrogen, 1.04 mg/L. One stream in the Blue Ridge had a mean total nitrogen concentration of 0.94 mg/L. The mean total nitrogen concentration was not substantially different between large rivers and smaller tributaries (0.83 and 0.90 mg/L respectively).

Four sites, draining forest mixed with agriculture or coal mining, ranked among the best sites in the Nation in a national Algal Status Index. This index measures the proportion of algal samples that belong to species that are tolerant of high nutrient concentrations and siltation.

Pesticides were Detected at Low Concentrations in Surface Water

Pesticides were sampled for 9 to 25 times at four sites in 1997. Two sites were on main-stem, large streams. The other two sites on tributary streams drained basins with more than 30 percent agricultural



Figure 20. Because much of the Kanawha–New River Basin is forested, surface water and ground water contain low concentrations of nutrients and few pesticides.

land and some urban land. (See Study Unit Design, p. 20). Time of sampling covered the seasonal spectrum of both climate and pesticide application. The pesticides detected at all sites are routinely detected at agricultural sites across the Nation.

Surface-water samples in the Kanawha–New River Basin contained only a few pesticides at low levels. In all, 23 of 83 pesticides analyzed for were detected (Ward and others, 1998). All pesticide detections were less than 1 $\mu\text{g/L}$; concentrations detected did not exceed USEPA drinking-water standards or aquatic-life criteria. The most commonly detected pesticides were atrazine, deethylatrazine (a breakdown product of atrazine), metolachlor, prometon, simazine, and tebuthiuron. Atrazine, deethylatrazine, metolachlor and simazine were detected in more than 90 percent of samples.

Dioxin is a particularly toxic contaminant in certain herbicides formerly manufactured near Charleston and is a known contaminant in the lower Kanawha River, but it was not analyzed for this study. Dioxin in the lower Kanawha River is the target of ongoing regulatory investigations by USEPA and other agencies.

Many VOCs Detected in the Lower Kanawha River

Numerous volatile organic compounds (VOCs) have been detected routinely at low concentrations in the Kanawha River downstream from the Charleston metropolitan area (Tennant and others, 1992). In this study, more than 20 VOCs were detected, at concentrations ranging from 0.015 to 0.3 $\mu\text{g/L}$, in each of two samples collected in late 1997 from the Kanawha River at Winfield. Each sample was analyzed for 85 compounds (Ward and others, 1998). The compounds detected at

Winfield, downstream from Charleston, included chloroform, motor fuel and aromatic compounds such as benzene, and industrial compounds such as ethers. In contrast, only a single compound was detected in one of two samples collected from the Kanawha River upstream at Kanawha Falls.

During 1987–96, one or more of 21 VOCs were detected in 50 percent of all daily samples collected for the Ohio River Valley Water Sanitation Commission (ORSANCO) from an industrial water intake at St. Albans, downstream from Charleston (Lundgren and Lopes, 1999). Benzene and toluene were the two most frequently detected compounds, and a maximum of 11 compounds was detected in a single sample. Median concentrations ranged from 0.1 to 2.3 $\mu\text{g/L}$. Gasoline spills or leaks of as little as 10 gallons per day that reach the river could produce the concentrations measured at St. Albans.

Radon Concentrations and Bacterial Contamination are the Principal Ground-Water-Quality Concerns

Physiographic Province, Geology, Well Construction, and Land Use Affect the Quality of Water from Domestic Wells

Ground water from private wells provides domestic supply for 30 percent of the people in the Kanawha–New River Basin. High concentrations of radon are a concern in the Blue Ridge (p. 18), and private wells can be contaminated by fecal bacteria throughout the basin (p. 19), but the occurrence of other contaminants differs among the physiographic provinces.

APPALACHIAN PLATEAUS PHYSIOGRAPHIC PROVINCE

In the layered sedimentary rocks of the Appalachian Plateaus, ground water moves mostly in a network of narrow fractures within a few hundred feet of the land surface (Wyrick and Borchers, 1981; Harlow and LeCain, 1993). Individual fractures typically connect to only a few others, and a well normally taps only a few of the many fractures nearby. Recharge comes from rain and melting snow. Ground water flows generally toward the nearest stream, forming local aquifers bounded by the ridgetops. Contamination of a local aquifer and its stream is most likely to come from local sources.

Water samples were collected from 30 newer domestic wells or similar-capacity public-supply wells throughout the Appalachian Plateaus (Sheets and Kozar, 2000) and from 28 generally older domestic wells close to surface coal mines where reclamation was completed between 1986 and 1996. Wells near active mines were not sampled. Most of the wells were between 40 and 200 feet deep, and most water levels were between 10 and 90 feet below land surface.

Concentrations of iron and manganese exceeded USEPA drinking-water guidelines in 40 and 57 percent, respectively, of the wells throughout the Appalachian Plateaus and in about 70 percent of wells near reclaimed mines. Water that exceeds these guidelines is unpleasant to drink and can stain laundry and plumbing fixtures, but it is not a health hazard.

Potentially hazardous concentrations of methane, an odorless component of natural gas that is often associated with coal seams, were detected in water at 7 percent of the wells. At concentrations greater than about 10 mg/L, methane can bubble out of water pumped from a well. If enough gas collects in a confined space, an explosion is possible. In the West Virginia coal fields, any well water that bubbles is a potential methane explosion hazard.

Other chemical analyses of ground water samples collected as part of this study showed the following water-quality characteristics and conditions. Water from 61 percent of the wells near reclaimed mines was slightly acidic (pH less than 6.5) and could leach lead or copper from water pipes in homes. Only 23 percent of other Appalachian Plateaus wells produced acidic water. Radon exceeded the proposed USEPA standard at half the wells throughout the Appalachian Plateaus (p. 18). Water from half the wells exceeded 20 mg/L of sodium, the upper limit that USEPA suggests for peo-

ple on a sodium-restricted diet. Arsenic in water from 7 percent of the wells exceeded the 10- $\mu\text{g/L}$ standard set in January 2001, but none exceeded the previous 50- $\mu\text{g/L}$ standard. Concentrations of radon, sodium, and arsenic were lower in wells near reclaimed mines than in wells remote from reclaimed mines. Home water-treatment techniques can remove lead, copper, sodium, and arsenic from drinking water.

BLUE RIDGE PHYSIOGRAPHIC PROVINCE

In the igneous and metamorphic bedrock of the Blue Ridge, as in the Appalachian Plateaus, ground water moves in a network of shallow fractures. Local aquifers generally drain toward the nearest stream (Coble and others, 1985).

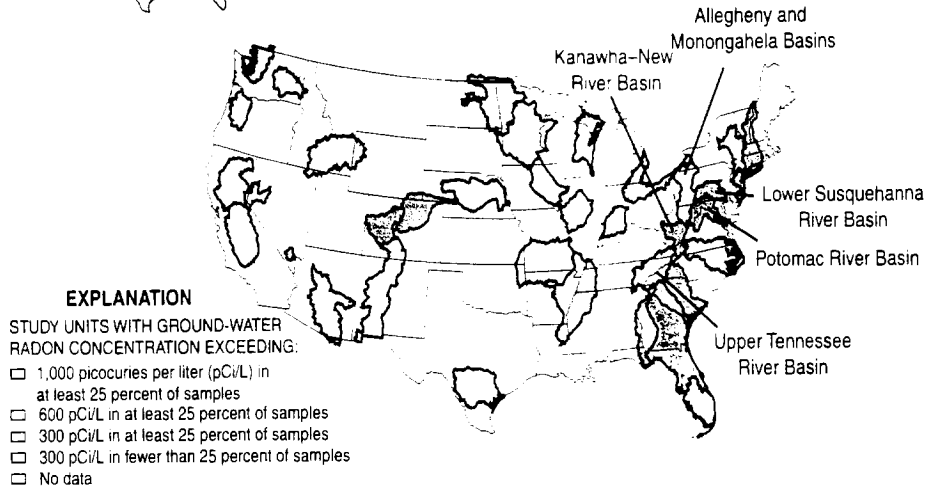
Water samples were collected from 30 newer domestic wells or similar low-capacity public-supply wells throughout the Blue Ridge. Most of the wells were between 100 and 350 feet deep, and most water levels were between 10 and 70 feet below land surface.

Ground water in the Blue Ridge is susceptible to contamination. Chlorofluorocarbon concentrations showed that the water in 89 percent of the wells had been recharged within the previous 20 years, indicating that contaminants could be transmitted readily into the fractured rock aquifers (Kozar and others, 2001).

Chemical analyses of ground water samples collected as part of this study indicated that concentrations of radon were among the highest in the Nation (p. 18); iron and manganese concentrations exceeded guidelines at only 17 percent of the wells; sodium exceeded 20 mg/L at 3 percent of the wells; and arsenic did not exceed 1 $\mu\text{g/L}$ at any of the sites. Pesticides were detected at 57 percent of the wells. The presence of the common agricultural herbicide atrazine in ground water, even in low concentrations, shows that potential contaminants could move quickly from the land surface into the drinking-water aquifer.

Valley and Ridge Physiographic Province ground-water conditions can be inferred from studies in similar settings in the Potomac River Basin, which was one of the 1991 NAWQA study units. See Lindsey and Ator, 1996 and Ator and others, 1998 for more details.

Radon concentrations in ground water were among the highest in the Nation



Radon is a radioactive gas that forms during the decay of natural uranium. Igneous and metamorphic rocks, like those in the Blue Ridge, commonly contain more uranium than other rock types. Radon in the air in homes is the second leading cause of lung cancer; and radon causes 2–3 percent of all cancer deaths in the United States. Homes can be designed or remodeled to

remove radon from both drinking water and interior air. The only way to determine if an individual well or home exceeds standards, however, is to have the water or air tested. Information on radon testing and removal is available at <http://www.epa.gov/safewater/radon/qa1.html> and other Web sites.

Radon concentration exceeds 1,000 pCi/L (picocuries per liter) in at least 25 percent of ground-water samples collected in many areas of the Eastern United States. In the Kanawha–New River Basin, 30 percent of samples exceeded 1,000 pCi/L (Appendix, p. 27), making the basin comparable to the Potomac and Lower Susquehanna River Basins to the northeast. Within the basin, however, radon in two-thirds of samples from wells in the Blue Ridge exceeded 1,000 pCi/L, but only in 10 percent of samples from the Appalachian Plateaus. The northern part of the basin, therefore, is more comparable to the adjacent Allegheny and Monongahela Rivers and Upper Tennessee River Basins.

Ground-water Radon Concentrations were Highest in the Blue Ridge

Radon concentrations were greater than 300 pCi/L, the proposed drinking-water standard (U.S. Environmental Protection Agency, 1999), in 87 percent of wells sampled in the Blue Ridge (fig. 21). The maximum concentration detected was 30,900 pCi/L (Kozar and Sheets, 1997). Of the 30 wells sampled, 10 contained concentrations of radon greater than 4,000 pCi/L, the alternate standard USEPA has proposed for regions where action is taken to decrease airborne radon. As water is used in a home, radon in the water can lead to an increase in radon in the air, which is the major exposure path for people.

Radon concentrations exceeded 300 pCi/L at 50 percent of wells sampled throughout the Appalachian Plateaus. The maximum in any sample was 2,500 pCi/L (fig. 21). The area is underlain primarily by sandstone, shale, coal, and limestone sedimentary rocks, in which uranium is less common than in igneous and metamorphic rocks.

At 28 wells downgradient from recently reclaimed surface coal mines, the median radon concentration was just 115 pCi/L, and the maximum was 450 pCi/L.

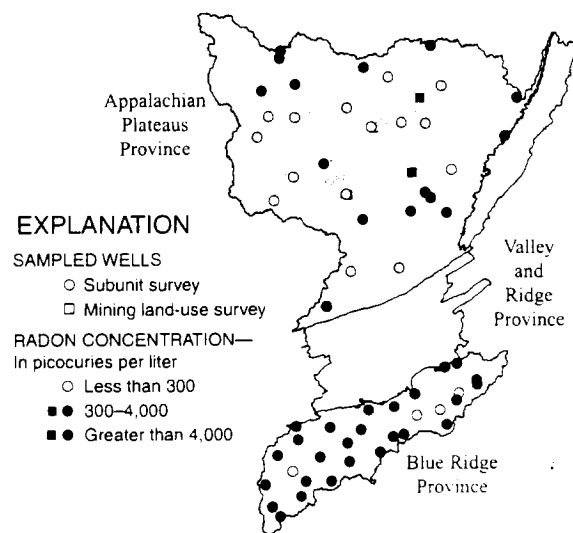


Figure 21. Radon concentrations vary greatly among physiographic provinces.

In comparison, at 15 wells in the same geologic units but not near mines, the median concentration was 200 pCi/L.

Modern Well Construction Can Prevent Fecal Bacteria from Reaching Drinking Water in Most Areas

Escherichia coli (*E. coli*) and the broader fecal coliform group of bacteria indicate the possible presence of disease-causing organisms. Standards for public drinking-water supplies do not permit the presence of any of these bacteria at detectable levels. Septic systems or livestock near a well are the probable sources of bacteria throughout the basin. Proper well construction can prevent bacteria from reaching the well water in some settings, and drinking water can be disinfected with chemicals or ultraviolet light.

Water from wells less than 25 years old in the Appalachian Plateaus and Blue Ridge was generally free from fecal bacteria (table 2). The sampled wells were generally in good condition, with a section of solid pipe at the top of the well sealed with concrete into the soil and rock (Sheets and Kozar, 1997). A residential septic system typically was nearby, but no heavy livestock use was within several hundred yards. Bacteria were found, however, at one fourth of the wells in a second study in the Appalachian Plateaus, which included some older wells and some without seals. Near these wells, there also may have been bacteria sources other than a septic system.

Table 2. *E. coli* or other fecal coliform bacteria were detected in few modern wells

Setting	Percentage of wells where bacteria were detected
Appalachian Plateaus:	
Newer wells	3
Older wells	26
Blue Ridge (newer wells only)	0

Most wells in limestone aquifers in the basin, including the Valley and Ridge, are at risk of contamination by bacteria (Boyer and Pasquarell, 1999), even if septic systems or livestock wastes are not nearby (Mathes, 2000), because ground water moves rapidly through solution channels in the rock. The wide valleys that typically overlie limestone aquifers are heavily used for livestock and agriculture.

Volatile Organic Compounds and Pesticides in Ground Water were Found in Low Concentrations

Both volatile organic compounds (VOCs) and pesticides were detected at low concentrations in the ground water of the Kanawha–New River Basin (Appendix, p. 27). Thirteen percent of samples (9 of 60) contained VOC concentrations greater than 0.1 µg/L. Of the seven detected VOCs, however, only three have established drinking-water standards. None of the VOCs identified in samples exceeded these standards. Pesticides were found above a detection limit of 0.001 µg/L in 32 percent of samples (19 of 60). Of the 12 detected pesticides, 4 have established drinking-water standards, none of which was exceeded.

Pesticides were detected in 17 of 30 wells sampled in the Blue Ridge, where 30 percent of the land was being used for agriculture in 1993. The most commonly detected pesticides, at one-third of the wells, were atrazine and its breakdown product deethylatrazine. The maximum concentration of all pesticides detected in a single sample was 0.14 µg/L. Two other pesticides, *p,p'*-DDE and simazine, were present in more than 10 percent of samples at a maximum concentration of 0.025 µg/L in this province. In the largely non agricultural Appalachian Plateaus, however, pesticides were detected only at two wells.

Nutrient Concentrations in Ground Water were At or Below National Background Levels

Nutrients were prevalent at relatively low concentrations in ground water of the Kanawha–New River Basin. Nitrate concentration in 1 of 88 wells sampled in this study exceeded the USEPA drinking-water standard of 10 mg/L (as nitrogen). Most ground water contained less nitrate than does precipitation in the basin. Concentrations of other nutrients measured were at or below national background levels. These findings are consistent with national findings on nutrients in the ground water of forested areas, and the Kanawha–New River Basin is about 80 percent forested.

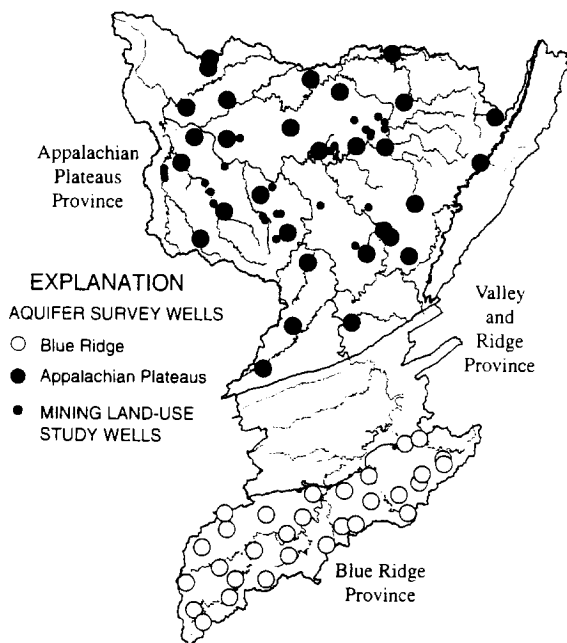
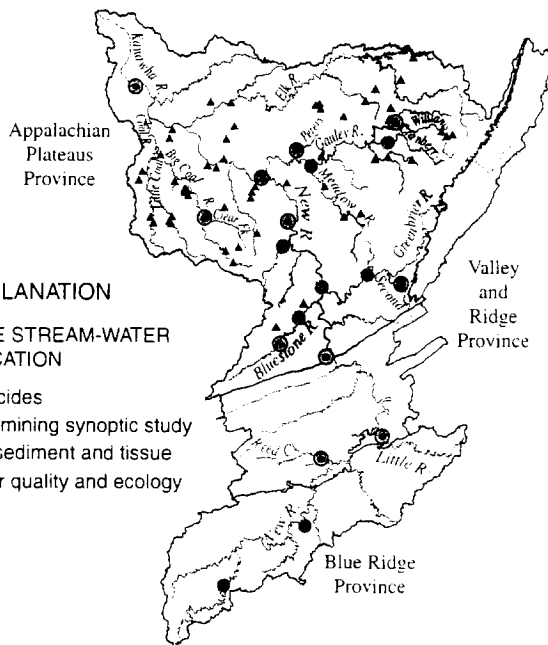
In the water of Appalachian Plateaus wells, the relatively high median ammonia concentration for a forested region—0.16 mg/L— is probably a result of mineralization of organic material. In contrast, ground water in the Blue Ridge, where a greater percentage of land is used for agriculture, had ground water with a higher median nitrate concentration (0.42 mg/L) and a higher median dissolved-oxygen concentration (5.1 mg/L).

STUDY UNIT DESIGN

Studies in the Kanawha–New River Basin were designed to describe the general quality of water and the aquatic ecosystem and to relate these conditions to natural and human influences (Gilliom and others, 1995). The design focused on the principal environmental settings—combinations of geohydrology, physiography, and land use—throughout the basin. The studies supplement assessment work by State agencies (Virginia Department of Environmental Quality, 1998; North Carolina Department of Environment and Natural Resources, 1999; West Virginia Division of Environmental Protection, 2000).

Stream Chemistry and Ecology

The sampling network was designed to characterize the effects of land use on stream quality at various scales. Water chemistry, fish and invertebrate communities, habitat, and bed-sediment and fish-tissue chemistry were used as indicators of stream quality. Fixed Sites were chosen on large rivers at the boundary between the Valley and Ridge and Appalachian Plateaus Physiographic Provinces, downstream from the Greenbrier and Gauley Rivers, and near the mouth of the Kanawha River. Fixed Sites also were chosen on tributaries to represent the effects of agriculture, coal mining, forest, and a relatively large human population in an otherwise rural setting.



Ground-Water Quality

The ground-water network was designed to broadly characterize the resource. Little previous information was available in the aquifer-survey areas. Aquifer surveys examined more constituents than any previous study and included a random component in site selection that allows estimates to be made for the whole population of similar wells. The land-use study targeted current effects of mining reclamation standards that have developed since around 1980.

Study component (Type of site)	What data were collected and why	Types of sites sampled	Number of sites	Sampling frequency and period
STREAM CHEMISTRY AND ECOLOGY				
Fixed sites— General quality of the water column	Concentration, seasonal variability, and load of major ions, common metals, nutrients, bacteria, organic carbon, dissolved oxygen, suspended sediment, pH, specific conductance, and temperature. Continuous streamflow monitoring.	Large rivers with mixed land use, draining 3,700 to 11,800 square miles at sites located between major tributaries or at boundaries of regional environmental settings. Tributary streams draining 40 to 300 square miles in basins with predominant land uses of agriculture, coal mining, forest, and rural residential.		Monthly plus storms: about 30 samples during October 1996 through September 1998.
Fixed sites— Dissolved pesticides	Concentration and seasonal variability of 86 organic compounds in addition to the general water-column constituents listed above.	One large river downstream from the Valley and Ridge Physiographic Province and one near the mouth of the Kanawha River. Tributary streams with extensive agricultural land use.		Semimonthly to monthly: 14 or 15 samples in 1997. Weekly to monthly, during 1997: 9 or 25 samples.
Fixed sites— General stream ecology and habitat	Fish, benthic invertebrate, and algae communities were sampled and physical habitat was described to determine the presence and community structure of aquatic species.	Fixed sites where general water-column samples were collected.	11	Once, in 1997; three reaches sampled at each of three tributary sites in 1998.
Contaminants in fish tissue	To determine the presence of potentially toxic compounds in food chains that can include humans. Data included 22 elements and 28 organic compounds. Samples were a composite of at least five fish from one species, usually rock bass or common carp.	Fixed sites where general water-column samples were collected, plus contrasting settings in three large basins with mixed land use and five tributaries.	19	1 or 2 samples per site and species, during 1996 or 1997: 27 total samples.
Contaminants in bed sediment	To determine the presence of potentially toxic compounds attached to sediments accessible to aquatic life. Data included 44 elements and more than 100 organic compounds.	Same as sites for contaminants in fish. Composite samples were collected from depositional zones, where fine-grained sediments transported within the past year settle out of the water.	19	One or 2 samples during 1996 or 1997: 21 total samples.
Synoptic sites— Coal mining	To assess the present effects of coal mining in Appalachian Plateaus streams and the change in stream chemistry since about 1980. Data included discharge, alkalinity, acidity, pH, specific conductance, sulfate, chloride, and dissolved and total iron, manganese, and aluminum. Coordinated with a similar study in the Allegheny-Monongahela study unit. Benthic invertebrate community, physical habitat, contaminants in bed sediment, and other major ions in addition to constituents listed above. Fish community, in addition to constituents listed above.	Streams draining 0.2 to 128 square miles in areas of known mining history, including unmined basins. Most of the sites were sampled for water-column chemistry during 1979-81. A subset of sites described above, draining 8.8 to 128 mi ² . A subset of benthic invertebrate sites.	19, including unmined sites	One sample during low flow, July 1998.
GROUND-WATER				
Aquifer Surveys— Blue Ridge and Appalachian Plateaus	General water quality, to determine the occurrence and distribution of contaminants. Data included major ions, nutrients, bacteria, organic carbon, 19 trace elements, 47 pesticides, 86 volatile organic compounds, dissolved oxygen, turbidity, pH, specific conductance, and temperature. Samples from the Blue Ridge were analyzed for an additional 39 pesticides.	Domestic and public supply wells 25 years old and younger, and in good condition.	60	Once in 1997.
Land-use effects, reclaimed surface coal mines	General water quality, to determine effects of present reclamation requirements. Data included the constituents from aquifer surveys, without pesticides or volatile organic compounds. Coordinated with a similar study in the Allegheny-Monongahela Study Unit.	Domestic wells within 3,100 feet downgradient from a fully reclaimed surface coal mine. Reclamation was complete between 2 and 12 years before sampling. None of the sites were near "mountaintop removal" mines. Included both old and new wells.	28, compared to 10 unmined aquifer survey sites.	Once in 1998.

GLOSSARY

- Aquatic-life criteria**—Water-quality guidelines for protection of aquatic life. Often refers to U.S. Environmental Protection Agency water-quality criteria for protection of aquatic organisms.
- Aquifer**— A water-bearing layer of soil, sand, gravel, or rock that will yield usable quantities of water to a well.
- Background concentration**— A concentration of a substance in a particular environment that is indicative of minimal influence by human (anthropogenic) sources.
- Bed sediment**— The material that temporarily is stationary in the bottom of a stream or other watercourse.
- Benthic**— Of, related to, or occurring on the bottom of a water body.
- Community**— In ecology, the species that interact in a common area.
- Constituent**— A chemical or biological substance in water, sediment, or biota that can be measured by an analytical method.
- Criterion**— A standard rule or test on which a judgment or decision can be based. Plural, **Criteria**.
- Cubic foot per second (ft³/s, or cfs)**— Rate of water discharge representing a volume of 1 cubic foot passing a given point during 1 second, equivalent to approximately 7.48 gallons per second, or 448.8 gallons per minute, or 0.02832 cubic meter per second.
- Detection limit**— The minimum concentration of a substance that can be identified, measured, and reported within 99 percent confidence that the analyte concentration is greater than zero; determined from analysis of a sample in a given matrix containing the analyte.
- Dissolved constituent**— Operationally defined as a constituent that passes through a 0.45-micrometer filter.
- Dissolved solids**— Amount of minerals, such as salt, that are dissolved in water; amount of dissolved solids is an indicator of salinity or hardness.
- Downgradient**— At or toward a location farther from the source of ground-water flow.
- Drainage basin**— The portion of the surface of the Earth that contributes water to a stream through overland runoff, including tributaries and impoundments.
- Drinking-water standard or guideline**— A threshold concentration in a public drinking-water supply, designed to protect human health. As defined here, standards are U.S. Environmental Protection Agency regulations that specify the maximum contaminate levels for public water systems required to protect the public welfare; guidelines have no regulatory status and are issued in an advisory capacity.
- Escherichia coli***—A common species of intestinal or fecal bacteria.
- Fecal bacteria**— Microscopic single-celled organisms (primarily fecal coliforms and fecal streptococci) found in the wastes of warm-blooded animals. Their presence in water is used to assess the sanitary quality of water for body-contact recreation or for consumption. Their presence indicates contamination by the wastes of warm-blooded animals and the possible presence of pathogenic (disease producing) organisms.
- Intolerant organisms**— Organisms that are not adaptable to human alterations to the environment and thus decline in numbers where human alterations occur. See also Tolerant species.
- Major ions**—Constituents commonly present in concentrations exceeding 1.0 milligram per liter. Dissolved cations generally are calcium, magnesium, sodium, and potassium; the major anions are sulfate, chloride, fluoride, nitrate, and those contributing to alkalinity, most generally bicarbonate and carbonate.
- Maximum contaminant level (MCL)**— Maximum permissible level of a contaminant in water that is delivered to any user of a public water system. MCLs are enforceable standards established by the U.S. Environmental Protection Agency.
- Micrograms per liter (µg/L)**— A unit expressing the concentration of constituents in solution as weight (micrograms) of solute per unit volume (liter) of water; equivalent to one part per billion in most streamwater and ground water. One thousand micrograms per liter equals 1 milligram per liter.
- Milligrams per liter (mg/L)**— A unit expressing the concentration of chemical constituents in solution as weight (milligrams) of solute per unit volume (liter) of water; equivalent to one part per million in most streamwater and ground water.
- Minimum reporting level (MRL)**— The smallest measured concentration of a constituent that may be reliably reported using a given analytical method. In many cases, the MRL is used when documentation for the detection limit is not available.
- Modified Hilsenhoff Biotic Index (MHBI)**— The Hilsenhoff Biotic Index (HBI) is a benthic invertebrate community index developed by W.L. Hilsenhoff. The HBI is determined by assigning a pollution tolerance value for each family of benthic invertebrates, then computing the average tolerance for a sample. In a modification of the HBI developed by R.W. Bode and M.A. Novak, pollution tolerance values are assigned by genus, which provides greater resolution in the average tolerance.
- Nutrient**— In aquatic systems, a substance that contributes to algal growth. Nutrients of concern include nitrogen and phosphorus compounds, but not elemental nitrogen.
- Picocurie (pCi)**— One trillionth (10^{12}) of the amount of radioactivity represented by a curie (Ci). A curie is the amount of radioactivity that yields 3.7×10^{10} radioactive disintegrations per second (dps). A picocurie yields 2.22 disintegrations per minute (dpm), or 0.037 dps.

Polycyclic aromatic hydrocarbon (PAH)— A class of organic compounds with a fused-ring (aromatic) structure. PAHs result from incomplete combustion of organic carbon (including wood), municipal solid waste, and fossil fuels, as well as from natural or anthropogenic introduction of uncombusted coal and oil. PAHs include benzo(a)pyrene, fluoranthene, and pyrene.

Recharge— Water that infiltrates the ground and reaches the saturated zone.

Secondary maximum contaminant level (SMCL)— The maximum contamination level in public water systems that, in the judgment of the U.S. Environmental Protection Agency (USEPA), is required to protect the public welfare. SMCLs are secondary (nonenforceable) drinking water regulations established by the USEPA for contaminants that may adversely affect the odor or appearance of such water.

Sediment— Particles, derived from rocks or biological materials, that have been transported by a fluid or other natural process, suspended or settled in water.

Specific conductance— A measure of the ability of a liquid to conduct an electrical current.

Suspended (as used in tables of chemical analyses)— The amount (concentration) of undissolved material in a water-sediment mixture. It is associated with the material retained on a 0.45-micrometer filter.

Suspended sediment— Particles of rock, sand, soil, and organic detritus carried in suspension in the water column, in contrast to sediment that moves on or near the streambed.

Taxon— Any identifiable group of taxonomically related organisms, such as a species or family. Plural, **Taxa**.

Tolerant species— Those species that are adaptable to (tolerant of) human alterations to the environment and often increase in number when human alterations occur.

Trace element— An element found in only minor amounts (concentrations less than 1.0 milligram per liter) in water or sediment; includes arsenic, cadmium, chromium, copper, lead, mercury, nickel, and zinc.

Upgradient— At or toward a location nearer to the source of ground-water flow.

Volatile organic compounds (VOCs)— Organic chemicals that have a high vapor pressure relative to their water solubility. VOCs include components of gasoline, fuel oils, and lubricants, as well as organic solvents, fumigants, some inert ingredients in pesticides, and some by-products of chlorine disinfection.

Water-quality standards— State-adopted and U.S. Environmental Protection Agency-approved ambient standards for water bodies. Standards include the use of the water body and the water-quality criteria that must be met to protect the designated use or uses.

Watershed— See Drainage basin.



Babcock Mill at Babcock State Park, WV.
Photograph by Douglas B. Chambers, USGS.

REFERENCES

- Ator, S.W., Blomquist, J.D., Brakebill, J.W., Denis, J.M., Ferrari, M.J., Miller, C.V., and Zappia, Humbert. 1998. Water quality in the Potomac River Basin. Maryland, Pennsylvania, Virginia, West Virginia, and the District of Columbia, 1992–96: U.S. Geological Survey Circular 1166, 38 p.
- Barbour, M.T., Gerritsen, J., Snyder, B.D., and Stribling, J.B., 1999, Rapid bioassessment protocols for use in streams and wadeable rivers—Periphyton, benthic macroinvertebrates, and fish (2d ed.): U.S. Environmental Protection Agency, Office of Water, EPA 841-B-99-002.
- Barwick, R.S., Levy, D.A., Craun, G.F., Beach, M.J., and Calderon, R.L., 2000, Surveillance for waterborne-disease outbreaks—United States, 1997–1998: Centers for Disease Control and Prevention, Morbidity and Mortality Weekly Report, May 26, v. 49, SS 4, p. 1–35.
- Boyer, D.G., and Pasquarell, G.C., 1999, Agricultural land use impacts on bacterial water quality in a karst ground-water aquifer: *Journal of the American Water Resources Association*, v. 35, no. 2, p. 291–300.
- Canadian Council of Ministers of the Environment, 1999, Canadian sediment quality guidelines for the protection of aquatic life: Winnipeg, Publication 1299, accessed January 19, 2001 at URL <http://www.ec.gc.ca/ceqg-rceq/>.
- Chambers, D.B., and Messinger, Terence, 1998, Selected contaminants in fish tissues in the Kanawha River Basin [abs]: *Proceedings of the West Virginia Academy of Science*, v. 70 no. 1, p. 20.
- Chambers, D.B., and Messinger, Terence, 2001, in press, Benthic macroinvertebrate communities in the Kanawha River Basin and their response to selected environmental factors: U.S. Geological Survey Water-Resources Investigations Report 01-4021.
- Chapman, P.M., Downie, James; Maynard, Allen; and Taylor, L.A., 1996, Coal and deodorizer residues in marine sediments—Contaminants or pollutants?: *Environmental Toxicology and Chemistry*, v. 15, no. 5, p. 638–642.
- Cincotta, D.A., Chambers, D.B., and Messinger, Terence, 1999, Recent changes in the distribution of fish species in the New River Basin in West Virginia and Virginia: *Proceedings of the New River Symposium*, Boone, N.C., p. 98–106.
- Clarkson, R.B., 1964, *Tumult on the mountains*: Parsons, W. Va., McClain Publishing Company, 401 p.
- Coble, R.W., Giese, G.L., and Eimers, J.L., 1985, North Carolina ground-water resources, in U.S. Geological Survey, 1985, National water summary 1984—Hydrologic events, selected water-quality trends, and ground-water resources: U.S. Geological Survey Water-Supply Paper 2275, p. 329–334.
- Crawford, J.K., and Luoma, S.N., 1993, Guidelines for studies of contaminants in biological tissue for the National Water-Quality Assessment Program: U.S. Geological Survey Open-File Report 92-494, 69 p.
- Cuffney, T.F., Gurtz, M.E., and Meador, M. R., 1993, Methods for collecting benthic invertebrate samples as part of the National Water-Quality Assessment Program: U.S. Geological Survey Open-File Report 93-406, 66 p.
- Ehlke, T.A., Runner, G.S., and Downs, S.C., 1982, Hydrology of Area 9, Eastern Coal Province, West Virginia: U.S. Geological Survey Water-Resources Investigations Open-File Report 81-803, 63 p.
- Eychaner, J.H., 1994, National Water-Quality Assessment Program—The Kanawha–New River Basin: U.S. Geological Survey Fact Sheet 94-019, 2 p.
- Eychaner, J.H., 1998, What's normal?—Constituent concentrations in West Virginia streamflow, ground water, streambed sediment, and fish tissue, in West Virginia Non-Point Source Conference, Charleston, 1998, *Proceedings: West Virginia Division of Environmental Protection and West Virginia Soil Conservation Agency*, p. 38–53.
- Eychaner, J.H., 1999, Progress of environmental studies in coal mining areas of western Pennsylvania and central West Virginia, in *Twentieth Annual West Virginia Surface Mine Drainage Task Force Symposium*, 1999, *Proceedings: Morgantown, W. Va.*, 6 p.
- Fedorko, Nick, and Blake, Mitch, 1998, A geologic overview of mountaintop removal mining in West Virginia: accessed October 20, 1999 at URL <http://www.wvgs.wvnet.edu/www/mtrm/>.
- Fenneman, N.M., 1938, *Physiography of Eastern United States*: New York, McGraw-Hill, 714 p.
- Gillies, W.N., Malakoff, D.A., Ailes, Nancy, and Constantz, George, 1998, *Greenbrier—A scientific portrait of a West Virginia river*: High View, W. Va., Cacapon Institute, 32 p.
- Gilliom, R.J., Alley, W.M., and Gurtz, M.E., 1995, Design of the National Water-Quality Assessment Program: Occurrence and distribution of water-quality conditions: U.S. Geological Survey Circular 1112, 33 p.
- Harlow, G.E., Jr., and LeCain, G.D., 1993, Hydraulic characteristics of, and ground-water flow in, coal-bearing rocks of southwestern Virginia: U.S. Geological Survey Water-Supply Paper 2388, 36 p.
- Henry, E.N., 1981, Phased cleanup, Kanawha River: Presented at Ohio River Valley Water Sanitation Commission (ORSANCO) meeting, May 1981, Charleston W. Va, 60 p.
- Herb, W.J., Shaw, L.C., and Brown, D.E., 1981a, Hydrology of Area 3, Eastern Coal Province, Pennsylvania: U.S. Geological Survey Water-Resources Investigations Open-File Report 81-537, 88 p.

- _____. 1981b, Hydrology of Area 5. Eastern Coal Province. Pennsylvania, Maryland, and West Virginia: U.S. Geological Survey Water-Resources Investigations Open-File Report 81-538, 92 p.
- Herb, W.J., Brown, D.E., Shaw, L.C., Stoner, J.D. and Felbinger, J.K., 1983, Hydrology of Area 2. Eastern Coal Province, Pennsylvania and New York: U.S. Geological Survey Water-Resources Investigations Open-File Report 82-647, 93 p.
- Hocutt, C.H., Denoncourt, R.F., and Stauffer, J.R., 1978. Fishes of the Greenbrier River, West Virginia, with drainage history of the Central Appalachians: *Journal of Biogeography*, v. 5, no. 1, p. 59-80.
- _____. 1979. Fishes of the Gauley River, West Virginia: *Brimleyana*, no. 1, p. 47-80.
- Jenkins, R.E., and Burkhead, N.M., 1994. Freshwater fishes of Virginia: Bethesda, Md., American Fisheries Society, 1079 p.
- Kanetsky, C.A., 1988. A study of dioxin contamination in sediments of the Kanawha River basin — Final report: U.S. Environmental Protection Agency, in cooperation with West Virginia Department of Natural Resources, 12 p., appendix.
- Karr, J.R., and Chu, E.W., 1999. Restoring life in running waters—Better biological monitoring: Washington, D.C., Island Press, 206 p.
- Kozar, M.D., Sheets, C.J., and Hughes, C.A., 2001, Ground-water quality and geohydrology of the Blue Ridge physiographic province, New River Basin, Virginia and North Carolina: U.S. Geological Survey Water-Resources Investigations Report 00-4270, 36 p.
- Kozar, M.D., and Sheets, C.J., 1997, Radon in ground water in the Kanawha–New River Basin, West Virginia, Virginia, and North Carolina [abs.]: *Proceedings of the Ohio River Basin Consortium for Research and Education 13th Annual Scientific Symposium*, 1 p.
- Lindsey, B.D., and Ator, S.W., 1996, Radon in ground water of the Lower Susquehanna and Potomac River Basins: U.S. Geological Survey Water-Resources Investigations Report 96-4156, 6 p.
- Lundgren, R.F., and Lopes, T.J., 1999, Occurrence, distribution, and trends of volatile organic compounds in the Ohio River and its major tributaries, 1987-96: U.S. Geological Survey Water-Resources Investigations Report 99-4257, 89 p.
- Maret, T.R., 1997. Characteristics of fish assemblages and related environmental variables for streams of the Upper Snake River Basin. Idaho and western Wyoming, 1993-95: U.S. Geological Survey Water-Resources Investigations Report 97-4087, 46 p.
- Mathes, M.V., 2000. Relation of bacteria in limestone aquifers to septic systems in Berkeley County, West Virginia: U.S. Geological Survey Water-Resources Investigations Report 00-4229, 12 p.
- McColloch, G.H., Jr., 1998, Coal geology of West Virginia. *in* Sanda, A.P., ed., *Keystone coal industry manual*, 1998: Chicago, Intertec Publishing, p. 708-717.
- Meador, M.R., Cuffney, T.F., and Gurtz, M.E., 1993. Methods for sampling fish communities as part of the National Water-Quality Assessment Program: U.S. Geological Survey Open-File Report 93-104, 40 p.
- Meador, M.R., Hupp, C.R., Cuffney, T.F., and Gurtz, M.E., 1993, Methods for characterizing stream habitat as part of the National Water-Quality Assessment Program: U.S. Geological Survey Open-File Report 93-408, 48 p.
- Messinger, Terence, 1997, Water-quality assessment of the Kanawha–New River Basin. West Virginia, Virginia, and North Carolina —Review of water-quality literature through 1996: U.S. Geological Survey Water-Resources Investigations Report 97-4075, 27 p.
- Messinger, Terence, and Chambers, D. B., 1998, Selected contaminants in bed sediment in the Kanawha River Basin [abs]: *Proceedings of the West Virginia Academy of Science*, v. 70, no. 1, p. 20.
- Messinger, Terence, and Hughes, C.A., 2000, Environmental setting and its effects on water quality, Kanawha-New River Basin—West Virginia, Virginia, North Carolina: National Water-Quality Assessment Program: U.S. Geological Survey Water-Resources Investigations Report 00-4020, 75 p.
- Messinger, Terence, and Chambers, D.B., 2001, in press. Fish assemblages of the Kanawha River basin and their relation to the environment, 1997-98: U.S. Geological Survey Water-Resources Investigations Report 01-4048
- Multi-Resolution Land Characteristics Interagency Consortium, 1997. Federal regional land cover data sets: accessed September 19, 1999, at URL <http://www.epa.gov/mrlc/Regions.html/>.
- National Agriculture Statistics Service, 1999, Census of agriculture, 1997: accessed September 10, 1999, at URL <http://www.nass.usda.gov/census/>.
- North Carolina Department of Environment and Natural Resources, 1999, Basinwide assessment report, New River Basin, July 1999: Raleigh, Division of Water Quality, 99 p.
- Rose, A.W., and Cravotta, C.A., III, 1998. Geochemistry of coal mine drainage, *in* Brady, K.B.C., Smith, M.W., and Schueck, Joseph, eds., 1998. Coal mine drainage prediction and pollution prevention in Pennsylvania: Pennsylvania Department of Environmental Protection, p. 1-1 to 1-22.
- Sams, J.I., III, and Beer, K.M., 2000, Effects of coal-mine drainage on stream water quality in the Allegheny and Monongahela River basins—Sulfate transport and trends: U.S. Geological Survey Water-Resources Investigations Report 99-4208, 17 p.

- Sheets, C.J., and Kozar, M.D., 1997. Bacteria in water from domestic wells in fractured bedrock within the Kanawha–New River basin in West Virginia, Virginia, and North Carolina [abs.]: Proceedings of the Ohio River Basin Consortium for Research and Education 13th Annual Scientific Symposium. 1 p.
- Sheets, C.J., and Kozar, M.D., 2000. Ground-water quality in the Appalachian Plateaus, Kanawha River Basin. West Virginia: U.S. Geological Survey Water-Resources Investigations Report 99–4269. 25 p.
- Shelton, L.R., 1994. Field guide for collecting and processing stream-water samples for the National Water-Quality Assessment Program: U.S. Geological Survey Open-File Report 98–455, 42 p.
- Shelton, L.R., and Capel, P.D., 1994. Guidelines for the collecting and processing samples of stream bed sediment for analysis of trace elements and organic contaminants for the National Water-Quality Assessment Program: U.S. Geological Survey Open-File Report 94–458, 20 p.
- Skousen, J., and others, 1998. Handbook of technologies for avoidance and remediation of acid mine drainage: Morgantown, W. Va., National Mine Land Reclamation Center. 131 p.
- Solley, W.B., Pierce, R.B., and Perlman, H.A., 1998. Estimated use of water in the United States in 1995: U.S. Geological Survey Circular 1200, 71 p.
- Stauffer, J.R., Jr., Boltz, J.M., and White, L.R., 1995. The fishes of West Virginia: Academy of Natural Sciences of Philadelphia, 389 p.
- Strange, R.M., 1999. Historical biogeography, ecology, and fish distributions—Conceptual issues for establishing IBI criteria, *in* Simon, T.P. ed., *Assessing the sustainability and biological integrity of water resources using fish communities*: Boca Raton, Fla., CRC Press, p. 65–78.
- Tennant, P.A., Norman, C.G., and Vicory, A.H., Jr., 1992. The Ohio River Valley Water Sanitation Commission's toxic substances control program for the Ohio River: *Water Science and Technology*, v. 26, no. 7–8, p. 1779–1788.
- Tully, John, 1996. Coal fields of the conterminous United States: U.S. Geological Survey Open-File Report 96–92, 1 sheet, scale 1:5,000,000.
- U. S. Environmental Protection Agency, 1986. Quality criteria for water – 1986: Office of Water, EPA 440/5–89–001.
- _____, 1999. Proposed radon in drinking water rule: Office of Water, EPA 815–F–99–009. Accessed January 19, 2001 at URL <http://www.epa.gov/safewater/radon/> and scroll to proposal.html.
- _____, 2000. Mountaintop mining valley fill draft environmental impact statement: EPA/903/R–00/013, accessed January 18, 2001 at URL <http://www.epa.gov/region3/mnttop/>.
- U.S. Geological Survey, 1999. The quality of our Nation's waters—Nutrients and pesticides: U.S. Geological Survey Circular 1225, 82 p.
- _____, 2000. Annual coal production for West Virginia counties, 1900–1998: accessed August 15, 2000, at URL <http://wv.usgs.gov/nawqa/coal.html>.
- Virginia Department of Environmental Quality, 1998. 1998 water quality assessment report: accessed September 21, 2000, at URL <http://www.deq.state.va.us/water/>.
- Waite, I.R., and Carpenter, K.D., 2000. Associations among fish assemblage structure and environmental variables in Willamette Basin streams. *Oregon: Transactions of the American Fisheries Society*, v. 129, no. 3, p. 754–770.
- Ward, S.M., Taylor, B.C., and Crosby, G.R., 1997. Water resources data, West Virginia, water year 1996: U.S. Geological Survey Water-Data Report WV–96–1, 258 p.
- _____, 1998. Water resources data, West Virginia, water year 1997: U.S. Geological Survey Water-Data Report WV–97–1, 414 p.
- _____, 1999. Water resources data, West Virginia, water year 1998: U.S. Geological Survey Water-Data Report WV–98–1, 476 p.
- West Virginia Division of Environmental Protection, 2000. West Virginia water quality status assessment 2000: Charleston, Office of Water Resources. 202 p.
- Wyrick, G.G., and Borchers, J.W., 1981. Hydrologic effects of stress-relief fracturing in an Appalachian valley: U.S. Geological Survey Water-Supply Paper 2177, 51 p.



Greenbrier River near Seebert, WV.
 Photograph by Katherine S. Paybins, USGS.

APPENDIX—WATER-QUALITY DATA FROM THE KANAWHA–NEW RIVER BASIN IN A NATIONAL CONTEXT

For a complete view of Kanawha–New River Basin data and for additional information about specific benchmarks used, visit our Web site at <http://water.usgs.gov/nawqa/>. Also visit the NAWQA Data Warehouse for access to NAWQA data sets at <http://infotrek.er.usgs.gov/wdbcctx/nawqa/nawqa.home>.

This appendix is a summary of chemical concentrations and biological indicators assessed in the Kanawha–New River Basin. Selected results for this basin are graphically compared to results from as many as 36 NAWQA Study Units investigated from 1991 to 1998 and to national water-quality benchmarks for human health, aquatic life, or fish-eating wildlife. The chemical and biological indicators shown were selected on the basis of frequent detection, detection at concentrations above a national benchmark, or regulatory or scientific importance. The graphs illustrate how conditions associated with each land use sampled in the Kanawha–New River Basin compare to results from across the Nation, and how conditions compare among the several land uses. Graphs for chemicals show only detected concentrations and, thus, care must be taken to evaluate detection frequencies in addition to concentrations when comparing study-unit and national results. For example, simazine concentrations in Kanawha–New River Basin agricultural streams were similar to the national distribution, but the detection frequency was much higher (94 percent compared to 61 percent).

CHEMICALS IN WATER

Concentrations and detection frequencies, Kanawha–New River Basin, 1996–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals

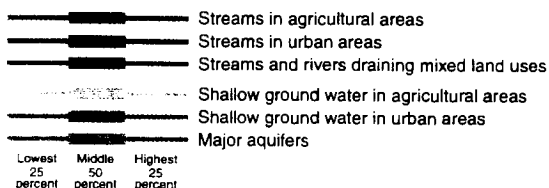
- ◆ Detected concentration in Study Unit

66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency

.. Not measured or sample size less than two

12 Study-unit sample size. For ground water, the number of samples is equal to the number of wells sampled

National ranges of detected concentrations, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

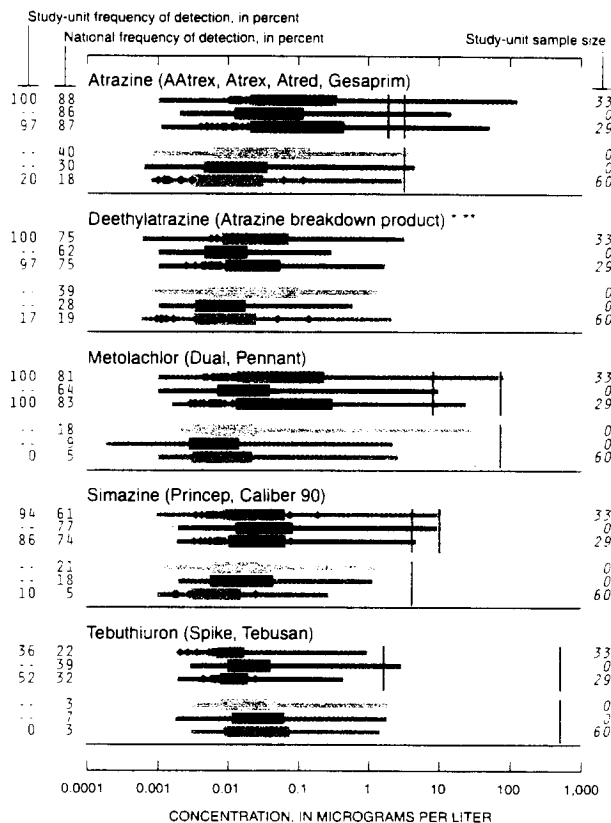


National water-quality benchmarks

National benchmarks include standards and guidelines related to drinking-water quality, criteria for protecting the health of aquatic life, and a goal for preventing stream eutrophication due to phosphorus. Sources include the U.S. Environmental Protection Agency and the Canadian Council of Ministers of the Environment

- | Drinking-water quality (applies to ground water and surface water)
- | Protection of aquatic life (applies to surface water only)
- | Prevention of eutrophication in streams not flowing directly into lakes or impoundments
- No benchmark for drinking-water quality
- .. No benchmark for protection of aquatic life

Pesticides in water—Herbicides



Other herbicides detected

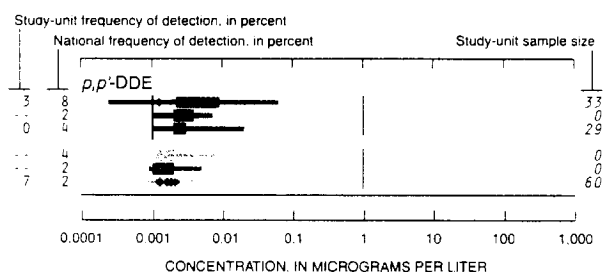
Acetochlor (Harness Plus, Surpass) **
 Alachlor (Lasso, Bronco, Lariat, Bullet) **
 Bentfluralin (Balan, Benefin, Bonalan) **
 Cyanazine (Bladex, Fortrol)
 DCPA (Dacthal, chlorthal-dimethyl) **
 2,6-Diethylaniline (Alachlor breakdown product) **
 Dinoseb (Dinosebe)
 Diuron (Crisuron, Karmex, Diurex) **
 EPTC (Eptam, Farmarox, Alirox) **
 Fenuron (Fenulon, Fenidim) **
 Molinate (Ordram) **
 Napropamide (Devrinol) **
 Oryzalin (Surflan, Dirimal) **
 Prometon (Pramitol, Princep) **
 Triallate (Far-Go, Avadex BW, Tri-allate) *
 Triclopyr (Garlon, Grandstand, Redeem, Remedy) **
 Trifluralin (Treflan, Gowan, Tri-4, Trific)

Herbicides not detected

Acifluorfen (Blazer, Tackle 2S) **
 Bentazon (Basagran, Bentazone) **
 Bromacil (Hyvar X, Urox B, Bromax)
 Bromoxynil (Buctril, Brominal) *
 Butylate (Sutan +, Genate Plus, Butilate) **
 Chloramben (Amiben, Amilon-WP, Vegiben) **
 Clopyralid (Stinger, Lontrel, Transline) **
 2,4-D (Aqua-Kleen, Lawn-Keep, Weed-B-Gone)
 2,4-DB (Butyrac, Butoxone, Embutox Plus, Embutone) **
 Dacthal mono-acid (Dacthal breakdown product) **
 Dicamba (Banvel, Dianat, Scotts Proturf)
 Dichlorprop (2,4-DP, Seritox 50, Lentemul) **
 Ethalfurailin (Sonalan, Curbit) **

Fluometuron (Flo-Met, Cotoran) **
 Linuron (Lorox, Linex, Sarclax, Linurex) ***
 MCPA (Rhomene, Rhonox, Chiptox)
 MCPB (Thistrol) * **
 Metribuzin (Lexone, Sencor)
 Neburon (Neburea, Neburyl, Noruben) ***
 Norflurazon (Evitai, Predict, Solicam, Zorial) ***
 Pebulate (Tillam, PEBC) * **
 Pendimethalin (Pre-M, Prowl, Stomp) * **
 Picloram (Grazon, Tordon)
 Pronamide (Kerb, Propyzamid) **
 Propachlor (Ramrod, Satecid) **
 Propanil (Stam, Stampede, Wham) * **
 Propham (Tuberite) **
 2,4,5-T **
 2,4,5-TP (Silvex, Fenoprop) **
 Terbacil (Sinbar) **
 Thiobencarb (Bolero, Saturn, Benthocarb) * **

Pesticides in water—Insecticides



Other insecticides detected

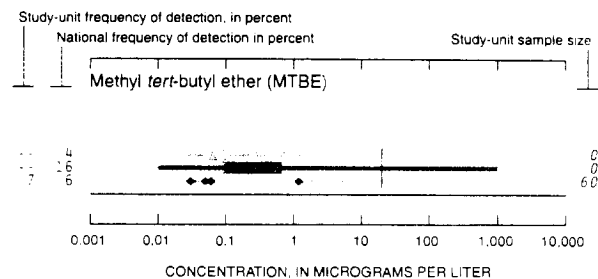
Carbaryl (Carbamine, Denapon, Sevin)
 Carbofuran (Furadan, Curater, Yaltox)
 Chlorpyrifos (Brodan, Dursban, Lorsban)
 Diazinon (Basudin, Diazatol, Neocidol, Knox Out)
 alpha-HCH (alpha-BHC, alpha-lindane) **
 gamma-HCH (Lindane, gamma-BHC)
 Malathion (Malathion)

Insecticides not detected

Aldicarb (Temik, Ambush, Pounce)
 Aldicarb sulfone (Standak, aldoxycarb)
 Aldicarb sulfoxide (Aldicarb breakdown product)
 Azinphos-methyl (Guthion, Gusathion M) *
 Dieldrin (Panoram D-31, Octalox, Compound 497)
 Disulfoton (Disyston, Di-Syston) **
 Ethoprop (Mocap, Ethoprophos) ***
 Fonofos (Dytonate, Capfos, Cudgel, Tycap) **
 3-Hydroxycarbofuran (Carbofuran breakdown product) ***
 Methiocarb (Slug-Geta, Grandslam, Mesuroil) ***
 Methomyl (Lanox, Lannate, Acinate) **
 Methyl parathion (Penncap-M, Folidol-M) **
 Oxamyl (Vydate L, Pratt) **
 Parathion (Roethyl-P, Alkron, Panthion, Phoskil) *
 cis-Permethrin (Ambush, Astro, Pounce) ***
 Phorate (Thimet, Granutox, Geomet, Rampart) ***
 Propargite (Comite, Omite, Ornamite) **
 Propoxur (Baygon, Blattanex, Unden, Proprotax) ***
 Terbufos (Contraven, Counter, Pilarfox) **

Volatile organic compounds (VOCs) in ground water

These graphs represent data from 16 Study Units, sampled from 1996 to 1998



Other VOCs detected

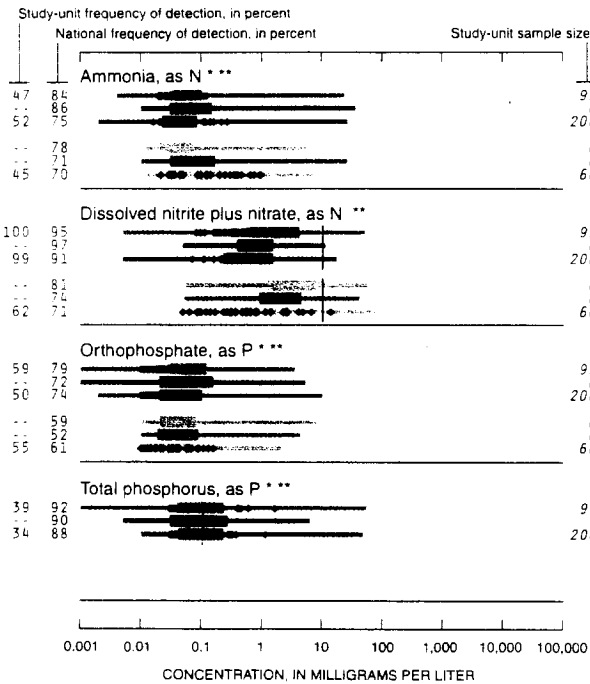
Benzene
 Bromodichloromethane (Dichlorobromomethane)
 2-Butanone (Methyl ethyl ketone (MEK)) *
 Carbon disulfide *
 Chlorodibromomethane (Dibromochloromethane)
 Chloromethane (Methyl chloride)
 1,4-Dichlorobenzene (*p*-Dichlorobenzene)
 Dichlorodifluoromethane (CFC 12, Freon 12)
 1,1-Dichloroethane (Ethylidene dichloride) *
 1,1-Dichloroethene (Vinylidene chloride)
cis-1,2-Dichloroethene ((*Z*)-1,2-Dichloroethene)
 Diisopropyl ether (Diisopropylether (DIPE)) *
 1,2-Dimethylbenzene (*o*-Xylene)
 1,3 & 1,4-Dimethylbenzene (*m*-&*p*-Xylene)
 1-4-Eooxy butane (Tetrahydrofuran, Diethylene oxide) *
 Ethylbenzene (Phenylethane)
 Iodomethane (Methyl iodide) *
 Isopropylbenzene (Cumene) *
 Methylbenzene (Toluene)
 2-Propanone (Acetone) *
 Tetrachloroethene (Perchloroethene)
 Tribromomethane (Bromoform)
 1,2,4-Trichlorobenzene
 1,1,1-Trichloroethane (Methylchloroform)
 Trichloroethene (TCE)
 Trichlorofluoromethane (CFC 11, Freon 11)
 Trichloromethane (Chloroform)
 1,2,4-Trimethylbenzene (Pseudocumene) *

VOCs not detected

tert-Amyl methyl ether (*tert*-amyl methyl ether (TAME)) *
 Bromobenzene (Phenyl bromide) *
 Bromochloromethane (Methylene chlorobromide)
 Bromoethene (Vinyl bromide) *
 Bromomethane (Methyl bromide)
n-Butylbenzene (1-Phenylbutane) *
sec-Butylbenzene *
tert-Butylbenzene *
 3-Chloro-1-propene (3-Chloropropene) *
 1-Chloro-2-methylbenzene (*o*-Chlorotoluene)
 1-Chloro-4-methylbenzene (*p*-Chlorotoluene)
 Chlorobenzene (Monochlorobenzene)
 Chloroethane (Ethyl chloride) *
 Chloroethene (Vinyl chloride)
 1,2-Dibromo-3-chloropropane (DBCP, Nemagon)
 1,2-Dibromoethane (Ethylene dibromide, EDB)
 Dibromomethane (Methylene dibromide) *
trans-1,4-Dichloro-2-butene ((*Z*)-1,4-Dichloro-2-butene) *
 1,2-Dichlorobenzene (*o*-Dichlorobenzene)
 1,3-Dichlorobenzene (*m*-Dichlorobenzene)
 1,2-Dichloroethane (Ethylene dichloride)
trans-1,2-Dichloroethene ((*E*)-1,2-Dichloroethene)
 Dichloromethane (Methylene chloride)
 1,2-Dichloropropane (Propylene dichloride)
 2,2-Dichloropropane *
 1,3-Dichloropropane (Trimethylene dichloride) *
trans-1,3-Dichloropropene ((*E*)-1,3-Dichloropropene)
cis-1,3-Dichloropropene ((*Z*)-1,3-Dichloropropene)
 1,1-Dichloropropene *
 Diethyl ether (Ethyl ether) *
 Ethenylbenzene (Styrene)
 Ethyl methacrylate *

Ethyl *tert*-butyl ether (Ethyl-*t*-butyl ether (ETBE) *
 1-Ethyl-2-methylbenzene (2-Ethyltoluene) *
 Hexachlorobutadiene
 1,1,1,2,2,2-Hexachloroethane (Hexachloroethane)
 2-Hexanone (Methyl butyl ketone (MBK)) *
p-Isopropyltoluene (*p*-Cymene) *
 Methyl acrylonitrile *
 Methyl-2-methacrylate (Methyl methacrylate) *
 4-Methyl-2-pentanone (Methyl isobutyl ketone (MIBK)) *
 Methyl-2-propenoate (Methyl acrylate) *
 Naphthalene
 2-Propenenitrile (Acrylonitrile)
n-Propylbenzene (Isocumene) *
 1,1,1,2-Tetrachloroethane *
 1,1,1,2-Tetrachloroethane
 Tetrachloromethane (Carbon tetrachloride)
 1,2,3,4-Tetramethylbenzene (Prenhitene) *
 1,2,3,5-Tetramethylbenzene (Isodurene) *
 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon 113) *
 1,2,3-Trichlorobenzene *
 1,1,2-Trichloroethane (Vinyl trichloride)
 1,2,3-Trichloropropane (Allyl trichloride)
 1,2,3-Trimethylbenzene (Hemimellitene) *
 1,3,5-Trimethylbenzene (Mesitylene) *

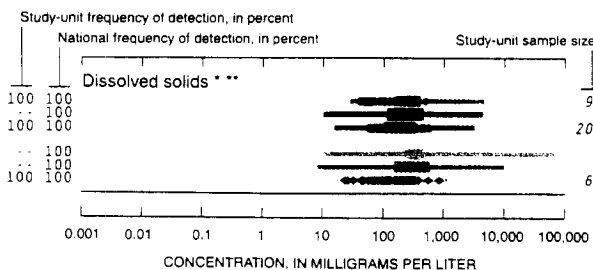
Nutrients in water



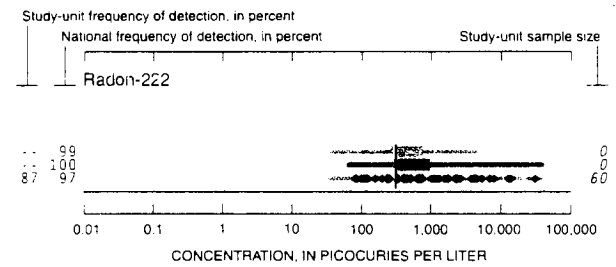
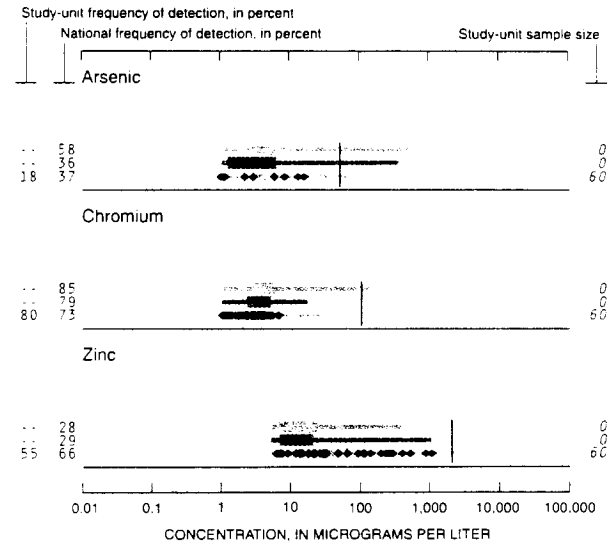
Nutrients not detected

Dissolved ammonia plus organic nitrogen as N ***

Dissolved solids in water



Trace elements in ground water



Other trace elements detected

Lead
Selenium
Uranium

Trace elements not detected

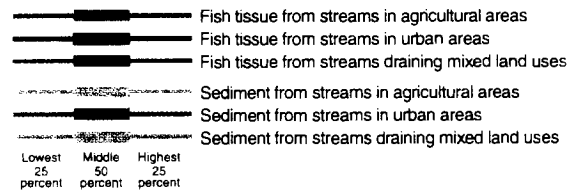
Cadmium

CHEMICALS IN FISH TISSUE AND BED SEDIMENT

Concentrations and detection frequencies, Kanawha–New River Basin, 1996–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals. Study-unit frequencies of detection are based on small sample sizes; the applicable sample size is specified in each graph

- ◆ Detected concentration in Study Unit
- 66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- Not measured or sample size less than two
- 12 Study-unit sample size

National ranges of concentrations detected, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

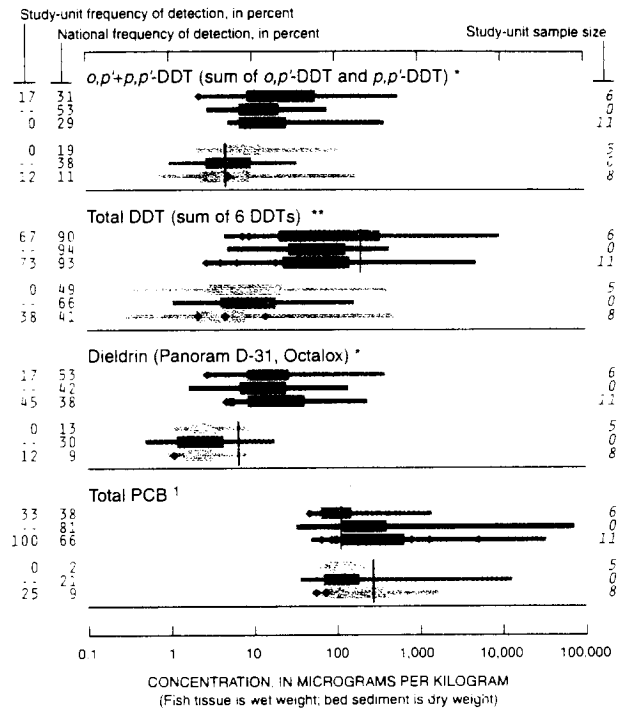
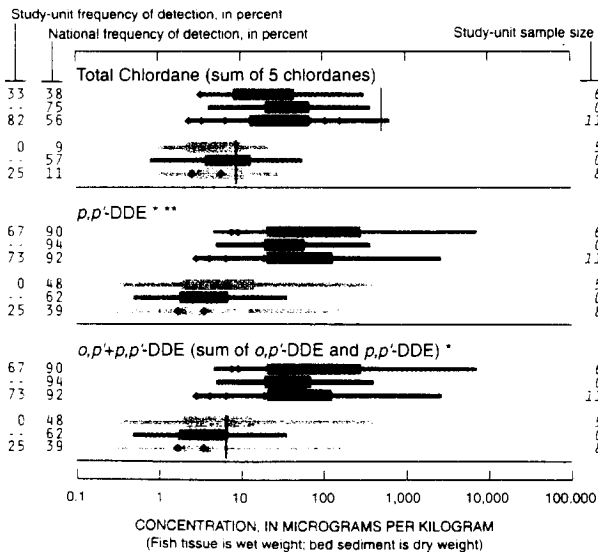


National benchmarks for fish tissue and bed sediment

National benchmarks include standards and guidelines related to criteria for protection of the health of fish-eating wildlife and aquatic organisms. Sources include the U.S. Environmental Protection Agency, other Federal and State agencies, and the Canadian Council of Ministers of the Environment

- | Protection of fish-eating wildlife (applies to fish tissue)
- | Protection of aquatic life (applies to bed sediment)
- No benchmark for protection of fish-eating wildlife
- No benchmark for protection of aquatic life

Organochlorines in fish tissue (whole body) and bed sediment



¹ The national detection frequencies for total PCB in sediment are biased low because about 30 percent of samples nationally had elevated detection levels compared to this Study Unit. See <http://water.usgs.gov/> for additional information

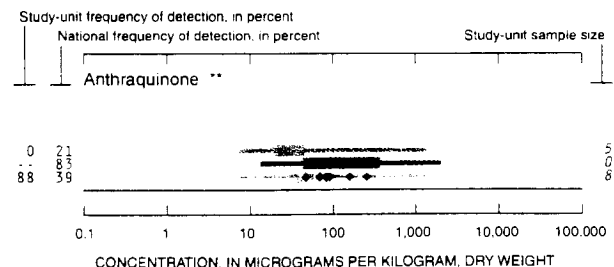
Other organochlorines detected

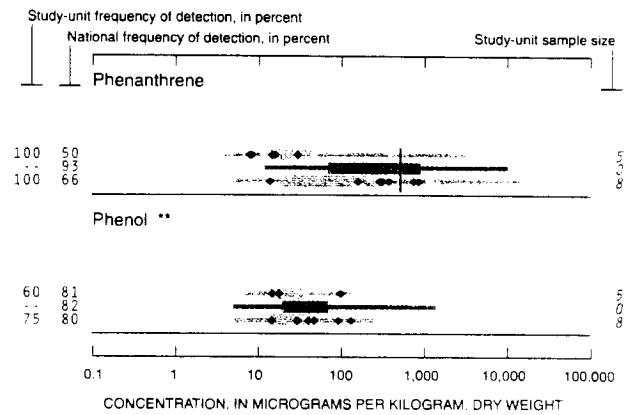
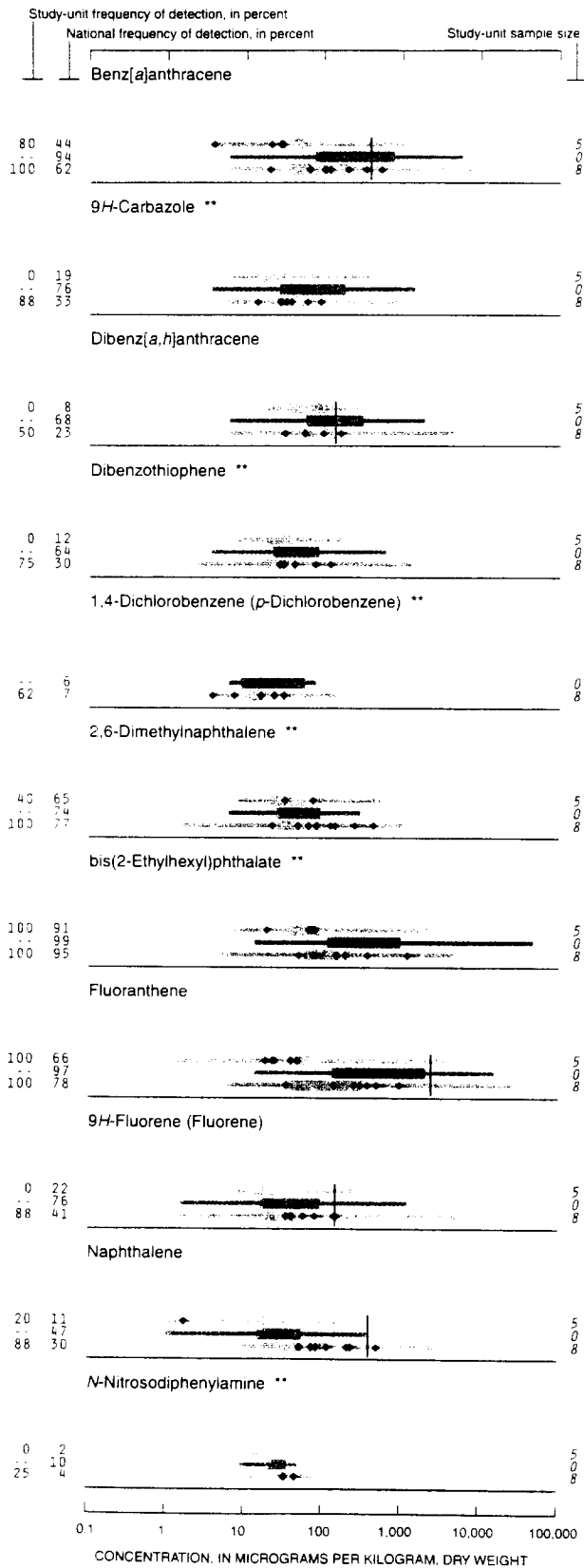
- o,p'+p,p'-DDD (sum of o,p'-DDD and p,p'-DDD) *
- Dieldrin+aldrin (sum of dieldrin and aldrin) **
- Heptachlor epoxide (Heptachlor breakdown product) *
- Heptachlor+heptachlor epoxide (sum of heptachlor and heptachlor epoxide) **

Organochlorines not detected

- Chloroneb (Chloronebe, Demosan) * **
- DCPA (Dacthal, chlorthal-dimethyl) * **
- Endosulfan I (alpha-Endosulfan, Thiodan) * **
- Endrin (Endrine)
- gamma-HCH (Lindane, gamma-BHC, Gammexane) *
- Total-HCH (sum of alpha-HCH, beta-HCH, gamma-HCH, and delta-HCH) **
- Hexachlorobenzene (HCB) **
- Isodrin (Isodrine, Compound 711) * **
- p,p'-Methoxychlor (Marlate, methoxychlore) * **
- o,p'-Methoxychlor * **
- Mirex (Dechlorane) **
- Pentachloroanisole (PCA) * **
- cis-Permethrin (Ambush, Astro, Pounce) * **
- trans-Permethrin (Ambush, Astro, Pounce) * **
- Toxaphene (Camphechlor, Hercules 3956) * **

Semivolatile organic compounds (SVOCs) in bed sediment





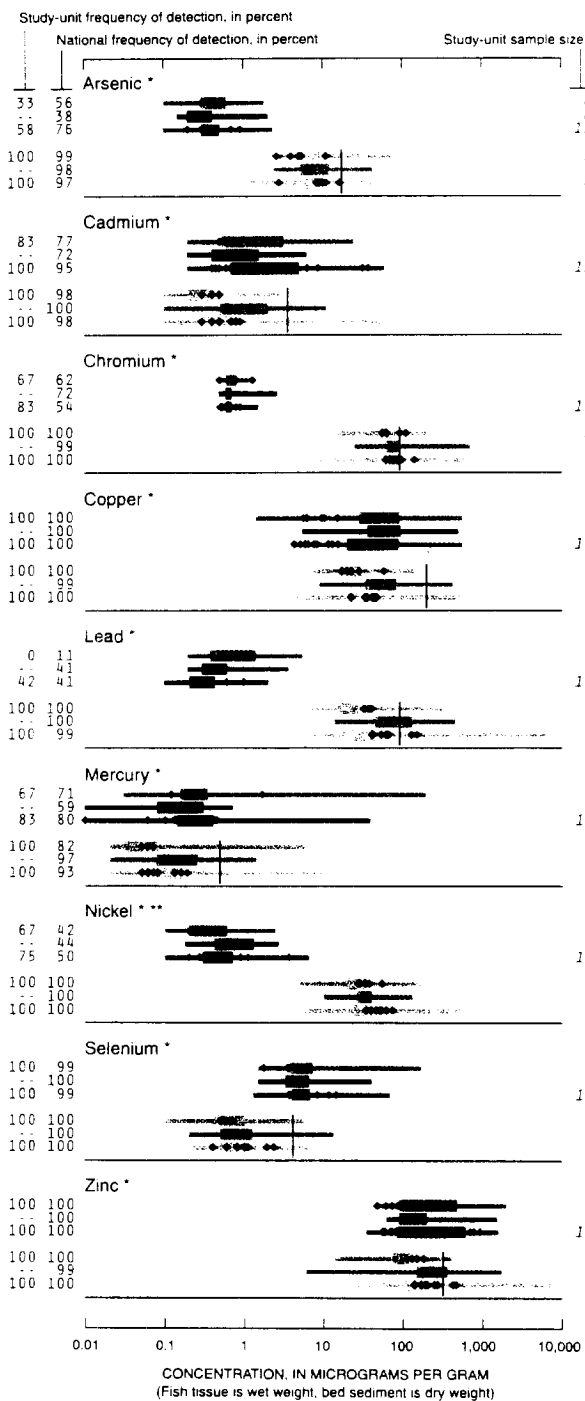
Other SVOCs detected

- Acenaphthene
- Acenaphthylene
- Acridine **
- C8-Alkylphenol **
- Anthracene
- Benzo[a]pyrene
- Benzo[b]fluoranthene **
- Benzo[ghi]perylene **
- Benzo[k]fluoranthene **
- Butylbenzylphthalate **
- Chrysene
- p-Cresol **
- Di-n-butylphthalate **
- 1,2-Dichlorobenzene (o-Dichlorobenzene) **
- Diethylphthalate **
- 1,2-Dimethylnaphthalene **
- 1,6-Dimethylnaphthalene **
- 3,5-Dimethylphenol **
- Dimethylphthalate **
- 2,4-Dinitrotoluene **
- Indeno[1,2,3-cd]pyrene **
- Isoquinoline **
- 1-Methyl-9H-fluorene **
- 2-Methylantracene **
- 4,5-Methylenephenanthrene **
- 1-Methylphenanthrene **
- 1-Methylpyrene **
- Phenanthridine **
- Pyrene
- Quinoline **
- 1,2,4-Trichlorobenzene **
- 2,3,6-Trimethylnaphthalene **

SVOCs not detected

- Azobenzene **
- Benzo[c]cinnoline **
- 2,2-Biquinoline **
- 4-Bromophenyl-phenylether **
- 4-Chloro-3-methylphenol **
- bis(2-Chloroethoxy)methane **
- 2-Chloronaphthalene **
- 2-Chlorophenol **
- 4-Chlorophenyl-phenylether **
- Di-n-octylphthalate **
- 1,3-Dichlorobenzene (m-Dichlorobenzene) **
- Isophorone **
- Nitrobenzene **
- N-Nitrosodi-n-propylamine **
- Pentachloronitrobenzene **

Trace elements in fish tissue (livers) and bed sediment



BIOLOGICAL INDICATORS

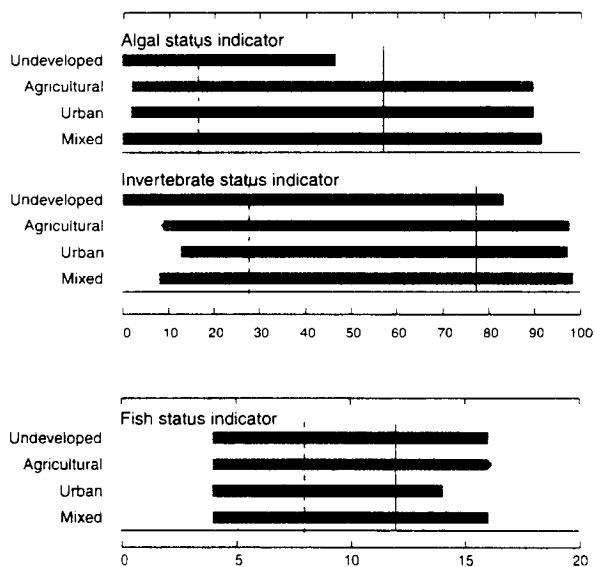
Higher national scores suggest habitat disturbance, water-quality degradation, or naturally harsh conditions. The status of algae, invertebrates (insects, worms, and clams), and fish provide a record of water-quality and stream conditions that water-chemistry indicators may not reveal. **Algal status** focuses on the changes in the percentage of certain algae in response to increasing siltation, and it often correlates with higher nutrient concentrations in some regions. **Invertebrate status** averages 11 metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. **Fish status** sums the scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that increase in association with water-quality degradation.

Biological indicator value, Kanawha–New River Basin, by land use, 1996–98

◆ Biological status assessed at a site

National ranges of biological indicators, in 16 NAWQA Study Units, 1994–98

- Streams in undeveloped areas
- Streams in agricultural areas
- Streams in urban areas
- Streams in mixed-land-use areas
- 75th percentile
- - - 25th percentile



A COORDINATED EFFORT

Coordination with agencies and organizations in the Kanawha-New River Basin was integral to the success of this water-quality assessment. We thank those who served as members of our liaison committee.

Federal Agencies

National Park Service
U.S. Army Corps of Engineers
U.S. Environmental Protection Agency
U.S. Fish and Wildlife Service
U.S. Office of Surface Mining
U.S. Department of Agriculture
 Agricultural Research Service
 Natural Resources Conservation Service
 Monongahela National Forest

State Agencies

North Carolina Division of Environmental Management
Virginia Department of Environmental Quality
Virginia Department of Game and Inland Fisheries
Virginia Department of Health
Virginia Division of Mineral Resources
Virginia Division of Soil and Water Conservation
West Virginia Bureau for Public Health
West Virginia Division of Environmental Protection
West Virginia Division of Natural Resources
West Virginia Geological and Economic Survey
West Virginia Soil Conservation Agency

Universities

Marshall University
Virginia Polytechnic Institute and State University
West Virginia University

Other public and private organizations

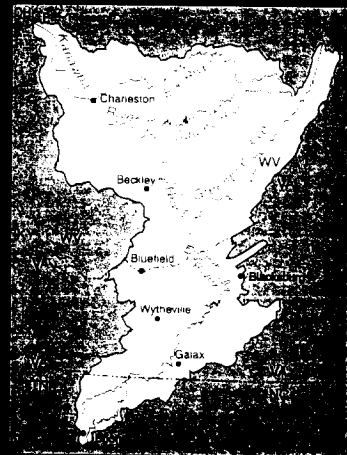
Cacapon Institute
Canaan Valley Institute
Greenbrier River Watershed Association
National Committee for the New River
New River Community Partners
Ohio River Valley Water Sanitation Commission
West Virginia American Water Company
West Virginia Citizens Action Group
West Virginia Coal Association
West Virginia Farm Bureau
West Virginia Highlands Conservancy
West Virginia Manufacturers Association
West Virginia Mining and Reclamation Association
West Virginia Rivers Coalition
West Virginia Rural Water Association

We gratefully acknowledge the cooperation of numerous property owners who provided access to sampling locations on their land. We also thank the following individuals for contributing data, knowledge, time, and expertise to this effort.

Dennis Adams, Billy Barton, Steven Bolssen, Melody Bova, Freddie Brogan, Charlynn Sheets Buchanan, John Buchanan, Daniel Cincotta, Matthew Cooke, Gary Crosby, David Eaton, Michael Eckenwiler, Ronald Evaldi, Carl Faulkenburg, Patsy Francisco, Georganne Gillespie, Wesley Gladwell, Jeffrey Hajenga, Kristi Hanson, Harold Henderlite, Curt Hughes, Donna Justus, Lisa Ham Lahti, Melvin Mathes, Kimberly Miller, Dawn Newell, Jesse Purvis, Brian Rasmussen, Lary Rogers, Tom Rosier, Benjamin Simerl, Kimberly Smith, Stephen Sorenson, Janet Steven, Joan Steven, Edward Vincent, Stephen Ward, David Wellman, Jeremy White, Matthew Wooten, Dennis Wyatt, Humbert Zappia

NAWQA

National Water-Quality Assessment (NAWQA) Program Kanawha–New River Basin



Paybins and others—Water Quality in the Kanawha–New River Basin
U.S. Geological Survey Circular 1204

ISBN 0-607-95412-4

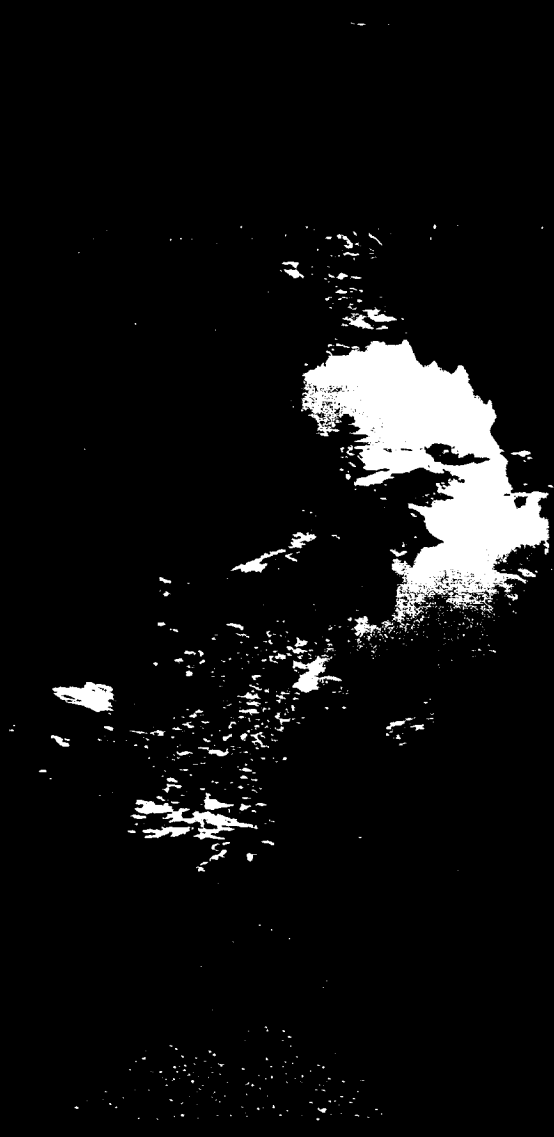


9 780607 954128

R0024275

Water Quality in the Upper Tennessee River Basin

Tennessee, North Carolina, Virginia, and Georgia, 1994–98



POINTS OF CONTACT AND ADDITIONAL INFORMATION

The companion Web site for NAWQA summary reports:

<http://water.usgs.gov/nawqa/>

Upper Tennessee River Basin contact and Web site:

District Chief
U.S. Geological Survey
Water Resources Division
640 Grassmere Park Drive, Suite 100
Nashville, TN 37211
e-mail: dc_tn@usgs.gov
<http://tn.water.usgs.gov/>

National NAWQA Program:

Chief, NAWQA Program
U.S. Geological Survey
Water Resources Division
12201 Sunrise Valley Drive, M.S. 413
Reston, VA 20192
<http://water.usgs.gov/nawqa/>

Other NAWQA summary reports

River Basin Assessments

Albemarle-Pamlico Drainage Basin (Circular 1157)
Allegheny and Monongahela River Basins (Circular 1202)
Apalachicola-Chattahoochee-Flint River Basin (Circular 1164)
Central Arizona Basins (Circular 1213)
Central Columbia Plateau (Circular 1144)
Central Nebraska Basins (Circular 1163)
Connecticut, Housatonic and Thames River Basins (Circular 1155)
Eastern Iowa Basins (Circular 1210)
Georgia-Florida Coastal Plain (Circular 1151)
Hudson River Basin (Circular 1165)
Kanawha-New River Basins (Circular 1204)
Lake Erie-Lake Saint Clair Drainages (Circular 1203)
Las Vegas Valley Area and the Carson and Truckee River Basins
(Circular 1170)
Lower Illinois River Basin (Circular 1209)
Long Island-New Jersey Coastal Drainages (Circular 1201)
Lower Susquehanna River Basin (Circular 1168)
Mississippi Embayment (Circular 1208)
Ozark Plateaus (Circular 1158)
Potomac River Basin (Circular 1166)
Puget Sound Basin (Circular 1216)

Red River of the North Basin (Circular 1169)
Rio Grande Valley (Circular 1162)
Sacramento River Basin (Circular 1215)
San Joaquin-Tulare Basins (Circular 1159)
Santee River Basin and Coastal Drainages (Circular 1206)
South-Central Texas (Circular 1212)
South Platte River Basin (Circular 1167)
Southern Florida (Circular 1207)
Trinity River Basin (Circular 1171)
Upper Colorado River Basin (Circular 1214)
Upper Mississippi River Basin (Circular 1211)
Upper Snake River Basin (Circular 1160)
Western Lake Michigan Drainages (Circular 1156)
White River Basin (Circular 1150)
Willamette Basin (Circular 1161)

National Assessments

The Quality of Our Nations' Waters—Nutrients and Pesticides (Circular 1225)

Front cover: Clear Creek, Tennessee (courtesy of National Park Service, Obed Wild and Scenic River).
Back cover: Tobacco in Cocke County, Tennessee (photograph by G.C. Johnson, U.S. Geological Survey); center, Whitewater rafting (photograph courtesy of National Park Service, Obed Wild and Scenic River); right, French Broad River Valley, North Carolina (photograph by P.S. Hampson, U.S. Geological Survey).

R0024277

Water Quality in the Upper Tennessee River Basin, Tennessee, North Carolina, Virginia, and Georgia 1994–98

By Paul S. Hampson, M.W. Treece, Jr., Gregory C. Johnson, Steven A. Ahlstedt,
and Joseph F. Connell

U.S. DEPARTMENT OF THE INTERIOR
GALE A. NORTON, SECRETARY

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

2000

Free on application to the
U.S. Geological Survey
Information Services
Box 25286 Federal Center
Denver, CO 80225

Or call: 1-888-ASK-USGS

Library of Congress Cataloging-in-Publications Data

Water quality in the Upper Tennessee River Basin, Tennessee, North Carolina, Virginia, and Georgia, 1995–98 /
by Paul S. Hampson...[et al.].

p. cm. -- (U.S. Geological Survey Circular ; 1205)

Includes bibliographical references.

ISBN 0-607-95546-5 (alk. paper)

1. Water quality--Tennessee River Watershed. I. Hampson, Paul S. II. Geological Survey (U.S.)
III. Series.

TD223.5 .W388 2000
363.739'42'09768--dc21

00-049460

R0024279

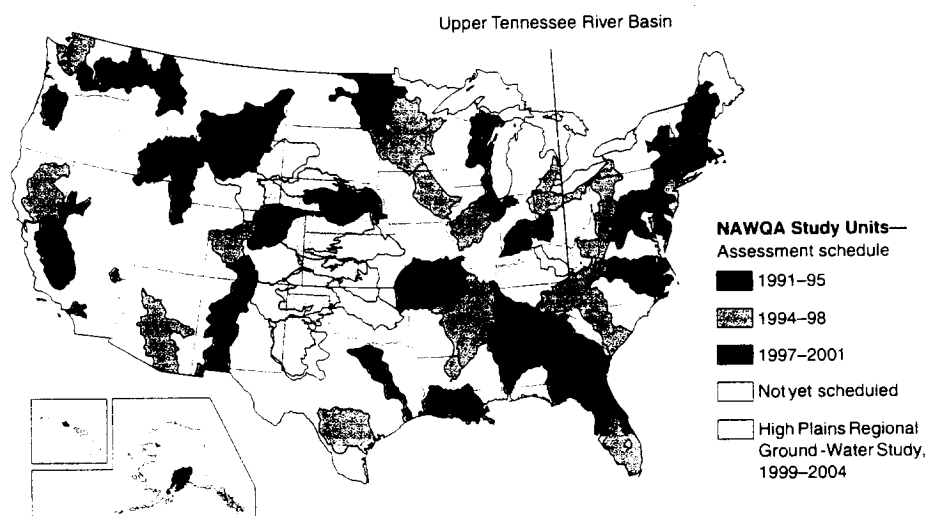
CONTENTS

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM.....	IV
SUMMARY OF MAJOR FINDINGS.....	1
Surface-Water Highlights	1
Ground-Water Highlights.....	2
INTRODUCTION TO THE UPPER TENNESSEE RIVER BASIN.....	3
MAJOR FINDINGS	7
Bacteria in the Upper Tennessee River Basin	7
Nutrients in the Upper Tennessee River Basin	9
NATIONAL PERSPECTIVE— Nutrient Concentrations in Upper Tennessee River Surface Waters Generally Are Lower Than National Median Concentrations.....	11
NATIONAL PERSPECTIVE— Nutrient Concentrations Generally Are Low in Upper Tennessee River Ground Water.....	13
Pesticides in the Upper Tennessee River Basin	14
NATIONAL PERSPECTIVE— Some Pesticides Were Detected More Frequently in the Upper Tennessee River Basin Than Nationally	16
Volatile Organic Compounds in Upper Tennessee River Ground Water.....	18
NATIONAL PERSPECTIVE— Upper Tennessee River Basin and National VOC Detection Frequencies Are Similar.....	18
Water-Quality Influences of Industry and Mining	19
Freshwater Mussels in the Clinch and Powell Rivers	21
NATIONAL PERSPECTIVE— Upper Tennessee River Biological Communities	21
Toxic Spills and Releases	22
STUDY UNIT DESIGN	23
GLOSSARY	25
REFERENCES	26
APPENDIX—WATER-QUALITY DATA FROM THE UPPER TENNESSEE RIVER BASIN IN A NATIONAL CONTEXT	27

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

THIS REPORT summarizes major findings about water quality in the Upper Tennessee River Basin that emerged from an assessment conducted between 1994 and 1998 by the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program. Water quality is discussed in terms of local and regional issues and compared to conditions found in all 36 NAWQA study areas, called Study Units, assessed to date. Findings are also explained in the context of selected national benchmarks, such as those for drinking-water quality and the protection of aquatic organisms. The NAWQA Program was not intended to assess the quality of the Nation's drinking water, such as by monitoring water from household taps. Rather, the assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water monitoring programs. The comparisons made in this report to drinking-water standards and guidelines are only in the context of the available untreated resource. Finally, this report includes information about the status of aquatic communities and the condition of in-stream habitats as elements of a complete water-quality assessment.

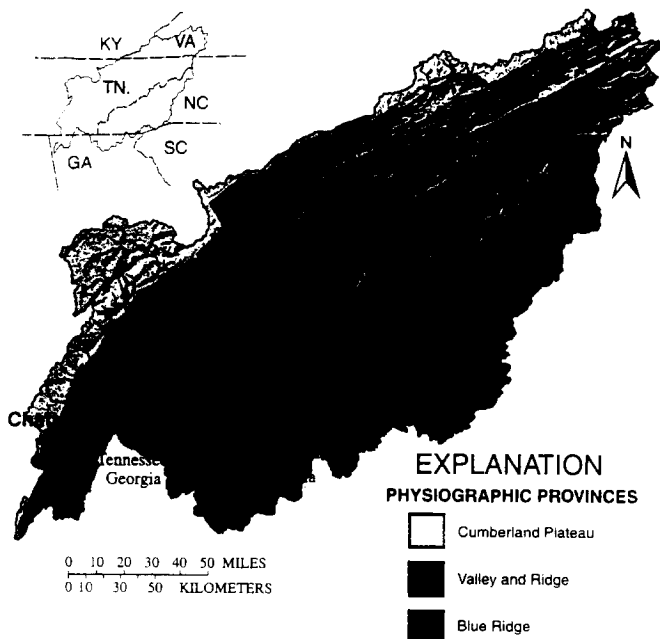
Many topics covered in this report reflect the concerns of officials of State and Federal agencies, water-resource managers, and members of stakeholder groups who provided advice and input during the Upper Tennessee River Basin assessment. Basin residents who wish to know more about water quality in the areas where they live will find this report informative as well.



THE NAWQA PROGRAM seeks to improve scientific and public understanding of water quality in the Nation's major river basins and ground-water systems. Better understanding facilitates effective resource management, accurate identification of water-quality priorities, and successful development of strategies that protect and restore water quality. Guided by a nationally consistent study design and shaped by ongoing communication with local, State, and Federal agencies, NAWQA assessments support the investigation of local issues and trends while providing a firm foundation for understanding water quality at regional and national scales. The ability to integrate local and national scales of data collection and analysis is a unique feature of the USGS NAWQA Program.

The Upper Tennessee River Basin Study Unit is one of 51 water-quality assessments initiated since 1991, when the U.S. Congress appropriated funds for the USGS to begin the NAWQA Program. As indicated on the map, 36 assessments have been completed, and 15 more assessments will conclude in 2001. Collectively, these assessments cover about one-half of the land area of the United States and include water resources that are available to more than 60 percent of the U.S. population.

SUMMARY OF MAJOR FINDINGS



The Upper Tennessee River Basin encompasses about 21,390 square miles and includes parts of four States: Tennessee, North Carolina, Virginia, and Georgia. Three major physiographic provinces are represented in the basin: the Cumberland Plateau, Valley and Ridge, and Blue Ridge Provinces. Most of the 2.4 million people residing in the basin live in the four metropolitan areas of Knoxville and Chattanooga, Tennessee; Asheville, North Carolina; and the Tri-Cities area of Tennessee and Virginia.

Surface-Water Highlights

The Upper Tennessee River Basin is characterized by an abundance of surface water that usually meets existing guidelines for drinking-water supply, recreation, and the protection of aquatic life. Bacteria levels, however, frequently exceed State standards for contact recreation both in agricultural and urban areas. In addition, mixtures of pesticides were detected at 67 of the 74 stream sites sampled. No pesticide concentrations exceeded drinking-water standards, but standards have not been determined for 11 of the 31 compounds detected.

- Bacteria levels frequently exceeded State standards in agricultural streams and streams in urban areas. Runoff from pasture land and direct livestock access to streams contribute to elevated bacterial counts in agricultural streams. Aging wastewater infrastructures are the most likely cause of elevated bacteria counts in urban streams.
- Inputs from urban and agricultural land uses have increased nutrient levels in streams. Yields of total nitrogen in streams are correlated to agricultural inputs, such as animal waste and fertilizer applications, whereas yields of total phosphorus are correlated with wastewater discharges. Tributary reservoirs serve as effective sinks for both nitrogen and phosphorus species in the basin.

- Herbicides and herbicide degradates were detected in 98 percent of the 428 total stream-water samples collected but at levels within drinking-water standards and aquatic-life guidelines. Insecticides used on agricultural fields, gardens, and lawns were detected infrequently (less than 12 percent of samples) and were at levels within drinking-water standards. Concentrations exceeding aquatic-life guidelines were observed, however, for carbaryl, diazinon, and lindane.
- Contamination from previous industrial and mining activities persists in parts of the basin resulting in fish-consumption advisories for PCB's (polychlorinated biphenyls), dioxin, and mercury in certain reservoirs and stream reaches. SVOC (semivolatile organic compounds) sediment concentrations exceeding aquatic-life guidelines were detected in some stream reaches draining coal mining areas.
- The Upper Tennessee River Basin is widely known for its aquatic diversity of fish and mussel species. While mussel populations are recovering in some parts of the basin, overall diversity is slowly declining.
- Releases and spills resulting in fish and mussel kills have occurred in many parts of the basin and pose a threat to isolated and endangered populations of aquatic species.

Selected Indicators of Stream-Water Quality

	Small Streams			Major Rivers
	Mixed Land Use	Agricultural	Forest	Mixed Land Use
Pesticides ¹	●	●	●	●
Phosphorus ²	●	●	●	●
Trace elements ³	●	●	●	●
Organochlorine compounds ⁴	●	●	●	●
Volatile organic compounds ⁵	—	●	—	—
Bacteria	●	●	●	●
Semivolatile organic compounds ⁶	●	●	●	●

● Percentage of samples with concentrations equal to or greater than a health-related national guideline for drinking water, aquatic life, or water-contact recreation; or below a national goal for preventing excess algal growth

● Percentage of samples with concentrations less than a health-related national guideline for drinking water, aquatic life, or water-contact recreation; or below a national goal for preventing excess algal growth

— Percentage of samples with no detection (^a Percentage is 1 or less and may not be clearly visible)

— Not assessed

¹ Insecticides, herbicides, and pesticide metabolites, sampled in water.

² Total phosphorus, sampled in water.

³ Arsenic, mercury, and metals, sampled in sediment.

⁴ Organochlorine compounds including DDT and PCBs, sampled in sediment.

⁵ Solvents, refrigerants, fumigants, and gasoline compounds, sampled in sediment.

⁶ Miscellaneous industrial chemicals and combustion by-products, sampled in sediment.

Trends in Stream-Water Quality

Because of water-treatment improvements, nitrogen and phosphorus levels for most of the streams in the Upper Tennessee River Basin remained unchanged or decreased from 1970 to 1993. Nitrogen concentrations, however, increased significantly for many streams in the Blue Ridge physiographic province because of nonurban residential development and aquaculture.

Trends in other water-quality constituents are difficult to assess because of changes in data-collection methods over time and an overall lack of data. Persistent organochlorine compounds such as DDE, a breakdown product of DDT, which was discontinued in 1973, and chlordane, which was discontinued in 1988, are still detected in fish tissues and bottom sediments in various parts of the basin.

Major Influences on Surface Waters

- Runoff from agricultural and urban areas
- Effluent from wastewater-treatment facilities
- Persistent sediment contamination
- Episodic spills and toxic releases

Ground-Water Highlights

Although ground-water use accounts for a little more than 3 percent of the total water use in the basin, over one-third of the population relies upon ground-water sources for drinking water. In the Upper Tennessee River Basin, ground-water studies focused on the carbonate rock formations of the Valley and Ridge physiographic province, which compose the most prolific aquifers in the basin and are the most susceptible to contamination. These aquifers typically provide water that meets all Federal and State drinking-water standards with the exceptions of nitrate and bacteria. Nitrate concentrations in domestic wells and springs used as drinking-water sources were within drinking-water standards and guidelines. Levels of nitrate exceeding drinking-water standards were detected only in shallow agricultural monitoring wells. Numerous pesticides and volatile organic compounds were detected in wells and springs, but none exceeded drinking-water standards.

- Bacteria levels exceeding finished drinking-water standards were detected in 11 of 30 wells used for untreated domestic drinking-water supply and in all 35 springs sampled. Bacteria levels in two springs exceeded State standards for recreation. Seventeen of the springs sampled are used for untreated drinking-water supplies.
- Nitrate was present in all domestic wells and springs but usually in concentrations well within the Federal drinking-water standard. Five of 30 monitoring wells that were installed adjacent to burley tobacco fields contained nitrate concentrations exceeding the drinking-water standard.

- Pesticides were detected in 40 percent of the agricultural wells, 43 percent of domestic water-supply wells, and 69 percent of the springs in relatively low concentrations. No pesticide concentrations exceeded drinking-water standards; however, 5 of the 18 compounds detected currently do not have standards. The most frequently detected pesticides were atrazine and metalaxyl (tobacco-specific) in the agricultural wells and atrazine, tebutiuron, and prometon in domestic wells and springs.
- Volatile organic compounds were detected in 86 percent of the springs and 67 percent of the domestic wells sampled. Trichloromethane was the most frequently detected compound of the 28 volatile organic compounds that were detected; but carbon disulfide, propanone, and methylbenzene generally were detected in the highest concentrations. None of the volatile organic compounds exceeded drinking-water standards or guidelines, but only 12 of the 28 currently have standards.

Major Influences on Ground Water

- Agricultural and urban land uses
- Permeability of soils and aquifer materials
- Bedrock fracture patterns and karst features

Selected Indicators of Ground-Water Quality

	Agricultural Shallow Ground Water	Domestic Supply Wells	Springs
Pesticides ¹			
Nitrate ²			
Radon ³			
Volatile organic compounds ⁴			
Bacteria			

Percentage of samples with concentrations equal to or greater than a health-related national guideline for drinking water

Percentage of samples with concentrations less than a health-related national guideline for drinking water

Percentage of samples with no detection

Not assessed

¹ Insecticides, herbicides, and pesticide metabolites, sampled in water.

² Nitrate (as nitrogen), sampled in water.

³ Radon, sampled in water.

⁴ Solvents, refrigerants, fumigants, and gasoline compounds, sampled in water.

INTRODUCTION TO THE UPPER TENNESSEE RIVER BASIN

The Upper Tennessee River Basin Study Unit encompasses about 21,390 square miles and includes the entire drainage area of the Tennessee River and its tributaries upstream from the USGS gaging station at Chattanooga, Tennessee. The study area includes parts of four States: Tennessee (11,500 square miles), North Carolina (5,480 square miles), Virginia (3,130 square miles), and Georgia (1,280 square miles). In 1990, the total population of the study area was about 2.4 million, of which about 1.6 million resided in the four metropolitan statistical areas of Chattanooga and Knoxville, Tennessee; Asheville, North Carolina; and the Tri-Cities area of Kingsport and Johnson City, Tennessee, and Bristol, Tennessee and Virginia.

Parts of three physiographic provinces—the Cumberland Plateau, Valley and Ridge, and Blue Ridge Provinces—compose the Upper Tennessee River Basin. Altitudes range from 621 feet above sea level at Chattanooga to 6,684 feet at Mount Mitchell, which is

just northeast of Asheville, North Carolina, and is the highest point in the Eastern United States. The Study Unit contains some of the most rugged terrain in the Eastern United States, including the Great Smoky Mountains range. The crest of the Smoky Mountains exceeds 5,000 feet for 34 miles along the Tennessee-North Carolina State line, has 16 peaks that exceed 6,000 feet, and is the most massive mountain range east of the Mississippi River.

The region generally has a temperate climate; temperatures and annual precipitation totals largely are dependent on land-surface elevations. Average annual temperatures in the area generally decrease by about 3 degrees Fahrenheit for every 1,000-foot increase in elevation. Average annual precipitation ranges from about 40 inches in some low-lying, sheltered areas in the Valley and Ridge province to more than 90 inches at elevations over 6,000 feet. Precipitation generally is distributed evenly throughout the year with no distinct dry and wet seasons.⁽¹⁾

Forests cover more than 67 percent of the Study Unit (fig. 1) and five National Forests—Jefferson, Pisgah, Cherokee, Nantahala, and Chattahoochee National Forests—wholly or partially lie within the basin. Agricultural land, predominantly pasture, is the second most common land use and accounts for more than 26 percent of the study area. Row crops account for only about 2.6 percent of the study area. Most of the agricultural land is located in the stream valleys and gently rolling parts of the Valley and Ridge physiographic province. The crests of steep ridges and more rugged areas of the basin remain forested. Less than 4.5 percent of the basin is developed. Row crops and developed areas, however, generally affect water-quality conditions much more than their small percentages would indicate.



Forest is the predominant land use in the Upper Tennessee River Basin.



Pasture is the predominant agricultural land use in the Upper Tennessee Basin.



Row crops account for only 2.6 percent of the Upper Tennessee River Basin.



Urban and industrial land uses have greater water-quality effects than their land-use percentages might indicate.

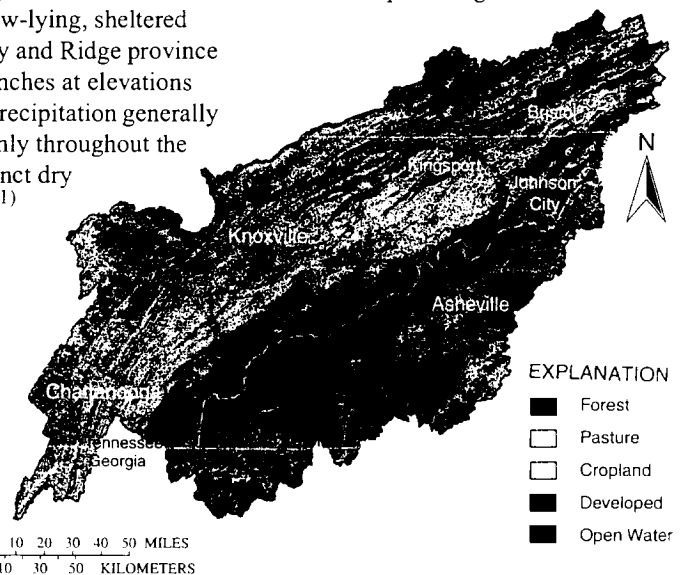


Figure 1. Water-quality conditions in the Upper Tennessee River Basin are influenced by land uses.

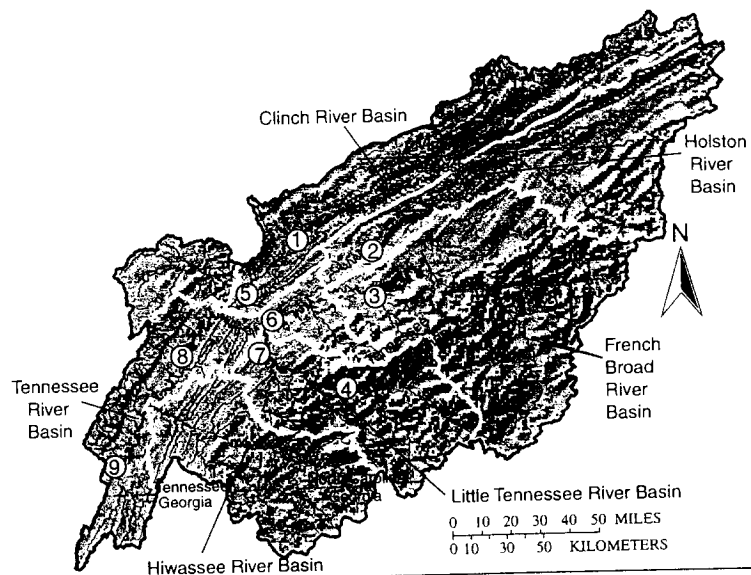
Surface-Water Features

The most prominent surface-water features of the Upper Tennessee River Basin are the tributary and main-stem reservoirs constructed and maintained by the Tennessee Valley Authority (TVA) and sometimes referred to as the "Great Lakes of the South." Four main-stem reservoirs are primarily flow-through systems that provide power generation and maintain navigational depths but provide little flood storage. These four reservoirs have a combined capacity of about 3.1 million acre-feet. Seventeen tributary reservoirs provide flood storage and power generation. These tributary reservoirs have a combined storage capacity of some 10 million acre-feet. An additional 17 privately owned and operated reservoirs also are located in the study area and have a combined storage capacity of about 0.6 million acre-feet.⁽²⁾

Five major tributaries (fig. 2) account for about 86 percent of the annual mean discharge of 35,450 cubic feet per second at the Tennessee River at Chattanooga and over 87 percent of the total area of the upper Tennessee River Basin. The Clinch (4,413 square miles), Holston (3,776 square miles), French Broad (5,124 square miles), Little Tennessee (2,627 square miles), and Hiwassee (2,700 square miles) Rivers each exhibit distinctive climatic and runoff characteristics. Average annual precipitation in these river basins ranges from about 45 inches in the Holston River Basin to almost 60 inches in the Little Tennessee River Basin, which receives the highest rainfall in the continental United States outside of the Puget Sound area of



Large reservoirs are the most prominent surface water features of the Upper Tennessee Basin.



Reservoir number	Reservoir name	Reservoir type	Surface area, in acres	Total capacity, in acre-feet
1	Norris	Tributary storage	34,200	969,000
2	Cherokee	Tributary storage	30,300	580,300
3	Douglas	Tributary storage	30,400	631,200
4	Fontana	Tributary storage	10,640	476,900
5	Melton Hill	Flow-through	5,960	16,100
6	Fort Loudon	Flow through	14,600	120,000
7	Tellico	Flow through	15,860	63,800
8	Watts Bar	Flow through	39,000	191,000
9	Chickamauga	Flow through	35,000	175,000

Figure 2. Two types of major reservoirs are on five major tributaries of the Upper Tennessee River.

Washington State.⁽³⁾ Average annual runoff totals have similar variations and range from about 18 inches in the Holston River Basin to more than 34 inches in the Little Tennessee River Basin.⁽⁴⁾

Water Use

In 1995, withdrawals of surface and ground water in the Upper Tennessee River Basin totaled about 4.8 billion gallons per day. Surface-water withdrawals for once-through cooling at thermoelectric plants accounted for about 3.5 billion gallons per day, or 73 percent of this total. Other uses (fig. 3) were commercial and industrial, 702 million gallons per day; public and domestic supply, 394 million gallons per day; agricultural, 203.3 million gallons per day; and mining, 10.4 million gallons per day, all of which were predominantly surface-water withdrawals.⁽⁵⁾ A total of 897 facilities were permitted to discharge wastewater in 1995 to area streams.

Total ground-water withdrawals in the basin for 1995 were about 138 million gallons per day and accounted for about 10.5 percent of the total non-thermoelectric water use in the basin. About 77 percent of the ground-water withdrawals were for public and domestic supply for over one-third of the basin's population.

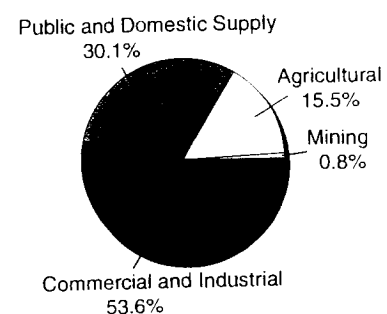


Figure 3. Nonthermoelectric water use in the Upper Tennessee River Basin, 1995. (Thermoelectric water use accounted for 73 percent of the total water use.)

Hydrologic Conditions

Understanding hydrologic variations over time is necessary for assessing water-quality conditions as well as for providing a context with which to evaluate trends. Overall, rainfall during the data-collection period was about 10 percent greater than the long-term mean values. Most of the excess rainfall occurred in the northern part of the basin, as the Knoxville and Tri-Cities weather stations both recorded about 4 inches per year more than their long-term averages of 46.7 and 41.3 inches, respectively. During this same period, rainfall at Chattanooga averaged only about 1 inch per year more than the long-term average of 53.3 inches.⁽⁶⁾

Although precipitation usually is distributed relatively evenly throughout the year in the Upper Tennessee River Basin with no pronounced dry or wet seasons, two relatively dry periods occurred in the late summer and fall of 1997 and 1998. These periods are reflected in the rainfall departures in figure 4 and streamflow discharges in figure 5.

Ground-Water Resources

Ground water in the Upper Tennessee River Basin occurs almost exclusively in unconfined water-table conditions with no regional flow systems. Ground-water flow systems usually are less than 10 square miles in areal extent and are largely controlled by the bedrock geology (fig. 6) and thickness of overlying regolith.

The Cumberland Plateau is characterized by hard, relatively impermeable sandstone of Pennsylvanian age generally overlain by thin soils. Well yields generally range from 5 to 50 gallons per minute from fractures, faults, and bedding-plane openings. Over much of the province, however, reliable ground-water supplies are not obtainable. Similarly, the Blue Ridge physiographic province is characterized by fractured crystalline igneous and metamorphic rock of low porosity and little storage capacity.

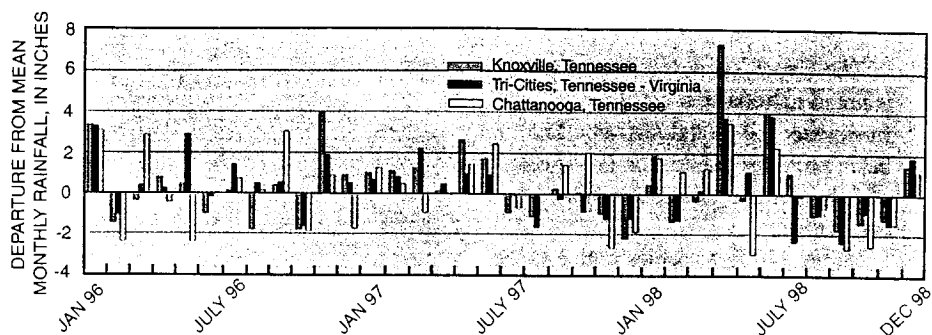


Figure 4. Departures from mean monthly rainfall at three stations in the Upper Tennessee River Basin reflect hydrologic conditions during the 1996-98 study period. (Data from National Weather Service, Morristown, Tenn.)

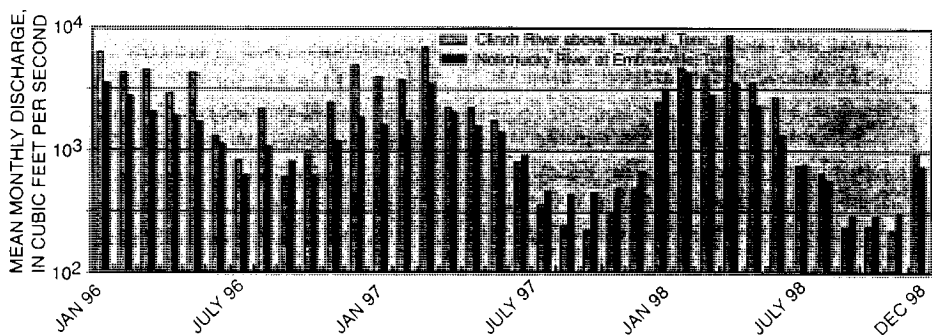


Figure 5. Mean monthly discharge for the Clinch and Nolichucky Rivers reflect the abnormally dry summers of 1997 and 1998.

Well yields depend upon interception of water-bearing fracture systems and usually range from 10 to 25 gallons of water per minute where available.

The Valley and Ridge physiographic province is underlain by folded and extensively faulted limestone, dolomite, shale, and sandstones that occur in long subparallel belts trending southwest to northeast. The principal water-bearing units are the carbonate-based dolomites and limestones, which provide water for many cities and industries. Yields generally range from

5 to 200 gallons per minute, but wells penetrating extensive solution features may yield as much as 2,000 gallons per minute.⁽⁷⁾ Solution features, such as caves and sinkholes with their inherent permeability, make the Valley and Ridge carbonate aquifers the most susceptible in the basin to contamination.

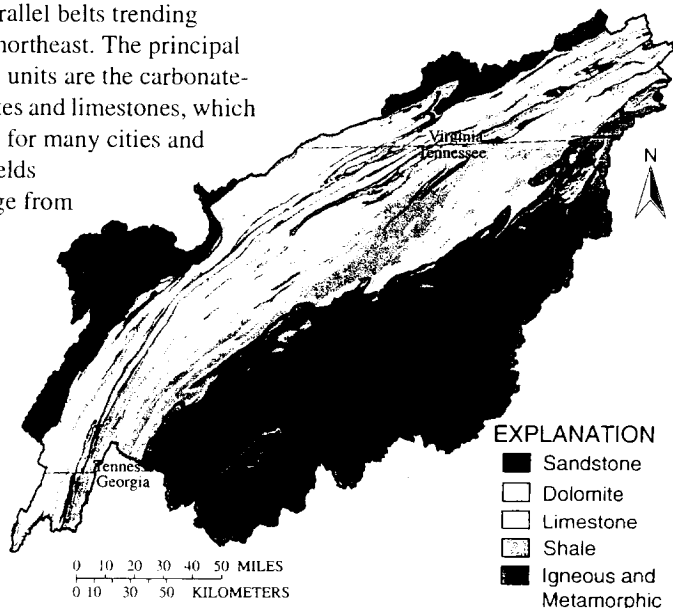


Figure 6. Ground-water availability is a function of surface geology in the Upper Tennessee River Basin.

Biological Diversity

The Upper Tennessee River Basin is noted nationally for its diversity of freshwater fishes and mussels. The basin provides habitat for 174 species of fish, including 25 species that are non-native.

Of the 149 fish species native to the Upper Tennessee River, 29 are found only in the Tennessee and adjacent Cumberland River Basins, and 15 are found only in the Upper Tennessee River. Fifteen fish species in the basin are federally listed as endangered or threatened and 50 species are listed under management categories used by the four States.

Most of the fish diversity in the basin is concentrated in the Valley and Ridge physiographic province, which includes 141 of the 149 native Upper Tennessee species, most notably in the Upper Clinch and lower Holston River Basins (fig. 7). The Clinch River alone is home to 126 Upper Tennessee River native species, 12 of which are federally protected and 41 of which are State listed. Four previously recorded fish species are no longer found in the Clinch River, the largest number of eliminated fish species for any Upper Tennessee drainage.

The Upper Tennessee River also includes one of the most diverse freshwater mussel fauna in the world with 85 different species having historically been recorded. Twenty-five of these species are no longer found in the basin, mostly because of habitat destruction associated with reservoir



The Upper Tennessee River Basin includes one of the world's most diverse freshwater mussel faunas. (Photograph courtesy of Richard Neves, Virginia Polytechnic and State University.)

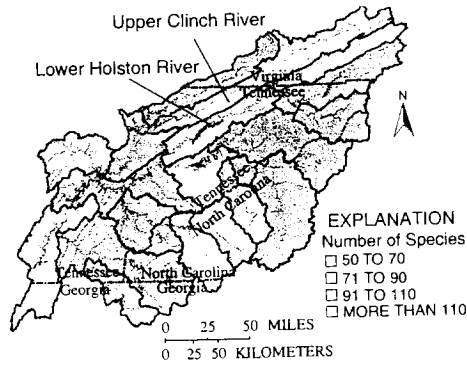


Figure 7. Fish diversity is highest in the Lower Holston and Upper Clinch River systems.

impoundment, and 11 are now believed to be extinct. Of the 60 freshwater mussel species now found in the Upper Tennessee River Basin, 30 species are under Federal protection and 52 species are listed by the States.

As with fishes, most of the freshwater mussel diversity is associated with the Valley and Ridge physiographic province, especially the Clinch River system (fig. 8). The Clinch River is now home to about 52 species of a previously recorded total of 79. Of the current total, 28 are federally listed and 38 are listed by the States.

Home to more than 300 globally rare species, the Upper Clinch River system, which includes the Powell River, has attracted attention from a number of environmental organizations including the designation as one of the "Last Great Places" by the Nature Conservancy. The Clinch River system also is considered to be one of the more biologically threatened river systems in the country (fig. 9). Of the 178 freshwater fish and mussel species presently inhabiting the Clinch River Basin, more than one-fourth are considered to be at-risk.⁽⁸⁾



The yellowfin madtom is one of the threatened fish species in the Upper Tennessee River Basin. (Photograph courtesy of the Tennessee Valley Authority.)

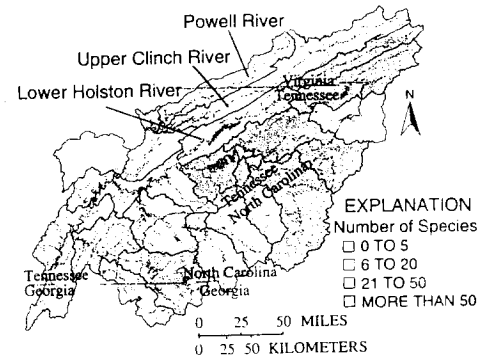


Figure 8. Freshwater mussel diversity is highest in the Valley and Ridge physiographic province.

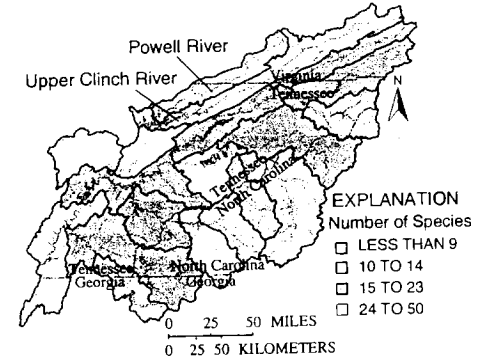


Figure 9. The Upper Clinch and Powell Rivers have the highest numbers of freshwater fish and mussel species considered to be at risk.⁽⁸⁾

Study Unit Design Focuses on Land Use.

Chemical and biological samples were collected from selected rivers and streams draining different land-use areas to assess overall quality as well as the effects of specific land uses. The study focused on agricultural land use and unregulated streams in the Valley and Ridge physiographic province. At Basic Fixed Sites, water samples were collected monthly and during storms to assess runoff conditions. Synoptic sites were sampled only once during periods of average flow.

Springs, domestic wells, and specifically installed agricultural monitoring wells were sampled to assess overall ground-water quality in the basin. Ground-water studies focused on the dolomite and limestone areas of the Valley and Ridge province, which provide the best aquifers and are the most susceptible areas in the basin to ground-water contamination. (See Study Unit Design, page 23, for details.)

MAJOR FINDINGS

Bacteria in the Upper Tennessee River Basin

Fecal indicator bacteria are the most frequent and widespread water-quality standard exceedances involving potential adverse effects to human health in the Upper Tennessee River Basin. The indicator bacteria themselves usually are harmless and easy to detect, but they are indicators of the presence of fecal material and have been shown to be associated with some waterborne disease-causing organisms. The presence of indicator bacteria, however, cannot be considered direct proof of any threat to human health, and research is underway to find better indicators.

Bacterial Counts Frequently Exceed Standards

The State of Tennessee's current water-quality standards are based on a total fecal coliform level of 200 colonies per 100 milliliters of water, as a mean value.⁽⁹⁾ This value is commonly exceeded in agricultural and urban streams in the Upper Tennessee River Basin (fig. 10). In agricultural areas, livestock waste is the most likely bacterial source both from allowing livestock direct access to streams and runoff from animal-waste areas. Bacterial counts generally increase during higher streamflows associated with runoff events in the agricultural areas (fig. 11).

Deteriorated and leaky sewage systems, faulty sewage treatment plants, urban runoff, and combined sewer overflow systems are among the



Livestock are a major contributor to fecal coliform levels in area streams.

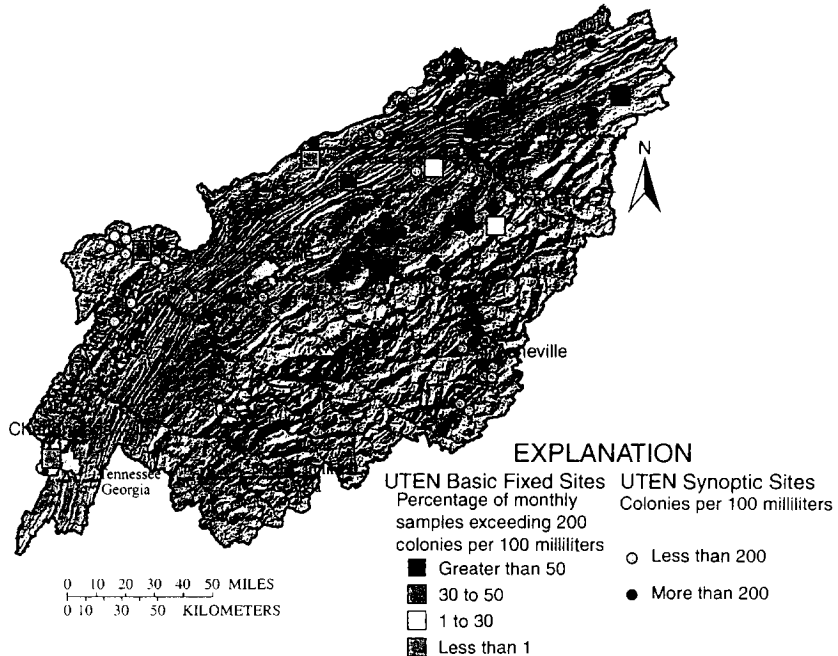


Figure 10. Fecal coliform bacteria frequently exceed standards in Upper Tennessee River (UTEN) streams.

sources of bacterial contamination in many urban streams. For example, all of the urban streams draining the central Knoxville, Tennessee, area regularly exceed bacterial standards⁽¹⁰⁾ because of widespread leakage from very old and deteriorating sewer systems in the older parts of the city. Replacement in 1998 of an obsolete combined sewer overflow system for one city neighborhood, however, has improved conditions for that neighborhood and adjacent parts of Fort Loudon Reservoir. These conditions highlight the continuing need for infrastructure improvements, especially in older urban areas.

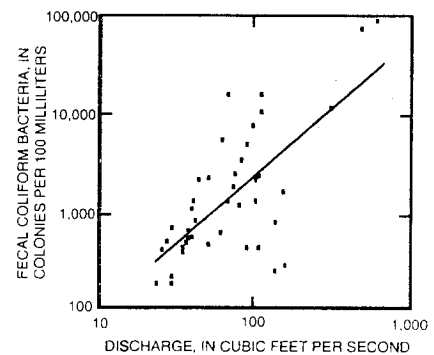


Figure 11. Fecal coliform counts vary with streamflow at Big Limestone Creek in Tennessee.



In Upper Tennessee River Basin urban areas, deteriorated sewerage systems and combined sewer overflows produce elevated fecal coliform levels.

Bacteria Frequently Are Detected in Domestic Wells and Springs

A common misconception is that untreated ground water from wells and springs generally is safe for consumption because percolation through the soil removes most contaminants. While the soil can act as a natural filter, this does not guarantee the absence of contaminants. In fact, about half of the waterborne-disease outbreaks in the United States since 1900 have involved contaminated ground water.⁽¹¹⁾

Ground-water systems such as the carbonate systems of the Upper Tennessee River Basin are particularly susceptible to contamination from surface sources. Ground-water flow paths in these systems usually are shallow, principally involving the upper 10 to 20 feet of highly fractured and heavily weathered rock. In addition, the common presence of bedrock outcrops, areas of thin overburden, and karst features such as sinkholes provide direct avenues for aquifer contamination (fig. 12). Other potential sources for bacterial contamination include faulty or poorly placed septic systems and poor well construction or sanitation practices.

For finished drinking water, the detection of as few as 4 coliform bacteria colonies per 100 milliliters (col/100 mL) or the detection of 1 col/100 mL of fecal coliform bacteria, or *E. coli*, warrants concern for human health.⁽¹²⁾ Of 30 domestic wells used as sources for untreated drinking water, 11 (37 percent) exceeded the total coliform drinking-water standard and 9 (30 percent)

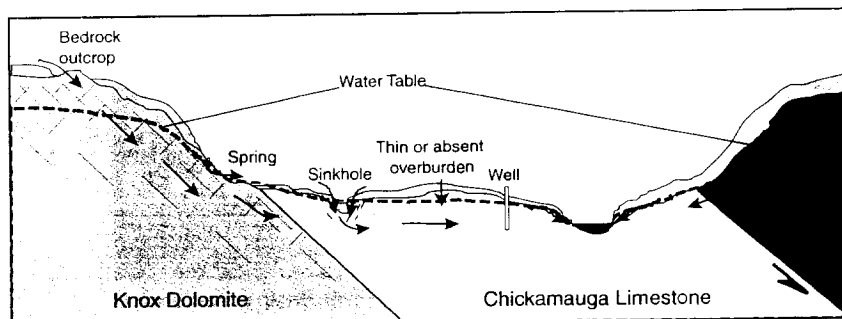


Figure 12. Upper Tennessee ground-water flow systems can be affected by a number of potential contamination sources such as sinkholes, outcrops of bedrock, and areas with thin overburden. (Not to scale)

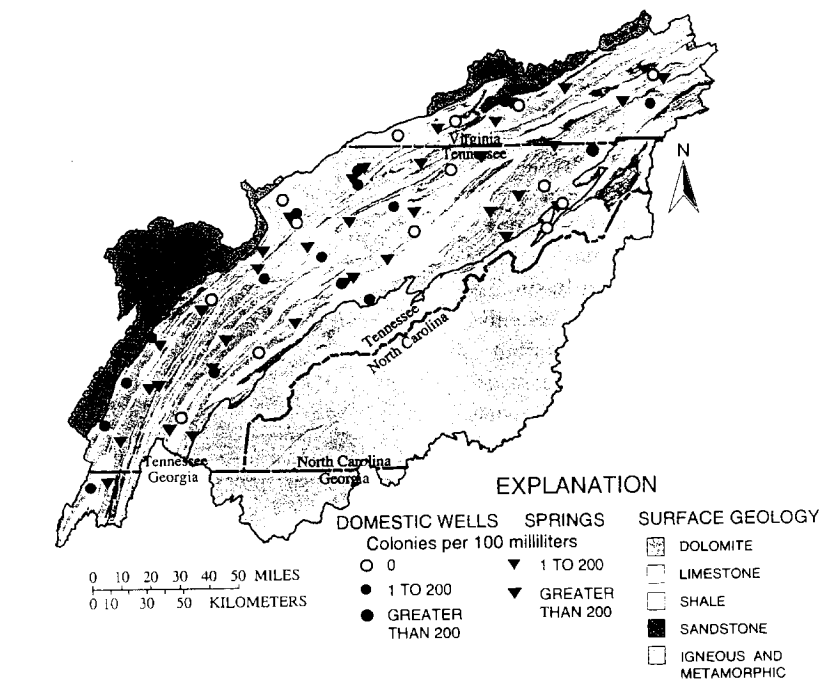
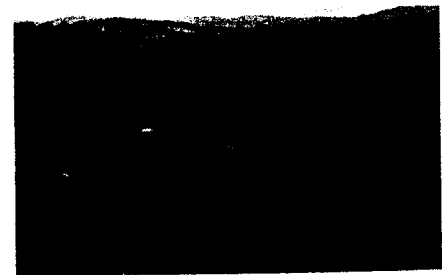


Figure 13. Coliform bacteria are often detected in Upper Tennessee River Basin ground water.

the *E. coli* drinking-water standard (fig. 13). The highest *E. coli* value detected was 1,600 col/100 mL.

Total coliform values for 35 springs sampled in the Upper Tennessee River Basin ranged from 10 to 1,900 col/100 mL and *E. coli* ranged from 0 to 660 col/100 mL. All of the springs tested exceeded drinking-water standards for total coliform bacteria, and 95 percent of the springs exceeded the *E. coli* standard. Two springs exceeded the *E. coli* body-contact standard of 126 col/100 mL. Sixteen of the 35 springs are used as domestic water supplies and others are used for filling water containers by the roadside with what usually is believed to be "clean mountain spring water."



Most of the rural population in the Upper Tennessee River Basin depend on shallow domestic wells for water supply.



Although much of the public perceives them as clean sources of drinking water, springs are very susceptible to contamination.

Nutrients in the Upper Tennessee River Basin

Nutrients are nitrogen and phosphorus compounds that are essential for plant growth. When found at elevated concentrations, however, nutrients can degrade water quality. The enrichment of a water body with nutrients, called eutrophication, can result in dense, rapidly multiplying growths, or blooms, of algal species and other nuisance aquatic plants. These can clog water intake pipes and filters and interfere with recreational activities, such as fishing, swimming, and boating. Subsequent decay of algal blooms can overload water bodies with oxidizable debris and result in foul odors, bad taste, and reduced dissolved oxygen levels, which are harmful to other aquatic life.⁽¹³⁾

Nutrients in the Upper Tennessee River Basin originate from point and nonpoint sources. Point sources are typically piped discharges from wastewater-treatment facilities and large urban and industrial stormwater systems. Nonpoint sources include stormwater runoff from urban and agricultural areas. In the Upper Tennessee River Basin, applications of synthetic fertilizers and manure are major sources.

Nutrient Loadings and Yields Vary among Upper Tennessee River Subbasins

Nutrient loadings in the Upper Tennessee River subbasins are primarily influenced by land use and streamflow conditions. Loads were estimated by using a constituent transport model and multiple regression to relate streamflow to the concentration of a water-quality constituent to derive loads.⁽¹⁴⁾ Twenty-three stations with adequate streamflow and chemical records were used for nitrogen calculations and 20 for total phosphorus.

The highest yields in the study area for both nutrient species were detected in the French Broad River Basin, particularly the upstream portion that includes Asheville, North Carolina (figs. 14 and 15). The French Broad River, as a whole, accounted for about 40 percent of the 138,000 pounds per day (lb/d) average annual total nitrogen load⁽¹⁵⁾ and about 25 percent of the 13,500 lb/d average annual total phosphorus load,⁽¹⁶⁾ leaving the basin at Chattanooga, Tennessee. The Holston River Basin added another 22 percent of the total nitrogen load but only 8 percent of the total phosphorus load.

A combination of agricultural and urban runoff is probably responsible for conditions in the French Broad River. In addition, the French Broad River and its tributaries have a history of water-quality problems associated with industrial point-source discharges. These basins also had the highest yields and loadings in the Upper Tennessee River Basin for total ammonia and organic forms of nitrogen.

Nutrient loadings and yields generally were lowest in those basins with relatively low percentages of agricultural land use and at sites directly downstream from tributary reservoirs. The fate of nutrients in the reservoirs depends on the physical characteristics of the reservoir (volume, surface area, depth, and hydraulic retention time) and its trophic state.⁽¹⁷⁾ The tributary reservoirs in the Upper Tennessee River Basin commonly function as sinks for nutrient species by providing a favorable environment for nitrogen transformation and by efficiently trapping both dissolved and sediment-bound phosphorus. Outflow loads of total phosphorus below Norris Lake on the Clinch River, for example, were 37 percent of the inflow load from the Clinch and Powell River Basins. Load estimates for the Holston River upstream and downstream from

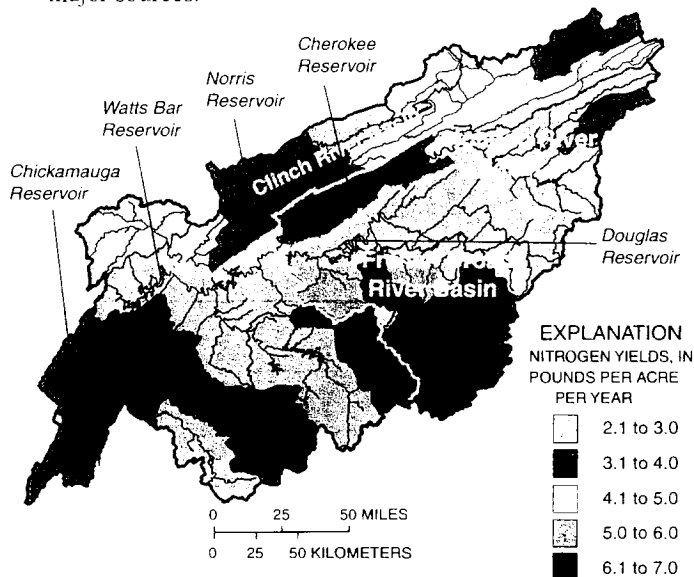


Figure 14. Mean annual total nitrogen yields between 1973 and 1993 were highest in the upper French Broad and upper Clinch River Basins.

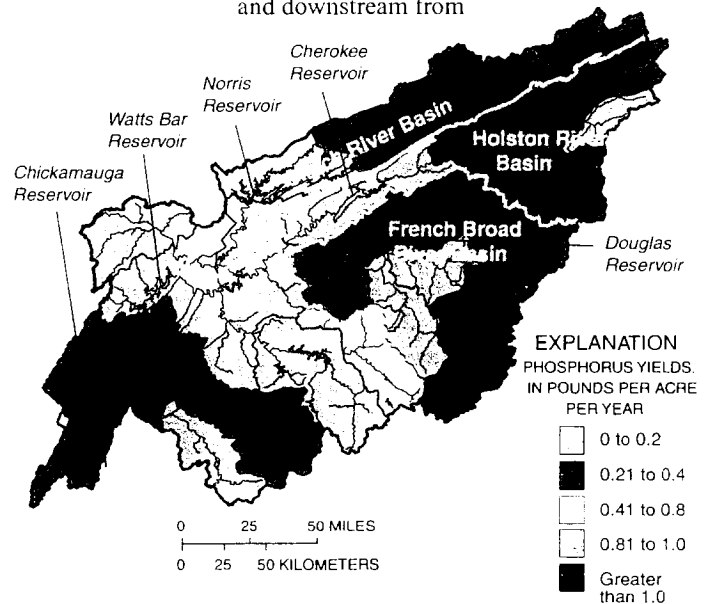


Figure 15. Mean annual total phosphorus yields between 1973 and 1993 were highest in the upper French Broad River Basin.

Cherokee Reservoir similarly indicate that the reservoir traps about 46 percent of the incoming load of total phosphorus. In contrast, less trapping occurs in the main-stem reservoirs, which are predominantly flow-through systems with limited storage capacity and relatively short residence times. Outflow phosphorus loads downstream from Chickamauga and Watts Bar Reservoirs significantly exceeded the inflow loads from upstream drainages. The increased loads can be attributed to low rates of trapping as well as additional input from ungaged areas adjacent to the reservoirs.⁽¹⁶⁾

Nutrient Concentrations and Yields Vary with Land Use

The relation between total nitrogen concentrations and land-use percentages was investigated for 87 sites in the Upper Tennessee River Basin and was found to be statistically significant. Stations in forested watersheds had the lowest concentrations of total nitrogen, whereas stations in agricultural areas had the highest. Concentrations of nitrogen in urban and mixed land-use areas were significantly greater than forested watersheds but were somewhat less than nitrogen concentrations in agricultural watersheds. Total nitrogen concentrations tended to increase with increased development whether agricultural or urban (fig. 16).⁽¹⁵⁾

Nitrogen sources also were investigated by using regression analysis between annual basin yields and total annual inputs from fertilizer, animal waste, wastewater discharges, and atmospheric deposition. For total nitrogen, basin yields significantly and positively correlated with agricultural inputs but only weakly correlated with wastewater discharges and atmospheric inputs. This tends to identify agricultural land use as the major contributor to annual instream nitrogen yields.⁽¹⁸⁾

The relation between total phosphorus concentrations and land-use percentages also were investigated for 83

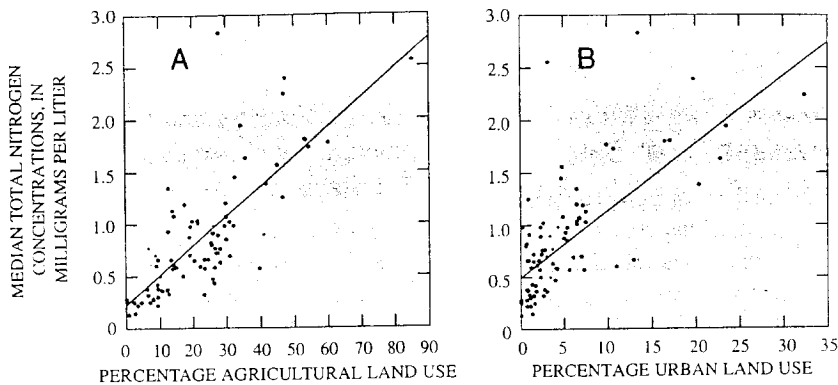


Figure 16. Median total nitrogen concentrations can be related to (A) agricultural, and (B) urban land uses.

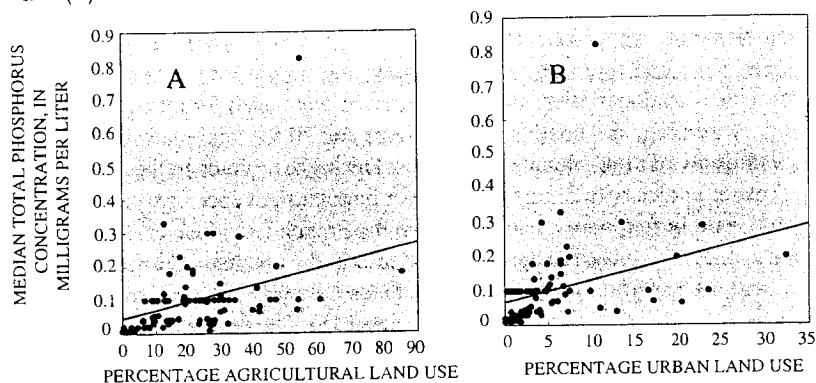


Figure 17. Median total phosphorus concentrations can be related to (A) agricultural and (B) urban land uses.

sites in the Upper Tennessee River Basin. Although the relation was not quite as clear as with nitrogen, statistically significant increases in total phosphorus concentrations also accompanied increased development whether urban or agricultural (fig. 17). As with total nitrogen, the lowest phosphorus concentrations were detected at sites in predominantly forested watersheds, whereas sites in urban and agricultural areas had the highest phosphorus concentrations.⁽¹⁶⁾

Phosphorus sources also were investigated by using calculated basin yields and total annual inputs from fertilizer,



Agricultural land uses appear to account for most of the total nitrogen loads to area streams.

animal waste, wastewater discharges, and the atmosphere. Phosphorus yields were found to strongly correlate with wastewater discharges but not with the agriculturally related input categories. This suggests that wastewater discharges may account for most of the total phosphorus load in basin streams (J.F. Connell, U.S. Geological Survey, written comun., October 20, 2000). Agriculturally applied phosphorus may be assimilated quickly by area soils thereby reaching area streams slowly if at all.



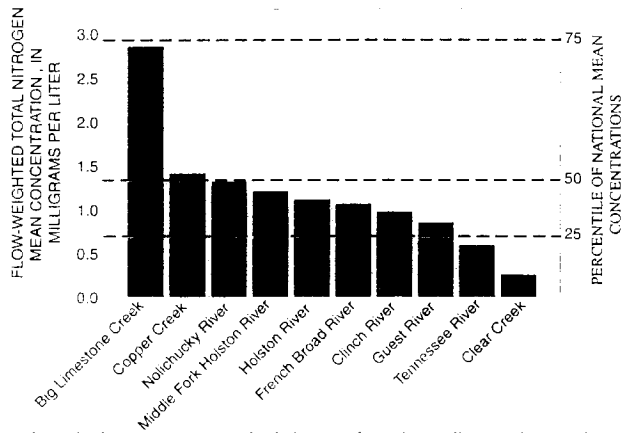
Wastewater discharges appear to account for most of the total phosphorus in Upper Tennessee River Basin streams.



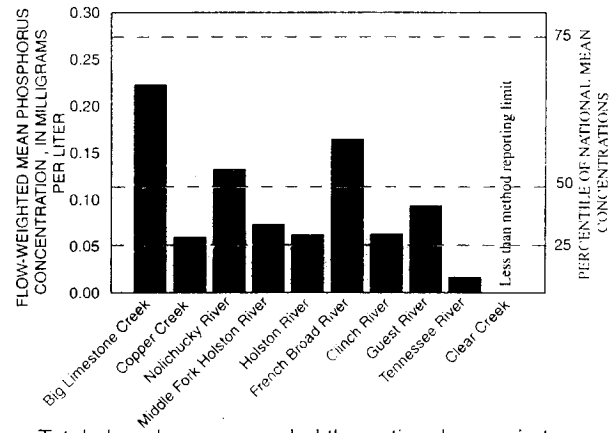
Nutrient Concentrations in Upper Tennessee River Basin Surface Waters Generally Are Lower Than National Median Concentrations

Although nutrient concentrations and loadings are a concern in parts of the Upper Tennessee River Basin, concentrations generally are low for most area subbasins when compared with national averages. Mean total nitrogen concentrations exceeded or equaled the national median values only for three agricultural sites: Big Limestone Creek (83 percent agricultural), Copper Creek (51 percent agricultural), and the Nolichucky River (39 percent agricultural). Similar results

were obtained for total phosphorus at Big Limestone Creek and the Nolichucky River, but the French Broad River flowing into Tennessee from North Carolina also exceeded the national median value. Although relatively low, mean total phosphorus concentrations at most sites exceeded the U.S. Environmental Protection Agency (USEPA) goal of 0.05 mg/L total phosphorus for surface water entering reservoirs.



Total nitrogen exceeded the national median value only at the two intensively sampled agricultural sites.



Total phosphorus exceeded the national mean in two heavily agricultural drainages and also in the French Broad River below the Asheville, N.C., urban area.

Nitrogen Species Changed by Wastewater Treatment

Prior to the widespread implementation of wastewater treatment, nitrogen loadings for most Upper Tennessee River Basin streams primarily consisted of reduced species such as ammonia and various organic forms. These nitrogen species generally are undesirable in surface water because of associated color changes and decreases in dissolved oxygen levels. In addition, under certain conditions, ammonia nitrogen can be highly toxic to aquatic life. Wastewater-treatment facilities convert these undesirable forms to the oxidized species, nitrite and nitrate.

At the Tennessee River at Chattanooga, Tennessee, as with most major streams in the Upper Tennessee River Basin, the ratio of reduced to oxidized nitrogen species began to change in the late 1970s (fig. 18), corresponding to

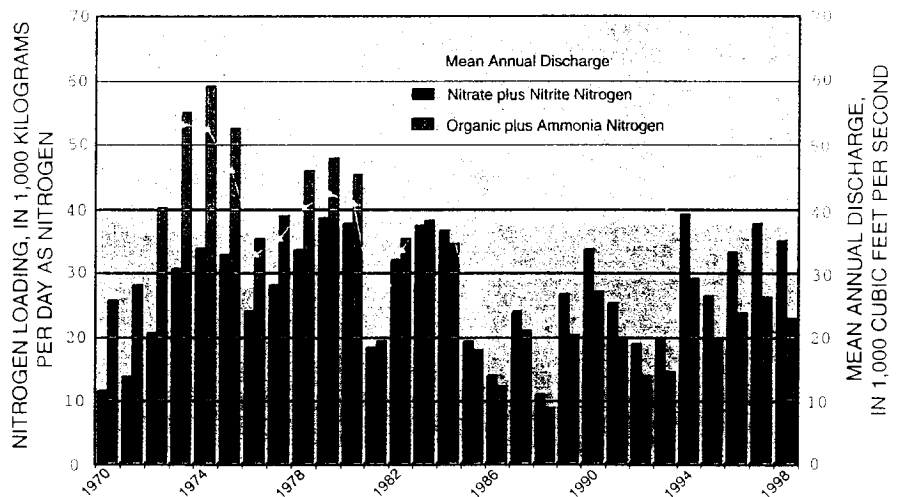


Figure 18. The predominant nitrogen species changed at the Tennessee River at Chattanooga, Tennessee, between 1970 and 1998.

the implementation of wastewater-treatment facilities. By about 1983, the oxidized nitrogen species, nitrate and nitrite, became the predominant forms of nitrogen discharged from the basin, a trend which has continued to the present.

Loading Trends Increase in Parts of the Upper Tennessee Basin

Trend analyses for 56 stations using the seasonal Kendall statistical analysis test indicated significant increases in total nitrogen at seven sites in the Upper Tennessee River Basin and significant decreases at eight sites (fig. 19). Sites showing decreases were all on relatively major streams (average drainage area, 2,600 square miles) or below major impoundments. Of the seven sites showing increases, six are in the Blue Ridge physiographic province and six drain basins with forests accounting for more than 75 percent of the total land use. The exception is Beaver Creek, which drains the Bristol, Tennessee and Virginia, urban area in the Valley and Ridge Province. The average area of basins showing nitrogen increases was only 276 square miles.⁽¹⁵⁾

Of the seven sites showing increases, five are in the Blue Ridge in North Carolina—two sites on the French Broad River and one each on the Little Tennessee River and tributaries to the Hiwassee and Pigeon Rivers. Much of this area is undergoing nonurban residential development in the form of vacation homes. Nitrogen loads are probably increased by the sewage and fertilizer use associated with this development.



Nonurban residential development in the Blue Ridge Mountains is most likely the largest contributor to increasing total nitrogen concentrations.

Similar trend analyses for 42 sites to detect changes in total phosphorus concentrations yielded only one site with significant increases (fig. 20). West Chickamauga Creek, which drains a major industrial and urban setting, showed high concentrations for the entire period of record. Most (33) sites showed no trend, and eight sites

showed significant decreases. These sites are dominated for the most part by pasture and forest; however, three sites are downstream from major wastewater discharges.⁽¹⁶⁾ For sites in these more urbanized basins, improvements in wastewater-treatment processes are clearly responsible for the downward phosphorus trends.

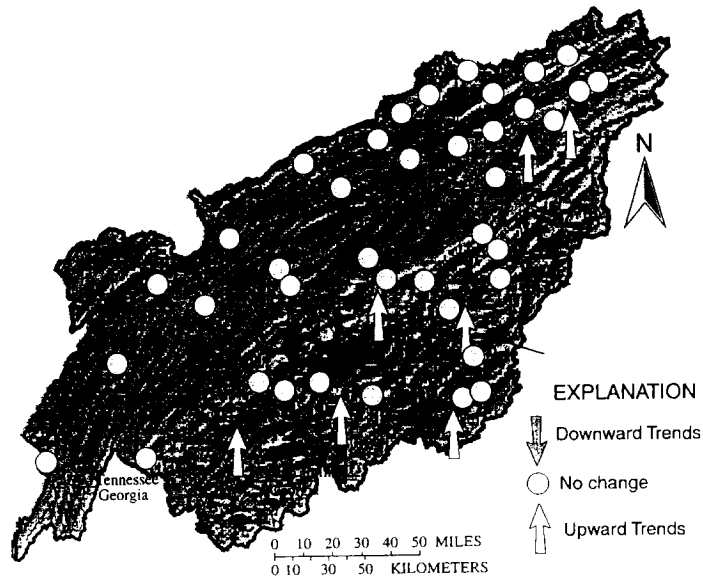


Figure 19. Total nitrogen increased in parts of the Upper Tennessee River Basin between 1970 and 1993.

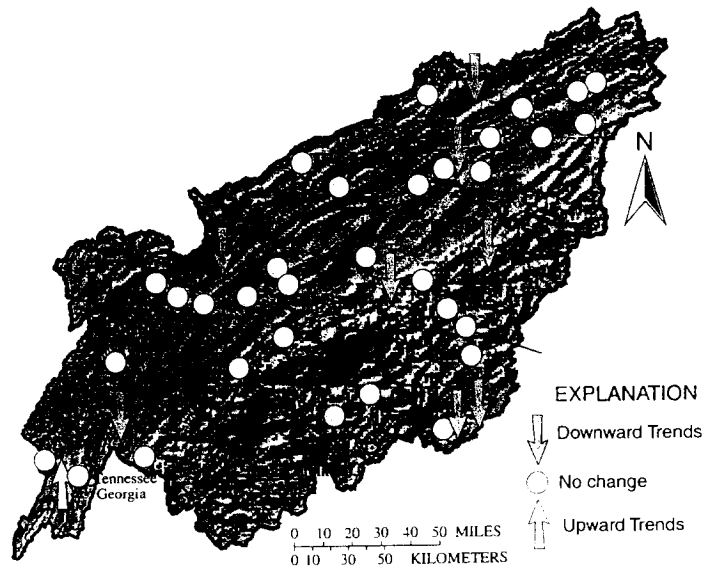


Figure 20. Total phosphorus decreased or remained unchanged in the Upper Tennessee River Basin between 1970 and 1993.



Nutrient Concentrations Generally Are Low in Upper Tennessee River Basin Ground Water

All of the nutrients measured in the Upper Tennessee River Basin ground water were relatively low, as is usually typical of ground water. Most nutrient species are retained by soil particles or organic matter, taken up by plants, or utilized by soil bacteria and never enter the ground-water flow system. Exceptions are the nitrate and ammonia forms of nitrogen; however, only nitrate has a drinking-water standard, which is 10 mg/L. Drinking water containing nitrate concentrations higher than the standard can cause methemoglobinemia, a life-threatening illness in infants.

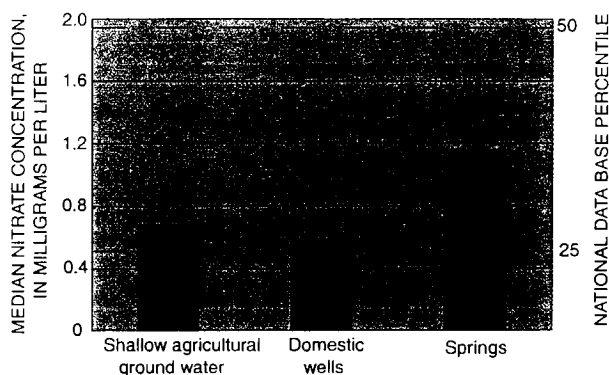
Nitrate was present in all wells and springs sampled in the Upper Tennessee River Basin but usually at concentrations of 3 mg/L or less. This included all of the 30 domestic wells used for drinking-water supply that were sampled and the 35 springs sampled across the basin. The median nitrate concentration for domestic wells was 0.59 mg/L, slightly more than the 25th percentile value nationally; the median nitrate concentration for springs was 1.16 mg/L, which was significantly lower than the national 50th percentile. The higher concentrations detected in springs most likely reflect the predominance of relatively short ground-water flow paths associated with localized recharge and runoff. No nitrate concentrations in excess of the 10-mg/L standard were detected in any domestic wells or springs.

Nitrate concentrations in excess of the 10 mg/L standard were detected in 5 of the 30 wells installed during the study period to monitor shallow ground-water quality under and adjacent to tobacco fields. Tobacco is the main cash crop in the Upper Tennessee River Basin and is usually grown in small but intensively fertilized and cultivated plots. In general, fertilizer applications for tobacco cultivation are much greater than for any other row-type crop raised in the Upper Tennessee River Basin.

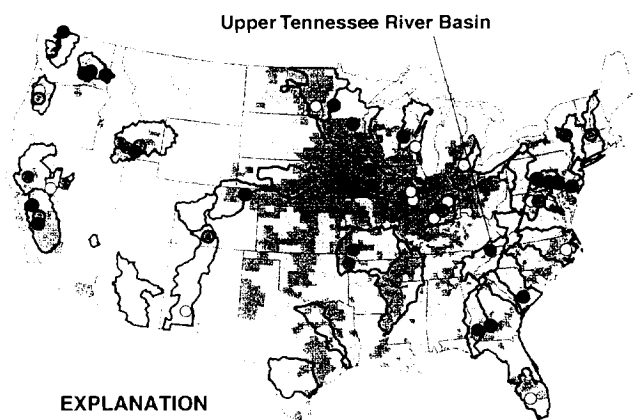
The median nitrate concentration in the shallow agricultural monitoring wells, however, was 0.68 mg/L—only slightly more than the median concentration for domestic wells and the national 25th percentile. Among the concentrations found nationally for agricultural and urban land uses, this value falls in the lower end of the medium range as shown in the accompanying figure.

The results indicate that nitrate contamination of extensive areas of ground water in the Upper Tennessee River Basin is very unlikely. High nitrate concentrations relative to the 10 mg/L drinking-water standard were detected only in shallow ground water directly under heavily fertilized tobacco plots. Tobacco fields typically

cover only about 2 acres and are widely scattered across the study area. The potential for nitrate contamination of drinking-water sources is, therefore, very low outside of the immediate vicinity of tobacco fields.



Median ground-water nitrate concentrations in the Upper Tennessee River Basin were significantly lower than the national median



EXPLANATION

Median concentration of nitrate— in milligrams per liter. Each circle represents a ground-water study

- Highest (greater than 5.1)
- Medium (0.48 to 5.1)
- Lowest (less than 0.48)

Background concentration

- Bold outline indicates median values greater than background concentration (2 milligrams per liter)

Average annual total nitrogen input— in pounds per acre, by county, for 1995-98. Inputs are from fertilizer, manure, and the atmosphere

- Greater than 25 pounds per acre
- 6 to 25 pounds per acre
- Less than 6 pounds per acre

Median nitrate concentrations in shallow ground water beneath agricultural land use in the Upper Tennessee River Basin were in the medium range on a national basis.

Pesticides in the Upper Tennessee River Basin

Pesticides are widely used in the Upper Tennessee River Basin to control insects, fungi, weeds, and other undesirable organisms. These compounds vary in their toxicity, persistence in the environment, and transport characteristics. Use of some of the more persistent organochlorine compounds, such as DDT, chlordane, dieldrin, and aldrin has been discontinued in the United States, but their residues are still detected in the environment. Although pesticides usually are applied to specific areas and directed at specific organisms, these compounds often become widely distributed and pose hazards to nontarget organisms. Of 18 sites sampled for organochlorine residues in bottom material and biota in the Upper Tennessee River Basin, chlordane was detected at three sites and dieldrin and DDT-related residues at two sites.

Pesticides were Frequently Detected in Surface Water

Pesticide use in the Upper Tennessee River Basin is primarily for agricultural purposes. Herbicides, including atrazine and its degradation product, deethylatrazine, had some of the highest application rates and were also among the most frequently detected pesticides in the basin. Herbicides were detected in 98 percent of the 428 surface-water samples collected; atrazine was found in 91 percent and deethyl-

atrazine in 86 percent. Metolachlor and simazine were detected in 62 and 40 percent, respectively. Tebuthiuron and prometon, which are used most commonly in noncrop areas, were also among the most frequently detected herbicides (in 58 and 31 percent of the samples collected, respectively). The most frequently detected insecticides were diazinon (12 percent), carbaryl (10 percent), and chlorpyrifos (10 percent), all of which are used on a variety of crops to control pests.

Detection frequencies for 27 pesticides detected at 3 intensively sampled agricultural sites in the Upper Tennessee River Basin (fig. 21) generally illustrate the results obtained at all 13 Basic Fixed Sites from which surface-water samples were collected. Overall, a total of 32 pesticides were detected. Chlorothalonil, alpha-BHC, and terbacil each were detected once and ethoprop was detected twice.

Some differences among the three sites are notable and probably reflect different agricultural practices and hydrologic conditions. For example, at the Nolichucky River site, compounds generally not found at other sites such as cyanazine, alachlor, DCPA, metribuzin, bromacil, and diazinon, were detected. Molinate, trifluralin, and p,p'-DDE were detected only at the Copper Creek site, which also had a significantly higher frequency of detection for tebuthiuron. Pesticide detection frequencies at Big Limestone Creek and the Nolichucky River were, as expected, similar for several compounds



Pesticides are widely used in the Upper Tennessee Basin for control of insects, fungi, weeds, and other undesirable organisms.

including metolachlor, simazine, prometon, and napropamide. Big Limestone Creek is a tributary to the Nolichucky River, and both drain the same general agriculturally dominated area. The Big Limestone Creek drainage basin, however, contains more dairy operations than other parts of the Nolichucky drainage basin, which may account for some of the differences between the two sites.

Table 1. Major pesticides used in the Upper Tennessee River Basin, listed in order of estimated total pounds of active ingredient applied annually (1991-94) ⁽¹⁸⁾

Insecticides	Herbicides	Fungicides			
Oil	256,000	Atrazine	116,000	Methyl bromide..	423,000
Acephate	80,700	2-4-D	55,600	1-3-D	342,000
Chlorpyrifos.....	71,500	Metolachlor	46,300	Captan.....	108,000
Carbaryl	27,200	Alachlor	40,900	Ziram	69,500
Fenamiphos.....	17,200	Pebulate	31,400	Sulfur	58,700
Carbofuran.....	17,000	Pendimethalin	25,200	Chloropicrin	45,100
Formetanate.....	16,300	Butylate	24,800	Mancozeb	40,400
Azinphos-methyl..	14,400	Simazine.....	23,800	Metalaxyl	28,100
Phosmet.....	9,420	Glyphosate.....	16,100	Manab	21,500

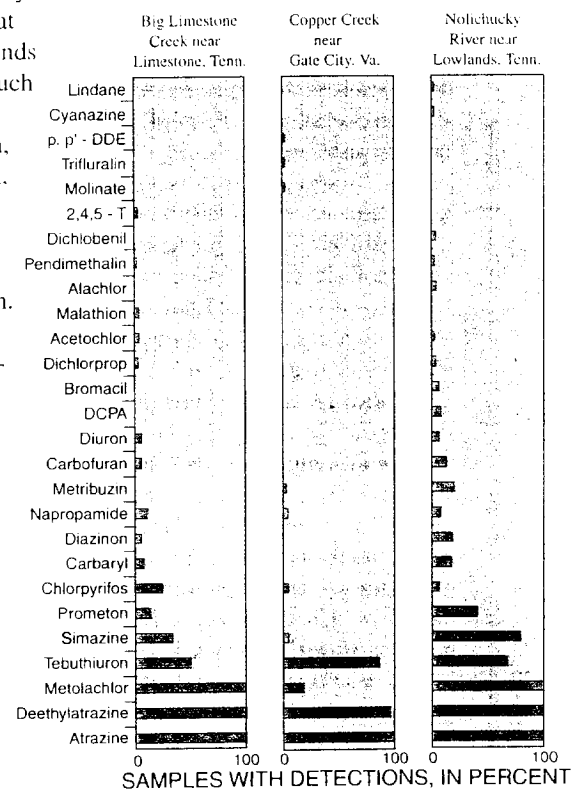


Figure 21. Pesticide detections at three agriculturally dominated sites followed similar patterns.

Mixtures of Pesticides Are Common

Pesticides were seldom detected alone in surface-water samples and usually occurred as mixtures of several compounds. Generally, the effects of pesticide mixtures on biota or humans are not included in water-quality criteria, which are most commonly based on single-species, single-chemical toxicity tests conducted under laboratory conditions. As a result, potential adverse effects on biota may be underestimated.

Of the 163 samples collected at the three intensive sites, only 2 samples at Copper Creek contained only one detectable pesticide compound, and only 5 total samples contained only two compounds (fig. 22). Among the intensively sampled sites, samples from the Nolichucky River at Lowlands, Tennessee, generally contained more detectable pesticide compounds than samples from the other sites, but usually at lower concentrations. This reflects the larger drainage area of the Nolichucky River (1,687 square miles) as compared to the drainage areas of the other intensive sites (79 and 106 square miles for Big Limestone and Copper Creeks, respectively). Similarly, more pesticides also were detected in sam-

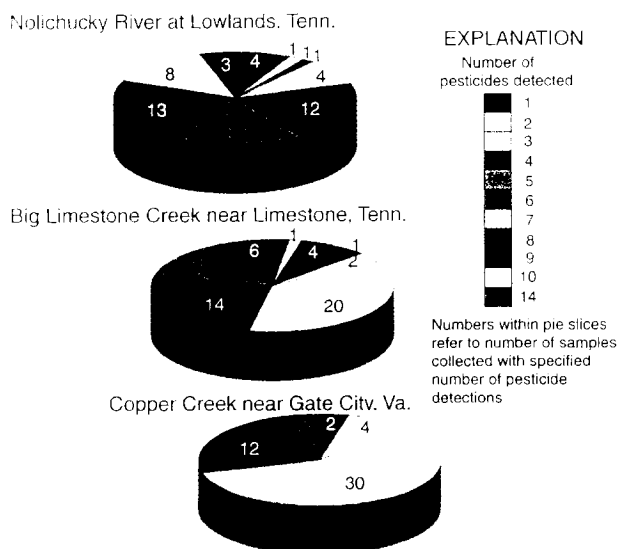


Figure 22. Pesticides usually were detected as mixtures of different compounds.

ples from Big Limestone Creek, which has a larger percentage of agricultural land use and a greater variety of crops than the Copper Creek Basin in Virginia.

Peak Pesticide Concentrations Are Seasonal

Pesticide concentrations were found to be seasonal and closely related to land use. The highest concentrations occurred in the more heavily agricultural basins in late spring and early summer, coinciding with crop applications. Results of weekly sampling results at the three intensively sampled agricultural sites illustrate the seasonality and short-lived nature of the peak concentrations in streams draining agricultural areas (fig. 23). Peak concentrations coincided with the first substantial runoff event following agricultural applications in May 1996, after which concentrations declined relatively rapidly to near-background levels. Less frequent sampling would have made it less likely to

have noted the existence of the peaks. Because these streams are "flashy" in that peak discharges come and go very quickly, it is possible that even higher concentrations can occur for short periods of time. Seasonality also was evident at sites not characterized or directly influenced by intense agricultural activities. Atrazine and metolachlor concentrations at Clear Creek at Lilly Bridge, a predominantly forested watershed and part of the Obed National Wild and Scenic River watershed, also showed a distinct seasonality but with much lower concentrations (fig. 24). The seasonal pattern at this site is more gradual, suggesting atmospheric input more than runoff from agricultural activity.

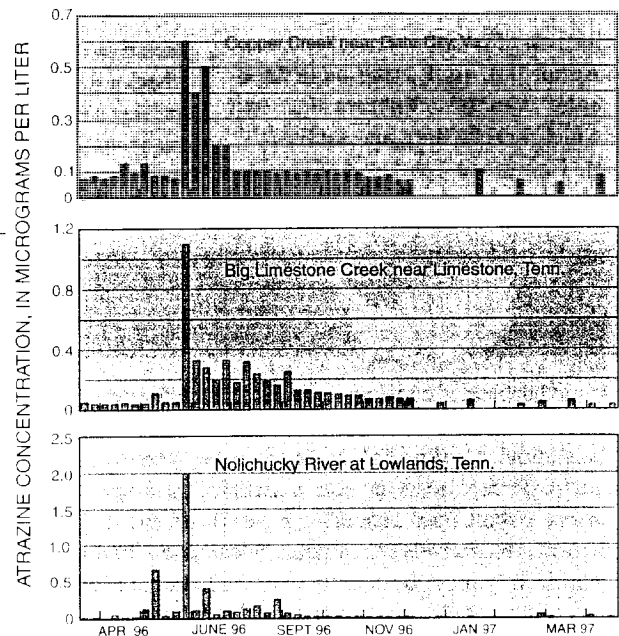


Figure 23. Atrazine concentrations were seasonal in the Upper Tennessee River Basin intensively sampled sites, March 1996 - April 1997.

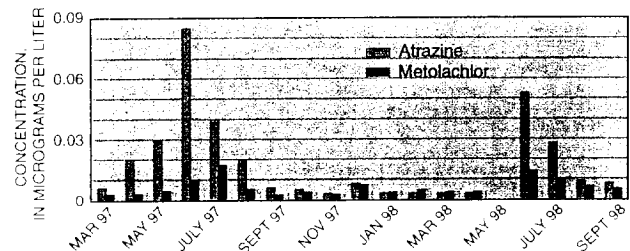


Figure 24. Atrazine and metolachlor concentrations were seasonal in monthly samples at Clear Creek at Lilly Bridge, March 1997 - September 1998.

Pesticide Concentrations Usually Meet Guidelines

Although most of the water samples collected contained detectable concentrations of one or more pesticides, no concentrations exceeded any drinking-water standards or guidelines. Only 20 of the 31 pesticides detected, however, have established guidelines. Of the 15 compounds that have aquatic-life guidelines, four were detected at concentrations higher than the guidelines. Carbaryl concentrations in excess of the 0.20- $\mu\text{g}/\text{L}$ (micrograms per liter) aquatic-life criterion⁽¹⁹⁾ were found in four samples—two each from the Guest River near Millers Yard, Virginia, and the Nolichucky River at Lowlands, Tennessee (fig. 25). Lindane, an organochlorine used primarily for the protection of tobacco transplants, was above the 0.01- $\mu\text{g}/\text{L}$ criterion in three samples from three different sites, two of which were in the same subbasin - Little Limestone Creek and the Nolichucky River at Lowlands, Tennessee.

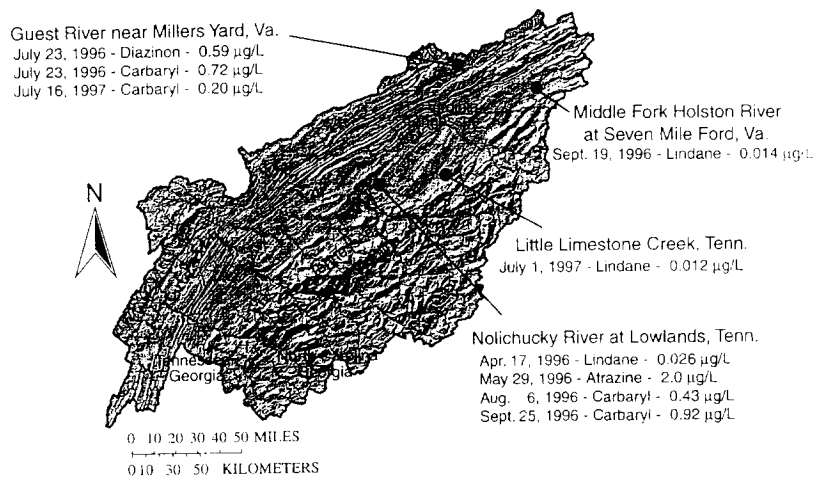
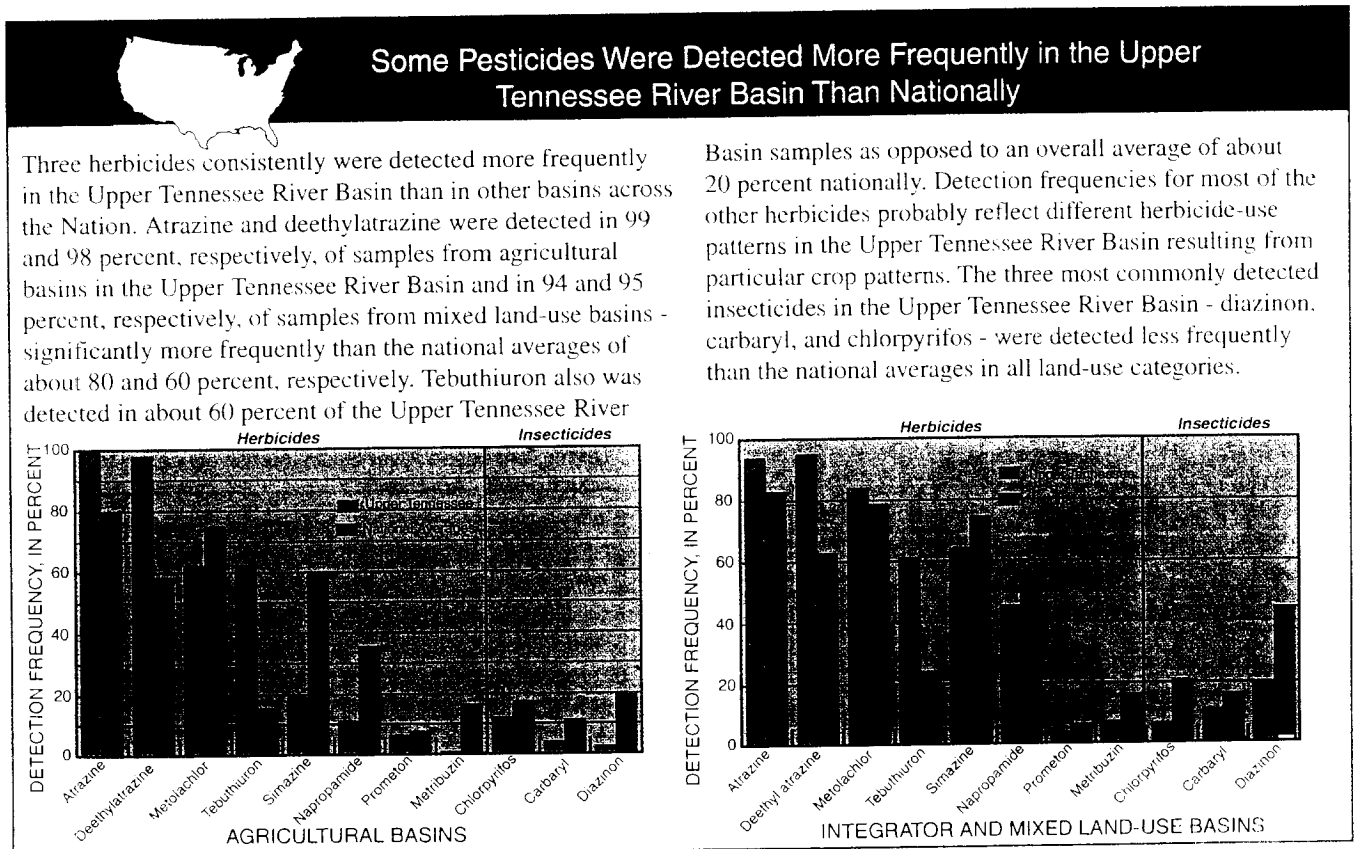


Figure 25. Pesticide concentrations, in micrograms per liter ($\mu\text{g}/\text{L}$), infrequently exceeded aquatic-life criteria in the Upper Tennessee River Basin, 1996-98.

An atrazine concentration higher than the 0.18- $\mu\text{g}/\text{L}$ criterion⁽²⁰⁾ also was detected in one sample taken at the Nolichucky River at Lowlands, Tennessee, in May 1996. This was the only criterion exceedance noted for any herbicide even though herbicides were detected much more frequently than the other

pesticide types. One sample collected at the Guest River near Millers Yard, Virginia, contained a diazinon concentration that was not only greater than the aquatic-life guideline of 0.08 $\mu\text{g}/\text{L}$ but approached the USEPA lifetime health advisory level of 0.60 $\mu\text{g}/\text{L}$ for drinking water.



Pesticides Were Detected at Low Levels in Ground Water

Pesticides were detected in Upper Tennessee River Basin ground-water samples more often than not, but generally at concentrations less than 0.01 µg/L. Pesticide concentrations in ground water did not exceed any drinking-water standards or guidelines. Usually, however, pesticides occur in mixtures for which criteria are not available. In addition, 5 of the 11 pesticides detected have no established guidelines or criteria.

Pesticides were detected in springs significantly more often and in more pesticide detections per sample than in other ground-water sources sampled (fig. 26). This probably reflects the greater vulnerability of springs to surface contamination either from the immediate area or karst features in the carbonate bedrocks. More frequent detections also may reflect the larger drainage areas from which springs capture ground water as opposed to wells. Of the 35 springs sampled, 24 (69 percent) contained detectable pesticide concentrations, and 12 (34 percent) contained detectable quantities of three or more different compounds. Detection frequencies in agricultural and domestic wells, by contrast, were significantly lower and similar to one another; 12 of 30 (40 percent) agricultural wells and 13 of 30 (43 percent) domestic wells contained detectable pesticide concentrations. Of these detections, only three (10 percent) samples from agricultural wells had detections of three or more pesticides. Eight (27 percent) domestic wells, however, had detections of three or more compounds.



Pesticides were detected more frequently in Upper Tennessee River Basin springs than in other sources of ground water.

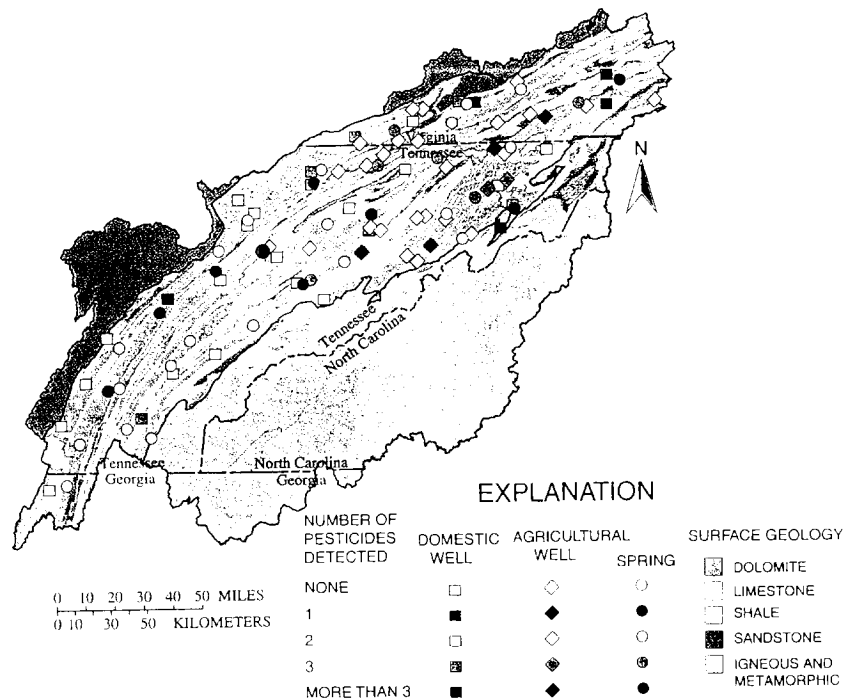


Figure 26. Pesticides were detected at low concentrations in Upper Tennessee River Basin ground water.

Atrazine and its degradation product, deethylatrazine, were the pesticides most commonly detected in all ground-water samples but were detected twice as frequently in springs as in other ground-water sources (fig. 27). Tebuthiuron, the third most frequently detected pesticide, also was detected more than twice as frequently in springs as in domestic wells. The different pesticide mixtures typical of the agricultural wells sampled reflect the focus on tobacco in this phase of the study. In general, a different suite of pesticides are used for tobacco than for most other crops. For example, atrazine and other broadleaf herbicides are toxic to tobacco.

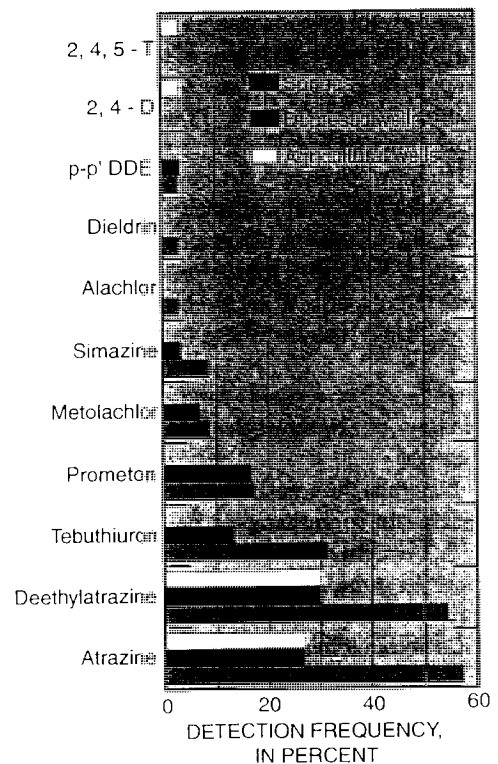


Figure 27. Pesticides were detected more frequently in springs than in wells in the Upper Tennessee River Basin.

Volatile Organic Compounds Were Frequently Detected in Ground Water

Ground-water samples were collected from 30 domestic wells and 35 springs tapping carbonate strata in the Upper Tennessee River Basin. Volatile organic compounds (VOCs) were detected in most of the ground-water samples (fig. 28) but generally at very low concentrations—often in orders of magnitude below the established reporting limit. Twenty-eight different VOCs were detected during sampling, 12 of which have drinking-water standards. No measured concentrations, however, exceeded these standards.

VOCs were detected more frequently in springs (86 percent) than in domestic wells (67 percent) and generally at slightly higher concentrations. Of the 20 samples with one or more concentrations greater than 0.1 µg/L, 14 were taken from springs and only 6 from wells. Similarly, of the 28 compounds detected, 22 were detected in spring samples and only 18 were detected in domestic wells.

The most frequently detected VOCs were trichloromethane (51 percent), chloromethane (28 percent), styrene (23 percent), tetrachloroethane (18 percent), carbon disulfide (11 percent), and trichloroethene (9 percent). The remaining 22 compounds were detected in three or fewer samples (less than 5 percent).

Other than the greater detection frequencies for spring samples, no areal or other occurrence patterns could be found. As is the case nationally, the source for many of the most common VOCs detected in ground water, such as trichloromethane, is unclear. The greater occurrence of detections in springs as well as the widespread but random pattern of occurrence suggests the possibility of atmospheric origins, but no definite source can be identified at present.

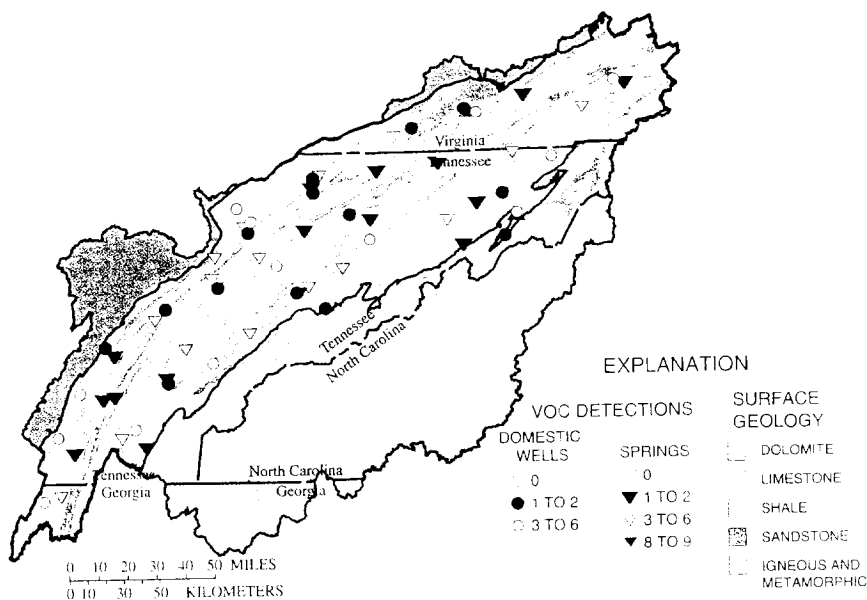
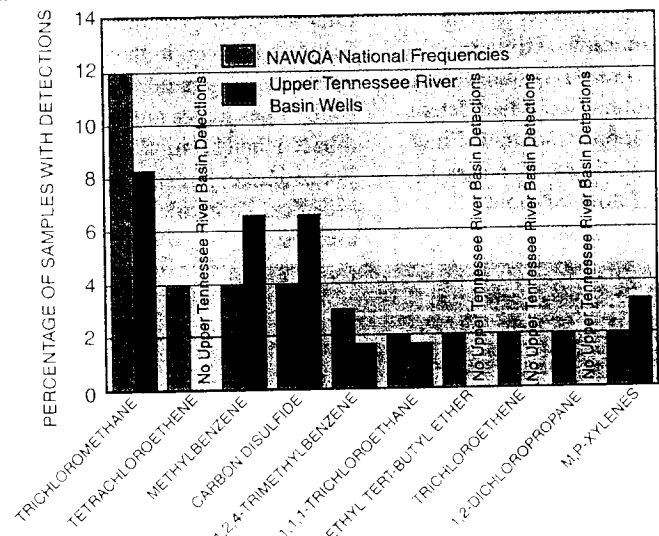


Figure 28. Volatile organic compounds (VOCs) are often detected in Upper Tennessee River Basin ground water.

Upper Tennessee River Basin and National VOC Detection Frequencies Are Similar

Detection frequencies in Upper Tennessee River Basin wells for the 10 most commonly detected VOCs nationally were similar to national detection frequencies found for ambient ground water in all land-use settings. All compounds were assessed at a common detection level of 0.1 mg/L. Trichloromethane was the most

commonly detected compound nationally as well as in the Upper Tennessee River Basin but typically was detected at concentrations far below drinking-water standards. The results are consistent with the mixed urban and rural land uses surrounding most Upper Tennessee River Basin ground-water sites.



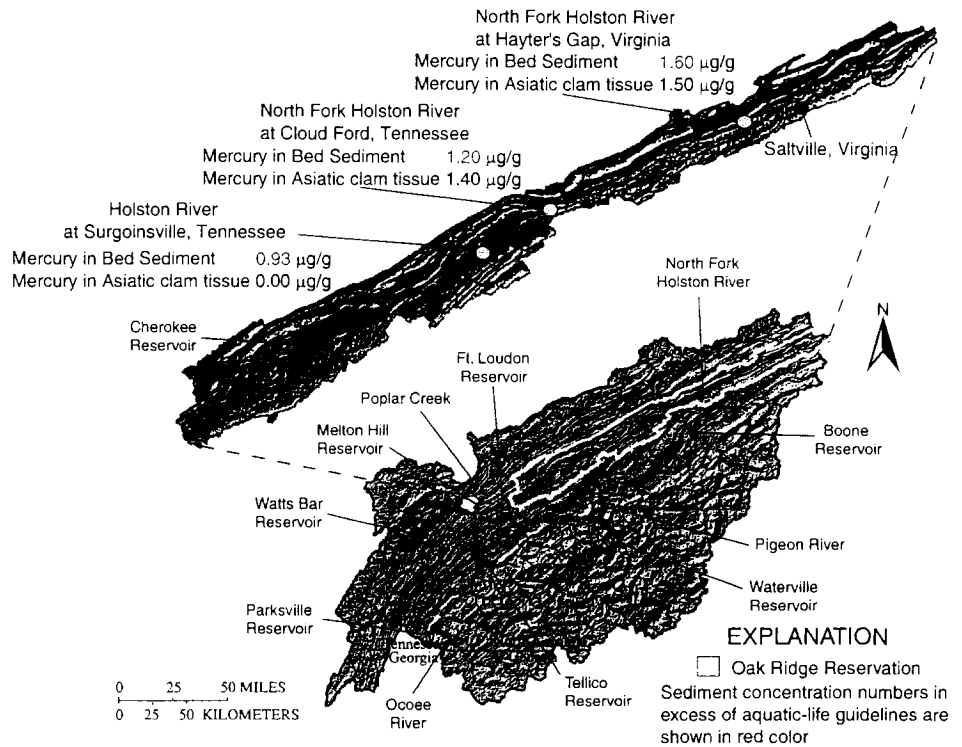
Upper Tennessee River Basin detection frequencies were similar to national results for the 10 most commonly detected VOCs nationally. (Assessment level of 0.1 microgram per liter)

Water-Quality Influences of Industry and Mining

Industrial and mining activities prior to the passage of the Clean Water Act in 1972 have left a legacy of contaminated sediment that continues to affect water quality in parts of the Upper Tennessee River Basin. The most widespread contaminants are PCBs (polychlorinated biphenyls) and mercury, mostly from industrial activities dating from 1950 to 1972. Sources for some of the other contaminants, however, such as those affecting the Pigeon and Ocoee Rivers date back as far as 1908 and 1843, respectively.

Mercury in the North Fork Holston River is a result of the operation of a chlor-alkali plant on the banks of the river from 1950 through 1971. An estimated 75 pounds of mercury per day were discharged either directly to the river or into unlined holding ponds along the riverbank.⁽²¹⁾ Although soils at the site have been remediated, the site continues to discharge mercury.⁽²²⁾ Bed-sediment and tissue samples taken from the Holston River system (fig. 29) were the only samples taken during the study that were above the Canadian guideline for aquatic-life protection (0.486 micrograms per gram total mercury). Although tissue samples in the main-stem Holston River site at Surgoinsville were free of mercury, the bed-sediment results suggest that mercury may be migrating farther downstream than previously thought and may eventually reach Cherokee Reservoir.

Mercury is also a major contaminant in the drainages downstream from the Department of Energy's 35,585-acre Oak Ridge Reservation (ORR), such as East Fork Poplar Creek, the White Oak Creek watershed, and the lower Clinch River - Watts Bar Reservoir. The ORR, established in 1942 as part of the Manhattan Project to develop the atomic bomb, encompasses three major facilities — X-10, originally for weapons research but now Oak Ridge National Laboratory (ORNL); Y-12, for the fabrication of nuclear weapons compo-



Location	Contaminant	Source	Comment
N. Fk. Holston River	Mercury	Industrial point source	Fish consumption advisory
Boone Reservoir	PCBs, Chlordane	Not identified	Fish consumption advisory
Pigeon River	Dioxin	Industrial point source	Fish consumption advisory
Waterville Reservoir	Dioxin	Industrial point source	Fish consumption advisory
Melton Hill Reservoir	PCBs, Chlordane	Industrial point source	Fish consumption advisory
E. Fk. Poplar Creek	PCBs, Mercury	Industrial point source	Fish consumption advisory
Watts Bar Reservoir	PCBs, Mercury	Industrial point source	Fish consumption advisory
Ft. Loudon Reservoir	PCBs	Industrial point source	Fish consumption advisory
Tellico Reservoir	PCBs	Not identified	Fish consumption advisory
Parksville Reservoir	Metals	Abandoned mining area	None
Ocoee River	Metals	Abandoned mining area	None

Figure 29. Mercury, in micrograms per gram, and organic contaminants persist in bed sediments and biological tissues in parts of the Upper Tennessee River Basin.

nents; and K-25, for uranium enrichment by gaseous diffusion. As a result of these operations, about 527 sites covering approximately 15 percent of the total ORR area have been identified as contaminated with metals, including mercury, radionuclides, a variety of VOCs, and nitrates.⁽²³⁾

Most of the contamination has remained confined within the ORR, which was added in its entirety to USEPA's National Priorities List in 1989. A number of contaminants, most notably mercury, PCBs, and cesium-137, however, have migrated to downstream areas. The State of Tennessee has posted a fish-consumption advisory for ORR drainages as well as Watts

Bar Reservoir as a result of bioaccumulation of mercury and PCBs in some fish species.

A 1983 inventory estimated that about 2 million pounds (1,088 metric tons) of mercury was 'lost' from operations related to thermonuclear bomb development on the ORR.⁽²⁴⁾ Most of this mercury is believed to have volatilized into the atmosphere, but much remains within ORR facilities and in Watts Bar Reservoir sediments. Analyses of sediment cores indicate that the highest discharges of mercury and cesium-137 occurred during the 1950s, and that about 76 metric tons of mercury has accumulated in Watts Bar sediments. About 91 percent of the 335

curies of cesium-137 released from the ORR have also been retained by the lake sediments. The concentrations detected are not believed to pose an imminent human health risk, especially if the deep sediments are not disturbed.⁽²⁵⁾

Mining of the massive sulfide deposits in the Copper Basin along the Ocoee River began in 1843. Copper was the primary metal extracted, but iron, sulfur, zinc, and small amounts of gold and silver also were produced. Before 1900, Copper Basin was the largest metal-mining district in the Southeast. The last mine was closed in 1987.⁽²⁶⁾

High concentrations of sulfur dioxide produced by smelting operations devastated the surrounding environment, resulting in a "moonscape" of about 25 square miles. Erosion of the area resulted in high sediment and associated metal loads to area streams. Although thousands of acres have been revegetated and the landscape is being slowly transformed back to forest, relatively high metal concentrations remain in the upper reaches of Parksville Reservoir and the Ocoee River.

Discharge of essentially untreated paper-mill effluent to the Pigeon River began in early 1908 and continued until plant improvements were instituted in the 1990s. Dioxins were first detected in fish samples from the river in 1988 (dioxin detection methods were not available until 1985) and became an immediate priority with respect to human health effects.⁽²⁷⁾ Dioxins have not been detected in recent samples, including bed-sediment and tissue samples taken during the Upper Tennessee NAWQA study. The State of Tennessee, however, continues a precautionary fish-consumption advisory for the Tennessee portion of the river.

Even though discussions regarding the Pigeon River continue between the States of Tennessee and North Carolina, all parties agree that conditions have improved significantly. Once nearly devoid of aquatic life, benthic invertebrate and fish populations in the

Tennessee portion of the Pigeon River are showing signs of recovery. Water-ville Lake, however, still retains tons of contaminated sediments deposited since the dam became operational in 1930, and these sediments remain a potential source of dioxin and other contaminants.

Polycyclic aromatic hydrocarbons (PAHs) commonly are detected as pollutants in soils and sediments, occur naturally in crude oil and coal, and also can result from the incomplete combustion of fossil fuels and forest fires.⁽²⁸⁾ In the upper Clinch River Basin, PAH concentrations reflect the presence of coal fines from upstream mining activities.

Twenty-nine PAHs were found in upper Clinch River bed-sediment samples and, with only a few exceptions, were not detected in the 12 samples taken from other parts of the Upper Tennessee Basin. Although PAHs are known to be toxic to fish, mussels, and aquatic insects, sediment-quality guidelines for the protection of aquatic life have been established for only 12 of the compounds detected. Of these, only two compounds – naphthalene and phenanthrene – exceeded their respec-

tive Canadian probable-effect levels of 391 µg/kg (micrograms per kilogram) and 515 µg/kg (fig. 30). The probable-effect levels define concentrations above which adverse effects are expected. A third compound, benzo(a)anthracene, occurred in concentrations very near its guideline of 385 µg/kg, and a number of compounds lacking guidelines were found at concentrations of 1,000 µg/kg or greater.

The highest concentrations generally follow the results for naphthalene and phenanthrene and occurred in the major river sites nearest, on a relative basis, to upstream mining activities. For example, concentrations at the Powell River and Pendleton Island sites exceeded those found at the Clinch River near Tazewell, which is farther removed from active mining in terms of river miles. Higher gradients and water velocities in the tributaries to the major streams prevent the accumulation of fine-grained sediment and coal fines. The main river channels, however, contain large pools and backwater areas where fine-grained material and associated constituents are deposited.

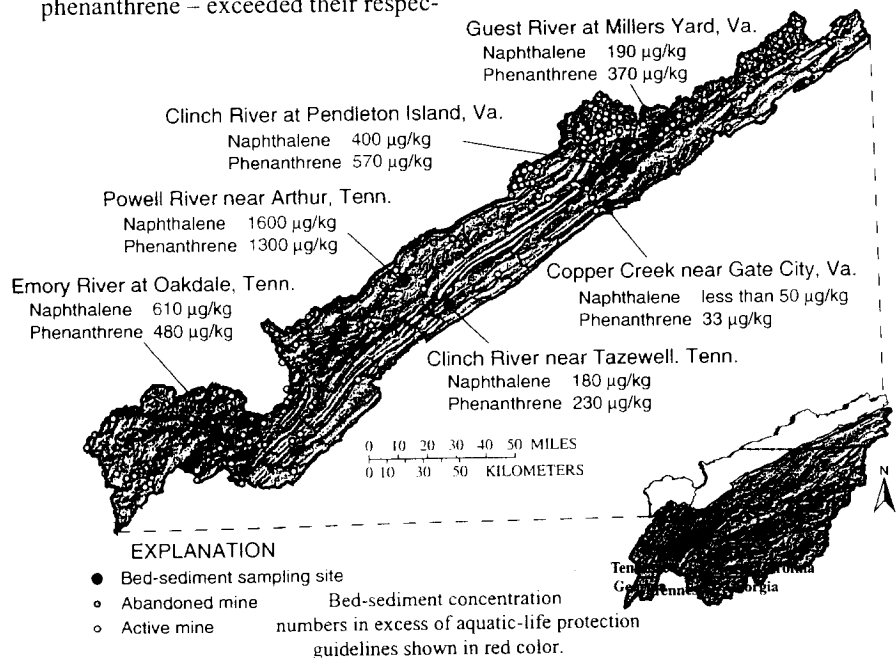


Figure 30. Relatively high polycyclic aromatic hydrocarbon (PAH) concentrations, in micrograms per kilogram, are common in bed sediments in the upper Clinch River Basin.

Freshwater Mussels in the Clinch and Powell Rivers

Freshwater mussel species diversity has been slowly declining in the Clinch and Powell Rivers of Tennessee and Virginia over the past 100 years. The numbers of mussel species found in these rivers are shown in figure 31 for selected periods of time and illustrate the long-term trend in loss of species diversity. The numbers do not precisely show numbers of species lost but reflect difficulties in finding specimens as species decline and, in some cases, difficulty with basin access. For example, prior to 1915, the upper parts of the river basins were inaccessible and remained unsurveyed.

Although some forms were lost, survey results from 1963 to 1971 indicate that the fauna survived TVA impoundment largely intact. Mussel declines became apparent, however, in the mid-1970's, and by that time many previously common mussel species had become rare, extirpated, or extinct.

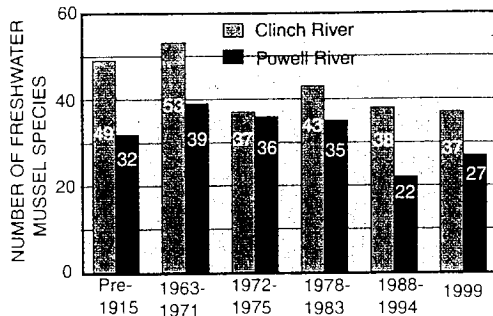


Figure 31. Freshwater mussel species diversity in the Clinch and Powell Rivers, Tennessee and Virginia, 1899-1999.

The greatest declines in mussel abundance occurred during the record drought from 1983 to 1988 (fig. 32). Since that time, the Clinch River in Tennessee has shown remarkable recovery, both in mussel densities and species numbers. The Virginia parts of the Clinch and the Powell Rivers, however, have recovered to only a little more than half the densities recorded in 1979, mostly reflecting recovery of the three most abundant species. Most of the rare and more sensitive species continue to decline in the Powell River and in the Virginia part of the Clinch River.⁽²⁹⁾

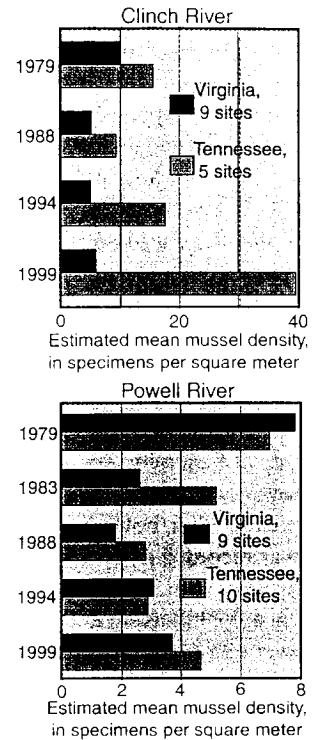


Figure 32. Estimated mean densities of freshwater mussel specimens in the Clinch and Powell Rivers, Tennessee and Virginia, 1979-99.⁽²⁹⁾



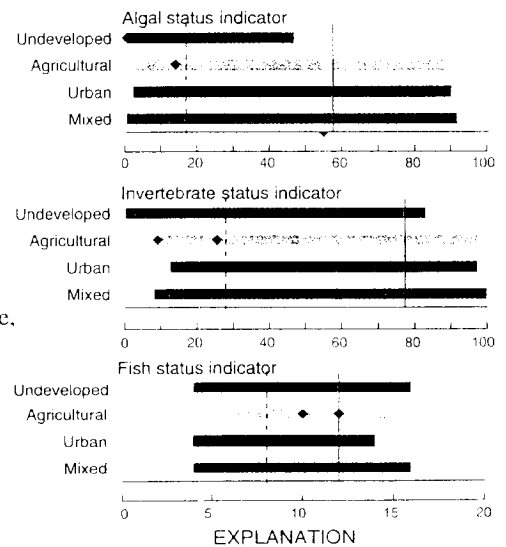
Upper Tennessee River Biological Communities in a National Context

Three biological indicators, which typically respond to changes in stream degradation, illustrate the relation of Upper Tennessee River Basin sites to the overall range of NAWQA sites nationwide. For all indicators, higher values suggest a more degraded stream site.

Algal status focuses on the changes in the percentage of certain algae in response to increasing siltation. Within the Upper Tennessee River Basin, the only sites in the highest 25 percent nationally are Big Limestone Creek, which drains predominantly agricultural land use, and the Pigeon River, which has been heavily affected by industrial wastes.

Invertebrate status is the average of 11 invertebrate (primarily insects, worms, crayfish, clams) metrics that summarize changes in richness, tolerance, trophic conditions, and dominance commonly associated with water-quality degradation. Among the Upper Tennessee River Basin sites, the two that rank highest on the index are the Pigeon and French Broad Rivers. The Pigeon River is recovering from decades of receiving industrial wastes. The French Broad River is principally affected by urban development in the Asheville, North Carolina, area and agriculture in the lower part of the basin.

Fish status is the sum of scores of four fish metrics (percentage of tolerant, omnivorous, non-native individuals, and percentage of individuals with external anomalies) that tend to increase in association with water-quality degradation. The Holston River at Surgoinsville, Tennessee, which ranked highest on this index, is characterized by relatively high concentrations of mercury and copper in bed sediments, probably derived from upstream industrial activities.



EXPLANATION

- 25th percentile of sites in 16 NAWQA Study Units, 1994 - 98.
- 75th percentile of sites in 16 NAWQA Study Units, 1994 - 98.
- ◆ Biological status assessed at a site

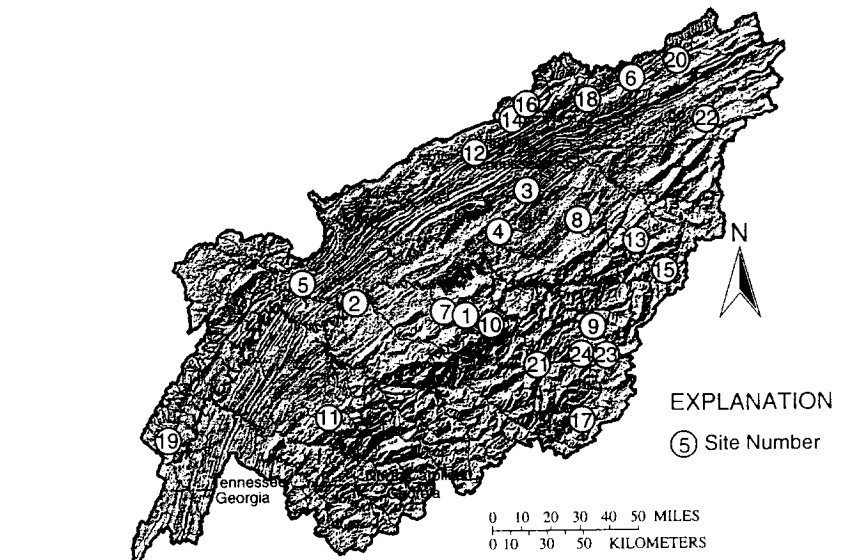
Toxic Spills and Releases

The NAWQA Program, like most water-quality assessments, is designed to gather information on general water-quality conditions and analyze problems that tend to be chronic as opposed to episodic. Even though the NAWQA Program provides for sampling during storm events in order to achieve a more complete "picture" of water-quality conditions, detection of every instance of water contamination is clearly beyond the program's defined scope. In general, this is true of every other ongoing State or Federal water-quality assessment.

In late May 1996, however, a toxic release was recorded at the Big Limestone Creek site (fig. 33, number 8) that resulted in a fishkill over several miles in the lower end of the stream. The apparent cause was excessive ammonia concentrations that were traced to agricultural activities upstream. If not for the sampling activity being conducted at the site, the kill most likely would have gone unreported. Given the relatively remote nature of many biologically diverse stream reaches in the Upper Tennessee River Basin, it is possible that many similar episodes go unreported as well.

The number of relatively rare and threatened aquatic species in the Upper Tennessee River Basin make accidental spills and releases a particular concern in parts of the basin. Habitat modifications resulting from human activities, such as impoundments and pollution, have restricted the greatest numbers and variety of aquatic fauna to only a few tributaries.⁽³²⁾ In addition, impoundments have effectively separated once contiguous biological communities into smaller, more vulnerable sub-units.

The upper Clinch and Powell watersheds are home to the most diverse fish and mussel fauna in the Upper Tennessee River Basin. These two subbasins are effectively separated from biological interaction, however, by Norris Lake and are very vulnerable to coal-



Number and Date of incident	Site name	Spill source	Contaminant
1 Apr 95	West Prong, Bird Creek, Tenn.	Industrial	Concrete
2 May 95	Second Creek, Tenn.	Urban	Raw sewage
3 Sept. 95	Crockett Creek, Tenn.	Urban	Unknown
4 Sept. 95	Flat Creek, Tenn.	Industrial	Sulfuric acid, zinc
5a Aug. 95	E. Fork Poplar Creek	Industrial	Styrene
5b July 97	E. Fork Poplar Creek	Industrial	Sodium bisulfite
6a Nov 95	Clinch River, Va.	Mining	Coal fines
6b Feb. 96	Clinch River, Va.	Mining	Coal fines
6c Mar. 96	Clinch River, Va.	Mining	Coal fines
7a May 96	W. Prong, Little Pigeon River, Tenn.	Urban	Sodium hypochlorite
7b July 96	W. Prong, Little Pigeon River, Tenn.	Urban	Sodium hypochlorite
8 May 96	Big Limestone Creek, Tenn.	Agriculture	Ammonia
9 June 96	Reems Creek, N.C.	Urban	Raw sewage
10 July 96	Webb Creek, Tenn.	Industrial	Asphalt sealant
11 July 96	Citico Creek, Tenn.	Agricultural	Concrete
12 Oct. 96	Powell River, Va.	Mining	Coal fines
13 Mar. 97	North Indian Creek, Tenn.	Urban	Chlorine
14 July 97	North Fork Powell River, Va.	Mining	Coal fines
15 July 97	North Toe River, N.C.	Urban	Chlorine
16 Aug. 97	Straight Creek, Va.	Mining	Acid mine drainage
17 Aug. 97	French Broad River, N.C.	Unknown	Unknown
18 Sept. 97	Stone Creek, Va.	Mining	Oil
19 Feb. 98	Rogers Branch, Tenn.	Urban	Pesticide
20 Aug. 98	Clinch River, Va.	Industrial	Octocure 554
21 Nov 98	Richland Creek, N.C.	Other	Oxygen demand
22 June 99	M. Fork Holston River, Va.	Unknown	Pesticide
23 Sept. 99	Swannanoa River, N.C.	Urban	Chlorine
24 Nov. 99	Smith Mill Creek, N.C.	Urban	Unknown

Figure 33. Contaminant releases have resulted in fish and mussel kills in the Upper Tennessee River Basin, 1995-99. (30, 31)

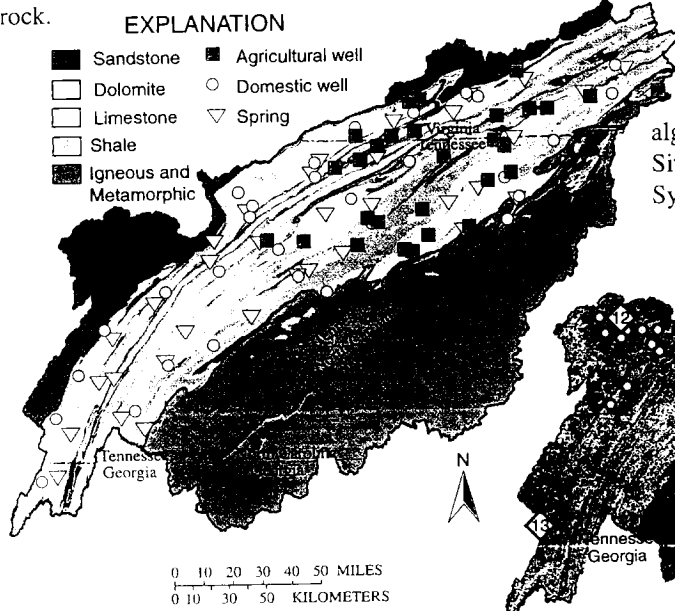
fine spills from numerous active and abandoned mining sites in their headwaters. At least five coal-fine spills occurred during the 1995-99 study period (G. Heffinger, U.S. Fish and Wildlife Service, written commun., April 17, 2000).

Mussel species generally are of the greatest concern because of their lack of mobility and the longer times typi-

cally required for populations to recover. For example, data collected in 1971 following a very large 1967 fly-ash spill in the Clinch River found that fish and aquatic insects were reestablished relatively quickly. Mussels, however, have yet to recolonize the 9- to 10-mile reach directly downstream from the spill site.⁽³³⁾

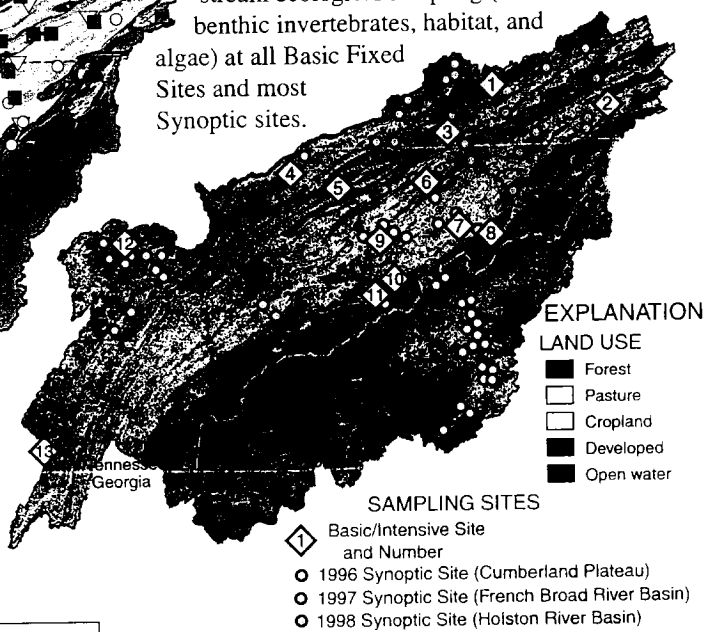
STUDY UNIT DESIGN

Study designs for both ground-water and surface-water components focused principally on the Valley and Ridge province. The Valley and Ridge is home to the majority of the Study Unit population and is the most highly developed in terms of agriculture and urban land uses. Ground-water studies focused on the carbonate-based dolomites and limestones of the Valley and Ridge. These geologic units form the most prolific aquifers in the Upper Tennessee River Basin and also are the most susceptible to contamination because of their associated karst and solution features. Ground-water resources are very limited in the Blue Ridge and Cumberland Plateau provinces because of the relatively impermeable nature of the bedrock and the low water-storage capacity of the thin soils that overlie the bedrock.



Surface-water studies focused on the unregulated portions of the Upper Tennessee River Basin principally in the Valley and Ridge province, which contains the most intense agricultural activity in the basin. Thirteen basic fixed stream-sampling sites were operated during the study to monitor water-quality conditions with time in various parts of the basin. Data-collection sites were selected to cover the major subbasins of the Upper Tennessee River and to encompass the major land uses. An additional 61 sites were sampled during the study as part of three synoptic networks designed to better describe areal water-quality variations of the subbasins. In keeping with the NAWQA multiple lines of evidence approach to describe water-quality conditions,⁽³⁴⁾ data-collection activities included water-column chemistry

at all sites, bed-sediment and Asiatic clam tissue samples at Basic Fixed Sites, and stream ecological sampling (fish communities, benthic invertebrates, habitat, and algae) at all Basic Fixed Sites and most Synoptic sites.



Site number	Site name	Site type	Physio-graphic province*
1	Guest River near Millers Yard, Virginia	Indicator, Mining	CP
2	Middle Fork Holston River at Seven-Mile Ford, Virginia	Indicator, Mixed	VR
3	Copper Creek near Gate City, Virginia	Indicator, Agriculture	VR
4	Powell River near Arthur, Tennessee	Integrator	CP-VR
5	Clinch River at Tazewell, Tennessee	Integrator	VR-CP
6	Holston River at Surgoinsville, Tennessee	Integrator	VR
7	Big Limestone Creek near Limestone, Tennessee	Indicator, Agriculture	VR

Site number	Site name	Site type	Physio-graphic province*
8	Nolichucky River at Embreeville, Tennessee	Indicator, Mining	BR
9	Nolichucky River at Lowlands, Tennessee	Indicator, Mixed	BR-VR
10	French Broad River near Newport, Tennessee	Indicator, Agriculture	BR
11	Pigeon River at Newport, Tennessee	Integrator	BR-VR
12	Clear Creek at Lilly Bridge, Tennessee	Integrator	CP
13	Tennessee River at Chattanooga, Tennessee	Integrator	CP-VR-BR

* CP - Cumberland Plateau, BR - Blue Ridge, VR - Valley and Ridge

SUMMARY OF DATA COLLECTION IN THE UPPER TENNESSEE RIVER BASIN, 1994–98

Study component	What data were collected and why	Types of sites sampled	Number of sites	Sampling frequency and period
Stream Chemistry				
Bottom-sediment survey	Sediment in depositional zones was sampled for pesticides, other synthetic organic compounds, and trace elements to determine the presence of potentially toxic compounds. Water-quality samples also were taken at each site, including major ions, nutrients, organic carbon, pesticides, bacteria, and suspended sediment.	Selected rivers and streams.	15	Once (1995, 1996, 1998)
Water-chemistry sites	Water-chemistry data, including major ions, nutrients, organic carbon, pesticides, bacteria, and suspended sediment, were used to describe concentrations and loads.	Sampling occurred near selected continuous streamflow sites.	13	Variable (1996–98)
Storm sampling program	Water-chemistry data, including major ions, nutrients, organic carbon, pesticides, bacteria, and suspended sediment, were used to describe concentrations and loads.	Samples were taken at water-chemistry sites during high-flow conditions.	variable	Variable (1996–98)
Nutrient/pesticide synoptic studies	Water-chemistry data, including major ions, nutrients, organic carbon, pesticides, bacteria, and suspended sediment, were used to describe concentrations of selected constituents.	Surface-water sampling sites in the Cumberland Plateau, French Broad River Basin, and the Valley and Ridge were selected to describe conditions across the Study Unit.	64	Variable (1996, 1997, 1998)
Intensive pesticide sampling	Pesticides, major ions, organic carbon, suspended sediment, bacteria, and nutrients were analyzed to determine seasonal variations in concentrations and loads.	Water-chemistry sites located in intensive agricultural basins or mixed land-use basins.	3	Biweekly (March–Nov., 1996)
Stream Ecology				
Contaminants in Asiatic clams	Asiatic clams were sampled for pesticides, other synthetic organic compounds, and trace elements to determine the presence of potentially toxic compounds.	Selected rivers and streams.	15	Once (1995, 1996, 1998)
Aquatic biology	Biological communities and stream habitat were assessed and fish, macroinvertebrates, and algae were quantitatively sampled.	Biological communities and habitat at basic fixed water-chemistry sites, and biological communities at synoptic sites.	13 fixed sites, 63 synoptic sites	Once (1995–98)
Spring synoptic study	Macroinvertebrates were qualitatively sampled.	Spring sites.	35	Once (Aug.–Nov., 1997)
Ground-Water Chemistry				
Agricultural land-use survey	Water-chemistry data, including major ions, nutrients, organic carbon, pesticides, and radionuclides, were analyzed to determine the effects of timely tobacco production on shallow ground-water quality.	Shallow 2-inch monitoring wells were installed adjacent to tobacco fields in the Valley and Ridge in northeastern Tennessee and southwestern Virginia.	30	Once (June and July, 1997)
Shallow spring survey	Water-chemistry data, including major ions, nutrients, organic carbon, pesticides, bacteria, and radionuclides, were analyzed to determine the quality of ground water.	Randomly selected springs in the Valley and Ridge.	35 springs	Once (Aug.–Nov., 1997)
Shallow well survey	Water-chemistry data, including major ions, nutrients, organic carbon, pesticides, bacteria, and radionuclides, were analyzed to determine the quality of ground water.	Randomly selected wells in the Valley and Ridge.	20 wells	Once (Aug.–Nov., 1997)

Aquatic-life criteria—Water-quality guidelines for protection of aquatic life. Often refers to U.S. Environmental Protection Agency water-quality criteria for protection of aquatic organisms.

Aquifer—A water-bearing layer of soil, sand, gravel, or rock that will yield usable quantities of water to a well.

Basic Fixed Sites—Sites on streams at which streamflow is measured and samples are collected for temperature, salinity, suspended sediment, major ions and metals, nutrients, and organic carbon to assess the broad-scale spatial and temporal character and transport of inorganic constituents of stream water in relation to hydrologic conditions and environmental settings.

Bed sediment—The material that temporarily is stationary in the bottom of a stream or other watercourse.

Bed sediment and tissue studies—Assessment of concentrations and distributions of trace elements and hydrophobic organic contaminants in streambed sediment and tissues of aquatic organisms to identify potential sources and to assess spatial distribution.

Benthic invertebrates—Insects, mollusks, crustaceans, worms, and other organisms without a backbone that live in, on, or near the bottom of lakes, streams, or oceans.

Constituent—A chemical or biological substance in water, sediment, or biota that can be measured by an analytical method.

Contamination—Degradation of water quality compared to original or natural conditions and due to human activity.

Cubic foot per second (ft³/s, or cfs)—Rate of water discharge representing a volume of 1 cubic foot passing a given point during 1 second, equivalent to approximately 7.48 gallons per second or 448.8 gallons per minute or 0.02832 cubic meter per second.

Degradation products—Compounds resulting from transformation of an organic substance through chemical, photochemical, and/or biochemical reactions.

Detection limit—The minimum concentration of a substance that can be identified, measured, and reported with 99 percent confidence that the analyte concentration is greater than zero; determined from analysis of a sample in a given matrix containing the analyte.

Discharge—Rate of fluid flow passing a given point at a given moment in time, expressed as volume per unit of time.

Drainage area—The drainage area of a stream at a specified location is that area, measured in a horizontal plane, which is enclosed by a drainage divide.

Drinking-water standard or guideline—A threshold concentration in a public drinking-water supply, designed to protect human health. As defined here, standards are U.S. Environmental Protection Agency regulations that specify the maximum contamination levels for public water systems required to protect the public welfare; guidelines have no regulatory status and are issued in an advisory capacity.

Indicator sites—Stream sampling sites located at outlets of drainage basins with relatively homogeneous land use and physiographic conditions; most indicator-site basins have drainage areas ranging from 20 to 200 square miles.

Integrator or Mixed-use site—Stream sampling site located at an outlet of a drainage basin that contains multiple environmental settings. Most integrator sites are on major streams with relatively large drainage areas.

Intensive Fixed Sites—Basic Fixed Sites with increased sampling frequency during selected seasonal periods and analysis of dissolved pesticides for 1 year. Most NAWQA Study Units have one to two integrator Intensive Fixed Sites and one to four indicator Intensive Fixed Sites.

Karst—A type of topography that results from dissolution and collapse of carbonate rocks such as limestone and dolomite, and characterized by closed depressions or sinkholes, caves, and underground drainage.

Load—General term that refers to a material or constituent in solution, in suspension, or in transport; usually expressed in terms of mass or volume.

Main stem—The principal course of a river or a stream.

Metamorphic rock—Rock that has formed in the solid state in response to pronounced changes of temperature, pressure, and chemical environment.

Micrograms per liter (µg/L)—A unit expressing the concentration of constituents in solution as weight (micrograms) of solute per unit volume (liter) of water; equivalent to one part per billion in most stream water and ground water. One thousand micrograms per liter equals 1 mg/L.

Milligrams per liter (mg/L)—A unit expressing the concentration of chemical constituents in solution as weight (milligrams) of solute per unit volume (liter) of water; equivalent to one part per million in most stream water and ground water.

Nonpoint source—A pollution source that cannot be defined as originating from discrete points such as pipe discharge. Areas of fertilizer and pesticide applications, atmospheric deposition, manure, and natural inputs from plants and trees are types of nonpoint source pollution.

Point source—A source at a discrete location such as a discharge pipe, drainage ditch, tunnel, well, concentrated livestock operation, or floating craft.

Synoptic sites—Sites sampled during a short-term investigation of specific water-quality conditions during selected seasonal or hydrologic conditions to provide improved spatial resolution for critical water-quality conditions.

Tributary—A river or stream flowing into a larger river, stream, or lake.

Volatile organic compounds (VOCs)—Organic chemicals that have a high vapor pressure relative to their water solubility. VOCs include components of gasoline, fuel oils, and lubricants, as well as organic solvents, fumigants, some inert ingredients in pesticides, and some by-products of chlorine disinfection.

Water-quality standards—State-adopted and U.S. Environmental Protection Agency-approved ambient standards for water bodies. Standards include the use of the water body and the water-quality criteria that must be met to protect the designated use or uses.

Water table—The point below the land surface where ground water is first encountered and below which the earth is saturated. Depth to the water table varies widely across the country.

Yield—The mass of material or constituent transported by a river in a specified period of time divided by the drainage area of the river basin.

REFERENCES

- 1 Murless, Dick, and Stallings, Constance, 1973, Hiker's guide to the Smokies: San Francisco, Calif., Sierra Club Books, 375 p.
- 2 Tennessee Valley Authority, 1987, Tennessee Valley Authority handbook—Dams and reservoirs, Knoxville, Tenn., 265 p.
- 3 Brewer, Alberta, and Brewer, Carson, 1975, Valley so wild: Knoxville, Tenn., East Tennessee Historical Society, 382 p.
- 4 Hampson, P.S., 1995, National Water-Quality Assessment Program - the Upper Tennessee River Basin: U.S. Geological Survey Fact Sheet FS-150-95, 2 p.
- 5 U.S. Geological Survey, 1999, Downloading 1995 water-use data: accessed on October 20, 2000, at <http://water.usgs.gov/watuse/spread95.html>
- 6 National Weather Service, 2000, East Tennessee climatological data, Monthly climate data for Chattanooga, Knoxville, and Tri-Cities: accessed on October 20, 2000, at <http://www.srh.noaa.gov/ftproot/>
- 7 Zurawski, Ann, 1978, Summary appraisals of the Nation's ground-water resources—Tennessee Region: U.S. Geological Survey Professional Paper 813-L, 35 p.
- 8 Master, L.L., Flack, S.R., and Stein, B.A., eds., 1998, Rivers of life—critical watersheds for protecting freshwater biodiversity: Arlington, Va., The Nature Conservancy, 71 p.
- 9 State of Tennessee, 1999, Rules of the Tennessee Department of Environment and Conservation, Chapter 1200-4-3, General Water Quality Criteria, 25 p.
- 10 State of Tennessee, 1998, Final 1998 303(d) list for the State of Tennessee: accessed on October 20, 2000, at <http://www.state.tn.us/environment/water.htm>
- 11 Yates, M.V., and Yates, S.R., 1993, Pathogens, in Alley, W.M., ed., Regional ground-water quality: New York, Van Nostrand Reinhold, p. 383-404.
- 12 U.S. Environmental Protection Agency, 1986, Ambient water-quality criteria for bacteria—1986: U.S. Environmental Protection Agency 44015-84-002, 18 p.
- 13 U.S. Geological Survey, 1999, The quality of our Nation's waters—Nutrients and pesticides: U.S. Geological Survey Circular 1225, 82 p.
- 14 Helsel, D.R., 1993, Hydrology of stream quality—Statistical analysis of water-quality data, in National Water Summary 1990-91—Hydrologic events and stream-water quality: U.S. Geological Survey Water-Supply Paper 2400, p. 93-100.
- 15 Treece, M.W., Jr., and Johnson, G.C., 1997, Nitrogen in streams of the Upper Tennessee River Basin, 1970-93, U.S. Geological Survey Open-File Report 97-117, 4 p.
- 16 Johnson, G.C., and Treece, M.W., Jr., 1998, Phosphorus in streams of the Upper Tennessee River Basin, 1970-93, U.S. Geological Survey Open-File Report 98-532, 6 p.
- 17 Tennessee Valley Authority, 1991, Reservoir vital signs monitoring—1990, physical and chemical characteristics of water and sediment: Tennessee Valley Authority TVA/WR/WQ-91/10, 142 p.
- 18 Thelin, G.P., 1999, County-level pesticide use estimates (1991-1993, 1995) in the conterminous United States (metadata): U.S. Geological Survey, accessed on October 20, 2000, at <ftp://ftpdcascr.wr.usgs.gov/nsp/thelin/ncfap.html>
- 19 U.S. Environmental Protection Agency, 1999, National recommended water quality criteria—Correction: EPA 822-Z-99-001, 26 p. accessed on October 20, 2000, at (<http://www.epa.gov/ost/pc/revcom.pdf>)
- 20 Environment Canada, 1999, Canadian sediment quality guidelines: accessed on October 20, 2000, at (www.ec.gc.ca/ceqg-rcqe/sediment.htm)
- 21 Milligan, J.D., and Ruane, R.J., 1978, Analysis of mercury data collected from the North Fork of the Holston River: Tennessee Valley Authority, Division of Environmental Planning, TVA/EP-78112, 32 p.
- 22 Seivard, L.D., Stilwell, D.A., Rice, Stephen O., and Seeley, K.R., 1993, Geographic distribution of mercury in asiatic clams, *Corbicula fluminea*, from the North Fork Holston River, Virginia: U.S. Fish and Wildlife Service, Environmental Contaminants Division, 16 p.
- 23 State of Tennessee, 1999, Tennessee Department of Environment and Conservation, DOE Oversight Division: Status report to the public, 72 p.
- 24 U.S. Department of Energy, 2000, Y-12 Mercury Task Force files—A guide to record series of the Department of Energy and its contractors: accessed on October 20, 2000, at <http://www.tis.eh.doe.gov/ohre/new/findingaids/epidemiologic/oakridge1/intro.html>
- 25 Olsen, C.R., Larsen, I.L., Lowry, P.D., Moriones, C.R., Ford, C.J., Deerstone, K.C., Turner, R.R., Kimmel, B.L., and Brandt, C.C., 1992, Transport and accumulation of cesium-137 and mercury in the Clinch River and Watts Bar Reservoir system: Oak Ridge National Laboratory, Environmental Sciences Division Publication 3471.
- 26 U.S. Geological Survey, 1996, The Ocoee River story: accessed on October 20, 2000, at <http://pubs.usgs.gov/gip/ocoe2>
- 27 Bartlett, R.A., 1995, Troubled waters—Champion International and the Pigeon River controversy: Knoxville, Tennessee, University of Tennessee Press, 348 p.
- 28 Swartz, R.C., 1999, Consensus sediment quality guidelines for polycyclic aromatic hydrocarbon mixtures: Environmental Toxicology and Chemistry, v. 18, no. 4, p. 780-787.
- 29 Ahlstedt, S.A. and Tuberville, J.D., 1997, Quantitative reassessment of the freshwater mussel fauna in the Clinch and Powell Rivers, Tennessee and Virginia, in Cummings, K.S., Buchanan, A.C., Mayer, C.A., and Naimo, T.J., eds., Conservation and management of freshwater mussels II. Proceedings of a Upper Mississippi River Conservation Committee symposium, October 16-18, 1995, St. Louis, Mo., p. 71-77.
- 30 State of Tennessee, 2000, Fish kill report database: Tennessee Wildlife Resources Agency.
- 31 State of North Carolina, 2000, North Carolina Division of Water Quality 2000 Fish kill event update: accessed on October 20, 2000, at <http://www.esb.enr.state.nc.us/Fishkill/fishkill00.htm>
- 32 Parmalee, P.W. and Bogan, A.E., 1998, The freshwater mussels of Tennessee: Knoxville, Tenn., University of Tennessee Press, 328 p.
- 33 The Nature Conservancy, 1993, Clinch River Valley Bioreserve, Strategic Plan: The Nature Conservancy, Abingdon, Va., 135 p.
- 34 Gilliom, R.J., Alley, W.M., and Gurtz, M.E., 1995, Design of the National Water-Quality Assessment Program: Occurrence and distribution of water-quality conditions: U.S. Geological Survey Circular 1112, 33 p.

APPENDIX—WATER-QUALITY DATA FROM THE UPPER TENNESSEE RIVER BASIN IN A NATIONAL CONTEXT

For a complete view of Upper Tennessee River Basin data and for additional information about specific benchmarks used, visit our Web site at <http://water.usgs.gov/nawqa/>. Also visit the NAWQA Data Warehouse for access to NAWQA data sets at <http://infotrek.er.usgs.gov/wdbctx/nawqa/nawqa.home>.

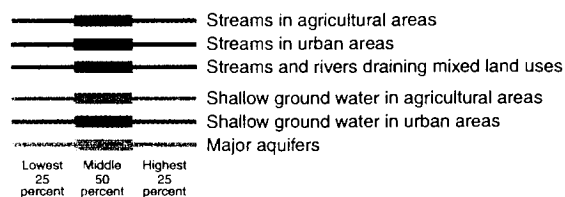
This appendix is a summary of chemical concentrations and biological indicators assessed in the Upper Tennessee River Basin. Selected results for this Basin are graphically compared to results from as many as 36 NAWQA Study Units investigated from 1991 to 1998 and to national water-quality benchmarks for human health, aquatic life, or fish-eating wildlife. The chemical and biological indicators shown were selected on the basis of frequent detection, detection at concentrations above a national benchmark, or regulatory or scientific importance. The graphs illustrate how conditions associated with each land use sampled in the Upper Tennessee River Basin compare to results from across the Nation, and how conditions compare among the several land uses. Graphs for chemicals show only detected concentrations and, thus, care must be taken to evaluate detection frequencies in addition to concentrations when comparing study-unit and national results. For example, tebuthiuron concentrations in Upper Tennessee River Basin major aquifers were similar to the national distribution, but the detection frequency was much higher (31 percent compared to 3 percent).

CHEMICALS IN WATER

Concentrations and detection frequencies, Upper Tennessee River Basin, 1995–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals

- ◆ Detected concentration in Study Unit
- 66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- Not measured or sample size less than two
- 12 Study-unit sample size. For ground water, the number of samples is equal to the number of wells sampled

National ranges of detected concentrations, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

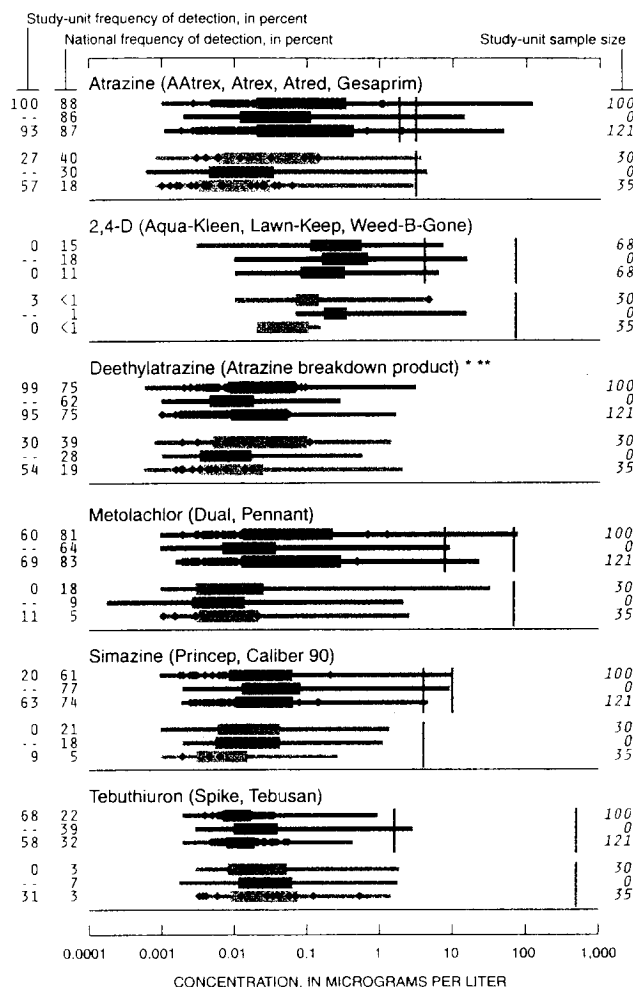


National water-quality benchmarks

National benchmarks include standards and guidelines related to drinking-water quality, criteria for protecting the health of aquatic life, and a goal for preventing stream eutrophication due to phosphorus. Sources include the U.S. Environmental Protection Agency and the Canadian Council of Ministers of the Environment

- | Drinking-water quality (applies to ground water and surface water)
- | Protection of aquatic life (applies to surface water only)
- | Prevention of eutrophication in streams not flowing directly into lakes or impoundments
- No benchmark for drinking-water quality
- ** No benchmark for protection of aquatic life

Pesticides in water—Herbicides



Other herbicides detected

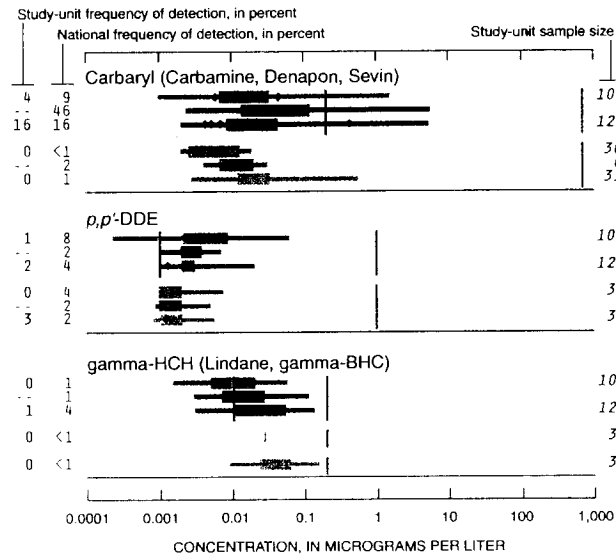
- Acetochlor (Harness Plus, Surpass) **
- Alachlor (Lasso, Bronco, Lariat, Bullet) **
- Bromacil (Hyvar X, Urox B, Bromax)
- Cyanazine (Bladex, Fortrol)
- DCPA (Dacthal, chlorthal-dimethyl) **
- Dichlorprop (2,4-DP, Seritox 50, Lentemul) **
- Diuron (Crisuron, Karmex, Diurex) **
- Metribuzin (Lexone, Sencor)
- Molinate (Ordram) ***
- Napropamide (Devrinol) **
- Pendimethalin (Pre-M, Prowl, Stomp) **
- Prometon (Pramitol, Princep) **
- 2,4,5-T **
- 2,4,5-TP (Silvex, Fenoprop) **
- Trifluralin (Treflan, Gowan, Tri-4, Trific)

Herbicides not detected

- Acifluorfen (Blazer, Tackle 2S) **
- Benfluralin (Balan, Benefin, Bonalan) **
- Bentazon (Basagran, Bentazone) **
- Bromoxynil (Buctril, Brominal) *
- Butylate (Sutan +, Genate Plus, Butilate) **
- Chloramben (Amiben, Amilon-WP, Vegiben) **
- Clopyralid (Stinger, Lontrel, Transline) ***
- 2,4-DB (Butyrac, Butoxone, Embutox Plus, Embutone) ***
- Dacthal mono-acid (Dacthal breakdown product) ***

Dicamba (Banvel, Dianat, Scotts Proturf)
 2,6-Diethylaniline (Alachlor breakdown product) * **
 Dinoseb (Dinosebe)
 EPTC (Eptam, Farmarox, Alirox) * **
 Ethalfuralin (Sonalan, Curbit) * **
 Fenuron (Fenuion, Fenidim) * **
 Fluometuron (Flo-Met, Cotoran) **
 Linuron (Lorox, Linex, Sarclex, Linurex, Afalon) *
 MCPA (Rhomene, Rhonox, Chiptox)
 MCPB (Thistrol) * **
 Neburon (Neburea, Neburyl, Noruben) * **
 Norflurazon (E vital, Predict, Solicam, Zorial) * **
 Oryzalin (Surflan, Dirimal) * **
 Pebulate (Tillam, PEBC) * **
 Picloram (Grazon, Tordon)
 Pronamide (Kerb, Propyzamid) **
 Propachlor (Ramrod, Satecid) **
 Propanil (Stam, Stampede, Wham) * **
 Propham (Tuberite) **
 Terbacil (Sinbar) **
 Thiobencarb (Bolero, Saturn, Benthiocarb) * **
 Triallate (Far-Go, Avadex BW, Tri-allate) *
 Triclopyr (Garlon, Grandstand, Redeem, Remedy) * **

Pesticides in water—Insecticides



Other insecticides detected

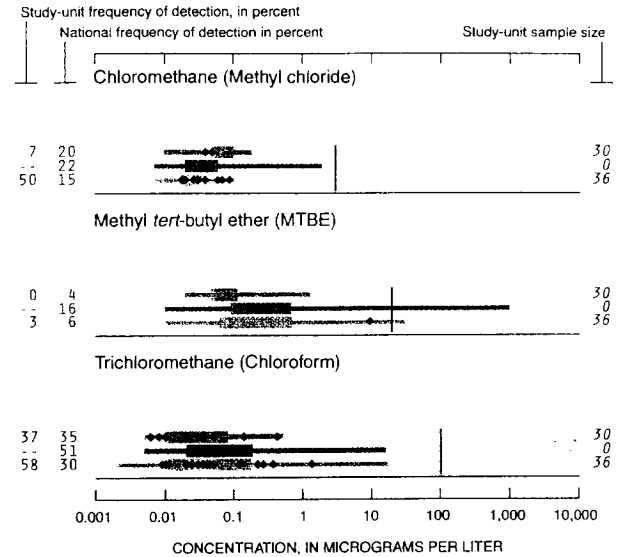
Carbofuran (Furadan, Curater, Yaltox)
 Chlorpyrifos (Brodan, Dursban, Lorsban)
 Diazinon (Basudin, Diazatol, Neocidol, Knox Out)
 Malathion (Malathion)

Insecticides not detected

Aldicarb (Temik, Ambush, Pounce)
 Aldicarb sulfone (Standak, aldoxycarb)
 Aldicarb sulfoxide (Aldicarb breakdown product)
 Azinphos-methyl (Guthion, Gusathion M) *
 Dieldrin (Panoram D-31, Octalox, Compound 497)
 Disulfoton (Disyston, Di-Syston) **
 Ethoprop (Mocap, Ethoprophos) * **
 Fonofos (Dyfonate, Capfos, Cudgel, Tycap) **
 alpha-HCH (alpha-BHC, alpha-lindane) **
 3-Hydroxycarbofuran (Carbofuran breakdown product) * **
 Methiocarb (Slug-Geta, Grandslam, Mesuroil) * **
 Methomyl (Lanox, Lannate, Acinate) **
 Methyl parathion (Pennacp-M, Folidol-M) **
 Oxamyl (Vydate L, Pratt) **
 Parathion (Roethyl-P, Alkron, Panthion, Phoskil) *
 cis-Permethrin (Ambush, Astro, Pounce) * **
 Phorate (Thimet, Granutox, Geomet, Rampart) * **
 Propargite (Comite, Omite, Ornamate) * **
 Propoxur (Baygon, Blattanex, Unden, Proprotox) * **
 Terbufos (Contraven, Counter, Pilarfox) **

Volatile organic compounds (VOCs) in ground water

These graphs represent data from 16 Study Units, sampled from 1996 to 1998



Other VOCs detected

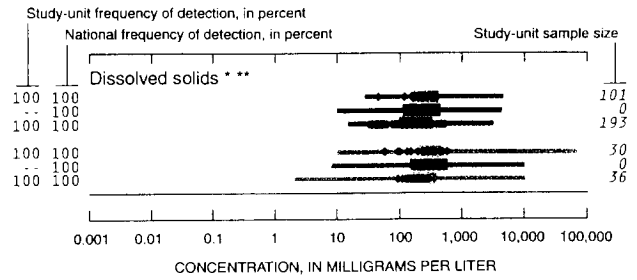
tert-Amylmeylether (tert-amyl methyl ether (TAME)) *
 Benzene
 Bromodichloromethane (Dichlorobromomethane)
 2-Butanone (Methyl ethyl ketone (MEK)) *
 n-Butylbenzene (1-Phenylbutane) *
 Carbon disulfide *
 Chlorobenzene (Monochlorobenzene)
 Chloroethane (Ethyl chloride) *
 1,3-Dichlorobenzene (m-Dichlorobenzene)
 1,4-Dichlorobenzene (p-Dichlorobenzene)
 Dichlorodifluoromethane (CFC 12, Freon 12)
 1,1-Dichloroethane (Ethylidene dichloride) *
 1,1-Dichloroethene (Vinylidene chloride)
 cis-1,2-Dichloroethene ((Z)-1,2-Dichloroethene)
 Diethyl ether (Ethyl ether) *
 Diisopropyl ether (Diisopropylether (DIPE)) *
 1,2-Dimethylbenzene (o-Xylene)
 1,3 & 1,4-Dimethylbenzene (m-&p-Xylene)
 Ethenylbenzene (Styrene)
 1-Ethyl-2-methylbenzene (2-Ethyltoluene) *
 Ethylbenzene (Phenylethane)
 Iodomethane (Methyl iodide) *
 Isopropylbenzene (Cumene) *
 p-Isopropyltoluene (p-Cymene) *
 Methylbenzene (Toluene)
 2-Propanone (Acetone) *
 n-Propylbenzene (Isocumene) *
 Tetrachloroethene (Perchloroethene)
 1,2,3,4-Tetramethylbenzene (Prehnitene) *
 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon 113) *
 1,1,1-Trichloroethane (Methylchloroform)
 Trichloroethene (TCE)
 1,2,3-Trimethylbenzene (Hemimellitene) *
 1,2,4-Trimethylbenzene (Pseudocumene) *
 1,3,5-Trimethylbenzene (Mesitylene) *

VOCs not detected

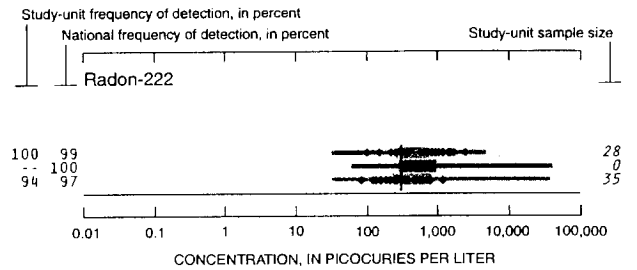
Bromobenzene (Phenyl bromide) *
 Bromochloromethane (Methylene chlorobromide)
 Bromoethene (Vinyl bromide) *
 Bromomethane (Methyl bromide)
 sec-Butylbenzene *
 tert-Butylbenzene *
 3-Chloro-1-propene (3-Chloropropene) *
 1-Chloro-2-methylbenzene (o-Chlorotoluene)
 1-Chloro-4-methylbenzene (p-Chlorotoluene)
 Chlorodibromomethane (Dibromochloromethane)
 Chloroethene (Vinyl chloride)
 1,2-Dibromo-3-chloropropane (DBCP, Nemagon)
 1,2-Dibromoethane (Ethylene dibromide, EDB)

- Dibromomethane (Methylene dibromide) *
- trans*-1,4-Dichloro-2-butene ((Z)-1,4-Dichloro-2-butene) *
- 1,2-Dichlorobenzene (*o*-Dichlorobenzene)
- 1,2-Dichloroethane (Ethylene dichloride)
- trans*-1,2-Dichloroethene ((E)-1,2-Dichloroethene)
- Dichloromethane (Methylene chloride)
- 1,2-Dichloropropane (Propylene dichloride)
- 2,2-Dichloropropane *
- 1,3-Dichloropropane (Trimethylene dichloride) *
- trans*-1,3-Dichloropropene ((E)-1,3-Dichloropropene)
- cis*-1,3-Dichloropropene ((Z)-1,3-Dichloropropene)
- 1,1-Dichloropropene *
- 1-4-Epoxy butane (Tetrahydrofuran, Diethylene oxide) *
- Ethyl methacrylate *
- Ethyl *tert*-butyl ether (Ethyl-*t*-butyl ether (ETBE)) *
- Hexachlorobutadiene
- 1,1,1,2,2,2-Hexachloroethane (Hexachloroethane)
- 2-Hexanone (Methyl butyl ketone (MBK)) *
- Methyl acrylonitrile *
- Methyl-2-methacrylate (Methyl methacrylate) *
- 4-Methyl-2-pentanone (Methyl isobutyl ketone (MIBK)) *
- Methyl-2-propenoate (Methyl acrylate) *
- Naphthalene
- 2-Propenenitrile (Acrylonitrile)
- 1,1,2,2-Tetrachloroethane *
- 1,1,1,2-Tetrachloroethane
- Tetrachloromethane (Carbon tetrachloride)
- 1,2,3,5-Tetramethylbenzene (Isodurene) *
- Tribromomethane (Bromoform)
- 1,2,4-Trichlorobenzene
- 1,2,3-Trichlorobenzene *
- 1,1,2-Trichloroethane (Vinyl trichloride)
- Trichlorofluoromethane (CFC 11, Freon 11)
- 1,2,3-Trichloropropane (Allyl trichloride)

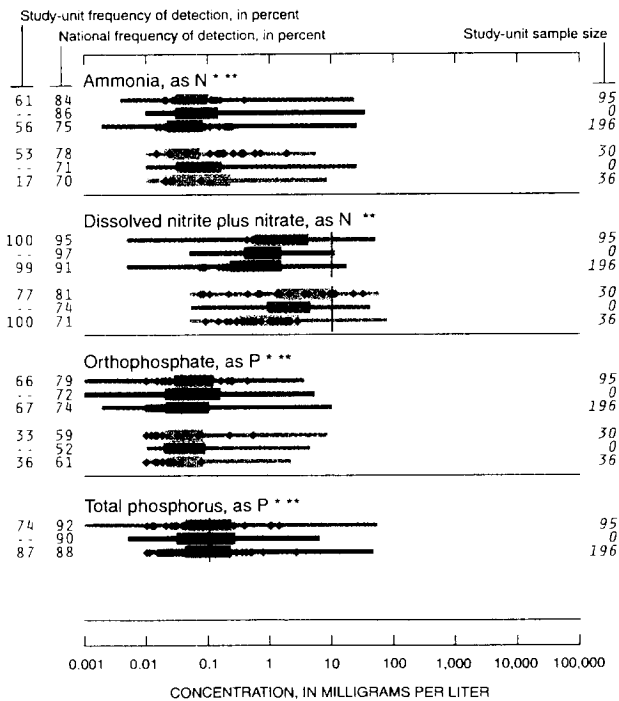
Dissolved solids in water



Trace elements in ground water



Nutrients in water



Other nutrients detected

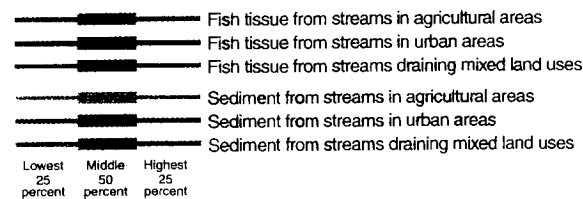
Dissolved ammonia plus organic nitrogen as N ***

CHEMICALS IN FISH TISSUE AND BED SEDIMENT

Concentrations and detection frequencies, Upper Tennessee River Basin, 1995–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals. Study-unit frequencies of detection are based on small sample sizes; the applicable sample size is specified in each graph

- ◆ Detected concentration in Study Unit
- 66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- Not measured or sample size less than two
- 12 Study-unit sample size

National ranges of concentrations detected, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

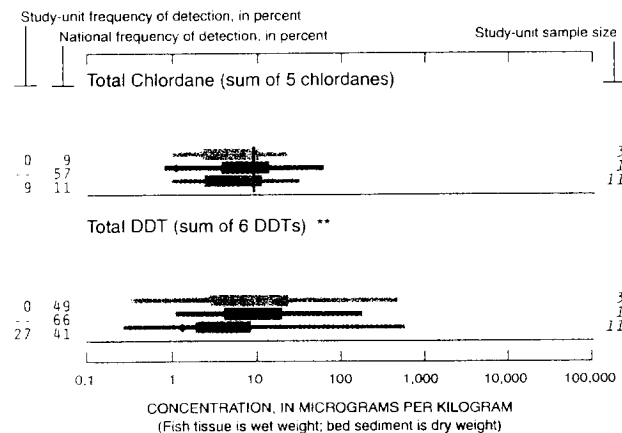


National benchmarks for fish tissue and bed sediment

National benchmarks include standards and guidelines related to criteria for protection of the health of fish-eating wildlife and aquatic organisms. Sources include the U.S. Environmental Protection Agency, other Federal and State agencies, and the Canadian Council of Ministers of the Environment

- | Protection of fish-eating wildlife (applies to fish tissue)
- | Protection of aquatic life (applies to bed sediment)
- No benchmark for protection of fish-eating wildlife
- ** No benchmark for protection of aquatic life

Organochlorines in fish tissue (whole body) and bed sediment



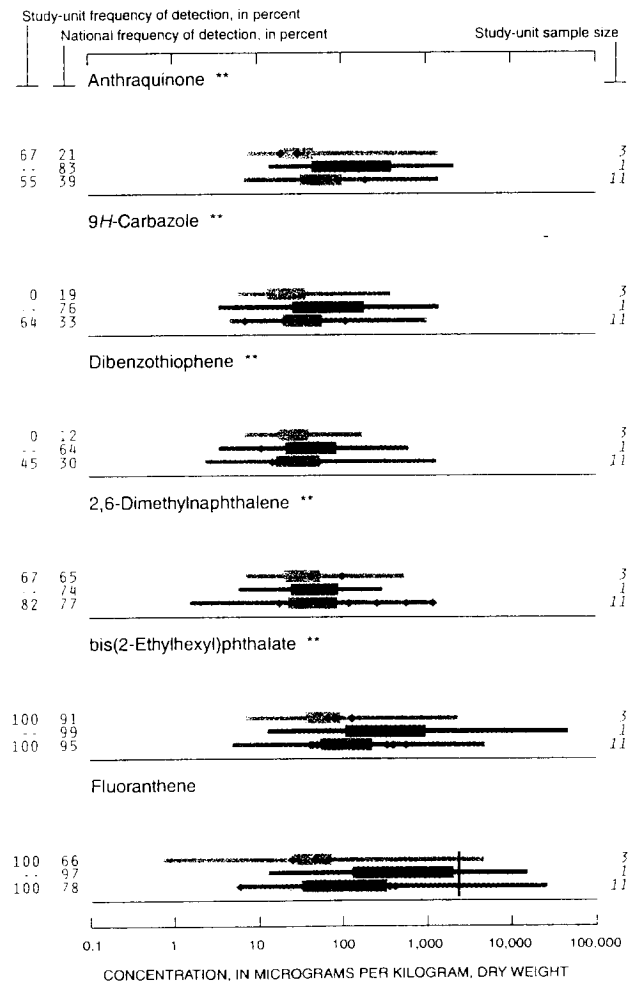
Other organochlorines detected

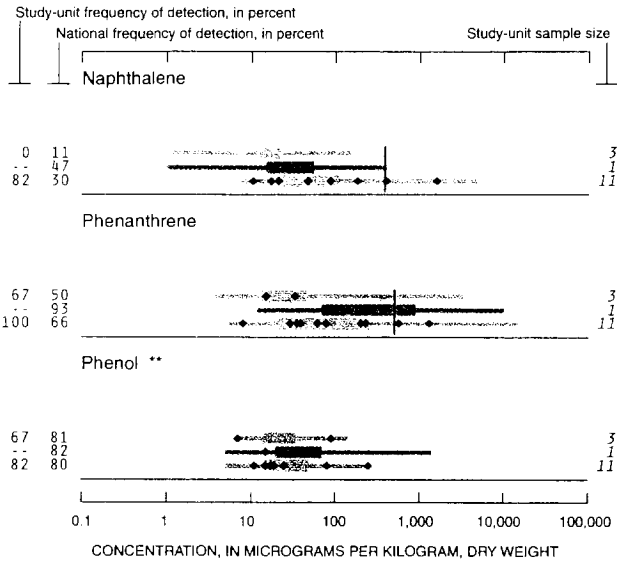
- o,p'*+*p,p'*-DDD (sum of *o,p'*-DDD and *p,p'*-DDD) *
- p,p'*-DDE * **
- o,p'*+*p,p'*-DDE (sum of *o,p'*-DDE and *p,p'*-DDE) *
- o,p'*+*p,p'*-DDT (sum of *o,p'*-DDT and *p,p'*-DDT) *

Organochlorines not detected

- Chloroneb (Chloronebe, Demosan) * **
- DCPA (Dacthal, chlorthal-dimethyl) * **
- Endosulfan I (alpha-Endosulfan, Thiodan) * **
- Endrin (Endrine)
- gamma-HCH (Lindane, gamma-BHC, Gammexane) *
- Heptachlor epoxide (Heptachlor breakdown product) *
- Heptachlor+heptachlor epoxide (sum of heptachlor and heptachlor epoxide) **
- Hexachlorobenzene (HCB) **
- Isodrin (Isodrine, Compound 711) * **
- p,p'*-Methoxychlor (Marlate, methoxychlore) * **
- o,p'*-Methoxychlor * **
- Mirex (Dechlorane) **
- Total PCB
- Pentachloroanisole (PCA) * **
- cis*-Permethrin (Ambush, Astro, Pounce) * **
- trans*-Permethrin (Ambush, Astro, Pounce) * **
- Toxaphene (Camphechlor, Hercules 3956) * **

Semivolatile organic compounds (SVOCs) in bed sediment





Other SVOCs detected

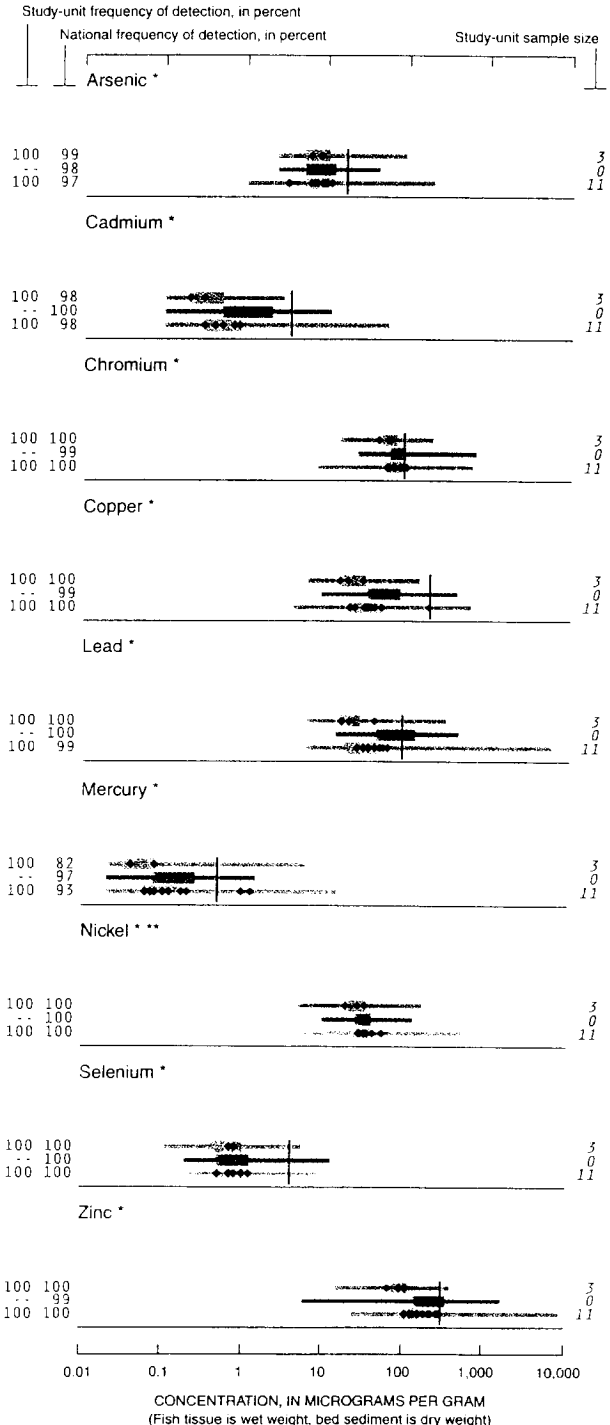
- Acenaphthene
- Acenaphthylene
- Acridine **
- C8-Alkylphenol **
- Anthracene
- Benz[a]anthracene
- Benzo[a]pyrene
- Benzo[b]fluoranthene **
- Benzo[ghi]perylene **
- Benzo[k]fluoranthene **
- 2,2-Biquinoline **
- Butylbenzylphthalate **
- Chrysene
- p-Cresol **
- Di-n-butylphthalate **
- Di-n-octylphthalate **
- Dibenz[a,h]anthracene
- Diethylphthalate **
- 1,2-Dimethylnaphthalene **
- 1,6-Dimethylnaphthalene **
- Dimethylphthalate **
- 2-Ethynaphthalene **
- 9H-Fluorene (Fluorene)
- Indeno[1,2,3-cd]pyrene **
- Isophorone **
- Isoquinoline **
- 1-Methyl-9H-fluorene **
- 2-Methylantracene **
- 4,5-Methylenephenanthrene **
- 1-Methylphenanthrene **
- 1-Methylpyrene **
- Phenanthridine **
- Pyrene
- Quinoline **
- 2,3,6-Trimethylnaphthalene **

SVOCs not detected

- Azobenzene **
- Benzo[c]cinnoline **
- 4-Bromophenyl-phenylether **
- 4-Chloro-3-methylphenol **
- bis(2-Chloroethoxy)methane **
- bis(2-Chloroethyl)ether **
- 2-Chloronaphthalene **
- 2-Chlorophenol **
- 4-Chlorophenyl-phenylether **
- 1,2-Dichlorobenzene (o-Dichlorobenzene) **
- 1,3-Dichlorobenzene (m-Dichlorobenzene) **
- 1,4-Dichlorobenzene (p-Dichlorobenzene) **
- 3,5-Dimethylphenol **
- 2,4-Dinitrotoluene **

- Nitrobenzene **
- N-Nitrosodi-n-propylamine **
- N-Nitrosodiphenylamine **
- Pentachloronitrobenzene **
- 1,2,4-Trichlorobenzene **

Trace elements in fish tissue (livers) and bed sediment



BIOLOGICAL INDICATORS

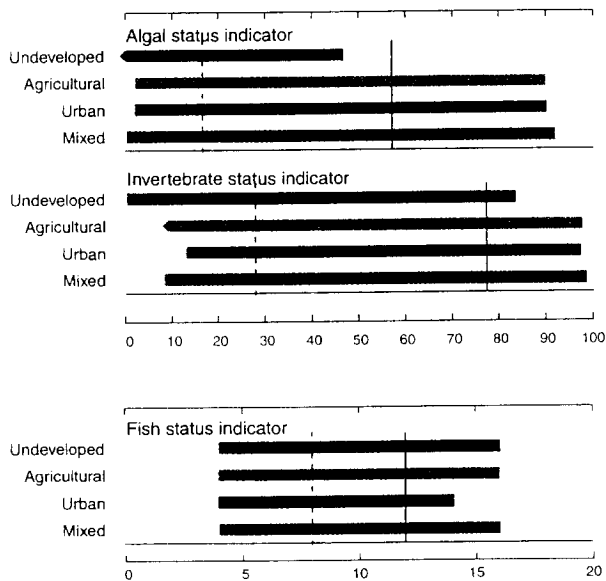
Higher national scores suggest habitat disturbance, water-quality degradation, or naturally harsh conditions. The status of algae, invertebrates (insects, worms, and clams), and fish provide a record of water-quality and stream conditions that water-chemistry indicators may not reveal. **Algal status** focuses on the changes in the percentage of certain algae in response to increasing siltation, and it often correlates with higher nutrient concentrations in some regions. **Invertebrate status** averages 11 metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. **Fish status** sums the scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that increase in association with water-quality degradation

Biological indicator value, Upper Tennessee River Basin, by land use, 1995-98

- ◆ Biological status assessed at a site

National ranges of biological indicators, in 16 NAWQA Study Units, 1994-98

- Streams in undeveloped areas
- Streams in agricultural areas
- Streams in urban areas
- Streams in mixed-land-use areas
- 75th percentile
- - - 25th percentile



A COORDINATED EFFORT

Coordination with agencies and organizations in the Upper Tennessee River Basin was integral to the success of this water-quality assessment. We thank those who served as members of our liaison committee.

Federal Agencies

Tennessee Valley Authority
U.S. Fish and Wildlife Service
National Park Service
U.S. Department of Energy,
 Oak Ridge National Laboratory
U.S. Environmental Protection Agency
U.S. Forest Service
U.S. Department of Agriculture,
 Natural Resources Conservation Service

State Agencies

Tennessee Wildlife Resources Agency
Tennessee Department of Environment and
 Conservation
Tennessee Department of Agriculture
North Carolina Department of Environment and
 Natural Resources
North Carolina Wildlife Resources Commission
Virginia Department of Environmental Quality
Virginia Department of Game and Inland Fisheries
Virginia Department of Mines, Minerals, and Energy

Local Agencies

Knox County, Tennessee
City of Johnson City, Tennessee

Universities

University of Tennessee
Virginia Polytechnic and State University
Tennessee Technological University

Other public and private organizations

Southern Appalachian Man and the Biosphere
 Program
Nature Conservancy

We thank the following individuals for contributing to this effort.

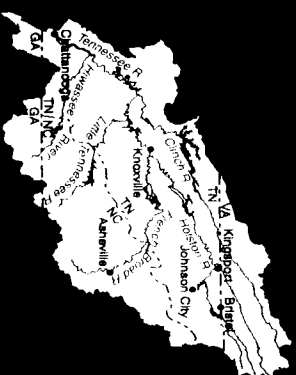
Edward Oaksford, Ben McPherson, Michael Woodside, Rebecca Deckard, and Sandra Cooper (USGS), Roberta Hylton (U.S. Fish and Wildlife Service), Karen Koehn and Celia Hampson (Knox County, Tennessee) for reviewing the report.

Charles Saylor and Edward Scott (Tennessee Valley Authority) for assistance in site selection and data collection.

The numerous property owners that allowed the use of their property by the USGS for access to specific stream reaches, the installation of monitoring wells, or the sampling of existing wells.

NAWQA

National Water-Quality Assessment (NAWQA) Program
Upper Tennessee River Basin



Hampson and others—Water Quality in the Upper Tennessee River Basin
U.S. Geological Survey Circular 1205



ISBN 0-607-95545-5
9 1780607195546 0

R0024315

Water Quality in the Santee River Basin and Coastal Drainages

North and South Carolina, 1995–98



POINTS OF CONTACT AND ADDITIONAL INFORMATION

The companion Web site for NAWQA Study Unit summary reports:

<http://water.usgs.gov/nawqa/>

Santee River Basin and coastal drainages contact and Web site:

USGS State Representative
U.S. Geological Survey
Water Resources Division
Stephenson Center, Suite 129
720 Gracern Road
Columbia, SC 29210-7651
e-mail: msdavenp@usgs.gov
<http://sc.water.usgs.gov/nawqa/santhome.html>

National NAWQA Program:

Chief, NAWQA Program
U.S. Geological Survey
Water Resources Division
12201 Sunrise Valley Drive, M.S. 413
Reston, VA 20192
<http://water.usgs.gov/nawqa/>

Other NAWQA summary reports

River Basin Assessments

Albemarle-Pamlico Drainage Basin (Circular 1157)
Allegheny and Monongahela River Basins (Circular 1202)
Apalachicola-Chattahoochee-Flint River Basin (Circular 1164)
Central Arizona Basins (Circular 1213)
Central Columbia Plateau (Circular 1144)
Central Nebraska Basins (Circular 1163)
Connecticut, Housatonic and Thames River Basins (Circular 1155)
Eastern Iowa Basins (Circular 1210)
Georgia-Florida Coastal Plain (Circular 1151)
Hudson River Basin (Circular 1165)
Kanawha-New River Basins (Circular 1204)
Lake Erie-Lake Saint Clair Drainages (Circular 1203)
Las Vegas Valley Area and the Carson and Truckee River Basins (Circular 1170)
Lower Illinois River Basin (Circular 1209)
Long Island-New Jersey Coastal Drainages (Circular 1201)
Lower Susquehanna River Basin (Circular 1168)
Mississippi Embayment (Circular 1208)
Ozark Plateaus (Circular 1158)
Potomac River Basin (Circular 1166)
Puget Sound Basin (Circular 1216)

Red River of the North Basin (Circular 1169)
Rio Grande Valley (Circular 1162)
Sacramento River Basin (Circular 1215)
San Joaquin-Tulare Basins (Circular 1159)
South-Central Texas (Circular 1212)
South Platte River Basin (Circular 1167)
Southern Florida (Circular 1207)
Trinity River Basin (Circular 1171)
Upper Colorado River Basin (Circular 1214)
Upper Mississippi River Basin (Circular 1211)
Upper Snake River Basin (Circular 1160)
Upper Tennessee River Basin (Circular 1205)
Western Lake Michigan Drainages (Circular 1156)
White River Basin (Circular 1150)
Willamette Basin (Circular 1161)

National Assessments

The Quality of Our Nation's Waters—Nutrients and Pesticides (Circular 1225)

Front cover: Edisto River near Givhans, South Carolina.

Back cover: Left—Surface-water sampling, Indian Creek, North Carolina; Middle—Fish identification, Jacob Fork River, North Carolina.

All photographs in this report were taken by members of the Santee River Basin and coastal drainages Study Unit, U.S. Geological Survey.

Errata Sheet

U.S. Geological Survey Circular 1206

Subsequent to publication of U.S. Geological Survey Circular 1206, "Water Quality in the Santee River Basin and Coastal Drainages, North and South Carolina, 1995-98," errors were found on pages 15 and 20.

On page 15, the unit of measurement used in figure 15 to express the median concentration of trace elements in bed sediment and tissue should have been micrograms per gram, rather than micrograms per kilogram. A correct figure 15 is below.

On page 20, the label on the left side of the top graph of figure 18 for the unit of measurement used to express the median concentration of atrazine erroneously included ammonia. The label for the unit of measurement used to express the median concentration of ammonia should have been on the right side of the top graph. A correct figure 18 is below.

We apologize for any inconvenience these errors may have caused.

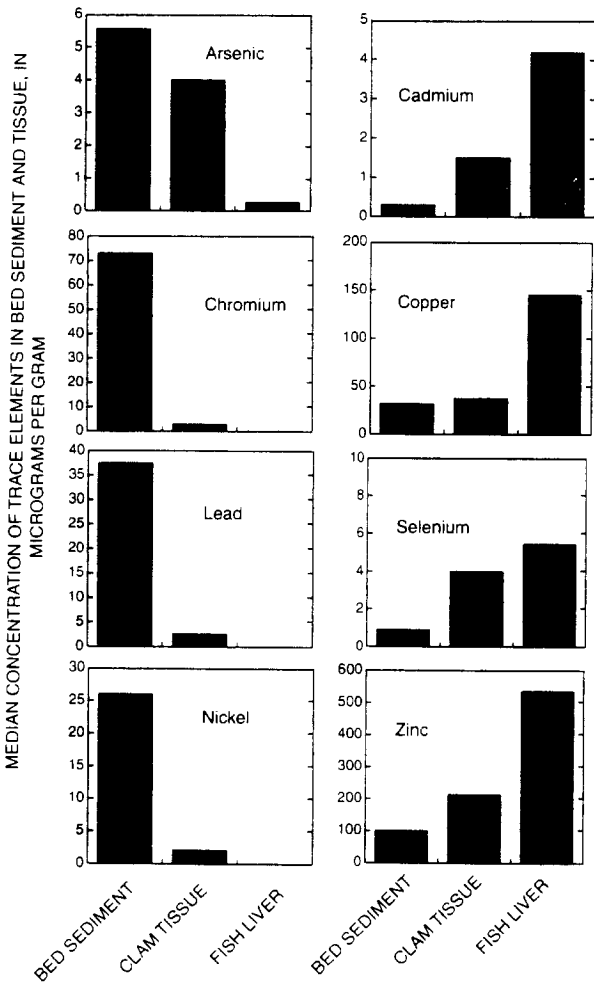


Figure 15. Cadmium, copper, selenium, and zinc were detected at higher concentrations in clam and fish tissue than in sediment, suggesting that they accumulate in the tissues. Conversely, arsenic, chromium, lead, and nickel were detected in lower concentrations in tissues than in sediment, indicating that these metals do not accumulate in the tissues.

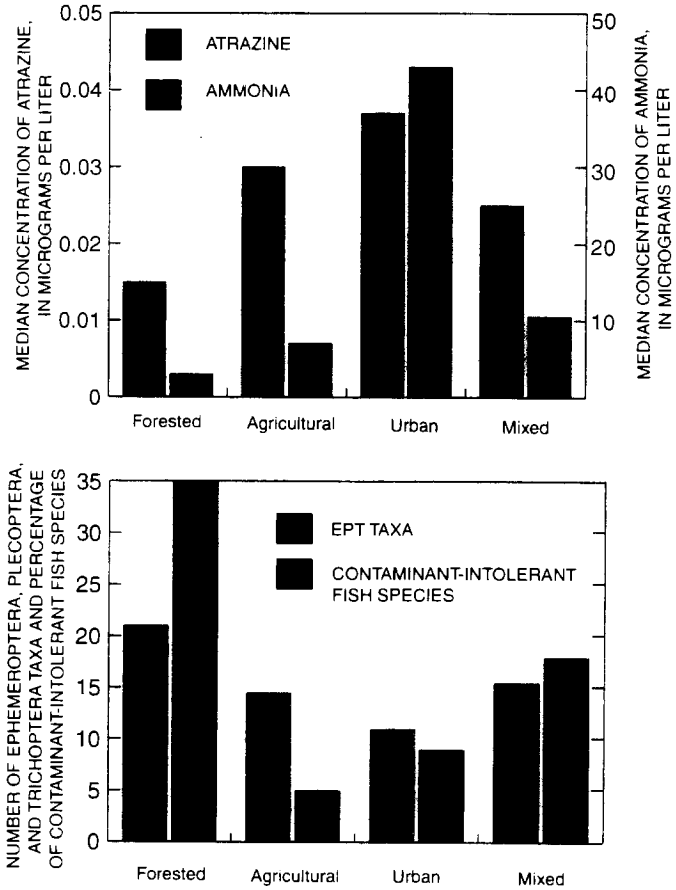


Figure 18. Compared to forested sites, urban and agricultural sites have higher concentrations of atrazine and ammonia as well as lower numbers of fish and invertebrates that are intolerant of contamination.

Water Quality in the Santee River Basin and Coastal Drainages, North and South Carolina, 1995–98

By W. Brian Hughes, Thomas A. Abrahamsen, Terry L. Maluk, Eric J. Reuber, *and* Lance J. Wilhelm

U.S. DEPARTMENT OF THE INTERIOR
BRUCE BABBITT, Secretary

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

2000

Free on application to the
U.S. Geological Survey
Information Services
Box 25286 Federal Center
Denver, CO 80225

Or call: 1-888-ASK-USGS

Library of Congress Cataloging-in-Publications Data

Water-quality assessment of the Santee River Basin and coastal drainages, North and South Carolina, 1995-98
/ by W. Brian Hughes...[et al].
p. cm. -- (U.S. Geological Survey Circular ; 1206)
Includes bibliographical references.
ISBN 0-607-95422-1 (alk. paper)
1. Water quality--South Carolina--Santee River Watershed. 2. Water quality--North Carolina. I. Hughes, W. Brian. II. Geological Survey (U.S.) III. Series.

TD224.S6 W384 2000
363.739'42'097578--dc21

00-049461

R0024320

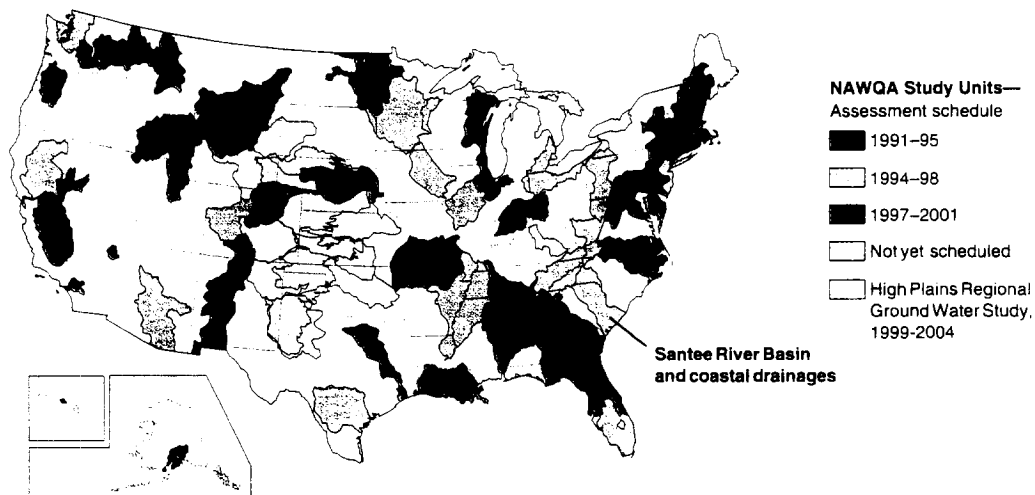
CONTENTS

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM.....	IV
SUMMARY OF MAJOR FINDINGS	1
Stream and River Highlights	1
Ground-Water Highlights	2
INTRODUCTION TO THE SANTEE RIVER BASIN AND COASTAL DRAINAGES	3
Physiography and Water Quality	3
Land Use and Water Quality.....	3
Climate Conditions and Water Quality	4
Water Use.....	4
Flow Regulation, Impoundments, and Surface-Water Quality	5
Aquifer Characteristics and Ground-Water Quality.....	5
MAJOR FINDINGS	6
Pesticides Commonly Were Detected in Santee Basin Streams.....	6
Few Pesticides Were Detected in Drinking-Water Supply Aquifers	7
Pesticides Were Common in Shallow Ground Water in Urban and Agricultural Areas.....	8
Organochlorine Pesticides Were Detected in Bed Sediment and Tissues	8
NATIONAL PERSPECTIVE—Santee Basin Pesticide Findings Were Similar to National Findings.....	9
Phosphorus Concentrations in Streams Frequently Exceeded USEPA Goals	10
Water-Supply Aquifers Rarely Exceeded Drinking-Water Standards for Nitrate.....	11
NATIONAL PERSPECTIVE—Phosphorus Concentrations in the Santee Basin Were in the Middle Range of National Results	13
Bed Sediment Had Low Concentrations of Trace Elements.....	14
NATIONAL PERSPECTIVE—Mercury Contamination Potential in Santee Basin Was Highest in Nation....	14
Trace Elements Accumulated in Clam Tissue and Fish Livers	15
Radon Exceeded Proposed Standards in Many Wells	16
NATIONAL PERSPECTIVE—Radon Concentrations Were High in the Santee Basin	16
Volatile Organic Compounds Were Common in Urban Ground Water.....	17
Drinking-Water Aquifers Had Low Concentrations of Volatile Organic Compounds.....	18
Urban and Agricultural Streams Had High Concentrations of Bacteria	19
Biological Communities Reflected Land-Use Differences	20
NATIONAL PERSPECTIVE—Santee Basin Streams were in the Middle of National Biological Rankings..	21
STUDY UNIT DESIGN.....	22
GLOSSARY	24
REFERENCES.....	25
APPENDIX—WATER-QUALITY DATA FROM THE SANTEE RIVER BASIN AND COASTAL DRAINAGES IN A NATIONAL CONTEXT	26

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

THIS REPORT summarizes major findings about water quality in the Santee River Basin and coastal drainages that emerged from an assessment conducted between 1995 and 1998 by the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program. Water quality is discussed in terms of local and regional issues and compared to conditions found in all 36 NAWQA study areas, called Study Units, assessed to date. Findings are also explained in the context of selected national benchmarks, such as those for drinking-water quality and the protection of aquatic organisms. The NAWQA program was not intended to assess the quality of the Nation's drinking water, such as by monitoring water from household taps. Rather, the assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water monitoring programs. The comparisons made in this report to drinking-water standards and guidelines are only in the context of the available untreated resource. Finally, this report includes information about the status of aquatic communities and the condition of in-stream habitats as elements of a complete water-quality assessment.

Many topics covered in this report reflect the concerns of officials of State and Federal agencies, water-resource managers, and members of stakeholder groups who provided advice and input during the Santee River Basin and coastal drainages assessment. Basin residents who wish to know more about water quality in the areas where they live will find this report informative as well.



THE NAWQA PROGRAM seeks to improve scientific and public understanding of water quality in the Nation's major river basins and ground-water systems. Better understanding facilitates effective resource management, accurate identification of water-quality priorities, and successful development of strategies that protect and restore water quality. Guided by a nationally consistent study design and shaped by ongoing communication with local, State, and Federal agencies, NAWQA assessments support the investigation of local issues and trends while providing a firm foundation for understanding water quality at regional and national scales. The ability to integrate local and national scales of data collection and analysis is a unique feature of the USGS NAWQA Program.

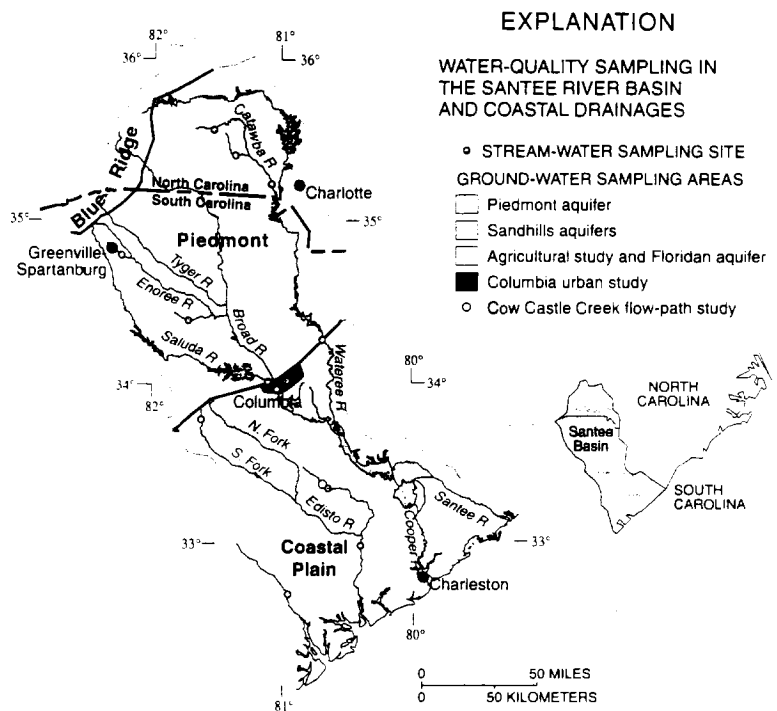
The Santee River Basin and coastal drainages is one of 51 water-quality assessments initiated since 1991, when the U.S. Congress appropriated funds for the USGS to begin the NAWQA Program. As indicated on the map, 36 assessments have been completed, and 15 more assessments will conclude in 2001. Collectively, these assessments cover about one-half of the land area of the United States and include water resources that are available to more than 60 percent of the U.S. population.

SUMMARY OF MAJOR FINDINGS

Stream and River Highlights

Surface water sampled in the Santee River Basin and coastal drainages generally meets existing Federal and State guidelines for drinking-water quality and protection of aquatic life. However, urban and agricultural land uses have affected water quality, as indicated by elevated concentrations of bacteria, pesticides, and nutrients in basins dominated by these land uses.

- The herbicides atrazine, simazine, and tebuthiuron were detected in almost every stream in the Santee Basin, including those in forested areas, at levels below aquatic-life and drinking-water guidelines. Four insecticides—malathion, diazinon, chlorpyrifos, and parathion—exceeded aquatic-life guidelines. No pesticides exceeded drinking-water standards, though 7 of the 30 compounds detected do not have drinking-water standards and 13 do not have aquatic guidelines. Pesticide concentrations had seasonal patterns,



Selected Indicators of Stream-Water Quality

	Small Streams			Major Rivers
	Urban	Agricultural	Undeveloped/Forest	Mixed Land Uses
Pesticides ¹	~	~	~	~
Nutrients ²	~	~	~	~
Trace elements ³	~	~	~	~
Organochlorines ⁴	~	~	~	~
Volatile organics ⁵	~	~	~	~
Bacteria ⁶	~	~	~	~
Semivolatile organics ⁷	~	~	~	~

Percentage of samples with concentrations equal to or greater than a health-related national guideline for drinking water, aquatic life, or contact recreation; or above a national goal for preventing excess algal growth

Percentage of samples with concentrations less than a health-related national guideline for drinking water, aquatic life, or water-contact recreation; or below a national goal for preventing excess algal growth

Percentage of samples with no detection

Not assessed ~ Insufficient data

¹ Insecticides, herbicides, and pesticide metabolites, sampled in water.
² Total phosphorus and nitrate (as nitrogen), sampled in water.
³ Arsenic, mercury, and metals, sampled in sediment.
⁴ Organochlorine compounds including DDT and PCB's, sampled in sediment.
⁵ Solvents, refrigerants, fumigants, and gasoline compounds, sampled in water.
⁶ Fecal coliform bacteria, sampled in water.
⁷ Miscellaneous industrial chemicals and combustion by-products, sampled in sediment.

The Santee River Basin and coastal drainages (the "Santee Basin") is an approximately 24,000-square-mile area in North and South Carolina that encompasses the Blue Ridge Mountains, the Piedmont, and the Coastal Plain (Fenneman, 1946). Most of the 3.5 million people in the Santee Basin live in urban areas. Eighty-six percent of the water used in homes and for industry is treated surface water withdrawn from rivers or reservoirs. Ground water is the main water source for rural households.

with the highest concentrations measured in the spring following application.

- Nitrate concentrations did not exceed drinking-water standards in any streams sampled. Average total phosphorus concentrations in four streams were above the U.S. Environmental Protection Agency (USEPA) recommended goal to prevent nuisance aquatic growth. The South Fork Catawba River had an average total phosphorus concentration that was four times higher than the USEPA goal and is a significant source of phosphorus to downstream lakes. Wastewater discharge and agricultural runoff are major sources of nitrogen and phosphorus.
- Trace metals were detected frequently in bed sediment and tissue, mostly at concentrations within aquatic-life guidelines. Arsenic, chromium, and lead exceeded guidelines in a few samples. Although concentrations were not high in sediment samples, data suggest that mercury is accumulating in fish and clams in concentrations that are harmful to humans or animals that eat them. Sampling by State agencies has

resulted in fish-consumption advisories for mercury in 49 rivers and reservoirs in South Carolina.

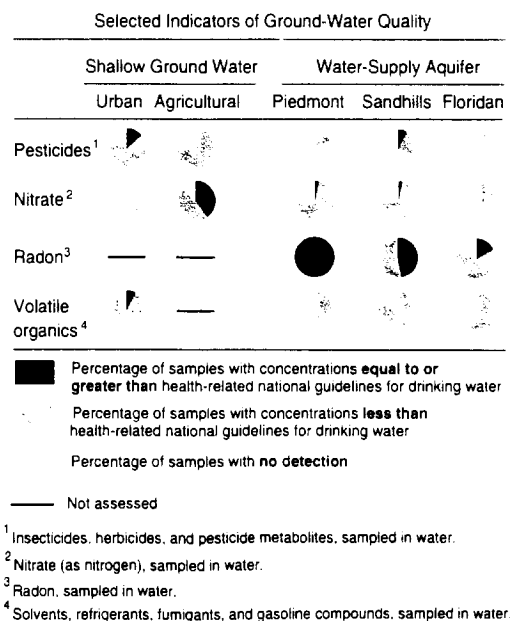
- Organochlorine pesticides were detected frequently in bed sediment and tissue. Most of these compounds have been discontinued for use for many years but continue to be detected because they are persistent in the environment. A derivative of DDT was detected at concentrations exceeding aquatic-life standards in sediment at three agricultural sites.
- Volatile organic compounds (VOCs) known to occur in the aquifer adjacent to Gills Creek, an urban stream in Columbia, S.C., were frequently detected in the creek as well. Although no existing Federal or State drinking-water standards or aquatic guidelines were exceeded, this finding is consistent with the important influence of ground-water quality on stream-water quality.
- Bacteria levels frequently exceeded South Carolina standards for contact recreation in streams in forested, urban, and agricultural areas. Standards were exceeded more frequently in small streams than in large rivers.
- Biological communities in urban and agricultural streams had fewer species of fish and invertebrates that can tolerate contamination than forested and mixed land-use streams. This suggests that contaminants resulting from these land uses affect the natural communities that live in these areas, although factors such as habitat alteration can cause similar changes in biological communities.

Ground-Water Highlights

Ground water in the Santee Basin generally meets existing Federal and State standards for drinking water except with respect to nitrate, which failed to meet the drinking-water standard in almost one-half of the shallow monitoring wells sampled in agricultural areas, and radon, which did not meet proposed standards in about one-half the drinking-water wells sampled basinwide. Pesticides were detected frequently in urban, agricultural, and drinking-water supply wells, but only two samples exceeded drinking-water standards. Many wells contained low concentrations of numerous synthetic chemicals related to industry, household use, and motor vehicles, and a few of these chemicals were at levels above drinking-water standards.

- Pesticides were detected in 17 of 90 wells sampled in drinking-water supply aquifers. Of the 34 detected, only 2 pesticides exceeded drinking-water standards, but 11 of the detected pesticides do not have standards. Most wells in agricultural and urban areas contained at least one pesticide, but only two wells in urban areas had concentrations that exceeded drinking-water standards.

- Nitrate is the only nutrient that was detected in significant concentrations in ground water, and it exceeded drinking-water standards in almost one-half of the shallow monitoring wells in agricultural areas. Although this finding indicates significant contamination of shallow ground water, most drinking-water wells are located in deeper aquifers. Nitrate in the Piedmont and Sandhills aquifers was elevated above natural concentrations but exceeded standards in only two wells. The Floridan aquifer, which is protected for the most part by confining units in the study area, had relatively low concentrations of nitrate.
- Radon, a naturally occurring radioactive gas, was detected in almost all wells sampled in drinking-water aquifers. Over one-half of the wells had concentrations that exceeded proposed Federal drinking-water standards. Most of the wells with radon concentrations that exceeded the proposed standard are located in the Piedmont aquifer.
- VOCs were detected in 27 of 30 monitoring wells in an urban setting. Most compounds were detected at extremely low levels; however, the concentrations of trichloroethylene, a solvent, and methyl *tert*-butyl ether, a gasoline additive, were above a drinking-water standard and an advisory level, respectively, in one well each. Thirteen of the 35 compounds detected do not have drinking-water standards. VOCs in drinking-water supply aquifers were detected throughout the Santee Basin, but no concentrations exceeded drinking-water standards.



INTRODUCTION TO THE SANTEE RIVER BASIN AND COASTAL DRAINAGES

The Santee River Basin and coastal drainages includes about 24,000 square miles in North and South Carolina. The Santee River has the second largest drainage area in the Eastern United States, and its basin makes up 70 percent of the study area. The basins of the Cooper, Edisto, and numerous smaller rivers make up the remainder of the study area. Throughout this report, the study area, including these smaller river basins, will be collectively referred to as the "Santee Basin."

Physiography and Water Quality

The rugged mountains of the Blue Ridge physiographic province are sparsely populated. Land-use effects on water quality are minimal because the area is largely undeveloped. Consequently, the Blue Ridge has more pristine water quality and intact stream ecosystems than other parts of the study area.

The rolling hills and abundant water resources of the Piedmont have attracted industrial development and human population growth. Many of the areas experiencing urban growth are in the Piedmont, near the headwaters of rivers that supply drinking water and also receive treated waste water. Since flows in these headwater streams are smaller than they are farther along the stream courses, they have a limited capacity to assimilate large quantities of wastewater and nonpoint-source inputs from the urban areas.

The flat-lying topography and fertile soils of the Coastal Plain are ideal for agricultural use. Development for shipping, industry, and tourism is mostly limited to land within a few miles of the coast.

Most of the Coastal Plain is characterized by slow-moving, low-gradient streams that commonly are bordered by extensive swamps (Smock and Gilinsky, 1992). The combination of slow-moving water and large quantities of organic matter in the swamps results in a characteristically dark-colored water called "blackwater." Under natural conditions, blackwater streams have low pH and contain low concentrations of dissolved oxygen. These conditions can make the stream particularly susceptible to water-quality degradation by the contribution of oxygen-consuming chemicals in wastewater discharge or nonpoint contamination.

Land Use and Water Quality

The Santee Basin has a rapidly growing population of about 3.5 million people. Most of the people live in the urban areas of Charlotte, N.C., and Greenville-Spartanburg, Columbia, and Charleston, S.C. (fig. 1). The most common types of urban development are commercial and residential.

As urban areas develop, increased use of pesticides and fertilizers on lawns and landscaped areas can lead to increased concentrations of these chemicals in ground and surface waters. Commercial and residential use of solvents and fuel products can result in their introduction to ground and

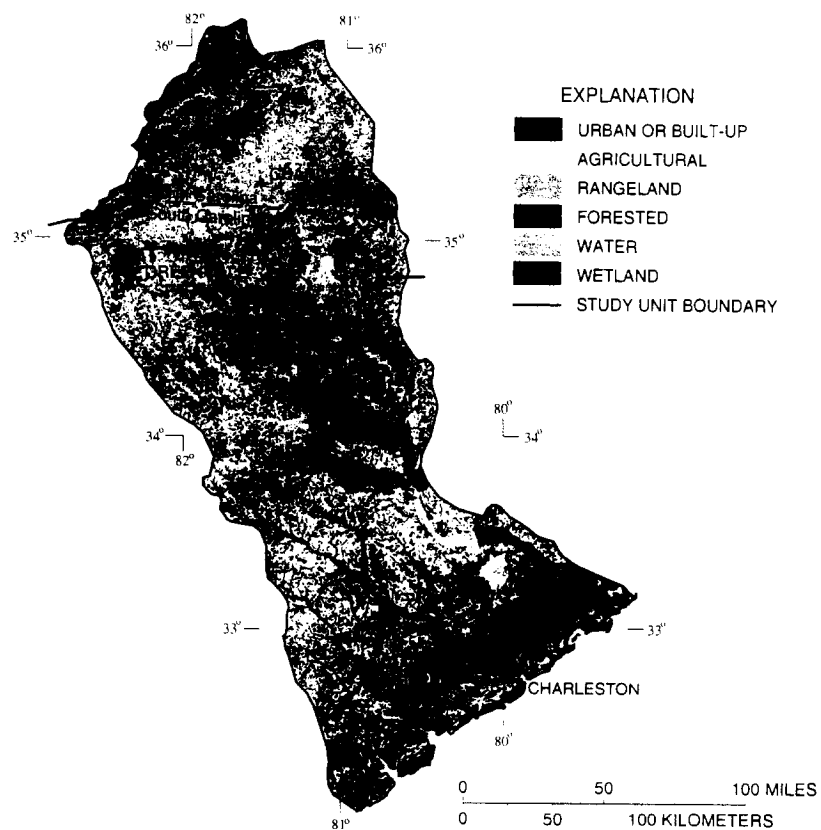


Figure 1. Land use in the Santee Basin includes about 60 percent forested, 30 percent agricultural, and 6 percent urban lands.

surface water through accidental spills or leaking storage tanks. Bacteria and nutrients can enter water through leaking sewer lines, malfunctioning septic tanks, and from runoff of pet and waterfowl wastes. Although thoroughly regulated, discharges from wastewater treatment plants increase as population grows, increasing the loading of nutrients to streams.

Agriculture is an important economic activity throughout the Santee Basin. Row crop agriculture is most common in the Coastal Plain, where corn, soybeans, and cotton are the most common crops (South Carolina Agricultural Statistics Service, 1999). Pasture for hay and for grazing cattle is typical of agricultural land in the Piedmont. These agricultural activities can result in elevated nutrient and pesticide levels in streams and ground water from runoff or infiltration of manure, fertilizer, or pesticides.

Most of the land in the Santee Basin is forested. Forests range from largely unaltered hardwoods in the Blue Ridge and mixed pine and hardwood stands in the Piedmont to intensively managed pine plantations and forested wetlands in the Piedmont and Coastal Plain. Trees are commercially harvested in all of these areas, producing various levels of soil disturbance, erosion, and increased sediment loads in the streams.

Climate Conditions and Water Quality

The major climatic factors affecting water quality are seasonal and areal distributions of precipitation. The amount of rainfall affects water quality because areas with higher rainfall generally have greater runoff and more infiltration to ground water. However,

increased flows also can help to dilute concentrations of chemicals in ground water and surface water. The distribution of rainfall in the Santee Basin is fairly uniform except for very high precipitation in the Blue Ridge (South Carolina Water Resources Commission, 1983; fig. 2).

Seasonal variability of rainfall also is important. Generally the highest concentrations of pesticides in streams occur when rainfall immediately follows pesticide applications. Rainfall is highest in spring and summer, typically when agricultural and residential lawn pesticides are applied.

Rainfall also is important because it contains nutrients and metals that contribute to concentrations of these compounds in surface water. Atmospheric deposition accounts for the majority of ammonia and nitrate nitrogen in streams (Maluk and others, 1998). A study also has suggested that mercury contamination in the Santee Basin results from atmospheric deposition (Krabbenhoft and others, 1999).

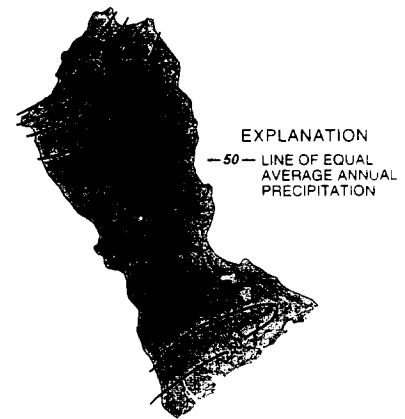


Figure 2. Precipitation affects water quality by producing runoff to streams and infiltration to aquifers.

Water Use

Most of the 7 billion gallons of water used each day in the Santee Basin is surface water (fig. 3). About 85 percent of this water is used in the production of electricity, and the remainder is used for public water supplies, commercial and industrial uses, irrigation of crops, and watering livestock.

Ground water accounts for only about 14 percent of total water use but is a very important resource. Private domestic wells are the only

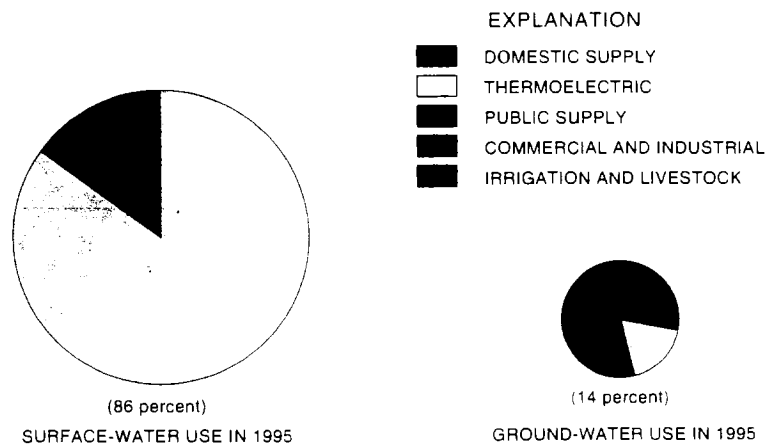


Figure 3. Most water used in the Santee Basin is supplied by surface water. Ground water is important because it is the major source of domestic water supply in rural areas. Relative size of pie charts represents relative percentage water use.

viable sources of water in areas not served by public water supplies

Flow Regulation, Impoundments, and Surface-Water Quality

The regulation of flow in the Santee Basin has altered the historical seasonal flow patterns in the rivers. High peak flows and extreme low flows downstream from major reservoirs generally are less common than they were prior to construction of the reservoirs. The alteration of flow primarily affects the physical habitat of the rivers and also can affect stream temperature. Populations of aquatic organisms are altered due to changed conditions in the streams downstream from reservoirs. For example, cold water discharged from the bottom of Lake Murray makes it possible for trout to survive nearly 100 miles beyond their normal range.

Reservoirs are especially affected by stream chemistry because they trap sediment and the phosphorus that attaches to the sediment. This trapping process can cause lakes to become eutrophic, or

nutrient enriched, and cause algal blooms. Occasionally, fishkills result when the artificially large algal population dies and the dissolved oxygen, which is necessary for fish survival, is consumed during the decaying process. Of 11 major lakes in the study area, 9 contained areas with "excessive nutrients, extremely high productivity" and were "susceptible to nuisance macrophyte growth and algal blooms" (Stecker and Crocker, 1991).

Aquifer Characteristics and Ground-Water Quality

Shallow ground water (generally less than 50 feet below land surface) is vulnerable to contamination in much of the Santee Basin. Fertilizers, pesticides, and spills or leaks of chemicals at or near the land surface can move rapidly to the water table. Areas with sandy soils are particularly susceptible to contamination because these coarse-grained soils allow rapid transport and provide little opportunity for filtration or degradation of contaminants.

Sandy soils typically are present in parts of the Coastal Plain and to a lesser degree in the Piedmont.

Deep aquifers also can be susceptible to contamination, depending on their degree of connection to the surface. The Piedmont, Sandhills, and Floridan aquifers supply most of the ground water in the Santee Basin (fig. 4). Ground water in the Piedmont aquifer occurs in fractures or cracks in the hard crystalline bedrock. In most areas, the bedrock is overlain by clay soils of variable thickness. Sandhills aquifers are unconfined; that is, they have no clay layer above them to inhibit the downward movement of contaminants to the aquifer. The Floridan aquifer is confined toward the coast but is unconfined farther inland. Of the three aquifers, the Sandhills aquifer is the most susceptible to contamination because of its sandy soils and lack of confinement. The Floridan aquifer, near the coast, is the least susceptible because it is confined.

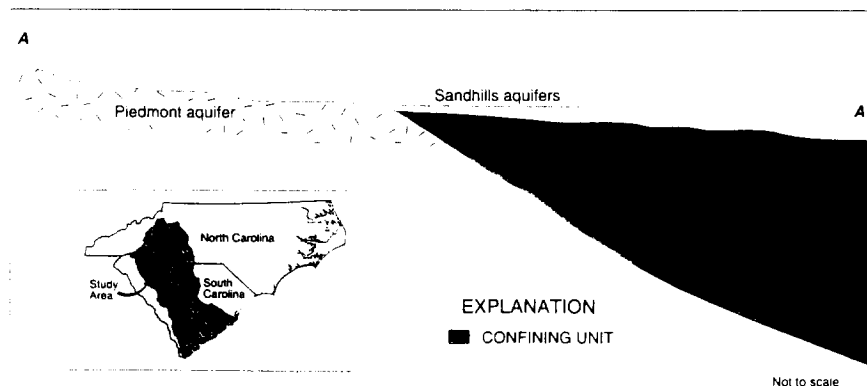


Figure 4. Aquifers sampled in the Santee Basin include the surficial, Piedmont, Sandhills, and Floridan aquifers (modified from Aucott and others, 1987). The Black Creek, Middendorf, and Cape Fear aquifers are not used much in the study area because of the cost of drilling deep wells and poor water quality.

MAJOR FINDINGS

Pesticides Commonly Were Detected in Santee Basin Streams

Thirty pesticides, including 22 herbicides and 8 insecticides, were detected in streams in the Santee Basin (Maluk and Kelley, 1998). Of the 161 surface-water samples collected, 141 contained at least one pesticide. At least one pesticide was detected at all of the sites, including forested sites that have little influence from humans.

Although detections were frequent, concentrations tended to be low, with no herbicides and only four insecticides and one metabolite exceeding aquatic-life criteria. None of the pesticide concentrations exceeded drinking-water standards. Thirteen of the 30 pesticides detected do not have aquatic criteria and 7 do not have drinking-water standards (U.S. Environmental Protection Agency, 1996).

Herbicides were the most commonly detected pesticides in streams. Atrazine, an herbicide used on corn as well as turfgrasses and golf courses, was detected at the most sites, occurring at 11 of the 13 sampling sites. Other frequently detected herbicides were simazine, metolachlor, and prometon. Tebuthiuron, an herbicide that generally is used to control weeds on highway and railroad rights-of-way, also was detected frequently.

Insecticides were detected much less frequently than herbicides, accounting for less than one-third of the pesticides detected. Most insecticides detected are used on agricultural and ornamental crops, lawns, livestock, and in homes and gardens. Those most commonly detected included chlorpyrifos, diazinon, malathion, and carbaryl.



Insecticides were detected most frequently in urban streams, such as Gills Creek in Columbia, S.C.

Urban Streams Contain More Pesticides than Other Streams

Pesticides were detected more frequently in urban streams than in agricultural streams (fig. 5). The most commonly detected herbicides—simazine, prometon, atrazine, and tebuthiuron—were detected nearly twice as frequently in water samples collected at urban sites than at agricultural sites. Conversely, some herbicides such as metolachlor and alachlor were detected almost exclusively at agricultural sites.

Insecticides were detected about four times more frequently at urban stream sites than at agricultural stream sites, and aquatic-life criteria were exceeded in nine samples collected at urban sites and in three samples collected at agricultural sites. The insecticide diazinon was detected only in urban streams, whereas chlorpyrifos was detected frequently in both urban and agricultural streams. This agrees with national findings in which insecticides were detected more frequently in urban than in

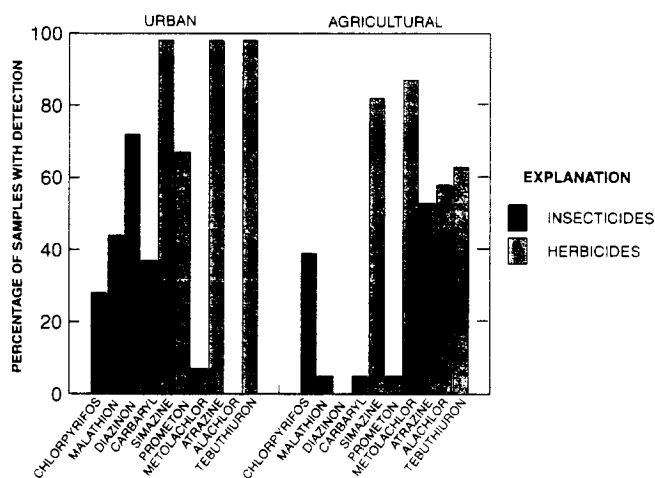


Figure 5. Pesticides, particularly insecticides, were detected more frequently in urban streams than in agricultural streams.

agricultural streams (U.S. Geological Survey, 1999).

These findings indicate that while agricultural activities contribute pesticides to surface water, concentrations are rarely high enough to affect aquatic life. The consequences of urban and suburban use of pesticides is much more significant, with concentrations of insecticides frequently at levels that can affect aquatic life.

Pesticides Showed Seasonal Patterns in Streams

Some herbicides showed patterns of occurrence that can be related to seasonal applications and weather patterns. At Gills Creek, an urban stream, concentrations of herbicides, such as atrazine, simazine, and tebuthiuron, peaked in the spring following application and gradually decreased over the summer (fig. 6). Atrazine and simazine followed a similar pattern at Cow Castle Creek, an agricultural stream, with the addition of a second peak in early fall. The highest concentrations at all sites

were observed during storms that followed applications.

Understanding these patterns of occurrence is important because sampling programs need to be designed so that critical periods of high pesticide concentrations are monitored. This information also can be used by environmental managers to assess risk associated with agricultural chemical use.

Few Pesticides Were Detected in Drinking-Water Supply Aquifers

Of the 90 drinking-water, industrial, and irrigation supply wells sampled in the Floridan, Piedmont, and Sandhills aquifers, 17 had detectable concentrations of pesticides; of those, only two wells had pesticide concentrations that exceeded USEPA drinking-water standards (U.S. Environmental Protection Agency, 1996). Eleven of the 34 pesticides detected in drinking-water supply aquifers did not have water-quality standards.

The Sandhills aquifers are more susceptible to contamination than

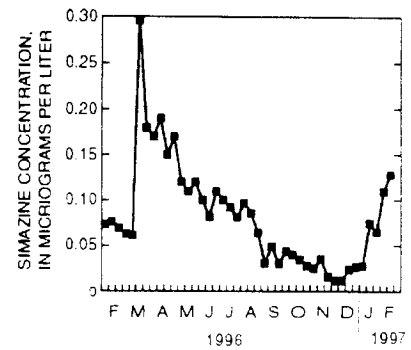


Figure 6. Herbicide concentrations generally peaked following spring applications in Gills Creek, an urban stream.

the other aquifers, as is illustrated by the larger number of pesticides detected and the higher detection frequency in the Sandhills aquifers than in the Piedmont and Floridan aquifers (fig. 7). In addition, the two wells that had pesticide concentrations exceeding USEPA drinking-water standards were located in the Sandhills aquifers. Drinking water obtained from the Piedmont and Floridan aquifers is probably unlikely to contain pesticides at harmful levels; however, the high rate of detection and large number of pesticides detected in the Sandhills aquifers are a cause for concern.



Ground-water samples are pumped directly into a mobile laboratory for processing samples and conducting field analyses.

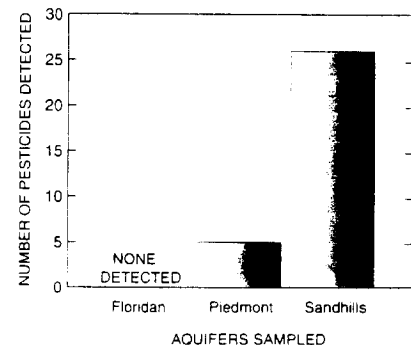


Figure 7. The large number of pesticides detected in the Sandhills aquifers illustrates its relatively high susceptibility to contamination.

The differences in the rate and number of detections in the aquifers may be related to differences in aquifer properties. The Sandhills are largely composed of layers of coarse and fine sand with various quantities of clay. No continuous confining unit or soil layer overlies the aquifer to impede the movement of contaminants into ground water. At the other extreme, the Floridan aquifer has an overlying clay confining layer that impedes vertical movement of contaminants throughout much of its extent (Aucott and others, 1987). The Piedmont aquifer has an overlying layer of weathered bedrock that contains abundant clay. The thickness of this unit is highly variable, but generally it impedes rapid vertical movement of contaminants from land surface to ground water.

Pesticides Were Common in Shallow Ground Water in Urban and Agricultural Areas

Pesticides were detected in 24 of 30 shallow wells in urban areas and in 22 of 30 wells in agricultural areas (Reuber, 1999). Dieldrin concentrations exceeded drinking-water standards in four urban wells; however, the wells sampled were installed for monitoring purposes only and are not drinking-water supply wells. Dieldrin also exceeded aquatic-life standards in these four urban wells, and tebutiuron concentrations exceeded aquatic-life standards in one agricultural well. Generally, ground-water concentrations are not compared to aquatic-life standards; however, these concentrations can be of concern because shallow ground water can discharge to streams, elevating the pesticide concentrations in surface water.

Pesticides were detected in ground water in urban areas about twice as frequently as in agricultural areas (fig. 8). Some pesticides detected in urban and agricultural ground water were the same, although insecticides were detected more frequently in urban ground water. This was similar to national NAWQA findings. The insecticide most frequently detected at urban sites was dieldrin. Although its agricultural use was canceled in the mid-1970s, dieldrin (and aldrin, which breaks down to dieldrin) was used for termite control until the mid-1980s and is a persistent compound (Barbash and Resek, 1996). Dieldrin was also the most commonly detected pesticide in urban ground water nationally (U.S. Geological Survey, 1999).



Shallow wells for the urban ground-water study were installed by the U.S. Geological Survey with assistance from the South Carolina Geological Survey.

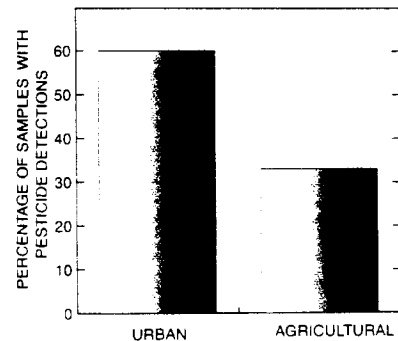


Figure 8. Pesticides were detected more frequently at urban ground-water sites than at agricultural sites.

Organochlorine Pesticides Were Detected in Bed Sediment and Tissues

Fourteen pesticides were detected in streambed-sediment samples, and 21 of 24 sites sampled had at least one detectable pesticide. All of the pesticides detected in sediment were organochlorine insecticides, such as chlordane, dieldrin, mirex, and DDT and its derivatives. Many of these compounds had their agricultural uses cancelled more than 20 years ago, yet they still appear in sediment samples. The reason for this is because the compounds are persistent, or are highly resistant to chemical breakdown.

Of the compounds detected, only DDE, a breakdown product of DDT, exceeded guidelines for the protection of aquatic life. The guidelines were exceeded at three sites, all of which are in basins with a high percentage of agricultural land. In addition, DDT concentrations were only slightly below the aquatic guidelines at several sites. Comparisons of land use to concentrations of organochlorine pesticides generally did not show strong relations, primarily because these insecticides were used in both urban and agricultural settings (fig. 9). The only prominent rela-

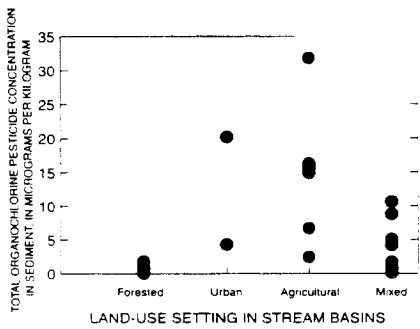


Figure 9. Bed sediment in forested settings has significantly lower total organochlorine pesticide concentrations than bed sediment in other land-use settings.

tion is the uniformly low levels of these pesticides detected at forested sites where these chemicals were less likely to be used.

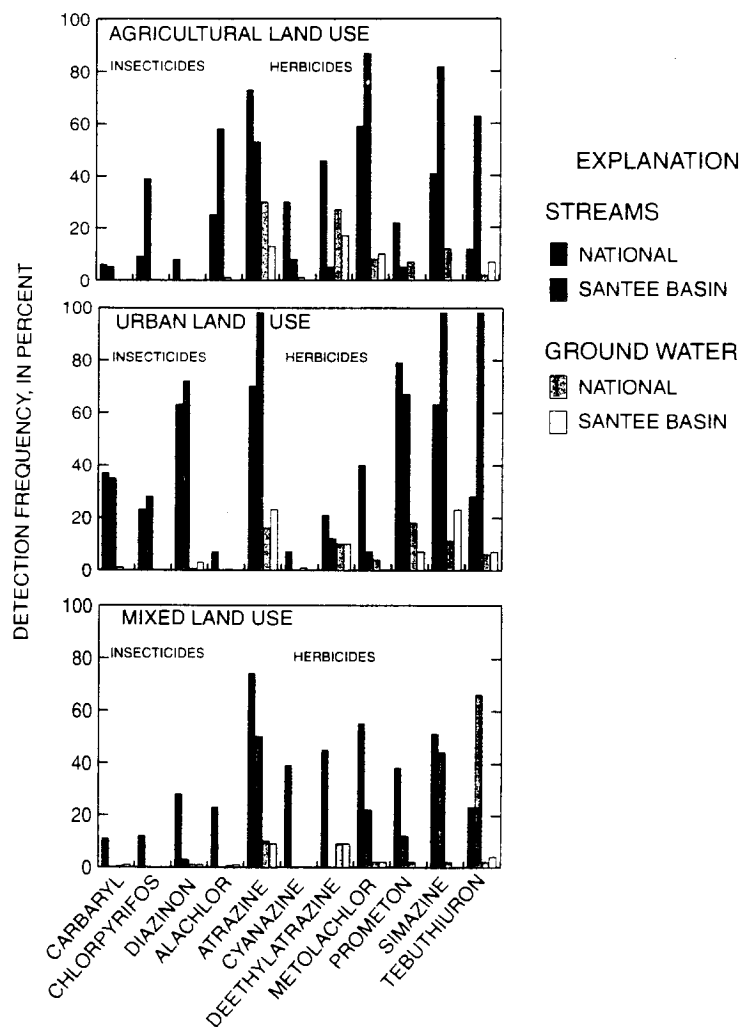
The same pesticides detected in sediments were also in clam and fish tissues. Concentrations measured were generally much higher in tissue than in sediment; however, a direct comparison of these concentrations may not be valid because of potential differences in exposure rates of sediment and fish, differences in uptake by sediment organic carbon and tissue, and partitioning in fish tissue.



Organochlorine pesticides, such as DDT, can be found in streambed sediment and can accumulate in tissues of fish, such as this carp.

SANTEE BASIN PESTICIDE FINDINGS WERE SIMILAR TO NATIONAL FINDINGS

The most common pesticides detected in the Santee Basin included atrazine, simazine, tebutiuron, prometon, and metolachlor. These pesticides were among the top 11 pesticides detected nationally. Consistent with national findings, herbicides were the most common type of pesticide detected in streams and aquifers in agricultural areas in the Santee Basin, whereas insecticides were more prevalent in urban areas. Overall, streams and aquifers that integrate different land uses had lower concentrations of pesticides than those that are dominated by either agricultural or urban land uses. Detections of pesticides in mixed land-use streams in the Santee Basin overall were much less frequent than national detections. The most striking differences between national and Santee Basin findings are the more frequent detections of alachlor, diazinon, and metolachlor in agricultural streams, simazine in agricultural and urban streams, and tebutiuron in all land-use settings in the Santee Basin. The greater detection frequency does not appear to result from higher use of these compounds in the Santee Basin.



Phosphorus Concentrations in Streams Frequently Exceeded USEPA Goals

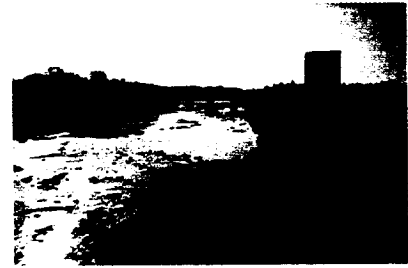
Nutrients measured in streams in the Santee Basin, such as ammonia, nitrite, nitrate, phosphate, and orthophosphate, were elevated above background concentrations in areas affected by agricultural and urban runoff. Of these nutrients, the only one governed by a drinking-water standard is nitrite-plus-nitrate nitrogen (hereinafter referred to as nitrate) because it is the only nutrient that directly affects human health. None of the surface-water samples had concentrations of nitrate that were above the drinking-water standard of 10 milligrams per liter (mg/L) (U.S. Environmental Protection Agency, 1996).

Phosphorus concentrations were above the USEPA goal in several rivers in the Santee Basin. For example, the flow-weighted mean annual concentration of total phosphorus in the South Fork Catawba River is about four times higher than the USEPA goal for streams entering a reservoir (U.S. Environmental Protection Agency, 1986) (fig. 10). This is important because many of the reservoirs in the Santee Basin are eutrophic; that is, they have high levels of nutrients that can result in excessive growth of algae (Stecker and Crocker, 1991). Much of the phosphorus and nitrogen that feeds the algae is carried into the reservoirs by major rivers. The South Fork Catawba River flows into Lake Wylie directly downstream of the sampling site and is the only stream sampled that enters directly into a reservoir.

The USEPA also has a goal of 0.1 mg/L total phosphorus for

streams that do not directly discharge to reservoirs (U.S. Environmental Protection Agency, 1986). The purpose of this goal is to prevent excessive plant growth in streams. Indian Creek, N.C., Congaree River, and Brushy Creek do not meet this goal based on mean annual concentrations (fig. 10). Only two of the streams sampled did not have at least one sample above the goal. These were Jacob Fork River and McTier Creek, both of which drain forested watersheds. The remaining streams had percentages of individual samples that exceeded the goal, ranging from 4 to 96 percent.

An analysis of nutrient data collected by State monitoring agencies during 1973–93 (Maluk and others, 1998) showed that all but 3 of 90 stream and lake sites exceeded the applicable phosphorus goal at least once, and 23 sites had median concentrations that exceeded the goal.



The Congaree River is one of several rivers in the Santee Basin that exceeded the U.S. Environmental Protection Agency goal for phosphorus.

Although 34 of the 90 sites showed decreasing trends in phosphorus concentrations, 53 showed no trend, and 3 had increasing trends.

Agricultural Runoff and Industrial Discharges are Sources of Nutrients in Surface Water

For all the streams in the study unit, except in the South Fork Catawba River, there is a strong relation between orthophosphate

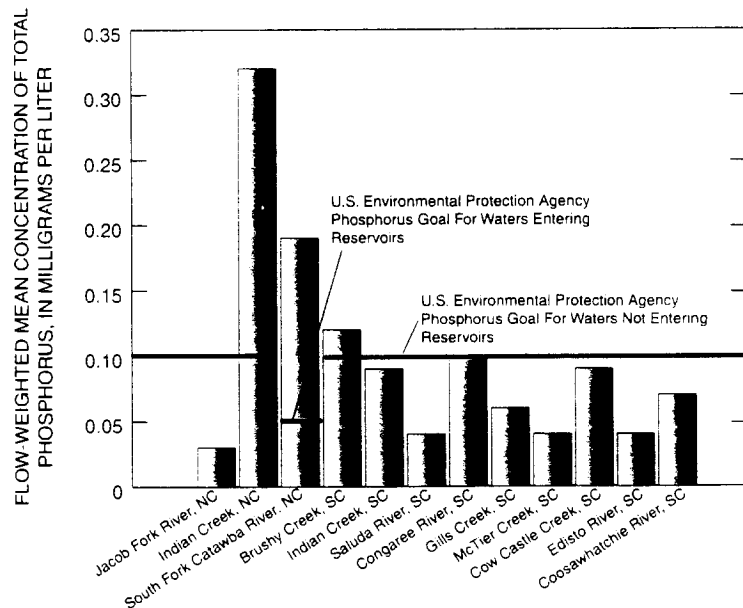


Figure 10. Three streams in the Santee Basin frequently exceeded the U.S. Environmental Protection Agency goal for phosphorus in surface waters not entering reservoirs, and one exceeded the goal for waters entering reservoirs.

(the predominant form of dissolved phosphorus in streams) concentrations and the percentage of agricultural land in the basins sampled (fig. 11). This relation most likely results from the runoff of phosphate-containing chemical fertilizer and manure from agricultural lands. The relation generally is not influenced by municipal waste-water discharges because phosphate-containing detergents have been banned for domestic use in the Santee Basin since the late 1990s (Litke, 1999).

The South Fork Catawba River has much higher concentrations of orthophosphate than would be predicted from the amount of agricultural land in the basin (fig. 11). This may result from a lack of a phosphorus ban on industrial users. The South Fork Catawba River Basin contains a large concentration of industries that use phosphate detergents, which are a potential source for the high orthophosphate levels in the South Fork Catawba River (Lindsey and Lewis, 1994).

Water-Supply Aquifers Rarely Exceeded Drinking-Water Standards for Nitrate

With the exception of nitrate, most nutrient concentrations in ground water in the Santee Basin were low. This is fairly typical of ground water in which most forms of nitrogen and phosphorus are negligible (Nolan and Stoner, 2000).

Nitrate concentrations exceeded the USEPA (1996) drinking-water standard of 10 mg/L in 14 of the 150 wells sampled. Drinking water containing concentrations of nitrate above the standard can result in methemoglobinemia, a life-threatening illness. All but two of the

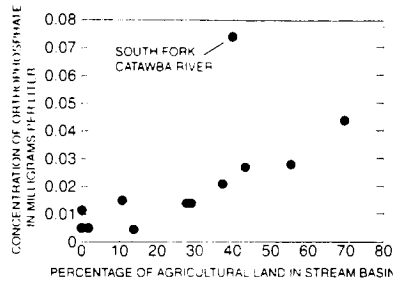


Figure 11. The concentration of orthophosphate in streams is directly related to the percentage of agricultural land in the stream basin except for the South Fork Catawba River.

wells that exceeded the standard were located in the shallow aquifer beneath agricultural land in the Coastal Plain. In fact, wells in the agricultural land-use study had the highest concentrations of nitrate overall, with concentrations up to 23 mg/L and a median concentration about double the national NAWQA median for agricultural land use (fig. 12). Although the shallow aquifer generally is not used for drinking-water supplies, the potential for movement of nitrate-enriched water to deeper aquifers used for water supply is a cause for concern. Wells beneath urban land had lower median con-

centrations of nitrate than wells in agricultural lands and were lower than the national NAWQA median for urban land use.

Nitrate concentrations measured in the three drinking-water supply aquifers sampled in the Santee Basin were variable. The Piedmont had the highest nitrate concentrations, followed by the Sandhills and Floridan aquifers (fig. 13). Only two wells exceeded the drinking-water standard for nitrate, one each in the Piedmont and Sandhills aquifers. With the exception of these two wells, most concentrations measured were well within the standard. One of the wells with a concentration above the standard was an irrigation well located in the middle of a corn and soybean field; the other was adjacent to a golf course. These results suggest that most wells in these three aquifers are safe from high levels of nitrate, but some concern is justified for wells located near areas with high fertilizer use.

The higher nitrate concentrations in the Piedmont and Sandhills aquifers are related to the lack of con-

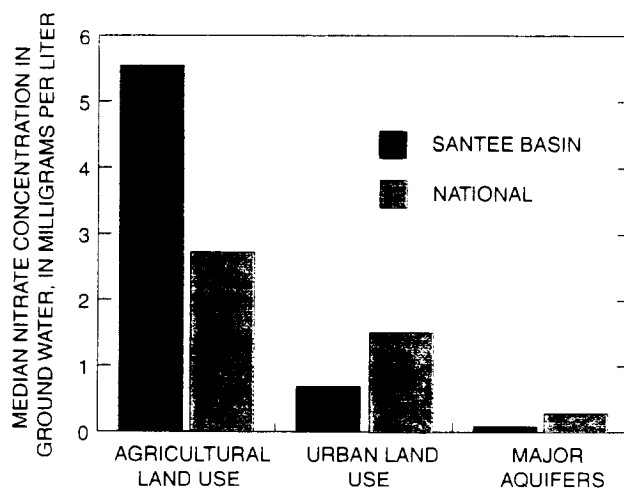


Figure 12. Nitrate concentrations in shallow ground water in agricultural areas were higher than those in urban areas and in major aquifers.

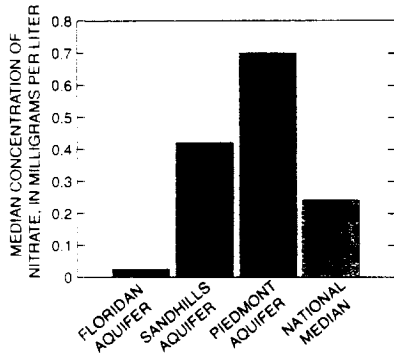


Figure 13. The Piedmont and Sandhills aquifers had higher median nitrate concentrations than the Floridan aquifer.

finement for these aquifers, which readily allows the downward movement of surficial contaminants. Water in these aquifers often has high dissolved oxygen concentrations, which prevents denitrification (the removal of nitrate by conversion to nitrogen gas). By comparison, the Floridan aquifer has the lowest nitrate concentrations because it is confined, meaning little water moves vertically into the aquifer, and it has little dissolved oxygen, a condition which can promote denitrification.

Nitrate in Ground Water Can Affect Nitrate Concentrations in Streams

Local conditions can strongly affect the nitrogen concentrations in ground water and how much nitrate discharges from ground water to surface water. A study of the transport of nitrate in ground water was conducted at an agricultural site adjacent to Cow Castle Creek, S.C. (fig. 14). At this site, ground water beneath a corn field had concentrations of nitrate more than 28 mg/L, nearly three times the drinking-water standard. Along the ground-water flow path, nitrate concentrations decreased to less than 5 mg/L.

Directly below the streambed, nitrate concentrations were above 4 mg/L. However, as ground water moves upward to the stream, it passes through an organic-rich zone containing little dissolved oxygen. Denitrification occurring in this zone results in water with a nitrate concentration of only 0.4 mg/L.

Denitrification may not always be effective in removing nitrate at

all locations where ground water discharges to Cow Castle Creek. This is evidenced by the high concentrations of nitrate measured in Cow Castle Creek during low flow when most streamflow is attributed to ground-water discharge.



Water levels were measured simultaneously in Cow Castle Creek and at multiple depths in the aquifer below the creek. Under most streamflow conditions, water from the aquifer was moving upward into the creek.

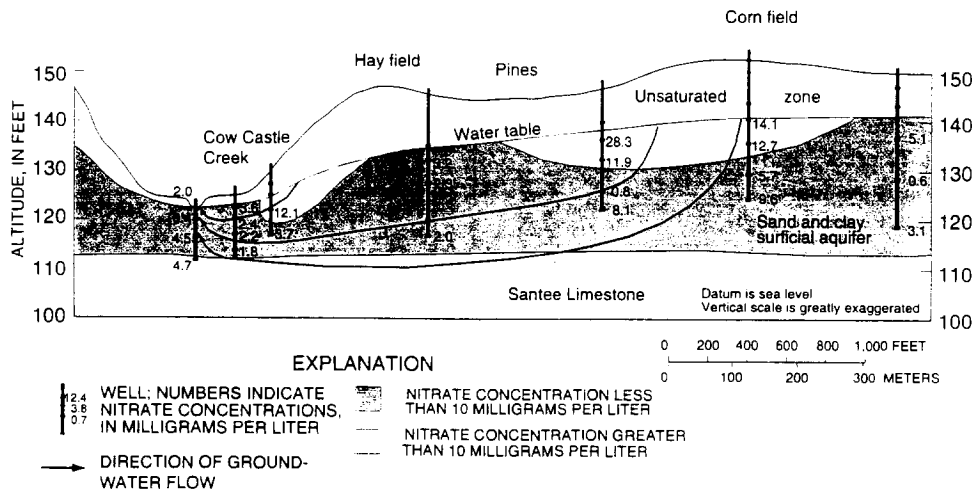


Figure 14. Concentrations of nitrate are reduced through denitrification as ground water flows to Cow Castle Creek. Typical wells used for water supply in this area are greater than 100 feet deep. Nitrate concentrations at that depth generally are not above drinking-water standards.



PHOSPHORUS CONCENTRATIONS IN THE SANTEE BASIN WERE IN THE MIDDLE RANGE OF NATIONAL RESULTS

Six of the 12 streams sampled in the Santee Basin had average concentrations of total phosphorus that were within the medium concentration range of all streams sampled in the NAWQA Program. The exceptions were an agricultural area drained by Indian Creek in North Carolina, which ranked high nationally, and two mixed land-use streams, which were in the lowest category for phosphorus. Nationally, higher concentrations of phosphorus corresponded to areas of the Midwest that also had high inputs of phosphorus from fertilizer and manure that were applied to agricultural lands. Phosphorus inputs from agriculture in the Santee Basin were typically in the low to middle range. The two mixed land-use sites that had the lowest phosphorus concentrations compared to national results were located on the Saluda and Edisto Rivers. The Saluda River site is located downstream from a major reservoir that traps much of the phosphorus carried into the system. The Edisto River is located in the Coastal Plain and mostly drains forested and agricultural lands. Extensive wetlands border the Edisto River and may act as local traps for phosphorus attached to sediment.

EXPLANATION

AVERAGE ANNUAL CONCENTRATION OF TOTAL PHOSPHORUS--IN MILLIGRAMS PER LITER

- Highest (greater than 0.27)
- Medium (0.05 to 0.27)
- Lowest (less than 0.05)

BACKGROUND CONCENTRATIONS

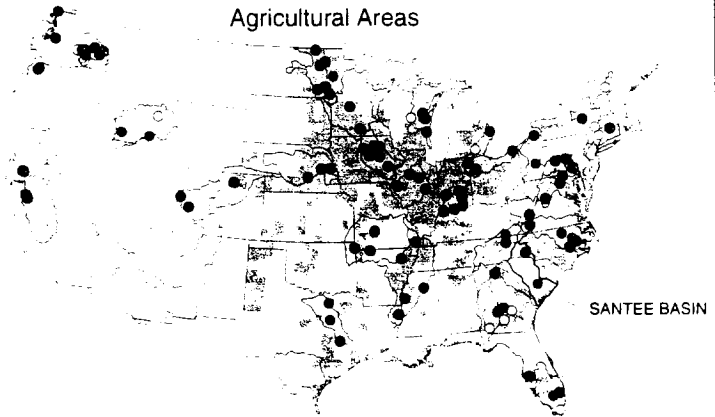
- Bold outline indicates median values greater than USEPA desired goal of 0.1 milligrams per liter for prevention of nuisance plant growth in flowing waters not discharging directly to lakes and impoundments

AVERAGE ANNUAL TOTAL PHOSPHORUS INPUT--IN POUNDS PER ACRE, BY COUNTY, FOR 1995-98. INPUTS ARE FROM FERTILIZER AND MANURE

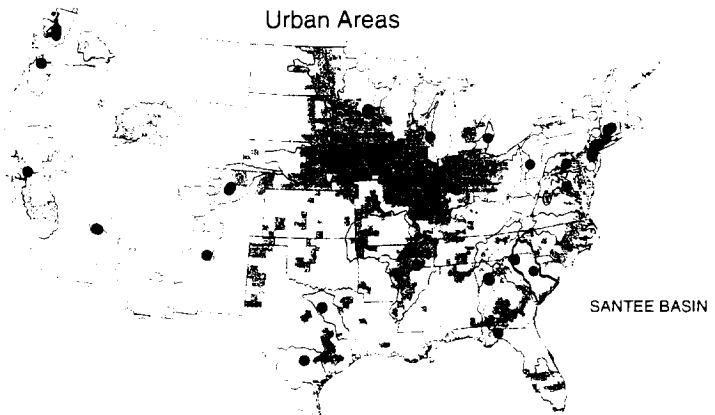
- ▣ Greater than 5 pounds per acre
- ▢ 2 to 5 pounds per acre
- ▢ Less than 2 pounds per acre

TOTAL PHOSPHORUS IN STREAMS

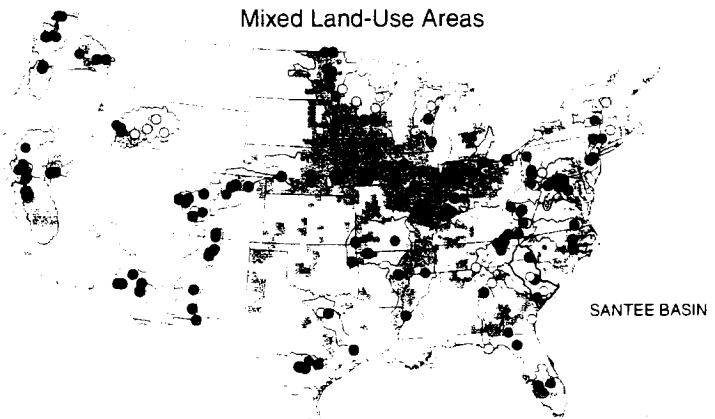
Agricultural Areas



Urban Areas



Mixed Land-Use Areas



Bed Sediment Had Low Concentrations of Trace Elements

Trace elements in bed sediment were detected frequently but mostly at concentrations below those expected to affect aquatic life (Canadian Council of Ministers of the Environment, 1995). Arsenic and lead exceeded aquatic standards in one sample each, and chromium exceeded standards in four samples. The samples with elevated chromium concentrations were not associated with any particular land use; however, most of these samples were collected at sites in the Piedmont. This suggests that the elevated concentrations may be naturally occurring as a result of geologic conditions in the Piedmont.

Regional differences in bed sediment trace-metal concentrations were observed in the study area. In general, a decrease in the bed-sediment concentrations of arsenic, chromium, copper, nickel, and zinc occurs from the Blue Ridge southward across the Piedmont to the Coastal Plain. In the same direction, an increase in the bed-sediment concentrations of lead, mercury, and selenium occurs. These differences are likely a result of geologic differences among the areas (Abrahamsen, 1999).

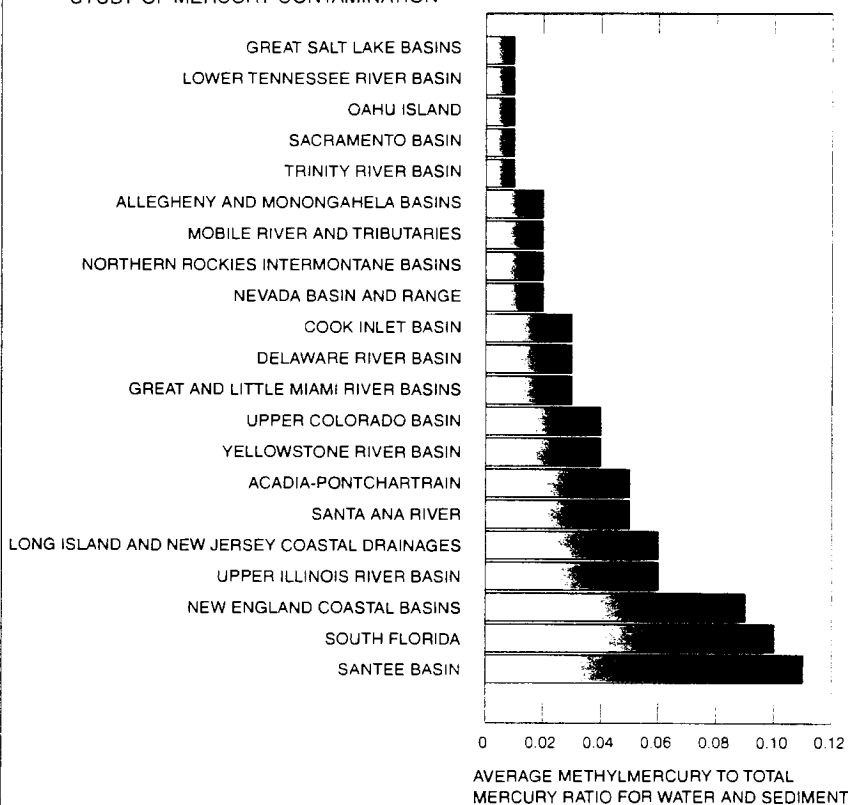
A comparison of land use with bed sediment trace-element concentrations indicates that lead is significantly higher in sediment from urban streams than in sediment from forested streams. Neither agricultural nor mixed land-use streams had significantly higher concentrations of lead than forested streams (Abrahamsen, 1999).



MERCURY CONTAMINATION POTENTIAL IN SANTEE BASIN WAS HIGHEST IN NATION

Mercury is a widespread and extremely toxic contaminant that affects the Nation's aquatic ecosystems. Approximately 80 percent of the fish consumption advisories in the Nation were issued as a result of contamination by mercury (U.S. Environmental Protection Agency, 1998). Methylmercury is the form of mercury that is easily accumulated in the tissues of organisms. As part of the NAWQA Program, a nationwide investigation of methylmercury in water, sediment, and fish tissues was conducted (Krabbenhoft and others, 1999). Analysis of samples collected in the Santee Basin indicated that they had the greatest methylation efficiency (the ratio of methylmercury to total mercury) in the Nation. In other words, conditions at the sites sampled in the Santee Basin are such that a relatively small amount of elemental mercury in water can result in high concentrations of methylmercury being available for accumulation in fish tissue. The types of sites sampled were primarily blackwater streams, which typically have large amounts of dissolved organic carbon and low dissolved oxygen concentrations. The bacteria that convert elemental mercury to methylmercury thrive under these conditions. Many of the streams that are covered by State fish-consumption advisories are blackwater streams.

NAWQA STUDY AREAS SAMPLED FOR THE NATIONAL PILOT STUDY OF MERCURY CONTAMINATION



Trace Elements Accumulated in Clam Tissue and Fish Livers

Trace metals are naturally occurring and were detected in all fish liver and clam tissue samples collected. Nine trace elements (arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, and zinc) have been classified as priority pollutants because they are toxic to aquatic organisms in low concentrations (Code of Federal Regulations, 1996). Of these nine metals, concentrations of cadmium, copper, selenium, and zinc were higher in clams and fish liver tissue than those measured in sediment. Carp liver tissue contained significantly higher concentrations of these metals than those in clams and bed sediment, indicating that the metals accumulate in fish livers. Concentrations of arsenic, chromium, lead, and nickel were significantly lower in tissues than in sediment, suggesting that these metals do not accumulate in tissues (fig. 15). Concentrations of mercury were higher in clam tissue than in sediment and fish livers. Although data suggest that some metals accumulate in tissues, most metals do not have criteria for assessing risk to human health or aquatic wildlife associated with fish consumption.

Concentrations of mercury in clams and fish liver tissue from the Edisto River were 24 and 8 times greater, respectively, than the South Carolina action level for issuance of a fish-consumption advisory. Data collected in the NAWQA Program cannot be used to assess potential risk to human health because fish livers were used for analyses, whereas fish filets are needed to assess human-

health risk. Consumption advisories generally are not applied to clams because few humans consume them. The South Carolina Department of Health and Environmental Control (2000) has issued fish-consumption advisories

because of high levels of mercury in 49 rivers and reservoirs in the Santee Basin, including the Edisto River.

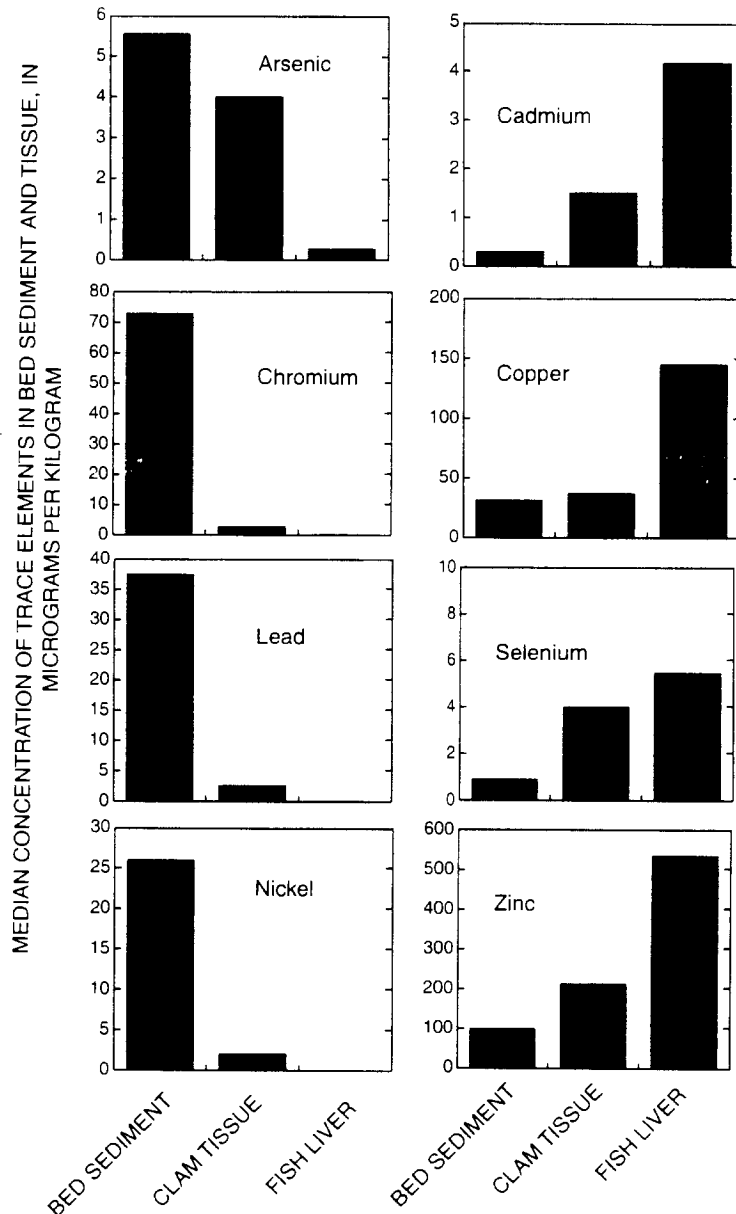


Figure 15. Cadmium, copper, selenium, and zinc were detected at higher concentrations in clam and fish tissue than in sediment, suggesting that they accumulate in the tissues. Conversely, arsenic, chromium, lead, and nickel were detected in lower concentrations in tissues than in sediment, indicating that these metals do not accumulate in the tissues.

Radon Exceeded Proposed Standards in Many Wells

Ninety-six percent of the 90 wells sampled in drinking-water supply aquifers of the Santee Basin contained measurable quantities of radon, a colorless, odorless gas that can cause cancer in humans. The gas results from the radioactive decay of uranium in rocks and soil and can enter homes directly from the soil or in drinking water supplied by wells. Radon is a health risk through direct inhalation of the gas and from drinking water contaminated with radon.

Of the 90 wells sampled, radon exceeded the USEPA's (1999) proposed maximum contaminant level (MCL) of 300 picocuries per liter (pCi/L) in 100, 47, and 17 percent of the wells in the Piedmont, Sandhills, and Floridan aquifers, respectively (fig. 16). For wells not meeting the MCL, the USEPA has proposed an Alternative Maximum Contaminant Level (AMCL) of 4,000 pCi/L. To comply with the

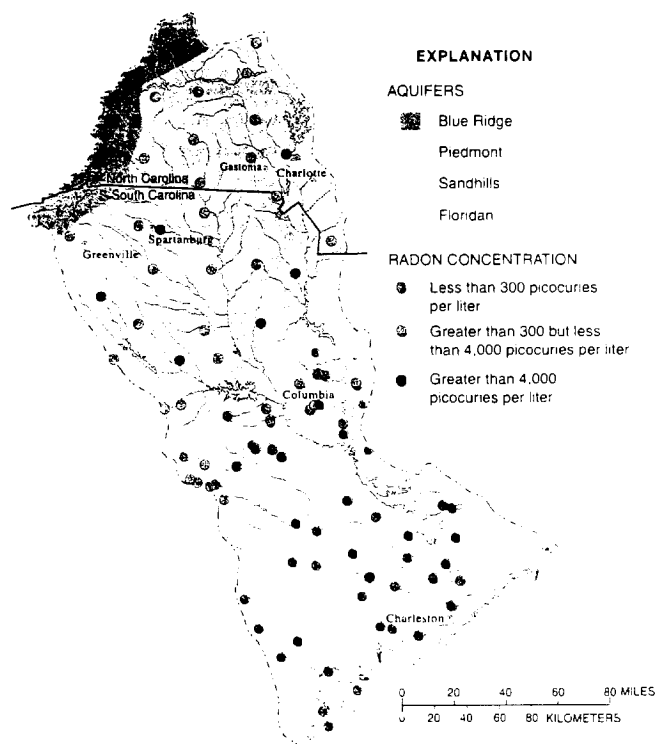
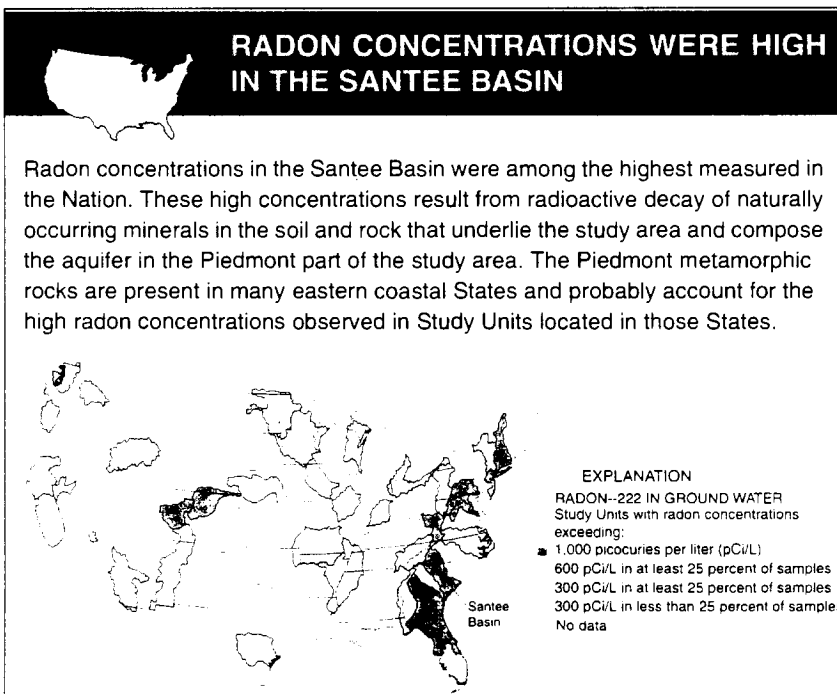


Figure 16. Radon concentrations were highest in the Piedmont and Sandhills aquifers, resulting from naturally high levels of uranium in near-surface rocks and sediment.



AMCL, the State or local water utility must develop indoor air radon-reduction programs and reduce radon levels in drinking water to 4,000 pCi/L. Of the wells that were sampled in the Piedmont, Sandhills, and Floridan aquifers, 20 percent, zero percent, and less than 1 percent, respectively, exceeded the proposed AMCL.

Wells in the Piedmont had much higher concentrations of radon, on average, than wells sampled in the Sandhills and Floridan aquifers (fig. 16). This results from the greater relative abundance of minerals containing uranium in the metamorphic rocks that compose the Piedmont aquifer. The same bedrock underlies the Sandhills and Floridan aquifers, but generally at depths ranging from several hundred to several thousand feet.

Volatile Organic Compounds Were Common in Urban Ground Water

All but 3 of 30 monitoring wells installed in commercial and residential areas of Columbia, S.C., contained a variety of volatile organic compounds (VOCs), a group of chemicals that includes gasoline additives, solvents, and disinfection by-products (Reuber, 1999). Thirty-five such compounds were detected. Most wells contained three or more VOCs, and one well contained 15 different VOCs.

Most of the VOC detections were at extremely low levels. The five VOCs detected with the highest concentrations were methyl *tert*-butyl ether (MTBE), trichloroethene (TCE), acetone, *tert*-amyl methyl ether (TAME), and trichloromethane. Of the 35 detected VOCs, 21 have established drinking-water standards; of these, only TCE exceeded the standard. Currently there is no standard for MTBE, but the MTBE concentra-

tion in one well exceeded a drinking-water advisory.

Some of the most frequently detected compounds included trichloromethane, chloromethane, and bromodichloromethane. These compounds can result from the chlorination of drinking water and can enter ground water by infiltration from irrigation systems or from leaky water-supply lines. Other VOCs that were detected were solvents, such as TCE, tetrachloroethene, and acetone. These compounds have commercial and industrial uses as degreasers and dry-cleaning solvents, but they are often used in households for similar purposes. The gasoline additives MTBE and TAME and the gasoline component benzene also were detected in Columbia's ground water. These compounds can enter ground water from leaking gasoline storage tanks, spills, and potentially through atmospheric deposition (Lopes, 1998).

The diffuse, nonpoint nature of sources of VOCs in ground water

makes it difficult to attribute detected compounds to particular homes or businesses, posing a problem for scientists who seek to establish the relative importance of these sources and for regulators who seek to educate the public or control the release of toxic substances.

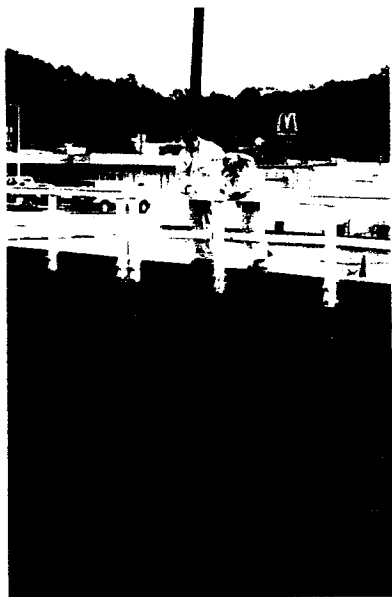
Volatile Organic Compounds in Streams Can Result from Ground-Water Discharge and Urban Runoff

Ten VOCs were detected in seven monthly stream-water samples that were collected from Gills Creek, an urban stream in the Columbia metropolitan area. MTBE, chloromethane, methylbenzene, chlorobenzene, and acetone were detected most frequently. None of the VOCs detected in Gills Creek were at concentrations exceeding drinking-water or aquatic-life standards. Six of the VOCs detected do not have standards.

VOCs have many sources, including contaminated precipitation, surface-water runoff, and ground-water discharge. Though the sources are diffuse and hard to measure directly, inferences can be made about the likely sources. For example, samples collected in Gills Creek while the water level was rising during a rainstorm indicate that as streamflow increased, the concentration of acetone increased. This suggests that the source of acetone in the samples was from contaminated precipitation or stormwater runoff (Lopes and others, 2000). If the acetone resulted from continuously discharging ground water, the concentration would be expected to decrease as dilution by rainfall and runoff increased.



Shallow ground water in the Columbia, S.C., metropolitan area contains a variety of volatile organic compounds, but mostly at low concentrations.



Stormwater samples for volatile organic compounds were collected by using special samplers developed for the NAWQA Program.

During the summer of 1996, 20 different VOCs were detected at low concentrations in individual samples collected at 16 surface-water sites scattered throughout the Gills Creek Basin. The compounds detected in the highest concentrations included MTBE, 1,1-dichloroethene, trichloroethene, methylbenzene, and 1,2-dibromo-3-chloropropane—all solvents except for MTBE, which is a gasoline additive. Fifteen of the compounds detected in surface water also were present in the Columbia ground-water samples. This result is not surprising because the samples were collected during low streamflow conditions when the ground-water contribution to Gills Creek is greatest; consequently, VOCs from contaminated ground water most likely would be detected in stream samples.

Drinking-Water Aquifers Had Low Concentrations of Volatile Organic Compounds

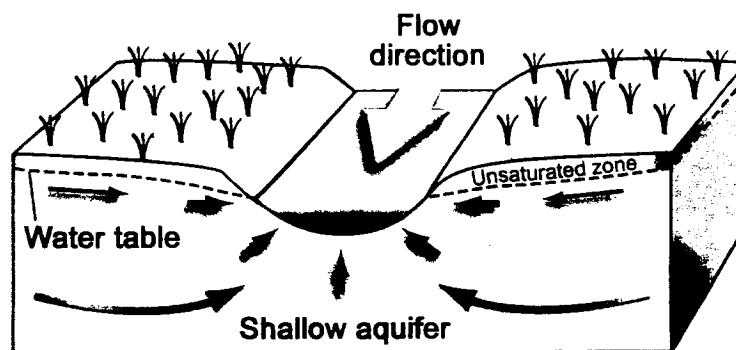
Of the 90 wells sampled in the Piedmont, Sandhills, and Floridan aquifers, 62 contained detectable concentrations of VOCs. All of the 28 compounds detected met USEPA drinking-water standards; however, 19 of the compounds did not have standards. The VOCs measured had widely ranging detection limits, making comparisons among compounds and aquifers difficult. A subset of the compounds having detection limits of 0.05 microgram per liter ($\mu\text{g/L}$) or lower was used to make comparisons. This comparison shows that

about twice as many wells in the Sandhills aquifers contain VOCs as those in the Floridan and Piedmont aquifers. The more frequent occurrence of VOCs in the Sandhills aquifers probably relates to their greater susceptibility to contamination.

Data on VOCs in drinking-water aquifers indicate that although these compounds are widespread, concentrations are sufficiently low that human health is not immediately at risk. However, the fact that detections were so frequent suggests that aquifers are susceptible to contamination and should be carefully monitored.

Ground water affects the quality of surface water

Presently, surface water is used for most public water supplies in the Columbia metropolitan area. However, the quality of these supplies is largely controlled by ground-water quality, especially during summer months. In areas of the basin with sandy soils, such as the Sandhills, flow in streams is contributed mostly by discharge from ground water. Discharging ground water transports contaminants, including VOCs, that can result in water-quality problems in streams and lakes. Consequently, the health of aquatic organisms and surface-water drinking supplies can be affected by the chemical quality of shallow ground water discharged to streams and reservoirs. Several streams in the metropolitan area, including Penn Branch and Jackson Creek, have high concentrations of nutrients and pesticides that evidence suggests result from ground-water discharge (Maluk, 1999).



Urban and Agricultural Streams Had High Concentrations of Bacteria

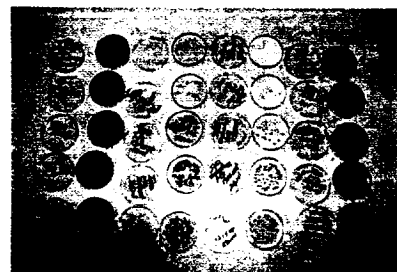
Thirteen of 17 streams sampled for fecal coliform bacteria had at least one sample that exceeded the South Carolina single-sample standard of 400 colonies per 100 milliliters (col/100 mL; South Carolina Department of Health and Environmental Control, 1992). This standard was implemented to reduce the risk of gastrointestinal disorders that are associated with recreational contact with water containing elevated levels of bacteria. All concentrations measured in this report were compared to South Carolina standards for consistency; North Carolina does not recognize a single-sample standard.

Urban and agricultural streams had more concentrations of bacteria that exceeded standards than forested and mixed land-use streams (fig. 17). Several creeks repeatedly had high concentrations of bacteria. For example, one of the urban streams sampled, Brushy Creek, exceeded the standard in 60 percent of the samples collected.

The highest bacterial concentrations measured were at agricultural sites, such as Cow Castle Creek and Indian Creek, N.C., which had concentrations of 21,600 and 12,000 col/100 mL, respectively. The highest concentrations observed in urban streams were much lower—around 2,000 col/100 mL. Forested streams generally had the lowest peak concentrations, ranging from about 500 to 1,000 col/100 mL. Samples were collected under all flow conditions, and higher concentrations as well as standard exceedances tended to occur at higher streamflows.

Stream Size is Important

All of the regularly monitored small streams exceeded the South Carolina single-sample standard for bacteria. Most large rivers did not exceed the standard during the period sampled, including the Wateree, Saluda, Congaree, and Edisto Rivers. The median bacterial concentrations in streams with drainage areas larger than 100 square miles (mi²) were significantly lower than those in streams with drainage areas less than 100



Bacterial cultures were prepared and counted in a mobile field laboratory and in the USGS South Carolina District laboratory.

mi². Because major rivers and small streams have similar sources of bacteria, the most likely reason for the differences in bacterial levels is dilution by the larger flows in the major rivers.

Sources of Bacteria

A comparison of bacterial concentrations to the physical and chemical parameters of the water indicate that surface-water runoff accounts for much of the elevated fecal coliform concentrations (Wilhelm and Maluk, 1998). Bacterial concentrations increased as streamflow, organic nitrogen, organic carbon, phosphorus, and suspended-sediment concentrations increased. Because increases in these parameters usually result from surface-water runoff, the implication is that the increase in bacteria also resulted from runoff.

In agricultural areas, bacteria in runoff may result from applications of manure to fields and from animal holding and feeding areas. Urban sources include runoff from lawns containing pet wastes, leaking or failed septic tanks and sewer lines, and municipal or industrial discharges. Bacterial contamination in forested areas most likely results from fecal contamination by wildlife.

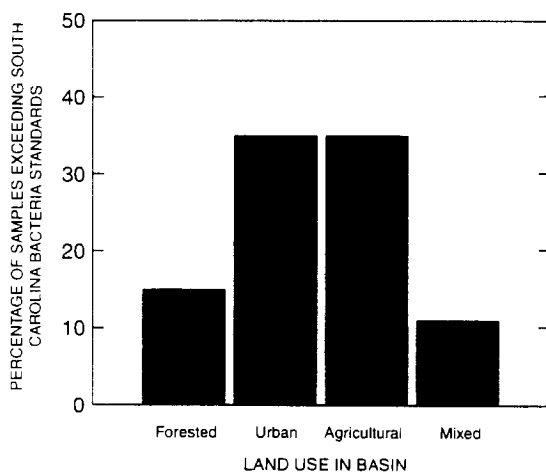


Figure 17. Urban and agricultural streams had more concentrations of bacteria that exceeded South Carolina State standards than forested and mixed land-use streams.

Biological Communities Reflected Land-Use Differences

Biological communities that inhabit streams in Santee Basin agricultural and urban areas were indicative of degraded water quality compared to those that inhabit streams that drain forested areas. Fish that have a low tolerance for contamination make up a smaller percentage of the fish community at agricultural and urban sites than at forested sites (fig. 18). This can result because fish such as darters and shiners that are sensitive to contaminants do not thrive at degraded sites. Other species such as catfish, redbreast sunfish, and some minnows that are relatively unaffected by contaminants will take the place of the more sensitive fish.

Urban and agricultural streams also had lower numbers of invertebrate species that are intolerant of contaminants than forested and mixed land-use streams. This is evidenced by the lower numbers of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa, a group of aquatic insects that are relatively intolerant of contamination, at urban and agricultural sites. The USEPA (Plafkin and others, 1989)



Over 85 fish species were identified during ecological sampling in the Santee Basin.

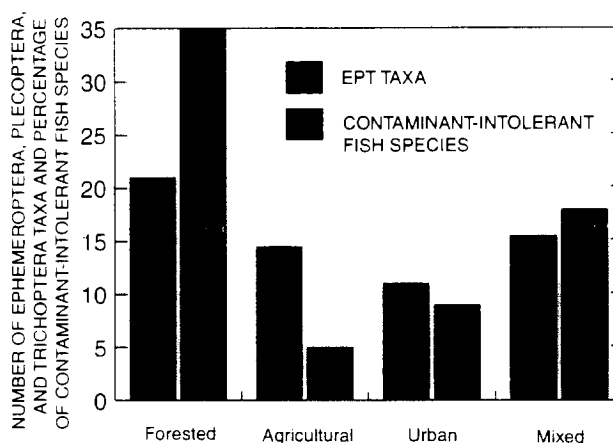
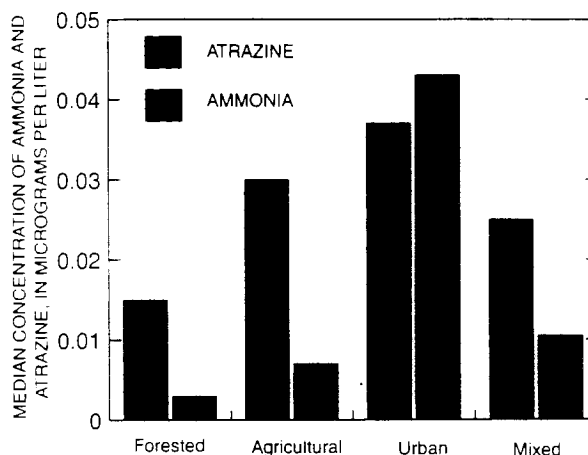


Figure 18. Compared to forested sites, urban and agricultural sites have higher concentrations of atrazine and ammonia as well as lower numbers of fish and invertebrates that are intolerant of contamination.

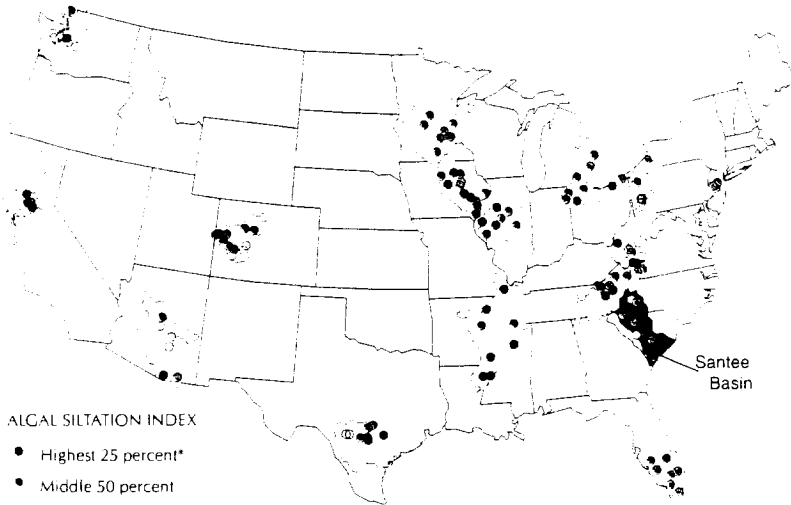
uses the presence or absence of EPT taxa as an indicator of aquatic community health.

Water-quality constituents and contaminant-intolerant species are related (fig. 18). Median concentrations of ammonia, a nutrient associated with wastewater discharges, and atrazine, an agricultural and turfgrass herbicide, are highest at urban and agricultural sites, corresponding to low numbers of contaminant-intolerant fish and invertebrate species. This suggests that these water-quality constituents have an effect on the aquatic community; however, other factors, primarily those associated with

aquatic habitats, can affect aquatic community health in ways that are similar to those that result from changes in water quality. In addition, sample sites were located in several different physiographic provinces, including the Blue Ridge, Piedmont, and Coastal Plain. Differences in species distributions and habitat in these different settings can make comparisons difficult to interpret. Most likely, a combination of water quality and habitat disturbance associated with agricultural and urban land uses results in the observed differences in biological communities.



SANTEE BASIN STREAMS WERE IN THE MIDDLE OF NATIONAL BIOLOGICAL RANKINGS



ALGAL SILTATION INDEX

- Highest 25 percent*
- Middle 50 percent
- ◌ Lowest 25 percent

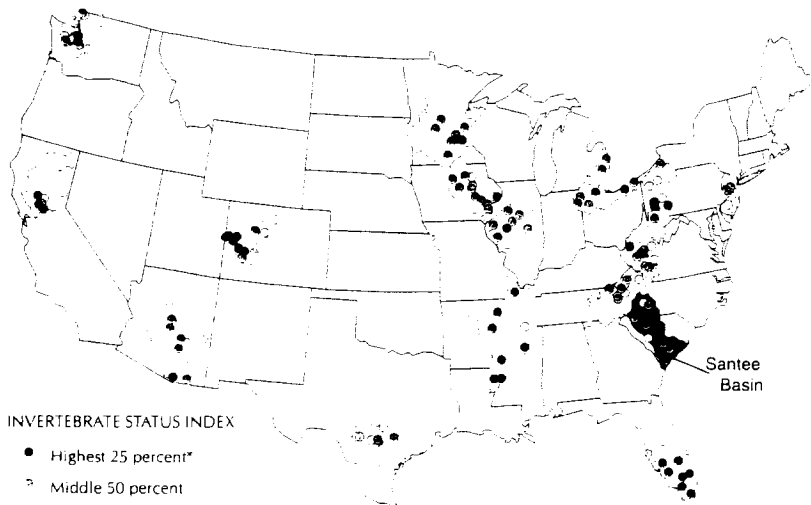
*Higher values suggest a more degraded stream site

Algal Siltation Index

Algal data collected in each of the NAWQA Study Units were combined and compared to show differences in the percentage of species that are able to avoid burial by sediment (Bahls and others, 1992). High percentages of these mobile algae are indicative of siltation of benthic habitats. Two sites in the Santee Basin were in the highest 25-percent category. One of these sites is the Edisto River, a blackwater river that has low light penetration. The high ranking at this site may result from low-light conditions that favor mobile algal species. Other variables that can lower light levels in streams can affect this index, including turbidity and forest canopy closure.

Invertebrate Status Index

A multimetric index was developed to compare invertebrate populations nationally. The index is an average of 11 metrics that summarize changes in richness, tolerance, trophic condition, and dominance associated with water-quality degradation. Only one site in the Santee Basin, an urban stream, was placed in the highest 25-percent category. The remaining Santee Basin sites were in the middle category, except for Jacob Fork River and McTier Creek, which were in the lowest category. Both of these sites represent the least degraded water quality in the study area and were located in areas with the least effects of human activity. Nationally, urban and agricultural sites tended to have the most degraded invertebrate communities according to this index.



INVERTEBRATE STATUS INDEX

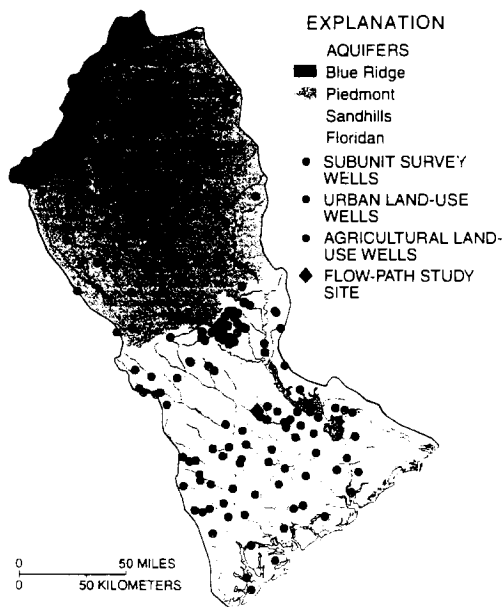
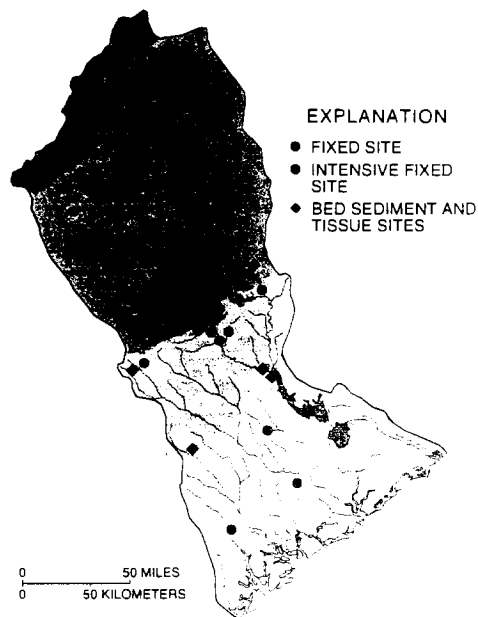
- Highest 25 percent*
- Middle 50 percent
- ◌ Lowest 25 percent

*Higher values suggest a more degraded stream site

STUDY UNIT DESIGN

STREAM CHEMISTRY AND BIOLOGY

Fixed sites were sampled to examine differences in streamwater quality due to the environmental setting, a combination of land use, geology, physiography, and climate. Intensive fixed sites were a subset of fixed sites that were sampled more frequently to determine the occurrence and seasonal variability of pesticides. Aquatic community structure, including algae, fish, and macroinvertebrates, was studied at each fixed site to quantify the effects of water quality on stream biota. Synoptic studies focused on low streamflow conditions in an urban setting in Gills Creek, S.C., and a mixed land-use setting in the South Fork Catawba River Basin, N.C. Streambed sediments and fish and clam tissues were sampled to determine the occurrence and distribution of trace elements and organic compounds.

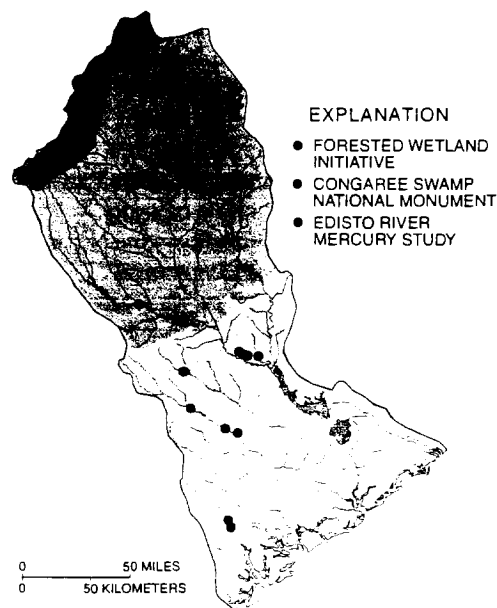


GROUND-WATER CHEMISTRY

Subunit surveys were conducted in three drinking-water supply aquifers to assess overall water quality. Land-use studies in urban and agricultural settings evaluated the effects of these land uses on shallow ground water. An agricultural flow-path study examined the transport and fate of nutrients and pesticides in shallow ground water.

SPECIAL STUDIES

The effects of a forested wetland on nutrient concentrations in stream water were studied as part of the Forested Wetland Initiative, a joint research project with the U.S. Forest Service. Baseline data on water quality and aquatic communities were collected in cooperation with the National Park Service at Congaree Swamp National Monument (Maluk and Abrahamsen, 1999). A study to determine the accumulation of mercury in fish tissues was conducted in the Edisto River Basin.



SUMMARY OF DATA COLLECTION IN THE SANTEE RIVER BASIN AND COASTAL DRAINAGES, 1995-98

Study component	What data were collected and why	Types of sites sampled	Number of sites	Sampling frequency and period
Stream Chemistry and Biology				
Fixed sites	Streamflow was measured continuously and samples were collected monthly for major ions, nutrients, organic carbon, suspended sediment, bacteria, pesticides, and physical properties to describe concentration, seasonal variability, and loads.	Streams draining basins ranging in size from 14 to 7,850 square miles and representing forested, agricultural, urban, and mixed land uses.	13	Monthly, plus 3-6 storms (October 1995-September 1997) Pesticides (February, May, August 1996)
Intensive fixed sites	In addition to the above constituents, samples were analyzed for dissolved pesticides to describe concentration and seasonal variability.	A subset of fixed sites draining agricultural, urban, and mixed land uses.	3	Weekly during growing season (February 1996-October 1997)
Urban synoptic study	Streamflow, major ions, nutrients, organic carbon, suspended sediment, bacteria, pesticides, volatile organic compounds, and physical properties were determined under low-flow conditions to describe concentrations and spatial distributions.	Urban streams draining basins ranging in size from 0.5 to 59.4 square miles.	16	September 1996
South Fork Catawba River synoptic study	Streamflow, major ions, nutrients, organic carbon, suspended sediment, bacteria, and physical properties were determined under low-flow conditions to describe concentrations and spatial distributions.	Mixed land use streams draining basins ranging in size from 5 to 350 square miles.	20	October 1997
Streambed sediments	Streambed sediments were analyzed for trace elements and hydrophobic pesticides and other organic compounds to determine occurrence and spatial distribution.	Sediment depositional zones at all fixed sites and other selected sites.	20	Summer 1995
Aquatic biota	Clams and fish livers were analyzed for trace metals, and clams and whole fish were analyzed for organic compounds to determine occurrence and spatial distribution.	All fixed sites and other selected sites.	20	Summers 1995, 1996
Fixed site reach assessment	Fish, benthic invertebrates, algae, and aquatic and riparian habitats were sampled and described to assess aquatic biological community structure in different land uses and associated habitats.	Stream reaches located at or near fixed sites. Sites represent the variety of land uses, geology, and physiography within the Santee Basin	13	Once in 1996 or 1997; Multiyear sites, once during 1996-98
Ground-Water Chemistry				
Study unit survey	Major ions, nutrients, pesticides, volatile organic compounds, dissolved organic carbon, trace metals, radon and physical parameters analyzed in three major drinking-water aquifers to determine ground-water quality.	Randomly chosen existing public supply, private domestic, irrigation, and industrial supply wells in the Piedmont, Sandhills, and Floridan aquifers.	90 (30 per aquifer)	Sandhills—Summer 1996 Floridan—Spring 1999 Piedmont—Fall 2000
Urban land-use study	Major ions, nutrients, pesticides, volatile organic compounds, dissolved organic carbon, and physical parameters analyzed in shallow ground water in the Columbia, South Carolina, metropolitan area.	Wells installed at randomly chosen commercial/residential land-use areas.	30	Summer 1996
Agricultural land-use study	Major ions, nutrients, pesticides, dissolved organic carbon, and physical parameters analyzed in shallow ground water in row-crop agricultural lands of the lower Coastal Plain.	Wells installed at randomly chosen row-crop land-use areas.	30	Summer 1997
Flow-path study	Major ions, nutrients, pesticides, dissolved organic carbon, and physical parameters analyzed to determine fate and transport of pesticides and nutrients in an agricultural setting.	Multidepth wells installed along a ground-water flow path in the lower Coastal Plain.	34	November 1997; April and August 1998
Special Studies				
Forested Wetland Initiative	Major ions, nutrients, and organic carbon analyzed upstream and downstream from a forested wetland to compare changes in concentrations.	Coosawhatchie River in the lower Coastal Plain.	2	Approximately quarterly, 1996-97
Congaree Swamp National Monument	Streamflow, major ions, nutrients, organic carbon, suspended sediment, bacteria, pesticides, and physical properties to describe concentrations and seasonal variability. Fish, benthic invertebrates, algae, and aquatic habitat described to assess aquatic biological community structure.	Streams draining basins ranging in size from 32 to 70 square miles with agricultural, urban, and forested land uses that drain into Congaree Swamp.	4	Quarterly 1996-98
Edisto mercury study	Sediment, stream water, aquatic insects, and vegetation, and whole fish analyzed for elemental and methyl mercury to examine bioaccumulation of mercury.	Blackwater streams in the Edisto River Basin and a reference site in the Piedmont	5	Once in 1998

GLOSSARY

- Ammonia**—A compound of nitrogen and hydrogen (NH_3) that is a common by-product of animal waste. Ammonia readily converts to nitrate in soils and streams.
- Aquatic-life criteria**—Water-quality guidelines for protection of aquatic life. Often refers to U.S. Environmental Protection Agency water-quality criteria for protection of aquatic organisms.
- Aquifer**—A water-bearing layer of soil, sand, gravel, or rock that will yield usable quantities of water to a well.
- Atmospheric deposition**—The transfer of substances from the air to the surface of the Earth, either in wet form (rain, fog, snow, dew, frost, hail) or in dry form (gases, aerosols, particles).
- Bedrock**—General term for consolidated (solid) rock that underlies soils or other unconsolidated material.
- Bed sediment**—The material that temporarily is stationary in the bottom of a stream or other watercourse.
- Concentration**—The amount or mass of a substance present in a given volume or mass of sample. Usually expressed as microgram per liter (water sample) or micrograms per kilogram (sediment or tissue sample).
- Confined aquifer (artesian aquifer)**—An aquifer that is completely filled with water under pressure and that is overlain by material that restricts the movement of water.
- Confining layer**—A layer of sediment or lithologic unit of low permeability that bounds an aquifer.
- Contamination**—Degradation of water quality compared to original or natural conditions due to human activity.
- Degradation products**—Compounds resulting from transformation of an organic substance through chemical, photochemical, and/or biochemical reactions.
- Denitrification**—A process by which oxidized forms of nitrogen such as nitrate (NO_3^-) are reduced to form nitrites, nitrogen oxides, ammonia, or free nitrogen; commonly brought about by the action of denitrifying bacteria and usually resulting in the escape of nitrogen to the air.
- Detect**—To determine the presence of a compound.
- DDT**—Dichloro-diphenyl-trichloroethane. An organochlorine insecticide no longer registered for use in the United States.
- Drainage area**—The drainage area of a stream at a specified location is that area, measured in a horizontal plane, which is enclosed by a drainage divide.
- Drinking-water standard or guideline**—A threshold concentration in a public drinking-water supply, designed to protect human health. As defined here, standards are U.S. Environmental Protection Agency regulations that specify the maximum contamination levels for public water systems required to protect the public welfare; guidelines have no regulatory status and are issued in an advisory capacity.
- Eutrophication**—The process by which water becomes enriched with plant nutrients, most commonly phosphorus and nitrogen.
- Fecal bacteria**—Microscopic single-celled organisms (primarily fecal coliforms and fecal streptococci) found in the wastes of warm-blooded animals. Their presence in water is used to assess the sanitary quality of water for body-contact recreation or for consumption. Their presence indicates contamination by the wastes of warm-blooded animals and the possible presence of pathogenic (disease producing) organisms.
- Flow path**—An underground route for ground-water movement, extending from a recharge (intake) zone to a discharge (output) zone such as a shallow stream.
- Ground water**—In general, any water that exists beneath the land surface, but more commonly applied to water in fully saturated soils and geologic formations.
- Herbicide**—A chemical or other agent applied for the purpose of killing undesirable plants. See also Pesticide.
- Insecticide**—A substance or mixture of substances intended to destroy or repel insects.
- Intolerant organisms**—Organisms that are not adaptable to human alterations to the environment and thus decline in numbers where human alterations occur.
- Invertebrate**—An animal having no backbone or spinal column.
- Major ions**—Constituents commonly present in concentrations exceeding 1.0 milligram per liter. Dissolved cations generally are calcium, magnesium, sodium, and potassium; the major anions are sulfate, chloride, fluoride, nitrate, and those contributing to alkalinity, most generally assumed to be bicarbonate and carbonate.
- Maximum contaminant level (MCL)**—Maximum permissible level of a contaminant in water that is delivered to any user of a public water system. MCLs are enforceable standards established by the U.S. Environmental Protection Agency.
- Median**—The middle or central value in a distribution of data ranked in order of magnitude. The median is also known as the 50th percentile.
- Metamorphic rock**—Rock that has formed in the solid state in response to pronounced changes of temperature, pressure, and chemical environment.
- Method detection limit**—The minimum concentration of a substance that can be accurately identified and measured with present laboratory technologies.
- Micrograms per liter ($\mu\text{g/L}$)**—A unit expressing the concentration of constituents in solution as weight (micrograms) of solute per unit volume (liter) of water; equivalent to one part per billion in most streamwater and ground water. One thousand micrograms per liter equals 1 mg/L.
- Milligrams per liter (mg/L)**—A unit expressing the concentration of chemical constituents in solution as weight (milligrams) of solute per unit volume (liter) of water; equivalent to one part per million in most streamwater and ground water. One thousand micrograms per liter equals 1 mg/L.
- Minimum reporting level (MRL)**—The smallest measured concentration of a constituent that may be reliably reported using a given analytical method. In many cases, the MRL is used when documentation for the method detection limit is not available.
- Monitoring well**—A well designed for measuring water levels and testing ground-water quality.
- Nitrate**—An ion consisting of nitrogen and oxygen (NO_3^-). Nitrate is a plant nutrient and is very mobile in soils.
- Nutrient**—Element or compound essential for animal and plant growth. Common nutrients in fertilizer include nitrogen, phosphorus, and potassium.
- Pesticide**—A chemical applied to crops, rights of way, lawns, or residences to control weeds, insects, fungi, nematodes, rodents or other "pests."
- Phosphorus**—A nutrient essential for growth that can play a key role in stimulating aquatic growth in lakes and streams.
- Study Unit**—A major hydrologic system of the United States in which NAWQA studies are focused. Study Units are geographically defined by a combination of ground- and surface-water features and generally encompass more than 4,000 square miles of land area.
- Trace element**—An element found in only minor amounts (concentrations less than 1.0 milligram per liter) in water or sediment; includes arsenic, cadmium, chromium, copper, lead, mercury, nickel, and zinc.
- Unconfined aquifer**—An aquifer whose upper surface is a water table; an aquifer containing unconfined ground water.
- Volatile organic compounds (VOCs)**—Organic chemicals that have a high vapor pressure relative to their water solubility. VOCs include components of gasoline, fuel oils, and lubricants, as well as organic solvents, fumigants, some inert ingredients in pesticides, and some by-products of chlorine disinfection.
- Water table**—The point below the land surface where ground water is first encountered and below which the earth is saturated. Depth to the water table varies widely across the country.

REFERENCES

- Abrahamsen, T.A., 1999. Trace elements in bed sediments and biota from streams in the Santee River Basin and coastal drainages, North and South Carolina, 1995–97: U.S. Geological Survey Water-Resources Investigations Report 99–4179, 26 p.
- Aucott, W.R., Davis, M.E., and Speiran, G.K., 1987. Geohydrologic framework of the Coastal Plain aquifers of South Carolina: U.S. Geological Survey Water-Resources Investigations Report 85–4271, 7 sheets.
- Bahls, L.R., Burkantis, R., and Tralles, S., 1992. Benchmark biology of Montana reference streams: Department of Health and Environmental Science, Water Quality Bureau, Helena, Mont., 47 p.
- Barbash, J.E., and Resek, E.A., 1996. Pesticides in ground water: Distribution, trends, and governing factors, v. 2—Pesticides in the hydrologic system: Chelsea, Mich., Ann Arbor Press, 590 p.
- Canadian Council of Ministers of the Environment, 1995. Protocol for the derivation of Canadian sediment quality guidelines for the protection of aquatic life: Report CCME EPC–98E, 38 p.
- Code of Federal Regulations, 1996. Title 40 Protection of the environment, chapter 1: Environmental Protection Agency, Part 401, Section 401.15, Toxic Pollutants.
- Fenneman, N.M., 1946. Physical divisions of the United States: Washington, D.C., U.S. Geological Survey special map, scale 1:7,000,000.
- Krabbenhoft, D.P., Wiener, J.G., Brumbaugh, W.G., Olson, M.L., DeWild, J.F., and Sabin, T.J., 1999. A national pilot study of mercury contamination of aquatic ecosystems along multiple gradients. *in* Morganwalp, D.W., and Buxton, H.T., eds., 1999. U.S. Geological Survey Toxic Substances Hydrology Program—Proceedings of the Technical Meeting, Charleston, S.C., March 8–12, 1999, v. 2—Contamination of hydrologic systems and related ecosystems: U.S. Geological Survey Water-Resources Investigations Report 99–4018-B, p. 147–160.
- Lindsey, Ginny, and Lewis, Ann, 1994. The rainbow river—Stain on the Piedmont: Asheville, N.C., Clean Water Fund of North Carolina, 52 p.
- Litke, D.W., 1999. Review of phosphorus control measures in the United States and their effects on water quality: U.S. Geological Survey Water-Resources Investigations Report 99–4007, 38 p.
- Lopes, T.J., 1998. Nonpoint sources of volatile organic compounds in urban areas—Relative importance of urban land surfaces and air: *Environmental Pollution*, v. 101, p. 221–230.
- Lopes, T.J., Fallon, J.D., and Maluk, T.L., 2000. Compositing water samples for analysis of volatile organic compounds: *Journal of Environmental Engineering*, v. 126, no. 8, p. 769–773.
- Maluk, T.L., 1999. Pesticides and bacteria in an urban stream—Gills Creek, Columbia, South Carolina: U.S. Geological Survey Fact Sheet FS–131–98, 4 p.
- Maluk, T.L. and Abrahamsen, T.A., 1999. Results of water-quality sampling and ecological characterization of streams of Congaree Swamp, South Carolina 1995–98: U.S. Geological Survey Water-Resources Investigation Report 99–4121, 76 p.
- Maluk, T.L., and Kelley, R.E. II, 1998. Pesticides in surface waters of the Santee River Basin and coastal drainages, North and South Carolina: U.S. Geological Survey Fact Sheet FS–007–98, 6 p.
- Maluk, T.L., Reuber, E.J., and Hughes, W.B., 1998. Nutrients in waters of the Santee River Basin and coastal drainages, North and South Carolina, 1973–93: U.S. Geological Survey Water-Resources Investigations Report 97–4172, 60 p.
- Nolan, B.T., and Stoner, J.D., 2000. Nutrients in ground waters of the conterminous United States 1992–1995: *Environmental Science and Technology*, v. 34, no. 7, p. 1156–1165.
- Plafkin, J.L., Barbour, M.T., Porter, K.D., Gross, S.K., and Hughes, R.M., 1989. Rapid bioassessment protocols for use in streams and rivers: USEPA/440/4–89–001, variously paged.
- Reuber, E.J., 1999. Shallow ground-water quality in the Coastal Plain of Columbia, South Carolina, 1996: U.S. Geological Survey Fact Sheet FS–130–98, 6 p.
- Smock, L.A., and Gilinsky, Ellen, 1992. Coastal Plain blackwater streams. *in* Hackney, C.T., Adams, S.M., and Marshall, W.H., eds., *Biodiversity of the southeastern United States aquatic communities*: New York, John Wiley, p. 271–313.
- South Carolina Agricultural Statistics Service, 1999. South Carolina Agricultural Statistics—Crops, livestock, poultry 1997–1999: Columbia, South Carolina Agricultural Statistics Service, 84 p.
- South Carolina Department of Health and Environmental Control, 1992. Water classifications and standards (Regulation 61–68): South Carolina Department of Health and Environmental Control, 35 p.
- _____, 2000. Fish consumption advisory tables: accessed May 25, 2000, at <http://www.state.sc.us/dhec/eqc/>
- South Carolina Water Resources Commission, 1983. South Carolina State Water Assessment: South Carolina Water Resources Commission Report No. 140, 367 p.
- Stecker, K.K., and Crocker, Michael, 1991. South Carolina lake classification survey: South Carolina Department of Health and Environmental Control Technical Report No. 006–91, 509 p.
- U.S. Environmental Protection Agency, 1996. Drinking water regulation and health advisories: U.S. Environmental Protection Agency report EPA–822B–96/002, 11 p.
- _____, 1999. Proposed radon in drinking water rule: U.S. Environmental Protection Agency Fact Sheet EPA 815–F–99–006, 7 p.
- _____, 1986. Quality criteria for water—1986: U.S. Environmental Protection Agency report EPA 440/5–86–002 (variously paged).
- _____, 1998. Update—Listing of fish and wildlife advisories: U.S. Environmental Protection Agency, LFWA Fact Sheet EPA–823–98–009, 4 p.
- U.S. Geological Survey, 1999. The quality of our Nation's waters—Nutrients and pesticides: U.S. Geological Survey Circular 1225, 82 p.
- Wilhelm, L.J., and Maluk, T.L., 1998. Fecal-indicator bacteria in surface waters of the Santee River Basin and coastal drainages, North and South Carolina, 1995–98: U.S. Geological Survey Fact Sheet FS–085–98, 6 p.

APPENDIX—WATER-QUALITY DATA FROM THE SANTEE RIVER BASIN AND COASTAL DRAINAGES IN A NATIONAL CONTEXT

For a complete view of Santee River Basin and coastal drainages data and for additional information about specific benchmarks used, visit our Web site at <http://water.usgs.gov/nawqa/>. Also visit the NAWQA Data Warehouse for access to NAWQA data sets at <http://infotrek.er.usgs.gov/wdbctx/nawqa/nawqa.home>.

This appendix is a summary of chemical concentrations and biological indicators assessed in the Santee River Basin and coastal drainages. Selected results for this Study Unit are graphically compared to results from as many as 36 NAWQA Study Units investigated from 1991 to 1998 and to national water-quality benchmarks for human health, aquatic life, or fish-eating wildlife. The chemical and biological indicators shown were selected on the basis of frequent detection, detection at concentrations above a national benchmark, or regulatory or scientific importance. The graphs illustrate how conditions associated with each land use sampled in the Santee River Basin and coastal drainages compare to results from across the Nation, and how conditions compare among the several land uses. Graphs for chemicals show only detected concentrations and, thus, care must be taken to evaluate detection frequencies in addition to concentrations when comparing study-unit and national results. For example, tebuthiuron concentrations in Santee River Basin and coastal drainages agricultural streams were similar to the national distribution, but the detection frequency was much higher (66 percent compared to 22 percent).

CHEMICALS IN WATER

Concentrations and detection frequencies, Santee River Basin and coastal drainages, 1995–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable

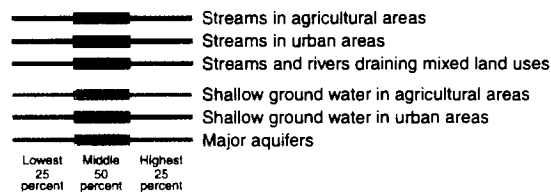
◆ Detected concentration in Study Unit

66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency

◆◆ Not measured or sample size less than two

12 Study-unit sample size. For ground water, the number of samples is equal to the number of wells sampled

National ranges of detected concentrations, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

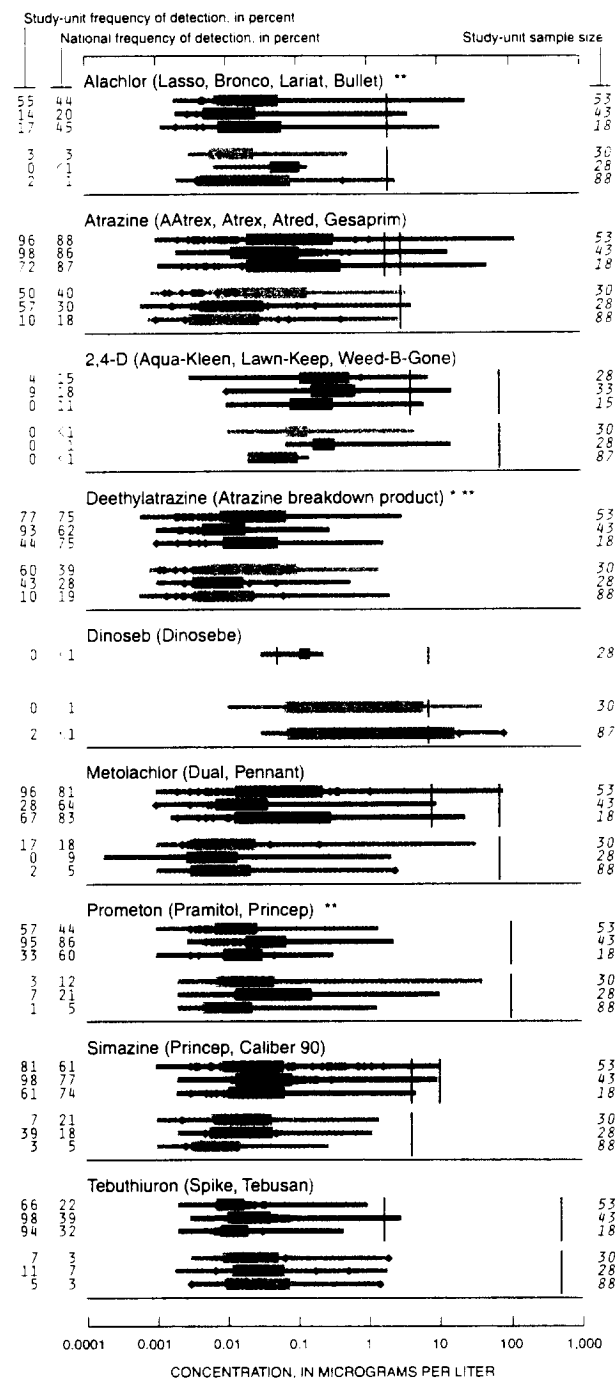


National water-quality benchmarks

National benchmarks include standards and guidelines related to drinking-water quality, criteria for protecting the health of aquatic life, and a goal for preventing stream eutrophication due to phosphorus. Sources include the U.S. Environmental Protection Agency and the Canadian Council of Ministers of the Environment

- | Drinking-water quality (applies to ground water and surface water)
- | Protection of aquatic life (applies to surface water only)
- | Prevention of eutrophication in streams not flowing directly into lakes or impoundments
- No benchmark for drinking-water quality
- ◆◆ No benchmark for protection of aquatic life

Pesticides in water—Herbicides



Other herbicides detected

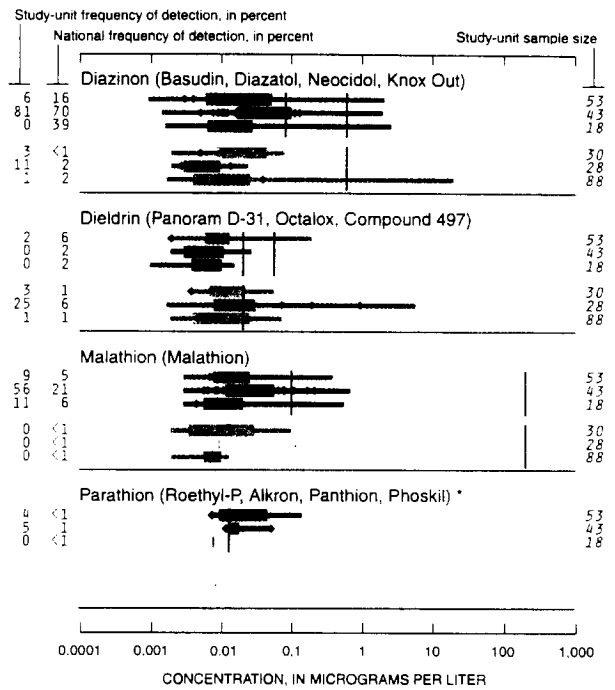
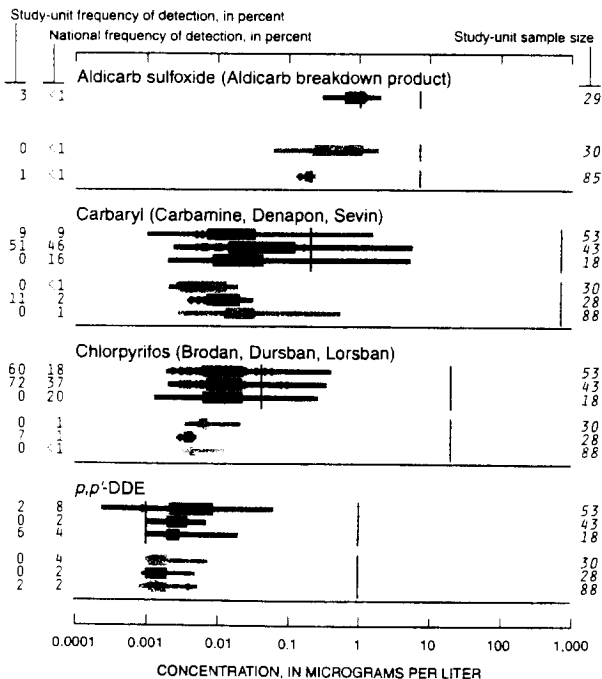
- Acifluorfen (Blazer, Tackle 2S) **
- Benfluralin (Balan, Benefin, Bonalan) * **
- Bentazon (Basagran, Bentazone) **
- Bromacil (Hyvar X, Urox B, Bromax)
- Butylate (Sutan +, Genate Plus, Butilate) **

Cyanazine (Bladex, Fortrol)
 DCPA (Dacthal, chlorthal-dimethyl) * **
 2,6-Diethylaniline (Alachlor breakdown product) * **
 Diuron (Crisuron, Karmex, Diurex) **
 EPTC (Eptam, Farmarox, Alirox) * **
 Ethalfuralin (Sonalan, Curbit) * **
 Fenuron (Fenuon, Fenidim) * **
 Fluometuron (Flo-Met, Cotoran) **
 Linuron (Lorox, Linex, Sarclex, Linurex, Afalon) *
 Metribuzin (Lexone, Sencor)
 Molinate (Ordran) * **
 Neburon (Neburea, Neburyl, Noruben) * **
 Norflurazon (Evital, Predict, Solicam, Zorial) * **
 Oryzalin (Surflan, Dirimal) * **
 Pendimethalin (Pre-M, Prowl, Stomp) * **
 Pronamide (Kerb, Propyzamid) **
 Propham (Tuberite) **
 Terbacil (Sinbar) **
 Trifluralin (Treflan, Gowan, Tri-4, Trific)

Herbicides not detected

Acetochlor (Harness Plus, Surpass) * **
 Bromoxynil (Buctril, Brominal) *
 Chloramben (Amiben, Amilon-WP, Vegiben) **
 Clopyralid (Stinger, Lontrel, Transline) * **
 2,4-DB (Butyrac, Butoxone, Embutox Plus, Embutone) * **
 Dacthal mono-acid (Dacthal breakdown product) * **
 Dicamba (Banvel, Dianat, Scotts Proturf)
 Dichlorprop (2,4-DP, Seritox 50, Lentemul) * **
 MCPA (Rhomene, Rhonox, Chiptox)
 MCPB (Thistrol) * **
 Napropamide (Devrinol) * **
 Pebulate (Tillam, PEBC) * **
 Picloram (Grazon, Tordon)
 Propachlor (Ramrod, Satecid) **
 Propanil (Stam, Stampede, Wham) * **
 2,4,5-T **
 2,4,5-TP (Silvex, Fenoprop) **
 Thiobencarb (Bolero, Saturn, Benthicarb) * **
 Triallate (Far-Go, Avadex BW, Tri-allate) *
 Triclopyr (Garlon, Grandstand, Redeem, Remedy) * **

Pesticides in water—Insecticides



Other insecticides detected

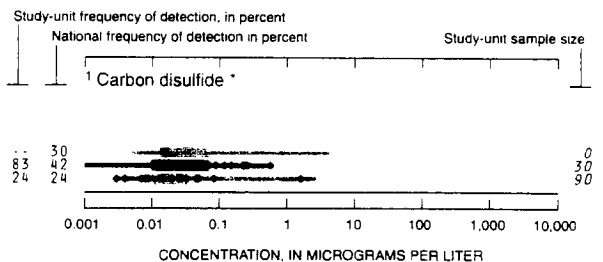
Aldicarb (Temik, Ambush, Pounce)
 Aldicarb sulfone (Standak, aldoxycarb)
 Carbofuran (Furadan, Curaterr, Yaltox)
 Fonofos (Dyfonate, Capfos, Cudgel, Tycap) **
 3-Hydroxycarbofuran (Carbofuran breakdown product) * **
 Methiocarb (Slug-Geta, Grandslam, Mesuro) * **
 Methomyl (Lanox, Lannate, Acinate) **
 Oxamyl (Vydate L, Pratt) **
 cis-Permethrin (Ambush, Astro, Pounce) * **
 Propoxur (Baygon, Blattanex, Uden, Proprotox) * **

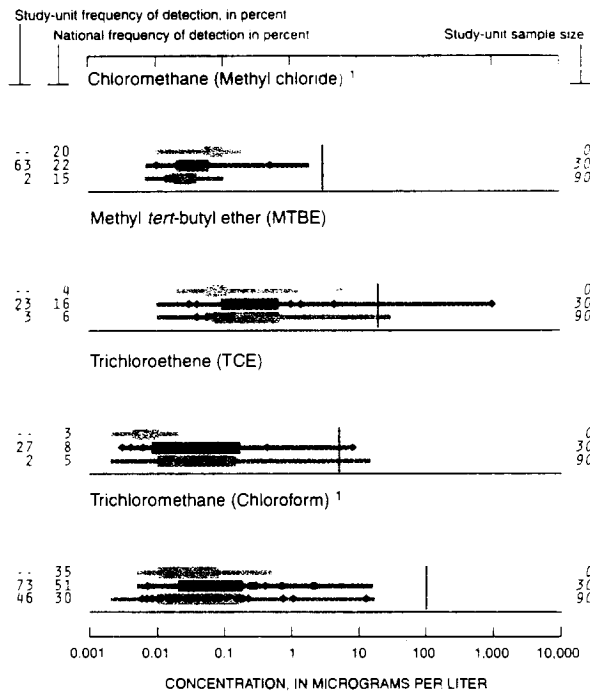
Insecticides not detected

Azinphos-methyl (Guthion, Gusathion M) *
 Disulfoton (Disyston, Di-Syston) **
 Ethoprop (Mocap, Ethoprophos) * **
 alpha-HCH (alpha-BHC, alpha-lindane) **
 gamma-HCH (Lindane, gamma-BHC)
 Methyl parathion (Penncap-M, Folidol-M) **
 Phorate (Thimet, Granutox, Geomet, Rampart) * **
 Propargite (Comite, Omite, Ornamite) * **
 Terbufos (Conraven, Counter, Pilarfox) **

Volatile organic compounds (VOCs) in ground water

These graphs represent data from 16 Study Units, sampled from 1996 to 1998





¹ Many of the samples in this study were diluted prior to laboratory analysis and therefore the actual detection frequency may be larger than the value listed.

Other VOCs detected

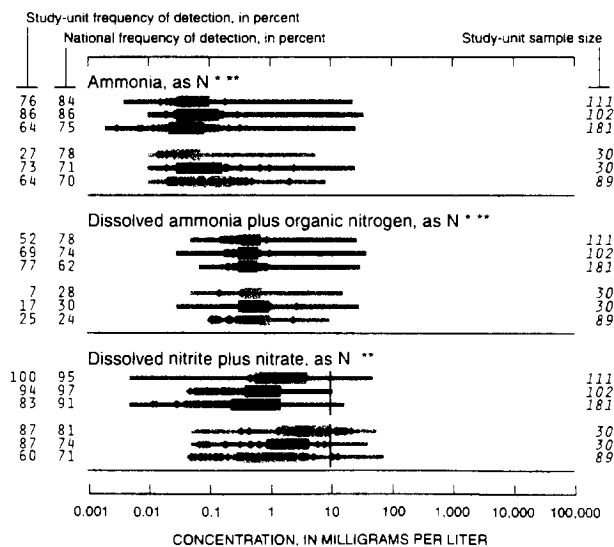
- tert*-Amyl methyl ether (*tert*-amyl methyl ether (TAME)) *
- Benzene
- Bromodichloromethane (Dichlorobromomethane)
- Bromomethane (Methyl bromide)
- 2-Butanone (Methyl ethyl ketone (MEK)) *
- sec*-Butylbenzene *
- tert*-Butylbenzene *
- Chlorobenzene (Monochlorobenzene)
- Chlorodibromomethane (Dibromochloromethane)
- Chloroethane (Ethyl chloride) *
- Chloroethene (Vinyl chloride)
- 1,2-Dichlorobenzene (*o*-Dichlorobenzene)
- 1,3-Dichlorobenzene (*m*-Dichlorobenzene)
- 1,4-Dichlorobenzene (*p*-Dichlorobenzene)
- Dichlorodifluoromethane (CFC 12, Freon 12)
- 1,2-Dichloroethane (Ethylene dichloride)
- 1,1-Dichloroethane (Ethylidene dichloride) *
- 1,1-Dichloroethene (Vinylidene chloride)
- trans*-1,2-Dichloroethene ((*E*)-1,2-Dichloroethene)
- cis*-1,2-Dichloroethene ((*Z*)-1,2-Dichloroethene)
- Dichloromethane (Methylene chloride)
- 1,2-Dichloropropane (Propylene dichloride)
- Diethyl ether (Ethyl ether) *
- 1,2-Dimethylbenzene (*o*-Xylene)
- 1,3 & 1,4-Dimethylbenzene (*m*-&*p*-Xylene)
- 1-4-Epoxy butane (Tetrahydrofuran, Diethylene oxide) *
- Ethylbenzene (Phenylethane)
- Iodomethane (Methyl iodide) *
- Isopropylbenzene (Cumene) *
- p*-Isopropyltoluene (*p*-Cymene) *
- Methylbenzene (Toluene)
- 2-Propanone (Acetone) *
- 1,1,2,2-Tetrachloroethane *
- Tetrachloroethene (Perchloroethene)
- Tetrachloromethane (Carbon tetrachloride)
- 1,2,3,5-Tetramethylbenzene (Isodurene) *
- Tribromomethane (Bromoform)

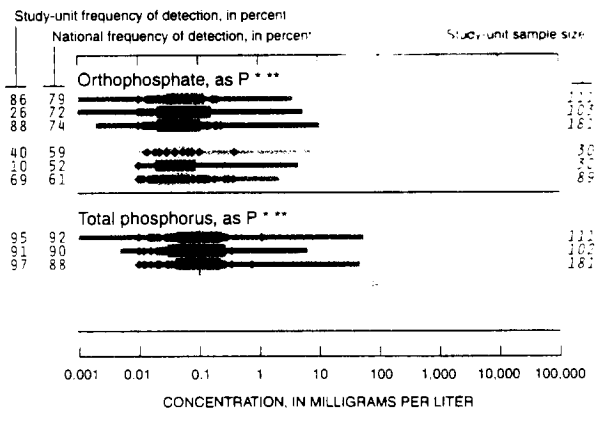
- 1,1,1-Trichloroethane (Methylchloroform)
- Trichlorofluoromethane (CFC 11, Freon 11)

VOCs not detected

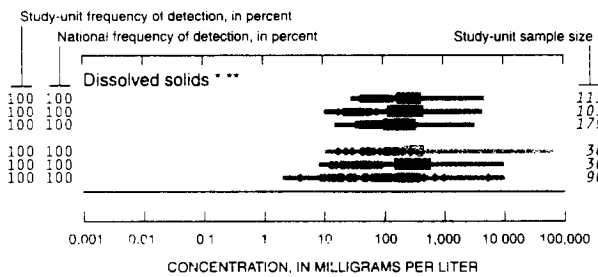
- Bromobenzene (Phenyl bromide) *
- Bromochloromethane (Methylene chlorobromide)
- Bromoethene (Vinyl bromide) *
- n*-Butylbenzene (1-Phenylbutane) *
- 3-Chloro-1-propene (3-Chloropropene) *
- 1-Chloro-2-methylbenzene (*o*-Chlorotoluene)
- 1-Chloro-4-methylbenzene (*p*-Chlorotoluene)
- 1,2-Dibromo-3-chloropropane (DBCP, Nemagon)
- 1,2-Dibromoethane (Ethylene dibromide, EDB)
- Dibromomethane (Methylene dibromide) *
- trans*-1,4-Dichloro-2-butene ((*Z*)-1,4-Dichloro-2-butene) *
- 2,2-Dichloropropane *
- 1,3-Dichloropropane (Trimethylene dichloride) *
- trans*-1,3-Dichloropropene ((*E*)-1,3-Dichloropropene)
- cis*-1,3-Dichloropropene ((*Z*)-1,3-Dichloropropene)
- 1,1-Dichloropropene *
- Diisopropyl ether (Diisopropylether (DIPE)) *
- Ethenylbenzene (Styrene)
- Ethyl methacrylate *
- Ethyl *tert*-butyl ether (Ethyl-*t*-butyl ether (ETBE)) *
- 1-Ethyl-2-methylbenzene (2-Ethyltoluene) *
- Hexachlorobutadiene
- 1,1,1,2,2,2-Hexachloroethane (Hexachloroethane)
- 2-Hexanone (Methyl butyl ketone (MBK)) *
- Methyl acrylonitrile *
- Methyl-2-methacrylate (Methyl methacrylate) *
- 4-Methyl-2-pentanone (Methyl isobutyl ketone (MIBK)) *
- Methyl-2-propenoate (Methyl acrylate) *
- Naphthalene
- 2-Propenenitrile (Acrylonitrile)
- n*-Propylbenzene (Isocumene) *
- 1,1,1,2-Tetrachloroethane
- 1,2,3,4-Tetramethylbenzene (Prenhitene) *
- 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon 113) *
- 1,2,4-Trichlorobenzene
- 1,2,3-Trichlorobenzene *
- 1,1,2-Trichloroethane (Vinyl trichloride)
- 1,2,3-Trichloropropane (Allyl trichloride)
- 1,2,3-Trimethylbenzene (Hemimellitene) *
- 1,2,4-Trimethylbenzene (Pseudocumene) *
- 1,3,5-Trimethylbenzene (Mesitylene) *

Nutrients in water

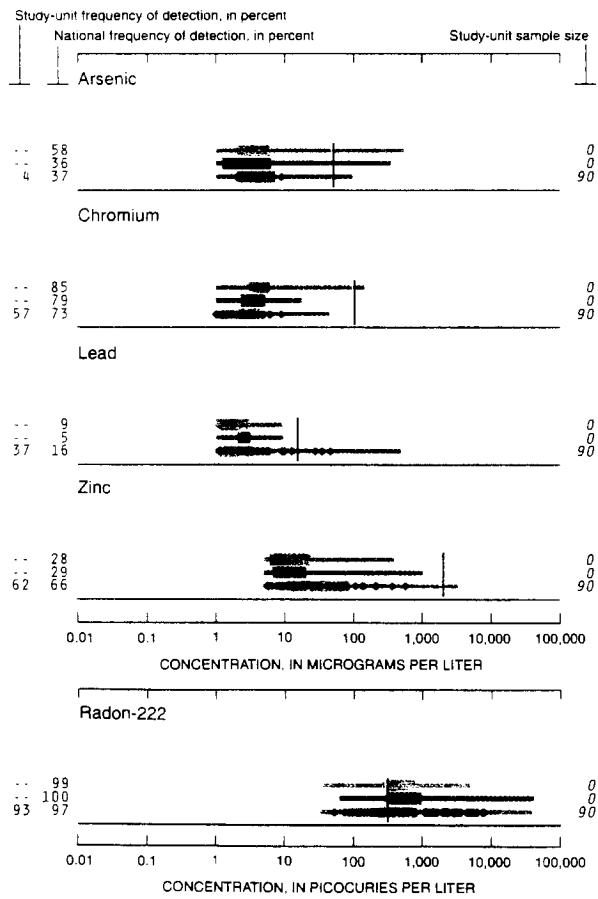




Dissolved solids in water



Trace elements in ground water



Other trace elements detected

Selenium
Uranium

Trace elements not detected

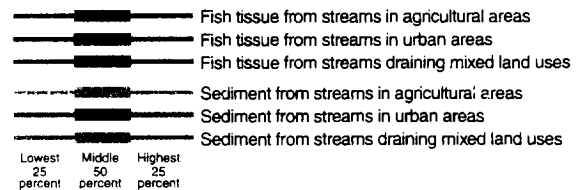
Cadmium

CHEMICALS IN FISH TISSUE AND BED SEDIMENT

Concentrations and detection frequencies, Santee River Basin and coastal drainages, 1995-98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals. Study-unit frequencies of detection are based on small sample sizes; the applicable sample size is specified in each graph

- ◆ Detected concentration in Study Unit
- 56 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- Not measured or sample size less than two
- 22 Study-unit sample size

National ranges of concentrations detected, by land use, in 36 NAWQA Study Units, 1991-98—Ranges include only samples in which a chemical was detected

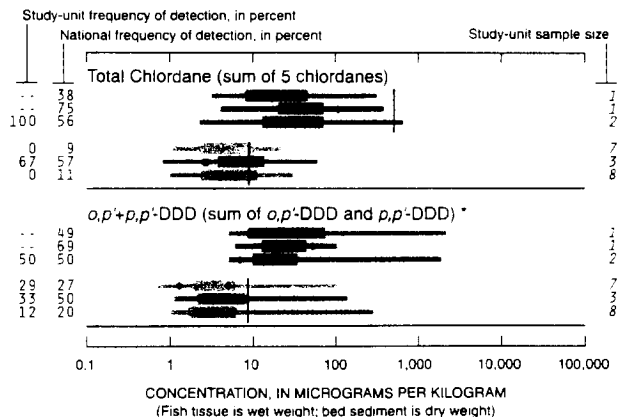


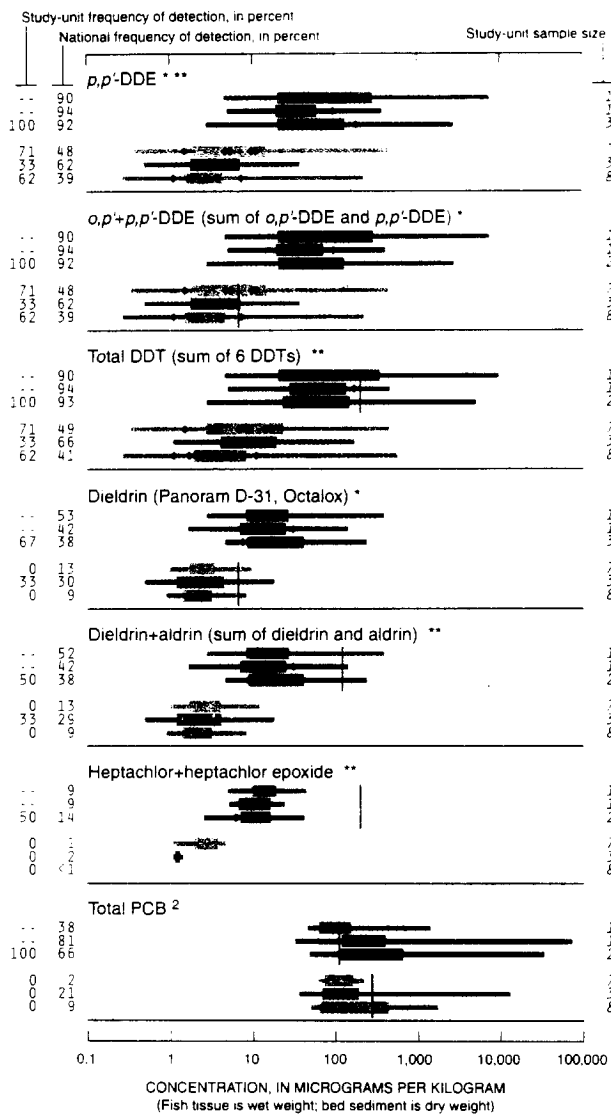
National benchmarks for fish tissue and bed sediment

National benchmarks include standards and guidelines related to criteria for protection of the health of fish-eating wildlife and aquatic organisms. Sources include the U.S. Environmental Protection Agency, other Federal and State agencies, and the Canadian Council of Ministers of the Environment

- | Protection of fish-eating wildlife (applies to fish tissue)
- | Protection of aquatic life (applies to bed sediment)
- No benchmark for protection of fish-eating wildlife
- No benchmark for protection of aquatic life

Organochlorines in fish tissue (whole body) and bed sediment





² The national detection frequencies for total PCB in sediment are biased low because about 30 percent of samples nationally had elevated detection levels compared to this Study Unit. See <http://water.usgs.gov/nawqa/> for additional information.

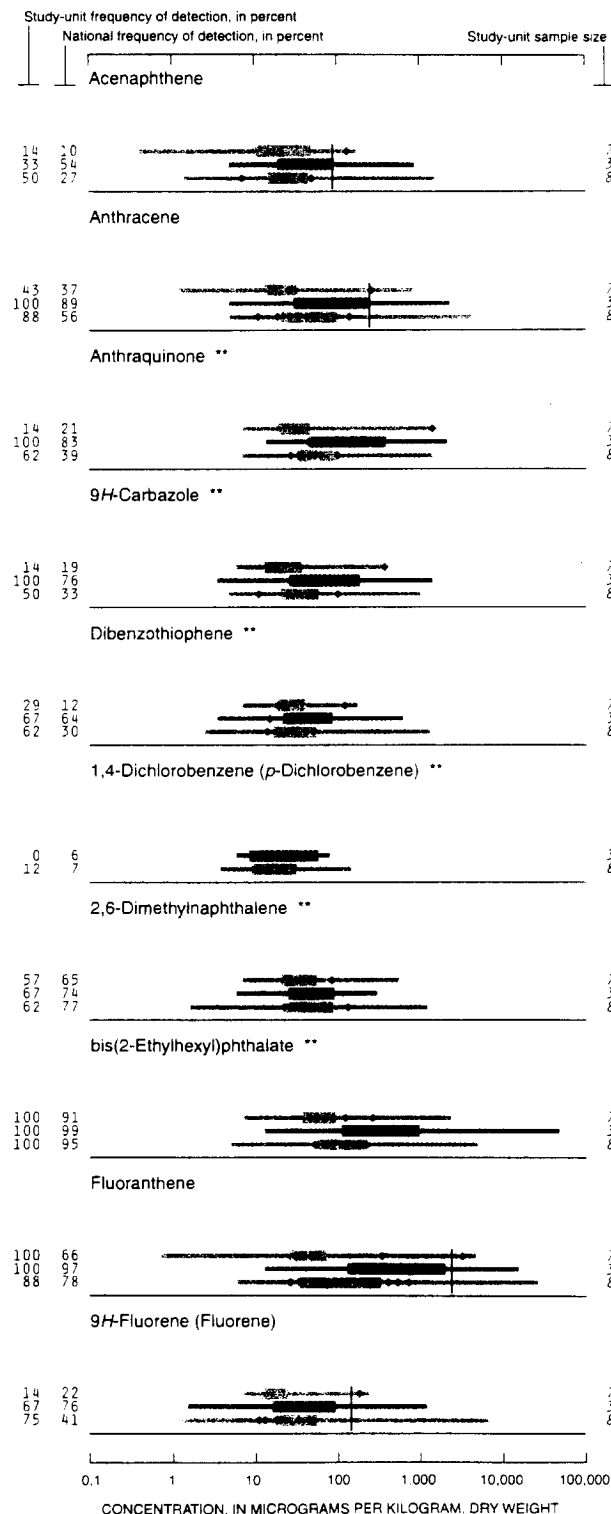
Other organochlorines detected

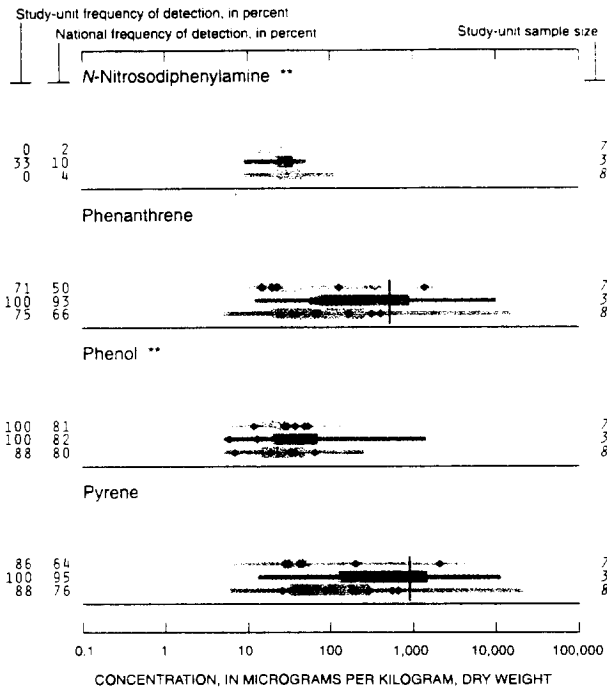
- DCPA (Dacthal, chlorthal-dimethyl) * **
- o,p'*+*p,p'*-DDT (sum of *o,p'*-DDT and *p,p'*-DDT) *
- Heptachlor epoxide (Heptachlor breakdown product) *
- Hexachlorobenzene (HCB) **
- Mirex (Dechlorane) **

Organochlorines not detected

- Chloroneb (Chloronebe, Demosan) * **
- Endosulfan I (alpha-Endosulfan, Thiodan) * **
- Endrin (Endrine)
- gamma-HCH (Lindane, gamma-BHC, Gammexane) *
- Total-HCH (sum of alpha-HCH, beta-HCH, gamma-HCH, and delta-HCH) **
- Isodrin (Isodrine, Compound 711) * **
- p,p'*-Methoxychlor (Mariate, methoxychlor) * **
- o,p'*-Methoxychlor * **
- cis*-Permethrin (Ambush, Astro, Pounce) * **
- trans*-Permethrin (Ambush, Astro, Pounce) * **
- Toxaphene (Camphechlor, Hercules 3956) * **

Semivolatile organic compounds (SVOCs) in bed sediment





Other SVOCs detected

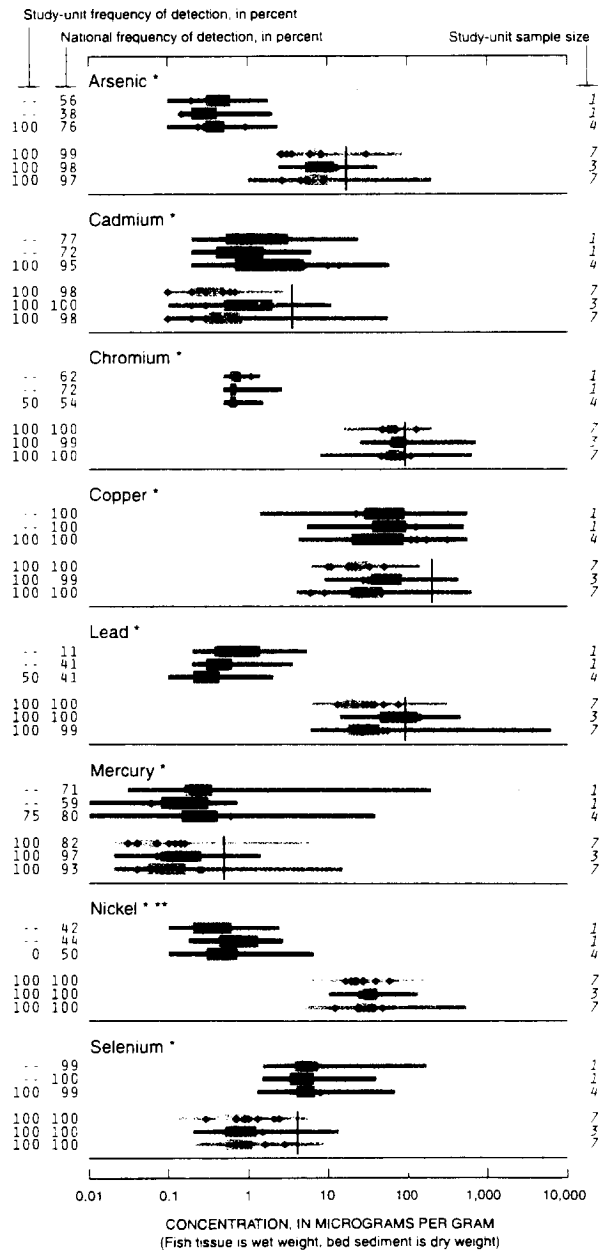
- Acenaphthylene
- Acridine **
- Benzo[a]anthracene
- Benzo[a]pyrene
- Benzo[b]fluoranthene **
- Benzo[ghi]perylene **
- Benzo[k]fluoranthene **
- Butylbenzylphthalate **
- Chrysene
- p-Cresol **
- Di-n-butylphthalate **
- Di-n-octylphthalate **
- Dibenz[a,h]anthracene
- Diethylphthalate **
- 1,2-Dimethylnaphthalene **
- 1,6-Dimethylnaphthalene **
- Dimethylphthalate **
- 2-Ethyl-naphthalene **
- Indeno[1,2,3-cd]pyrene **
- Isoquinoline **
- 1-Methyl-9H-fluorene **
- 2-Methylanthracene **
- 4,5-Methylenepheneanthrene **
- 1-Methylphenanthrene **
- 1-Methylpyrene **
- Naphthalene
- Phenanthridine **
- Quinoline **
- 2,3,6-Trimethylnaphthalene **

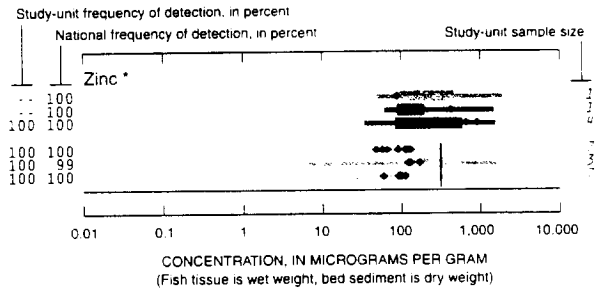
SVOCs not detected

- C8-Alkylphenol **
- Azobenzene **
- Benzo[c]cinnoline **
- 2,2-Biquinoline **
- 4-Bromophenyl-phenylether **
- 4-Chloro-3-methylphenol **
- bis(2-Chloroethoxy)methane **
- 2-Chloronaphthalene **
- 2-Chlorophenol **

- 4-Chlorophenyl-phenylether **
- 1,2-Dichlorobenzene (o-Dichlorobenzene) **
- 1,3-Dichlorobenzene (m-Dichlorobenzene) **
- 3,5-Dimethylphenol **
- 2,4-Dinitrotoluene **
- Isophorone **
- Nitrobenzene **
- N-Nitrosodi-n-propylamine **
- Pentachloronitrobenzene **
- 1,2,4-Trichlorobenzene **

Trace elements in fish tissue (livers) and bed sediment





BIOLOGICAL INDICATORS

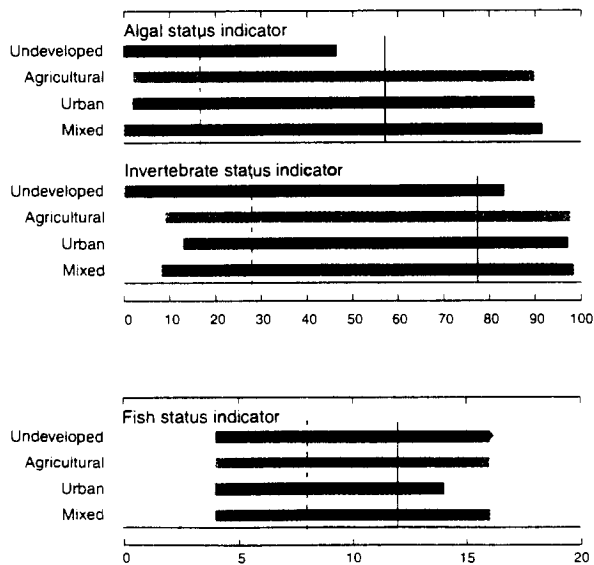
Higher national scores suggest habitat disturbance, water-quality degradation, or naturally harsh conditions. The status of algae, invertebrates (insects, worms, and clams), and fish provide a record of water-quality and stream conditions that water-chemistry indicators may not reveal. **Algal status** focuses on the changes in the percentage of certain algae in response to increasing siltation, and it often correlates with higher nutrient concentrations in some regions. **Invertebrate status** averages 11 metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. **Fish status** sums the scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that increase in association with water-quality degradation

Biological indicator value, Santee River Basin and coastal drainages, by land use, 1995-98

- ◆ Biological status assessed at a site

National ranges of biological indicators, in 16 NAWQA Study Units, 1994-98

- ▬ Streams in undeveloped areas
- ▬ Streams in agricultural areas
- ▬ Streams in urban areas
- ▬ Streams in mixed-land-use areas
- 75th percentile
- - - 25th percentile



A COORDINATED EFFORT

Coordination with agencies and organizations in the Santee River Basin and coastal drainages was integral to the success of this water-quality assessment. We thank those who served as members of our liaison committee.

Federal Agencies

National Park Service, Congaree Swamp National Monument
Natural Resources Conservation Service
U.S. Army Corps of Engineers
U.S. Fish and Wildlife Service
U.S. Forest Service

State Agencies

North Carolina Department of Environment and Natural Resources
Ocean and Coastal Resource Management
South Carolina Department of Health and Environmental Control
South Carolina Department of Natural Resources
South Carolina Forestry Commission
South Carolina Geological Survey
South Carolina Sea Grant Consortium
South Carolina Water Resources Research Institute

Local Agencies

Catawba Regional Planning Council
Charlotte-Mecklenburg Utility Department

Gaston County Cooperative Extension Service
Greenville County Soil and Water Conservation District
Western Piedmont Council of Governments

Universities

Columbia College
South Carolina State University
University of North Carolina at Charlotte
University of South Carolina

Other public and private organizations

Catawba Nation
Clean Water Fund of North Carolina
Duke Energy
National Audubon Society
South Carolina Electric and Gas Company
South Carolina Rural Water Association

We thank the following individuals for contributing to this effort.

Barbara Kleiss, Ted Campbell, and Sandra Cooper (USGS); Barry Beasley (South Carolina Department of Natural Resources); and Oscar Penegar (Environmental Advocate) for reviewing the report.

Jeannie Eidson (South Carolina Department of Environmental Control) for providing support and data for geographic information systems.

Larry Bradham (South Carolina Well Drillers Association) for assisting in locating wells for sampling.

Gary Taylor and Ralph Willoughby (South Carolina Geological Survey) for assisting in drilling wells.

The numerous property owners who allowed the USGS to install monitoring wells or sample existing wells on their property.

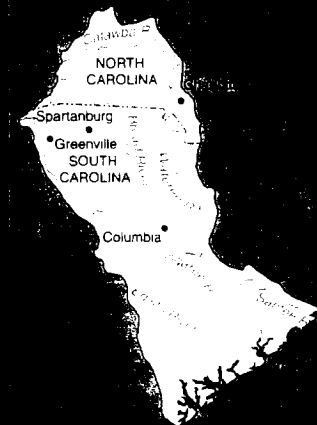
Ben Abercrombie, Boyce Blanks, Wade Bryant, Kristen Hein, Cliff Hupp, Robert Kelley, Donald Leary, Krystal Lynn, Brent Means, Larry Puckett, Whitney Stringfield, Robert Thorn, and Carlton Wood for providing invaluable technical and field support for the study.

Dick Christie (South Carolina Department of Natural Resources), Ginny Lindsey (North Carolina Clean Water Fund), and Charlie Zemp (South Carolina Department of Natural Resources) for providing guided tours of parts of the Santee Basin.

This report is dedicated to the memory of Don Leary; his quiet strength, hard work, and good humor helped make this study a success.

NAWQA

National Water-Quality Assessment (NAWQA) Program Santee River Basin and Coastal Drainages



Hughes and others—Water Quality in the Santee River Basin and Coastal Drainages
U.S. Geological Survey Circular 1206

ISBN 0-607-95422-1



9 780607 954227

R0024356

Water Quality in Southern Florida

Florida, 1996–98



U.S. Department of the Interior
U.S. Geological Survey

Circular 1207

R0024357

POINTS OF CONTACT AND ADDITIONAL INFORMATION

The companion Web site for NAWQA summary reports:

<http://water.usgs.gov/nawqa/>

Southern Florida contact and Web site:

USGS State Representative: Carl Goodwin
U.S. Geological Survey
Water Resources Division
227 N. Bronough St., Suite 3015
Tallahassee, FL 32301
state rep fl@usgs.gov
<http://water.usgs.gov/pubs/nawqa/>

National NAWQA Program:

Chief, NAWQA Program
U.S. Geological Survey
Water Resources Division
12201 Sunrise Valley Drive, M.S. 413
Reston, VA 20192
<http://water.usgs.gov/nawqa/>

Other NAWQA summary reports

River Basin Assessments

Albemarle-Pamlico Drainage Basin (Circular 1157)
Allegheny and Monongahela River Basins (Circular 1202)
Apalachicola-Chattahoochee-Flint River Basin (Circular 1164)
Central Arizona Basins (Circular 1213)
Central Columbia Plateau (Circular 1144)
Central Nebraska Basins (Circular 1163)
Connecticut, Housatonic and Thames River Basins (Circular 1155)
Eastern Iowa Basins (Circular 1210)
Georgia-Florida Coastal Plain (Circular 1151)
Hudson River Basin (Circular 1165)
Kanawha-New River Basin (Circular 1204)
Lake Erie-Lake Saint Clair Drainages (Circular 1203)
Las Vegas Valley Area and the Carson and Truckee River Basins (Circular 1170)
Lower Illinois River Basin (Circular 1209)
Long Island-New Jersey Coastal Drainages (Circular 1201)
Lower Susquehanna River Basin (Circular 1168)
Mississippi Embayment (Circular 1208)
Ozark Plateaus (Circular 1158)
Potomac River Basin (Circular 1166)

Puget Sound Basin (Circular 1216)
Red River of the North Basin (Circular 1169)
Rio Grande Valley (Circular 1162)
Sacramento River Basin (Circular 1215)
San Joaquin-Tulare Basins (Circular 1159)
Santee River Basin and Coastal Drainages (Circular 1206)
South-Central Texas (Circular 1212)
South Platte River Basin (Circular 1167)
Trinity River Basin (Circular 1171)
Upper Colorado River Basin (Circular 1214)
Upper Mississippi River Basin (Circular 1211)
Upper Snake River Basin (Circular 1160)
Upper Tennessee River Basin (Circular 1205)
Western Lake Michigan Drainages (Circular 1156)
White River Basin (Circular 1150)
Willamette Basin (Circular 1161)

National Assessments

The Quality of Our Nation's Waters—Nutrients and Pesticides (Circular 1225)

Front cover: Everglades National Park (photograph by Benjamin F. McPherson).

Back cover: Left, Southern Everglades C-111 agricultural basin, irrigation of crops; center, Northern Everglades, slough and tree islands; right, areal view of Miami (photographs by Benjamin F. McPherson).



United States Department of the Interior

U.S. GEOLOGICAL SURVEY
WATER RESOURCES DIVISION
227 N. BRONOUGH STREET, SUITE 3015
TALLAHASSEE, FLORIDA 32301

TELEPHONE (850) 942-9500
FAX: (850) 942-9521
<http://fl-water.usgs.gov>

January 18, 2001

Dear Sir or Madam:

We are pleased to enclose "Water Quality in Southern Florida, 1996-98," a publication from the U.S. Geological Survey Circular series which addresses water-quality issues of regional and national concern. This series is designed to present results on surface- and ground-water in major river basins across the Nation. The report presents significant findings on multiple nutrients and pesticides in surface and ground water from agricultural and urban areas. The conditions in south Florida were documented during 1996-98 for the Southern Florida study of the National Water Quality Assessment Program (NAWQA). Results indicate that the Everglades ecosystem has been altered by human activities.

We hope that the report will be useful to you and provide new insights on existing water-resource conditions in south Florida. If you or your organization would like additional copies of the report, or if you would like to discuss a particular aspect of the report or the NAWQA Program, please contact Ben McPherson, Chief of the Southern Florida NAWQA Program, by telephone at (813) 884-9336, by mail at U.S. Geological Survey, 4710 Eisenhower Blvd., Suite B-5, Tampa, FL, 33634, or by e-mail at bmcphers@usgs.gov.

We also encourage you to visit the NAWQA home page on the World Wide Web (<http://water.usgs.gov/nawqa>) to take advantage of other NAWQA products, including other publications, data sets, and maps of chemical use and occurrence. The site not only provides direct access to our national studies, but also to our individual assessments in more than 50 major river basins and aquifer systems across the Nation.

Thank you for your continued commitment to our Nation's water-resource needs.

Sincerely yours,

Carl R. Goodwin
USGS State Representative, Florida

Enclosure: "Water Quality in Southern Florida, Florida, 1996-98" by Benjamin F. McPherson, Ronald L. Miller, Kim H. Haag, and Anne Bradner. U.S. Geological Survey Circular 1207.

R0024359

Water Quality in Southern Florida

Florida, 1996–98

By Benjamin F. McPherson, Ronald L. Miller, Kim H. Haag, *and* Anne Bradner

U.S. GEOLOGICAL SURVEY CIRCULAR 1207

R0024360

U.S. DEPARTMENT OF THE INTERIOR
BRUCE BABBITT, Secretary

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

2000

Free on application to the
U.S. Geological Survey
Information Services
Box 25286 Federal Center
Denver, CO 80225

Or call: 1-888-ASK-USGS

Library of Congress Cataloging-in-Publications Data

Water quality in Southern Florida, Florida, 1996-98 / by Benjamin F. McPherson...[et al.].
p. cm. -- (U.S. Geological Survey Circular ; 1207)

Includes bibliographical references.

ISBN 0-607-95413-2 (alk. paper)

1. Water quality--Florida--Everglades. 2. Watersheds--Florida. I. McPherson, Benjamin F. II. Geological Survey (U.S.) III. Series.

TD224.F6 W3723 2000
363.739'42'0975939--dc21

00-049463

R0024361

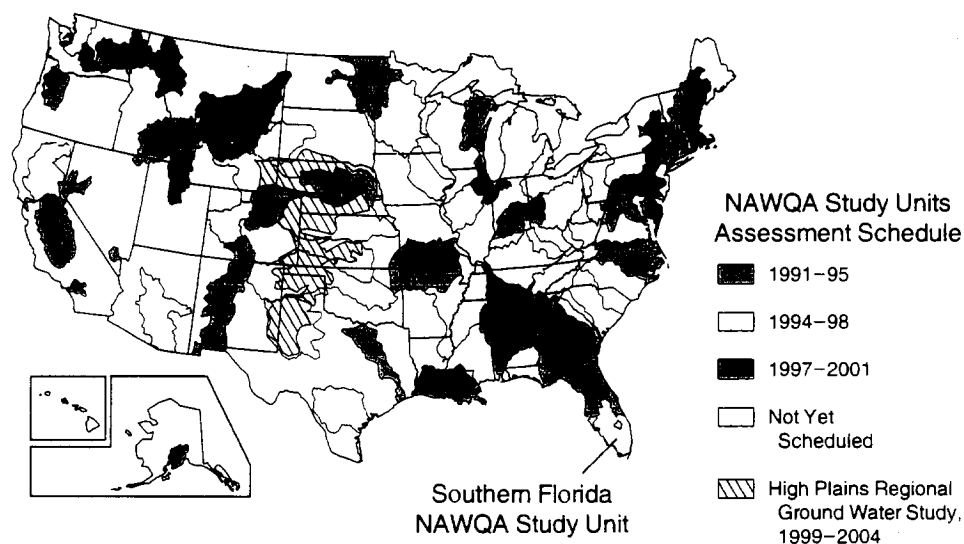
CONTENTS

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM.....	IV
SUMMARY OF MAJOR FINDINGS.....	1
Surface-Water Highlights	1
Ground-Water Highlights.....	2
INTRODUCTION TO THE SOUTHERN FLORIDA NAWQA STUDY UNIT	3
Rainfall	6
MAJOR FINDINGS	7
Nutrient enrichment is prevalent in surface water	7
NATIONAL PERSPECTIVE—Nutrient concentrations vary widely in southern Florida and the Nation.....	7
Nutrient concentrations in ground water are highly variable	9
Dissolved organic carbon concentrations are often high	9
Pesticides are present in most surface-water samples	10
NATIONAL PERSPECTIVE—Pesticide detections vary with land use in southern Florida and the Nation.....	11
Regional patterns of pesticides, VOCs and trace elements are evident in ground water	12
NATIONAL PERSPECTIVE—What combinations of pesticides occur most frequently in southern Florida and the Nation?.....	13
Pesticides, PCBs, other organics and trace elements have accumulated in bottom sediment and fish	15
Mercury is a contaminant in the Southern Florida Study Unit.....	16
Biological communities are influenced by water quality	18
NATIONAL PERSPECTIVE—Southern Florida aquatic communities in a national context	20
Exotic species are a threat to native biota	21
STUDY UNIT DESIGN.....	22
GLOSSARY	24
REFERENCES	25
APPENDIX A—WATER-QUALITY DATA FROM SOUTHERN FLORIDA IN A NATIONAL CONTEXT.....	27

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

THIS REPORT summarizes major findings about water quality in the southern Florida area studied by the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program between 1996 and 1998. Water quality is discussed in terms of local and regional issues and compared to conditions found in all 36 NAWQA study areas, called Study Units, assessed to date. Findings also are explained in the context of selected national benchmarks, such as those for drinking-water quality and the protection of aquatic organisms. The NAWQA Program was not intended to assess the quality of the Nation's drinking water, such as by monitoring water from household taps. Rather, the assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water monitoring programs. The comparisons made in this report to drinking-water standards and guidelines are only in the context of the available untreated resource. Finally, this report includes information about the status of aquatic communities and the condition of stream habitats as elements of a complete water-quality assessment.

Many topics covered in this report reflect the concerns of officials of State and Federal agencies, water-resource managers, and members of stakeholder groups who provided advice and input during the Southern Florida assessment. Basin residents who wish to know more about water quality in the areas where they live will find this report informative as well.



THE NAWQA PROGRAM seeks to improve scientific and public understanding of water quality in the Nation's major river basins and ground-water systems. Better understanding facilitates effective resource management, accurate identification of water-quality priorities, and successful development of strategies that protect and restore water quality. Guided by a nationally consistent study design and shaped by ongoing communication with local, State, and Federal agencies, NAWQA assessments support the investigation of local issues and trends while providing a firm foundation for understanding water quality at regional and national scales. The ability to integrate local and national scales of data collection and analysis is a unique feature of the USGS NAWQA Program.

The southern Florida area is one of 51 water-quality assessments initiated since 1991, when the U.S. Congress appropriated funds for the USGS to begin the NAWQA Program. As indicated on the map, 36 assessments have been completed, and 15 more assessments will conclude in 2001. Collectively, these assessments cover about one-half of the land area of the United States and include water resources that are available to more than 60 percent of the U.S. population.

SUMMARY OF MAJOR FINDINGS

Surface-Water Highlights

The environment in southern Florida is being degraded by human activities. Native biota have been reduced greatly in abundance and diversity by drainage, development, alteration of water flows, degradation of water quality, and by continuing invasions of exotic species. The Everglades ecosystem, which is adapted to water that has an extremely low phosphorus concentration, is being altered by agricultural activities that produce high levels of phosphorus in water. Nutrient loading in the major rivers is contributing to over-enrichment of Lake Okeechobee and estuaries such as Charlotte Harbor. Mercury has accumulated in Everglades game fish, and consumption of the fish poses a potential human health risk. Mercury has accumulated

in the Everglades food web because natural conditions and human influences enhance methylation of mercury to its organic form and because high atmospheric mercury deposition rates (among the highest in the Nation) sustain mercury methylation.

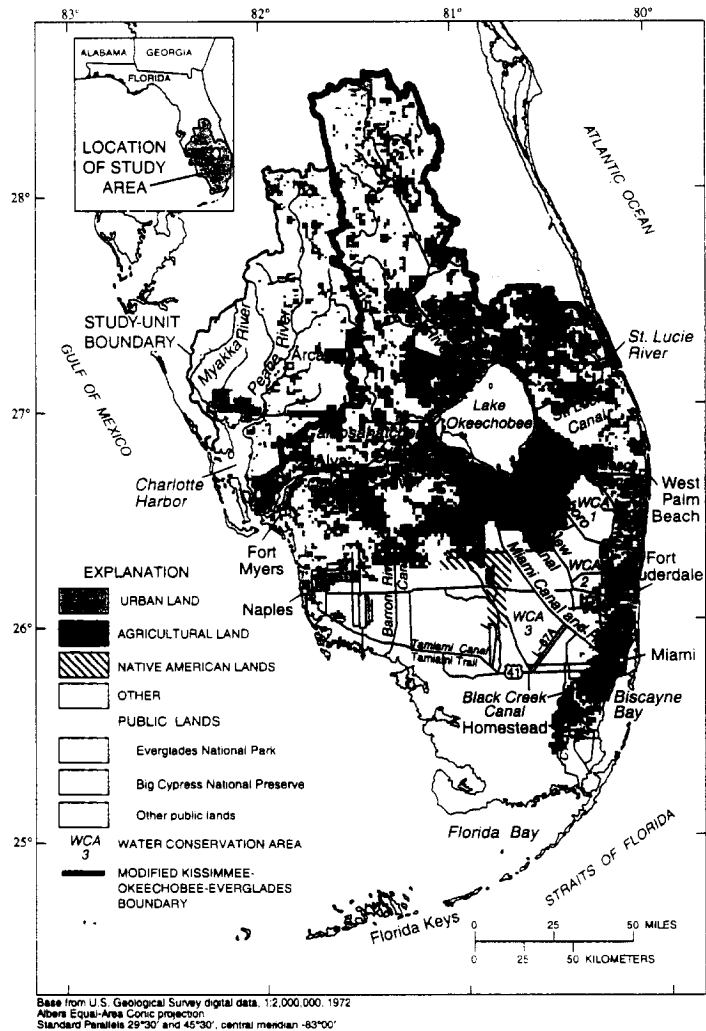
Federal and State agencies and environmental groups agree that parts of southern Florida should be restored to predevelopment conditions. Restoration will require massive changes in the water-management system to restore pre-development drainage patterns, improve water quality, and protect native biota.

Major findings on water quality and biology from this study:

- Concentrations of total phosphorus (TP) at the Southern Florida (SOFL) National Water-Quality Assessment (NAWQA) Program sites were above Everglades background levels and exceeded the U.S. Environmental Protection Agency's (USEPA) Everglades water-quality standard of 0.01 milligram per liter (mg/L). A major source of the high TP is fertilizer from agriculture.
- Concentrations of dissolved organic carbon (DOC) in southern Florida water were relatively high compared with those in other waters of the Nation. High DOC concentrations provide food for bacteria to grow, reduce light penetration in the water, and enhance transport and cycling of pesticides and trace elements such as mercury.
- Pesticides were detected in almost all SOFL samples. Most concentrations were below aquatic-life criteria; however, the criteria do not address potential effects of mixtures of

pesticides and their degradation products, which were common in the samples.

- Organochlorine pesticides, such as DDT and its degradation products, are still prevalent in bottom sediment and fish tissue at the SOFL sites, even though most uses of these compounds have been discontinued in recent decades. The mobilization of these pesticides by the reflooding of Everglades farm lands could lead to food-web contamination.
- Of 21 NAWQA basins nationwide, the Everglades has the second highest ratio of methylmercury to mercury in sediment. This enrichment in methylmercury enhances mercury uptake by the biota.
- The frequency of external anomalies (lesions, ulcers, and tumors) on fish collected at two SOFL agricultural canal sites was in the top 25 percent of



Urban, agricultural, Native American, public lands and other important features in the Southern Florida NAWQA Study Unit (McPherson and Halley, 1996).

144 NAWQA sites sampled nationwide. Anomalies can be indications that fish are stressed by contamination.

- Exotic animals and plants are a threat to native biota. Ten of the 54 exotic fish species established in the region were collected at the SOFL sites. Several herbicides used to control exotic plants were detected in surface water.

Major Influences on Surface Water and Ecology

- Drainage modifications and wetland destruction.
- Runoff from agricultural and urban areas.
- High concentrations of DOC and its effects on the transport of mercury and the attenuation of light.
- Deliberate or accidental release of exotic species.

Ground-Water Highlights

In much of the SOFL region, ground water in the surficial aquifers, such as the Biscayne aquifer, is of good quality and usually meets Federal and State drinking-water quality standards. Contaminants are usually in low concentrations, presumably because of rapid flushing and recharge as a result of high annual rainfall (about 55 inches) and shallow aquifers and porous limestone that allow the easy interchange of surface and ground water. However, because of the shallow aquifers and porous limestone, ground water is vulnerable to surface contamination and to saltwater intrusion.

Major findings on ground-water quality from this study include the following:

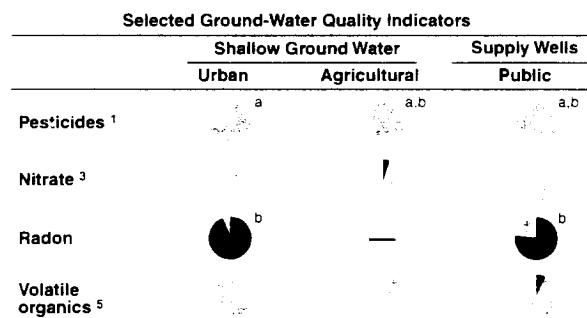
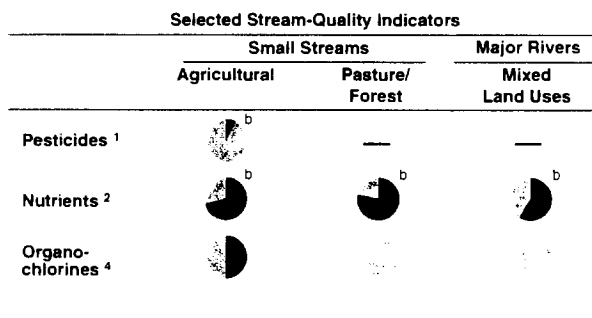
- Nitrate concentrations were below the drinking-water standard (10 mg/L) in 108 SOFL wells

(Biscayne and other surficial aquifers), except for two shallow wells in the unnamed surficial aquifer of the citrus area.

- Pesticides were detected in more than 85 percent of the SOFL wells and beneath every type of land use studied, but no concentrations exceeded any USEPA or State of Florida drinking-water standard.
- Pesticides detected in shallow ground water were associated with specific land uses. For example, the herbicides bromacil and norflurazon were detected almost exclusively in citrus areas. Metolachlor and simazine were common in mixed agricultural areas near the southern Everglades.
- Volatile organic carbon compounds (VOCs) commonly were detected in water from shallow and deep wells in the Biscayne aquifer. Concentrations of one industrial VOC, vinyl chloride, exceeded the USEPA maximum contaminant level (MCL) of 2 micrograms per liter ($\mu\text{g/L}$) for drinking water in two samples.
- Radon-222 radioactivity exceeded the proposed MCL (300 picocuries per liter (pCi/L)) in the majority of samples from the Biscayne aquifer, including untreated water from the public-supply wells.

Major Influences on Ground Water

- Porous, shallow limestone aquifers overlain by thin layers of sandy, permeable soils.
- Water-management practices involving canals, pumps, gates, locks, and saltwater-control structures.
- Agricultural and urban land-use practices and aquatic-weed control.



Percentage of samples with concentrations **equal to or greater than** a health-related national guideline for drinking water, aquatic life, or water-contact recreation; or above a national goal for preventing excess algal growth (^a Percentage is 1 or less and may not be clearly visible)

Percentage of samples with concentrations **less than** a health-related national guideline for drinking water, aquatic life, or water-contact recreation; or below a national goal for preventing excess algal growth

Percentage of samples with **no detection** (^b Percentage is 1 or less and may not be clearly visible)

Not assessed

¹ Insecticides, herbicides, and pesticide metabolites, sampled in water.

² Total phosphorus (TP) sampled in water. Aquatic-life guidelines based on USEPA Everglades standard of 0.01 mg/L TP for small Everglades streams and on USEPA national goal of 0.1 mg/L TP for major rivers.

³ Nitrate (as nitrogen), sampled in water.

⁴ Organochlorine compounds including DDT and PCBs, sampled in fish tissue.

⁵ Solvents, refrigerants, fumigants, and gasoline compounds, sampled in water.

INTRODUCTION TO THE SOUTHERN FLORIDA NAWQA STUDY UNIT

The Everglades are remarkable for the quantity of the water there, which continually bubbles and ebbs clear as crystal. —Alonso Sotomayor, 1513, p. 11. Nov., Dec. 1911.

Both these grand rivers (Miami and Neuse) possess a gaily beautiful clear water, also the Florida (fringed on either side by the wild swamps of hammocks, pine and palmetto) forming the eastern scenery of the South. —W. MacGonigal, 1842, p. 10. National Geographic Magazine, Vol. 10, p. 10.

In the mid-1800s southern Florida was a lush, subtropical wilderness of pine forest, hardwood hammocks, swamps, marshes, estuaries, and bays. Wetlands dominated the landscape. The region contained one of the largest wetlands in the continental United States, the Everglades, which was part of a larger watershed—the Kissimmee-Okeechobee-Everglades—which extended more than half the length of the Florida peninsula (fig. 1). Wetlands of the Everglades, Big Cypress Swamp, and Mangrove and Coastal Glades stretched continuously across much of the southern part of the peninsula south of Lake Okeechobee (fig. 1). To the north, much of the Flatwoods physiographic province also was wetlands; upland habitats were primarily on the narrow Lake Wales and Atlantic Coastal Ridges.

Freshwater in the Everglades and other wetlands generally moved as

sheetflow in marshes, sloughs, and cypress strands. Numerous small streams and rivers near the coast, such as the Miami River, drained into mangrove forests and tidal waters and provided the freshwater that sustained the highly productive and abundant coastal fisheries around the southern end of the peninsula (McIvor and others, 1994).

The wetlands of southern Florida made much of the region inhospitable for human habitation. Settlers and developers in the late 1800s and early 1900s began to drain the wetlands for commercial and safety

reasons. Loss of lives as a result of hurricane flooding in the 1920s accelerated drainage projects. Today, many of the region's original wetlands have been drained. Water in the region is now intensively managed, with more than 1,400 miles of primary canals and more

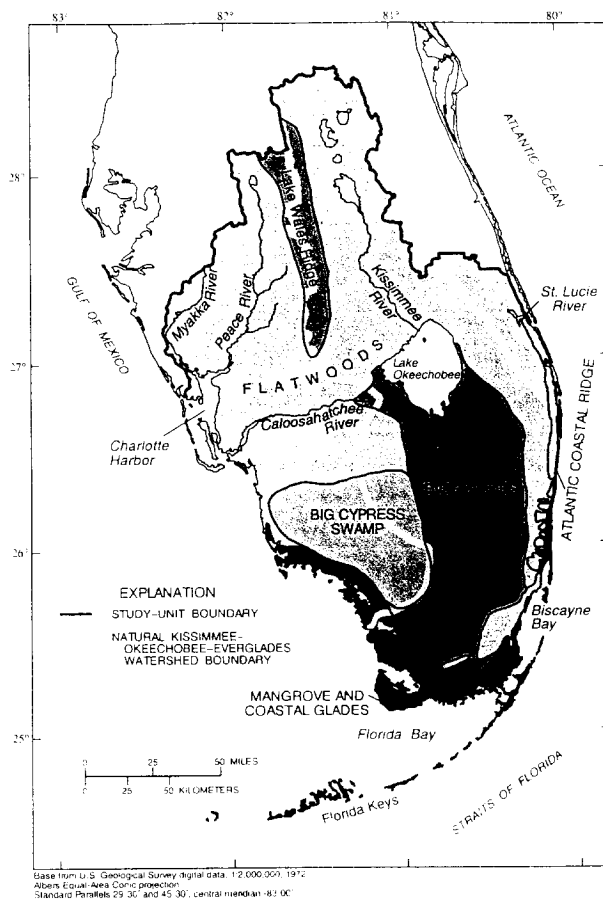


Figure 1. Physiographic provinces of southern Florida. (Modified from Davis, 1943; Parker and others, 1955.)

The Southern Florida (SOFL) National Water-Quality Assessment study encompasses about 19,500 square miles. It is part of a regional ecosystem that includes coastal waters between Charlotte Harbor on the Gulf of Mexico and the St. Lucie River on the Atlantic Ocean and the lands that drain into these waters. The elevation in the study area ranges from about 300 feet above sea level to sea level along the coast. It includes a large and rapidly growing urban population (about 5 million people) along the Atlantic coast and a less rapidly growing population (over a million people) along the gulf coast, areas of intense agricultural development around Lake Okeechobee and along the southeastern edge of the Everglades, and vast regions of wetlands, including the Everglades National Park (ENP), Big Cypress National Preserve, and other parks, preserves, and conservation areas that are mostly in public ownership. Ground water from the shallow, highly porous Biscayne aquifer is the source of most of the drinking water for the densely populated southeast coast.



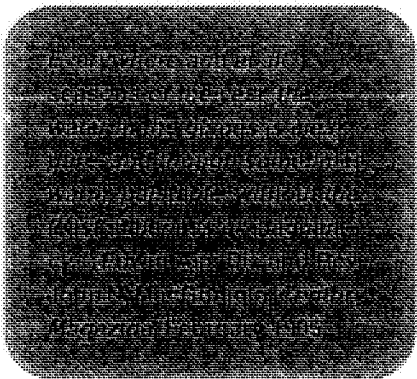
Complex canal system in northern Everglades.



Northern Big Cypress Swamp.

than 100 water-control structures. The larger rivers, such as the Kissimmee and Caloosahatchee Rivers, have been canalized and controlled to enhance their ability to move water. About half the Everglades have been lost to drainage and development since the early 1900s; the remaining Everglades, included in the Everglades National Park (ENP), conservation areas, and the Loxahatchee National Wildlife Refuge, are protected from physical destruction, but it has been degraded by altered quantity, quality, and timing of freshwater inflows.

Drainage and development of wetlands have adversely affected water quality and ecology throughout southern Florida. Water



pumped into canals from agricultural lands commonly has high concentrations of nutrients and pesticides. The high nutrient concentrations and loads entering Lake Okeechobee and the Everglades from farms and cattle lands have degraded water quality. Phosphorus concentrations in Lake Okeechobee have increased two and one-half times since the 1970s, and massive algal blooms have become more frequent and persistent. The increased nutrient loading to the Everglades is stressing native vegetative communities. Sawgrass, which is adapted to a low-nutrient environment, is being replaced by cattails in parts of the northern Everglades where nutrient loading has been excessive. Drainage and development also has resulted in loss of peat soils, contamination by pesticides, saltwater intrusion into aquifers near the coast, mercury buildup in the biota, fragmentation of landscape, loss of wetland functions, widespread invasion by exotic species, increased algal blooming, seagrass die-off, and declines in fishing resources in coastal waters.

An abundant and uncontaminated supply of freshwater was a primary environmental characteristic of southern Florida in predevelopment times. Increased human population and activity have brought not only increased need for water but also a decrease in water supply and deterioration in water quality. These changes in the hydrologic system, wrought by growth and development, are thought to be the major causes of the substantial declines in the health of the remaining natural ecosystem.

A consensus has begun to emerge among environmental groups and Federal and State agencies that southern Florida, and particularly the Everglades, should be restored to the extent possible to the predevelopment ecosystem. A first and primary step in this undertaking is the restoration of predevelopment hydrologic conditions to the remaining natural system. Plans are to change the manmade water-conveyance system and restore the natural hydrologic cycle of the predevelopment Everglades as a means of contributing to overall ecosystem restoration.

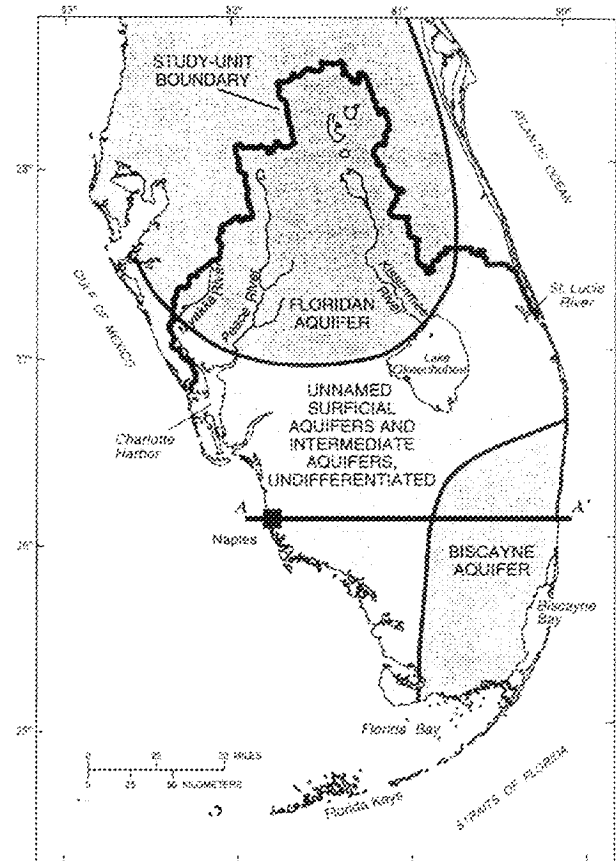
Aquifers and Water Use

Southern Florida, which is underlain by shallow marine carbonate sediments to depths of 20,000 feet, contains three major aquifer systems: the Floridan, the intermediate, and the surficial aquifer systems (figs. 2a,b). The confined Floridan aquifer system is the principal source of water for human use in the northern part of the south Florida area, but water from this aquifer is too mineralized for most uses in the southern part of the area. The semiconfining layers of the intermediate aquifer system, which overlies the Floridan, serves as the confining unit for the Floridan aquifer, a source of freshwater for public supply along the east coast. The surficial aquifer system includes the highly permeable Biscayne aquifer, which is the principal source of potable water for the more than 5 million people in southeastern Florida. The Biscayne aquifer has been designated as a sole source drinking water supply by the USFPA.

Most of the potable water supply in southern Florida is withdrawn from shallow aquifers, generally from wells less than 250 feet deep. Ground water supplied 94 percent of 2.2 million gallons per day (MGD) of the water used by most of the 1.8 million people in the SDF. South Florida 1990 Water Use Report (in 1991, 2,735 MGD) (USFPA, 1991). The USFPA also provides water for agricultural irrigation. Richard J. Marzullo, USFPA, "Groundwater Resources of Southern Florida," 1990.



Agricultural irrigation, eastern Everglades area.



Base from U.S. Geological Survey digital data, 1:2,000,000, 1972
Albers Equal-Area Conic projection
Standard Parallels 29° 30' and 45° 30', central meridian -85° 00'

Figure 2a. The three main aquifer systems of southern Florida.

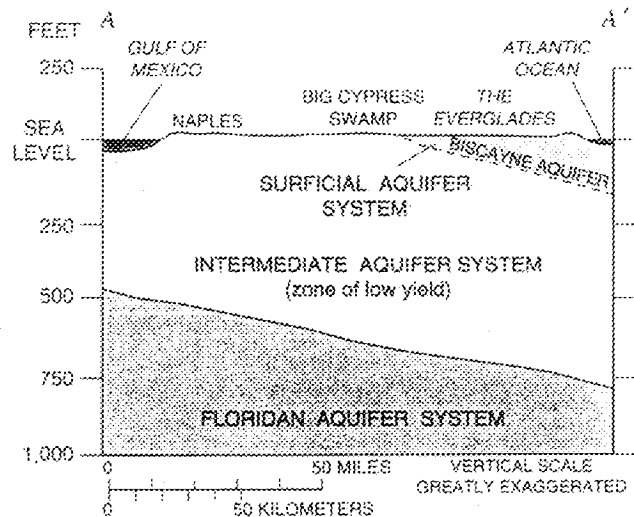


Figure 2b. Generalized subsurface section A-A' showing aquifers of southern Florida (Klein and others, 1975).

Rainfall

Annual rainfall in southern Florida ranges from about 40 to 65 inches. The east coast usually receives the greatest amount of rainfall, whereas the Florida Keys

and areas near Lake Okeechobee and Charlotte Harbor usually receive the least. More than half the rainfall occurs from June through September and is associated with thunderstorms and tropical cyclones. Rainfall during the

remainder of the year usually is the result of large frontal systems and is broadly distributed rather than localized. April and May typically have the least rainfall. Annual and seasonal rainfalls vary from year to year (fig. 3).

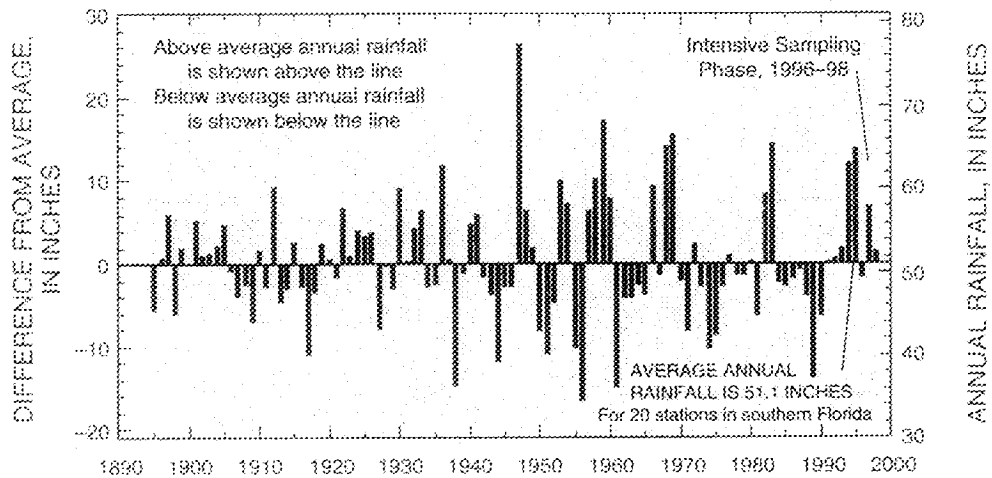


Figure 3. Annual rainfall was above average during the 1996-98 sampling period. (Data from the National Oceanic and Atmospheric Administration's National Climate Center.)

HOW IS THE SOUTHERN FLORIDA NAWQA DIFFERENT FROM OTHER NAWQA STUDY UNITS?

- Drainage basins are poorly defined.
- Surface and ground water are closely connected.
- Sheetflow is common through the "River of Grass" and other wetlands.
- Surface-water flow in canals and rivers is highly managed and regulated.
- Organic soils (peats) are abundant, but much has been lost to oxidation.
- The farming season is in winter.
- Coastal meteorological effects often dominate.
- Tropical storms are common.
- Nutrient concentrations are naturally low in the Everglades and other pristine wetlands.
- Dissolved organic carbon concentrations are high.
- Water color is dark in some rivers and wetlands.
- There are extensive subtropical wetlands and public lands, including four national parks, preserves, or refuges.
- Many exotic species thrive in the subtropical climate.

MAJOR FINDINGS

Nutrient enrichment is prevalent in surface water

Water quality has been degraded in large parts of southern Florida by human activities that result in high nutrient concentrations and over-enrichment. Nutrient concentrations at the SOFL NAWQA sites are elevated when compared with Everglades background concentrations (see box below). The high nutrient concentrations, primarily from agricultural runoff, have contributed to overenrichment of surface water, including Lake Okeechobee, estuaries such as Charlotte Harbor, and the northern Everglades. The high phosphorus concentrations in agricultural runoff entering the northern Everglades are a significant cause of ecosystem degradation.

The USEPA recently (May 26, 1999) approved a new water-quality standard of 0.01 mg/L or less for phosphorus in the Miccosukee Federal Indian reservation lands of the Everglades. The State of Florida is reviewing additional scientific information and plans to adopt a numerical phosphorus standard for other parts of the Everglades.

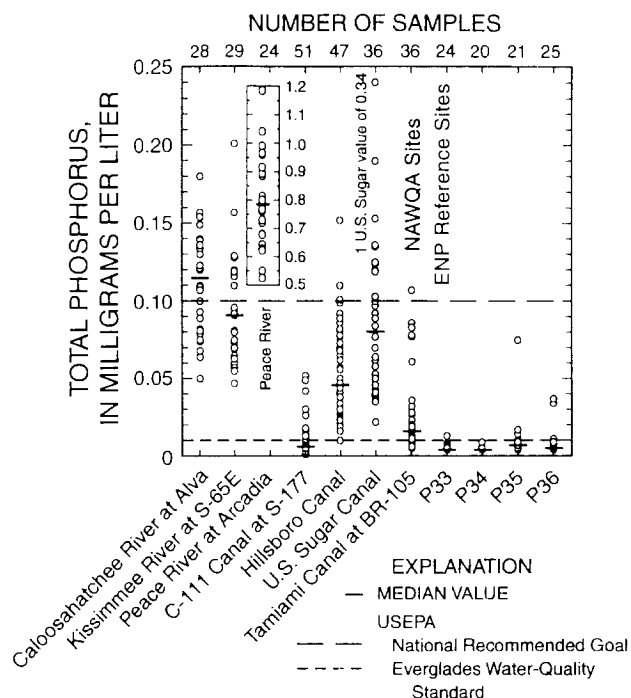


Figure 4. Concentrations of total phosphorus (TP) at SOFL NAWQA sites during 1996–98 frequently exceeded the USEPA recommended goal of 0.1 mg/L and Everglades standard of 0.01 mg/L. Concentrations of TP at Everglades National Park reference sites were near or below 0.01 mg/L.

NUTRIENT CONCENTRATIONS VARY WIDELY IN SOUTHERN FLORIDA AND THE NATION

Concentrations of nutrients in surface water at the SOFL sites vary widely compared with national NAWQA median concentrations. Generally, the high nutrient concentrations in southern Florida are at canal and river sites north of the Tamiami Trail and are associated with agricultural activities. Concentrations of phosphorus in surface water of the more pristine Everglades (south of the Tamiami Trail) are more than an order of magnitude lower than national background concentrations. The Everglades ecosystem is adapted to water that has extremely low phosphorus concentrations and is being adversely affected by agricultural runoff with high concentrations of phosphorus. Concentrations of phosphorus in southern Florida are highest in the Peace River (fig. 4) and are well above national median concentrations. The high concentrations in the Peace River are associated with natural sources (phosphatic deposits) as well as mining and agricultural activities in the basin.

Concentrations of nutrients in shallow ground water at SOFL wells generally are lower than or comparable with the national background concentrations. Median con-

centrations of nitrate in the southern Florida wells are one-fourth the national background concentration and median concentrations of orthophosphate are comparable to the national background concentration. Concentrations of nutrients at the SOFL sites are plotted against corresponding national nutrient ranges for different land uses in Appendix A.

Nutrient concentrations of the national NAWQA and the SOFL NAWQA study areas, national background, and Everglades background sites in mg/L [SW, surface water; GW, shallow ground water; <, less than; --, no data]

Total phosphorus (SW)	0.12	0.006–0.79	0.1	<0.004
Total nitrogen (SW)	1.3	0.7–2.6	1.0	1.1
Nitrite plus nitrate (SW)	0.71	0.006–0.59	0.6	0.005
Ammonia (SW)	0.046	0.017–0.25	0.1	<0.015
Ammonia (GW)	0.02	0.24–0.438	--	--
Nitrite plus nitrate (GW)	1.85	<0.05–0.44	2.0	--
Orthophosphate (GW)	0.11	0.013–0.028	0.02	--

¹U.S. Geological Survey, 1999.

²Flow-weighted concentrations.

³Medians for routinely sampled (BFS) sites (SW) and for land-use categories (GW).

⁴Everglades National Park, site P-34, medians; data from South Florida Water Management District (1992).



Ground-water sampling, eastern Everglades C-111 agricultural basin.

All routinely sampled SOFL sites (BFS sites, see Glossary) had phosphorus concentrations that exceeded the USEPA Everglades standard of 0.01 mg/L, but the two southern sites, Canal C-111 and Tamiami Canal at bridge 105, had median concentrations near the 0.01-mg/L Everglades standard (fig. 4). The ENP reference sites had median phosphorus concentrations below 0.01 mg/L and are characteristic of pristine Everglades water.

Seasonal changes in total phosphorus concentrations often are related to changes in water levels

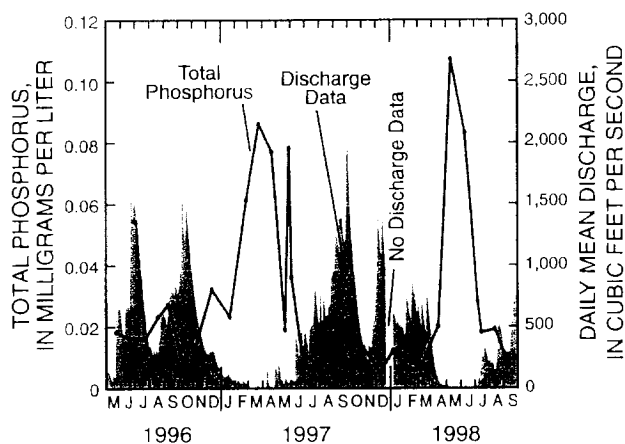


Figure 5. Total phosphorus concentrations increased as discharge and water levels declined at Tamiami Canal at Bridge 105, Big Cypress National Preserve.

and flows. Generally, concentrations were below 0.02 milligram per liter (mg/L) at the Big Cypress Swamp reference site (Br 105) during 1996–98, but increased to more than 0.10 mg/L as water flows and levels declined during the dry season (fig. 5). The increase in phosphorus concentrations occurs when fish, wading birds, and other aquatic organisms congregate in ponded waters and their wastes contribute nutrients to the remaining inundated areas. The relatively greater ground-water contributions to surface water during the dry season also may increase nutrient concentrations. The background marsh sites in the ENP had very low concentrations of phosphorus (less than 0.01 mg/L) throughout the

1996–98 sampling period, and effects of seasonal low water levels on nutrients were not evident.

The dominant source of phosphorus loading in southern Florida is fertilizer (fig. 6). Manure and atmospheric sources are also important in some subbasins. Annual phosphorus loads estimated for selected canals and rivers were highest in the Peace River and lowest in the eastern Big Cypress Swamp. Much more phosphorus has been transported seaward from the northern Everglades and Okeechobee basins by the Caloosahatchee River, St. Lucie Canal, and major canals of Palm Beach County than is transported seaward in the southern Everglades (Haag and others, 1996).

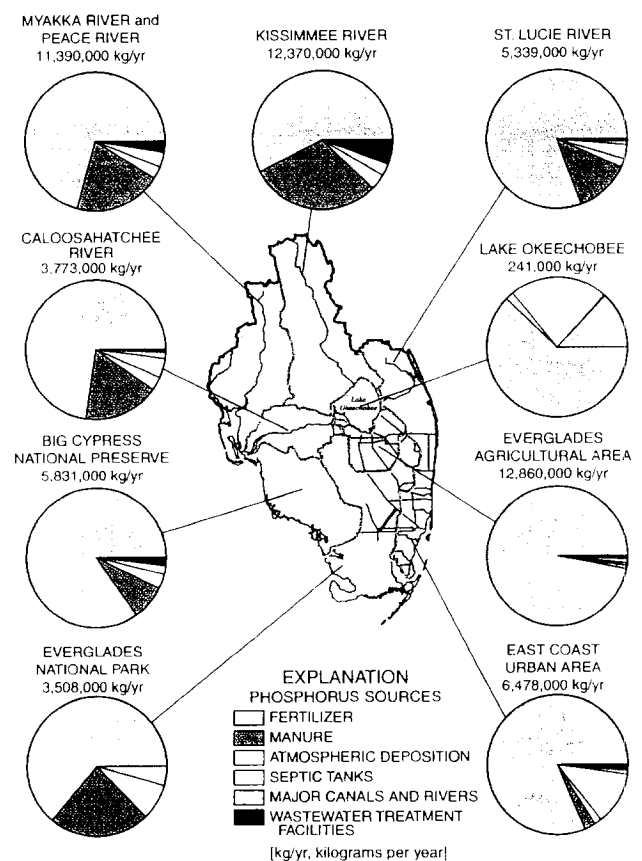


Figure 6. Estimated phosphorus loading from point and nonpoint sources in surface-water basins in southern Florida (Haag and others, 1996).

Nutrient concentrations in ground water are highly variable

High nitrate concentrations (greater than 10 mg/L) in drinking water can cause a life-threatening illness in infants known as "blue-baby syndrome." Nitrate concentrations in ground water were low (commonly not detected) in most of the 108 wells sampled in the SOFL Study Unit, but a few occasionally were elevated. In areas with agricultural land use. Wells located in or near the mixed-agricultural land-use area (including public-supply wells) tended to yield water with detectable concentrations of nitrate, but no nitrate concentrations exceeded the drinking-water standard of 10 mg/L. More than half of the wells located in citrus groves yielded water with no detectable nitrate concentrations, but two wells had nitrate concentrations above the drinking-water standard. With a few exceptions, shallow Biscayne aquifer wells in the urban land-use area contained water with relatively low concentrations of nitrate.

Ammonia concentrations in ground water in the SOFL Study Unit were relatively high compared with concentrations in other NAWQA Study Units across the Nation. The median ammonia concentration (0.396 mg/L) in the

SOFL citrus land-use survey was the highest of 47 NAWQA agricultural surveys nationwide. Median concentrations of ammonia in the SOFL urban and study-unit survey (public water supply) ranked second and fifth highest, respectively, in these categories nationally.

Dissolved phosphorus concentrations in SOFL wells ranged from 0.001 to 0.79 mg/L; most land-use areas had median concentrations less than 0.01 mg/L (table 1). Concentrations above 0.05 mg/L occurred in some wells in all land-use and public-supply surveys, but highest concentrations generally were in the deeper public-supply wells of the Biscayne aquifer and in the citrus land-use area. The source of the relatively high concentrations of phosphorus in ground water may be fertilizers or naturally occurring phosphatic materials associated with silt and clays.

Dissolved organic carbon concentrations are commonly high

Dissolved organic carbon (DOC) originates from natural sources, such as living and decaying plants, and from human sources. DOC represents about half the dissolved organic matter (DOM) in natural waters (Hem, 1985). In southern Florida, DOC concentrations commonly are high compared with

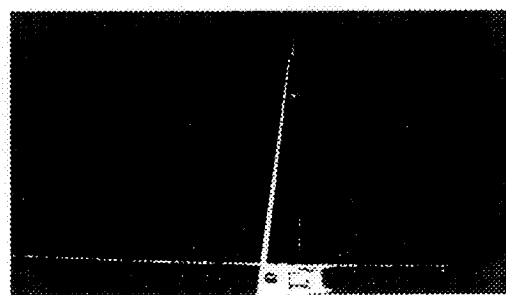
other natural waters in the Nation and may constitute, as DOM, a significant fraction of the dissolved solids (table 2).

Concentrations of DOC ranged from 4.8 to 52.0 mg/L at the SOFL surface-water sites, and from 0.6 to 80 mg/L at the ground-water sites (table 4). The highest DOC concentrations (median of 34.0 mg/L) occurred at Hillsboro Canal at S-6 in the northern Everglades down-gradient from the Everglades Agricultural Area, which has highly organic muck soils that are thought to be a source of DOC. In contrast, the Big Cypress Swamp reference site at Tamiami Trail Bridge 105 had the lowest concentrations (median of 9.5 mg/L) of the routinely sampled sites. This site is characterized by natural swamp vegetation and thin carbonate soils and rock. Concentrations of DOC at sites along the Tamiami Trail (1996-97) ranged from 4.8 to 26.9 mg/L (table 2), and tended to be low in the central and eastern Big Cypress Swamp and higher in the western Big Cypress and to the east in the Everglades (Miller and others, 1999).

The amount of DOC in water is significant. It can (1) contribute to water color, which absorbs sunlight and reduces the amount of light available for use by submerged aquatic plants and phytoplankton,

Table 1. Dissolved phosphorus concentrations varied widely in shallow wells in southern Florida during 1997-98. (Different detection levels were reported for different survey types during the period)
(B, Biscayne aquifer; S, unnamed surficial aquifer; <, less than)

Survey type	n	Median (mg/L)	95th Percentile (mg/L)
Urban/residential (B)	32	<0.01-0.30	<0.01
Urban background (B)	3	<0.01-0.07	<0.01
Citrus (S)	31	<0.01-0.79	0.023
Citrus background (S)	5	<0.004-0.036	<0.01
Mixed agricultural (Canal C-111) (B)	7	0.001-0.019	0.003
Public supply (B)	30	<0.01-0.34	<0.01



Citrus groves north of Big Cypress Swamp.

Table 2. Concentrations of dissolved organic carbon, in milligrams per liter, at surface- and ground-water sites in southern Florida, 1996-98
 [Dissolved organic carbon (DOC) in natural water typically is 45 to 55 percent of the dissolved organic matter (DOM; Robert Wershaw and Larry Leenheer, oral commun., 1997) and averages about 50 percent of the DOM that also includes other elements such as oxygen, hydrogen, nitrogen, phosphorus; --, no data]

	Surface Water	Ground Water	Number of DOC samples	DOC as a percentage of dissolved solids
Kissimmee River at S-65E	13.0-22.0	16.0	29	20-39
Peace River Arcadia	8.6-26.0	15.0	30	6-36
Caloosahatchee River at Alva	13.0-22.0	16.0	27	5-15
Hillsboro Canal at S-6	15.0-52.0	34.0	48	7-13
U.S. Sugar	17.0-30.0	20.0	37	11-16
Tamiami Canal at Bridge 105	7.1-14.0	9.5	35	5-18
Canal C-111 at S-177	5.0-18.0	6.6	51	4-9
All 7 Basic Fixed Sites	5.0-52.0	16.0	257	4-39
Tamiami Trail, SOFL data (1996-97)	4.8-26.9	13.5	43	--
	Ground Water			
Urban/residential land-use wells	1.9-36	11.5	32	--
Citrus wells	2.7-80	24	29	--
Public supply wells	0.6-22	9.4	30	--

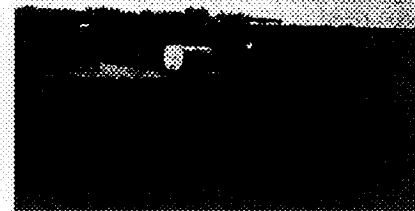
(2) serve as a source of carbon for bacterial growth; (3) form complexes with trace elements, such as mercury, and make them more soluble and mobile in water (Reddy and others, 1999); (4) reduce bio-availability of nonionic organic compounds through sorption, entrapment, or sequestering the compounds (Nowell and others, 1999, p. 296); (5) increase the solubilities of relatively insoluble compounds, such as p,p'-DDT, PCBs, and lindane (Chiou and others, 1986); and (6) react with chemicals used to disinfect public water supplies and produce undesirable by-products, such as chloroform and other trihalomethanes.

Naturally occurring dissolved organic matter (DOM) causes the tea-colored water in many of Florida's rivers and swamps. Water color can have significant ecological effects by decreasing available light in the water column and reducing algal and aquatic plant productivity and growth. In the Peace River and upper Charlotte Harbor, for example, color can contribute up to about half of the total light attenuation in the water column, and this high attenuation limits algal phytoplankton and seagrass growth (McPherson and Miller, 1987). The range in color for southern Florida water, including the NAWQA sites, is shown in table 3.

that may have harmful effects on human health.

Pesticides are present in most surface-water samples

Pesticides are widely used in southern Florida to control insects, fungi, weeds, and other undesirable organisms. These compounds vary in their toxicity, persistence, and transport. Some of the more persistent pesticides, such as DDT, chlor-dane, dieldrin, and aldrin, have been discontinued for use in Florida, but their residues persist in the environment. Although pesticides usually are applied to specific areas and directed at specific organisms, these compounds often become widely



Pesticide application in vegetable farming area.

distributed and pose potential hazards to nontarget organisms.

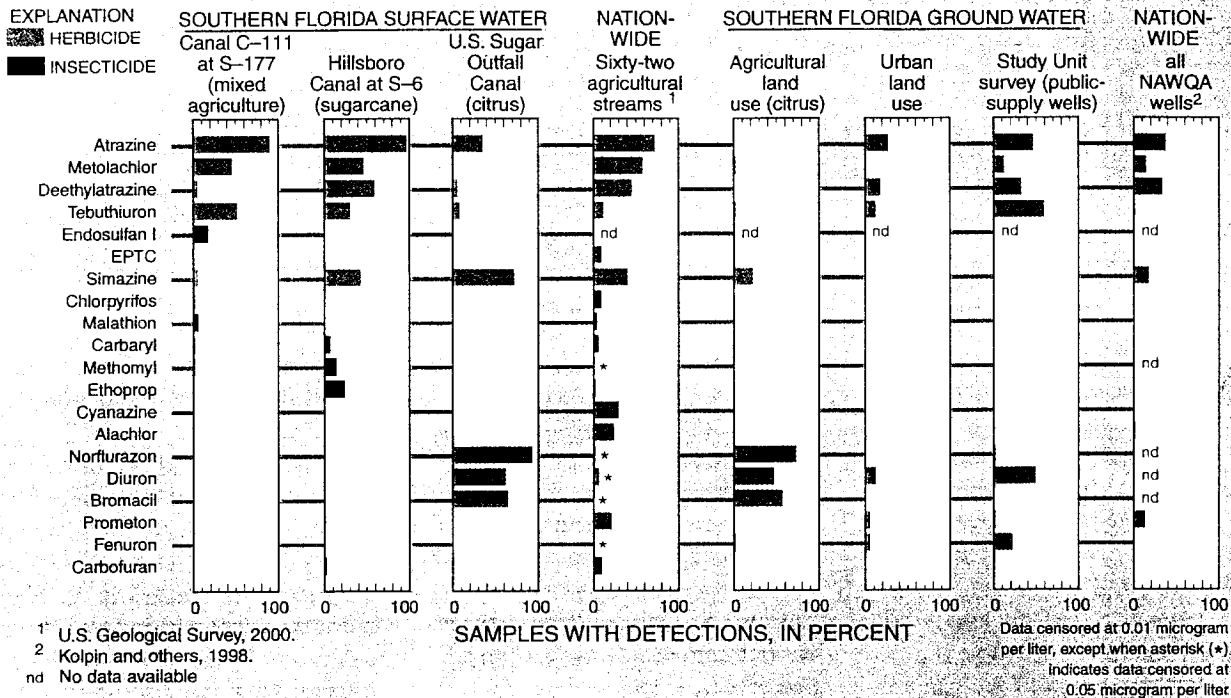
Intensive pesticide sampling (up to weekly sampling) was conducted at three SOFL sites (IFS sites, see Glossary) that represented three different agricultural land uses — mixed vegetable crops (C-111 basin), sugarcane (S-6), and citrus (U.S. Sugar). Pesticides were detected in all but one sample from the three intensive-sampling sites during 1996-98. The most frequently detected pesticides included those with highest annual application rates, such as the herbicides atrazine, bromacil, simazine, 2,4-D, and diuron. Atrazine, the most frequently detected pesticide overall, was detected in about 90 percent of all samples. The other most frequently detected pesticides overall were metachlor, simazine, tebuthiuron, norflurazon, bromacil, and diuron.

Table 3. Water color for southern Florida surface-water sites, 1913-1999 (Color is reported in platinum-cobalt units. Data from U.S. Geological Survey data bases)

Site	Color Range	Mean	SD	Period
Kissimmee River, S-65E	25-240	85	51	1959-75
Peace River, Arcadia	0-320	82.5	414	1930-99
Caloosahatchee River at S-79	5-500	62.5	144	1953-88
Hillsboro Canal at S-6	30-560	120	89	1945-84
Tamiami Canal at Bridge 105	10-100	30	56	1967-99
Canal, C-111 at S-18-C	0-40	5	40	1970-83
Everglades P-33	10-120	45	82	1959-85
878 southern Florida sites in Miami USGS data base	0-640	55	11,457	1913-99



PESTICIDE DETECTIONS VARY WITH LAND USE IN SOUTHERN FLORIDA AND THE NATION



Different patterns of pesticide detections were evident in surface and ground water of the different land-use areas of southern Florida. For example, norflurazon and bromacil frequently were detected in citrus land-use areas but seldom detected in other agricultural and urban areas. Detections of prometon and fenuron were relatively high in the public-supply wells of the Biscayne aquifer but were much lower in other areas. Many of the same pesticides were detected in surface and ground water nationwide, but a few such as alachlor, cyanazine, and carbofuran were mostly absent in southern Florida. The most commonly detected pesticides, such as atrazine, simazine, and metolachlor, were more common in surface water than in ground water in both southern Florida and the Nation. Concentrations of selected herbicides at SOFL sites are plotted against corresponding national herbicide ranges for different land uses in Appendix A.

Concentrations of pesticides in water were seasonal and related to land use. Concentrations of atrazine peaked at all three sites in late winter and spring (fig. 7). Concentrations were highest at the S-6 site, where some samples had atrazine concentrations that exceeded the Canadian aquatic-life criterion of 2 µg/L and the USEPA MCL of 3 µg/L. Concentrations at the other two sites were significantly lower, with maximum values less than 1 µg/L.

A pesticide of particular concern, endosulfan, was detected mainly at the Canal C-111 site (fig. 8). Endosulfan was detected by the South Florida Water Management

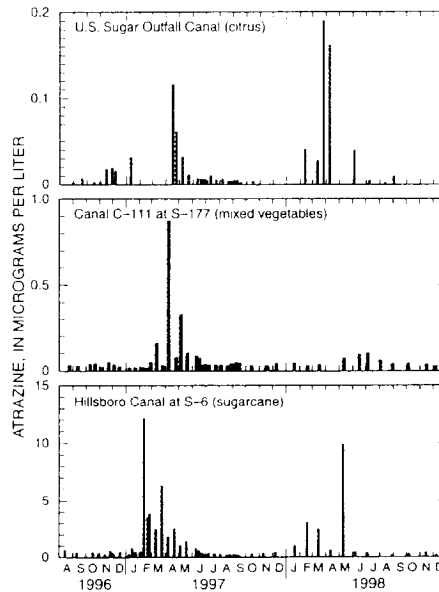


Figure 7. Concentrations of atrazine at the Intensive Fixed Sites, August 1996–December 1998, showing similar seasonal occurrence patterns but different concentrations.

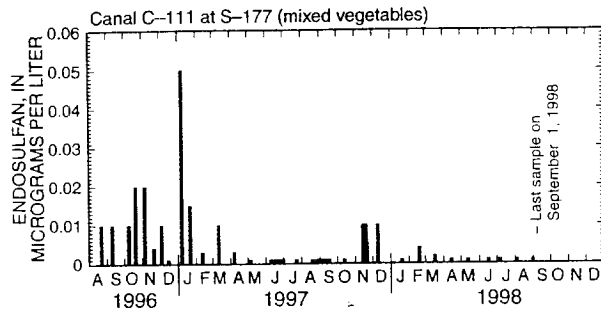


Figure 8. Concentrations of endosulfan at Canal C-111 at S-177, August 1996-September 1998.



Residential land near Fort Lauderdale.

District (SFWMD) over a number of years in the C-111 basin at levels considered to be a threat to aquatic life in the basin and in nearby Florida Bay (Miles and Pfeuffer, 1997). During the intensive sampling period of the SOFL study (1996-98), endosulfan concentrations were 0.05 $\mu\text{g/L}$ or less (fig. 8), which is just below the Florida Department of Environmental Protection criterion (0.056 mg/L) for Class III (recreation, propagation and maintenance of a healthy, well-balanced population of fish and wildlife) freshwater. Detections of endosulfan were frequent in 1996 but became less frequent in the following 2 years, as the use of this pesticide was discouraged and an alternate, imidacloprid, was introduced.

Mixtures of pesticides were common in samples from SOFL and other national NAWQA sites

The NAWQA Program uses Federal drinking-water standards and guidelines to assess the quality of drinking water in potential surface- or ground-water sources prior to treatment and distribution. This complements many ongoing Federal, State, and local monitoring programs that assess drinking water after treatment and distribution.

(see figure on page 13). The effects of pesticide mixtures on biota or humans are not included in criteria, which are based on the results of single-species, single-chemical toxicity tests conducted in the laboratory. As a result, analyses of individual pesticides may underestimate potential adverse effects of contaminants on biota (Nowell and others, 1999).

Regional patterns of pesticides, VOCs, and trace elements are evident in ground water

Pesticides were detected in ground water from more than 85 percent of the 108 SOFL wells and beneath every type of land use studied. No pesticide concentration

exceeded USEPA or State of Florida drinking-water standards or health advisories.

VOCs commonly were detected in water from shallow, residential land-use wells and deeper public-supply (study-unit survey) wells in the Biscayne aquifer (fig. 9). Vinyl chloride, trichloroethylene, *tetra*-chloroethylene, *cis*-1,2-dichloroethylene, and methyl *tert*-butyl ether (MTBE) were detected more commonly in the older residential and industrial areas and in public-supply wells near mixed agricultural lands. Toluene, *p*-isopropyltoluene, and 1,2,4-trimethyl-benzene commonly were detected in the newer residential areas and in public-supply wells near mixed agricultural lands. Two samples from public-supply wells in more industrial-

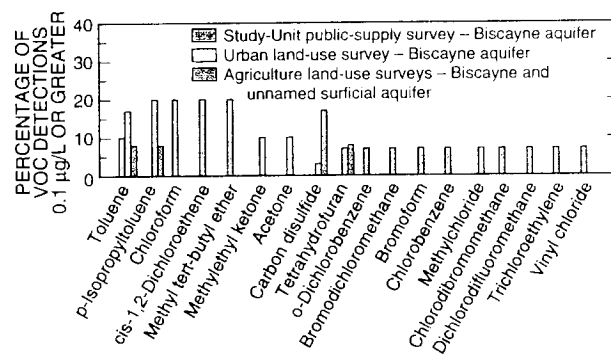


Figure 9. Percentages of common volatile organic carbons (VOCs) detected in water from shallow urban (residential) and agricultural wells and from deeper public-supply wells (see Study Unit Design on page 22 for locations).

ized areas had vinyl chloride concentrations (4.68 and 3.18 µg/L) slightly above the USEPA MCL of 2 µg/L.

Urban and agricultural activities are sources of trace-element contamination in ground water. Arsenic and copper are used as fungicides in citrus groves. Arsenic (in the herbicide monosodium meth-ane arsonate) is used in turf-grass maintenance on golf courses, and concentrations of arsenic in shallow ground water are sometimes elevated (Swancar, 1996).

Arsenic has been implicated as causing several cancers. Because of this health concern, the USEPA is considering lowering the MCL for arsenic from 50 to about 5 µg/L. Concentrations of arsenic in shallow ground water exceeded 5 µg/L in some of the urban and citrus land-use SOFL wells. Concentrations of copper in the SOFL ground-water samples reached 19 µg/L, which is well below the drinking-water MCL of 1,300 µg/L.

Uranium and radon-222, two naturally occurring radioactive

elements that are potential carcinogens, exceeded drinking-water standards in some shallow ground water in the SOFL Study Unit.

Uranium exceeded the MCL in 5 of 116 samples. Radon-222, a gaseous radionuclide that, when released to the air and inhaled is a significant cause of lung cancer, exceeded the proposed MCL of 300 picocuries per liter (pCi/L) in more than 75 percent of the samples from the Biscayne aquifer.

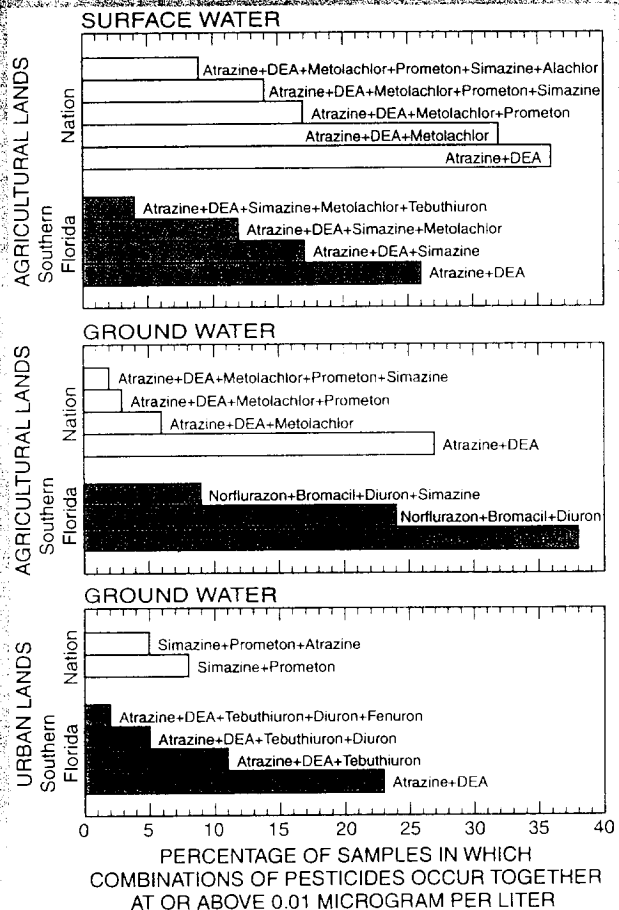


WHAT COMBINATIONS OF PESTICIDES OCCUR MOST FREQUENTLY IN SOUTHERN FLORIDA AND THE NATION?

The composition of the most common pesticide mixtures in surface water of agricultural areas in the SOFL Study Unit is similar to, but generally lower than, that of mixtures in agricultural areas nationwide. Atrazine, deethylatrazine (DEA), simazine, and metolachlor are found together most frequently in both southern Florida and the Nation.

Ground water in the SOFL Study Unit, however, has a different composition of pesticide mixtures than is found in ground water in other areas of the Nation. Norflurazon and bromacil were found together most frequently as mixtures in the SOFL Study Unit; these pesticides commonly are used on citrus crops.

The composition of the most common pesticide mixtures in ground water in urban areas in the SOFL Study Unit also differs from that throughout the rest of the Nation: the most common pesticides in mixtures were atrazine and DEA in the SOFL Study Unit compared with simazine and prometon nationwide. The atrazine and DEA in shallow ground water of the southern Florida urban study area may originate from local residential lawn herbicide applications or may be transported from nearby agricultural lands, either through the atmosphere or in canals. Canals that drain agricultural lands recharge shallow ground water in this urban area.



DEA is deethylatrazine. National data from U.S. Geological Survey, 1999.

Transport of Herbicides and their Breakdown Products

To evaluate the geochemical transport of herbicides, water and bed-sediment samples were collected in May 1997 and February 1998 from six SOFL sites (fig. 10) representing different land uses. The samples were analyzed after the methods of Thurman and others (1990) and Meyer and others (1993) for a suite of herbicides and breakdown products. Low levels (0.05 to 2.5 $\mu\text{g/L}$) of one or more herbicides were detected in water at all sites, including atrazine at every site. Other herbicides detected include ametryn, prometryn, and metolachlor at the sugarcane site, simazine and metolachlor at a mixed-agricultural (vegetable) site, and ametryn, simazine, and terbutryn at a citrus site. Atrazine (at trace levels) and ametryn (exceeding 40 micrograms per kilogram ($\mu\text{g/kg}$)) were detected in the sediment samples from the sugarcane site (S-6) in both years. The only other herbicides detected in sediments were trace levels of ametryn and alachlor at the mixed-agricultural (vegetable) site (Canal C-111). A breakdown product of alachlor, 2,6-diethylaniline, was detected in water at the same site. At the sugarcane site, S-6, the ratio of sediment-to-water concentration for ametryn was 240 (1997) and 580 (1998) and was zero for atrazine both years, which indicates that ametryn is transported primarily in sediment and atrazine is transported primarily in water.

The ratio of the concentration of deethylatrazine to the parent herbicide atrazine (DAR) has been used as an indicator of herbicide transport and surface- and ground-water interaction in the Midwestern States. Generally, a ratio greater than 1.0 indicates slow unsaturated zone transport and ground-water contributions to surface water (Adams and Thurman, 1991; Thurman and others, 1991; 1992). An elevated ratio also can be caused by photodecomposition of atrazine to deethylatrazine during atmospheric transport of herbicides (Goolsby and others, 1997). A ratio less than 0.1 indicates rapid overland-flow transport to surface water shortly after herbicide application. In southern Florida, DAR values were less than 0.1 at the sugarcane site (S-6), suggesting rapid transport of the herbicide into canal water shortly after herbicide application. The DAR at three other sites was between 0.1 and 1.0, which suggests post-application runoff. The DAR value of 1.0 at the background site (Br-105) presumably is from low-level atmospheric transport and photodecomposition because this site is remote from any farm runoff or ground-water sources of atrazine. The higher values above 1.0 at the citrus site (U.S. Sugar) and Canal C-111 at S-178 could indicate ground-water contributions (as in the Midwest) or more rapid photodegradation of atrazine in southern Florida because of higher temperatures, stronger sunlight, or greater soil organic carbon content and soil moisture than in the Midwest.

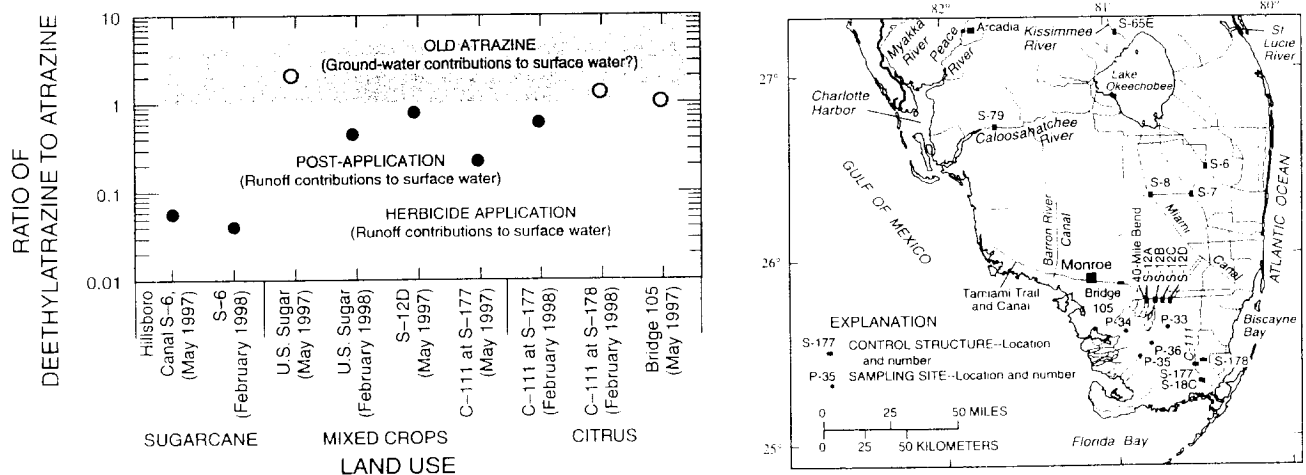


Figure 10. Ratio of deethylatrazine to atrazine (DAR) in surface water at selected sites, 1997–98 (M.T. Meyer, U.S. Geological Survey, written commun., 1998).

Pesticides, PCBs, other organics, and trace elements have accumulated in bottom sediment and fish

Many pesticides, other trace organic compounds, and trace elements are hydrophobic; that is, in aquatic environments they tend to be associated with sediment particles and biological tissues rather than dissolved in water. For this reason, sampling bottom sediment and fish is an effective way to assess the occurrence of these contaminants in the aquatic environment.

The most frequently detected pesticides in bottom sediment at the SOFL sites during 1996–98 were DDT and its breakdown products DDE and DDD, ranging from 2.4 to 670 micrograms per kilogram ($\mu\text{g}/\text{kg}$). DDE exceeded the Canadian sediment quality guidelines probable effects level (PEL) of $6.75 \mu\text{g}/\text{kg}$ (Environment Canada, 1999) at four of the seven fixed sites, including the Hillsboro Canal at S-6 ($308 \mu\text{g}/\text{kg}$) and the Kissimmee River at S-65E ($670 \mu\text{g}/\text{kg}$). More than 40 organic compounds were detected in bed-sediment from the Hillsboro Canal at S-6, including ametryn, chlordane, DDT compounds, dieldrin,



Row crops, C-111 basin, southern Florida.

endosulfan, and other semivolatile organic compounds, including polycyclic aromatic hydrocarbons (PAHs).

The most frequently detected pesticides in fish at 15 SOFL sites also were DDT and its breakdown products. Largemouth bass and (or) Florida gar were collected at each site, and one or more samples (5–8 whole fish) were analyzed for pesticides. DDT compounds were detected in 25 of the 27 composited fish samples. Concentrations of total DDT ranged from less than 5 to $1,170 \mu\text{g}/\text{kg}$ in Florida gar and from less than 5 to $610 \mu\text{g}/\text{kg}$ in largemouth bass. The most commonly detected and abundant DDT product was *p,p'*-DDE. Total DDT concentrations exceeded the $200 \mu\text{g}/\text{kg}$ guidelines (Newell and others, 1987) for the protection of fish-eating wildlife in 4 of 27 fish samples. Highest concentrations of total DDT were in canals of the northern Everglades near agricultural lands. For comparison, during 1970–73 concentrations of total DDT in 49 composite fish samples from 12 sites in southern Florida ranged from 6 to $800 \mu\text{g}/\text{kg}$. In 1978, total DDT concentrations in 23 fish samples ranged from 3 to $1,650 \mu\text{g}/\text{kg}$ (Haag and McPherson, 1997).



Fish sampling in Big Cypress Swamp as part of the southern Florida NAWQA study.

Other organochlorine compounds and PCBs detected in the composite whole-fish samples collected during 1995–96 include polychlorinated biphenyls (PCBs) and the pesticides dieldrin, mirex, and various compounds of chlordane (*cis*-chlordane, *oxy*-chlordane, *trans*-chlordane, *trans*-nonachlor, and *cis*-nonachlor). Maximum concentrations of pesticides occurred primarily in fish collected in Hillsboro Canal at S-6. PCBs were detected in three separate fish samples; the maximum concentration ($140 \mu\text{g}/\text{kg}$) was in fish collected from Black Creek Canal. Dieldrin and toxaphene were two other pesticides commonly detected in composite fish samples from the Everglades in the early 1970s. Concentrations of dieldrin in largemouth bass were as high as $130 \mu\text{g}/\text{kg}$, and concentrations of toxaphene were as high as $5,000 \mu\text{g}/\text{kg}$. In 1995, dieldrin concentrations ranged from less than 5 to $18 \mu\text{g}/\text{kg}$, and 5 of 27 fish samples had detectable dieldrin. Toxaphene was not detected in fish collected during 1995–96, but the analytical method was not very sensitive for toxaphene (only levels greater than $200 \mu\text{g}/\text{kg}$ could be detected).

The types and amounts of pesticides used in Florida have changed over the years because of new technology, land use, and State and Federal regulations. One of the most frequently detected herbicides in bed sediment in southern Florida, ametryn, is used in relatively small amounts on sugarcane crops (6 tons per year [tons/y]) (Miles and Pfeuffer, 1997). By far, the greatest frequency of insecticide detection was the organochlorine insecticides, such as DDD, DDE, DDT, dieldrin, and heptachlor. DDT was banned for most uses in the Nation in 1973. These insecticides also are the most frequently detected pesticides in bottom sediments (Shahane, 1994). Although most organochlorine pesticides such as DDT and chlordane are no longer sold in the United States, they persist in the environment and continue to pose potential threats to wildlife and humans. Persistent organochlorine pesticides were detected beginning in the late 1960s and early 1970s (Kolipinski and Higer, 1969; McPherson, 1973) in bottom sediment and fish that are a part of the food chain in the Everglades. Reflooding of farm lands for Everglades restoration potentially could lead to mobilization of persistent organochlorine pesticides and foodweb contamination, as occurred in Lake Apoka just north of the study area. Many organochlorine pesticides and PCBs also have been linked to hormone disruption and reproductive problems in aquatic animals (Colborn and others, 1993).

Polynuclear aromatic hydrocarbons (PAHs) are contaminants in soils and sediments and originate from such sources as crude oil and tar and from forest fires and incom-

plete combustion of fossil fuels. Bottom-sediment samples were collected at 10 sites in a survey of the Barron River Canal in 1998 to evaluate the occurrence of PAHs and other semivolatile organic compounds in the vicinity of the Big Cypress National Preserve (Miller and McPherson, U.S. Geological Survey, in press). PAHs normalized to organic carbon had patterns of distribution that indicated sources to be roads, vehicles, or an old creosote wood-treatment facility. Concentrations of phthalate esters and the trace elements arsenic, cadmium, and zinc in the Barron River Canal appear to have a nonpoint source and to be influenced by local bed-sediment properties, such as sediment particle size and organic content. At some Barron River Canal sites, lead, copper, and zinc, normalized to aluminum, exceeded background levels and may be enriched by human activities. Trace elements in bottom-sediment samples from the Barron River Canal sites did not exceed the Canadian PEL for freshwater sediment.

Mercury is a contaminant in the Southern Florida Study Unit

Game fish in the Everglades have concentrations of mercury that exceed recommended levels (1.5 micrograms per gram, $\mu\text{g/g}$) for human consumption (Ware and others, 1990; fig. 11). The maximum concentrations of mercury found in edible portions of largemouth bass (4.4 $\mu\text{g/g}$) collected from the Everglades exceeded mercury concentrations in game

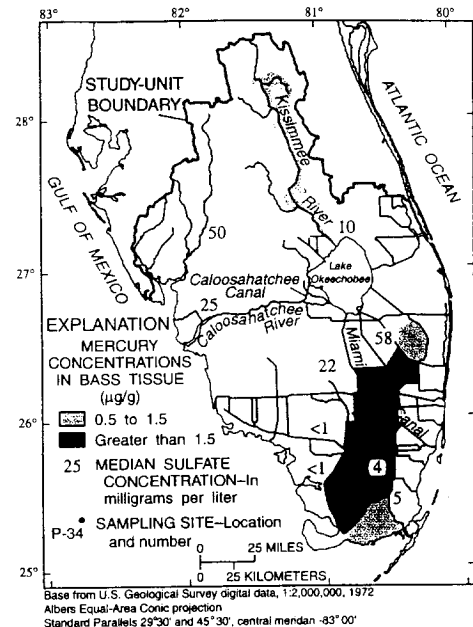


Figure 11. Mercury concentrations in largemouth bass tissue equaled or exceeded 0.5 $\mu\text{g/g}$ in an area in southern Florida. (Lambou and others, 1991). Median sulfate concentrations at SOFL sites, P-33 and P-34, in mg/L, 1996-98 (P-site data from South Florida Water Management District).

fish from all other parts of the State (Stober and others, 1995). The NAWQA Program specifies that trace elements (including mercury) be determined for fish livers. Thus, the data are not directly comparable to data from studies analyzing fish filets (the edible portion of the fish). Mercury concentrations in composite samples of largemouth bass livers from the SOFL sites ranged from 0.4 $\mu\text{g/g}$ in Black Creek Canal to 42 $\mu\text{g/g}$ in Miami Canal at S-8. Mercury concentrations in Florida gar livers ranged from 2.1 $\mu\text{g/g}$ in the Caloosahatchee River at Alva to 190 $\mu\text{g/g}$ in the Miami Canal at S-8. Largemouth bass and gar are top predators; on average, mercury concentrations in gar livers were about four times higher than mercury concentrations in largemouth bass livers.

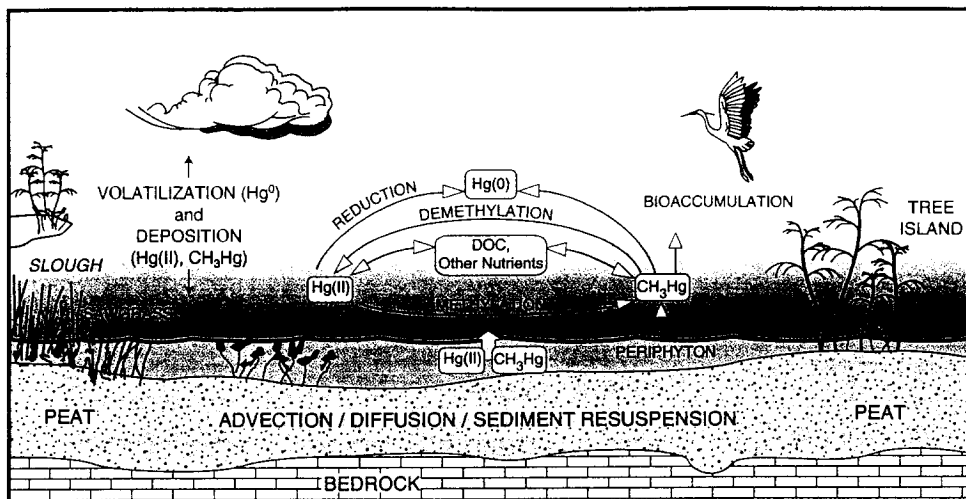


Figure 12. High concentrations of mercury in Everglades fish, birds, and other organisms is a result of food web bioaccumulation of methylmercury, which originates primarily at the sediment and periphyton-water interface (modified from Krabbenhoft, 1996).

The high concentrations of mercury in fish result from food-web bioaccumulation of methylmercury, the most biologically available form of mercury in the environment (fig. 12). Buildup in the Everglades food web begins with high rates of bacterial mercury methylation at the sediment-water interface and subsequent transport of methylmercury from this interface into the water column. The Everglades region had one of the highest methylmercury to mercury ratios in sediment of the 21 NAWQA basins sampled in 1998 (Krabbenhoft and others, 1999a).

Mercury methylation rates in the Everglades are affected by complex physical-chemical-biological processes that vary widely from day to night, from season to season, and along spatial gradients. Two of the most important controls on mercury methylation are availability of sulfur and bacterial cycling of sulfur. Sulfur inputs to the Everglades have increased over background levels during the 20th century as a result of runoff containing agricul-

tural fertilizers. Large inputs of sulfur in parts of the northern Everglades have stimulated sulfate reduction that would normally favor methylation, but very high levels of sulfide in the eutrophic areas have an inhibitory effect on methylation rates. Lower levels of sulfur contamination in the central Everglades (fig. 11) have increased sulfate reduction and mercury methylation without the inhibitory effects of excess sulfide on methylation rates (Benoit and others, 1999; Orem and others, 1999, p. 79). Concentrations of mercury are higher in fish and other organisms in the more remote and low-nutrient waters of the central and southern Everglades than in the high-nutrient waters of the northern Everglades (Cleckner and others, 1998; Gilmour and others, 1998; Hurley and others, 1998).

Degradation of methylmercury also is a dynamic process that varies from day to night and spatially across the Everglades. Microbial demethylation rates in the sediments do not show strong spatial patterns, whereas photo-

demethylation rates in the water column do show a strong north to south upward trend, ranging from 2 to 15 percent per day, as a result of decreasing DOM in the water column from north to south and the resulting increase in light penetration in the south (Krabbenhoft and others, 1999b).

Atmospheric sources supply mercury that sustains methylation and food-web biomagnification in the Everglades. The southern Florida area has one of the highest atmospheric mercury deposition rates (25 micrograms per square meter per year, $\mu\text{g}/\text{m}^2/\text{y}$) in the United States (Krabbenhoft and others, 1999a). Atmospheric inputs have local sources, such as medical, municipal, and industrial incinerators, landfills, power plants, and other urban activities, and global sources. Local sources are considered by most investigators to be primarily responsible for the relatively high atmospheric mercury inputs in southern Florida.

Biological communities are influenced by water quality

The distribution of fish, invertebrates, and algae within a river or canal is influenced by natural conditions and human activities that affect water quality and available habitat. Generally, a diverse aquatic community composed of a variety of species and dominated by no single species or group of species is an indicator of favorable biotic conditions and an absence of contaminants and other environmental stresses. Although human activities have caused significant changes in many southern Florida freshwater habitats, it is important to understand how unique natural conditions in southern Florida influence the composition of aquatic communities there.

Sixty-three species of fish in 26 families were collected in the seven canals and rivers sampled in the SOFL Study Unit during 1996–98. The fish community included 43 native species and 10 exotic or non-native species. Additionally, 10 species of marine fish that periodically inhabit portions of canals and rivers in southern Florida also were collected. The chemical composition of freshwater in southern

Florida, in particular the concentrations of sodium, chloride, and calcium, facilitates the invasion and occasional establishment of fishes with broad salinity tolerance (Loftus and Kushlan, 1987). No fish species listed as threatened or endangered in the United States or Florida were collected in southern Florida during the NAWQA study.

Florida rivers have smaller drainage basins and fewer fish species than rivers in the adjacent Southeastern United States (Swift and others, 1986). Natural conditions, in this case the repeated rise and fall of sea level over geologic time, have reduced the number of freshwater fish species in the Florida peninsula. The present fish community represents the most recent reinvasion following sea-level withdrawal (Bass, 1990). The most important natural factors limiting the diversity of the fish community are unsuitability of habitat and climate for temperate species (Loftus and Kushlan, 1987).

The rivers in southern Florida have very low gradients, typically only a few inches per mile (McPherson and Halley, 1996), as well as slow current velocities, high water temperatures, and low

dissolved-oxygen concentrations. Fish and invertebrate communities are characterized by species that prefer these conditions and by the absence of related species that are found only in cool, swiftly flowing, well-oxygenated streams. The number of fish species collected at SOFL sites, or the species richness of the community, tended to increase with increasing mean annual dissolved-oxygen concentrations. Some game-fish species, such as bass, were absent at sites with the lowest dissolved-oxygen concentrations. One entire group of invertebrates (Plecoptera: stoneflies), which prefer cool running waters, usually are not found anywhere in the waters of southern Florida. Diptera (true flies) was the dominant insect group at all sites (fig. 13). Many of the Diptera were species in the family Chironomidae (midges), which are adapted to aquatic environments with sandy substrates and very low dissolved-oxygen concentrations, such as those prevalent in southern Florida surface waters. Species in this family are also tolerant of nutrient enrichment and contaminants.

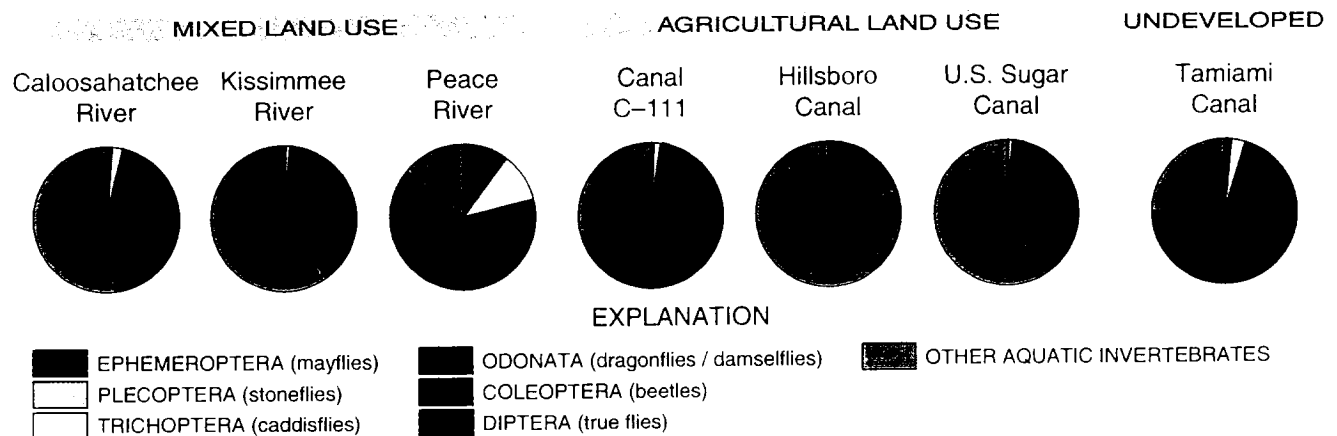


Figure 13. Diptera (true flies) dominated the insect community at the Southern Florida National Water-Quality Assessment Study Unit Sites, 1996–98.

In the canals and some of the rivers of southern Florida, habitat alteration is significant. The canal channels are rectangular in cross-section, and they have been channelized and are routinely dredged to facilitate navigation and the movement of water. There are few areas of shallow water (littoral zone) to provide sufficient light for attached algae (periphyton) and suitable spawning areas for fish. Moreover, aquatic vegetation is periodically removed for water-management purposes, further reducing habitat for small fish and invertebrates that serve as food for fish. The species richness of fish and the total numbers of fish collected generally were lowest in the canals of southern Florida in basins with agricultural land use (fig. 14). The canals and channelized rivers have little suitable habitat of any kind for many aquatic insects that prefer hard substrate, such as caddisflies (Trichoptera) and some mayflies (Ephemeroptera); snags (woody debris) are scarce because bankside vegetation is usually cleared, and hard surfaces, such as rock, are limited to porous limestone outcroppings.

Fish can be categorized on the basis of their ability to tolerate a range of environmental conditions. There were no intolerant fish species collected in any of the canals in southern Florida (fig. 14). The presence of intolerant species in the Peace River during the SOFL study, as well as in earlier studies (Champeau, 1990), indicates favorable biotic conditions. The Peace River is unique among the rivers sampled in southern Florida because it has not been

channelized, and the main channel is connected to the flood plain during periods of high water. The flood plain provides a refuge for fish during high flow, perhaps enhancing survival and reproduction. Also, periodic flooding results in input of particulate organic matter and dissolved organic carbon from the flood plain, which increases the productivity of the river and increases its ability to support a diverse fish community.

The environmental conditions in the Peace River also supported a diverse aquatic insect community compared to most other SOFL sites (fig. 13). For example, the Peace River had the highest mean dissolved-oxygen concentration of any southern Florida site. Hard substrates, such as the woody snags preferred by many aquatic insects, were abundant. Seasonal input of leaf debris (particulate organic matter) from the flood plain provides a valuable food source for many aquatic invertebrates. In contrast, Hillsboro Canal, a relatively degraded SOFL site, had insects from a number of different groups, but few of the pollution-

sensitive insects, including caddisflies, were present.

In many parts of the United States, the abundance of a group of invertebrate taxa referred to as the EPT taxa (Ephemeroptera, Plecoptera, Trichoptera) is used to assess the biotic condition of streams and rivers. Species in these insect groups generally thrive in conditions of flowing water, moderate to high dissolved oxygen, and low suspended sediment and are intolerant of contaminants and habitat disturbance. The relative abundance of EPT taxa compared to the more tolerant Diptera taxa can be used to assess and compare sites. However, this metric must be used with caution because Plecoptera usually are not found in southern Florida. The Peace River had the highest EPT/Diptera ratio compared to the other SOFL sites. The Peace River also supported a diverse algae community with the greatest species richness (220 species) of all the SOFL sites. Total phosphorus and nitrogen concentrations were high in the Peace River compared to other SOFL sites because of natural conditions and human activities. The avail-

ability of these essential nutrients, combined with an unmodified habitat and more shallow-water areas, may contribute to a more diverse algal community.

The Tamiami Canal is unique among the SOFL canal sites because land within its basin is mostly undeveloped, limiting the input of contaminants. However, the environmental conditions are stressful because the shallow canal has almost no flow, the water is warm, and dissolved-oxygen concentra-

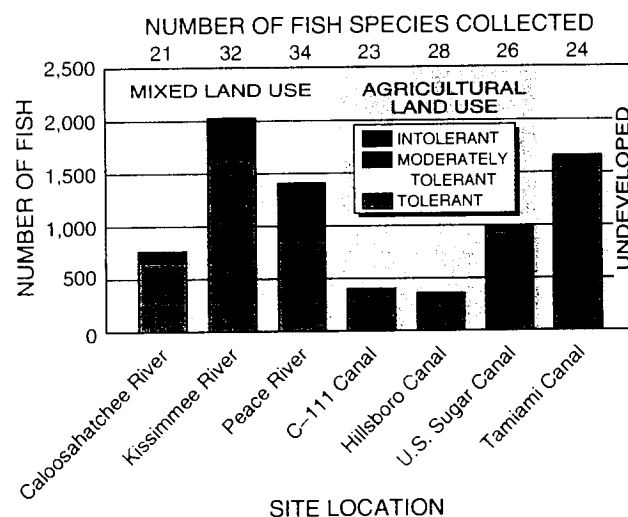


Figure 14. Environmentally tolerant fish are common in southern Florida canals and rivers.

tions typically are very low. Nutrient concentrations were among the lowest of the SOFL sites. The insect community was dominated by Diptera, although some Trichoptera that are adapted to still or slow-flowing conditions were collected. Bluegreen algae were relatively abundant. The fish community showed little similarity to any of the other sites and was

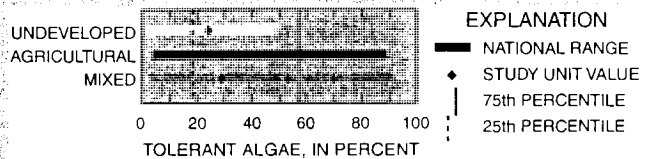
characterized by an abundance of minnows and Florida gar. The Tamiami Canal had the smallest proportion (5 percent) of fish with external anomalies. External anomalies, such as eroded fins, lesions, ulcers, tumors, and external parasites, can be an indication that fish are stressed by environmental conditions or contaminants. The highest

percentages of fish with external anomalies were from the Hillsboro (32 percent) and C-111 Canals (22 percent). Anomalies at these two sites were primarily external parasites, eroded fins, and anatomical deformities. These two sites were in the top 25 percent of 144 NAWQA sites sampled nationwide.

SOUTHERN FLORIDA AQUATIC COMMUNITIES IN A NATIONAL CONTEXT

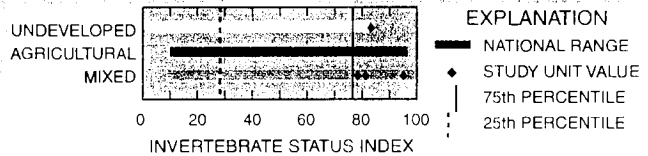
Algae Siltation Index

Algae collected at the seven southern Florida NAWQA sites were compared to algae at 140 national NAWQA sites by using the Siltation Index (Bahls and others, 1992). This index is the relative abundance of the motile diatoms *Navicula*, *Nitzschia*, *Cylindrotheca*, and *Surirella* in a diatom count. These diatoms are able to move through silt particles, and because they are able to avoid being buried, they are considered more tolerant of sedimentation than other diatoms. The values for this index are higher for streams in agricultural basins. Based on the relative abundance of these diatoms, the undeveloped Southern Florida site (Tamiami Trail Canal) falls into the less degraded stream category, whereas the agricultural sites are between the 25th and 50th percentile, and two of the three major river sites in the mixed land-use category are near or above the 75th percentile. See pages 18–20 for how algae communities compare at sites within the SOFL Study Unit.



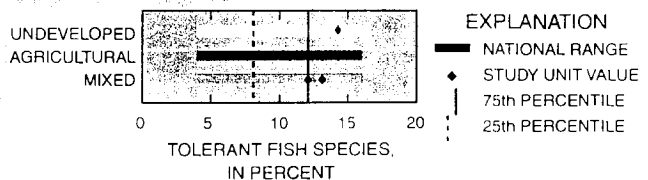
Invertebrate Community Status Index (ICSI)

Invertebrate communities at the seven southern Florida NAWQA sites were compared to invertebrate communities at NAWQA sites nationwide by using a multimetric index. The index combines 11 metrics, including ones that assess taxa richness and diversity, taxa richness of mayflies, stoneflies, and caddisflies, and various measures of environmental tolerance. Using the ICSI, all seven of the Southern Florida sites ranked above the 75th percentile, indicating that they are among the most degraded sites in the Nation. Use of the ICSI in southern Florida must be approached with caution because background environmental conditions prevalent at all sites, including slow water velocities and low dissolved-oxygen concentrations, probably contribute to the high index values and may obscure differences in invertebrate communities because of other factors, such as land use or changes in habitat. The dominance of Diptera (true flies), insects that are tolerant of degraded environmental conditions, and low diversity of insects overall contribute to the high ICSI scores for Southern Florida. See pages 18–20 for how insect communities compare at sites within the SOFL Study Unit.



Fish Status Index

To facilitate comparisons, fish collected at NAWQA sites across the Nation were classified based on four fish metrics (percentage of tolerant, omnivorous, non-native individuals, and percentage of individuals with external anomalies). Nationwide, higher values suggest a more degraded stream site. Six of the SOFL sites, including the reference site, ranked in the highest 25th percentile category for the Nation. Environmental conditions mentioned above for invertebrates also may contribute to stress in the fish community at sites including the reference site. This environmental stress can lead to dominance of tolerant fish, favorable conditions for exotic species, and increased external anomalies at some sites in the SOFL Study Unit, leading to higher values compared with other sites nationwide. See pages 18–20 for a discussion of how fish communities compare at sites within the SOFL Study Unit.



Exotic species are a threat to native biota

Exotic or non-native species of fish, as well as other animals and plants, represent a major threat to native biota in southern Florida. Exotic fish compete with native species for food and habitat. Exotic plants crowd out native species and can form dense monocultures that alter habitat for birds, fish, and other native biota. Ten exotic fish species were collected at the SOFL sites during 1996–98 (fig. 15). Cichlids were the most numerous and widespread exotic species, including the black acara, Mayan cichlid, peacock cichlid, blue tilapia, spotted tilapia, and the oscar. The other exotic species collected (pike killifish, grass carp, walking catfish, and sailfin catfish) were all rare and found at only one or two sites each. Fifty-four exotic fish species have been recorded in southern Florida, and the total statewide exceeded 125 species by 1998 (Fuller and others, 1999). Nearly all the exotic fish species were originally imported for the aquarium trade and either escaped from fish farms or were released by individuals with home aquariums. Exotic fish usually are most abun-

dant in streams that have been altered by human activity (Moyle, 1986). Generally, the canal sites in southern Florida had more exotic species than the major rivers. The number of species of exotic fish at SOFL sites ranked in the top one-third of NAWQA sites nationwide.

The swamp eel (*Monopterus albus*), recently introduced from Southeast Asia, is a potentially dangerous invader. The swamp eel can breathe air for extended periods of time, enabling it to inhabit stagnant water and even live out of water. This characteristic also enhances its ability to disperse widely. It is a voracious general predator, making it a threat to native fishes, amphibians, and aquatic invertebrates. The swamp eel was not collected at any of the SOFL sites during 1996–98. However, this species has affected regional water-management practices. Selected water-control structures in the vicinity of established populations are not being opened to prevent or at least retard dispersal, particularly into waters of ENP.

In addition to fish, other exotic species are of great concern in Florida. At least 25 percent of all plant species in the State are exotic

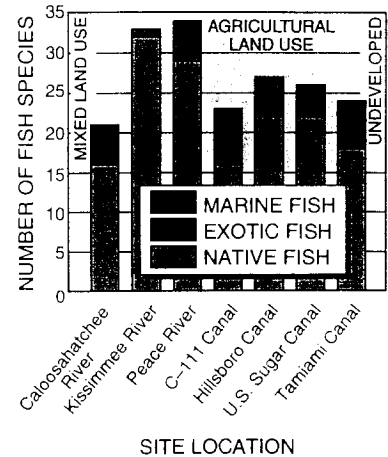
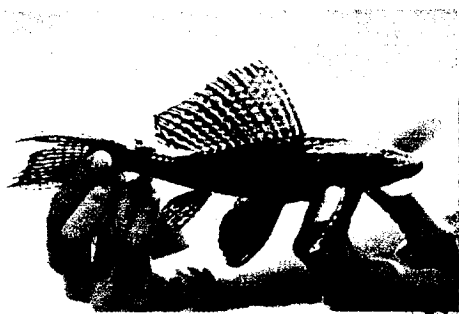
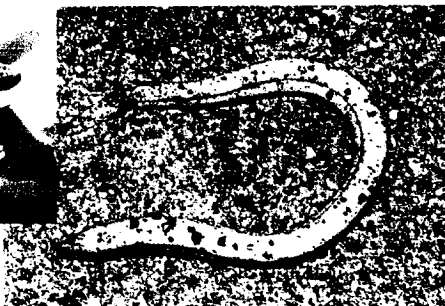


Figure 15. Exotic fish species were collected at all SOFL sites, 1996–98.

(Langeland, 1998). Melaleuca, a wetland tree, covers almost 400,000 acres in the State and is particularly troublesome in the Water Conservation Areas (Laroche, 1994). Brazilian pepper, a wetland shrub, has infested more than 100,000 acres in the ENP (Ferriter, 1997). Hydrilla, water hyacinth, water milfoil, and water lettuce are the most abundant exotic aquatic plant species infesting canals throughout southern Florida. Efforts to control or eradicate exotic plant species involve the use of mechanical, chemical, and biological control methods. Numerous herbicides are applied, including 2,4-D, copper sulfate, diquat, endothall, fluridone, glyphosate, imazapyr, triclopyr, and others. At least two of these compounds (2,4-D and triclopyr) have been detected in surface water at the SOFL sites.

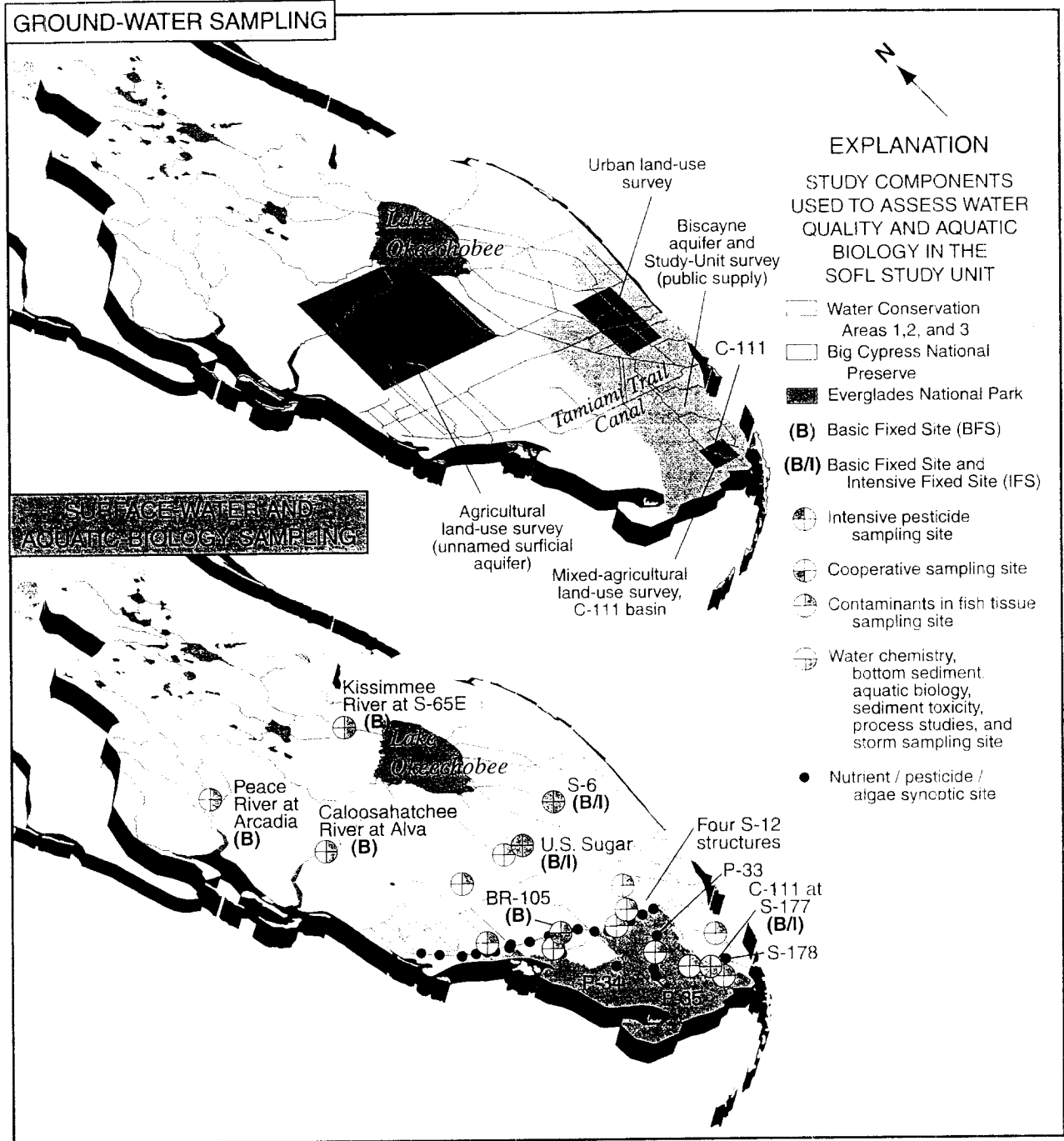


The exotic sailfin catfish, *Pterygoplichthys disjunctivus*, is a South American invader.



The Asian swamp eel, *Monopterus albus*, continues to spread.

STUDY UNIT DESIGN



Study component (Type of site)	Objectives	Brief description and water-quality parameters	Number of sites	Sampling frequency during water year	Historical data available
SURFACE WATER					
Bottom-sediment survey (BFS)	Determine presence of potentially toxic compounds in sediment at selected sites	Sample the depositional zone for pesticides, other synthetic organic compounds, and trace elements at water-chemistry sites	7	1 (1996)	some
Water-chemistry sites (BFS)	Describe concentrations and loads of major ions and nutrients at selected sites	Sample near-continuous streamflow sites for major ions, nutrients, organic carbon, and suspended sediment	7	2-6 in 1996 12-31 in 1997 12 in 1998	data at some sites
Cooperative sampling program (BFS)	Assess the influence of sampling method and location on water-quality data	Sample water-chemistry sites on the same day as the SFWMD using both USGS methods and SFWMD methods	5	2-6 in 1996 12-31 in 1997 12 in 1998	none
Storm sampling (BFS)	Assess effects of runoff during high-flow conditions on surface-water quality	Sample water-chemistry sites during high-flow conditions for major ions, nutrients, organic carbon, and suspended sediment	variable	variable (1997 and 1998)	some
Nutrient/pesticide synoptic studies	Determine the concentration of nutrients and pesticides across southern end of Study Unit during high and low flow	Sample sites along the Tamiami Trail Canal during low flow (June) and high flow (August) for major ions, mercury, nutrients, organic carbon, and pesticides	30	3 (Aug. 1996, June and August 1997)	limited
Process studies	Describe processes controlling the degradation and biological availability of pesticides	Deploy and collect semipermeable membrane devices at selected sites. Collect water and bottom-sediment samples	variable	4 (1997-98)	undetermined
Intensive pesticide sampling (IFS)	Determine seasonal variation in the occurrence and concentrations of nutrients and pesticides at water-chemistry sites	Sample water-chemistry sites that are located in specified agricultural land-use areas (citrus, sugarcane, and vegetables) for nutrients and pesticides	3	2 in 1996; weekly/biweekly in 1997; monthly in 1998	some
GROUND WATER					
Urban land-use survey	Describe water quality in a residential and light commercial land-use area in a shallow aquifer susceptible to contamination	Sample monitoring wells for major ions, nutrients, organic carbon, pesticides, radionuclides, trace elements, and volatile organic compounds (VOCs)	38 wells	1 (1997)	some
Ground-water and surface-water synoptic survey	Determine which water-quality constituents are transported through shallow ground water into surface water in an agricultural land-use area	Sample shallow wells and surface-water sites for major ions, nutrients, pesticides, and VOCs before and during a surface-water release in a canal that drains agricultural land	7 wells; 9 surface-water sites	1-5 (1997)	none
Agricultural land-use survey	Describe the effects of citrus land use on water quality in a shallow aquifer.	Sample new wells for major ions, nutrients, pesticides, radionuclides, and trace elements	30 wells	1 (1998)	some
Study-unit survey (public supply)	Describe overall water quality in the Biscayne aquifer, which is used for drinking-water supply	Sample existing wells for major ions, nutrients, organic carbon, pesticides, radionuclides, trace elements, and VOCs	30 wells	1 (1998)	some
AQUATIC BIOLOGY					
Contaminants in fish tissue	Determine the presence of contaminants that can accumulate in fish tissue	Collect largemouth bass and Florida gar at a large number of sites. Sample composites of whole fish for organic compounds and fish livers for trace elements	15	1 (1996)	limited
Sediment toxicity testing	Evaluate potential toxicity of bed sediment at the water-chemistry sites	Collect bed-sediment samples and evaluate their potential toxicity by using ASTM methods and <i>Hyallela azteca</i>	7	1 (1996)	none
Aquatic biology	Assess biological communities and stream habitat at the Basic Fixed Sites	Quantitatively sample fish, macroinvertebrates, and algae near water-chemistry sites. Quantitatively describe stream habitat for these organisms	7	3 (1 per year during 1996-98)	limited
Algae synoptic studies	Determine the influence of nutrients on the algal community along a nutrient gradient in the southern part of the basin	Collect algae using artificial substrate samplers at sites across the Tamiami Trail Canal	10-13	3 (June, August, November, 1997)	none

GLOSSARY

- Aquatic-life criteria** - Water-quality guidelines for protection of aquatic life. Often refers to U.S. Environmental Protection Agency water-quality criteria for protection of aquatic organisms.
- Aquifer** - A water-bearing layer of soil, sand, gravel, or rock that will yield usable quantities of water to a well.
- Basic Fixed Sites (BFS)** - Sites on streams or canals at which streamflow is measured and samples are routinely collected for temperature, salinity, major ions, nutrients, and organic carbon. Samples may also be periodically collected for contaminants in bottom sediment, biota, storm effects, or other constituents.
- Breakdown products** - Compounds resulting from transformation of an organic substance through chemical, photochemical, and/or biochemical reactions.
- DDT** - Dichloro-diphenyl-trichloroethane. An organochlorine insecticide no longer registered for use in the United States. DDT degradation products include DDE and DDD.
- Drinking-water standard or guideline** - A threshold concentration in a public drinking-water supply, designed to protect human health.
- Ecosystem** - The interacting populations of plants, animals, and microorganisms occupying an area, plus their physical environment.
- Intensive Fixed Sites (IFS)** - Basic Fixed Sites with increased sampling frequency during selected seasonal periods and analysis of dissolved pesticides for 1 year.
- Load** - General term that refers to a material or constituent in solution, in suspension, or in transport; usually expressed in terms of mass or volume.
- Maximum contaminant level (MCL)** - Maximum permissible level of a contaminant in water that is delivered to any user of a public water system. MCLs are enforceable standards established by the U.S. Environmental Protection Agency.
- Methylation** - The addition of a methyl group ($-CH_3$) to a molecule or atom through a chemical reaction.
- Micrograms per liter ($\mu\text{g/L}$)** - A unit expressing the concentration of constituents in solution as weight (micrograms) of solute per unit volume (liter) of water; equivalent to one part per billion in most stream water and ground water.
- Nutrient** - Element or compound essential for animal and plant growth. Common nutrients in fertilizer include nitrogen, phosphorus, and potassium.
- Organochlorine insecticide** - A class of organic insecticides containing a high percentage of chlorine. Includes dichlorodiphenylethanes (such as DDT), chlorinated cyclo-dienes (such as chlordane), and chlorinated benzenes (such as lindane).
- Periphyton** - Organisms that grow on underwater surfaces; periphyton include algae, bacteria, fungi, protozoa, and other organisms.
- Pesticide** - A chemical applied to crops, rights of way, lawns, or residences to control weeds, insects, fungi, nematodes, rodents or other "pests."
- Phosphorus** - A nutrient essential for growth that can play a key role in stimulating aquatic growth in lakes and streams.
- Phthalates** - A class of organic compounds containing phthalic acid esters [$C_6H_4(COOR)_2$] and derivatives. Used as plasticizers in plastics. Also used in many other products (such as detergents, cosmetics) and industrial processes (such as defoaming agents during paper and paperboard manufacture, and dielectrics in capacitors).
- Polychlorinated biphenyls (PCBs)** - A mixture of chlorinated derivatives of biphenyl, marketed under the trade name Aroclor with a number designating the chlorine content (such as Aroclor 1260). PCBs were used in transformers and capacitors for insulating purposes and in gas pipeline systems as a lubricant. Further sale for new use was banned by law in 1979.
- Polycyclic aromatic hydrocarbon (PAH)** - A class of organic compounds with a fused-ring aromatic structure. PAHs result from incomplete combustion of organic carbon (including wood), municipal solid waste, and fossil fuels, as well as from natural or anthropogenic introduction of uncombusted coal and oil. PAHs include benzo(a)pyrene, fluoranthene, and pyrene.
- Recharge** - Water that infiltrates the ground and reaches the saturated zone.
- Reference site** - A sampling site selected for its relatively undisturbed conditions.
- Trace element** - An element found in only minor amounts (concentrations less than 1.0 mg/L) in water or sediment; includes arsenic, cadmium, chromium, copper, lead, mercury, nickel, and zinc.
- Volatile organic compounds (VOCs)** - Organic chemicals that have a high vapor pressure relative to their water solubility. VOCs include components of gasoline, fuel oils, and lubricants, as well as organic solvents, fumigants, some inert ingredients in pesticides, and some by-products of chlorine disinfection.
- Water-quality criteria** - Specific levels of water quality which, if reached, are expected to render a body of water unsuitable for its designated use.
- Water-quality standards** - State-adopted and U.S. Environmental Protection Agency-approved ambient standards for water bodies. Standards include the use of the water body and the water-quality criteria that must be met to protect the designated use or uses.
- Water year** - The 12-month period October 1 through September 30, designated by the calendar year in which it ends.

REFERENCES

- Adams, C.D., and Thurman, E.M., 1991, Formation and transport of deethylatrazine in the soil and vadose zone: *Journal of Environmental Quality*, v. 20, no. 3, p. 540–547.
- Bahls, L.R., Burkantis, R., and Tralles, S., 1992, Benchmark biology of Montana reference streams: Helena, Montana, Department of Health and Environmental Science, Water Quality Bureau, 47 p.
- Bass, D.G., Jr., 1990, Monitoring Florida's riverine fish communities: *Florida Scientist*, v. 53, no. 1, p. 1–10.
- Benoit, J.M., Gilmour, C.C., Mason, R.P., and Heyes, A., 1999, Sulfide controls on mercury speciation and bioavailability to methylating bacteria in sediment and pore waters: *Environmental Science and Technology*, v. 33, p. 951–957.
- Champeau, T.R., 1990, Ichthyofaunal evaluation of the Peace River, Florida: *Florida Scientist*, v. 53, p. 302–311.
- Chiou, C.T., Malcolm, R.L., Brinton, T.I., and Kile, D.E., 1986, Water solubility enhancement of some organic pollutants and pesticides by dissolved humic and fulvic acids: *Environmental Science and Technology*, v. 20, p. 502–508.
- Cleckner, L.A., Garrison, P.J., Hurley, J.P., Olson, M.O., and Krabbenhoft, D.P., 1998, Trophic transfer of methyl mercury in the northern Florida Everglades: *Biogeochemistry*, v. 40, p. 347–361.
- Colborn, T., vom Saal, F.S., and Soto, A.M., 1993, Developmental effects of endocrine-disrupting chemicals in wildlife and humans: *Environmental Health Perspectives*, v. 101, p. 378–384.
- Davis, J.H., Jr., 1943, The natural features of southern Florida, especially the vegetation, and the Everglades: *Florida Geological Survey Bulletin* 25, 311 p.
- Environment Canada, 1999, Canadian sediment quality guidelines for the protection of aquatic life. Summary tables: accessed September 17, 1999 at <http://www.ec.gc.ca/ceqg-rcqe/sediment.htm>.
- Ferriter, A.P., 1997, Brazilian pepper management plan for Florida: West Palm Beach, Fla., Florida Exotic Pest Plant Council, 38 p.
- Fuller, P.L., Nico, L.G., and Williams, J.D., 1999, Nonindigenous fishes introduced into inland waters of the United States: Bethesda, Md., American Fisheries Society, Special Publication, v. 27, 613 p.
- Gilmour, C.C., Riedel, G.S., Ederington, M.C., Bell, J.T., Benoit, J.M., Gill, G.A., and Stordal, M.C., 1998, Mercury methylation and sulfur cycling across a trophic gradient in the northern Everglades: *Biogeochemistry*, v. 40, p. 327–345.
- Goolsby, D.A., Thurman, E.M., Pomes, M.L., Meyer, M.T., and Battaglin, W.A., 1997, Herbicides and their metabolites in rainfall: Origin, transport, and deposition patterns across the Midwestern and Northeastern United States, 1990–1991: *Environmental Science and Technology*, v. 31, p. 1325–1333.
- Haag, K.H., and McPherson, B.F., 1997, Organochlorine pesticides and PCBs in southern Florida fishes—Then and Now: U.S. Geological Survey Fact Sheet FS-110-97, 4 p.
- Haag, K.H., Miller, R.L., Bradner, L.A., and McCulloch, D.S., 1996, Water-quality assessment of southern Florida—An overview of available information on surface and ground water and ecology: U.S. Geological Survey Water-Resources Investigations Report 96-4177, 42 p.
- Hem, J.D., 1985, Study and interpretation of the chemical characteristics of natural water (3d ed.): U.S. Geological Survey Water-Supply Paper 2254, 263 p.
- Hurley, J.P., Krabbenhoft, D.P., Cleckner, L.B., Olson, M.L., Aiken, G., and Rawlik, P.S., Jr., 1998, System controls on aqueous mercury distribution in the northern Everglades: *Biogeochemistry* 40, p. 293–311.
- Klein, Howard, Armbruster, J.T., McPherson, B.F., and Freiberger, H.J., 1975, Water and the south Florida environment: U.S. Geological Survey Water-Resources Investigation 24-75, 165 p.
- Kolipinski, M.C., and Higer, A.L., 1969, Some aspects of the effects of the quantity and quality of water on biological communities in Everglades National Park: U.S. Geological Survey Open-File Report, 97 p.
- Kolpin, D.W., Barbash, J.E., and Gilliom, R.J., 1998, Occurrence of pesticides in shallow ground water of the United States—Initial results from the National Water-Quality Assessment Program: *Environmental Science and Technology*, v. 32, 558–566 p.
- Krabbenhoft, D.P., 1996, Mercury studies in the Florida Everglades: U.S. Geological Survey Fact Sheet FS-166-96, 4 p.
- Krabbenhoft, D.P., Wiener, J.G., Brumbaugh, W.G., Olson, M.L., DeWild, J.F., and Sabin, T.J., 1999a, A national pilot study of mercury contamination of aquatic ecosystems along multiple gradients, in Morganwalp, D.W., and Buxton, H.T., eds., U.S. Geological Survey Toxic Substances Hydrology Program—Proceedings of the Technical Meeting, Charleston, SC, March 8–12, 1999—Contamination of hydrologic systems and related ecosystems: U.S. Geological Survey Water-Resources Investigations Report 99-4018B, v. 2, p. 147–160.
- Krabbenhoft, D.P., Hurley, J.P., Marvin-DiPasquale, M., Orem, W.H., Aiken, G.R., Schuster, P.J., Gilmour, C.C., and Harris, R., 1999b, The aquatic cycling of mercury in the Everglades (ACME) project: A process-based investigation of mercury biogeochemistry in a complex environmental setting, in Gerould, Sarah, and Higer, Aaron, compilers, U.S. Geological Survey, Program on the South Florida Ecosystem—Proceedings of South Florida Restoration Science Forum, May 17–19, 1999, Boca Raton, Florida: U.S. Geological Survey Open-File Report 99-181, p. 54–56.
- Lambou, V.W., Barkay, T., Braman, R.S., Delfino, J.J., Jansen, J.J., Nimmo, D., Parks, J.W., Porcella, D.B., Rudd, J., Schultz, D., Stober, J., Watras, C., Wiener, J.G., Gill, G., Huckabee, J., and Rood, B.E., 1991, Mercury technical advisory committee interim report to the Florida Governor's Mercury in Fish and Wildlife Taskforce and the Department of Environmental Regulation: Tallahassee, Fla., Center for Biomedical and Toxicological Research and Waste Management, 60 p.
- Langeland, K.A., 1998, Help protect Florida's natural areas from non-native invasive plants: Gainesville, University

- of Florida, Cooperative Extension Service, Institute of Food and Agricultural Sciences, Circular 1204, 6 p.
- Laroche, F.B., ed., 1994, Melaleuca management plan for Florida: West Palm Beach, Florida Exotic Pest Plant Council, 88 p.
- Loftus, W.F., and Kushlan, J.A., 1987, Freshwater fishes of southern Florida: Gainesville, Fla., Bulletin of the Florida State Museum of Biological Sciences, v. 31, no. 4, 344 p.
- McIvor, C.C., Ley, J.A., and Bjork, R.D., 1994, Changes in freshwater inflow from the Everglades to Florida Bay including effects on biota and biotic processes—A review, *in* Davis, S.M., and Ogden, J.C., eds., Everglades—The ecosystem and its restoration: Delray Beach, Fla., St. Lucie Press, p. 117–146.
- McPherson, B.F., 1973, Water quality in the conservation areas of the central and southern Florida Flood Control District, 1970–72: U.S. Geological Survey Open-File Report 73–014, 39 p.
- McPherson, B.F., and Halley, Robert, 1996, The south Florida environment—A region under stress: U.S. Geological Survey Circular 1134, 61 p.
- McPherson, B.F., and Miller, R.L., 1987, The vertical attenuation of light in Charlotte Harbor, a shallow, subtropical estuary, southwestern Florida: *Estuarine, Coastal, and Shelf Science*, v. 25, p. 721–737.
- Meyer, M.T., Mills, M.S., and Thurman, E.M., 1993, Automated solid-phase extraction of herbicides from water for gas chromatographic-mass spectrometric analysis: *Journal of Chromatography*, v. 629, p. 55–59.
- Miles, C.J., and Pfeuffer, R.J., 1997, Pesticides in canals of south Florida: *Archives of Environmental Contamination and Toxicology*, v. 32, p. 337–345.
- Miller, R.L., and McPherson, B.F., in press, Occurrence and distribution of contaminants in bottom sediment and water of the Barron River Canal, Big Cypress National Preserve, Florida. October 1998: *Florida Scientist*, v. 64, no. 1.
- Miller, R.L., McPherson, B.F., and Haag, K.L., 1999, Water quality in the southern Everglades and Big Cypress Swamp in the vicinity of the Tamiami Trail, 1996–97: U.S. Geological Survey Water-Resources Investigations Report 99–4062, 16 p.
- Moyle, P.B., 1986, Fish introductions into North America—Patterns and ecological impact, *in* Mooney, H.A., and Drake, J.A., eds., *Ecology of biological invasions of North America and Hawaii*: New York, Springer-Verlag, p. 27–43.
- Newell, A.J., Johnson, D.W., and Allen, L.K., 1987, Niagara River Project—Fish flesh criteria for piscivorous wildlife: New York State Department of Environmental Conservation Technical Report 87–3.
- Nowell, L.H., Capel, P.D., and Dileanis, P.D., 1999, Pesticides in stream sediment and aquatic biota—Distribution, trends, and governing factors: Boca Raton, Fla., Lewis Publishers, 1001 p.
- Orem, W.H., Bates, A.L., Lerch, H.E., Corum, Margo, and Boylan, Ann, 1999, Sulfur contamination in the Everglades and its relation to mercury methylation, *in* Gerould, Sarah, and Higer, Aaron, comps., U.S. Geological Survey Program on the South Florida Ecosystem—Proceedings of South Florida Restoration Science Forum, May 17–19, 1999, Boca Raton, Florida: U.S. Geological Survey Open-File Report 99–181, 121 p.
- Parker, G.G., Ferguson, G.E., Love, S.K., and others, 1955, Water resources of southeastern Florida, with special reference to geology and ground water of the Miami area: U.S. Geological Survey Water-Supply Paper 1255, 965 p.
- Reddy, M.M., Aiken, G.R., and Schuster, P.F., 1999, Mercury-dissolved organic carbon interactions in the Florida Everglades—A field and laboratory investigation, *in* U.S. Geological Survey, Program on the south Florida ecosystem, Proceedings of South Florida Restoration Science Forum, May 17–19, 1999, Boca Raton, Fla.: U.S. Geological Survey Open-File Report 99–181, p. 86–87.
- Shahane, A.N., 1994, Pesticide detections in surface waters of Florida—Toxic substances and the hydrologic sciences: American Institute of Hydrology, p. 408–416.
- Stober, Q.J., Jones, R.D., and Scheidt, D.J., 1995, Ultra trace level mercury in the Everglades ecosystem, a multi-media canal pilot study: *Water, Air, and Soil Pollution*, v. 80, p. 991–1001.
- Swancar, Amy, 1996, Water quality, pesticide occurrence, and effects of irrigation with reclaimed water at golf courses in Florida: U.S. Geological Survey Water-Resources Investigations Report 95–4250, 85 p.
- Swift, C.C., Gilbert, C.R., Bortone, S.A., Burgess, G.H., and Yerger, R.W., 1986, Zoogeography of the freshwater fishes of the southeastern United States—Savannah River to Lake Pontchartrain, *in* Hocutt, C.H. and Wiley, E.O., eds., *The zoogeography of northern American freshwater fishes*: New York, John Wiley and Sons, p. 213–265.
- Thurman, E.M., Goolsby, D.A., Meyer, M.T., and Kolpin, D.W., 1991, Herbicides in surface waters of the Midwestern United States—The effect of the spring flush: *Environmental Science and Technology*, v. 25, p. 1794–1796.
- Thurman, E.M., Goolsby, D.A., Meyer, M.T., Mills, M.S., Pomes, M.L., and Koplin, D.W., 1992, A reconnaissance study of herbicides and their metabolites in surface water of the Midwestern United States using immunoassay and gas chromatography/mass spectrometry: *Environmental Science and Technology*, v. 26, p. 2440–2447.
- Thurman, E.M., Meyer, M.T., Pomes, M.L., Perry, C.A., and Schwab, P., 1990, Comparison of an enzyme-linked immunosorbent assay and gas chromatography/mass spectrometry for the analysis of herbicides in water: *Analytical Chemistry*, v. 62, p. 2043–2048.
- U.S. Geological Survey, 2000, Pesticides in surface and ground water of the United States: Summary of results of the National Water-Quality Assessment Program (NAWQA): accessed Aug. 18, 2000, at http://wwwwd-cascr.wr.usgs.gov/pnsp/allsum_sw.html.
- 1999, The quality of our Nation's waters: U.S. Geological Survey Circular 1225, 82 p.
- Ware, F. J., Royals, H., and Lange, T., 1990, Mercury contamination in Florida largemouth bass: Proceedings of the Annual Conference Southeastern Association of Fish and Wildlife Agencies, no. 44, p. 5–12.

APPENDIX—WATER-QUALITY DATA FROM SOUTHERN FLORIDA IN A NATIONAL CONTEXT

For a complete view of Southern Florida data and for additional information about specific benchmarks used, visit our Web site at <http://water.usgs.gov/nawqa/>. Also visit the NAWQA Data Warehouse for access to NAWQA data sets at <http://infotrek.er.usgs.gov/wdbcx/nawqa/nawqa.home>.

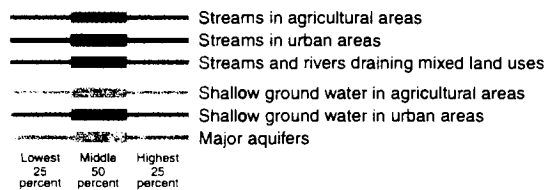
This appendix is a summary of chemical concentrations and biological indicators assessed in Southern Florida. Selected results for this Study Unit are graphically compared to results from as many as 36 NAWQA Study Units investigated from 1991 to 1998 and to national water-quality benchmarks for human health, aquatic life, or fish-eating wildlife. The chemical and biological indicators shown were selected on the basis of frequent detection, detection at concentrations above a national benchmark, or regulatory or scientific importance. The graphs illustrate how conditions associated with each land use sampled in Southern Florida compare to results from across the Nation, and how conditions compare among the several land uses. Graphs show only detected concentrations and, thus, care must be taken to evaluate detection frequencies in addition to concentrations when comparing study-unit and national results. For example, norflurazon concentrations in Southern Florida ground water in agricultural areas were similar to the national distribution, but the detection frequency was much higher (72 percent compared to 2 percent).

CHEMICALS IN WATER

Concentrations and detection frequencies, Southern Florida, 1996–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals

- ◆ Detected concentration in Study Unit
- 66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- Not measured or sample size less than two
- 12 Study-unit sample size. For ground water, the number of samples is equal to the number of wells sampled

National ranges of detected concentrations, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

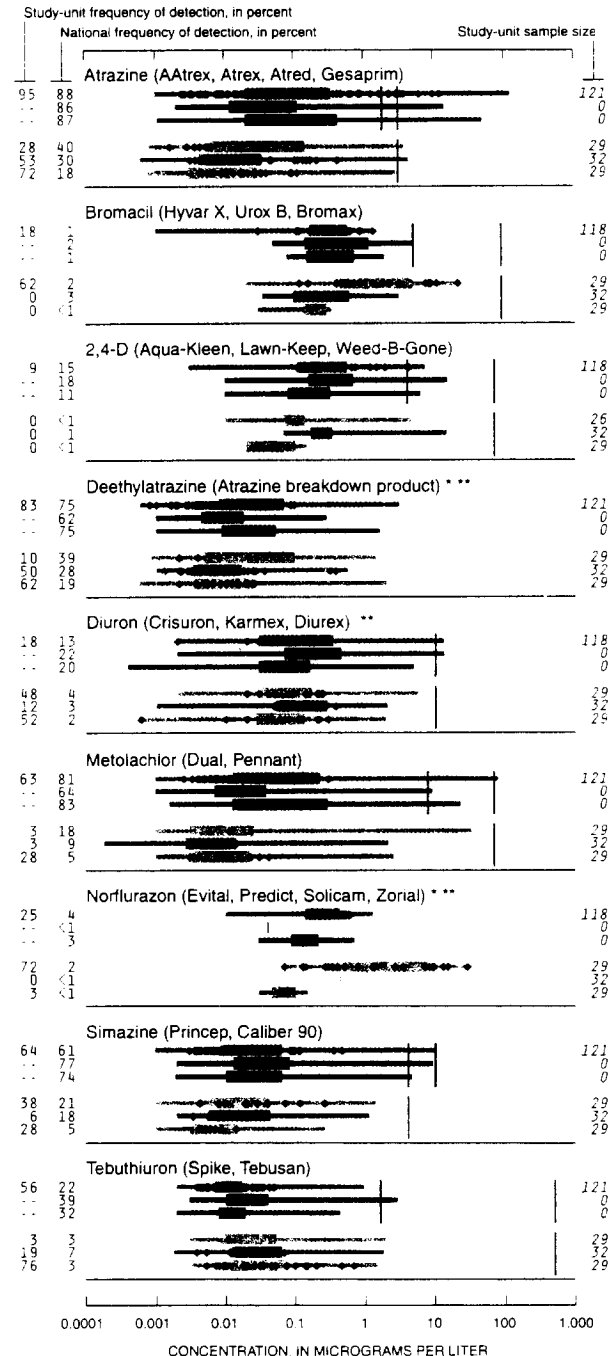


National water-quality benchmarks

National benchmarks include standards and guidelines related to drinking-water quality, criteria for protecting the health of aquatic life, and a goal for preventing stream eutrophication due to phosphorus. Sources include the U.S. Environmental Protection Agency and the Canadian Council of Ministers of the Environment

- | Drinking-water quality (applies to ground water and surface water)
- | Protection of aquatic life (applies to surface water only)
- | Prevention of eutrophication in streams not flowing directly into lakes or impoundments
- No benchmark for drinking-water quality
- No benchmark for protection of aquatic life

Pesticides in water—Herbicides



Other herbicides detected

- Alachlor (Lasso, Bronco, Lariat, Bullet) **
- Benfluralin (Balan, Benefin, Bonalan) ***
- Bentazon (Basagran, Bentazone) **
- Butylate (Sutan +, Genate Plus, Butylate) **
- Cyanazine (Bladex, Fortrol)

DCPA (Dacthal, chlorthal-dimethyl) * **
 Dicamba (Barvel, Dianat, Scotts Proturf)
 2,6-Diethylaniline (Alachlor breakdown product) * **
 EPTC (Eptam, Farmarox, Alirox) * **
 Fenuron (Fenulon, Fenidim) * **
 Linuron (Lorox, Linex, Sarclax, Linurex, Afalon) *
 Metribuzin (Lexone, Sencor)
 Napropamide (Devrinol) * **
 Prometon (Pramitol, Princep) **
 Pronamide (Kerb, Propyzamid) **
 Propanil (Stam, Stampede, Wham) * **
 Terbacil (Sinbar) **
 Triclopyr (Garlon, Grandstand, Redeem, Remedy) * **

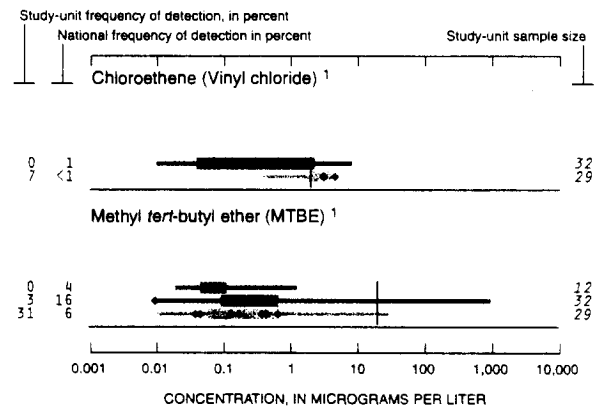
Herbicides not detected

Acetochlor (Harness Plus, Surpass) * **
 Acifluorfen (Blazer, Tackle 2S) **
 Bromoxynil (Buctril, Brominal) *
 Chloramben (Amiben, Amilon-WP, Vegiben) **
 Clopyralid (Stinger, Lontrel, Transline) * **
 2,4-DB (Butyrac, Butoxone, Embutox Plus, Embutone) * **
 Dacthal mono-acid (Dacthal breakdown product) * **
 Dichlorprop (2,4-DP, Seritox 50, Lentemul) * **
 Dinoseb (Dinosebe)
 Ethalfurailin (Sonalan, Curbit) * **
 Fluometuron (Flo-Met, Cotoran) **
 MCPA (Rhomene, Rhonox, Chiptox)
 MCPB (Thistrol) * **
 Molinate (Ordram) * **
 Neburon (Neburea, Neburyl, Noruben) * **
 Oryzalin (Surflan, Dirimal) * **
 Pebulate (Tillam, PEBC) * **
 Pendimethalin (Pre-M, Prowl, Stomp) * **
 Picloram (Grazon, Tordon)
 Propachlor (Ramrod, Satecid) **
 Prophanil (Tuberite) **
 2,4,5-T **
 2,4,5-TP (Silvex, Fenoprop) **
 Thiobencarb (Bolero, Saturn, Benthiocarb) * **
 Triallate (Far-Go, Avadex BW, Tri-allate) *
 Trifluralin (Treflan, Gowan, Tri-4, Trific)

Azinphos-methyl (Guthion, Gusathion M) *
 Diazinon (Basudin, Diazatol, Neocidol, Knox Out)
 Disulfoton (Disyston, Di-Syston) **
 Fonofos (Dyfonate, Capfos, Cudgel, Tycap) **
 alpha-HCH (alpha-BHC, alpha-lindane) **
 gamma-HCH (Lindane, gamma-BHC)
 3-Hydroxycarbofuran (Carbofuran breakdown product) * **
 Methiocarb (Slug-Geta, Grandslam, Mesurol) * **
 Oxamyli (Vydate L, Pratt) **
 Parathion (Roethyl-P, Alkron, Panthion, Phoskil) *
 cis-Permethrin (Ambush, Astro, Pounce) * **
 Phorate (Thimet, Granutox, Geomet, Rampart) * **
 Propargite (Comite, Omite, Ornamite) * **
 Terbufos (Contraven, Counter, Pilarfox) **

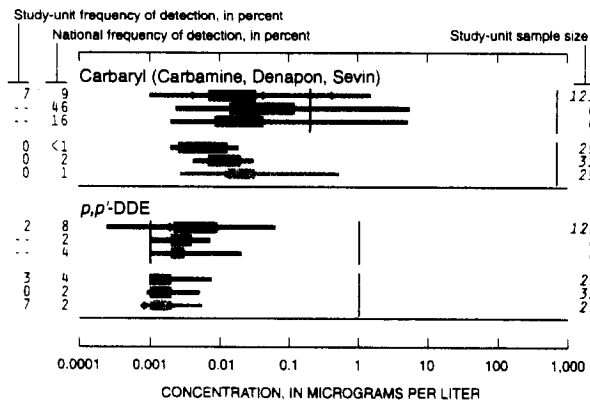
Volatile organic compounds (VOCs) in ground water

These graphs represent data from 16 Study Units, sampled from 1996 to 1998



¹ Many of the samples in this study were diluted prior to laboratory analysis and therefore the actual detection frequency may be larger than the value listed.

Pesticides in water—Insecticides



Other insecticides detected

Carbofuran (Furadan, Curaterr, Yaltox)
 Chlorpyrifos (Brodan, Dursban, Lorsban)
 Dieldrin (Panoram D-31, Octalox, Compound 497)
 Ethoprop (Mocap, Ethoprophos) * **
 Malathion (Malathion)
 Methomyl (Lanox, Lannate, Acinate) **
 Methyl parathion (Pennacp-M, Folidol-M) **
 Propoxur (Baygon, Blattanex, Unden, Propotox) * **

Insecticides not detected

Aldicarb (Temik, Ambush, Pounce)
 Aldicarb sulfone (Standak, aldoxycarb)
 Aldicarb sulfoxide (Aldicarb breakdown product)

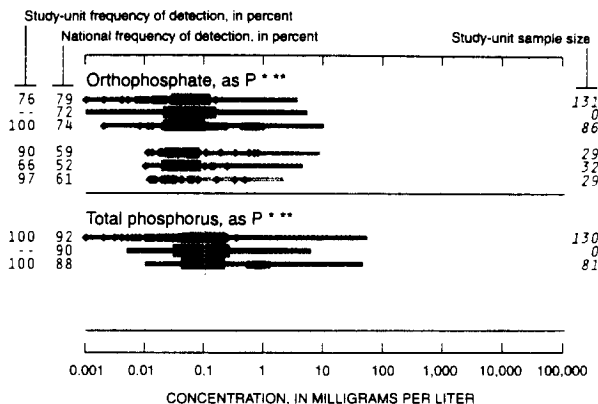
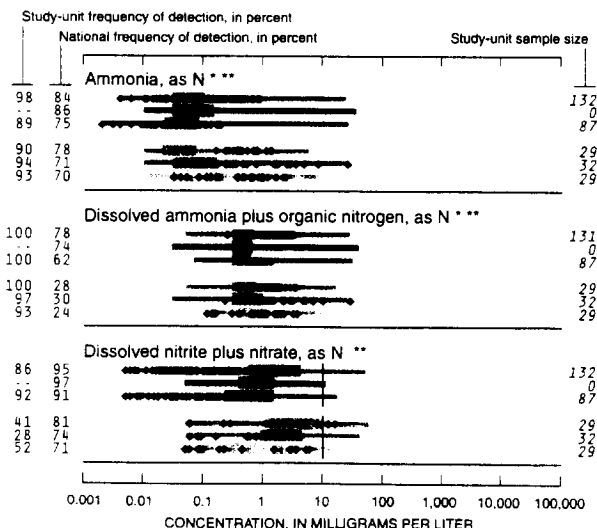
Other VOCs detected

tert-Amylmethylether (tert-amyl methyl ether (TAME)) *
 Benzene
 Bromochloromethane (Methylene chlorobromide)
 Bromodichloromethane (Dichlorobromomethane)
 2-Butanone (Methyl ethyl ketone (MEK)) *
 Carbon disulfide *
 1-Chloro-2-methylbenzene (o-Chlorotoluene)
 Chlorobenzene (Monochlorobenzene)
 Chlorodibromomethane (Dibromochloromethane)
 Chloromethane (Methyl chloride)
 Dibromomethane (Methylene dibromide) *
 1,3-Dichlorobenzene (m-Dichlorobenzene)
 1,4-Dichlorobenzene (p-Dichlorobenzene)
 Dichlorodifluoromethane (CFC 12, Freon 12)
 1,1-Dichloroethane (Ethylidene dichloride) *
 1,1-Dichloroethene (Vinylidene chloride)
 trans-1,2-Dichloroethene ((E)-1,2-Dichloroethene)
 cis-1,2-Dichloroethene ((Z)-1,2-Dichloroethene)
 Dichloromethane (Methylene chloride)
 1,2-Dichloropropane (Propylene dichloride)
 Diethyl ether (Ethyl ether) *
 1,2-Dimethylbenzene (o-Xylene)
 1,3 & 1,4-Dimethylbenzene (m-&p -Xylene)
 1,4-Epoxy butane (Tetrahydrofuran, Diethylene oxide) *
 Ethylbenzene (Phenylethane)
 p-Isopropyltoluene (p-Cymene) *
 4-Methyl-2-pentanone (Methyl isobutyl ketone (MIBK)) *
 Methylbenzene (Toluene)
 Naphthalene
 2-Propanone (Acetone) *
 Tetrachloroethene (Perchloroethene)
 Tribromomethane (Bromofom)
 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon 113) *
 Trichloroethene (TCE)
 Trichloromethane (Chlorofom)
 1,2,4-Trimethylbenzene (Pseudocumene) *

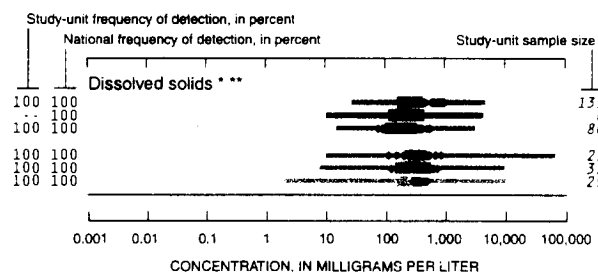
VOCs not detected

- Bromobenzene (Phenyl bromide) *
- Bromoethene (Vinyl bromide) *
- Bromomethane (Methyl bromide)
- n*-Butylbenzene (1-Phenylbutane) *
- sec*-Butylbenzene *
- tert*-Butylbenzene *
- 3-Chloro-1-propene (3-Chloropropene) *
- 1-Chloro-4-methylbenzene (*p*-Chlorotoluene)
- Chloroethane (Ethyl chloride) *
- 1,2-Dibromo-3-chloropropane (DBCP, Nemagon)
- 1,2-Dibromoethane (Ethylene dibromide, EDB)
- trans*-1,4-Dichloro-2-butene ((*Z*)-1,4-Dichloro-2-butene) *
- 1,2-Dichlorobenzene (*o*-Dichlorobenzene)
- 1,2-Dichloroethane (Ethylene dichloride)
- 2,2-Dichloropropane *
- 1,3-Dichloropropane (Trimethylene dichloride) *
- trans*-1,3-Dichloropropene ((*E*)-1,3-Dichloropropene)
- cis*-1,3-Dichloropropene ((*Z*)-1,3-Dichloropropene)
- 1,1-Dichloropropene *
- Diisopropyl ether (Diisopropylether (DIPE)) *
- Ethenylbenzene (Styrene)
- Ethyl methacrylate *
- Ethyl *tert*-butyl ether (Ethyl-*t*-butyl ether (ETBE)) *
- 1-Ethyl-2-methylbenzene (2-Ethyltoluene) *
- Hexachlorobutadiene
- 1,1,1,2,2,2-Hexachloroethane (Hexachloroethane)
- 2-Hexanone (Methyl butyl ketone (MBK)) *
- Iodomethane (Methyl iodide) *
- Isopropylbenzene (Cumene) *
- Methyl acrylonitrile *
- Methyl-2-methacrylate (Methyl methacrylate) *
- Methyl-2-propenoate (Methyl acrylate) *
- 2-Propenenitrile (Acrylonitrile)
- n*-Propylbenzene (Isocumene) *
- 1,1,2,2-Tetrachloroethane *
- 1,1,1,2-Tetrachloroethane
- Tetrachloromethane (Carbon tetrachloride)
- 1,2,3,4-Tetramethylbenzene (Prehnitene) *
- 1,2,3,5-Tetramethylbenzene (Isodurene) *
- 1,2,4-Trichlorobenzene
- 1,2,3-Trichlorobenzene *
- 1,1,1-Trichloroethane (Methylchloroform)
- 1,1,2-Trichloroethane (Vinyl trichloride)
- Trichlorofluoromethane (CFC 11, Freon 11)
- 1,2,3-Trichloropropane (Allyl trichloride)
- 1,2,3-Trimethylbenzene (Hemimellitene) *
- 1,3,5-Trimethylbenzene (Mesitylene) *

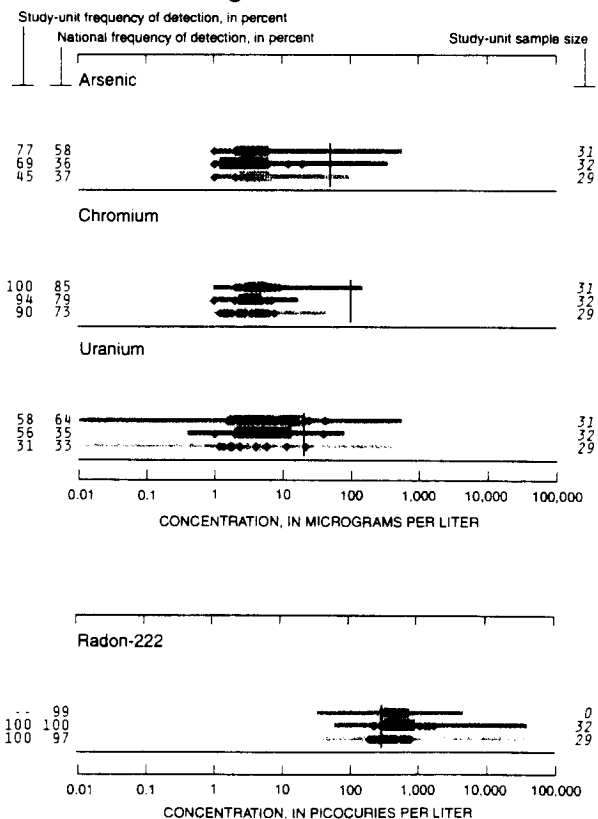
Nutrients in water



Dissolved solids in water



Trace elements in ground water



Other trace elements detected

Lead
Selenium
Zinc

Trace elements not detected

Cadmium

CHEMICALS IN FISH TISSUE AND BED SEDIMENT

Concentrations and detection frequencies, Southern Florida, 1996-98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals. Study-unit frequencies of detection are based on small sample sizes; the applicable sample size is specified in each graph

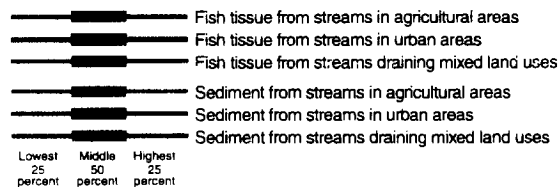
◆ Detected concentration in Study Unit

66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency

-- Not measured or sample size less than two

12 Study-unit sample size

National ranges of concentrations detected, by land use, in 36 NAWQA Study Units, 1991-98—Ranges include only samples in which a chemical was detected

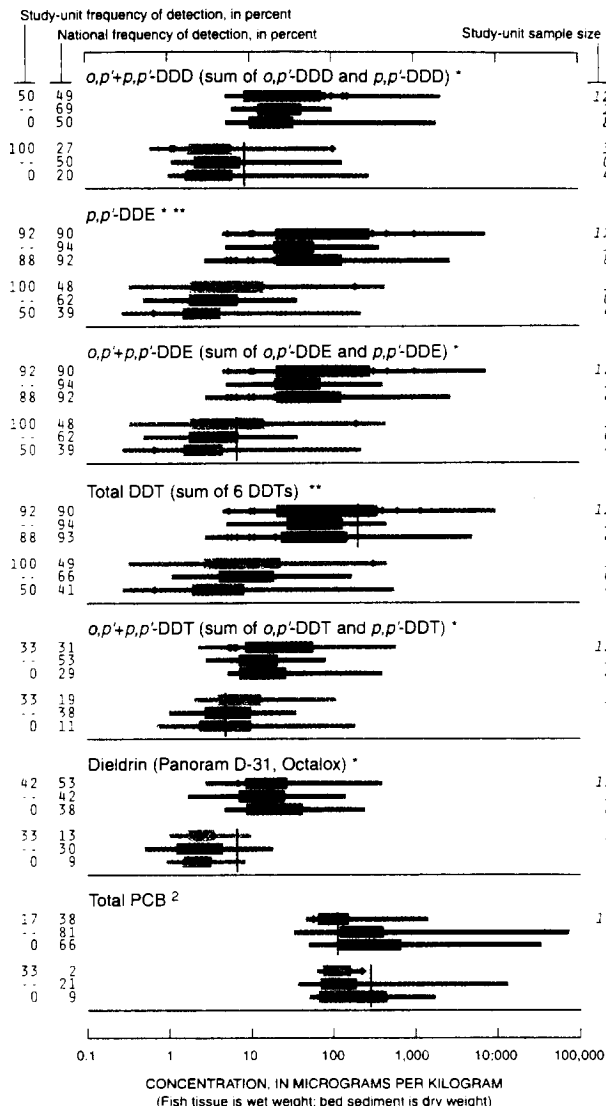
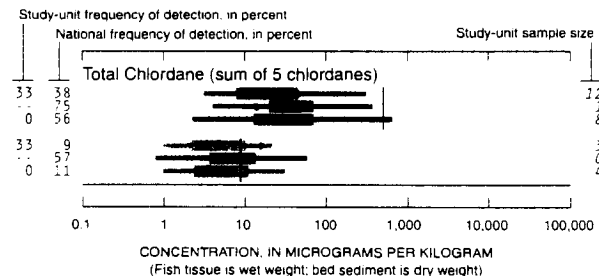


National benchmarks for fish tissue and bed sediment

National benchmarks include standards and guidelines related to criteria for protection of the health of fish-eating wildlife and aquatic organisms. Sources include the U.S. Environmental Protection Agency, other Federal and state agencies, and the Canadian Council of Ministers of the Environment

- | Protection of fish-eating wildlife (applies to fish tissue)
- | Protection of aquatic life (applies to bed sediment)
- No benchmark for protection of fish-eating wildlife
- ** No benchmark for protection of aquatic life

Organochlorines in fish tissue (whole body) and bed sediment



² The national detection frequencies for total PCB in sediment are biased low because about 30 percent of samples nationally had elevated detection levels compared to this Study Unit. See <http://water.usgs.gov/nawqa/> for additional information.

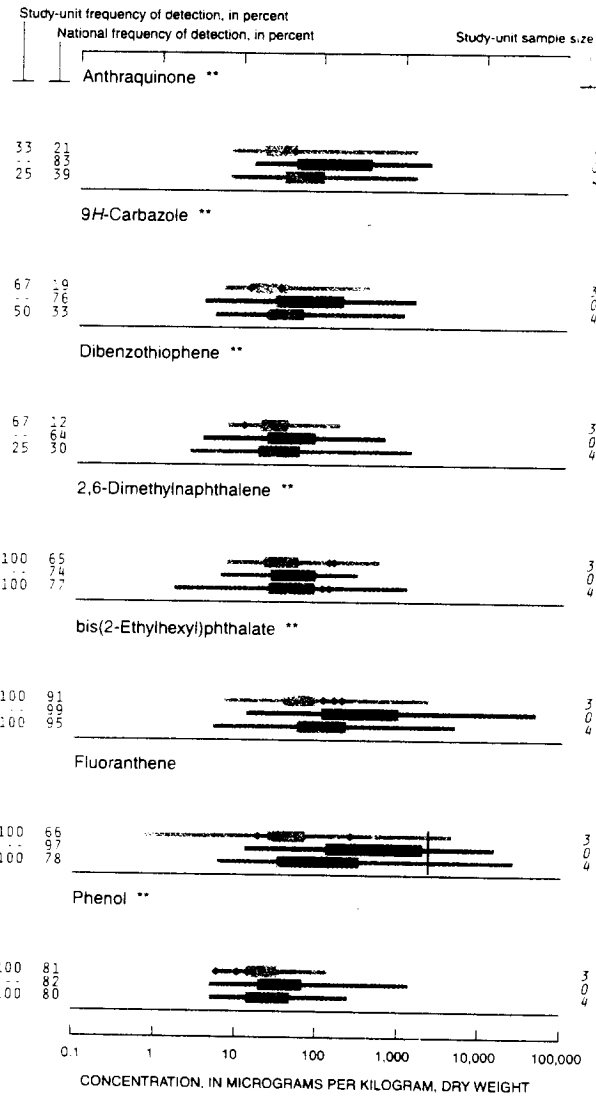
Other organochlorines detected

Dieldrin+aldrin (sum of dieldrin and aldrin) **
Endosulfan I (alpha-Endosulfan, Thiodan) ***
Mirex (Dechlorane) **

Organochlorines not detected

Chloroneb (Chloronebe, Demosan) ***
DCPA (Dacthal, chlorthal-dimethyl) ***
Endrin (Endrine)
gamma-HCH (Lindane, gamma-BHC, Gammexane) *
total-HCH (sum of alpha-HCH, beta-HCH, gamma-HCH, and delta-HCH) **
Heptachlor epoxide (Heptachlor breakdown product) *
Heptachlor+heptachlor epoxide (sum of heptachlor and heptachlor epoxide) **
Hexachlorobenzene (HCB) **
Isodrin (Isodrine, Compound 711) ***
p,p'-Methoxychlor (Mariate, methoxychlore) ***
o,p'-Methoxychlor ***
Pentachloroanisole (PCA) ***
cis-Permethrin (Ambush, Astro, Pounce) ***
trans-Permethrin (Ambush, Astro, Pounce) ***
Toxaphene (Camphechlor, Hercules 3956) ***

Semivolatile organic compounds (SVOCs) in bed sediment



Other SVOCs detected

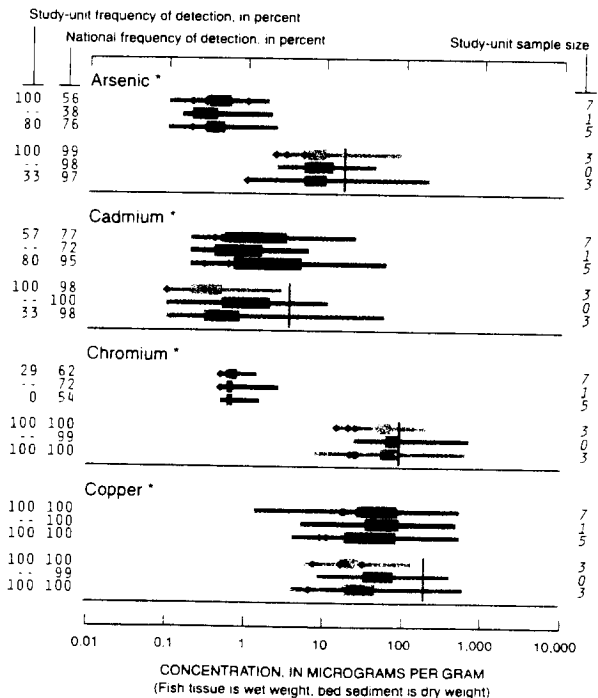
Acenaphthene
 Acenaphthylene
 Acridine **
 Anthracene
 Benz[*a*]anthracene
 Benzo[*a*]pyrene
 Benzo[*b*]fluoranthene **
 Benzo[*ghi*]perylene **
 Benzo[*k*]fluoranthene **
 Butylbenzylphthalate **
 Chrysene
p-Cresol **
 Di-*n*-butylphthalate **
 Di-*n*-octylphthalate **
 Dibenz[*a,h*]anthracene
 Diethylphthalate **
 1,6-Dimethylnaphthalene **
 Dimethylphthalate **
 9H-Fluorene (Fluorene)

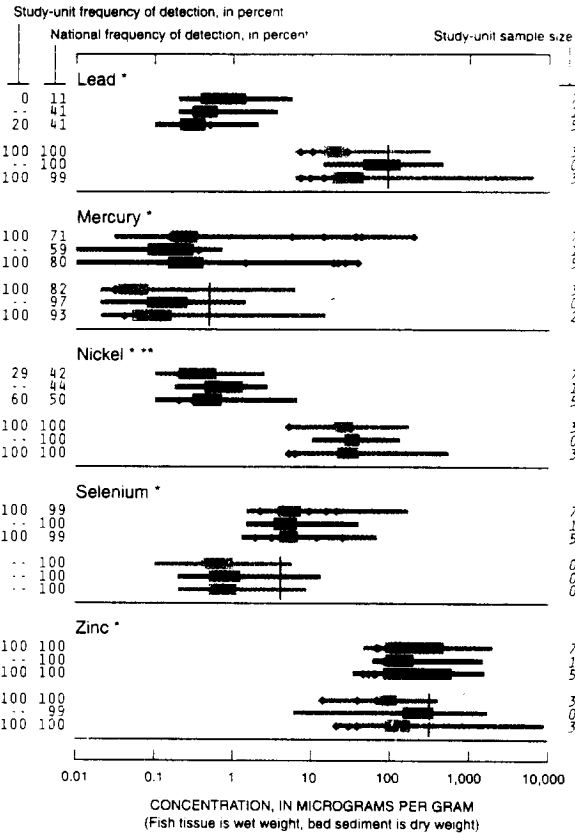
Indeno[1,2,3-*cd*]pyrene **
 1-Methyl-9H-fluorene **
 2-Methylantracene **
 4,5-Methylenephenanthrene **
 1-Methylphenanthrene **
 1-Methylpyrene **
 Naphthalene
 Phenanthrene
 Phenanthridine **
 Pyrene
 2,3,6-Trimethylnaphthalene **

SVOCs not detected

C8-Alkylphenol **
 Azobenzene **
 Benzo[*c*]cinnoline **
 2,2-Biquinoline **
 4-Bromophenyl-phenylether **
 4-Chloro-3-methylphenol **
 bis(2-Chloroethoxy)methane **
 2-Chloronaphthalene **
 2-Chlorophenol **
 4-Chlorophenyl-phenylether **
 1,2-Dichlorobenzene (*o*-Dichlorobenzene) **
 1,3-Dichlorobenzene (*m*-Dichlorobenzene) **
 1,4-Dichlorobenzene (*p*-Dichlorobenzene) **
 1,2-Dimethylnaphthalene **
 3,5-Dimethylphenol **
 2,4-Dinitrotoluene **
 2-Ethyl-naphthalene **
 Isophorone **
 Isoquinoline **
 Nitrobenzene **
N-Nitrosodi-*n*-propylamine **
N-Nitrosodiphenylamine **
 Pentachloronitrobenzene **
 Quinoline **
 1,2,4-Trichlorobenzene **

Trace elements in fish tissue (livers) and bed sediment





BIOLOGICAL INDICATORS

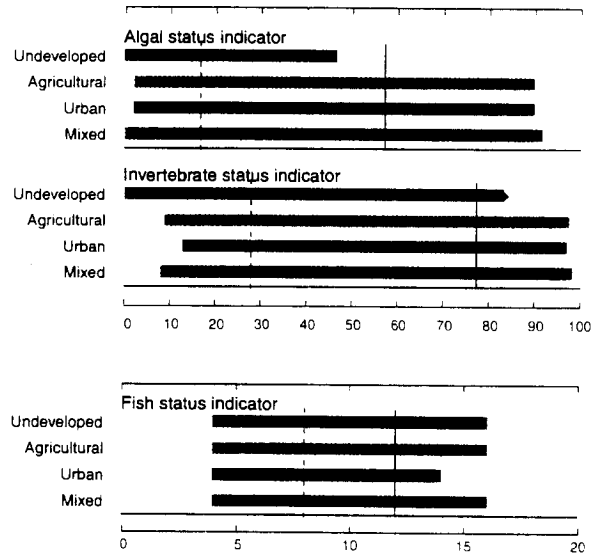
Higher national scores suggest habitat disturbance, water-quality degradation, or naturally harsh conditions. The status of algae, invertebrates (insects, worms, and clams), and fish provide a record of water-quality and stream conditions that water-chemistry indicators may not reveal. **Algal status** focuses on the changes in the percentage of certain algae in response to increasing siltation, and it often correlates with higher nutrient concentrations in some regions. **Invertebrate status** averages 11 metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. **Fish status** sums the scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that increase in association with water-quality degradation.

Biological indicator value, Southern Florida, by land use, 1996-98

- ◆ Biological status assessed at a site

National ranges of biological indicators, in 16 NAWQA Study Units, 1994-98

- Streams in undeveloped areas
- Streams in agricultural areas
- Streams in urban areas
- Streams in mixed-land-use areas
- 75th percentile
- - - 25th percentile



A COORDINATED EFFORT

Coordination among agencies and organizations is an integral part of the NAWQA Program. We thank the following agencies and organizations who contributed data used in this report or participated in the Study Unit liaison committee.

Federal Organizations

National Park Service
National Oceanic and Atmospheric Administration
U.S. Army Corps of Engineers
U.S. Department of Agriculture
U.S. Environmental Protection Agency
U.S. Fish and Wildlife Service
National Marine Fisheries Service

State Agencies

Florida Marine Research Institute
Florida Department of Agriculture and Consumer Services
Florida Department of Environmental Protection
Florida Game and Freshwater Fish Commission
Florida Geological Survey
Florida Park Service
South Florida Water Management District
Southwest Florida Water Management District

Private Organizations

Nature Conservancy

Native American

Miccosukee Tribe of Indians of Florida
Seminole Tribe of Florida

Universities

University of Miami
University of Florida
Florida International University

Local Agencies

City of Naples
Metropolitan Dade County
Palm Beach County
Broward County
Collier County

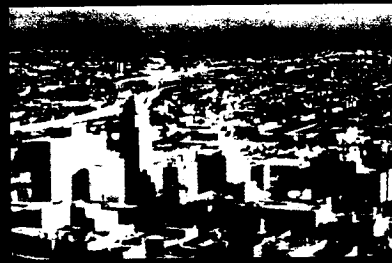
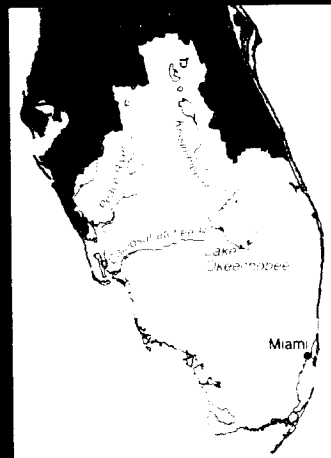
Special thanks to U.S. Geological Survey employees for their contributions:

Bruce Bernard, lead technician for SOFL, John Byrnes, Mark Zucker for data collection. Becky Deckard, Teresa Embry, Pat Mixson, and Ron Spencer for production of the report, including editing, layout, and illustrations. Michael Meyers for his contribution on pesticide degradation.

Appreciation also is extended to those individuals and agencies that reviewed this report: Edward Oaksford, U.S. Geological Survey, Michael Thurman, U.S. Geological Survey, Richard Pfeuffer, South Florida Water Management District.

NAWQA

National Water-Quality Assessment (NAWQA) Program Southern Florida



McPherson and others—Water Quality in Southern Florida
U.S. Geological Survey Circular 1207

ISBN 0-607-95413-2



9 780607 954135

R0024397

Water Quality in the Mississippi Embayment

Mississippi, Louisiana, Arkansas, Missouri, Tennessee, and Kentucky
1995–98



POINTS OF CONTACT AND ADDITIONAL INFORMATION

The companion Web site for NAWQA summary reports:

<http://water.usgs.gov/nawqa/>

MISE contact and Web site:

USGS State Representative
U.S. Geological Survey
Water Resources Division
308 South Airport Road
Pearl, MS 39208-6649
e-mail: dc.ms@usgs.gov
<http://ms.water.usgs.gov/misenawqa/>

National NAWQA Program:

Chief, NAWQA Program
U.S. Geological Survey
Water Resources Division
12201 Sunrise Valley Drive, M.S. 413
Reston, VA 20192
<http://water.usgs.gov/nawqa/>

Other NAWQA summary reports

River Basin Assessments

Albemarle-Pamlico Drainage Basin (Circular 1157)
Allegheny and Monongahela River Basins (Circular 1202)
Apalachicola-Chattahoochee-Flint River Basin (Circular 1164)
Central Arizona Basins (Circular 1213)
Central Columbia Plateau (Circular 1144)
Central Nebraska Basins (Circular 1163)
Connecticut, Housatonic, and Thames River Basins (Circular 1155)
Eastern Iowa Basins (Circular 1210)
Georgia-Florida Coastal Plain (Circular 1151)
Hudson River Basin (Circular 1165)
Kanawha-New River Basins (Circular 1204)
Lake Erie-Lake Saint Clair Drainages (Circular 1203)
Las Vegas Valley Area and the Carson and Truckee River Basins
(Circular 1170)
Lower Illinois River Basin (Circular 1209)
Long Island-New Jersey Coastal Drainages (Circular 1201)
Lower Susquehanna River Basin (Circular 1168)
Ozark Plateaus (Circular 1158)
Potomac River Basin (Circular 1166)
Puget Sound Basin (Circular 1216)
Red River of the North Basin (Circular 1169)

Rio Grande Valley (Circular 1162)
Sacramento River Basin (Circular 1215)
San Joaquin-Tulare Basins (Circular 1159)
Santee River Basin and Coastal Drainages (Circular 1206)
South-Central Texas (Circular 1212)
South Platte River Basin (Circular 1167)
Southern Florida (Circular 1207)
Trinity River Basin (Circular 1171)
Upper Colorado River Basin (Circular 1214)
Upper Mississippi River Basin (Circular 1211)
Upper Snake River Basin (Circular 1160)
Upper Tennessee River Basin (Circular 1205)
Western Lake Michigan Drainages (Circular 1156)
White River Basin (Circular 1150)
Willamette Basin (Circular 1161)

National Assessments

The Quality of Our Nation's Waters—Nutrients and Pesticides (Circular 1225)

Front cover: Photograph of the Black Swamp, Cache River Basin, near Gregory, Arkansas.

Back cover: Left, cotton grows on the banks of bayous in the Yazoo Basin; center, many of the rivers in the bootheel of Missouri have been channelized; right, soybeans are a major crop in the Mississippi Embayment Study Unit.

Photographs in this report were all taken by members of the MISE NAWQA Study Unit, U.S. Geological Survey.

Water Quality in the Mississippi Embayment, Mississippi, Louisiana, Arkansas, Missouri, Tennessee, and Kentucky, 1995–98

By Barbara A. Kleiss, Richard H. Coupe, Gerard J. Gonthier, *and* Billy G. Justus

U.S. DEPARTMENT OF THE INTERIOR
BRUCE BABBITT, Secretary

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

2000

Free on application to the
U.S. Geological Survey
Information Services
Box 25286 Federal Center
Denver, CO 80225

Or call: 1-888-ASK-USGS

Library of Congress Cataloging-in-Publications Data

Water quality in the Mississippi Embayment, Mississippi, Louisiana, Arkansas, Missouri, Tennessee, and Kentucky, 1995-98 / by Barbara A. Kleiss...[et al.].

p. cm. -- (U.S. Geological Survey Circular ; 1208)

Includes bibliographical references.

ISBN 0-607-95414-0 (alk. paper)

1. Water quality--Mississippi Embayment. I. Kleiss, Barbara A., 1958- II. Geological Survey (U.S.) III. Series.

TD225.M64 W375 2000
363.739'42'09633--dc21

00-049462

R0024401

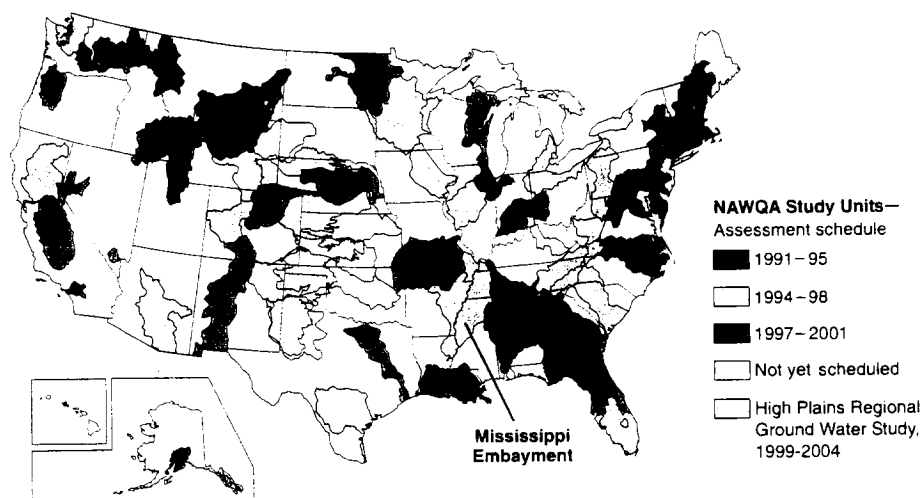
CONTENTS

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM	IV
SUMMARY OF MAJOR FINDINGS	1
INTRODUCTION TO THE MISSISSIPPI EMBAYMENT	3
MAJOR FINDINGS	7
Nutrient Contributions to the Mississippi River System	7
NATIONAL PERSPECTIVE—National Comparison of Total Nitrogen in Streams.....	8
Pesticides Commonly Found in Mississippi Embayment Surface Water	10
NATIONAL PERSPECTIVE—Pesticides in Streams Across the United States.....	11
Tertiary Aquifers Have High-Quality Drinking Water	12
Few Pesticides Detected in Memphis Shallow Aquifers.....	12
Water-Quality Differences in Geological Subunits of the Alluvial Aquifer.....	13
NATIONAL PERSPECTIVE—Radon Levels in Ground Water Low in Mississippi Embayment Study Unit	14
Aquatic Communities Show Environmental Stress	15
NATIONAL PERSPECTIVE—Comparison of Mississippi Embayment Aquatic Communities to National Results.....	17
Organochlorine Pesticides Persist in Fish Tissue	18
Pesticides in Air and Rain Samples from Agricultural and Urban Sites	20
Integration of Biological, Geological, and Chemical Data Improves Understanding of Aquatic Systems	22
STUDY UNIT DESIGN.....	24
GLOSSARY	26
REFERENCES	28
APPENDIX —WATER-QUALITY DATA FROM THE MISSISSIPPI EMBAYMENT IN A NATIONAL CONTEXT	30

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

THIS REPORT summarizes major findings about water quality in the Mississippi Embayment that emerged from an assessment conducted between 1995 and 1998 by the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program. Water quality is discussed in terms of local and regional issues and compared to conditions found in all 36 NAWQA study areas, called Study Units, assessed to date. Findings are also explained in the context of selected national benchmarks, such as those for drinking-water quality and the protection of aquatic organisms. The NAWQA Program was not intended to assess the quality of the Nation's drinking water, such as by monitoring water from household taps. Rather, the assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water monitoring programs. The comparisons made in this report to drinking-water standards and guidelines are only in the context of the available untreated resource. Finally, this report includes information about the status of aquatic communities and the condition of in-stream habitats as elements of a complete water-quality assessment.

Many topics covered in this report reflect the concerns of officials of State and Federal agencies, water-resource managers, and members of stakeholder groups who provided advice and input during the Mississippi Embayment assessment. Basin residents who wish to know more about water quality in the areas where they live will find this report informative as well.



THE NAWQA PROGRAM seeks to improve scientific and public understanding of water quality in the Nation's major river basins and ground-water systems. Better understanding facilitates effective resource management, accurate identification of water-quality priorities, and successful development of strategies that protect and restore water quality. Guided by a nationally consistent study design and shaped by ongoing communication with local, State, and Federal agencies, NAWQA assessments support the investigation of local issues and trends while providing a firm foundation for understanding water quality at regional and national scales. The ability to integrate local and national scales of data collection and analysis is a unique feature of the USGS NAWQA Program.

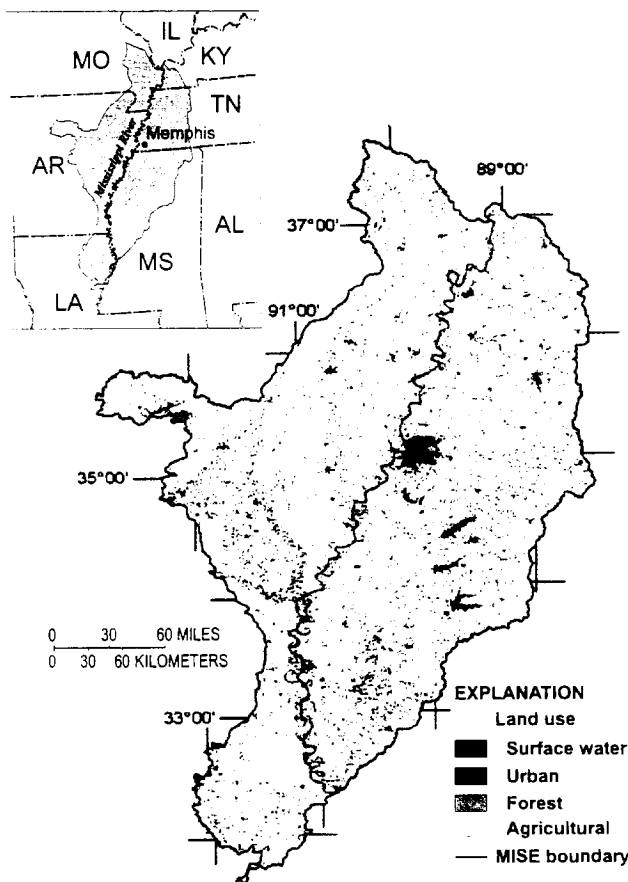
The Mississippi Embayment is one of 51 water-quality assessments initiated since 1991, when the U.S. Congress appropriated funds for the USGS to begin the NAWQA Program. As indicated on the map, 36 assessments have been completed, and 15 more assessments will conclude in 2001. Collectively, these assessments cover about one-half of the land area of the United States and include water resources that are available to more than 60 percent of the U.S. population.

SUMMARY OF MAJOR FINDINGS

Stream and River Highlights

The climate, rainfall, soil type, and surficial geology of the Mississippi Embayment (MISE) Study Unit strongly influence land use in the basin and subsequently influence water quality. About 62 percent of the Study Unit is used for agriculture. In areas of intensive row-crop production, as much as 90 percent of the land is used for agriculture. This influence from agricultural land use, with additional contributions from urban areas, has resulted in streams that often have high turbidities, mixtures of pesticides, and degraded riparian habitat. Biological communities in the streams commonly are stressed. However, human activities on the Earth's surface seem to have a limited effect on the ground-water resources, which supply the vast majority of the region's drinking water.

- Herbicides frequently were detected in streams draining agricultural or mixed land-use basins; insecticides were detected less often. Pesticides in over 60 percent of samples collected from these streams exceeded aquatic-life guidelines. Insecticides frequently were detected in samples from the urban stream; diazinon and chlorpyrifos were detected in every sample, usually in concentrations above aquatic-life guidelines.
- Nitrogen concentrations in the MISE generally were in the middle range of the national data, whereas total phosphorus concentrations were in the 67th to 93d percentile. The phosphorus concentrations in the Study Unit probably were related to many factors, such as rainfall amounts, soils, and artificial drainage of agricultural fields. No sample exceeded the guidelines and standards for nitrate or ammonia, but most exceeded the U.S. Environmental Protection Agency's (USEPA) goal of 0.1 mg/L (milligram per liter) of phosphorus for the prevention of plant nuisances in streams.
- Although the sale of the organochlorine insecticide DDT was discontinued in 1972, DDT and metabolites (chemicals resulting from the breakdown of DDT) were widespread within the MISE. DDT, or one of its metabolites, was found in every fish tissue sample collected and was found in 67 percent of the streambed-sediment samples. Detectable levels of a metabolite of DDT were measured in 14 percent of surface-water samples.
- Although volatile and semivolatile organic compounds often were detected in urban stream water and in bed sediment, they were rarely at levels of concern.
- Aquatic organisms present in the MISE streams were typical of those found in impacted or degraded streams. Fish



The Mississippi Embayment (MISE) Study Unit is an approximately 49,800-square-mile area in the six States of Arkansas, Kentucky, Louisiana, Mississippi, Missouri, and Tennessee. Land use in the MISE is principally agricultural. Approximately 62 percent of the study area is agricultural, 33 percent is forested, and 5 percent represents other land uses. The land use in some of the smaller drainage basins sampled is greater than 90 percent agricultural.

communities in most of the streams were dominated by fish tolerant of poor water quality conditions. The aquatic insects and algal communities generally were tolerant of turbid, silty conditions.

- Methyl parathion, a metabolite of DDT, and several other pesticides were detected in air and rain samples collected in an agricultural area and in the urban area of Jackson, Mississippi.

Major Influences on Streams and Rivers

- Runoff from agricultural and urban areas
- Drainage modifications and channelization of streams
- Modification or elimination of riparian habitat

Selected Stream-Quality Indicators

	Small Streams			Major Rivers
	Urban	Agricultural	Undeveloped	Mixed Land Uses
Pesticides ¹			—	
Total phosphorus ²			—	
Nitrate ³			—	
Trace elements ⁴	—		—	
Organo-chlorines ⁵	—		—	
Volatile organics ⁶		—	—	—
Semivolatile organics ⁷	—		—	

Percentage of samples with concentrations **equal to or greater than** a health-related national guideline for drinking water, aquatic life, or water-contact recreation; or above a national goal for preventing excess algal growth

Percentage of samples with concentrations **less than** a health-related national guideline for drinking water, aquatic life or water-contact recreation; or below a national goal for preventing excess algal growth

— Not assessed

¹ Insecticides, herbicides, and pesticide metabolites, sampled in water.

² Total phosphorus, sampled in water.

³ Nitrate (as nitrogen), sampled in water.

⁴ Arsenic, mercury, and metals sampled in sediment.

⁵ DDT and PCBs sampled in fish tissue.

⁶ Solvents, refrigerants, fumigants, and gasoline compounds sampled in water.

⁷ By-products of fossil-fuel combustion; components of coal and crude oil sampled in sediment.

Ground-Water Highlights

Ground-water quality in the Mississippi Embayment Study Unit generally is very good. Ground water in the deep Tertiary aquifers, which supply most of the region's drinking water, generally is isolated from surface activities by thick "confining layers" of clays. Surface activities influence ground water where shallow deposits cover the hills in the eastern part of the Study Unit and in the Memphis shallow aquifers more than in the deeper aquifers. The abundant ground water in the alluvial aquifer of the Mississippi River valley is near the land surface but is covered by dense clays.

- Pesticides, such as atrazine, simazine, and metolachlor, were detected most frequently in the ground water in the shallow deposits that cover the hills in the eastern part of the Study Unit and in ground water underlying urban areas.

Bentazon, molinate, and fluometuron were the pesticides most frequently detected in the alluvial aquifer. Atrazine and dieldrin were detected one time each in shallow urban wells at levels above the drinking-water standards and guidelines.

- Nutrient concentrations in the ground water in the MISE generally were low. All nitrate concentrations were below the USEPA drinking-water standard of 10 mg/L.
- Radon is naturally occurring and was detected in almost every well sampled. Concentrations above the USEPA-proposed drinking water standard of 300 picocuries per liter were found in water from only 16 of 109 wells. These levels are low, relative to levels detected in other NAWQA Study Units.
- Volatile organic compounds (VOCs) were detected in ground water throughout the Study Unit; however, concentrations were well within drinking-water standards. The most frequently detected VOCs were 1,2,4-trimethylbenzene and carbon disulfide.

Major Influences on Ground Water

- Ground water is commonly protected from surface activities by thick, regional clay layers.

Selected Ground-Water Quality Indicators

	Shallow Ground Water		Supply Wells	
	Urban	Agricultural	Domestic	Public
Pesticides ¹				
Nitrate ²				
Radon				
Volatile organics ³		—	—	

Percentage of samples with concentrations **equal to or greater than** a health-related national guideline for drinking water, aquatic life, or water-contact recreation; or above a national goal for preventing excess algal growth

Percentage of samples with concentrations **less than** a health-related national guideline for drinking water, aquatic life or water-contact recreation; or above a national goal for preventing excess algal growth

— Percentage of samples with **no detection**

— Not assessed

¹ Insecticides, herbicides, and pesticide metabolites, sampled in water.

² Nitrate (as nitrogen), sampled in water.

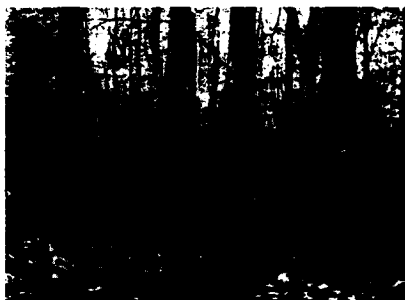
³ Solvents, refrigerants, fumigants, and gasoline compounds sampled in water.

INTRODUCTION TO THE MISSISSIPPI EMBAYMENT

In 1821, while painting a peregrine falcon, John James Audubon described the Yazoo River, the largest river wholly contained within the Mississippi Embayment Study Unit, as “a beautiful stream of transparent water, covered by thousands of geese and ducks and filled with fish” (Smith, 1954). Since that time, the bottomland hardwood forests that covered the Mississippi River Alluvial Plain have been cleared for agricultural use of the rich alluvial soils for the production of cotton, soybeans, rice, and corn. This clearing of the land has exposed the fine alluvial soils to erosion. Over time, the clays, along with nutrients and agricultural chemicals sorbed to the clay surfaces, were washed into the rivers and streams, thus greatly changing the water quality of the area.

Physiography and Ecoregions

Within the Mississippi Embayment (MISE) Study Unit, the surficial geology is the underlying controlling factor for the physiography, land use, biological communities, and water quality of the area. Therefore, the areas defined as physiographic regions (Fenneman, 1938) strongly correspond to



Much of the Mississippi Embayment Study Unit was bottomland hardwood forests and wetlands well into the 20th century.



FIG. 1. Peregrine Falcon (Kinglet's Hawk)

John James Audubon's painting of the peregrine falcon that he worked on while visiting the Yazoo River area in 1821. (Reprinted courtesy of the National Audubon Society.)

the related ecoregions defined by Omernik in 1987 (fig. 1).

About 57 percent of the MISE Study Unit lies within the Mississippi Alluvial Plain physiographic province and ecoregion. This area has been dominated by the flow and flooding of the Mississippi River during the past 2 million years or more. The Mississippi Alluvial Plain is an area of little topographic relief with an average slope of about 0.5 foot per mile toward the Gulf of Mexico. One of the distinct features of the alluvial plain is the formation of natural levees along the banks of the rivers, and the associated backswamp deposits that are dominated by dense alluvial clays and historically have supported extensive wetland areas. These clays have created low permeability soils, which limit the ability of rainwater to infiltrate the ground surface and may cause runoff from agricultural fields to rapidly enter rivers and streams. These clays also seem to limit the susceptibility of the ground water to surface activities in intense agricultural areas.

Thirty-five percent of the remainder of the Study Unit lies in

the Gulf Coastal Plains physiographic province, which includes the area identified as the Mississippi Valley Loess Plains and Southeastern Plains ecoregions (fig. 1). The Gulf Coastal Plains are separated from the eastern edge of the Mississippi Alluvial Plain by the Loess Hills, which extend most of the length of the Study Unit. These hills are made of wind-blown silts, rise a few hundred feet above the Mississippi Alluvial Plain, and average about 15 miles in width. The remaining part of the Gulf Coastal Plains uplands and Southeastern Plains ecoregion generally is rolling to hilly with low to moderate topographic relief. The soils are composed, in part, of silts and are more permeable than the alluvial clays; there are indications that this allows for the downward infiltration of precipitation. This may partly protect the streams and rivers from compounds carried in runoff but may make the ground water slightly more susceptible to surface contamination. These coarser soils on steeper slopes are more erodible than alluvial soils, and large amounts of soil from the Gulf Coastal Plains uplands have

eroded into the Mississippi Alluvial Plain.

In the west, the Study Unit abuts small areas of the Ozark Highlands. Limited sampling was done in these areas during this project.

The land surface generally slopes toward the Mississippi River from both the eastern and western sides of the Study Unit and to the south toward the Gulf of Mexico. Thus, nearly all of the activities in this Study Unit that influence water quality ultimately influence the water quality of the Mississippi River and the Gulf of Mexico.

Geology

The Mississippi Alluvial Plain is in the northern part of the Missis-

issippi Embayment, a geologic structural trough in which the underlying crust of the Earth forms a deep valley. Large rivers, such as the Mississippi, Arkansas, and Ohio Rivers, have flowed through this region, carved the surface, and deposited clay, silt, sand, and gravel, collectively called alluvium. During the past 2 million years, up to 300 feet of alluvium has filled this valley. The alluvium can be grouped into three major units: the Pleistocene Prairie Complex, Pleistocene valley trains, and the Holocene alluvium (see fig. 7 for map; Autin and others, 1991; Saucier, 1994).

The Prairie Complex is older than the Pleistocene valley trains and the Holocene alluvium. Sauc-

ier (1994) suggested that the Prairie Complex was deposited between about 120,000 years ago and the time of the greatest extent of the last glacier, about 18,000 years ago. The Pleistocene valley trains were mostly deposited during two time periods, between about 60,000 and 25,000 years ago and during the waning phase of the latest glacial period between 18,000 and 10,000 years ago. Glacial outwash (melting) flowing from north to south provided enough energy to cause a braided stream depositional environment to form in the Lower Mississippi River Valley during this time. By about 9,000 years ago, the rate of glacial outwash in the Lower Mississippi River Valley declined, and valley train deposi-

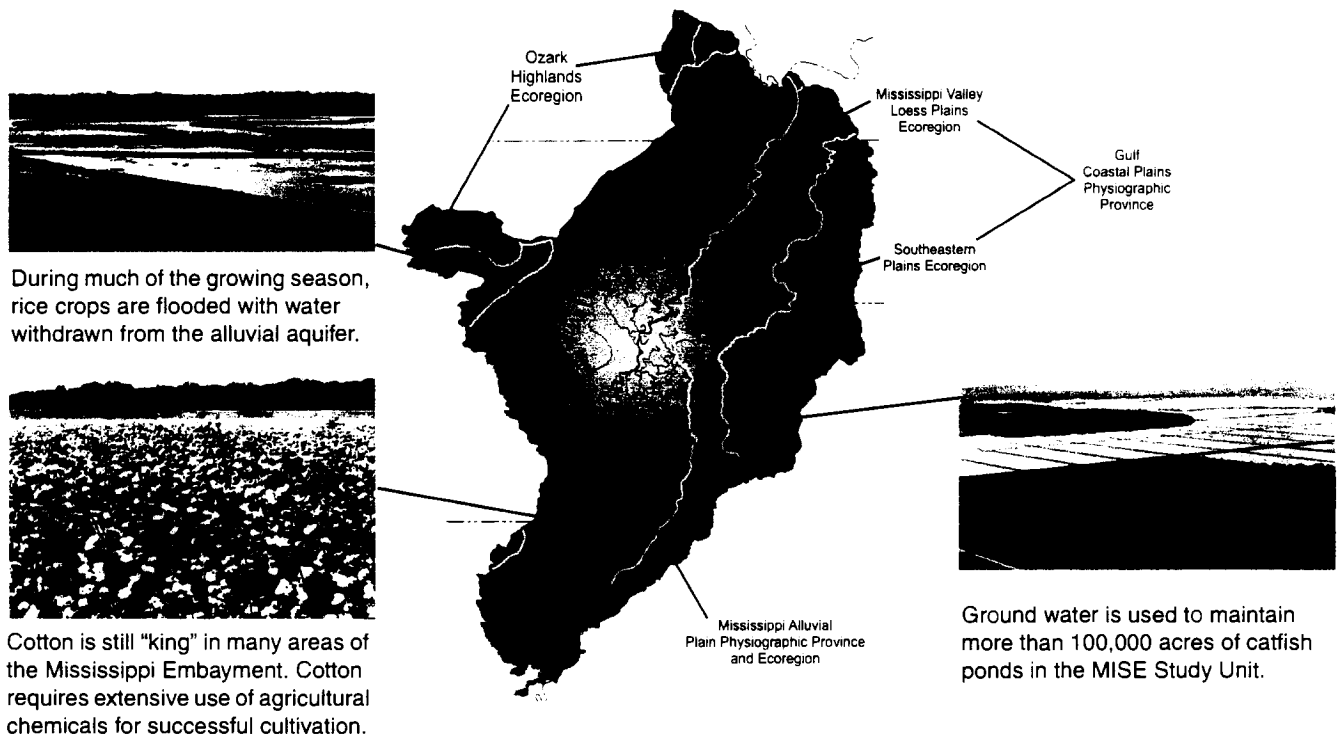


Figure 1. Boundaries for Fenneman's (1938) physiographic regions are very similar to Omernik's Level III ecoregions (1987), at least in part because the surficial geology is a controlling factor in the MISE Study Unit. The only major metropolitan area in the Study Unit is Memphis, Tennessee. The area has many rivers, as well as several large river systems, including the Yazoo and St. Francis Rivers and parts of the White and Arkansas Rivers. Major crops include soybeans, cotton, rice, and corn. Catfish farms are a major component of the landscape as well as a principal user of ground water.

tion ceased. The braided stream depositional process of the Pleistocene epoch was replaced by the lower energy meander stream depositional process of the Holocene epoch near major rivers, such as the Mississippi and Arkansas Rivers. (See fig. 2 for more explanation.) Autin and others (1991) reported that the depositional transition from Pleistocene valley trains (braided streams) to Holocene alluvium (meander streams) started near Baton Rouge, Louisiana, around 12,000 years ago and migrated northward to near Cairo, Illinois, by 9,000 years ago.

The Pleistocene valley train deposits generally have a coarser grain size than the Holocene alluvium. Also, water well drillers' logs indicate that the clay and silt layer near the surface is thicker in the Holocene alluvium, whereas the underlying sand and gravel layer (alluvial aquifer) is thicker in the Pleistocene valley train deposits.

Hydrogeology

Two principal aquifer systems provide drinking-water supplies in the Mississippi Embayment—the Tertiary and the alluvial aquifers (fig. 3).

Tertiary Aquifers

The geologic groups associated with the deep Tertiary aquifers are the Midway, Wilcox, Claiborne, and the Jackson groups. The deep Tertiary aquifers sampled in this study are thick sand deposits within the Wilcox and Claiborne groups. The names of the aquifers, from youngest to oldest, include the Cockfield, Sparta, Winona-Tallahatta, Memphis, Meridian-upper Wilcox, and Wilcox.



Figure 2. Although the photograph on the left was recently taken of a stream in the Western United States, its braided condition is representative of what streams in the Mississippi Valley may have looked like during the Pleistocene geologic period. These high-energy systems allow sand and gravel carried by the stream to be deposited in the flood plain. The photograph on the right depicts a classic meandering stream. Streams like this are low-energy systems and primarily deposit clay, silt, and fine sand in the flood plains adjacent to the streams. This depositional pattern is present today and has been the dominant form of deposition in the Lower Mississippi River Valley during the last 9,000 to 12,000 years. These differences in depositional environments appear to influence the chemistry of the ground water, the bioaccumulation of pesticides, and biological communities.

The natural regional flow of ground water in the Mississippi Embayment in the Tertiary aquifers is from the outcrop areas in the upper Gulf Coastal Plain, laterally along the aquifers toward the embayment axis, and then upward through overlying confining units and aquifers to the surface of the Mississippi Alluvial Plain (Grubb, 1986; Ackerman, 1989). Pumping

in the deep Tertiary aquifers has caused recharge rates to increase in the outcrop and production areas of the aquifer (Williamson and others, 1990).

Alluvial Aquifers

The Mississippi River confining unit is composed of the upper silt and clay of the Quaternary allu-

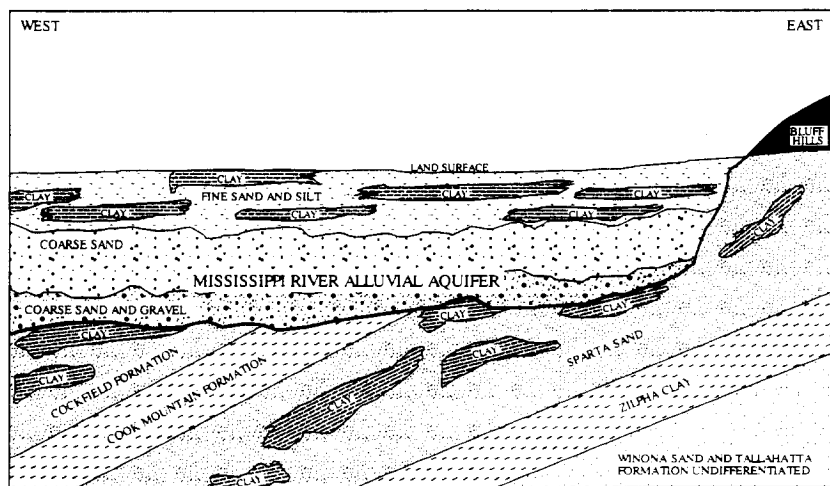


Figure 3. Generalized geohydrologic section of the Mississippi River alluvial aquifer and underlying Tertiary aquifers (from Arthur, 1994).

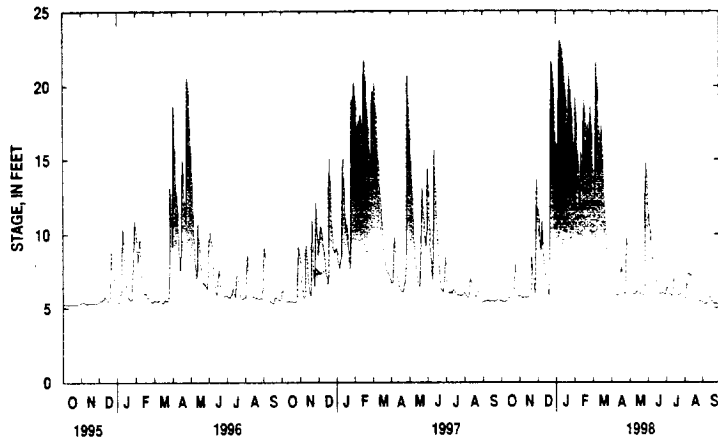


Figure 4. This hydrograph is representative of streams in the Mississippi Alluvial Plain. Often streams in this Study Unit flood rapidly, remain at high levels for long periods, and have very low streamflows in the fall. During the years of this project, 1996 was drier than usual, and 1997 and 1998 were wetter than usual.

vium, whereas the Mississippi River alluvial aquifer is composed of the lower sand and gravel of the Quaternary alluvium (Boswell and others, 1968; Ackerman, 1989). Overlying silt and clay of the confining unit impedes recharge into the alluvial aquifer. Confining unit thickness generally ranges from 10 to 50 feet and generally increases from north to south within the MISE Study Unit. The thickness of the alluvial aquifer ranges from 60 to 140 feet. Wells screened in the alluvial aquifer typically yield between 1,000 and 2,000 gallons per minute (Whitfield, 1975). Prior to development, ground-water flow is believed to have been generally from the older adjacent and underlying aquifers toward the alluvial aquifer (Williamson and others, 1990).

Climate

Climate in the MISE Study Unit varies from humid, temperate in the northern part to humid, subtropical in the southern part. This warm climate results in a long growing season and few killing

frosts, which influences the types of crops that can be grown and the amount of pesticides that generally are applied. Mean annual precipitation ranges from about 48 inches per year in the northern part of the Study Unit to 56 inches per year in the southern part. Precipitation generally is greatest in April and least in October but is distributed fairly evenly throughout the year. This causes minor drought conditions to occur frequently during the

summer, when the evapotranspiration rate is higher than the precipitation rate. These conditions also result in streams which flood rapidly, remain at high levels for long periods of time, and have low flows in the fall (fig. 4).

Water Use

In general, about three times as much ground water is used compared to surface water in the MISE Study Unit (fig. 5). During the summer months, both ground and surface water are used for irrigating crops. Most (in excess of 7 billion gallons per day) of the irrigation water is withdrawn from the alluvial aquifer. This aquifer is also used for domestic drinking water, aquaculture (primarily for catfish ponds), power production, and other commercial and industrial needs. Ground water, primarily from the Tertiary aquifers, is used for public supply. The principal use of surface water is for power production where it is used for cooling water for electric power generation. The second largest use of surface water is for irrigation.

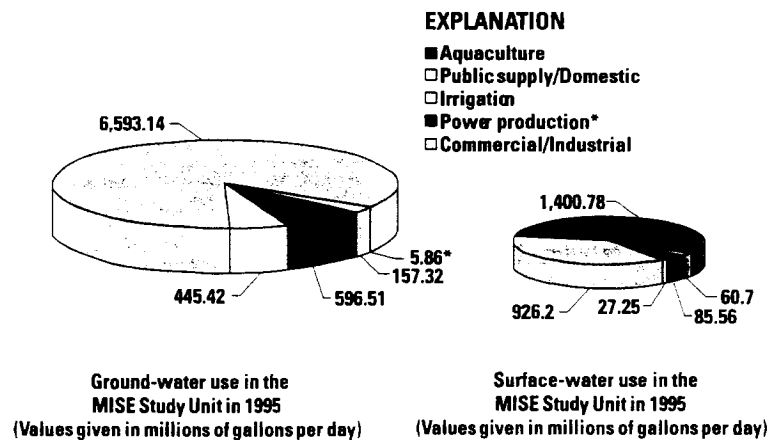


Figure 5. Ground-water use in the Mississippi Embayment (MISE) Study Unit is dominated by irrigation usage. Surface water is also used for irrigation, but more is used for cooling water for electrical power production.

MAJOR FINDINGS

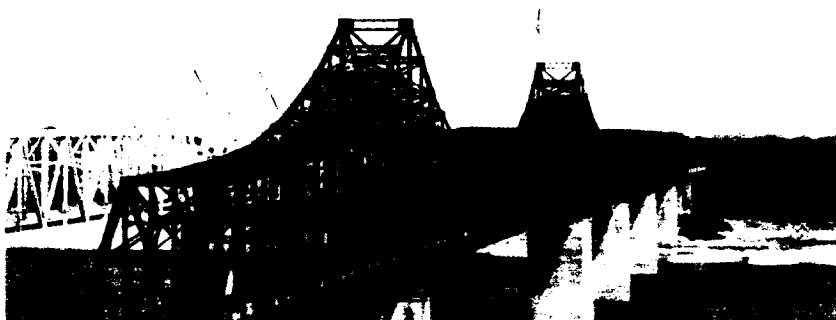
Nutrient Contributions to the Mississippi River System

Although nitrogen and phosphorus, as well as silica and other nutrients, are natural and important parts of a healthy ecosystem, severe water-quality problems can arise if an ecosystem becomes enriched, or overloaded, with nutrients. In recent years, scientists have become aware of a large area of low dissolved oxygen that develops off the coast of Louisiana and Texas each summer. The extent and duration of this area of low dissolved oxygen has been related to the amount of nutrients, especially nitrogen, and freshwater flowing from the Mississippi River into the Gulf of Mexico (Goolsby and others, 1999). The proximity of the MISE Study Unit to the Gulf and the use of nitrogen fertilizer in the agricultural areas of the Study Unit, especially the Yazoo River Basin, have led to speculation that the surface waters of the Study Unit may be contributing a disproportionate amount of nitrogen and phosphorus to the Mississippi River and ultimately to the Gulf of Mexico.

In the MISE Study Unit investigations have shown that concentrations of nutrients (except total phosphorus) are higher in the Mississippi River at Vicksburg, Mississippi than near the mouth of the Yazoo River (table 1). The annual load of nitrogen and phosphorus from the Yazoo River for the 1996–97 calendar years, while significant, was only a small percentage of the load carried by the Mississippi River (Coupe, 1998).

Water-Quality Standards

Concentrations of nitrogen and phosphorus were measured from weekly to at least monthly at nine



The Yazoo River, the river with the largest drainage area wholly contained in the Mississippi Embayment Study Unit, enters the Mississippi River just north of these bridges at Vicksburg, Mississippi.

stream-sampling sites in the MISE Study Unit. Nitrate concentrations never exceeded the drinking-water standard of 10 mg/L in any sample, and ammonia concentrations did not exceed aquatic-life guidelines. However, the USEPA goal of 0.1 mg/L or less total phosphorus for streams not entering reservoirs was exceeded in every sample from the urban stream and in more than 50 percent of the samples from five streams located in the Mississippi Alluvial Plain. Samples from the

streams located in the Gulf Plains exceeded the recommended goal of 0.1 mg/L or less total phosphorus in less than 50 percent of the samples.

Comparison of Nitrogen and Phosphorus in Streams in the Mississippi River Basin

The yield of nitrogen (mass per unit area), from streams in the MISE Study Unit during 1995–96 was compared to the average yield during 1980–96 from streams in

Table 1. Concentrations of nutrients near the mouth of the Yazoo River compared to the Mississippi River at Vicksburg, Mississippi.

[Concentrations are in milligrams per liter]

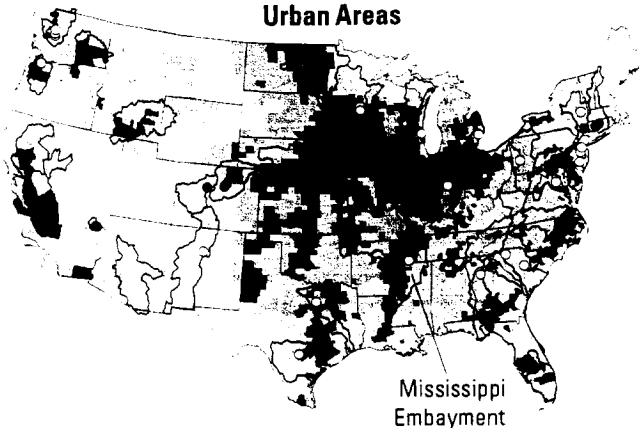
Constituent	Site	Year	Maximum	Minimum	Mean
Total nitrogen	Yazoo River	1996–97	3.3	0.57	1.3
	Mississippi River	1984–93	3.8	1.1	2.3
Nitrate as N	Yazoo River	1996–97	1.2	0.16	0.45
	Mississippi River	1984–93	2.7	0.70	1.5
Total phosphorus	Yazoo River	1996–97	0.89	0.12	0.26
	Mississippi River	1984–93	0.38	0.04	0.16
Orthophosphate as P	Yazoo River	1996–97	0.10	0.01	0.043
	Mississippi River	1984–93	0.13	0.02	0.058



National Comparison of Total Nitrogen in Streams

TOTAL NITROGEN IN STREAMS

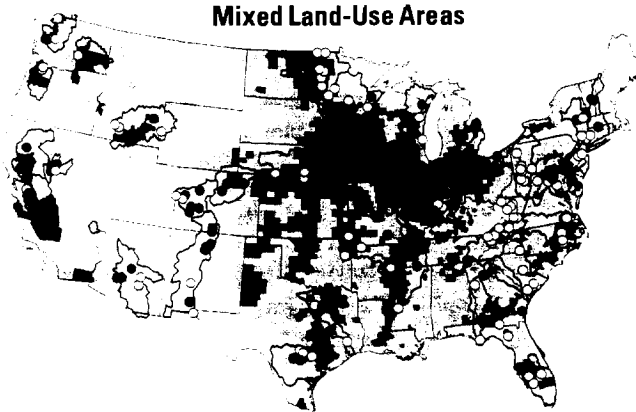
Urban Areas



Nitrogen is a natural and important component of a healthy stream; however, too much nitrogen can lead to degraded stream-water quality, affecting both the aquatic ecosystem and its use as a recreational or drinking-water source for humans. The sources of nitrogen in surface water are many and include atmospheric deposition, municipal and industrial wastewater, and fixation of nitrogen from the atmosphere by plants and some species of algae.

By far, the biggest source of nitrogen in an agricultural setting, such as the Mississippi Embayment (MISE) Study Unit, is from the application of fertilizer to crops. For most of the Study Unit, the average annual total nitrogen input from fertilizer, manure, and the atmosphere combined is greater than 25 pounds per acre. Most of the agriculturally productive Midwest receives the same amount.

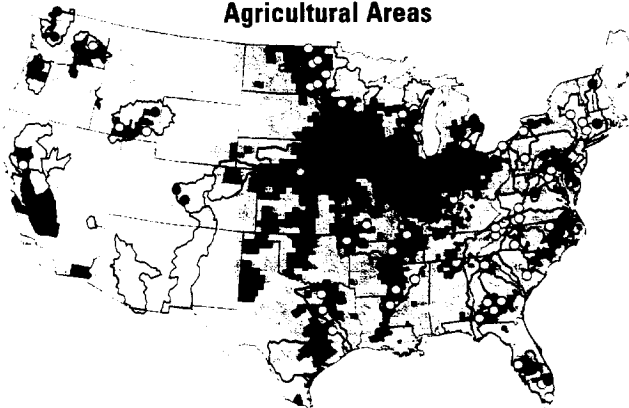
Mixed Land-Use Areas



The average annual concentration of total nitrogen from agricultural and mixed land-use streams in the MISE Study Unit is in the medium range, whereas nationally, most streams that drain areas with greater than 25 pounds per acre of nitrogen input are in the high range. The lower concentrations of total nitrogen in the MISE Study Unit may be due to the milder climate that increases microbial activity in the winter and to the increased uptake of nitrogen by vegetation during the longer growing season.

Nationally, the average annual total nitrogen concentrations in urban streams, including the one urban site in the Mississippi Embayment Study Unit, tend to fall into the "medium" classification.

Agricultural Areas



EXPLANATION

Average annual concentration of total nitrogen —
in milligrams per liter

- Highest (greater than 2.9)
- Medium (0.64 to 2.9)
- Lowest (less than 0.64)

Average annual total nitrogen input —
in pounds per acre, by county, for 1995–98.
Inputs are from fertilizer, manure, and the atmosphere

- Greater than 25 pounds per acre
- 6 to 25 pounds per acre
- Less than 6 pounds per acre

the Mississippi River Basin. These data indicate that the yield of nitrogen from the MISE Study Unit was less than the average yield from streams in intensive agricultural areas of the Midwest, but more than from streams in the drier West or in the less agricultural Upper Mississippi River Basin. The flow-weighted mean total nitrogen concentrations for streams in the MISE Study Unit were generally in the 50th to 60th percentile for all data collected in the national NAWQA Program (372 stream sites). The exception was the mean nitrogen concentration at the smallest mixed land-use site which was near the 20th percentile, nationally.

The yields of total phosphorus generally were higher in the MISE Study Unit than from most other areas in the Mississippi River Basin, and the percentile ranking of

the flow-weighted mean total phosphorus concentration generally was quite high (67th to 93d). Again, the exception was the smallest mixed land-use site, where the mean total phosphorus concentration was near the 40th percentile. These high phosphorus yields were somewhat unexpected, as the soils in the MISE Study Unit, while fertile, do not contain excessive amounts of phosphorus. Also, phosphorus is used less as a fertilizer in the MISE Study Unit than in many parts of the Midwest (Battaglin and Goolsby, 1995), and due to the rural nature of the MISE, there are few significant point sources. One hypothesis for the high yields and concentrations of phosphorus in the MISE involves a combination of factors, such as soils, rainfall, and agricultural drainage. The sediment in the rivers of the MISE Study

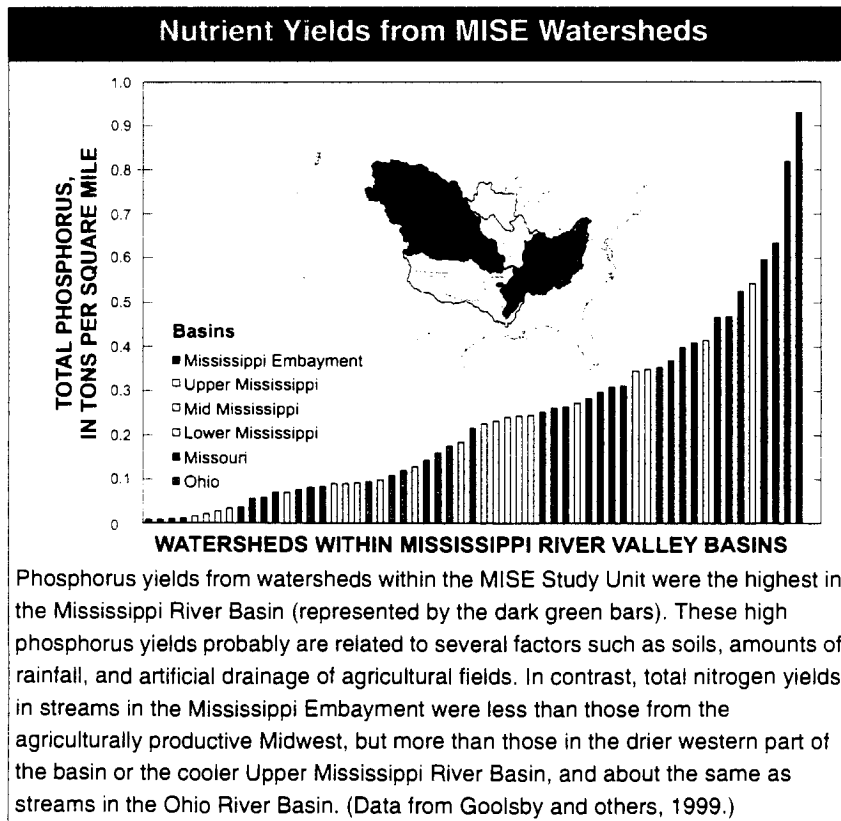
Unit is composed of fine, clay-sized particles to which phosphorus can sorb. Heavy rainfalls in the Study Unit increase the potential for erosion and the movement of these fine clay-sized particles from agricultural fields into the streams. Additionally, because of the large amount of rain, the tight clays that decrease infiltration of water, and the relatively flat terrain, much of the Study Unit has artificial drainage to expedite the movement of water. Most of this artificial drainage is surface drainage, which has been shown to decrease nitrate concentrations but to increase total phosphorus concentrations.

The Effects of Land Use and Geology on Nutrient Concentrations and Yields

Generally, total nitrogen and phosphorus concentrations and yields were higher in streams with predominantly agricultural land use in the Mississippi Alluvial Plain, but the highest nutrient concentrations and yields were from the urban stream. However, one stream located in the Gulf Plains in an area with no urban land use and only a moderate amount of agricultural land use had comparatively high total nitrogen and phosphorus yields. The high yields in this stream are reflective of the steep topography of the area and channelization of the stream for flood-control purposes.

More details on nutrients in the Yazoo River can be found in the report: Coupe, R.H., 1998, Concentrations and loads of nitrogen and phosphorus in the Yazoo River, northwestern Mississippi, 1996-97: U.S. Geological Survey Water-Resources Investigations Report 98-4219, 17 p.

The report also can be downloaded at: <http://ms.water.usgs.gov/misenawqa/>

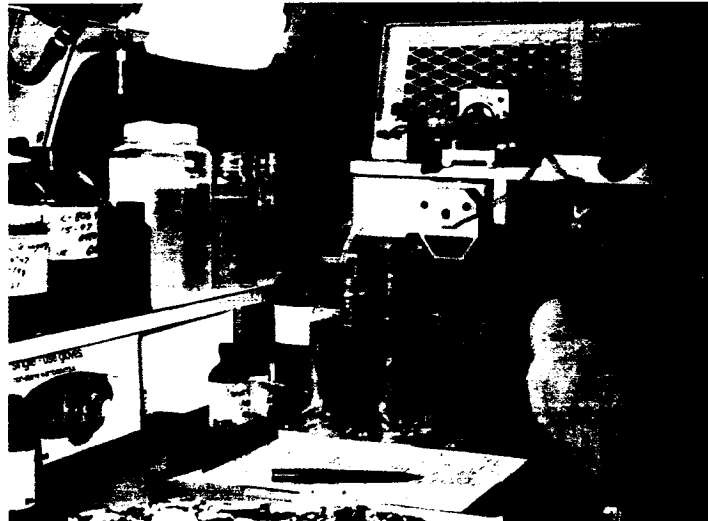


Pesticides Commonly Found in Mississippi Embayment Surface Water

The occurrence and temporal distribution of more than 80 pesticides and pesticide metabolites were determined at five stream-sampling sites from 1996 to 1998 in the MISE Study Unit. More than 230 stream samples were collected and analyzed. The five rivers sampled included three rivers with small, primarily agricultural watersheds; one river with a small, urban watershed; and one large river (the Yazoo River) with mixed land use (row-crop agriculture, forest, pasture, and a small amount of urban). Pesticides, usually herbicides, frequently were detected in water samples from all five rivers sampled. Aquatic-life guidelines were frequently exceeded in the urban stream and occasionally exceeded in all of the rivers sampled in the MISE Study Unit.

Agricultural Streams

The pesticides detected in the rivers that drain the agricultural watersheds in the MISE Study Unit



Water samples are filtered and processed in a mobile laboratory immediately after sample collection.

showed distinct seasonal patterns that corresponded to the type of crops grown in the basin and the use of pesticides on those crops. For instance, the highest concentration of the pre-emergent herbicide atrazine frequently was found early in the growing season (April-May) corresponding to its application prior to the planting of corn and grain sorghum (fig. 6). The highest concentrations of herbicides that are used on other crops (cotton and

rice) with later planting dates, or herbicides that are used after the crop has emerged from the ground, such as fluometuron and molinate, were detected later in the growing season (June–July). The concentrations of most of these herbicides were well below any acute toxicity; however, the long-term effects of chronic exposure to low levels of multiple herbicides are not well known.

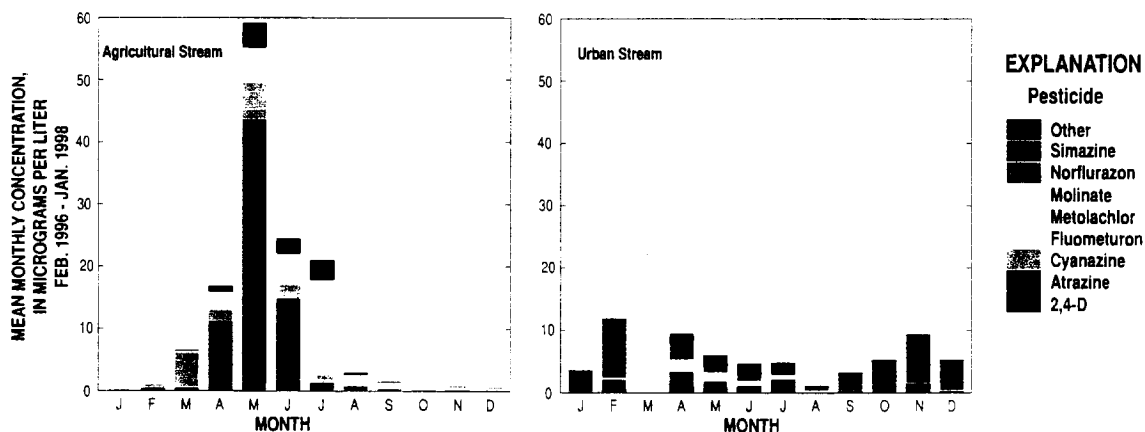
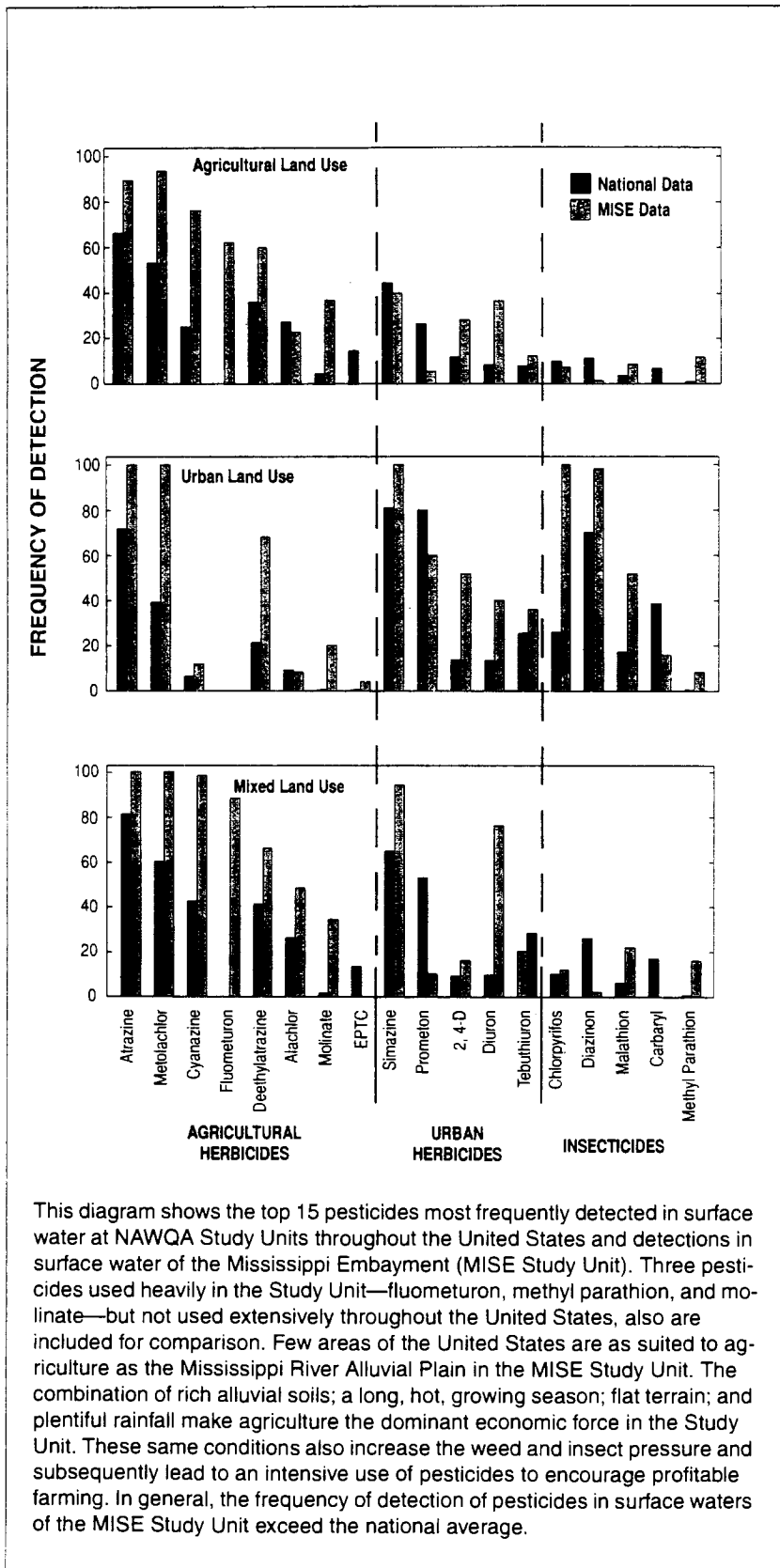


Figure 6. Herbicides in agricultural streams in the Mississippi Embayment Study Unit had higher concentrations, showed clear seasonal patterns, and contained different compounds than herbicides in urban streams. The urban stream samples were dominated by simazine, a turf grass herbicide. Concentrations of herbicides in the urban stream remained fairly constant throughout the year, whereas agricultural sites had concentrations that peaked in the spring shortly after application. Agricultural sites also were dominated by different herbicides (in this case atrazine).



Urban Stream

The type, amounts, and timing of the occurrence of pesticides in the stream draining an urban watershed are much different from those in the agricultural streams (fig. 6). The herbicides most frequently occurring in the urban stream such as atrazine, 2,4-D, simazine, and prometon, are those used in lawn care and in the maintenance of rights-of-way. The urban stream was also the only stream with frequent occurrences of insecticides: chlorpyrifos and diazinon were detected in every urban stream sample collected and exceeded the aquatic-life guidelines in 12 of 25 and 24 of 25 samples, respectively.



Many pesticides are applied by aircraft in the Study Unit.

DDT Metabolites in Surface Water

Although DDT strongly sorbs to sediments rather than readily dissolving in water, detectable levels of DDE, a metabolite of DDT, were found in 14 percent of the filtered stream-water samples analyzed.

More details on pesticides in streams in the MISE can be found in the report: Coupe, R.H., 2000. Occurrence of pesticides in five rivers of the Mississippi Embayment Study Unit, 1996–98: U.S. Geological Survey Water-Resources Investigations Report 99–4159, 69 p.

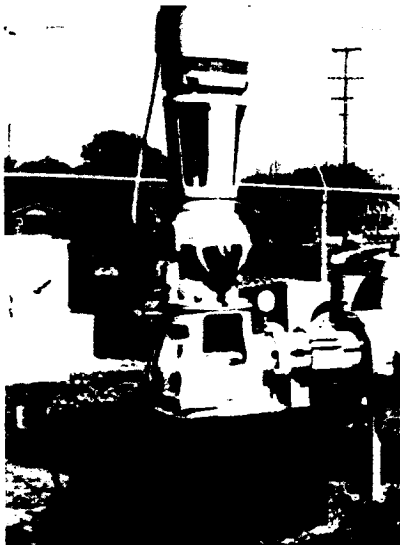
The report also can be downloaded at: <http://ms.water.usgs.gov/misenawqa/>

Tertiary Aquifers Have High-Quality Drinking Water

During the spring of 1996, water samples were collected from 30 public-supply wells in the deep Tertiary aquifers in the MISE Study Unit. The most significant finding from this part of the study is that all of the samples analyzed from these public-supply wells met all existing drinking-water standards and guidelines. Concentrations of most of the constituents measured that could adversely affect water quality, including nutrients, pesticides, radon, and volatile organic compounds, were below drinking-water standards and guidelines.

The Deep Tertiary Aquifers

The deep Tertiary aquifers underlie about 80 percent of the MISE Study Unit. Much of the population in this part of the country depends on these aquifers for drinking water. Wells sampled



Ground-water samples commonly were collected from municipal drinking-water facilities, such as the one pictured above.

range in depth from 208 to 1,460 feet below the ground surface.

Sample Results

Water from wells in the deep Tertiary aquifers had low nutrient concentrations. The highest nutrient concentration measured in a sample was 3.8 mg/L of nitrite plus nitrate nitrogen, which is less than half the drinking-water guideline of 10 mg/L. Pesticides were detected in water from only one of the wells. Water from the shallowest well sampled had a 0.16- $\mu\text{g/L}$ (microgram per liter) concentration of the herbicide bromacil and a 0.004- $\mu\text{g/L}$ concentration of deethylatrazine, a metabolite of the herbicide atrazine. Volatile organic compounds (VOCs) are compounds that have a high vapor pressure relative to their water solubility and include such things as components of gasoline and organic solvents. VOCs were detected frequently in the MISE, but concentrations were far below drinking-water guidelines. Samples from 26 of the 30 public-supply wells had at least one VOC detection. The VOCs most commonly detected were methyl-ethylketone and 1,2,4-trimethylbenzene, detected in 23 and 7 percent of the wells, respectively. Nutrients, pesticides, and VOCs generally enter the ground water from surface contamination; however, public-supply wells were generally deep enough to avoid elevated levels of these compounds.

Radon in water from the public-supply wells ranged from 54 to 270 picocuries per liter; none exceeded guidelines. Radon levels found in the ground water in the MISE were the second lowest of the 16 Study Units sampled during 1996–98.



Ground-water sample preparation occurred inside plastic enclosures in order to minimize sample contamination from chemicals in the atmosphere.

Few Pesticides Detected in Memphis Shallow Aquifers

In addition to the Tertiary aquifers, 32 shallow monitoring wells (not public-supply, drinking-water wells) were sampled in the shallow aquifers near Memphis, Tennessee. Results were similar to those from the deep Tertiary aquifer study except that pesticides were more frequently detected and radon concentrations were higher. An atrazine concentration of 3.14 $\mu\text{g/L}$ was measured in one well, which narrowly exceeded the drinking-water guideline of 3.0 $\mu\text{g/L}$, and dieldrin was measured above the drinking-water guideline of 0.02 $\mu\text{g/L}$ in another well.

More details on ground-water quality in the deep Tertiary aquifers can be found in the report:

Gonthier, G.J., 2000, Water quality in the deep Tertiary aquifers of the Mississippi Embayment, 1996: U.S. Geological Survey Water-Resources Investigations Report 99-4131, 91 p.

The report also can be downloaded at: <http://ms.water.usgs.gov/misenawqa/>

Water-Quality Differences in Geological Subunits of the Alluvial Aquifer

Previous researchers have studied the alluvial aquifer as a single Quaternary feature (Grubb, 1986; Ackerman, 1989). However, during this NAWQA investigation, the results of the water-chemistry studies were examined by dividing the area into different major geologic units, two of which are the Pleistocene valley trains and the Holocene alluvium (Saucier, 1994). The data collected suggest that the differences in the geology influence the chemical makeup of the

ground water in the alluvial aquifer.

The Alluvial Aquifer

The alluvial aquifer is a large, underground, water-bearing layer of sand and gravel in the Lower Mississippi River Valley (fig. 7). Water use from the alluvial aquifer is enormous; pumpage from the aquifer is about 7 billion gallons per day (Mesko and others, 1990). Most of the water pumped from the alluvial aquifer is used to irrigate crops or to maintain aquaculture, but the ground water also is used for public supply and industry.



Many of the water samples collected from the alluvial aquifer were taken from irrigation wells in agricultural areas.

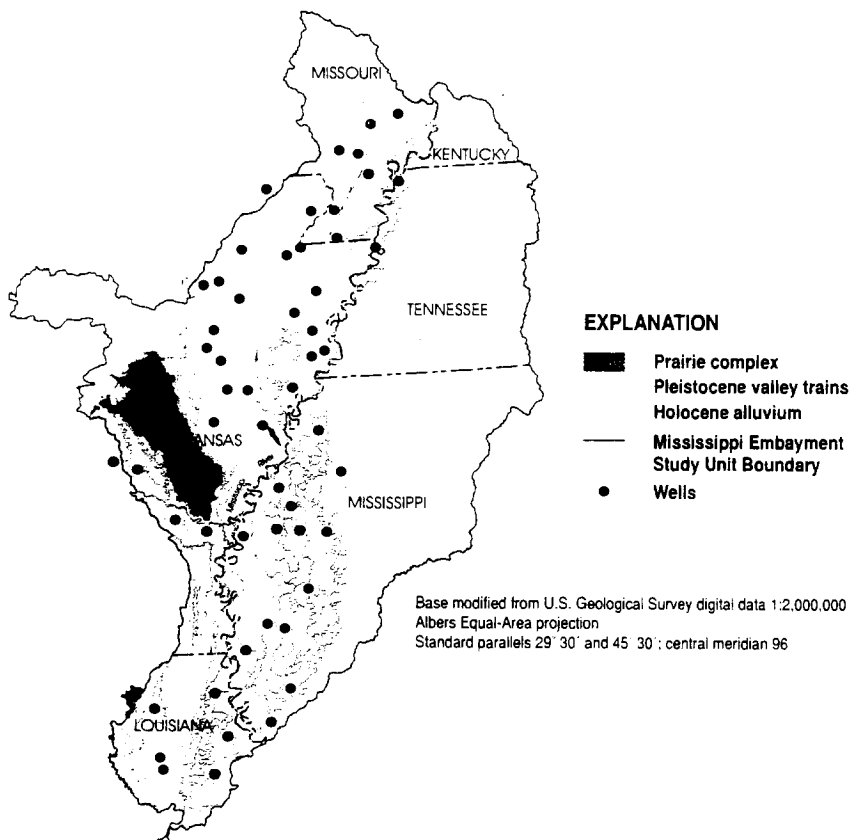


Figure 7. The colored part of this map depicts the areal extent of the three main geologic subunits of the alluvial aquifer and the location of the 54 wells that were sampled as part of this study. Statistically significant differences exist in the nutrient, carbon, and metal chemistry of the water sampled in wells located in the Pleistocene valley trains as compared to the Holocene alluvium, demonstrating the effect of geology on other components of the environment.

Water-Chemistry Analysis Results

Twenty-nine wells in the Pleistocene valley trains and 25 wells in the Holocene alluvium were sampled during the summer of 1998. At least one pesticide was detected in water from 19 of the 54 wells, but none of the concentrations were above drinking-water standards or guidelines. The most frequently detected pesticide was bentazon, an herbicide used to control weeds in soybean fields. Other pesticides detected in the alluvial aquifer in very low concentrations were molinate, fluometuron, 2,4-D, fenuron, atrazine, deethylatrazine, metolachlor, propanil, and *p,p'* DDE. At least one VOC was detected in water from 25 of 46 wells; 1,2,4-trimethylbenzene was detected most frequently. However, all of the VOC concentrations were well below drinking-water standards or guidelines.

Pleistocene Valley Trains and the Holocene Alluvium

The two subunits of the alluvial aquifer, the Pleistocene valley trains and the Holocene alluvium, have different lithological charac-

teristics. The Pleistocene valley trains generally have coarser grain sizes than the Holocene alluvium, whereas the Holocene alluvium has a thicker clay and silt surficial unit. These characteristics indicate that ground-water flow may be more active in the Pleistocene valley trains.

Results of the ground-water chemistry showed that sulfate, pH, tritium, chloride, and radon-222 were present in higher concentrations in water from wells in the Pleistocene valley trains, whereas dissolved organic carbon, iron, ammonia, fluoride, potassium, bicarbonate, magnesium, radium-226, barium, calcium, chromium, and dissolved solids were present in higher concentrations in water from wells in the Holocene alluvium.

Examples of some of these differences are shown in figure 8. Water in the Holocene alluvium tends to be older than water in the Pleistocene valley trains; that is, it has been underground longer. This increases possible contact with buried organics, resulting in less dissolved oxygen, which could influence concentrations of other chemical constituents.

Arsenic Concentrations

Arsenic, a compound that has been implicated in causing several cancers, was found at concentrations that exceeded current drinking-water guidelines in water from only one irrigation well that pumped water from the alluvial aquifer. However, concentrations at several other wells were high

enough to justify additional testing if proposals to lower the standards are promulgated.

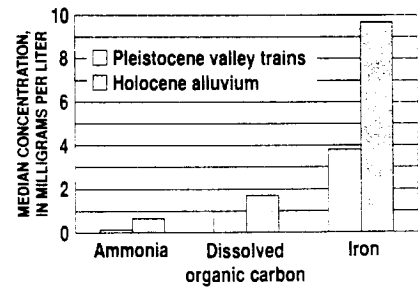
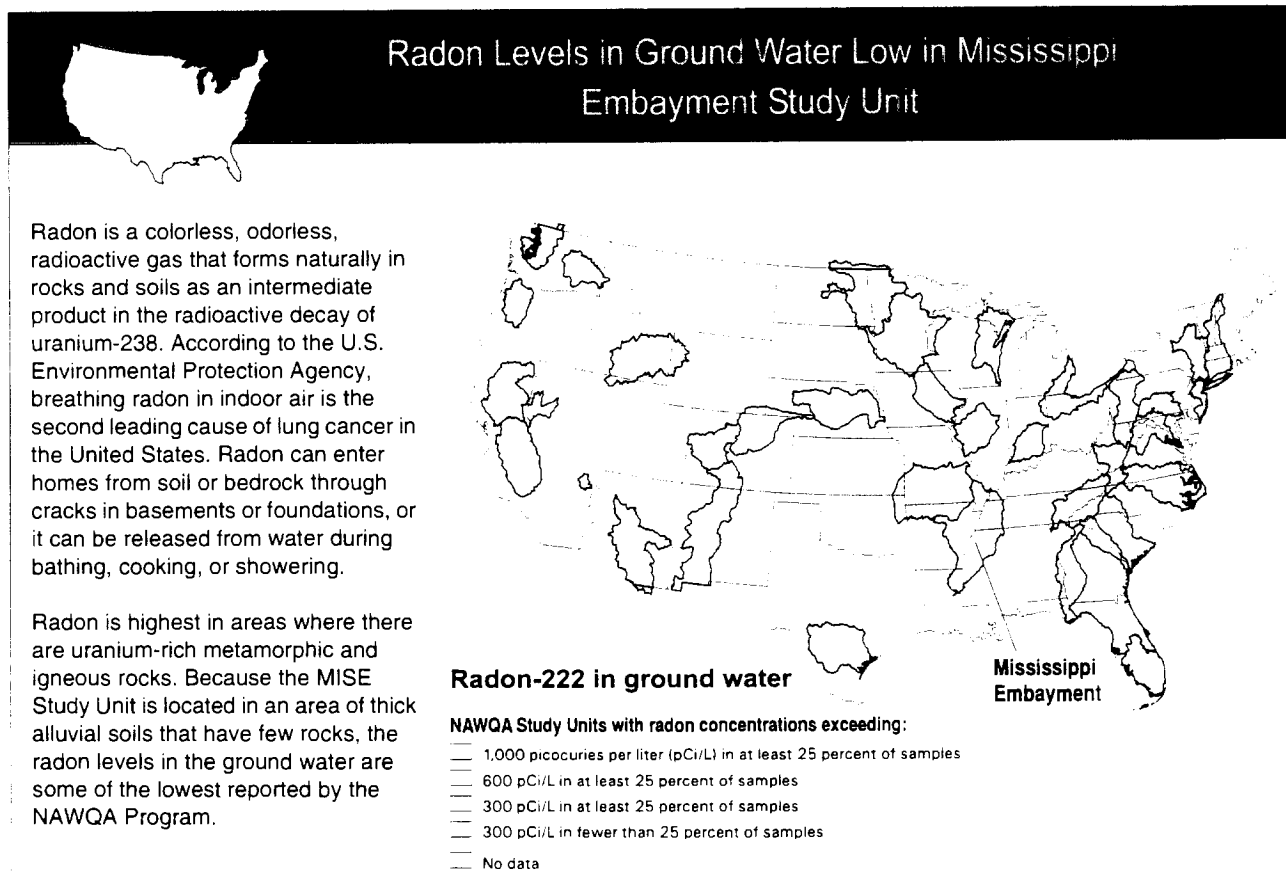


Figure 8. This graph shows some of the differences in chemistry in water from the Pleistocene valley trains and from the Holocene alluvium for a few selected constituents. Water from the wells in the Holocene deposits tended to be older and had lower oxygen levels. Low dissolved-oxygen concentrations in ground water may be associated with the presence of ammonia, dissolved organic carbon, and iron.



Aquatic Communities Show Environmental Stress

A combination of natural and anthropogenic (human-related) factors results in stream conditions that stress aquatic communities in the Mississippi Embayment Study Unit. The streams have naturally low gradients that result in sluggish flows and slow rates of reaeration (the ability of oxygen to enter the water). Also, the streams, in their natural conditions, have an abundance of streamside vegetation and swamps, resulting in an abundance of organic material in the water. This organic material is a good source of food for invertebrates (aquatic insects, crayfish, and freshwater shrimp) that inhabit the streams, but decay of the organic material and seasonally high water temperatures contribute to low dissolved-oxygen concentrations. The combination of low flow, high organic concentrations, and high temperatures results in a natural environment in which the organisms are often stressed by low dissolved-oxygen concentrations.

These natural conditions are coupled with many anthropogenic conditions in the area. These include stream channelization, which can eliminate riparian vegetation and degrade stream habitat; agricultural runoff into streams, which adds sediments, pesticides, and fertilizers to the aquatic environment; and the decline in the base flow of the streams due to ground-water withdrawal, which reduces the quantity of water available for organisms. This combination of natural and anthropogenic conditions affects each of the major biotic communities differently.

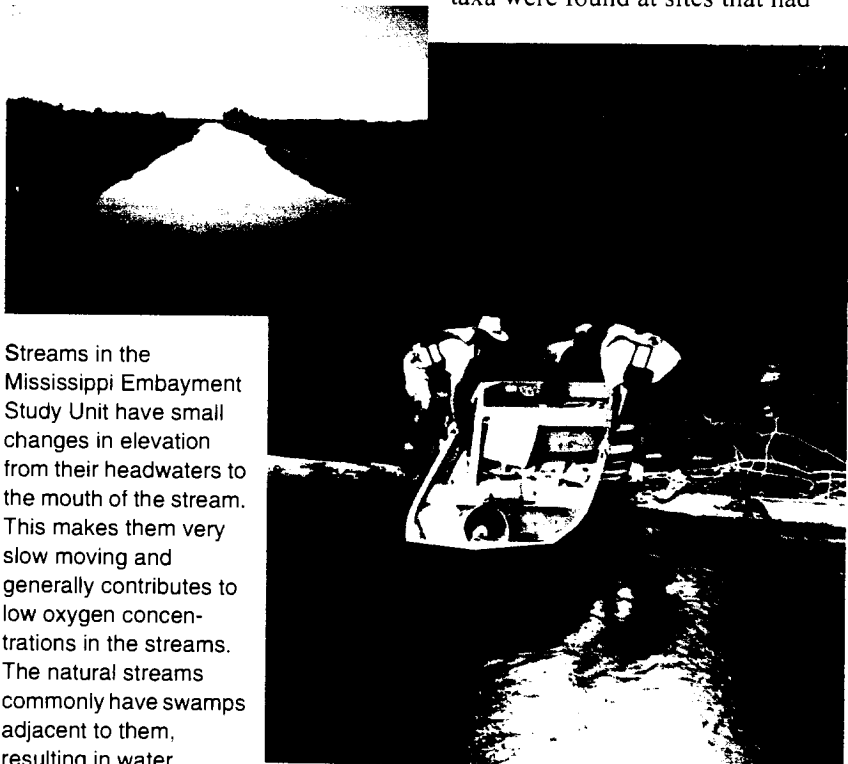
Algae

As a result of the dominant agricultural land use, fine alluvial soils, and limited vegetation in streamside or riparian areas, most of the streams in the MISE Study Unit are very turbid. Turbidity refers to the reduced clarity of surface water due to particles (usually sediment) suspended in the water. Many of the streams within the MISE also have moderately high phosphorus levels, which encourage algal growth. However, the algal growth in the streams is often more limited by the inability of light to penetrate the turbid waters, than by lack of nutrients. One indication of this can be seen in the Algal Siltation Index on page 17. This index uses the relative abundance of diatom species, which are able to move

and avoid being buried under large amounts of sediment, as an indication of stream-water quality. Levels of this index are high in agricultural areas of the MISE and moderate in mixed land-use areas.

Aquatic Invertebrates

Aquatic invertebrates in the MISE are influenced by habitat quantity and quality as well as by water chemistry. Often, the loss of stream and riparian habitat in the MISE is associated with stream channelization, where streams have been cleared, ditched, and straightened to facilitate the movement of floodwaters. These activities also result in the loss of microhabitats that are essential to aquatic invertebrates for food sources and refuge. Lower numbers of invertebrate taxa were found at sites that had



Streams in the Mississippi Embayment Study Unit have small changes in elevation from their headwaters to the mouth of the stream. This makes them very slow moving and generally contributes to low oxygen concentrations in the streams. The natural streams commonly have swamps adjacent to them, resulting in water stained with organics, ample habitat, and difficult sampling conditions, as shown above. The many channelized streams in the area (see upper left photograph) have commonly lost all of their riparian vegetation, and the streams have little habitat for aquatic organisms.



Some of the samples for both algae and invertebrates were collected by scraping organisms off submerged sticks in the streams. These sticks are commonly some of the best remaining habitats in many streams and rivers in the Mississippi Embayment Study Unit.

lower habitat quality (fig. 9). Other factors, including turbidity, low dissolved-oxygen concentrations, and the introduction of contaminants to the water, also affect invertebrate communities.

When compared to samples collected in other NAWQA Study Units across the United States (p. 17), invertebrate communities in MISE agricultural areas have low numbers of species and high proportions of tolerant organisms.

Fish

The condition of the fish communities in the MISE Study Unit was related to both water quality

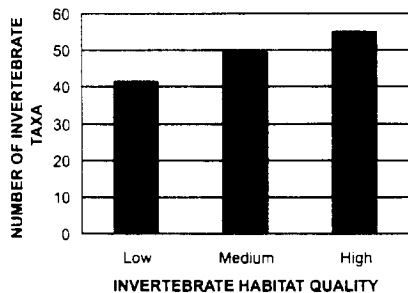


Figure 9. The number of invertebrate taxa increased as the quality of the invertebrate habitat increased in MISE streams.



A prime habitat for invertebrates in the Mississippi Embayment Study Unit is this "undercut bank" where insects and other organisms hide in the roots and sticks. This habitat is removed or disturbed by channelization of streams and rivers or other habitat alterations.

and land-use activities. The best quality fish communities occurred where instream variables such as turbidity, total ammonia, the average number of herbicides detected, and total DDT in fish tissue were lowest. As shown in figure 10, the number of bass, an important game fish in the MISE, was highest at the sites where the turbidity was lowest. The condition of the fish communities also was correlated to landscape-level variables such as insecticide application rates and soil permeability.

When compared to fish communities collected at NAWQA Study Units across the Nation (p. 17), the MISE had more fish that are omnivores (that is, fish that eat whatever is available and therefore are more tolerant than fish that have more restrictive diets) and more fish that are considered tolerant of poor water-quality conditions. However, there were fewer anomalies

(sores, parasites and other abnormalities on the fish) and non-native fish found in the MISE than in other locations. This resulted in an overall national fish ranking of MISE agricultural streams slightly below the midpoint between the most degraded and least degraded streams in the NAWQA Program, whereas mixed land-use streams ranged from good to less than average.

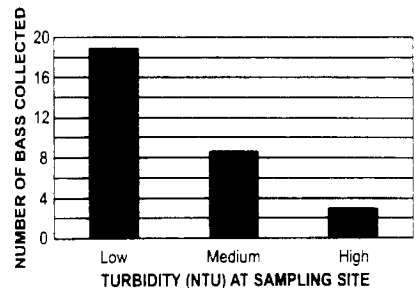


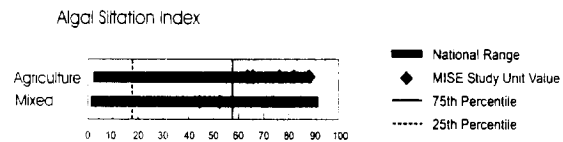
Figure 10. The number of bass collected at a stream site was highest in those streams where the turbidity was the lowest.



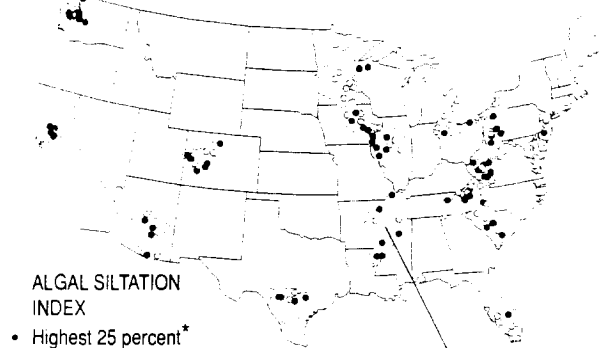
Comparison of Mississippi Embayment Aquatic Communities to National Results

Algal Siltation Index

Algae collected at the 8 MISE basic fixed ecology sites were compared to algae at 140 other NAWQA sites by use of the Siltation Index (Bahls and others, 1992). This index is the relative abundance of motile diatoms, the species *Navicula*, *Nitzschia*, *Cylindrotheca*, and *Surirella* in a diatom count. These diatoms are able to move through silt particles and are associated with fine sediments. Because they are able to avoid being buried, they are considered more tolerant of sedimentation than other diatoms. Generally, this index tends to be higher for streams in agricultural basins. Relative to this index, all of the MISE agricultural sites fall into the most-degraded category for streams along with one of the mixed land-use sites, whereas the other two sites fall into the middle 50 percent.

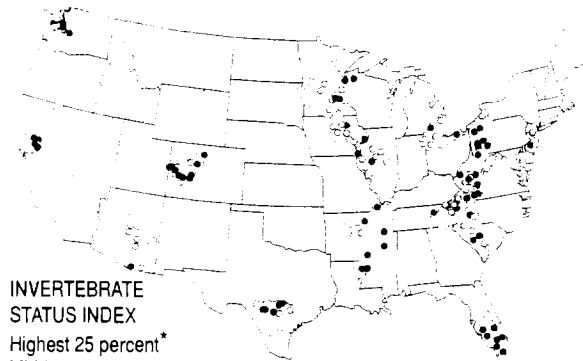


A National Ranking of ALGAL STATUS in Streams



- ALGAL SILTATION INDEX
- Highest 25 percent*
 - Middle 50 percent
 - Lowest 25 percent
- * Higher values suggest a more degraded stream site
- Mississippi Embayment

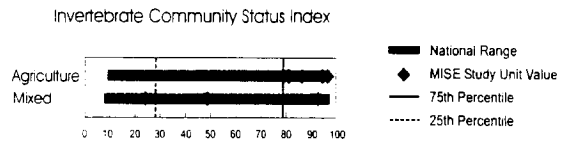
A National Ranking of INVERTEBRATE STATUS in Streams



- INVERTEBRATE STATUS INDEX
- Highest 25 percent*
 - Middle 50 percent
 - Lowest 25 percent
- * Higher values suggest a more degraded stream site

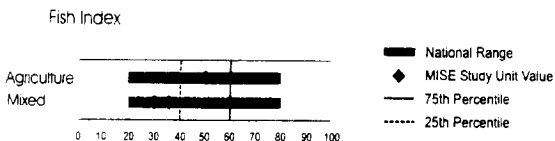
Invertebrate Community Status Index

To compare the invertebrate communities at MISE ecological basic fixed sites to national sites, a multimetric index called the Invertebrate Community Status Index was developed. The index combines 11 metrics, including ones that address taxa richness and diversity, richness of mayflies, stoneflies, and caddisflies, and tolerance metrics. Relative to this index, MISE agricultural streams are all ranked within the 25-percent most-degraded category for streams, whereas the mixed land-use streams have one site that falls into each category. Across the United States, both agricultural and urban sites tend to fall into the more degraded category.

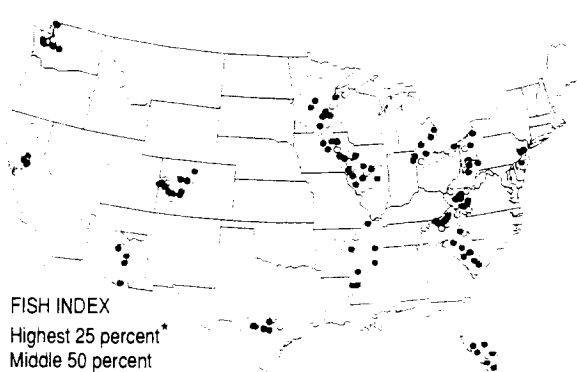


Fish Community Index

Similar in concept to the invertebrates, a multimetric fish index was developed to facilitate the comparison of fish communities across the United States. This index included four metrics: the percentage of tolerant individuals, the percentage of omnivorous individuals, the percentage of non-native individuals, and the percentage of individuals with anomalies. In all four metrics, a high percentage of individuals with the characteristic is typical of degraded sites. Agricultural sites in the MISE Study Unit generally fell at the boundary between the most-degraded category and the middle 50 percent (four sites are represented by the symbol at the 60-percent point). The mixed land-use sites ranged from the least-degraded category to the highly degraded category.



A National Ranking of FISH STATUS in Streams

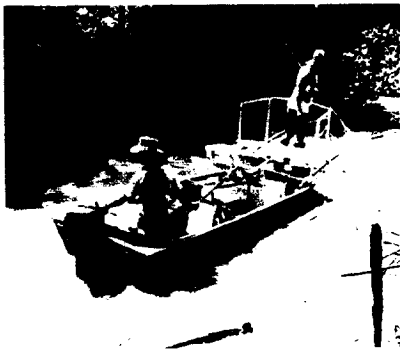


- FISH INDEX
- Highest 25 percent*
 - Middle 50 percent
 - Lowest 25 percent
- * Higher values suggest a more degraded stream site

Organochlorine Pesticides Persist in Fish Tissue

During the 1950s and 1960s, organochlorine pesticides were heavily used in the MISE Study Unit to control insects, particularly those associated with the cultivation of cotton. One of the most widely used organochlorine insecticides was DDT. In 1972, the sale of DDT was discontinued, and the use of most of the persistent organochlorine pesticides was discontinued during the 1970s. Nearly 30 years later, however, DDT or its metabolites were detected in fish tissue from all 41 sites sampled in the Study Unit. Thirty of these sites have total DDT levels in excess of the 0.2-mg/kg (milligram per kilogram) or 200 µg/kg (micrograms per kilogram) criterion set by the State of New York for the protection of fish-eating wildlife (Newell and others, 1987).

Although somewhat less prevalent and at lower concentrations, other organochlorine compounds also were detected in fish, including chlordane (at 33 percent of the sites), dieldrin (79 percent), heptachlor (7 percent), mirex (29 percent), and toxaphene (56 percent).



Most of the rivers in the MISE Study Unit were too deep to wade, so boat-mounted electroshocking equipment was used to collect fish (usually common carp) for tissue analysis.

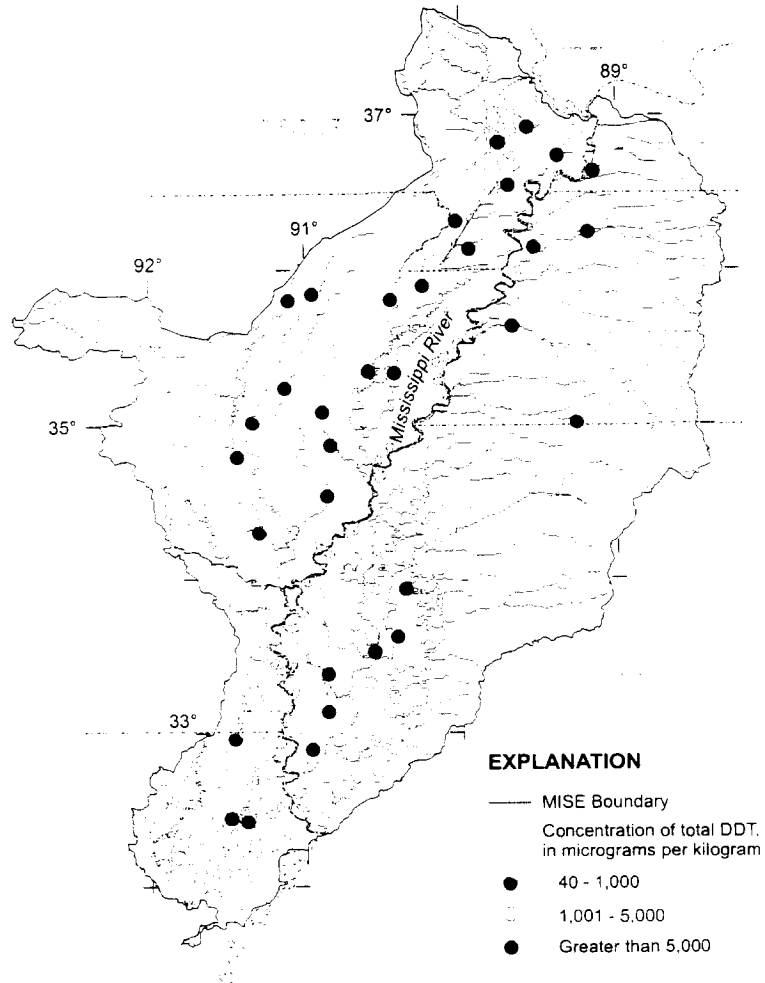


Figure 11. The concentrations of organochlorine pesticides were measured in fish tissue collected at 41 sites in the MISE Study Unit. Total DDT, an insecticide that has not been sold in the United States since 1972, was detected at all sites and occurs in concentrations that are of possible human health concern at some sites.

Of the 506 sites sampled thus far in the National Water-Quality Assessment Program, the highest concentrations of total DDT and toxaphene were found in fish collected in the Mississippi Embayment Study Unit (see pages 33 and 34 in Appendix A).

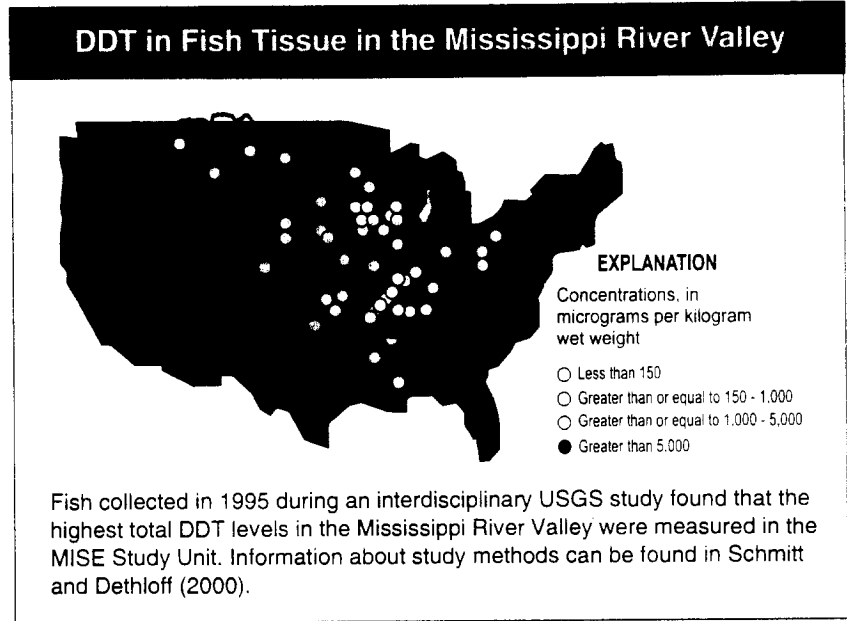
Distribution of Concentrations of DDT Within the Mississippi Embayment Study Unit

Although DDT was detected in fish tissue at every sampling site,

tissue samples from sites in the southern part of the Study Unit (fig. 11) had much higher concentrations of total DDT. On a national scale, Nowell and others, (1999) presented a positive correlation between median total DDT concentrations in whole fish and agricultural use of DDT in 1966. In the MISE, however, there seem to be additional factors influencing the persistence of DDT in fish tissue. Fish tissue data were compared to dozens of land-use and water-chemistry variables. Good correla-

tions can be made between total DDT concentrations and dissolved ammonia, nitrate, total phosphorus, and turbidity. The correlations with land-use variables, such as the percentage of the basin in agricultural production or the percentage of the basin in which cotton is grown, are much weaker, suggesting that factors in addition to land use and pesticide use are contributing to the distribution of DDT.

A relation exists between the concentrations of DDT in fish tissue and the geology of the area. Total DDT concentrations are much higher at sites in the Holocene alluvium than in the Pleistocene valley trains (fig. 12). The soils and geologic deposits in the Holocene alluvium are largely composed of dense clays that may cause agricultural runoff from these areas to enter the streams directly rather than infiltrate through the soils. Also, DDT tends to sorb onto clay particles and may be carried into streams attached to



these particles. Once in the stream, the cohesive nature of clay tends to prevent the material from moving downstream. These combined mechanisms may cause greater amounts of DDT to be in streams and available to organisms in Holocene areas than in streams in the Pleistocene valley trains.

Total DDT concentrations were detected in streambed sediments at all 15 sites measured. However, the concentrations in the sediments were many times lower than those in fish tissue.

Human Health Considerations

Because the NAWQA sampling was designed to measure the occurrence and distribution of compounds in the environment rather than address human health issues, whole fish (usually common carp) rather than fish fillets or other edible fish parts were analyzed for organochlorine compounds. However, the high concentrations of organochlorines that were detected at some of the sites in the MISE Study Unit suggest the need for further investigation. In the years since the sale of DDT and toxaphene has been discontinued, concerns have been raised about not only the toxicity of the pesticides but also the carcinogenic nature of organochlorine compounds.

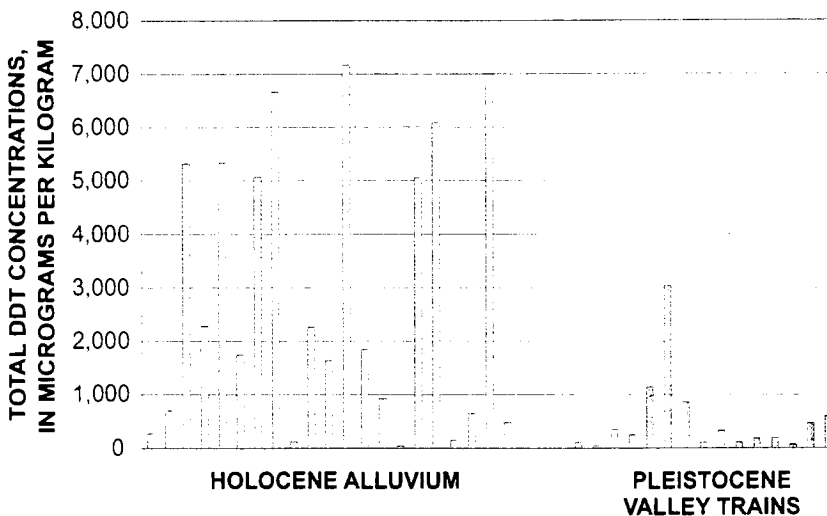


Figure 12. The concentration of total DDT in fish tissue is higher at Holocene alluvium sampling sites than at sites in the Pleistocene valley trains. The mean concentration for the Holocene sites is 2,895 micrograms per kilogram, while the mean concentration for the Pleistocene sites is 522 micrograms per kilogram.

Pesticides in Air and Rain Samples from Agricultural and Urban Sites

During the 1995 growing season, weekly air and rain samples were collected from two sites in Mississippi and analyzed for 49 pesticides and pesticide metabolites. The two sites represented an agricultural area and an urban area. Every air and rain sample had detectable levels of multiple pesticides; the pesticides detected and the frequency of detection varied between the two sites and were related to the types of pesticides used nearby. However, long-range transport appears to have an effect, as some pesticides that are not registered for use in an urban setting were found in air and rain samples at the urban site. These findings demonstrate that small amounts of pesticides can be transported through the atmosphere and deposited into aquatic and terrestrial ecosystems at a substantial distance from their point of use.

Pesticides in the Air

Pesticides can enter the atmosphere through volatilization, wind

erosion of soil particles to which pesticides are attached, and direct spraying into the atmosphere during pesticide application.

Seasonal distribution of pesticides at the agricultural site is related to local application times of individual compounds. At the start of the study in April, the herbicides pendimethalin and trifluralin were the pesticides found in the highest concentrations in the air. In May, the two pesticides with the highest measured concentrations were the herbicides propanil and thiobencarb, which typically are used on rice. By August, the insecticide methyl parathion was detected in the highest concentration of any of the pesticides in the air at the agricultural site.

In contrast, diazinon was detected in the highest concentration in the air at the urban site. Chlorpyrifos also was detected frequently, and carbaryl, methyl parathion, and trifluralin were found less often. Although most of these compounds are used commonly in residential settings for such purposes as termite control, methyl parathion is used only for insect control in agricultural settings, thus



The yellow polyurethane foam plug was used to collect pesticides distributed into the gas phase of the atmosphere.

suggesting the long distance atmospheric transport of this compound.

Pesticides in the Rain

Once in the atmosphere, pesticides can be degraded, transported, and (or) redeposited. Deposition can be either wet, as with rain or snow, or dry, as with gaseous sorption and particle fallout.



The urban atmospheric sampling site on the left is in a residential area in southern Jackson, Mississippi, a metropolitan area of about 400,000 people. The site is more than 10 miles from the nearest agricultural field. The agricultural site on the right is located in the center of a catfish pond complex in Sharkey County, Mississippi. This site was selected to measure pesticides in the atmosphere in an area with minimal influence of direct application of pesticides to nearby fields. The nearest agricultural field is about one-half mile away, and the major crops grown in this area are soybeans, cotton, corn, and rice.

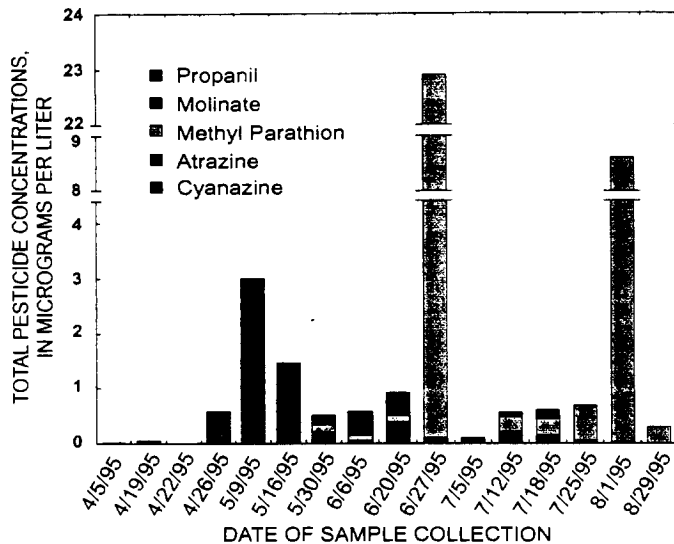


Figure 13. Pesticide concentrations in rainfall samples collected at an agricultural site in Mississippi during the 1995 growing season. In the spring, atrazine and propanil were detected in higher concentrations, but by summer the dominant pesticide measured was methyl parathion.

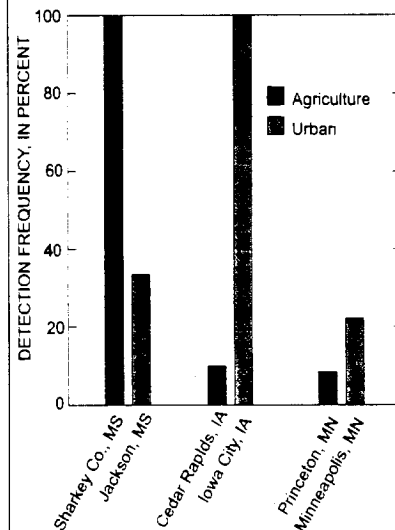
The concentrations of selected pesticides detected in rainfall samples from the agricultural site are shown in figure 13. The pesticides detected in the highest concentrations during the early part of the growing season were the herbicides atrazine, used on corn and grain sorghum, and propanil, used mainly on rice. Later in the growing season, the pesticide found in the highest concentrations was the insecticide methyl parathion.

The pesticides detected in rainfall samples from the urban site— atrazine, carbaryl, methyl parathion, and propanil—were similar to those found at the agricultural site, but the concentrations were much lower. Because methyl parathion and propanil do not have legal urban uses, it is assumed that these pesticides were transported in the atmosphere from their application areas. Methyl parathion and propanil are the first and sixth most heavily used agricultural pesticides in Mississippi.

Metabolites of DDT in the Air

In order to fully understand the fate, transport, and environmental effects of a pesticide, major metabolites of the pesticide commonly are included in the sampling program. One of the major metabolites of the organochlorine insecticide DDT is *p,p'*-DDE. In 1971, Stanley and others (1971) detected *p,p'*-DDE in the air at an agricultural site near Stoneville, Miss. (about 60 miles north of the Sharkey County site) in concentrations ranging from 2.6 to 7.1 ng/m³ (nanograms per cubic meter). Twenty-four years later, the range of *p,p'*-DDE in air samples from the agricultural site ranged from 0.13 to 1.1 ng/m³, lower than reported by Stanley and others (1971) but still significant considering that there is no current use of DDT in the area. These results indicate that a persistent metabolite of *p,p'*-DDT was still measurable in the air more than two decades after the sale of DDT was discontinued in the United States.

DDT Metabolites in the Air Across Mid-America



The air concentrations of *p,p'*-DDE, a metabolite of DDT, were determined at paired urban and agricultural sites in Mississippi, Iowa, and Minnesota (Foreman and others, 2000). The agricultural site in Mississippi and the urban site in Iowa had detectable levels of *p,p'*-DDE in every sample. Approximately 10 to 30 percent of the samples collected at the other sites had detectable levels of *p,p'*-DDE. It is likely that these concentrations of *p,p'*-DDE are from past local use of DDT. However, DDT is still being used in other parts of the world, and it is possible that some proportion of this *p,p'*-DDE originated elsewhere and was transported in the atmosphere.

More details on pesticides in the atmosphere can be found in the paper: Coupe, R.H., Manning, M.A., Foreman, W.T., Goolsby, D.A., and Majewski, M.S., 2000, Occurrence of pesticides in urban and agricultural areas of Mississippi, April–September 1995: *Science of the Total Environment*, v. 248, no. 2–3, p. 227.

The paper also can be downloaded at: <http://ms.water.usgs.gov/misenawqa/>

Integration of Biological, Geological, and Chemical Data Improves Understanding of Aquatic Systems

One of the most significant advances in the aquatic sciences during the last century is the growing understanding that aquatic systems, such as lakes and rivers, are profoundly influenced by the biological, chemical, and physical aspects of the drainage basin in which the water body is located (Wetzel, 1983). Although the conceptual framework for these ideas is decades old, scientists are still developing the technologies and accumulating the data bases necessary to more fully understand the relations between a body of water, the biotic components within it, and the chemical and physical influences of the drainage basin.

In the NAWQA Program, researchers often collect biological, geological, and chemical information from the same stream sites and analyze land use from the corresponding drainage basins. This information presents an opportunity to examine biogeochemical relations within the watersheds. This interdisciplinary approach has yielded at least three broad findings in the MISE Study Unit. Three findings, which need to be examined in more detail to guide scientific understanding and management in this region, are: (1) DDT and its metabolites remain detectable in many parts of the environment; (2) numerous interrelations exist between the biological, geological, and chemical components of the MISE Study Unit; and (3) evidence exists that the geology of the Study Unit exerts an influence

on many aspects of the environment.

DDT and Its Metabolites Are Present Throughout the Environment

Nearly 30 years after the sale of DDT was discontinued, the pesticide and its metabolites were detected in all parts of the MISE Study Unit environment. While residual total DDT is found throughout the world, the concentrations in the MISE are among the highest in the United States. (See water, fish tissue, and bed sediment sections of the national comparison tables on pages 31, 33, and 34 for a comparison of MISE results to other NAWQA Study Units in the United States.) A metabolite of DDT, DDE persists in various concentrations in many different parts of the MISE environment, including the air, water, streambed sediment, and fish tissue, generating interesting questions about its transport, accumulation, and persistence.

Biological and Chemical Relations

The MISE Study Unit has numerous complex interrelations among stream biology, basin geology, stream chemistry and land use. Some of these interrelations are illustrated by a variety of linear relations among biological, geological, and chemical components of 36 stream sites and their drainage basins in the MISE Study Unit (fig. 14). Figures 14A and 14C illustrate how differences in stream chemistry can affect biological communities in the stream. Figure 14A shows that the number of insectivore fish taxa decreases as total ammonia increases, whereas figure 14C

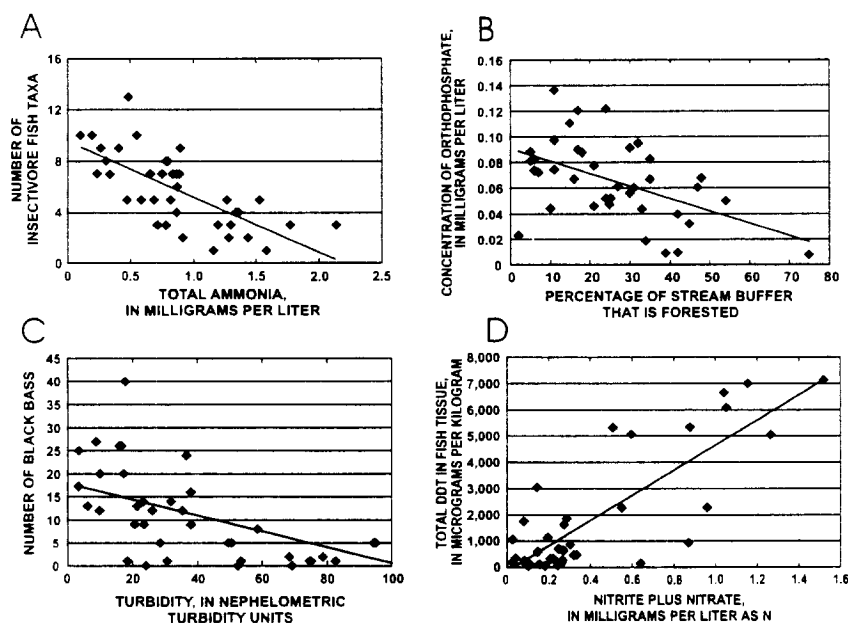


Figure 14. There are numerous interrelations among stream biology, basin geology, stream chemistry, and land use in the Mississippi Embayment Study Unit. The lines in these graphs represent linear relations between some of the variables.

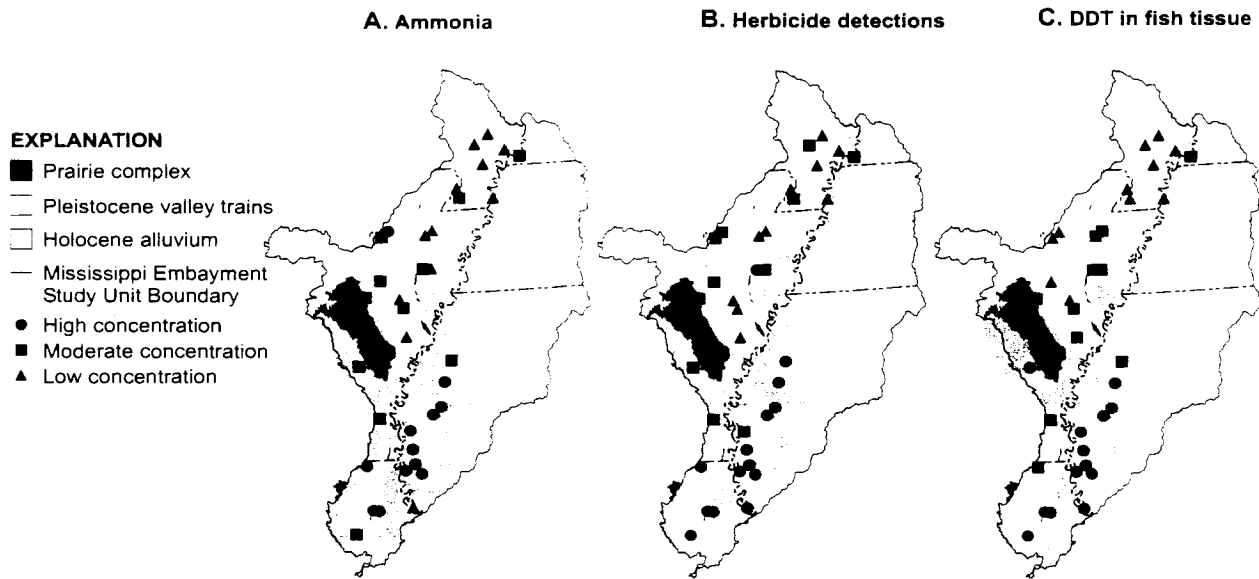


Figure 15. Several variables in the MISE Study Unit vary spatially and correspond generally to the Quaternary geology of the Mississippi Alluvial Plain. Soils formed from the Holocene alluvium tend to be dense, thick clays, whereas the Pleistocene features tend to have soils that have more silts and sands. These soils influence permeability and runoff, which in turn, appear to affect the amount and persistence of agricultural chemicals in the streams.

shows that the number of black bass decreases as turbidity increases. Figures 14B and 14D show how factors in the drainage area, but not in the stream, can influence the stream. Figure 14B shows that as the percentage of forested stream buffer (the area within 60 meters of the streams) increases, concentrations of orthophosphate in the stream decrease, indicating the value of vegetated stream buffers in minimizing the amount of runoff entering streams and rivers. Finally, figure 14D illustrates that as the concentration of dissolved nitrite plus nitrate increases, the amount of total DDT in fish tissue increases, suggesting that conditions that facilitate agricultural runoff affect nutrient levels in the streams and the long-term bioaccumulation of organochlorine pesticides in fish tissue.

The sum of these various relations shows the complexity of the system, underscores the need for

multidisciplinary studies in order to thoroughly understand the Mississippi Embayment Study Unit, and discourages simplistic, one-dimensional management solutions.

Geology Influences Many Variables

Within the Mississippi Embayment Study Unit, the geology of the land surface appears to exert an overarching influence on many components of the environment. The dense, tight clays of the Holocene alluvium dictate the types of crops grown and cause high runoff potential in the area. These factors help to determine the types of agricultural chemicals that are applied and the amounts of these chemicals that are transported into rivers and streams. Once in the streams, the clays tend to cause chemicals sorbed to sediments to remain locally in the bed sediment rather than to be washed downstream. Sites where high concentrations of ammonia were found are

clustered in the Holocene alluvium (fig. 15A), whereas similar clusters occurred for the number of herbicides detected in the water (fig. 15B) and the amount of total DDT found in fish tissue (fig. 15C). In turn, many biological metrics were lower at sites located in the Holocene alluvium, and differences in ground-water chemistry were detected (pages 13 and 14).

This information reaffirms that a better understanding of the underlying geology of the Earth itself is needed to fully understand the effects of human activities on the environment.

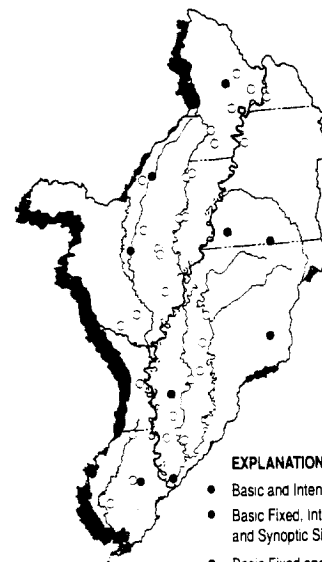


Carp used for tissue analysis.

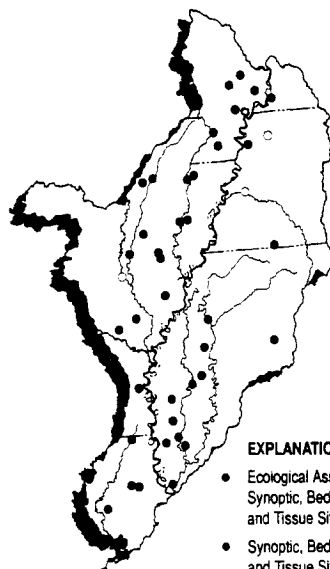
STUDY UNIT DESIGN

Stream Chemistry

The stream-chemistry network was designed to measure the effects of land use (primarily cropland) on stream quality and to integrate the effects of multiple land uses and hydrogeologic settings on water quality. Data from the Basic Fixed Sites were used to examine differences in water quality from one basin to another. Data from Intensive Fixed Sites, which were sampled more often, were used to examine seasonal changes in stream quality and to calculate fluxes from the basin. Also, detailed data on pesticides dissolved in the water were collected at intensive sites. A synoptic study was conducted to measure a selected set of constituents at 38 sites in the Study Unit in order to better understand spatial differences among the basins and to compare water chemistry to biological and landscape parameters.



- EXPLANATION**
- Basic and Intensive Fixed Sites
 - Basic Fixed, Intensive Fixed, and Synoptic Sites
 - Basic Fixed and Synoptic Sites Synoptic Sites



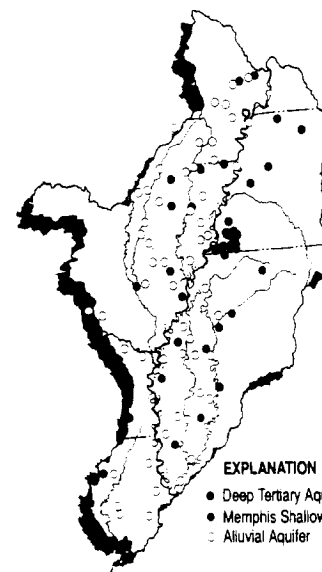
- EXPLANATION**
- Ecological Assessment, Synoptic, Bed Sediment, and Tissue Sites
 - Synoptic, Bed Sediment, and Tissue Sites
 - Synoptic and Tissue Sites
 - Bed Sediment and Tissue Sites

Stream Ecology

Ecological assessments were done annually at the basic and intensive stream-chemistry sites. The objective of these studies was to investigate biological, chemical, and physical data as multiple lines of evidence to assess water quality. Some of the assessments examined multiple reaches of a stream to determine spatial variations in the community structure of the aquatic organisms. Synoptic studies were designed to examine spatial variability in biological communities in the Study Unit and to relate this variability to stream chemistry and landscape variables. Early in the project, bed sediment and fish tissue were sampled at a subset of sites; later in the project, fish tissue was sampled for organochlorine concentrations at all of the synoptic sites.

Ground-Water Chemistry

Three surveys examined the effects of land use on ground water in different aquifer settings. The deep Tertiary aquifers are the deepest aquifers studied and the ones that provide drinking water to the greatest number of people in the Study Unit. The Memphis shallow aquifer survey was designed to assess the ground-water quality in commercial and residential areas. The alluvial aquifer survey examined ground-water quality from a shallow aquifer that is heavily used for agriculture.



- EXPLANATION**
- Deep Tertiary Aquifers
 - Memphis Shallow Aquifer
 - Alluvial Aquifer

SUMMARY OF DATA COLLECTION IN THE MISSISSIPPI EMBAYMENT STUDY UNIT, 1995-98

Study component	What data were collected and why	Types of sites sampled	Number of sites	Sampling frequency and period
Stream Chemistry				
Basic Fixed Sites	Continuous streamflow, nutrients, major ions, organic carbon, suspended sediment, and physical parameters were measured to describe concentrations and seasonal variations.	Sites were selected to represent the range of ecoregions, physiographic regions, and land uses present in the Study Unit.	9	Samples were collected monthly February 1996 - January 1998.
Intensive Fixed Sites	In addition to the above list of data collected at the Basic Fixed Sites, information on 82 dissolved pesticides was collected to determine concentrations, seasonal variations, and loads. Volatile organic compounds also were measured at the urban site.	Three of the sites were located at streams that drained intensive agricultural areas but were dominated by different crop types. One site was a large river site with mixed land use. The remaining site was located in a rapidly growing urban area.	5	Sampling frequency ranged from biweekly during the growing season to weekly throughout the remainder of the period February 1996 - January 1998.
Synoptic Sites—Water Chemistry	Nutrients, pesticides, and physical properties were measured to broaden the spatial coverage of water-quality information in the Study Unit.	Synoptic sites were selected in an effort to sample streams that drained all major crop types grown in the Study Unit.	38	Samples were collected once during May, July, and August 1997.
Stream Ecology				
Ecological Assessment Sites	Fish, macroinvertebrate, algae, and habitat data were collected to examine relations among the biological community and water chemistry, land use, and physical components of the landscape and drainage basin.	Sites were selected to represent the range of ecoregions, physiographic regions, and land uses present in the Study Unit.	8	Samples were collected once during low-flow conditions in the summers of 1996-98; at two sites, three reaches were sampled in 1996.
Synoptic Sites—Ecology	Fish, macroinvertebrate, habitat, and streamflow data were collected once at a larger number of sites to develop a better understanding of the spatial aspects of aquatic communities in the Study Unit.	Synoptic sites were selected in an effort to sample streams that drained all major crop types grown in the Study Unit.	38	Samples were collected July-September 1997 for macroinvertebrates; July-September 1998 for fish.
Contaminants in Bed Sediments	Total PCB's, organochlorine pesticides, semivolatile organic compounds, and trace elements were measured in order to determine the occurrence and distribution of contaminants in stream sediments.	Sites were selected to represent the range of ecoregions, physiographic regions, and land uses present in the Study Unit.	15	Samples were collected during August and September 1995.
Contaminants in Fish Tissue	Organochlorine pesticides were measured in whole fish, and trace elements were measured in fish liver.	Sites were selected in an effort to sample streams that drained all major crop types grown in the Study Unit.	41 (pesticides) 15 (trace elements)	Samples were collected during late summer low-flow conditions in 1995-98.
Ground-Water Chemistry				
Deep Tertiary Aquifers	Nutrients, major ions, pesticides, volatile organic compounds, radioisotopes, stable isotopes, and physical parameters were measured to determine overall water quality in a deep aquifer used for drinking water.	Public-supply wells screened in the deep Tertiary aquifers (Claiborne and Wilcox Groups) were sampled.	30	Samples were collected once during April and May 1996.
Memphis Shallow Aquifers	Nutrients, major ions, trace elements, pesticides, volatile organic compounds, radioisotopes, stable isotopes, and physical parameters were measured to determine overall water quality in shallow aquifers in a rapidly developing urban area.	Twenty-four monitoring wells screened in the shallow water-table aquifer and eight monitoring wells screened in the upper part of the Memphis aquifer were sampled.	32	Samples were collected once during April and May 1997.
Alluvial Aquifer	Nutrients, major ions, trace elements, pesticides, volatile organic compounds, radioisotopes, stable isotopes, and physical parameters were measured to determine overall water quality in an aquifer largely used for agricultural irrigation, but also for public supply and industry.	Twenty-five irrigation wells screened in the Holocene alluvium and 29 irrigation wells screened in the Pleistocene valley trains were sampled.	54	Samples were collected once during the summer of 1998.
Special Studies				
Pesticides in the Atmosphere	Pesticides were measured in the air and in the rain.	Two sites were sampled—one in an agricultural area in Sharkey County, Miss., and one in an urban area in Jackson, Miss.	2	Samples were collected April-September 1995.
Pesticides in Fish Tissue in the Mississippi River	Organochlorine pesticides in whole fish (common carp) were measured to determine if pesticides leaving the Yazoo River Basin could be detected in fish in the Mississippi River.	Four sites were selected within an area 100 miles upstream and another four sites were selected 100 miles downstream from the confluence of the Yazoo River and the Mississippi River.	8	Samples were collected once during November 1997.

GLOSSARY

- Algae** - Chlorophyll-bearing nonvascular, primarily aquatic species that have no true roots, stems, or leaves; most algae are microscopic, but some species can be as large as vascular plants.
- Alluvial aquifer** - A water-bearing deposit of unconsolidated material (sand and gravel) left behind by a river or other flowing water.
- Alluvium** - Deposits of clay, silt, sand, gravel or other particulate rock material left by a river in a streambed, on a flood plain, delta, or at the base of a mountain.
- Aquatic-life criteria** - Water-quality guidelines for protection of aquatic life. Often refers to U.S. Environmental Protection Agency water-quality criteria for protection of aquatic organisms. See also Water-quality guidelines, Water-quality criteria, and Freshwater chronic criteria.
- Aquifer** - A water-bearing layer of soil, sand, gravel, or rock that will yield usable quantities of water to a well.
- Atmospheric deposition** - The transfer of substances from the air to the surface of the Earth, either in wet form (rain, fog, snow, dew, frost, hail) or in dry form (gases, aerosols, particles).
- Basic Fixed Sites** - Sites on streams at which streamflow is measured and samples are collected for temperature, salinity, suspended sediment, major ions and metals, nutrients, and organic carbon to assess the broad-scale spatial and temporal character and transport of inorganic constituents of streamwater in relation to hydrologic conditions and environmental settings.
- Bed sediment** - The material that temporarily is stationary in the bottom of a stream or other watercourse.
- Bioaccumulation** - The biological sequestering of a substance at a higher concentration than that at which it occurs in the surrounding environment or medium. Also, the process whereby a substance enters organisms through the gills, epithelia tissues, dietary, or other sources.
- Biota** - Living organisms.
- Channelization** - Modification of a stream, typically by straightening the channel, to provide more uniform flow; often done for flood control or for improved agricultural drainage or irrigation.
- Community** - In ecology, the species that interact in a common area.
- Concentration** - The amount or mass of a substance present in a given volume or mass of sample. Usually expressed as microgram per liter (water sample) or microgram per kilogram (sediment or tissue sample).
- Criterion** - A standard rule or test on which a judgment or decision can be based.
- Degradation products** - Compounds resulting from transformation of an organic substance through chemical, photochemical, and/or biochemical reactions.
- Detection limit** - The minimum concentration of a substance that can be identified, measured, and reported within 99 percent confidence that the analyte concentration is greater than zero; determined from analysis of a sample in a given matrix containing the analyte.
- DDT** - Dichloro-diphenyl-trichloroethane. An organochlorine insecticide no longer registered for use in the United States.
- Dissolved solids** - Amount of minerals, such as salt, that are dissolved in water; amount of dissolved solids is an indicator of salinity or hardness.
- Drainage area** - The drainage area of a stream at a specified location is that area, measured in a horizontal plane, which is enclosed by a drainage divide.
- Drinking-water standard or guideline** - A threshold concentration in a public drinking-water supply, designed to protect human health. As defined here, standards are U.S. Environmental Protection Agency regulations that specify the maximum contamination levels for public water systems required to protect the public welfare; guidelines have no regulatory status and are issued in an advisory capacity.
- Ecosystem** - The interacting populations of plants, animals, and microorganisms occupying an area, plus their physical environment.
- Eutrophication** - The process by which water becomes enriched with plant nutrients, most commonly phosphorus and nitrogen.
- Fixed Sites** - NAWQA's most comprehensive monitoring sites. See also Basic Fixed Sites and Intensive Fixed Sites.
- Ground water** - In general, any water that exists beneath the land surface, but more commonly applied to water in fully saturated soils and geologic formations.
- Habitat** - The part of the physical environment where plants and animals live.
- Herbicide** - A chemical or other agent applied for the purpose of killing undesirable plants. See also Pesticide.
- Holocene** - A subdivision of geologic time which began at the end of the Pleistocene (approximately 9,000 to 11,000 years ago) and extends to the present.
- Human health advisory** - Guidance provided by U.S. Environmental Protection Agency, State agencies or scientific organizations, in the absence of regulatory limits, to describe acceptable contaminant levels in drinking water or edible fish.
- Intensive Fixed Sites** - Basic Fixed Sites with increased sampling frequency during selected seasonal periods and analysis of dissolved pesticides for 1 year. Most NAWQA Study Units have one to two integrator Intensive Fixed Sites and one to four indicator Intensive Fixed Sites.
- Load** - General term that refers to a material or constituent in solution, in suspension, or in transport; usually expressed in terms of mass or volume.
- Loess** - Homogeneous, fine-grained sediment made up primarily of silt and clay, and deposited over a wide area (probably by wind).
- Maximum contaminant level (MCL)** - Maximum permissible level of a contaminant in water that is delivered to any user of a public water system. MCLs are enforceable standards established by the U.S. Environmental Protection Agency.
- Median** - The middle or central value in a distribution of data ranked in order of magnitude. The median is also known as the 50th percentile.
- Metabolite** - A compound derived by chemical, biological, or physical action upon a pesticide. The breakdown is a natural process which may result in a more toxic or a less toxic compound and a more persistent or less persistent compound.
- Method detection limit** - The minimum concentration of a substance that can be accurately identified and measured with present laboratory technologies.

- Micrograms per liter ($\mu\text{g/L}$)** - A unit expressing the concentration of constituents in solution as weight (micrograms) of solute per unit volume (liter) of water; equivalent to one part per billion in most streamwater and ground water. One thousand micrograms per liter equals 1 mg/L.
- Minimum reporting level (MRL)** - The smallest measured concentration of a constituent that may be reliably reported using a given analytical method. In many cases, the MRL is used when documentation for the method detection limit is not available.
- Nitrate** - An ion consisting of nitrogen and oxygen (NO_3^-). Nitrate is a plant nutrient and is very mobile in soils.
- Nonpoint source** - A pollution source that cannot be defined as originating from discrete points such as pipe discharge. Areas of fertilizer and pesticide applications, atmospheric deposition, manure, and natural inputs from plants and trees are types of nonpoint source pollution.
- Nutrient** - Element or compound essential for animal and plant growth. Common nutrients in fertilizer include nitrogen, phosphorus, and potassium.
- Organochlorine insecticide** - A class of organic insecticides containing a high percentage of chlorine. Includes dichlorodiphenylethanes (such as DDT), chlorinated cyclodienes (such as chlordane), and chlorinated benzenes (such as lindane). Most organochlorine insecticides were banned because of their carcinogenicity, tendency to bioaccumulate, and toxicity to wildlife.
- Periphyton** - Organisms that grow on underwater surfaces; periphyton include algae, bacteria, fungi, protozoa, and other organisms.
- Phosphorus** - A nutrient essential for growth that can play a key role in stimulating aquatic growth in lakes and streams.
- Physiography** - A description of the surface features of the Earth, with an emphasis on the origin of landforms.
- Pleistocene** - A subdivision of geologic time which began about 2 million years ago and ended at the beginning of the Holocene epoch, approximately 9,000 to 11,000 years ago.
- Quaternary** - A subdivision of geologic time which began about 2 million years ago and extends to the present. The Quaternary period is further divided into two epochs, the Pleistocene and the Holocene.
- Recharge** - Water that infiltrates the ground and reaches the saturated zone.
- Sediment quality guideline** - Threshold concentration above which there is a high probability of adverse effects on aquatic life from sediment contamination, determined using modified USEPA (1996) procedures.
- Semivolatile organic compound (SVOC)** - Operationally defined as a group of synthetic organic compounds that are solvent-extractable and can be determined by gas chromatography/mass spectrometry. SVOCs include phenols, phthalates, and polycyclic aromatic hydrocarbons (PAHs).
- Species diversity** - An ecological concept that incorporates both the number of species in a particular sampling area and the evenness with which individuals are distributed among the various species.
- Species (taxa) richness** - The number of species (taxa) present in a defined area or sampling unit.
- Stream reach** - A continuous part of a stream between two specified points.
- Study Unit** - A major hydrologic system of the United States in which NAWQA studies are focused. Study Units are geographically defined by a combination of ground- and surface-water features and generally encompass more than 4,000 square miles of land area.
- Study-Unit Survey** - Broad assessment of the water-quality conditions of the major aquifer systems of each Study Unit. The Study-Unit Survey relies primarily on sampling existing wells and, wherever possible, on existing data collected by other agencies and programs. Typically, 20 to 30 wells are sampled in each of three to five aquifer subunits.
- Synoptic sites** - Sites sampled during a short-term investigation of specific water-quality conditions during selected seasonal or hydrologic conditions to provide improved spatial resolution for critical water-quality conditions.
- Taxon (plural taxa)** - Any identifiable group of taxonomically related organisms.
- Total DDT** - The sum of DDT and its metabolites (breakdown products), including DDD and DDE.
- Trace element** - An element found in only minor amounts (concentrations less than 1.0 milligram per liter) in water or sediment; includes arsenic, cadmium, chromium, copper, lead, mercury, nickel, and zinc.
- Triazine herbicide** - A class of herbicides containing a symmetrical triazine ring (a nitrogen-heterocyclic ring composed of three nitrogens and three carbons in an alternating sequence). Examples include atrazine, propazine, and simazine.
- Turbidity** - Reduced clarity of surface water because of suspended particles, usually sediment.
- Volatile organic compounds (VOCs)** - Organic chemicals that have a high vapor pressure relative to their water solubility. VOCs include components of gasoline, fuel oils, and lubricants, as well as organic solvents, fumigants, some inert ingredients in pesticides, and some by-products of chlorine disinfection.
- Water-quality criteria** - Specific levels of water quality which, if reached, are expected to render a body of water unsuitable for its designated use. Commonly refers to water-quality criteria established by the U.S. Environmental Protection Agency. Water-quality criteria are based on specific levels of pollutants that would make the water harmful if used for drinking, swimming, farming, fish production, or industrial processes.
- Water-quality guidelines** - Specific levels of water quality which, if reached, may adversely affect human health or aquatic life. These are nonenforceable guidelines issued by a governmental agency or other institution.
- Water-quality standards** - State-adopted and U.S. Environmental Protection Agency-approved ambient standards for water bodies. Standards include the use of the water body and the water-quality criteria that must be met to protect the designated use or uses.
- Water year** - The continuous 12-month period, October 1 through September 30, in U.S. Geological Survey reports dealing with the surface-water supply. The water year is designated by the calendar year in which it ends and which includes 9 of the 12 months. Thus, the year ending September 30, 1980, is referred to as the water year 1980.
- Wetlands** - Ecosystems whose soil is saturated for long periods seasonally or continuously, including marshes, swamps, and ephemeral ponds.

REFERENCES

- Ackerman, D.J., 1989, Hydrology of the Mississippi River Valley alluvial aquifer, south-central United States—A preliminary assessment of the regional flow system: U.S. Geological Survey Water-Resources Investigations Report 88-4028, 74 p.
- Arthur, J.K., 1994, Thickness of the upper and lower confining units of the Mississippi River alluvial aquifer in northwestern Mississippi: U.S. Geological Survey Water-Resources Investigations Report 94-4172, 1 p.
- Autin, W.J., Burns, S.F., Miller, B.J., Saucier, R.T., and Snead, J.I., 1991, Quaternary geology of the Lower Mississippi Valley, in Morrison, R.B., ed., Quaternary nonglacial geology—Conterminous U.S., in *The Geological of North America: Boulder, Colo.*, Geological Society of America, *The Geology of North America*, v. K-2, p. 547-582.
- Bahls, L.R., Burkantis, R., and Tralles, S., 1992, Benchmark biology of Montana reference streams: Helena, Montana, Department of Health and Environmental Science, Water Quality Bureau, 47 p.
- Battaglin, W.A., and Goolsby, D.A., 1995, Spatial data in geographic information system format on agricultural chemical use, land use, and cropping practices in the United States: U.S. Geological Survey Water-Resources Investigation Reports 94-4176, 87 p.
- Boswell, E.H., Cushing, E.M., and Hosman, R.L., 1968, Quaternary aquifer in the Mississippi Embayment *with a discussion of Quality of the water*, by Jeffery, H.G.: U.S. Geological Survey Professional Paper 448-E, 15 p., 2 pls.
- Coupe, R.H., 1998, Concentrations and loads of nitrogen and phosphorus in the Yazoo River, northwestern Mississippi, 1996-97: U.S. Geological Survey Water-Resources Investigations Report 98-4219, 17 p.
- _____, 2000, Occurrence of pesticides in five rivers of the Mississippi Embayment Study Unit, 1996-98: U.S. Geological Survey Water-Resources Investigations Report 99-4159, 69 p.
- Coupe, R.H., Manning, M.A., Foreman, W.T., Goolsby, D.A., and Majewski, M.S., 2000, Occurrence of pesticides in urban and agricultural areas of Mississippi, April-September, 1995: *Science of the Total Environment*, v. 248, no. 2-3, p. 227.
- Fenneman, N.M., 1938, *Physiography of the eastern United States*: New York, McGraw-Hill, 714 p.
- Foreman, W.T., Majewski, M.S., Goolsby, D.A., Wiebe, F.W., and Coupe, R.H., 2000, Pesticides in the atmosphere of the Mississippi River Valley, Part II—Air: *Science of the Total Environment*, v. 248, no. 2-3, p. 213-226.
- Gonthier, G.J., 2000, Water quality in the deep Tertiary aquifers of the Mississippi Embayment, 1996: U.S. Geological Survey Water-Resources Investigations Report 99-4131, 91 p.
- Goolsby, D.A., Battaglin, W.A., Lawrence, G.B., Artz, R.S., Aulenbach, B.T., Hooper, R.P., Keeney, D.R., and Stensland, G.J., 1999, Fluxes and sources of nutrients in the Mississippi-Atchafalaya River Basin—Topic 3 Report for the integrated assessment on hypoxia in the Gulf of Mexico: Silver Spring, Md., NOAA Coastal Ocean Program Decision Analysis Series No. 17, 130 p.
- Grubb, H.F., 1986, Gulf Coastal Plain regional aquifer-system study, in Sun, R.J., ed., *Regional Aquifer-System Analysis Program of the U.S. Geological Survey; summary of projects, 1978-84*: U.S. Geological Survey Circular 1002, p. 152-161.
- Mesko, T.O., Williams, T.A., Ackerman, D.J., and Williamson, A.K., 1990, Ground-water pumpage for the Gulf Coast aquifer systems, 1960-85, south-central United States: U.S. Geological Survey Water-Resources Investigations Report 89-4180, 177 p.
- Newell, A.J., Johnson, D.W., and Allen, L.K., 1987, Niagara River biota contamination project—Fish flesh criteria for piscivorous wildlife: New York State Department of Environmental Conservation, Division of Fish and Wildlife, Bureau of Environmental Protection Technical Report 87-3, 182 p.

- Nowell, L.H., Capel, P.D., Dileanis, P.D., 1999, Pesticides in stream sediment and aquatic biota—Distribution, trends, and governing factors: Boca Raton, Fla., Lewis Publishers, 1001 p.
- Omernik, J.M., 1987, Ecoregions of the conterminous United States: *Annals of the Association of American Geographers*, v. 77, no. 1, p. 118–125, scale 1:7,500,000.
- Saucier, R.T., 1994, *Geomorphology and Quaternary geologic history of the Lower Mississippi Valley*: Vicksburg, Miss., U.S. Army Engineer Waterways Experiment Station, v. I, 398 p.
- Schmitt, C.J., and Dethloff, G.M., eds., 2000, *Biomonitoring of environmental status and trends (BEST) program—Selected methods for monitoring chemical contaminants and their effects in aquatic ecosystems*: U.S. Geological Survey Information and Technology Report USGS/BRD–2000–0005, 81 p.
- Smith, F.E., 1954, *The Yazoo River*: Jackson, Miss., University Press of Mississippi, 362 p.
- Stanley, C.W., Barney, J.E., II, Helton, M.R., and Yobs, A.R., 1971, Measurement of atmospheric levels of pesticides: *Environmental Science & Technology*, v. 5, no. 5, p. 430–435.
- Wetzel, R.G., 1983, *Limnology* [2d ed.]: Philadelphia, Pa., Saunders College Publishing, 767 p.
- Whitfield, M.S., Jr., 1975, *Geohydrology and water quality of the Mississippi River alluvial aquifer, northeastern Louisiana*: U.S. Geological Survey Water-Resources Technical Report 10, 29 p.
- Williamson, A.K., Grubb, H.F., and Weiss, J.S., 1990, *Ground-water flow in the Gulf Coast aquifer systems, south-central United States—A preliminary analysis*: U.S. Geological Survey Water-Resources Investigations Report 89–4071, 124 p.

APPENDIX—WATER-QUALITY DATA FROM THE MISSISSIPPI EMBAYMENT IN A NATIONAL CONTEXT

For a complete view of Mississippi Embayment data and for additional information about specific benchmarks used, visit our Web site at <http://water.usgs.gov/nawqa/>. Also visit the NAWQA Data Warehouse for access to NAWQA data sets at <http://infotrek.er.usgs.gov/wdbctx/nawqa/nawqa.home>.

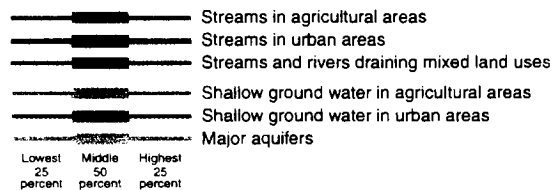
This appendix is a summary of chemical concentrations and biological indicators assessed in the Mississippi Embayment. Selected results for this Study Unit are graphically compared to results from as many as 36 NAWQA Study Units investigated from 1991 to 1998 and to national water-quality benchmarks for human health, aquatic life, or fish-eating wildlife. The chemical and biological indicators shown were selected on the basis of frequent detection, detection at concentrations above a national benchmark, or regulatory or scientific importance. The graphs illustrate how conditions associated with each land use sampled in the Mississippi Embayment compare to results from across the Nation, and how conditions compare among the several land uses. Graphs for chemicals show only detected concentrations and, thus, care must be taken to evaluate detection frequencies in addition to concentrations when comparing study-unit and national results. For example, fluometuron concentrations in Mississippi Embayment agricultural streams were similar to the national distribution, but the detection frequency was much higher (63 percent compared to 8 percent).

CHEMICALS IN WATER

Concentrations and detection frequencies, Mississippi Embayment, 1995–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals

- ◆ Detected concentration in Study Unit
- 66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- .. Not measured or sample size less than two
- 12 Study-unit sample size. For ground water, the number of samples is equal to the number of wells sampled

National ranges of detected concentrations, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

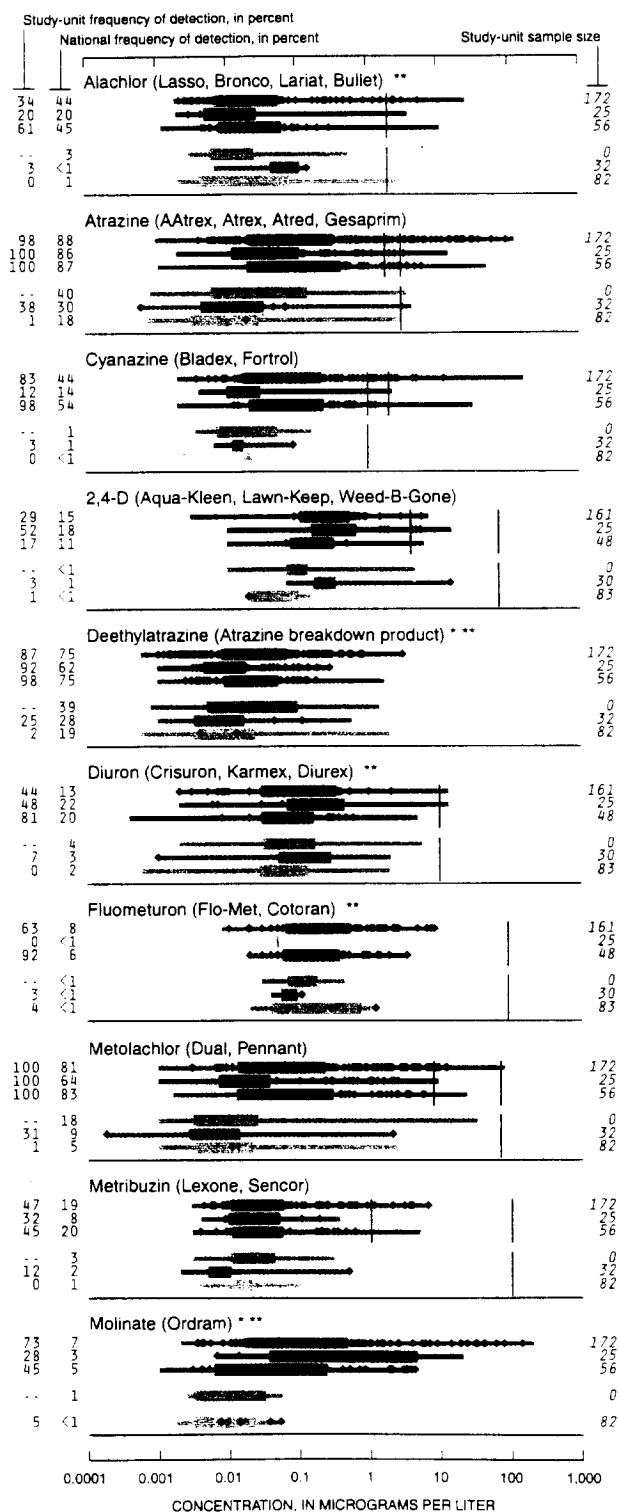


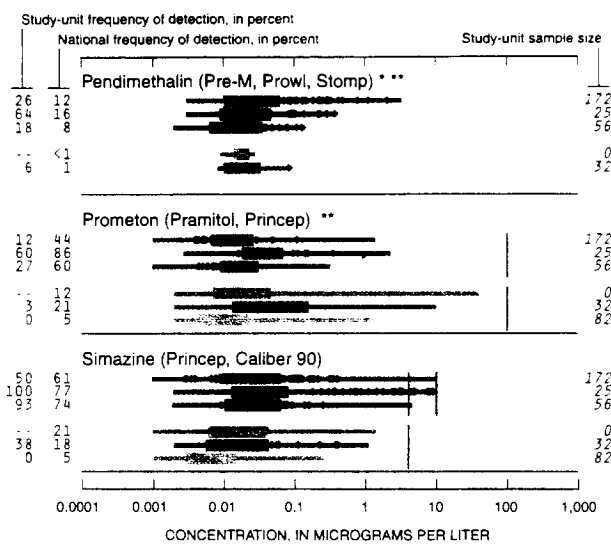
National water-quality benchmarks

National benchmarks include standards and guidelines related to drinking-water quality, criteria for protecting the health of aquatic life, and a goal for preventing stream eutrophication due to phosphorus. Sources include the U.S. Environmental Protection Agency and the Canadian Council of Ministers of the Environment

- | Drinking-water quality (applies to ground water and surface water)
- | Protection of aquatic life (applies to surface water only)
- | Prevention of eutrophication in streams not flowing directly into lakes or impoundments
- No benchmark for drinking-water quality
- .. No benchmark for protection of aquatic life

Pesticides in water—Herbicides





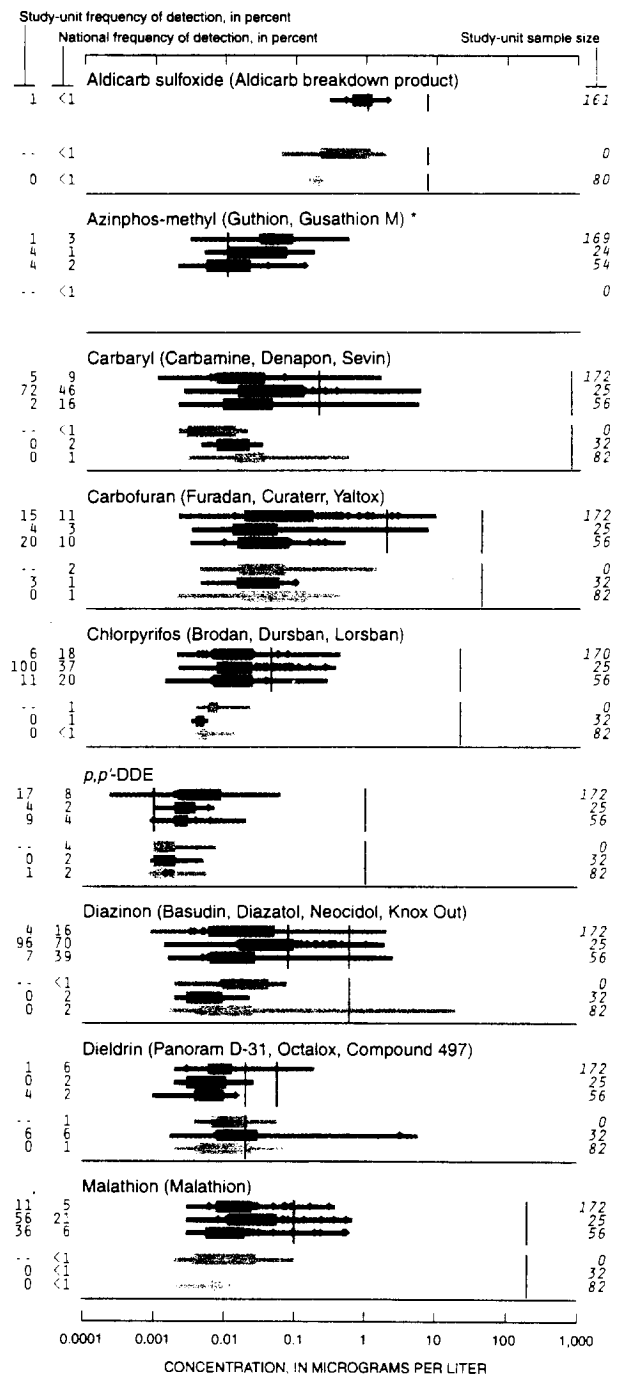
Other herbicides detected

- Acetochlor (Harness Plus, Surpass) ***
- Acifluorfen (Blazer, Tackle 2S) **
- Benfluralin (Balan, Benefin, Bonalan) ***
- Bentazon (Basagran, Bentazone) **
- Bromacil (Hyvar X, Urox B, Bromax)
- Bromoxynil (Buctril, Brominal) *
- 2,4-DB (Butyrac, Butoxone, Embutox Plus, Embutone) ***
- DCPA (Dacthal, chlorthal-dimethyl) ***
- Dacthal mono-acid (Dacthal breakdown product) ***
- Dicamba (Banvel, Dianat, Scotts Proturf)
- Dichlorprop (2,4-DP, Seritox 50, Lentemul) ***
- Dinoseb (Dinosebe)
- EPTC (Eptam, Farmarox, Alirox) ***
- Fenuron (Fenulon, Fenidim) ***
- Linuron (Lorox, Linex, Sarclex, Linurex, Afalon) *
- MCPA (Rhomene, Rhonox, Chiptox)
- Napropamide (Devrinol) ***
- Neburon (Neburea, Neburyl, Noruben) ***
- Norflurazon (Evitall, Predict, Solicam, Zorial) ***
- Pebulate (Tillam, PEBC) ***
- Picloram (Grazon, Tordon)
- Pronamide (Kerb, Propyzamid) **
- Propachlor (Ramrod, Satecid) **
- Propanil (Stam, Stampede, Wham) ***
- Tebuthiuron (Spike, Tebusan)
- Terbacil (Sinbar) **
- Thiobencarb (Bolero, Saturn, Benthicarb) ***
- Triclopyr (Garlon, Grandstand, Redeem, Remedy) ***
- Trifluralin (Treflan, Gowan, Tri-4, Trific)

Herbicides not detected

- Butylate (Sutan +, Genate Plus, Butilate) **
- Chloramben (Amiben, Amilon-WP, Vegiben) **
- Clopyralid (Stinger, Lontrel, Transline) ***
- 2,6-Diethylaniline (Alachlor breakdown product) ***
- Ethalfuralin (Sonalan, Curbit) ***
- MCPB (Thistrol) ***
- Oryzalin (Surflan, Dirimal) **
- Propham (Tuberite) **
- 2,4,5-T **
- 2,4,5-TP (Silvex, Fenoprop) **
- OTrallate (Far-Go, Avadex BW, Tri-allate) *

Pesticides in water—Insecticides



Other insecticides detected

- Disulfoton (Disyston, Di-Syston) **
- Methomyl (Lanox, Lannate, Acinate) **
- Methyl parathion (Penncap-M, Folidol-M) **
- Propargite (Comite, Omite, Ornamite) ***

Insecticides not detected

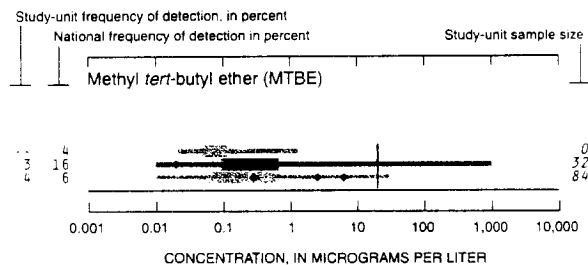
- Aldicarb (Temik, Ambush, Pounce)
- Aldicarb sulfone (Standak, aldoxycarb)
- Ethoprop (Mocap, Ethoprophos) ***

Fonofos (Dyfonate, Capfos, Cudgel Tyca) **
 alpha-HCH (alpha-BHC, alpha-lindane) **
 gamma-HCH (Lindane, gamma-BHC)
 3-Hydroxycarbofuran (Carbofuran breakdown product) ***
 Methiocarb (Slug-Geta, Grandslam, Mesuro) **
 Oxamyl (Vydate L, Pratt) **
 Parathion (Roethyl-P, Alkron, Panthion, Phoskil) *
 cis-Permethrin (Ambush, Astro, Pounce) ***
 Phorate (Thimet, Granutox, Geomet, Rampart) **
 Propoxur (Baygon, Blattanex, Uden, Proprox) ***
 Terbufos (Contraven, Counter, Pilarfox) **

1,2-Dibromo-3-chloropropane (DBCP, Nemagon)
 1,2-Dibromoethane (Ethylene dibromide, EDB)
 Dibromomethane (Methylene dibromide) *
 trans-1,4-Dichloro-2-butene ((Z)-1,4-Dichloro-2-butene) *
 1,2-Dichlorobenzene (o-Dichlorobenzene)
 1,3-Dichlorobenzene (m-Dichlorobenzene)
 1,2-Dichloropropane (Propylene dichloride)
 2,2-Dichloropropane *
 1,3-Dichloropropane (Trimethylene dichloride) *
 trans-1,3-Dichloropropene ((E)-1,3-Dichloropropene)
 cis-1,3-Dichloropropene ((Z)-1,3-Dichloropropene)
 1,1-Dichloropropene *
 Ethyl methacrylate *
 Ethyl tert-butyl ether (Ethyl-t-butyl ether (ETBE)) *
 Hexachlorobutadiene
 1,1,1,2,2,2-Hexachloroethane (Hexachloroethane)
 2-Hexanone (Methyl butyl ketone (MBK)) *
 Iodomethane (Methyl iodide) *
 Isopropylbenzene (Cumene) *
 p-Isopropyltoluene (p-Cymene) *
 Methyl acrylonitrile *
 Methyl-2-methacrylate (Methyl methacrylate) *
 Methyl-2-propenoate (Methyl acrylate) *
 Naphthalene
 2-Propenenitrile (Acrylonitrile)
 n-Propylbenzene (Isocumene) *
 1,1,2,2-Tetrachloroethane *
 1,1,1,2-Tetrachloroethane
 Tetrachloromethane (Carbon tetrachloride)
 1,2,3,4-Tetramethylbenzene (Prehnitene) *
 1,2,3,5-Tetramethylbenzene (Isodurene) *
 Tribromomethane (Bromoform)
 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon 113) *
 1,2,4-Trichlorobenzene
 1,2,3-Trichlorobenzene *
 1,1,2-Trichloroethane (Vinyl trichloride)
 1,2,3-Trichloropropane (Allyl trichloride)

Volatile organic compounds (VOCs) in ground water

These graphs represent data from 16 Study Units, sampled from 1996 to 1998



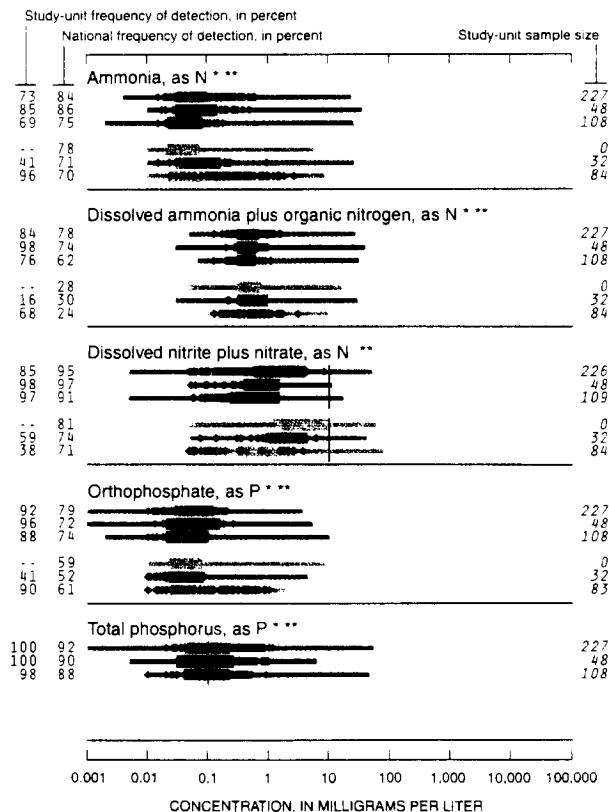
Other VOCs detected

tert-Amylmethylether (tert-amyl methyl ether (TAME)) *
 Benzene
 Bromodichloromethane (Dichlorobromomethane)
 2-Butanone (Methyl ethyl ketone (MEK)) *
 Carbon disulfide *
 Chlorobenzene (Monochlorobenzene)
 Chloroethane (Ethyl chloride) *
 Chloromethane (Methyl chloride)
 1,4-Dichlorobenzene (p-Dichlorobenzene)
 Dichlorodifluoromethane (CFC 12, Freon 12)
 1,2-Dichloroethane (Ethylene dichloride)
 1,1-Dichloroethane (Ethylidene dichloride) *
 1,1-Dichloroethene (Vinylidene chloride)
 trans-1,2-Dichloroethene ((E)-1,2-Dichloroethene)
 cis-1,2-Dichloroethene ((Z)-1,2-Dichloroethene)
 Dichloromethane (Methylene chloride)
 Diethyl ether (Ethyl ether) *
 Diisopropyl ether (Diisopropylether (DIPE)) *
 1,2-Dimethylbenzene (o-Xylene)
 1,3 & 1,4-Dimethylbenzene (m-&p-Xylene)
 1,4-Epoxy butane (Tetrahydrofuran, Diethylene oxide) *
 Ethenylbenzene (Styrene)
 1-Ethyl-2-methylbenzene (2-Ethyltoluene) *
 Ethylbenzene (Phenylethane)
 4-Methyl-2-pentanone (Methyl isobutyl ketone (MIBK)) *
 Methylbenzene (Toluene)
 2-Propanone (Acetone) *
 Tetrachloroethene (Perchloroethene)
 1,1,1-Trichloroethane (Methylchloroform)
 Trichloroethene (TCE)
 Trichlorofluoromethane (CFC 11, Freon 11)
 Trichloromethane (Chloroform)
 1,2,3-Trimethylbenzene (Hemimellitene) *
 1,2,4-Trimethylbenzene (Pseudocumene) *
 1,3,5-Trimethylbenzene (Mesitylene) *

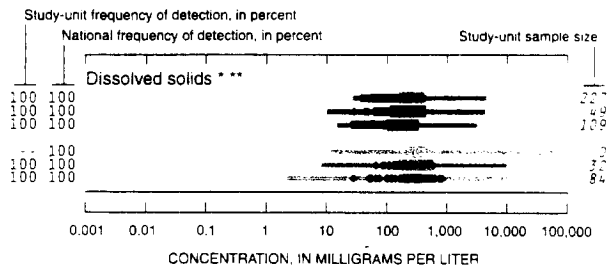
VOCs not detected

Bromobenzene (Phenyl bromide) *
 Bromochloromethane (Methylene chlorobromide)
 Bromoethene (Vinyl bromide) *
 Bromomethane (Methyl bromide)
 n-Butylbenzene (1-Phenylbutane) *
 sec-Butylbenzene *
 tert-Butylbenzene *
 3-Chloro-1-propene (3-Chloropropene) *
 1-Chloro-2-methylbenzene (o-Chlorotoluene)
 1-Chloro-4-methylbenzene (p-Chlorotoluene)
 Chlorodibromomethane (Dibromochloromethane)
 Chloroethene (Vinyl chloride)

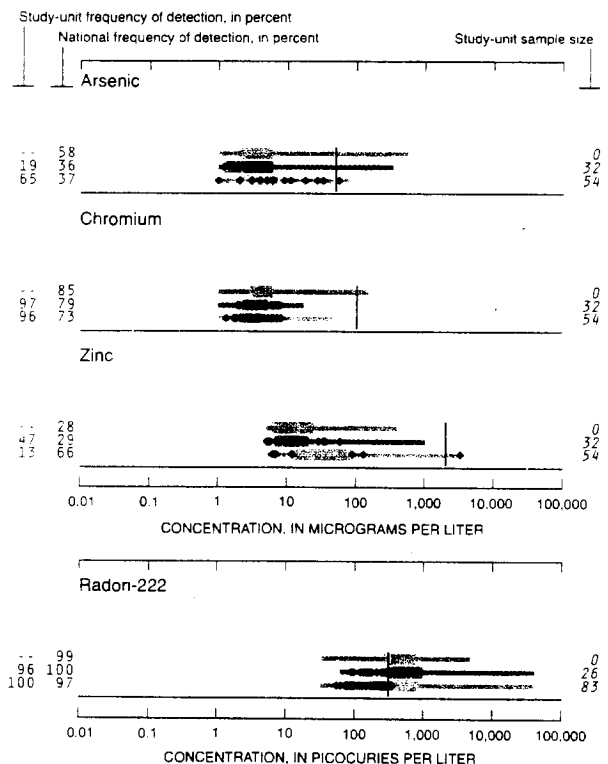
Nutrients in water



Dissolved solids in water



Trace elements in ground water



Other trace elements detected

Selenium
Uranium

Trace elements not detected

Cadmium
Lead

CHEMICALS IN FISH TISSUE AND BED SEDIMENT

Concentrations and detection frequencies, Mississippi Embayment, 1995-98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals. Study-unit frequencies of detection are based on small sample sizes; the applicable sample size is specified in each graph

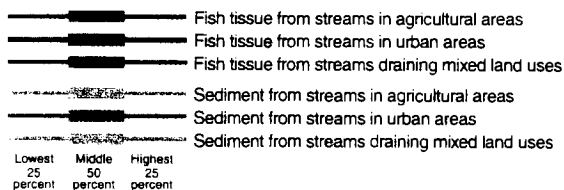
◆ Detected concentration in Study Unit

66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency

.. Not measured or sample size less than two

12 Study-unit sample size

National ranges of concentrations detected, by land use, in 36 NAWQA Study Units, 1991-98—Ranges include only samples in which a chemical was detected

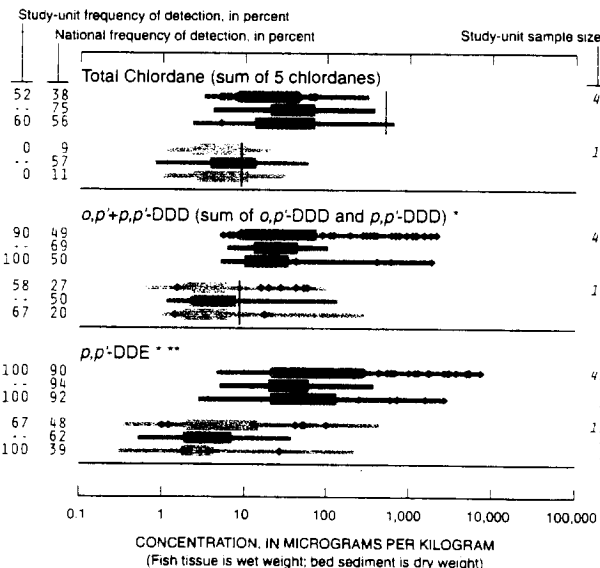


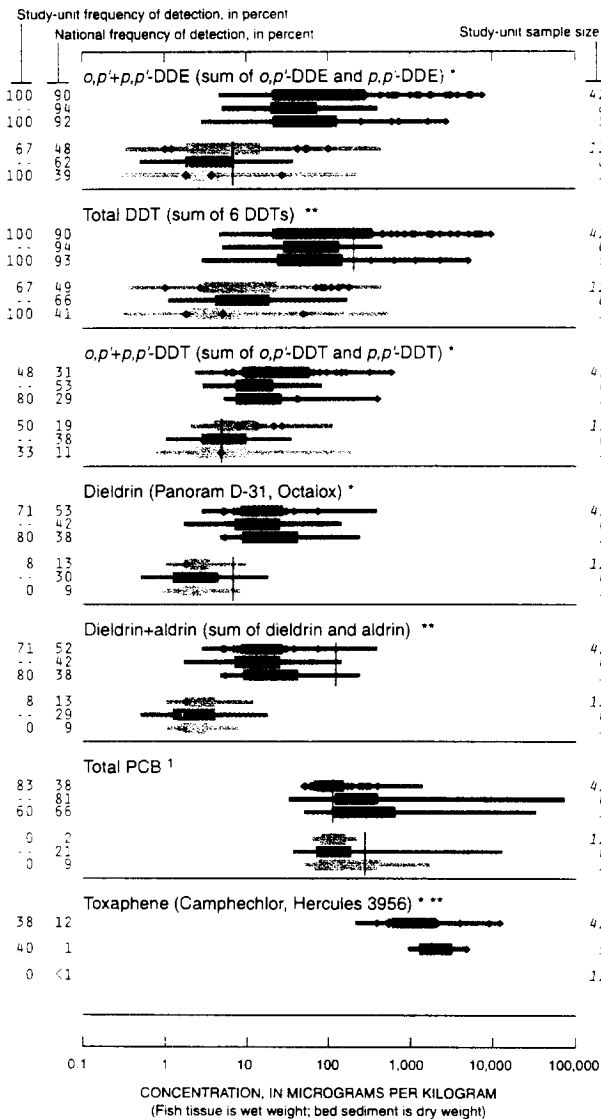
National benchmarks for fish tissue and bed sediment

National benchmarks include standards and guidelines related to criteria for protection of the health of fish-eating wildlife and aquatic organisms. Sources include the U.S. Environmental Protection Agency, other Federal and State agencies, and the Canadian Council of Ministers of the Environment

- | Protection of fish-eating wildlife (applies to fish tissue)
- | Protection of aquatic life (applies to bed sediment)
- No benchmark for protection of fish-eating wildlife
- .. No benchmark for protection of aquatic life

Organochlorines in fish tissue (whole body) and bed sediment





¹ The national detection frequencies for total PCB in sediment are biased low because about 30 percent of samples nationally had elevated detection levels compared to this Study Unit. See <http://water.usgs.gov/nawqa/> for additional information.

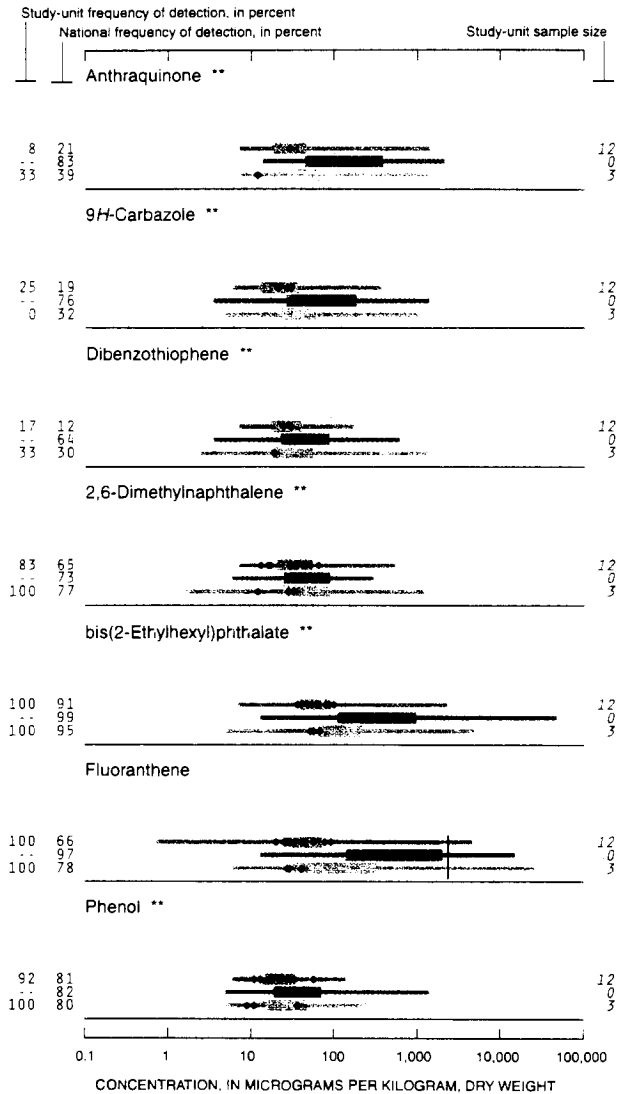
Other organochlorines detected

- Total-HCH (sum of alpha-HCH, beta-HCH, gamma-HCH, and delta-HCH) **
- Heptachlor+heptachlor epoxide (sum of heptachlor and heptachlor epoxide) **
- Hexachlorobenzene (HCB) **
- Mirex (Dechlorane) **
- Pentachloroanisole (PCA) **
- Toxaphene (Camphechlor, Hercules 3956) ***

Organochlorines not detected

- Chloroneb (Chloronebe, Demosan) ***
- DCPA (Dacthal, chlorthal-dimethyl) ***
- Endosulfan I (alpha-Endosulfan, Thiodan) ***
- Endrin (Endrine)
- gamma-HCH (Lindane, gamma-BHC, Gammexane) *
- Heptachlor epoxide (Heptachlor breakdown product) *
- Isodrin (Isodrine, Compound 711) ***
- p,p'*-Methoxychlor (Mariate, methoxychlore) ***
- o,p'*-Methoxychlor ***
- cis*-Permethrin (Ambush, Astro, Pounce) ***
- trans*-Permethrin (Ambush, Astro, Pounce) ***

Semivolatile organic compounds (SVOCs) in bed sediment



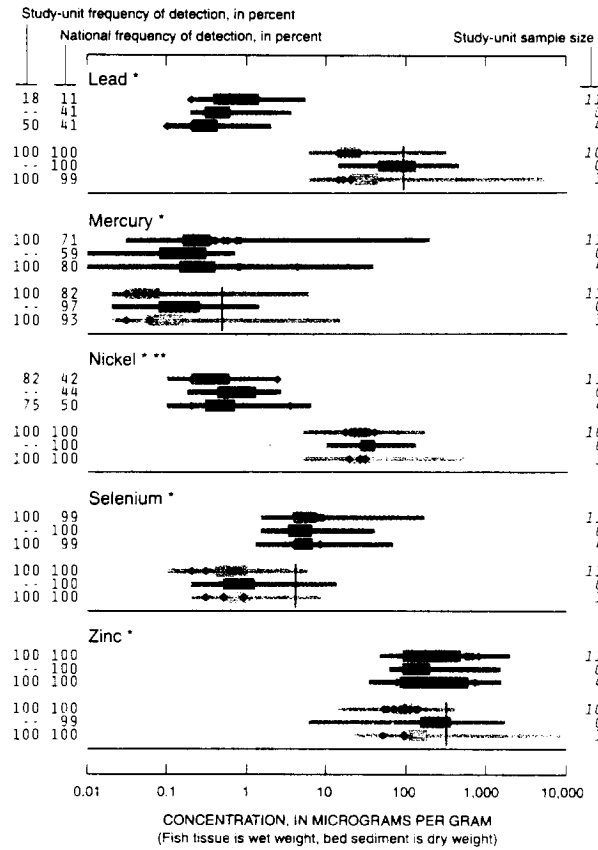
Other SVOCs detected

- Acenaphthene
- Acenaphthylene
- Acridine **
- Anthracene
- Benz[*a*]anthracene
- Benzo[*a*]pyrene
- Benzo[*b*]fluoranthene **
- Benzo[*ghi*]perylene **
- Benzo[*k*]fluoranthene **
- 2,2-Biquinoline **
- Butylbenzylphthalate **
- Chrysene
- p*-Cresol **
- Di-*n*-butylphthalate **
- Di-*n*-octylphthalate **
- Diethylphthalate **
- 1,6-Dimethylnaphthalene **
- Dimethylphthalate **
- 9H-Fluorene (Fluorene)
- Indeno[1,2,3-*cd*]pyrene **

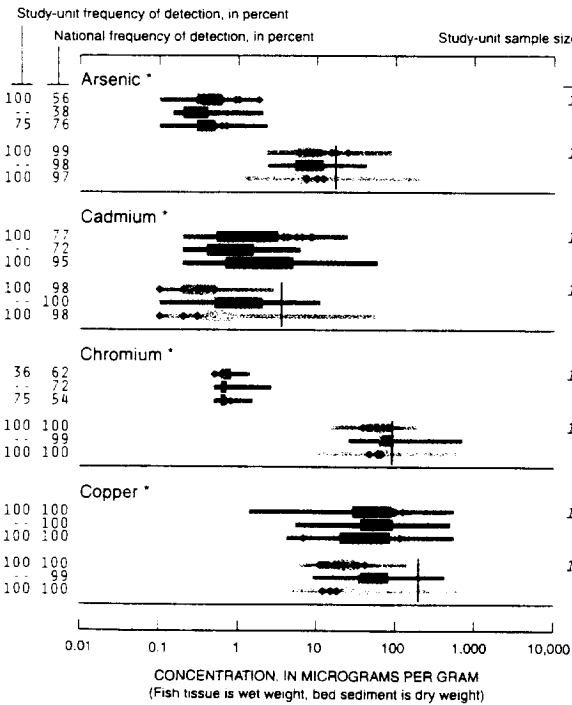
Isoquinoline **
 2-Methylantracene **
 4,5-Methylenephenanthrene **
 1-Methylphenanthrene **
 1-Methylpyrene **
 Naphthalene
 Phenanthrene
 Pyrene

SVOCs not detected

C8-Alkylphenol **
 Azobenzene **
 Benzo[c]cinnoline **
 4-Bromophenyl-phenylether **
 4-Chloro-3-methylphenol **
 bis(2-Chloroethoxy)methane **
 2-Chloronaphthalene **
 2-Chlorophenol **
 4-Chlorophenyl-phenylether **
 Dibenz[a,h]anthracene
 1,2-Dichlorobenzene (*o*-Dichlorobenzene) **
 1,3-Dichlorobenzene (*m*-Dichlorobenzene) **
 1,4-Dichlorobenzene (*p*-Dichlorobenzene) **
 1,2-Dimethylnaphthalene **
 3,5-Dimethylphenol **
 2,4-Dinitrotoluene **
 2-Ethyl-naphthalene **
 Isophorone **
 1-Methyl-9H-fluorene **
 Nitrobenzene **
 N-Nitrosodi-*n*-propylamine **
 N-Nitrosodiphenylamine **
 Pentachloronitrobenzene **
 Phenanthridine **
 Quinoline **
 1,2,4-Trichlorobenzene **
 2,3,6-Trimethylnaphthalene **



Trace elements in fish tissue (livers) and bed sediment



BIOLOGICAL INDICATORS

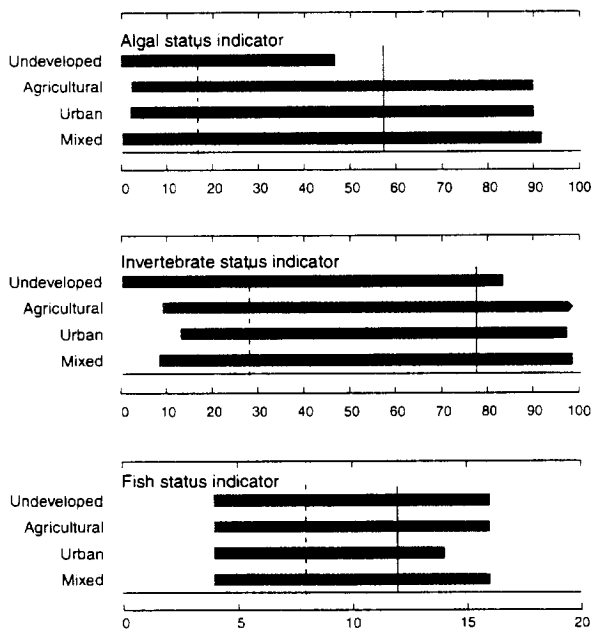
Higher national scores suggest habitat disturbance, water-quality degradation, or naturally harsh conditions. The status of algae, invertebrates (insects, worms, and clams), and fish provide a record of water-quality and stream conditions that water-chemistry indicators may not reveal. **Algal status** focuses on the changes in the percentage of certain algae in response to increasing siltation, and it often correlates with higher nutrient concentrations in some regions. **Invertebrate status** averages 11 metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. **Fish status** sums the scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that increase in association with water-quality degradation

Biological indicator value, Mississippi Embayment, by land use, 1995-98

- ◆ Biological status assessed at a site

National ranges of biological indicators, in 16 NAWQA Study Units, 1994-98

- ▬ Streams in undeveloped areas
- ▬ Streams in agricultural areas
- ▬ Streams in urban areas
- ▬ Streams in mixed-land-use areas
- 75th percentile
- - - 25th percentile



A COORDINATED EFFORT

Coordination with agencies and organizations in the Mississippi Embayment was integral to the success of this water-quality assessment. We thank those who served as members of our liaison committee.

Federal Agencies

U.S. Army Corps of Engineers
U.S. Department of Agriculture
U.S. Environmental Protection Agency
U.S. Fish and Wildlife Service
U.S. Forest Service
U.S. Geological Survey, BRD

State Agencies

Mississippi Department of Environmental Quality
Mississippi Department of Wildlife, Fisheries, and Parks

Local Agencies

Delta Council
Yazoo-Mississippi Delta Joint Water Management District

Universities

Arkansas State University
Mississippi State University
University of Arkansas
University of Mississippi

Other public and private organizations

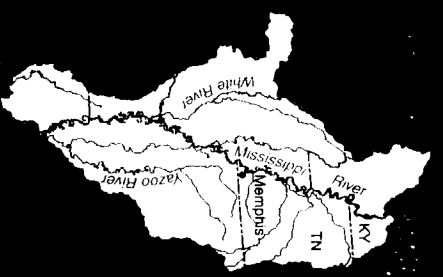
Wolf River Conservancy

We thank the following individuals, agencies, and organizations for contributing to the success of this study.

- Chip Bray, Al Gibson, David Loch, Michael Beiser, and David Singleton (Mississippi Department of Environmental Quality), and Charles Knight (Mississippi Department of Wildlife, Fisheries and Parks) for help with biological field work
- Michael Mallory for his work as the first Project Chief for this Study Unit
- Memphis Light, Gas and Water Utilities, the Mississippi Department of Transportation, and numerous towns, cities, and individuals who allowed samples to be collected on their property
- Brian Hughes and Larry Slack (USGS), Henry Folmar and Richard Ingram (Mississippi Department of Environmental Quality), and Bruce Reid (The National Audubon Society) for providing technical review of the report and Carol Moss, Rebecca Deckard, Sandra Cooper and Mary Kidd (USGS) for editorial input
- Michael Wade and Lance Cooper for graphics and layout for this report
- Todd Slack, Stephen Ross, and Neil Douglas for verifying and vouchering fish specimens
- Michael Thurman (USGS, Kansas District) for contributing analyses of selected pesticides
- USGS employees Michael Manning, Brian Caskey, Robert Joseph, Charles Lee, Darryl Wilson, Susan McIntosh, Desmond Funchess, Paul Floyd, John Storm, Kenny Burks, David Massingill, Bobby Richards, Larry Thomas, Lisa Zimmerman, Jim Alvis, Allen Roberts, Rachel Pike, Jennifer Mahaney, Shane Stocks, Michelle Wates, Fred Morris III, John Ewing, Eric Strom, Wade Bryant, Anuba Dayama, Jan Jones, Tony Schrader, Marsha Gipson, Larry Remsing, Terry Holland, Dwight Lasker, Phil Stephens, Aaron Pugh and others for assisting with collection and compilation of data

NAWQA

National Water-Quality Assessment (NAWQA) Program
Mississippi Embayment



Kleiss and others—Water Quality in the Mississippi Embayment
U.S. Geological Survey Circular 1208



ISBN 0-607-95414-0
9 780607 954142

R0024441

Water Quality in the Lower Illinois River Basin

Illinois, 1995–98



POINTS OF CONTACT AND ADDITIONAL INFORMATION

The companion Web site for NAWQA summary reports:

<http://water.usgs.gov/nawqa/>

Lower Illinois River Basin contact and Web site:

USGS State Representative
U.S. Geological Survey
Water Resources Division
221 N. Broadway Ave., Urbana, IL 61801
e-mail: dc_il@usgs.gov
<http://il.water.usgs.gov/proj/lirb>

National NAWQA Program:

Chief, NAWQA Program
U.S. Geological Survey
Water Resources Division
12201 Sunrise Valley Drive, M.S. 413
Reston, VA 20192
<http://water.usgs.gov/nawqa/>

Other NAWQA summary reports

River Basin Assessments

Albemarle-Pamlico Drainage Basin (Circular 1157)
Allegheny and Monongahela River Basins (Circular 1202)
Apalachicola-Chattahoochee-Flint River Basin (Circular 1164)
Central Arizona Basins (Circular 1213)
Central Columbia Plateau (Circular 1144)
Central Nebraska Basins (Circular 1163)
Connecticut, Housatonic and Thames River Basins (Circular 1155)
Eastern Iowa Basins (Circular 1210)
Georgia-Florida Coastal Plain (Circular 1151)
Hudson River Basin (Circular 1165)
Kanawha - New River Basins (Circular 1204)
Lake Erie - Lake Saint Clair Drainages (Circular 1203)
Las Vegas Valley Area and the Carson and Truckee River Basins (Circular 1170)
Long Island-New Jersey Coastal Drainages (Circular 1201)
Lower Susquehanna River Basin (Circular 1168)
Mississippi Embayment (Circular 1208)
Ozark Plateaus (Circular 1158)
Potomac River Basin (Circular 1166)
Puget Sound Basin (Circular 1216)

Red River of the North Basin (Circular 1169)
Rio Grande Valley (Circular 1162)
Sacramento River Basin (Circular 1215)
San Joaquin-Tulare Basins (Circular 1159)
Santee River Basin and Coastal Drainages (Circular 1206)
South-Central Texas (Circular 1212)
South Platte River Basin (Circular 1167)
Southern Florida (Circular 1207)
Trinity River Basin (Circular 1171)
Upper Colorado River Basin (Circular 1214)
Upper Mississippi River Basin (Circular 1211)
Upper Snake River Basin (Circular 1160)
Upper Tennessee River Basin (Circular 1205)
Western Lake Michigan Drainages (Circular 1156)
White River Basin (Circular 1150)
Willamette Basin (Circular 1161)

National Assessments

The Quality of Our Nation's Waters—Nutrients and Pesticides (Circular 1225)

Front cover: Summer wildflowers at Indian Creek near Wyoming, Ill. (photograph by David J. Fazio).

Back cover: Left, hydrologists collecting a water sample from Panther Creek near El Paso, Ill. (photograph by Paul J. Terrio); right, biologists collecting invertebrate samples from the Illinois River at Valley City, Ill. (photograph by David J. Fazio).

R0024443

Water Quality in the Lower Illinois River Basin, Illinois, 1995–98

By George E. Groscheñ, Mitchell A. Harris, Robin B. King, Paul J. Terrio, *and*
Kelly L. Warner

U.S. DEPARTMENT OF THE INTERIOR
BRUCE BABBITT, Secretary

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

2000

Free on application to the
U.S. Geological Survey
Information Services
Box 25286 Federal Center
Denver, CO 80225

Or call: 1-888-ASK-USGS

Library of Congress Cataloging-in-Publications Data

Water quality in the Lower Illinois River Basin, Illinois, 1995-98 / by George E. Groschen...[et al].
p. cm. -- (U.S. Geological Survey Circular ; 1209)

Includes bibliographical references.

ISBN 0-607-95423-X (alk. paper)

1. Water quality--Illinois--Illinois River Watershed. I. Groschen, G. E. II. Geological Survey (U.S.)
III. Series.

TD224.I3 W37 2000
363.739'42'097735--dc21

00-049464

R0024445

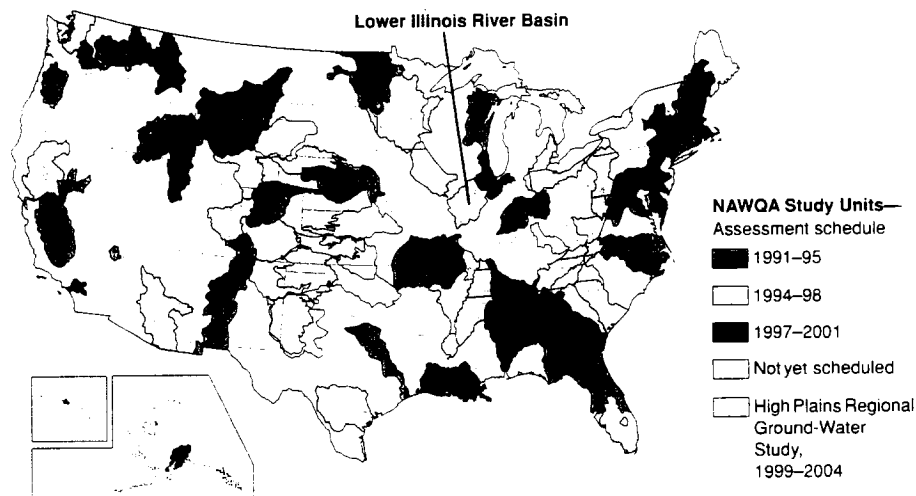
CONTENTS

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM	IV
SUMMARY OF MAJOR FINDINGS	1
Stream and river highlights	1
Ground-water highlights	2
INTRODUCTION TO THE LOWER ILLINOIS RIVER BASIN	3
Land use	3
Study design in relation to land use	4
Water use	4
Hydrologic conditions during the study period	4
The lower Illinois River	5
MAJOR FINDINGS	6
Nutrient concentrations in streams and rivers are a major concern	6
Stream and river samples often exceeded the nitrate drinking-water standard and phosphorus guideline	7
Nitrate concentrations were lower than in other areas of the Corn Belt during late summer	7
Nitrate concentrations differed upstream and downstream from water-supply reservoirs	8
NATIONAL PERSPECTIVE—Algal index indicates comparatively poor water quality in corn belt	9
Riparian buffer zones influence the quality of Midwestern streams and rivers	10
NATIONAL PERSPECTIVE—Nutrient concentrations were among the highest in the Nation	11
Nitrate concentrations in some shallow wells exceeded the drinking-water standard	12
Some pesticides were always detected in streams and rivers	12
Pesticide concentrations in streams periodically exceeded drinking-water standards	13
Pesticide concentrations in streams have a seasonal pattern	14
Many factors affect the presence and concentration of pesticides in streams	14
Concentrations of agricultural chemicals generally reflect application rates	15
Stream size affects pesticide concentrations	15
Concentrations of alachlor and cyanazine in the Illinois River decreased	16
Pesticide breakdown products were detected more frequently than the parent compound	16
Pesticides were not frequently detected in ground water	17
Insecticides discontinued in the 1970s were still detected in fish and sediment	17
NATIONAL PERSPECTIVE—Basin and national results were similar for pesticides	19
Other organochlorine compounds are persistent in sediments and fish	20
Common stream animals indicate poor water-quality conditions	20
Highest concentrations of polycyclic aromatic hydrocarbons were in bed sediment near urban areas	21
Arsenic exceeded the drinking-water standard in the Mahomet aquifer	22
NATIONAL PERSPECTIVE—Arsenic in ground water is a national concern	22
Radon exceeded the proposed drinking-water standard in almost one-half of shallow wells	23
Volatile organic compounds were detected in ground water at low concentrations	24
Fish and sediment in the Illinois River have high concentrations of trace elements	24
STUDY UNIT DESIGN	25
GLOSSARY	27
REFERENCES	28
APPENDIX—WATER QUALITY FROM THE LOWER ILLINOIS RIVER BASIN IN A NATIONAL CONTEXT	30

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

THIS REPORT summarizes major findings about water quality in the lower Illinois River Basin that emerged from an assessment conducted between 1995 and 1998 by the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program. Water quality is discussed in terms of local and regional issues and compared to conditions found in all 36 NAWQA study areas, called Study Units, assessed to date. Findings are also explained in the context of selected national benchmarks, such as those for drinking-water quality and the protection of aquatic organisms. The NAWQA Program was not intended to assess the quality of the Nation's drinking water, such as by monitoring water from household taps. Rather, the assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water monitoring programs. The comparisons made in this report to drinking-water standards and guidelines are only in the context of the available untreated resource. Finally, this report includes information about the status of aquatic communities and the condition of in-stream habitats as elements of a complete water-quality assessment.

Many topics covered in this report reflect the concerns of officials of State and Federal agencies, water-resource managers, and members of stakeholder groups who provided advice and input during the lower Illinois River Basin assessment. Basin residents who wish to know more about water quality in the areas where they live will find this report informative as well.



THE NAWQA PROGRAM seeks to improve scientific and public understanding of water quality in the Nation's major river basins and ground-water systems. Better understanding facilitates effective resource management, accurate identification of water-quality priorities, and successful development of strategies that protect and restore water quality. Guided by a nationally consistent study design and shaped by ongoing communication with local, State, and Federal agencies, NAWQA assessments support the investigation of local issues and trends while providing a firm foundation for understanding water quality at regional and national scales. The ability to integrate local and national scales of data collection and analysis is a unique feature of the USGS NAWQA program.

The lower Illinois River Basin is one of 51 water-quality assessments initiated since 1991, when the U.S. Congress appropriated funds for the USGS to begin the NAWQA Program. As indicated on the map, 36 assessments have been completed, and 15 more assessments will conclude in 2001. Collectively, these assessments cover about one-half of the land area of the United States and include water resources that are available to more than 60 percent of the U.S. population.

SUMMARY OF MAJOR FINDINGS

Stream and river highlights

During the past century, agricultural runoff, channel and drainage modifications, urbanization, and other activities have altered water quality, aquatic biological communities, and aquatic habitat in the lower Illinois River Basin. The water quality of large rivers, such as the Illinois and Sangamon Rivers, was more likely to meet drinking-water standards than water quality of small streams during 1995–98. In spring, concentrations of nitrate—especially in small streams—exceeded the Maximum Contaminant Level (MCL) of 10 mg/L (milligrams per liter) for drinking water. Some of these streams replenish drinking-water reservoirs. In samples collected during runoff from spring and early summer storms, concentrations of herbicides and a few insecticides exceeded drinking-water standards or guidelines, or guidelines to protect aquatic life. In a few samples from small streams, concentrations of commonly used agricultural pesticides were among the highest nationally. Although most concentrations were low with respect to existing drinking-water standards or guidelines, criteria for the protection of human health or wildlife have not been established for more than one-half of the chemicals detected.

- Nitrogen and phosphorus concentrations were among the highest in the Nation. The highest concentrations in the basin were found in small streams in agricultural areas. The MCL for nitrate was exceeded in 15 percent of samples from all streams and rivers. The concentration of total phosphorus in most samples (79 percent) from all streams and rivers exceeded the 0.1-mg/L guideline recommended by U.S. Environmental Protection Agency (USEPA) to prevent excess algal growth in streams. (See pages 6–8.)
- Nitrate concentrations in the Illinois River at the inflow to the basin (Ottawa) and outflow from the basin (Valley City) were similar; however, approximately twice the amount of nitrogen was transported out of the basin (124,000 tons per year) as was transported into the basin (66,000 tons per year). (See page 6.)
- During August 1997, concentrations of nitrate in streams were lower in the basin than in other NAWQA study-area streams of the upper Midwest. Algal communities may have incorporated much of the instream nitrogen, resulting in lower nitrate concentrations in the water during late summer. (See pages 7–10.)
- Three herbicides commonly used by farmers to protect corn and soybean crops—atrazine, metolachlor, and cyanazine—were detected in every sample collected during 1995–98. During periods of spring runoff, these herbicides exceeded drinking-water standards or guidelines or aquatic-life guidelines. Another herbicide, acetochlor, was detected in most samples (81 percent). (See pages 12–17.)
- Pesticide breakdown products were detected much more frequently than the parent compound, and generally



The lower Illinois River Basin occupies approximately 18,000 square miles in central and west-central Illinois. The basin lies almost entirely within the Till Plains physiographic section. The glacial materials account for the flat prairie landscape and the thick, rich soils. The three physiographic subsections influence water quality indirectly by influencing the type and intensity of agricultural activities. [1]

at higher concentrations and for a longer period of time after application. (See pages 16–17.)

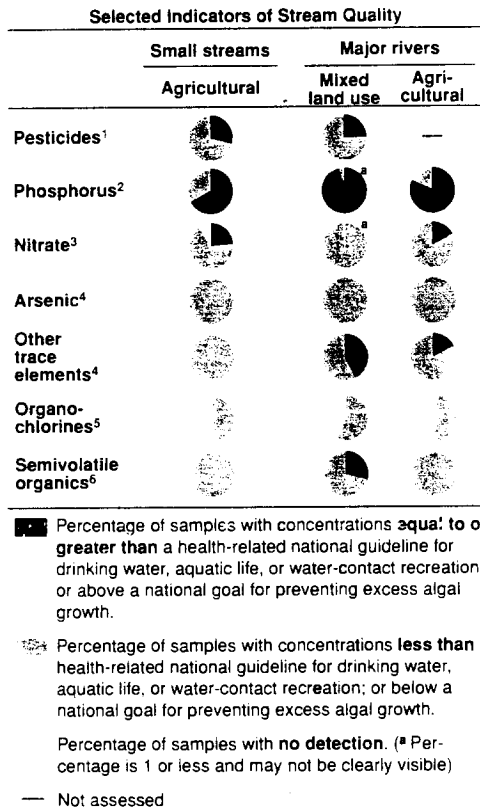
- Organochlorine compounds (including insecticides no longer used) were detected in fish tissue and sediment, and polycyclic aromatic hydrocarbons were detected in sediments at levels of concern. (See pages 17–22.)
- The biological communities of streams in the basin comprised many organisms that are tolerant of poor water quality; however, some high-velocity streams had fairly diverse invertebrate communities. (See pages 20–21.)

Trends in stream-water quality

For decades, fertilizers and many pesticides have been applied to crops and land. The persistence of pesticides and breakdown products in the soil, water, and sediment within a watershed is not well understood. Concentrations of the herbicides alachlor and cyanazine, however, decreased from the 1991–92 period to the 1996–98 period in the Illinois River because of reduced application rates since the early 1990s. (See page 16.)

Major influences on streams and rivers

- Subsurface (tile) drainage from agricultural areas
- Agricultural and urban land-surface runoff
- Drainage modification to streams and channels



Ground-water highlights

In contrast to the water quality of streams and rivers in the basin and the quality of ground water in other areas across the Nation, agricultural chemicals in ground-water samples from shallow monitoring wells (generally less than 100 feet deep) and drinking-water wells only rarely exceeded the nitrate MCL. Except for radon and nitrate, shallow ground water in the lower Illinois River Basin generally met drinking-water standards or guidelines. Except for radon and arsenic, the water quality in the Mahomet aquifer (greater than 200 feet deep) meets all drinking-water standards or guidelines.

- Major corn and soybean herbicides were not as frequently detected in ground-water samples as they were in stream-water samples. No ground-water sample exceeded

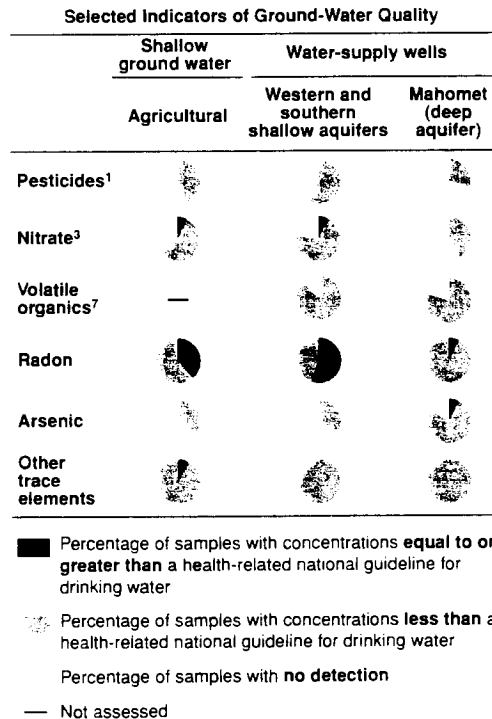
¹ Insecticides, herbicides, and pesticide breakdown products, sampled in water.
² Total phosphorus, sampled in water.
³ Nitrate (as nitrogen), sampled in water.
⁴ Arsenic, mercury, and trace metals, sampled in sediment.
⁵ Organochlorine compounds including DDT and PCBs, sampled in sediment.
⁶ Miscellaneous industrial chemicals and combustion by-products, sampled in sediment.
⁷ Solvents, refrigerants, fumigants, and gasoline compounds, sampled in water.

drinking-water standards or guidelines for pesticides. (See page 17.)

- Naturally occurring arsenic exceeded the current MCL of 50 µg/L (micrograms per liter) in 2 of 30 wells sampled in the Mahomet aquifer, a major drinking-water source. If the MCL is lowered to 5 µg/L, as proposed by the USEPA, samples from 60 percent (18 of 30) of the domestic (private household) and public-supply (publicly owned wells generally serving a community) wells sampled would probably exceed the lower standard. (See page 22.)
- Geologic materials underlying the basin indicate that it is an area of potentially high radon concentrations in ground water. In about one-half of the samples of shallow ground water, radon exceeded the proposed MCL of 300 pCi/L (picocuries per liter). Only 2 of 30 samples from the Mahomet aquifer exceeded the proposed radon MCL. (See pages 23–24.)
- Volatile organic compounds (VOCs) were detected in samples from 83 percent of shallow domestic and public-supply wells, but no samples exceeded drinking-water standards or guidelines. VOCs were detected in 80 percent of samples from the Mahomet aquifer but at concentrations near the method detection limit and well below drinking-water standards and guidelines. (See page 24.)

Major influences on ground water

- Agricultural and urban land uses
- Permeability of soil and aquifers
- Minerals in geological materials



INTRODUCTION TO THE LOWER ILLINOIS RIVER BASIN

The lower Illinois River Basin (the basin) encompasses 18,000 mi² (square miles) of central and west-central Illinois between the upper end at Ottawa and the confluence of the Illinois River with the Mississippi River near Grafton. The basin includes all of 22 counties and parts of 19 counties.

As of 1990, 1.3 million people lived in the basin. More than 50 percent of the population lived in the counties of Macon, McLean, Peoria, Sangamon, and Tazewell. The four most populated cities are Peoria, Springfield, Decatur, and Bloomington. The basin population decreased by 7 percent from 1980 to 1990 [2].

Land use

Agriculture is the predominant land use—typically corn and soybean row crops. Agriculture accounts for 88 percent of the overall land area, whereas forests account for 7 percent and urban areas account for about 2 percent (fig. 1). The remaining land use, about 3 percent, is mostly grassland, wetland, or water.

The small-stream basins that were sampled during 1995–98 have even higher percentages of land devoted to farming; for example, the Panther Creek Basin is 99 percent agricultural land (fig. 2). The lower Illinois River Basin has some of the most highly productive and intensively cultivated farmland in the world. As a result, application rates of farm-related chemicals are among the highest in the country in support of corn and soybean production. Runoff from these vast, flat farm areas carries relatively high concentrations of nutrients and pesticides into the

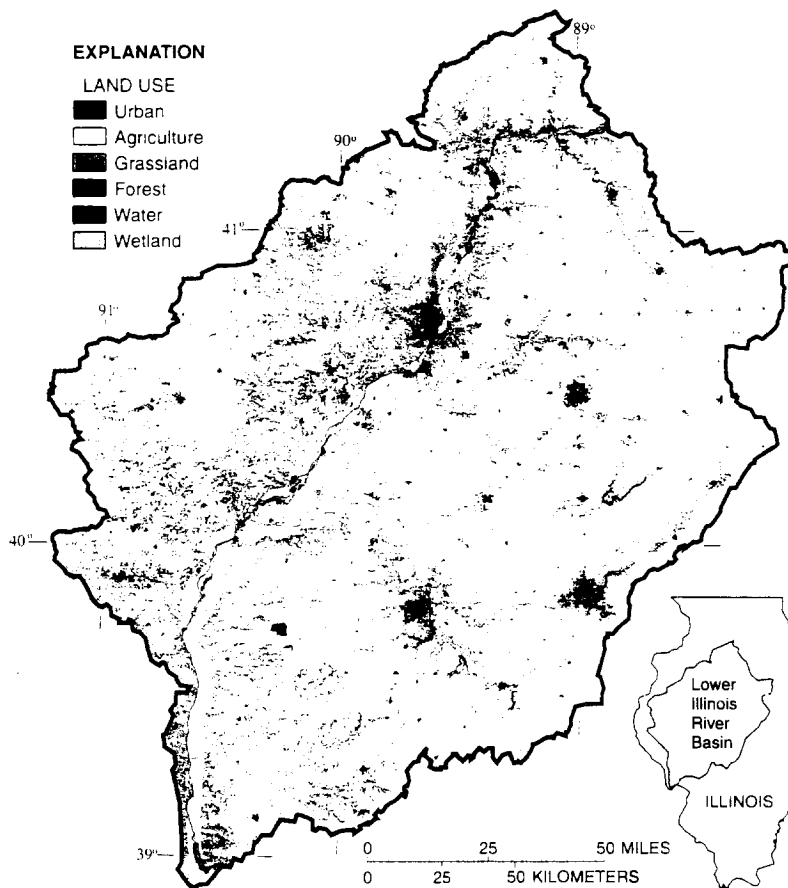


Figure 1. Land use in the lower Illinois River Basin is predominantly agriculture. Farming is most intensive in the part east of the Illinois River. Streambanks generally are steeper west of the Illinois River and are more forested than streambanks east of the river.

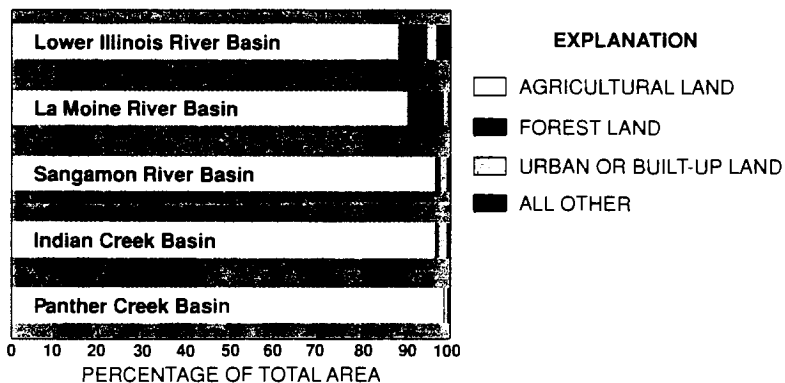


Figure 2. Proportion of land devoted to agriculture varies only slightly among basins—from 88 percent in the entire lower Illinois River Basin to 99 percent in the Panther Creek Basin.

small streams and major rivers of the basin.

Animal farming is also a significant industry in the basin [2], and runoff from manure spreading is a contributor to some of the highest levels of nitrate and total nitrogen found in the rivers and streams of the United States.

Study design in relation to land use

Chemical and biological samples were collected from a range of river and stream sizes in different landscape types and from shallow aquifers. A deep aquifer (Mahomet aquifer) was sampled to assess the overall water quality. A group of the shallow large-bore wells used for domestic (private household) supply was sampled, as well as other shallow domestic wells, to assess overall water quality and determine the occurrence of human-induced contaminants and the extent of land-use effects on recently recharged ground water.

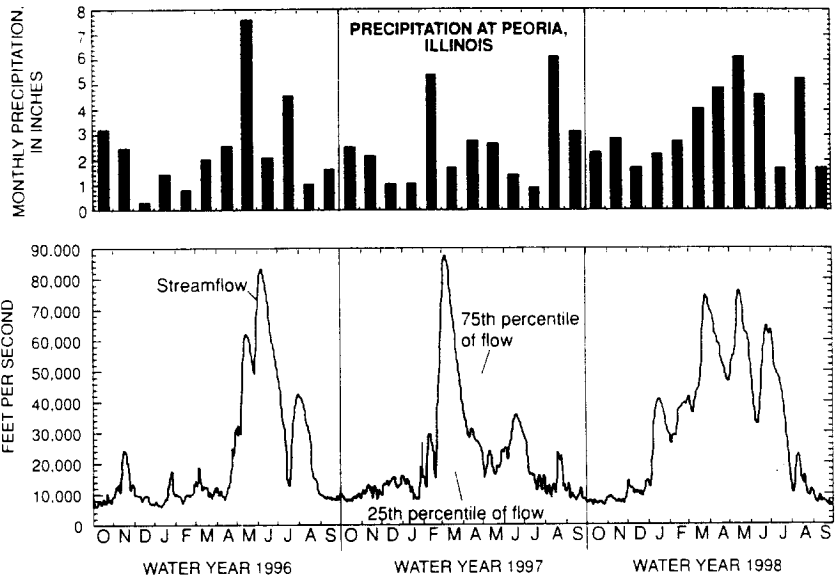


Figure 4. Hydrologic conditions during the data-collection period were near normal. Some relatively dry periods resulted in below-normal flow, and several wet periods resulted in above-normal flow on the Illinois River at Valley City.

Water use

In the lower Illinois River Basin, 48 percent of the public- or municipal-supply water is drawn from ground-water sources and 52 percent from surface-water sources (fig. 3). All private-supply

water, usually for domestic use, is from ground water. Rural residents (about 25 percent of the population) generally are self-supplied from ground water [2]. In most of the western and southern parts of the basin, homeowners rely on cisterns or large-diameter dug or bored wells [3]. These large-diameter wells are highly susceptible to contamination from surface-water runoff and shallow ground water.

Irrigation, which is confined to sandy soils near the center of the basin, is entirely supplied by ground water.

Hydrologic conditions during the study period

Rainfall generally was near normal during the data-collection period (fig. 4). Typically, the largest rainstorms are during April to July. The generally normal rainfall had exceptions: in particular, water year 1997 (October 1996–September 1997) was slightly drier than

Figure 3. Water withdrawals in the lower Illinois River Basin are dominated by thermoelectric withdrawals (cooling water for power generation). Drinking water is supplied roughly equally from surface-water reservoirs and ground-water supplies.

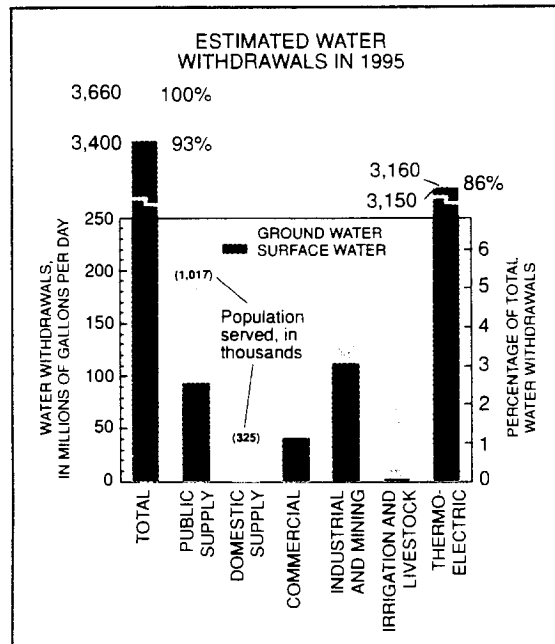


Figure 5. Ground-water conditions were near normal during 1989–98, including the data-collection period. In 1993, the observation-well water level shows the effect of the significant 1993 flood on the shallow ground water. (Data from K. Hlinka, Illinois State Water Survey, and National Weather Service.)

normal, and spring 1996 had fewer storms than normal.

Ground-water levels reflected the near-normal rainfall conditions, particularly in the southern part of the basin (fig. 5). Shallow ground water typically is recharged in late winter and early spring, then the level falls during the summer growing season. Water level in a representative well is shown in figure 5, along with 10 years of annual rainfall data collected near Peoria, which lies near the center of the basin.

The lower Illinois River

The Illinois River is a navigable waterway that connects the Mississippi River to Chicago and the Great Lakes. Barges carrying grain and other commodities are a significant influence on suspended sediment and water quality. The mean annual flow of the Illinois River increases from about 12,600 ft³/s (cubic feet per second) at Ottawa to 22,600 ft³/s at Valley City [2]. Five major tributaries—Vermilion River, Spoon River, Mackinaw River, La Moine River, and Sangamon River—and many smaller streams join the Illinois River (fig. 6). Flow gaging and sampling of the outflow from the basin is done at Valley City because the water level of the Mississippi River, at times, affects the flow of the Illinois River downstream from Valley City.

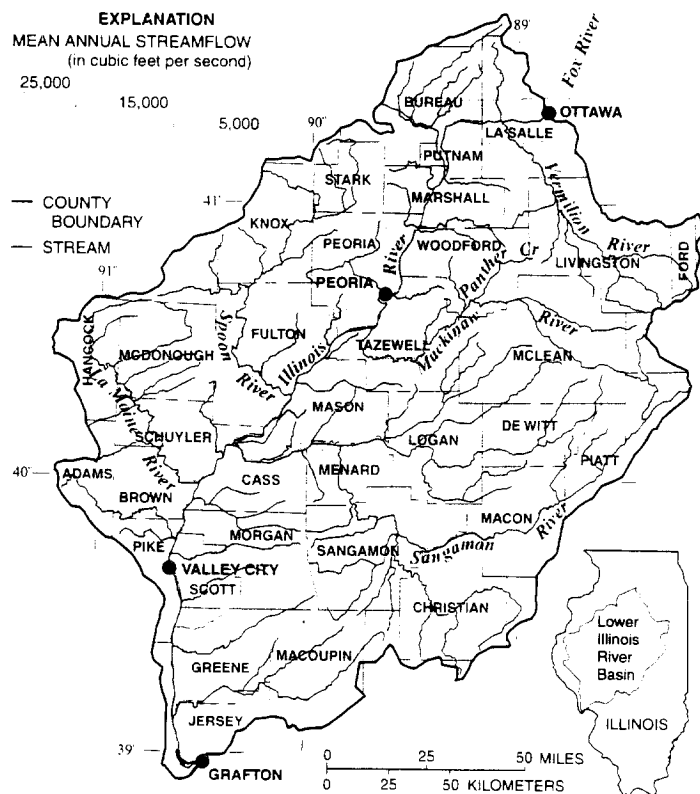
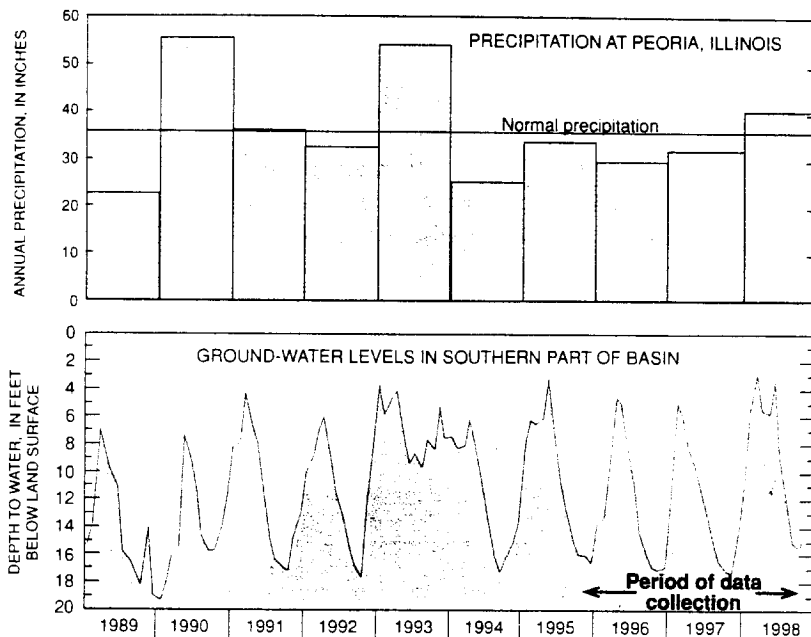


Figure 6. The lower Illinois River has five major tributaries. The inflow to the basin is at Ottawa, Ill., where the Fox River joins the Illinois River. Where the Illinois River leaves the basin and enters the Mississippi River near Grafton, Ill., the outflow is almost double that of the inflow.

MAJOR FINDINGS

Nutrient concentrations in streams and rivers are a major concern

Agriculture is the most important activity in the lower Illinois River Basin from an economic and water-quality perspective (fig. 7). Application rates of nitrogen and phosphorus in synthetic fertilizer and manure are among the highest in the Nation. The high application rates of agrichemicals in the basin were reflected by the kinds and amounts of contaminants found in streams, rivers, and ground water. Nevertheless, contaminants associated with urban or industrial activities also were detected in some streams, rivers, and wells.



Figure 7. Farming and related businesses are major activities in the lower Illinois River Basin. Agriculture has a significant effect on water quality and wildlife habitat. (Photograph by David J. Fazio, USGS.)

Flow-weighted concentrations of nitrate (nitrate as nitrogen) were generally highest in small streams in intensive agricultural areas and east of the Illinois River (fig. 8). The lowest flow-weighted mean concentration of nitrate was for the Illinois River at Valley City—the outflow from the basin. Concentrations of nutrients were similar at the basin inflow (Illinois River at Ottawa) and the basin outflow (Illinois River at Valley City). Urban sources, primarily wastewater effluent, however, were major contributors at the inflow [4], whereas agriculture provided most of the nutrient contribution to the water resources within the basin.

The nitrate load (amount of nitrate in water that moves past a location on the river) measured at the outflow from the basin, 124,000 ton/yr (tons per year), was nearly twice the load measured at the inflow, 66,000 ton/yr (fig. 9). Flow-weighted mean concentrations of nitrate are similar between the two Illinois River locations; therefore, the difference in loads is

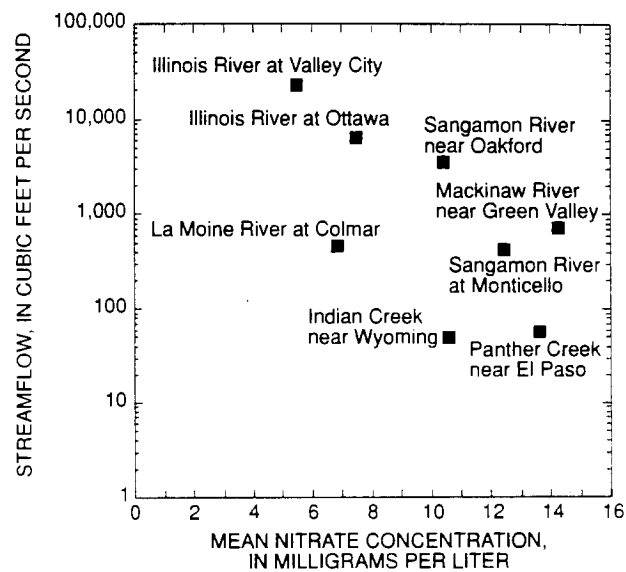


Figure 8. Mean nitrate concentrations generally are lowest in the Illinois River. Nitrate concentrations in the three largest rivers show a downward trend with increasing streamflow from the Sangamon River near Oakford to the Illinois River at Ottawa and to the Illinois River at Valley City. The La Moine River, which is the only river or stream sampled that lies entirely within the Galesburg Plain physiographic subsection, does not fit the general trend. The nitrate concentrations are flow-weighted means.

directly related to the difference in streamflow (fig. 6). The lower Illinois River Basin contributes roughly the same amount of nutrients to the Mississippi River as does the upper Illinois River Basin, which includes the

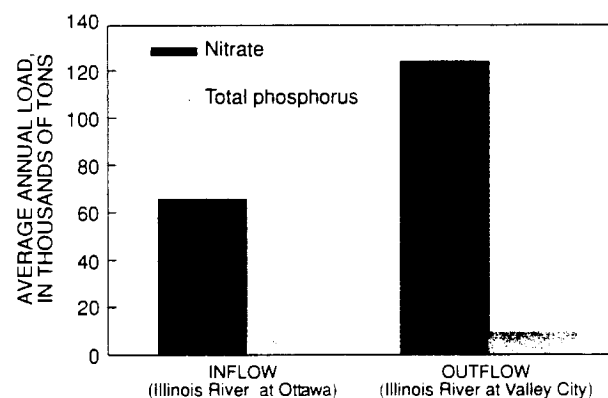


Figure 9. About twice as much nitrate and total phosphorus leaves the lower Illinois River as enters it. The loads for the upper Illinois River Basin—the basin upstream from Ottawa—are about one-half of the respective loads for the lower Illinois River Basin.

What Was Assessed?

The NAWQA Program was not intended to assess the quality of the Nation's drinking water, such as by monitoring water from household taps. Rather, NAWQA assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water monitoring programs. Comparisons made in this report to drinking-water standards and guidelines are made only in the context of the available untreated resource.

Chicago urban area and rural areas in Illinois, Indiana, and Wisconsin.

The annual nitrogen yield (amount of nitrogen in water that flows out of the watershed divided by the area of the watershed) measured at the river and stream locations ranged from 4.6 to 12 (ton/mi²)/yr (tons per square mile per year), the smallest yields being measured in the largest river and the largest yields measured in the smallest streams. The overall yield for the basin (Illinois River at Valley City) was 4.6 (ton/mi²)/yr during 1995–98.

Substantial differences among nutrient concentrations in small agricultural basins and the lower nutrient concentrations in large rivers indicate that hydrological and biochemical processes reduce the nitrate concentration as nitrate moves through the basin and into the Mississippi River. Instantaneous concentrations of nutrients varied with season: nitrate concentrations were highest during the winter and spring and lowest in the late summer and fall.

Sites on the largest rivers—the Illinois River at Ottawa and Valley City and the Sangamon River near Oakford—had the highest flow-weighted mean concentrations of

total organic nitrogen, ammonia nitrogen, and phosphorus; the latter two are indicators of urban wastewater effluent [4]. The Sangamon River near Oakford had the highest flow-weighted mean concentration of total organic nitrogen, ammonia nitrogen, and phosphorus. Samples from middle-sized La Moine River at Colmar generally had higher flow-weighted mean concentrations of most nutrients compared with samples from the Sangamon River at Monticello, which also is middle sized but is at the eastern edge of the basin. Flow-weighted mean concentrations of nitrate in samples from the Sangamon River at Monticello, however, were greater than those in samples from the La Moine River at Colmar. Panther Creek near El Paso, a small-basin site, generally had higher concentrations of most nutrients than did Indian Creek near Wyoming, another small-basin site.

Stream and river samples often exceeded the nitrate drinking-water standard and phosphorus guideline

Flow-weighted mean concentrations of nitrate for five of the eight sample locations in the basin, including two locations on the Sangamon River (used for public water supplies), were higher than the Maximum Contaminant Level (MCL) of 10 mg/L nitrate as nitrogen established for drinking water.

The concentration of total phosphorus in most samples (80 percent) from all streams and rivers exceeded the 0.1 mg/L guideline recommended by USEPA to prevent eutrophication in streams. Samples from Indian Creek near Wyoming had the highest concentrations of total phos-

Streamflow-Weighted Concentrations

Drinking-water standards are generally based on the yearly average of samples collected quarterly. This is a legal definition of yearly average concentrations. Truly representative “average” water-quality conditions of a stream would require daily samples for some streams because of the variability in the concentration and in streamflow from day to day—especially over a period of storm runoff. During a storm, the levels of contaminants in a stream can vary 10 times or more from beginning to end. A method is needed to address this variability and “weigh” the concentrations found in individual “point” samples from a stream. A process is used to estimate the daily load or amount of a chemical that flows past a point. The daily loads are summed and then divided by the amount of streamflow to get an average for the period of interest. This flow weighting produces a more accurate mean concentration of a specific chemical for a period such as a year or, as for this report, for the 2–3 year period when samples were collected.

phorus, and samples from Panther Creek near El Paso had the smallest. Indian Creek near Wyoming is downstream from a small wastewater-treatment plant, which is likely the reason for the higher total phosphorus at this site.

Nitrate concentrations were lower than in other areas of the Corn Belt during late summer

During August 1997, concentrations of dissolved nitrate and total nitrogen were found to be lower in streams and rivers of the lower Illinois River Basin than in some other areas of the Midwest. A synoptic study was done at 70 sites with drainage areas between 100

and 1,000 mi² in Illinois, Iowa, and Minnesota in August 1997, when streamflow had approximately reached the annual minimum [5]. Median concentrations of dissolved nitrate in streams generally increased northwestward, from Illinois (0.52 mg/L) to Iowa (1.5 mg/L) to Minnesota (3.5 mg/L). No significant relations between nitrate concentrations and streambank characteristics or soil permeability were identified. Streams that have formed on windblown glacial materials, however, generally had lower concentrations than streams formed on glacial till (unsorted material left behind by continental glaciers). The windblown materials typically have slightly higher permeability than the glacial till and are more likely to allow rainwater to infiltrate rather than run off to streams.

In Illinois, rainfall and thus runoff to streams was low before and during synoptic sampling; therefore, most streamflow was derived from ground-water seepage. Thus the low nutrient concentrations in ground water in the basin—much lower than those in streams and rivers—are a major reason why low nitrate concentrations were detected in the streams during August 1997 in the basin.

Differences in moisture conditions and precipitation during the 3-month period before the August sampling (wettest in Minnesota and driest in Illinois) probably also contributed to the differences in concentrations across the three-State region. The percentages of total nitrogen present as organic nitrogen and ammonia were higher in Illinois than in other parts of the study area. Nitrogen incorporation in algae also was a factor in the distribution of dissolved nitrate. Algal communities, particularly phytoplankton, were more stable and productive in Illinois than in other parts of the three-State synoptic study, and much of the nitrogen in the aquatic system was incorporated into these plant communities in the basin.

Nitrate concentrations differed upstream and downstream from water-supply reservoirs

Dissolved nitrate concentrations in samples from the Sangamon River (at Monticello) were found to be more variable upstream from several water-supply reservoirs than downstream from the reservoirs (fig. 10). From April through June, nitrate concentrations were frequently above the MCL of 10 mg/L (highest concentration, 17 mg/L) in the Sangamon River at Monticello, 29 mi upstream from Lake Decatur, the drinking-water source for the city of Decatur.

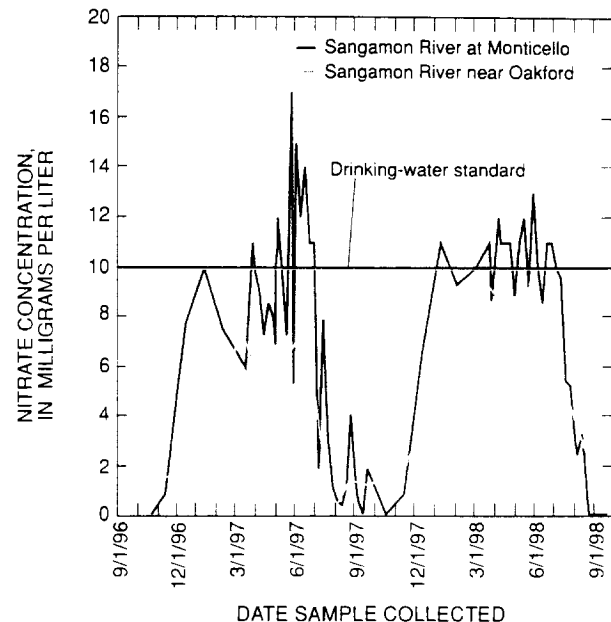


Figure 10. Nitrate concentration in the Sangamon River is affected by reservoirs and other factors. Monticello lies upstream from several water-supply reservoirs and the cities of Springfield and Decatur. Oakford lies downstream from the reservoirs. The reservoirs are for water supply, and during spring they are replenished by river water that exceeds the drinking-water standard for nitrate. Water leaving the reservoirs usually does not exceed the nitrate standard and has less variability in nitrate concentrations.

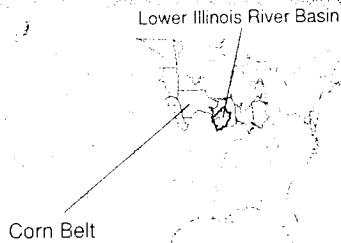
The city of Decatur often must treat water for nitrate above the MCL during spring and early summer or mix the high-nitrate surface water with ground water to lower the nitrate concentration. Nitrate concentrations also were highest in the spring in the Sangamon River near Oakford, which is downstream from Lake Decatur and other water-supply reservoirs used by the city of Springfield. Concentrations in samples from the Sangamon River near Oakford, however, seldom exceeded the nitrate MCL.

Conversely, concentrations of nitrate usually were less than 1 mg/L in the Sangamon River at Monticello from August through November, but typically remained higher than 1 mg/L at the Oakford site during the same period. Data from Scribner and others [6] indicate that nitrate concentrations of reservoirs in corn- and soybean-producing areas of the Midwest have nutrient concentrations and patterns similar to those in the Sangamon River near Oakford.

Algal index indicates comparatively poor water quality in Corn Belt



Small streams in the Corn Belt have both poorer water quality, as indicated by the algal status index, and higher nitrate concentrations than small streams in other parts of the Nation. The algal status index reflects a greater relative abundance of benthic algal species that are considered tolerant of poor water-quality conditions, including high nutrient concentrations.



The lower Illinois River Basin lies within the eastern one-half of the Corn Belt.

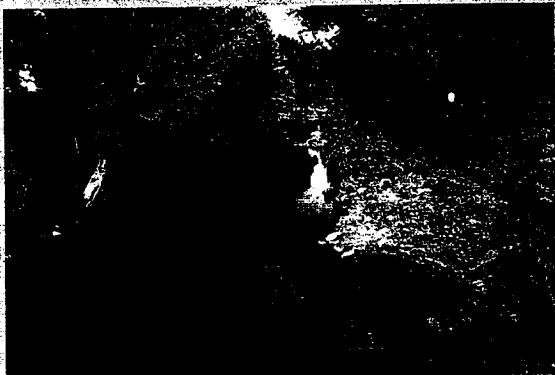
Results for this study are from data collected in 16 NAWQA Study Units from 1996 to 1998.

Algae grow on virtually all stream surfaces receiving light. Algal communities are the foundation of the food web of many streams, serving as a food source for invertebrates and fish.

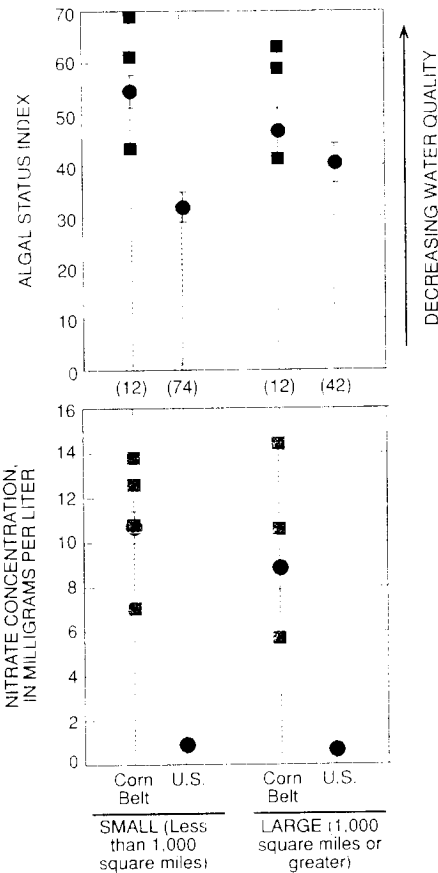
Nuisance algal growth can result from excess nutrients and can lead to large variations in dissolved oxygen concentrations.

In large streams, flow-weighted mean concentrations of nitrate were also higher in the Corn Belt than in the rest of the Nation, but the algal status index was not significantly higher. The dynamics of the algal community change with stream size, and the importance of phytoplankton (floating algae) increases as streams get larger. The algal status index does not reflect the composition of the phytoplankton community, and this may explain the differences observed between small and large streams.

Chemical, physical, and biological factors all influence the algal community and local stream conditions. The algal status index at individual lower Illinois River Basin streams was similar to that at other streams in the Corn Belt.



Excess nutrients often cause algal blooms that can be seen in streams and rivers in the basin. Algae reduce the amount of nutrients that remain in the water by converting nutrients into algal biomass. (Photograph by Paul J. Terrio, USGS.)



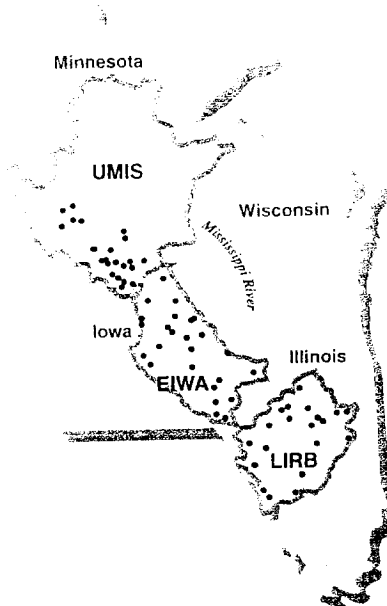
GEOGRAPHIC REGION AND STREAM BASIN SIZE EXPLANATION

- Algal index mean for Corn Belt or U.S. (●) Two standard errors of the mean
 - Nitrate mean for Corn Belt or U.S. (●) Two standard errors of the mean
 - Algal index for samples from lower Illinois River Basin streams or rivers
 - ▣ Nitrate mean for samples from lower Illinois River Basin streams or rivers
- Numbers in parentheses are samples in each group

Algal status index values and nitrate concentrations for the lower Illinois River Basin, and for Study Units in the Corn Belt, are higher than mean values for Study Units in the Nation.

Riparian buffer zones influence the quality of midwestern streams and rivers

Despite similar land use throughout the Corn Belt region of the Midwest, streams flowing through cropland differ considerably in their ecological characteristics, in part because of differences in riparian buffer zones (see text boxes). This conclusion is based on an investigation of 70 streams and rivers within three NAWQA Study Units in the upper Midwest during August 1997 (figure at right) [5; 7]. Specifically, increases in tree cover in buffer zones were associated with aquatic biological communities indicative of good stream quality, reduced nuisance algal growths, and maintenance of sufficient dissolved oxygen concentrations to support diverse communities of aquatic organisms. For example, the number of aquatic insects indicative of good stream quality tended to increase with increases in percentage of tree cover, especially in sites where streamflow and dissolved oxygen conditions were favorable. Fish communities, sampled at 24 sites in the UMIS Study Unit, also indicated better overall conditions in streams with wooded riparian zones than those with more open canopy [8].

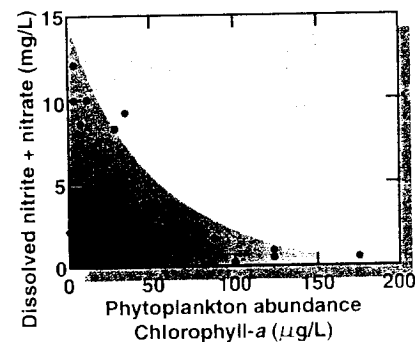


The influence of riparian buffer zones on the quality of 70 midwestern streams and rivers was evaluated in the Upper Mississippi River (UMIS), Eastern Iowa (EIWA), and lower Illinois River Basins (LIRB).

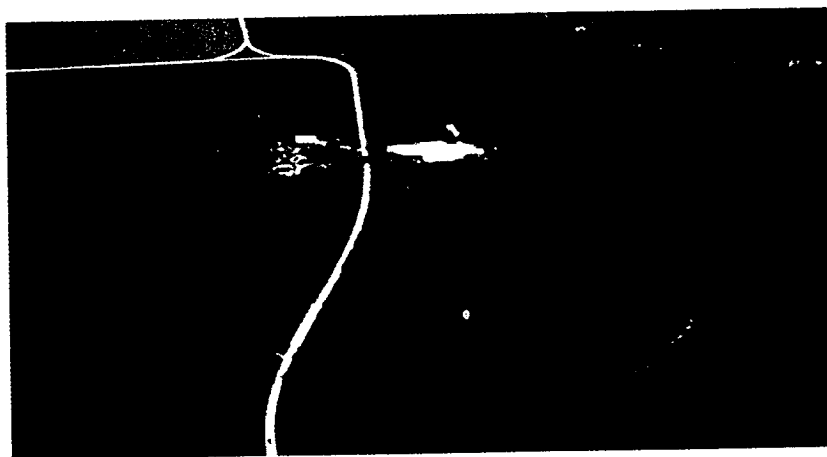
Streams with less tree cover, and thus less shading, contained relatively large growths of phytoplankton (algae suspended in the water) at levels considered indicative of eutrophication [9].

Organic enrichment resulting from excessive algal production in some midwestern streams may reduce dissolved oxygen concentrations and be detrimental to other requirements of aquatic organisms.

Shading from tree cover in riparian buffer zones may influence nutrient concentrations indirectly by reducing the growth of phytoplankton. In streams where phytoplankton were abundant (often where buffer zones were thin or lacking), dissolved nitrate concentrations were significantly lower (figure below) [9]. The lower nutrient concentrations may result from uptake by the abundant phytoplankton. Thus, assessments of eutrophication would benefit from consideration of biological communities and the riparian zone, rather than being based solely on nutrient concentrations in the water.



Dissolved nutrient concentrations decreased in eutrophic streams with excessive algal productivity. Rates of nutrient uptake by the algae can exceed rates at which nutrients are transported by streams during low-flow conditions.



Digital images derived from USGS topographic maps were used to estimate the percentage of trees in a riparian buffer zone (a 100-meter width on each side of the stream) for 2- to 3-mile segments upstream from each sampling site, supplemented by vegetation surveys at the sampling site [5].

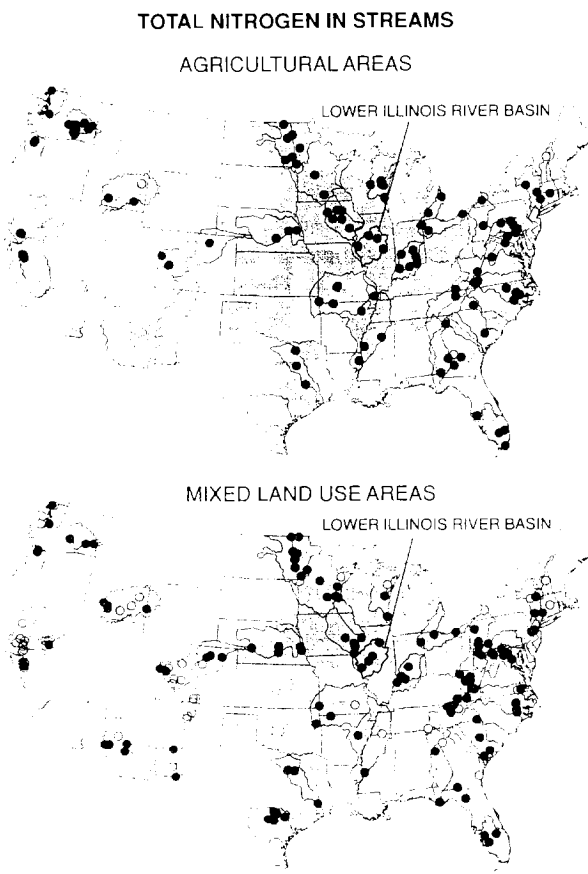
Resource agencies, including the U.S. Department of Agriculture, encourage maintenance of strips of trees or grass between cropland and streams as a best management practice. These "riparian buffer zones" are thought to intercept runoff of sediment and chemicals from fields, promote bank stability, and provide shading and habitat for aquatic life [10]. Riparian buffer zones should be considered along with other important factors that affect chemical and biological indicators of stream quality, such as soil drainage properties and stream hydrology [7].



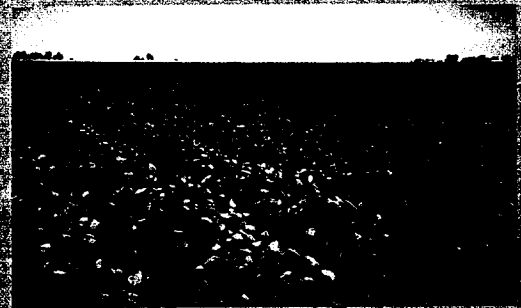
Nutrient concentrations were among the highest in the Nation

Concentrations, loads, and yields of nutrients in the lower Illinois River Basin were among the highest in the Nation. When compared with data from NAWQA studies nationwide, nitrate concentrations in the lower Illinois River Basin were in the highest 10 percent at every sampling location, and total phosphorus concentrations were in the highest 25 percent at seven of the eight locations.

Soybeans, which are a major crop in the lower Illinois River Basin, are important for nutrient management and the nitrogen balance of agricultural watersheds. The roots of soybeans harbor bacteria that fix nitrogen from the atmosphere into the soil in a form that can be used by the plant, and by other crops the following year. (Photograph by Kelly L. Warner, USGS.)



- EXPLANATION**
- AVERAGE ANNUAL TOTAL NITROGEN INPUT, IN POUNDS PER ACRE, BY COUNTY, FOR 1995-98**
 INPUTS ARE FROM FERTILIZER, MANURE, AND THE ATMOSPHERE
- GREATER THAN 25 POUNDS PER ACRE
 - 6 TO 25 POUNDS PER ACRE
 - LESS THAN 6 POUNDS PER ACRE
- AVERAGE ANNUAL CONCENTRATION OF TOTAL NITROGEN, IN MILLIGRAMS PER LITER**
- HIGHEST (GREATER THAN 2.9)
 - MEDIUM (0.64 TO 2.9)
 - LOWEST (LESS THAN 0.64)



Of the 10 largest rivers monitored by NAWQA, greater than 20,000 cfs mean daily streamflow, the lower Illinois River Basin site at Valley City had the highest flow-weighted mean nitrate and total nitrogen concentrations (see Streamflow-Weighted Concentrations). The concentrations were greater than twice the concentrations of both nitrate and total nitrogen of the next highest ranked location—Mississippi River at Hastings, Minnesota—and of a comparable size river location—Mississippi River at Red Wing, Minnesota.

Goolsby, Battaglin, and others [11, 12] concluded that the basin is one of the areas of highest nitrogen yield (amount of nitrogen runoff into streams per unit area) contributing to the Mississippi River. Alexander and others [13] estimated that 50 percent to greater than 90 percent of the nitrogen exported from much of the basin is transported to the Gulf of Mexico, where excess nutrients help create hypoxia—oxygen concentrations below the level required by most aquatic life in the Gulf. The Illinois River contributes from 15 to 20 percent of the total nitrogen that goes into the Gulf of Mexico from the Mississippi River [14, 12].

Roughly between 61,000 and 124,000 tons of nitrogen is transported annually from the lower Illinois River Basin to the Gulf of Mexico. More than one-half of the excess nitrogen entering the Gulf from the Mississippi River basin thought to come from the upper Mississippi basin—basin that is an upstream from St. Louis, Missouri. The Illinois River joins the Mississippi River just north of St. Louis.

Nitrate concentrations in some shallow wells exceeded the drinking-water standard

The 117-well network sampled in the basin consisted of 57 shallow monitoring wells installed for the study (fig. 11) and 60 water-supply wells of various types. Although the concentrations of nitrate in none of the 30 samples from the deep Mahomet aquifer exceeded the MCL (all concentrations were below 0.10 mg/L), nitrate concentrations in the shallow wells exceeded the MCL in 7 percent (6 of 87) of the shallow-well samples. Concentrations above the MCL ranged from 11 to 77 mg/L. All exceedances of the nitrate MCL were in samples from wells less than 50 feet deep.

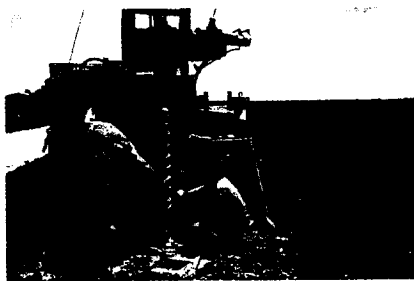


Figure 11. Study-Unit staff installed 57 wells to sample recently recharged ground water. Assistance and geological expertise was provided by the Illinois State Geological Survey. (Photograph by William S. Morrow, Jr., USGS.)

An important reason for fewer detections and much lower concentrations of nitrate and other agrichemicals in ground water is artificial drainage (where present) and the geological materials from which the soils developed. Most farmland in Illinois is artificially drained by ditches and subsurface drains (tile drains) because of the poor natural drainage of the soils

and their parent materials (fig. 12). Artificial drainage short circuits infiltrating water to streams through the subsurface drains rather than allowing the water to move downward to the water table or to deeper aquifers [15]. Subsurface drainage also decreases the amount of potential runoff across land surface to streams because the soil is less saturated and ponding is less likely [15]. Another factor that helps prevent agrichemicals from reaching wells is that the fine-grained materials may be conducive to rapid breakdown of many of the chemicals.

One-half of the shallow wells installed for monitoring recently recharged ground water (28 of 57) were installed in the geological materials overlying the Mahomet aquifer. The shallow ground water in this area has a high potential for eventually recharging the underlying Mahomet aquifer [2]. Sampling results from these wells indicate the Mahomet aquifer is fairly well protected from contamination at land surface.

Some pesticides were always detected in streams and rivers

Herbicides are applied during spring planting to virtually all corn and soybean crops in the basin. Insecticides are applied during the summer to about 10 to 30 percent of the corn crop, depending on weather conditions. Approximately 6,000 to 6,700 tons of agricultural pesticides were applied annually in the lower Illinois River Basin during 1996–98 (data from [16–18]).

Atrazine, which is commonly applied to corn in the basin, and its breakdown products (hydroxyatrazine, deisopropylatrazine, and



Figure 12. Subsurface drainage allows water with high concentrations of agrichemicals to be transported directly to ditches and streams. The outlet pipe from a subsurface drain at center of photograph is draining into the stream. (Photograph by Kelly L. Warner, USGS.)

deethylatrazine) were the most frequently detected pesticide and related compounds in streams throughout the basin and shallow ground water beneath cropland. Atrazine was detected in every stream sample collected from the basin, even during winter. Prometon, which is not used in appreciable amounts on cropland but commonly is applied for weed control in rights-of-way in the basin, was detected frequently in streams (93 percent of samples) and infrequently in shallow ground water (7 percent of samples).

In samples collected at Illinois River at Valley City (basin outflow), 40 pesticides or pesticide breakdown products were detected. Atrazine and metolachlor were always present, and cyanazine was detected in 97 percent of the samples.

Because the method detection limit varies widely from one pesticide or related compound to another, a common reporting level is used to compare detections of the pesticides and compounds on an equal basis. In figure 13, the frequency of detection of pesticides in stream and river samples and ground-water samples is shown

at a common reporting level of 0.05 µg/L. Of the pesticides detected in 1 percent or more of samples, those shown in figure 13, only 2,4-D had a higher method detection limit than 0.05 µg/L; therefore the detection frequency of 2,4-D may be underestimated in figure 13. Even at the common reporting level, atrazine was detected in almost all stream and river samples.

Pesticide concentrations in streams periodically exceeded drinking-water standards

Concentrations of atrazine and cyanazine in some individual samples exceeded USEPA drinking-water standards or guidelines. Annual average concentrations of each compound in streams and rivers, however, were below their respective MCL (atrazine: 3 µg/L) or lifetime health advisory level (HAL) (cyanazine: 1 µg/L). Carbofuran, diazinon, chlorpyrifos, and carbaryl were the only insecticides detected. Diazinon concentration exceeded the Great Lakes water-quality objective (0.08 µg/L) [19] in one sample at Illinois River at Valley City during August 1998.

Some pesticide concentrations that exceeded human health standards or guidelines also were potentially toxic to aquatic life. For example, an atrazine concentration of 110 µg/L in a single sample from the La Moine River at Colmar (fig. 14) exceeded not only the USEPA MCL (3 µg/L) but also the Canadian guideline for the protection of freshwater aquatic life (1.8 µg/L) [20]. In samples from the Illinois River at Ottawa and at Valley City and the Sangamon River at Oakford, all exceedances of the cyanazine HAL and the

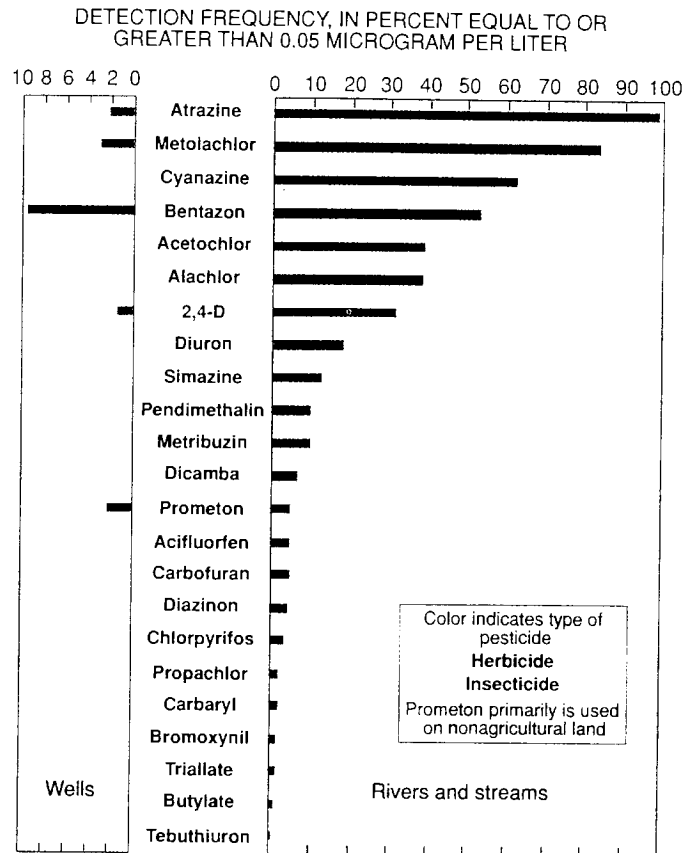


Figure 13. Many commonly applied pesticides were frequently detected in streams and rivers in the lower Illinois River Basin. Detections in ground water were far fewer, and some of the pesticides frequently detected in surface water (such as cyanazine, acetochlor, and alachlor) were not detected in shallow ground water at or above the common reporting level of 0.05 micrograms per liter.

metolachlor aquatic-life guideline (7.8 µg/L) were concurrent with an exceedance of the atrazine MCL. In contrast, diazinon concentrations twice exceeded the Great Lakes water-quality objective and one chlorpyrifos concentration exceeded the aquatic-life guideline (0.041 µg/L) independent of all other pesticide exceedances.

In samples from the La Moine River at Colmar and the Sangamon River at Monticello, for every pesticide other than atrazine, concentrations exceeded a drinking-water standard or guideline concurrent with an atrazine MCL exceedance. The lone exception was cyanazine, which exceeded the HAL independent of all other standard or guideline exceedances.

Currently (2000), drinking-water standards or guidelines and aquatic life guidelines have been established only for individual pesticides. Furthermore, USEPA drinking-water MCLs or HALs are established for only 25 of the 40 pesticides detected in the lower Illinois River Basin. Pesticides, however, commonly are found in mixtures of as many as 21 compounds in surface water (fig. 15). Although most samples of shallow ground water contained relatively few detectable concentrations of pesticides compared to samples from streams and rivers, two or more pesticides were detected

in all samples where any of pesticides were detected at all. The health effects of such combinations of pesticides in drinking water, even in low concentrations, are not well understood.

Pesticide concentrations in streams have a seasonal pattern

Herbicide concentrations in stream and river samples are highest in late spring to early summer (fig. 16), and insecticide concentrations typically peak in midsummer. The highest pesticide concentrations typically occur during the first one or two periods of storm runoff after pesticide application. During seasonal peaks, concentrations of pesticides in samples of stream water exceeded drinking-water standards or guidelines or aquatic-life guidelines. For example, for about 6 weeks after herbicide application in the spring, concentrations of atrazine at Illinois River at Valley City were greater than the MCL of 3 µg/L. The annual mean atrazine concentration in the Illinois River at Valley City for 1996 through 1998, however, never exceeded the MCL. Peak concentrations of the insecticides diazinon and chlorpyrifos were measured during July or August of 1997 or 1998.

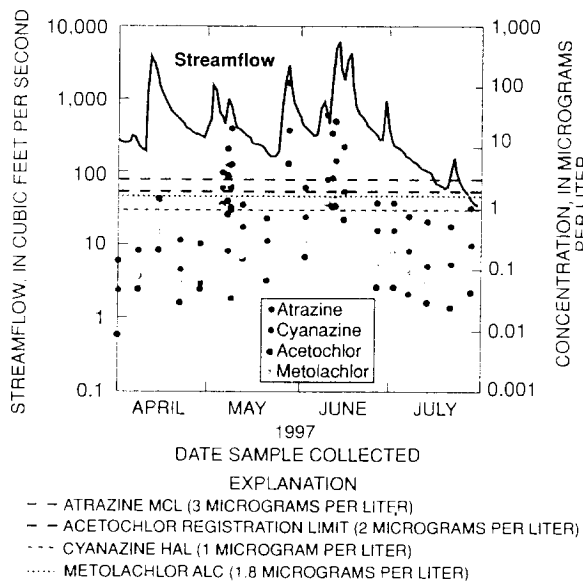


Figure 14. During late spring and early summer, many pesticides exceeded drinking-water standards or guidelines or aquatic-life guidelines. The graph shows concentrations of selected pesticides in the La Moine River at Colmar during part of 1997. (Registration limit for acetochlor is from [21]; ALC, Aquatic-Life Criterion from [20]; HAL, Health Advisory Level.)

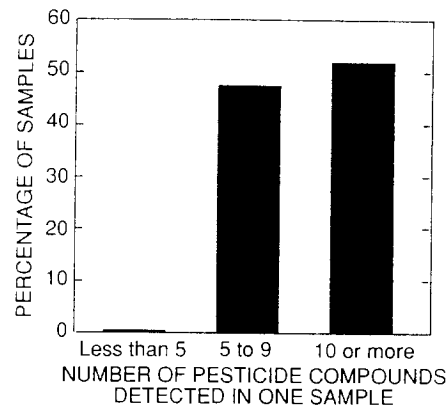


Figure 15. Most samples from streams and rivers contained multiple pesticides and related compounds. Five or more compounds were detected in more than 90 percent of samples. The maximum number of compounds detected at or above the method detection limit in one sample was 21.

Insecticides are often applied later in the growing season than herbicides.

Many factors affect the presence and concentration of pesticides in streams

The presence and concentration of pesticides in streams in the lower Illinois River Basin are influenced

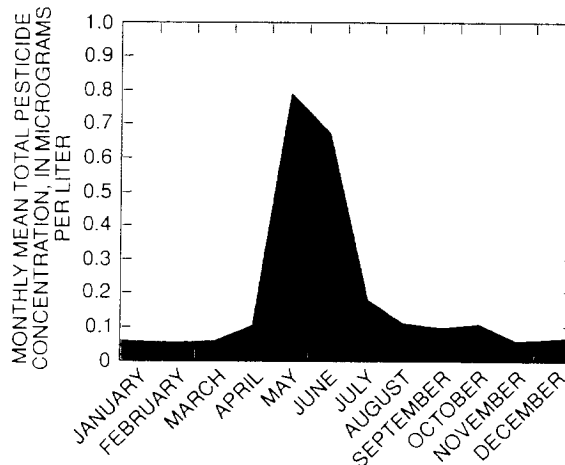


Figure 16. Pesticide concentrations vary seasonally. The sum of pesticide concentrations in samples peaks in late spring during runoff from rainstorms. Often, runoff from spring rains supplies reservoirs on some rivers and streams with a substantial amount of the water storage for the remainder of the year. Monthly mean totals are for all locations sampled for pesticides during October 1996–September 1998.

by land use and agricultural practices (fig. 17) such as pesticide application rate, drainage of crop land, and cropping methods. Natural factors, such as the size of the streams, type of terrain, and soil characteristics, also are important determinants. In addition, the distribution of individual pesticides in streams throughout a watershed depends on the physical and chemical properties of the pesticides that are in use—properties such as sorption (the ability of a pesticide to stick to soil particles), transformation (the ease with which a pesticide breaks down in the environment), and volatilization (the tendency of a pesticide to become gaseous and rise into the atmosphere).

Concentrations of agricultural chemicals generally reflect application rates

The most heavily applied pesticides (more than 1 million pounds of active ingredient per year;

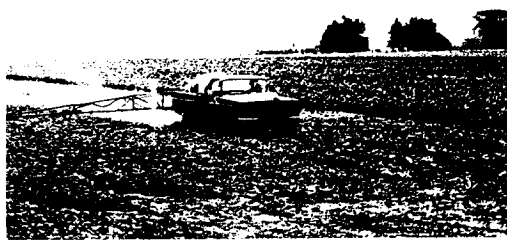


Figure 17. Pesticides are used to protect crops and increase production. Pesticides are often sprayed directly on the plants in the field. Other pesticides are incorporated into the soil, partly to limit the amount of chemical that leaves the field. (Photograph by Kelly L. Warner, USGS.)

acetochlor, atrazine, cyanazine, and metolachlor) were generally the most frequently detected in streams and shallow aquifers. Herbicides such as alachlor and simazine, which are no longer applied as heavily as in previous decades, still are detected at relatively high rates. Bentazon is not applied as heavily as other herbicides, yet it is frequently detected in stream and well samples (fig. 18).

Pesticide concentrations in water also were commonly related to drainage of water over and through agricultural soils, pesticide uptake by plants and microbes, and attachment to soil particles. Pesticide and nutrient persistence and regional differences in soils, geology, and climate govern the distribution of

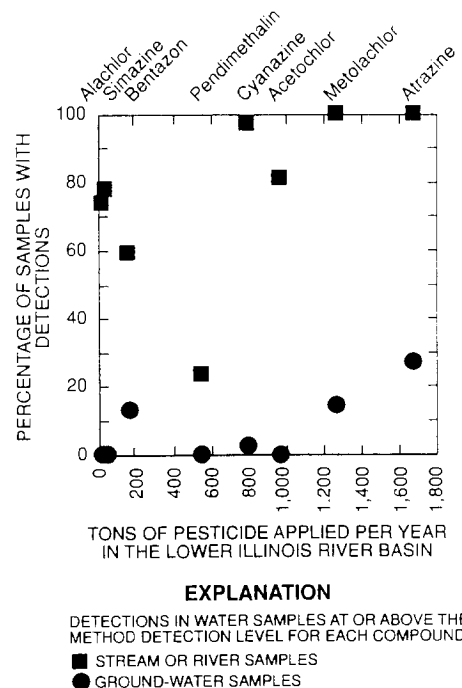


Figure 18. Detection rates in streams are mainly affected by the amount of pesticide applied in the basin. Detection rates in ground water, even for those pesticides that are heavily applied, are much lower.

these water-quality indicators in the lower Illinois River Basin.

The relatively infrequent detection of metolachlor in well samples may be related to the following factors:

1. Metolachlor is applied after weeds sprout and emerge, and it is taken up by plants and is converted to other compounds.
2. Soil microbes effectively degrade metolachlor.
3. Soils retain metolachlor instead of allowing it to run off or seep downward to ground water.

Stream size affects pesticide concentrations

In the lower Illinois River Basin, pesticide concentrations in small streams typically range from the highest to the lowest observed concentrations, whereas concentrations in large rivers are moderate and less variable. For example, during the period when most pesticides are washed into streams (May through July), the maximum concentration of atrazine measured in the Sangamon River at Monticello was about twice as high as that measured in the Illinois River. The highest concentrations of several pesticides, including atrazine (110 µg/L; May 1997), acetochlor (12 µg/L; May 1998), cyanazine (16 µg/L; May 1997), and 2,4-D (7.3 µg/L; May 1998) were detected in samples from the La Moine River at Colmar. In contrast,

in months other than May or June, 70 percent (35 of 50) of samples from Sangamon River at Monticello had total pesticide concentrations that were less than 1 µg/L, as compared with 79 percent (26 of 33) of the samples from Illinois River at Valley City. Samples for analysis of pesticides were not collected from the smallest streams monitored.

Concentrations of alachlor and cyanazine in the Illinois River decreased

Application rates of cyanazine decreased during the 1996–98 sampling period in anticipation of its use being eliminated in 2003. Alachlor use has declined since the early 1990s—in fact, no use of alachlor was reported for 1998 in the basin. The use of alachlor, by voluntary agreement [21], is decreasing as acetochlor replaces it (fig. 19).

Cyanazine was detected in 97 percent of the river and stream samples and alachlor was detected in 74 percent of the samples during 1996–98. A comparison of mean monthly concentrations of those detections, however, shows a different story. During 1991–92, samples that were comparable to those collected as part of this NAWQA study were collected from the Illinois River at Valley City. The mean monthly concentrations for the two periods, separated by about 4–5 years or longer, indicate that concentrations of alachlor and cyanazine have substantially decreased (fig. 20).

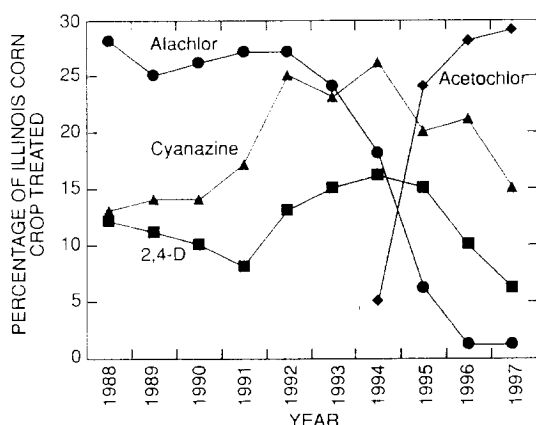


Figure 19. Use of herbicides on corn and soybeans in Illinois changed during 1988–97. Use of three common herbicides has declined from their peak use, but the use of acetochlor has increased substantially. (Data from [22].)

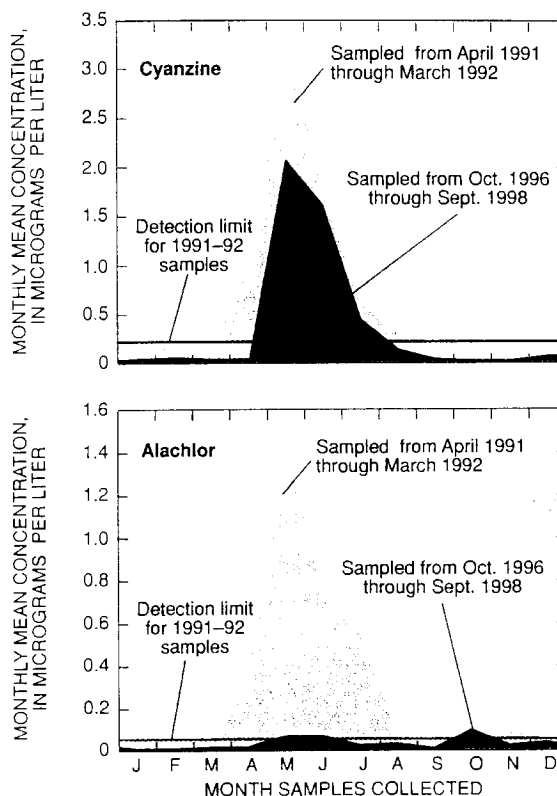


Figure 20. Monthly mean concentrations of cyanazine and alachlor declined substantially in water samples collected from the Illinois River at Valley City.

Pesticide breakdown products were detected more frequently than the parent compound

In stream and river samples (fig. 21) and ground-water samples (fig. 22), pesticide breakdown products, such as metolachlor ethanesulfonic acid (a breakdown product of metolachlor) and deethylatrazine (a breakdown product of atrazine) were detected more frequently at higher concentrations than their respective parent compounds. The toxicity of most of these breakdown products generally is unknown. Also unknown is the combined effect of parent compounds and the breakdown products. Drinking-water standards or guidelines or aquatic-life guidelines have been established for only a few pesticide breakdown products.

Most of the residue of metolachlor detected in ground water consisted of breakdown products (fig. 22). Metolachlor was frequently detected in streams and rivers. Because primarily breakdown products of metolachlor were found in ground water, metolachlor must break down before it can reach ground water.

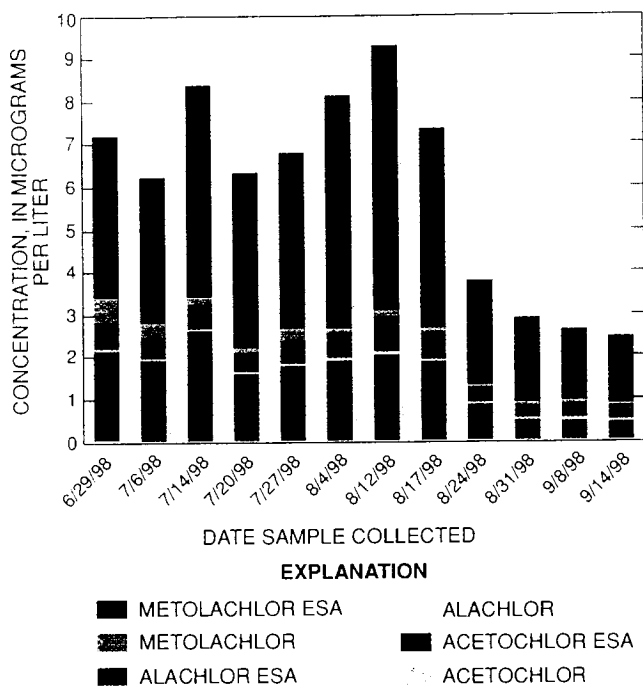


Figure 21. Breakdown products of common herbicides account for most of the analyzed herbicide residue in river water. Long after the parent compound drops below detection, breakdown products can still be detected. Earlier in the spring, acetochlor concentrations peaked at 9 micrograms per liter at this location, Sangamon River at Monticello. (ESA, ethanesulfonic acid)

Pesticides were not frequently detected in ground water

Pesticides are much less common in ground water than in streams. Six compounds, all herbicides (atrazine, metolachlor, prometon, bentazon, cyanazine, and dicamba), were detected in monitoring and water-supply wells. No insecticides were detected in ground-water samples. Only four of these pesticides were detected more than once. The maximum number of pesticides detected in a single well sample was four. Concentrations of all pesticides detected were very low (less than 0.44 µg/L). In ground-water samples, no pesticide came close to exceeding an existing drinking-water standard or guideline.

Insecticides discontinued in the 1970s were still detected in fish and sediment

Several organochlorine compounds (dieldrin, heptachlor epoxide, chlordane, and DDT) were commonly detected in fish tissue. Fish samples usually comprised 8 specimens of common carp, but at several locations

Standards for water, biota, and sediment quality

The USEPA [23–25] is responsible for setting drinking-water standards and guidelines to protect human health. Maximum Contaminant Level (MCL) is the maximum allowable concentration of a contaminant in water delivered to any user of public water systems. MCLs are enforceable standards based on an average annual concentration taken from four quarterly samples of finished drinking water. Health Advisory Level (HAL) is a nonenforceable, risk-based guideline. HALs indicate contaminant exposures below which no short- or long-term human-health effects are expected, based on drinking a specific amount of water for a specific period of time. Risk of illness increases with exposure time and concentration. Of 88 pesticides analyzed for in water, 14 have MCLs and 38 have HALs.

Guidelines to protect aquatic life have been set by several agencies including the USEPA, Environment Canada [26, 20], and the International Joint Commission (IJC) [19]. The aquatic-life guidelines developed by USEPA are based on 4-day average concentrations, are intended to protect 95 percent of the aquatic species, and should not be exceeded more than once in 3 years. The Canadian and IJC aquatic-life guidelines, which are more stringent than those of the USEPA, indicate a single maximum concentration that should never be exceeded. Aquatic-life guidelines have been established for only 18 pesticides.

Probable Effect Levels (PELs), are used to indicate contaminant concentrations in freshwater sediment associated with adverse effects on aquatic life for the Great Lakes region. The PEL is an estimate of the concentration above which adverse biological effects frequently occur [19, 27]. PELs are established for 18 of the 43 organochlorine and semivolatile organic compounds and 8 of 29 trace elements analyzed for this study.

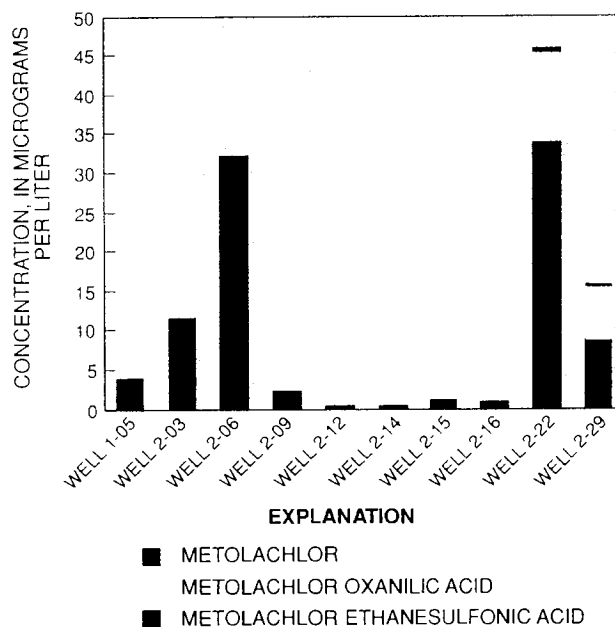


Figure 22. Breakdown products account for most of the residue of metolachlor in ground water. In these 10 shallow wells, metolachlor or its residue was detected. Metolachlor never exceeded the HAL (70 µg/L) in well samples; no drinking-water standards or guidelines have been established for the breakdown products.

white suckers were collected. These compounds are insecticides or insecticide breakdown products. Most of the organochlorine insecticides are no longer used because their registration was canceled by USEPA in the 1970s. Dieldrin, which was applied to corn and around buildings against termites, is a transformation product of aldrin, another insecticide that was applied to corn before use was discontinued during the early 1970s [28].

Dieldrin was detected in every tissue sample and one-third of the bed-sediment samples collected

in the lower Illinois River Basin (table 1). The highest dieldrin concentration in fish tissue in the NAWQA Program across the Nation was detected in fish from the Sangamon River. Every concentration detected in the basin was in the highest 10 percent of the national NAWQA results. Aquatic-life guidelines were exceeded in 30 percent of the samples.

Heptachlor epoxide is a breakdown product of the insecticide heptachlor. The highest heptachlor epoxide concentration in fish tissue in the NAWQA Program nationally was detected in the La Moine River.

Heptachlor epoxide was detected throughout the basin but at only one location in the Illinois River. The concentration of heptachlor epoxide in all fish tissue samples from the lower Illinois River Basin in which the compound was detected exceeded the concentration in more than 95 percent of all NAWQA samples. Heptachlor epoxide was not detected in any bed-sediment samples.

Chlordane was found in every tissue sample and in two bed-sediment samples. No concentrations exceeded applicable guidelines. Total chlordane, as defined for this report, is the sum of five individual components of technical chlordane [28]. Among NAWQA studies nationwide, the highest concentration of oxychlordane (a breakdown product of chlordane) in fish tissue was found in fish from the Sangamon River at Monticello.

DDT was once a widely used insecticide in agricultural and urban areas of the United States. Legal use of DDT ended in 1972. Adverse effects of DDT on fish and wildlife include growth impairment, reproductive failure, and inhibition of thyroid activity, and recent studies have shown DDT to be disruptive of reproductive cycles in animals [28].

Breakdown products of DDT were detected in every fish-tissue sample and in almost one-half the bed-sediment samples. The highest total DDT concentrations in the basin in fish tissue and sediment were in samples from Illinois River locations. Concentrations in fish tissue also were high at two Sangamon River locations (in Monticello and downstream from Springfield) compared to national NAWQA data. In bed sediment, excluding

Table 1. Summary of total chlordane, total DDT, dieldrin, heptachlor epoxide, DDD, DDE, and total PCB concentrations in fish tissue (carp or white suckers) and bed sediment. The detections are compared to guidelines for the protection of fish-eating wildlife. DDD and DDE are breakdown products of DDT

[µg/kg, micrograms per kilogram (equivalent to parts per billion); NYSDEC, New York State Department of Environmental Conservation [27]; PEL, Probable Effect Level [26]; –, not applicable]

Fish Tissue					
Compound detected in fish tissue	Number of sites with detections (20 sites sampled)	Minimum detected concentration (µg/kg)	Maximum detected concentration (µg/kg)	NYSDEC whole-fish non-carcinogenic criteria recommended for the protection of fish-eating wildlife (µg/kg)	Number of sites with concentrations in excess of NYSDEC criteria
Dieldrin	20	24	300	120	6
Heptachlor epoxide	13	13	44	200	0
Chlordane, total	20	20	310	500	0
DDT, total	20	11	550	200	1
PCBs, total	15	130	4,400	110	15

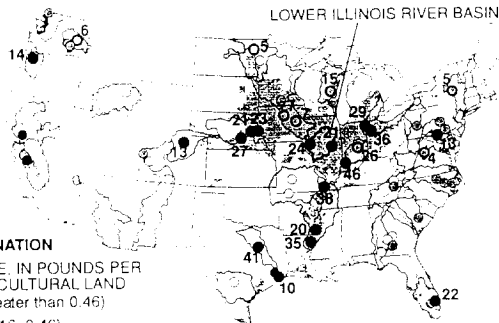
Bed Sediment					
Compound detected in stream sediments	Number of sites with detections (21 sites sampled)	Minimum detected concentration (µg/kg)	Maximum detected concentration (µg/kg)	Canadian guidelines PEL (µg/kg)	Number of sites exceeding PEL
Dieldrin	7	1.4	5.5	6.7	0
Heptachlor epoxide	0	–	–	2.7	0
Chlordane, total	2	2.6	3.4	8.9	0
DDD	5	1.1	4.1	8.5	0
DDE	5	1.0	2.6	6.8	0
DDT	0	–	–	4.8	0
PCBs, total	1	57	57	277	0



Basin and national results were similar for pesticides

HERBICIDES

AGRICULTURAL STREAMS



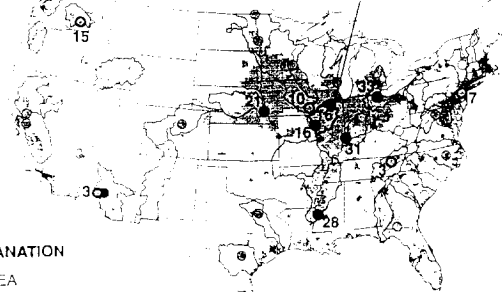
EXPLANATION

HERBICIDE USE, IN POUNDS PER ACRE OF AGRICULTURAL LAND

- Highest (greater than 0.46)
- Medium (0.16-0.46)
- Lowest (less than 0.16)
- No reported use

STREAMS WITH MIXED LAND USE AREAS

LOWER ILLINOIS RIVER BASIN



EXPLANATION

■ URBAN AREA

SUM OF HERBICIDE CONCENTRATIONS (75TH PERCENTILE, IN MICROGRAMS PER LITER)

- Highest 25 percent (greater than 1.3)
- Middle 50 percent (0.11-1.3)
- Lowest 25 percent (less than 0.11)

□ AQUATIC-LIFE GUIDELINES

□ Bold outline indicates exceedance by one or more herbicides. Number is percentage of samples that exceeded a guideline within 3-year period.

National results from river and stream sampling reflect results found for pesticides in the Lower Illinois River Basin [29]. Atrazine and metolachlor were the most frequently detected herbicides nationally and in the basin. Roundup and simazine were the next most frequent detections in the Nation, whereas cyanazine, prometon, and alachlor were the next most frequent detections in the basin. Roundup has no listed agricultural use and is mainly applied to lawns and roadsides.

The frequent detection of these herbicides in streams and rivers is due, at least in part, to the high level of application to crops and land areas. However, the major differences between the national and basin herbicide use patterns is that the frequency of detections and detection concentrations were generally higher in the basin. Basin agriculture, especially corn and soybean production, has the highest rates of chemical applications

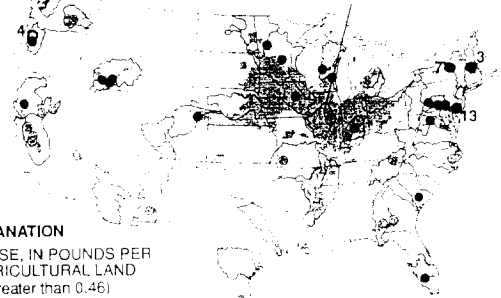
in the United States. Comparing only watersheds that are primarily agricultural, the results of the basin are similar to those from the rest of the Nation in frequency of detection.

For ground water, national and Lower Illinois River Basin pesticide sampling results also are similar. Nationally, detections of pesticides are less frequent and the concentrations are lower in wells than in streams. However, the detections of pesticides in ground water in the basin are even less frequent than in well samples for other agricultural areas in the United States. Most of the Lower Illinois River Basin is underlain by glacially derived material that is poorly drained. Subsurface drainage diverts water to streams rather than allowing it to move downward (p. 12). The fine-grained, glacially derived material is conducive to rapid breakdown of many of the pesticides. These factors help prevent pesticides from reaching wells, thereby keeping the concentrations lower than those found in ground water in other parts of the Nation.

HERBICIDES

SHALLOW GROUND WATER IN AGRICULTURAL AREAS

LOWER ILLINOIS RIVER BASIN



EXPLANATION

HERBICIDE USE, IN POUNDS PER ACRE OF AGRICULTURAL LAND

- Highest (greater than 0.46)
- Medium (0.16-0.46)
- Lowest (less than 0.16)
- No reported use

MAJOR AQUIFERS

LOWER ILLINOIS RIVER BASIN



EXPLANATION

HIGHEST DETECTION FREQUENCY

- Highest 25 percent (greater than 62)
- Middle 50 percent (19-62)
- Lowest 25 percent (less than 19)

□ DRINKING-WATER STANDARDS OR GUIDELINES

□ Bold outline indicates exceedance by one or more herbicides. Number is percentage of wells that exceeded a standard or guideline.

Illinois River locations, DDT breakdown products were detected only in a small stream in Decatur.

Other organochlorine compounds are persistent in sediments and fish

Polychlorinated biphenyls (PCBs) are a mixture of organochlorine compounds used in industrial processes. Even though production and use of PCBs was banned in the United States in 1979, PCBs commonly are found in fish and sediments because large amounts were produced, they degrade slowly, and they are transported in the atmosphere [28]. PCBs were detected in 75 percent of the fish-tissue samples in the basin, and these detections were geographically widespread. All detections in the basin exceeded the water-quality guidelines for the protection of fish-eating wildlife developed by the New York State Department of Environmental Conservation [27]. The highest PCB concentrations in fish tissue were in samples from Illinois River locations, with concentrations decreasing with increasing downstream distance from Chicago. PCBs were detected at only one location in the bed sediment.

In sufficiently high concentrations, most organochlorine compounds can be toxic to fish and other animals. These compounds bioaccumulate in animal tissue and may result in tumors, hormonal and behavioral problems, immune- and respiratory-system suppression, and abnormal development. Available evidence indicates that concentrations of these compounds have decreased over time and may be leveling off nationally [30, 28].

Common stream animals indicate poor water-quality conditions

Degradation of streams and habitat is caused by more than just chemical contamination. Biological community measures are used to assess water quality because the living components of stream ecosystems reflect the physical, chemical, and biological attributes of a stream. Measures of the algal, benthic-macroinvertebrate (fig. 23), and fish communities in streams each respond to different aspects of water quality over differing time periods and geographic areas.

Algal, benthic macroinvertebrate, and fish indices (table 2)



Figure 23. Freshwater mussels are relatively long-lived animals that are indicators of long-term water quality. These mussels were from the North Fork Vermilion River in central Illinois. Mussels bury themselves in the bottom where they filter organic matter from the water. During the twentieth century, freshwater mussel populations declined drastically in the lower Illinois River Basin. (Photograph by David J. Fazio, USGS.)

Table 2. Selected biological indicators of water quality—comparison of lower Illinois River Basin sites to national network sites¹

Location sampled	National Network Status		
	Fish status	Invertebrate status	Algal status
Panther Creek near El Paso	■		
Indian Creek near Wyoming	■		■
Sangamon River at Monticello	■		
La Moine River at Colmar	■		■
Mackinaw River near Green Valley			
Sangamon River near Oakford	■	■	■
Illinois River at Valley City	■		■

¹ Represents 140 sites in the NAWQA national network that have algal, invertebrate, and fish data.

were used to assess water quality as part of the NAWQA Program [32]. The higher the index, the more degraded the stream conditions. (See Appendix.) The algal status index reflects the relative abundance of benthic algal species that are considered tolerant of poor water-quality conditions, including high nutrient concentrations. In some regions, this index correlates with transport and deposition of silt.

Eleven community characteristics were used in the invertebrate community status index to assess water quality. The fish status indicator is the sum of scores of four fish-population characteristics (percent tolerant species, percent omnivorous fish, percent non-native fish, and individuals with external anomalies) that increase in concert with water-quality degradation (fig. 24).

In relation to the rest of the Nation, the invertebrate status index in the lower Illinois River Basin was intermediate at all sites except one (table 2). Macroinvertebrate communities of streams in

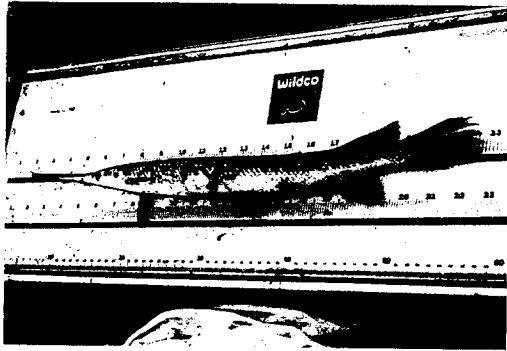


Figure 24. Longnose gar from the Sangamon River in central Illinois. In the basin, the Longnose gar is the most common gar found in small rivers. Many fish were collected, measured, and released to determine the health of the fish communities in the streams and rivers sampled. (Photograph by David J. Fazio, USGS.)



Figure 25. Basin streams with fast current had the most diverse macroinvertebrate communities. (Photograph by David J. Fazio, USGS.)

the basin are dominated by many species that are widely distributed, most of which are tolerant of the effects of excess nutrients. Excessive quantities of nutrients in runoff to streams can be harmful to intolerant species by stimulating algal growth that may

decrease dissolved-oxygen concentrations at night. The highest proportions of sensitive macroinvertebrate species were in streams with comparatively swift currents because increased water velocity can increase rates of stream aeration and thus maintain higher dissolved-oxygen concentrations (fig. 25).

Algae index values were either high or intermediate in relation to the entire Nation. The algal status index indicates that general algal conditions (as an indicator of water quality) in the Corn Belt are poor relative to the rest of the country. (See pages 9–10.)

One of the lowest fish-index scores in the country (indicating a healthy fish community) was at Panther Creek near El Paso. This small, shaded stream held a variety of habitats and a diverse fish population. All the biological index scores were high (degraded community) for the Sangamon River near Oakford. Little habitat diversity was found at this site, and concentrations of nutrients were high.

Highest concentrations of polycyclic aromatic hydrocarbons were in bed sediment near urban areas

Polycyclic aromatic hydrocarbons (PAHs) were detected in concentrations at or above a common reporting level of 50 µg/kg in as much as 50 percent of the bed-sediment samples (fig. 26). PAHs are by-products of combustion; sources include fires, manufacturing, power generation, and vehicle emissions. With sufficient exposure, many PAHs are carcinogenic, causing tumors in fish and other animals, and many PAHs are toxic to some organisms [33]. In the basin, the highest PAH concentrations were in the bed sediment of the Illinois River at locations upstream from Peoria. The highest PAH concentrations at other sites in the basin were near urban areas: specifically, in

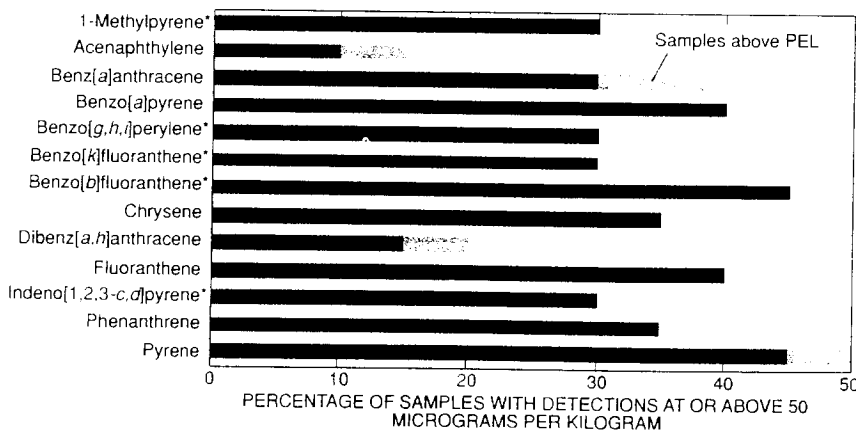
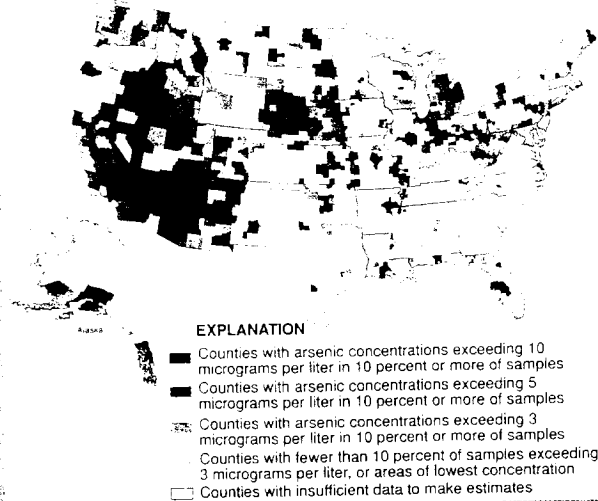


Figure 26. Four polycyclic aromatic hydrocarbons (PAHs) were detected at concentrations that may be detrimental to wildlife that live in bed sediments. Many PAHs are prevalent in sediments of the basin. Shown are percent detections at or above 50 micrograms per kilogram in bed-sediment samples and percent detections that exceed Canadian guidelines for protection of aquatic life [26]. [*], no guideline available; PEL, Probable Effect Level]



Arsenic in ground water is a national concern

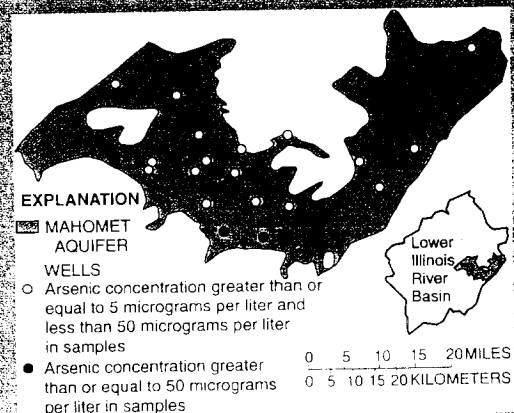


Arsenic is a naturally occurring trace element in natural waters. With sufficient exposure, arsenic can cause skin, lung, bladder, liver, and kidney cancer in humans [34, 21]. It also has been reported to impair development and hearing; to adversely affect the vascular, gastrointestinal, and nervous systems; and to be associated with diabetes. The current (2000) arsenic MCL applies to the sum of all arsenic forms in finished drinking water. The USEPA has proposed a new MCL of 5 µg/L.

Areas in the Western United States and upper Midwest have relatively high arsenic concentrations in ground water. In Illinois, the dark areas on the map generally indicate the counties

overlying the Mahomet aquifer, where water from about 7 percent of wells sampled (2 of 30) exceeded the current MCL (50 µg/L) and 60 percent would exceed the lower MCL. Only few concentrations of arsenic above 5 µg/L and none above 50 µg/L in shallow wells (less than 100 feet deep) were found at wells in the western and southern shallow aquifers and other areas in the lower Illinois River Basin.

Nationally, water from 1 percent of wells sampled exceeded the current MCL [35], and only 16 percent would exceed the proposed new standard in Illinois, if they do not treat the water for arsenic. In Illinois, people who draw water from the Mahomet aquifer have greater risk of drinking arsenic in water than ground-water users in the rest of the United States.



a small stream in the city of Decatur, in the Vermilion River near its confluence with the Illinois River (downstream from Streator and Pontiac), and in the Sangamon River downstream from Springfield. Four detections of PAH compounds exceeded the Canadian interim freshwater sediment-quality guidelines for the protection of aquatic life [26].

Arsenic exceeded the drinking-water standard in the Mahomet aquifer

The Mahomet aquifer is overlain by more than 100 feet of clay-rich glacial material. Contamination of this aquifer by human activities is minimal [31]. Although arsenic resulting from human activities can contaminate water, high concentrations of arsenic in the Mahomet aquifer are from natural sources. Arsenic concentrations in parts of the Mahomet aquifer

exceeded the USEPA current MCL of 50 µg/L. In 1999, the National Academy of Science [34] concluded that this current standard does not sufficiently protect public health. The USEPA has proposed lowering the MCL to 5 µg/L. If the arsenic MCL were lowered to 5 µg/L, then water from more than 60 percent of the wells sampled (18 of 30) would exceed the new level.

Within the lower Illinois River Basin, the percentage of private-well samples with concentrations of arsenic above 50 µg/L was approximately the same as that of samples from public-supply wells. Statewide data from the Illinois Environmental Protection Agency [31], however, indicate the percentage of concentrations above 5 µg/L was different between the public wells and private wells statewide.

ILLINOIS RIVER: A history of degradation and recovery

In Illinois, 7 percent of the fish species have disappeared since about 1900, and many more species are in decline [36]. Smith [37] listed eight conditions responsible for the loss of many fish populations in Illinois streams. Some of these conditions can be found in the basin: siltation; drainage of natural lakes, swamps, and prairie marshes; desiccation during drought; species interactions, including competition from non-native fish; industrial, domestic, and agricultural pollution other than siltation; dams and reservoirs; and increased water temperature.

One of the most notable effects of human activity on stream life occurred between 1900 and 1920 in the Illinois River downstream from Chicago, after the Chicago Sanitary and Shipping Canal opened. The biological repercussions of industrial and domestic wastes being diverted to the Illinois River peaked in about 1920, when plants, benthic organisms, and fish practically were eliminated from the Illinois River downstream from Chicago to a point downstream from Peoria [38]. Since the 1920s, water quality in this reach of the Illinois River has gradually improved because of the improved wastewater-treatment practices upstream. Monitoring by the Illinois Natural History Survey (INHS) has documented partial recovery of fish communities. Between 1957 and 1992,

common carp and goldfish dominated fish communities in the main stem of the upper Illinois River. **By 1993, however, native species had returned to the main stem, and the relative abundance of common carp and goldfish had declined significantly** [39]. In contrast, fish life in the bottomland lakes along the lower Illinois River has not increased to the degree anticipated from improvement in dissolved-oxygen conditions [40, 41].

Compared to urban areas upstream from the basin, streams in predominantly agricultural areas generally have higher biotic integrity and more diverse fish communities that commonly include various intolerant fish species [42, 43, and 37].



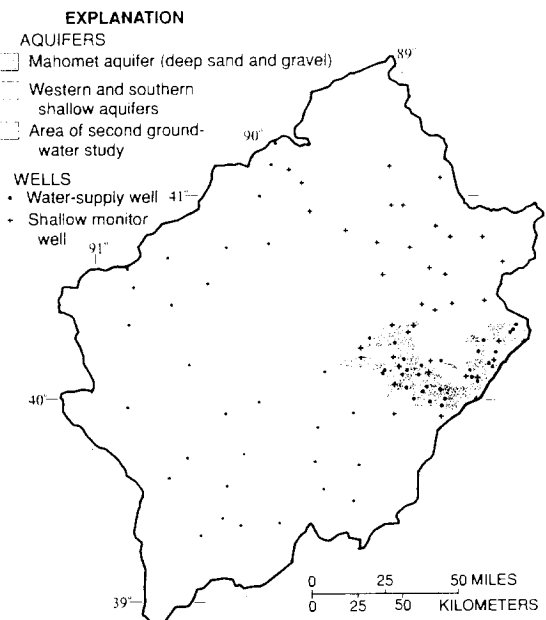
The Illinois River historically was an important wildlife resource. It is gradually recovering that status.

Radon exceeded the proposed drinking-water standard in almost one-half of shallow wells

The bedrock underlying the glacial materials in the lower Illinois River Basin and especially the glacial material itself contain a high concentration of the source elements that decay into radon. As a result, the potential for high concentrations of radon in ground water and soil gas in the basin is greater than the national average (W.S. Morrow, Jr., U.S. Geological Survey, written commun., 2000).

Radon was detected in all 30 water samples from the Mahomet aquifer (fig. 27). Concentrations of radon ranged from 110 to 730 picocuries per liter (pCi/L), and the mean concentration was 190 pCi/L.

Figure 27. Ground-water studies encompassed most of the area of the lower Illinois River Basin. Two studies of water-supply wells and two studies of shallow, recently recharged ground water were completed. The first set of shallow monitor wells was drilled in the material overlying the Mahomet aquifer.



The samples from the Mahomet aquifer, however, had the lowest average radon concentrations of the four groups of study wells. (See pages 25–26.) Concentrations in 7 percent of Mahomet aquifer samples (2 of 30) exceeded the proposed USEPA radon MCL of 300 pCi/L. No sample concentrations exceeded the USEPA-proposed Alternative Maximum Contaminant Level (AMCL) for radon of 4,000 pCi/L. The MCL is proposed for drinking water where there is no other indoor-air radon-mitigation strategy. The AMCL is proposed for drinking water where a multimedia indoor-air mitigation strategy is used to reduce radon exposure.

Radon concentration was higher in samples from the 57 shallow monitoring wells and the 30 water-supply wells in the western and southern shallow aquifers than in the Mahomet aquifer. In the samples from the western and southern shallow aquifers, 55 percent (16 of 29) exceeded the proposed MCL, but none exceeded the proposed higher AMCL. Radon was highest in the samples from shallow monitoring wells in the glacial materials overlying the Mahomet aquifer (fig. 27). In the shallow monitoring wells, 38 percent (22 of 57) of the samples exceeded the radon MCL, although no sample exceeded the AMCL.

Volatile organic compounds were detected in ground water at low concentrations

Samples were collected from 60 private and public water-supply wells in the basin to determine the extent

of contamination by volatile organic compounds (VOCs); 30 of the wells were in the Mahomet aquifer and the other 30 were in the western and southern shallow aquifers. Detections were similar in samples from wells in the western and southern shallow aquifers (83 percent) and in the Mahomet aquifer (80 percent). One sample from a well in the area of western and southern shallow aquifers contained 0.57 µg/L of methyl *tert*-butyl ether (MTBE)—well below the HAL for MTBE of 20 µg/L. The concentration of VOCs in almost all of the samples was at or near the method detection limit and was usually 10 to 100 times lower than applicable standards or guidelines for drinking water. The source of these VOCs—especially those in the Mahomet aquifer, which is largely protected from atmospheric or land-use contamination—is uncertain.

Fish and sediment in the Illinois River have high concentrations of trace elements

The Canadian interim freshwater sediment-quality guidelines for the protection of aquatic life [26] were exceeded at four locations for various trace elements. Three of these locations were on the Illinois River close to Chicago. In the Illinois River, bed-sediment concentrations of seven metals (chromium, copper, lead, mercury, nickel, selenium, and zinc) were highest near Ottawa and decreased downstream [44] (fig. 28).

In many samples of fish tissue collected in the lower Illinois River Basin, cadmium concentrations were high compared to those in all samples analyzed in the rest of the Nation. (See Appendix.) Concentrations of trace elements were examined in carp liver collected at 17 locations in the basin. The highest cadmium concentration in carp liver from across the Nation (52 milligrams per kilogram dry weight) was detected in the Vermilion River near its confluence with the Illinois River. Concentrations of cadmium in 10 carp-liver samples from the basin were in the upper one-third of the concentrations in all carp-liver samples collected in the Nation. Of those 10 samples, 6 were collected from the Illinois River main stem. Fitzpatrick and others [45] found cadmium at higher than background concentrations in sediment or biota at urban and agricultural locations in the upper Illinois River Basin. Long-term chronic exposure to cadmium can cause kidney and bone damage in humans. Mining and activities related to mining are the main sources of cadmium in the environment from human activities, but other sources include waste incineration, sewage sludge, fertilizers, batteries, tires, and many industrial processes [23].

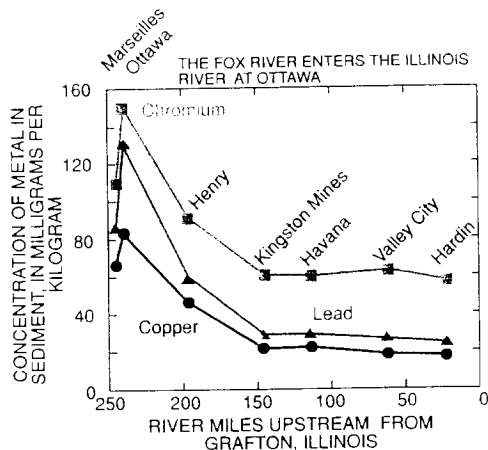
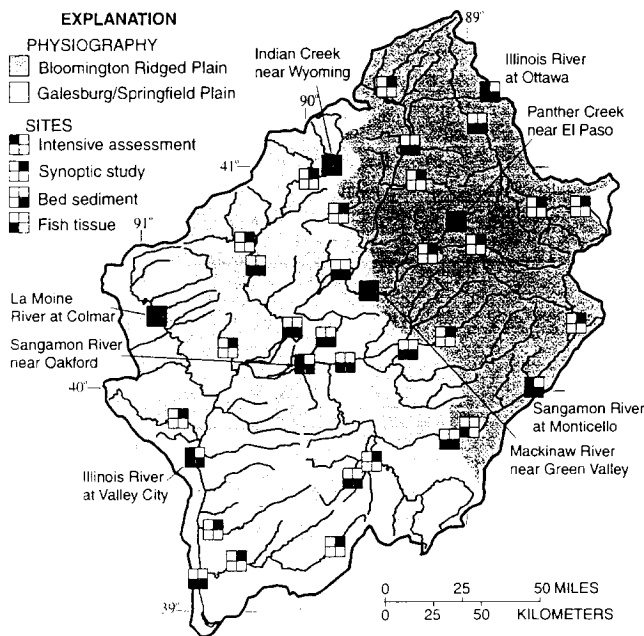


Figure 28. Concentrations of three metals in bed sediments decreased in the Illinois River from Ottawa, Ill., to Kingston Mines, Ill. The concentrations of metals were highest near the confluence of the Fox River and the Illinois River at Ottawa.

STUDY UNIT DESIGN

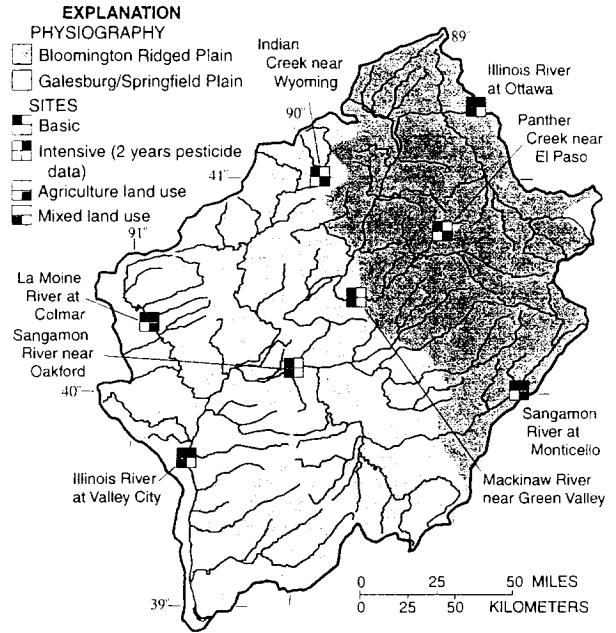
Stream chemistry

The main objective of the stream-chemistry component of the study was to assess the relation between agricultural land use and basin size and chemical constituents of surface water. Stream and river sampling locations were distributed among subbasins and on the Illinois River. Sampling locations included Basic and Intensive Fixed Sites—sampled at regular intervals for a period of 2 to 3 years—and Synoptic Sites that were sampled during a 2-week period in August 1997.



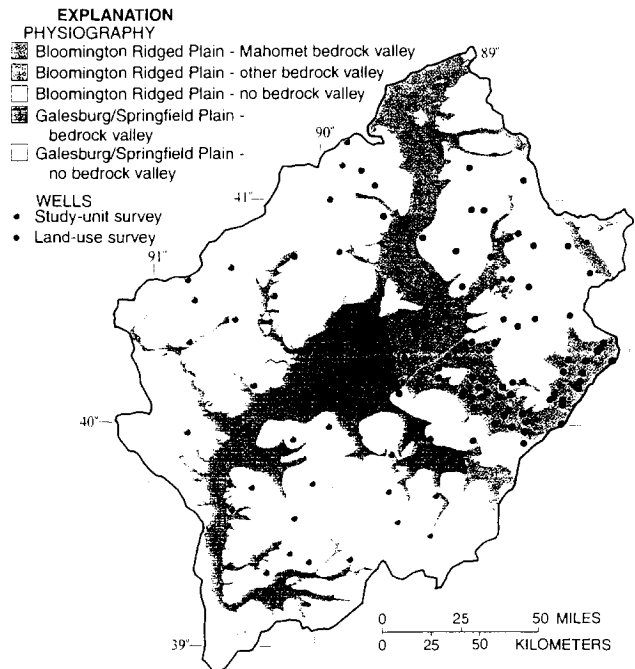
Ground-water chemistry

One objective of the ground-water chemistry component of the study was to determine whether chemical constituents of ground water were related to agricultural land uses (land-use surveys). Another objective in two surveys of domestic (household) and public-supply (municipal) wells (Study-Unit surveys) was to assess the overall water quality in selected drinking-water-source aquifers (Mahomet aquifer and the western and southern shallow aquifers) and to determine whether they are affected by land-use practices. The full extent of the Mahomet aquifer (in the Mahomet bedrock valley) is not shown. Results for the Mahomet aquifer are representative only for that part shown.



Stream ecology

The main objective of the ecology component of the study was to assess surface-water quality by integrating the physical, chemical, and biological factors. Ecology Sites were distributed at the basic fixed sites and at smaller subbasins that are all predominantly agricultural land. Sites were classified as either intensive or synoptic on the basis of the level and schedule of sampling effort.



Summary of data collection in the lower Illinois River Basin, 1995-98

Study component	What data were collected and why	Types of sites sampled	Number of sites	Sampling frequency and period
Stream Chemistry				
Basic Sites—general water quality	Concentrations, seasonal variation, and annual loads. Data included streamflow, nutrients, major ions, organic carbon, suspended sediment, water temperature, specific conductance, alkalinity, pH, and dissolved oxygen.	Basic Fixed Sites: representative of common land uses, as well as major tributary and basin outflow sites.	8	Monthly plus storms; 2 years
Intensive Sites—pesticides	Concentrations and seasonal variations in pesticides and breakdown products. Data included 75 pesticides and 13 breakdown products of the most heavily applied herbicides.	Subset of Basic Sites: Two watersheds with greater than 90 percent row-crop agriculture and the basin inflow and outflow sites.	4	Weekly to monthly Jan. 1996–June 1998
Synoptic Sites—water chemistry and pesticides	Spatial distribution of nutrients, pesticides, and pesticide breakdown products. Data included nutrients and 24 pesticides and pesticide breakdown products.	Basic Sites Additional sites	4 17	Once in August 1997 during minimum flow
Contaminants in bed sediments	Occurrence and distribution of contaminants in bed sediment. Data included total PCBs, 32 organochlorine pesticides, 63 semivolatile organic compounds, and 44 trace elements.	Depositional zones of Basic Fixed Sites and some additional sites.	21	Once; June–October in 1996–97
Contaminants in fish	Occurrence and distribution of contaminants in biota. Data included total PCBs, 30 organochlorine pesticides in whole fish, and trace elements in fish livers.	Most stream sites also sampled for bed sediments.	20	Once; June–October in 1996–97
Stream Ecology				
Intensive assessments	Relations among biological communities and water chemistry, physical habitat, and land use. Includes spatial relations, along with spatial variation. Data included algae, invertebrates, fish communities, and physical habitat.	Basic Fixed Sites	8	1 reach, 1 year (1996) 1 reach, 3 years (1996–98) 3 reaches, 1 year (1996 or 1997)
Synoptic studies	Better spatial assessment. Data collected were same as above.	Synoptic Sites	21	Once in August 1997
Ground-Water Chemistry				
Aquifer survey—domestic and public supply wells	Occurrence and distribution of chemicals in domestic and public supply wells. Data included major ions, nutrients, 75 pesticides, 13 pesticide breakdown products, 73 volatile organic compounds, dissolved organic carbon, trace elements, and radon.	Domestic and public supply wells across the subunit aquifers	56	Once in November in 1997
Resample selected aquifer survey wells	Confirmation of low-level or questionable detections of pesticides, breakdown products, and age-dating chemicals.	A subset of the wells above	10	Once in 1998
Resample selected aquifer survey wells	Confirmation of high-level detections of arsenic and to determine the chemical nature of the dissolved arsenic. Data included major ions and selected dissolved metals in addition to two chemical forms of arsenic.	A subset of wells from the deep aquifer survey with arsenic that exceeded the drinking water standard.	10	Once in 1997
Land-use effects—corn and soybean row crops	Describe the effects of agricultural land use on shallow ground water in the area underlain by most recent glacial geologic material. Data included major ions, nutrients, 75 pesticides, 13 pesticide breakdown products, dissolved organic carbon, trace elements, and radon.	Very shallow monitoring wells installed for this study	56	Once in 1997

GLOSSARY

- Algae**—Chlorophyll-bearing nonvascular, primarily aquatic species that have no true roots, stems, or leaves; most algae are microscopic, but some species can be as large as vascular plants.
- Aquifer**—A water-bearing layer of soil, sand, gravel, or rock that will yield usable quantities of water to a well.
- Bed sediment**—The material that temporarily is stationary in the bottom of a stream or other watercourse.
- Benthic**—Refers to plants or animals that live on the bottom of lakes, streams, or oceans.
- Breakdown product**—A compound derived by chemical, biological, or physical action upon a pesticide. The breakdown is a natural process that may result in a more toxic or a less toxic compound and a more persistent or less persistent compound.
- Concentration**—The amount or mass of a substance present in a given volume or mass of sample. Usually expressed as microgram per liter (water sample) or micrograms per kilogram (sediment or tissue sample).
- Cubic foot per second (ft³/s, or cfs)**—Rate of water discharge representing a volume of 1 cubic foot passing a given point during 1 second, equivalent to approximately 7.48 gallons per second or 448.8 gallons per minute or 0.02832 cubic meter per second.
- Ground water**—In general, any water that exists beneath the land surface, but more commonly applied to water in fully saturated soils and geologic formations.
- Habitat**—The part of the physical environment where plants and animals live.
- Herbicide**—A chemical or other agent applied for the purpose of killing undesirable plants. See also Pesticide.
- Insecticide**—A substance or mixture of substances intended to destroy or repel insects. See also Pesticide.
- Intolerant species**—Those species that are not adaptable to human alterations to the environment and thus decline in numbers where human alterations occur. See also Tolerant species.
- Invertebrate**—An animal having no backbone or spinal column.
- Load**—General term that refers to a material or constituent in solution, in suspension, or in transport; usually expressed in terms of mass or volume.
- Mean**—The average of a set of observations, unless otherwise specified.
- Median**—The middle or central value in a distribution of data ranked in order of magnitude. The median is also known as the 50th percentile.
- Method detection limit**—The minimum concentration of a substance that can be accurately identified and measured with present laboratory technologies.
- Micrograms per liter (µg/L)**—A unit expressing the concentration of constituents in solution as weight (micrograms) of solute per unit volume (liter) of water; equivalent to one part per billion in most stream water and ground water. One thousand micrograms per liter equals 1 milligram per liter.
- Milligrams per liter (mg/L)**—A unit expressing the concentration of chemical constituents in solution as weight (milligrams) of solute per unit volume (liter) of water; equivalent to one part per million in most stream water and ground water. One thousand micrograms per liter equals 1 milligram per liter.
- Nitrate**—An ion consisting of nitrogen and oxygen (NO₃⁻). Nitrate is a plant nutrient and is very mobile in soils.
- Organochlorine compound**—Synthetic organic compounds containing chlorine. As generally used, this term refers to compounds containing mostly or exclusively carbon, hydrogen, and chlorine. Examples include organochlorine insecticides, polychlorinated biphenyls, and some solvents containing chlorine.
- Periphyton**—Organisms that grow on underwater surfaces; periphyton include algae, bacteria, fungi, protozoa, and other organisms.
- Pesticide**—A chemical applied to crops, rights-of-way, lawns, or residences to control weeds, insects, fungi, nematodes, rodents or other "pests."
- Phosphorus**—A nutrient essential for growth that can play a key role in stimulating aquatic growth in lakes and streams.
- Phytoplankton**—See Plankton.
- Picocurie (pCi)**—One trillionth (10⁻¹²) of the amount of radioactivity represented by a curie (Ci). A curie is the amount of radioactivity that yields 3.7 × 10¹⁰ radioactive disintegrations per second (dps). A picocurie yields 2.22 disintegrations per minute (dpm) or 0.037 dps.
- Plankton**—Floating or weakly swimming organisms at the mercy of the waves and currents. Animals of the group are called zooplankton, and the plants are called phytoplankton.
- Point source**—A source at a discrete location such as a discharge pipe, drainage ditch, tunnel, well, concentrated livestock operation, or floating craft.
- Relative abundance**—The number of organisms of a particular kind present in a sample relative to the total number of organisms in the sample.
- Riparian**—Areas adjacent to rivers and streams often with a high density, diversity, and productivity of plant and animal species relative to nearby uplands.
- Semivolatile organic compound (SVOC)**—Operationally defined as a group of synthetic organic compounds that are solvent-extractable and can be determined by gas chromatography/mass spectrometry. SVOCs include phenols, phthalates, and polycyclic aromatic hydrocarbons (PAHs).
- Sorption**—General term for the interaction (binding or association) of a solute ion or molecule with a solid.
- Subsurface drain**—A shallow drain installed in an irrigated field to intercept the rising ground-water level and maintain the water table at an acceptable depth below the land surface.
- Tolerant species**—Those species that are adaptable to (tolerant of) human alterations to the environment and often increase in number when human alterations occur.
- Trace element**—An element found in only minor amounts (concentrations less than 1.0 milligram per liter) in water or sediment; includes arsenic, cadmium, chromium, copper, lead, mercury, nickel, and zinc.
- Volatile organic compounds (VOCs)**—Organic chemicals that have a high vapor pressure relative to their water solubility. VOCs include components of gasoline, fuel oils, and lubricants, as well as organic solvents, fumigants, some inert ingredients in pesticides, and some by-products of chlorine disinfection.
- Yield**—The mass of material or constituent transported by a river in a specified period of time divided by the drainage area of the river basin.

REFERENCES

1. Willman, H.B., Atherton, Elwood, Buschbach, T.C., Collinos, Charles, Frye, J.C., Hopkins, M.E., Lineback, J.A., and Simon, J.A., 1975. Handbook of Illinois Stratigraphy: Illinois State Geological Survey Bulletin 95, 261 p.
2. Warner, K.L., 1998. Water-quality assessment of the lower Illinois River Basin—Environmental setting: U.S. Geological Survey Water-Resources Investigations Report 97-4165, 50 p.
3. McKenna, D.P., and Keefer, D.A., 1991. Potential for agricultural chemical contamination of aquifers in Illinois: Illinois State Geological Survey, Open-File Series 1991-7R, 16 p.
4. Sullivan, D.J., 2000. Nutrients and suspended solids in surface waters of the Upper Illinois River Basin, Illinois, Indiana, and Wisconsin, 1978-97: U.S. Geological Survey Water-Resources Investigations Report 99-4275, 57 p.
5. Sorenson, S.K., Porter, S.D., Akers, K.K.B., Harris, M.A., Roberts, L.R., and Terrio, P.J., 1999. Water quality and habitat conditions in upper Midwest streams relative to riparian vegetation and soil characteristics, August 1997—Study design, methods and data: U.S. Geological Survey Open-File Report 99-202, 53 p.
6. Scribner, E.A., Goolsby, D.A., Thurman, E.M., Meyer, M.T., and Battaglin, W.A., 1996. Concentrations of selected herbicides, herbicide metabolites, and nutrients in outflow from selected Midwestern reservoirs, April 1992 through September 1993: U.S. Geological Survey Open-File Report 96-393, 128 p.
7. Porter, S.D., 2000. Upper Midwest River Systems—Algal and nutrient conditions in streams and rivers in the upper Midwest region during seasonal low-flow conditions, *in* Nutrient criteria technical guidance manual, rivers and streams: Washington, D.C., U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology, EPA-822-B-00-002, p. A-25—A-42.
8. Stauffer, J.C., Goldstein, R.M., and Newman, R.M., 2000. Relationship of wooded riparian zones and runoff potential to fish community composition in agricultural streams: Canadian Journal of Fisheries and Aquatic Sciences, v. 57, p. 307-316.
9. Porter, S.D., Harris, M.A., and Kalkhoff, S.J., 2000. The influence of natural factors on the quality of Midwestern streams and rivers: U.S. Geological Survey Water-Resources Investigations Report 00-4288, 8 p.
10. Osborne, L.L. and Kovacic, D.A., 1993. Riparian vegetated buffer strips in water-quality restoration and stream management: Freshwater Biology, v. 29, p. 243-258.
11. Goolsby, D.A., 2000. Mississippi Basin nitrogen flux believed to cause Gulf hypoxia: EOS, Transactions, American Geophysical Union, v. 81, no. 29, p 321-327.
12. Goolsby, D.A., Battaglin, W.A., Lawrence, G.B., Artz, R.S., Aulenbach, B.T., Hooper, R.P., Keeney, D.R., and Stensland, G.J., 1999. Flux and sources of nutrients in the Mississippi-Atchafalaya River Basin—Topic 3 Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico: Silver Spring, Md., NOAA Coastal Ocean Program Decision Analysis Series no. 17, 130 p.
13. Alexander, R.B., Smith, R.A., and Schwarz, G.E., 2000. Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico: Nature, v. 403, p. 758-761.
14. David, M.B., and Gentry, L.E., 2000. Anthropogenic inputs of nitrogen and phosphorus and riverine export for Illinois. USA: Journal of Environmental Quality, v. 29, p. 494-508.
15. Zucker, L.A. and Brown, L.C., 1998. Agricultural drainage—Water quality impacts and subsurface drainage studies in the Midwest: Ohio State University Extension Bulletin 871, 40 p.
16. Illinois Agricultural Statistics Service, 1997. Annual summary: Bulletin 97-1, 127 p.
17. Illinois Agricultural Statistics Service, 1998. Annual summary: Bulletin 98-1, 125 p.
18. Illinois Agricultural Statistics Service, 1999. Annual summary: Bulletin 99-1, 117 p.
19. International Joint Commission, 1999. Revised Great Lakes Water Quality Agreement of 1978, as amended by Protocol signed November 18, 1987: International Joint Commission United States and Canada, September, 1989 (<http://www.ijc.org/agree/quality.html#art5>). See also Annex I, Specific objectives (<http://www.ijc.org/agree/quality.html#ann1>) accessed November 2000.
20. Canadian Council of Resource and Environment Ministers, 1997. Canadian water quality guidelines (rev. ed.): Ottawa, Ont., Task Force on Water Quality Guidelines, Canadian Council of Resource and Environment Ministers, looseleaf (originally published 1987), 23 appendixes.
21. Acetochlor Registration Partnership, at URL <http://www.arpinfo.com/> (Accessed July 12, 2000).
22. University of Illinois, College of Agriculture, Consumer, and Environmental Sciences, Pesticide Impact Assessment Program: Web site accessed July 15, 2000, at URL <http://ext.agn.uiuc.edu/piap/>
23. U.S. Environmental Protection Agency, 2000. Volunteer estuary monitoring, a methods manual: Web site <http://www.epa.gov/owow/wtr1/estuaries/monitor/index.html>, accessed August 29, 2000 [site revised August 21, 2000].
24. U.S. Environmental Protection Agency, 2000. National primary drinking water regulations—Arsenic and clarifications to compliance and new source contaminants monitoring: Federal Register, v. 65, no. 121, 28

- p. 38887–38983.
25. U.S. Environmental Protection Agency. 1999. Drinking water and health advisories: U.S. Environmental Protection Agency. Office of Water: accessed March 27, 2000, at URL <http://www.epa.gov/ost/drinking/standards> [site revised October 17, 2000]
 26. Canadian Council of Ministers of the Environment. 1999. Canadian sediment quality guidelines for the protection of aquatic life: Introduction and summary tables, *in* Canadian environmental quality guidelines: Canadian Council of Ministers of the Environment. Winnipeg, Canada. accessed December 5, 2000 at URL <http://www.ec.gc.ca/ceqg-rcqe>
 27. Newell, A.J., Johnson, D.W., and Allen, L.K.. 1987. Niagara River biota contamination project—Fish flesh criteria for piscivorous wildlife: New York State Department of Environmental Conservation, Division of Fish and Wildlife, Bureau of Environmental Protection Technical Report 87–3, 182 p.
 28. Wong, C.S., Capel, P.D., and Nowell, L.H.. 2000. Organochlorine pesticides and PCBs in stream sediment and aquatic biota—Initial results from the National Water-Quality Assessment Program, 1992–1995: U.S. Geological Survey Water-Resources Investigations Report 00–4053, 88 p.
 29. Fuhrer, G.J., Gilliom, R.J., Hamilton, P.A., Morace, J.L., Nowell, L.H., Rinella, J.F., Stoner, J.D., and Wentz, D.A., 1999. The quality of our Nation's waters—Nutrients and pesticides: U.S. Geological Survey Circular 1225, 82 p.
 30. Nowell, L.H., Capel, P.D., and Dileanis, P.D., 1999. Pesticides in stream sediment and aquatic biota—Distribution, trends, and governing factors: Boca Raton, Fla., CRC Press, Pesticides in the Hydrologic Systems series, v. 4, 1040 p.
 31. Warner, K.L., 2000. Analysis of nutrients, selected inorganic constituents, and trace elements in water from Illinois community-supply wells, 1984–91: U.S. Geological Survey Water-Resources Investigations Report 99–4152, 47 p.
 32. Gilliom, J.G., Alley, W.M., and Gurtz, M.E., 1995. Design of the National Water-Quality Assessment Program—Occurrence and distribution of water-quality conditions: U.S. Geological Survey Circular 1112, 33 p.
 33. Eisler, Ronald, 1987. Polycyclic aromatic hydrocarbon hazards to fish, wildlife, and invertebrates—A synoptic review: Laurel, Md., Patuxent Wildlife Research Center, U.S. Fish and Wildlife Service, Biological Report 85 (1.11), 81 p.
 34. National Research Council, 1999. Arsenic in drinking water: Washington D.C., National Academy Press, 310 p.
 35. Focazio, M.J., Welch, A.H., Watkins, S.A., Helsel, D.R., and Horn, M.A., 1999. A retrospective analysis on the occurrence of arsenic in ground-water resources of the United States and limitations in drinking-water-supply characterizations: U.S. Geological Survey Water-Resources Investigations Report 99–4279, 21 p.
 36. Illinois Department of Energy and Natural Resources and The Nature of Illinois Foundation. 1994. The changing Illinois environment: Critical trends: summary of the critical trends assessment project: Springfield, Ill., Illinois Department of Energy and Natural Resources, 89 p.
 37. Smith, P.W., 1971. Illinois streams—A classification based on their fishes and an analysis of factors responsible for the disappearance of native species: Illinois Natural History Survey Biological Note, v. 76, p. 1–14.
 38. Richardson, R.E., 1928. The bottom fauna of the middle Illinois River, 1913–1925—Its distribution, abundance, valuation, and index value in the study of stream pollution: Illinois State Natural History Survey, v. 7, no. 2, p.189–192.
 39. Sparks, R.E., and Lerczak, T.V., 1993. Recent trends in the Illinois River indicated by fish populations: Havana, Ill., Illinois Natural History Survey River Research Laboratory of the Forbes Biological Station, 34 p.
 40. Mills, H.B., Starrett, W.C., and Bellrose, F.C., 1966. Man's effect on the fish and wildlife of the Illinois River: Illinois Natural History Survey Biological Notes no. 57, 24 p.
 41. Sparks, R.E., and Starrett, W.C., 1975. An electrofishing survey of the Illinois River, 1959–1974: Illinois Natural History Survey Bulletin no. 31, p. 317–380.
 42. Bertrand, W.A., 1984. Des Plaines River Basin fisheries assessment: Illinois Department of Conservation, Division of Fish and Wildlife Resources, 43 p.
 43. Ruhl, P.M., 1995. Surface-water-quality assessment of the upper Illinois River Basin in Illinois, Indiana, and Wisconsin—Analysis of relations between fish-community structure and environmental conditions in the Fox, Des Plaines, and Du Page River Basins in Illinois: U.S. Geological Survey Water-Resources Investigations Report 94–4094, 50 p.
 44. Colman, J.A., and Sanzalone, R.F., 1992. Geochemical characterization of streambed sediment in the Upper Illinois River Basin: Water Resources Bulletin, v. 28, no. 5, p. 933–950.
 45. Fitzpatrick, F.A., Scudder, B.C., Crawford, J.K., Schmidt, A.R., Sieverling, J.B., and others, 1995. Surface water-quality assessment of the upper Illinois River Basin in Illinois, Indiana, and Wisconsin—Major and trace elements in water, sediment, and biota, 1978–90: U.S. Geological Survey Water-Resources Investigations Report 95–4045, 254 p.

APPENDIX—WATER-QUALITY DATA FROM THE LOWER ILLINOIS RIVER BASIN IN A NATIONAL CONTEXT

For a complete view of lower Illinois River Basin data and for additional information about specific benchmarks used, visit our Web site at <http://water.usgs.gov/nawqa/>. Also visit the NAWQA Data Warehouse for access to NAWQA data sets at <http://infotrek.er.usgs.gov/wdbctx/nawqa/nawqa.home>.

This appendix is a summary of chemical concentrations and biological indicators assessed in the lower Illinois River Basin. Selected results for this basin are graphically compared to results from as many as 36 NAWQA Study Units investigated from 1991 to 1998 and to national water-quality benchmarks for human health, aquatic life, or fish-eating wildlife. The chemical and biological indicators shown were selected on the basis of frequent detection, detection at concentrations above a national benchmark, or regulatory or scientific importance. The graphs illustrate how conditions associated with each land use sampled in the lower Illinois River Basin compare to results from across the Nation, and how conditions compare among the several land uses. Graphs for chemicals show only detected concentrations and, thus, care must be taken to evaluate detection frequencies in addition to concentrations when comparing study-unit and national results. For example, acetochlor concentrations in lower Illinois River Basin agricultural streams were similar to the national distribution, but the detection frequency was much higher (83 percent compared to 33 percent).

CHEMICALS IN WATER

Concentrations and detection frequencies, lower Illinois River Basin, 1995–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals

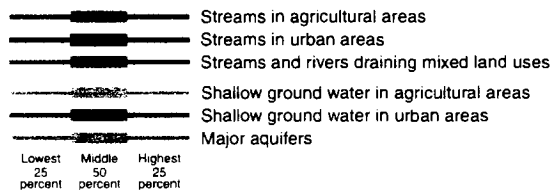
◆ Detected concentration in Study Unit

66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency

— Not measured or sample size less than two

12 Study-unit sample size. For ground water, the number of samples is equal to the number of wells sampled

National ranges of detected concentrations, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

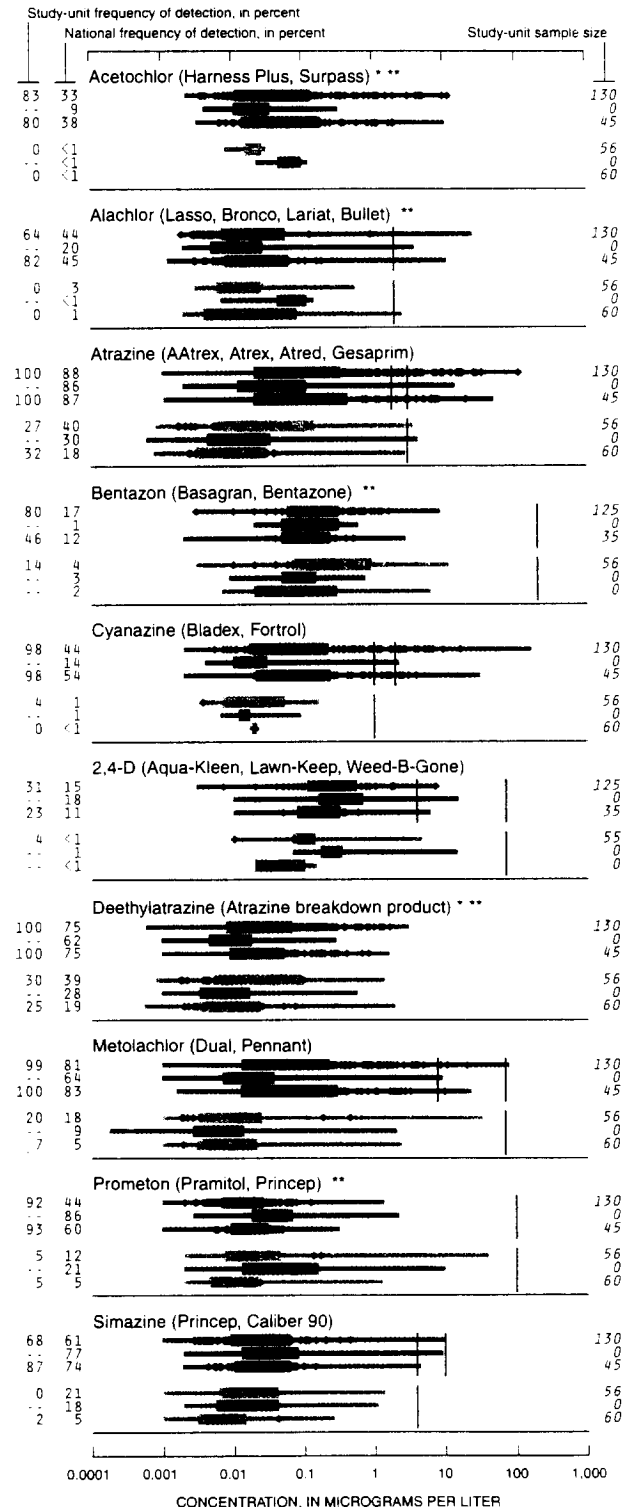


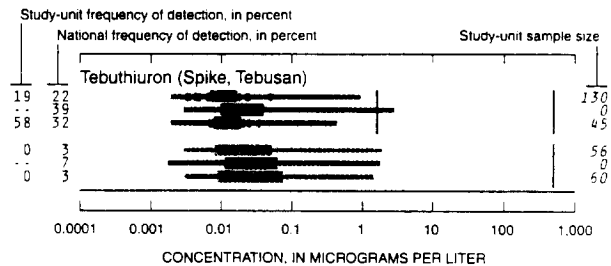
National water-quality benchmarks

National benchmarks include standards and guidelines related to drinking-water quality, criteria for protecting the health of aquatic life, and a goal for preventing stream eutrophication due to phosphorus. Sources include the U.S. Environmental Protection Agency and the Canadian Council of Ministers of the Environment

- Drinking-water quality (applies to ground water and surface water)
- Protection of aquatic life (applies to surface water only)
- Prevention of eutrophication in streams not flowing directly into lakes or impoundments
- No benchmark for drinking-water quality
- No benchmark for protection of aquatic life

Pesticides in water—Herbicides





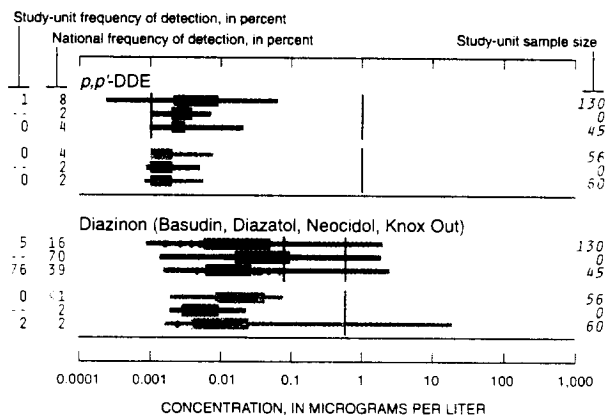
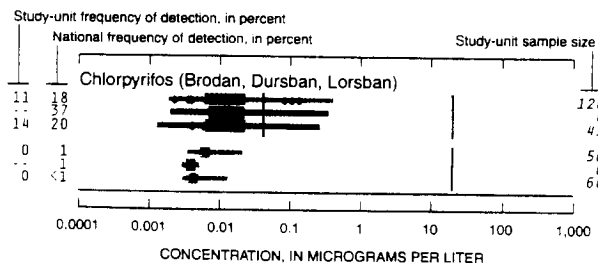
Other herbicides detected

- Acifluorfen (Blazer, Tackle 2S) **
- Benfluralin (Balan, Benefin, Bonalan) * **
- Bromoxynil (Buctril, Brominal) *
- Butylate (Sutan +, Genate Plus, Butilate) **
- DCPA (Dacthal, chlorthal-dimethyl) * **
- Dacthal mono-acid (Dacthal breakdown product) * **
- Dicamba (Banvel, Dianat, Scotts Proturf)
- 2,6-Diethylaniline (Alachlor breakdown product) * **
- Diuron (Crisuron, Karmex, Diurex) **
- EPTC (Eptam, Farmarox, Alirox) * **
- Linuron (Lorox, Linex, Sarclex, Linurex, Afalon) *
- MCPA (Rhomene, Rhonox, Chiptox)
- Metribuzin (Lexone, Sencor)
- Molinate (Ordram) * **
- Pendimethalin (Pre-M, Prowl, Stomp) * **
- Propachlor (Ramrod, Satecid) **
- Terbacil (Sinbar) **
- Triallate (Far-Go, Avadex BW, Tri-allate) *
- Triclopyr (Garlon, Grandstand, Redeem, Remedy) * **
- Trifluralin (Treflan, Gowan, Tri-4, Trific)

Herbicides not detected

- Bromacil (Hyvar X, Urox B, Bromax)
- Chloramben (Amiben, Amilon-WP, Vegiben) **
- Clopyralid (Stinger, Lontrel, Transline) * **
- 2,4-DB (Butyrac, Butoxone, Embutox Plus, Embutone) * **
- Dichlorprop (2,4-DP, Seritox 50, Lentemul) * **
- Dinoseb (Dinosebe)
- Ethalfuralin (Sonalan, Curbit) * **
- Fenuron (Fenulon, Fenidim) * **
- Fluometuron (Flo-Met, Cotoran) **
- MCPB (Thistrol) * **
- Napropamide (Devrinol) * **
- Neburon (Neburea, Neburyl, Noruben) * **
- Norflurazon (E vital, Predict, Solicam, Zorial) * **
- Oryzalin (Surflan, Dirimal) * **
- Pebulate (Tillam, PEBC) * **
- Picloram (Grazon, Tordon)
- Pronamide (Kerb, Propyzamid) **
- Propanil (Stam, Stampede, Wham) * **
- Propham (Tuberite) **
- 2,4,5-T **
- 2,4,5-TP (Silvex, Fenoprop) **
- Thiobencarb (Bolero, Saturn, Benthiocarb) * **

Pesticides in water—Insecticides



Other insecticides detected

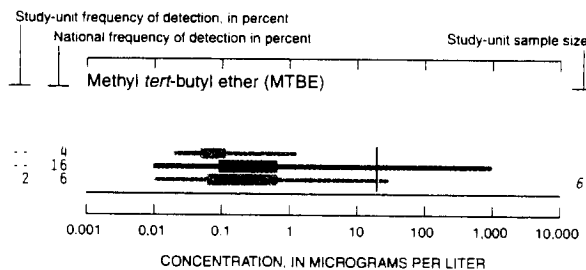
- Carbaryl (Carbamine, Denapon, Sevin)
- Carbofuran (Furadan, Curaterr, Yaltox)
- Dieldrin (Panoram D-31, Octalox, Compound 497)
- Fonofos (Dyfonate, Capfos, Cudgel, Tycap) **
- Malathion (Malathion)
- Methyl parathion (Pencap-M, Folidol-M) **
- Oxamyl (Vydate L, Pratt) **
- Propargite (Comite, Omite, Ornamate) * **
- Terbufos (Contraven, Counter, Pilarfox) **

Insecticides not detected

- Aldicarb (Temik, Ambush, Pounce)
- Aldicarb sulfone (Standak, aldoxycarb)
- Aldicarb sulfoxide (Aldicarb breakdown product)
- Azinphos-methyl (Guthion, Gusathion M) *
- Disulfoton (Disyston, Di-Syston) **
- Ethoprop (Mocap, Ethoprophos) * **
- alpha-HCH (alpha-BHC, alpha-lindane) **
- gamma-HCH (Lindane, gamma-BHC)
- 3-Hydroxycarbofuran (Carbofuran breakdown product) * **
- Methiocarb (Slug-Geta, Grandslam, Mesuroil) * **
- Methomyl (Lanox, Lannate, Acinate) **
- Parathion (Roethyl-P, Alkron, Panthion, Phoskil) *
- cis-Permethrin (Ambush, Astro, Pounce) * **
- Phorate (Thimet, Granutox, Geomet, Rampart) * **
- Propoxur (Baygon, Blattanex, Unden, Proprotox) * **

Volatile organic compounds (VOCs) in ground water

These graphs represent data from 16 Study Units, sampled from 1996 to 1998



Other VOCs detected

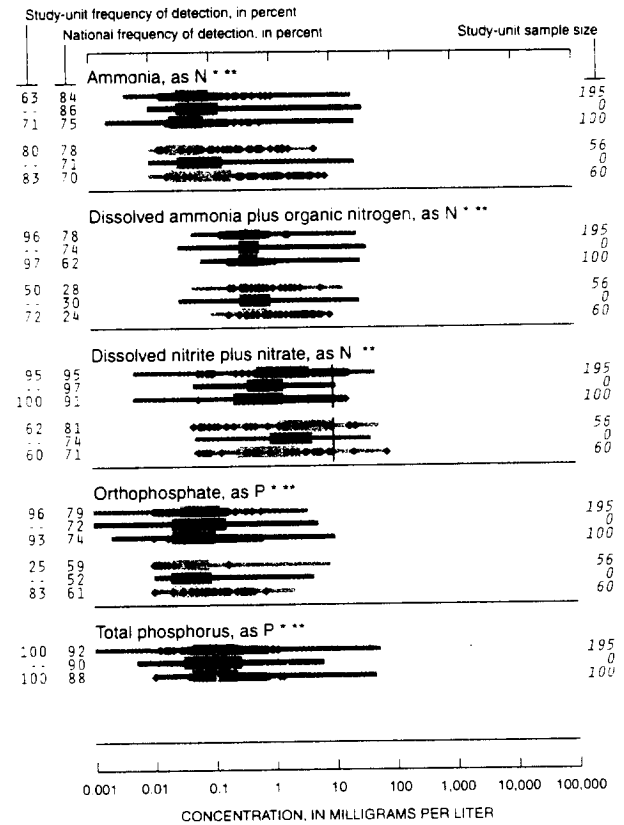
- Bromodichloromethane (Dichlorobromomethane)
- 2-Butanone (Methyl ethyl ketone (MEK)) *
- Carbon disulfide *
- Chlorobenzene (Monochlorobenzene)
- Chlorodibromomethane (Dibromochloromethane)
- Chloroethane (Ethyl chloride) *
- Chloromethane (Methyl chloride)
- Dichlorodifluoromethane (CFC 12, Freon 12)
- Dichloromethane (Methylene chloride)

- 1,3 & 1,4-Dimethylbenzene (*m*-&*p*-Xylene)
- 1,4-Epoxy butane (Tetrahydrofuran, Diethylene oxide)
- 1-Ethyl-2-methylbenzene (2-Ethyltoluene)
- Hexachlorobutadiene
- Iodomethane (Methyl iodide)
- 4-Methyl-2-pentanone (Methyl isobutyl ketone (MIBK))
- Methylbenzene (Toluene)
- 2-Propanone (Acetone)
- Tetrachloroethene (Perchloroethene)
- Tetrachloromethane (Carbon tetrachloride)
- 1,2,3,4-Tetramethylbenzene (Prehnitene)
- 1,2,3,5-Tetramethylbenzene (Isodurene)
- Tribromomethane (Bromoform)
- 1,2,3-Trichlorobenzene
- 1,1,1-Trichloroethane (Methylchloroform)
- Trichloromethane (Chloroform)
- 1,2,3-Trimethylbenzene (Hemimellitene)
- 1,2,4-Trimethylbenzene (Pseudocumene)

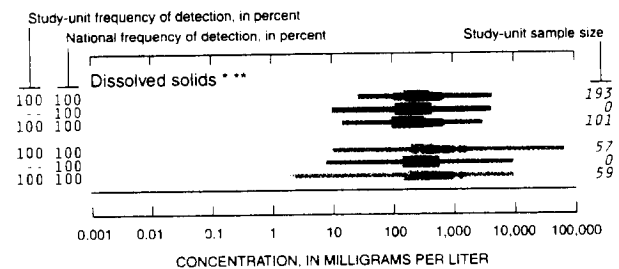
VOCs not detected

- tert*-Amyl methyl ether (*tert*-amyl methyl ether (TAME))
- Benzene
- Bromobenzene (Phenyl bromide)
- Bromochloromethane (Methylene chlorobromide)
- Bromoethene (Vinyl bromide)
- Bromomethane (Methyl bromide)
- n*-Butylbenzene (1-Phenylbutane)
- sec*-Butylbenzene
- tert*-Butylbenzene
- 3-Chloro-1-propene (3-Chloropropene)
- 1-Chloro-2-methylbenzene (*o*-Chlorotoluene)
- 1-Chloro-4-methylbenzene (*p*-Chlorotoluene)
- Chloroethene (Vinyl chloride)
- 1,2-Dibromo-3-chloropropane (DBCP, Nemagon)
- 1,2-Dibromoethane (Ethylene dibromide, EDB)
- Dibromomethane (Methylene dibromide)
- trans*-1,4-Dichloro-2-butene ((*Z*)-1,4-Dichloro-2-butene)
- 1,2-Dichlorobenzene (*o*-Dichlorobenzene)
- 1,3-Dichlorobenzene (*m*-Dichlorobenzene)
- 1,4-Dichlorobenzene (*p*-Dichlorobenzene)
- 1,2-Dichloroethane (Ethylene dichloride)
- 1,1-Dichloroethane (Ethylidene dichloride)
- 1,1-Dichloroethene (Vinylidene chloride)
- trans*-1,2-Dichloroethene ((*E*)-1,2-Dichloroethene)
- cis*-1,2-Dichloroethene ((*Z*)-1,2-Dichloroethene)
- 1,2-Dichloropropane (Propylene dichloride)
- 2,2-Dichloropropane
- 1,3-Dichloropropane (Trimethylene dichloride)
- trans*-1,3-Dichloropropene ((*E*)-1,3-Dichloropropene)
- cis*-1,3-Dichloropropene ((*Z*)-1,3-Dichloropropene)
- 1,1-Dichloropropene
- Diethyl ether (Ethyl ether)
- Diisopropyl ether (Diisopropylether (DIPE))
- 1,2-Dimethylbenzene (*o*-Xylene)
- Ethylbenzene (Styrene)
- Ethyl methacrylate
- Ethyl *tert*-butyl ether (Ethyl-*t*-butyl ether (ETBE))
- Ethylbenzene (Phenylethane)
- 1,1,1,2,2,2-Hexachloroethane (Hexachloroethane)
- 2-Hexanone (Methyl butyl ketone (MBK))
- Isopropylbenzene (Cumene)
- p*-Isopropyltoluene (*p*-Cymene)
- Methyl acrylonitrile
- Methyl-2-methacrylate (Methyl methacrylate)
- Methyl-2-propenoate (Methyl acrylate)
- Naphthalene
- 2-Propenenitrile (Acrylonitrile)
- n*-Propylbenzene (Isocumene)
- 1,1,2,2-Tetrachloroethane
- 1,1,1,2-Tetrachloroethane
- 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon 113)
- 1,2,4-Trichlorobenzene
- 1,1,2-Trichloroethane (Vinyl trichloride)
- Trichloroethene (TCE)
- Trichlorofluoromethane (CFC 11, Freon 11)
- 1,2,3-Trichloropropane (Allyl trichloride)
- 1,3,5-Trimethylbenzene (Mesitylene)

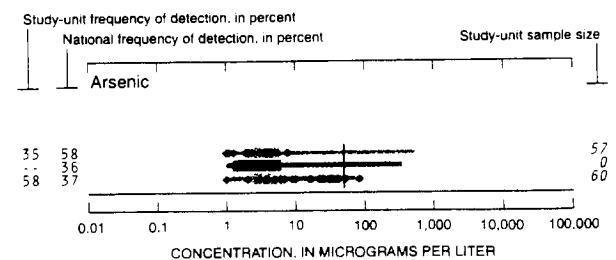
Nutrients in water

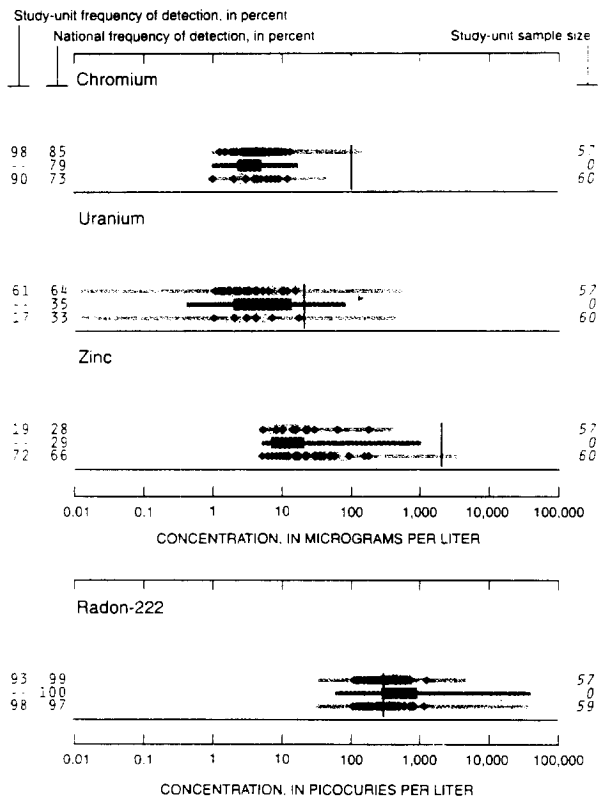


Dissolved solids in water



Trace elements in ground water





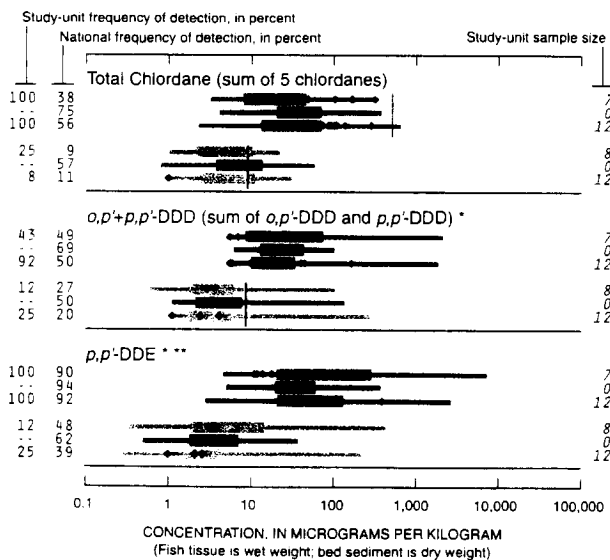
Other trace elements detected

Lead
Selenium

Trace elements not detected

Cadmium

Organochlorines in fish tissue (whole body) and bed sediment



CHEMICALS IN FISH TISSUE AND BED SEDIMENT

Concentrations and detection frequencies, lower Illinois River Basin, 1995-98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals. Study-unit frequencies of detection are based on small sample sizes; the applicable sample size is specified in each graph

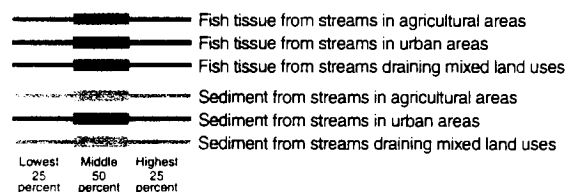
◆ Detected concentration in Study Unit

66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency

-- Not measured or sample size less than two

12 Study-unit sample size

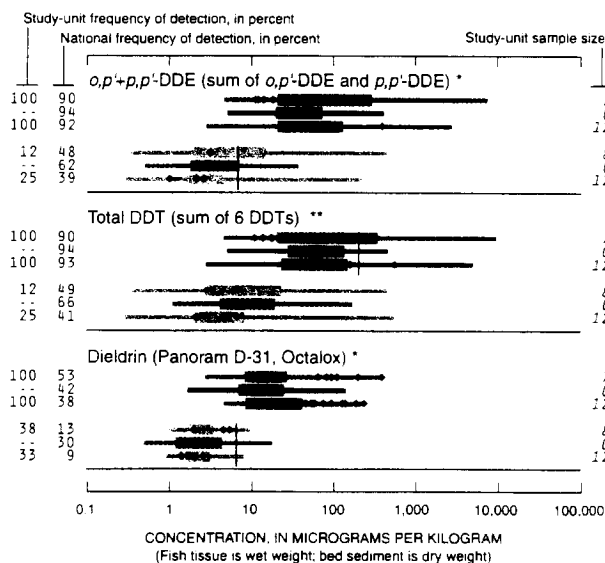
National range of concentrations detected, by land use, in 36 NAWQA Study Units, 1991-98—Ranges include only samples in which a chemical was detected

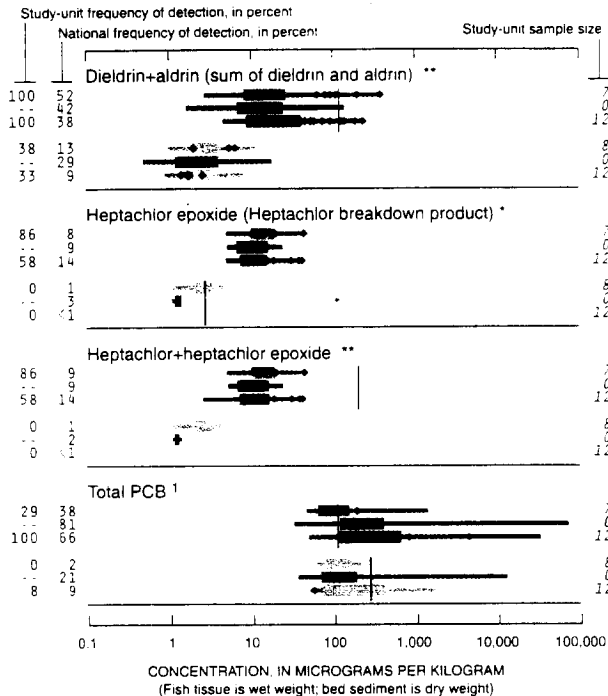


National benchmarks for fish tissue and bed sediment

National benchmarks include standards and guidelines related to criteria for protection of the health of fish-eating wildlife and aquatic organisms. Sources include the U.S. Environmental Protection Agency, other Federal and State agencies, and the Canadian Council of Ministers of the Environment

- | Protection of fish-eating wildlife (applies to fish tissue)
- | Protection of aquatic life (applies to bed sediment)
- No benchmark for protection of fish-eating wildlife
- No benchmark for protection of aquatic life





¹ The national detection frequencies for total PCB in sediment are biased low because about 30 percent of samples nationally had elevated detection levels compared to this Study Unit. See <http://water.usgs.gov/nawqa/> for additional information.

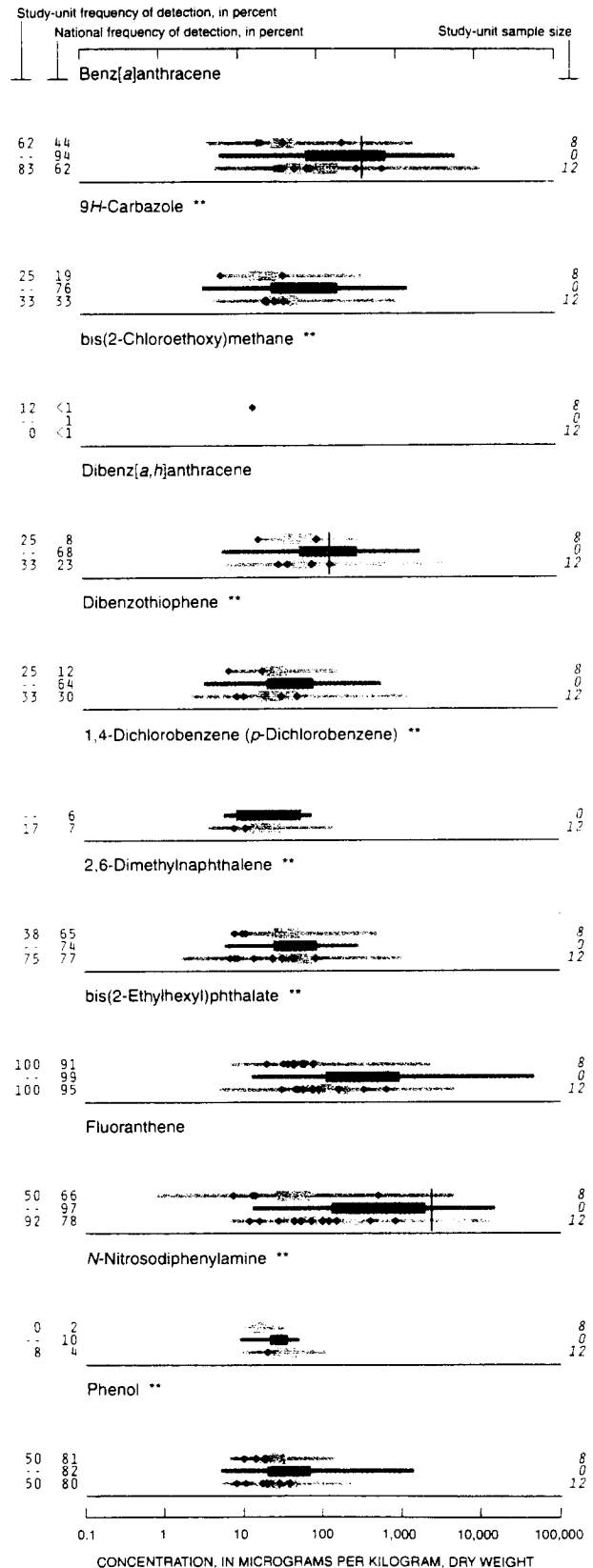
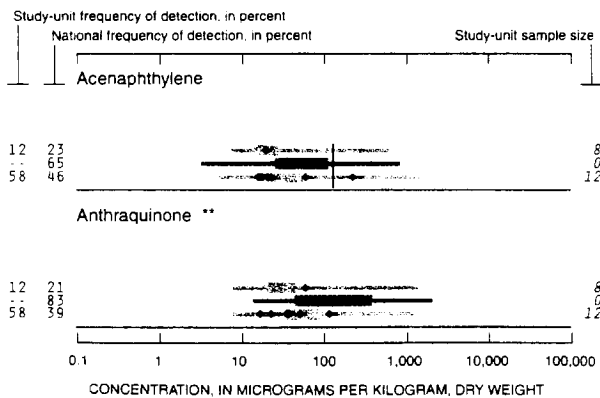
Other organochlorines detected

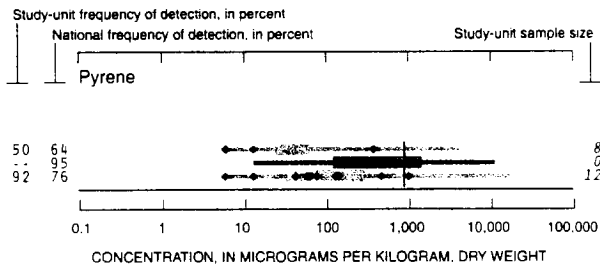
- DCPA (Dacthal, chlorthal-dimethyl) ***
- Hexachlorobenzene (HCB) **
- Pentachloroanisole (PCA) **

Organochlorines not detected

- Chloroneb (Chloronebe, Demosan) ***
- o,p'*+*p,p'*-DDT (sum of *o,p'*-DDT and *p,p'*-DDT) *
- Endosulfan I (alpha-Endosulfan, Thiodan) ***
- Endrin (Endrine)
- gamma-HCH (Lindane, gamma-BHC, Gammexane) *
- Total-HCH (sum of alpha-HCH, beta-HCH, gamma-HCH, and delta-HCH) **
- Isodrin (Isodrine, Compound 711) ***
- p,p'*-Methoxychlor (Marlate, methoxychlore) **
- o,p'*-Methoxychlor **
- Mirex (Dechlorane) **
- cis*-Permethrin (Ambush, Astro, Pounce) ***
- trans*-Permethrin (Ambush, Astro, Pounce) **
- Toxaphene (Camphechlor, Hercules 3956) ***

Semivolatile organic compounds (SVOCs) in bed sediment

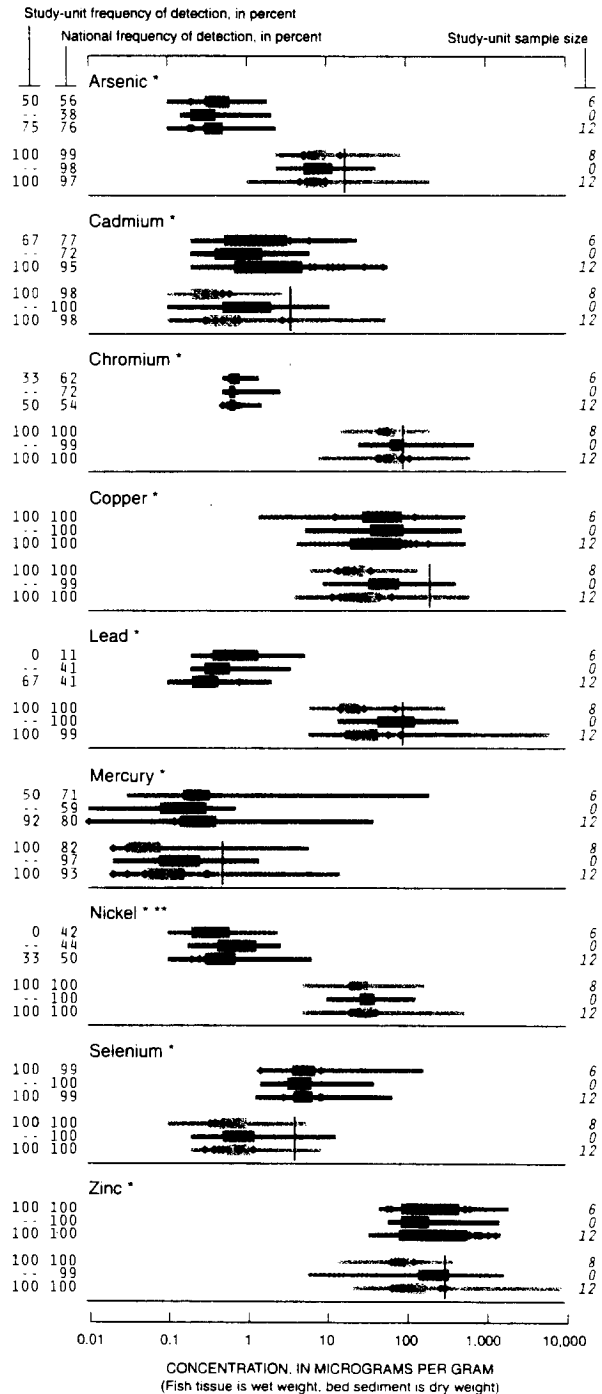




Other SVOCs detected

- Acenaphthene
 - Acridine **
 - Anthracene
 - Benzo[a]pyrene
 - Benzo[b]fluoranthene **
 - Benzo[ghi]perylene **
 - Benzo[k]fluoranthene **
 - Butylbenzylphthalate **
 - 4-Chloro-3-methylphenol **
 - 4-Chlorophenyl-phenylether **
 - Chrysene
 - p-Cresol **
 - Di-n-butylphthalate **
 - Di-n-octylphthalate **
 - 1,3-Dichlorobenzene (m-Dichlorobenzene) **
 - Diethylphthalate **
 - 1,2-Dimethylnaphthalene **
 - 1,6-Dimethylnaphthalene **
 - 3,5-Dimethylphenol **
 - Dimethylphthalate **
 - 2-Ethynaphthalene **
 - 9H-Fluorene (Fluorene)
 - Indeno[1,2,3-cd]pyrene **
 - Isoquinoline **
 - 1-Methyl-9H-fluorene **
 - 2-Methylantracene **
 - 4,5-Methylenephenanthrene **
 - 1-Methylphenanthrene **
 - 1-Methylpyrene **
 - Naphthalene
 - Nitrobenzene **
 - Phenanthrene
 - Phenanthridine **
 - 1,2,4-Trichlorobenzene **
 - 2,3,6-Trimethylnaphthalene **
- SVOCs not detected**
- C8-Alkylphenol **
 - Azobenzene **
 - Benzo[c]cinnoline **
 - 2,2-Biquinoline **
 - 4-Bromophenyl-phenylether **
 - 2-Chloronaphthalene **
 - 2-Chlorophenol **
 - 1,2-Dichlorobenzene (o-Dichlorobenzene) **
 - 2,4-Dinitrotoluene **
 - Isophorone **
 - N-Nitrosodi-n-propylamine **
 - Pentachloronitrobenzene **
 - Quinoline **

Trace elements in fish tissue (livers) and bed sediment



BIOLOGICAL INDICATORS

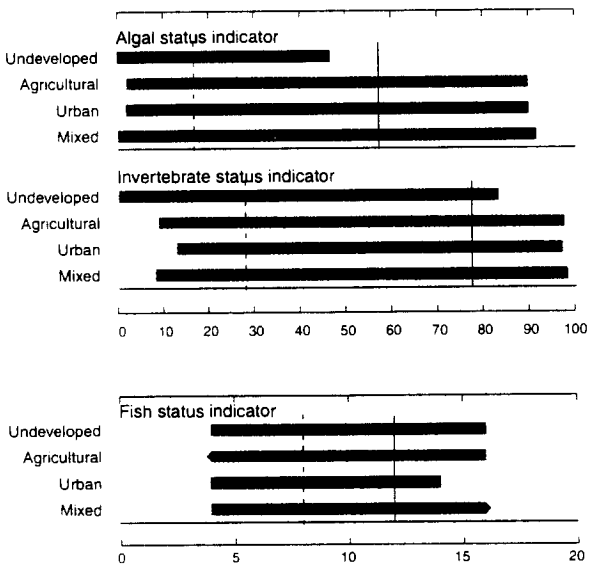
Higher national scores suggest habitat disturbance, water-quality degradation, or naturally harsh conditions. The status of algae, invertebrates (insects, worms, and clams), and fish provide a record of water-quality and stream conditions that water-chemistry indicators may not reveal. **Algal status** focuses on the changes in the percentage of certain algae in response to increasing siltation, and it often correlates with higher nutrient concentrations in some regions. **Invertebrate status** averages 11 metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. **Fish status** sums the scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that increase in association with water-quality degradation

Biological indicator value, lower Illinois River Basin, by land use, 1995-98

- ◆ Biological status assessed at a site

National ranges of biological indicators, 16 NAWQA Study Units, 1994-98

- Streams in undeveloped areas
- Streams in agricultural areas
- Streams in urban areas
- Streams in mixed-land-use areas
- 75th percentile
- - - 25th percentile



A COORDINATED EFFORT

Coordination with agencies and organizations in the lower Illinois River Basin was integral to the success of this water-quality assessment. We thank those who served as members of our liaison committee.

Federal Agencies

U.S. Environmental Protection Agency
Natural Resources Conservation Service
U.S. Fish and Wildlife Service

Springfield City Water, Light, and Power
Mahomet Valley Water Authority
Peoria Health Department
Sanitary District of Decatur

State Agencies

Illinois Environmental Protection Agency
Illinois Natural History Survey
Illinois Department of Natural Resources
Illinois State Geological Survey
Illinois State Water Survey
Illinois Department of Agriculture
Illinois Hazardous Waste Research Center
Illinois Abandoned Lands Council
Illinois Department of Public Health

Universities

University of Illinois
Water Resources Center, University of Illinois
Natural Resources and Environmental Sciences,
University of Illinois
College of Agriculture, Consumer and Environmental
Sciences, University of Illinois
College of Engineering, University of Illinois

Local Agencies

Bloomington and Normal Water Reclamation District
Metropolitan Water Reclamation District of Greater
Chicago
Bloomington Water Department
Greater Peoria Sanitary District
City of Bloomington

Other public and private organizations

Tri-County River Front Action Forum
The Nature Conservancy, Illinois Chapter
Heartland Water Resources

We thank the following individuals for contributing to this effort.

Steve Gough, Gough and Associates, for assisting the biology team: field-data collection

David J. Fazio, U.S. Geological Survey, for most maps in this report

Kymm Akers, U.S. Geological Survey, for drafting two maps used in this report

William S. Morrow, Jr., David H. Dupré, Debbie L. Adolphson, Tim A. Brown and David G. Frothingham,
U.S. Geological Survey, for data collection

Ken Hlinka, Illinois State Water Survey, for providing ground-water-level data

Sherry Reeves, U.S. Geological Survey, for editorial review and report layout

Jim Stark and Stephen Kalkhoff, U.S. Geological Survey, for review of manuscript

Matthew Short, Illinois Environmental Protection Agency, for review of manuscript

Gretchen Bonfert, McKnight Foundation, for review of manuscript

Michael Eberle, Leah Hout, and Joyce McClure, U.S. Geological Survey, for editorial review

NAWQA

National Water-Quality Assessment (NAWQA) Program
Lower Illinois River Basin

Groschen and others—Water Quality in the Lower Illinois River Basin
U.S. Geological Survey Circular 1209



ISBN 0-607-95423-X



9 780607 954234

R0024485

Water Quality in the Eastern Iowa Basins Iowa and Minnesota, 1996-98



U.S. Department of the Interior
U.S. Geological Survey

Circular 1210

R0024486

POINTS OF CONTACT AND ADDITIONAL INFORMATION

The companion Web site for NAWQA summary reports:

<http://water.usgs.gov/nawqa/>

Eastern Iowa Basins contact and Web site:

USGS State Representative
U.S. Geological Survey
Water Resources Division
PO Box 1230
400 South Clinton St., Rm 269
Iowa City, IA 52244
e-mail: dc_ia@usgs.gov
<http://iowa.usgs.gov/nawqa/index.html>

National NAWQA Program:

Chief, NAWQA Program
U.S. Geological Survey
Water Resources Division
12201 Sunrise Valley Drive, M.S. 413
Reston, VA 20192
<http://water.usgs.gov/nawqa/>

Other NAWQA summary reports

River Basin Assessments

Albemarle-Pamlico Drainage Basin (Circular 1157)
Allegheny and Monongahela River Basins (Circular 1202)
Apalachicola-Chattahoochee-Flint River Basin (Circular 1164)
Central Arizona Basins (Circular 1213)
Central Columbia Plateau (Circular 1144)
Central Nebraska Basins (Circular 1163)
Connecticut, Housatonic and Thames River Basins (Circular 1155)
Georgia-Florida Coastal Plain (Circular 1151)
Hudson River Basin (Circular 1165)
Kanawha - New River Basins (Circular 1204)
Lake Erie - Lake Saint Clair Drainages (Circular 1203)
Las Vegas Valley Area and the Carson and Truckee River Basins (Circular 1170)
Lower Illinois River Basin (Circular 1209)
Long Island - New Jersey Coastal Drainages (Circular 1201)
Lower Susquehanna River Basin (Circular 1168)
Mississippi Embayment (Circular 1208)
Ozark Plateaus (Circular 1158)
Potomac River Basin (Circular 1166)
Puget Sound Basin (Circular 1216)
Red River of the North Basin (Circular 1169)

Rio Grande Valley (Circular 1162)
Sacramento River Basin (Circular 1215)
San Joaquin-Tulare Basins (Circular 1159)
Santee River Basin and Coastal Drainages (Circular 1206)
South-Central Texas (Circular 1212)
South Platte River Basin (Circular 1167)
Southern Florida (Circular 1207)
Trinity River Basin (Circular 1171)
Upper Colorado River Basin (Circular 1214)
Upper Mississippi River Basin (Circular 1211)
Upper Snake River Basin (Circular 1160)
Upper Tennessee River Basin (Circular 1205)
Western Lake Michigan Drainages (Circular 1156)
White River Basin (Circular 1150)
Willamette Basin (Circular 1161)

National Assessments

The Quality of Our Nation's Waters—Nutrients and Pesticides (Circular 1225)

Front cover: Aerial view of the Old Mans Creek basin showing pattern of wooded riparian buffers and cropland typical of the Southern Iowa Drift Plain landform in Iowa (*photograph by Doug Schnoebelen, USGS*).

Back cover: Left, study unit biologist measuring water transparency (*photo by Stephen Porter, USGS*); center, study unit staff collecting aquatic organisms from woody debris (*photograph by Stephen Porter*); right, scientists measuring water temperature and dissolved oxygen in Old Mans Creek in preparation for sample collection (*photograph by Debra Sneck-Fahrer, USGS*).

Water Quality in the Eastern Iowa Basins, Iowa and Minnesota, 1996–98

By Stephen J. Kalkhoff, Kimberlee K. Barnes, Kent D. Becher,
Mark E. Savoca, Douglas J. Schnoebelen, Eric M. Sadorf, Stephen D. Porter,
and Daniel J. Sullivan

U.S. DEPARTMENT OF THE INTERIOR
GALE A. NORTON, SECRETARY

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

2000

Free on application to the
U.S. Geological Survey
Information Services
Box 25286 Federal Center
Denver, CO 80225

Or call: 1-888-ASK-USGS

Library of Congress Cataloging-in-Publications Data

Water quality in the eastern Iowa basins, Iowa and Minnesota, 1996-98 / by Stephen J. Kalkhoff...[et al.].
p. cm. -- (U.S. Geological Survey Circular ; 1210)

Includes bibliographical references.

ISBN 0-607-95415-9 (alk. paper)

1. Water quality--Iowa. 2. Water quality--Minnesota. 3. Watersheds--Iowa. 4. Watersheds--Minnesota. I. Kalkhoff, Stephen J. II. Geological Survey (U.S.) III. Series.

TD224.I8 W274 2000
363.739'42'097776--dc21

00-049465

R0024489

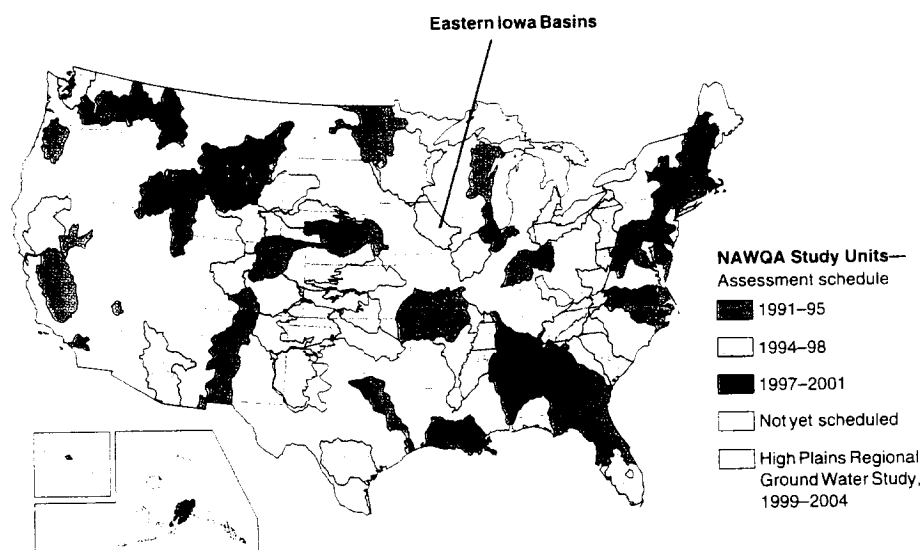
CONTENTS

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM	IV
SUMMARY OF MAJOR FINDINGS	1
Stream and River Highlights	1
Ground-Water Highlights	2
INTRODUCTION TO THE EASTERN IOWA BASINS	3
MAJOR FINDINGS	7
Nutrients in Ground Water and Streams	7
NATIONAL PERSPECTIVE—Nutrient concentrations in streams were among the highest in the Nation	8
Animal Feeding Operations contribute additional nutrients to streams	13
Pesticides in Ground Water and Streams	14
NATIONAL PERSPECTIVE—Pesticide concentrations rank higher in ground water than in streams and rivers	15
What is a Pesticide Degradate?	17
Multiple pesticide compounds occur more frequently in streams than in ground water	20
Other Organic Compounds	21
NATIONAL PERSPECTIVE—MTBE detection rates are similar to the average detection rate in the Nation	21
NATIONAL PERSPECTIVE—Biological communities in streams consist of organisms that are moderately to highly tolerant of environmental degradation	22
Riparian buffer zones influence the quality of Midwestern streams and rivers	25
STUDY UNIT DESIGN	26
GLOSSARY	28
REFERENCES	30
APPENDIX—WATER-QUALITY DATA FROM THE EASTERN IOWA BASINS IN A NATIONAL CONTEXT	32

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

THIS REPORT summarizes major findings about water quality in the Eastern Iowa Basins that emerged from an assessment conducted between 1996 and 1998 by the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program. Water quality is discussed in terms of local and regional issues and compared to conditions found in all 36 NAWQA study areas, called Study Units, assessed to date. Findings also are explained in the context of selected national benchmarks, such as those for drinking-water quality and the protection of aquatic organisms. The NAWQA Program was not intended to assess the quality of the Nation's drinking water, such as by monitoring water from household taps. Rather, the assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water monitoring programs. The comparisons made in this report to drinking-water standards and guidelines are only in the context of the available untreated resource. Finally, this report includes information about the status of aquatic communities and the condition of in-stream habitats as elements of a complete water-quality assessment.

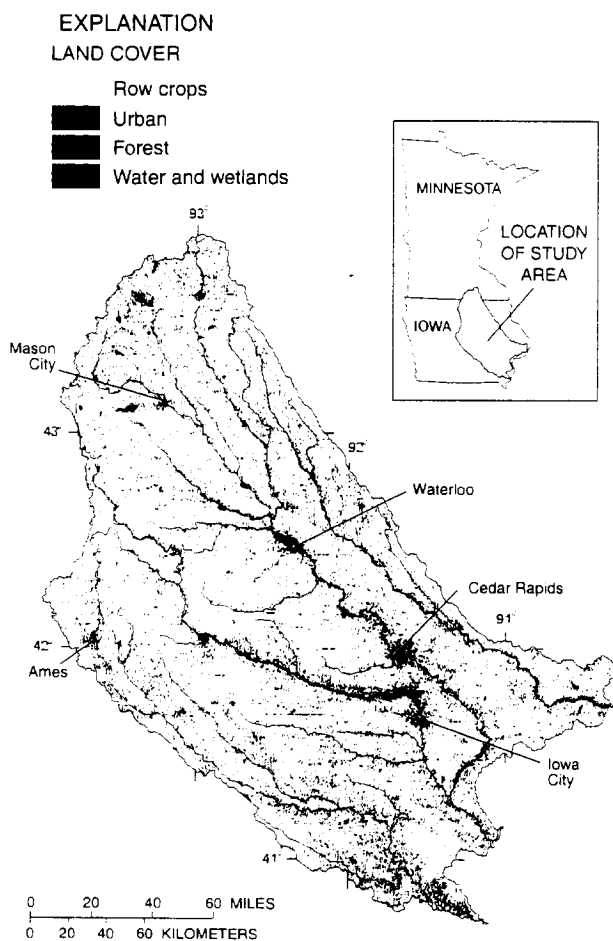
Many topics covered in this report reflect the concerns of officials of State and Federal agencies, water-resource managers, and members of stakeholder groups who provided advice and input during the Eastern Iowa Basins assessment. Basin residents who wish to know more about water quality in the areas where they live will find this report informative as well.



THE NAWQA PROGRAM seeks to improve scientific and public understanding of water quality in the Nation's major river basins and ground-water systems. Better understanding facilitates effective resource management, accurate identification of water-quality priorities, and successful development of strategies that protect and restore water quality. Guided by a nationally consistent study design and shaped by ongoing communication with local, State, and Federal agencies, NAWQA assessments support the investigation of local issues and trends while providing a firm foundation for understanding water quality at regional and national scales. The ability to integrate local and national scales of data collection and analysis is a unique feature of the USGS NAWQA Program.

The Eastern Iowa Basins is one of 51 water-quality assessments initiated since 1991, when the U.S. Congress appropriated funds for the USGS to begin the NAWQA Program. As indicated on the map, 36 assessments have been completed, and 15 more assessments will conclude in 2001. Collectively, these assessments cover about one-half of the land area of the United States and include water resources that are available to more than 60 percent of the U.S. population.

SUMMARY OF MAJOR FINDINGS



The Eastern Iowa Basins Study Unit encompasses the Wapsipinicon, the Cedar, the Iowa, and the Skunk River Basins and covers about 19,500 mi² in eastern Iowa and southern Minnesota. In 1990, about 40 percent of the more than 1 million people in the Study Unit were concentrated in cities with populations of greater than 20,000 people. Cedar Rapids is the only city with a population greater than 100,000. Ground water is the major source for municipal, industrial, and domestic supplies. During the study, Iowa City was the largest municipal user of surface water. Over 90 percent of the land in the Study Unit is used for agricultural purposes. Forested areas account for only 4 percent of the land. Data from Eros Data Center, 1994.

Stream And River Highlights

Nitrogen and phosphorus concentrations in streams in the Eastern Iowa Basins Study Unit rank as some of the highest in the Corn Belt (see map, p. 8) as well as the Nation and were higher than the drinking-water standard in many samples. These conditions reflect intensive use of the land for growing crops and dense populations of livestock in some basins.

- Nitrate concentrations in 22 percent of the stream samples exceeded the U.S. Environmental Protection Agency (USEPA) drinking-water standard of 10 mg/L (milligrams per liter). The standard was most frequently exceeded during June, soon after spring fertilizer application. Although many

of the streams sampled are not currently used for drinking-water supplies, the Cedar and Iowa Rivers are the direct or indirect source for Cedar Rapids and Iowa City—two of the largest cities in the study area.

- The highest nitrate concentrations occurred in medium-sized streams draining basins with the most intensive row-crop agriculture and in a stream draining a basin with both intensive row-crop agriculture and dense concentrations of large-scale animal feeding operations. Nitrate concentrations in these streams exceeded 10 mg/L in almost 50 percent of the samples. Conversely, nitrate concentrations were lowest in basins that had greater percentages of pasture, grassland, and forest.
- Total phosphorus concentrations frequently (75 percent of the samples) exceeded the 0.1-mg/L USEPA-recommended goal to minimize algal growth in rivers. Total phosphorus concentrations were greatest in streams and rivers that drain basins with more highly erodible soils and in large river basins that contain the largest cities and towns in the Study Unit.
- The large amounts of nitrogen and phosphorus that are transported to the Mississippi River from the Study Unit represent an economic loss to farmers and a potential environmental threat to downstream waters. The estimated annual loss of 17 to 41 lb/acre (pounds per acre) of nitrogen and 1.2 to 1.5 lb/acre of phosphorus represents a potential loss in crop yield or the cost of additional fertilizer needed to compensate for that flushed from the fields. Nutrients transported to the Mississippi River likely reach the Gulf of Mexico where they contribute to eutrophication and hypoxia.
- Although the use of herbicides and insecticides in the Study Unit is among the most intense nationwide, herbicide concentrations in streams were not among the highest 25 percent nationally, and insecticide concentrations were in the lowest 25 percent nationally. Breakdown compounds (degradates), whose widespread occurrence has only recently been discovered and about which little is known of the human and environmental effects, generally accounted for the majority of the pesticide compounds present in rivers and streams.
- The most commonly used herbicides were the most frequently detected and were generally present in the greatest concentrations. Atrazine and metolachlor were detected in all stream samples. Concentrations generally ranged from 0.1 to 1.0 µg/L (microgram per liter). Atrazine concentrations exceeded the USEPA 3.0-µg/L drinking-water standard in about 10 percent of the samples; exceedances occurred mainly during late-spring runoff.
- Acetochlor, a conditionally registered herbicide that is intended to replace several other commonly used herbicides, was frequently detected, but concentrations were less than 0.1 µg/L in 75 percent of the samples. Mean annual acetochlor concentrations did not exceed the 2.0-µg/L USEPA registration requirement at any site, but concentrations did exceed that level in about 3 percent of the individual samples. The maximum concentration measured during the study (10.6 µg/L) exceeded the level that would require biweekly sampling by water-supply systems.
- Alachlor, metolachlor, and acetochlor degradates are present in relatively high concentrations throughout the year, indicating that they are more persistent than their parent compounds.
- Carbofuran and chlorpyrifos, insecticides that have been identified as posing a high risk to aquatic insects and mussels, were present in as much as 60 percent of the monthly samples during the summer when these insecticides are normally applied.
- Riparian buffer zones influence the quality of water in streams and rivers. Biological communities respond to tree density in riparian buffer zones. Invertebrate taxa indicative of good stream quality increased with increased numbers of trees. In contrast, streams that were not shaded by trees contained large algal growths considered indicative of eutrophication.

Selected Indicators of Stream-Water Quality

	Small Streams		Major Rivers
	Urban	Agricultural	Mixed Land Uses
Pesticides	—		
Pesticide degradates	—		
Nitrate ¹	—		
Total phosphorus ¹	—		
Trace elements ²			
Organo-chlorines ³			
Semivolatile organics ⁵			

- Percentage of samples with concentrations **greater than or equal to** health-related national guidelines for drinking water, protection of aquatic life, or contact recreation
- Percentage of samples with detected concentrations **less than** health-related national guidelines for drinking water, protection of aquatic life, or contact recreation
- Percentage of samples with **no detection**
- Not assessed

Major influences on streams and rivers

- Agricultural storm runoff
- Animal feeding operations
- Tile-line drainage
- Urban areas

Ground-Water Highlights

Compared to surface water, ground water in the Eastern Iowa Basins had substantially lower nutrient concentrations and less frequent detections. Land use, however, had a substantial effect on the quality of water in alluvial aquifers. Pesticide degradates were some of the most commonly detected constituents in these aquifers. Nitrate and methyl *tert*-butyl ether (MTBE) exceeded the USEPA standard or advisory in some of the samples. In contrast, bedrock aquifers, which are generally protected by clay and shale layers, typically had low nitrate concentrations and low frequency of pesticide detections.

- Nitrate concentrations generally decreased with depth in the alluvial aquifers. Biological denitrification may result in decreased nitrate concentration with depth, but it is also possible that the deeper water infiltrated during past years when less fertilizer was used for crops.

- Nutrients move from ground water to streams by natural drainage and tile lines. Nitrate concentrations in 24 of 25 medium-sized streams exceeded 10 mg/L during the sampling period in May 1998 when streamflow originated primarily from ground-water discharge. Nitrate concentrations consistently exceeded 10 mg/L in water from a selected tile line draining to the Iowa River.
- Pesticides were detected in alluvial aquifers underlying both agricultural and urban areas, but shallow ground water in agricultural areas contained greater concentrations than urban areas. A greater variety of pesticide compounds was detected in urban areas than agricultural areas, reflecting a more diverse usage.
- Pesticides most commonly detected in the alluvial aquifers underlying urban areas were atrazine, prometon, and metolachlor. Pesticide concentrations did not exceed established drinking-water standards.
- With the exception of atrazine and metolachlor and prometon in urban areas, pesticides were infrequently detected in alluvial aquifers. Pesticide degradates generally were more commonly detected in the alluvial aquifers than their parent compounds. The greater presence of degradates indicates that many pesticides break down in the soil and that the resulting pesticide degradates are transported to the shallow aquifers.
- MTBE, a common gasoline additive used to increase the octane content or ensure cleaner burning, was detected in 23 percent of samples from alluvial aquifers in urban areas. Concentrations exceeded the USEPA drinking-water advisory in samples from 6 percent of the wells.

Major influences on ground water

- Lawn, garden, and agricultural fertilizers
- Agricultural and urban pesticides
- Leaking underground fuel-storage tanks

Selected Indicators of Ground-Water Quality

	Shallow Ground Water		Supply Wells	
	Urban	Agricultural	Domestic	Public
Pesticides				
Nitrate				
Radon				
Volatile organics ⁴				

- Percentage of samples with concentrations **greater than or equal to** health-related national guidelines for drinking water, protection of aquatic life, or contact recreation (**Detected in 1 percent or less of samples)
- Percentage of samples with concentrations **less than** health-related national guidelines for drinking water, protection of aquatic life, or contact recreation
- Percentage of samples with **no detection** (*Not detected in 1 percent or less of samples)
- Not assessed

¹ Phosphorus and nitrogen, sampled in water.
² Arsenic, mercury, and metals, sampled in sediment, fish tissue, and water.
³ DDT and PCBs, sampled in sediment and fish tissue
⁴ Solvents, refrigerants, fumigants, and gasoline compounds, sampled in water.
⁵ By-products of fossil-fuel combustion; components of coal and crude oil, sampled in sediment and fish tissue.

INTRODUCTION TO THE EASTERN IOWA BASINS

The Eastern Iowa Basins Study Unit encompasses the Wapsipinicon, the Cedar, the Iowa, and the Skunk River Basins and encompasses about 19,500 mi² (square miles) in eastern Iowa and southern Minnesota (fig. 1). The four major rivers in the Study Unit generally flow in a southeasterly direction and empty into the Mississippi River. The basins of these four major rivers are relatively long and narrow. The Wapsipinicon River originates in southeastern Minnesota and extends about 225 mi (miles) to its confluence with the Mississippi River. The Wapsipinicon River Basin has a drainage area of 2,540 mi². The Cedar River originates in southern Minnesota and joins the Iowa River about 30 mi upstream from the mouth of

the Iowa River. The Iowa River originates in north-central Iowa. The Iowa and the Cedar River Basins cover 12,640 mi², more than 90 percent of which is in Iowa. The Skunk River originates in central Iowa and drains about 4,350 mi².

Geology

Glaciers created a land surface with three distinct regions in the Eastern Iowa Basins Study Unit: the Des Moines Lobe, the Iowan Surface, and the Southern Iowa Drift Plain (fig. 1). The Des Moines Lobe is characterized by low relief with some distinct ridges near the eastern boundary and occasional depressions that form lakes, ponds, and swamps. Glacial till is the dominant surficial mate-

rial, and alluvium is present along the streams. Poorly drained soils have developed on the tills. The Iowan Surface has gently rolling topography with long slopes, low relief, and a mature drainage pattern. The surficial material is primarily glacial drift with thin layers of windblown loess on the ridges and alluvium near the streams. A subregion of the Iowan Surface (Iowan Karst) was defined for this study in an area where bedrock is close to the land surface. In the Southern Iowa Drift Plain, streams have eroded deeply into the glacial drift and loess mantle to produce a steeply rolling terrain with broad, flat drainage divides. Moderately well-drained soils have developed on the loess.

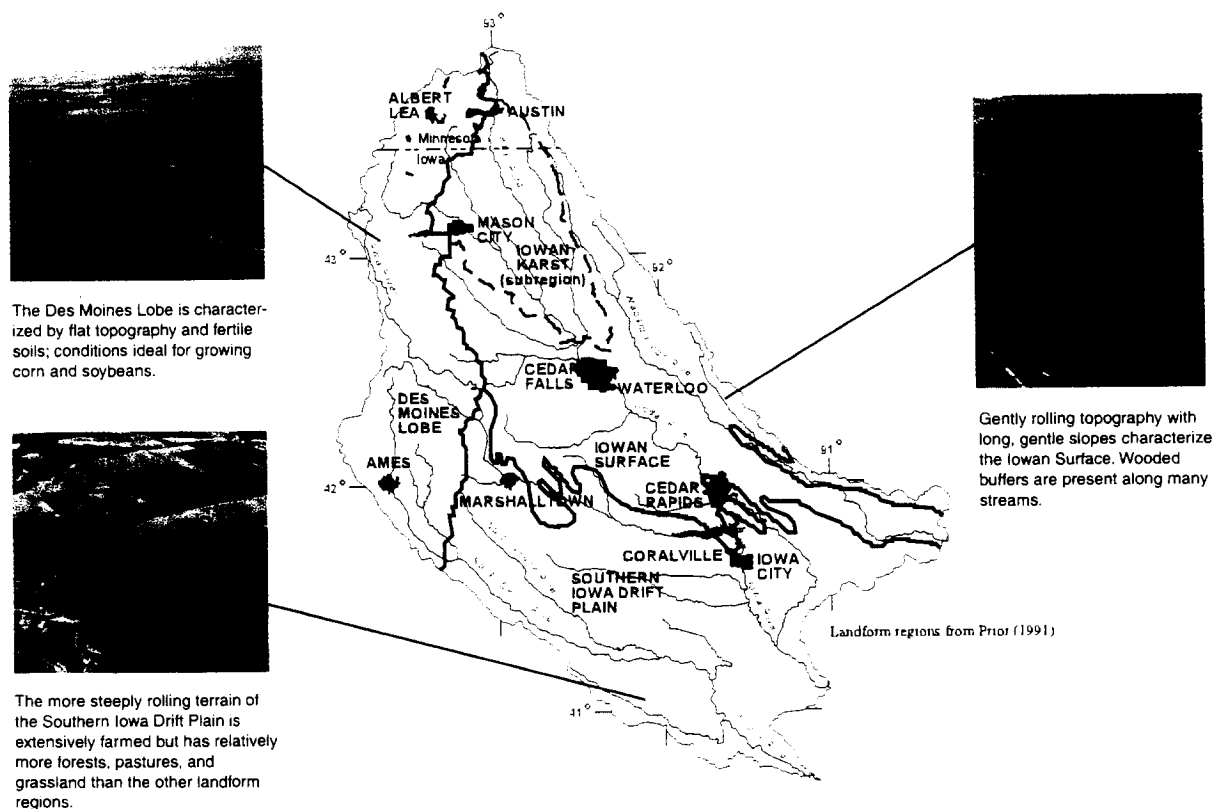


Figure 1. Glacial deposits formed three major landforms with characteristic soils and topography in the Eastern Iowa Basins.

Climate

Water originates as rainfall in late spring through late fall and as snow during winter and early spring. Average annual precipitation in the basins ranges from about 30 inches in the northwestern part of the Study Unit to about 36 inches in the southeast. The greatest rainfall typically occurs during the growing season in spring and summer. The mean April-to-October precipitation is about 25 inches. The most intense 24-hour rainfall (5-year recurrence interval) can be more than 4 inches. Snowfall has been recorded from September to May. The greatest 24-hour snowfall seldom (less than 25 percent of the years) exceeds 10 inches. Yearly rainfall during the study period ranged from below average in 1996 to about average in 1997 and above average in 1998.

Land Use

Because water flows over the land surface or infiltrates the soil,

human activities may have a substantial effect on the quality of ground and surface water. The production of row crops, such as corn, and cover crops, such as alfalfa and small grains, constitutes the major land use in the Study Unit. Land near the streams and rivers has a combination of crops and forests. About 40 percent of the more than 1 million people in the Study Unit are concentrated in cities with populations greater than 20,000.

Water Use

Water used for household, municipal, commercial, industrial, and agriculture purposes originates primarily from ground water. Surface water, although an important supply for several larger cities including Cedar Rapids and Iowa City, is used primarily for cooling water in the generation of electric power.

Water that infiltrates through the soil into underlying sand and gravel deposits and ultimately into

the underlying bedrock formations is used as a water supply for about 94 percent of the population in the Eastern Iowa Basins Study Unit. Rivers and streams are the source of public-water supplies for about 6 percent of the population (fig. 2).

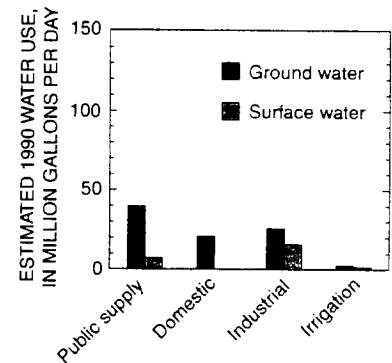


Figure 2. An abundant ground-water resource provides water to municipalities, homes, and industry. Ground water is used by more than 90 percent of the population in the Study Unit.

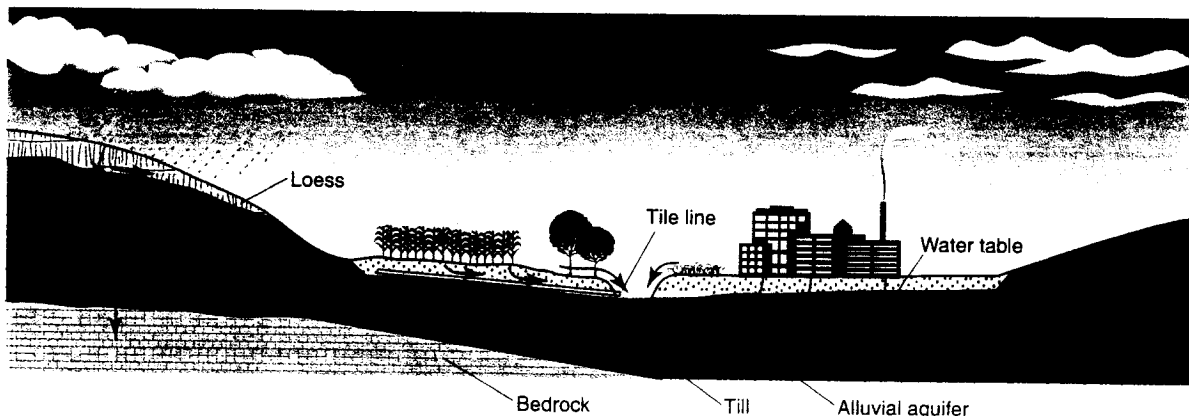


Figure 3. In the Eastern Iowa Basins, water originating from precipitation flows overland or through loess, till, and alluvial deposits to nearby streams. Areas with high water tables and poor natural drainage have commonly been artificially drained with tile lines. (Graphic created by Suzanne Roberts, U.S. Geological Survey.)

Surface Water

Excess precipitation that does not infiltrate into the soil or evaporate runs off to the streams (fig. 3). Generally, poorly permeable till soils typical of the Des Moines Lobe and steeper slopes typical of the Southern Iowa Drift Plain generate greater overland flow than the moderately well drained loess soils and gentle slopes typical of the Iowa Surface. Overland flow to streams is slowed or reduced by grass, perennials, shrubs, and trees (riparian buffers) where present near streambanks. Runoff to streams averages about 25 percent of the annual precipitation and ranges from less than 7 inches per year in the northern part of the Study Unit to about 9 inches per year in the southeastern part. Overland flow and ground-water discharge are the major sources of streamflow. However, tile lines may be an important source to streams during base flow in areas where they have been installed to

remove standing water from surface depressions and to lower the water table.

Long-term yearly discharge from the Eastern Iowa Basins Study Unit averages about 9.2 million acre-feet. The overall increase in rainfall from 1996 to 1998 was reflected in the substantial increase in yearly stream discharge (fig. 4). Yearly discharge from the Eastern Iowa Basins increased from about 8.6 million acre-feet in 1996 to almost 13.8 million acre-feet in 1998. Yearly discharge was not uniform in the major basins. Discharge from the Wapsipinicon and Iowa River Basins increased in 1997 and 1998, and the discharge from the Skunk River Basin decreased from 1996 to 1998.

Ground Water

Water from rainfall infiltrates through the soil and, depending on whether permeable sand, gravel, and fractured bedrock are present, may continue to move to the deeper aquifers (fig. 3). If low-permeabil-

ity clay or shale lies below the unconsolidated surficial materials, water may move laterally to a nearby stream.

Alluvial material that has been deposited by rivers and streams commonly consists of sand and gravel layers that store and transmit water readily. The alluvial aquifers (fig. 5) are the most frequently used source of ground water in the Eastern Iowa Basins because they are near land surface and can supply large amounts of water. The same properties (shallow depth and permeable material) that make alluvial aquifers excellent sources of water also make the alluvial aquifers susceptible to contamination from surface activities.

Two additional surficial deposits, not assessed during this study, provide water for domestic and municipal supplies. Sand and gravel deposits in low-permeability glacial till generally yield small quantities of water that are used mostly for rural domestic and stock supplies. Also, deep sand and

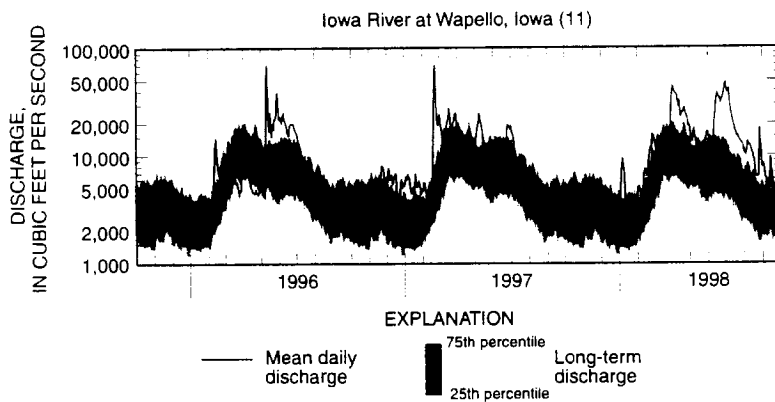


Figure 4. Stream discharge in the Eastern Iowa Basins ranged from below normal in 1996 to near normal in 1997 and above normal in 1998 (site number in parentheses; see site map in "Study Unit Design" section).

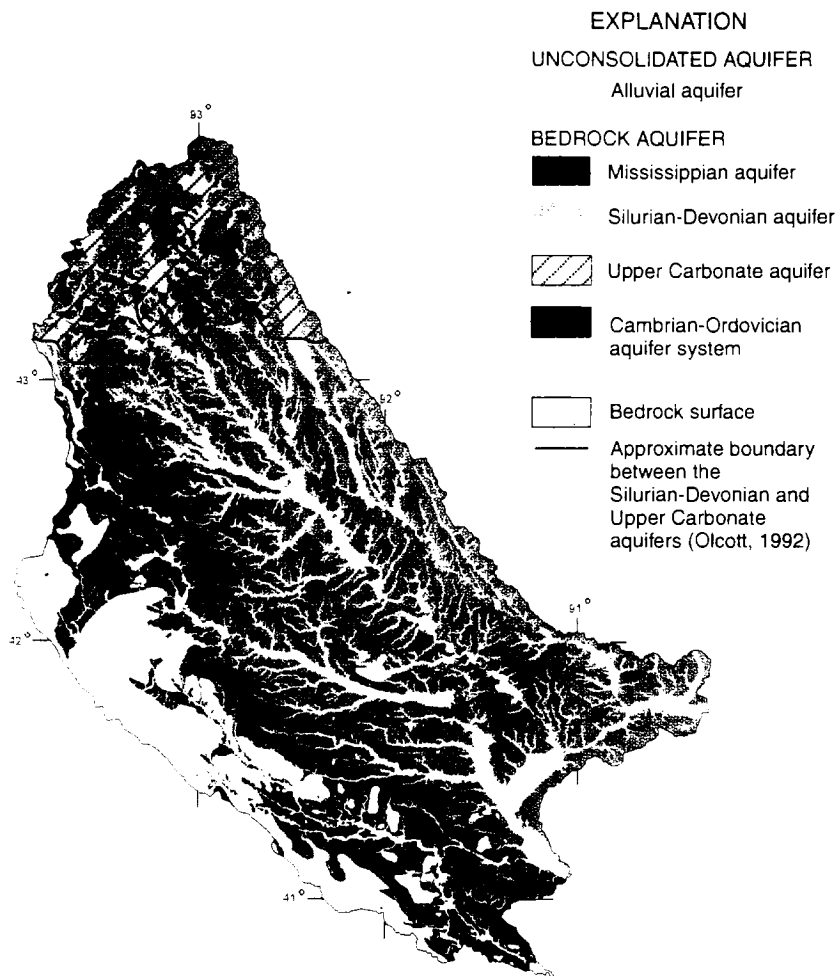


Figure 5. Several major bedrock and unconsolidated aquifers are used as sources of water for domestic, municipal, and industrial supplies. Only the most heavily used aquifers, the Silurian-Devonian and Upper Carbonate aquifers and the alluvial aquifers, were sampled.

gravel deposited in bedrock valleys before the last glacial advance is an important source of water in parts of the Eastern Iowa Basins.

Rock formations (bedrock) generally underlie the clay, silt, sand, and gravel surficial materials and can provide water for use. Bedrock aquifers are generally deep and are protected from surficial contamination. However, in areas such as the

Iowan Karst, bedrock is exposed or is covered by very thin unconsolidated deposits and is susceptible to contamination from urban and agricultural land uses.

The most extensively used bedrock aquifers are the Silurian-Devonian and Upper Carbonate aquifers. The Silurian-Devonian and Upper Carbonate aquifers consist mainly of limestone and

dolomite with locally interbedded shale and evaporite beds. Bedrock aquifers that are used as a source of water in the Study Unit but were not evaluated during this study are the Mississippian aquifer and the Cambrian-Ordovician aquifer system.

MAJOR FINDINGS

Nutrients in Ground Water and Streams

Two naturally occurring nutrients, nitrogen and phosphorus, commonly are applied in the form of fertilizers and manure in agricultural areas to increase the yield of crops and as fertilizer in cities and towns to enhance the appearance of residential lawns, city parks, and golf courses. Nitrogen and phosphorus also are commonly discharged from wastewater-treatment facilities. Nutrients that are not used by plants or attached to soil particles can move to shallow ground water or can be washed into nearby streams during intense rains. Runoff from rainfall and ground-water inflow can transport excess nutrients to streams, causing algal blooms that deplete oxygen for fish and other aquatic organisms. High concentrations of nitrogen in the form of nitrate may make untreated water unsuitable for human consumption. The USEPA has established a Maximum Contaminant Level (MCL) of 10 mg/L (milligrams per liter) for nitrate as nitrogen in drinking water (U.S. Environmental Protection Agency, 1996). High concentrations of nitrogen in the form of ammonia may kill fish and other aquatic organisms. To minimize algal growth in streams and reservoirs, a total phosphorus concentration of 0.10 mg/L or less has been recommended by the USEPA (1986).

Nutrients in Ground Water

Nitrogen and phosphorus are prevalent in ground water. Nitrogen readily moves from the land surface to ground water in the shallow alluvial aquifers but not as readily to most parts of the deeper Silurian-Devonian and Upper Carbonate bedrock aquifers in the

Eastern Iowa Basins. Nitrogen, in the form of nitrate and ammonia, and dissolved phosphorus were detected in more than 65 percent of 124 ground-water samples. Nitrate concentrations were significantly higher in the shallow alluvial aquifers, which have been identified as being susceptible to contamination (Hoyer and Hallberg, 1991), than in the Silurian-Devonian and Upper Carbonate bedrock aquifers (fig. 6). Nitrate concentrations in water samples from monitoring wells screened near the water table in urban areas were higher than concentrations in samples from domestic wells, which generally are screened deeper in the aquifers.

Human and natural factors affect movement of nutrients to ground water. Downward movement of water containing nitrogen and phosphorus is slowed by fine-grained materials overlying the aquifers. The presence of clay in the soil and in the shallow subsurface and low-permeability rocks, such as shale, can slow ground-

water infiltration rates and limit the amount of nutrients at depth. Surficial contamination most affects water near the top of the alluvial aquifers where the clay layer above the water table is thin or nonexistent (see fig. 8, p. 8).

In the Silurian-Devonian and Upper Carbonate aquifers, ground-water ages (determined by analysis of tritium concentrations) were significantly younger and nitrate concentrations were significantly higher in samples from areas where an overlying low-permeability shale layer was absent or where less than 100 feet of unconsolidated deposits overlies the aquifer (Savoca and others, 1999). These two results indicate that longer flow paths (deeper sample depth and thicker clay layer) increase opportunities for sorption, degradation, and dispersion and may contribute to decreases in nutrient concentrations with depth. Alternatively, water deeper in the aquifer may have infiltrated in years when fertilizer use was not as prevalent.

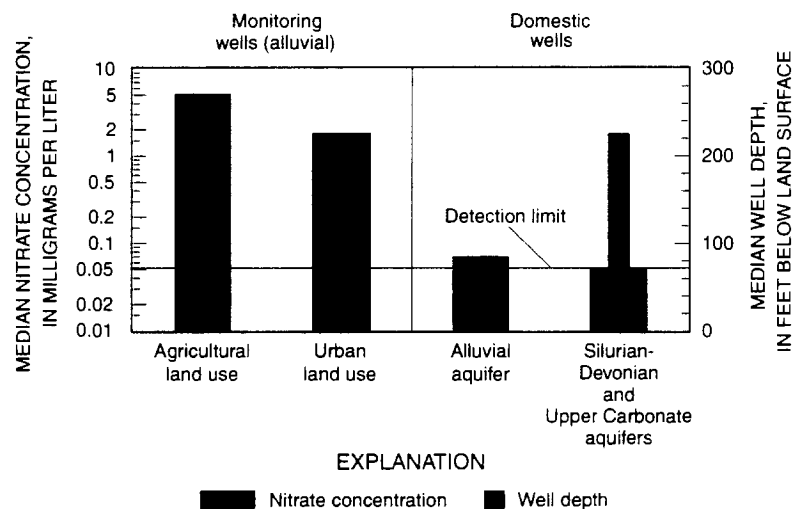


Figure 6. The shallowest ground water is most heavily affected by current land-use practices. Agricultural practices result in higher levels of nitrate in ground water than urban activities.

NUTRIENT CONCENTRATIONS IN STREAMS WERE AMONG THE HIGHEST IN THE NATION



Concentrations of nitrate and total phosphorus in streams in Eastern Iowa Basins rank among the highest in the Nation (fig. 7). The median discharge-weighted nitrate concentrations also were significantly greater in Eastern Iowa Basins streams than in streams sampled elsewhere in the Corn Belt and Northern Great Plains ecoregion (Omernik, 2000)—an area of similar climate, topography, regional geology and soils, and broad land-use patterns. Overall, total nitrogen concentrations were similar to the streams sampled in Illinois and to an agricultural stream sampled in southern Minnesota. Nitrate concentrations were in the upper 25th percentile nationally for 9 of 11 Basic Fixed Sites. The defined reference site (Wapsipinicon River near Tripoli) and the site on the stream draining the Iowan Karst landform (Flood Creek near Powersville) were within the middle 50 percent nationally. Concentrations of total phosphorus ranged from moderate to high in relation to other streams sampled in the United States. Average discharge-weighted total phosphorus concentrations at two sites (Old Mans Creek near Iowa City and Skunk River at Augusta) in the southern part of the Eastern Iowa Basins, where highly erodible loess soils cover most of the basins, were in the upper fifth percentile nationally.

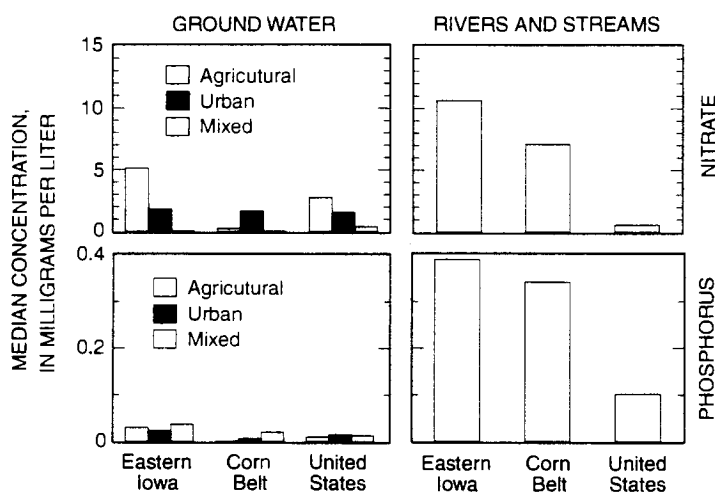
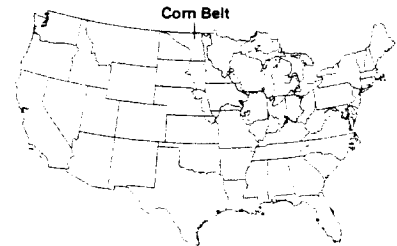


Figure 7. Nitrate and phosphorus concentrations in Eastern Iowa Basins streams and ground water are higher when compared with a wide variety of land uses across the Nation and when compared to similar land use in other parts of the Corn Belt. (Phosphorus in ground water as dissolved orthophosphorus and phosphorus in streams as total phosphorus)

Although substantially lower than in surface water, nitrate concentrations in the alluvial aquifers in agricultural areas in Eastern Iowa Basins are greater than those in ground water sampled elsewhere in the Corn Belt and the Nation. The alluvial aquifer is a relatively susceptible aquifer in contrast to aquifers within glacial till sampled in Illinois; therefore, the ranking of concentrations may not be directly comparable. However, nitrate concentrations in urban areas and in aquifers that serve as a source of water supply for municipal and domestic use in the Eastern Iowa Basins are comparable to those from the rest of the Nation. Dissolved phosphorus concentration in the alluvial aquifers in agricultural areas of Eastern Iowa Basins was greater than in ground water in the rest of the Nation. Increased susceptibility and greater fertilizer use may account for the higher phosphorus concentrations.

Natural processes may remove nitrogen from alluvial aquifers. The relation between nitrogen, dissolved oxygen, and organic carbon concentrations suggests that nitrogen is being biologically trans-

formed and removed from the alluvial aquifers. Nitrate remaining after water moves through or around clay layers may be converted to nitrogen gas in a low-oxygen or oxygen-free environ-

ment in the presence of organic carbon. This process is called denitrification. Nitrate concentrations were significantly higher in alluvial ground water having dissolved oxygen concentrations

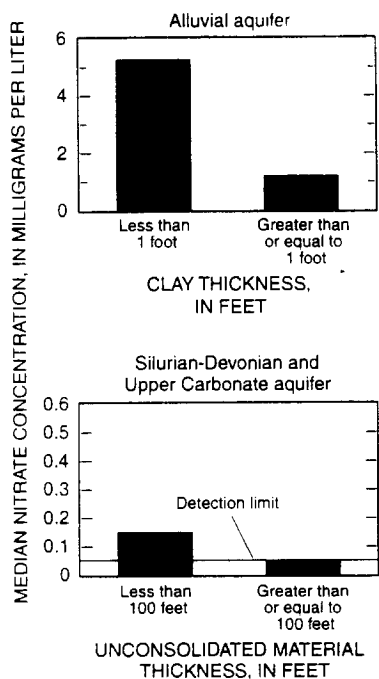


Figure 8. Overlying clay and other low-permeability material hinder movement of nitrate to ground water.

greater than 0.5 mg/L and were significantly lower in ground-water samples having increased dissolved organic carbon concentrations (Savoca and others, 2000). Denitrification may be an important natural process that reduces nitrate concentrations before water moves to ground-water supplies or is discharged to streams.

The type of land use affects shallow ground-water quality.

Different uses, whether the land is used for agriculture, homes, business, or industry, are reflected in the nutrient concentrations in the ground water. Animal feeding operations can further affect water quality in areas with intensive row-crop agriculture (fig. 9). Nitrate was detected more frequently (94 percent of samples) and in greater concentrations (median of 5.1 mg/L) in shallow alluvial

aquifers in agricultural areas than in urban areas. Nitrate was detected in 77 percent of samples at a median concentration of 1.8 mg/L from urban areas (fig. 6). Nitrate exceeded the USEPA MCL (10 mg/L as N) in 39 percent of samples from agricultural areas and in none of the samples from urban areas. Dissolved phosphorus concentrations tended to be higher in samples from agricultural areas (median of 0.03 mg/L) than urban areas (median of 0.01 mg/L); however, the difference was not statistically significant. The higher reliance on fertilizers in agricultural areas than in urban areas most likely contributed to the higher nitrate and phosphorus concentrations.

Ammonia is prevalent in shallow urban ground water.

Dissolved ammonia concentrations were significantly higher in samples from urban areas than in samples from agricultural areas (fig. 9). The median dissolved ammonia concentration in samples from shallow urban ground water (0.025 mg/L) was more than twice the median concentration in samples from agricultural areas (0.010 mg/L). Anhydrous ammonia is used frequently as a nitrogen fertilizer for corn, but soil microbes quickly convert ammonia to nitrate.

Conditions in the alluvial aquifers (fig. 9) indicate that ammonia in shallow urban ground water originates, at least partly, from degradation of organic matter most likely derived from human activities. Substantial amounts of organic matter are available in the soil, silt, and clay above the aquifer and dissolved in water within the aquifer. Dissolved oxygen concentrations were significantly lower in urban

areas than in agricultural areas. Microbial degradation of abundant organic matter would result in the decrease in dissolved oxygen concentrations and in the increase of dissolved ammonia concentrations in shallow ground water.

Nutrients move from ground water to streams by natural drainage and tile lines.

At times, ground water contributes a substantial amount of nitrogen to streams in the Eastern Iowa Basins Study Unit. In most parts of the Study

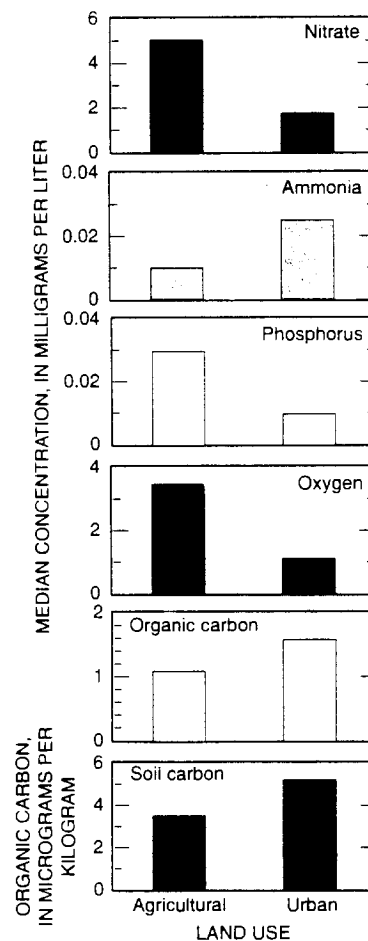


Figure 9. Agricultural and urban land uses contribute nitrogen to shallow alluvial aquifers. Conditions are suitable in urban areas for the formation of ammonia in the organic-enriched aquifers.

Unit, water naturally flows from alluvial aquifers to streams through the streambanks and streambeds. During high base flow in May 1998, nitrate concentrations in samples from 24 of 25 streams in the Eastern Iowa Basins were higher than 10 mg/L. In contrast, nitrate concentrations in these streams ranged from less than 0.05 to 8.3 mg/L during low base flow in August 1997. The high concentrations detected during the spring may be due to the increased ground-water inflow after fertilizer was applied and was readily available for transport.

Shallow tile lines have been installed to remove excess water from the land and to lower the water table in many parts of the Study Unit. These tile lines typically drain water from the upper part of the water table, which generally contains the highest nitrate concentrations to nearby streams. During the summer, the water table may decline below the tile lines due to decreased rainfall and increased evapotranspiration. Tile-line flow and nutrient transport then cease. For example, nitrate concentrations were consistently higher than 10 mg/L in tile-line discharge from April until the end of August when flow in this and other local tile lines ceased (fig. 10). After tile flow ceased, nitrate concentrations in the Iowa River decreased.

When tile flow ceases, streamflow originates from natural ground-water inflow, which traverses deeper and longer flow paths. Water that originates from deep in the alluvial aquifers contains low nitrate concentrations (see previous discussion) due to transformation in the aquifer or

streambed and plant uptake as it passes beneath riparian buffers. These results are similar to those of previous studies (Cambardella and others, 1999; Soenksen, 1996) that documented the importance of tile-line discharge on stream-water quality.

Increased algal growth in late summer and early fall also can contribute to decreased nitrate concentrations in streams (Porter, 2000). Tile-line drainage is an important hydrologic factor that may serve to protect shallow ground water by removing contaminants before they move down into the aquifers; however, tile lines also may enhance the contamination of streams by short circuiting natural processes that remove nitrogen from ground water.

Nutrients in Streams

The concentrations of nutrients in a stream or river are the result of the interaction of human activities and natural factors in the basin. Human activities, agricultural and urban, generally increase the input

of nitrogen and phosphorus into the basin, alter the land surface and drainage patterns (which may affect the amount and timing of rainfall runoff), and alter vegetation on the land and habitat in streams and rivers, resulting in a more rapid flushing of nutrients from the land and downstream in streams and rivers. Natural factors including the amount and timing of rainfall, soil types, and land-surface slope affect the amount of nitrogen and phosphorus washed off fields into nearby streams.

Nitrogen and phosphorus are almost always present in streams and rivers. Nitrate, the most common form of nitrogen in streams and rivers in the Eastern Iowa Basins, was present in more than 98 percent of the samples at concentrations of 0.05 mg/L or greater and was most frequently (80 percent of the samples) present in the range of 1 to 10 mg/L (fig. 11). Nitrate concentrations less than about 0.5 mg/L were generally associated with low streamflow in late summer when nitrogen inputs

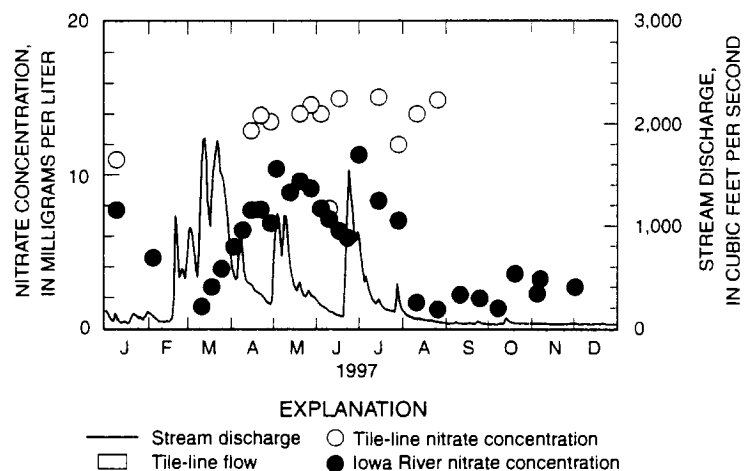


Figure 10. Tile lines can contribute substantial amounts of nitrate to streams and rivers. Nitrate concentrations in the Iowa River near Rowan decreased as tile-line discharge ceased.

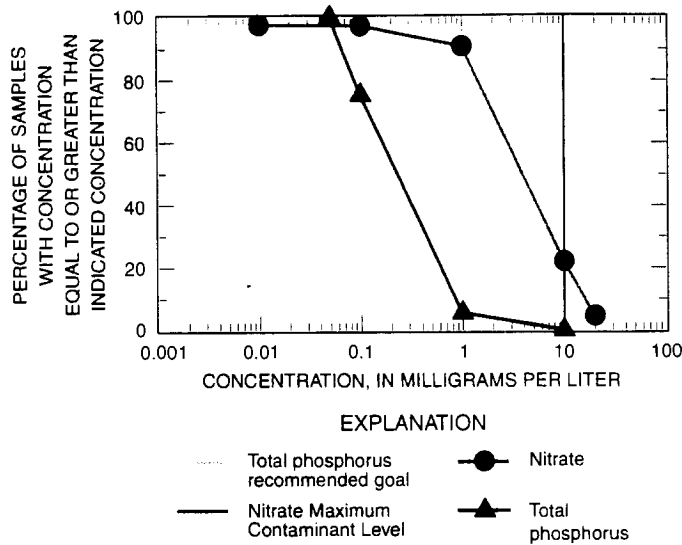


Figure 11. Nitrate concentrations in streams were most often in the range of 1 to 10 mg/L and equaled or exceeded the Maximum Contaminant Level in 22 percent of the samples. In contrast, total phosphorus concentrations equaled or exceeded the 0.1-mg/L goal for minimization of algal growth in 75 percent of the samples.

were low and algal uptake was high. About 22 percent of the samples (fig. 12) contained nitrate concentrations at or above the USEPA's drinking-water standard of 10 mg/L (U.S. Environmental Protection Agency, 1996).

Phosphorus, dissolved in the stream water and attached to sediment particles or organic compounds transported in the stream, was present in all samples. Total phosphorus concentrations exceeded the USEPA goal of 0.1 mg/L (U.S. Environmental Protection Agency, 1986) or less to minimize plant and algal growth about 75 percent of the time. Streams with large total phosphorus concentrations generally contained large suspended-sediment concentrations.

Landform and land use affect nitrogen and phosphorus concentrations. Nitrogen and phosphorus concentrations in streams differed

among landform areas and with land use (fig. 12). Basins of the size where human activities are minimal (background) are not present in eastern Iowa and southern Minnesota. The Wapsipicon River near Tripoli was identified as a reference site, however, because although agricultural land use constitutes more than 90 percent of the basin, substantial areas of riparian forests and wetlands remain. Samples from the reference site generally had the lowest median nutrient concentrations of any stream site studied. Median suspended-sediment concentrations also were the lowest. Biological indicators of water quality at the reference site were better than all other sites in the Study Unit. Extensive floodplain vegetation may decrease direct transport of nutrients from fields to the Wapsipicon River, and extensively wooded banks on

the nonchannelized river may decrease streambank erosion and enhance biological habitat.

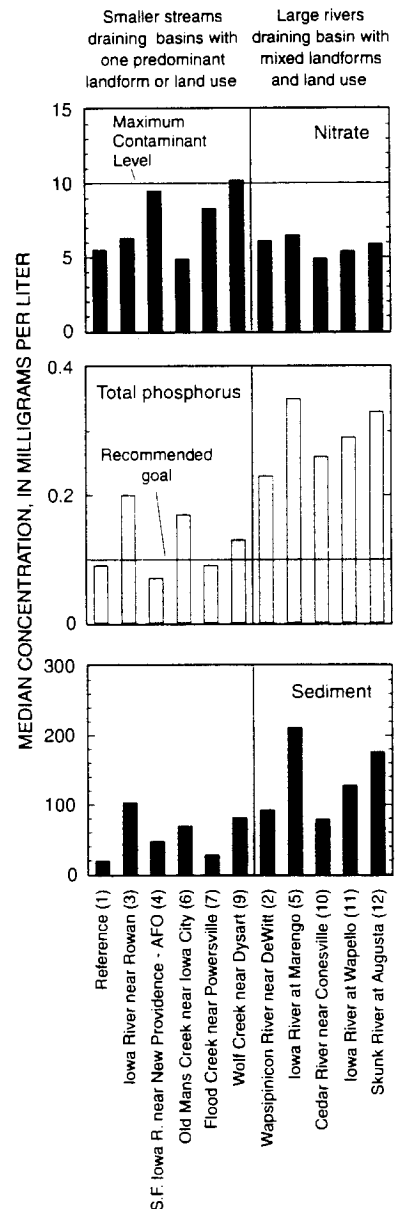


Figure 12. Nitrate concentrations generally were lower in large rivers than in smaller streams. In contrast, total phosphorus concentrations were greater in large rivers than in streams. (Site number in parentheses; see site map in Study Unit Design section. AFO is animal feeding operation.)

Median nitrate concentrations were highest in samples from Wolf Creek, a stream draining an Iowan Surface basin that has more than 80-percent row-crop agriculture, and in samples from the South Fork Iowa River, a stream draining a Des Moines Lobe basin that has concentrated animal-feeding operations (AFO). The median nitrate concentration in samples from Wolf Creek (10.2 mg/L) exceeded the 10-mg/L USEPA MCL. Algal status and biomass in Wolf Creek indicated degraded conditions (from a national perspective), and fish and invertebrate communities were dominated by species tolerant to nutrient and organic enrichment. Wolf Creek flows into the upper end of a reach of the Cedar River that has been listed on the 1998 USEPA Section 303(d) list (Iowa Department of Natural Resources, 2000) as an impaired water body. Wolf Creek may provide substantial amounts of nitrogen that contribute to the degradation of the Cedar River between Waterloo and Cedar Rapids.

Landform features affect phosphorus concentrations. Phosphorus and suspended-sediment concentrations were typically larger in streams that drain the Southern Iowa Drift Plain and the Des Moines Lobe than in other streams. The Southern Iowa Drift Plain typically has steeper slopes than the rest of the Study Unit and contains loess deposits that are easily eroded (Schwarz and Alexander, 1995). Many Des Moines Lobe streams have been extensively channelized; to a great extent, the streambanks and flood plains contain little riparian vegetation, resulting in a greater dominance of phytoplankton than benthic algae and relatively higher rates of stream metabolism that can

influence dissolved-oxygen conditions during base-flow conditions (Porter, 2000).

Intensive row-crop agriculture contributes to greater nitrate concentration. The intensity of row-crop agriculture is partly responsible for the variability of nitrate concentrations in streams. Typically, streams in basins that have a higher percentage of corn and soybeans and less pasture, forest, and CRP (Conservation Reserve Program) acres had higher total nitrogen concentrations (fig. 13). However, large rivers in basins that had a slightly lower percentage of row crops and a slightly larger proportion of urban areas typically had higher total phosphorus concentrations. Large rivers generally contained higher suspended-sediment concentrations than streams, and most large potential point-source phosphorus contributors (large cities) are located on the large rivers.

Increased availability results in greater nutrient concentrations during the spring and early summer. Concentrations of nitrogen and phosphorus varied seasonally in streams in eastern Iowa (fig. 14). Median total dissolved

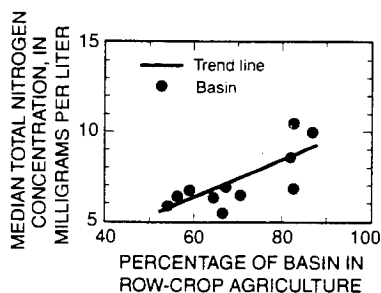


Figure 13. Streams draining basins that have higher percentages of pasture, grassland, and forests and less land planted in row crops tended to have lower nitrogen concentrations.

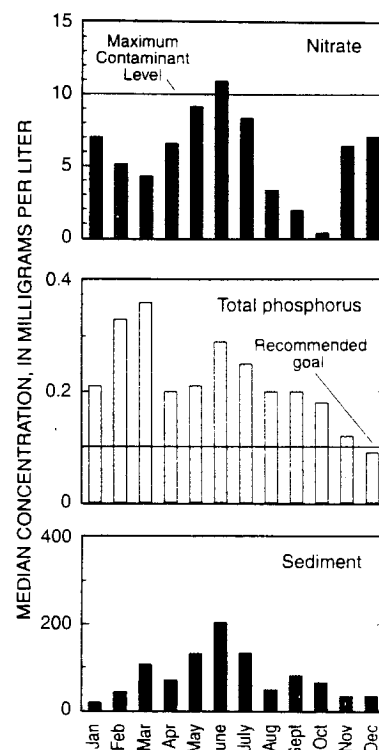


Figure 14. Nitrate concentrations that commonly increase to levels that exceed the Maximum Contaminant Level in June can decrease to levels below detection in October. Total phosphorus concentrations were rarely below the recommended goal for reduction of algal growth.

nitrogen concentrations were highest in June (11.5 mg/L) when nitrogen is transported to streams and rivers by spring and early summer runoff from rainfall. Nitrogen probably originates from that accumulated in the soil following application of chemical fertilizers and manure during the spring and previous fall. Nitrate was the predominant form of nitrogen detected during this period. Total dissolved nitrogen concentrations in streams decreased through the summer as nitrogen was removed from the soil by plant uptake, runoff, and leaching to shallow ground

ANIMAL FEEDING OPERATIONS CONTRIBUTE ADDITIONAL NUTRIENTS TO STREAMS

The median nitrate concentration in samples from the South Fork of the Iowa River (9.5 mg/L) were significantly higher than those in samples from the Iowa River near Rowan (median of 6.3 mg/L), though both streams drain the same landform (Des Moines Lobe) and have similar crop patterns (greater than 80 percent row crops). In addition to row-crop agriculture, the basin of the South Fork Iowa River contains substantially more permitted animal feeding operations (AFO) (29) than the Iowa River Basin upstream from Rowan (8). The density of hogs (fig. 15) in the South Fork basin is more than twice that in the Iowa River Basin (Sorenson and others, 1999). The manure generated from AFOs is commonly applied on fields in substitution for chemical fertilizer. However, in areas where there is dense concentration of AFOs, sufficient land may not always be available to economically dispose of animal wastes, and the potential for overapplication exists. Excess nutrients seep into the ground water and are washed into nearby streams. About 1.8 times more nitrogen and about 2.5 times more phosphorus were transported by the South Fork Iowa River than by the Iowa River (fig. 15). The higher nitrate concentrations and the greater nitrogen and phosphorus yields from the South Fork Iowa River compared to the Iowa River near Rowan may indicate that a reduction in chemical fertilizer application equivalent to the increased manure application has not occurred in the South Fork Basin.

water. Algal and plant growth in the streams during low streamflow in late summer and early fall can result in increased uptake of dissolved nutrients (Porter, 2000). The combination of reduced source loading and instream processing

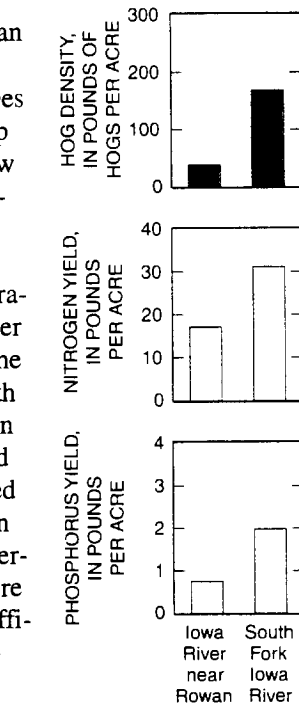


Figure 15. Large-scale hog production adds to the nitrogen and phosphorus in streams draining basins in the Des Moines Lobe landform. [Hog density data calculated from Iowa Department of Natural Resources (1999) waste-control facility permits.]

results in lowest nitrate concentrations in October. Fall rains, in combination with fertilizer and manure applications, were associated with another rise in nitrogen concentrations beginning in November and peaking in January.

Although nitrate remained the primary form of nitrogen, ammonia and organic nitrogen were more prevalent during the winter.

Runoff from agricultural and urban areas transports substantial amounts of phosphorus to streams, but the timing of peak phosphorus concentrations does not always correspond to peak nitrate concentrations. Phosphorus concentrations peak during periods of high runoff in early spring when substantial amounts of soil are eroded into the streams. The highest median total phosphorus concentrations occurred in February (0.33 mg/L) and March (0.36 mg/L) with a secondary peak in June (0.29 mg/L) (fig. 14). Maximum concentrations corresponded with early seasonal runoff from snowmelt or rainfall. Early summer rains produced runoff that accounted for a secondary peak in total phosphorus concentrations during June.

Transport of nutrients represent an economic loss and a potential environmental threat. The large amounts of nitrogen and phosphorus that are transported from the Study Unit by the Wapsipinicon, Cedar, Iowa, and Skunk Rivers represent an economic loss to farmers and an environmental concern for downstream water.

Nitrogen and phosphorus transported to the Mississippi River increased yearly from 1996 through 1998 (fig. 16). Increased streamflow was a major factor in the increased loads. The estimated mass of nitrogen (load) increased from about 106,000 tons in 1996 to more than 257,000 tons in 1998, and the estimated total phosphorus load increased from 7,500 tons in 1996 to 9,700 tons in 1998.

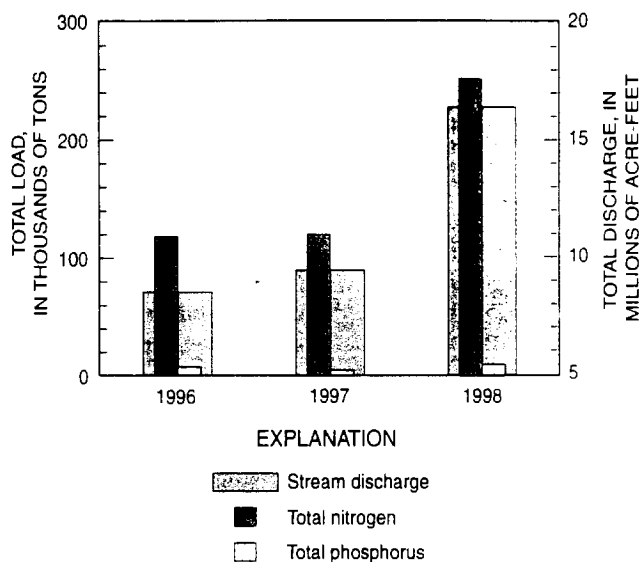


Figure 16. Increased streamflow from 1996 to 1998 resulted in larger amounts of nitrogen and phosphorus transported from the Study Unit to the Mississippi River.

Transport of nutrients from the Eastern Iowa Basins represents a potential loss in crop yield or the cost of additional fertilizer needed to compensate for that flushed from the fields. Nutrient loads transported from the Eastern Iowa Basins represent an average loss of 17 lb/acre/year of nitrogen and 1.2 lb/acre/year of phosphorus in 1996 and 42 lb/acre/year of nitrogen and 1.6 lb/acre/year of phosphorus in 1998.


Iowa, including the Eastern Iowa Basins, has been identified as a major source of nutrients (Goolsby and others, 1999) that contribute to eutrophication and hypoxia in the Gulf of Mexico (Rabalais, 1996). Alexander and others (2000) estimated that more than 90 percent of the nitrogen reaching the Mississippi River from the Eastern Iowa Basins is transported to the Gulf of Mexico.

Pesticides in Ground Water and Streams

Pesticides (herbicides and insecticides) are used extensively in agricultural and urban settings in the Eastern Iowa Basins to control unwanted vegetation and insects. Triazine (atrazine and cyanazine) and chloroacetanilide (alachlor and metolachlor) herbicides were the most extensively used pesticides during 1996–98. However, in 1994, acetochlor, a herbicide conditionally registered by the USEPA, began to replace alachlor. In 1998, acetochlor use exceeded all other herbicides in Iowa (U.S. Department of Agriculture, 1999). Other classes of low-use herbicides (for example, sulfonylurea and imidazolinone herbicides) also had become more popular. Genetically altered corn and soybeans that are resistant to glyphosate and genetically

altered “Bt corn” that contains the genes allowing corn to produce compounds toxic to damaging insects also were beginning to change pesticide-use patterns. Although urban pesticide-use data were not available, herbicides commonly are used on lawns, golf courses, and road rights-of-way for weed control, and a wide range of insecticides are used for insect control.

Transport of pesticides from the site of application is dependent on the persistence of the compound, its solubility in water, and its tendency to adsorb to soil particles. Pesticide compounds break down at various rates due to biological and chemical processes. Pesticides can be broken down in the soil by bacteria and fungi and nonaffected plants and in streams by microorganisms, algae, and aquatic plants. The older organochlorine pesticides, most of which have been banned (for example, DDT) strongly attach to soil particles and are transported to streams primarily with sediment. In contrast, many herbicides commonly used in the 1990’s (for example, atrazine) are relatively soluble and are transported almost exclusively dissolved in water. Water is the primary mechanism by which most pesticides and their breakdown products (degradates) are leached to ground water. Water transports pesticides and their degradates to streams by overland flow, tile drains, and ground-water inflow.



PESTICIDE CONCENTRATIONS RANK HIGHER IN GROUND WATER THAN IN STREAMS AND RIVERS

Herbicide and insecticide concentrations in streams do not rank among the highest 25th percentile nationally (fig. 17) even though use in the Eastern Iowa Basins is high. Herbicide concentrations rank in the middle 50 percent nationally and in the middle one-third in the Corn Belt. Insecticide concentrations are among the lowest in the Corn Belt and Nation. Reasons for lower pesticide concentrations in the Eastern Iowa Basins compared to other parts of the Nation are not entirely clear but may include these: (1) although pesticide use is high in Eastern Iowa Basins it may be higher in other corn, cotton, and other crop-growing areas of the Nation and (2) soils in Eastern Iowa Basins may be relatively conducive to leaching to ground water or conditions are favorable for breakdown of the pesticides.

In contrast, herbicide and insecticide concentrations in the alluvial aquifers in Eastern Iowa Basins are among the highest 25th percentile in the Corn Belt and Nation (fig. 17). Water in these aquifers has infiltrated within the last 20 years when the use of the analyzed pesticides was the greatest. Because water from the Silurian-Devonian and Upper Carbonate aquifers is generally deeper and has infiltrated before pesticides were commonly used, pesticide concentrations from these aquifers generally were among the lowest in the Nation.

Pesticides in Ground Water

Herbicides are prevalent in the shallow alluvial aquifers. Herbicides are prevalent in the shallow alluvial aquifers but are not common in the deeper Silurian-Devonian and Upper Carbonate bedrock aquifers in the Eastern Iowa Basins. Atrazine and metolachlor were the most frequently detected herbicides in the alluvial aquifers and the Silurian-Devonian and Upper Carbonate aquifers. Atrazine was detected in more than 50 percent of the alluvial aquifer samples

and in 18 percent of the Silurian-Devonian and Upper Carbonate aquifer samples. Metolachlor was detected in 20 percent of the alluvial samples and in 12 percent of the Silurian-Devonian and Upper Carbonate samples. Acetochlor was detected in slightly less than 2 percent of the alluvial aquifer samples but at concentrations less than 0.2 µg/L, which is the concentration of concern for conditional registration. Acetochlor was not detected in samples from the Silurian-Devonian and Upper Carbonate aquifers.

Pesticide degradates commonly constitute the majority of the pesticide mass analyzed. Many of the detected pesticides and pesticide degradates have no established drinking-water standard or aquatic-life criteria, and the potential for these compounds to affect humans or aquatic organisms is unknown. Alachlor ethanesulfonic acid (ESA), atrazine, and metolachlor ESA were detected in more than 30 percent of samples from shallow alluvial (fig. 18) and deep bedrock aquifers.

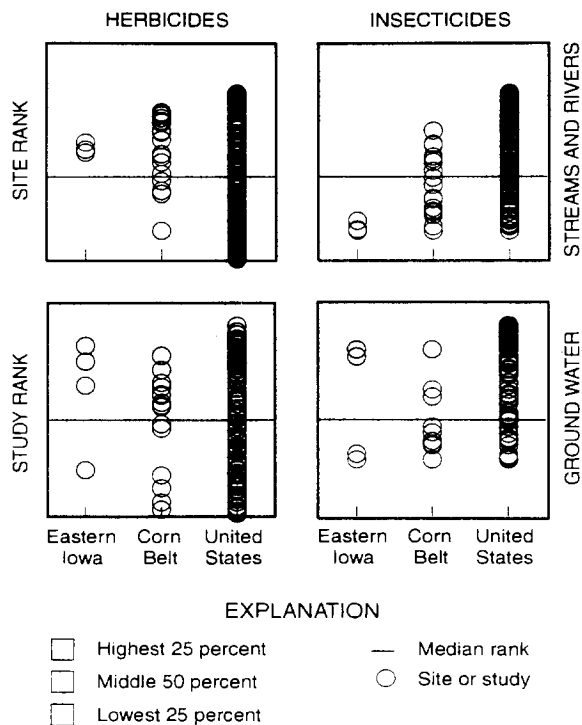


Figure 17. Although pesticides are heavily used, concentrations in the Eastern Iowa Basins streams are not among the highest in the Nation or the Corn Belt. The vulnerability of the alluvial aquifers to contamination is seen in pesticide concentrations in ground-water samples that are among the highest in the Nation.

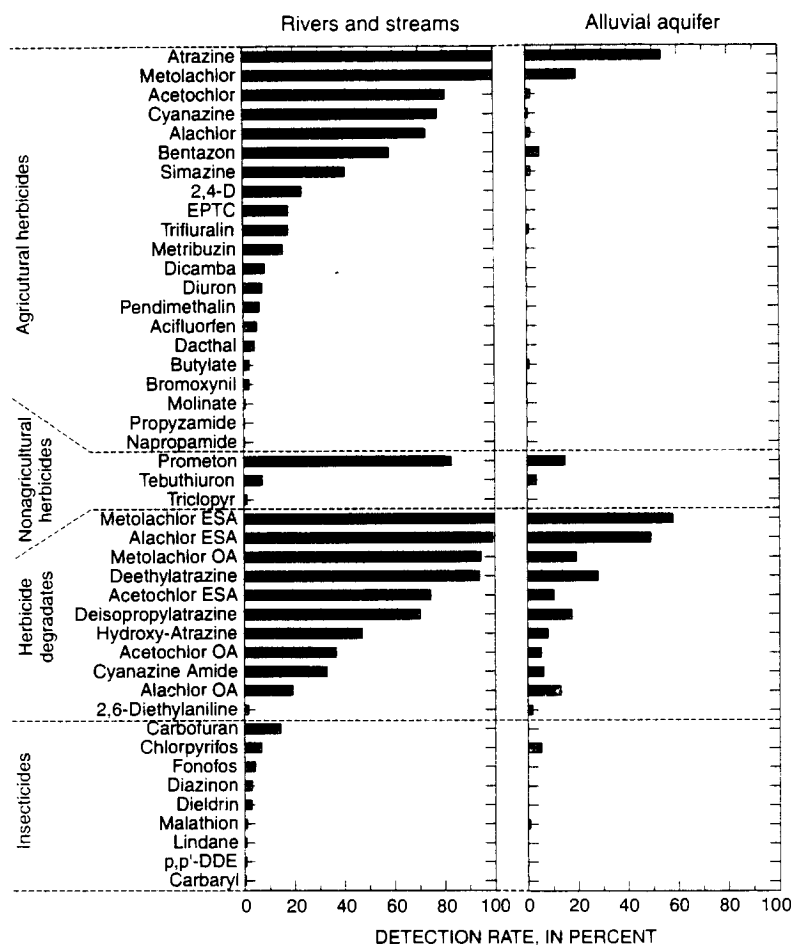


Figure 18. The pesticide compounds present in streams and rivers and the alluvial aquifers, which are hydraulically connected, were similar but were detected less frequently in ground water. Agricultural herbicides and their degradates were most frequently detected.

Agricultural and urban applications have resulted in sporadic low levels of insecticide contamination in the alluvial aquifers. Chlorpyrifos, an insecticide that was recently reevaluated by the USEPA for safety as part of the Food Quality Protection Act (U.S. Environmental Protection Agency, 2000b), was detected in about 7 percent of the urban land-use samples and about 13 percent of the agricultural land-use samples. The maximum

concentration in urban areas (0.005 $\mu\text{g/L}$) was about four times lower than the maximum concentration in agricultural areas (0.021 $\mu\text{g/L}$). Malathion, an insecticide also currently (2000) under USEPA review, was detected in one sample from an urban land-use well.

Pesticides do not occur everywhere in the Study Unit, and when they do occur, concentrations can be highly variable. This variability in occurrence and concentration of

various pesticide compounds in ground water is probably associated with several factors including the presence of overlying protective layers and land-use practices.

Clay and shale materials hinder movement of pesticides to aquifers. Clay and shale materials hinder movement of pesticides by slowing the movement of water containing pesticides to underlying aquifers. In some instances, water that has infiltrated the ground during the last 40 years when many of the studied compounds were used has not yet reached the deeper parts of the alluvial and bedrock aquifers. The age of the ground water was significantly younger and pesticide concentrations were significantly higher in samples from areas where an overlying bedrock confining unit was absent or where less than 100 feet of unconsolidated deposits overlies the Silurian-Devonian and Upper Carbonate aquifers. Water is most affected by surficial contamination near the top of alluvial aquifers where an overlying clay layer is thin or absent.

Triazine pesticides and degradate concentrations were significantly higher in samples from wells in areas with a thin clay layer above the screened interval than those from wells in areas with thick overlying clay layers. Concentrations decreased with increasing depth in the alluvial aquifers. As with nitrate, these two results indicate that longer and slower flow paths (deeper sample and thicker clay layer) increase opportunities for sorption, degradation, and dispersion of pesticides and may contribute to decreases in pesticide concentrations with depth.

WHAT IS A PESTICIDE DEGRADATE?

Pesticides released into the environment break down into intermediate compounds and, over time, into their constituent molecules (fig. 19). The intermediate compounds are pesticide degradates that may be short-lived or persist for years. The occurrence of atrazine degradates in ground water and surface water is relatively well known, but the occurrence of alachlor, acetochlor, and metolachlor degradates in Iowa have only recently been investigated (Kalkhoff and others, 1998; Kolpin and others, 1997). Little is known about their possible effects on human health and aquatic life (Heydens and others, 2000; Stamper and Tuovinen, 1998). The occurrence and distribution of several common degradates of atrazine, alachlor, acetochlor, cyanazine, and metolachlor were documented in the Eastern Iowa Basins to develop a better understanding of the fate and transport of these compounds and their possible effect on the environment. These compounds generally are the products of the first step in the breakdown of the most commonly used herbicides. Breakdown occurs in the soil but also can take place in plants and in ground water and streams.

Modification of a side carbon chain (atrazine degradation) or the replacement of the halogen (chlorine) with a sulfonic acid or carboxylic acid (acetochlor, alachlor, metolachlor degradation) is commonly the first step in the breakdown of these herbicides. This first step represents a small molecular change that may affect the toxicity, solubility, and persistence of the resulting degradate.

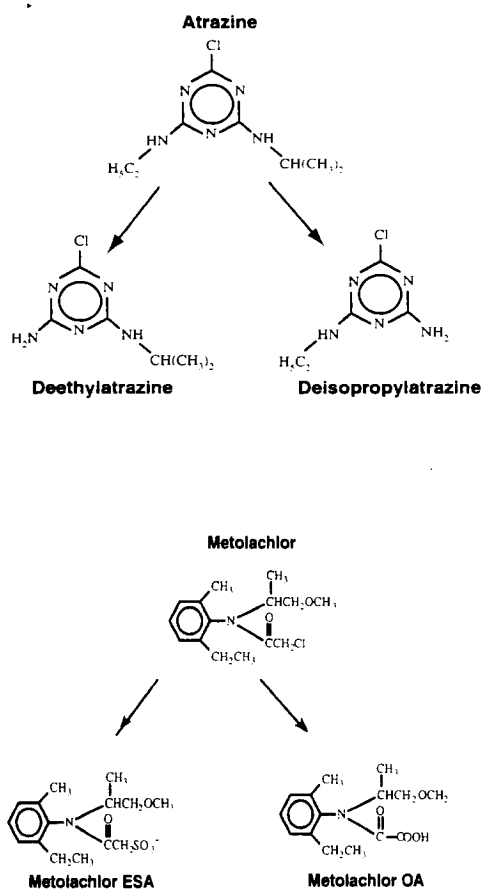


Figure 19. Small molecular changes (shown in red) occur during the initial breakdown of atrazine and metolachlor.

Pesticides are present in alluvial aquifers both in urban and agricultural areas. Pesticides were prevalent in the shallow alluvial aquifers both in agricultural and urban areas (fig. 20). Pesticides were detected in 84 percent of the samples from agricultural areas and in 70 percent of the samples from urban areas. Atrazine and metolachlor were the most frequently detected pesticides in samples from agricultural areas; atrazine and prometon were the most frequently detected pesticides in samples from urban areas. None of the concentrations exceeded USEPA MCLs, but many pesticides do not have MCLs.

Samples from alluvial aquifers in agricultural areas contained an average total pesticide concentration (1.3 µg/L) that was more than seven times the concentration in samples from urban wells (0.17 µg/L). Although total pesticide concentrations were higher in samples from agricultural areas, more compounds were detected in samples from urban areas. Seventeen compounds were detected in samples from urban areas compared to 12 compounds detected in samples from agricultural areas. Herbicide degradates were detected in 94 percent of samples from agricultural areas and 53 percent of the samples from urban areas. Alachlor ESA, metolachlor ESA, and metolachlor oxanilic acid (OA) were the most frequently detected degradates in samples from both agricultural and urban areas; deethylatrazine and deisopropylatrazine were detected frequently in agricultural areas.

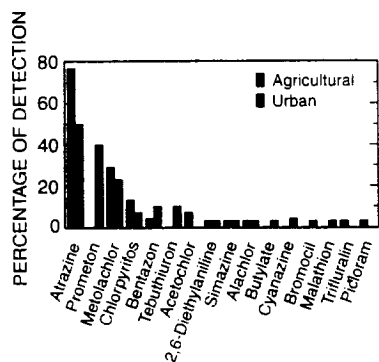


Figure 20. Pesticides were frequently detected in the alluvial aquifers in both agricultural and urban areas. Although the total concentrations of pesticides were higher in agricultural areas than in urban areas, more compounds were detected in urban than in agricultural areas.

Pesticides in Streams

The most commonly used herbicides were the most frequently detected. Atrazine and metolachlor, the two most commonly used herbicides in Iowa for row-crop agriculture during 1996–98, were detected in all stream samples (fig. 18). Acetochlor, alachlor, and cyanazine were detected in more than 70 percent of the samples. Most atrazine concentrations (76 percent) exceeded 0.1 µg/L with almost 60 percent of the samples in the 0.1 to 1.0 µg/L range (fig. 21). About 10 percent of the samples exceeded the atrazine MCL level of 3.0 µg/L. Almost half of the samples had metolachlor concentrations in the range from 0.1 to 1.0 µg/L (fig. 21). Other less frequently detected pesticides were carbofuran, 2,4-D, dicamba, EPTC, metribuzin, and trifluralin; these were applied in Iowa at a rate of only 0.6 to 30 percent of the amount applied for atrazine (Sands and Holden, 1996).

Acetochlor, a conditionally registered herbicide that is intended to replace a number of other commonly used herbicides, was frequently detected but most times (75 percent) at concentrations less than 0.1 µg/L. Acetochlor concentrations did not exceed the 2.0-µg/L annual mean concentration registration requirement at any site but did exceed this level in about 3 percent of the individual samples. The maximum concentration (10.6 µg/L) measured during the study exceeded the level that would trigger requirements for biweekly sampling for water-supply systems (U.S. Environmental Protection Agency, 1994).

Few nonagricultural herbicide compounds were detected. Many pesticides are used both in agricultural and urban settings, but only three herbicides used almost exclusively in non-row-crop agriculture and urban settings (prometon, tebuthiuron, and triclopyr) were present in streams in eastern Iowa and southern Minnesota from 1996 through 1998 (fig. 18). Prometon, a

herbicide used for weed control around buildings, fence rows, and under asphalt, was present at very low concentrations (less than 0.1 µg/L) in more than 80 percent of the samples. Prometon is extremely persistent (half-life of hundreds to thousands of days), which may explain the relatively high detection rate relative to its low use. Tebuthiuron, used on road rights-of-way and industrial sites, was detected in 7 percent of the samples. Triclopyr, used on road rights-of-way, industrial sites, and turf grass, was detected in 1 percent of the samples.

Insecticides were detected mainly during the summer. A number of insecticides that have been identified as posing a high risk to aquatic invertebrates were detected in streams from May through September, the months when most application normally occurs (fig. 22). Carbofuran was the most frequently detected insecticide (16 percent of all samples). Although detected in less than 20 percent of all samples, carbofu-

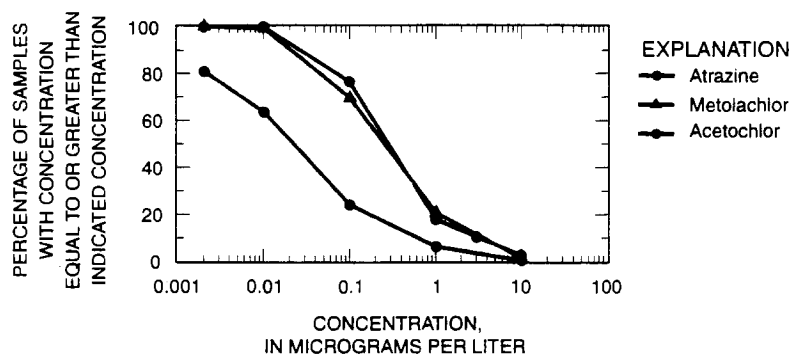


Figure 21. Atrazine and metolachlor were present in more than 50 percent of the samples from rivers and streams at concentrations between 0.1 and 1.0 µg/L. In contrast, acetochlor was present in 76 percent of the samples at concentrations less than 0.1 µg/L.

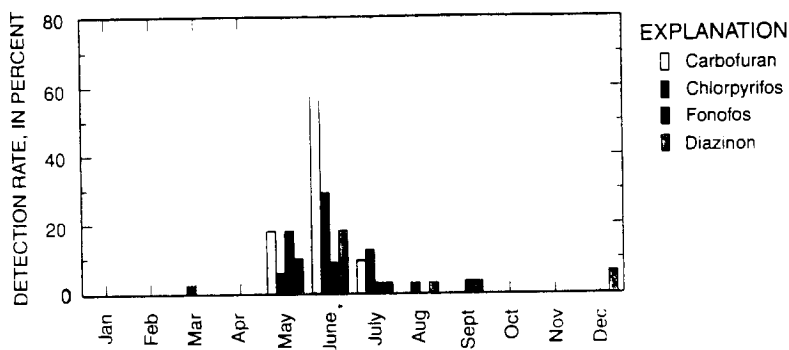


Figure 22. Insecticides were prevalent in streams only during the late spring and summer. Carbofuran and chlorpyrifos were the most frequently detected insecticides.

ran was detected in 68 percent of the samples collected in June. When present, carbofuran concentrations generally were less than 0.80 µg/L. Chlorpyrifos was detected in about 7 percent of the samples. As with the other insecticides, chlorpyrifos was detected most frequently in June (about 30 percent of the samples). The highest concentration was 0.06 µg/L. Malathion was detected in only three samples in spring and early summer at concentrations that ranged from 0.023 to 0.078 µg/L. Insecticides were detected more frequently in streams than in shallow ground water in the alluvial aquifers. Lower use relative to the herbicides, short half-life, and application during periods of reduced runoff may account for the overall low detection rate and low concentrations of insecticides in rivers and streams.

Pesticide degradates constitute the majority of the pesticide compounds in streams. The pesticide degradate compounds were some of the most frequently detected pesticide compounds in streams (fig. 18) and on average constituted the majority of the pesticide mass in water samples. Acetochlor ESA, alachlor ESA,

metolachlor ESA, and metolachlor OA were detected in more than 75 percent of the samples. The degradates were detected much more frequently than their parent compounds with the exception of atrazine and two of its degradates—deethylatrazine and deisopropylatrazine (fig. 18).

On average, approximately 83 percent of the total pesticide mass (parent compounds and degradates) can be accounted for by 10 common degradates of acetochlor, alachlor, atrazine, cyanazine, and metolachlor. Concentrations of acetochlor ESA, alachlor ESA, and metolachlor ESA commonly were more than 10 times higher than their parent compounds. Although herbicide degradates frequently occur in substantial concentrations in streams, only limited research has been conducted on the acute and chronic human and environmental effects of these compounds (Heydens and others, 2000).

The occurrence of degradates and the ratio of degradate to parent compound were substantially different for the triazine (atrazine and cyanazine) than the chloroacetanilide (acetochlor, alachlor, and metolachlor) compounds. Concentrations of the triazine degradates

tended to follow the pattern of the parent compounds—highest concentrations during the early summer followed by decreasing concentrations during the late summer and fall (fig. 23). However, concentrations of the triazine degradates were lower than their parent compounds except in the fall and winter when they were slightly higher. In contrast, the ESA and OA degradates of alachlor, acetochlor, and metolachlor were present in higher concentrations than their parent compounds throughout the year.

Occurrence of pesticides are related to landform type. The triazine herbicides—atrazine and cyanazine—and their degradates were present in significantly higher concentrations in streams draining soils dominated by windblown

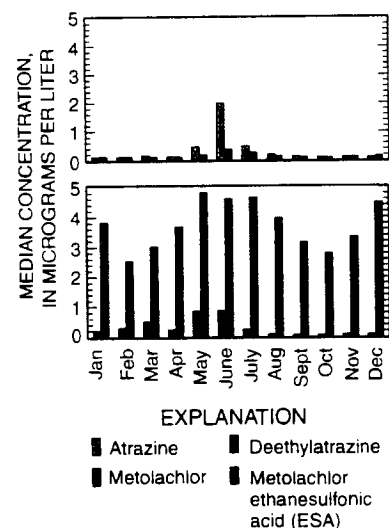


Figure 23. Pesticides and their degradates are readily available for transport to streams and rivers in late spring and early summer after application. A common metolachlor degradate persists at higher concentrations than a common atrazine degradate throughout the year.

Multiple Pesticide Compounds Occur More Frequently in Streams Than in Ground Water

The use of a wide variety of pesticides in the Eastern Iowa Basins is reflected by the presence of multiple compounds in streams and ground water. Two or more compounds were detected in every sample, and five or more compounds were detected in

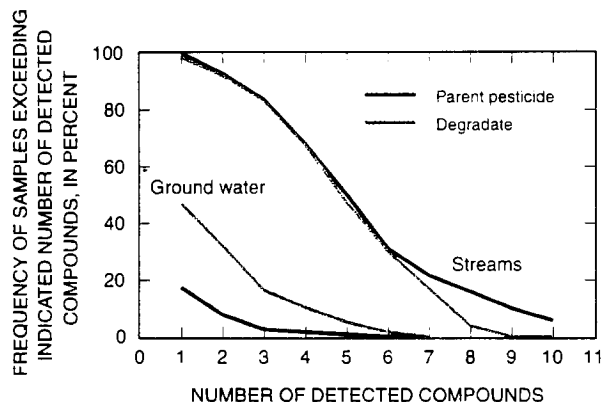


Figure 24. More than one pesticide compound was always present and more than five compounds were detected in about 80 percent of the stream samples. Multiple degradates were more common than multiple parent compounds in ground water.

more than 80 percent of the stream samples (fig. 24). As many as 16 pesticide compounds were detected in a single stream sample. In contrast, 2 or more compounds were detected in only 17 percent and 5 or more pesticides in less than 2 percent of the ground-water samples.

Multiple degradates were much more frequently detected than multiple parent compounds in ground water. Many pesticide compounds attach to soil particles and are not readily leached to the ground water. However, the pesticide degradates sampled are generally more water soluble and once formed in the soil may move readily to the shallow ground water. Also, several degradates may form from each pesticide, accounting for additional degradate compounds.

The importance of multiple pesticide compounds for human and environmental health is currently unclear. Most toxicity assessments are based on results from a single contaminant.

loess (Southern Iowa Drift Plains) than in streams draining till soils (Des Moines Lobe or the Iowan Surface). Because of differences in soil properties, triazine pesticide-use rates are apparently less in areas dominated by till soils such as the Des Moines Lobe and Iowan Surface landforms (Stoltenberg and Pope, 1990). The high pH of the till soils (increased calcium carbonate content) results in more triazine

herbicides available for plants and greater persistence. Less triazine herbicides have been applied to till soils (particularly in the Des Moines Lobe) because the effects of triazine herbicides can “carry over” to soybeans planted after corn. Results from a regional low-flow synoptic study (Sorenson and others, 1999) indicate that the percentage of blue-green algae in stream periphyton communities

increases with total (parent compounds plus degradates) triazine herbicide concentrations (Porter and others, 2001).

Runoff from rains soon after application washes large quantities of pesticides into streams.

Concentrations of pesticides and their degradates were highest in the late spring and early summer when intense rains occurred soon after application (fig. 23). Pesticides and their degradates are transported from the Eastern Iowa Basins in largest quantities during the late spring and early summer. Pesticide degradates make up the largest part of the pesticide load transported to the Mississippi River. Pesticide data from the Iowa River at Wapello, Iowa, illustrate that the monthly degradate load was higher than the loads of the parent compounds throughout the year (fig. 25) but were particularly dominant during the spring and early summer and late fall and winter periods. Typically, the loads for all compounds were largest during the spring and early summer (March through June) when significant runoff events transported pesticide compounds to streams. In the fall

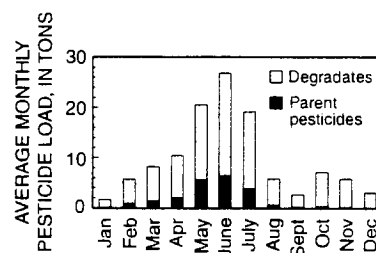


Figure 25. The most frequently used pesticides begin breaking down before being transported from the Study Unit to the Mississippi River. More than 80 percent of the yearly pesticide load in the Iowa River at Wapello is pesticide degradates. Of the yearly load, about 57 percent was transported from the Study Unit during May, June, and July.

and winter months, however, the pesticide degradates accounted for nearly all of the pesticide loads (fig. 25). Parent compounds accounted for only 3 percent (December) to 27 percent (May) of the total pesticide compounds transported from the Iowa River. Much of the water and dissolved pesticide compounds originated from ground-water inflow during the relatively dry late fall and winter months. The presence of relatively high concentrations of alachlor and metolachlor degradates in the fall and winter, several months after pesticide application when parent compounds have all but disappeared, indicates that these compounds are relatively stable.

Other Organic Compounds

The gasoline additive MTBE was the most commonly detected volatile organic compound in ground water. Shallow alluvial ground water in urban areas is susceptible to contamination from organic compounds resulting from activities such as transportation, manufacturing, and service industries. Volatile organic compounds (VOCs) were detected in 40 percent of alluvial samples from urban areas and 10 percent of the samples from agricultural areas. Methyl *tert*-butyl ether was the most frequently detected VOC and was present in 23 percent of samples from urban areas (fig. 26). Although most detected concentra-

tions were low, MTBE concentrations from two urban monitoring wells were at levels of potential concern for human health (USEPA drinking-water advisory of 20 to 40 µg/L). MTBE was commonly found with other gasoline compounds (benzene, ethylbenzene, toluene, and xylene), indicating that contamination likely originated from leaking fuel storage tanks and possibly from surface spills. MTBE was not detected in agricultural areas. Solvents were detected in less than 15 percent of the samples, and other VOCs were detected in less than 10 percent of the samples.

Water from deeper in the alluvial aquifers and in the deeper Silurian-Devonian and Upper Carbonate



MTBE DETECTION RATES ARE SIMILAR TO THE AVERAGE DETECTION RATE IN THE NATION

Methyl *tert*-butyl ether (MTBE), a compound added to gasoline to enhance the octane content or to ensure cleaner burning with reduced carbon monoxide emissions, was detected in the alluvial aquifers underlying towns and cities in the Eastern Iowa Basins at a rate similar to the average detection rate in the United States (Squillace and others, 1999). However, detection rates (fig. 26) were not as great as in shallow, vulnerable aquifers in Denver, Colorado (Bruce and McMahon, 1996), and other areas where MTBE has been used extensively to reduce carbon monoxide emissions. MTBE also has been detected in areas where MTBE is used only to enhance the octane content (Zorgorski and others, 1997). Although ethanol is commonly added to gasoline in Iowa, MTBE has also been added to increase the octane level. Legislatively mandated sampling of soil and ground water from leaking underground storage tank (LUST) sites in Iowa verified that MTBE is prevalent in gasoline-contaminated areas (Iowa Department of Natural Resources, 2000b). MTBE was present in ground water at about 59 percent of the 523 LUST sites in 248 Iowa towns and cities.

The effects of MTBE on human health are not fully understood, but MTBE has been linked to headaches, nausea, dizziness, and breathing difficulties (Melnick and others, 1997). Experimental studies indicate that MTBE is carcinogenic in rats and mice (Melnick and others, 1997).

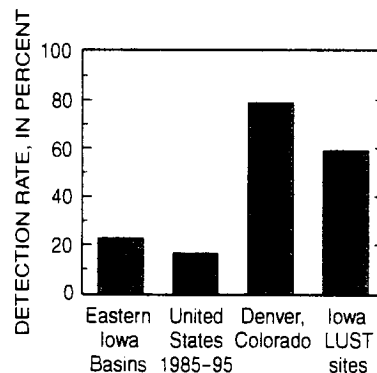


Figure 26. The presence of the gasoline additive MTBE in alluvial aquifers in urban areas reflects the common occurrence of MTBE at gasoline-contaminated sites in Iowa.



BIOLOGICAL COMMUNITIES IN STREAMS CONSIST OF ORGANISMS THAT ARE MODERATELY TO HIGHLY TOLERANT OF ENVIRONMENTAL DEGRADATION

Biological communities (algae, fish, and invertebrates) in streams in the Eastern Iowa Basins generally rank in the middle 50th to highest 25th percentile most tolerant of environmental degradation nationally (fig. 27). Commonly, biological communities in streams change in response to environmental changes (degradation). Degradation can result from a variety of factors that modify habitat or other environmental features such as land use, water chemistry, stream-flow, and others. The biological community may change from few individuals of many species to a community of many individuals of a few tolerant species.

Algal status focuses on the changes in the percentage of certain algae in response to increasing siltation and seems to correlate relatively well with higher nutrient concentrations in many regions. Invertebrate status is the average of 11 invertebrate (primarily insects, worms, crayfish, and clams) metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. Fish status is the sum of scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that change (increase) in association with water-quality degradation. For all indicators, higher values indicate a more degraded stream site.

Community differences occur among streams in the Eastern Iowa Basins, but these differences are small in relation to differences across the Nation. From a national perspective, algal status was moderate to high in all Study Unit streams and rivers and corresponds with high concentrations of dissolved and total nutrients. With one exception (Wapsipinicon River near DeWitt), invertebrate status tended to rank in the middle to lowest 25th percentile, indicating a moderate degree of degradation in relation to the rest of the Nation. Because of nutrient enrichment and subsequent abundance of algae and organic material, an abundant food supply is present to support a diverse invertebrate community. From a local and regional perspective, invertebrate status indicated greater degradation in large rivers than small streams. However, lower status in the Skunk River may be associated more with better habitat at that site than any quantitative differences in water chemistry among other large rivers. Fish communities tended to be ranked highest (most degraded) to moderate when compared nationally. Based on the presence of more species sensitive to environmental stress, the fish population at Wapsipinicon River near DeWitt was minimally affected by environmental degradation in relation to other sites nationally.

Landform/land-use setting	Stream or river (see p. 26)	Algal status	Invertebrate status	Fish status
Streams				
Reference site	Wapsipinicon River near Tripoli	■	■	■
Des Moines Lobe	Iowa River near Rowan	■	■	■
Des Moines Lobe with concentrated AFOs	South Fork Iowa River near New Providence	■	■	■
Southern Iowa Drift Plain	Old Mans Creek near Iowa City	■	■	■
Iowan Karst	Flood Creek near Powersville	■	■	■
Iowan Surface	Wolf Creek near Dysart	■	■	■
Large Rivers				
Mixed	Wapsipinicon River near DeWitt	■	■	■
Mixed	Iowa River at Marengo	■	■	■
Mixed	Cedar River near Conesville	■	■	■
Mixed	Iowa River at Wapello	■	■	■
Mixed	Skunk River at Augusta	■	■	■

- Highest 25 percent nationally
- Middle 50 percent nationally
- Lowest 25 percent nationally

National network represents 140 sites in the NAWQA national basic fixed-site network that have algal, invertebrate (primarily insects, worms, crayfish, clams), and fish data

Figure 27. Status of biological communities—comparison of Eastern Iowa Basins sites to selected national network sites.

bedrock aquifers used for rural domestic supply rarely contained detectable concentrations of VOCs, even though gasoline is commonly stored and solvents are at times used in the vicinity of these wells.

Pesticides banned or restricted in the 1970s and 1980s are still present in fish tissue. Although no longer in use, residue from organochlorine insecticides (chlordane, DDT, dieldrin, and heptachlor epoxide) is still found in fish-tissue samples collected from eastern Iowa streams (Roberts, 1997) but not at levels of concern for human health. Concentrations were relatively higher in agricultural than urban streams, and fish from the Wapsipinicon River Basin contained lower concentrations of contaminants than fish from the Cedar or Iowa River Basins. Although aldrin, an organochlorine pesticide, has been banned since the 1970s, its degradate, dieldrin, was present in five stream samples during or immediately following spring runoff. Dieldrin was present in tissue samples collected from carp at 15 of 16 sites in eastern Iowa, and concentrations in fish seem to be associated with agricultural settings (Roberts, 1997). Concentrations that peaked in sediment deposited in Coralville Reservoir during the 1993 flood indicate that dieldrin is still transported in rivers during high flow (Kalkhoff and Van Metre, 1997). In contrast to the apparent agricultural source of dieldrin, the occurrence and concentration of PCBs (polychlorinated biphenyls) in fish tissue indicate an urban source for this contaminant (Roberts, 1997).

Biological Communities and Stream Habitat

The quality of water entering from runoff and ground water inflow is an important factor that influences the community structure of fish, invertebrates, and algae that live in the rivers and streams of the Eastern Iowa Basins. Stream habitat, the physical conditions in and along streams and rivers, also influences the occurrence and distribution of aquatic organisms. Water-quality degradation and habitat alteration resulting from agricultural and urban activities can adversely affect aquatic communities.

In general, degradation results in the replacement of native species with those that are more tolerant of nutrient and organic enrichment. There also tends to be a reduction in numbers of sensitive species, increases in abundance and dominance of tolerant species, and decreases in the diversity and evenness of biological communities that are typically considered an indicator of environmental stress.

Biological metrics that contain measures of the composition, abundance, function, and tolerance of species to environmental stress are commonly used to evaluate aquatic communities in relation to water quality and habitat conditions.

Fish populations are related to stream size and water-quality conditions. During 1996–98, 67 fish species representing 15 families were collected in six streams and six rivers in the Eastern Iowa Basins (Sullivan, 2000). The species and trophic composition, as well as the abundance and condition of fish communities, varied between the smaller streams and large river sites in the Eastern

Iowa Basins. Measures of fish-community status, tolerance, structure, and index of biotic integrity (IBI) indicated that fish communities also differed among land uses and landforms in the basin (fig. 28). Habitat and water-quality data indicate that differences in species composition between rivers and streams are probably due to more than just stream size. Although differences in stream size and associated habitat differences favor the occurrence of different species, relatively higher concentrations of suspended sediment and phosphorus in large rivers may contribute to habitat degradation and eutrophication that may explain the relatively lower IBI scores at large river sites (Sullivan, 2000).

The two highest IBI-rated river sites were the Wapsipinicon River near DeWitt and the Cedar River at Gilbertville, an integrator site for

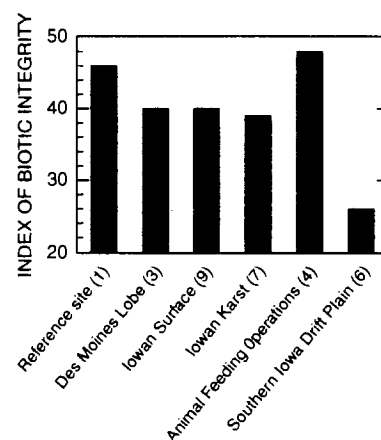


Figure 28. Healthy fish populations were present in one of the least affected (reference) and one of the most affected (animal feeding operations) streams in the study area. Factors other than water quality (stream habitat) also have an important effect on fish populations. (Site number in parentheses; see site map in “Study Unit Design” Section.)

the upper Cedar River watershed River sites with the lowest IBI scores include the Skunk River at Augusta and three sites in the Iowa River Basin: Iowa River at Marengo, Old Mans Creek near Iowa City, and the most downstream integrator site (Iowa River at Wapello).

Agriculture and subsequent organic enrichment of the streams have likely reduced habitat, cover, and water-quality requirements for the maintenance of diverse fish populations and communities. However, the small number of fish species in Flood Creek (fig. 27) may be reflective of the karst hydrology in the basin; extended reaches of Flood Creek may be dry during low-flow conditions or periods of drought, limiting fish communities to juvenile fish or those with rapid recolonization rates.

Macroinvertebrate communities in the Eastern Iowa Basins reflect cumulative effects of both land use and downstream succession—the natural sequence of communities from headwater streams to large rivers (Vannote and others, 1980). More than 250 benthic invertebrate taxa were found in collections from submerged woody debris at 12 Basic Fixed Sites (see Glossary). Aquatic insects, including mayflies, net-spinning caddisflies, and midges, accounted for more than one-half of the organisms collected. The abundance of highly tolerant midges (Chironomidae) and worms (Oligochaeta), as well as collector-filterers was

higher in large rivers, whereas the abundance of less-tolerant mayflies, stoneflies, and caddisflies (EPT taxa; see Glossary) was relatively larger in small tributary streams (Brigham and Sadorf, 2001). However, most invertebrate taxa in eastern Iowa streams and rivers are considered to be tolerant of nutrient and organic enrichment (Hilsenhoff, 1987). Stream velocity, rates of stream respiration, minimum dissolved oxygen, wooded-riparian zones, and the composition and abundance of periphyton were the primary factors associated with invertebrate community structure (M.A. Harris, U.S. Geological Survey, written commun., 2000).

In general, invertebrate diversity was lower in unshaded streams with silt, sand, or gravel bottoms (for example, Iowa River near Rowan and Old Mans Creek) than in shaded streams with bedrock or large rocks (for example, Flood Creek and Wapsipinicon River near Tripoli). Despite similarities in agricultural land-use intensity in the Midwestern Corn Belt, invertebrate communities reveal substantial differences in the quality of streams and rivers, corresponding to physical (modifications of stream channels and riparian zones) and hydrologic (rainfall-runoff and ground-water relations) differences among basins (M.A. Harris, U.S. Geological Survey, written commun., 2000).

Algal communities are dominated by nutrient- and sediment-tolerant taxa. More than 330 algal species were found in periphyton

collections from submerged woody debris. Algal communities were dominated by taxa that are tolerant to nutrients and other agricultural contaminants such as herbicides and sediment.

Relative differences are seen in algal status (an indicator of nutrient and sediment enrichment) and biomass among sites from a regional and local perspective. For example, algal biomass (the amount of attached algae) was relatively higher in Old Mans Creek, Flood Creek, and the Cedar River at Gilbertville than other sites. Within the Eastern Iowa Basins, algal status was relatively better in the Wapsipinicon River near Tripoli (the reference site) and in the Cedar River near Gilbertville and Conesville.

During low-flow conditions in August 1997, shaded streams that drain permeable soils (for example, the upper Cedar and Wapsipinicon River Basins) were dominated by periphyton taxa (diatoms, red algae, and green algae) that are a good food source for invertebrates and fish. In contrast, channelized streams and rivers with poor riparian shading, impermeable soils, and slow velocity were dominated by phytoplankton taxa (notably blue-green algae) that are avoided as food sources and contribute to organic enrichment, higher rates of stream respiration, and lower dissolved oxygen concentrations during the night (Sorenson and others, 1999; Porter, 2000).

RIPARIAN BUFFER ZONES INFLUENCE THE QUALITY OF MIDWESTERN STREAMS AND RIVERS

Despite similar land use throughout the Corn Belt region of the Midwest, streams flowing through cropland differ considerably in their ecological characteristics, in part because of differences in riparian buffer zones (see text box). This conclusion is based on an investigation of 70 streams and rivers within three NAWQA Study Units in the upper Midwest during August 1997 (fig. 29; Sorenson and others, 1999; Porter and others, 2001). Specifically, increases in tree cover in buffer zones were associated with aquatic biological communities indicative of good stream quality, reduced nuisance algal growths, and maintenance of sufficient dissolved oxygen concentrations to support diverse communities of aquatic organisms. For example, the number of aquatic insects indicative of good stream quality tended to increase with increases in percentage of tree cover, especially in sites where streamflow and dissolved oxygen conditions were favorable. Fish communities, sampled at 24 sites in the UMIS Study Unit, also indicated better overall conditions in streams with wooded riparian zones than those with more open canopy (Stauffer and others, 2000).

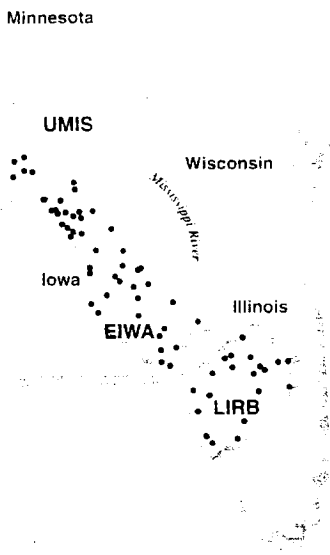


Figure 29. The influence of riparian buffer zones on the quality of 70 Midwest streams and rivers was evaluated in the Upper Mississippi River (UMIS), Eastern Iowa (EIWA), and Lower Illinois River Basins (LIRB).

Streams with less tree cover, and thus less shading, contained relatively large growths of phytoplankton (algae suspended in the water) at levels considered indicative of eutrophication (Porter, 2000). Organic enrichment

resulting from excessive algal production in some Midwestern streams may reduce dissolved oxygen concentrations and be detrimental to other requirements of aquatic organisms.

Shading from tree cover in riparian buffer zones may influence nutrient concentrations indirectly by reducing the growth of phytoplankton. In streams where phytoplankton were abundant (often where buffer zones were thin or lacking), dissolved nitrate concentrations were significantly lower (fig. 30; Porter, 2000). The lower nutrient concentrations may result from uptake by the abundant phytoplankton. Thus, assessments of eutrophication would benefit from consideration of biological communities and the riparian zone, rather than being based solely on nutrient concentrations in the water.

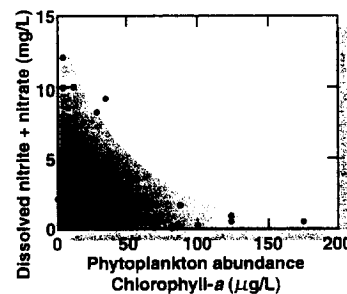
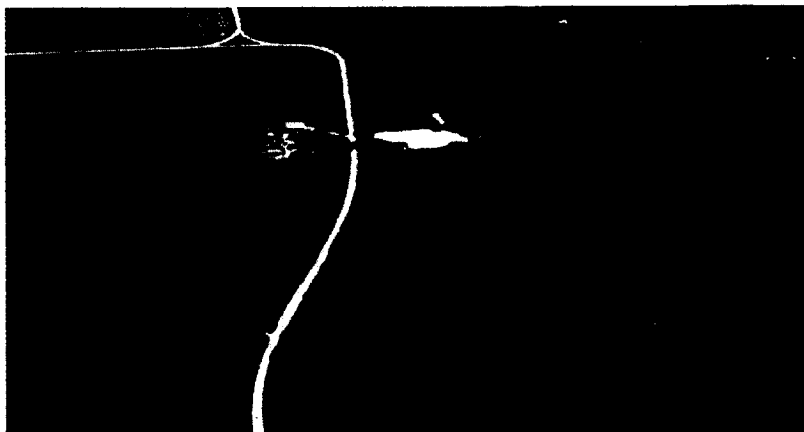


Figure 30. Dissolved nutrient concentrations decreased in eutrophic streams with excessive algal productivity. Rates of nutrient uptake by the algae can exceed rates at which nutrients are transported by streams during low-flow conditions.



Digital images derived from USGS topographic maps were used to estimate the percentage of trees in a riparian buffer zone (a 100-meter width on each side of the stream) for 2- to 3-mile segments upstream from each sampling site, supplemented by vegetation surveys at the sampling site (Sorenson and others, 1999).

Resource agencies, including the U.S. Department of Agriculture, encourage maintenance of strips of trees or grass between cropland and streams as a best management practice. These "riparian buffer zones" are thought to intercept runoff of sediment and chemicals from fields, promote bank stability, and provide shading and habitat for aquatic life (Osborne and Kovacic, 1993). Riparian buffer zones should be considered along with other important factors that affect chemical and biological indicators of stream quality, such as soil drainage properties and stream hydrology (Porter and others, 2001).

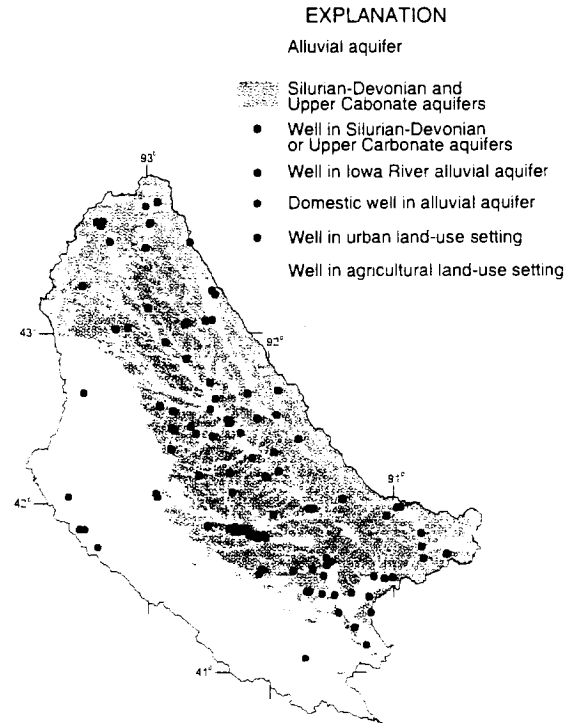
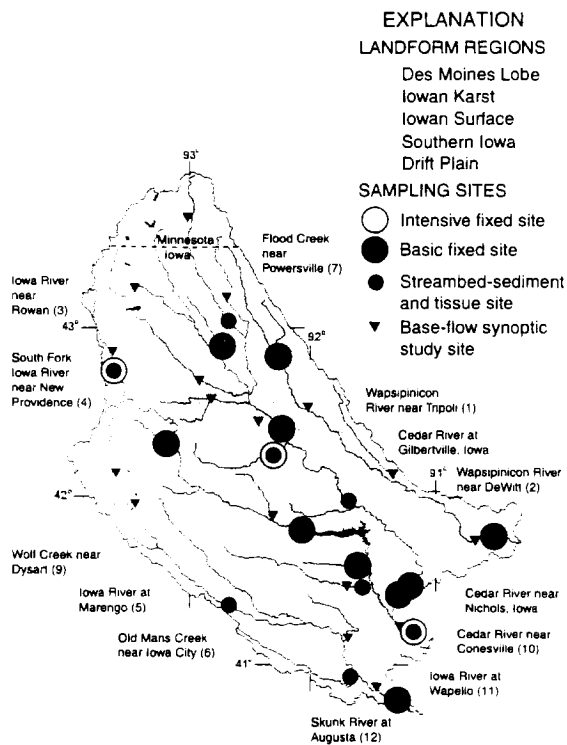
STUDY UNIT DESIGN

The objective of the Eastern Iowa Basins NAWQA study was to assess the water-quality conditions in streams and ground water in the Study Unit. The study design is based on a nationally consistent structure that incorporates an interdisciplinary approach (Gilliom and others, 1995). Stream-water quality was assessed using three interrelated components: stream chemistry, streambed-sediment chemistry, and stream ecology. Ground-water quality of the alluvial aquifers was selected for assessment because these aquifers are the major source of water for municipal and domestic supplies and they provide flow to streams. Water quality in the Silurian-Devonian and Upper Carbonate aquifers also was investigated.

Stream Chemistry

The Basic Fixed Site sampling network was designed to characterize the effects of physiographic differences on water quality in the primarily agricultural Study Unit. Water-chemistry, bed-sediment, and reservoir-core data were collected. Sites were selected on large rivers and smaller streams. Fixed sites on large rivers were located near the mouth of the four major rivers to characterize the integrated effects of differing land use and environmental setting on stream quality. Two additional large river sites were chosen to assess the upper part of the Cedar River and the Iowa River, before it flows into Coralville Reservoir. Fixed sites on streams were selected to characterize each of the physiographic areas. A reference site was selected on a watershed that retains a large amount of bottomland wetlands. Another site was selected to assess the effects of concentrated animal feeding operations on stream quality. A subset of the Basic Fixed Site network was intensively sampled (weekly to

biweekly) through the spring and summer of 1997. Two synoptic studies were conducted during base-flow conditions (high and low base flow) to improve the spatial resolution and to better evaluate the effects of soil type and riparian buffers on stream-water quality and biological communities.



Stream Ecology

Ecological data including fish-tissue chemistry and fish, macroinvertebrate, and algal community structure were collected to provide better understanding of the relations among physical, chemical, and biological characteristics of a stream. Data were collected at the Basic Fixed Site sampling network plus four additional sites to provide better spatial coverage.

Ground-Water Chemistry

The ground-water network was designed to characterize water quality in the most heavily used aquifers in the Study Unit. A Study Unit survey characterized the water quality in the Silurian-Devonian and Upper Carbonate bedrock aquifers, the second greatest source of municipal and domestic supplies in the Study Unit. Another Study Unit survey characterized the water quality of the alluvial aquifers using domestic wells. Land-use studies assessed the occurrence and distribution of water-quality constituents in recently recharged water in the alluvial aquifers. Agricultural and urban land-use effects on quality of shallow ground water was characterized by sampling two networks of monitoring wells constructed at randomly selected sites.

SUMMARY OF DATA COLLECTION IN THE EASTERN IOWA BASINS, 1996-98

Study Component	Constituents were collected and why	Where	When	Frequency
Stream chemistry				
Basic Fixed Sites—large rivers	Major ions, organic carbon, suspended sediment, nutrients, pesticides, and streamflow were determined to describe concentrations and the amount of selected constituents transported from the study area	Streams draining basins from about 2,300 to more than 12,000 square miles that integrate the effects of urban and agricultural land use and physiographic regions	6 in 1996, 5 in 1997-98	Monthly beginning in March 1996 and during selected flood events
Basic Fixed Sites—streams	Major ions, organic carbon, suspended sediment, nutrients, pesticides, and streamflow were determined to evaluate physiographic effects on stream-water quality	Streams draining basins from 120 to 418 square miles of homogeneous land use and physiography	6	Monthly beginning in March 1996 and during selected flood events
Intensive Fixed Sites	Major ions, organic carbon, suspended sediment, nutrients, pesticides, and streamflow were determined to define short-term temporal variability	One large river and two stream Basic Fixed Sites	3	Weekly during 1997 growing season; biweekly for remainder of the year
Base-flow synoptic study	Nutrients, pesticides, organic carbon, and streamflow were determined to refine spatial variability during both low and high base-flow conditions	Streams draining basins ranging from 120 to 530 square miles representing greater than 90 percent agricultural land use	25	August 1997 and May 1998
Bed-sediment chemistry				
Bed sediment and tissue	Trace elements, organochlorine, and semivolatile organic compounds in streambed sediment to determine presence of these potentially toxic, hydrophobic compounds	Ecological sites—Large river and tributary/headwater fixed sites plus four additional sites for better spatial coverage	16	September 1995
Reservoir core study	Trace elements and organochlorine compounds in sediment to determine the historical occurrence (from filling in 1958 to 1993)	Site in a deep depositional zone of the Coralville Reservoir about 1.5 miles upstream from the dam	1	November 1993
Stream ecology				
Bed sediment and tissue	Trace elements and organochlorine compounds in fish tissue to determine occurrence	Ecological sites	16	September 1995
Intensive assessments	Fish, benthic invertebrates, algae, and aquatic and riparian habitat were sampled and described to assess community structure and to document within stream and annual variation	Ecological sites	12	All fixed sites in 1996 and intensive sites in 1997-98
Ecological synoptic survey	Benthic invertebrates, algae, and aquatic and riparian habitat were sampled to assess biological responses in relation to water quality and hydrologic variability	Streams draining basins ranging from 120 to 530 square miles representing greater than 90 percent agricultural land use	25	August 1997
Ground-water chemistry				
Bedrock aquifer survey	Major ions, nutrients, pesticides, pesticide degradates, VOCs, and tritium were determined to assess quality in second most-used aquifer in Study Unit	Existing domestic wells completed in the Silurian-Devonian aquifer (32-700 feet deep)	33	June-July 1996
Alluvial aquifer survey	The same constituents as in bedrock aquifer survey were determined to assess quality in most-used aquifer in the Study Unit	Existing domestic wells completed in unconsolidated alluvial deposits	32	June-July 1998
Land-use effects study—agricultural and urban	The same constituents as in bedrock aquifer survey were determined to assess water-quality differences due to agricultural and urban land use	Newly constructed monitoring wells at sites randomly selected on alluvial deposits and completed at the water table (31 agricultural and 30 urban wells)	61	June-August 1997
Ground-water chemistry special study				
Changing land-use study	The same constituents as in bedrock aquifer survey were determined to assess changes in water-quality due to conversion of row crops to wetlands and prairie	Existing monitoring wells plus three new monitoring wells completed at various depths in the Iowa River alluvial aquifer	28	August 1996 and 1998

GLOSSARY

- Acre-foot**—A volume of water equal to 1 foot in depth and covering 1 acre; equivalent to 43,560 cubic feet or 325,851 gallons.
- Algae**—Chlorophyll-bearing nonvascular, primarily aquatic species that have no true roots, stems, or leaves; most algae are microscopic, but some species can be as large as vascular plants.
- Alluvial aquifer**—A water-bearing deposit of unconsolidated material (sand and gravel) left behind by a river or other flowing water.
- Alluvium**—A general term for clay, silt, sand, and gravel deposited by a river or stream in the bed of the stream or on its flood plain.
- Ammonia**—A compound of nitrogen and hydrogen (NH₃) that is a common by-product of animal waste. Ammonia readily converts to nitrate in soils and streams.
- Aquatic guidelines**—Specific levels of water quality which, if reached, may adversely affect aquatic life. These are nonenforceable guidelines issued by a governmental agency or other institution.
- Aquifer**—A water-bearing layer of soil, sand, gravel, or rock that will yield usable quantities of water to a well.
- Base flow**—Sustained, low flow in a stream; groundwater discharge is the source of base flow in most places.
- Basic Fixed Sites**—Sites on streams at which streamflow is measured and samples are collected for temperature, salinity, suspended sediment, major ions, nutrients, and organic carbon to assess the broad-scale spatial and temporal character and transport of inorganic constituents of streamwater in relation to hydrologic conditions and environmental settings.
- Breakdown product**—A compound derived by chemical, biological, or physical action upon a pesticide. The breakdown is a natural process which may result in a more toxic or a less toxic compound and a more or less persistent compound.
- Concentration**—The amount or mass of a substance present in a given volume or mass of sample. Usually expressed as micrograms per liter (water sample) or micrograms per kilogram (sediment or tissue sample).
- Constituent**—A chemical or biological substance in water, sediment, or biota that can be measured by an analytical method.
- Contamination**—Degradation of water quality compared to original or natural conditions due to human activity.
- Cubic foot per second (ft³/s, or cfs)**—Rate of water discharge representing a volume of 1 cubic foot passing a given point during 1 second, equivalent to approximately 7.48 gallons per second or 448.8 gallons per minute or 0.02832 cubic meter per second.
- Degradate**—See Breakdown product.
- Detection limit**—The minimum concentration of a substance that can be identified, measured, and reported within 99 percent confidence that the analyte concentration is greater than zero; determined from analysis of a sample in a given matrix containing the analyte.
- Discharge**—Rate of fluid flow passing a given point at a given moment in time, expressed as volume per unit of time.
- Drainage basin**—The portion of the surface of the Earth that contributes water to a stream through overland run-off, including tributaries and impoundments.
- Drinking-water standard or guideline**—A threshold concentration in a public drinking-water supply, designed to protect human health. As defined here, standards are U.S. Environmental Protection Agency regulations that specify the maximum contamination levels for public water systems required to protect the public welfare; guidelines have no regulatory status and are issued in an advisory capacity.
- Ecoregion**—An area of similar climate, landform, soil, potential natural vegetation, hydrology, or other ecologically relevant variables.
- EPT richness index**—An index based on the sum of the number of taxa in three insect orders, Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies), that are composed primarily of species considered to be relatively intolerant to environmental alterations.
- Eutrophication**—The process by which water becomes enriched with plant nutrients, most commonly phosphorus and nitrogen.
- Ground water**—In general, any water that exists beneath the land surface, but more commonly applied to water in fully saturated soils and geologic formations.
- Habitat**—The part of the physical environment where plants and animals live.
- Hypoxia**—Seasonally depleted dissolved oxygen concentrations (less than 2 milligrams per liter) in a water body.
- Index of Biotic Integrity (IBI)**—An aggregated number, or index, based on several attributes or metrics of a fish community that provides an assessment of biological conditions.
- Indicator sites**—Stream sampling sites located at outlets of drainage basins with relatively homogeneous land use and physiographic conditions: most indicator-site basins have drainage areas ranging from 100 to about 400 square miles.

- Integrator or Mixed-use site**—Stream sampling site located at an outlet of a drainage basin that contains multiple environmental settings. Most integrator sites are on major streams with relatively large drainage areas.
- Intolerant organisms**—Organisms that are not adaptable to human alterations to the environment and thus decline in numbers where human alterations occur. See also Tolerant species.
- Karst**—A type of topography that results from dissolution and collapse of carbonate rocks such as limestone and dolomite, and characterized by closed depressions or sinkholes, caves, and underground drainage.
- Leaching**—Refers to movement of pesticides or nutrients from land surface to ground water.
- Load**—General term that refers to a material or constituent in solution, in suspension, or in transport; usually expressed in terms of mass or volume.
- Loess**—Homogeneous, fine-grained sediment made up primarily of silt and clay, and deposited over a wide area (probably by wind).
- Maximum contaminant level (MCL)**—Maximum permissible level of a contaminant in water that is delivered to any user of a public water system. MCLs are enforceable standards established by the U.S. Environmental Protection Agency.
- Median**—The middle or central value in a distribution of data ranked in order of magnitude. The median is also known as the 50th percentile.
- Monitoring well**—A well designed for measuring water levels and testing ground-water quality.
- Mouth**—The place where a stream discharges to a larger stream, a lake, or the sea.
- Nutrient**—Element or compound essential for animal and plant growth. Common nutrients in fertilizer include nitrogen, phosphorus, and potassium.
- Overland flow**—The part of surface runoff flowing over land surfaces toward stream channels.
- Periphyton**—Organisms that grow on underwater surfaces; periphyton include algae, bacteria, fungi, protozoa, and other organisms.
- Pesticide**—A chemical applied to crops, rights-of-way, lawns, or residences to control weeds, insects, fungi, nematodes, rodents or other "pests."
- Physiography**—A description of the surface features of the Earth, with an emphasis on the origin of landforms.
- Plankton**—Floating or weakly swimming organisms at the mercy of the waves and currents. Animals of the group are called zooplankton and the plants are called phytoplankton.
- Radon**—A naturally occurring, colorless, odorless, radioactive gas formed by the disintegration of the element radium; damaging to human lungs when inhaled.
- Recharge**—Water that infiltrates the ground and reaches the saturated zone.
- Reference site**—A NAWQA sampling site selected for its relatively undisturbed conditions.
- Riparian zone**—Pertaining to or located on the bank of a body of water, especially a stream.
- Runoff**—Excess rainwater or snowmelt that is transported to streams by overland flow, tile drains, or ground water.
- Species diversity**—An ecological concept that incorporates both the number of species in a particular sampling area and the evenness with which individuals are distributed among the various species.
- Species (taxa) richness**—The number of species (taxa) present in a defined area or sampling unit.
- Study Unit**—A major hydrologic system of the United States in which NAWQA studies are focused. Study Units are geographically defined by a combination of ground- and surface-water features and generally encompass more than 4,000 square miles of land area.
- Tile drain**—A buried perforated pipe designed to remove excess water from soils.
- Tolerant species**—Those species that are adaptable to (tolerant of) human alterations to the environment and often increase in number when human alterations occur.
- Total concentration**—Refers to the concentration of a constituent regardless of its form (dissolved or bound) in a sample.
- Triazine herbicide**—A class of herbicides containing a symmetrical triazine ring (a nitrogen-heterocyclic ring composed of three nitrogens and three carbons in an alternating sequence). Examples include atrazine, propazine, and simazine.
- Tritium**—A radioactive form of hydrogen with atoms of three times the mass of ordinary hydrogen; can be used to determine the age of water.
- Unconsolidated deposit**—Deposit of loosely bound sediment that typically fills topographically low areas.
- Urban site**—A site that has greater than 50 percent urbanized and less than 25 percent agricultural area.
- Volatile organic compounds (VOCs)**—Organic chemicals that have a high vapor pressure relative to their water solubility. VOCs include components of gasoline, fuel oils, and lubricants, as well as organic solvents, fumigants, some inert ingredients in pesticides, and some by-products of chlorine disinfection.

REFERENCES

- Akers, K.K.B., Schnoebelen, D.J., Savoca, M.E., Roberts, L.R., and Becher, K.D., 1999. Water-quality assessment of the Eastern Iowa Basins—Hydrologic and biologic data. September 1995 through September 1996: U.S. Geological Survey Open-File Report 99–66, 154 p.
- Akers, K.K.B., Montgomery, D.L., Christiansen, D.E., Savoca, M.E., Schnoebelen, D.J., Becher, K.D., and Sadorf, E.M., 2000. Water-quality assessment of the Eastern Iowa Basins—Hydrologic and biologic data. October 1996 through September 1998: U.S. Geological Survey Open-File Report 00–67, 300 p.
- Alexander, R.B., Smith, R.A., and Schwarz, G.E., 2000. Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico: *Nature*, v. 403, p. 758–761.
- Becher, K.D., Schnoebelen, D.J., and Akers, K.K.B., 2000. Nutrient concentrations and yields in surface water in Eastern Iowa: *Journal of the American Water Resources Association*, v. 36, no. 1, p. 161–173.
- Brigham, A.R., and Sadorf, E.M., 2001. Benthic invertebrate assemblages and their relation to physical and chemical characteristics of streams in the Eastern Iowa Basins, 1996–1998: U.S. Geological Survey Water-Resources Investigations Report 00–4256, p.
- Bruce, B.W., and McMahon, P.B., 1996. Shallow groundwater quality beneath a major urban center—Denver, Colorado, USA: *Journal of Hydrology*, v. 186, p. 129–151.
- Cambardella, C.A., Moorman, T.B., Jaynes, D.B., Hatfield, J.L., Parkin, T.B., Simpkins, W.W., and Karlen, D.L., 1999. Water quality in Walnut Creek watershed—Nitrate-nitrogen in soils, subsurface drainage water, and shallow groundwater: *Journal of Environmental Quality*, v. 28, p. 25–34.
- Gilliom, R.J., Alley, W.M., and Gurtz, M.E., 1995. Design of the National Water-Quality Assessment Program—Occurrence and distribution of water-quality conditions: U.S. Geological Survey Circular 1112, 33 p.
- Goolsby, D.A., Battaglin, W.A., Lawrence, G.B., Artz, R.S., Aulenbach, B.T., Hooper, R.P., Keeney, D.R., and Stensland, G.J., 1999. Flux and sources of nutrients in the Mississippi—Atchafalaya River Basin Topic 3 Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico: National Oceanic and Atmospheric Administration Decision Analysis Series No. 17, 130 p.
- Heydens, W.F., Wilson, A.G., Kraus, L.J., Hopkins, W.E., and Hotz, K.J., 2000. Ethane sulfonate metabolite of alachlor—Assessment of oncogenic potential based on metabolic and mechanistic considerations: *Toxicological Sciences*, v. 55, p. 36–43.
- Hilsenhoff, W.L., 1987. An improved index of organic stream pollution: *The Great Lakes Entomologist*, v. 20, no. 1, p. 31–39.
- Hoyer, B.E., and Hallberg, G.R., 1991. Groundwater vulnerability regions of Iowa: Iowa Department of Natural Resources. Geological Survey Bureau Special Map Series II, 1 sheet.
- Iowa Department of Natural Resources, 1999. Animal waste control facilities with operating permits in Iowa, digital data, accessed August 1999 at URL <http://www.igsb.uiowa.edu/nrgis/gishome.htm>
- _____, 2000a, Final approved Iowa 1998 303(d) list: accessed September 2000 at URL <http://www.state.ia.us/epd/wtresrce/303dnotc.htm>
- _____, 2000b, Methyl tertiary-butyl ether (MTBE) occurrence in Iowa: Iowa Department of Natural Resources Report for the 2000 Session of the Seventy-Eighth General Assembly, 29 p. accessed June 2000 at URL <http://www.state.ia.us/government/dnr/organiza/epd/ust/gwprof/021500.htm>.
- Kalkhoff, S.J., Kolpin, D.W., and Thurman, E.M., 1998. Degradation of chloroacetanilide herbicides—The prevalence of sulfonic and oxanilic acid metabolites in Iowa ground and surface waters: *Environmental Science and Technology*, v. 32, no. 11, p. 1738–1740.
- Kalkhoff, S.J., and Van Metre, P.C., 1997. Organochlorine compounds in a sediment core from the Coralville Reservoir, Iowa: U.S. Geological Survey Fact Sheet 129–97, 4 p.
- Kolpin, D.W., Kalkhoff, S.J., Goolsby, D.A., Sneek-Fahrer, D.A., and Thurman, E.M., 1997. Occurrence of selected herbicides and herbicide degradation products in Iowa's ground water, 1995: *Ground Water*, v. 35, no. 4, p. 679–688.
- Melnick, R.L., and others, 1997. Interagency assessment of oxygenated fuels, Chap. 4, Potential health effects of oxygenated gasoline: Washington, D.C., Office of Science and Technology Policy, p. 4–1—4–38.
- Olcott, P.G., 1992. Ground water atlas of the United States—Segment 9, Iowa, Michigan, Minnesota, Wisconsin: U.S. Geological Survey Hydrologic Investigations Atlas 730–J, 31 p.
- Omernik, J.A., 2000. Draft aggregations of level III ecoregions for the National Nutrient Strategy: accessed September 2000 at URL <http://www.epa.gov/ost/standards/ecomap.html>.
- Osborne, L.L., and Kovacic, D.A., 1993. Riparian vegetated buffer strips in water-quality restoration and stream management: *Freshwater Biology*, v. 29, p. 243–258.
- Porter, S.D., Harris, M.A., and Kalkhoff, S.J., 2001. Influence of natural factors on the quality of midwestern streams and rivers: U.S. Geological Survey Water-Resources Investigations Report 00–4288.
- Porter, S.D., 2000. Upper Midwest river systems—Algal and nutrient conditions in streams and rivers in the upper Midwest region during seasonal low-flow conditions, *in* Nutrient criteria technical guidance manual, rivers and streams: Washington, D.C., U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology, EPA–822–B–00–002, p. A–25—A–42.
- Prior, J.C., 1991. Landforms of Iowa: Iowa City, University of Iowa Press, p. 30–75.

- Rabalais, N.N., 1996, Nutrient changes in the Mississippi River and system responses on the adjacent continental shelf: *Estuaries*, v. 19, p. 386–407.
- Roberts, L.R., 1997, Occurrence of selected organochlorine compounds in fish tissue from eastern Iowa streams, 1995: U.S. Geological Survey Fact Sheet 027–97, 4 p.
- Sadorf, E.M., Linhart, S.M., and Savoca, M.E., 2000, Water quality of alluvial aquifers in eastern Iowa and southern Minnesota, 1998: U.S. Geological Survey Water-Resources Investigations Report 00–4106, 46 p.
- Sands, J.K., and Holden, H.R., 1996, Iowa agricultural statistics: Des Moines, Iowa, U.S. Department of Agriculture and Iowa State University Extension, 97 p.
- Savoca, M.E., Tobias, J.L., Sadorf, E.M., and Birkenholtz, T.L., 1997, Herbicides and nitrates in the Iowa River alluvial aquifer prior to changing land use, Iowa County, Iowa, 1996: U.S. Geological Survey Fact Sheet 085–97, 4 p.
- Savoca, M.E., Sadorf, E.M., Akers, K.K., 1999, Groundwater quality in the eastern part of the Silurian-Devonian and Upper Carbonate aquifers in the Eastern Iowa Basins, Iowa and Minnesota, 1996: U.S. Geological Survey Water-Resources Investigations Report 98–4224, 31 p.
- Savoca, M.E., Sadorf, E.M., Linhart, S.M., and Akers, K.K.B., 2000, Effects of land use and hydrogeology on the water quality of alluvial aquifers in eastern Iowa and southern Minnesota, 1997: U.S. Geological Survey Water-Resources Investigations Report 99–4246, 38 p.
- Schnoebelen, D.J., Becher, K.D., Bobier, M.W., and Wilton, T., 1999, Selected nutrients and pesticides in streams of the Eastern Iowa Basins, 1970–95: U.S. Geological Survey Water-Resources Investigations Report 99–4028, 65 p.
- Schwarz, G.E., and Alexander, R.B., 1995, State soil geographic (STATSGO) data base for the conterminous United States: U.S. Geological Survey Open-File Report 95–489.
- Soenksen, P.J., 1996, Transport of agricultural chemicals in surface flow, tileflow, and streamflow of the Walnut Creek watershed near Ames, Iowa, April 1991–September 1993: U.S. Geological Survey Water-Resources Investigations Report 96–4017, 41 p.
- Sorenson, S.K., Porter, S.D., Akers, K.K.B., Harris, M.A., Kalkhoff, S.J., Lee, K.E., Roberts, L.R., and Terrio, P.J., 1999, Water quality and habitat conditions in upper Midwest streams relative to riparian vegetation and soil characteristics, August 1997—Study design, methods, and data: U.S. Geological Survey Open-File Report 99–202, 53 p.
- Squillace, P.J., Moran, M.J., Lapham, W.W., Price, C.V., Clawges, R.M., and Zogorski, J.S., 1999, Volatile organic compounds in untreated ambient groundwater of the United States, 1985–1995: *Environmental Science and Technology*, v. 33, no. 23, p. 4176–4187.
- Stamper, D.M., and Tuovinen, O.H., 1998, Biodegradation of the acetanilide herbicides alachlor, metolachlor, and propachlor: *Critical Reviews in microbiology*, v. 24, no. 1, p. 1–22.
- Stauffer, J.C., Goldstein, R.M., and Newman, R.M., 2000, Relationship of wooded riparian zones and runoff potential to fish community composition in agricultural streams: *Canadian Journal of Fisheries and Aquatic Sciences*, v. 57, p. 307–316.
- Stoltenberg, D., and Pope, R., 1990, Atrazine management rules for Iowa: Iowa State University Extension Pamphlet Pm–1390, 2 p.
- Sullivan, D.J., 2000, Fish communities and their relation to environmental factors in the Eastern Iowa Basins in Iowa and Minnesota, 1995–96: U.S. Geological Survey Water-Resources Investigations Report 00–4195, 20 p.
- U.S. Department of Agriculture, 1999, Agricultural chemical usage, 1998 field crops summary: National Agricultural Statistics Service and Economic Research Service accessed August 1999 at URL <http://usda.mannlib.cornell.edu/reports/nassr/other/pcu-bb>
- U.S. Environmental Protection Agency, 1986, Quality criteria for water 1986: Washington, D.C., report 440/5–86–001, 453 p.
- _____, 1994, Acetochlor registration agreement and addendums, accessed August, 2000 at URL <http://www.epa.gov/oppefed1/aceto/regagree.htm>
- _____, 1996, Drinking water regulations and health advisories: Washington D.C., Report 822–R–96–001, 16 p.
- _____, 2000, Chlorpyrifos revised risk assessment and agreement with registrants: U.S. Environmental Protection Agency Factsheet, Prevention, pesticides and toxic substances (7506C), 4 p., accessed September 2000 at URL <http://www.epa.gov/pesticides/announcement6800.htm>
- Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R., and Cushing, C.E., 1980, The river continuum concept: *Canadian Journal of Fisheries and Aquatic Sciences*, v. 37, p. 130–137.
- Zogorski and others, 1997, Interagency assessment of oxygenated fuels, chap. 2. Fuel oxygenates and water quality: Washington D.C., Office of Science and Technology Policy, p. 2–1—2–80.

APPENDIX—WATER-QUALITY DATA FROM THE EASTERN IOWA BASINS IN A NATIONAL CONTEXT

For a complete view of Eastern Iowa Basins data and for additional information about specific benchmarks used, visit our Web site at <http://water.usgs.gov/nawqa/>. Also visit the NAWQA Data Warehouse for access to NAWQA data sets at <http://infotrek.er.usgs.gov/wdbctx/nawqa/nawqa.home>.

This appendix is a summary of chemical concentrations and biological indicators assessed in the Eastern Iowa Basins. Selected results for this Study Unit are graphically compared to results from as many as 36 NAWQA Study Units investigated from 1991 to 1998 and to national water-quality benchmarks for human health, aquatic life, or fish-eating wildlife. The chemical and biological indicators shown were selected on the basis of frequent detection, detection at concentrations above a national benchmark, or regulatory or scientific importance. The graphs illustrate how conditions associated with each land use sampled in the Eastern Iowa Basins compare to results from across the Nation, and how conditions compare among the several land uses. Graphs for chemicals show only detected concentrations and, thus, care must be taken to evaluate detection frequencies in addition to concentrations when comparing study-unit and national results. For example, acetochlor concentrations in Eastern Iowa Basins agricultural streams were similar to the national distribution, but the detection frequency was much higher (79 percent compared to 33 percent).

CHEMICALS IN WATER

Concentrations and detection frequencies, Eastern Iowa Basins, 1996–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals

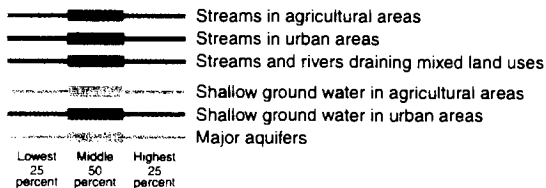
◆ Detected concentration in Study Unit

66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency

-- Not measured or sample size less than two

12 Study-unit sample size. For ground water, the number of samples is equal to the number of wells sampled

National ranges of detected concentrations, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

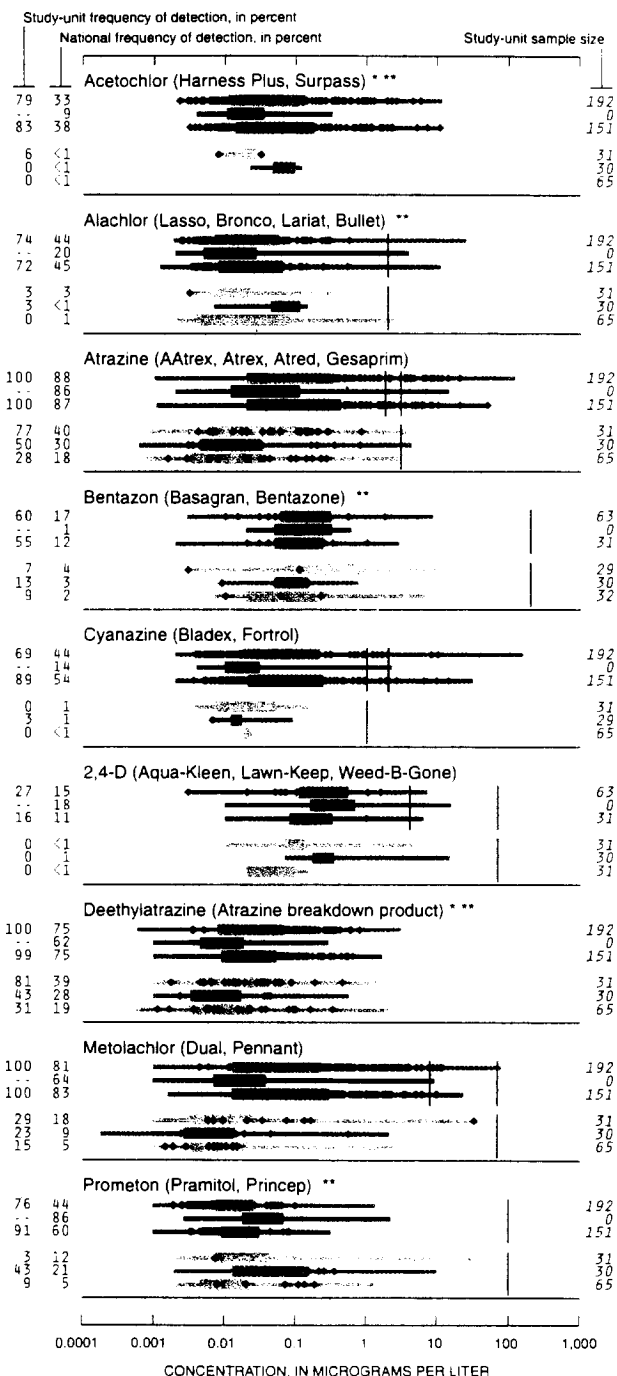


National water-quality benchmarks

National benchmarks include standards and guidelines related to drinking-water quality, criteria for protecting the health of aquatic life, and a goal for preventing stream eutrophication due to phosphorus. Sources include the U.S. Environmental Protection Agency and the Canadian Council of Ministers of the Environment

- | Drinking-water quality (applies to ground water and surface water)
- | Protection of aquatic life (applies to surface water only)
- | Prevention of eutrophication in streams not flowing directly into lakes or impoundments
- No benchmark for drinking-water quality
- No benchmark for protection of aquatic life

Pesticides in water—Herbicides



Other herbicides detected

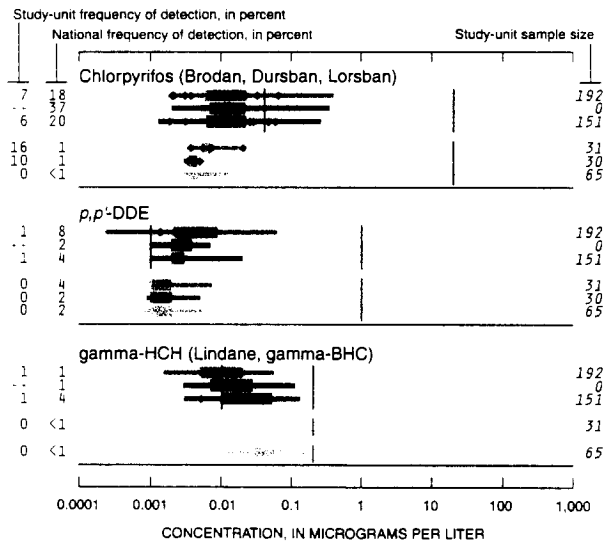
- Acifluorfen (Blazer, Tackle 2S) **
- Bromacil (Hyvar X, Urox B, Bromax)
- Bromoxynil (Buctril, Brominal) *
- Butylate (Sutan +, Genate Plus, Butilate) **
- DCPA (Dacthal, chlorthal-dimethyl) ***
- Dicamba (Banvel, Dianat, Scotts Proturf)

Dichlorprop (2,4-DP, Seritox 50, Lentemul) * **
 2,6-Diethylaniline (Alachlor breakdown product) * **
 Diuron (Crisuron, Karmex, Diurex) **
 EPTC (Eptam, Farmarox, Alirox) * **
 Metribuzin (Lexone, Sencor)
 Molinate (Ordram) * **
 Napropamide (Devrinol) * **
 Pendimethalin (Pre-M, Prowl, Stomp) * **
 Picloram (Grazon, Tordon)
 Pronamide (Kerb, Propyzamid) **
 Propachlor (Ramrod, Satecid) **
 Simazine (Princep, Caliber 90)
 Tebuthiuron (Spike, Tebusan)
 Triclopyr (Garlon, Grandstand, Redeem, Remedy) * **
 Trifluralin (Treflan, Gowan, Tri-4, Trific)

Herbicides not detected

Benfluralin (Balan, Benefin, Bonalan) * **
 Chloramben (Amiben, Amilon-WP, Vegiben) **
 Clopyralid (Stinger, Lontrel, Transline) * **
 2,4-DB (Butyrac, Butoxone, Embutox Plus, Embutone) * **
 Dacthal mono-acid (Dacthal breakdown product) * **
 Dinoseb (Dinosebe)
 Ethalfuralin (Sonalan, Curbit) * **
 Fenuron (Fenulon, Fendim) * **
 Fluometuron (Flo-Met, Cotoran) **
 Linuron (Lorox, Linex, Sarclax, Linurex, Afalon) *
 MCPA (Rhomene, Rhonox, Chiptox)
 MCPB (Thistrol) * **
 Neburon (Neburea, Neburyl, Noruben) * **
 Norflurazon (Evital, Predict, Solicam, Zorial) * **
 Oryzalin (Surflan, Dirimal) * **
 Pebulate (Tillam, PEBC) * **
 Propanil (Stam, Stampede, Wham) * **
 Propham (Tuberite) **
 2,4,5-T **
 2,4,5-TP (Silvex, Fenoprop) **
 Terbacil (Sinbar) **
 Thiobencarb (Bolero, Saturn, Benthicarb) * **
 Triallate (Far-Go, Avadex BW, Tri-allate) *

Pesticides in water—Insecticides



Other insecticides detected

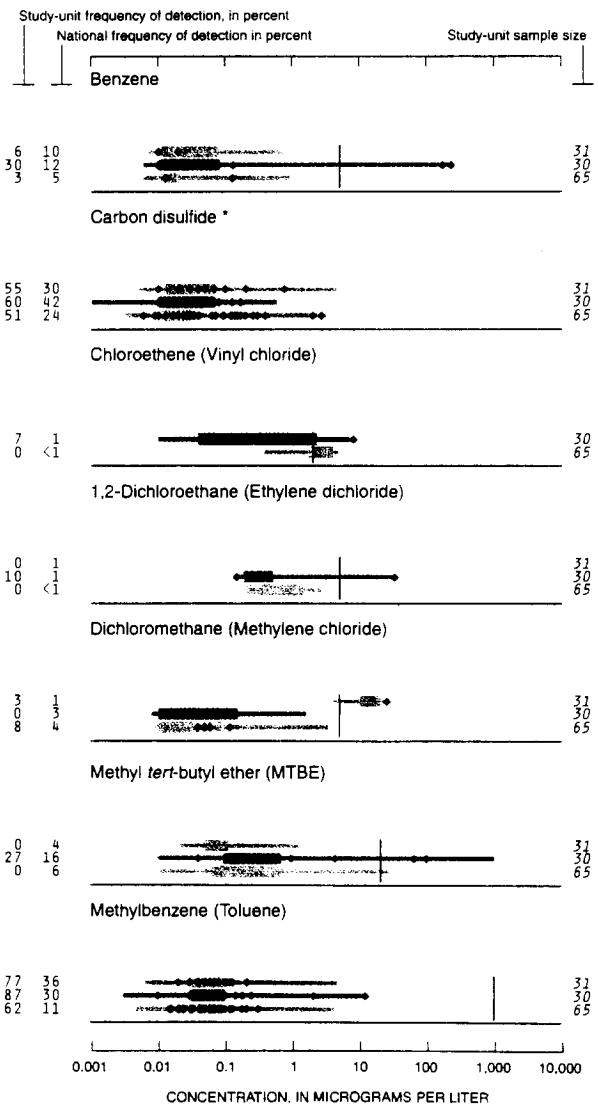
Carbaryl (Carbamine, Denapon, Sevin)
 Carbofuran (Furadan, Curaterr, Yaltox)
 Diazinon (Basudin, Diazatol, Neocidol, Knox Out)
 Dieldrin (Panoram D-31, Octalox, Compound 497)
 Fonofos (Dyfonate, Capfos, Cudgel, Tycap) **
 3-Hydroxycarbofuran (Carbofuran breakdown product) * **
 Malathion (Malathion)

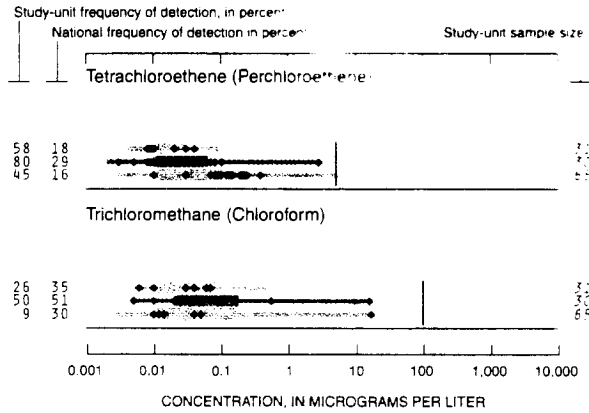
Insecticides not detected

Aldicarb (Temik, Ambush, Pounce)
 Aldicarb sulfone (Standak, aldoxycarb)
 Aldicarb sulfoxide (Aldicarb breakdown product)
 Azinphos-methyl (Guthion, Gusathion M) *
 Disulfoton (Disyston, Di-Syston) **
 Ethoprop (Mocap, Ethoprophos) * **
 alpha-HCH (alpha-BHC, alpha-lindane) **
 Methiocarb (Slug-Geta, Grandslam, Mesurol) * **
 Methomyl (Lanox, Lannate, Acinate) **
 Methyl parathion (Penncap-M, Folidol-M) **
 Oxamyl (Vydate L, Pratt) **
 Parathion (Roethyl-P, Alkron, Panthion, Phoskil) *
 cis-Permethrin (Ambush, Astro, Pounce) * **
 Phorate (Thimet, Granutox, Geomet, Rampart) * **
 Propargite (Comite, Omite, Ornamite) * **
 Propoxur (Baygon, Blattatex, Unden, Proprotox) * **
 Terbufos (Contraven, Counter, Pilarfox) **

Volatile organic compounds (VOCs) in ground water

These graphs represent data from 16 Study Units, sampled from 1996 to 1998





Other VOCs detected

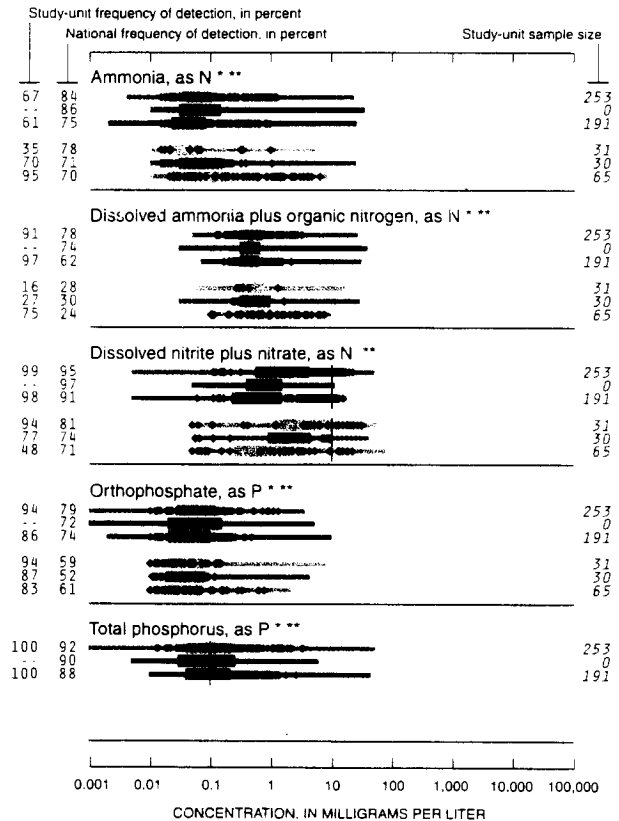
tert-Amylmethylether (*tert*-amyl methyl ether (TAME)) *
Bromodichloromethane (Dibromochloromethane)
Bromomethane (Methyl bromide)
2-Butanone (Methyl ethyl ketone (MEK)) *
n-Butylbenzene (1-Phenylbutane) *
sec-Butylbenzene *
tert-Butylbenzene *
Chlorobenzene (Monochlorobenzene)
Chlorodibromomethane (Dibromochloromethane)
Chloroethane (Ethyl chloride) *
Chloromethane (Methyl chloride)
1,3-Dichlorobenzene (*m*-Dichlorobenzene)
1,4-Dichlorobenzene (*p*-Dichlorobenzene)
Dichlorodifluoromethane (CFC 12, Freon 12)
1,1-Dichloroethane (Ethylidene dichloride) *
1,1-Dichloroethene (Vinylidene chloride)
trans-1,2-Dichloroethene ((E)-1,2-Dichloroethene)
cis-1,2-Dichloroethene ((Z)-1,2-Dichloroethene)
1,2-Dichloropropane (Propylene dichloride)
Diethyl ether (Ethyl ether) *
Diisopropyl ether (Diisopropylether (DIPE)) *
1,2-Dimethylbenzene (*o*-Xylene)
1,3 & 1,4-Dimethylbenzene (*m*-&*p*-Xylene)
1,4-Epoxy butane (Tetrahydrofuran, Diethylene oxide) *
Ethyl *tert*-butyl ether (Ethyl-*t*-butyl ether (ETBE)) *
1-Ethyl-2-methylbenzene (2-Ethyltoluene) *
Ethylbenzene (Phenylethane)
Iodomethane (Methyl iodide) *
Isopropylbenzene (Cumene) *
p-Isopropyltoluene (*p*-Cymene) *
Naphthalene
2-Propanone (Acetone) *
n-Propylbenzene (Isocumene) *
1,2,3,5-Tetramethylbenzene (Isodurene) *
Tribromomethane (Bromoform)
1,2,3-Trichlorobenzene *
1,1,1-Trichloroethane (Methylchloroform)
Trichloroethene (TCE)
1,2,3-Trichloropropane (Allyl trichloride)
1,2,3-Trimethylbenzene (Hemimellitene) *
1,2,4-Trimethylbenzene (Pseudocumene) *
1,3,5-Trimethylbenzene (Mesitylene) *

VOCs not detected

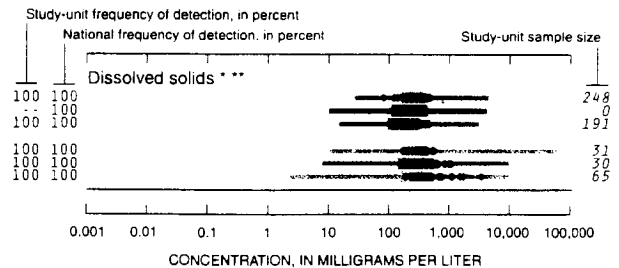
Bromobenzene (Phenyl bromide) *
Bromochloromethane (Methylene chlorobromide)
Bromoethene (Vinyl bromide) *
3-Chloro-1-propene (3-Chloropropene) *
1-Chloro-2-methylbenzene (*o*-Chlorotoluene)
1-Chloro-4-methylbenzene (*p*-Chlorotoluene)
1,2-Dibromo-3-chloropropane (DBCP, Nemagon)
1,2-Dibromoethane (Ethylene dibromide, EDB)
Dibromomethane (Methylene dibromide) *
trans-1,4-Dichloro-2-butene ((Z)-1,4-Dichloro-2-butene) *
1,2-Dichlorobenzene (*o*-Dichlorobenzene)
2,2-Dichloropropane *
1,3-Dichloropropane (Trimethylene dichloride) *
trans-1,3-Dichloropropene ((E)-1,3-Dichloropropene)

cis-1,3-Dichloropropene ((Z)-1,3-Dichloropropene)
1,1-Dichloropropene *
Ethenylbenzene (Styrene)
Ethyl methacrylate *
Hexachlorobutadiene
1,1,1,2,2,2-Hexachloroethane (Hexachloroethane)
2-Hexanone (Methyl butyl ketone (MBK)) *
Methyl acrylonitrile *
Methyl-2-methacrylate (Methyl methacrylate) *
4-Methyl-2-pentanone (Methyl isobutyl ketone (MIBK)) *
Methyl-2-propenoate (Methyl acrylate) *
2-Propenenitrile (Acrylonitrile)
1,1,2-Tetrachloroethane *
1,1,1,2-Tetrachloroethane
Tetrachloromethane (Carbon tetrachloride)
1,2,3,4-Tetramethylbenzene (Prehnitene) *
1,1,2-Trichloro-1,2,2-trifluoroethane (Freon 113) *
1,2,4-Trichlorobenzene
1,1,2-Trichloroethane (Vinyl trichloride)
Trichlorofluoromethane (CFC 11, Freon 11)

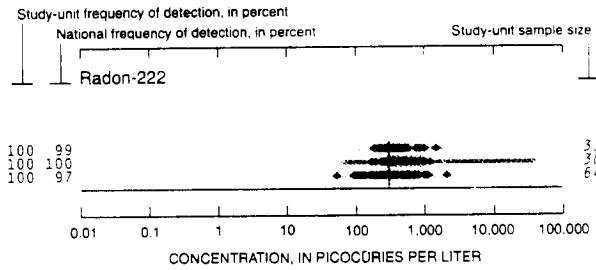
Nutrients in water



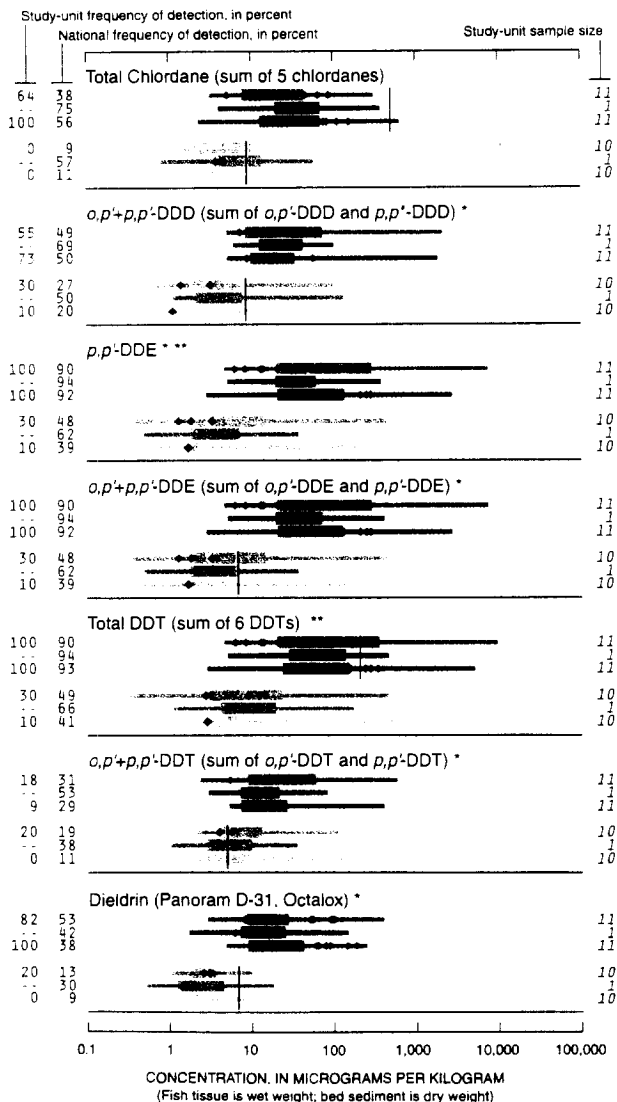
Dissolved solids in water



Trace elements in ground water



Organochlorines in fish tissue (whole body) and bed sediment

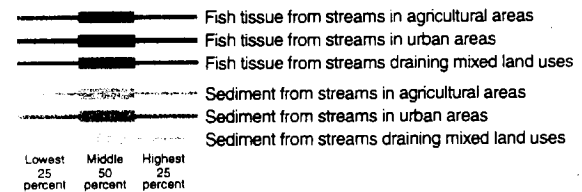


CHEMICALS IN FISH TISSUE AND BED SEDIMENT

Concentrations and detection frequencies, Eastern Iowa Basins, 1996-98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals. Study-unit frequencies of detection are based on small sample sizes; the applicable sample size is specified in each graph

- ◆ Detected concentration in Study Unit
- 66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- Not measured or sample size less than two
- 12 Study-unit sample size

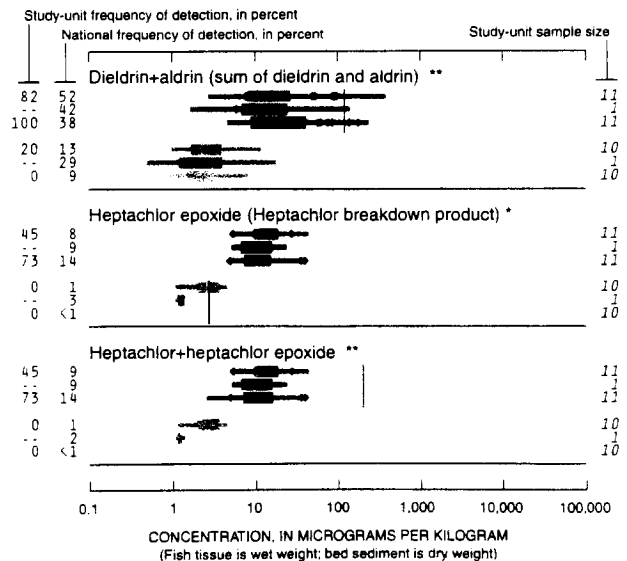
National ranges of concentrations detected, by land use, in 36 NAWQA Study Units, 1991-98—Ranges include only samples in which a chemical was detected

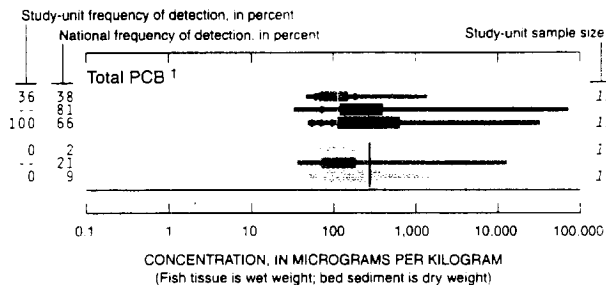


National benchmarks for fish tissue and bed sediment

National benchmarks include standards and guidelines related to criteria for protection of the health of fish-eating wildlife and aquatic organisms. Sources include the U.S. Environmental Protection Agency, other Federal and State agencies, and the Canadian Council of Ministers of the Environment

- | Protection of fish-eating wildlife (applies to fish tissue)
- | Protection of aquatic life (applies to bed sediment)
- No benchmark for protection of fish-eating wildlife
- ** No benchmark for protection of aquatic life





¹ The national detection frequencies for total PCB in sediment are biased low because about 30 percent of samples nationally had elevated detection levels compared to this Study Unit. See <http://water.usgs.gov/nawqa/> for additional information.

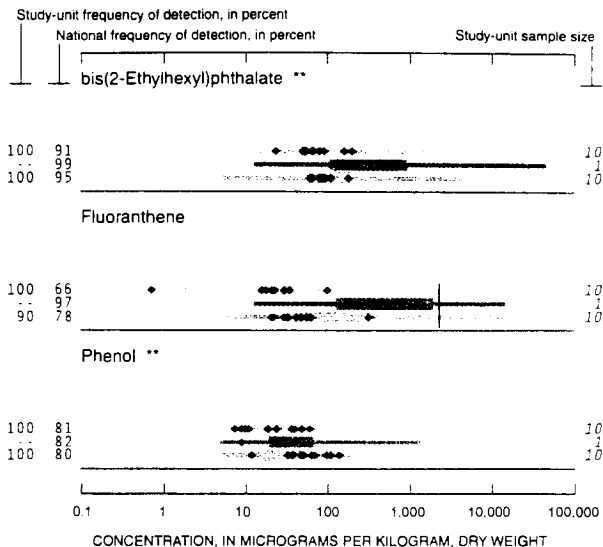
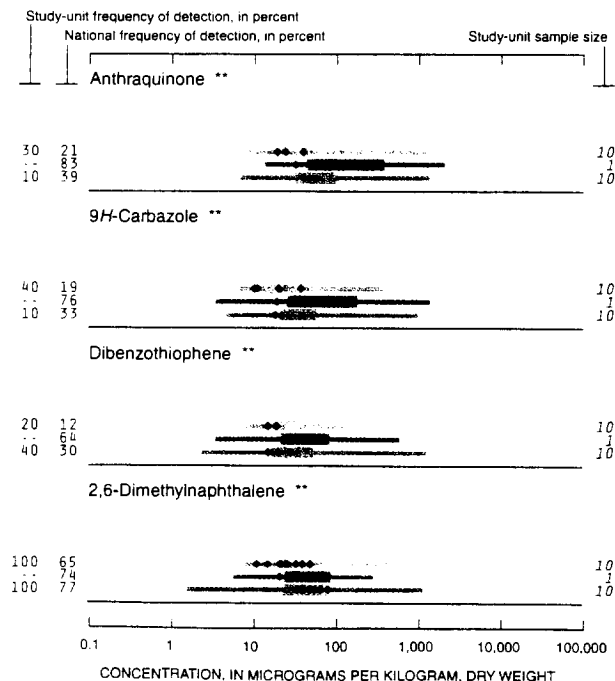
Other organochlorines detected

- DCPA (Dacthal, chlorthal-dimethyl) * **
- Total-HCH (sum of alpha-HCH, beta-HCH, gamma-HCH, and delta-HCH) **
- Mirex (Dechlorane) **
- Pentachloroanisole (PCA) * **

Organochlorines not detected

- Chloroneb (Chloronebe, Demosan) * **
- Endosulfan I (alpha-Endosulfan, Thiodan) * **
- Endrin (Endrine)
- gamma-HCH (Lindane, gamma-BHC, Gammexane) *
- Hexachlorobenzene (HCB) **
- Isodrin (Isodrine, Compound 711) * **
- p,p'-Methoxychlor (Marlate, methoxychlor) * **
- o,p'-Methoxychlor * **
- cis-Permethrin (Ambush, Astro, Pounce) * **
- trans-Permethrin (Ambush, Astro, Pounce) * **
- Toxaphene (Camphechlor, Hercules 3956) * **

Semivolatile organic compounds (SVOCs) in bed sediment



Other SVOCs detected

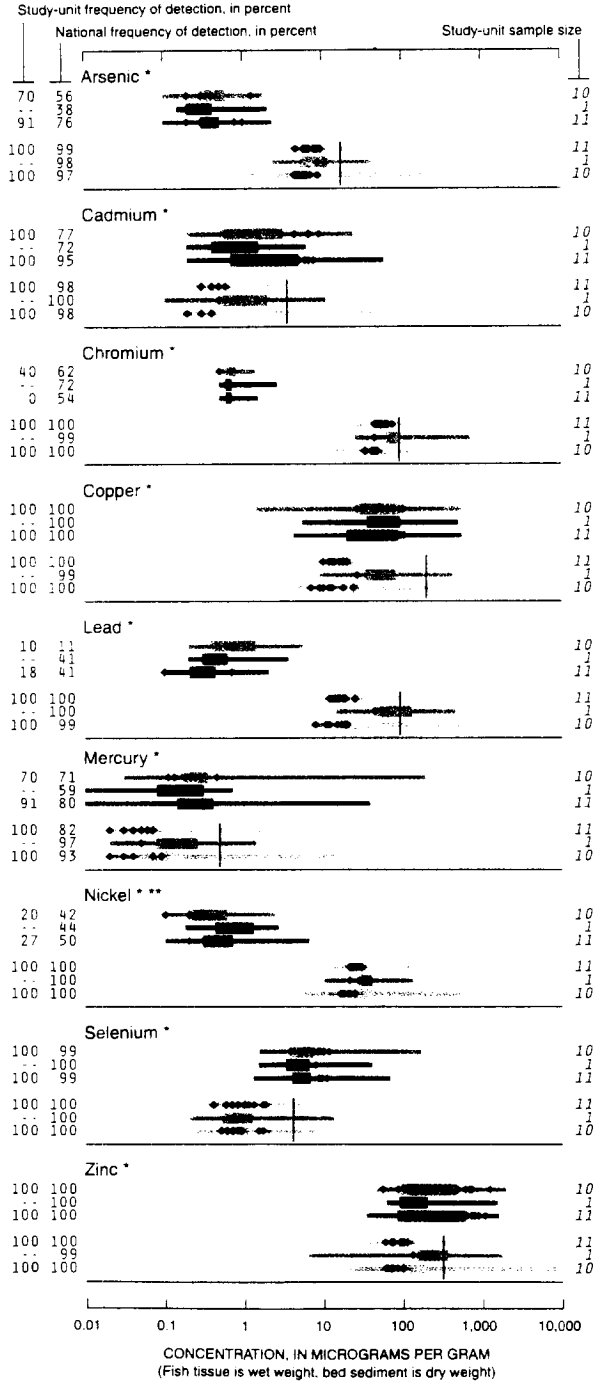
- Acenaphthene
- Acenaphthylene
- Acridine **
- Anthracene
- Benzo[a]anthracene
- Benzo[a]pyrene
- Benzo[b]fluoranthene **
- Benzo[ghi]perylene **
- Benzo[k]fluoranthene **
- Butylbenzylphthalate **
- Chrysene
- p-Cresol **
- Di-n-butylphthalate **
- Di-n-octylphthalate **
- Dibenz[a,h]anthracene
- Diethylphthalate **
- 1,2-Dimethylnaphthalene **
- 1,6-Dimethylnaphthalene **
- Dimethylphthalate **
- 2-Ethynaphthalene **
- 9H-Fluorene (Fluorene)
- Indeno[1,2,3-cd]pyrene **
- Isoquinoline **
- 1-Methyl-9H-fluorene **
- 2-Methylantracene **
- 4,5-Methylenephenanthrene **
- 1-Methylphenanthrene **
- 1-Methylpyrene **
- Naphthalene
- Phenanthrene
- Pyrene
- 2,3,6-Trimethylnaphthalene **

SVOCs not detected

- C8-Alkylphenol **
- Azobenzene **
- Benzo[c]cinnoline **
- 2,2-Biquinoline **
- 4-Bromophenyl-phenylether **
- 4-Chloro-3-methylphenol **
- bis(2-Chloroethoxy)methane **
- 2-Chloronaphthalene **
- 2-Chlorophenol **
- 4-Chlorophenyl-phenylether **
- 1,2-Dichlorobenzene (o-Dichlorobenzene) **
- 1,3-Dichlorobenzene (m-Dichlorobenzene) **
- 1,4-Dichlorobenzene (p-Dichlorobenzene) **
- 3,5-Dimethylphenol **
- 2,4-Dinitrotoluene **
- Isophorone **
- Nitrobenzene **

N-Nitrosodi-n-propylamine **
 N-Nitrosodiphenylamine **
 Pentachloronitrobenzene **
 Phenanthridine **
 Quinoline **
 1,2,4-Trichlorobenzene **

Trace elements in fish tissue (livers) and bed sediment



BIOLOGICAL INDICATORS

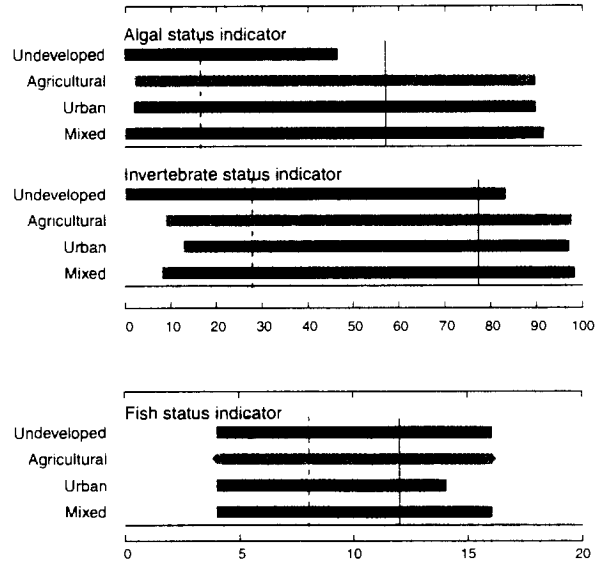
Higher national scores suggest habitat disturbance, water-quality degradation, or naturally harsh conditions. The status of algae, invertebrates (insects, worms, and clams), and fish provide a record of water-quality and stream conditions that water-chemistry indicators may not reveal. **Algal status** focuses on the changes in the percentage of certain algae in response to increasing siltation, and it often correlates with higher nutrient concentrations in some regions. **Invertebrate status** averages 11 metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. **Fish status** sums the scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that increase in association with water-quality degradation.

Biological indicator value, Eastern Iowa Basins, by land use, 1996-98

- ◆ Biological status assessed at a site

National ranges of biological indicators, in 16 NAWQA Study Units, 1994-98

- Streams in undeveloped areas
- Streams in agricultural areas
- Streams in urban areas
- Streams in mixed-land-use areas
- 75th percentile
- - - 25th percentile



A COORDINATED EFFORT

Coordination with agencies and organizations in the Eastern Iowa Basins was integral to the success of this water-quality assessment. We thank personnel from the following agencies and organizations who served as members of our liaison committee and participated in our liaison committee meetings.

U.S. Department of Agriculture Natural Resources Conservation Service Agricultural Research Service	University of Iowa Hygienics Laboratory Limnology Section
U.S. Environmental Protection Agency, Region VII	Environmental Research
U.S. Fish and Wildlife Service	Institute of Hydraulic Research
U.S. Geological Survey Biological Resources Division	Center for Health Effects on the Environment
Iowa Department of Agriculture and Land Stewardship Pesticide Bureau	University of Northern Iowa College of Natural Sciences
Iowa Department of Natural Resources Environmental Protection Division Geological Survey Bureau Fisheries Bureau	Wartburg College Biology Department
Minnesota Department of Natural Resources Minnesota Pollution Control Agency Minnesota Geological Survey	Iowa Farm Bureau Federation Iowa Environmental Council Iowa Ground Water Association
Linn County Iowa REAP	Izaak Walton League
Linn County Iowa Conservation Board	Sierra Club
Johnson County Iowa Board of Supervisors	Dow AgroSciences
Cedar Rapids Water Department	Dupont Agricultural Products
Iowa City Public Works	Novartis Crop Protection
Iowa State University Center for Sustainable Agriculture Department of Botany Extension Service	Monsanto American Cyanamid Company American Corn Growers Association Johnson County Farm Bureau

We thank the following individuals for contributing to this effort.

James Cerveny, Jon Nania, Joel Galloway, Jennifer Tobias, and Matthew Bobier, the primary field personnel, who worked in all conditions to obtain a high-quality data set without which this report would not be possible.

Linda Roberts, the Study Unit biologist, who planned and supervised the collection of algae, macroinvertebrate, and fish samples.

The many U.S. Geological Survey personnel (too many to list) from the Iowa District and surrounding States that assisted in the collection of biological samples and in the construction of equipment shelters.

Scott Yess and Ann Rundstrom of the Fish and Wildlife Service and J. Kent Johnson University of Iowa Hydraulics Institute, who assisted in the collection of fish.

Technicians and chemists at the USGS National Water-Quality Laboratory in Denver, Colorado, who provided analysis of all water samples. Mike Thurman and Elizabeth Scribner and the staff of the U.S. Geological Survey Organic Geochemistry Research laboratory in Lawrence, Kansas, who analyzed all pesticide samples for degradates. Researchers and technicians at the USGS Isotope Tracers Laboratory and the USGS National Research Program Laboratory, who provided tritium and stable isotope analysis.

Tom Wilton of the Iowa Department of Natural Resources who provided a wealth of information, data, and advice that was invaluable in the completion of the water-chemistry and biological parts of the study.

Reviewers—Michael Burkhart, USDA Soil Tilth Laboratory; Susan Heathcote, Iowa Environmental Council; Jim Ellerhoff, Iowa Department of Agriculture and Land Stewardship; Richard Robinson, Iowa Farm Bureau Federation; Bernard Hoyer, Iowa Department of Natural Resources; Geological Survey Bureau. The many colleagues in the U.S. Geological Survey whose technical suggestions substantially improved this report. The editorial reviewers, Lanna Combs and Betty Palcsak.

Editors and publication preparation: Alene Brogan, Mary Kidd, Robert Olmstead, and Ed Swibas.

NAWQA

National Water-Quality Assessment (NAWQA) Program Eastern Iowa Basins



Kalkhoff and others—Water Quality in the Eastern Iowa Basins
U.S. Geological Survey Circular 1210

ISBN 0-607-95415-9



9 780607 954159

R0024530

Water Quality in the Upper Mississippi River Basin

Minnesota, Wisconsin, South Dakota, Iowa, and North Dakota, 1995–98



U.S. Department of the Interior
U.S. Geological Survey

Circular 1211

R0024531

POINTS OF CONTACT AND ADDITIONAL INFORMATION

The companion Web site for NAWQA summary reports:

<http://water.usgs.gov/nawqa/>

Upper Mississippi River Basin contact and Web site:

USGS State Representative
U.S. Geological Survey
2280 Woodale Drive
Mounds View, MN 55112
e-mail: stark@usgs.gov
<http://mn.water.usgs.gov/umis/index.html>

National NAWQA Program:

Chief, NAWQA Program
U.S. Geological Survey
Water Resources Division
12201 Sunrise Valley Drive, M.S. 413
Reston, VA 20192
<http://water.usgs.gov/nawqa/>

Other NAWQA summary reports

River Basin Assessments

Albemarle-Pamlico Drainage Basin (Circular 1157)
Allegheny and Monongahela River Basins (Circular 1202)
Apalachicola-Chattahoochee-Flint River Basin (Circular 1164)
Central Arizona Basins (Circular 1213)
Central Columbia Plateau (Circular 1144)
Central Nebraska Basins (Circular 1163)
Connecticut, Housatonic and Thames River Basins (Circular 1155)
Eastern Iowa Basins (Circular 1210)
Georgia-Florida Coastal Plain (Circular 1151)
Hudson River Basin (Circular 1165)
Kanawha-New River Basins (Circular 1204)
Lake Erie-Lake Saint Clair Drainages (Circular 1203)
Las Vegas Valley Area and the Carson and Truckee River Basins (Circular 1170)
Lower Illinois River Basin (Circular 1209)
Long Island-New Jersey Coastal Drainages (Circular 1201)
Lower Susquehanna River Basin (Circular 1168)
Mississippi Embayment (Circular 1208)
Ozark Plateaus (Circular 1158)
Potomac River Basin (Circular 1166)
Puget Sound Basin (Circular 1216)

Red River of the North Basin (Circular 1169)
Rio Grande Valley (Circular 1162)
Sacramento River Basin (Circular 1215)
San Joaquin-Tulare Basins (Circular 1159)
Santee River Basin and Coastal Drainages (Circular 1206)
South-Central Texas (Circular 1212)
South Platte River Basin (Circular 1167)
Southern Florida (Circular 1207)
Trinity River Basin (Circular 1171)
Upper Colorado River Basin (Circular 1214)
Upper Snake River Basin (Circular 1160)
Upper Tennessee River Basin (Circular 1205)
Western Lake Michigan Drainages (Circular 1156)
White River Basin (Circular 1150)
Willamette Basin (Circular 1161)

National Assessments

The Quality of Our Nation's Waters—Nutrients and Pesticides (Circular 1225)

Front cover: Mississippi River, St. Paul, Minnesota (Bob Firth ©).

Back cover: Left, row crops in the Minnesota River Basin (Scott Murray Photography); middle, St. Paul skyline (Scott Murray Photography); right, St. Croix River valley (National Park Service).



News Release

U.S. Department of the Interior
U.S. Geological Survey

Address
Upper Mississippi River NAWQA
2280 Woodale Drive
Mounds View MN 55112

E-mail
stark@usgs.gov

Release Date
May 8, 2001

Contact
James Stark

Phone
(763) 783-3230

Fax
(612) 785-3103

Look at Surrounding Land, Soil, and Geology for Water Resource Quality

Water resources are affected by land use, soils, and geology in the Upper Mississippi River Basin, according to a recently published report by the U. S. Geological Survey (USGS). Urban, agricultural, and forested land use in combination with geology and soil type result in different degrees of water-resource quality. Surface-water quality was better in forested watersheds than in urban or agricultural watersheds. Ground-water quality was better in aquifers with overlying soils that reduced direct infiltration.

The study of the Upper Mississippi River Basin, conducted from 1995-98, as part of the USGS National Water-Quality Assessment Program, found that streams in forested watersheds like the St. Croix River Basin had less nutrients, sediment, and organic contaminants than streams in agricultural watersheds like the Minnesota River Basin or urban areas like the Twin Cities Metropolitan Area. "Land use a major factor affecting water quality in streams and ground water. However, the type of soils in the watershed or overlying shallow aquifers could have a significant effect", said Jim Stark, the project chief. The findings indicate that agriculture, urbanization, and natural land uses continue to affect water quality.

The quality of urban streams has been affected more than streams in agricultural or forested areas. One component of the study examined biological communities and habitat. Urban streams had reduced fish and invertebrate species richness (numbers of species) and biological diversity compared to agricultural and forested streams. Several factors may be responsible for the comparatively poor condition of aquatic resources in urban streams, including contaminants in the water and sediment, and reduced habitat.

Reduced habitat in urban streams was associated with loss of riparian vegetation, channelization, impoundments, and changes in the pattern of streamflow. The rise and fall of the water in urban streams after rainstorms or snowmelt are much more rapid than streams in other land-use settings. Impervious surfaces such as parking lots, roads, and building roofs do not allow precipitation to soak into the ground; therefore, runoff reaches the streams more quickly and streamflow increases and decreases rapidly. Higher flows scour the stream bottom, reducing habitat.

Agricultural streams in the study were found to receive nonpoint-source inputs of nutrients and sediment. Nitrate concentrations in streams in artificially drained agricultural areas exceeded the U.S. Environmental Protection Agency's (USEPA) drinking water standard of 10 milligrams per liter in about 20 percent of the samples. Stream channels and habitat were also affected by channelization and loss of riparian buffers. The importance of riparian buffers for agricultural streams was shown with a special study. This study found that streams with more trees and shrubs along the banks in riparian buffer zones had more fish and invertebrate species and better ecological integrity than streams without riparian buffer zones.

Streams in forested basins like those in the northern part of the study area were relatively unaffected except by localized activities such as minor wastewater effluent discharges, pesticide applications at tree farms, leaks from septic systems. Nutrients and pesticides did not exceed drinking water standards or criteria to protect aquatic life. Concentrations of nutrients and pesticides were

small compared to urban and agricultural streams.

Ground water supplies about 75 percent of the public and industrial water supply. The major source for this water is the Prairie du Chien-Jordan aquifer. The local quality of the water in the aquifer depends on land use and the type of bedrock or glacial deposits above it. In areas where water can readily seep through the overlying soils to the aquifer, 8 percent of the nitrate samples exceeded the USEPA drinking water standards. In areas where water could not readily reach the aquifer due to low-permeability glacial tills, nitrate standards were not exceeded. Some pesticides were detected in ground water. In areas overlain by till, atrazine (a herbicide used with corn) was detected in 36 percent of the wells, whereas in areas without a low-permeability layer, it was detected in 52 percent of the wells.

The USGS report, "Water Quality in the Upper Mississippi River Basin, Minnesota, Wisconsin, South Dakota, Iowa, and North Dakota, 1995-1998", published as USGS Circular 1211, is available on the World Wide Web as downloadable portable document files (PDF) at <http://water.usgs.gov/nawqa> or in printed form (single copies of the report are at no cost) from Branch of Information Services, P.O. Box 25286, Denver, CO 80225, or by fax request to 303-202-4693. Please specify USGS report C-1211.

The USGS assessment is part of a national program, currently releasing results on surface and ground water in 15 additional major river basins. Check the status and availability of the individual basin reports on the NAWQA website, as well as accessibility to other NAWQA publications and national data sets and maps.

As the nation's largest water, earth and biological science and civilian mapping agency, the USGS works in cooperation with more than 2,000 organizations across the country to provide reliable, impartial, scientific information to resource managers, planners, and other customers. This information is gathered in every state by USGS scientists to minimize the loss of life and property from natural disaster, contribute to the sound conservation, economic and physical development of the nation's natural resources, and enhance the quality of life by monitoring water, biological, energy, and mineral resources.

Selected Findings and Current Perspectives on Urban and Agricultural Water Quality by the National Water-Quality Assessment Program

Studies by the USGS National Water-Quality Assessment (NAWQA) program in the last decade describe water-quality conditions in nearly 120 agricultural and 35 urban watersheds ("urban" primarily refers to residential and commercial development over the last 50 years). The findings show that for both urban and agricultural areas, nonpoint chemical contamination is an issue. Much work still needs to be done in urban areas with point source contamination as well, including infrastructure improvements. Appreciable improvements in overall water quality, however, will depend upon effective management of point and nonpoint sources. The findings also show that water-quality conditions and aquatic health reflect a complex combination of land and chemical use, land-management practices, population density and watershed development, and natural features, such as soils, geology, hydrology, and climate. Contaminant concentrations vary from season to season and from watershed to watershed. Even among seemingly similar land uses and sources of contamination, different areas can have very different degrees of vulnerability and, therefore, have different rates at which improved treatment or management can lead to water-quality improvements.

Water Quality in Agricultural Watersheds

- Nitrogen and phosphorus in surface water commonly exceed levels that contribute to excessive algae. For example, average annual concentrations of phosphorus in nearly 80 percent of streams sampled in agricultural areas were greater than the U. S. Environmental Protection Agency (USEPA) desired goal for preventing nuisance plant growth in streams. Excessive plant growth can lead to low dissolved oxygen, which can be harmful to fish and other aquatic life.
- Nitrate is often elevated above background levels in shallow ground water underlying farmland. Concentrations in about 20 percent of shallow wells sampled in agricultural areas exceeded the USEPA drinking water standard. This result is a concern in rural areas where shallow ground water is used for domestic supply; these domestic wells are not regulated and owners often do not know the quality of their well water or whether their wells are vulnerable to contamination. Nitrate is most often elevated in karst (carbonate) areas or where soils and aquifers consist of sand and gravel. These natural features enable rapid infiltration and downward movement of water and chemicals. Some of the more vulnerable areas are the Central Valley of California, and parts of the Pacific Northwest, the Great Plains, and the Mid-Atlantic region. In contrast, ground-water contaminants underlying farmland in parts of the upper Midwest are barely detectable, despite similar high rates of chemical use. In these areas ground-water contamination may be limited because of relatively impermeable, poorly drained soils and glacial till that cover much of the region, and because tile drains provide quick pathways for runoff to streams.
- Pesticides are widespread. At least one pesticide was detected in more than 95 percent of stream samples. Pesticides were detected in more than 60 percent of shallow wells sampled in agricultural areas.
- Pesticides commonly occur in mixtures. Two-thirds of stream samples collected in agricultural areas contained 5 or more pesticides, and more than one-quarter of the samples contained 10 or more. Ground water contained fewer pesticides; about 30 percent of the wells sampled contained 2 or more.
- Concentrations of pesticides generally are low and below drinking-water standards. However, the risk to humans and the environment from present-day low levels of contaminant exposure remains unclear. For example, current standards and guidelines do not yet account for exposure to mixtures, and many pesticides and their breakdown products do not have standards or guidelines.
- Herbicides—most commonly atrazine and its breakdown product desethylatrazine, and metolachlor, cyanazine, and alachlor—occur more frequently and usually at higher concentrations in agricultural streams and ground water than in urban waters. Their occurrence is linked to their use; they rank in the top five in national herbicide use for agriculture.
- Insecticides that were used in the past still persist in agricultural streams and sediment. DDT was the most commonly detected organochlorine compound, followed by dieldrin and chlordane. Their uses were restricted in the 1970s and 1980s and, yet, more than 20 years later, one or more sediment-quality guidelines were exceeded at more than 20 percent of agricultural sites.

Water Quality in Urban Watersheds

- Concentrations of fecal coliform bacteria commonly exceed recommended standards for water-contact recreation.
- Concentrations of total phosphorus are generally as high in urban streams as in agricultural streams. More than 70 percent of sampled urban streams exceeded the USEPA desired goal for preventing nuisance plant growth.
- Insecticides, such as diazinon, carbaryl, chlorpyrifos, and malathion, occur more frequently, and usually at higher concentrations in urban streams than in agricultural streams. Concentrations are low in urban streams, rarely exceeding USEPA drinking-water standards. However, effects on aquatic life may be more of a concern. Concentrations of insecticides exceeded at least one guideline established to protect aquatic life in every sampled urban stream.
- Herbicides are widespread in surface water (detected in 99 percent of urban stream samples) and ground water (detected in more than 50 percent of sampled wells). Most common are those applied to lawns, golf courses, and road right-of-ways, such as atrazine, simazine, and prometon.
- Similar to agricultural areas, pesticides in urban waters commonly occur in mixtures; nearly 80 percent of stream samples contained 5 or more pesticides. Two of the most commonly detected insecticides in mixtures were diazinon and chlorpyrifos; common herbicides detected were simazine and prometon.
- Sediment in urban streams is associated with higher frequencies of occurrence of DDT, chlordane, and dieldrin and higher concentrations of chlordane and dieldrin than sediment in agricultural streams. Sediment-quality guidelines for organochlorine pesticides were exceeded at 36 percent of sampled urban sites.
- Volatile organic compounds, which are used in plastics, cleaning solvents, gasoline, and industrial operations, occur widely in shallow urban ground water. Some of the most frequently detected of the 60 analyzed compounds were the commercial and industrial solvents trichloroethene (TCE), tetrachloroethene (PCE), and methylene chloride; the gasoline additive methyl tert-butyl ether (MTBE); and the solvent and disinfection by-product of water treatment, trichloromethane (also known as chloroform).
- Concentrations of selected trace elements, such as cadmium, lead, zinc, and mercury, are elevated above background levels in populated urban settings, most likely caused by emissions from industrial and municipal activities and motor vehicles. Sediment cores from streambeds and reservoirs, which can be used to track changes over long time periods, indicate that lead increased from 1940s to the 1970s, and began to decrease after it was removed from gasoline. Concentrations are not yet down to background levels. Decreases also are noted for DDT and chlordane.
- In contrast to lead, DDT, and chlordane, sediment cores indicate that zinc and polycyclic aromatic hydrocarbons (PAHs, which result from fossil fuel combustion) are increasing. These increases most likely relate to increasing motor vehicle traffic in watersheds. Sediment-quality guidelines for PAHs were exceeded at more than 40 percent of urban sites.
- Toxic compounds in streambed sediment in urban areas, such as DDT, chlordane, dieldrin, and PCBs, also were found in fish tissue, often at higher concentrations than in the sediment. One or more organochlorine compounds were detected in 97 percent of whole fish samples collected at urban sites, and PCBs were detected in more than 80 percent of whole fish samples. Concentrations of organochlorine compounds exceeded guidelines to protect wildlife at more than 10 percent of urban sites; wildlife guidelines for PCBs were exceeded at nearly 70 percent of urban sites. These findings have contributed to decisions by some states to issue fish-consumption advisories.
- Deteriorated water quality and sediment, as well as habitat disturbances, contribute to degraded biological communities in urban streams. The greatest effects are seen in areas with the highest human population densities and watershed development. Pollution-tolerant algae and aquatic invertebrates (such as worms and midges), as well as omnivorous fish communities, prevail at the affected sites.

Contacts for additional information or questions:

Tim Miller (703) 648-6868 (tlmiller@usgs.gov)
Pixie Hamilton (804) 261-2602 (pahamilt@usgs.gov)

For Internet access to NAWQA publications, data, and maps:

<http://water.usgs.gov/nawqa>

R0024536

Upper Mississippi River Basin

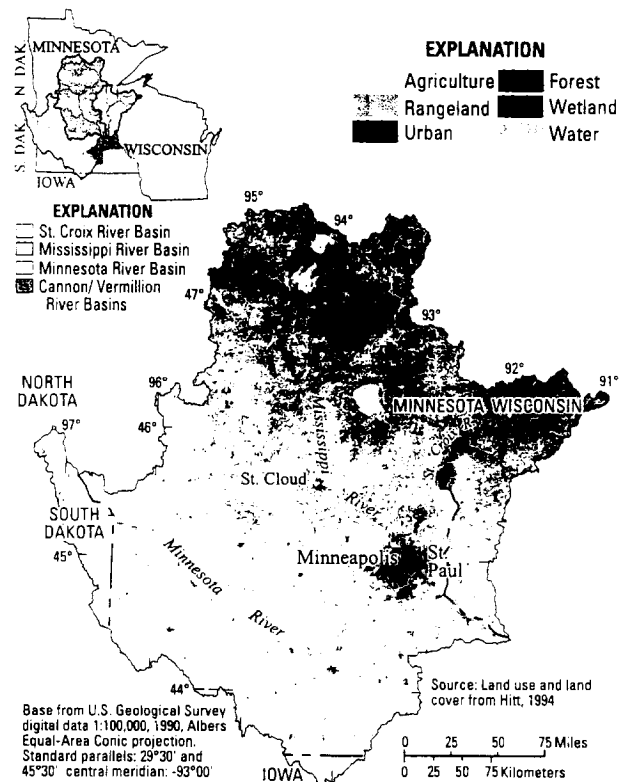
SUMMARY OF MAJOR FINDINGS

The Upper Mississippi River Basin Study Unit encompasses about 47,000 mi² (square miles) in Minnesota, Wisconsin, South Dakota, Iowa, and North Dakota and includes the Twin Cities (Minneapolis and St. Paul) metropolitan area (TCMA). The three major rivers in the Study Unit are the Mississippi, the Minnesota, and the St. Croix. In 1990, about 3.7 million people resided in the Study Unit, mostly in the TCMA. The Mississippi River is the primary source of drinking water for St. Cloud, Minneapolis, and St. Paul in Minnesota. Ground water is the primary source of drinking water in rural and suburban areas.

Highlights of Streams and Aquatic Biology

Elevated concentrations of nutrients (nitrogen and phosphorus) in water are potentially harmful to humans, livestock, and aquatic life. Major sources of nutrients to streams are commercial fertilizers applied to crops, lawns, and gardens; wastewater discharge; leaking septic systems; snowmelt runoff; and animal manure. The total amounts of nitrate and dissolved orthophosphate were greater in streams draining agricultural areas than in streams draining areas with other land uses. Although pesticides (herbicides and insecticides) were commonly detected, most concentrations were less than current drinking-water standards and guidelines and aquatic-life guidelines; however, not all pesticides detected currently have drinking-water standards and guidelines. Samples from most streams in the Study Unit met Federal and State drinking-water standards and guidelines and aquatic-life guidelines. Invertebrate and fish communities were most degraded in urban streams.

- Nitrate concentrations in streams in artificially drained agricultural areas exceeded the U.S. Environmental Protection Agency (USEPA) drinking-water standard of 10 mg/L (milligrams per liter) in about 20 percent of the samples.
- Insecticides and nonagricultural herbicides were detected most frequently in urban areas.
- Agricultural herbicides were detected in streams throughout the Study Unit.
- Urban streams have reduced invertebrate and fish species richness and diversity compared to agricultural streams.
- Algal productivity was greater in agricultural streams than in urban and forest streams, due in part to greater concentrations of nutrients.
- Agricultural streams with wooded riparian cover had greater fish and invertebrate species richness and diversity than agricultural streams lacking wooded riparian cover.



Land use and land cover in the Upper Mississippi River Basin study unit

Trends in Stream-Water Quality and Aquatic Biology

Assessing trends in water quality and aquatic biology is difficult because historical data sets are discontinuous and sampling objectives and analysis methods have varied. Some observable trends are increased nitrate concentrations, based on historical data, and decreased ammonia concentrations in streams in the TCMA during 1984–1993 primarily because of process changes at wastewater treatment facilities. Breakdown products of the pesticide DDT, the use of which was discontinued in the 1970's, are still detectable in fish, streams, and streambed sediment.

Major Influences on Streams and Aquatic Biology

- Application of pesticides and fertilizers in agricultural and urban areas
- Discharges from wastewater treatment facilities
- Runoff from agricultural and urban areas
- Stream modifications and artificial drainage
- Destruction of riparian cover along streambanks
- Contaminants in precipitation and in the atmosphere

Highlights of Conditions in Ground Water

Shallow ground water in the TCMA (less than 50 feet below land surface) commonly contained pesticides, nutrients, and industrial chemicals and detectable concentrations of numerous volatile organic compounds (VOCs). Deeper ground water, typically used for public supply (water supplied for the general public by municipal and private purveyors), contained few pesticides and lower nitrate concentrations. With the exception of naturally occurring radon, deeper ground water met drinking-water standards and guidelines for most chemicals.

- Nitrate concentrations in water from nearly one-half of shallow ground water sampled beneath agricultural areas exceeded the USEPA drinking-water standard (10 mg/L).
- Road salt constituents (sodium and chloride) were detected at greater concentrations in shallow ground water underlying urban areas than other areas.
- Agricultural pesticides were commonly detected in all land-use settings. Concentrations were greatest in agricultural areas.
- Atrazine was the most frequently detected agricultural pesticide. Concentrations were greater in shallow ground water than in deeper ground water.

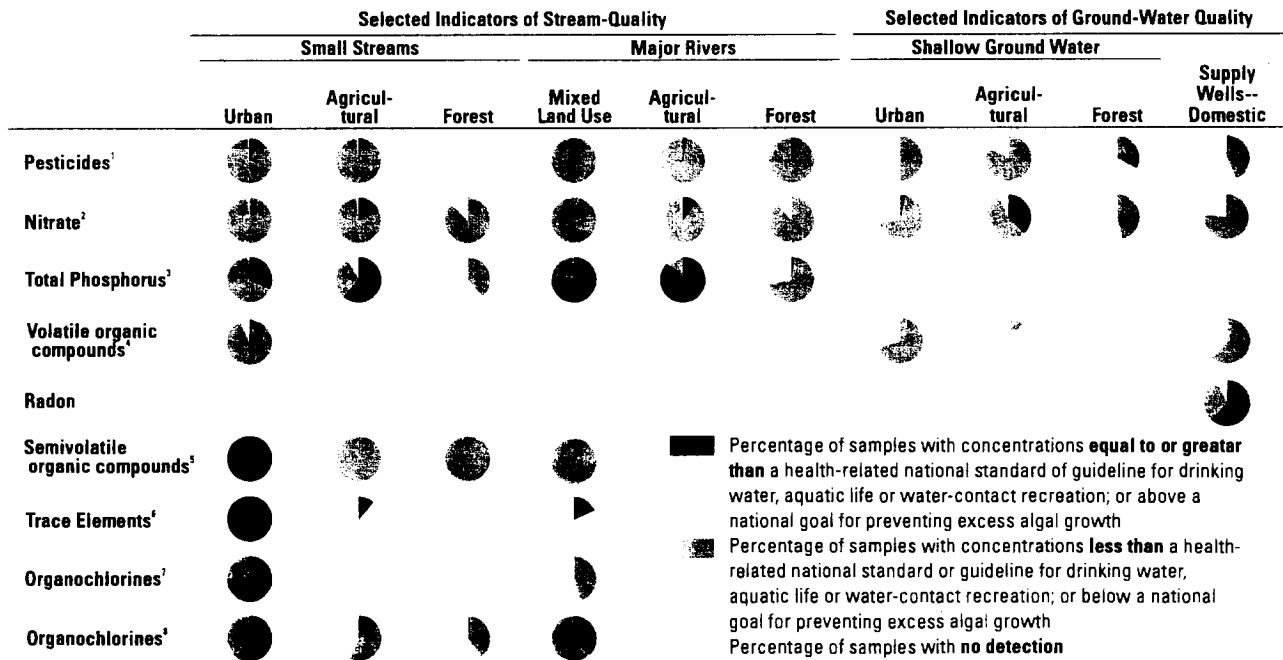
- Prometon was the most frequently detected herbicide in urban areas.
- Ground water in the Prairie du Chien-Jordan aquifer, an important source of drinking water, is protected by overlying confining units in some areas. Concentrations of nitrate, atrazine, and VOCs were lower in these areas than where confining units were absent.
- Radon exceeded the USEPA suspended drinking water standard of 300 pCi/L (picocuries per liter) in more than one-half of the water samples from the Prairie du Chien-Jordan aquifer.

Trends in Ground-Water Quality

Temporal trends in ground-water quality are difficult to define because limited information exists. Spatial trends include greater nitrate and pesticide concentrations in agricultural areas, greater VOC concentrations in urban areas, and few detections of pesticides or VOCs in forested areas.

Major Influences on Ground Water

- Application of pesticides and fertilizers
- Confining units and depth to water
- Urban contaminants (road salts, VOCs)
- Naturally occurring radon gas



¹Insecticides, herbicides, and pesticide metabolites, sampled in water. ²Nitrate (as nitrogen), sampled in water. ³Total phosphorus, sampled in water. ⁴Solvents, refrigerants, fumigants, and gasoline compounds, sampled in water. ⁵Byproducts of fossil-fuel combustion or components of coal and crude oil, sampled in sediment. ⁶Arsenic, mercury, and metals, sampled in sediment. ⁷Organochlorine compounds including DDT and PCBs, sampled in sediment. ⁸Organochlorine compounds including DDT and PCBs, sampled in fish tissue.

Water Quality in the Upper Mississippi River Basin, Minnesota, Wisconsin, South Dakota, Iowa, and North Dakota, 1995–98

By J.R. Stark, P.E. Hanson, R.M. Goldstein, J.D. Fallon, A.L. Fong, K.E. Lee, S.E. Kroening, *and* W.J. Andrews

U.S. DEPARTMENT OF THE INTERIOR
GALE A. NORTON, SECRETARY

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

Reston, Virginia
2000

Free on application to the
U.S. Geological Survey
Information Services
Box 25286 Federal Center
Denver, CO 80225

Or call: 1-888-ASK-USGS

Library of Congress Cataloging-in-Publications Data

Water quality in part of the Upper Mississippi Basin, Minnesota, Wisconsin, South Dakota, Iowa, and North Dakota, 1995-98 / by Stark, J.R....[et al.].

p. cm. -- (U.S. Geological Survey Circular ; 1211)

Includes bibliographical references.

ISBN 0-607-95416-7 (alk. paper)

1. Water quality--Mississippi River Watershed. I. Stark, James R. II. Geological Survey (U.S.) III. Series.

TD224.M64 W38 2000
363.739'42'0977--dc21

00-049466

R0024540

CONTENTS

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM	IV
SUMMARY OF MAJOR FINDINGS	1
Highlights of Streams and Aquatic Biology	1
Highlights of Conditions in Ground Water	2
INTRODUCTION TO THE UPPER MISSISSIPPI RIVER BASIN	3
Natural Factors Affect Water Quality and Aquatic Biology	3
Human Activities Affect Water Quality and Aquatic Biology	4
MAJOR FINDINGS	5
Land Use Influences Water Quality and Aquatic Biology	5
NATIONAL PERSPECTIVE—NITRATE IN A NATIONAL CONTEXT	7
RIPARIAN BUFFER ZONES AFFECT THE QUALITY OF MIDWESTERN STREAMS AND RIVERS	9
NATIONAL PERSPECTIVE—CHLORIDE IN A NATIONAL CONTEXT	11
NATIONAL PERSPECTIVE—ORGANIC CONTAMINANTS IN A NATIONAL CONTEXT	14
NATIONAL PERSPECTIVE—URBAN BIOLOGICAL COMMUNITIES IN A NATIONAL CONTEXT	15
STUDY UNIT DESIGN	23
GLOSSARY	25
REFERENCES	26
APPENDIX —WATER QUALITY DATA FROM THE UPPER MISSISSIPPI RIVER BASIN	29

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

THIS REPORT summarizes major findings about water quality in part of the Upper Mississippi River Basin (referred to as the Study Unit in this report) that emerged from an assessment conducted from 1995 to 1998 by the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program. Water quality is discussed in terms of local and regional issues and compared to conditions in 36 NAWQA study areas, called study units, assessed to date. Findings are also explained in the context of selected national benchmarks, such as those for drinking-water quality and the protection of aquatic organisms. The NAWQA Program was not intended to assess the quality to the Nation's drinking water, such as by monitoring water from household taps. Rather, the assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water monitoring programs. The comparisons made in this report to drinking-water standards and guidelines are only in the context of the available untreated resource. Finally, this report includes information about the status of aquatic communities and the condition of in-stream habitat as elements of the complete water-quality assessment.

Many topics covered in this report reflect the concerns of officials of State and Federal agencies, water-resource managers, and members of stakeholder groups who provided advice and input during the Study Unit assessment. Basin residents who wish to know more about water quality in the areas where they live will find this report informative as well.



THE NAWQA PROGRAM seeks to improve scientific and public understanding of water quality in the Nation's major river basins and ground-water systems. Better understanding encourages effective resource management, accurate identification of water-quality priorities, and successful development of strategies that protect and restore water quality. Guided by a nationally consistent study design and shaped by ongoing communication with local, State, and Federal agencies, NAWQA assessments support the investigation of local issues and trends, while providing a firm foundation for understanding water quality at regional and national scales. The ability to integrate local and national scales of data collection and analysis is a unique feature of the NAWQA Program.

The Upper Mississippi River Basin Study Unit is one of 51 water-quality assessments initiated since 1991, when the U.S. Congress appropriated funds for the USGS to begin the NAWQA Program. As indicated on the map, 36 assessments have been completed, and 15 assessments will conclude in 2001. Collectively, these assessments cover about one-half of the land area of the United States and include water resources that are available to more than 60 percent of the U.S. population.

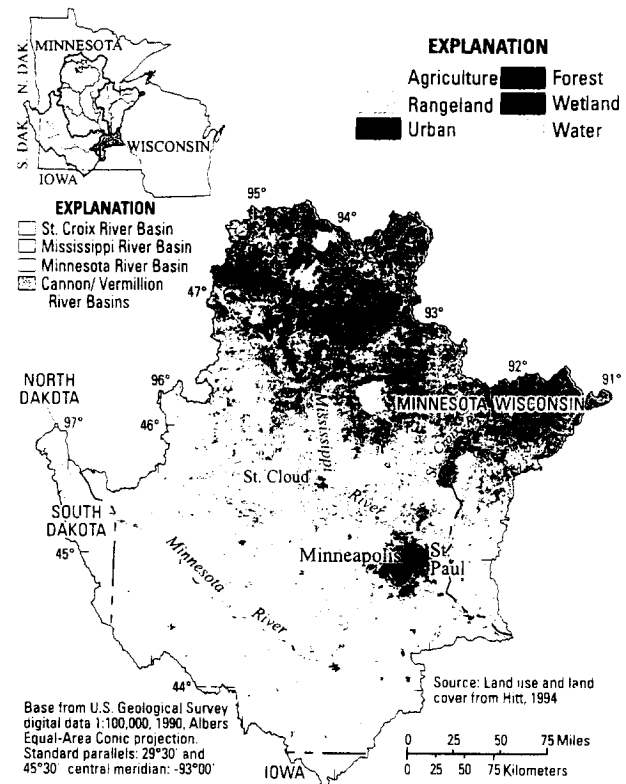
SUMMARY OF MAJOR FINDINGS

The Upper Mississippi River Basin Study Unit encompasses about 47,000 mi² (square miles) in Minnesota, Wisconsin, South Dakota, Iowa, and North Dakota and includes the Twin Cities (Minneapolis and St. Paul) metropolitan area (TCMA). The three major rivers in the Study Unit are the Mississippi, the Minnesota, and the St. Croix. In 1990, about 3.7 million people resided in the Study Unit, mostly in the TCMA. The Mississippi River is the primary source of drinking water for St. Cloud, Minneapolis, and St. Paul in Minnesota. Ground water is the primary source of drinking water in rural and suburban areas.

Highlights of Streams and Aquatic Biology

Elevated concentrations of nutrients (nitrogen and phosphorus) in water are potentially harmful to humans, livestock, and aquatic life. Major sources of nutrients to streams are commercial fertilizers applied to crops, lawns, and gardens; wastewater discharge; leaking septic systems; snowmelt runoff; and animal manure. The total amounts of nitrate and dissolved orthophosphate were greater in streams draining agricultural areas than in streams draining areas with other land uses. Although pesticides (herbicides and insecticides) were commonly detected, most concentrations were less than current drinking-water standards and guidelines and aquatic-life guidelines; however, not all pesticides detected currently have drinking-water standards and guidelines. Samples from most streams in the Study Unit met Federal and State drinking-water standards and guidelines and aquatic-life guidelines. Invertebrate and fish communities were most degraded in urban streams.

- Nitrate concentrations in streams in artificially drained agricultural areas exceeded the U.S. Environmental Protection Agency (USEPA) drinking-water standard of 10 mg/L (milligrams per liter) in about 20 percent of the samples.
- Insecticides and nonagricultural herbicides were detected most frequently in urban areas.
- Agricultural herbicides were detected in streams throughout the Study Unit.
- Urban streams have reduced invertebrate and fish species richness and diversity compared to agricultural streams.
- Algal productivity was greater in agricultural streams than in urban and forest streams, due in part to greater concentrations of nutrients.
- Agricultural streams with wooded riparian cover had greater fish and invertebrate species richness and diversity than agricultural streams lacking wooded riparian cover.



Land use and land cover in the Upper Mississippi River Basin study unit

Trends in Stream-Water Quality and Aquatic Biology

Assessing trends in water quality and aquatic biology is difficult because historical data sets are discontinuous and sampling objectives and analysis methods have varied. Some observable trends are increased nitrate concentrations, based on historical data, and decreased ammonia concentrations in streams in the TCMA during 1984–1993 primarily because of process changes at wastewater treatment facilities. Breakdown products of the pesticide DDT, the use of which was discontinued in the 1970's, are still detectable in fish, streams, and streambed sediment.

Major Influences on Streams and Aquatic Biology

- Application of pesticides and fertilizers in agricultural and urban areas
- Discharges from wastewater treatment facilities
- Runoff from agricultural and urban areas
- Stream modifications and artificial drainage
- Destruction of riparian cover along streambanks
- Contaminants in precipitation and in the atmosphere

Highlights of Conditions in Ground Water

Shallow ground water in the TCMA (less than 50 feet below land surface) commonly contained pesticides, nutrients, and industrial chemicals and detectable concentrations of numerous volatile organic compounds (VOCs). Deeper ground water, typically used for public supply (water supplied for the general public by municipal and private purveyors), contained few pesticides and lower nitrate concentrations. With the exception of naturally occurring radon, deeper ground water met drinking-water standards and guidelines for most chemicals.

- Nitrate concentrations in water from nearly one-half of shallow ground water sampled beneath agricultural areas exceeded the USEPA drinking-water standard (10 mg/L).
- Road salt constituents (sodium and chloride) were detected at greater concentrations in shallow ground water underlying urban areas than other areas.
- Agricultural pesticides were commonly detected in all land-use settings. Concentrations were greatest in agricultural areas.
- Atrazine was the most frequently detected agricultural pesticide. Concentrations were greater in shallow ground water than in deeper ground water.

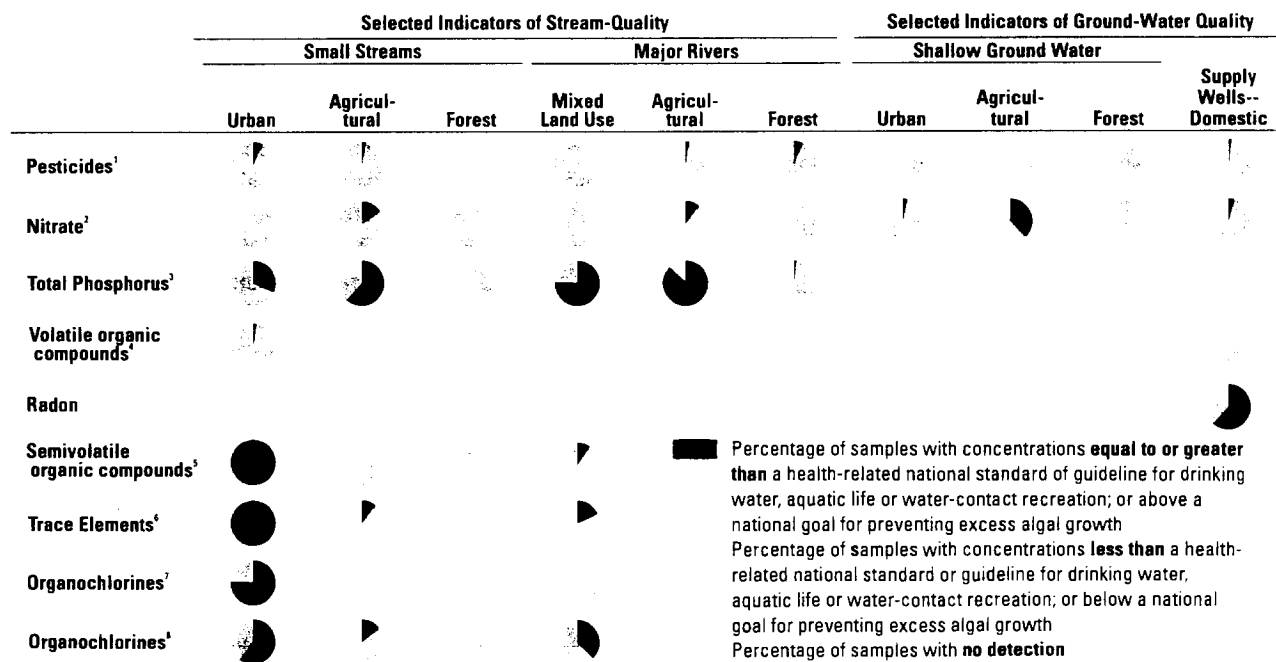
- Prometon was the most frequently detected herbicide in urban areas.
- Ground water in the Prairie du Chien-Jordan aquifer, an important source of drinking water, is protected by overlying confining units in some areas. Concentrations of nitrate, atrazine, and VOCs were lower in these areas than where confining units were absent.
- Radon exceeded the USEPA suspended drinking water standard of 300 pCi/L (picocuries per liter) in more than one-half of the water samples from the Prairie du Chien-Jordan aquifer.

Trends in Ground-Water Quality

Temporal trends in ground-water quality are difficult to define because limited information exists. Spatial trends include greater nitrate and pesticide concentrations in agricultural areas, greater VOC concentrations in urban areas, and few detections of pesticides or VOCs in forested areas.

Major Influences on Ground Water

- Application of pesticides and fertilizers
- Confining units and depth to water
- Urban contaminants (road salts, VOCs)
- Naturally occurring radon gas



¹Insecticides, herbicides, and pesticide metabolites, sampled in water. ²Nitrate (as nitrogen), sampled in water. ³Total phosphorus, sampled in water. ⁴Solvents, refrigerants, fumigants, and gasoline compounds, sampled in water. ⁵Byproducts of fossil-fuel combustion or components of coal and crude oil, sampled in sediment. ⁶Arsenic, mercury, and metals, sampled in sediment. ⁷Organochlorine compounds including DDT and PCBs, sampled in sediment. ⁸Organochlorine compounds including DDT and PCBs, sampled in fish tissue.

INTRODUCTION TO THE UPPER MISSISSIPPI RIVER BASIN

The Upper Mississippi River Basin Study Unit (Study Unit) includes the drainage of the Mississippi River from its source at Lake Itasca, Minnesota, and its major tributaries (the St. Croix and Minnesota Rivers) to the outflow of Lake Pepin, Minnesota (fig. 1). Natural and human factors (climate, hydrology, geology, water use, land use, and land cover) affect surface- and ground-water quality, and aquatic biology in rivers and streams.

Natural Factors Affect Water Quality and Aquatic Biology

Differences in precipitation, evaporation, evapotranspiration, air temperature (fig. 1), and drainage basin characteristics (drainage area, slope, geology, and the capacity of soils to transmit water) affect hydrology and water quality. These differ most from southwest to northeast. Mean annual runoff, which is related to precipitation and evaporation, ranges from less than 2 inches in the headwaters of the Minnesota River to greater than 14 inches in the headwaters of the St. Croix River.

The range from minimum to maximum streamflow is greatest in spring and early summer as a result of rain and melting snow. Streamflow variation is greatest during late summer and fall, when precipitation ranges from drought conditions to locally heavy rains (fig. 2). Streamflow varies least during winter, when ground-water discharge to streams is dominant. During the period of sampling (1996–98), precipitation was greater than the 30-year average, resulting in increased runoff and streamflow. As a result, the amount of sediment, nutrients, pesticides, and other contaminants reaching streams may have been

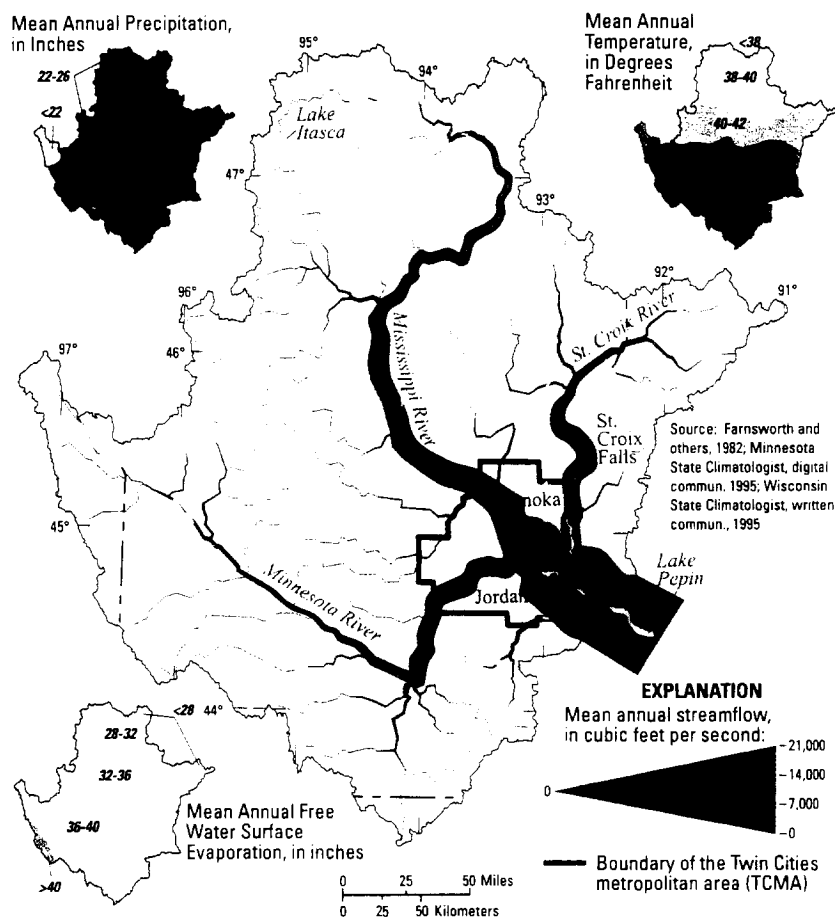


Figure 1. Climatic variables such as precipitation, temperature, and evaporation affected streamflow in the Study Unit, 1961-90.

greater than during periods of normal streamflow.

Water quality is also affected by geologic materials. Most streams in the Study Unit drain the Central

Lowland physiographic province, which is underlain by clay-rich, calcareous (calcium carbonate) glacial deposits (fig. 3). Fewer streams drain the Superior Upland physio-

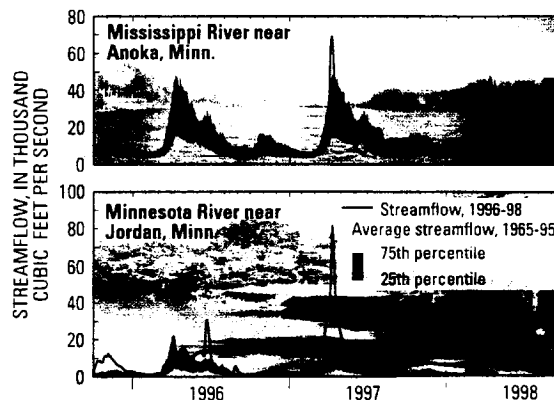


Figure 2. Streamflow during the sampling period (1996-98) in the large rivers in the Study Unit differed from their 30-year average, 1965-95.

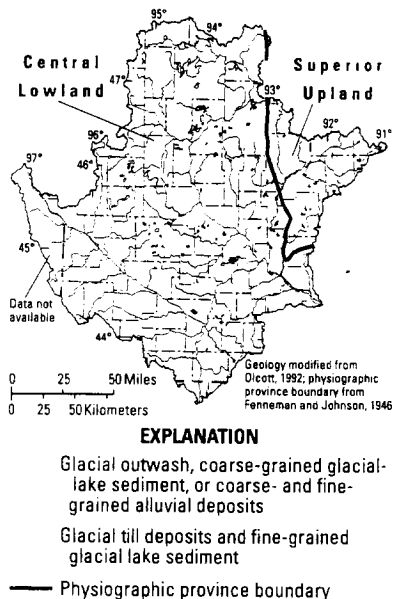


Figure 3. Surficial geology and physiographic provinces can affect water quality in the Study Unit.

graphic province, which is primarily underlain by siliceous (rich in silica), sandy glacial deposits. Water in streams draining the Central Lowland generally has greater alkalinity and greater concentrations of suspended sediment than water in streams draining the Superior Upland.

Human Activities Affect Water Quality and Aquatic Biology

The greatest effects on hydrology, water quality, and aquatic biology occur in areas with the greatest human population densities or where disruption to the natural land cover is substantial. The population of the Study Unit in 1990 was about 3.7 million—16-percent increase from 1970. Seventy-five percent of those people reside in the TCMA.

Land use and land cover in the Study Unit can be categorized into three zones: an agricultural zone across the southwest, a forested

zone across the northeast, and a transitional zone between these areas (fig. 4). About 63 percent of the Study Unit is agricultural (cropland and pasture). The remaining land use and land cover consists of forests (about 22 percent), water and wetlands (about 13 percent), urban (about 2 percent), and other categories (less than 1 percent).

The uses of water and the disposal of wastewater also can affect water quality and streamflow. Based on data from 1990, a daily average of 413 Mgal/d (million gallons of water

per day) was used for public supply (including drinking water) in the Study Unit—59 percent from ground water and 41 percent from surface water (fig. 5). The total of all water used for public supply is equal to about 7 percent of the average streamflow of the Mississippi River upstream from the TCMA, near Anoka, Minn. Wastewater is discharged to streams from about 270 facilities located throughout the Study Unit (Kroening and Andrews, 1997).

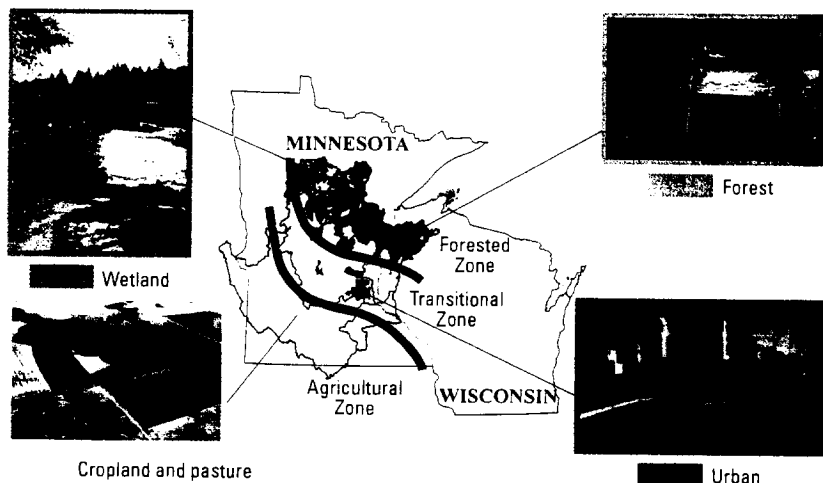


Figure 4. Land use and land cover can be categorized into three general zones in the Study Unit.

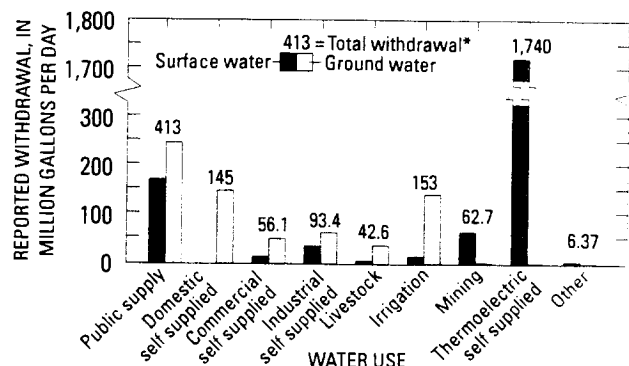


Figure 5. Ground water supplies the majority of the public drinking water in the Study Unit in 1990. (* Total refers to the combination of surface and ground water in each category.)

MAJOR FINDINGS

This report presents work by the U.S. Geological Survey's National Water-Quality Assessment Program to assess the quality of the Study Unit's water and aquatic resources (fig. 6). The report summarizes historical data and Study Unit data collected during 1995-98.

Land Use Influences Water Quality and Aquatic Biology

Point and nonpoint sources of nutrients, sediments, metals, and organic compounds from industrial, agricultural, and urban land uses are important water-quality issues in the Study Unit. Degradation of streams, including the loss of riparian habitat, reduction in fish populations, loss of habitat for bottom-dwelling organisms, eutrophication, and deterioration of the sanitary quality of streams is also important. Additional issues

include the introduction of toxic substances, such as organic compounds and trace elements that accumulate in sediments and aquatic biota of the rivers. These contaminants can adversely affect the health of aquatic biota and may biomagnify in fish-eating birds and mammals.

Water-quality issues in the TCMA and other urban areas include surface-water contamination from urban runoff and discharge from industrial and wastewater treatment facilities and the introduction of toxic substances to ground water from industrial activities and nonpoint sources. In agricultural areas, including the Minnesota River Basin, water-quality degradation from artificial drainage systems and point and nonpoint sources of sediment, nutrients, and pesticides are of concern. Both urban and agricultural

land uses contribute to the impairment of habitat and eutrophication in the Mississippi River in and downstream from the TCMA. In forested areas, including the St. Croix River Basin and upper reaches of the Mississippi River Basin, water is generally of better quality than elsewhere in the Study Unit. Maintaining the quality of water in the St. Croix River Basin is a priority for the National Park Service and the States of Minnesota and Wisconsin (Minnesota Department of Natural Resources and others, 1995).

Water Quality and Aquatic Biological Conditions Remain Relatively Undisturbed in Forested Areas

White pine forests originally covered much of the upper parts of the St. Croix River Basin and the Mississippi River Basin. These forests were logged during the mid 1800s to early 1900s and are now covered by second-growth forests.

Land-cover disturbances in these forested areas have been minimal, although small farms and towns are common, as is increased development for recreation. Water quality in these forested areas has been affected by minor applications of herbicides at small farms, tree farms, and in lakes (for weed reduction); discharges of wastewater effluent; leaks from septic systems; local stream-channel disturbances from forestry; and localized draining of wetlands. These activities result in small increases in nutrient, pesticide, suspended-sediment, and bacteria concentrations relative to natural conditions. Water-quality and aquatic-biological conditions have probably been affected less by human activities in the forested areas than in other

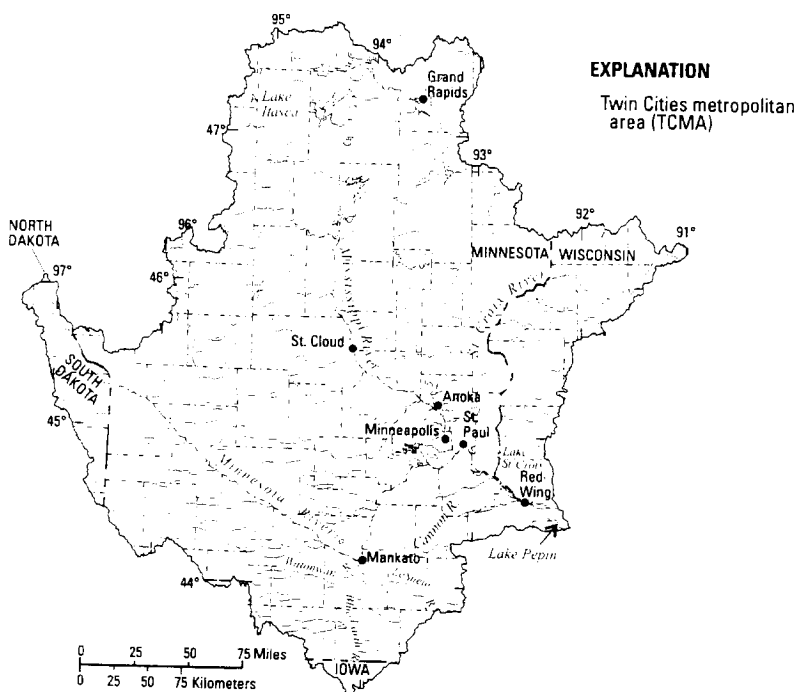


Figure 6. Upper Mississippi River Basin Study Unit, Twin Cities metropolitan area, major rivers and streams, and selected cities.

areas of the Study Unit. Nutrients and pesticides did not exceed drinking-water standards and guidelines for human consumption in streams and in ground water in forested areas. Nitrate and phosphorus yields were low in streams in forested areas (table 1). Suspended-sediment concentrations, which can contribute to degraded water quality and habitat, also were low in streams draining forested areas compared to the rest of the Study Unit.

Table 1. Nitrate and phosphorus yields in pounds per square mile per year in forest streams, 1996-98

STREAM	NITRATE	PHOSPHORUS
Namekagon River	260	— ^a
St. Croix River	160	50

^aYield was not calculated because concentrations were below the analytical reporting limits.

Pesticides were periodically detected in streams and in shallow ground water in forested areas, but concentrations and detection rates were lower than in the rest of the Study Unit (Fallon and others, 1997; Fong, 2000). Trace-element concentrations in streambed sediments corresponded to the composition of the surficial glacial deposits (Kroening and others, 2000). For example, increased concentrations of copper in the forested areas are attributed to naturally occurring sources. Although bacteria concentrations in streams in forested areas were below the USEPA criterion for swimming (Kroening, 1999; U.S. Environmental Protection Agency, 1986), these waters would not be suitable for human consumption without treatment because bacteria counts may occasionally exceed USEPA drinking-water standards.

Physical modifications to streams, such as stream dredging or channelization, have been minimal in forested areas of the Study Unit.

Consequently, aquatic communities are rich and diverse. Streams generally are more shaded than streams in other parts of the Study Unit, resulting in cooler water temperatures. Greater shading, cooler water, and lower concentrations of nutrients may limit algal productivity in these streams draining forested land. Algal communities in forest streams consist of species such as diatoms that are indicative of low nutrient and suspended-sediment concentrations.



Forests dominate the northern portion of the Study Unit. (Photograph courtesy of the National Park Service.)

Increased urbanization and development for recreation contribute to degraded water quality and aquatic life. Management practices that could benefit the quality of streams in these areas include restoration of natural wetlands and riparian vegetation. Eliminating these practices would improve stream habitat and hydraulic conditions and improve the diversity of fish and invertebrate communities.

Many programs and water-quality regulations are in place or are being considered to protect the quality of water in these areas, particularly in the St. Croix River Basin. One example is an effort to restrict increases in phosphorus to the St. Croix River to prevent excessive algal growth in Lake St. Croix (Holmberg and others, 1997).

Agricultural Activities Increase Nutrient and Pesticide Concentrations in Ground Water and Streams and Degrade Aquatic Biological Conditions

Agricultural areas of the Study Unit (fig. 4) include most of the Minnesota River Basin and parts of the Mississippi and St. Croix River Basins. In these areas, much of the land is used for production of row crops, primarily corn and soybeans. Many streams in agricultural areas have been straightened, ditches excavated, and land is commonly cultivated close to the streambanks. Most wetlands in agricultural areas have artificial drainage systems to increase crop production. Agricultural activities disrupt riparian zones in streams, contributing to erosion and runoff of agricultural chemicals and sediment.



Cultivation of land close to streams, artificial drainage, and stream straightening degrade water quality and aquatic habitat. (Photograph by James D. Fallon.)

Nutrient concentrations in surface water and ground water (much of which eventually discharges to streams) were greater in agricultural areas than in other parts of the Study Unit (Payne, 1994; Kroening and Andrews, 1997; Ruhl and others, 2000). Commercial fertilizers and animal manure applied to agricultural

land are sources of nutrients to streams and ground water (Kroening, 1998b; Ruhl and others, 2000). Nutrients that reach streams through artificial drainage or runoff accelerate the growth of algae and aquatic plants, resulting in eutrophication and diminished dissolved oxygen concentrations. In addition to affecting aquatic species, eutrophication also can cause taste and odor problems in water for domestic use.

Nitrate concentrations in streams draining the southern and southeastern parts of the Study Unit,

most notably in the Blue Earth, Le Sueur, and Watonwan Rivers (Payne, 1994), have exceeded the drinking-water standard of 10 mg/L set by the USEPA to prevent methemoglobinemia in infants. Greater than one-half of the samples collected by Payne (1994) exceeded that drinking-water standard. Nitrate yields were greatest in agricultural streams (table 2). Nitrate yields were about 10 times greater in streams draining artificially drained, fine-grained surficial geologic deposits compared to streams draining coarse-grained

deposits (Kroening, 1998a). Nitrate concentrations in shallow ground water are also greatest in the agricultural part of the Study Unit, and generally increased with the intensity of the agricultural activity and decreased with the water-table depth below land surface (Ruhl and others, 2000). (see "Nitrate in a National Context")

Table 2. Nitrate and phosphorus yields in pounds per square mile per year in agricultural streams, 1996-98

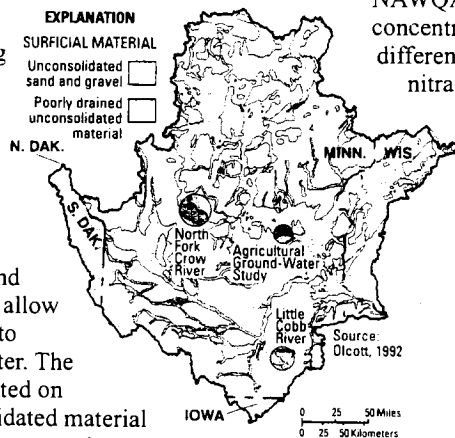
STREAM	NITRATE	PHOSPHORUS
North Fork Crow River	1,400	190
Little Cobb River	15,000	330



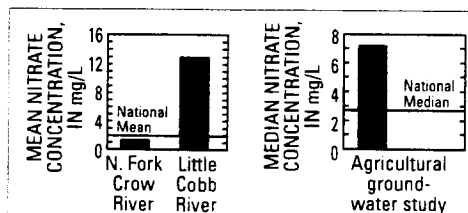
NITRATE IN A NATIONAL CONTEXT--CONCENTRATIONS RELATE TO HYDROGEOLOGY AND AGRICULTURAL DRAINAGE IN THE STUDY UNIT

Nitrate concentrations in the Study Unit are related to hydrogeologic setting and agricultural drainage. The application of commercial fertilizers and manure are sources of nitrate in streams and ground water. In general, nitrate concentrations in water are greatest in agricultural areas throughout the Nation (U.S. Geological Survey, 1999) including the Upper Mississippi River Basin. Yet, within agricultural areas within the Study Unit, nitrate concentrations vary due to the hydrogeologic setting.

Two rivers draining agricultural land in the Study Unit were frequently sampled for nitrate (1996-98). The North Fork Crow River is located in an area underlain by unconsolidated, coarse-grained sand and gravel deposits, that allow water and contaminants to infiltrate into ground water. The Little Cobb River is located on poorly drained unconsolidated material that limits the ability of water and contaminants to infiltrate into ground water. Artificial drainage systems (ditches and tiles) have been installed throughout these poorly drained soils to improve agricultural production. These systems also result in more direct transport of contaminants to nearby streams.



Although nitrate application rates from fertilizer and manure were similar in both river basins, nitrate concentrations in the streams were different. The nitrate concentration in the naturally well-drained North Fork Crow River was less than the national average for agricultural streams. In contrast, artificial drainage in the Little Cobb River Basin has contributed to nitrate concentrations in the stream, which rank among the top 2 percent of all streams sampled in the NAWQA Program. Differences between the nitrate concentrations in these two streams (see graph) reflect differences in their hydrogeologic settings. Although nitrate concentrations were low in streams draining surficial sand and gravel deposits, concentrations were greater in ground water--much greater than the national median. (see graph.)



To maintain water quality in streams and ground water, best management practices could include consideration of hydrogeologic setting.

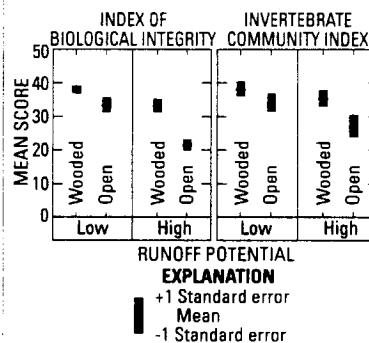
Phosphorus concentrations exceeding the goal of 0.1 mg/L recommended by the USEPA (1986) to prevent eutrophication were measured in agricultural streams (Kroening, 2000). Results from routine sampling showed this concentration was exceeded more frequently (about 75 percent of the samples) in streams fed by artificially drained soils that developed on fine-grained materials than in streams draining coarse-grained materials (about 30 percent of samples). Phosphorus yields were greatest in agricultural streams (table 2). Phosphorus yields were approximately 1.7 times greater in streams draining artificially drained, fine-grained surficial deposits than in streams draining coarse-grained deposits.

Median suspended-sediment concentrations typically ranged from 60 to 120 mg/L in agricultural streams (Kroening, 2000). Suspended-sediment concentrations were greater in streams in artificially drained, fine-grained surficial deposits compared to streams draining coarse-grained deposits. Physical disturbances to stream morphology, hydrology, and instream habitat have been caused by stream straightening, removal of riparian vegetation, drainage of wetlands, and tile drainage systems (see "Riparian Cover and Runoff Potential Affect Aquatic Biology," and "Riparian Buffer Zones Affect the Quality of Midwestern Streams and Rivers," p. 9). These disturbances also contribute to increased concentrations of suspended sediment, relative to streams in other land-use settings.

Pesticides frequently were detected in streams and shallow ground water in agricultural areas

RIPARIAN COVER AND RUNOFF POTENTIAL AFFECT AQUATIC BIOLOGY

An investigation of 24 streams in the Minnesota River Basin during August 1997 determined that there were differences in fish- and invertebrate-community compositions due to both riparian cover and runoff potential (which increases when water infiltration decreases) (Stauffer and others, 2000; ZumBerge, 1999). An Index of Biotic Integrity (IBI--a measure of biological conditions based on several fish-community attributes), an Invertebrate Community Index (ICI--a measure of biological conditions based on several invertebrate community attributes), and species richness were used as measures of resource quality. Streams with wooded riparian cover had better IBI scores, ICI scores, and greater fish and invertebrate species richness than streams with open riparian cover indicating better resource quality. Streams with low runoff potential had better IBI scores, ICI scores, and fish species richness than streams with high runoff potential.



These results suggest that streams with wooded riparian cover had greater resource quality as indicated by fish and invertebrate community measures.

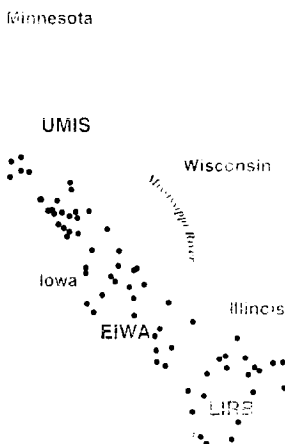
(Fallon and others, 1997; Fallon, 1998; Ruhl and others, 2000). Few concentrations exceeded applicable drinking-water standards and guidelines or aquatic-life guidelines. Herbicides were detected more

frequently than insecticides. Pesticide concentrations in streams typically were greatest from May to July (Fallon and others, 1997). Ground-water samples with detections of one or more pesticides usually coincided with areas of shallow ground water close to the land surface (Hanson, 1998; Ruhl and others, 2000). Organochlorine insecticides were detected in fish tissue but not in streambed sediment (fig. 7, and see "Concentrations of Degradation Products of Agricultural Herbicides were Greater than Their Parent Compounds in Little Cobb River Near Beauford, Minn., 1997," p. 10).

Algal, invertebrate, and fish communities have likely been affected by agriculture. Increased nutrients in agricultural streams have resulted in greater algal abundance and primary production. Algal communities were composed of a large proportion of blue-green algae that are commonly associated with high nutrient concentrations and are not suitable food sources for invertebrates (Kroening, 2000; Lee and ZumBerge, 2000). Contaminants from agricultural practices have likely affected invertebrate communities, which were moderately diverse and composed of mayflies and caddisflies that are relatively sensitive to contaminants. Total fish biomass was high in agricultural streams, probably in response to greater algal abundance and productivity. Although suspended-sediment concentrations were greater in the agricultural streams than in streams in other land-use settings, the presence of fish species such as stonecat and smallmouth bass indicate good water quality in terms of clarity (Goldstein and others, 1999).

RIPARIAN BUFFER ZONES INFLUENCE THE QUALITY OF MIDWESTERN STREAMS AND RIVERS

Despite similar land use throughout the Corn Belt region of the Midwest, streams flowing through cropland differ considerably in their ecological characteristics, in part because of differences in riparian buffer zones (see text boxes). This conclusion is based on an investigation of 70 streams and rivers within three NAWQA Study Units in the upper Midwest during August 1997 (map shown at right; Sorenson and others, 1999; Porter and others, 2001). Specifically, increases in tree cover in buffer zones were associated with aquatic biological communities indicative of good stream quality, reduced nuisance algal growths, and maintenance of sufficient dissolved oxygen concentrations to support diverse communities of aquatic organisms. For example, the number of aquatic insects indicative of good stream quality tended to increase with increases in percentage of tree cover, especially in sites where streamflow and dissolved oxygen conditions were favorable. Fish communities, sampled at 24 sites in the UMIS Study Unit, also indicated better overall conditions in streams with wooded riparian zones than those with more open canopy (Stauffer and others, 2000).

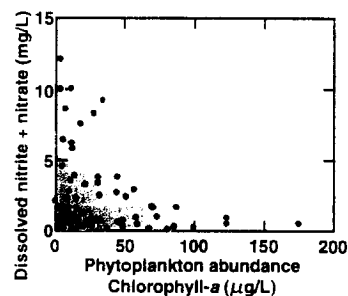


The influence of riparian buffer zones on the quality of 70 midwestern streams and rivers was evaluated in the Upper Mississippi River (UMIS), Eastern Iowa (EIWA), and Lower Illinois River Basins (LIRB).

Streams with less tree cover, and thus less shading, contained relatively large growths of phytoplankton (algae suspended in the water) at levels considered indicative of eutrophication (Porter, 2000). Organic enrichment

resulting from excessive algal production in some midwestern streams may reduce dissolved oxygen concentrations and be detrimental to other requirements of aquatic organisms.

Shading from tree cover in riparian buffer zones may influence nutrient concentrations indirectly by reducing the growth of phytoplankton. In streams where phytoplankton were abundant (often where buffer zones were thin or lacking), dissolved nitrate concentrations were significantly lower (graph shown below; Porter, 2000). The lower nutrient concentrations may result from uptake by the abundant phytoplankton. Thus, assessments of eutrophication would benefit from consideration of biological communities and the riparian zone, rather than being based solely on nutrient concentrations in the water.



Dissolved nutrient concentrations decreased in eutrophic streams with excessive algal productivity. Rates of nutrient uptake by the algae can exceed rates at which nutrients are transported by streams during low-flow conditions.



Digital images derived from USGS topographic maps were used to estimate the percentage of trees in a riparian buffer zone (a 100-meter width on each side of the stream) for 2- to 3-mile segments upstream from each sampling site, supplemented by vegetation surveys at the sampling site (Sorenson and others, 1999).

Resource agencies, including the U.S. Department of Agriculture, encourage maintenance of strips of trees or grass between cropland and streams as a best management practice. These "riparian buffer zones" are thought to intercept runoff of sediment and chemicals from fields, promote bank stability, and provide shading and habitat for aquatic life (Osborne and Kovacic, 1993). Riparian buffer zones should be considered along with other important factors that affect chemical and biological indicators of stream quality, such as soil drainage properties and stream hydrology (Porter and others, 2001).

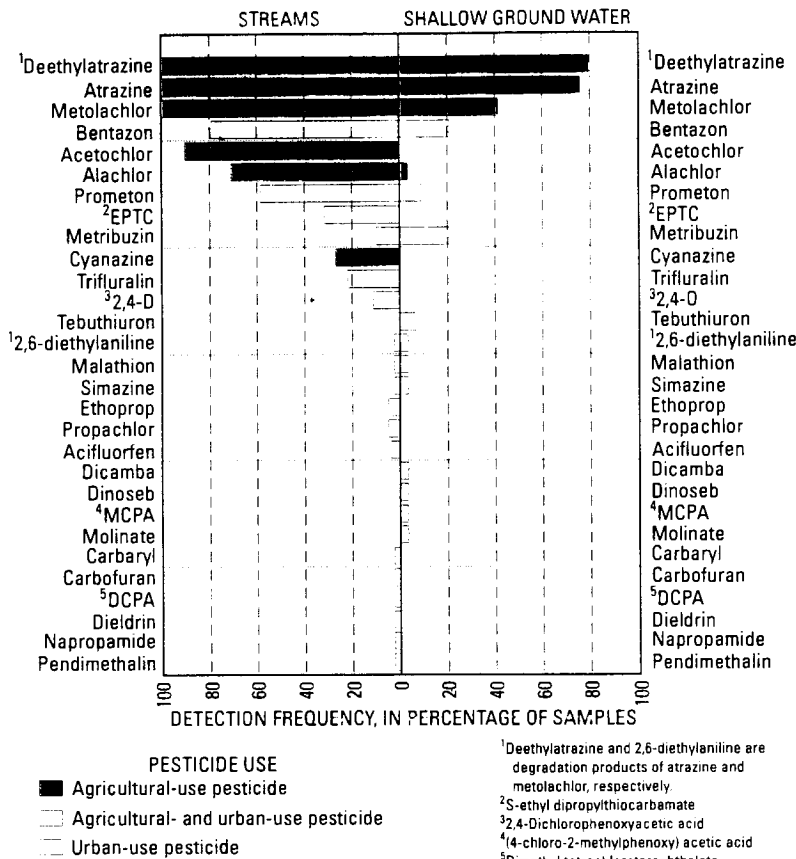
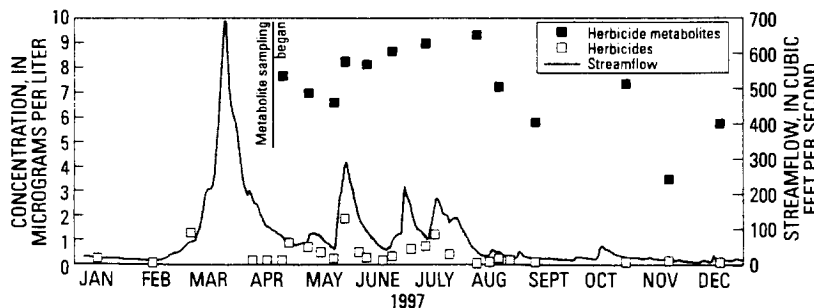


Figure 7. Atrazine and its degradation product deethylatrazine were the most frequently detected pesticides in streams and shallow ground water in agricultural areas in the Study Unit.

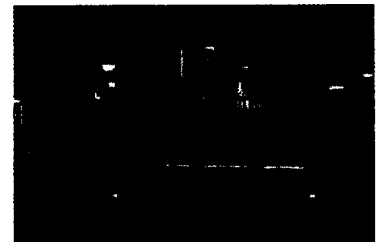
CONCENTRATIONS OF DEGRADATION PRODUCTS OF AGRICULTURAL HERBICIDES WERE GREATER THAN THEIR PARENT COMPOUNDS IN LITTLE COBB RIVER NEAR BEAUFORD, MINNESOTA, 1997

Eight degradation products (metabolites) of four commonly used agricultural herbicides (acetochlor, alachlor, atrazine, and metolachlor) were detected in samples collected from the Little Cobb River, an agricultural stream. Summed metabolite concentrations were always greater than summed parent compound concentrations. Metabolite concentrations were least during the fall and greatest during the summer. Four metabolites were present year round at substantial concentrations (metolachlor-ethane sulfonic acid and metolachlor-, acetochlor-, and alachlor-oxanylic acid). The affects of these metabolites on aquatic and human health are not known, their persistence and relatively high concentrations are a cause for concern.



Water Quality and Aquatic Biological Conditions are Adversely Affected in Urban Areas

The intensity of development in urban areas has adversely affected the quality of streams and ground water. Nonpoint-source contaminants to surface and ground water in urban areas originate from automobiles, road de-icing chemicals, construction, application of pesticides and fertilizers, atmospheric deposition, street debris in urban stream-water runoff, and animal and plant refuse (Hambrook and others, 1997). Major sources of contamination to ground water include spills or improper disposal of industrial or manufacturing chemicals, leachate from solid-waste landfills, and spills and leaks from petroleum storage areas and pipelines (Minnesota Pollution Control Agency, 1986).



Minneapolis, Minn., the largest city in the Study Unit. (Photograph by Scott Murray Photography.)

Several factors can affect the occurrence and distribution of contaminants in surface and ground water in urban areas. Factors affecting urban streams include impervious surfaces, drainage of wetlands, construction of detention ponds, loss of riparian cover, and stream-channel modifications (Riley, 1998). Impervious surfaces cause greater peak streamflow rates of shorter duration from runoff than would occur naturally, and increase transport of contaminants from

streets and parking lots to streams (Riley, 1998). These factors can increase water temperature and degrade habitat and water quality. Average water temperature in TCMA streams increased as the percentage of impervious surface increased (Talmage and others, 1999). Concentrations of nutrients, trace elements, chloride, sodium pesticides, and counts of bacteria were frequently greater in urban streams than those that occur naturally and may inhibit growth, reproduction, and diversity of aquatic biota (Klein, 1979; Pope and Putnam, 1997). Factors affecting shallow ground-water quality

include the composition of surficial material and depth to ground water. Sand and gravel surficial materials increase infiltration and impervious surfaces decrease infiltration to ground water. Shallow ground-water quality generally improves with depth.

Streams and ground water in shallow aquifers in the TCMA contained elevated concentrations of sodium and chloride (Andrews and others, 1998), a result of the application of road de-icers. (see "Chloride in a National Context")

Chloride concentrations in urban streams (Fallon and Chaplin, 2001) frequently exceeded the aquatic-

life criterion of 230 mg/L (U.S. Environmental Protection Agency, 1999). Chloride concentrations were greater in streams with greater percentages of impervious surfaces and may have adversely affected fish diversity. (see "Urbanization Affects Fish Communities and Water Quality in Urban Streams of the Study Unit," p. 12)

All nitrate concentrations in streams were less than the USEPA drinking-water standard of 10 mg/L (Kroening, 1998a, 2000). Less than 10 percent of nitrate concentrations in ground water exceeded the standard (Andrews and others, 1998; Fong and others,

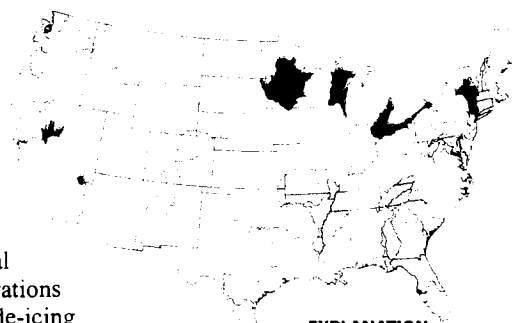


CHLORIDE IN A NATIONAL CONTEXT-- CONCENTRATIONS ARE GREATEST IN NORTHERN URBAN AREAS

Chloride concentrations in urban streams of the Study Unit were substantially greater than in most urban streams sampled throughout the Nation. Median chloride concentrations in ground water overlain by urban areas in the Study Unit were also greater than the national median, although not substantially. Elevated chloride concentrations result from runoff of de-icing chemicals applied to roads and highways during winter storms (Granato, 1996). Because winter conditions are similar across the North-Central and Northeastern United States, the greater median chloride concentrations in other northern study units may also be at least partly the result of de-icing compounds. Sodium chloride (salt) is the primary de-icing compound applied to roads and highways in the Study Unit (Minnesota Department of Transportation, electronic commun., 2000). The environmental setting of the urban portion of the Study Unit, much of it covered with permeable sandy soils, wetlands, and lakes, may allow chloride to be more readily transported to and stored in lakes, wetlands, and shallow ground water (where chloride can persist) as well as being flushed directly to streams. Talmage and others (1999) reported that chloride concentrations were positively correlated with impervious areas (buildings and paved surfaces) in 13 urban streams of the Study Unit. The source of elevated chloride concentrations in urban streams in arid study units are likely from naturally occurring salts concentrated by the evaporation of surface water (Hem, 1992).

Whereas de-icers are applied to roads in other study units throughout the Nation, concentrations in streams and ground water in this Study Unit are likely greater for several reasons. The amount of snowfall and seasonal duration of subfreezing temperatures may be greater in the Study Unit than most other study units. De-icing compounds other than sodium chloride may be used in other study units.

Many samples had chloride concentrations that exceeded the aquatic-life criteria established by the U.S. Environmental Protection Agency (1999). Elevated chloride concentrations in streams may affect biological communities by altering the species composition. Urban streams in the Study Unit were dominated by fish and invertebrate species that are tolerant to degraded physical and chemical conditions, compared to other streams in the Study Unit.



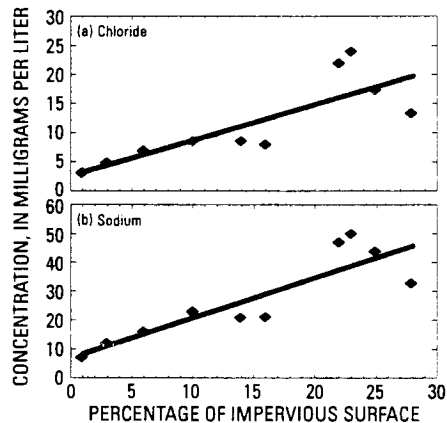
EXPLANATION
Median concentration in urban streams, in milligrams per liter (mg/L)

100.0 - 185.0
40.0 - 99.9
10.0 - 39.9
0.0 - 9.9

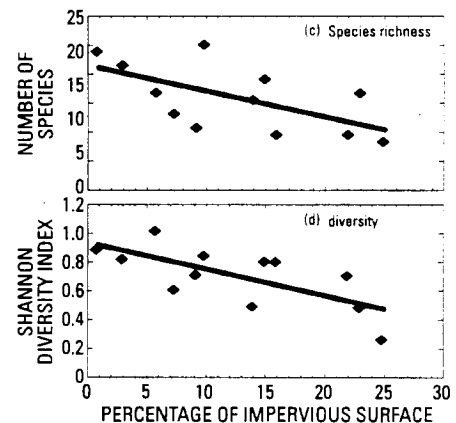
URBANIZATION AFFECTS FISH COMMUNITIES AND WATER QUALITY IN URBAN STREAMS OF THE STUDY UNIT

Water quality, instream habitat, and fish-community composition were characterized at urban streams of the Study Unit during low-flow conditions, September 1997. The density of impervious cover (roads, parking lots, and rooftops) generally increases as population density increases and was used as a measure of urbanization. Nutrient and pesticide concentrations were generally low, rarely exceeding concentrations found in agricultural streams. Nutrient concentrations did not change with the percentage of impervious area. In contrast, chloride (fig. a) and sodium (fig. b) (used for road de-icing) concentrations were generally elevated in urban streams and increased as the percentage impervious area increased.

Fish communities within most urban streams were characterized by species that are tolerant to degraded physical and chemical conditions, such as the central mudminnow, fathead minnow, and black bullhead. There were, however, differences in the fish communities among streams. Two measures of community health--the species richness and diversity--decreased as the percentage of impervious area increased (figs. c and d). Factors associated with impervious cover, such as reduced instream habitat, presence of contaminants in water and sediment, alterations to stream channels, and migration barriers, may directly affect fish-community composition.



Concentrations of chloride (a) and sodium (b) in relation to percentage of impervious surface in urban streams of the Study Unit, September 1997



Species richness (c) and diversity (d) in relation to percentage of impervious surface in urban streams of the Study Unit, September 1997

1998). Nitrate and phosphorus yields in urban streams (table 3) were less than in agricultural streams (table 2) and greater than in forest streams (table 1). About 30–37 percent of the total phosphorus concentrations in urban streams exceeded the USEPA's water-quality criterion of 0.1 mg/L (Kroening, 1998a, 2000). The greatest concentrations of nitrate in ground water were from samples of shallow ground water (unconfined surficial sand and gravel aquifers) (Kroening and Andrews, 1997). Areas with the greatest concentrations of nitrate are related to aquifer susceptibility and overlying land use. Nitrate concentrations tend to decrease with increased well depth (Hanson, 1998).

Table 3. Nitrate and phosphorus yields, in pounds per square mile per year in streams in urban areas, 1996-98

Stream	Nitrate (lb/sq mi/yr)	Phosphorus (lb/sq mi/yr)
Shingle Creek	400	130
Nine Mile Creek	510	140

Dissolved-oxygen concentrations in most urban streams usually were greater than the minimum 5 mg/L aquatic-life criterion (U.S. Environmental Protection Agency, 1986) necessary for the protection of aquatic life. Dissolved-oxygen saturation in urban streams during the growing season was generally greater than forest streams and less than agricultural streams.

Pesticides were frequently detected in urban streams and shallow ground water (fig. 8); however, concentrations seldom exceeded

applicable standards or guidelines (Andrews and others, 1998). Concentrations in shallow ground water were generally less than in surface water (Fallon and others, 1997; Andrews and others, 1998). Factors affecting pesticides in surface and ground water include land use, application methods, and atmospheric transport and deposition. In streams and shallow ground water, herbicides commonly used on road rights-of-way were detected (prometon and tebuthiuron), as were agricultural herbicides (atrazine and metolachlor). Insecticides were detected in almost 50 percent of stream water samples (Fallon, 1998) but in less than 5 percent of ground-water samples (fig. 8).

Volatile organic compounds (VOCs) were detected in surface and shallow ground water in the urban part of the Study Unit (fig. 9) (Andrews and others, 1995 and 1998). Some VOCs are suspected carcinogens and may be toxic to humans and wildlife. Although many VOCs were detected in urban streams, concentrations generally were below applicable standards and guidelines. The greatest concentrations occurred in stormwater runoff and winter low flows. The most frequently detected VOCs are components of petroleum products and by-products of petroleum combustion. These VOCs are contributed to streams from engine emissions to the atmosphere and from oil and gasoline leaks from vehicles to parking lots and roadways.

Other contaminants such as polycyclic aromatic hydrocarbons (PAHs), organochlorine compounds (OCs), and trace elements are common in urban streams, frequently at concentrations greater than aquatic-life guidelines (McNellis and others, 2000; Talmage and others, 1999) (see "Organic Contaminants in a National Context"). Urban activities and discharges also contribute to increased concentrations of trace elements (particularly cadmium, copper, lead, and zinc) in some urban streambed sediments. Elevated concentrations of some trace elements can be toxic to humans and aquatic life.

Fecal coliform counts differed widely among urban stream samples collected during September 1997, ranging from about 54 col/100mL (colonies per 100 mL) to more than 11,000 col/100 mL (Talmage and others, 1999). Fecal coliform counts at 8 of 13 sites

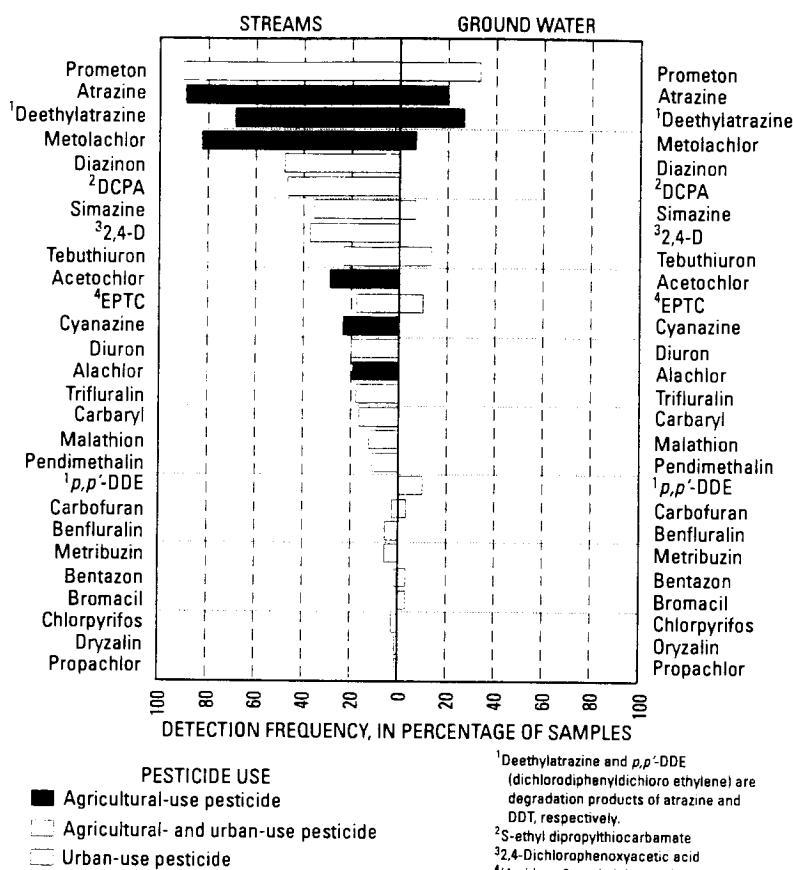


Figure 8. Pesticides typically used in agricultural areas were frequently detected in streams and ground water in urban areas in the Study Unit.

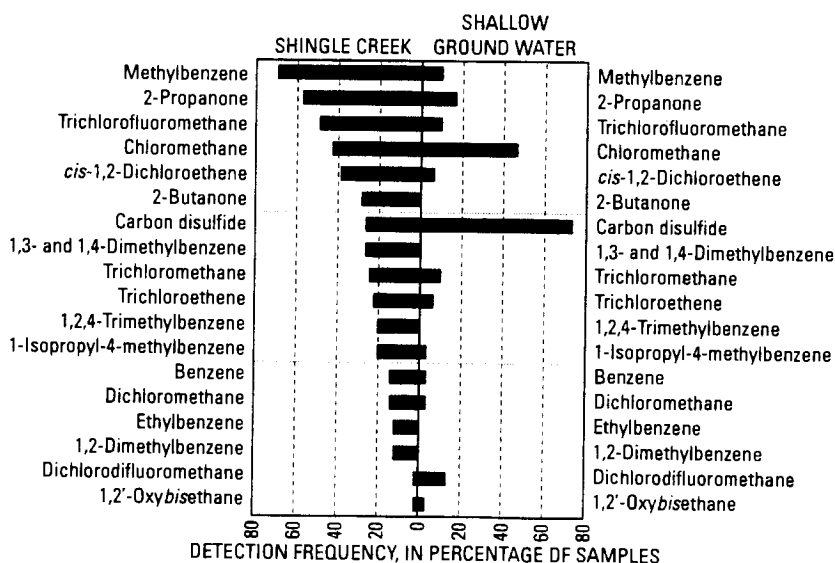


Figure 9. More volatile organic compounds (VOCs) were detected in Shingle Creek (an urban stream) than in the shallow ground water in the same land-use setting, indicating that many VOCs break down before infiltrating the shallow ground water.



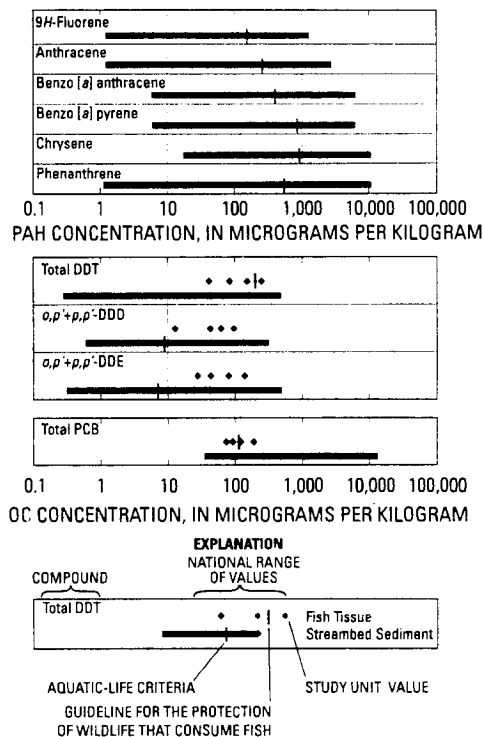
ORGANIC CONTAMINANTS IN A NATIONAL CONTEXT: CONCENTRATIONS WERE GREATEST IN URBAN STREAMS IN THE STUDY UNIT

Polycyclic aromatic hydrocarbon concentrations in streambed sediment in urban areas are among the greatest in the Nation.

Six polycyclic aromatic hydrocarbon compounds (PAHs) were detected at concentrations above U.S. Environmental Protection Agency (USEPA) aquatic-life criteria. Some are known carcinogens and are toxic to aquatic life. These compounds are generally by-products of combustion of fossil fuels or the burning of wood. Concentrations of PAHs at sites in other land uses were 10 to 100 times less than those in urban areas.

Organochlorine detections are prevalent in urban areas. Some sites had concentrations greater than recommended for the protection of aquatic life or wildlife.

Streambed sediment and fish tissue were analyzed for organochlorine compounds (OCs). Although uses of the insecticide DDT for mosquito control and polychlorinated biphenyls (PCBs) for industrial applications were discontinued in the 1970s, these compounds were still detected in urban streambed sediment in the Study Unit. Twelve of the 13 OCs (insecticides and PCBs) detected in streambed sediment in the Study Unit were found at urban sites. Three OCs including DDT, DDT metabolites (DDE and DDD), and total PCBs were detected in fish tissue at all urban sites in the Study Unit. Total DDT and metabolites in streambed sediment exceeded USEPA water-quality guidelines. PCB concentrations in fish exceeded USEPA standards for wildlife that consume fish.



exceeded the State of Minnesota freshwater standard for recreational use (200 col/100 mL) (Minnesota Pollution Control Agency, 1991). The greater bacteria counts may indicate localized leaking sewer or septic systems or animal waste.

Relatively low nutrient concentrations, stream shading, and contaminants may lead to low algal production in urban streams (Lee and others, 1999). However, nutrient concentrations are a concern because urban streams commonly drain to lakes that are more sensitive to eutrophication. The warmer temperatures and longer residence times of the water in lakes allow greater algal productivity. Invertebrate taxa that indicate good water

quality, such as mayflies and stoneflies, were absent (see "Urban Biological Communities in a National Context"). Fish communities were characterized by a large proportion of species that can tolerate degraded water-quality conditions, such as central mudminnows and fathead minnows (Goldstein and others, 1999; Talmage and others, 1999). Factors that affect biological communities in urban streams include water and sediment chemistry and physical conditions such as hydrology and instream habitat.

Physical alterations, such as channelization and the high percentage of impervious area in urban basins, contribute to greater hydrologic variability (rapid

streamflow increases and decreases during storm events). Waterfalls and dams are common in urban streams in the Study Unit and may be barriers to fish migration (Talmage and others, 1999). Migration barriers can limit the total number of fish species.

Water Quality and Aquatic Biological Conditions have Characteristics Indicative of Dominant Land Uses

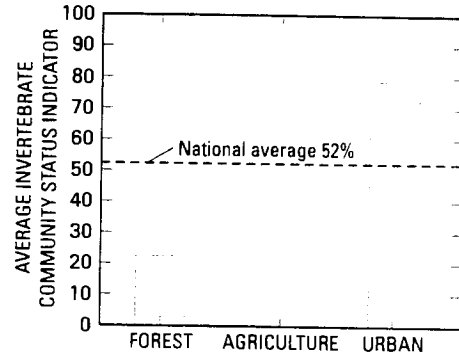
Sodium and chloride concentrations were greater in shallow ground water and streams in urban areas than in agricultural or forested areas. Chloride concentrations commonly exceeded the USEPA aquatic-life criteria of 230



URBAN BIOLOGICAL COMMUNITIES IN A NATIONAL CONTEXT- INVERTEBRATE COMMUNITIES REFLECT POOR RESOURCE QUALITY WITHIN URBAN STREAMS IN THE STUDY UNIT

Invertebrate communities indicated that the most degraded conditions occurred in 13 urban streams compared to 26 agricultural streams and 1 forest stream in the Study Unit. Urban streams were also among the most degraded in the Nation. Invertebrate communities in urban streams were composed of pollution tolerant species, such as true flies, with few sensitive species, such as mayflies and stoneflies.

Factors influencing invertebrate communities in urban streams may include elevated concentrations of PCBs, organochlorine pesticides (DDT, DDE and DDD), PAHs, and trace elements in streambed sediments. Concentrations of some of these compounds rank among the greatest in the Nation (McNellis and others, 2001; Kroening and others, 2000). In addition to chemical characteristics, modification to stream hydrology and removal of instream habitat may contribute to degraded conditions for aquatic communities in urban streams in the Study Unit.



Invertebrate Community Status Indicators (ICSI) scores were greatest in urban streams indicating poor aquatic resource (habitat and water) quality. The ICSI is a measure that summarizes species richness, tolerance, dominance, and trophic conditions, and that are associated with water-quality degradation. The indicator values increase with greater resource-quality degradation.

mg/L (Mitton and Payne, 1997; Fong, 2000; Fallon and Chaplin, 2001). Elevated sodium and chloride concentrations are the result of de-icers that are applied more heavily in urban areas.

Concentrations and yields of nutrients and suspended sediment in streams that drain agricultural areas were substantially greater than those that drain urban or forested areas (fig. 10). Increased nutrient concentrations have contributed to accelerated eutrophication and low dissolved-oxygen concentrations (Kroening, 2000), which adversely affect aquatic communities. Eutrophication has been most notable in the Minnesota River Basin. The greatest nitrate concentrations in the Minnesota River Basin were measured during rainfall runoff (Payne, 1994; Kroening and others, 2000). Exceedences of the USEPA drinking-water standard of 10 mg/L for nitrate occurred in less than 4 percent of urban and 38 percent of

agricultural ground-water samples, whereas nitrate was commonly undetected (less than 0.05 mg/L) in forested areas (fig. 11). Nitrate concentrations in shallow ground water increased with agricultural intensity, particularly in unconfined sand and gravel aquifers (Hanson, 1998), suggesting that underlying deeper aquifers, typically used for drinking water, have potential to be contaminated with nitrate (fig. 11) (Fong, 2000).

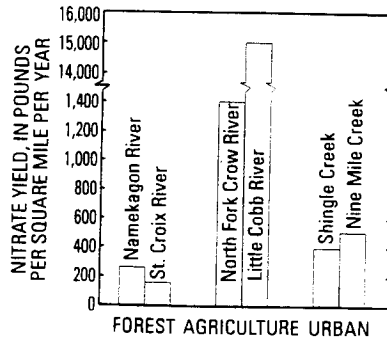


Figure 10. Nitrate yields were greatest in streams draining agricultural areas in the Study Unit.

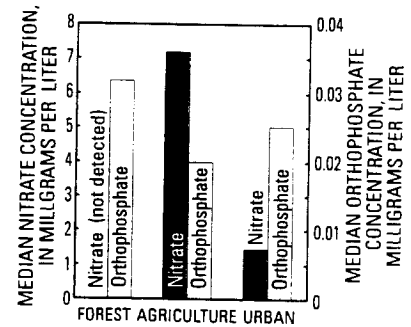


Figure 11. Nitrate concentrations were greatest in ground water in agricultural areas of the Study Unit.

The pesticides detected differed by land use. Herbicides were the most frequently detected in surface and ground water. Atrazine and its degradation product, deethylatrazine, were detected in all land-use settings (Fallon and others, 1997; Fong, 2000). Prometon, a herbicide used on road rights-of-way, was the most frequently detected herbicide in ground water in urban settings (Andrews and others, 1998). Organochlorine concentrations in streambed sediment were substantially

greater in urban streams than in agricultural or forest streams (McNellis and others, 2001).

Volatile organic compounds were most commonly detected in urban areas. In ground water, the most frequently detected VOCs (carbon disulfide and chloromethane) were in shallow aquifers in urban areas, but at concentrations generally less than 1 µg/L (fig. 12) (Andrews and others, 1998). VOCs also were detected in ground-water samples from agricultural areas, but at concentrations and detection frequencies less than urban areas (Ruhl and others, 2000). In urban streams, the greatest concentrations of VOCs were detected following storm runoff and during winter low flows.

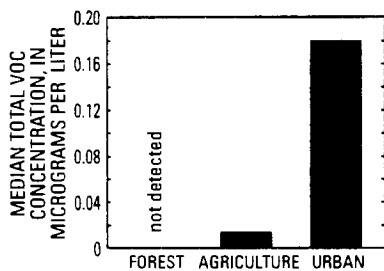


Figure 12. Total volatile organic compound concentrations were greatest in ground water in urban areas in the Study Unit.

Trace concentrations of PCBs and DDE (a degradation product of DDT) were detected in fish throughout the Study Unit (Biedron and Helwig, 1991). PCB concentrations in common carp fillet tissue have decreased at different rates in each land-use setting since their use was discontinued in the 1970s (Duffee, 1976) (fig. 13). Concentrations of these compounds were greater in fish and sediment from stream reaches near urban areas (Fallon and others, 1997; Lee and Anderson, 1998; McNellis and others, 2001).

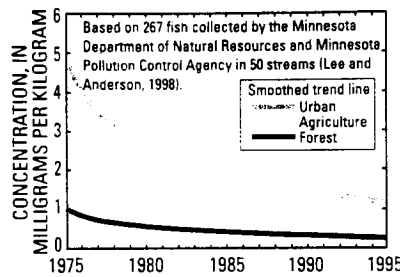


Figure 13. Polychlorinated biphenyl concentrations in common carp filets collected from streams in the Study Unit have decreased since 1975.

Streambed-sediment concentrations of lead, zinc (fig. 14), cadmium, and copper were greater in urban areas than other land-use settings (Kroening and others, 2000). In streams draining agricultural and forested areas, trace-element concentrations in streambed sediment probably reflected natural geochemistry. Mercury concentrations in fish livers were greater in streams draining land uses other than urban settings (Kroening and others, 2000). Agricultural and urban activities contribute to elevated suspended-sediment concentrations and bacteria counts in small streams. Suspended-sediment concentrations were greatest in agricultural streams.

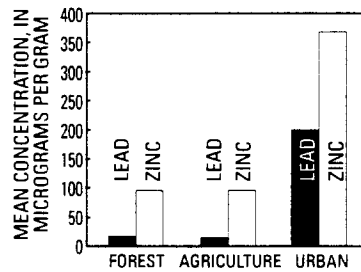


Figure 14. Lead and zinc concentrations were greatest in streambed sediments in urban areas in the Study Unit.

Aquatic biological communities are affected by chemical, hydrological, and physical conditions in streams and serve as good indicators of water quality. Community composition indicated more degraded con-

ditions in urban streams than in forest or agricultural streams (Lee and others, 1999; Talmage and others, 1999). Invertebrate communities in urban streams are composed of fewer mayflies, stoneflies, and caddisflies than streams draining agricultural and forested land (fig. 15) (Lee and others, 1999). Fish communities in urban streams were dominated by species tolerant of low dissolved-oxygen concentrations and warm temperatures (Goldstein and others, 1999; Talmage and others, 1999). Fish biomass and phytoplankton biovolume are indicators of stream productivity. The greatest fish biomass (usually in the form of species such as common carp) and phytoplankton biovolumes were measured in agricultural streams (fig. 16).

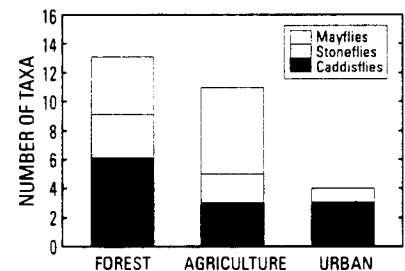


Figure 15. Total number of mayflies, stoneflies and caddisflies, indicators of good water-quality conditions, was greatest in streams draining forested areas in the Study Unit.

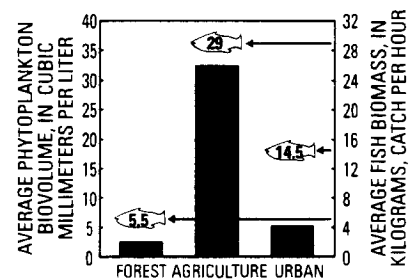


Figure 16. Phytoplankton biovolume and fish biomass were greatest in streams draining agricultural areas in the Study Unit.

Land Use Influences Water-Supply Aquifers

The Prairie du Chien-Jordan aquifer, which occurs in dolomite and sandstone of Cambrian to Ordovician age, is the principal bedrock aquifer throughout much of the Study Unit (fig. 17), supplying approximately 75 percent of the ground water withdrawn in the area for public and industrial supply. In certain areas, termed confined portion, bedrock or glacial deposits having low permeability overlie the aquifer. In other areas, termed unconfined portion, glacial sand and gravel deposits having greater permeability overlie the aquifer. The hydrogeologic characteristics of these overlying units affect the downward movement of water and contaminants from the land surface into the aquifer.

Water in the unconfined portion of the aquifer appears to be affected to a greater degree by human-related activities than water in the confined portion of the aquifer. Nitrate concentrations were greater in the unconfined portion of the aquifer. In the unconfined portion of the aquifer, nitrate in 8 percent of the wells sampled exceeded the USEPA drinking-water standard of 10 mg/L. In the confined portion of the aquifer, no samples exceeded 10 mg/L of nitrate. Phosphorus concentrations generally were about one-tenth of nitrate concentrations. In about 40 percent of water samples from confined and unconfined portions of the aquifer, concentrations of iron and manganese in water samples from confined and unconfined portions of the aquifer exceeded drinking-water guidelines.

Radon concentrations ranged from 100 to 2,700 pCi/L and

exceeded the suspended USEPA drinking-water standard of 300 pCi/L in 68 percent of the water samples from the unconfined portion of the aquifer and 64 percent from the confined portion of the aquifer. Tritium concentrations in ground water indicated that water in the unconfined portion of the aquifer was recharged more recently than water in the confined portion of the aquifer.

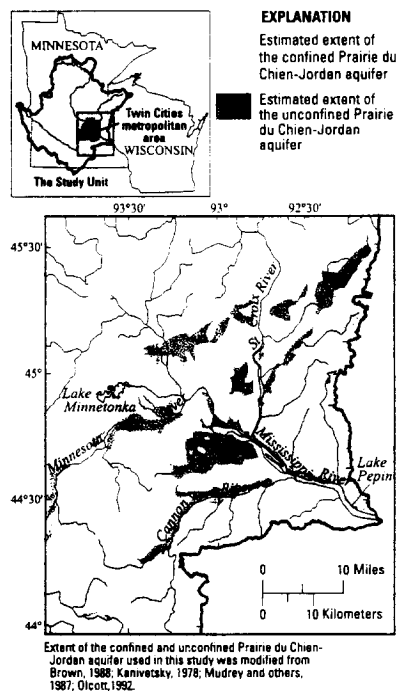


Figure 17. Estimated extent of the Prairie du Chien-Jordan aquifer in part of the Study Unit.

Arsenic concentrations in the confined and unconfined portions of the aquifer ranged from less than the method reporting limit (1 $\mu\text{g/L}$) to 7 $\mu\text{g/L}$. These concentrations do not exceed the current USEPA drinking-water standard of 50 $\mu\text{g/L}$.

Seven different pesticide compounds were detected in water samples. Atrazine and its degradation product, deethylatrazine, were most frequently detected. Atrazine was

detected in water from 36 percent of wells in the confined portion of the aquifer and 52 percent of wells in the unconfined portion of the aquifer. VOCs were detected in 82 percent of the water samples, but none at concentrations exceeding 1 $\mu\text{g/L}$. More VOCs were detected in water samples from the unconfined portion of the aquifer than from the confined portion.

Water Quality and Aquatic Biology of Large Rivers

Water quality and aquatic biology in the large rivers of the Study Unit (the Mississippi, Minnesota, and St. Croix) represent the cumulative quality of their tributaries. The tributaries of the Minnesota River drain primarily agricultural land, the tributaries of the St. Croix River drain primarily forested land, and the tributaries of the Mississippi River drain primarily agricultural and forested land. Because of agricultural activities and natural conditions, water in the Minnesota River contains elevated concentrations and yields of nutrients, suspended sediments, and pesticides (Fallon and others, 1997; Kroening, 2000). The aquatic biological community contains fewer invertebrate and algal taxa, but greater chlorophyll-*a* concentrations associated with greater nutrient concentrations (Kroening, 2000; Lee and ZumBerge, 2000) (fig. 18). In contrast, the St. Croix River and the Mississippi River upstream from the TCMA have low nutrient concentrations, relatively clear water, and low suspended-sediment and pesticide concentrations (Fallon and others, 1997; Fallon, 1998; Kroening, 2000). Downstream from the TCMA, and below the confluence of the Minnesota and St. Croix Rivers, water quality in

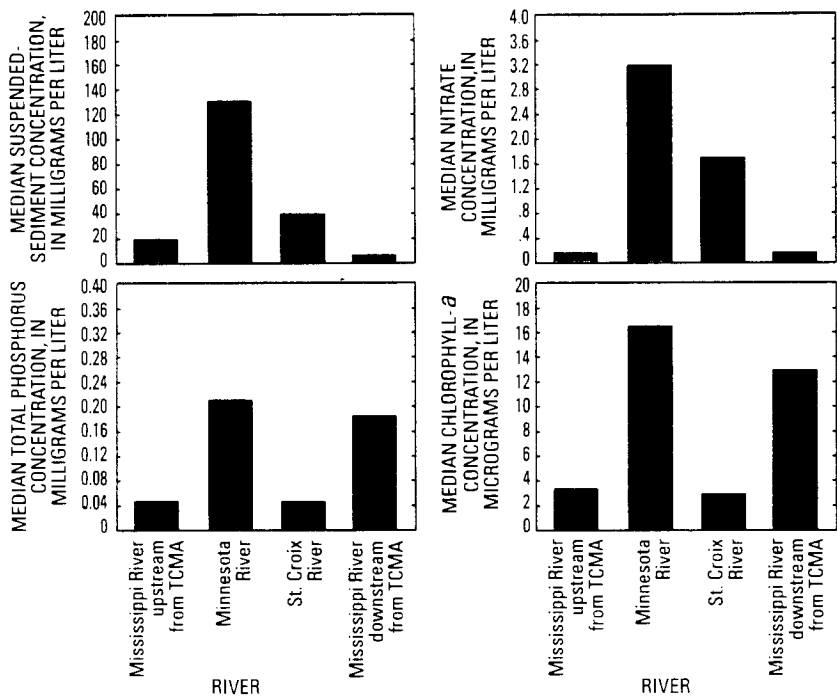
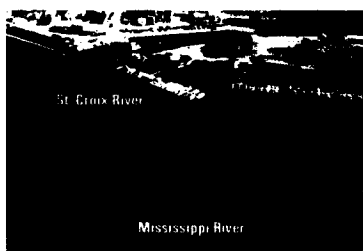


Figure 18. Median concentrations of nitrate, total phosphorus, suspended sediments, and chlorophyll-a were generally lower upstream from the Twin City Metropolitan Area (TCMA) and were greatest in the Minnesota River.

the Mississippi River results from a complex mixture of water and chemical constituents. Concentrations of nutrients, suspended sediments, and pesticides in the Mississippi River increase at the confluence with the Minnesota River and decrease slightly, due to dilution downstream from the confluence with the St. Croix River (fig. 19) (Fallon, 1998; Kroening, 2000).



Aerial view of the confluence of the St. Croix and the Mississippi Rivers. (Photograph by James R. Stark.)

Nitrate concentrations in the Mississippi and St. Croix Rivers did not exceed the USEPA drinking-water standard of 10 mg/L

(Kroening, 1998a, 2000). Eleven percent of the samples from the Minnesota River near Jordan, Minn., exceeded the standard. The most noticeable trends in the Mississippi, Minnesota, and St. Croix Rivers during 1984–93 were an increase in nitrate concentrations and a decrease in total ammonia concentrations in the TCMA (fig. 20) (Kroening and Andrews, 1997). These trends were not observed at other sites. These ammonia reductions are probably the result of nitrification processes used at the three largest wastewater treatment facilities in the TCMA, which convert ammonia-nitrogen to nitrate. This process has resulted in wastewater effluents that are less toxic to fish and other aquatic life. Nitrate concentrations, however, may contribute to eutrophication.

Total phosphorus concentrations in parts of the Minnesota River and in the Mississippi River downstream from the TCMA frequently exceeded the USEPA guideline of 0.1 mg/L to prevent eutrophication

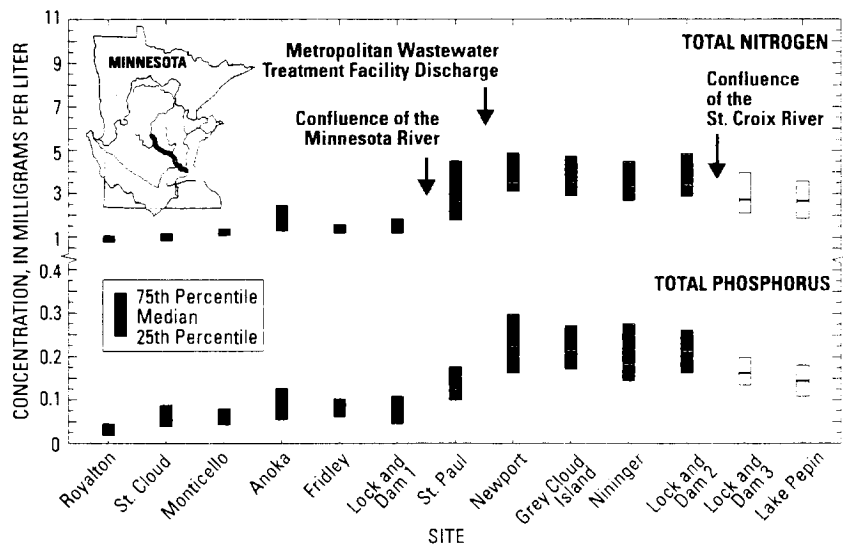


Figure 19. Total nitrogen and phosphorus concentrations in the Mississippi River increase downstream from the confluence of the Minnesota River and decrease downstream from the confluence of the St. Croix River.

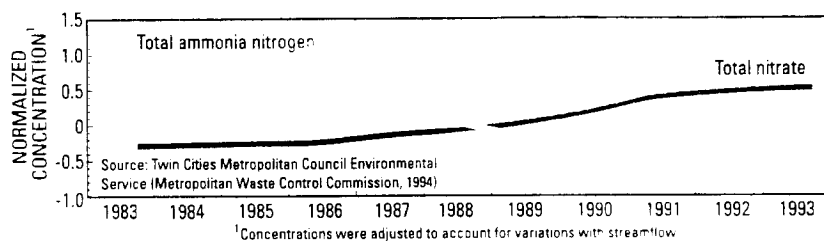
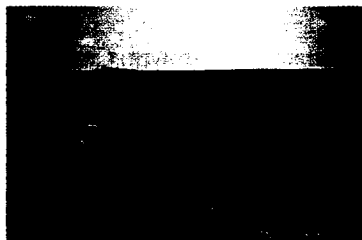


Figure 20. Modifications in wastewater-treatment processes have changed measured total nitrate and total ammonia nitrogen concentrations in the Mississippi River at Newport, Minn.

(Kroening, 1998b, 2000). Phosphorus concentrations and loads to the rivers originate from both point and nonpoint sources. The major point sources are wastewater treatment facilities, whereas the major nonpoint sources are from agriculture in the Minnesota River Basin. During low streamflow conditions, more phosphorus comes from wastewater treatment facilities, whereas during high streamflow conditions, nonpoint sources dominate. Dissolved orthophosphate concentrations generally were greatest at sites downstream from wastewater discharges in the TCMA (Kroening, 1998b, 2000). Eutrophication of Lake Pepin has been linked to elevated phosphorus concentrations in the Mississippi River (Minnesota Pollution Control Agency, 1989).



Wastewater treatment facilities introduce contaminants such as nutrients and chloride to streams. (Photograph by Scott Murray Photography.)

Biochemical oxygen demand (BOD) of materials discharged from wastewater treatment facilities

has resulted in dissolved-oxygen concentrations in the Mississippi and Minnesota Rivers (Johnson and Aasen, 1989; Minnesota Pollution Control Agency, 1985) that are sometimes less than the USEPA guideline of 5 mg/L for the protection of aquatic life (U.S. Environmental Protection Agency, 1986).



The runoff of agricultural chemicals and sediment affects water quality in nearby streams and rivers. (Photograph by Scott Murray Photography.)

Suspended sediment adversely affects aquatic life by limiting light and covering habitat. Suspended sediment also transports nutrients, trace elements, and organic compounds attached to particles. The

greatest concentration of suspended sediment in the large rivers was in the Minnesota River (Kroening, 2000). The primary contributors of suspended sediment to the Minnesota River are the tributary watersheds in the central and southeastern parts of the Minnesota River Basin (Payne, 1994). Concentrations were lower in the St. Croix River and in the upper reaches of the Mississippi River.

Pesticides frequently were detected in the large rivers, but no concentrations exceeded applicable drinking-water standards or guidelines (Fallon and others, 1997; Fallon, 1998). Herbicides detected in all large rivers include the row crop herbicides alachlor, atrazine, and its degradation product deethylatrazine, cyanazine, and metolachlor. In and downstream from the TCMA, insecticides were frequently detected in water, and although use was discontinued in the early 1970s, DDT and its degradation products DDE and DDD were frequently detected in fish tissue and bed sediment.

Streambed sediment in the Mississippi River within and downstream from the TCMA contained the greatest number of OCs (Fallon and others, 1997; Fallon, 1998). PCB concentrations in streambed sediments have decreased over time (Anderson and Perry, 1999). Fish tissue concentrations have paralleled this decline (Lee and Anderson, 1998).

Human activities have had a strong influence on the occurrence and distribution of trace elements in large rivers of the Study Unit. The TCMA is the largest source of trace elements to rivers in the Study Unit. Trace-element data collected in the TCMA during 1992 by the Metropolitan Waste Control Com-

Control Commission (1994) indicate that concentrations of most trace elements in the water were less than applicable standards and guidelines, with the periodic exceptions of mercury and copper. Concentrations of cadmium, lead, mercury, and zinc were greatest in streambed-sediment samples within or immediately downstream from the TCMA (Wiener and others, 1984; Kroening and others, 2000). An industrial pretreatment program that began in the early 1980s has reduced the amount of trace elements discharged to the Mississippi River. For example, zinc concentrations have decreased an average of 80 percent (Anderson and Perry, 1999) (fig. 21) since the pretreatment program began.

Treated wastewater and untreated animal waste in the Study Unit also contribute to increased counts of fecal bacteria in the large rivers. Fecal bacteria counts were greatest in the Minnesota River and in the Mississippi River as it flowed through the TCMA. Approximately 40 percent of samples collected in the Minnesota River Basin exceeded the Minnesota and Wisconsin State freshwater standards for recreational use of 200 col/100 mL (Payne, 1994; Wisconsin Department of Natural

Resources, 1997; Minnesota Pollution Control Agency, 1999). Data collected by the Metropolitan Waste Control Commission (1994) indicate that during 1992, 25 percent of the water samples collected in the Mississippi River immediately downstream from the Minnesota River and the Metropolitan Wastewater Treatment Plant outfall exceeded freshwater standards for recreational use regarding bacteria.

Changes in the habitat of the large rivers have been caused by the construction of locks and dams, dredging to maintain navigation channels, modifications to stream morphology, and changes in land use. (see "Riparian Buffer Zones Affect the Quality of Midwestern Streams and Rivers," p. 9). Instream habitat and fish community conditions in the large rivers differ among areas of forest, urban, and agricultural lands. Diverse aquatic biological communities and relatively undisturbed riffle-pool morphology are found in the St. Croix River and the upper reaches of the Mississippi River in forested areas. Drainage of wetlands, loss of riparian vegetation, and channel straightening in the Minnesota River Basin have reduced habitat, modified hydraulic conditions, and changed water quality.

In the Mississippi River, the construction and maintenance of locks and dams have altered physical habitat for fish, invertebrates, and algae by changing streamflow from free-flowing to impounded, and altering the natural hydrology and the physical structure of the channel. As a result, the river has changed from a meandering, flowing system, which periodically overran its banks and flood plain, to a series of impoundments connected by dredged channels where the streamflow and water levels are controlled. The impoundments change the physical structure of the river, the diversity of aquatic habitats, and water quality. Impoundments reduce the velocity and warm the water in the pools. Reduced velocity causes sediment to settle, changing the composition of the substrate on the bottom of impoundments to fine-grained material (sand and silt). Nutrients and contaminants associated with sediment particles are concentrated in the bottom sediments of the pools.

The addition of nutrients from wastewater treatment facilities and from agricultural activities, combined with greater water temperatures and greater light penetration, stimulate algal growth. Concentrations of chlorophyll-*a* and phytoplankton biovolume in the Minnesota River at Jordan, Minn., and in the lower Mississippi River sites at Hastings, Minn., and at Red Wing, Minn., are greater than twice the concentrations measured at the upper Mississippi River site at Royalton, Minn. (Kroening, 2000), indicating greater phytoplankton abundance and primary production (fig. 22). High concentrations of nutrients, coupled with the environmental conditions of sufficient

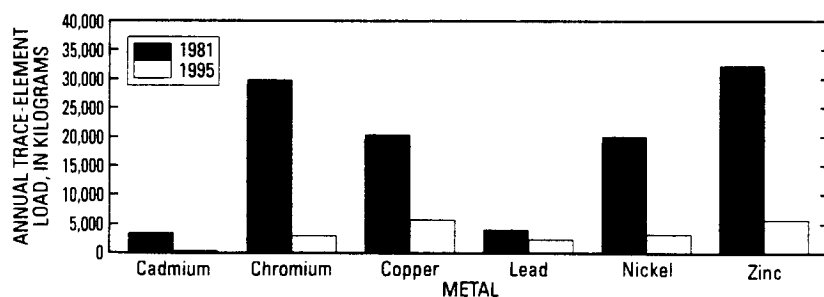


Figure 21. Annual trace-element load from the Metropolitan Wastewater Treatment Plant by industrial users has decreased since 1981.

light and temperature, can result in eutrophication and subsequent oxygen deficits. Blue-green algal blooms were suspected of causing low dissolved-oxygen concentrations in Lake Pepin during the summer of 1988 (an abnormally dry period) that resulted in fishkills (Minnesota Pollution Control Agency, 1989).

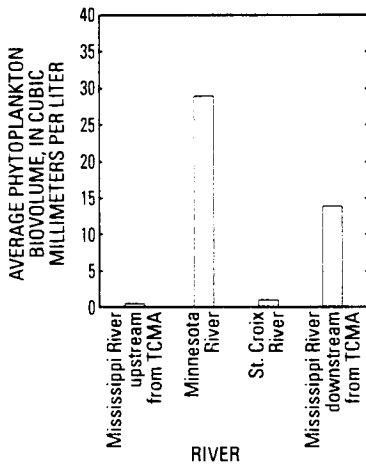


Figure 22. Phytoplankton biovolume was least in the Mississippi River upstream from the Twin Cities Metropolitan Area (TCMA) and was greatest in the Minnesota River.

Invertebrate communities also have been influenced by environmental and morphologic conditions in the large rivers of the Study Unit. Sensitive invertebrate species (mayflies, stoneflies, and caddisflies) were most abundant in the St. Croix River, which drains primarily forested land. These sensitive taxa were least abundant in and downstream from the TCMA (fig. 23), where tolerant taxa such as Diptera (true flies) and Oligochaeta (aquatic worms) composed a large portion of the invertebrate community. Several species of mollusks are no longer present, due to commercial harvesting, loss and modification of habitat, water contamination, deposition of silt, and the introduction of zebra

mussels (Mueller, 1993). Contaminants such as cadmium and mercury in the sediments have accumulated in burrowing mayflies and may present a substantial source of trace element contaminants to fish, particularly in Lake Pepin (Beauvais and others, 1995).

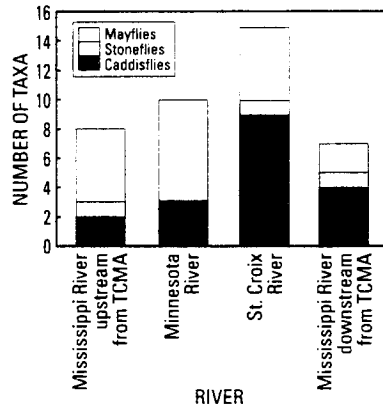


Figure 23. Total number of mayflies, stoneflies, and caddisflies was least downstream from the Twin Cities Metropolitan Area (TCMA) and greatest in the St. Croix River.

Several chemical and physical factors affect the abundance and distribution of fish species. St. Anthony Falls in Minneapolis, Minn., on the Mississippi River, and the dam at St. Croix Falls, Wis., on the St. Croix River, form two major barriers to fish migration. These barriers have resulted in differences in fish species composition (Underhill, 1989). More species occur downstream of the barriers (fig. 24) (Goldstein and others, 1999; Underhill, 1989).

Other differences in the fish community distribution exist among large rivers in the Study Unit. The Mississippi River upstream from the TCMA and the St. Croix River upstream from Taylors Falls have fish species that thrive in cold water. Fish commu-

nities at these river sections are dominated by cool water and riverine species such as redhorse and smallmouth bass. Farther downstream, particularly in the Mississippi River downstream from the TCMA, the fish community consists of catfish, buffalo fish, freshwater drum, carpsuckers, and gizzard shad that tolerate warm water. The pattern of thermal preference is also consistent in the Minnesota and St. Croix Rivers. Lake species that are adapted to still water with high thermal ranges are found in and downstream from the TCMA.

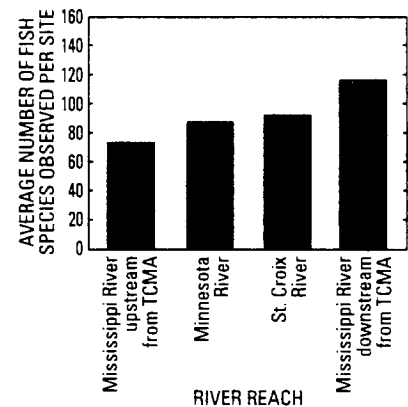


Figure 24. Total number of fish species was greatest in the Lower Mississippi River downstream from the Twin Cities Metropolitan Area (TCMA).

The distribution of fish also differs by trophic status in the large rivers. Upstream from the TCMA, fish (northern hogsucker, golden and shorthead redhorse, hornyhead chub, common shiner, smallmouth bass, and two species of darter) that primarily consume invertebrates species that require a gravel or cobble substrate were abundant compared to downstream from the TCMA where fish (common carp and buffalo fish) that primarily consume detritus were more abundant. Downstream from the TCMA, species that feed on detritus

rely on filter feeding and suctioning of the bottom sediments for fine particulate organic matter.

The reduction in river velocity resulting from hydrologic modifications, such as impoundments, also alters the composition of the fish communities in the rivers. Species downstream from the TCMA tend to be associated with still-water habitats, whereas species upstream from the TCMA are associated more with flowing-water habitat. The abundance of fish (gizzard shad and emerald shiner) that eat plankton in the Mississippi River downstream from the TCMA indicates that a plankton community more common to lakes exists in that part of the river.



Clean drinking water is important to everyone. (Photograph from U.S. Geological Survey files.)

An indicator of the general quality of aquatic resources is the presence of contaminants in fish. Two contaminants, PCBs and DDE (a degradation product of DDT), were the most frequently

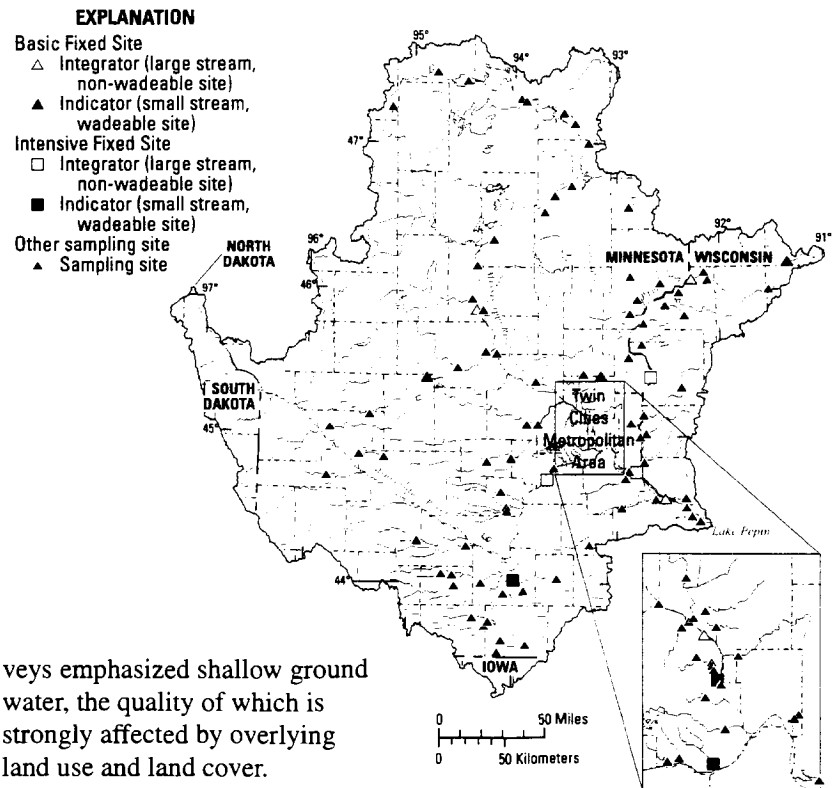
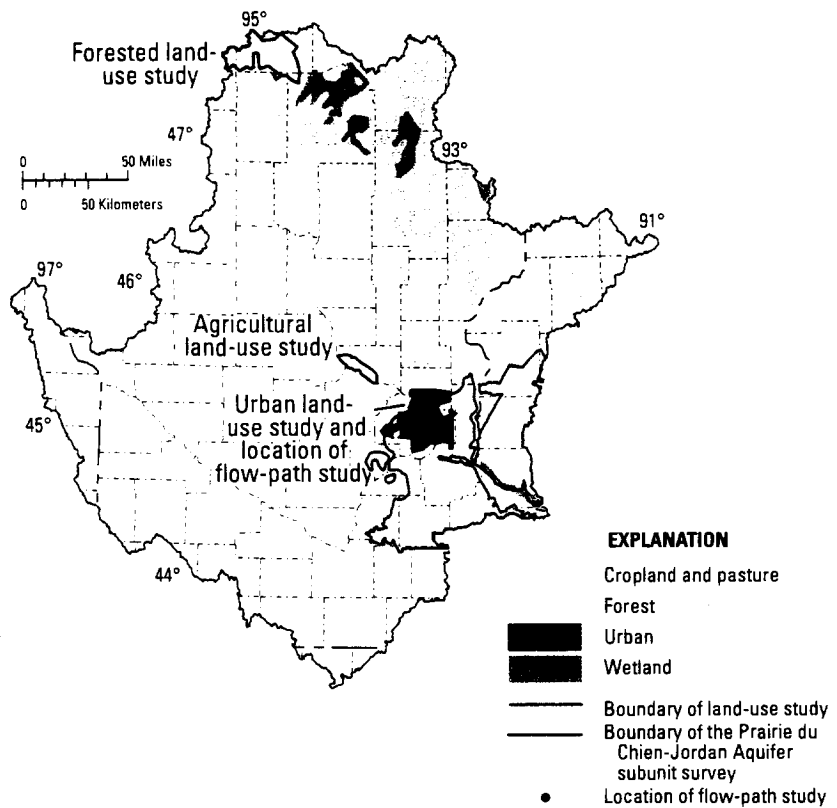
detected OCs in fish in the Study Unit. These contaminants in fish were greatest in the Mississippi River downstream from the TCMA. PCB and DDE concentrations in common carp tissue generally were greater in the Mississippi than in the Minnesota or St. Croix Rivers, and DDE concentrations generally increased in the Mississippi River main stem from Grand Rapids, Minn., downstream to Red Wing, Minn. Although concentrations have decreased over time (Lee and Anderson, 1998), PCBs and DDE continue to be detected in fish tissue, but at relatively low concentrations of less than 1 $\mu\text{g}/\text{kg}$ (micrograms per kilogram).

STUDY UNIT DESIGN

During 1996–98, about 4,200 water-quality aquatic-biological samples from about 240 sites were collected in the Study Unit, processed, and analyzed, using nationally consistent protocols and methods (Gilliom and others, 1995). The NAWQA design included physical, chemical, and aquatic-biological aspects of surface water and ground water for the entire Study Unit. Six sampling components were included in the sampling design. Each component involved measurements of water-quality or aquatic biological characteristics at one or more spatial or temporal scales. Three of the sampling components addressed surface water and aquatic biology, and three addressed ground water. A detailed description of the design and implementation of these water-quality studies is contained in Stark and others (1999).

Water quality in streams was assessed through water-chemistry and aquatic-biological studies. The surface-water and aquatic biology components included (1) stream sites that integrate multiple land uses and encompass large watersheds (integrator site network), (2) stream sites that indicate homogeneous and more specific land uses (indicator site network), and (3) stream sites sampled for special studies (synoptic surveys).

Ground-water quality was assessed for aquifer/land-use combinations using three sampling strategies: (1) a regional study of a selected major aquifer (subunit survey), (2) targeted-area studies in selected land uses (land-use studies), and (3) a localized study of processes occurring along shallow ground-water-flow paths (flow-path study). These studies and sur-



veys emphasized shallow ground water, the quality of which is strongly affected by overlying land use and land cover.

SUMMARY OF DATA COLLECTION IN THE UPPER MISSISSIPPI RIVER BASIN, 1995-98

Study component	Purpose of component and types of data collected	Types of sites sampled	Number of sites	Sampling frequency and period
Stream Chemistry				
Basic Fixed Sites— large rivers	Major ions, organic carbon, suspended sediment, nutrients, and streamflow were measured to describe concentrations and amounts of constituents transported in major tributaries in and from the Study Unit.	Sites on the Mississippi, Minnesota, and St. Croix Rivers draining 1,510 to 46,800 mi ² that integrate the effects of agricultural, urban, and forested land use and physiographic regions.	4 in 1996-97; 3 in 1998	Monthly beginning in March 1996 and during selected runoff events
Basic Fixed Sites— indicator tributaries	Major ions, organic carbon, suspended sediment, nutrients, and streamflow were measured to determine the effects of land use (undeveloped, urban, or agricultural) and surficial geology on stream-water quality.	Streams draining 27.3 to 232 mi ² of homogeneous agricultural, urban, or forested areas on unsorted or sorted surficial glacial deposits.	3 in 1996; 2 in 1997-98	Monthly beginning in March 1996 and during selected runoff events
Intensive Fixed Site— large rivers	Major ions, organic carbon, suspended sediment, nutrients, pesticides, and streamflow were determined to define short-term temporal variability.	Sites on the Mississippi, Minnesota, and St. Croix Rivers draining 6,150 to 37,000 mi ² .	3	Monthly beginning in March 1996 and during selected runoff events
Intensive Fixed Site— indicator tributaries	Major ions, organic carbon, suspended sediment, nutrients, pesticides, and streamflow were determined to define short-term temporal variability. Volatile organic compounds were determined at two urban sites.	Streams draining 28.2 to 130 mi ² in homogeneous agricultural and urban areas.	3	Weekly or biweekly during April through August 1997
Snowmelt synoptic survey	Nutrients and suspended sediment were determined using modified NAWQA protocols to characterize instantaneous concentrations and yields during increasing streamflow of snowmelt runoff.	Streams draining 10 to 46,800 mi ² .	41	Once in March or April 1997
Stream Ecology				
Bed sediment and tissue	Trace elements and hydrophobic-organic compounds in fish tissue and streambed sediment to determine occurrence and distribution of these compounds throughout the Study Unit.	Sites with drainage areas from 20 to 47,300 mi ² draining a variety of land use.	Fish sampled at 25 sites, streambed sediment at 27 sites.	1995-96
Basic Fixed Sites— indicator tributaries	Fish, benthic invertebrates, phytoplankton, periphyton, and instream habitat were sampled or characterized to determine the community structure and to evaluate the association between land use and aquatic communities.	Same as for stream chemistry	6 in 1996; 5 in 1997-98	One each fall 1996-98
Basic Fixed Sites— large rivers	Fish, benthic invertebrates, phytoplankton, periphyton, and instream habitat were sampled or characterized to determine the spatial distribution of aquatic communities and to evaluate the association between land use and aquatic communities.	Same as for stream chemistry	7	One each fall 1996-98
Urban synoptic study	Nutrients, suspended sediment, pesticides, organic carbon, phytoplankton, and chlorophyll- <i>a</i> were analyzed. Aquatic community sampling included fish and invertebrate community sampling and instream habitat to determine how water quality and aquatic communities differ in response to changes in population density.	Streams with drainage areas ranging from 9.9 to 152 mi ² draining urban areas in the Twin Cities metropolitan area.	13	September-October 1997
Mid-continent agricultural synoptic study	Nutrients, suspended sediment, pesticides, organic carbon, phytoplankton and chlorophyll- <i>a</i> were analyzed. Aquatic community sampling included fish and invertebrate community sampling and instream habitat characterization to determine how water quality and aquatic communities differ in response to changes in local-scale riparian cover and to basin-scale soils.	Sites with drainage areas from 60 to 317 mi ² draining land that was greater than 87 percent agricultural land use.	24	August 1997
Longitudinal synoptic study	Nutrients, suspended sediment, major ions, pesticides, organic carbon, chlorophyll- <i>a</i> , and organic compounds indicative of wastewater were analyzed. Aquatic community sampling included fish and invertebrates and instream habitat to characterize the water quality and aquatic communities along the Mississippi River.	Sites with drainage areas ranging from 32 to 46,800 mi ² along the Mississippi River main stem from Lake Itasca to Red Wing, Minnesota.	Sampled aquatic communities at 12 sites and water chemistry at 19 sites.	July and August of 1998
Ground-Water Chemistry				
Bedrock aquifer survey	Major ions, nutrients, dissolved organic carbon, trace elements, pesticides, volatile organic compounds, radon, and tritium were analyzed to describe the water quality and natural chemical patterns in unconfined and confined portions of the most frequently used bedrock aquifer in the Study Unit.	Existing domestic wells completed in the Prairie du Chien-Jordan aquifer.	25 wells in the unconfined portion 25 wells in the confined portion	July-September 1996
Land-use effects— surficial aquifer	Major ions, nutrients, dissolved organic carbon, pesticides, volatile organic compounds, and tritium were analyzed to determine the effects of specific land uses (urban, agricultural, and forested) on the quality of shallow ground water.	Monitoring wells completed at the water table in the surficial sand and gravel aquifer.	30 wells in the urban study 29 wells in the agricultural study 15 wells in the forested study	June-July 1996, May-September 1998, June 1998
Variations along flow— surficial aquifer	Major ions, nutrients, dissolved organic carbon, trace elements, pesticides, volatile organic compounds, radon, tritium, dissolved gases, and chlorofluorocarbons were analyzed to describe the effects of urban land use on the quality of shallow ground water along ground-water flow from an area of recharge to an area of discharge to a stream.	Monitoring and multipoint wells (open to the aquifer at different depths) completed in the surficial sand and gravel aquifer.	1 monitoring well and 6 multipoint wells	July 1997, October 1997, August 1998

GLOSSARY

Alkalinity - The alkalinity of a solution is the capacity for solutes it contains to react with and neutralize acid.

Aquatic-life criteria - Water-quality guidelines for protection of aquatic life. Often refers to U.S. Environmental Protection Agency water-quality criteria for protection of aquatic organisms. See also Water-quality guidelines and Water-quality criteria.

Bioaccumulation - The biological sequestering of a substance at a higher concentration than that at which it occurs in the surrounding environment or medium. Also, the process whereby a substance enters organisms through the gills, epithelial tissues, dietary, or other sources.

Confined aquifer (artesian aquifer) - An aquifer that is completely filled with water under pressure and that is overlain by material that restricts the movement of water.

Degradation products - Compounds resulting from transformation of an organic substance through chemical, photochemical, and/or biochemical reactions.

Drinking-water standard or guideline - A threshold concentration in a public drinking-water supply, designed to protect human health. As defined here, standards are U.S. Environmental Protection Agency regulations that specify the maximum contamination levels for public water systems required to protect the public welfare; guidelines have no regulatory status and are issued in an advisory capacity.

EPT richness index - An index based on the sum of the number of taxa in three insect orders, Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies), that are composed primarily of species considered to be relatively intolerant to environmental alterations.

Eutrophication - The process by which water becomes enriched with plant nutrients, most commonly phosphorus and nitrogen.

Human health advisory - Guidance provided by U.S. Environmental Protection Agency, State agencies or scientific organizations, in the absence of regulatory limits, to describe acceptable contaminant levels in drinking water or edible fish.

Index of Biotic Integrity (IBI) - An aggregated number, or index, based on several attributes or metrics of a fish community that provides an assessment of biological conditions.

Load - General term that refers to a material or constituent in solution, in suspension, or in transport; usually expressed in terms of mass or volume.

Nonpoint source - A pollution source that cannot be defined as originating from discrete points such as pipe discharge. Areas of fertilizer and pesticide applications, atmospheric deposition, manure, and natural inputs

from plants and trees are types of nonpoint source pollution.

Organochlorine compound - Synthetic organic compounds containing chlorine. As generally used, term refers to compounds containing mostly or exclusively carbon, hydrogen, and chlorine. Examples include organochlorine insecticides, polychlorinated biphenyls, and some solvents containing chlorine.

Point source - A source at a discrete location such as a discharge pipe, drainage ditch, tunnel, wells, concentrated livestock operation, or floating craft.

Polychlorinated biphenyls (PCBs) - A mixture of chlorinated derivatives of biphenyl, marketed under the trade name Aroclor with a number designating the chlorine content (such as Aroclor 1260). PCBs were used in transformers and capacitors for insulating purposes and in gas pipeline systems as a lubricant. Further sale for new use was banned by law in 1979.

Polycyclic aromatic hydrocarbon (PAH) - A class of organic compounds with a fused-ring aromatic structure. PAHs result from incomplete combustion of organic carbon (including wood), municipal solid waste, and fossil fuels, as well as from natural or anthropogenic introduction of uncombusted coal and oil. PAHs include benzo(a)pyrene, fluoranthene, and pyrene.

Tolerant species - Those species that are adaptable to (tolerant of) human alterations to the environment and often increase in number when human alterations occur.

Unconfined aquifer - An aquifer whose upper surface is a water table; an aquifer containing unconfined ground water.

Water-quality criteria - Specific levels of water quality which, if reached, are expected to render a body of water unsuitable for its designated use. Commonly refers to water-quality criteria established by the U.S. Environmental Protection Agency. Water-quality criteria are based on specific levels of contaminants that would make the water harmful if used for drinking, swimming, farming, fish production, or industrial processes.

Suspended sediment - Particles of rock, sand, soil, and organic detritus carried in suspension in the water column, in contrast to sediment that moves on or near the streambed.

Water-quality guidelines - Specific levels of water quality which, if reached, may adversely affect human health or aquatic life. These are nonenforceable guidelines issued by a governmental agency or other institution.

Yield - The mass of material or constituent transported by a river in a specified period of time divided by the drainage area of the river basin.

REFERENCES

- Anderson, J.P., and Perry, J., 1999, Comparison of temporal trends in ambient and compliance trace element and PCB data in Pool 2 of the Mississippi River, 1985–95: *Environmental Management*, v. 24, no. 4, p. 497–507.
- Andrews, W.J., Fallon, J.D., and Kroening, S.E., 1995, Water-quality assessment of part of the Upper Mississippi River Basin, Minnesota and Wisconsin—Volatile organic compounds in surface and ground water, 1978–94: U.S. Geological Survey Water-Resources Investigations Report 95–4216, 39 p.
- Andrews, W.J., Fong, A.L., Harrod, L., and Dittes, M.E., 1998, Water-quality assessment of part of the Upper Mississippi River Basin, Minnesota and Wisconsin—Ground-water quality in an urban part of the Twin Cities Metropolitan Area, Minnesota, 1996: U.S. Geological Survey Water-Resources Investigations Report 97–4248, 54 p.
- Beauvais, S.L., Wiener, J.G., and Atchison, G.J., 1995, Cadmium and mercury in sediment and burrowing mayfly nymphs (*Hexagenia*) in the Upper Mississippi River, USA: *Archives of Environmental Contamination and Toxicology* 28, p. 178–183.
- Biedron, C.J., and Helwig, D.D., 1991, PCBs in common carp in the Upper Mississippi River: St. Paul, Minnesota, Minnesota Pollution Control Agency, 41 p.
- Brown, B.A., 1988, Bedrock geology of Wisconsin: Wisconsin Geological and Natural History Survey Map 88–7, 1 sheet.
- Durfee, R.L., 1976, Production and usage of PCBs in the United States, in National Conference on Polychlorinated Biphenyls, November 19–21, 1976: U.S. Environmental Protection Agency Report EPA–56016–75–004, Washington, D.C., p. 103–107.
- Fallon, J.D., 1998, Pesticides and pesticide metabolites in selected streams of the Upper Mississippi River Basin, 1997 [abs.]: Minnesota Water '98—Protecting Minnesota's Water Supplies, 6th Biennial Conference on Minnesota Water Issues, May 5–6, 1998, University of Minnesota, Water Resources Research Center, p. 92.
- Fallon, J.D., and Chaplin, Brian, 2001, Chloride-related studies in streams and ground water of the Twin Cities metropolitan area, Minnesota 1996–98—A summary of established and new results [abs.]: Chloride Impacts to Local Waters, January 10, 2001, St. Paul, Minn.
- Fallon, J.D., Fong, A.L., and Andrews, W.J., 1997, Water-quality assessment of part of the Upper Mississippi River Basin, Minnesota and Wisconsin—Pesticides in streams, streambed sediment, and ground water, 1974–94: U.S. Geological Survey Water-Resources Investigations Report 97–4141, 53 p.
- Farnsworth, D.K., Thompson, E.S., and Peck, E.L., 1982, Annual free water surface (FWS) evaporation shallow lake, 1956–1970, map 3 in pocket, Evaporation atlas for the contiguous 48 United States: National Oceanic and Atmospheric Administration Technical Report NWS 33, scale 1:4,800,000.
- Fenneman, N.M., and Johnson, D.W., 1946, Physical divisions of the United States: U.S. Geological Survey, scale 1:7,000,000.
- Fong, A.L., 2000, Water-quality assessment of part of the Upper Mississippi River Basin, Minnesota and Wisconsin—Ground-water quality in three different land-use areas, 1996–98: U.S. Geological Survey Water-Resources Investigations Report 00–4131, 37 p.
- Fong, A.L., Andrews, W.J., and Stark, J.R., 1998, Water-quality assessment of part of the Upper Mississippi River Basin, Minnesota and Wisconsin—Ground-water quality in the Prairie du Chien-Jordan Aquifer, 1996: U.S. Geological Survey Water-Resources Investigations Report, 98–4248, 45 p.
- Gilliom, R.J., Alley, W.M., and Gurtz, M.E., 1995, Design of the National Water-Quality Assessment Program—Occurrence and distribution of water-quality conditions: U.S. Geological Survey Circular 1112, 33 p.
- Goldstein, R.M., Lee, Kathy, Talmage, Philip, Stauffer, J.C., and Anderson, J.P., 1999, Relation of fish community composition to environmental factors and land use in part of the Upper Mississippi River Basin, 1995–97: U.S. Geological Survey Water-Resources Investigations Report 99–4034, 32 p.
- Granato, G.E., 1996, De-icing chemicals as source of constituents of highway runoff: Transportation Research Record 1533, p. 50–58.
- Hambrook, J.A., Koltun, G.F., Palcsak, B.B., and Tertuliani, J.S., 1997, Hydrologic disturbance and response of aquatic biota in Big Darby Creek Basin, Ohio: U.S. Geological Survey Water-Resources Investigations Report 96–4315, 82 p.
- Hanson, P.E., 1998, Pesticides and nitrates in surficial sand and gravel aquifers as related to modeled contamination susceptibility in part of the Upper Mississippi River Basin: U.S. Geological Survey Fact Sheet FS–107–98, 4 p.
- Hem, J.D., 1992, Study and interpretation of the chemical characteristics of natural water: U.S. Geological Survey Water-Supply Paper 2254, 263 p.
- Hitt, K.J., 1994, Refining 1970's land-use data with 1990 population data to indicate new residential development: U.S. Geological Survey Water-Resources Investigations Report 94–4250, 15 p.
- Holmberg, K., Perry, J., Ferrin, R., and Sharrow, D., 1997, Water resources management plan—St. Croix National Scenic Riverway: National Park Service, St. Croix Falls, Wisc., 155 p.
- Johnson, D.K., and Aasen, P.W., 1989, The Metropolitan Wastewater Treatment Plant and the Mississippi River—50 years of improving river quality: *Journal of the Minnesota Academy of Science*, v. 55, p. 134–138.

- Kanivetsky, Roman, 1978, Hydrogeologic map of Minnesota, bedrock hydrogeology: Minnesota Geological Survey, State Map Series S-2, 2 sheets.
- Klein, P.D., 1979, Urbanization and stream quality impairment: *Water Resources Bulletin* v. 15, p. 948-963.
- Kroening, S.E., 1998a, Nitrate concentrations, loads, and yields in streams in part of the Upper Mississippi River Basin, Minnesota and Wisconsin, 1996-98 [abs.]: *Minnesota Water '98—Protecting Minnesota's Water Supplies*, 6th Biennial Conference on Minnesota's Critical Water Issues, May 5-6, 1998, University of Minnesota. Water Resources Research Center. p. 67.
- _____, 1998b, Nutrient sources within the Upper Mississippi River Basin, Minnesota and Wisconsin, 1991-93: U.S. Geological Survey Fact Sheet FS-121-98, 4 p.
- _____, 1999, Bacteria monitoring in the St. Croix National Scenic Riverway [abs.]: 11th Annual St. Croix River Research Rendezvous, October 19, 1999, Marine on St. Croix, Minnesota, variously paged.
- _____, 2000, Nutrients, suspended sediment, and algae in streams in part of the Upper Mississippi River Basin, 1984-98 [abs.]: *Minnesota Water 2000—A Watershed Year—Looking Back, Planning Ahead*, 7th Biennial Conference, April 25-26, 2000, University of Minnesota, Water Resources Research Center, p. 46.
- Kroening, S.E., and Andrews, W.J., 1997, Water-quality assessment of part of the Upper Mississippi River Basin, Minnesota and Wisconsin—Nitrogen and phosphorus in streams, streambed sediment, and ground water, 1971-94: U.S. Geological Survey Water-Resources Investigations Report 97-4107, 61 p.
- Kroening, S.E., Fallon, J.D., and Lee, K.E., 2000, Water-quality assessment of part of the Upper Mississippi River Basin, Minnesota and Wisconsin—Trace elements in streambed sediment and fish livers, 1995-96: U.S. Geological Survey Water-Resources Investigations Report, 00-4031, 26 p.
- Lee, K.E., and Anderson, J.P., 1998, Water-quality assessment of the Upper Mississippi River Basin, Minnesota and Wisconsin—Polychlorinated biphenyls in common carp and walleye filets, 1975-95: U.S. Geological Survey Water-Resources Investigations Report 98-4126, 27 p.
- Lee, K.E., Talmage, P.J., ZumBerge, J.R., and Stauffer, J.C., 1999, The influence of urban and agricultural land use on biological communities in part of the Upper Mississippi River Basin, 1997 [abs.]: *Bulletin of the North American Benthological Society* 47th Annual Meeting, Duluth, Minnesota, v. 16, no. 1, p. 136.
- Lee, K.E., and ZumBerge, Jeremy, 2000, Phytoplankton community composition in streams in part of the Upper Mississippi River Basin, April-September 1996 [abs.]: *Minnesota Water 2000—A Watershed Year—Looking Back, Planning Ahead*, 7th Biennial Conference, April 25-26, 2000, University of Minnesota, Water Resources Research Center, p. 46.
- McNellis and others, 2000, Water-quality assessment of part of the Upper Mississippi River Basin, Minnesota and Wisconsin—Organochlorine compounds in streambed sediment and fish tissues, 1995-97: U.S. Geological Survey Water-Resources Investigations Report 00-4213, p.
- Metropolitan Waste Control Commission, 1994, Water-quality analysis of the Lower Minnesota River and selected tributaries—River (1976-1991) and nonpoint source (1989-1992) monitoring, volume I: Water-Quality Division, Quality-Control Department Report No. QC-93-267, St. Paul, Minnesota, 142 p.
- Minnesota Department of Natural Resources, Minnesota Pollution Control Agency, National Park Service, and Wisconsin Department of Natural Resources, 1995, St. Croix River Basin Water Resources Management Plan, Phase II, 28 p.
- Minnesota Pollution Control Agency, 1985, Lower Minnesota River waste load allocation study: St. Paul, Minnesota: Minnesota Pollution Control Agency, 190 p.
- _____, 1986, Ground water in Minnesota—A user's guide to understanding Minnesota's ground water resource, St. Paul, Minnesota: Minnesota Pollution Control Agency, 47 p.
- _____, 1989, Review of water-quality conditions in Lake Pepin for the summer of 1988: St. Paul, Minnesota: Minnesota Pollution Control Agency, 50 p.
- _____, 1991 *Minnesota State Rules*: St. Paul, Minnesota: Minnesota Pollution Control Agency, [variously paged].
- _____, 1999, *Minnesota State Statutes*: St. Paul, Minnesota: Minnesota Pollution Control Agency, [variously paged].
- Mitton, G.B., and Payne, G.A., 1997, Quantity and quality of runoff from selected guttered and unguttered roadways in northeastern Ramsey County, Minnesota: U.S. Geological Survey Water-Resources Investigations Report 96-4284, 72 p.
- Mudrey, M.G., Jr., LaBarge, G.A., Myers, P.E., and Cordua, W.S., 1987, Bedrock geology of Wisconsin, Wisconsin Geological and Natural History Survey Map 87-11a, 2 sheets.
- Mueller, L., 1993, Winged maple leaf mussel and Higgins-eye pearly mussel—Freshwater mussels threatened with extinction: Minnesota Department of Agriculture, 19 p.
- Osborne, L.L., and Kovacic, D.A., 1993, Riparian vegetated buffer strips in water-quality restoration and stream management: *Freshwater Biology*, v. 29, p. 243-258.
- Olcott, P.G., 1992, Ground-water atlas of the United States, Segment 9, Iowa, Michigan, Minnesota, and Wisconsin: U.S. Geological Survey Hydrologic Investigations Atlas 730-J, 31 p., scale 1:250,000 and 1:500,000.

- Payne, G.A., 1994, Sources and transport of sediment, nutrients, and oxygen-demanding substances in the Minnesota River Basin, 1989–92: U.S. Geological Survey Water-Resources Investigations Report 93–4232, 71 p.
- Pope, L.M., and Putnam, J.E., 1997, Effects of urbanization on water quality in the Kansas River, Shunganunga Creek Basin, and Soldier Creek, Topeka, Kansas, October 1993 through September 1995: U.S. Geological Survey Water-Resources Investigations Report 97–4045, 84 p.
- Porter, 2000, Upper Midwest River Systems—Algal and nutrient conditions in streams and rivers in the Upper Midwest Region during seasonal low-flow conditions, *in* Nutrient Criteria Technical Guidance Manual, Rivers and Streams: Washington, D.C., U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology, EPA-822-B-00-002, p. A-25–A-42.
- Riley, A.L., 1998, Restoring streams in cities—A guide for planners, policy makers, and citizens: Washington, D.C., Island Press, 423 p.
- Ruhl, J.F., 1987, Hydrogeologic and water-quality characteristics of glacial-drift aquifers in Minnesota: U.S. Geological Survey Water-Resources Investigations Report 87–4224, 2 pls.
- Ruhl, J.F., Fong, A.L., Hanson, P.E., and Andrews, W.J., 2000, Water-quality assessment of part of the Upper Mississippi River Basin, Minnesota and Wisconsin—Ground-water quality in an agricultural area of Sherburne County, Minnesota, 1998: U.S. Geological Survey Water-Resources Investigations Report 00–4107, 33 p.
- Sorenson, S.K., Porter, S.D., Akers, K.B., Harris, M.A., Kalkhoff, S.J., Lee, K.E., Roberts, L.R., Terrio, P.J., 1999, Water quality and habitat conditions in Upper Midwest streams in relation to riparian vegetation and soil characteristics, August 1997—Study design, methods, and data: U.S. Geological Survey Open-File Report 99–202, 53 p.
- Stark, J.R., Fallon, J.D., Fong, A.L., Goldstein, R.M., Hanson, P.E., Kroening, S.E., and Lee, K.E., 1999, Water-quality assessment of part of the Upper Mississippi River Basin, Minnesota and Wisconsin—Design and implementation, 1995–98: U.S. Geological Survey Water-Resources Investigations Report 99–4135, 85 p.
- Stauffer, J.C., Goldstein, R.M., and Newman, R.M., 2000, Relation of wooded riparian zones and runoff potential to fish community composition in the agricultural streams: *Canadian Journal of Fisheries and Aquatic Sciences*, v. 57, p. 307–316
- Talmage, P.J., Lee, K.E., Goldstein, R.M., Anderson, J.P., and Fallon, J.D., 1999, Water quality, physical habitat, and fish community composition in streams in the Twin Cities metropolitan area, Minnesota, 1997–1998: U.S. Geological Survey Water-Resources Investigations Report 99–4247, 18 p.
- Underhill, J.C., 1989, The distribution of Minnesota fishes and late Pleistocene glaciation: *Journal of the Minnesota Academy of Sciences*, v. 55, p. 32–37.
- U.S. Environmental Protection Agency, 1986, Quality criteria for water 1986: U.S. Environmental Protection Agency, EPA 440–5–86–001.
- _____, 1999, National recommended water-quality criteria—Correction: USEPA 822–Z–99–col, 25 p.
- U.S. Geological Survey, 1999, The quality of our Nation's water—Nutrients and pesticides: U.S. Geological Survey Circular 1225, 82 p.
- Vershueren, K., 1983, Handbook of environmental data on organic chemicals: New York, Van Nostrand Reinhold, 1310 p.
- Wiener, J.G., Jackson, G.A., May, T.W., and Cole, B.P., 1984, Longitudinal distribution of trace elements (As, Cd, Cr, Hg, Pb, and Se) in fishes and sediments in the Upper Mississippi River, *in* Wiener, J.G., Anderson, R.V., and McConville, D.R., eds., Contaminants in the Upper Mississippi River, Proceedings of the 15th Annual Meeting of the Mississippi River Research Consortium, La Crosse, Wisconsin, April 14–15, 1982: Stoneham, Massachusetts, Butterworth Publishers, p. 139–170.
- Wisconsin Department of Natural Resources, 1997, Wisconsin state statutes, chapter NR–102, water-quality standards for Wisconsin surface waters: Accessed August 16, 1999, at URL <http://www.legis.state.wi.us/rsb/code/nr/nr100.html>
- Wisconsin Register, 1998, No. 506, chap. NR 102, subpart 4.
- ZumBerge, J.A., 1999, Influence of local riparian cover and basin runoff potential on benthic invertebrate communities in the Minnesota River Basin: Master's Thesis, University of Minnesota, October 1999, variously paged.

APPENDIX—WATER-QUALITY DATA FROM THE UPPER MISSISSIPPI RIVER BASIN IN A NATIONAL CONTEXT

For a complete view of Upper Mississippi River Basin data and for additional information about specific benchmarks used, visit our Web site at <http://water.usgs.gov/nawqa/>. Also visit the NAWQA Data Warehouse for access to NAWQA data sets at <http://infotrek.er.usgs.gov/wdbctx/nawqa/nawqa.home>.

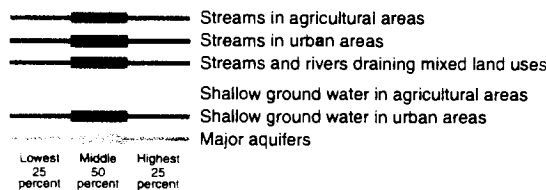
This appendix is a summary of chemical concentrations and biological indicators assessed in the Upper Mississippi River Basin. Selected results for this basin are graphically compared to results from as many as 36 NAWQA Study Units investigated from 1991 to 1998 and to national water-quality benchmarks for human health, aquatic life, or fish-eating wildlife. The chemical and biological indicators shown were selected on the basis of frequent detection, detection at concentrations above a national benchmark, or regulatory or scientific importance. The graphs illustrate how conditions associated with each land use sampled in the Upper Mississippi River Basin compare to results from across the Nation, and how conditions compare among the several land uses. Graphs for chemicals show only detected concentrations and, thus, care must be taken to evaluate detection frequencies in addition to concentrations when comparing study-unit and national results. For example, acetochlor concentrations in the Upper Mississippi River Basin agricultural streams were similar to the national distribution, but the detection frequency was much higher (90 percent compared to 33 percent).

CHEMICALS IN WATER

Concentrations and detection frequencies, Upper Mississippi River Basin, 1995–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals

- ◆ Detected concentration in Study Unit
- 66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- Not measured or sample size less than two
- 12 Study-unit sample size. For ground water, the number of samples is equal to the number of wells sampled

National ranges of detected concentrations, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

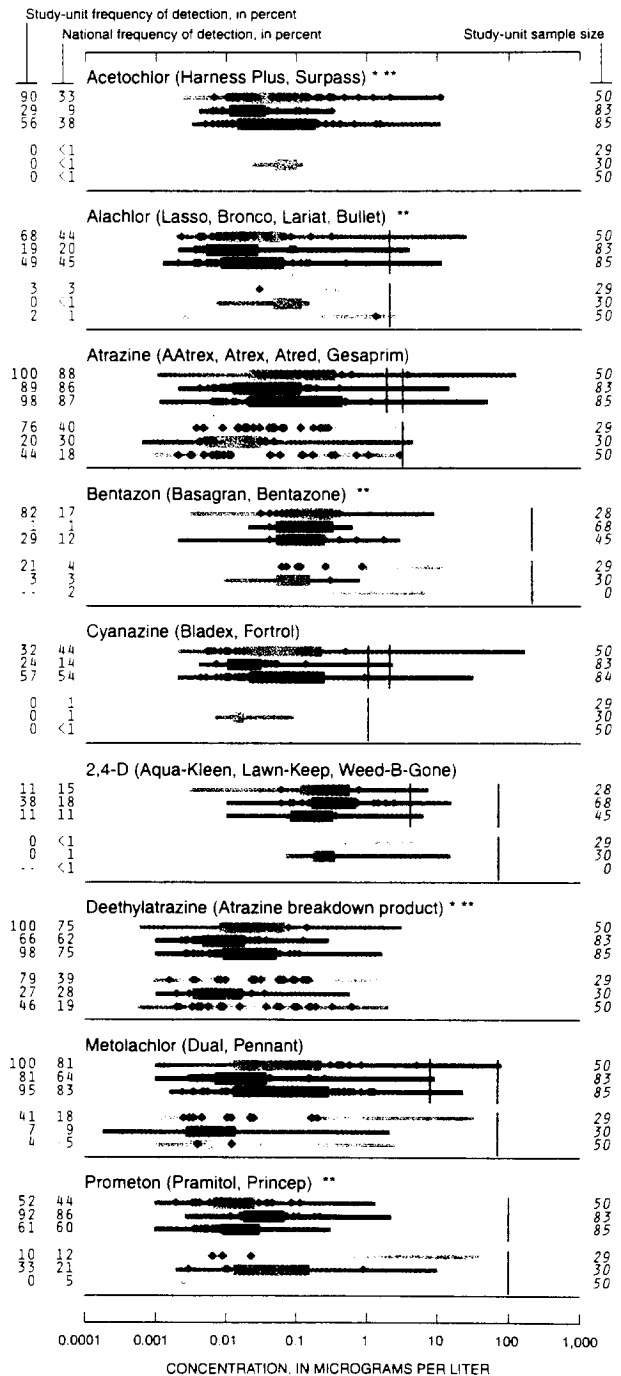


National water-quality benchmarks

National benchmarks include standards and guidelines related to drinking-water quality, criteria for protecting the health of aquatic life, and a goal for preventing stream eutrophication due to phosphorus. Sources include the U.S. Environmental Protection Agency and the Canadian Council of Ministers of the Environment

- | Drinking-water quality (applies to ground water and surface water)
- | Protection of aquatic life (applies to surface water only)
- | Prevention of eutrophication in streams not flowing directly into lakes or impoundments
- No benchmark for drinking-water quality
- ** No benchmark for protection of aquatic life

Pesticides in water—Herbicides



Other herbicides detected

- Acifluorfen (Blazer, Tackle 2S) **
- Benfluralin (Balan, Benefin, Bonalan) ***
- Bromacil (Hyvar X, Urox B, Bromax)
- Bromoxynil (Buctril, Brominal) *
- DCPA (Dacthal, chlorthal-dimethyl) ***

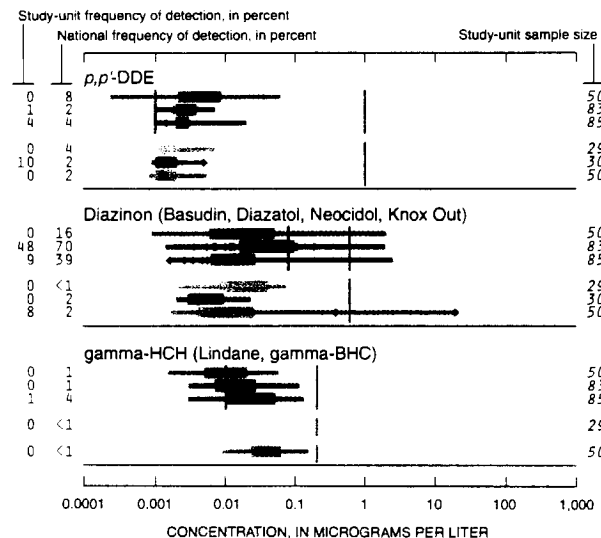
R0024571

Dicamba (Banvel, Dianat, Scotts Proturf)
 2,6-Diethylaniline (Alachlor breakdown product) * **
 Dinoseb (Dinosebe)
 Diuron (Crisuron, Karmex, Diurex) **
 EPTC (Eptam, Farmarox, Alirox) * **
 Metribuzin (Lexone, Sencor)
 Napropamide (Devrinol) * **
 Oryzalin (Surflan, Dirimal) * **
 Pendimethalin (Pre-M, Prowl, Stomp) * **
 Propachlor (Ramrod, Satecid) **
 Simazine (Princep, Caliber 90)
 Tebuthiuron (Spike, Tebusan)
 Thiobencarb (Bolero, Saturn, Benthicarb) * **
 Trifluralin (Treflan, Gowan, Tri-4, Trific)

Herbicides not detected

Butylate (Sutan +, Genate Plus, Butilate) **
 Chloramben (Amiben, Amilon-WP, Vegiben) **
 Clopyralid (Stinger, Lontrel, Transline) * **
 2,4-DB (Butyrac, Butoxone, Embutox Plus, Embutone) * **
 Dacthal mono-acid (Dacthal breakdown product) * **
 Dichlorprop (2,4-DP, Seritox 50, Lentemul) * **
 Ethalfuralin (Sonalan, Curbit) * **
 Fenuron (Fenulon, Fenidim) * **
 Fluometuron (Flo-Met, Cotoran) **
 Linuron (Lorox, Linex, Sarclax, Linurex, Afalon) *
 MCPA (Rhomene, Rhonox, Chiptox)
 MCPB (Thistrol) * **
 Molinate (Ordram) * **
 Neburon (Neburea, Neburyl, Noruben) * **
 Norflurazon (Evtal, Predict, Solicam, Zorial) * **
 Pebulate (Tillam, PEBC) * **
 Picloram (Grazon, Tordon)
 Pronamide (Kerb, Propyzamid) **
 Propanil (Stam, Stampede, Wham) * **
 Propham (Tuberite) **
 2,4,5-T **
 2,4,5-TP (Silvex, Fenoprop) **
 Terbacil (Sinbar) **
 Triallate (Far-Go, Avadex BW, Tri-allate) *
 Triclopyr (Garlon, Grandstand, Redeem, Remedy) * **

Pesticides in water—Insecticides



Other insecticides detected

Carbaryl (Carbamine, Denapon, Sevin)
 Carbofuran (Furadan, Curaterr, Yaltox)
 Chlorpyrifos (Brodan, Dursban, Lorsban)
 Dieldrin (Panoram D-31, Octalox, Compound 497)
 Ethoprop (Mocap, Ethoprophos) * **
 Malathion (Malathion)

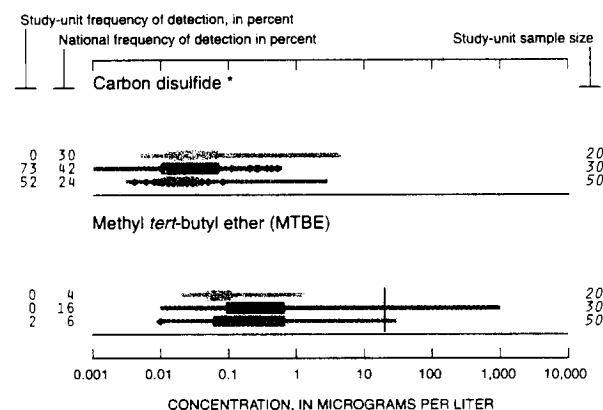
Methomyl (Lanox, Lannate, Acinate) **
 Oxamyl (Vydate L, Pratt) **
 Propargite (Comite, Omite, Ornamate) * **

Insecticides not detected

Aldicarb (Temik, Ambush, Pounce)
 Aldicarb sulfone (Standak, aldoxycarb)
 Aldicarb sulfoxide (Aldicarb breakdown product)
 Azinphos-methyl (Guthion, Gusathion M) *
 Disulfoton (Disyston, Di-Syston) **
 Fonofos (Dytonate, Capfos, Cudgel, Tycap) **
 alpha-HCH (alpha-BHC, alpha-lindane) **
 3-Hydroxycarbofuran (Carbofuran breakdown product) * **
 Methiocarb (Slug-Geta, Grandslam, Mesuro!) * **
 Methyl parathion (Pennacp-M, Folidol-M) **
 Parathion (Roethyl-P, Alkron, Panthion, Phoskil) *
 cis-Permethrin (Ambush, Astro, Pounce) * **
 Phorate (Thimet, Granutox, Geomet, Rampart) * **
 Propoxur (Baygon, Blattanex, Unden, Proprotox) * **
 Terbufos (Contrafen, Counter, Pilarfox) **

Volatile organic compounds (VOCs) in ground water

These graphs represent data from 16 Study Units, sampled from 1996 to 1998



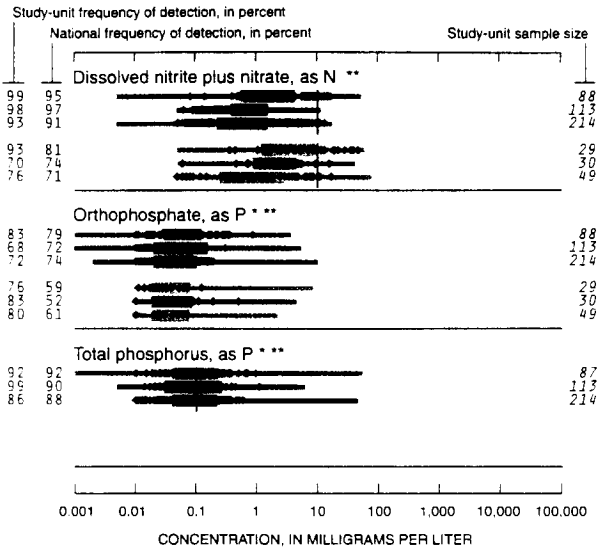
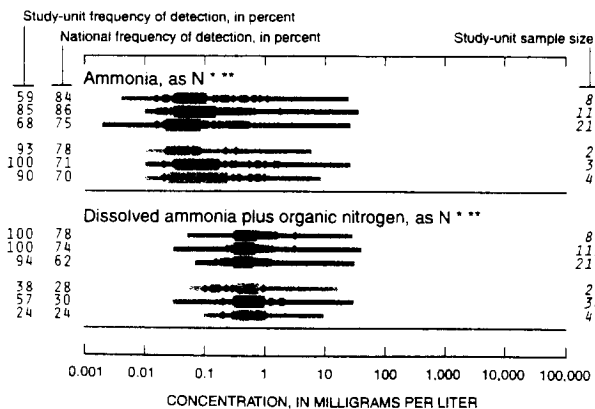
Other VOCs detected

Benzene
 Bromodichloromethane (Dichlorobromomethane)
 2-Butanone (Methyl ethyl ketone (MEK)) *
 Chlorobenzene (Monochlorobenzene)
 Chlorodibromomethane (Dibromochloromethane)
 Chloroethane (Ethyl chloride) *
 Chloromethane (Methyl chloride)
 Dichlorodifluoromethane (CFC 12, Freon 12)
 1,1-Dichloroethane (Ethylidene dichloride) *
 cis-1,2-Dichloroethene (Z)-1,2-Dichloroethene
 Dichloromethane (Methylene chloride)
 Diethyl ether (Ethyl ether) *
 1-4-Epoxy butane (Tetrahydrofuran, Diethylene oxide) *
 Ethenylbenzene (Styrene)
 Iodomethane (Methyl iodide) *
 p-Isopropyltoluene (p-Cymene) *
 4-Methyl-2-pentanone (Methyl isobutyl ketone (MIBK)) *
 Methylbenzene (Toluene)
 2-Propanone (Acetone) *
 Tetrachloroethene (Perchloroethene)
 1,2,3,4-Tetramethylbenzene (Prehnitene) *
 Tribromomethane (Bromoform)
 1,1,1-Trichloroethane (Methylchloroform)
 Trichloroethene (TCE)
 Trichlorofluoromethane (CFC 11, Freon 11)
 Trichloromethane (Chloroform)
 1,2,4-Trimethylbenzene (Pseudocumene) *

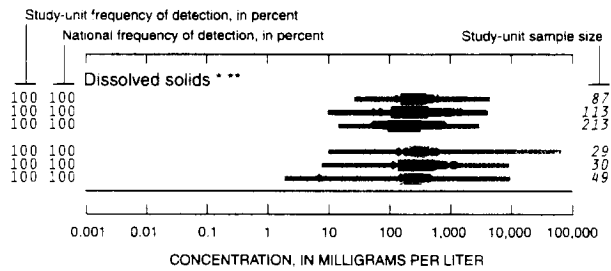
VOCs not detected

- tert*-Amylmethylether (*tert*-amyl methyl ether) *AME *
- Bromobenzene (Phenyl bromide) *
- Bromochloromethane (Methylene chlorobromide)
- Bromoethene (Vinyl bromide) *
- Bromomethane (Methyl bromide)
- n*-Butylbenzene (1-Phenylbutane) *
- sec*-Butylbenzene *
- tert*-Butylbenzene *
- 3-Chloro-1-propene (3-Chloropropene) *
- 1-Chloro-2-methylbenzene (*o*-Chlorotoluene)
- 1-Chloro-4-methylbenzene (*p*-Chlorotoluene)
- Chloroethene (Vinyl chloride)
- 1,2-Dibromo-3-chloropropane (DBCP, Nemagon)
- 1,2-Dibromoethane (Ethylene dibromide, EDB)
- Dibromomethane (Methylene dibromide) *
- trans*-1,4-Dichloro-2-butene ((*Z*)-1,4-Dichloro-2-butene) *
- 1,2-Dichlorobenzene (*o*-Dichlorobenzene)
- 1,3-Dichlorobenzene (*m*-Dichlorobenzene)
- 1,4-Dichlorobenzene (*p*-Dichlorobenzene)
- 1,2-Dichloroethane (Ethylene dichloride)
- 1,1-Dichloroethene (Vinylidene chloride)
- trans*-1,2-Dichloroethene ((*E*)-1,2-Dichloroethene)
- 1,2-Dichloropropane (Propylene dichloride)
- 2,2-Dichloropropane *
- 1,3-Dichloropropane (Trimethylene dichloride) *
- trans*-1,3-Dichloropropene ((*E*)-1,3-Dichloropropene)
- cis*-1,3-Dichloropropene ((*Z*)-1,3-Dichloropropene)
- 1,1-Dichloropropene *
- Diisopropyl ether (Diisopropylether (DIPE)) *
- 1,2-Dimethylbenzene (*o*-Xylene)
- 1,3 & 1,4-Dimethylbenzene (*m*-&*p*-Xylene)
- Ethyl methacrylate *
- Ethyl *tert*-butyl ether (Ethyl-*t*-butyl ether (ETBE)) *
- 1-Ethyl-2-methylbenzene (2-Ethyltoluene) *
- Ethylbenzene (Phenylethane)
- Hexachlorobutadiene
- 1,1,1,2,2,2-Hexachloroethane (Hexachloroethane)
- 2-Hexanone (Methyl butyl ketone (MBK)) *
- Isopropylbenzene (Cumene) *
- Methyl acrylonitrile *
- Methyl-2-methacrylate (Methyl methacrylate) *
- Methyl-2-propenoate (Methyl acrylate) *
- Naphthalene
- 2-Propenenitrile (Acrylonitrile)
- n*-Propylbenzene (Isocumene) *
- 1,1,2,2-Tetrachloroethane *
- 1,1,1,2-Tetrachloroethane
- Tetrachloromethane (Carbon tetrachloride)
- 1,2,3,5-Tetramethylbenzene (Isodurene) *
- 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon 113) *
- 1,2,4-Trichlorobenzene
- 1,2,3-Trichlorobenzene *
- 1,1,2-Trichloroethane (Vinyl trichloride)
- 1,2,3-Trichloropropane (Allyl trichloride)
- 1,2,3-Trimethylbenzene (Hemimellitene) *
- 1,3,5-Trimethylbenzene (Mesitylene) *

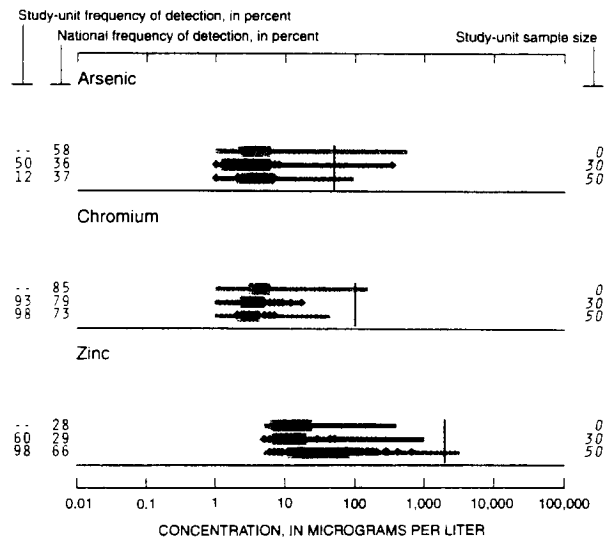
Nutrients in water

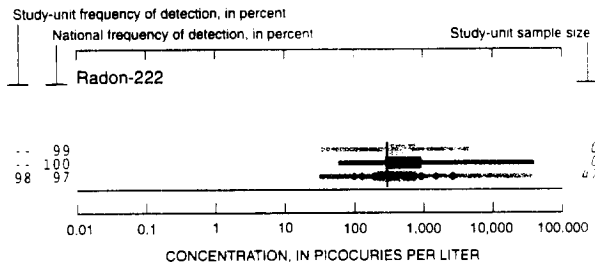


Dissolved solids in water



Trace elements in ground water





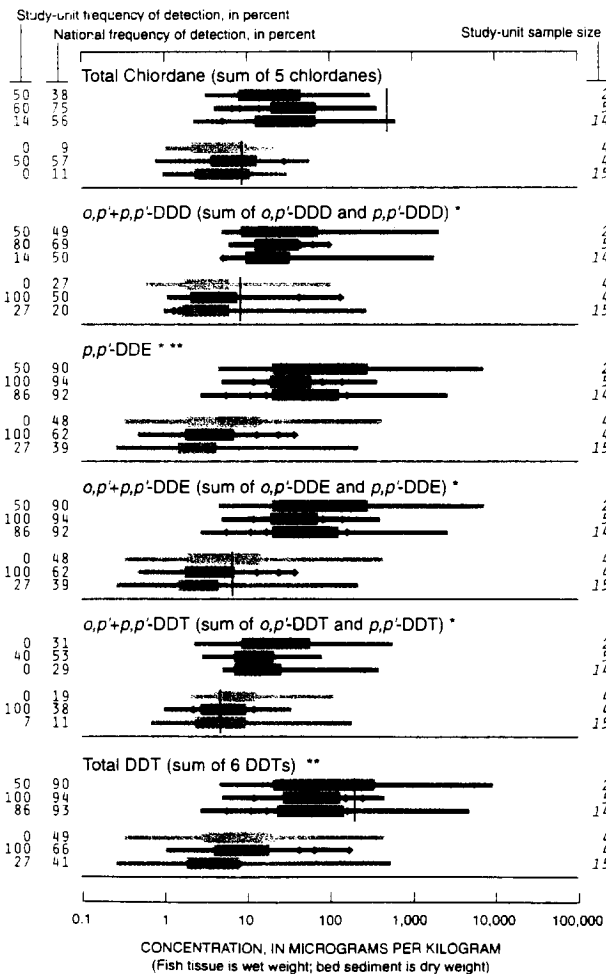
Other trace elements detected

Lead
Selenium
Uranium

Trace elements not detected

Cadmium

Organochlorines in fish tissue (whole body) and bed sediment

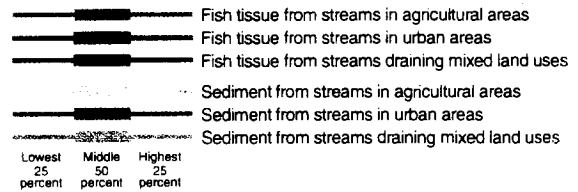


CHEMICALS IN FISH TISSUE AND BED SEDIMENT

Concentrations and detection frequencies, Upper Mississippi River Basin, 1995-98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals. Study-unit frequencies of detection are based on small sample sizes; the applicable sample size is specified in each graph

- ◆ Detected concentration in Study Unit
- 65 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- Not measured or sample size less than two
- 12 Study-unit sample size

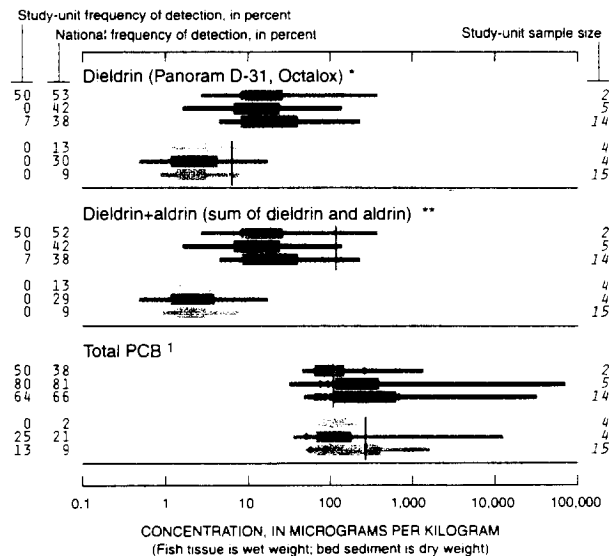
National ranges of concentrations detected, by land use, in 36 NAWQA Study Units, 1991-98—Ranges include only samples in which a chemical was detected



National benchmarks for fish tissue and bed sediment

National benchmarks include standards and guidelines related to criteria for protection of the health of fish-eating wildlife and aquatic organisms. Sources include the U.S. Environmental Protection Agency, other Federal and State agencies, and the Canadian Council of Ministers of the Environment

- | Protection of fish-eating wildlife (applies to fish tissue)
- | Protection of aquatic life (applies to bed sediment)
- No benchmark for protection of fish-eating wildlife
- No benchmark for protection of aquatic life



¹ The national detection frequencies for total PCB in sediment are biased low because about 30 percent of samples nationally had elevated detection levels compared to this Study Unit. See <http://water.usgs.gov/nawqa/> for additional information.

Other organochlorines detected

Endosulfan I (alpha-Endosulfan, Thiodan) * **

Organochlorines not detected

Chloroneb (Chloronebe, Demosan) * **

DCPA (Dacthal, chlorthal-dimethyl) * **

Endrin (Endrine)

gamma-HCH (Lindane, gamma-BHC, Gammexane) *

Total-HCH (sum of alpha-HCH, beta-HCH, gamma-HCH, and delta-HCH) **

Heptachlor epoxide (Heptachlor breakdown product) *

Heptachlor+heptachlor epoxide (sum of heptachlor and heptachlor epoxide) **

Hexachlorobenzene (HCB) **

Isodrin (Isodrine, Compound 711) * **

p,p'-Methoxychlor (Marlate, methoxychlore) * **;

o,p'-Methoxychlor * **

Mirex (Dechlorane) **

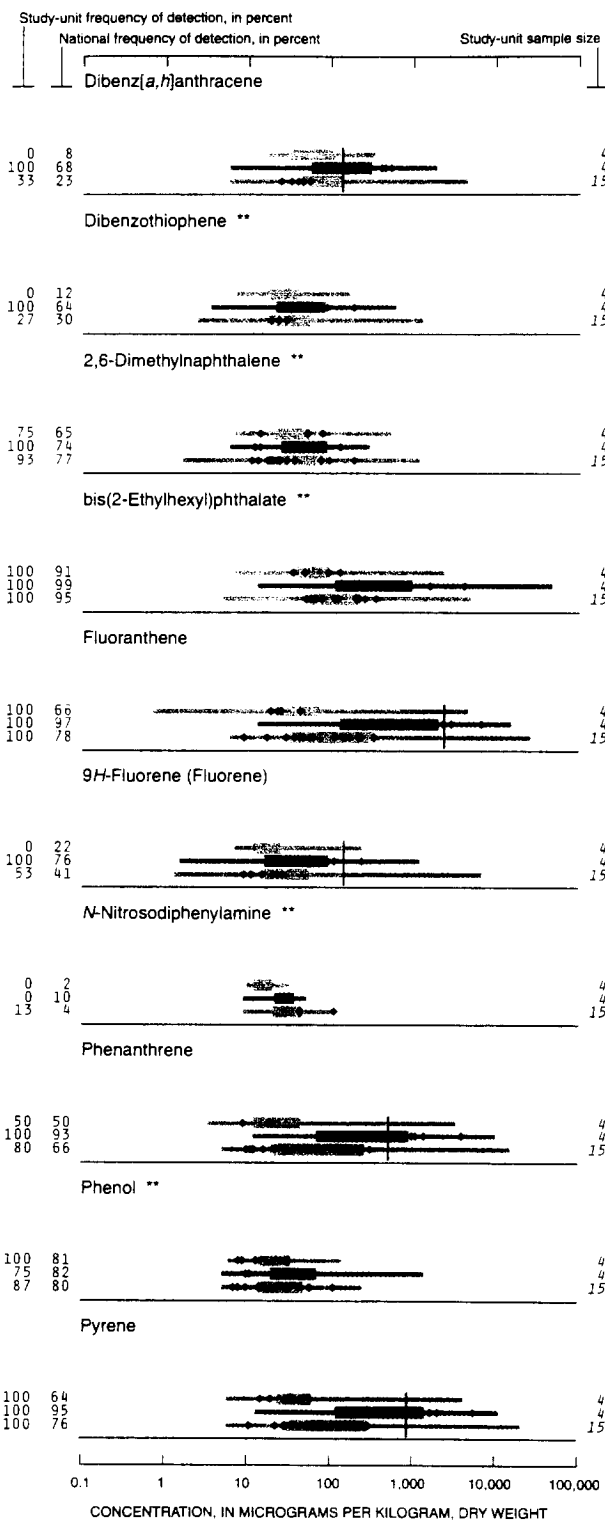
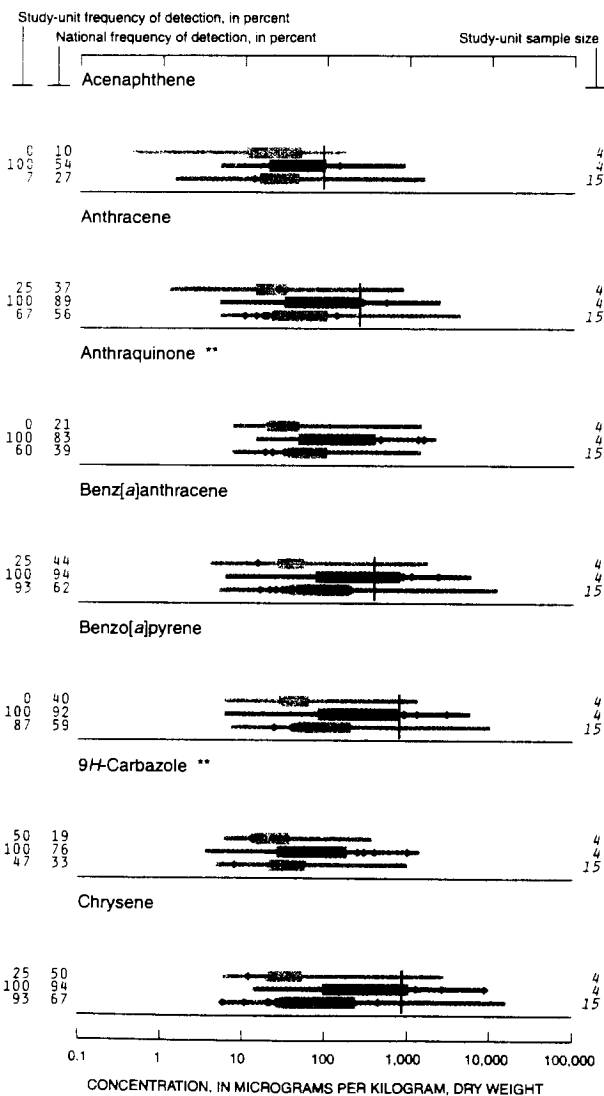
Pentachloroanisole (PCA) * **

cis-Permethrin (Ambush, Astro, Pounce) * **

trans-Permethrin (Ambush, Astro, Pounce) * **

Toxaphene (Camphechlor, Hercules 3956) * **

**Semivolatile organic compounds (SVOCs)
in bed sediment**



Other SVOCs detected

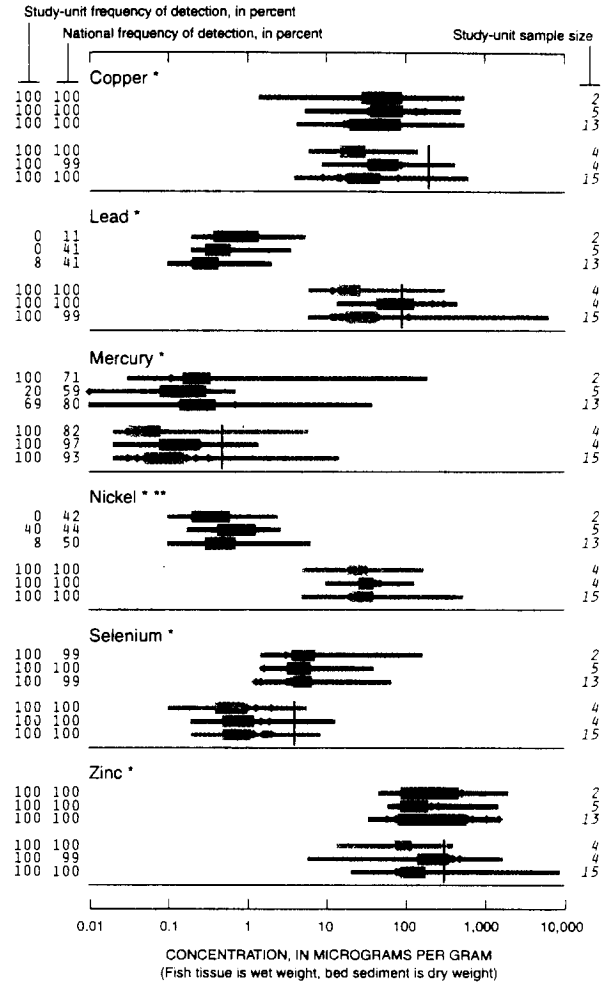
Acenaphthylene

Acridine **

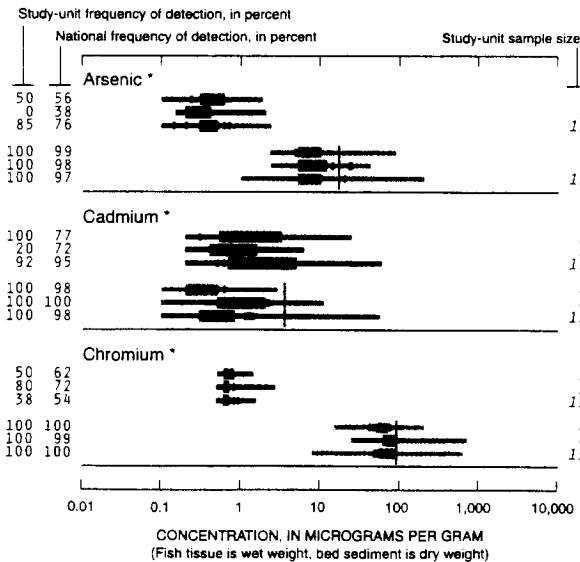
Azobenzene **

Benzo[b]fluoranthene **

- Benzo[ghi]perylene **
- Benzo[k]fluoranthene **
- 2,2-Biquinoline **
- Butylbenzylphthalate **
- 4-Chloro-3-methylphenol **
- p*-Cresol **
- Di-*n*-butylphthalate **
- Di-*n*-octylphthalate **
- Diethylphthalate **
- 1,6-Dimethylnaphthalene **
- Dimethylphthalate **
- 2-Ethyl-naphthalene **
- Indeno[1,2,3-*cd*]pyrene **
- Isoquinoline **
- 1-Methyl-9*H*-fluorene **
- 2-Methylantracene **
- 4,5-Methylenephenanthrene **
- 1-Methylphenanthrene **
- 1-Methylpyrene **
- Naphthalene
- Phenanthridine **
- SVOCs not detected**
- C8-Alkylphenol **
- Benzo[c]cinnoline **
- 4-Bromophenyl-phenylether **
- bis(2-Chloroethoxy)methane **
- 2-Chloronaphthalene **
- 2-Chlorophenol **
- 4-Chlorophenyl-phenylether **
- 1,2-Dichlorobenzene (*o*-Dichlorobenzene) **
- 1,3-Dichlorobenzene (*m*-Dichlorobenzene) **
- 1,4-Dichlorobenzene (*p*-Dichlorobenzene) **
- 1,2-Dimethylnaphthalene **
- 3,5-Dimethylphenol **
- 2,4-Dinitrotoluene **
- Isophorone **
- Nitrobenzene **
- N*-Nitrosodi-*n*-propylamine **
- Pentachloronitrobenzene **
- Quinoline **
- 1,2,4-Trichlorobenzene **
- 2,3,6-Trimethylnaphthalene **



Trace elements in fish tissue (livers) and bed sediment



BIOLOGICAL INDICATORS

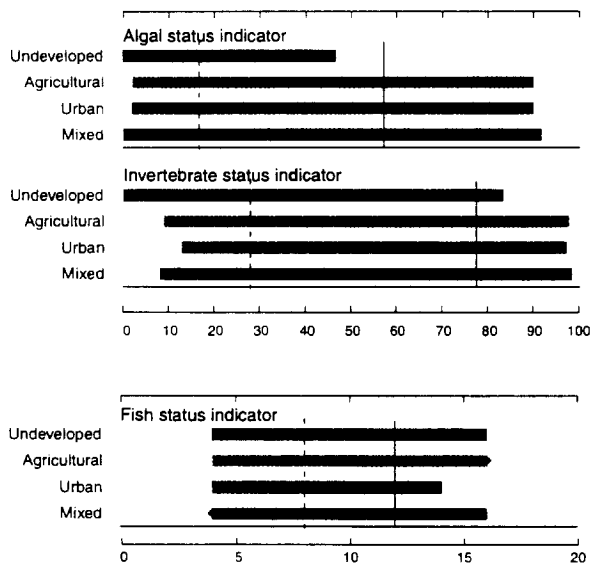
Higher national scores suggest habitat disturbance, water-quality degradation, or naturally harsh conditions. The status of algae, invertebrates (insects, worms, and clams), and fish provide a record of water-quality and stream conditions that water-chemistry indicators may not reveal. **Algal status** focuses on the changes in the percentage of certain algae in response to increasing siltation, and it often correlates with higher nutrient concentrations in some regions. **Invertebrate status** averages 11 metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. **Fish status** sums the scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that increase in association with water-quality degradation.

Biological indicator value, Upper Mississippi River Basin, by land use, 1995–98

- ◆ Biological status assessed at a site

National ranges of biological indicators, in 16 NAWQA Study Units, 1994–98

- ▬ Streams in undeveloped areas
- ▬ Streams in agricultural areas
- ▬ Streams in urban areas
- ▬ Streams in mixed-land-use areas
- 75th percentile
- - - 25th percentile



R0024577

A COORDINATED EFFORT

An integral part of the NAWQA Program is cooperation among agencies and organizations. We wish to thank the following agencies and organizations who contributed to this report or participated in the Study Unit liaison committee.

*American Water Works Association
Anoka County, Minnesota
Bell Museum of Natural History
Cedar Creek Natural History Area
Dakota County Planning Department
Elm Creek Watershed District
Friends of the Mississippi River
Hennepin Conservation District
Izaak Walton League
Legislative Commission on Minnesota Resources
McKnight Foundation
Metropolitan Council
Minneapolis Water Works
Minnesota Board of Water and Soil Resources
Minnesota Department of Agriculture
Minnesota Department of Health
Minnesota Department of Natural Resources
Minnesota Extension Service
Minnesota Geological Survey
Minnesota Pollution Control Agency
Minnesota State Planning Agency
Minnesota-Wisconsin Boundary Area Commission*

*Mississippi River Headwaters Board
Montgomery Watson
National Park Service
National Weather Service
Northern States Power Company
Rivers Council of Minnesota
St. Cloud State University
St. Paul Water Utility
Science Museum of Minnesota
Shingle Creek Watershed District
Sierra Club
University of Minnesota
University of Minnesota Water Resources Center
Upper Mississippi River Basin Association
U.S. Army Corps of Engineers
U.S. Department of Agriculture
U.S. Environmental Protection Agency
U.S. Fish and Wildlife Service
Wisconsin Department of Natural Resources*

Special thanks to U.S. Geological Survey employees for their contributions: Wallace Larson, Todd Schmitt, and Joshua Larson for data collection, and Ginger Amos, and Robert Borgstede for production of the report.

NAWQA

National Water-Quality Assessment (NAWQA) Program Upper Mississippi River Basin

Stark and others—Water Quality in the Upper Mississippi River Basin
U.S. Geological Survey Circular 1211



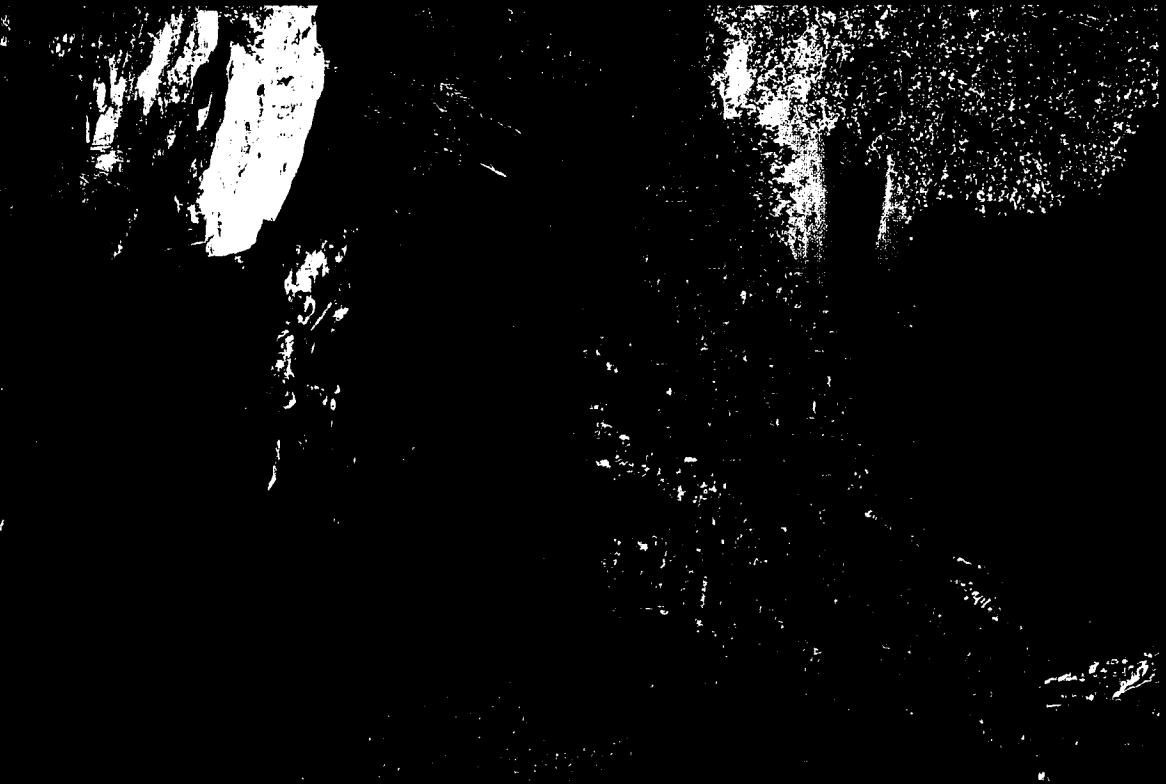
ISBN 0-607-95416-7



9 780607 954166

R0024579

Water Quality in the Central Arizona Basins Arizona, 1995–98



U.S. Department of the Interior
U.S. Geological Survey

Circular 1213

R0024580

POINTS OF CONTACT AND ADDITIONAL INFORMATION

The companion Web site for NAWQA summary reports:

http://water.usgs.gov/nawqa_summaryreports/

Central Arizona Basins contact and web site:

USGS State Representative
U.S. Geological Survey
Water Resources Division
520 N. Park Ave., Suite 221
Tucson, AZ 85719
e-mail: nmelcher@usgs.gov
<http://az.water.usgs.gov/cazb/>

National NAWQA Program:

Chief, NAWQA Program
U.S. Geological Survey
Water Resources Division
12201 Sunrise Valley Drive, M.S. 413
Reston, VA 20192
<http://water.usgs.gov/nawqa/>

Other NAWQA summary reports

River Basin Assessments

Albemarle-Pamlico Drainage Basin (Circular 1157)
Allegheny and Monongahela River Basins (Circular 1202)
Apalachicola-Chattahoochee-Flint River Basin (Circular 1164)
Central Columbia Plateau (Circular 1144)
Central Nebraska Basins (Circular 1163)
Connecticut, Housatonic and Thames River Basins (Circular 1155)
Eastern Iowa Basins (Circular 1210)
Georgia-Florida Coastal Plain (Circular 1151)
Hudson River Basin (Circular 1165)
Kanawha - New River Basins (Circular 1204)
Lake Erie - Lake Saint Clair Drainages (Circular 1203)
Las Vegas Valley Area and the Carson and Truckee River Basins
(Circular 1170)
Lower Illinois River Basin (Circular 1209)
Long Island - New Jersey Coastal Drainages (Circular 1201)
Lower Susquehanna River Basin (Circular 1168)
Mississippi Embayment (Circular 1208)
Ozark Plateaus (Circular 1158)
Potomac River Basin (Circular 1166)
Puget Sound Basin (Circular 1216)
Red River of the North Basin (Circular 1169)

Rio Grande Valley (Circular 1162)
Sacramento River Basin (Circular 1215)
San Joaquin-Tulare Basins (Circular 1159)
Santee River Basin and Coastal Drainages (Circular 1206)
South-Central Texas (Circular 1212)
South Platte River Basin (Circular 1167)
Southern Florida (Circular 1207)
Trinity River Basin (Circular 1171)
Upper Colorado River Basin (Circular 1214)
Upper Mississippi River Basin (Circular 1211)
Upper Snake River Basin (Circular 1160)
Upper Tennessee River Basin (Circular 1205)
Western Lake Michigan Drainages (Circular 1156)
White River Basin (Circular 1150)
Willamette Basin (Circular 1161)

National Assessments

The Quality of Our Nation's Waters—Nutrients and Pesticides (Circular 1225)

Front cover: Sabino Creek in Sabino Canyon near Tucson. The water is colored brown by natural tannin from plant material in the stream. (Photograph by Gail E. Cordy.)

Back cover: Left, view of Tucson from "A" Mountain; right, view of west side of the Whetstone Mountains, southeast of Tucson. (Photographs by Alissa L. Coes.)

Water Quality in the Central Arizona Basins, Arizona, 1995–98

By Gail E. Cordy, D.J. Gellenbeck, Joseph B. Gebler, David W. Anning,
Alissa L. Coes, R.J. Edmonds, Julie A. H. Rees, *and* H.W. Sanger

U.S. DEPARTMENT OF THE INTERIOR
GALE A. NORTON, SECRETARY

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

2000

Free on application to the
U.S. Geological Survey
Information Services
Box 25286 Federal Center
Denver, CO 80225

Or call: 1-888-ASK-USGS

R0024583

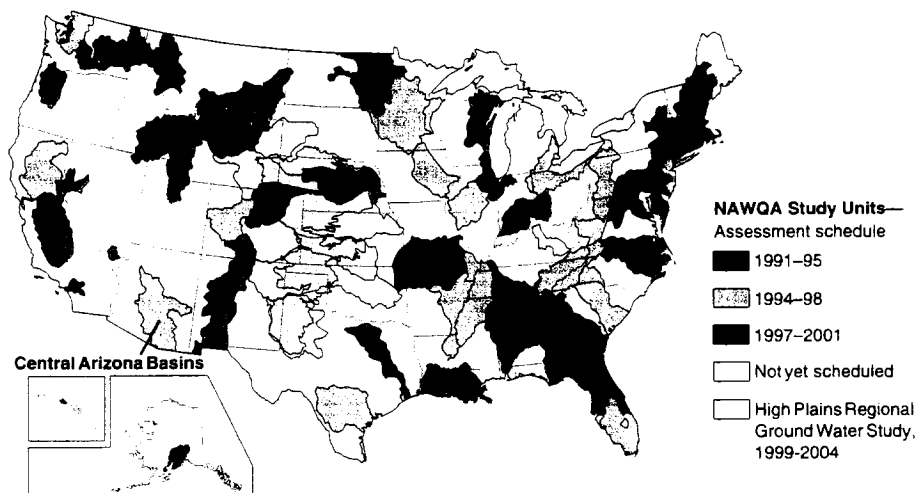
CONTENTS

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM.....	IV
SUMMARY OF MAJOR FINDINGS.....	1
Stream and river highlights	1
Ground-water highlights.....	2
INTRODUCTION TO THE CENTRAL ARIZONA BASINS	3
MAJOR FINDINGS	7
Natural stream water quality	7
Effects of human activities on stream water quality	9
NATIONAL PERSPECTIVE—In the CAZB Study Unit, insecticide concentrations in agricultural/urban streams are among the highest in the Nation	16
Natural ground-water quality.....	17
Effects of human activities on ground-water quality.....	19
NATIONAL PERSPECTIVE—Proposed drinking-water standards for arsenic, radon, and uranium have major implications for ground-water supplies.....	20
NATIONAL PERSPECTIVE—Nitrate concentrations in ground water in the West Salt River Valley are among the highest in the Nation	21
NATIONAL PERSPECTIVE—Trichloromethane was the most commonly detected VOC in the Nation and in the CAZB Study Unit.....	25
STUDY UNIT DESIGN	26
GLOSSARY	28
REFERENCES	30
APPENDIX—WATER-QUALITY DATA FROM THE CENTRAL ARIZONA BASINS IN A NATIONAL CONTEXT.....	33

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

THIS REPORT summarizes major findings about water quality in the Central Arizona Basins Study Unit that emerged from an assessment conducted between 1995 and 1998 by the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program. Water quality is discussed in terms of local and regional issues and compared to conditions found in the 36 NAWQA study areas, called Study Units, assessed to date. Findings are also explained in the context of selected national benchmarks, such as those for drinking-water quality and the protection of aquatic organisms. The NAWQA Program was not intended to assess the quality of the Nation's drinking water, such as by monitoring water from household taps. Rather, the assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water-monitoring programs. The comparisons made in this report to drinking-water standards and guidelines are only in the context of the available untreated resource. Finally, this report includes information about the status of aquatic communities and the condition of in-stream habitats as elements of a complete water-quality assessment.

Many topics covered in this report reflect the concerns of officials of State and Federal agencies, water-resource managers, and members of stakeholder groups who provided advice and input during the Central Arizona Basins assessment. Basin residents who wish to know more about water quality in the areas where they live will find this report informative as well.



THE NAWQA PROGRAM seeks to improve scientific and public understanding of water quality in the Nation's major river basins and ground-water systems. Better understanding facilitates effective resource management, accurate identification of water-quality priorities, and successful development of strategies that protect and restore water quality. Guided by a nationally consistent study design and shaped by ongoing communication with local, State, and Federal agencies, NAWQA assessments support the investigation of local issues and trends while providing a firm foundation for understanding water quality at regional and national scales. The ability to integrate local and national scales of data collection and analysis is a unique feature of the USGS NAWQA Program.

The Central Arizona Basins Study Unit is one of 51 water-quality assessments initiated since 1991, when the U.S. Congress appropriated funds for the USGS to begin the NAWQA Program. As indicated on the map, 36 assessments have been completed, and 15 more assessments will conclude in 2001. Collectively, these assessments cover about one-half of the land area of the United States and include water resources that are available to more than 60 percent of the U.S. population.

SUMMARY OF MAJOR FINDINGS



The Central Arizona Basins (CAZB) Study Unit of the National Water-Quality Assessment (NAWQA) Program covers 34,700 square miles in the Central Highlands and Basin and Range Lowlands hydrologic provinces. Phoenix was America's fastest growing city during the 1990s, and a population of about 3.8 million people is concentrated around the cities of Phoenix and Tucson. The climate is arid to semiarid, and dams on major perennial streams in the Central Highlands collect water for use in the Phoenix area. More than 50 percent of the water used in the Study Unit is ground water, which is often the sole source available. More than 70 percent of the water is used for agriculture, which accounts for 5 percent of the land use.

Stream and River Highlights

Most of the perennial streams in the Central Arizona Basins (CAZB) Study Unit drain relatively undeveloped basins in the Central Highlands that are covered by forests and (or) rangeland. The water quality of these forest/rangeland streams is primarily determined by natural factors, such as chemical weathering of rocks and soils. About 24 percent of samples from forest/rangeland streams had concentrations of phosphorus that exceeded the U.S. Environmental Protection Agency's (USEPA) desired goal for prevention of nuisance plant growth (eutrophication), whereas nitrate concentrations were typically less than the background levels for streams nationally. More than 75 percent of samples from the Salt River (above reservoirs) exceeded the USEPA drinking-water guideline for dissolved solids; however, rainfall and snowmelt runoff helped dilute these concentrations in reservoirs and in streamflow leaving the reservoirs.

In the Basin and Range Lowlands, streams typically flow only when it rains (ephemeral streams). Consequently, a small fraction of the nutrients and dissolved solids applied to the land surface by human, animal, and natural sources is transported to streams. The remaining dissolved solids and nutrients are accumulating in basins and can degrade ground-water quality.

Urban streams with perennial flow are sustained by the discharge of treated wastewater (effluent-dependent). Agricultural/urban streams are a combination of wastewater and irrigation return flows. All samples from both the effluent-dependent urban and agricultural/urban streams exceeded the USEPA's desired phosphorus goal for prevention of nuisance plant growth, and dissolved-oxygen concentrations were minimal for fish survival. Organochlorine compounds in streambed sediment and fish tissue from urban and agricultural/urban streams exceeded guidelines for protection of aquatic health and fish-eating wildlife.

- Effluent-dependent urban streams are valuable water resources; however, the water quality is poor.
- Organochlorine insecticides from past agricultural use persist in streams, streambed sediment, and fish tissue and are a concern because they exceed guidelines for protection of aquatic life and fish-eating wildlife.
- Insecticide concentrations in water from streams affected by agricultural and urban land uses were among the highest in the Nation.

Selected Indicators of Stream-Water Quality

	Urban ¹ (effluent-dependent)	Agricultural/ urban ²	Undeveloped ³ (forest/ rangeland)
Pesticides ⁴	—	●	—
Phosphorus ⁵	●	●	●
Nitrate ⁶	●	●	●
Organo-chlorines ⁷	● ⁺	●	—

■ Percentage of samples with concentrations equal to or greater than a health-related national guideline for drinking water, aquatic life, or water-contact recreation; or above a national goal for preventing excess algal growth

▒ Percentage of samples with concentrations less than a health-related national guideline for drinking water, aquatic life, or water-contact recreation; or below a national goal for preventing excess algal growth

○ Percentage of samples with no detection

— Not assessed

¹ 91st Avenue Wastewater Treatment Plant, Santa Cruz River at Cortaro, Santa Cruz River at Tubac, Santa Cruz River near Nogales International Wastewater Treatment Plant (bed sediment only).
² Buckeye Canal near Avondale (surface water only), Hassayampa River near Arlington (surface water only), Buckeye Canal near Hassayampa (bed sediment only).
³ San Pedro River at Charleston, Gila River at Kelvin, Salt River near Roosevelt, Verde River above West Clear Creek, Verde River below Tangle Creek, West Clear Creek.
⁴ Insecticides, herbicides, and pesticide metabolites, sampled in water.
⁵ Total phosphorus, sampled in water.
⁶ Nitrate (as nitrogen), sampled in water.
⁷ Organochlorine compounds including DDT and PCBs, sampled in bed sediment.
⁺ Although the 91st Avenue Wastewater Treatment Plant outfall is classified as urban, past agricultural land use in the area is the source of most organochlorine compounds at this site.

Trends in stream water quality

Water quality of forest/rangeland streams generally is improving over time. From 1950-90, dissolved-solids concentrations decreased in outflow from reservoirs as a result of dilution from increased precipitation and physical and chemical processes in reservoirs. A decrease in nutrient concentrations in forest/rangeland streams in the early 1980s to 1999 could be attributed to decreased contributions from natural sources, better land-use management practices upstream, or increased nitrogen use by aquatic life.

Major Influences on Streams and Rivers

- Natural factors such as chemical weathering of rocks and soil
- Precipitation
- Reservoirs
- Runoff from agricultural and urban lands
- Discharge of treated wastewater to streams

Ground-Water Highlights

Most of the ground water used in the CAZB Study Unit is pumped from basin-fill aquifers in the Basin and Range Lowlands. Water from major aquifers (basinwide) in the West Salt River Valley (WSRV), the Upper Santa Cruz Basin (USCB), and the Sierra Vista subbasin (SVS) generally meets existing USEPA standards and guidelines for drinking water with some exceptions. Nitrate and dissolved-solids concentrations in some samples from the WSRV and USCB exceeded USEPA drinking-water standards and guidelines. Shallow ground water from an agricultural area in the WSRV exceeded drinking-water standards and guidelines for nitrate and dissolved solids in more than 75 percent of samples. More than 90 percent of ground-water samples from the three basins exceeded the USEPA's proposed drinking-water standard for radon. A small percentage of samples exceeded drinking-water standards for arsenic, fluoride, and molybdenum. Samples from urban and agricultural areas contained low concentrations of numerous chemicals (pesticides and volatile organic compounds) that can be linked to household, industrial, and agricultural uses.

- Most of the deep wells yield old ground water that generally has not been affected by land uses in the last 50 years.
- Use of fertilizers and treated wastewater on agricultural and urban lands and the evaporation of irrigation water have resulted in the accumulation of nitrate and dissolved solids in shallow ground water.

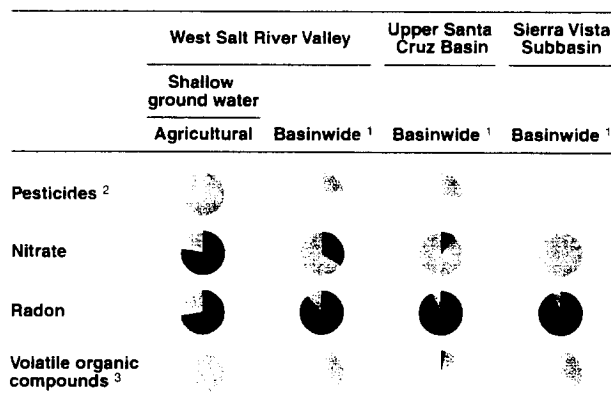
- Adoption of draft or proposed USEPA drinking-water regulations for arsenic, radon, and uranium—constituents that occur naturally in the study area—will require most water suppliers and municipalities to treat their water to remove these constituents or find alternative supplies.
- Pesticides detected in ground-water basins with substantial agricultural and (or) urban development did not exceed USEPA drinking-water standards and guidelines.

Though trends in ground-water quality over time were not determined for the CAZB Study Unit, the data indicate possible future changes. As urban land use spreads with the growing population in the area, ground-water quality is likely to deteriorate, as indicated by detections of pesticides and volatile organic compounds in urban areas. Nitrate and dissolved solids accumulating in shallow ground water in the WSRV have the potential to degrade the quality of deeper drinking-water supplies.

Major Influences on Ground Water

- Geohydrology
- Dissolution of evaporites and other minerals
- Irrigation of agricultural and urban lands
- Agricultural and urban fertilizer and pesticide use

Selected Indicators of Ground-Water Quality



Percentage of samples with concentrations equal to or greater than a health-related national guideline or proposed regulation for drinking water

Percentage of samples with concentrations less than a health-related national guideline or proposed regulation for drinking water

Percentage of samples with no detection

¹ Most wells sampled as part of basinwide surveys were existing domestic (household) wells.

² Insecticides, herbicides, and pesticide metabolites, sampled in water.

³ Solvents, refrigerants, fumigants, and gasoline compounds, sampled in water.

INTRODUCTION TO THE CENTRAL ARIZONA BASINS

The Central Arizona Basins (CAZB) Study Unit encompasses a 34,700-square-mile area in central and southern Arizona and northern Mexico (fig. 1). The Study Unit includes large parts of two hydrologic provinces—the Central Highlands in the north and the Basin and Range Lowlands in the south (U.S. Geological Survey, 1969). Climate, hydrology, geology, land use, and water use are distinctly different in these two provinces.

The Central Highlands (fig. 1) have minimal development and are generally representative of natural conditions. Mountainous terrain with shallow, narrow intermountain basins predominates in

the Central Highlands (Cordy and others, 1998). Forests and rangeland cover most of the province. The largest population is in the town of Prescott—35,785 (Arizona Department of Economic Security, rev. July 7, 2000), and small rural towns dot the region. Agricultural development is minimal except in the northernmost tip of the CAZB.

Most of the perennial streams in the Study Unit are in the Central Highlands (fig. 2). These streams derive their flow from mean annual precipitation of more than 25 inches in the mountains and from rainfall and snowmelt along the Mogollon Rim, which forms the

northeastern border of the CAZB Study Unit.

Major streams having their headwaters in the Central Highlands include the Salt, Verde, and Agua Fria Rivers (fig. 2). These rivers flow year around (perennial) in their upper reaches but are captured for water supply for metropolitan Phoenix, power generation, and flood control before they reach the Basin and Range Lowlands.

Though streams provide most of the water for agricultural use in the Central Highlands, ground water is the main source for municipal and industrial supply (fig. 3). Much of the ground water is pumped from sedimentary deposits of limited

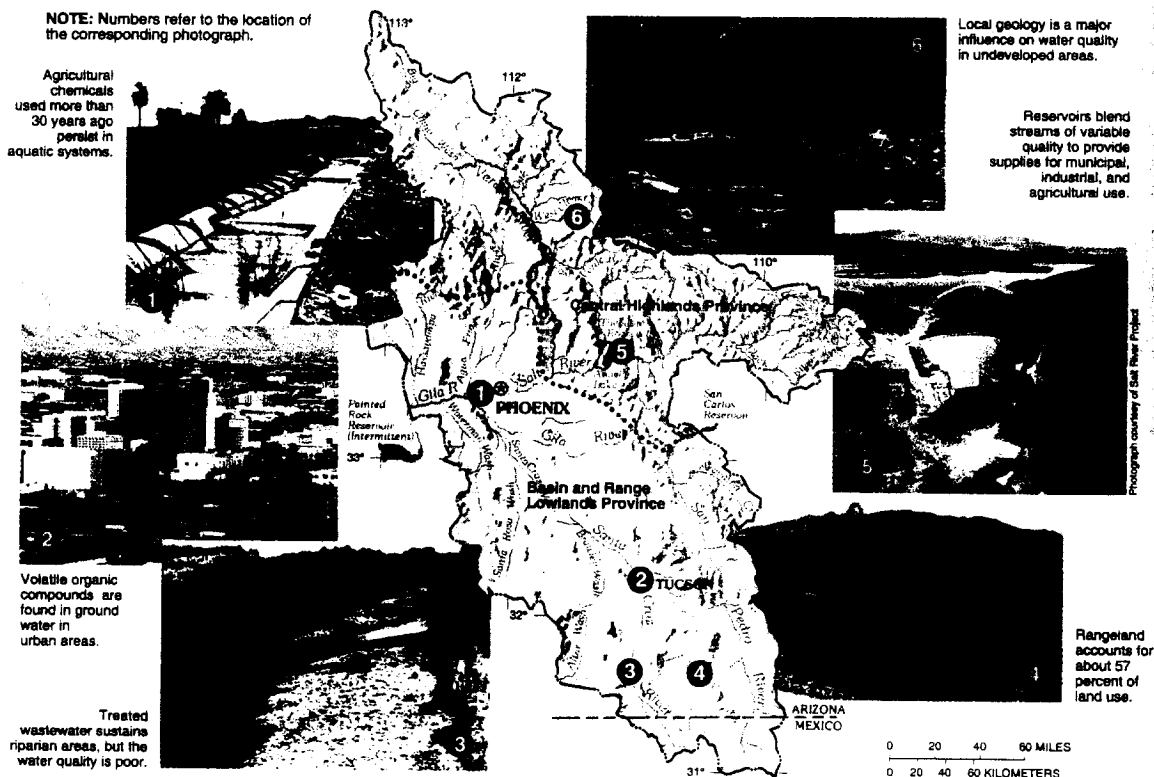


Figure 1. The Central Highlands hydrologic province is mountainous compared to the large, elongate alluvial basins of the Basin and Range Lowlands. Reservoirs capture the perennial streams of the Central Highlands to provide water supplies for the Basin and Range Lowlands.

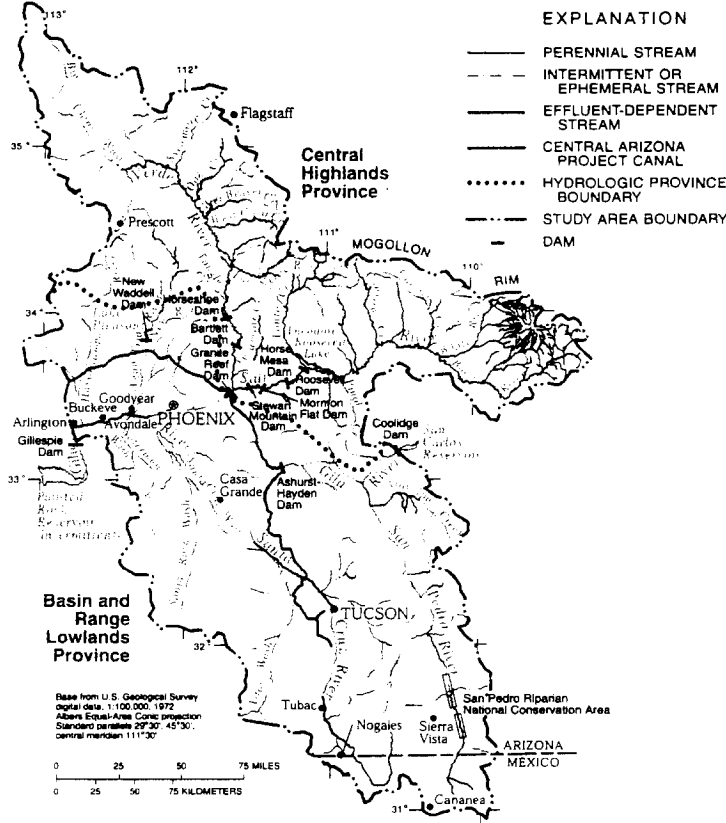


Figure 2. Perennial streams in the Central Highlands, Colorado River water from the Central Arizona Project Canal, ground water, and treated sewage effluent fulfill water demands in the Basin and Range Lowlands.

extent in the valleys. As a result, some of the fastest-growing towns are being forced to seek alternative water supplies (Arizona Department of Water Resources, 1994). Natural factors such as dissolution of minerals in rocks and basin sediments are major influences on ground-water quality in the Central Highlands (Owen-Joyce and Bell, 1983; Marsh, 2000); however, activities such as mining have affected water quality locally (Brown and Favor, 1996).

The Basin and Range Lowlands (fig. 1) are characterized by a lack of perennial streams, the largest water demands, and reli-

ance on ground water. Deep, broad alluvial basins separated by mountain ranges of small areal extent characterize this hydrologic province. The basins are filled with thick deposits of gravel, sand, silt, and clay and include interbedded evaporite deposits and volcanic rocks in places (Anderson and others, 1992). These basin-fill sediments can be 2,000 feet to as much as 12,000 feet thick and constitute the major aquifers that are often referred to as “basin-fill aquifers.” The basin-fill aquifers contain large reserves of ground water that were recharged when Arizona’s climate was much wetter than at

present, possibly thousands of years ago.

Ephemeral streams are characteristic of the Basin and Range Lowlands (fig. 2). Very little natural streamflow is generated because the average annual rainfall is less than 10 to 15 inches except at the highest elevations. With the exception of some small, higher elevation streams and sections of the San Pedro River, most perennial streams in the Basin and Range Lowlands are effluent-dependent; that is, their flow is sustained all year by treated wastewater (fig. 2). Effluent-dependent streams have beneficial uses. They support riparian and aquatic communities where those communities would not otherwise exist. By recharging effluent, cities can accrue “credits” toward pumping of ground water from other locations in a basin (Arizona Department of Water Resources, 1994).

Rangeland is the predominant land use in the Basin and Range Lowlands. The two largest urban areas—Phoenix and Tucson—account for about 5 percent of the land use and include 75 percent of Arizona’s 4.9 million people (Arizona Department of Economic Security, rev. July 7, 2000). Agricultural development, which is mostly west and south of Phoenix, is about 5 percent of the land use (Cordy and others, 1998). Cropland is the primary agricultural land use, and cotton is the main crop.

Water use in the Basin and Range Lowlands represents 96 percent of all water use in the CAZB Study Unit (Cordy and others, 1998). Agriculture is the largest water user (73 percent in 1990; fig. 3). Because of the general lack of surface-water

resources in the Basin and Range Lowlands, ground-water is relied upon heavily to meet agricultural and municipal demands (fig. 3). In areas with substantial agricultural and (or) urban development, ground water has been and continues to be used more quickly than it can be replenished naturally. Ground-water levels have declined several hundred feet in areas with the heaviest pumping, and land subsidence has resulted in a loss in aquifer storage capacity (Arizona Department of Water Resources, 1994). To mitigate some of the

problems caused by overpumping of ground water, Colorado River water is delivered to central Arizona by the Central Arizona Project (CAP) canal (fig. 2). CAP water is used for aquifer recharge and municipal and agricultural purposes.

The study design focused on the effects of land use on water quality. Water, sediment, and biological samples were collected from streams in urban, agricultural, forest, and rangeland areas of the CAZB Study Unit to assess the overall quality of streams as well as the effects of specific land-use practices on stream-water quality (U.S. Geological Survey, 1999). At most sites, water samples were collected monthly from late 1995

through early 1998, and at some stream sites additional samples were collected during storms to assess the effects of stormwater runoff on water quality. Two stream sites were sampled twice monthly for 1 year to determine the occurrence and distribution of pesticides. A single round of sampling for contaminants in streambed sediment and fish tissue was completed in 1995–96 (See “Study Unit Design,” p. 26).

Ground water was sampled from wells in three alluvial basins in the Basin and Range Lowlands—the West Salt River Valley, the Upper Santa Cruz Basin, and the Sierra Vista subbasin. Existing wells were sampled in the three basins to assess overall water quality as well

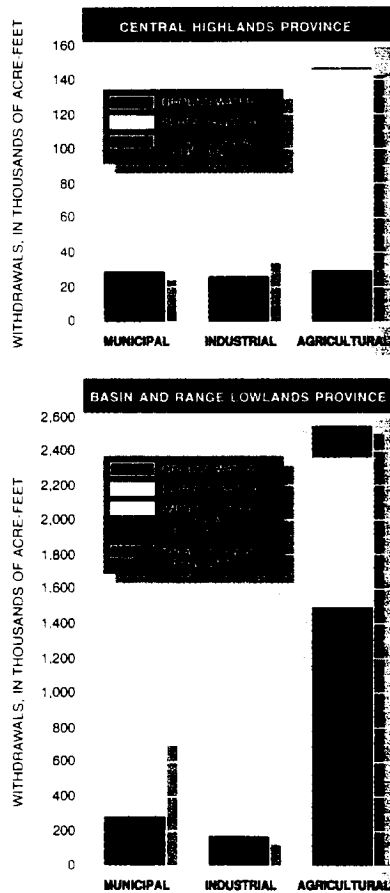
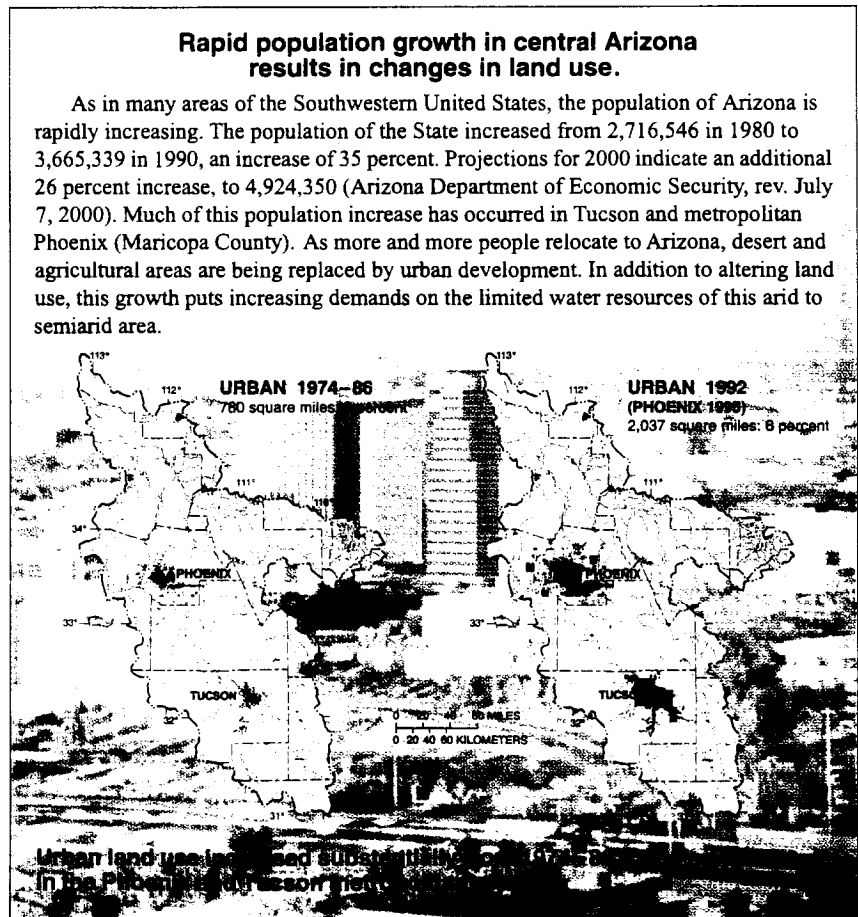


Figure 3. Water-use data for 1990 show the many sources of water used to meet demands in the CAZB Study Unit.



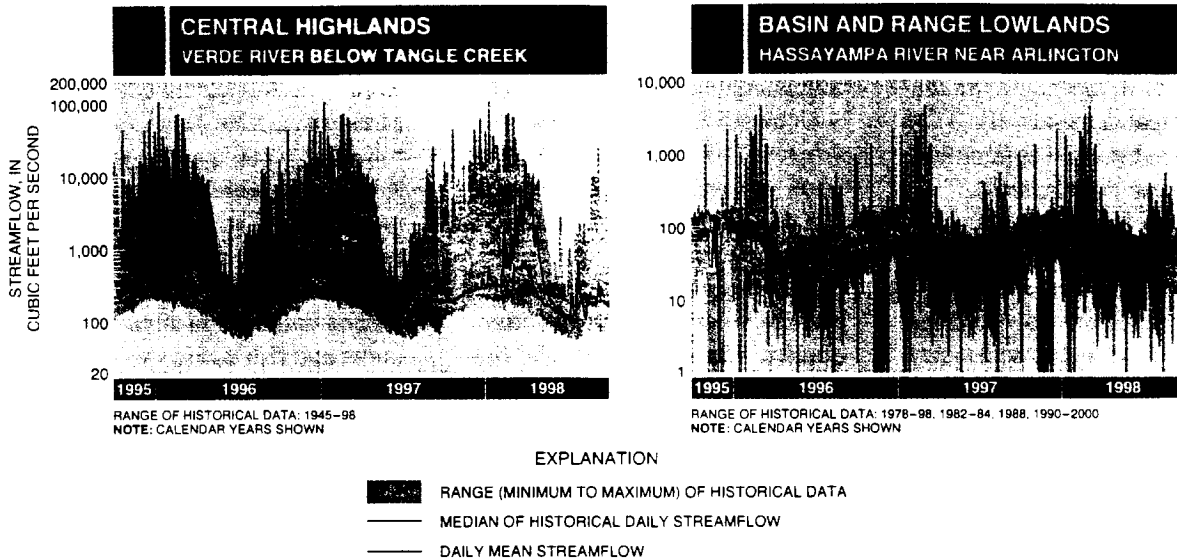


Figure 4. Streamflow in the Central Highlands increased each year from 1996–98 as indicated by the Verde River below Tangle Creek. In the Basin and Range Lowlands, streamflow is difficult to characterize because it is controlled by dams and (or) wastewater-treatment plants. For the Hassayampa River near Arlington, a Basin and Range Lowlands stream, summer streamflow in 1996 and 1997 was greater than the median historical daily value.

as the effects of human activities on water quality. In the West Salt River Valley, shallow monitoring wells were installed and sampled to determine the effects of irrigated agriculture on shallow groundwater quality. Existing groundwater-quality data were used to assess overall water quality in alluvial basins of the Basin and Range Lowlands that were not sampled.

This report is organized into sections on stream-water quality and ground-water quality. In each section, natural water quality, that is water that has been minimally affected by agricultural or urban development, is discussed followed by a discussion of the effects of human activities on water quality. This organization is designed to assist the reader in understanding the changes in natural water quality that result from human activities.

Understanding climatic and hydrologic conditions during the

sampling period, 1995–98, is useful in interpreting the CAZB study results. The climate of the Study Unit is characterized by variability from place to place and also by large differences in precipitation from one year to the next. Precipitation can be three times greater in wet years than in dry years (Cordy and others, 1998).

In Central Highlands streams, represented by the Verde River below Tangle Creek (fig. 4), daily mean streamflow was successively higher from 1996 through 1998. Streamflow in 1998 generally was greater than the median of historical daily streamflow, and streamflow in 1996 was less than the median of historical daily streamflow (fig. 4).

Streamflow in the Basin and Range Lowlands is difficult to characterize because it is controlled by dams and (or) wastewater-treatment plants. The Hassayampa

River near Arlington is an example of a Basin and Range Lowlands stream that is a combination of effluent and irrigation return flows most of the time, supplemented by flows from storm runoff (fig. 4). Streamflow at the site typically was less than the median historical daily streamflow during 1996 and 1997; however, summer streamflow in those years was greater than the median historical daily streamflow because of increased summer thundershowers. Streamflow during 1998 was about the same as the median of historical daily streamflow.

When streamflow exceeds baseflow as a result of rainfall or snowmelt runoff, dissolved-solids concentrations decrease in streams and reservoirs because of dilution. Nutrient concentrations increase with increased streamflow because precipitation and runoff carry more nutrients to streams.

MAJOR FINDINGS

Natural Stream Water Quality

In the CAZB Study Unit, perennial streams draining areas with little or no agricultural or urban land use represent baseline or “natural” conditions in the basins. These natural streams are referred to as “forest/rangeland streams” in this report because they drain basins that are 93 to 100 percent forest and (or) rangeland. Examples of forest/rangeland streams include the upper Verde, upper Salt, and upper Gila Rivers and West Clear Creek in the Central Highlands province and the upper San Pedro River in the Basin and Range Lowlands province. Because some of the forest/rangeland streams provide drinking water for Phoenix or recharge aquifers used for drinking water, the quality of these streams is compared to drinking-water standards and guidelines as well as to other water-quality criteria.

The water quality of forest/rangeland streams is primarily determined by natural factors. Processes such as chemical weathering of bedrock and soils, biological activity in soils (Likens and others, 1977), ground-

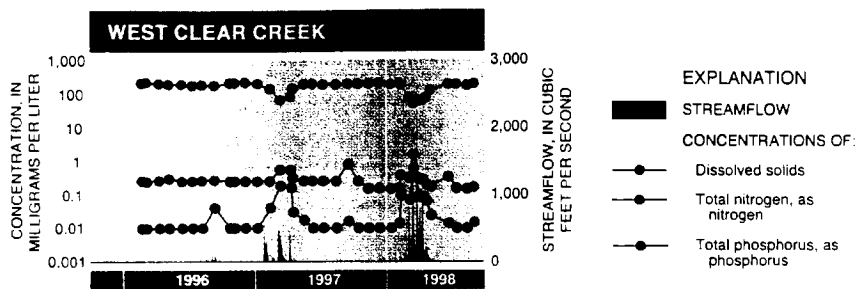


Figure 5. Concentrations of nutrients increase and concentrations of dissolved solids decrease during rainfall or snowmelt runoff.

water discharge to streams, and runoff determine the water quality of these streams. Locally, stream-water quality may be affected by agriculture, mining, or urban land use.

Nutrient and dissolved-solids concentrations fluctuate seasonally in forest/rangeland streams. The patterns of rainfall and snowmelt runoff account for the seasonal fluctuations in concentrations of nutrients (fig. 5). Nutrient concentrations increase in streams during times of rainfall and snowmelt runoff because runoff carries nutrients washed off the land surface to streams, thereby increasing concentrations. Nitrogen in rainfall

and snowmelt also adds to nutrient concentrations in streams. Conversely, during low streamflows, nutrient concentrations are lower because very little runoff reaches streams, and aquatic life in the streams take up the available nutrients.

Seasonal patterns of dissolved-solids concentrations are opposite to those of nutrients. During periods of low flow, the sources of streamflow are springs, which in some areas, such as the upper Salt River Basin, are quite saline (Feth and Hem, 1963). During periods of runoff, flow in streams is diluted, which lowers the dissolved-solids concentrations (fig. 5).

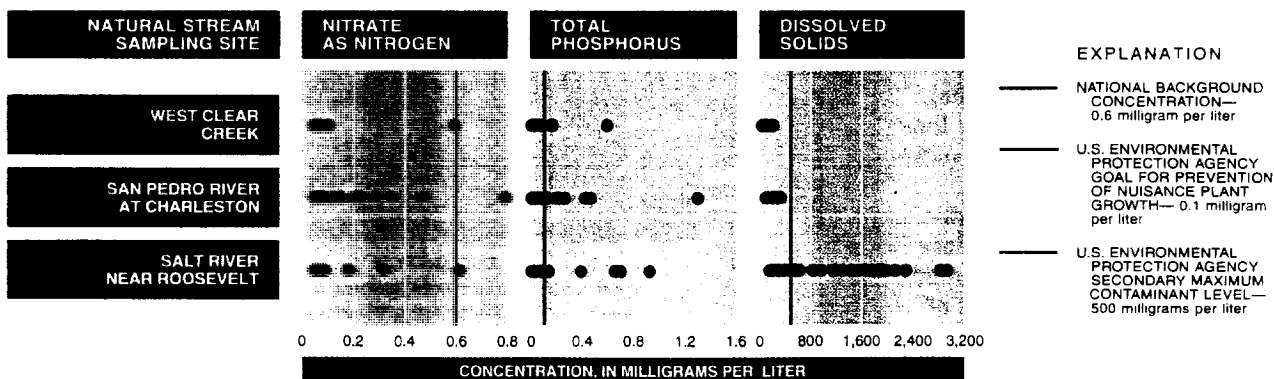


Figure 6. Nitrate concentrations in forest/rangeland streams are significantly lower than the maximum contaminant level of 10 mg/L. Most water samples from the upper Salt River exceeded the secondary maximum contaminant level for dissolved solids (500 mg/L) because saline springs sustain streamflow during periods of low flow.

Nitrate concentrations in forest/rangeland streams were significantly lower than the U.S. Environmental Protection Agency's (USEPA) Maximum Contaminant Level (MCL) of 10 mg/L. Nitrate was detected in 43 percent of the samples from forest/rangeland streams. None of the nitrate concentrations exceeded the MCL, which was established for the protection of human health (fig. 6), and less than 2 percent of the samples had concentrations of nitrate that were greater than the estimated national background concentration in streams of 0.6 mg/L (U.S. Geological Survey, 1999). Concentrations greater than background levels are generally considered to be the result of human activities. Samples that exceeded the background concentration were collected during high flows associated with rainfall or snowmelt runoff.

Twenty-four percent of the samples from forest/rangeland streams exceeded the USEPA desired goal for total phosphorus of 0.1 mg/L for the prevention of nuisance plant growth (fig. 6). The USEPA desired goal of 0.1 mg/L is the same as the estimated national background concentration for phosphorus (U.S. Geological Survey, 1999). Phosphorus enrichment in streams can lead to eutrophication; however, in the forest/rangeland streams, phosphorus concentrations exceeding the USEPA goal are generally limited to periods of rainfall and snowmelt runoff.

Dissolved-solids concentrations exceeded the USEPA Secondary Maximum Contaminant Level (SMCL) in 76 percent of samples from the upper Salt River. None of the samples from the upper San Pedro River or West Clear Creek exceeded the SMCL of 500 mg/L that is based on taste of

drinking water (fig. 6). Samples that exceeded this drinking-water guideline were collected at times when streamflow was sustained by flow from springs. Saline springs drain into the upper Salt River (Feth and Hem, 1963), which accounts for the particularly large number of samples that exceeded the SMCL.

Total DDT concentrations in fish tissue samples from forest/rangeland streams were significantly less than the New York State guideline (Newell and others, 1987) for the protection of fish-eating wildlife. None of the other organochlorine pesticides and PCBs analyzed for were detected in fish tissue from forest/rangeland streams (Gebler, 2000). In addition, organochlorine compounds and PCBs were not detected in streambed sediment from these streams.

Stream water quality generally is improving on the basis of nutrient and dissolved-solids concentrations in forest/rangeland streams. Statistical analysis of nitrogen data for forest/rangeland streams indicates that concentrations have

generally declined since the early 1980s (fig. 7). Phosphorus concentration data showed the same trend as nitrogen. In the upper, undeveloped parts of the Salt and Verde River Basins (upstream from reservoirs) the decrease in nutrients could be from a decrease in contributions from natural sources (see p. 10), a decrease as a result of better land-use management practices upstream, and (or) an increase in nitrogen use by aquatic life.

Dissolved-solids concentrations decreased substantially in outflow from reservoirs on the Verde River from 1950–90 (fig. 7). This downward trend, also seen on the Salt River, probably is caused by both increased rainfall and snowmelt runoff diluting the dissolved-solids concentrations and physical and chemical processes in the reservoirs that remove some dissolved solids from solution.

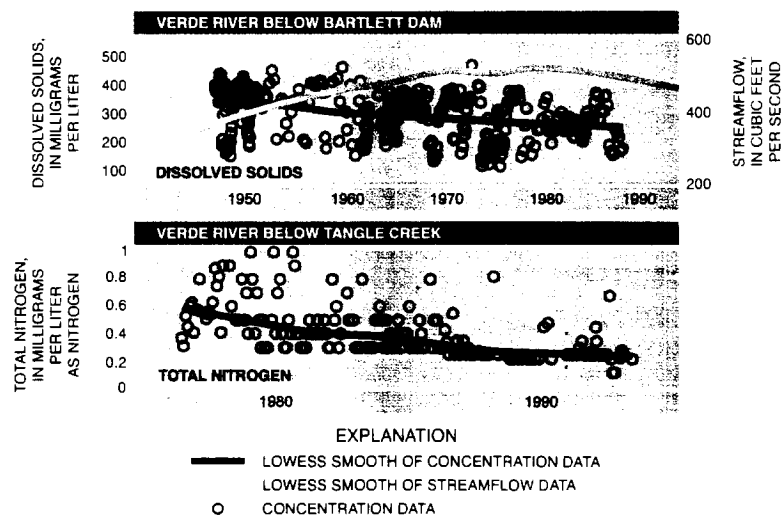


Figure 7. Water quality of forest/rangeland streams has improved, on the basis of decreases in dissolved-solids and nutrient concentrations during the past 30 to 50 years.

Effects of Human Activities on Stream Water Quality

Streams affected by human activities may have elevated concentrations of dissolved solids and nutrients from a variety of activities including urban and agricultural runoff. Manmade compounds such as pesticides and volatile organic compounds (VOCs) in streams are a direct result of human activities. To determine the factors affecting water quality in the CAZB, annual stream loads of dissolved solids (the mass of material transported in the water) entering the basins were compared to annual stream loads leaving the basins (see story at right). In addition, the quantifiable sources of nitrogen and phosphorus (nutrients) coming into major basins and leaving in streamflow were used to identify basins where water quality is affected by human activities (see p. 10). Water-quality characteristics of affected streams are indicative of the local effects of human activities.

Streams sampled in the CAZB that are affected by human activities can be divided into two main categories—**effluent-dependent and agricultural/urban**. Streamflow in effluent-dependent streams is almost entirely treated sewage effluent discharged from wastewater-treatment plants (WWTPs). These streams are referred to in this report as “effluent-dependent” or “effluent-dependent urban” streams (see p. 12) because the effluent reflects urban land uses. Some sampling sites in the CAZB receive irrigation return flows and rainfall runoff from agricultural fields as well as treated effluent, and these streams are referred to as “agricultural/urban” streams.

Dissolved solids are accumulating in basins with agricultural and urban irrigation.

Data collected as part of the CAZB NAWQA study indicate that in 1997, about 1.6 billion kilograms (kg) (1.76 million tons) of dissolved solids were carried into the Basin and Range Lowlands by streams draining the Central Highlands (Verde, Salt, and Gila Rivers) and by the Central Arizona Project (fig. 8). Only 440 million kg (0.48 million tons) were transported out of the study area in streams. The remaining 1.16 billion kg (1.28 million tons) are accumulating in soils, the unsaturated zone, and ground water in irrigated agricultural and urban areas.

Much of the streamflow from the Central Highlands and the Central Arizona Project Canal is used for irrigating agricultural fields and urban landscape. When plants are irrigated in the Basin and Range Lowlands, 50 to 80 percent of the water evaporates or is transpired by plants as pure water. The dissolved solids that were in the evapotranspired water remain in the soil or are concentrated in the water that remains. Over time these salts build up in soils and ground water. To prevent crop damage from salt accumulation, excess irrigation water is commonly applied to leach the salts out of the root zone.

Excess water that percolates below the root zone carries a higher concentration of salts than the original irrigation water (Cordy and Bouwer, 1999). If this deep-percolation water reaches the ground water, the upper part of the aquifer can be contaminated by dissolved solids, nutrients, and pesticide residues. Because deep-percolation water moves slowly through the unsaturated zone and ground water is several hundred feet deep in basins with substantial agricultural and urban development, the effects of the contamination may not be seen in ground water for years or decades after irrigation has declined or ceased.

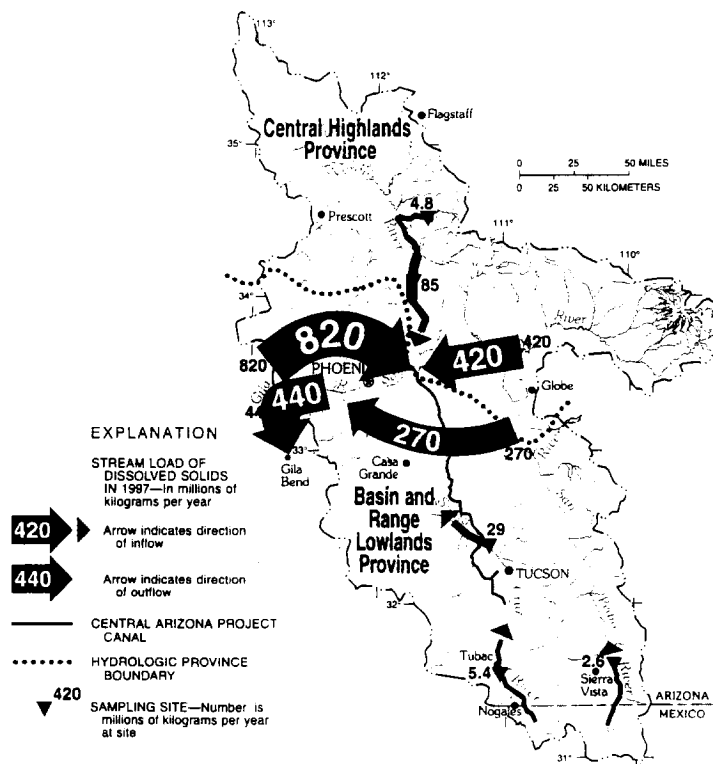


Figure 8. Streams and the CAP canal brought 1.6 billion kilograms of dissolved solids into the Basin and Range Lowlands in 1997, but only 440 million kilograms left the area in streams. The remaining dissolved solids accumulated in soils, the unsaturated zone, and ground water.

What are the sources of nitrogen and phosphorus in basins?

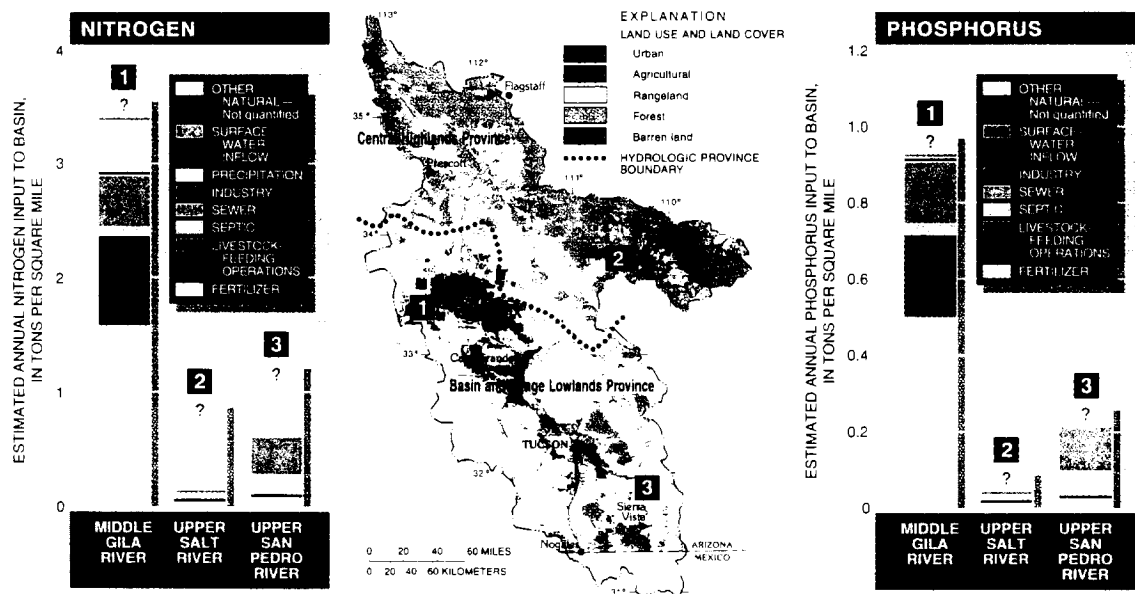
Major sources of nitrogen and phosphorus that can be quantified include fertilizers, livestock-feeding operations (commercial feedlot and dairy operations), inputs to sewer and septic systems, atmospheric deposition, industrial wastes, and streamflow into basins. For the CAZB Study Unit, the quantities of nitrogen and phosphorus contributed by each source annually were determined for selected drainage basins. Sources were quantified using records of fertilizer sales by county, livestock population counts, population and housing census information, National Atmospheric Deposition data for Arizona, USEPA Toxic Release Inventory data, and stream water-quality data collected in the CAZB for the NAWQA Program (Anning, 1998). The quantifiable sources of nitrogen and phosphorus in three basins are shown in figure 9.

Many sources of nitrogen and (or) phosphorus, such as the weathering of geologic formations and soils or the decomposition of vegetation, contribute nutrients to the basins but are difficult, if not impossible, to quantify. As a result, these and other unquantifiable sources of nitrogen and phosphorus are shown in figure 9 with question marks (?) to indicate that the quantities are unknown and may actually exceed the quantifiable sources.

The quantifiable and unquantifiable sources of nitrogen and phosphorus in basins represent potential contributors of nutrients to streams; however, the quantity of nutrients contributed annually from each source does not necessarily reach the streams. Some nutrients may be taken up in terrestrial ecosystems, transported to the ground water, or volatilized to the atmosphere. Conversely, nutrients can enter streams directly when treated sewage effluent from WWTPs is discharged to stream channels or excess irrigation water from agricultural areas discharges to streams. Best management practices and regulation of point-source pollution are methods used to reduce or control the quantity of nutrients entering streams.

In the drainage basins of the upper San Pedro River, upper Salt River, and other perennial streams with minimal agricultural and urban land use, the largest quantifiable source of nitrogen coming into these basins is from precipitation (fig. 9). Sewer and septic systems, livestock-feeding operations, and fertilizers are the largest quantifiable sources of phosphorus in these basins.

In basins with substantial agricultural and (or) urban land use such as the middle Gila River, the quantities of nitrogen and phosphorus from quantifiable sources are much greater per unit area than those for basins with little or no agricultural or urban land use. Additionally, fertilizers, livestock-feeding operations, and sewer (WWTPs) and septic systems account for a larger part of the total nutrients in basins with agricultural and (or) urban land use than in basins without these land uses.



Effluent-dependent streams

Nutrient concentrations in effluent-dependent streams exceeded the background concentrations found in forest/rangeland streams (see fig. 11).

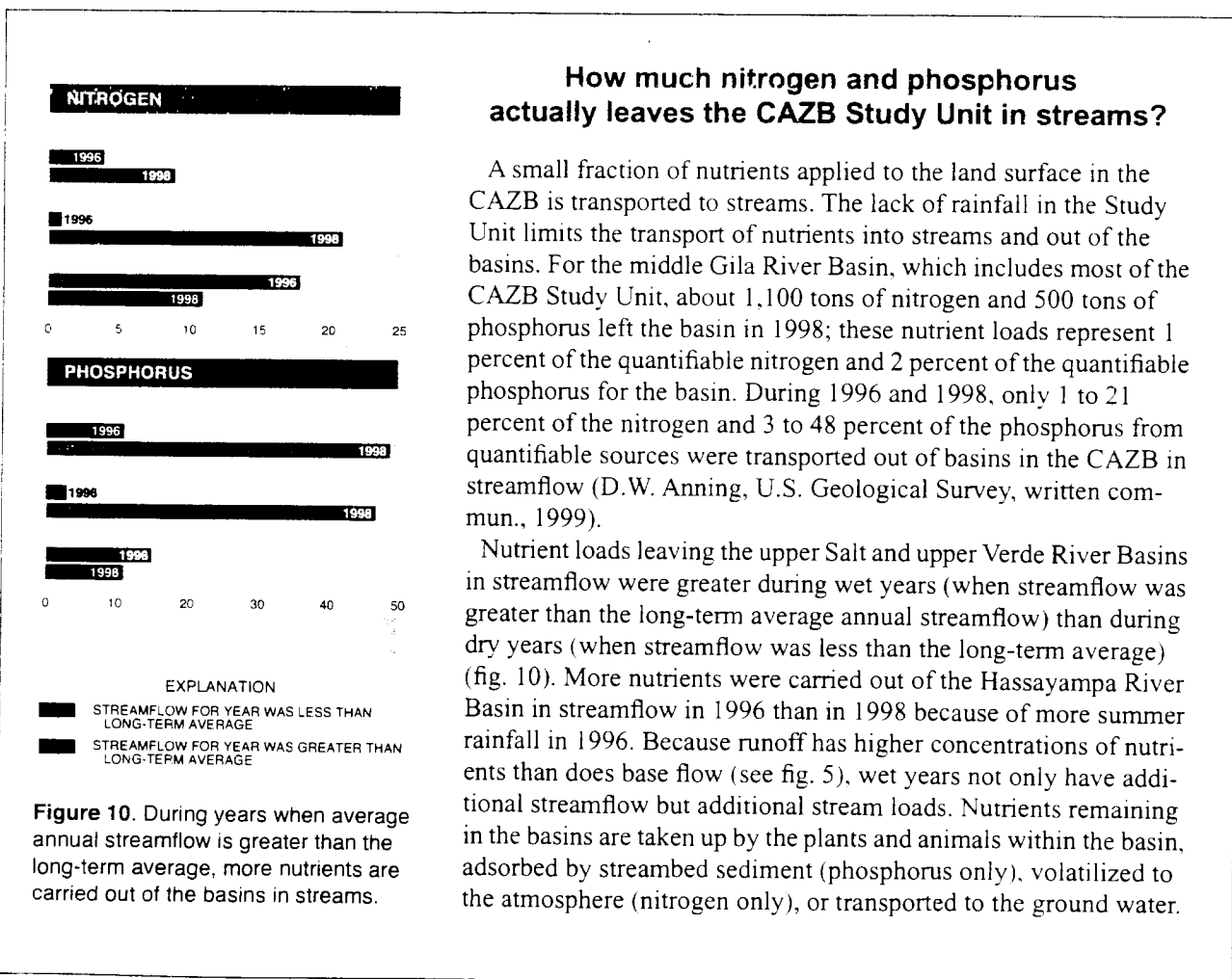
The 91st Avenue WWTP outfall near Phoenix and the Santa Cruz River at Tubac and at Cortaro Road (Tucson; see p. 26 for location of sites) are effluent-dependent streams that were sampled in the CAZB. Data from the San Pedro River at Charleston and the Salt River near Roosevelt represent background values for nutrients in the CAZB because these streams drain areas with relatively little urban or agricultural land use. By

comparison, the nutrient concentrations at the effluent-dependent sites in the CAZB are elevated because the effluent discharged directly into the stream channels is a major source of nitrogen and phosphorus (U.S. Geological Survey, 1999).

Effluent-dependent streams can sustain riparian communities and aquatic life, but the water quality is poor.

Some effluent-dependent streams in the CAZB can support valuable riparian communities with high biodiversity of terrestrial plants and animals; however, dissolved oxygen and phosphorus concentrations in these streams indicate that the water quality is poor. At a minimum, most fish need 3 to 5 mg/L of dis-

solved oxygen (DO) over a long period of time to survive (Swenson and Baldwin, 1965). At the Santa Cruz River at Cortaro, DO concentrations were commonly lower than 3 mg/L (fig. 11), whereas concentrations at the other two effluent-dependent sites were in the minimal range. All the samples from the effluent-dependent streams exceeded the USEPA's desired goal for phosphorus of 0.1 mg/L for prevention of nuisance plant growth (eutrophication) (U.S. Environmental Protection Agency, 1986). Excessive algae and aquatic plant growth can lead to low DO concentrations (U.S. Geological Survey, 1999).



How much nitrogen and phosphorus actually leaves the CAZB Study Unit in streams?

A small fraction of nutrients applied to the land surface in the CAZB is transported to streams. The lack of rainfall in the Study Unit limits the transport of nutrients into streams and out of the basins. For the middle Gila River Basin, which includes most of the CAZB Study Unit, about 1,100 tons of nitrogen and 500 tons of phosphorus left the basin in 1998; these nutrient loads represent 1 percent of the quantifiable nitrogen and 2 percent of the quantifiable phosphorus for the basin. During 1996 and 1998, only 1 to 21 percent of the nitrogen and 3 to 48 percent of the phosphorus from quantifiable sources were transported out of basins in the CAZB in streamflow (D.W. Anning, U.S. Geological Survey, written commun., 1999).

Nutrient loads leaving the upper Salt and upper Verde River Basins in streamflow were greater during wet years (when streamflow was greater than the long-term average annual streamflow) than during dry years (when streamflow was less than the long-term average) (fig. 10). More nutrients were carried out of the Hassayampa River Basin in streamflow in 1996 than in 1998 because of more summer rainfall in 1996. Because runoff has higher concentrations of nutrients than does base flow (see fig. 5), wet years not only have additional streamflow but additional stream loads. Nutrients remaining in the basins are taken up by the plants and animals within the basin, adsorbed by streambed sediment (phosphorus only), volatilized to the atmosphere (nitrogen only), or transported to the ground water.

Effluent-dependent streams are valuable water resources in the CAZB

In many of the urban areas in Arizona, treated sewage effluent from wastewater-treatment plants is discharged into otherwise dry streambeds. Less than a century ago, some of these "effluent-dependent" streams, such as the Santa Cruz River in Tucson (see below) and the Salt River in Phoenix, had natural perennial streamflow, but ground-water pumping, damming of rivers, or other human activities have resulted in a loss of natural streamflow and associated riparian and aquatic communities (Tellman and others, 1997).

In the dry climate of Arizona, effluent-dependent streams provide perennial water resources with a variety of benefits. Effluent-dependent streams can support riparian communities with high biodiversity of terrestrial plants and animals. These streams support limited aquatic invertebrate and fish communities, which are food for organisms higher in the food chain. Riparian plant communities along these streams can help stabilize streambanks, reducing erosion and sedimentation. Trees and bushes provide plant material, creating habitat and food for aquatic organisms and shade that reduces evaporation. Effluent in streams is particularly important to cities and towns in Arizona because it recharges ground water in aquifers and can be used by cities to accrue "recharge credits" that allow for pumping elsewhere in the ground-water basin (Gelt and others, 1999).

Currently (2000), the water quality of effluent-dependent streams limits restoration of instream communities. If the water quality of these streams is improved by upgrading wastewater-treatment methods, it is likely that the streams would be able to support a greater number of aquatic species, and aquatic communities could even begin to resemble those of streams such as the upper San Pedro or upper Salt Rivers.



Newly restored Santa Cruz River near Tubac, Arizona. The Santa Cruz River sustained riparian and aquatic communities in the past (Tellman and others, 1997). Photograph by Walter Hader, Arizona Historical Society, Tucson, 1986.

Santa Cruz River near Tubac, 1997. The effluent-dependent stream at this location is sustained by treated effluent from the local municipal wastewater treatment plant. The thick forest of riparian trees is maintained by perennial streamflow.

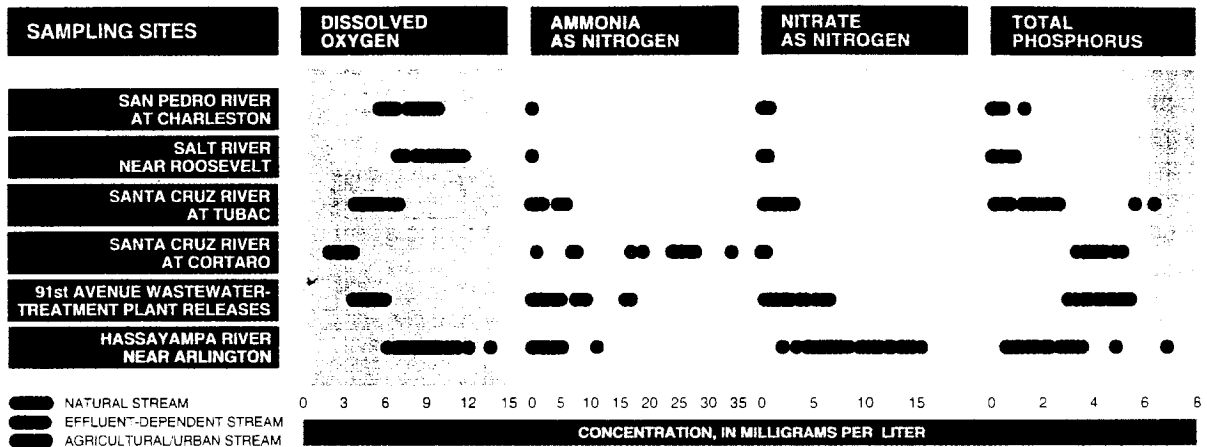


Figure 11. Nutrient enrichment in effluent-dependent streams contributes to abundant algal growth, which results in decreased dissolved oxygen and limited aquatic communities.

The level of sewage treatment and the distance effluent travels downstream from the discharge point influence the water quality of effluent-dependent streams.

Ammonia concentrations in effluent at the Santa Cruz River at Cortaro are extremely variable and typically higher than those in the Santa Cruz River at Tubac or the 91st Avenue WWTP (fig. 11). Effluent at the Cortaro site has had secondary treatment, which results in nitrogen remaining in the effluent as ammonia (David Garrett, Pima County Wastewater, oral commun., 2000). In contrast, effluent sampled at the discharge point from the 91st Avenue WWTP has had tertiary treatment in which the ammonia is converted to nitrate. Converting ammonia to nitrate during treatment limits the direct threat of toxicity to fish that ammonia presents, but it does not change the potential for eutrophication of the stream (Mueller and others, 1996).

The lowest nutrient concentrations in effluent-dependent streams were at the Santa Cruz River at Tubac (fig. 11). Effluent in this stream receives secondary treatment and travels about 15 miles downstream to Tubac. As the effluent moves downstream, ammonia is lost to the atmosphere or converted to nitrate,

some nitrate and phosphorus are taken up by plants and aquatic life, and phosphorus may be adsorbed by streambed sediments. Each of these processes reduces concentrations of nitrogen and (or) phosphorus, resulting in lower concentrations with distance downstream from the WWTP.

Abundant algal growth from nutrient enrichment in effluent-dependent streams may adversely affect aquatic organisms. Phosphorus, nitrate, and ammonia in effluent-dependent streams encourage algal growth. Chlorophyll *a* concentrations (fig. 12), which are indicators of the quantity of algae in a stream, were much higher in effluent-dependent streams than in forest/rangeland streams (Gebler, 1998).

Abundant algal growth and the resulting increase in decaying organic material in effluent-dependent streams can cause decreased DO concentrations, particularly at night when plants cease photosynthesis and decrease their oxygen production. The decreased DO can adversely affect aquatic invertebrates and fish.

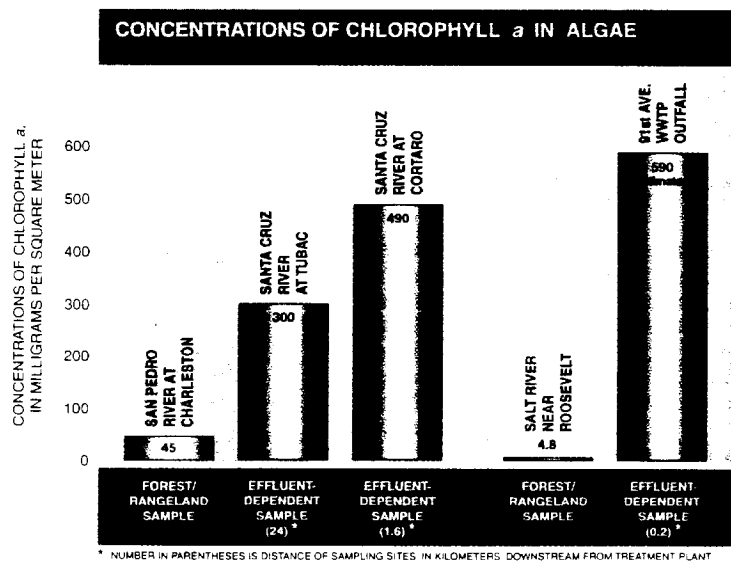


Figure 12. Nutrients in effluent-dependent streams encourage algal growth, as indicated by chlorophyll *a* concentrations.

Effluent-dependent streams support limited instream communities of aquatic invertebrates. The diversity of pollution-sensitive aquatic invertebrates such as mayflies, stoneflies, and caddisflies in effluent-dependent streams is very low, especially when compared to the high diversity in forest/rangeland streams (fig. 13). Pollution-tolerant species of aquatic worms and midges account for more than 90 percent of the numbers of aquatic invertebrates in effluent-dependent stream reaches sampled in the

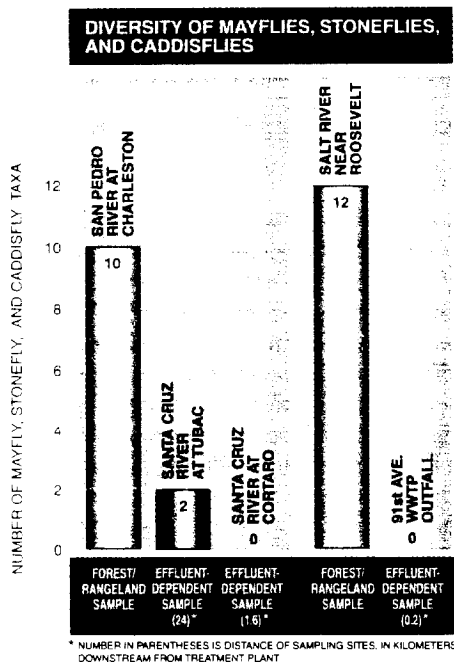


Figure 13. Aquatic invertebrate communities in effluent-dependent streams lack diversity.

CAZB (fig. 14). In forest/rangeland streams, mayflies, stoneflies, and caddisflies were the most abundant of all aquatic invertebrate groups, which is consistent with good water quality and instream habitat (Gebler, 1998).

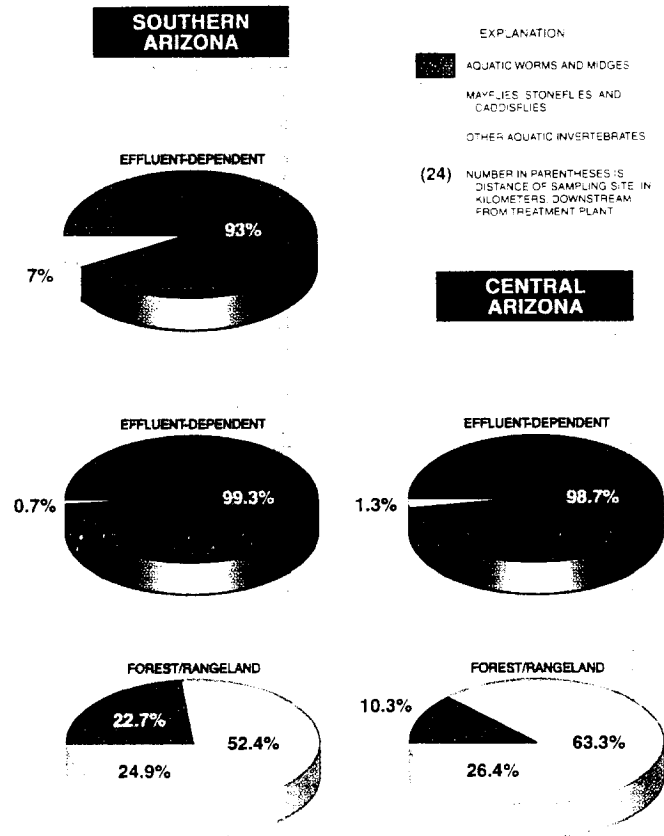


Figure 14. Pollution-tolerant aquatic invertebrates are most abundant in effluent-dependent streams.

Aquatic invertebrates are indica

Aquatic invertebrates are animals such as worms and insects that live in water. Fly fishermen know that game fish such as trout and bass eat insects such as mayflies, stoneflies, and caddisflies. Biologists who study water quality have found that some aquatic invertebrates, such as certain aquatic worms and midges, can tolerate poor water quality. Many types of mayflies, stoneflies, and caddisflies are sensitive to water-quality degradation and are most abundant in streams with good water quality. Biologists can sample for aquatic invertebrates and determine the relative quality of the water by the numbers and types of invertebrates found.



Organochlorine pesticides and PCBs in streambed sediment and fish tissue from effluent-dependent streams exceeded guidelines for protection of aquatic life and fish-eating wildlife. Probable effect levels (PELs) for sediment (Canadian Council of Ministers of the Environment, 1999) were exceeded for DDE and total chlordane in samples from the 91st Avenue WWTP and at a site near the discharge point from the Nogales WWTP into the Santa Cruz River. The PEL is a concentration above which adverse effects to aquatic organisms are predicted to occur frequently. Exceedance of the PEL concentrations indicates that bottom-dwelling aquatic organisms may be adversely affected by toxicity. Total DDT (91st Avenue WWTP) and PCBs (Santa Cruz River at Tubac) in fish-tissue samples exceeded New York State guidelines (Newell and others, 1987) for the protection of fish-eating wildlife. These guidelines are being applied to findings from NAWQA Study Units nationwide. DDT, which breaks down to form DDE and DDD, is associated with past use of DDT in agricultural areas. Use of DDT was discontinued in Arizona in 1969. PCBs were primarily used in industrial and urban settings, but their use was discontinued in 1979. Exceedances of tissue guidelines can result in reduced reproductive ability and other possible adverse effects in wildlife that eat contaminated fish (Faber and Hickey, 1973).

Agricultural/urban streams

As in effluent-dependent streams, nutrient concentrations in agricultural/urban streams were elevated compared with

concentrations in forest/rangeland streams (fig. 11). This is no surprise given that the two agricultural/urban streams—Buckeye Canal near Avondale and Hassayampa River near Arlington—receive effluent from the 91st Avenue WWTP in Phoenix. The effluent is mixed with ground water in Buckeye Canal and used to irrigate cotton and other crops. Downstream, effluent and irrigation return flows in Buckeye Canal are discharged into the Hassayampa River near Arlington. At this point, the water has been used and reused for agricultural irrigation, and nitrate concentrations are typically higher than those in the original effluent (fig. 11) because of the use of fertilizers in the agricultural area near Buckeye.

Herbicides were detected in streams soon after application to agricultural lands, but concentrations did not exceed guidelines for protection of aquatic life. In the West Salt River Valley west of Phoenix, the pre-emergent herbicides dacthal, EPTC, simazine, and trifluralin are applied to tilled fields prior to cotton planting in the early spring to control weeds. They may

be reapplied in the fall to fields where winter crops are grown. These herbicides were detected in surface-water samples from the agricultural/urban streams in the early spring and fall, soon after application. Changes in concentrations of dacthal at the Hassayampa River near Arlington (fig. 15) are representative of the patterns seen for herbicide concentrations at both sites. Agricultural and rainfall runoff carry these pesticides to streams. Because streamflow at these sites is not used for drinking water but does sustain aquatic life, guidelines for the protection of aquatic life were used to evaluate water quality. Aquatic-life guidelines for simazine and trifluralin were not exceeded in any samples from these sites. There are no aquatic-life guidelines for dacthal and EPTC.

Organochlorine pesticides that persist in streambed sediment and in fish tissue from an agricultural/urban stream are a concern for aquatic ecosystem health. PEL concentrations for sediment were exceeded for DDE and DDT at the agricultural/urban stream site on the Buckeye Canal

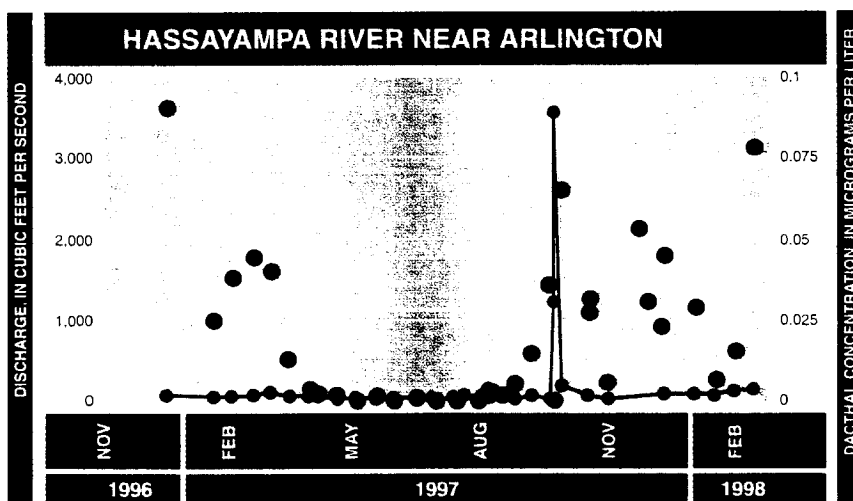


Figure 15. Herbicides were detected in streams soon after being applied to crops.

near the Hassayampa River (adjacent to the Hassayampa River at Arlington site, see p. 26). Concentrations of DDE in fish-tissue samples from this site exceeded guidelines established by New York State (Newell and others, 1987) for the protection of fish-eating wildlife. Concentrations of toxaphene in two out of three fish-tissue samples from the same site

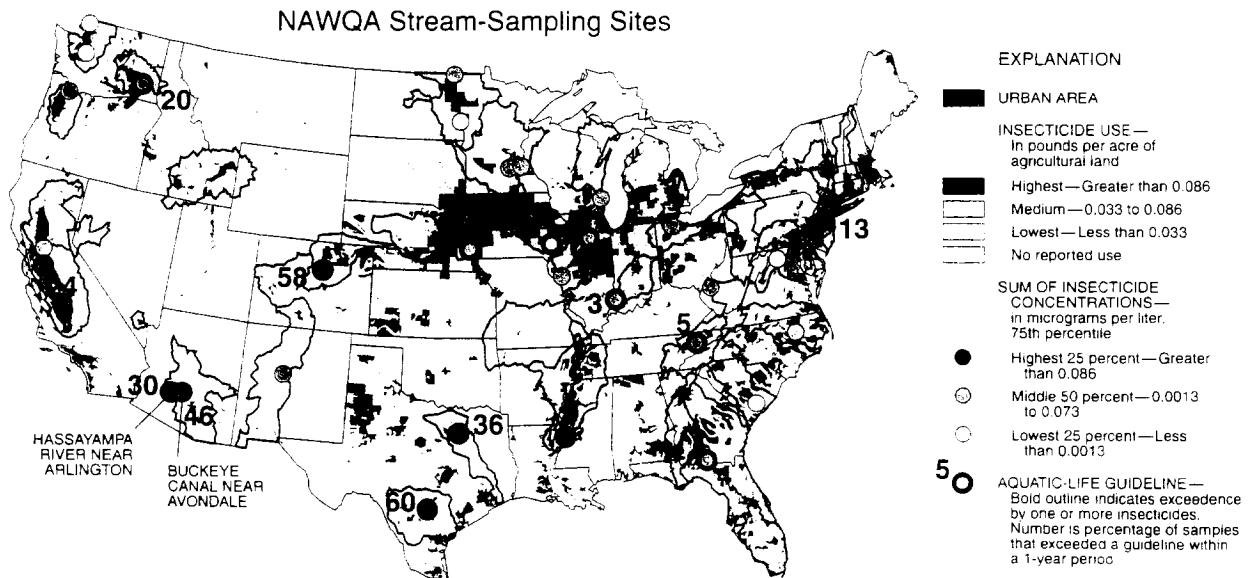
exceeded the National Academy of Science/National Academy of Engineering (1973) guideline for the protection of fish-eating wildlife. Past use of pesticides including DDT, toxaphene, and others on agricultural areas in the West Salt River Valley is the source of these pesticides. Though use of these pesticides was discontinued decades ago, the pesticides persist

over time and their breakdown products continue to enter streams by erosion of contaminated soils, surface-water runoff, and atmospheric deposition. Exceedances of tissue guidelines indicate possible adverse effects, such as reduced reproductive ability and eggshell thinning, to birds and other wildlife that eat contaminated fish (Faber and Hickey, 1973).

In the CAZB Study Unit, insecticide concentrations in agricultural/urban streams are among the highest in the Nation

Pesticides in water were measured at 117 sampling sites on 114 rivers and streams across the United States as part of the NAWQA Program from 1992–1998. At each site, concentrations of all insecticides detected during a 1-year period were summed and categorized as low (lowest 25 percent), middle (middle 50 percent), and high (highest 25 percent) compared to concentrations at all of the sites monitored (see figure below). This information was compared, by county, to insecticide use during the early to mid-1990's on agricultural lands.

In the CAZB Study Unit, insecticide concentrations in streams with mixed agricultural/urban land use were among the highest in the Nation. These sites—the Buckeye Canal near Avondale and the Hassayampa River near Arlington in the West Salt River Valley—are dominated by treated effluent and irrigation return flows that contain insecticides from urban and agricultural land uses. Nearly one-half the samples (46 percent) collected from the Buckeye Canal during 1 year exceeded aquatic-life guidelines for one or more of the following insecticides: diazinon, malathion, lindane, and chlorpyrifos. At the Hassayampa River site, 30 percent of samples collected in a 1-year period exceeded aquatic-life guidelines for one or more of the following pesticides: chlorpyrifos, azinphos-methyl, DDE, dinoseb, malathion, diazinon, and parathion. Although these streams are not used for drinking water, the water quality does present a potential hazard to aquatic life. In addition, little is known about the effects of mixtures of pesticides, even at low concentrations, on aquatic life (Gilliom, 1999).



Insecticide concentrations in streams in the West Salt River Valley near Phoenix are among the highest in the Nation.

Natural Ground-Water Quality

It is important to understand how natural processes affect ground-water quality in order to identify the effects of urban and agricultural development under similar hydrogeologic conditions. In the CAZB, the majority of ground-water basins do not have significant urban or agricultural development. The ground-water quality in these basins is primarily a product of natural processes such as the interaction of ground water with rocks and sediment in the basins (Robertson, 1991).

Natural sources of dissolved-solids and nitrate can control ground-water quality in basins with minimal urban development. Specific-conductance values (an indirect measure of the dissolved-solids concentration) and nitrate concentrations for ground water in basins with minimal urban development increase northward from southeastern Arizona toward the central part of the State (figs. 16 and 17). The increasing specific-conductance values can be attributed to a corresponding increase in evaporite deposits in basin sediments from southeast to northwest (Gellenbeck and Coes, 1999). Evaporite deposits in the basins contain minerals such as halite (salt) and gypsum that can be easily dissolved in ground water. (Robertson, 1991). The increasing nitrate concentrations can be largely attributed to natural sources; however, human activities such as agriculture can be a source in some basins. In some locations in the CAZB, high nitrate concentrations in ground water reported prior to any agricultural or urban development indicate that natural

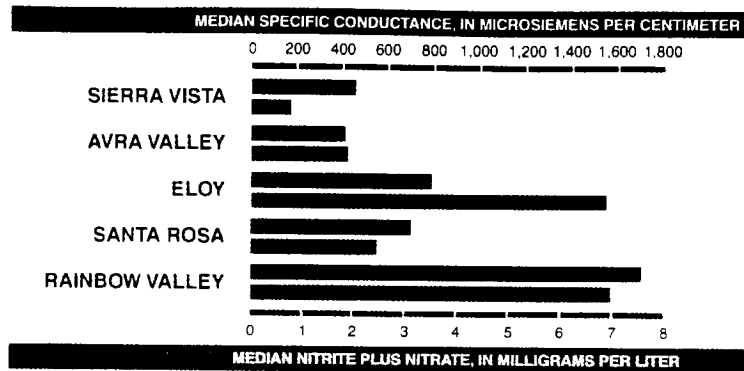


Figure 16. Nitrate concentrations and specific conductance values in ground water from basins with minimal urban development increase from the southeast to the northwest. (Basins shown below in figure 17.)

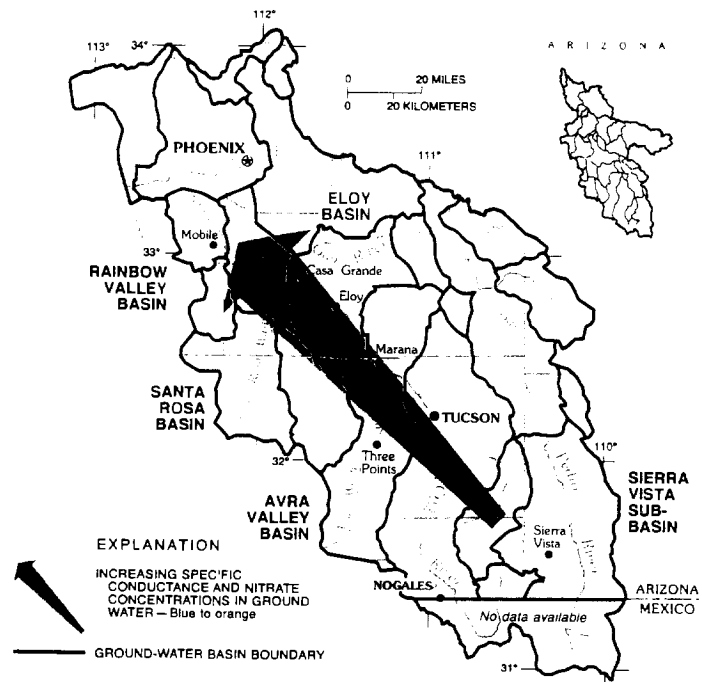


Figure 17. Increasing specific conductance values in ground water from southeast to northwest can be attributed to an increase in soluble evaporite deposits in basin sediments. Increasing nitrate concentrations in the same direction may be the result of naturally occurring nitrate and of human activities that include agriculture.

sources of nitrate are present in some basins (Hem, 1985; Robertson, 1991; Gellenbeck, 1994; Gellenbeck and Coes, 1999). Dissolution of evaporite deposits, decay of buried organic matter, precipitation, weathering of rocks and soils, and fixation by microorganisms are just a few of the possible sources of naturally occurring nitrate in ground water.

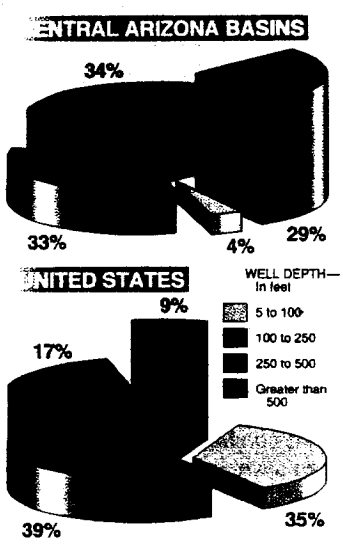


Figure 18. Ground water sampled in the CAZB Study Unit generally is from greater depths than ground water sampled in NAWQA Study Units across the Nation.

Quality of deep, older ground water unaffected by human activities

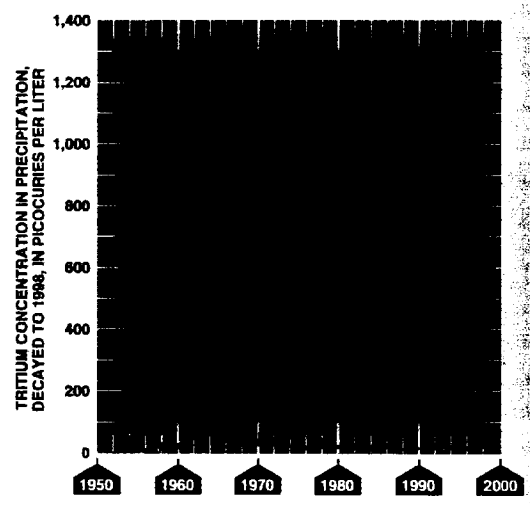
In general, ground water in Arizona is replenished (recharged) at very slow rates because of little precipitation, high evaporation losses, and the long distance water must travel to recharge deep aquifers. In the CAZB, 63 percent of the wells sampled that draw water from aquifers used for drinking water were at least 250 feet deep compared to 26 percent of NAWQA wells sampled nationwide (fig. 18). Recharge takes longer to reach deep aquifers than shallow aquifers; therefore, deep ground water typically was recharged earlier and is older than shallow ground water. Tritium age dating of ground water (see below) confirms that 55 percent of the wells sampled in the CAZB yielded ground water that was recharged prior to 1953 and possibly thousands of years earlier. For example, some ground water in the Upper Santa Cruz Basin was determined to be about 6,500 years old (Kalin, 1994). Across the Nation, only 27 percent of the NAWQA wells sampled for tritium yielded ground water that was recharged prior to 1953.

The age and depth of ground water in the CAZB have important implications for water quality and quantity. Because much of the deep ground water sampled in the CAZB was recharged prior to 1953 and has not mixed with younger recharge, drinking-water quality generally has not been substantially affected by human activities that took place after 1953. The movement of contaminants from the land surface to the deep ground water is hindered by the thickness of basin-fill sediments through which contaminants must travel. Ground-water quantity is affected because ground water pumped from deep aquifers is not being replaced by recharge (see p. 5), resulting in a net decrease in the quantity of ground water available for consumption.

Method for dating ground water

Tritium is a radioactive isotope that can be used to estimate whether ground water has been recharged before or after 1953. Large quantities of tritium were released to the atmosphere during testing of thermonuclear weapons from 1952 until the late 1960s. Atmospheric tritium is incorporated into water molecules in the atmosphere prior to precipitation and recharge to ground water. The concentration of tritium in ground water at a given time is controlled by both the quantity of tritium in the atmosphere when precipitation and recharge occur and the radioactive decay rate of tritium. Ground water that does not contain detectable tritium (less than 2.5 picocuries of tritium per liter) can be assumed to have been recharged prior to 1953, and ground water that does contain detectable tritium (more than 2.5 picocuries of tritium per liter) is assumed to contain some component of ground water that was recharged after 1953.

Tritium concentrations in precipitation are a guide for determining when ground water was recharged.



Concentrations of arsenic, fluoride, and molybdenum exceeded drinking-water standards in samples from major aquifers. The median arsenic concentration in ground water for the three CAZB basins sampled was 4 µg/L. One sample from the Upper Santa Cruz Basin and one sample from the West Salt River Valley exceeded the current MCL for arsenic of 50 µg/L; however, a new, lower standard of 5 µg/L has been proposed by the USEPA because of the cancer risk posed by arsenic in drinking water (U.S. Environmental Protection Agency, rev. August 25, 2000). When arsenic concentrations in ground water sampled in the CAZB are compared to the proposed standard, more than 50 percent of samples from

aquifers in West Salt River Valley that are used for drinking water exceed 5 µg/L. Seventeen percent of samples in the Upper Santa Cruz Basin and 10.5 percent of samples in the Sierra Vista subbasin exceed 5 µg/L. The USEPA may not settle on 5 µg/L, but the new standard is likely to be significantly lower than the current MCL.

The median concentration of fluoride was 0.5 µg/L; about 2 percent of the samples exceeded the current MCL for fluoride of 4 µg/L. The median concentration of molybdenum was 3 µg/L; about 1 percent of the samples exceeded the current lifetime health advisory for molybdenum of 40 µg/L established by the USEPA.

Radon and uranium are detected in most ground-water samples. Radon is a colorless and odorless radioactive gas that is carried in the water pumped from wells (fig. 19) and released to indoor air by activities such as cooking and showering. Breathing radon increases the risk of lung cancer (U.S. Environmental Protection Agency, rev. October 18, 1999). Radon is naturally formed in rocks and soils from the radioactive decay of radium, an intermediate product in the uranium decay process. In the CAZB Study Unit, radon was present in 100 percent of the samples, and uranium was detected in 90 percent of the samples. The median concentrations for radon and uranium were 584 picocuries per liter and 3 micrograms per liter, respectively. Currently (2000), there are no USEPA MCLs for radon and uranium; however, proposed MCLs could result in increased costs for water suppliers to treat drinking water for these constituents or find alternate supplies. Additional costs would probably be passed on to the water user (see information on proposed standards for arsenic, radon, and uranium on p. 20).



Figure 19. Samples are collected at the well head for radon analysis to prevent possible sample contamination from exposure to the atmosphere.

Effects of Human Activities on Ground-Water Quality

The contamination of major aquifers is largely controlled by hydrology and land use (U.S. Geological Survey, 1999). In the CAZB Study Unit, deep ground water that was recharged prior to 1953 typically has not been affected by human activities (see p. 18). In areas with recent recharge (after 1953), ground water is more likely to be contaminated by nutrients and man-made chemicals associated with urban and agricultural land uses.

Ground-water quality deteriorates in irrigated areas. Irrigation water that seeps downward is a principal source of ground-water recharge in irrigated areas of the CAZB. Dissolved-solids concentrations in seepage can be as much as five times those in the original irrigation water (Bouwer, 1990) because of concentration by evaporation and plant use (see p. 9). The greater the dissolved-solids concentration in the applied irrigation water, the greater the concentration in the seepage moving downward to the ground water.

To determine the effects of irrigated agriculture on shallow ground-water quality, nine monitoring wells

were drilled and sampled in the southwestern part of the West Salt River Valley (see "Study Unit Design," p. 26). Because the average depth to ground water in the nine wells is 32 feet (table 1) compared to 230 feet for wells sampled basinwide, irrigation seepage does not have to travel far to reach the shallow ground water in the agricultural area. Sources of irrigation water in this area include treated sewage effluent, water from the Salt River and CAP canal, irrigation return flows, and ground water. Dissolved-solids concentrations of these sources range from about 900 mg/L for treated sewage effluent (Tadayon and others, 1998) to 650 mg/L for CAP water and 470 mg/L for Salt River water (Salt River Project, 1997).

The median dissolved-solids concentration in water from the nine shallow wells exceeded 3,000 mg/L (table 1). In addition, the effects of nitrate from fertilizer applications and reuse of irrigation return flows were evident from the median nitrate concentration that was nearly twice the MCL of 10 mg/L (table 1).

Table 1. Median concentrations of nitrate and dissolved solids were highest in shallow ground water from an agricultural area in the West Salt River Valley

Study area	Median concentration, in milligrams per liter		Average depth to ground water, in feet
	Nitrate	Dissolved solids	
<i>West Salt River Valley</i>			
<i>Agricultural area</i>	19.0	3,050	32
<i>Basinwide</i>	2.7	560	230
<i>Upper Santa Cruz Basin</i>	1.5	305	230
<i>Sierra Vista subbasin</i>	0.78	262	171
<i>U.S. Environmental Protection Agency drinking-water standard</i>	¹ 10	² 500	

¹ Maximum Contaminant Level.

² Secondary Maximum Contaminant Level.

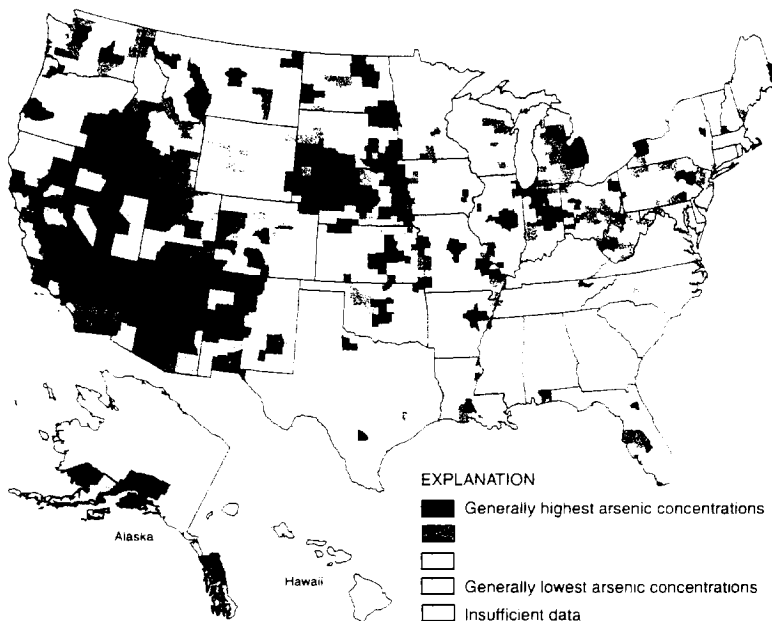


Proposed drinking-water standards for arsenic, radon, and uranium have major implications for ground-water supplies

During 1991–98, arsenic, radon, and uranium were measured in ground-water samples from 36 NAWQA Study Units across the United States. If the ground-water samples from these Study Units are representative of ground water across the Nation, MCLs (see Glossary) proposed by the USEPA for these constituents will affect many water suppliers and municipalities in the United States. Because ground-water supplies in many parts of the Nation will likely exceed the proposed MCLs, public-water systems would be required to either specifically treat their water to decrease concentrations of the constituents or find alternative sources of supply. Costs of these options would probably be passed on to water users.

Arsenic

The current USEPA MCL for arsenic in drinking water, 50 µg/L, is under review after recognition of the risks of developing cancers (National Research Council, 1999). In 2000, the USEPA proposed a new, lower arsenic MCL of 5 µg/L (U.S. Environmental Protection Agency, rev. June 2, 2000). In samples collected in three CAZB basins, only 2 percent exceeded the current USEPA MCL; however, 32 percent exceeded the lower, proposed MCL. In samples collected across the Nation, only 0.6 percent exceeded the current USEPA MCL; however, 14 percent exceeded the lower proposed MCL. Arizona, including the CAZB, is among the areas in the Nation where 10 percent or more of ground-water samples are likely to exceed the lower MCL (figure at right; Welch and others, 2000).



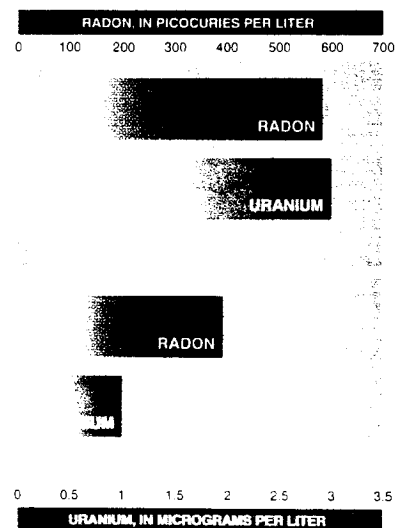
High concentrations of arsenic in ground water are more widespread in the West.

Radon and uranium

Currently, USEPA MCLs for radon and uranium do not exist. Because of public health concerns, including increased risks for developing lung cancer, the USEPA proposed an MCL of 20 µg/L for uranium in 1991. In 1999, the USEPA proposed an MCL of 300 picocuries per liter (pCi/L) for radon; however, if States or water suppliers implement methods to lower radon levels in indoor air, they would only be required to meet an Alternate MCL (AMCL) of 4,000 pCi/L (U.S. Environmental Protection Agency, rev. April 21, 2000; U.S. Environmental Protection Agency, rev. October 18, 1999). Of the ground-water samples collected in the three CAZB basins, 9 percent exceeded the proposed uranium

MCL, 91 percent exceeded the proposed radon MCL, and 1 percent exceeded the proposed radon AMCL. In the NAWQA samples collected nationwide, 4 percent exceeded the proposed uranium MCL, 61 percent exceeded the proposed radon MCL, and 4 percent exceeded the proposed radon AMCL (Dennis Wentz and others, U.S. Geological Survey, written commun., 1999). The Study Units with the highest radon concentrations were in the Colorado Rockies and the Eastern United States. Median concentrations of radon and uranium in the CAZB Study Unit were higher than median concentrations for the United States (figure at right).

Median concentrations of radon and uranium in ground water in the CAZB exceeded those for the Nation.



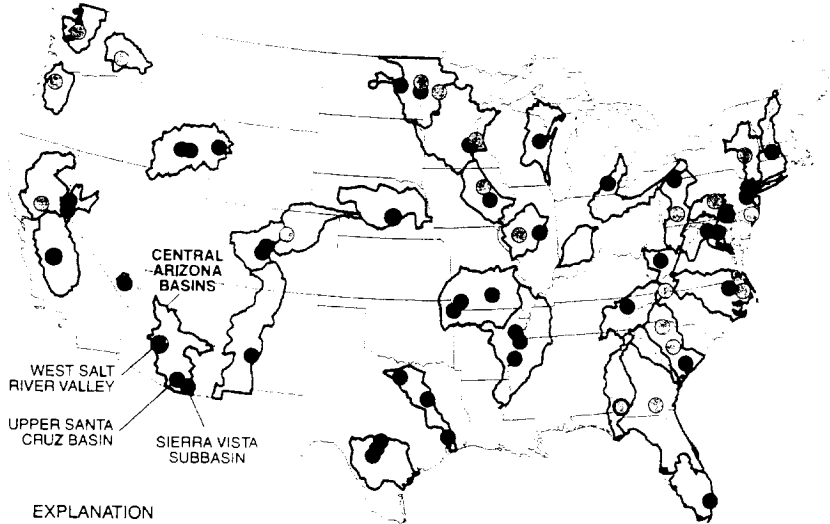


Nitrate concentrations in ground water in the West Salt River Valley are among the highest in the Nation

The study of water quality in deep aquifers that provide drinking water in the West Salt River Valley is one of eight NAWQA ground-water studies nationwide that had more than 10 percent of samples that exceeded the USEPA MCL for nitrate of 10 mg/L (figure at right). Of 35 samples collected basinwide in the West Salt River Valley, 34 percent had concentrations greater than the USEPA MCL. Seventy-eight percent of shallow ground-water samples from the agricultural land-use study in the West Salt River Valley had concentrations greater than the USEPA MCL for nitrate. Only 10 percent of samples from the Upper Santa Cruz Basin and none of the samples from the Sierra Vista subbasin had concentrations that exceeded the USEPA MCL for nitrate.

Excessive nitrate in drinking water is a health concern for children and adults. In children, high nitrate concentrations can result in "blue-baby syndrome," in which oxygen levels in the blood of infants are low, sometimes fatally so (National Governor's Association, 1991).

Birth defects also have been attributed to high nitrate concentrations (National Governor's Association, 1991). In adults, high nitrate concentrations have been associated with cancer (National Academy of Sciences, 1977).



EXPLANATION

PERCENTAGE OF SAMPLES EXCEEDING DRINKING-WATER STANDARD FOR NITRATE OF 10 MILLIGRAMS PER LITER—
Each dot represents a major aquifer

- Greater than 10 percent
- ◐ Less than 10 percent
- Zero samples exceed standard

○ BACKGROUND CONCENTRATION—
Bold outline indicates median values greater than background concentration of 2 milligrams per liter

The CAZB is one of eight Study Units in the Nation with nitrate concentrations in ground water that exceed the drinking-water standard in more than 10 percent of samples.

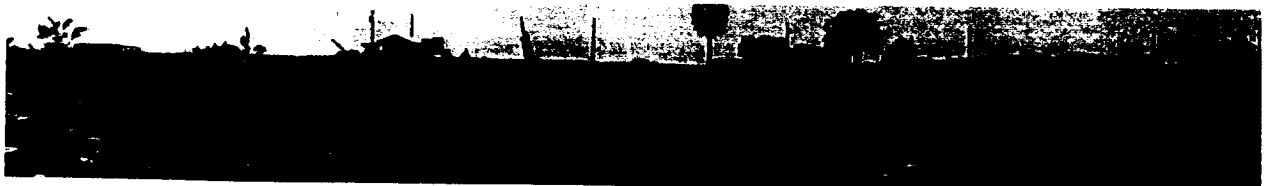
Sources of nitrate in the Central Arizona Basins Study Unit include evaporite deposits in basin sediments, precipitation, agricultural fertilizers, animal-feeding operations, WWTP outflow, and others

(see p. 10). In areas with agricultural and (or) urban development, sources of nitrate related to human activities are prevalent, whereas precipitation and geologic sources of nitrate predominate in undeveloped areas.

The highest concentrations of nitrate and dissolved solids were in shallow ground water beneath an irrigated agricultural area. Shallow ground water from the agricultural land-use study area in the West Salt River Valley had median concentrations of nitrate (19 mg/L) and dissolved solids (3,050 mg/L) that exceeded the USEPA MCL and SMCL, respectively (table 1). Nitrate and dissolved solids from irrigation and agricultural practices are accumulating in shallow ground water (see p. 9 and 11). The shallow ground water in this area is not used for drinking water, and clay beds reduce the

likelihood of contamination of the aquifers below that are used for drinking water (see p. 22).

Deeper ground water from urban, rangeland, and agricultural areas in other parts of the West Salt River Valley had a median nitrate concentration that was less than the MCL of 10 mg/L; however, the median concentration of dissolved solids exceeded the SMCL of 500 mg/L (table 1). Median concentrations of nitrate from the Upper Santa Cruz Basin and the Sierra Vista subbasin also were less than the MCL, and median concentrations of dissolved solids were less than the SMCL (table 1).



Clay beds that currently protect deep ground water from contamination may not do so in the future.

In the agricultural land-use study area of the West Salt River Valley (fig. 20), the tops of low-permeability clay beds are about 150 to 400 ft below the land surface. These clay beds impede the downward movement of irrigation seepage and reduce the likelihood of contaminants reaching deeper drinking-water supplies. Domestic wells in the area yield water from beneath the protective clay beds. Ground water above the clay beds has higher nitrate and dissolved-solids concentrations than ground water from beneath the clay beds (fig. 21). In this area, ground-water samples from above the clay beds had a median dissolved-solids concentration of 3,050 mg/L and a median nitrate concentration of 19.0 mg/L (table 1). Ground-water samples from below the clay beds had a median dissolved-solids concentration of 702 mg/L and a median nitrate concentration of 1.9 mg/L. Care must be taken in drilling and completing drinking-water wells below the clay beds to ensure that shallow ground water above the clay beds does not contaminate the well and aquifer below.



Figure 20. Low-permeability clay beds in an agricultural area in the West Salt River Valley reduce the likelihood of contamination reaching deeper drinking-water supplies.

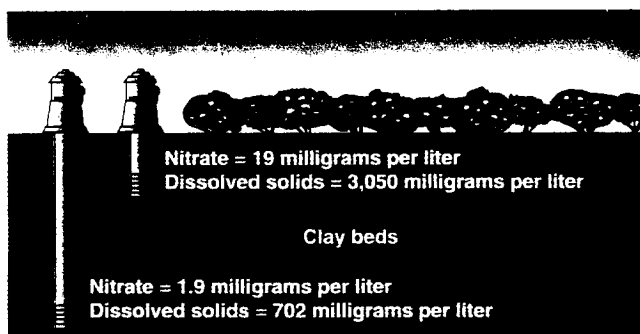


Figure 21. Ground water above the clay beds was recharged after 1953 and has been affected by agricultural activities.

Analyses of the tritium from ground water in this area indicated that water above the clay beds generally had been recharged after 1953, and water below the clay beds generally had been recharged before 1953 (see information about age dating ground water on p. 18). Although the clay beds currently reduce the likelihood that irrigation seepage will contaminate the ground water below, future large-scale withdrawals of ground water from below the clay beds could possibly result in the movement of shallow, poor quality water through the clay beds and into the domestic ground-water supply.

Occurrence and distribution of pesticides in ground water in the CAZB reflect both agricultural and urban land uses. Ten pesticides were detected in shallow ground water from the agricultural land-use study area in the West Salt River Valley, west of Phoenix (fig. 22). In other parts of the West Salt River Valley, consisting of agricultural, urban, and rangeland areas, eight pesticides were detected in ground water. Five pesticides were detected in ground water from the Upper Santa Cruz Basin, where there is a mixture of land-use types, but 60 percent of the basin is undeveloped rangeland (Coes and others, 2000). In the Sierra Vista subbasin, where urban and agricultural land uses are minimal (3.3 percent of basin; Coes and others, 1999) and have been minimal in the past, no pesticides were detected in ground-water samples. During 1996–98, the largest quantities of pesticides used among the three basins were for agriculture in the West Salt River Valley (Ken Agnew, University of Arizona, Pesticide Information and Training Office, written commun., 1999).

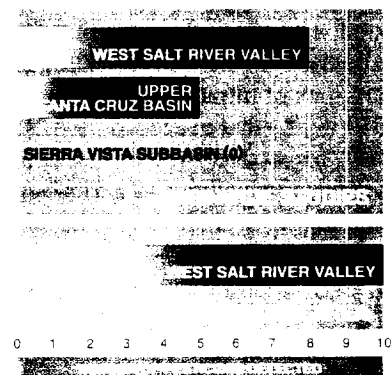


Figure 22. The largest number of pesticides was detected in an agricultural area in the West Salt River Valley.

Most of the pesticides detected in ground water in the CAZB were herbicides used to control unwanted plants in urban and agricultural areas (fig. 23). Herbicide use in urban areas is indicated by detections of simazine and prometon in the West Salt River Valley and prometon and 2,4-D in the Upper Santa Cruz Basin. These herbicides are used primarily in nonagricultural areas (U.S. Geological Survey, 1999). Detections of atrazine and deethylatrazine (a breakdown product of atrazine) in the West Salt River Valley and the Upper Santa Cruz Basin are an indication that herbicides used in areas of present and historical agriculture are affecting ground-water quality. Atrazine is one of the most heavily used herbicides in agricultural areas in the United States (U.S. Geological Survey, 1999).

Concentrations of pesticides in ground water did not exceed drinking-water standards or guidelines. Although deethylatrazine, simazine, prometon, DDE, atrazine, and diuron were detected in more than 30 percent of the ground-water samples from the agricultural land-use study area of the West Salt River Valley, none of the concentrations exceeded drinking-water standards or guidelines. Similarly, pesticides detected in ground water from the basinwide sampling in the West Salt River Valley during 1996–98 did not exceed drinking-water standards or guidelines.

DDE was detected in 10 (56 percent) of the shallow ground-water samples from the agricultural land-use study area in the West Salt River Valley. Detections of DDE in this area are the result of the persistence of this insecticide breakdown product in the environment and the physical characteristics of the ground-water system in

the area. In particular, the shallow depth to ground water in the agricultural land-use study area means that irrigation seepage and recharge, containing pesticides and their breakdown products, do not have to travel far to contaminate the ground water. Clay layers impede the movement of pesticides into the deeper aquifers in the area. The soils in the agricultural area have been identified as a source of DDE for the ground water (Brown, 1993). The only detection of DDE in the West Salt River Valley outside of the agricultural area was in a sample from the northern part of the Phoenix metropolitan area. DDE was not detected in samples from the Upper Santa Cruz Basin or the Sierra Vista subbasin. The large depths to ground water and small amounts of DDT used in most of the West Salt River Valley, the Upper Santa Cruz Basin, and the Sierra Vista subbasin limit the potential for introduction of DDE to the ground water.

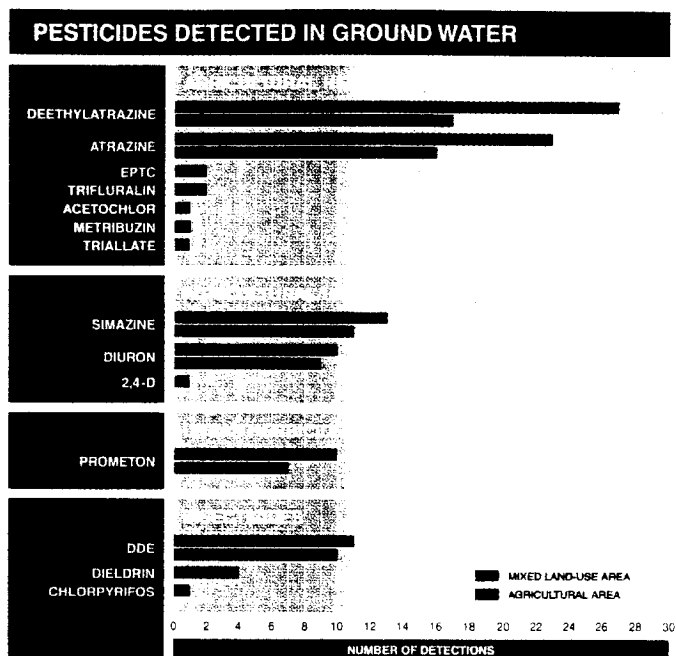


Figure 23. Most of the pesticides detected in ground water in the CAZB were herbicides used in agricultural and urban environments.

Pesticides and Human Health

Laboratory studies, mostly on animals, have shown that pesticides can cause health problems such as birth defects, nerve damage, cancer, and disruption of the endocrine system (USEPA, rev. June 12, 2000). The health effects on humans are not adequately understood, particularly when estimating the risks of exposure to mixtures of pesticides in water (U.S. Geological Survey, 1999). The USEPA determines risk on the basis of toxicity and exposure to a single pesticide (U.S. Environmental Protection Agency, rev. November 17, 1999), whereas NAWQA studies in the CAZB (see p. 24) and nationwide have shown that most contamination in water occurs as pesticide mixtures (U.S. Geological Survey, 1999). The effects of exposure to low concentrations of pesticide mixtures in drinking water is not known and will require further study to determine if standards or guidelines can be developed.

Detections of multiple pesticides indicate the complexity of contamination from land-surface activities. No standards or guidelines currently exist for mixtures of pesticides in drinking water because their effect on human health is not known (U.S. Geological Survey, 1999). All 9 wells in the agricultural land-use study area had 3 or more pesticides detected, whereas only 3 of the 35 wells sampled basinwide in the West Salt River Valley had 3 or more pesticides detected, and none of the wells in the Upper Santa Cruz Basin had 3 or more pesticides detected (fig. 24). No pesticides were detected in the Sierra Vista subbasin.

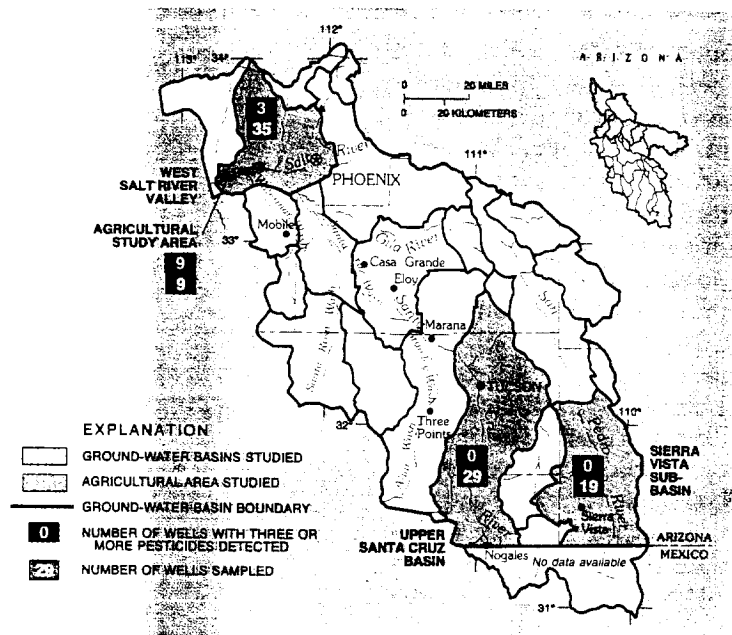


Figure 24. Multiple pesticides were detected in all nine monitoring wells in the agricultural land-use study area of the West Salt River Valley.

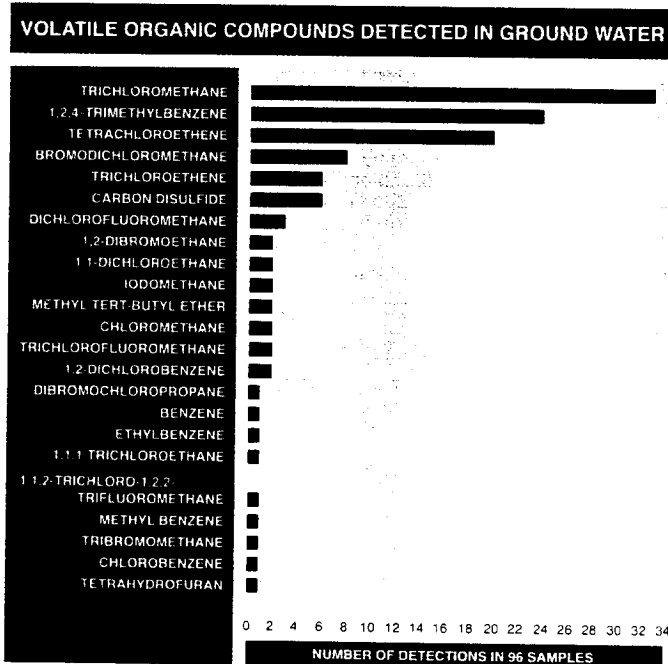


Figure 25. The VOCs trichloromethane, 1,2,4-trimethylbenzene, and tetrachloroethene were detected most frequently in ground water.

Volatile organic compounds (VOCs), including gasoline compounds, solvents, and refrigerants, have been identified as a major concern for ground-water contamination in Arizona (Marsh, 1994). Leaking underground storage tanks and disposal of solvents have been linked to most of the documented cases of ground-water contamination by VOCs. Electronic- and aerospace-manufacturing facilities use solvents for degreasing and are known to be sources of some of the largest VOC contamination problems in Arizona. Disposal of solvents from these types of facilities has occurred since the 1950s (Marsh, 1994). Dry-cleaning facilities also have been identified as sources of recent ground-water contamination by VOCs. Some municipal supply wells in the urban areas of Phoenix and Tucson are no longer used because of contamination by VOCs (Marsh, 1994).

VOCs were detected in ground water from all three basins sampled during 1996-98 (fig. 25). Of the 96 samples collected, 33 (34 percent) contained trichloromethane, 24 (24 percent) contained 1,2,4-

trimethylbenzene, and 20 (21 percent) contained tetrachloroethene (otherwise known as perchloroethylene, PCE, a solvent commonly used in dry cleaning). Only two VOC detections exceeded drinking-water regulations—PCE (5.48 µg/L) in the Upper Santa Cruz Basin and 1,2-dibromoethane (0.080 µg/L) in shallow ground water in the agricultural area of the West Salt River Valley.

Shallow ground water from the nine wells in the agricultural land-use study area had the largest number of VOC detections (35). Ground water from the other 35 wells in the West Salt River Valley had 32 detections. The Upper Santa Cruz Basin (18) and the

Sierra Vista subbasin (13) had fewer detections. The larger area of urban land use in the West Salt River Valley appears to be the reason for the greater number of detections there than in the other basins sampled.

Three wells that had five or more VOCs detected in ground water were located in the metropolitan area of Phoenix in the West Salt River Valley. The VOCs detected in these wells were either refrigerants, solvents and chemicals used to make solvents, or gasoline additives. These detections are typical of detections found in small-capacity wells in the metropolitan Phoenix area (Marsh, 1994). Combinations of solvents and gasoline additives are often detected in

ground water because their use is widespread, not necessarily because they are from the same source (Squillace and others, 1999).

Detections of VOCs in ground water in the relatively undeveloped Sierra Vista subbasin indicate that ground water in localized areas of the subbasin may be affected by human activities.

These detections are not widespread; therefore, the effects of human activity on present-day ground-water quality are not considered significant for the entire subbasin. These detections are an “early warning” of what could occur in the future in a basin that is presently considered minimally affected by urban activities.



Trichloromethane was the most commonly detected VOC in the Nation and in the CAZB Study Unit

Trichloromethane (chloroform), tetrachloroethene (PCE), and 1,2,4-trimethylbenzene were three of the five most commonly detected VOCs in the Nation and in the CAZB when concentrations above an assessment level of 0.1 µg/L were considered. The national data collected by the NAWQA Program during 1996–99 represent ambient ground water for all land-use types. Trichloromethane and PCE have been shown to cause cancer in laboratory animals from long-term exposure at concentrations greater than USEPA MCLs.

Trichloromethane is a by-product created during the use of chlorine to disinfect water, a solvent, and a degradation product of carbon tetrachloride. It can enter ground water from lawn irrigation, leaking sewers and water mains, and spills or improper disposal at industrial sites. The use of treated effluent from sewage-treatment plants for irrigation also provides a way for trichloromethane to reach the ground water in the CAZB, specifically in the agricultural land-use study area in the West Salt River Valley.

PCE is a solvent used primarily for degreasing and at dry-cleaning facilities. 1,2,4-trimethylbenzene is used to make trimellitic anhydride, dyes, and pharmaceuticals. Because there are many individual sources of these compounds in urban areas of the CAZB, it is difficult to identify the exact sources of ground-water contamination without site-specific studies, which were beyond the scope of the NAWQA sampling program.

Five most frequently detected volatile organic compounds in the CAZB and the Nation

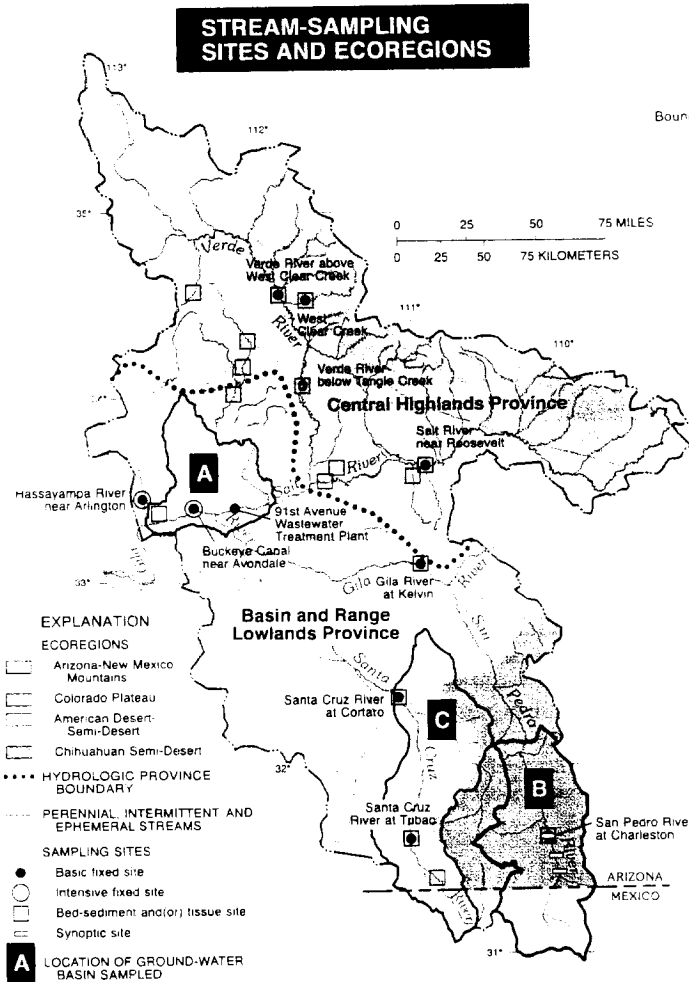
Central Arizona Basins Study Unit		Nation	
Compound name	Frequency of detection, in percent	Compound name	Frequency of detection, in percent
Trichloromethane	16	Trichloromethane	12
PCE	6	Toluene	4
1,2,4-Trimethylbenzene	5	Tetrachloroethene	4
1,2-Dibromoethane	4	Diethylsulfide	4
1,1,1-Trichloroethane	2	1,2,4-Trimethylbenzene	4

(Data include all land-use types; assessment level of 0.1 microgram per liter)

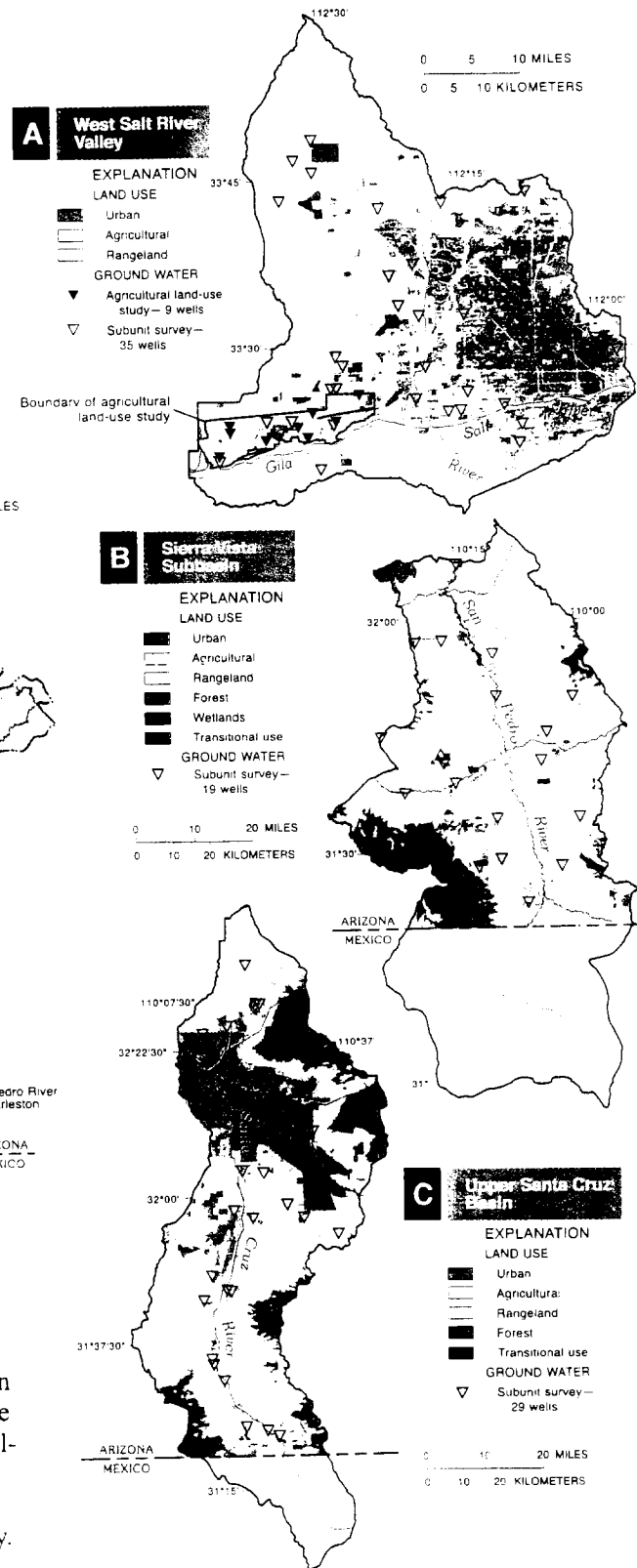
STUDY UNIT DESIGN

The Central Arizona Basins study was designed to provide nationally comparable water-quality data and address local and national questions about water quality. The primary goal of the study was to understand the human and natural factors that affect the chemistry of ground and surface water and communities of aquatic organisms.

Surface-water chemistry and biological-sampling sites were divided between the two main hydrologic



GROUND-WATER-QUALITY-SAMPLING SITES AND LAND USE



provinces in the study area: the Central Highlands and the Basin and Range Lowlands (see above). Ground-water samples were collected in three basins in the Basin and Range Lowlands. Human activities were least in the Sierra Vista subbasin, greatest in the West Salt River Valley, and intermediate in the Upper Santa Cruz Basin. Effects of agricultural activities on ground water were studied in the western part of the West Salt River Valley.

SUMMARY OF DATA COLLECTION IN THE CENTRAL ARIZONA BASINS, 1995-98

Study component	What data were collected and why	Types of sites sampled	Number of sites	Sampling frequency and period
Stream Chemistry				
Basic fixed sites, general water quality	Streamflow, dissolved oxygen, pH, alkalinity, specific conductance, temperature, nutrients, major ions, organic carbon, and suspended sediment were measured to determine occurrence and distribution.	Streams selected to represent urban, mixed agricultural/urban, and forest/rangeland land uses were distributed throughout the study area. Basins ranged from 0 miles (at point sources) to 18,011 square miles.	9	Monthly plus high flows Oct. 1995-Apr. 1998
Intensive fixed sites	Above constituents plus 87 pesticides and 85 volatile organic compounds.	Sites selected closer to urban and (or) agricultural areas so as to be more likely to reflect those land uses.	2	Monthly Jan. 1996-Dec. 1996, increased sampling frequency to approximately twice a month Dec. 1996-Feb. 1998
Fixed sites, dissolved organic carbon	Spectral characteristics of dissolved organic carbon from surface water were measured to determine sources.	Same sites as basic fixed sites and intensive fixed sites.	11	Monthly Jan. 1996-August 1997
Synoptic	Same as basic fixed sites, plus pesticides.	Three locations collocated with key sites for stream ecology synoptic.	3	Quarterly Jan. 97-Oct. 97
Contaminants in bed sediment	Trace elements and (or) organic compounds to determine occurrence and distribution in streambed sediments.	Depositional zones of most basic and intensive stream-chemistry sites plus additional sites.	17	Once May and June 1996
Contaminants in tissues of aquatic biota	Trace elements and (or) organic compounds to determine occurrence and distribution in tissues of fish, clams, and crayfish.	Same sites as sediment samples.	15	Once May and June 1996
Stream Ecology				
Basic sites	Communities of algae, invertebrates, and fish; and instream and riparian habitats surveyed to assess biological conditions of the study area.	Sites collocated with most basic and intensive stream-chemistry sites.	7	Once Oct. 1995-Jan. 1996
			2	Annually 1995-1997
Synoptic	Communities of algae, invertebrates, and fish; and instream and riparian habitats surveyed to evaluate spatial variability.	Nine reaches along one segment of a stream with minimal anthropogenic influences.	9	Once Oct.-Dec. 1996
Ground-Water Chemistry				
Study Unit West Salt River Valley - mixed land use	Nutrients, major ions, trace elements, volatile organic compounds, radon, dissolved organic carbon, and pesticides to assess water quality of the basin's aquifers.	Existing domestic, public-supply, irrigation, livestock, and industrial wells.	35	Once 1996-1997
Study Unit Upper Santa Cruz Basin - mixed land use	Nutrients, major ions, trace elements, volatile organic compounds, radon, dissolved organic carbon, and pesticides to assess water quality of the basin's aquifers.	Existing domestic, public-supply, irrigation, livestock, and industrial wells.	29	Once 1998
Study Unit Sierra Vista subbasin - mixed land use	Nutrients, major ions, trace elements, volatile organic compounds, radon, dissolved organic carbon, and pesticides to assess water quality of the basin's aquifers.	Existing domestic, public-supply, irrigation, and livestock wells.	19	Once 1996
Land use West Salt River Valley - agricultural	Nutrients, major ions, trace elements, volatile organic compounds, radon, dissolved organic carbon, and pesticides to determine effects of agricultural land use on shallow ground-water quality.	Shallow monitoring wells.	9	Twice Aug. 1997 Feb. 1998

GLOSSARY

- Anthropogenic**—A condition or occurrence that is the result of, or is influenced by, human activity.
- Aquatic-life criteria**—Water-quality guidelines for protection of aquatic life. Typically refers to U.S. Environmental Protection Agency water-quality criteria for protection of aquatic organisms.
- Aquifer**—A water-bearing layer of soil, sand, gravel, or rock that will yield usable quantities of water to a well.
- Background concentration**—A concentration of a substance in a particular environment that is indicative of minimal influence by human (anthropogenic) sources.
- Base flow**—Sustained, low flow in a stream; ground-water discharge is the source of base flow in most places.
- Basic fixed sites**—Sites on streams at which streamflow is measured and samples are collected for temperature, salinity, suspended sediment, major ions and metals, nutrients, and organic carbon to assess the broad-scale spatial and temporal character and transport of inorganic constituents of streamwater in relation to hydrologic conditions and environmental settings.
- Bed sediment**—The material that temporarily is stationary in the bottom of a stream or other watercourse.
- Bioaccumulation**—The biological sequestering of a substance at a higher concentration than that at which it occurs in the surrounding environment or medium. Also, the process whereby a substance enters organisms through the gills, epithelial tissues, or dietary or other sources.
- Biomass**—The amount of living matter, in the form of organisms, present in a particular habitat, usually expressed as weight per unit area.
- Breakdown product**—A compound derived by chemical, biological, or physical action upon a pesticide. The breakdown is a natural process that may result in a more toxic or a less toxic compound and a more persistent or less persistent compound.
- Concentration**—The amount or mass of a substance present in a given volume or mass of sample. Usually expressed as milligrams per liter or micrograms per liter (water sample) or micrograms per kilogram (sediment or tissue sample).
- Confining layer**—A layer of sediment or lithologic unit of low permeability that bounds an aquifer.
- Cubic foot per second (ft³/s or cfs)**—Rate of water discharge representing a volume of 1 cubic foot passing a given point during 1 second, equivalent to approximately 7.48 gallons per second or 448.8 gallons per minute or 0.02832 cubic meter per second.
- Dissolved solids**—Amount of minerals, such as salt, that are dissolved in water; amount of dissolved solids is an indicator of salinity or hardness.
- Drainage basin**—The portion of the surface of the Earth that contributes water to a stream through overland runoff, including tributaries and impoundments.
- Drinking-water guideline**—Nonenforceable Federal guideline regarding cosmetic (tooth or skin discoloration) or aesthetic effects (such as taste, color, odor).
- Drinking-water standard**—A threshold concentration in a public drinking-water supply, designed to protect human health or as defined here, standards are U.S. Environmental Protection Agency regulations that specify the maximum contamination levels for public water systems required to protect the public welfare.
- Ecoregion**—An area of similar climate, landform, soil, potential natural vegetation, hydrology, or other ecologically relevant variables.
- Effluent**—Outflow from a particular source, such as a stream that flows from a lake or liquid waste that flows from a factory or sewage-treatment plant.
- Ephemeral stream**—A stream or part of a stream that flows only in direct response to precipitation or snowmelt. Its channel is above the water table at all times.
- Eutrophication**—The process by which water becomes enriched with plant nutrients, most commonly phosphorus and nitrogen.
- Evaporite minerals (deposits)**—Minerals or deposits of minerals formed by evaporation of water containing salts. These deposits are common in arid climates.
- Evapotranspiration**—A collective term that includes water lost through evaporation from the soil and surface-water bodies and by plant transpiration.
- Infiltration**—Movement of water, typically downward, into soil or porous rock.
- Intensive fixed sites**—Basic Fixed Sites with increased sampling frequency during selected seasonal periods and analysis of dissolved pesticides for 1 year. Most NAWQA Study Units have one to two integrator Intensive Fixed Sites and one to four indicator Intensive Fixed Sites.
- Intermittent stream**—A stream that flows only when it receives water from rainfall runoff or springs, or from some surface source such as melting snow.
- Invertebrate**—An animal having no backbone or spinal column.
- Irrigation return flow**—The part of irrigation applied to the surface that is not consumed by evapotranspiration or uptake by plants and that migrates to an aquifer or surface-water body.
- Land subsidence**—Compression of soft aquifer materials in a confined aquifer due to pumping of water from the aquifer.
- Leaching**—The removal of materials in solution from soil or rock to ground water; refers to movement of pesticides or nutrients from land surface to ground water.
- Load**—General term that refers to a material or constituent in solution, in suspension, or in transport; usually expressed in terms of mass or volume.

LOWESS smooth—LOcally WEighted Scatterplot

Smoothing is a statistical method of defining a smooth curve through the middle of a scatterplot to highlight trends or patterns in the data.

Maximum Contaminant Level (MCL)—Maximum permissible level of a contaminant in water that is delivered to any user of a public water system. MCLs are enforceable standards established by the U.S. Environmental Protection Agency.

Median—The middle or central value in a distribution of data ranked in order of magnitude. The median is also known as the 50th percentile.

Metabolite—A substance produced in or by biological processes.

Micrograms per liter ($\mu\text{g/L}$)—A unit expressing the concentration of constituents in solution as weight (micrograms) of solute per unit volume (liter) of water; equivalent to one part per billion in most stream water and ground water. One thousand micrograms per liter equals 1 mg/L.

Milligrams per liter (mg/L)—A unit expressing the concentration of chemical constituents in solution as mass (milligrams) of solute per unit volume (liter) of water; equivalent to one part per million in most stream water and ground water. One thousand micrograms per liter equals 1 mg/L.

Nutrient—Element or compound essential for animal and plant growth. Common nutrients in fertilizer include nitrogen, phosphorus, and potassium.

Organochlorine insecticide—A class of organic insecticides containing a high percentage of chlorine. Includes dichlorodiphenylethanes (such as DDT), chlorinated cyclodienes (such as chlordane), and chlorinated benzenes (such as lindane). Most organochlorine insecticides were banned because of their carcinogenicity, tendency to bioaccumulate, and toxicity to wildlife.

Perennial stream—A stream that normally has water in its channel at all times.

Pesticide—A chemical applied to crops, rights of way, lawns, or residences to control weeds, insects, fungi, nematodes, rodents or other "pests."

Picocurie (pCi)—One trillionth (10^{-12}) of the amount of radioactivity represented by a curie (Ci). A curie is the quantity of any radioactive nuclide in which the number of disintegrations is 3.7×10^{10} per second (dps). A picocurie yields 2.22 disintegrations per minute (dpm) or 0.037 dps.

Public-supply withdrawals—Water withdrawn by public and private water suppliers for use within a general community. Water is used for a variety of purposes such as domestic, commercial, industrial, and public water use.

Recharge—Water that infiltrates the ground and reaches the saturated zone.

Riparian—Areas adjacent to rivers and streams with a high density, diversity, and productivity of plant and animal species relative to nearby uplands.

Runoff—Excess rainwater or snowmelt that is transported to streams by overland flow, tile drains, or ground water.

Secondary maximum contaminant level (SMCL)—The maximum contamination level in public water systems that, in the judgment of the U.S. Environmental Protection Agency (USEPA), is acceptable to protect the public welfare. SMCLs are secondary (nonenforceable) drinking water regulations established by the USEPA for contaminants that may adversely affect the odor or appearance of such water.

Specific conductance—A measure of the ability of a liquid to conduct an electrical current.

Tolerant species—Those species that are adaptable to (tolerant of) human alterations to the environment and often increase in number when human alterations occur.

Trace element—An element typically found in only minor amounts (concentrations less than 1.0 milligram per liter) in water; includes arsenic, cadmium, chromium, copper, lead, mercury, nickel, and zinc.

Volatile organic compounds (VOCs)—Organic chemicals that have a high vapor pressure relative to their water solubility. VOCs include components of gasoline, fuel oils, and lubricants, as well as organic solvents, fumigants, some inert ingredients in pesticides, and some by-products of chlorine disinfection.

REFERENCES

- Anderson, T.W., Freethey, G.W., and Tucci, Patrick, 1992, Geohydrology and water resources of alluvial basins in south-central Arizona and parts of adjacent states: U.S. Geological Survey Professional Paper 1406-B, 67 p.
- Anning, D.W., 1998, Sources of nitrogen and phosphorus in drainage basins of central Arizona, *in* water at the confluence of science, law, and public policy: Proceedings of the Arizona Hydrological Society, Eleventh Annual Symposium, Sept. 24-26, 1998, Tucson, Arizona, p. 8.
- Arizona Department of Economic Security, rev. July 7, 2000, Latest population projections—State and counties: accessed July 13, 2000, at URL <http://www.de.state.us/links/economic/webpage/page2.html>
- Arizona Department of Water Resources, 1994, Arizona water resources assessment, volume 1, Inventory and analysis: Phoenix, Arizona, 253 p.
- Brown, S.L., 1993, Organochlorine pesticide levels associated with historic agricultural areas within the middle Gila River basin: Tempe, Arizona, Arizona State University, Master's thesis, 48 p.
- Brown, J.G. and Favor, Barbara, 1996, Hydrology and geochemistry of aquifer and stream contamination related to acidic water in Pinal Creek Basin near Globe, Arizona: U.S. Geological Survey Water-Supply Paper 2466, 103 p.
- Bouwer, Herman, 1990, Agricultural chemicals and ground-water quality: *Journal of Soil and Water Conservation*, v. 45, no. 2, p. 184-189.
- Canadian Council of Ministers of the Environment, 1999, Canadian sediment quality guidelines for the protection of aquatic life — Summary tables, *in* Canadian environmental quality guidelines: Winnipeg, Canada.
- Cordy, G.E., and Bouwer, Herman, 1999, Where do the salts go? The potential effects and management of salt accumulation in south-central Arizona: U.S. Geological Survey Fact Sheet 170-98, 4 p.
- Cordy, G.E., Rees, J.A., Edmonds, R.J., Gebler, J.B., Wirt, Laurie, Gellenbeck, D.J., and Anning, D.W., 1998, Water-quality assessment of the Central Arizona basins—Environmental setting and overview of water quality: U.S. Geological Survey Water-Resources Investigations Report 98-4097, 72 p.
- Coes, A.L., Gellenbeck, D.J., and Towne, D.C., 1999, Ground-water quality in the Sierra Vista subbasin, Arizona, 1996-97: U.S. Geological Survey Water-Resources Investigations Report 99-4056, 50 p.
- Coes, A.L., Gellenbeck, D.J., Towne, D.C., and Freark, M.C., 2000, Ground-water quality in the Upper Santa Cruz Basin, Arizona, 1998: U.S. Geological Survey Water-Resources Investigations Report 00-4117, 55 p.
- Faber, R., and Hickey, J., 1973, Eggshell thinning, chlorinated hydrocarbons, and mercury in inland aquatic bird eggs, 1969 and 1970: *Pesticide Monitoring Journal*, v. 7, p. 27-36.
- Feth, J.H., and Hem, J.D., 1963, Reconnaissance of head-water springs in the Gila River drainage basin, Arizona: U.S. Geological Survey Water-Supply Paper 1619-H, 54 p.
- Gebler, J.B., 1998, Water-quality of selected effluent-dependent stream reaches in southern Arizona as indicated by concentrations of periphytic chlorophyll *a* and aquatic-invertebrate communities: U.S. Geological Survey Water-Resources Investigations Report 98-4199, 12 p.
- _____, 2000, Organochlorine compounds in streambed sediment and in biological tissue from streams and their relations to land use, central Arizona: U.S. Geological Survey Water-Resources Investigations Report 00-4041, 21 p.
- Gellenbeck, D.J., 1994, Isotopic composition and sources of nitrate in ground water from western Salt River Valley, Arizona: U.S. Geological Survey Water-Resources Investigations Report 94-4063, 53 p.
- Gellenbeck, D.J., and Coes, A.L., 1999, Ground-water quality in alluvial basins that have minimal urban development, south-central Arizona: U.S. Geological Survey Water-Resources Investigations Report 99-4056, 27 p.
- Gelt, Joe, Henderson, Jim, Seasholes, Kenneth, Tellman, Barbara, and Woodard, Gary, 1999, Water in the Tucson area—Seeking sustainability: University of Arizona Water Resources Research Center Issue Paper #20, 155 p.
- Gilliom, R.J., rev. March 19, 1999, Pesticides in the Nation's water resources—Initial National Water-Quality Assessment (NAWQA) Program Results: U.S. Geological Survey publication: accessed July 6, 2000, at URL <http://www.capp.er.usgs.gov/nawqa/pesticides.html>

- Hem, J.D., 1985, Study and interpretation of the chemical characteristics of natural water. U.S. Geological Survey Water-Supply Paper 2254, 263 p.
- Kalin, Robert M., 1994, Hydrogeochemical evolution of the groundwater of the Tucson basin with application to 3-dimensional groundwater flow modeling: Tucson, University of Arizona, Department of Geosciences, Ph.D. dissertation, 510 p.
- Likens, G.E., Bormann, F.H., Pierce, R.S., Eaton, J.S., and Johnson, N.M., 1977, Biogeochemistry of a forested ecosystem: New York, Springer-Verlag, 146 p.
- Marsh, Diana, ed., 1994, Arizona water-quality assessment 1994: Phoenix, Arizona Department of Environmental Quality Report EQR 94-3, 246 p.
- Marsh, Diana, 2000, The status of water quality in Arizona, Clean Water Act section 305(b) report 2000: Arizona Department of Environmental Quality Report EQR-00-03, [variously paged].
- Mueller, D.K., Hamilton, P.A., Helsel, D.R., Hitt, K.J., and Ruddy, B.C., 1996, Nutrients in ground water and surface water of the United States: NWQEP Notes, March 1996, no. 76, accessed October 18, 2000, at URL <http://www5.bae.ncsu.edu/programs/extension/wqg/issues/76.html#1>
- National Academy of Sciences, 1977, Drinking water and health: Washington, D.C., National Academy of Sciences. Safe Drinking Water Committee, 939 p.
- National Academy of Sciences and National Academy of Engineering, 1973, Water quality criteria, 1972: Washington, D.C., U.S. Government Printing Office, Ecological Research Series EPA-R3-73-033, 594 p.
- National Governor's Association, 1991, The health effects of nitrate in ground water: Ground Water Bulletin, v. 2, no. 3, p. 3.
- National Research Council, 1999, Arsenic in drinking water: Washington, D.C., National Academy Press, 273 p.
- Newell, A.J., Johnson, D.W., and Allen, L.K., 1987, Niagara River biota contamination project—Fish flesh criteria for piscivorous wildlife: Albany, New York State Department of Environmental Protection, Division of Fish and Wildlife, Bureau of Environmental Protection Technical Report 87-3, 182 p.
- Owen-Joyce, S.J., and Bell, C.K., 1983, Appraisal of water resources in the upper Verde River area, Yavapai and Coconino Counties, Arizona: Arizona Department of Water Resources Bulletin 2, 219 p.
- Robertson, F.N., 1991, Geochemistry of ground water in alluvial basins of Arizona and adjacent parts of Nevada, New Mexico, and California: U.S. Geological Survey Professional Paper 1406-C, 90 p.
- Salt River Project, 1997, Salt River Project 1996 annual water quality report: Phoenix, Arizona, Salt River Project, [variously paged].
- Squillace, P.J., Moran, M.J., Lapham, W.W., Price, C.W., Clawges, R.M., and Zogorski, J.S., 1999, Volatile organic compounds in untreated ambient groundwater of the United States, 1985-95: Environmental Science and Technology, v. 33, no. 23, p. 4167-4187.
- Swenson, H.A., and Baldwin, H.L., 1965, A primer on water quality: U.S. Geological Survey booklet, 27 p.
- Tadayon, S., Duet, N.R., Fisk, G.G., McCormack, H.F., Pope, G.L., and Rigas, P.D., 1998, Water resources data, Arizona, water year 1997: U.S. Geological Survey Water-Data Report AZ-97-1, 416 p.
- Tellman, Barbara, Yarde, Richard, and Wallace, Mary G., 1997, Arizona's changing rivers: Water Resources Research Center Issue Paper #19, 198 p.
- U.S. Environmental Protection Agency, 1986, Quality criteria for water—1986: U.S. Environmental Protection Agency report EPA 440/5-86-001 [variously paged].
- _____, rev. October 18, 1999, Proposed radon in drinking water rule: U.S. Environmental Protection Agency report EPA 815-F-99-006, accessed June 6, 2000, at URL <http://www.epa.gov/OGWDW/radon/fact.html>
- _____, rev. November 17, 1999, Assessing health risks from pesticides: U.S. Environmental Protection Agency Report 735-F-99-002, accessed October 11, 2000, at URL <http://www.epa.gov/opp00001/citizens/risk-assess.htm>
- _____, rev. April 21, 2000, Radionuclides—Notice of data availability—Summary: accessed June 6, 2000, at URL <http://www.epa.gov/fedrgstr/EPA-WATER/2000/April/Day-21/w9654.htm>
- _____, rev. June 2, 2000, Proposed revision to arsenic drinking water standard: U.S. Environmental Protection Agency

report EPA 815-F-00-012. accessed June 6, 2000, at URL <http://www.epa.gov/safewater/ars/proposalfs.html>

_____. rev. June 12, 2000, Pesticides and food, health problems pesticides may pose: U.S. Environmental Protection Agency, accessed October 11, 2000, at URL <http://www.epa.gov/pesticides/food/risks.html>

_____. rev. August 25, 2000, Drinking water priority rulemaking—Arsenic, accessed October 5, 2000, at URL <http://www.epa.gov/OGWDW/ars/arsenic.htm>

U.S. Geological Survey, 1969, Annual report on ground water in Arizona, spring 1967 to spring 1969: Arizona State Land Department Water Resources Report 38, 54 p.

_____. 1999, The quality of our Nation's waters—Nutrients and pesticides: U.S. Geological Survey Circular 1225, 82 p.

Welch, A.H., Watkins, S.A., Helsel, D.R., and Focazio, M.J., 2000, Arsenic in ground-water resources of the United States: U.S. Geological Survey Fact Sheet 063-00,

accessed June 6, 2000, at URL <http://co.water.usgs.gov/trace/pubs/fs-063-00/index.html>

Additional References

Beaulieu, K.M., Capesius, J.P., and Gebler, J.B., 2000, Physical-habitat and geomorphic data for selected river reaches in central Arizona basins, 1995–98: U.S. Geological Survey Open-File Report 00-90, 197 p.

Coes, A.L., and Gellenbeck, D.J., 1997, Collection and analysis of ground-water samples in the Sierra Vista basin, Arizona, 1996: U.S. Geological Survey Fact Sheet 107-97, 1 sheet.

Cordy, G.E., 1994, National water-quality assessment program—Central Arizona Basins: U.S. Geological Survey Fact Sheet 94-016, 1 sheet.

Gilliom, R.J., Alley, W.M., and Gurtz, M.E., 1995, Design of the National Water-Quality Assessment Program: Occurrence and distribution of water-quality conditions: U.S. Geological Survey Circular 1112, 33 p.

APPENDIX—WATER-QUALITY DATA FROM THE CENTRAL ARIZONA BASINS IN A NATIONAL CONTEXT

For a complete view of Central Arizona Basins data and for additional information about specific benchmarks used, visit our Web site at <http://water.usgs.gov/nawqa/>. Also visit the NAWQA Data Warehouse for access to NAWQA data sets at <http://infotrek.er.usgs.gov/wdbctx/nawqa/nawqa.home>.

This appendix is a summary of chemical concentrations and biological indicators assessed in the Central Arizona Basins. Selected results for this Study Unit are graphically compared to results from as many as 36 NAWQA Study Units investigated from 1991 to 1998 and to national water-quality benchmarks for human health, aquatic life, or fish-eating wildlife. The chemical and biological indicators shown were selected on the basis of frequent detection, detection at concentrations above a national benchmark, or regulatory or scientific importance. The graphs illustrate how conditions associated with each land use sampled in the Central Arizona Basins compare to results from across the Nation, and how conditions compare among the several land uses. Graphs for chemicals show only detected concentrations and, thus, care must be taken to evaluate detection frequencies in addition to concentrations when comparing study-unit and national results. For example, trifluralin concentrations in Central Arizona Basins agricultural streams were similar to the national distribution, but the detection frequency was much higher (76 percent compared to 21 percent).

CHEMICALS IN WATER

Concentrations and detection frequencies, Central Arizona Basins, 1995–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals

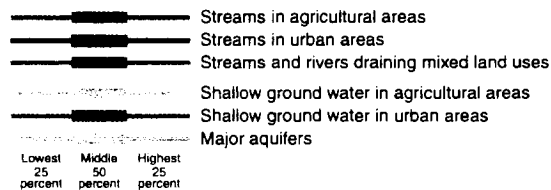
◆ Detected concentration in Study Unit

66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency

--- Not measured or sample size less than two

12 Study-unit sample size. For ground water, the number of samples is equal to the number of wells sampled

National ranges of detected concentrations, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

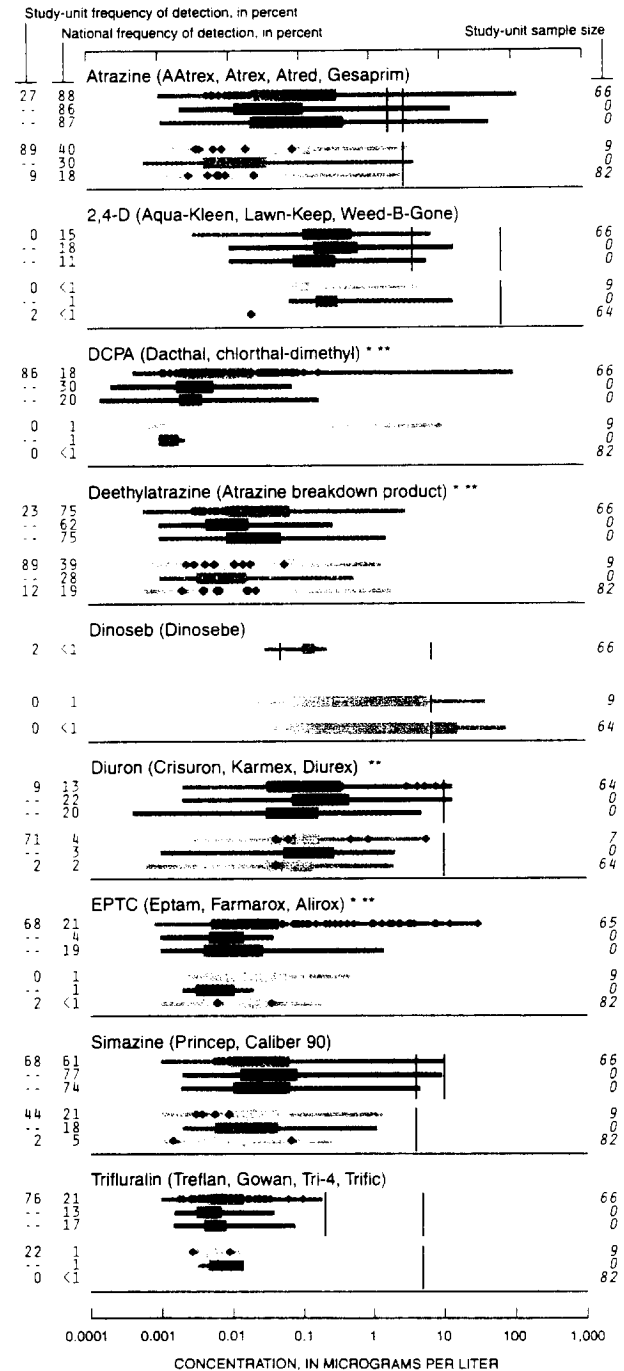


National water-quality benchmarks

National benchmarks include standards and guidelines related to drinking-water quality, criteria for protecting the health of aquatic life, and a goal for preventing stream eutrophication due to phosphorus. Sources include the U.S. Environmental Protection Agency and the Canadian Council of Ministers of the Environment

- | Drinking-water quality (applies to ground water and surface water)
- | Protection of aquatic life (applies to surface water only)
- | Prevention of eutrophication in streams not flowing directly into lakes or impoundments
- No benchmark for drinking-water quality
- No benchmark for protection of aquatic life

Pesticides in water—Herbicides



Other herbicides detected

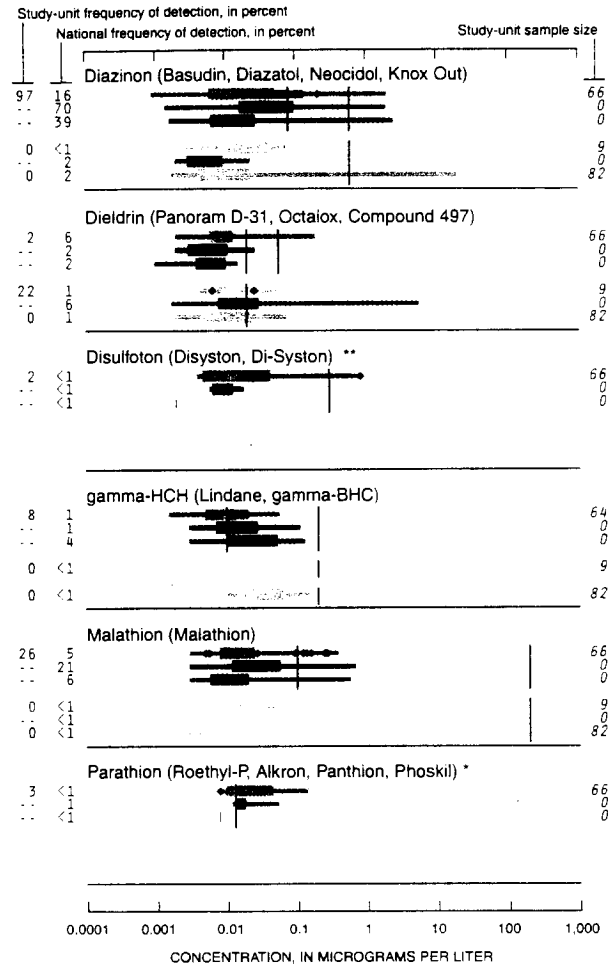
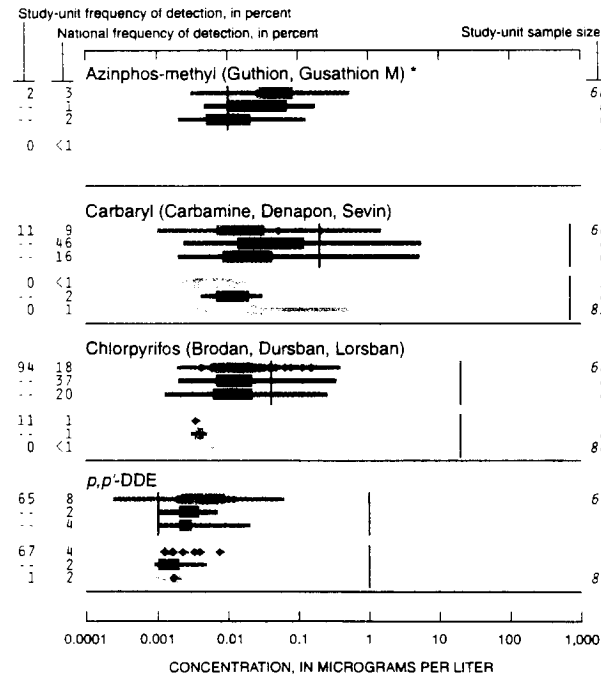
- Acetochlor (Harness Plus, Surpass) * **
- Benfluralin (Balan, Benefin, Bonalan) * **
- Cyanazine (Bladex, Fortrol)
- Linuron (Lorox, Linex, Sarclex, Linurex, Afalon) *
- Metribuzin (Lexone, Sencor)

Molinate (Ordram) * **
 Pendimethalin (Pre-M, Prowl, Stomp) * **
 Prometon (Pramitol, Princep) **
 Pronamide (Kerb, Propyzamid) **
 Propachlor (Ramrod, Satecid) **
 Tebuthiuron (Spike, Tebusan)
 Triallate (Far-Go, Avadex BW, Tri-alleate) *

Herbicides not detected

Acifluorfen (Blazer, Tackle 2S) **
 Alachlor (Lasso, Bronco, Lariat, Bullet) **
 Bentazon (Basagran, Bentazone) **
 Bromacil (Hyvar X, Urox B, Bromax)
 Bromoxynil (Buctril, Brominal) *
 Butylate (Sutan +, Genate Plus, Butilate) **
 Chloramben (Amiben, Amilon-WP, Vegiben) **
 Clopyralid (Stinger, Lontrel, Transline) * **
 2,4-DB (Butyrac, Butoxone, Embutox Plus, Embutone) * **
 Dacthal mono-acid (Dacthal breakdown product) * **
 Dicamba (Banvel, Dianat, Scotts Proturf)
 Dichlorprop (2,4-DP, Seritox 50, Lentemul) * **
 2,6-Diethylaniline (Alachlor breakdown product) * **
 Ethalfuralin (Sonalan, Curbit) * **
 Fenuron (Fenulon, Fenidim) * **
 Fluometuron (Flo-Met, Cotoran) **
 MCPA (Rhomene, Rhonox, Chiptox)
 MCPB (Thistrol) * **
 Metolachlor (Dual, Pennant)
 Napropamide (Devrinol) * **
 Neburon (Neburea, Neburyl, Noruben) * **
 Norflurazon (Evital, Predict, Solicam, Zorial) * **
 Oryzalin (Surflan, Dirimal) * **
 Pebulate (Tillam, PEBC) * **
 Picloram (Grazon, Tordon)
 Propanil (Stam, Stampede, Wham) * **
 Propham (Tuberite) **
 2,4,5-T **
 2,4,5-TP (Silvex, Fenoprop) **
 Terbacil (Sinbar) **
 Thiobencarb (Bolero, Saturn, Benthicarb) * **
 Triclopyr (Garlon, Grandstand, Redeem, Remedy) * **

Pesticides in water—Insecticides



Other insecticides detected

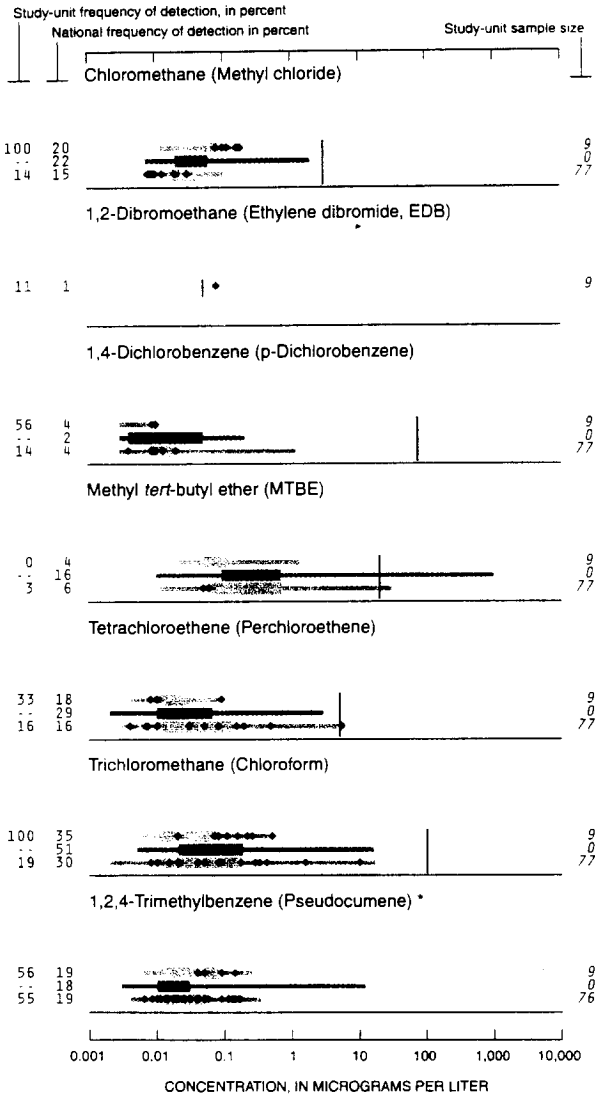
Carbofuran (Furadan, Curaterr, Yaltox)
 Methomyl (Lanox, Lannate, Acinate) **
 Methyl parathion (Penncap-M, Folidol-M) **
 Phorate (Thimet, Granutox, Geomet, Rampart) * **
 Propoxur (Baygon, Blattanex, Uden, Proprotax) * **

Insecticides not detected

Aldicarb (Temik, Ambush, Pounce)
 Aldicarb sulfone (Standak, aldoxycarb)
 Aldicarb sulfoxide (Aldicarb breakdown product)
 Ethoprop (Mocap, Ethoprophos) * **
 Fonofos (Dyfonate, Capfos, Cudgel, Tycap) **
 alpha-HCH (alpha-BHC, alpha-lindane) **
 3-Hydroxycarbofuran (Carbofuran breakdown product) * **
 Methiocarb (Slug-Geta, Grandslam, Mesurol) * **
 Oxamyi (Vydate L, Pratt) **
 cis-Permethrin (Ambush, Astro, Pounce) * **
 Propargite (Comite, Ornite, Ornamite) * **
 Terbufos (Conraven, Counter, Pilarfox) **

Volatile organic compounds (VOCs) in ground water

These graphs represent data from 16 Study Units, sampled from 1996 to 1998



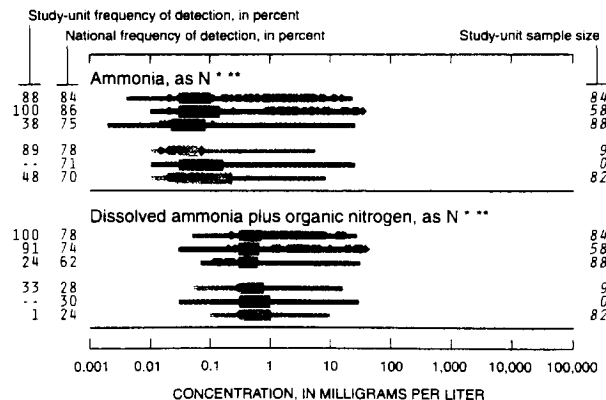
Other VOCs detected

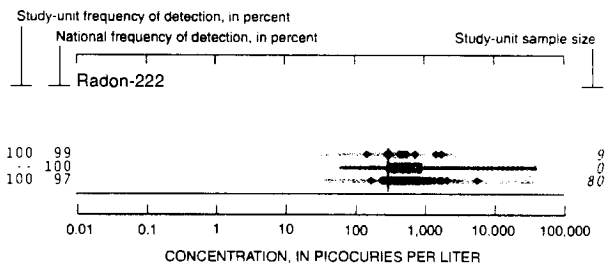
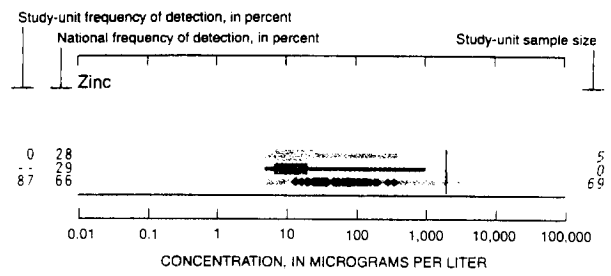
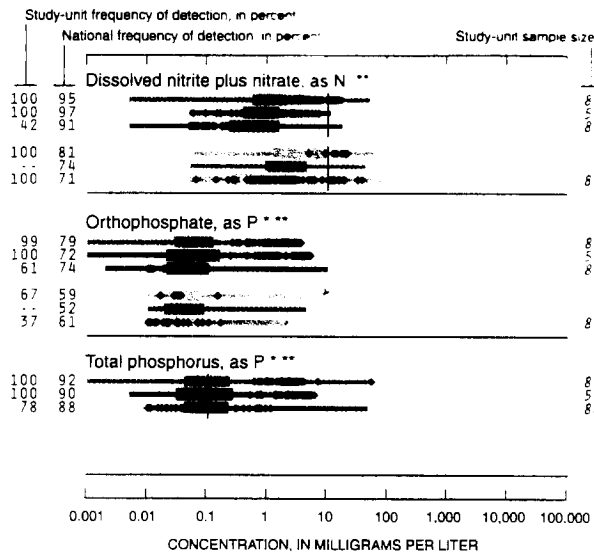
Benzene
 Bromodichloromethane (Dichlorobromomethane)
 Carbon disulfide *
 1-Chloro-2-methylbenzene (o-Chlorotoluene)
 Chlorobenzene (Monochlorobenzene)
 Dichlorodifluoromethane (CFC 12, Freon 12)
 1,1-Dichloroethane (Ethylidene dichloride) *
 1,1-Dichloroethene (Vinylidene chloride)
 Dichloromethane (Methylene chloride)
 1,2-Dimethylbenzene (o-Xylene)
 1,3 & 1,4-Dimethylbenzene (m-&p-Xylene)
 1-4-Epoxy butane (Tetrahydrofuran, Diethylene oxide) *
 Ethylbenzene (Phenylethane)
 Iodomethane (Methyl iodide) *
 Methylbenzene (Toluene)
 2-Propanone (Acetone) *
 Tribromomethane (Bromoform)
 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon 113) *
 1,1,1-Trichloroethane (Methylchloroform)
 Trichloroethene (TCE)
 Trichlorofluoromethane (CFC 11, Freon 11)

VOCs not detected

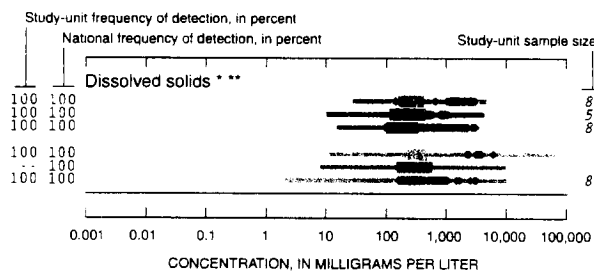
tert-Amylmethylether (tert-amyl methyl ether (TAME)) *
 Bromobenzene (Phenyl bromide) *
 Bromochloromethane (Methylene chlorobromide)
 Bromoethene (Vinyl bromide) *
 Bromomethane (Methyl bromide)
 2-Butanone (Methyl ethyl ketone (MEK)) *
 n-Butylbenzene (1-Phenylbutane) *
 sec-Butylbenzene *
 tert-Butylbenzene *
 3-Chloro-1-propene (3-Chloropropene) *
 1-Chloro-4-methylbenzene (p-Chlorotoluene)
 Chlorodibromomethane (Dibromochloromethane)
 Chloroethane (Ethyl chloride) *
 Chloroethene (Vinyl chloride)
 1,2-Dibromo-3-chloropropane (DBCP, Nemagon)
 Dibromomethane (Methylene dibromide) *
 trans-1,4-Dichloro-2-butene ((Z)-1,4-Dichloro-2-butene) *
 1,2-Dichlorobenzene (o-Dichlorobenzene)
 1,3-Dichlorobenzene (m-Dichlorobenzene)
 1,2-Dichloroethane (Ethylene dichloride)
 trans-1,2-Dichloroethene ((E)-1,2-Dichloroethene)
 cis-1,2-Dichloroethene ((Z)-1,2-Dichloroethene)
 1,2-Dichloropropane (Propylene dichloride)
 2,2-Dichloropropane *
 1,3-Dichloropropane (Trimethylene dichloride) *
 trans-1,3-Dichloropropane ((E)-1,3-Dichloropropane)
 cis-1,3-Dichloropropane ((Z)-1,3-Dichloropropane)
 1,1-Dichloropropane *
 Diethyl ether (Ethyl ether) *
 Diisopropyl ether (Diisopropylether (DIPE)) *
 Dimethylbenzenes (Xylenes (total))
 Ethenylbenzene (Styrene)
 Ethyl methacrylate *
 Ethyl tert-butyl ether (Ethyl-t-butyl ether (ETBE)) *
 1-Ethyl-2-methylbenzene (2-Ethyltoluene) *
 Hexachlorobutadiene
 1,1,1,2,2,2-Hexachloroethane (Hexachloroethane)
 2-Hexanone (Methyl butyl ketone (MBK)) *
 Isopropylbenzene (Cumene) *
 p-Isopropyltoluene (p-Cymene) *
 Methyl acrylonitrile *
 Methyl-2-methacrylate (Methyl methacrylate) *
 4-Methyl-2-pentanone (Methyl isobutyl ketone (MIBK)) *
 Methyl-2-propenoate (Methyl acrylate) *
 Naphthalene
 2-Propenenitrile (Acrylonitrile)
 n-Propylbenzene (Isocumene) *
 1,1,1,2-Tetrachloroethane *
 1,1,1,2-Tetrachloroethane
 Tetrachloromethane (Carbon tetrachloride)
 1,2,3,4-Tetramethylbenzene (Prehnitene) *
 1,2,3,5-Tetramethylbenzene (Isodurene) *
 1,2,4-Trichlorobenzene
 1,2,3-Trichlorobenzene *
 1,1,2-Trichloroethane (Vinyl trichloride)
 1,2,3-Trichloropropane (Allyl trichloride)
 1,2,3-Trimethylbenzene (Hemimellitene) *
 1,3,5-Trimethylbenzene (Mesitylene) *

Nutrients in water





Dissolved solids in water



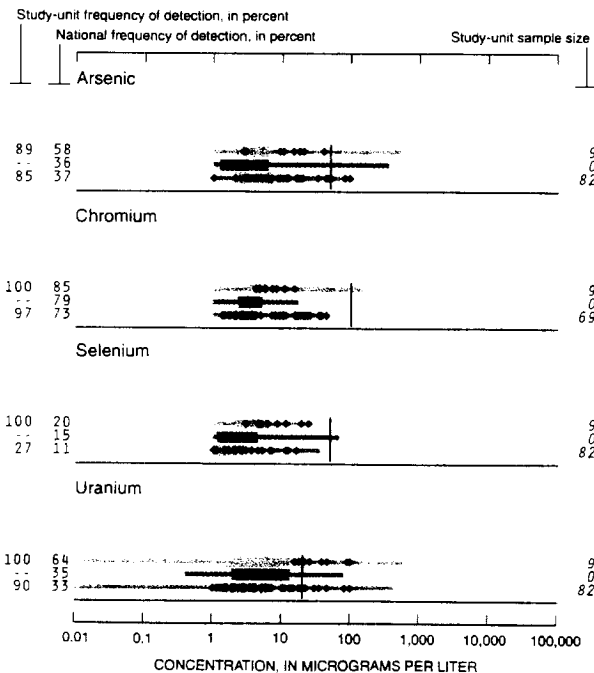
Other trace elements detected

Lead

Trace elements not detected

Cadmium

Trace elements in ground water

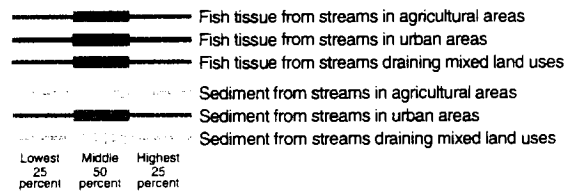


CHEMICALS IN FISH TISSUE AND BED SEDIMENT

Concentrations and detection frequencies, Central Arizona Basins, 1995–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals. Study-unit frequencies of detection are based on small sample sizes; the applicable sample size is specified in each graph

- ◆ Detected concentration in Study Unit
- 66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- Not measured or sample size less than two
- 12 Study-unit sample size

National ranges of concentrations detected, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

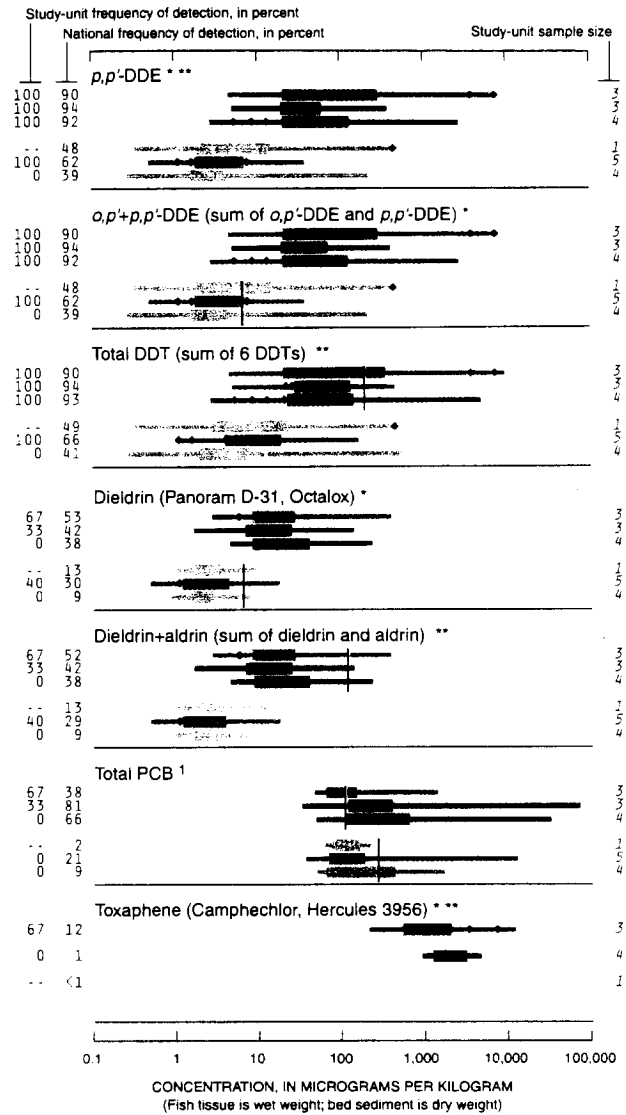
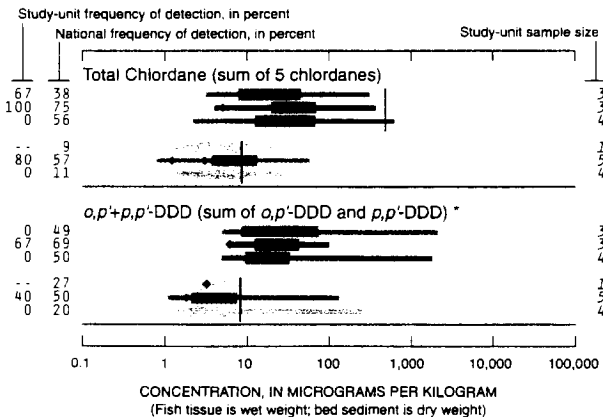


National benchmarks for fish tissue and bed sediment¹

National benchmarks include standards and guidelines related to criteria for protection of the health of fish-eating wildlife and aquatic organisms. Sources include the U.S. Environmental Protection Agency, other Federal and State agencies, and the Canadian Council of Ministers of the Environment

- | Protection of fish-eating wildlife (applies to fish tissue)
- | Protection of aquatic life (applies to bed sediment)
- No benchmark for protection of fish-eating wildlife
- ** No benchmark for protection of aquatic life

Organochlorines in fish tissue (whole body) and bed sediment



¹ The national detection frequencies for total PCB in sediment are biased low because about 30 percent of samples nationally had elevated detection levels compared to this Study Unit. See <http://water.usgs.gov/nawqa/> for additional information.

Other organochlorines detected

- Pentachloroanisole (PCA) ***
- cis*-Permethrin (Ambush, Astro, Pounce) **
- trans*-Permethrin (Ambush, Astro, Pounce) **

Organochlorines not detected

- Chloroneb (Chloronebe, Demosan) **
- DCPA (Dacthal, chlorthal-dimethyl) ***
- Endosulfan I (alpha-Endosulfan, Thiodan) **
- Endrin (Endrine)
- gamma-HCH (Lindane, gamma-BHC, Gammexane) *
- Total-HCH (sum of alpha-HCH, beta-HCH, gamma-HCH, and delta-HCH) **
- Heptachlor epoxide (Heptachlor breakdown product) *
- Heptachlor+heptachlor epoxide (sum of heptachlor and heptachlor epoxide) **
- Hexachlorobenzene (HCB) **
- Isodrin (Isodrine, Compound 711) **
- p,p'*-Methoxychlor (Mariate, methoxychlor) **
- α,p' -Methoxychlor ***
- Mirex (Dechlorane) **

BIOLOGICAL INDICATORS

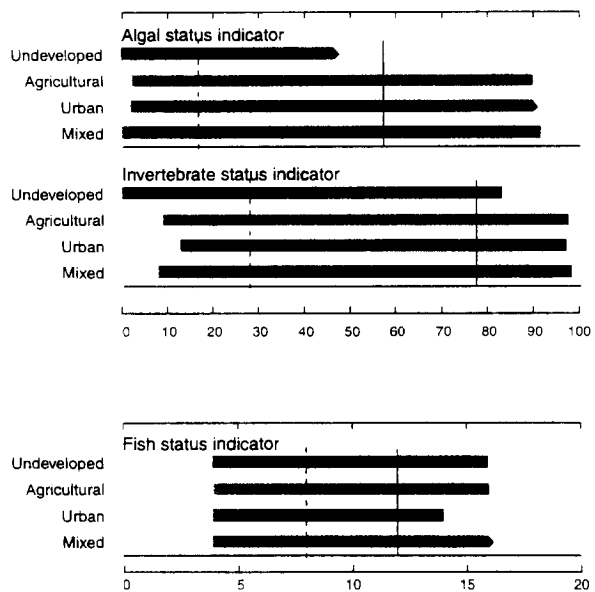
Higher national scores suggest habitat disturbance, water-quality degradation, or naturally harsh conditions. The status of algae, invertebrates (insects, worms, and clams), and fish provide a record of water-quality and stream conditions that water-chemistry indicators may not reveal. **Algal status** focuses on the changes in the percentage of certain algae in response to increasing siltation, and it often correlates with higher nutrient concentrations in some regions. **Invertebrate status** averages 11 metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. **Fish status** sums the scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that increase in association with water-quality degradation.

Biological indicator value, Central Arizona Basins, by land use, 1995–98

- ◆ Biological status assessed at a site

National ranges of biological indicators, in 16 NAWQA Study Units, 1994–98

- Streams in undeveloped areas
- ▨ Streams in agricultural areas
- ▩ Streams in urban areas
- ▧ Streams in mixed-land-use areas
- 75th percentile
- - - 25th percentile



A COORDINATED EFFORT

Coordination with agencies and organizations in the Central Arizona Basins Study Unit was integral to the success of this water-quality assessment. We thank those who served as members of our liaison committee.

Federal Agencies

Bureau of Indian Affairs
Bureau of Reclamation
Centro de Investigación y Estudios Ambientales
Comisión Nacional del Agua
International Boundary and Water Commission
National Park Service
Natural Resources Conservation Service
Salt River-Pima Indian Community
Tohono O'odham Nation
U.S. Environmental Protection Agency
U.S. Fish and Wildlife Service (USFWS)
U.S. Department of Agriculture, Forest Service
U.S. Department of Agriculture, Water Conservation Laboratory

State Agencies

Arizona Department of Environmental Quality (ADEQ)
Arizona Department of Water Resources (ADWR)
Arizona Game and Fish Department (AzGF)
Arizona Geological Survey

Local Agencies

City of Phoenix
City of Tucson
Maricopa County
Pima Association of Governments
Pima County
Southern Arizona Association of Governments

Universities

Arizona State University
University of Arizona

Other public and private organizations

Arizona Toxics Information
Friends of the Santa Cruz River
Salt River Project
Southern Arizona Water Resources Association
The Nature Conservancy

We thank the following individuals and organizations for contributing to this effort.

Laurie Wirt (USGS) designed and guided the surface-water-quality sampling program for the CAZB from 1994 to 1996.

Doug Towne and Maureen Freark (ADEQ) coordinated with CAZB to design cooperative ground-water studies in the Upper Santa Cruz Basin and the Sierra Vista subbasin.

Salt River Project, ADEQ, and ADWR provided valuable data for our study.

Buckeye Water Conservation and Drainage District, City of Phoenix, City of Peoria, Town of Buckeye, City of Goodyear, Roosevelt Irrigation District, and numerous individual landowners allowed us access to their wells and data.

Terry Short, Lisa H. Nowell, A.B. Richards, and Steve Goodbred provided invaluable assistance and guidance for the CAZB biological data collection and reports.

Patrice Spindler (ADEQ), Kirke King (USFW), Kirk Young (AzGF), W.L. Minckley, and Paul Marsh (ASU) provided information and expertise for the biological aspects of this project.

Karen Beaulieu, Dave Peyton, Joe Capesius, Christie O'Day, Ann Tillery, Melissa Butler, Todd Ingersol, Ray Davis, David Graham, Ken Galyean, Frank Oliver, Rodrigo Morales, Tasha Lewis, Dawn McDoniel, Herb Pierce, Cory Angeroth (USGS), Tom Rees (volunteer), and Brian Popadac (volunteer) assisted with data collection and compilation.

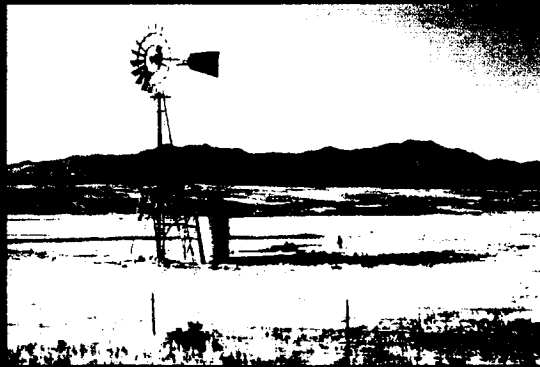
Sid Alwin, Pat Rigas, Doug Cummings, and John Callahan (USGS) contributed their talents to the preparation of this report.

Norm Spahr and Joe Domagalski (USGS), Marlene Baker (Concerned Citizens About Responsible Environment), Jeanmarie Haney (Tucson Regional Water Council), and many NAWQA Program staff provided valuable reviews of this report.

We extend special thanks and appreciation to our spouses, families, and friends, without whose support we could not have accomplished the work described herein.

NAWQA

National Water-Quality Assessment (NAWQA) Program
Central Arizona Basins



Cordy and others—Water Quality in the Central Arizona Basins
U.S. Geological Survey Circular 1213

ISBN 0-607-95418-3



9 780607 954180

R0024625

Water Quality in the Upper Colorado River Basin

Colorado, 1996–98



U.S. Department of the Interior
U.S. Geological Survey

Circular 1214

R0024626

POINTS OF CONTACT AND ADDITIONAL INFORMATION

The companion Web site for NAWQA summary reports:

<http://water.usgs.gov/nawqa/>

Upper Colorado River Basin contact and Web site:

USGS State Representative
U.S. Geological Survey
Water Resources Division
Denver Federal Center
Building 53, MS 415
Lakewood, CO 80225
e-mail: dc_co@usgs.gov
<http://co.water.usgs.gov>

National NAWQA Program:

Chief, NAWQA Program
U.S. Geological Survey
Water Resources Division
12201 Sunrise Valley Drive, M.S. 413
Reston, VA 20192
<http://water.usgs.gov/nawqa/>

Other NAWQA summary reports

River Basin Assessments

Albemarle-Pamlico Drainage Basin (Circular 1157)
Allegheny and Monongahela River Basins (Circular 1202)
Apalachicola-Chattahoochee-Flint River Basin (Circular 1164)
Central Arizona Basins (Circular 1213)
Central Columbia Plateau (Circular 1144)
Central Nebraska Basins (Circular 1163)
Connecticut, Housatonic and Thames River Basins (Circular 1155)
Eastern Iowa Basins (Circular 1210)
Georgia-Florida Coastal Plain (Circular 1151)
Hudson River Basin (Circular 1165)
Kanawha-New River Basins (Circular 1204)
Lake Erie-Lake Saint Clair Drainages (Circular 1203)
Las Vegas Valley Area and the Carson and Truckee River Basins (Circular 1170)
Lower Illinois River Basin (Circular 1209)
Long Island - New Jersey Coastal Drainages (Circular 1201)
Lower Susquehanna River Basin (Circular 1168)
Mississippi Embayment (Circular 1208)
Ozark Plateaus (Circular 1158)
Potomac River Basin (Circular 1166)
Puget Sound Basin (Circular 1216)

Red River of the North Basin (Circular 1169)
Rio Grande Valley (Circular 1162)
Sacramento River Basin (Circular 1215)
San Joaquin-Tulare Basins (Circular 1159)
Santee River Basin and Coastal Drainages (Circular 1206)
South-Central Texas (Circular 1212)
South Platte River Basin (Circular 1167)
Southern Florida (Circular 1207)
Trinity River Basin (Circular 1171)
Upper Mississippi River Basin (Circular 1211)
Upper Snake River Basin (Circular 1160)
Upper Tennessee River Basin (Circular 1205)
Western Lake Michigan Drainages (Circular 1156)
White River Basin (Circular 1150)
Willamette Basin (Circular 1161)

National Assessments

The Quality of Our Nation's Waters—Nutrients and Pesticides (Circular 1225)

Front cover: Maroon Creek in the Elk Mountains above Aspen, Colorado. (Photograph by Michael Collier.)

Back cover: Left, Drilling sampling well (photograph by Lori Apodaca); center, sampling the Colorado River near Dotsero (photograph by Norman Spahr); right, collecting invertebrate samples (photograph by Jeffrey Deacon).

Water Quality in the Upper Colorado River Basin, Colorado, 1996–98

By Norman E. Spahr, Lori E. Apodaca, Jeffrey R. Deacon, Jeffrey B. Bails, Nancy J. Bauch, C. Michelle Smith, *and* Nancy E. Driver

U.S. DEPARTMENT OF THE INTERIOR
BRUCE BABBITT, Secretary

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

2000

Free on application to the
U.S. Geological Survey
Information Services
Box 25286 Federal Center
Denver, CO 80225

Or call: 1-888-ASK-USGS

Library of Congress Cataloging-in-Publications Data

Water quality in the Upper Colorado River Basin, Colorado, 1996-98 / by Norman E. Spahr...[et al].
p. cm. -- (U.S. Geological Survey Circular ; 1214)

Includes bibliographical references.

ISBN 0-607-95424-8 (alk. paper)

1. Water quality--Colorado River Watershed (Colo.-Mexico) I. Spahr, Norman E. II. Geological Survey (U.S.) III. Series.

TD224.9 W38 2000
363.739'42'097913--dc21

00-049454

R0024629

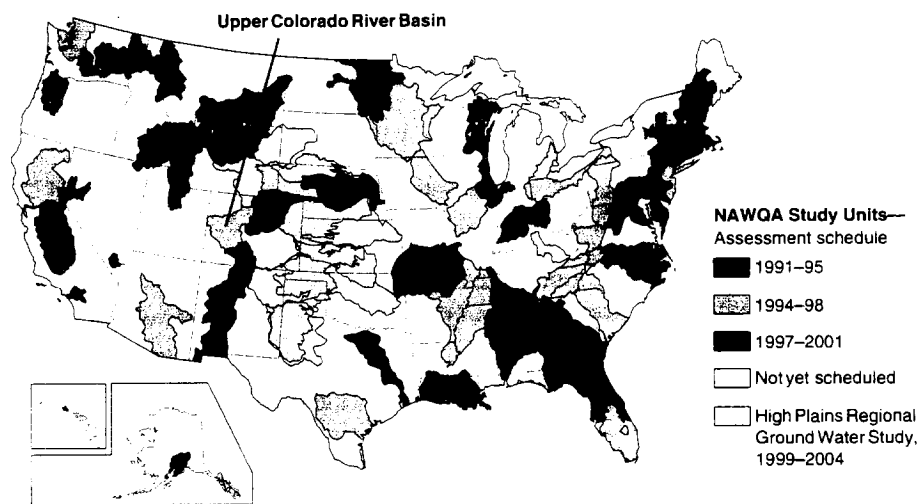
CONTENTS

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM	IV
SUMMARY OF MAJOR FINDINGS	1
Stream and river highlights	1
Ground-water highlights	2
INTRODUCTION TO THE UPPER COLORADO RIVER BASIN.....	3
MAJOR FINDINGS	6
Urban Development is a Water-Quality Issue Within the Southern Rocky Mountains	6
NATIONAL PERSPECTIVE—Biological measures indicate that urban sites in the UCOL are less degraded when compared to other urban sites nationally.....	8
NATIONAL PERSPECTIVE—Radon concentrations in UCOL ground water were generally higher than other areas of the Nation	9
The Quality of Streams and Rivers is Affected by Historical Mining Activities	11
NATIONAL PERSPECTIVE—Trace elements in streambed sediments of the UCOL were often greater than concentrations found at other sites nationally	12
Water-Quality Characteristics of Agricultural Areas	15
NATIONAL PERSPECTIVE—Herbicides commonly detected in the UCOL were also commonly detected in other agricultural areas nationally	17
NATIONAL PERSPECTIVE—Nitrogen concentrations at the UCOL agricultural sites were greater than concentrations for many agricultural sites nationwide.....	18
NATIONAL PERSPECTIVE—Biological measures indicate that UCOL agricultural and Colorado Plateau mixed land use sites rank among the more degraded sites nationally	19
STUDY UNIT DESIGN.....	21
GLOSSARY	23
REFERENCES	25
APPENDIX—WATER-QUALITY DATA FROM THE UPPER COLORADO RIVER BASIN IN A NATIONAL CONTEXT	28

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

THIS REPORT summarizes major findings about water quality in the Upper Colorado River Basin that emerged from an assessment conducted between 1996 and 1998 by the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program. Water quality is discussed in terms of local and regional issues and compared to conditions found in all 36 NAWQA study areas, called Study Units, assessed to date. Findings are also explained in the context of selected national benchmarks, such as those for drinking water quality and the protection of aquatic organisms. The NAWQA Program was not intended to assess the quality of the Nation's drinking water, such as by monitoring water from household taps. Rather, the assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water monitoring programs. The comparisons made in this report to drinking-water standards and guidelines are only in the context of the available untreated resource. Finally, this report includes information about the status of aquatic communities and the condition of instream habitats as elements of a complete water-quality assessment.

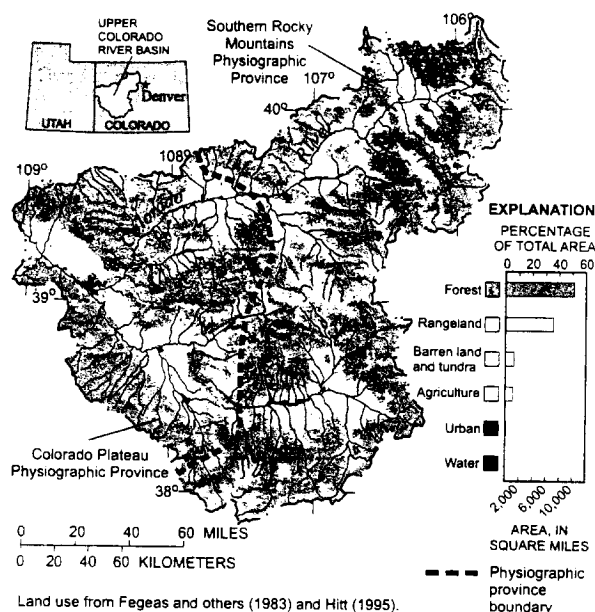
Many topics covered in this report reflect the concerns of officials of State and Federal agencies, water-resource managers, and members of stakeholder groups who provided advice and input during the Upper Colorado River Basin assessment. Basin residents who wish to know more about water quality in the areas where they live will find this report informative as well.



THE NAWQA PROGRAM seeks to improve scientific and public understanding of water quality in the Nation's major river basins and ground-water systems. Better understanding facilitates effective resource management, accurate identification of water-quality priorities, and successful development of strategies that protect and restore water quality. Guided by a nationally consistent study design and shaped by ongoing communication with local, State, and Federal agencies, NAWQA assessments support the investigation of local issues and trends while providing a firm foundation for understanding water quality at regional and national scales. The ability to integrate local and national scales of data collection and analysis is a unique feature of the USGS NAWQA Program.

The Upper Colorado River Basin is one of 51 water-quality assessments initiated since 1991, when the U.S. Congress appropriated funds for the USGS to begin the NAWQA Program. As indicated on the map, 36 assessments have been completed, and 15 more assessments will conclude in 2001. Collectively, these assessments cover about one-half of the land area of the United States and include water resources that are available to more than 60 percent of the U.S. population.

SUMMARY OF MAJOR FINDINGS



The Upper Colorado River Basin (UCOL) of the National Water-Quality Assessment (NAWQA) Program includes the 17,800-square-mile drainage basin of the Colorado River upstream from the Colorado-Utah State line. The study area is almost equally divided between the Southern Rocky Mountains and the Colorado Plateau Physiographic Provinces. Population in the basin is approximately 308,000. The major use of water is irrigation, but transmountain diversions provide water to more than 1 million people in the eastern part of Colorado (outside of the study area).

Stream and River Highlights

Streams and rivers in the Upper Colorado River Basin (UCOL) are very different in the two major physiographic provinces. In general, streams within the Southern Rocky Mountains are characterized by lower sediment and dissolved-solids concentrations, cooler temperatures, and somewhat higher gradients than streams in the Colorado Plateau. Sediment, salinity, and nutrient (nitrogen and phosphorus) concentrations increase along the major rivers as the water flows from the upstream areas in the Southern Rocky Mountains down through the Colorado Plateau.

Coupled with the general differences due to physiography and geology are the effects of different land uses. Recreation and urban development are becoming major land-use issues throughout the basin, precious metal mining was historically prevalent in the Southern Rocky Mountains, and intensive agriculture is located in the valleys of the Colorado Plateau.

Most of the streams and rivers sampled within the UCOL met State and Federal water-quality guidelines. Major exceptions to this statement were trace-element concentrations in some streams in the Southern Rocky Mountains and selenium concentrations in some streams in the Colorado Plateau.

- In the Southern Rocky Mountains, concentrations of nutrients (nitrogen and phosphorus) were generally low but were greater in urban streams than in streams in areas with minimal development (p. 6).
- Urban streams in the Southern Rocky Mountains had greater amounts of algae and a change in the invertebrate community from pollution-sensitive insects to pollution-tolerant insects compared to streams in areas with minimal development (p. 6–7). Similarly, in some mining areas of the Southern Rocky Mountains, the invertebrate community was also composed of pollution-tolerant insects, indicating more degraded sites (p. 13).

Selected indicators of Stream-Water Quality

	Southern Rocky Mountains		Colorado Plateau		
	Small Streams	Major Rivers	Small Streams	Major Rivers	Outlet of Study Unit
	Urban	Mining	Agri-culture		
Pesticides ¹	—	—	—	—	—
Nitrate ²	●	●	●	●	●
Total Phosphorus ³	●	●	●	●	●
Trace Elements ⁴	—	●	● ^a	●	●

■ Percentage of samples with concentrations **greater than or equal to** health-related national guidelines for drinking water, protection of aquatic life, or contact recreation

▒ Percentage of samples with concentrations **less than** health-related national guidelines for drinking water, protection of aquatic life, or contact recreation

○ Percentage of samples with **no detection** (^a Percentage is 1 or less and may not be clearly visible)

— Not assessed

¹Insecticides, herbicides, and pesticide metabolites, sampled in water.

²Nitrate (as nitrogen), sampled in water.

³Total phosphorus, sampled in water.

⁴Selenium and metals (such as cadmium, lead, and zinc), sampled in water.

- Concentrations of trace elements, such as cadmium, zinc, copper, and lead, in streambed sediments in many historical mining areas were greater than guidelines for the protection of aquatic life (p. 11–12).
- Pesticides were commonly detected in streams in agricultural areas of the Colorado Plateau during the growing season; however, the concentrations were typically low. Pesticide concentrations that exceeded guidelines for the protection of aquatic life were detected in only 5 of 90 samples (p. 16). Not all detected pesticides have established guidelines.
- The herbicides atrazine and alachlor were detected in more than one-half of the water samples collected in agricultural areas of the Colorado Plateau. These compounds, commonly used for weed control in corn, were also commonly detected in agricultural areas nationwide (p. 17).
- Nutrient and suspended-sediment concentrations in streams in the Colorado Plateau were typically greater than concentrations found in streams in other areas of the UCOL (p. 18–19). These concentrations can generally be associated with a more degraded status of algae, invertebrates, and fish (p. 19).
- Ground water in urban areas recharged in the late 1980s or 1990s tends to have higher concentrations of nitrate than ground water recharged before the 1980s (p. 10).
- Pesticides and volatile organic compounds were detected infrequently and generally at concentrations less than drinking-water standards. In only one sample, dichloromethane and tetrachloroethene, which are solvents, were detected at concentrations greater than their drinking-water standards (p. 10). Low concentrations of methyl *tert*-butyl ether (MTBE), a gasoline additive, were detected in shallow ground water in four of the five urban areas sampled (p. 9).
- Total coliform bacteria were detected in 21 percent of the shallow ground-water samples collected in urban areas in the Southern Rocky Mountains; none of the samples contained the potentially pathogenic *Escherichia coli* (*E. coli*) bacteria (p. 10).
- Radon, a naturally occurring radioactive gas, was detected in all wells sampled in urban areas in the Southern Rocky Mountains. Concentrations were greater than the proposed USEPA drinking-water standard of 300 picocuries per liter. Currently (2000), radon in drinking water is not regulated; however, if a new drinking-water regulation is implemented, treatment of drinking water for radon may be required in the UCOL (p. 9).

Major Influences on Streams and Rivers

- Urban development
- Abandoned/inactive mines
- Agricultural return flows

Ground-Water Highlights

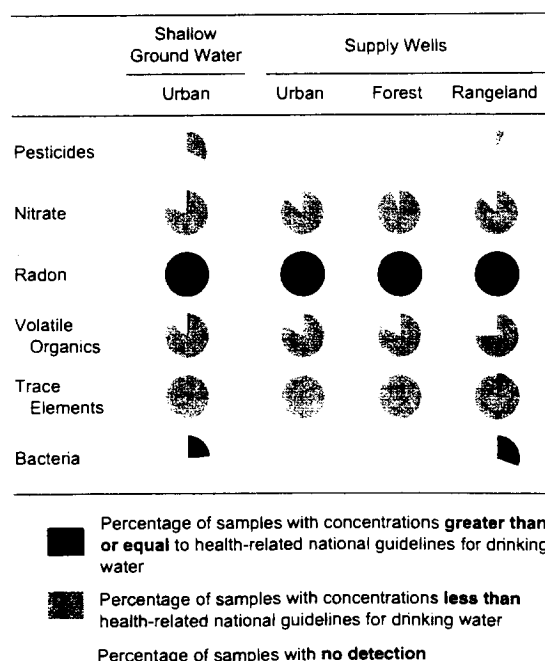
Historical ground-water-quality data for the UCOL are limited. The studies completed in the UCOL by NAWQA provide baseline information that can be used for identifying future water-quality changes. With the exception of radon, ground-water quality in the urban areas of the Southern Rocky Mountains generally met Federal and State standards for drinking water. The presence of a few elevated nitrate concentrations, a few pesticides, and generally low concentrations of volatile organic compounds indicate some influence on the quality of ground water from human activities. Bacteria were detected in ground-water samples and can occur naturally or indicate human influences.

- A concentration of nitrate greater than the U.S. Environmental Protection Agency (USEPA) drinking-water standard was found in 1 of 57 shallow ground-water samples collected in urban land-use settings (p. 8).

Major Influences on Ground Water

- Urban development in the Southern Rocky Mountains
- Natural background conditions

Selected Indicators of Ground-Water Quality



INTRODUCTION TO THE UPPER COLORADO RIVER BASIN

The Upper Colorado River Basin encompasses about 17,800 square miles. The primary river, the Colorado River, originates in the mountains of central Colorado and flows about 230 miles southwest into Utah. The basin is composed of two physiographic provinces: the Southern Rocky Mountains and the Colorado Plateau (fig. 1). The topography varies from rugged mountainous regions in the east to

high plateaus bordered by steep cliffs along valleys in the west. Because of differences in altitude of about 10,000 feet from east to west, the climate ranges from alpine conditions to semiarid/arid conditions. Precipitation ranges from 40 inches or more per year at high elevations in the eastern part of the basin to less than 10 inches per year at low elevations in the western part of the basin (fig. 2).

Snowmelt Runoff Dominates the Streamflow in Many Areas of the UCOL

The amount of water derived from the winter snowpack generally determines the magnitude and quality of streamflow for the UCOL. Streamflows are typically highest in the spring and lowest



The Colorado Plateau contains warm, saline streams that drain sedimentary geologic settings. (Photograph by Jeffrey Deacon, U.S. Geological Survey.)



Urban development is occurring in areas of the Southern Rocky Mountains. (Photograph by Scott Mize, U.S. Geological Survey.)



Cool and clear mountain streams are present in the forested areas of the Southern Rocky Mountains. (Photograph by Scott Mize, U.S. Geological Survey.)

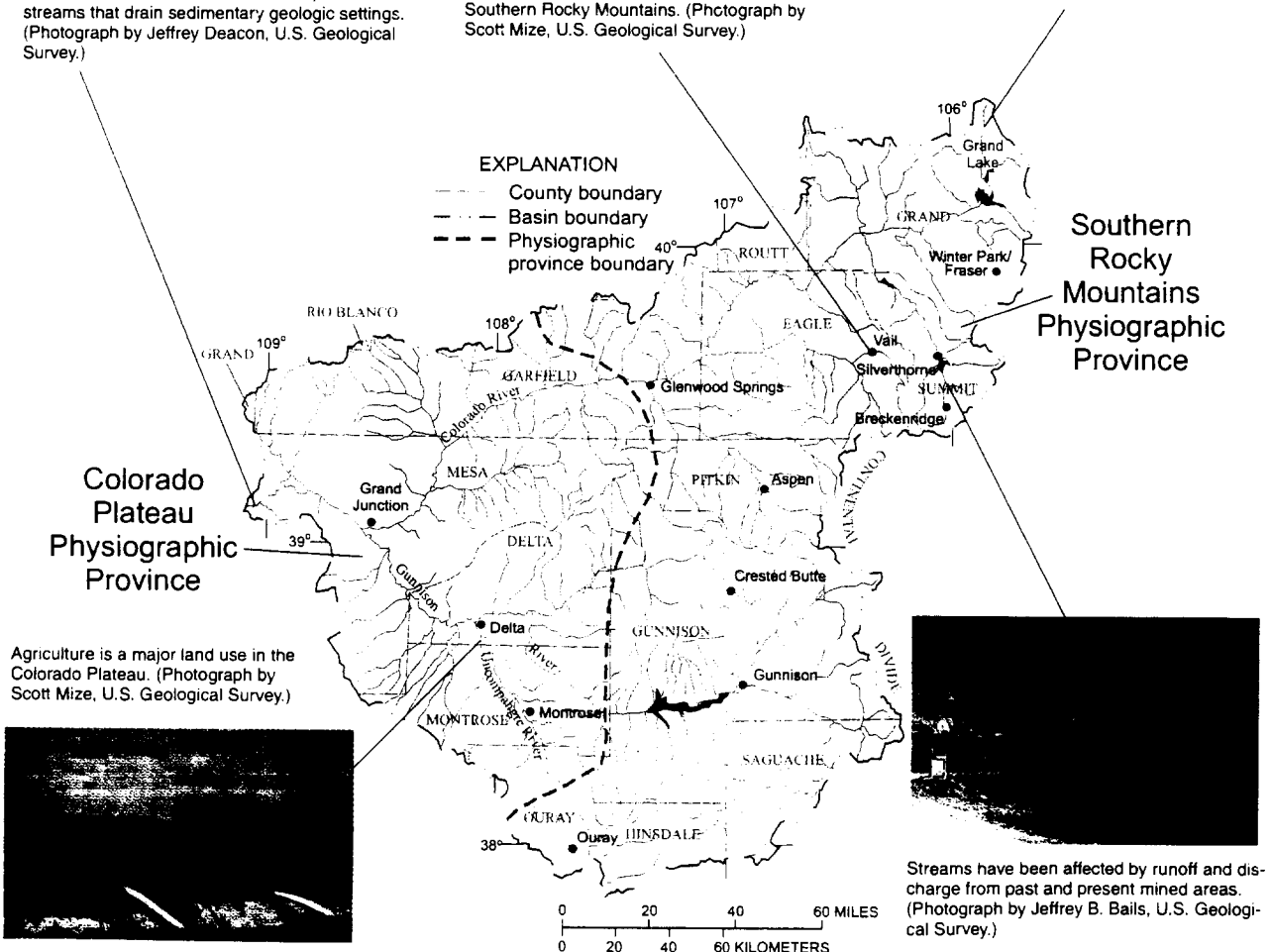


Figure 1. The combinations of physiography and land use produce different environments found in the UCOL. Physiographic provinces from Fenneman, 1946.

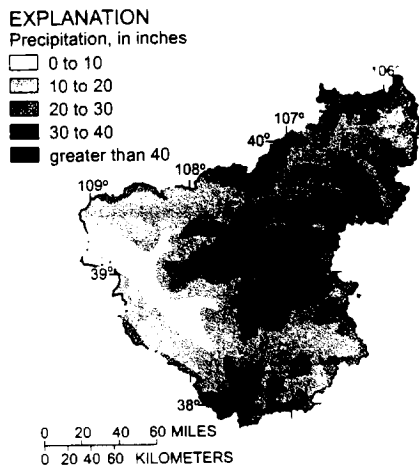


Figure 2. Average annual precipitation (1951–80) in the UCOL (Colorado Climate Center, 1984).

during the winter (fig. 3). The primary data-collection period for the UCOL study was water years 1996–98. Streamflow during water years 1996–97 was above average for most of the basin. Streamflow during water year 1998 was about average. High streamflows will dilute many water-quality constituents, resulting in lower concentrations. Suspended sediment and constituents associated with sediment may have larger concentrations during periods of high flow than during low flow.

Irrigation is the Principal Water Use Within the UCOL

Irrigation accounts for 97 percent of the water use in the UCOL (fig. 4). Ninety-nine percent of the water used in the Study Unit is derived from surface-water sources (U.S. Geological Survey, 1995). Ground water accounts for 1 percent of water use and is an important resource in remote and rural areas where the water is used primarily for domestic purposes.

Water diverted eastward from the UCOL, through transmountain

diversions, is used by many municipalities in the eastern plains of Colorado. This diverted water from the UCOL constitutes about 35 percent of the water supply for the city of Denver (Denver Water Department, 1999) and about 65 percent of the water supply for Colorado Springs (Scott Campbell, Colorado Springs Water Utility, oral commun., 2000). In addition, the Colorado Big Thompson project, using water diverted from the UCOL, provides complete or

partial supply for more than 30 cities and towns in northern Colorado (Northern Colorado Water Conservancy District, 2000). Transmountain diversions can affect the water quality in the basin because the diversions can account for a substantial portion of the local streamflow in upstream areas. In addition, the diverted water commonly has low salinity that is no longer available to dilute more mineralized water in the downstream part of the Study Unit.

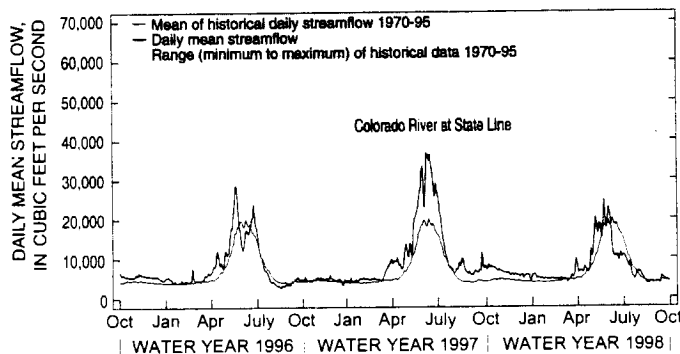


Figure 3. Streamflows in the UCOL were above average in water years 1996 and 1997. Water year 1998 streamflow was near average.

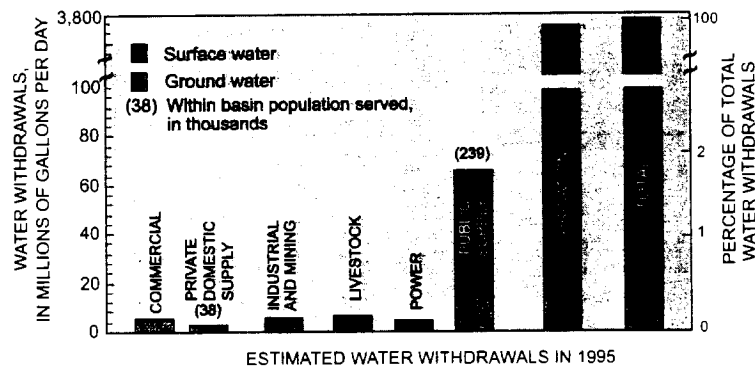


Figure 4. Water use in the basin is primarily derived from surface water; however, ground water is used for some domestic and public water supplies. Transmountain diversions to eastern Colorado from the UCOL accounted for about 451 million gallons per day in 1995 (Upper Colorado River Commission, 1999).

Water Quality is Influenced by Geologic Factors

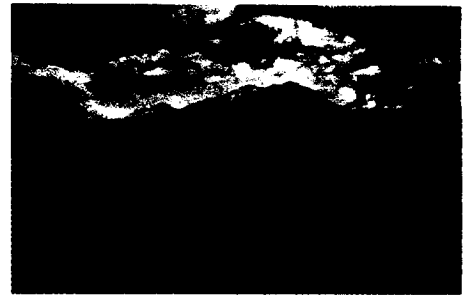
The underlying bedrock in the Study Unit is made up of crystalline and sedimentary rocks (fig. 5). Alluvium consisting of stream, landslide, terrace, and glacial deposits is present in valleys throughout the basin. Weathering of the different geologic units affects water-quality conditions in the basin. The sedimentary, igneous, and metamorphic rocks contribute material such as salts and trace elements to the streams. High concentrations of some materials, particularly salts, are derived from the sedimentary rocks, which are more common in the western part of the basin. Also, highly mineralized areas in the upper basin contribute trace metals to surface and ground water. Selenium occurs naturally in the shale bedrock of the middle and lower reaches of the basin and is present in surface and ground water. In addition, mineral hot springs located primarily in carbonate rock units in the north-

central part of the basin contribute about 13 percent of the total salt load at the outlet of the UCOL (U.S. Department of the Interior, 1995; Butler, 1996).

Water Quality is Influenced by Land Use

The UCOL study was designed to investigate land-use influences on water quality (see "Study Unit Design," page 21, for details). Urban development, mining, and agricultural were the three primary land uses investigated in the UCOL.

Urban development has the potential to affect the quality of surface and ground water by adding nutrients, bacteria, pesticides, hydrocarbons, trace elements, and salts from point and nonpoint sources and by changes to the natural landscape. Urban land use accounts for only 1 percent in the UCOL, which has a population of about 308,000 people (Bureau of the Census, 1999). By the year 2020, the population is projected to increase to more than 500,000



Topography representative of the Southern Rocky Mountains. (Photograph by Jeffrey Deacon, U.S. Geological Survey.)

(Colorado Department of Local Affairs, 2000). Resident population increases do not reflect development and services for vacation properties. There are large seasonal fluctuations in nonresident populations within the basin due to recreational activities. During the 1996–97 ski season, more than 9 million skiers visited ski areas within the UCOL (Colorado Ski Country USA, 2000). Effects on water quality from urban development are evident in some communities in the Southern Rocky Mountains.

Lode and placer mining, a historically significant land use in the UCOL, was common throughout the Southern Rocky Mountains. Streams and ground water have been affected by point-source mine discharge and nonpoint-source runoff from mined areas.

Areas of intensive agriculture are located primarily in the Colorado Plateau. Salinity, sediment, nutrients, pesticides, and selenium and other trace elements are common constituents in agricultural runoff. These constituents can have an adverse effect on the surface water, ground water, and aquatic life.

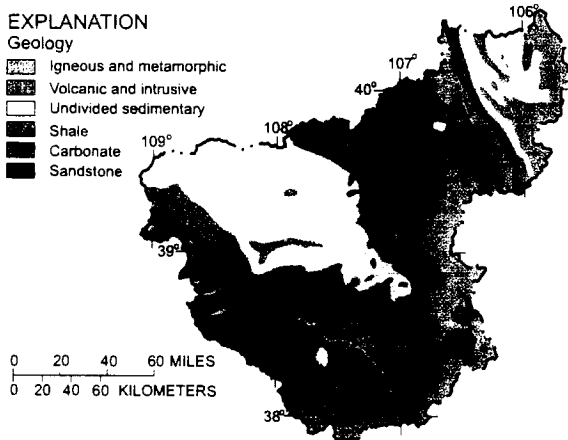


Figure 5. The bedrock geology within the UCOL changes from predominately igneous, metamorphic, and volcanic rock types in the eastern and central areas to predominantly sedimentary rock types in the western areas (Tweto, 1979; Green, 1992).

MAJOR FINDINGS

Urban Development is a Water-Quality Issue Within the Southern Rocky Mountains

Recreation in the mountains of Colorado is becoming a significant land-use activity in many areas. Urban development and the infrastructure to support recreation have increased substantially in the last decade. Between 1990 and 1997, the population increase in Summit and Eagle Counties exceeded 40 percent (<http://www.colorado.edu/libraries/govpubs/colonumb/counties.htm>). In addition to resident population increases, seasonal population fluxes due to recreation are large. These temporary population increases occur in winter during periods of extreme low flow and minimal dilution in rivers and streams.

Although urban development may be only a small percentage of total watershed area, development and transportation systems tend to be adjacent to riparian areas in mountainous terrain (fig. 6). This land-use pattern has placed human populations in locations that have the greatest effects on the quality of the water resources. Surface- and ground-water studies were implemented to investigate the water-quality issues in urban areas of the Southern Rocky Mountains.

Surface-water studies investigated the effects of urban land use. Two stream-monitoring sites were located in areas of urban development (fig. 6). The Gore Creek site is downstream from Vail, and the East River site is downstream from Crested Butte. Development in Vail is approaching build-out, whereas there is potential for large increases of development in the Crested Butte area. The Colorado River below

Baker Gulch, a reference site with little urban development in the Southern Rocky Mountains, was also sampled monthly to provide a comparison with other sites.

Nutrient concentrations at sites in areas of urban development were greater than concentrations at the reference site. Concentrations of nitrite plus nitrate and total phosphorus were slightly elevated at Gore Creek and East River compared to reference concentrations (fig. 7). Concentrations for nitrate were below the 10-mg/L Colorado instream standard at all three sites. Un-ionized ammonia concentrations, computed from dissolved ammonia, pH, and temperature, did not exceed State instream standards. Dissolved and orthophosphate

phosphorus concentrations were greater at Gore Creek than the other sites. Gore Creek and East River are not wastewater-effluent-dominated streams such as might be found in large metropolitan areas; however, small amounts of nitrogen and phosphorus can increase algal growth and eutrophication processes.

The amount of algae and the types of aquatic invertebrates (insects) are influenced by nutrient enrichment from urban sources. The amount of algae (algal biovolume) determined from algae samples collected in 1996 and 1997 was largest in Gore Creek (fig. 8), where nutrient levels were higher. The percentage of the invertebrate community represented by pollution sensitive insects

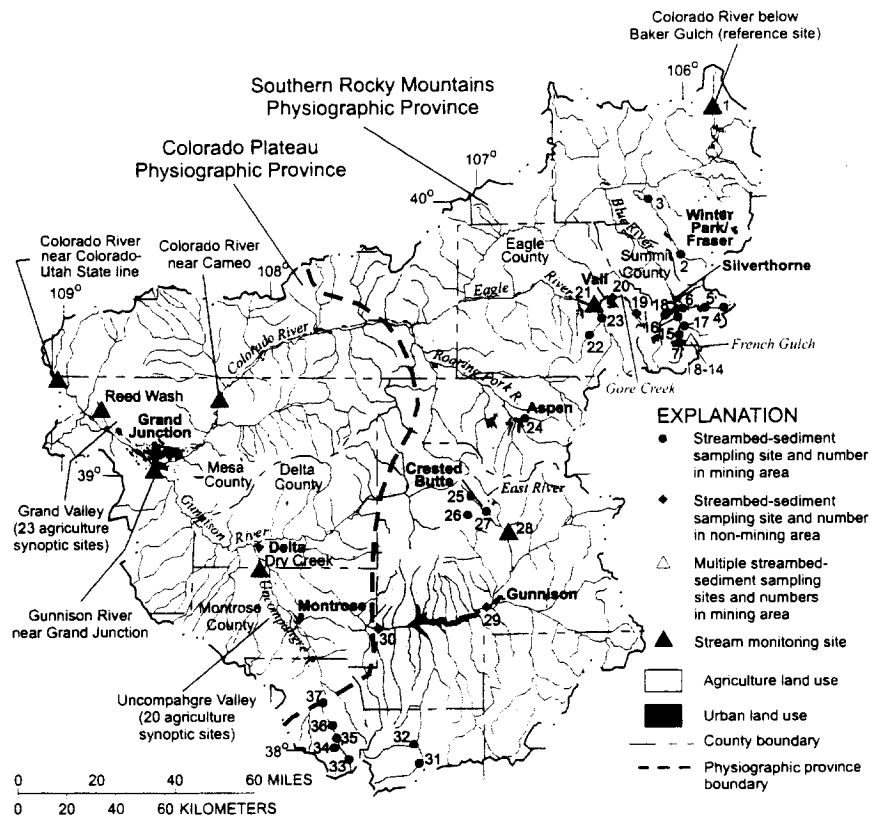


Figure 6. Sampling sites and study areas were selected to assess effects of urban, mining, and agricultural land uses on water quality.

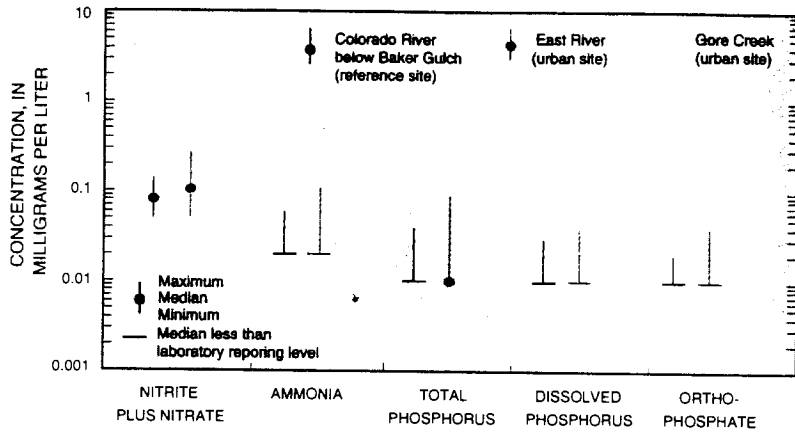


Figure 7. Concentrations of nitrite plus nitrate and total phosphorus were greater at sites with urban development (Gore Creek and East River) than at the reference site (Colorado River below Baker Gulch). Sites shown in order of increasing urban development. A log scale is used due to the large range of concentrations.

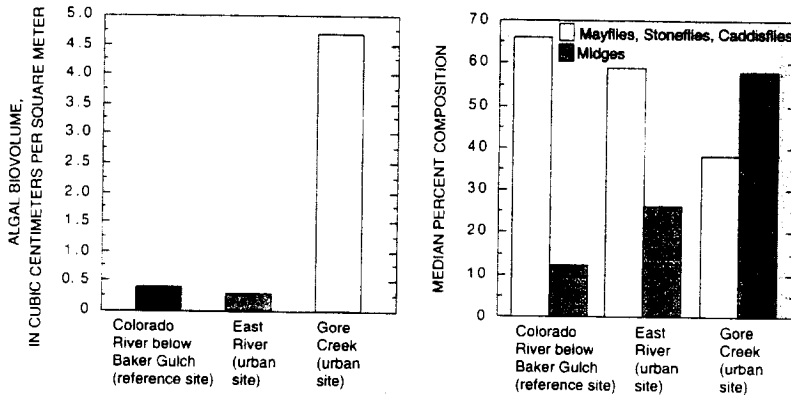


Figure 8. Algal biovolume (the amount of algae) was greater at Gore Creek, which also has greater urban development. The percentage of mayflies, stoneflies, and caddisflies decreased, and the percentage of midges (indicative of more degraded water quality) increased with urban development. Sites shown in order of increasing urban development.

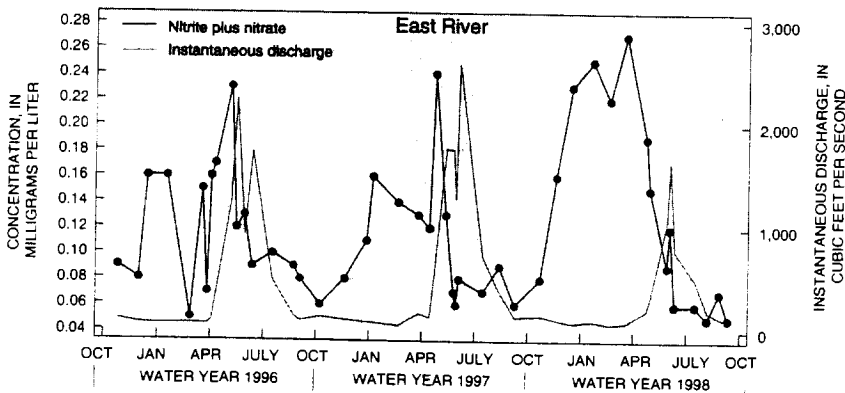


Figure 9. Nutrient concentrations were greatest during the late winter low-flow period prior to peak snowmelt runoff. This pattern was similar for total phosphorus at East River and for nitrite plus nitrate and total phosphorus at Gore Creek.

(mayflies, stoneflies, and caddisflies) was greater at sites with less urban influence (Colorado River below Baker Gulch, fig. 8). The percentage of midges (insects generally tolerant of pollution) increased with increasing urban influence (East River and Gore Creek sites, fig. 8). Even though the percentage of midges was greater at Gore Creek, the invertebrate status of Gore Creek was less degraded than other urban sites in the NAWQA Program (p. 8).

Nutrient concentrations were generally largest during winter.

Concentrations of nitrite plus nitrate and dissolved and total phosphorus were greatest during the late winter low-flow period prior to snowmelt runoff (fig. 9). With the onset of snowmelt runoff, nutrient concentrations were diluted. Algal uptake of nutrients during warm weather (July-October) probably lower concentrations during the summer. With reduced dilution and diminished algal uptake, nutrient concentrations increase again during the winter low-flow period.

Ground-water studies investigated the effects of urban land use on water in selected alluvial aquifers.

Although many communities in the UCOL rely on surface water as their primary source of drinking water, a few mountain towns and many thousands of individual homes in the Southern Rocky Mountains use ground water as their primary water source. Effects of urban land use can be indicated by elevated concentrations of nitrate, detections of synthetic organic compounds (pesticides and VOCs), and bacteria.



BIOLOGICAL MEASURES INDICATE THAT URBAN SITES IN THE UCOL ARE LESS DEGRADED WHEN COMPARED TO OTHER URBAN SITES NATIONALLY

Within the UCOL, there are changes in the algal and invertebrate communities in areas of urban development. Algal biovolume (amount of algae) and midges (insects considered more tolerant of degraded conditions) were greater at sites with urban development than sites with little urban development. However, with the exception of algal status at the Gore Creek site, biological indices for UCOL urban sites are in the lowest 25 percent of urban sites from other NAWQA Study Units. The Gore Creek watershed is more urbanized than the East River watershed, and the changes in the algal community reflect increased nutrient concentrations at this site, making it more typical of other urban sites nationally. The invertebrate and fish communities are ranked among the least degraded nationally at both UCOL urban sites.

Site name	Biological indicators		
	Algal status	Invertebrate status	Fish status
Colorado River below Baker Gulch (reference site)	■	■	■
East River (area of urban development)	■	■	■
Gore Creek (area of urban development)	□	■	■

EXPLANATION

■ Lowest 25 percent nationally, least degraded sites

□ Middle 50 percent nationally

Explanation of Biological Rankings

The three selected biological indicators respond to changes in stream degradation. Degradation can result from a variety of factors that modify habitat or other environmental features such as land use, water chemistry, and flow. Algal status focuses on the changes in the percentage of certain algae in response to increasing siltation and often is positively correlated with higher nutrient concentrations in many regions of the Nation. Invertebrate status is the average of 11 invertebrate metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. Fish status focuses on changes in the percentage of tolerant fish species that make up the total number of fish. "Tolerant" fish are reported to thrive in degraded water quality. For all indicators, higher values indicate degraded water quality.

Ground water was sampled from shallow monitoring wells in the Crested Butte, Gunnison, Silverthorne, Vail, and Winter Park/Fraser areas (fig. 6). Domestic (household and private) and public supply drinking-water wells of various depths were also sampled in urban areas throughout the Southern Rocky Mountains.

Some ground-water samples collected in urban areas contained elevated levels of nitrate. Nutrients in ground water can originate from various natural sources (such as atmospheric deposition or

dissolution of geologic materials); however, elevated concentrations in ground water are often related to human activities, such as effluent from septic systems or the application of fertilizers. Nitrate concentrations greater than the USEPA Maximum Contaminant Level (MCL) drinking-water standard of 10 mg/L (U.S. Environmental Protection Agency, 1996) were detected in one sample collected from a shallow monitoring well in an urban area but in none of the drinking-water wells. Nutrient concentrations in undeveloped areas

(determined from data from the 20 NAWQA Study Units that began in 1991) indicate a national background concentration for nitrate of 2.0 mg/L (U.S. Geological Survey, 1999). The undeveloped areas are considered to be minimally affected by agriculture, urban development, or associated land uses. Twenty percent of the samples collected from monitoring wells (and none of the samples collected from drinking-water wells) contained nitrate concentrations greater than 2.0 mg/L. These findings indicate that portions of the

shallow ground-water system have been affected by urban land use, but these effects were not found in the deeper ground water used for drinking water.

Generally, low levels of pesticides and volatile organic compounds were detected in ground water from shallow alluvial aquifers in urban areas. Most agriculture (and pesticide use) in the basin occurs in the Colorado Plateau where ground-water studies were not conducted. The pesticides that

were detected in the ground water sampled in the Southern Rocky Mountains were primarily herbicides used for controlling grasses and weeds in nonagricultural areas. Concentrations were very low (less than 0.1 µg/L) for most of the pesticides detected. Two herbicides, bromacil and prometon, were detected in urban areas at concentrations less than the USEPA drinking-water guidelines (U.S. Environmental Protection Agency, 1996).

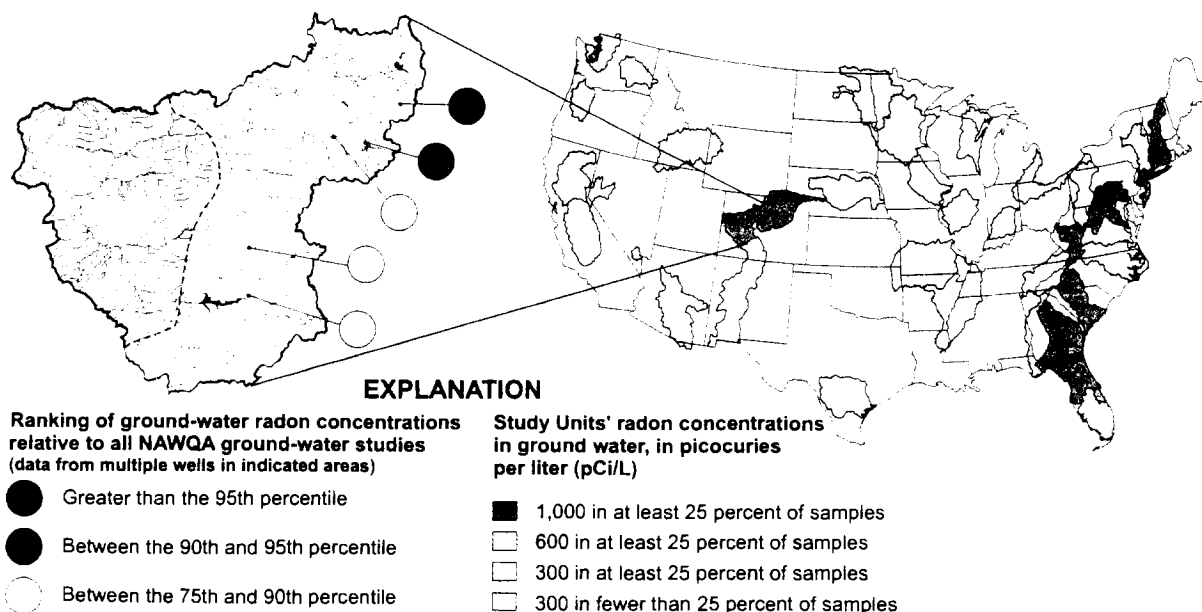
Volatile organic compounds (VOCs) were detected in ground-water samples at generally low concentrations. The six most frequently detected VOCs were methyl *tert*-butyl ether (MTBE), tetrachloroethene, chloroform, 1,1,1-trichloroethane, 1,2,4-trimethylbenzene, and dichloromethane (fig. 10). MTBE, a gasoline additive, was detected in at least one well in four of the five urban areas where shallow ground water was sampled. In addition,



RADON CONCENTRATIONS IN UCOL GROUND WATER WERE GENERALLY HIGHER THAN OTHER AREAS OF THE NATION

Radon concentrations in alluvial aquifers of the UCOL are among the highest of all NAWQA ground-water samples nationwide (S.J. Ryker, USGS, written commun., 1999). Radon is a naturally occurring, colorless, odorless, radioactive gas derived from the decay of uranium. Radon has been identified as a human carcinogen and can enter a building from seepage of air through dirt floors, cracks in concrete floors or walls, floor drains, or through the use of water supplied from wells. A greater risk of exposure to radon exists in urban areas due to the prevalence of enclosed structures, such as basements, which can trap radon gas.

Though the USEPA does not currently (2000) regulate radon in drinking water, there is a proposed maximum contaminant level (MCL) of 300 picocuries per liter (pCi/L) for radon in drinking water (U.S. Environmental Protection Agency, 1999). Radon concentrations were greater than the proposed MCL for all wells sampled in the Southern Rocky Mountains. Higher radon concentrations were measured in the Winter Park/Fraser and Silverthorne areas. Igneous and metamorphic rocks, which can contain uranium-bearing minerals, are common rock types in these areas.



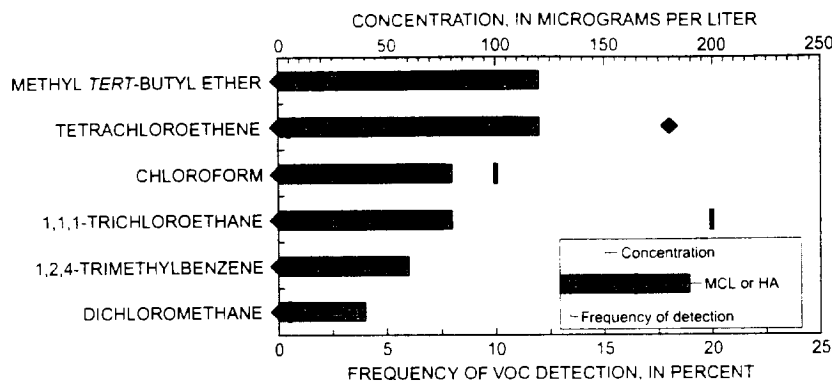


Figure 10. Volatile organic compounds were detected at low frequencies in monitoring and drinking-water wells sampled in the Southern Rocky Mountains. With a few exceptions (2 samples), concentrations of VOCs were substantially less than water-quality standards. All concentrations shown are greater than 0 µg/L. (MCL, maximum contaminant level; HA, health advisory.)

MTBE was detected in water from one municipal well. MTBE concentrations greater than the drinking-water lifetime health advisory of 20–40 µg/L (U.S. Environmental Protection Agency, 1997) were not detected. However, once MTBE enters the ground water, it is less vulnerable to decomposition and travels farther than other

gasoline compounds (Squillace and others, 1996).

Tetrachloroethene is present in solvents for adhesives and is also a by-product of dry cleaning. Chloroform can occur naturally or result from the chlorination of drinking water. 1,1,1-Trichloroethane is present in solvents and cleaning agents. 1,2,4-Trimethylbenzene is present in petroleum by-products.

Dichloromethane is predominantly used as a solvent in paint strippers and removers. Concentrations of dichloromethane and tetrachloroethene were greater than the USEPA drinking-water standard of 5 µg/L at one site in Silverthorne.

Total coliform bacteria were detected in 21 percent of the ground-water samples collected in urban areas. Total coliform bacteria were detected only in samples collected from the shallow monitoring wells and in none of the water samples from drinking-water wells. Bacteria in ground water may occur naturally in soils or may be related to human or animal waste and, therefore, may be an indication of the sanitary quality of the water (Myers and Sylvester, 1997). However, the presence of total coliform bacteria does not necessarily indicate the presence of potentially pathogenic bacteria such as *Escherichia coli* (*E. coli*), which was not detected in any of the samples.



Urban land use in the Southern Rocky Mountains, near Fraser, Colorado. (Photograph by Jeffrey B. Bails, U.S. Geological Survey.)

Chlorofluorocarbons (CFCs) were used to determine the age of ground water

Shallow ground water, which is used for drinking water in part of the Southern Rocky Mountains, is generally young (less than 10 years old) and is more susceptible to contamination as a result of land-use practices. The age of the ground water refers to the time from when the water recharged, or entered, the aquifer to the time it was withdrawn from the aquifer. The presence of contaminants, if any, in ground water can reflect land-use conditions at the time of recharge. The age of the ground water was determined in five urban areas where water-quality samples were collected from shallow alluvial aquifers. The age of ground water at most of the sites sampled ranged from 0 to 10 years old. Ground-water ages of greater than 10 years were found in 7 of 25 wells.

In the samples where CFC dates were obtained, four of the five nitrate concentrations greater than 2.0 mg/L were from recently recharged ground water (10 years or younger). This is an indication that current land use may be affecting water quality. Increasing urban development may add nitrate to the ground water from different sources, such as septic systems, application of fertilizers, and domestic animal wastes. Effects of increased urban development on ground-water resources are not extensive in the study area at present.

The Quality of Streams and Rivers is Affected by Historical Mining Activities

Streams affected by mine drainage, from abandoned/inactive mines, are present throughout much of the Rocky Mountains of Colorado. Metal mining in the UCOL has been an important part of the economy of Colorado since the late 1800s. As a result, many streams in the upper part of the basin contain heavy metals and other toxic elements that affect stream-water quality. Knowledge of mine-drainage effects on a body of water is essential for assessing water quality with regard to human consumption, recreation, and aquatic life.

Because of the potential exposure of aquatic organisms to trace elements in streambed sediment, trace-element concentrations in streambed sediment serve as an indicator of potential toxicity to aquatic life. Sampling trace ele-

ments in streambed sediment can be used to identify stream reaches affected by mining and can be used to determine sources of trace elements. The occurrence and distribution of trace elements were characterized by collecting streambed-sediment samples at 29 sites in mining districts (fig. 6 and table 1) and at 8 background (non-mining) sites in the Southern Rocky Mountains.

Concentrations of cadmium, copper, lead, and zinc in streambed sediment were high at some sites downstream from mining areas (Deacon and Driver, 1999). Median concentrations of cadmium (Cd) and zinc (Zn) in samples from most mining districts exceeded the Canadian Sediment Quality Guidelines Probable Effect Level (PEL), which is a sediment guideline (fig. 11). The PEL defines the concentration level above which adverse effects to aquatic biota are predicted to occur



Mining land use in the Southern Rocky Mountains. (Photograph by Jeffrey Deacon, U.S. Geological Survey.)

frequently (Canadian Council of Ministers of the Environment, 1999). National sediment concentration guidelines have not been established for the United States. Concentrations of copper and lead were also elevated and exceeded the PEL in several mining districts. The mining and mineralized areas of the UCOL generally result in higher trace-element concentrations in streambed sediment than are detected in other areas studied within the NAWQA Program (p. 12).

Collecting samples from several types of media provides a better understanding of trace elements in the environment. In addition to streambed sediment, which can store trace elements, the overlying stream water can also be a source of trace-element exposure to aquatic organisms. Because invertebrates are continuously exposed to water-quality conditions, these organisms integrate effects of contaminants over time and provide a measurement of water quality. Invertebrate indicators of streams contaminated by trace elements include reduced abundance and a shift in commu-

Table 1. Mining districts and sampling sites in the Southern Rocky Mountains

Mining district ¹	River/Stream [site number(s) in figure 6] ²
Aspen	Hunter Creek (24)
Breckenridge	Blue River, French Gulch, Swan River (7, 10, 11, 12, 13, 14, 15, 16, 17)
Climax	Tenmile Creek (18)
Crested Butte	Oh-Be-Joyful Creek, Coal Creek, Slate, East Rivers (25, 26, 27, 28)
Gilman	Cross Creek, Eagle River (22, 23)
Lake City (Carson, Burrows, Sherman Districts)	Lake Fork of the Gunnison River, Hensen Creek (31, 32)
Montezuma	Peru Creek, Snake River (4, 5, 6)
Ouray (Red Mountain, Sneffels, Uncompahgre Districts)	Canyon Creek, Red Mountain Creek, Uncompahgre River (33, 34, 35, 36, 37)
Urad-Henderson	South Fork of Williams Fork, Williams Fork (2, 3)

¹Information from Davis and Streufert (1990).

²Background (non-mining) sites 1, 8, 9, 19, 20, 21, 29, 30.

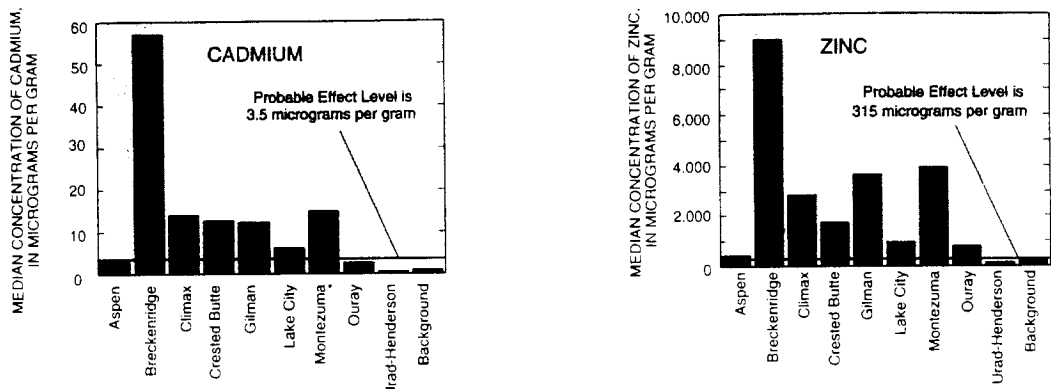


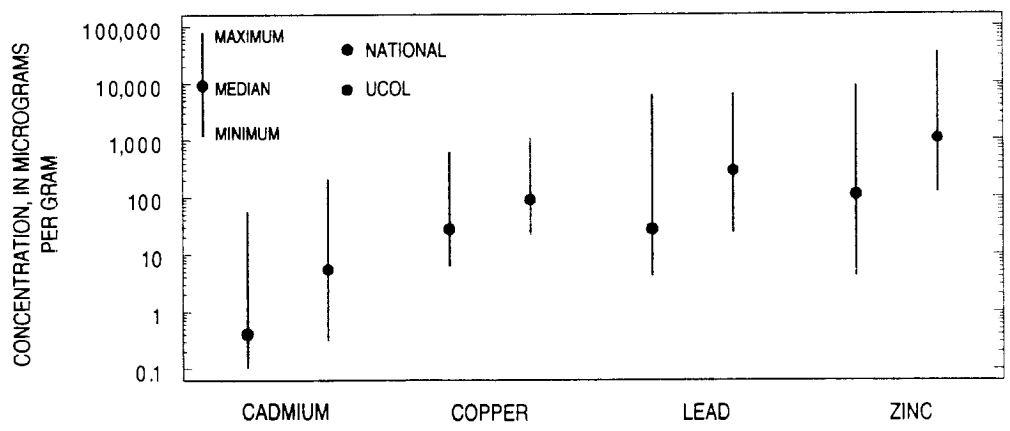
Figure 11. Median concentrations of cadmium and zinc in streambed sediments were greater than the Canadian Probable Effect Level in most mining districts of the UCOL.

nity composition from sensitive insects, such as mayflies, to more tolerant insects, such as midges. Lower mayfly abundance can be related to increased trace-element concentrations (Clements and Kiffney, 1995; Clements, 1994;

Kiffney and Clements, 1994). Concentrations of trace elements in transplanted aquatic moss also provide information about the occurrence and bioavailability of trace elements (Carter and Porter, 1997). In order to assess the source(s) and environmental effects of trace

elements, a more detailed study was conducted in the Blue River Basin; this study included the collection of water, streambed sediments, reservoir sediments, and aquatic moss samples and an assessment of the habitat and invertebrate communities.

TRACE ELEMENTS IN STREAMBED SEDIMENTS OF THE UCOL WERE OFTEN GREATER THAN CONCENTRATIONS DETECTED AT OTHER SITES NATIONALLY



The median concentrations for cadmium, copper, lead, and zinc in streambed sediment from sites in the UCOL were higher than the median concentrations of the same trace elements sampled in other Study Units of the national NAWQA Program. The data available on trace elements include 541 samples from the NAWQA Study Units (Rice, 1999) and 37 samples from the UCOL. Mineralized areas and historical mining activities in the UCOL have resulted in higher concentrations of some trace elements in streams than those concentrations that typically are found in other parts of the United States.



Collecting sediment core samples on Dillon Reservoir. (Photograph by Norman Spahr, U.S. Geological Survey.)

Mining activities have affected trace-element concentrations and aquatic invertebrates in French Gulch and the Blue River. Samples from different media were used to investigate the effects of mining activities on the water quality and associated biota at 10 sites along the Blue River and French Gulch, a tributary of the Blue River (fig. 12). Sites 2 and 3 (fig. 12) are background sites on French Gulch and are minimally affected by mining activities. Zinc concentrations in water and streambed sediment and zinc accumulated by aquatic moss at these background sites were lower than those from sites downstream from the mined areas. The invertebrate community structure (as indicated by mayfly abundance and percent midges) was minimally affected at the background sites.

Although habitat conditions at site 4 were degraded and zinc concentrations were greater than those upstream, the invertebrate community was not found to be signifi-

cantly degraded. Farther downstream, where underground mine seepage contributes to surface runoff (site 6), zinc concentrations in the water, streambed sediment, and aquatic moss were high and the invertebrate communities were degraded. The most affected site along French Gulch was site 7, located near the confluence with the Blue River.

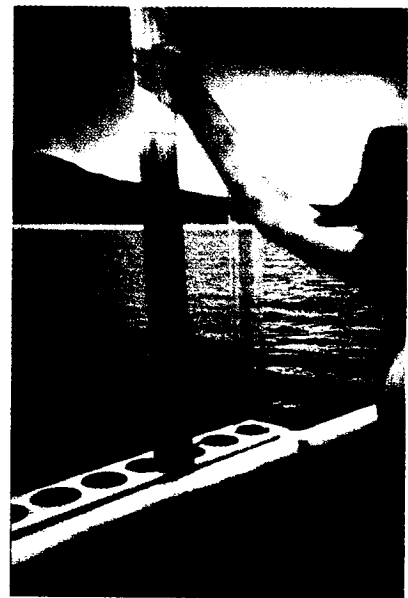
Site 1, upstream from French Gulch on the Blue River, was characterized by lower zinc concentrations in water, moss, and sediment than downstream sites, indicating that the French Gulch basin is contributing zinc to the Blue River. Downstream from the confluence of French Gulch with the Blue River (site 8), concentrations of zinc in water, moss, and streambed sediment were higher than background levels but slightly lower than in French Gulch. Also at site 8, brown trout livers had the highest concentrations of zinc of any sites sampled in the Southern Rocky Mountains (Deacon and Stephens, 1998). The invertebrate community in the Blue River downstream from French Gulch was degraded (site 8). The site conditions remain moderately affected downstream to Dillon Reservoir (site 10).

Mining activities have affected trace-element concentrations in the bottom sediment of Dillon Reservoir. Dillon Reservoir was constructed in 1963 in an area with a long history of mining activity. Sources of trace elements in Dillon Reservoir, a drinking-water supply for the city of Denver, are located in the Blue River, Snake River, and Tenmile Creek Basins. The reservoir is accumulating some trace elements. For example, loads calculated using streamflow and

water concentration indicate that only 37 percent of the zinc that enters Dillon Reservoir leaves the reservoir (fig. 12, top left graph).

Sediment cores were collected at several locations in Dillon Reservoir. Concentrations of lead and zinc throughout the core collected near the dam exceeded the PEL. Cadmium, lead, and zinc concentrations in sediment cores from the Blue River, Snake River, and Tenmile Creek arms of the reservoir also were above the PEL (values for zinc are shown in the top of fig. 12).

Although bottom sediment concentrations of some trace elements exceeded the PEL, concentrations in the water column were not high. The concentrations of trace elements in the reservoir water column did not exceed the Colorado surface-water-quality standards.



Sediment core from Dillon Reservoir. (Photograph by Norman Spahr, U.S. Geological Survey.)

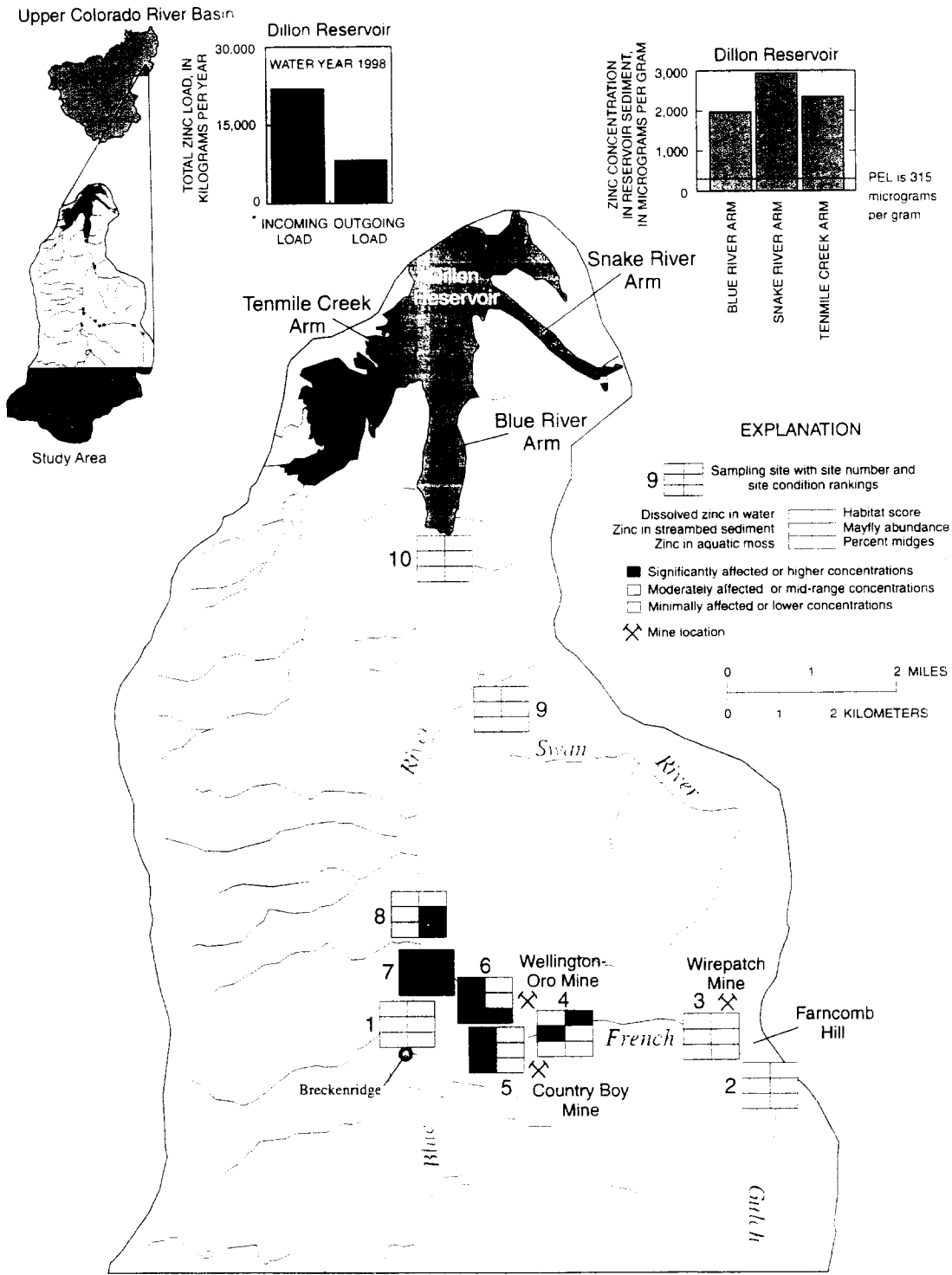
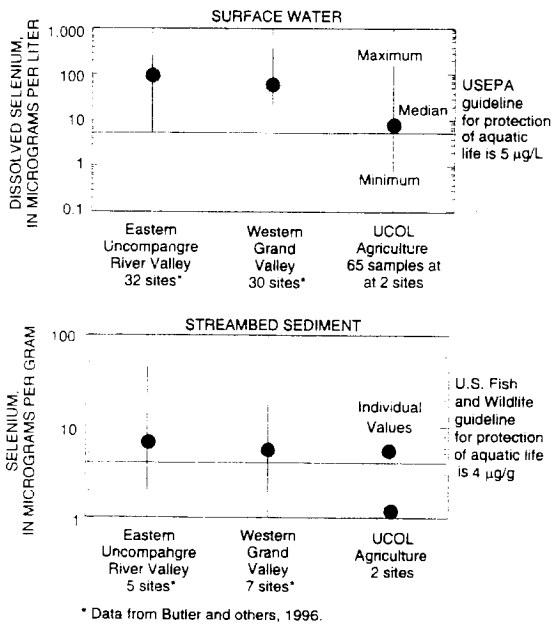


Figure 12. Different types of sampling media provide a better understanding of conditions at sites sampled in the Blue River Basin. Dillon Reservoir sediments are accumulating some trace elements originating from mining areas.

Selenium in Western Colorado

Selenium, a naturally occurring trace element, is common throughout the Western United States in marine sedimentary rocks. It is an essential micronutrient for birds, fish, and animals (Mayland, 1994) but at high concentrations can be highly toxic to fish and wildlife. Selenium can be very mobile in the environment and the mobility can be accelerated by irrigation. As irrigation water is applied to soils containing selenium, the selenium is leached out of the soils and into surface and ground water. Selenium in wetlands, ponds, and lakes is incorporated into bed sediment and can be bioaccumulated by wildlife, including fish and birds (Ohlendorf and others, 1986). Areas of the Western United States susceptible to selenium contamination from irrigation, which include



the Grand and Uncompahgre Valleys in western Colorado, have been identified by Seiler and others (1999). Extensive irrigated agriculture is present in the Grand and Uncompahgre Valleys of the Colorado Plateau in western Colorado (fig. 6). Irrigation drainage from these areas may account for as much as 75 percent of the selenium load in the Colorado River near the Colorado-Utah State line (Butler and others, 1996). It is estimated that 61 percent of the selenium load to Lake Powell in Utah originates from these agricultural areas in the UCOL (Engberg, 1999). Primary source areas of selenium in western Colorado are the western one-half of the Grand Valley and the eastern side of the Uncompahgre River Valley where the residual soils and alluvium are derived primarily from the Mancos Shale, a marine shale containing selenium.

A study in 1991–93 of irrigation drainage in the Grand and Uncompahgre River Valleys detected relatively high concentrations of selenium in many surface-water and streambed-sediment samples (Butler and others, 1996). Concentrations of selenium greater than the aquatic-life guidelines were also detected in the UCOL surface-water sampling in agricultural areas during 1995–98 and at one streambed-sediment sampling site in 1995. Investigations of selenium in western Colorado for remediation planning are continuing through the U.S. Department of the Interior National Irrigation Water Quality Program and the Gunnison Basin Selenium Task Force.

Water-Quality Characteristics of Agricultural Areas

Agricultural land use constitutes about 5 percent of the Study Unit area (fig. 6). Within the Southern Rocky Mountains, agricultural land is almost exclusively hay meadows. The agricultural areas of the Colorado Plateau (principally the Grand Valley near Grand Junction in Mesa County and the Uncompahgre Valley near Delta and Montrose in Delta and Montrose Counties, fig. 6) produce hay, corn, small grains, dry beans, onions, melons, fruit, and grapes. The Grand Valley has about 70,000 acres of irrigated land, and the Uncompahgre Valley contains about 86,000 irrigated acres (Butler and others, 1996). The market value of agricultural products produced in Delta, Mesa, and Montrose Counties was about 145 million dollars in 1992 (Bureau of the Census, 1994). Pesticides, nutrients, and sediment are water-quality issues commonly associated with agricultural land use.

Pesticides were sampled in different media. Pesticides were investigated in surface water by periodic monitoring at 2 agricultural sites and a one-time synoptic sampling at 43 agricultural sites. Fish were collected once at three agricultural sites and the tissues analyzed for organochlorine pesticides. Organochlorine pesticides in streambed sediments were sampled once at six agricultural sites. Ground-water samples were not collected in the agricultural areas of the UCOL because ground water is generally not used for public water supply in these areas.



Collecting water samples at an agricultural drain in the Grand Valley. (Photograph by Norman Spahr, U.S. Geological Survey.)

Pesticides were commonly detected in agricultural areas of the UCOL, but concentrations were generally low. Most pesticide detections in surface water were not at concentrations of concern. Freshwater aquatic-life guidelines were exceeded occasionally; however, guidelines have not been established for all compounds. Only 5 of the 90 samples collected in the agricultural areas contained pesticides that exceeded established guidelines. The pesticides that exceeded guidelines for protection of freshwater aquatic life are azinphos-methyl, 1 of 24 samples at Reed Wash; carbaryl, 2 synoptic sites, Indian Wash and Orchard Mesa Drain in the Grand Valley; diazinon, 1 synoptic site, Indian Wash; diuron, 1 synoptic site, Indian Wash; and *gamma*-HCH, 1 synoptic site, the drain at Blossom Road in the Uncompahgre Valley.

Pesticide detections and concentrations showed seasonal patterns. Periodic sampling at two

agricultural streams showed that the total number of pesticides detected was greatest during April through August (Bauch and Spahr, 2000). Pesticide detections per month for Dry Creek are shown in figure 13. Concentrations of atrazine were found to be greater from May through August than during other periods of the year (fig. 13). The seasonal pattern of concentrations was similar for other pesticides and reflects the growing season for the Grand and Uncompahgre Valleys.

Pesticides were detected in 40 of 43 agricultural streams.

Thirty-one pesticides (21 herbicides and 10 insecticides) were detected at least once during a May 1998 agricultural stream synoptic study in the Grand Valley and Uncompahgre Valley areas. Atrazine and alachlor were detected in more than 50 percent of the samples. Concentrations of atrazine did not exceed aquatic-life guidelines. Guidelines are not available for alachlor.

Some insecticides have persisted in streambed sediment and fish tissue, although their use has been banned or restricted. DDT was banned in 1972, but DDT or its



Agriculture and the San Juan Mountains near Montrose. (Photograph by Norman Spahr, U.S. Geological Survey.)

breakdown products, DDE and DDD, were detected in streambed sediments at five of the six agricultural sites sampled and in fish tissue at all three of the agricultural sites where fish tissue samples were collected (Stephens and Deacon, 1998). Concentrations of DDT and DDE in streambed sediment at two sites exceeded the Canadian Sediment Quality Guidelines PEL (Canadian Council of Ministers of the Environment, 1999). The DDD concentration at one site exceeded the PEL. Dieldrin (a restricted use insecticide since 1974) was detected in streambed sediments at two of the six sites and in fish tissue at all three sites. Insecticides detected in fish tissue and streambed sediments were generally not detected in

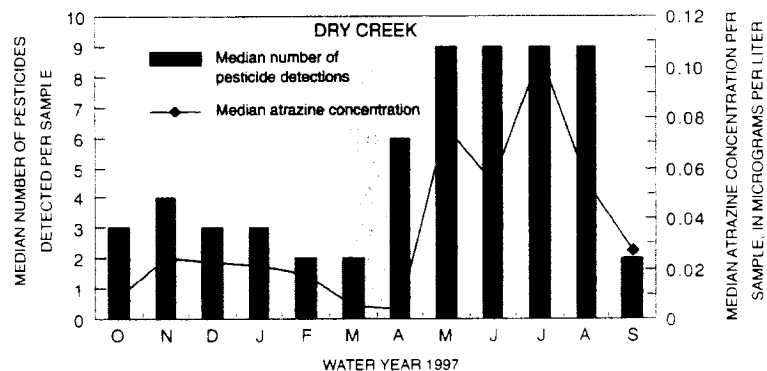
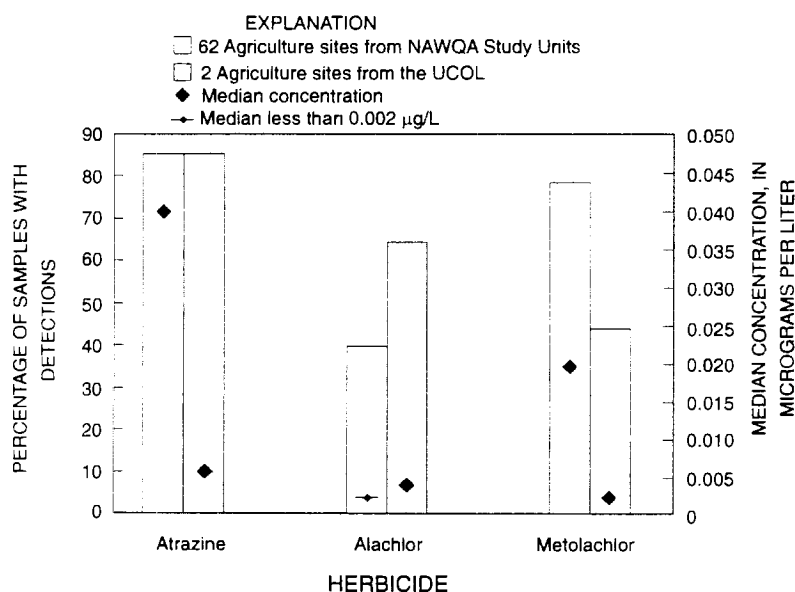


Figure 13. The number of pesticide detections and median atrazine concentrations in surface water are greater during the spring and summer months. Atrazine is commonly used for season-long weed control in corn.



HERBICIDES COMMONLY DETECTED IN THE UCOL WERE ALSO COMMONLY DETECTED IN OTHER AGRICULTURAL AREAS NATIONALLY

The three most commonly detected herbicides at the two agricultural monitoring sites in the UCOL (one site each in the Grand and Uncompahgre Valleys) also were among the top 10 herbicides detected in surface-water samples at 62 agricultural sites in 35 nationally distributed NAWQA Study Units. The percentages of samples with detections for atrazine, alachlor, and metolachlor are shown below and are based on more than 1,550 samples for the national sites and 39 samples at the UCOL sites. Atrazine, alachlor, and metolachlor are commonly used for weed control in corn. Alachlor and metolachlor are also used for weed control in dry beans. Other herbi-



cides that were frequently detected (present in over one-third of the samples) in the UCOL but not shown in the graph include trifluralin, DCPA, 2,4-D, cyanazine, and deethylatrazine (a breakdown product of atrazine).

These herbicides also were commonly detected at other agricultural sites across the Nation. Median concentrations of the commonly detected herbicides in the UCOL were less than or similar to the median concentrations for the national sites. Concentrations of atrazine and metolachlor were less than the Canadian guidelines for the protection of aquatic life [1.8 µg/L for atrazine and 7.8 µg/L for metolachlor (Environment Canada, 1999)]. Guidelines have not been established for alachlor.

water samples from the same site, probably because these compounds are relatively insoluble in water. Occurrence of organochlorine pesticides, even at low concentrations, is becoming increasingly relevant because of recent evidence linking these compounds to endocrine disruption (Goodbred and others, 1997).

Nutrient concentrations in streams and rivers reflect point and nonpoint sources. National background concentrations have been determined for some forms of nutrients: total nitrogen in streams (1.0 mg/L), nitrate in streams (0.6 mg/L), and total phosphorus in

streams (0.1 mg/L) (U.S. Geological Survey, 1999). Relative concentrations of nutrients in surface water are linked to the amounts and types of substances used and discharged in different land-use settings. These substances can then reach the stream through point sources (such as wastewater discharge) or nonpoint sources (such as precipitation or runoff from agricultural areas). Estimated amounts of fertilizer applied during 1997 in Delta, Mesa, and Montrose Counties were about 14,100,000 pounds of nitrogen and 1,800,000 pounds of phosphorus (Jeffrey Stoner, U.S. Geological Survey, written

commun., 2000). Fertilizer use in these counties accounted for about 77 percent of the total estimated fertilizer usage for the UCOL.

Nutrient concentrations in areas of agricultural land use were generally greater than in areas of other land uses. Median concentrations of ammonia, nitrite plus nitrate, total phosphorus, dissolved phosphorus, and orthophosphate were greater at agriculture sites than at the Colorado Plateau reference and mixed land-use sites (fig. 14). The Colorado Plateau reference site has some limited agriculture upstream (livestock/hay meadows), so is not representa-

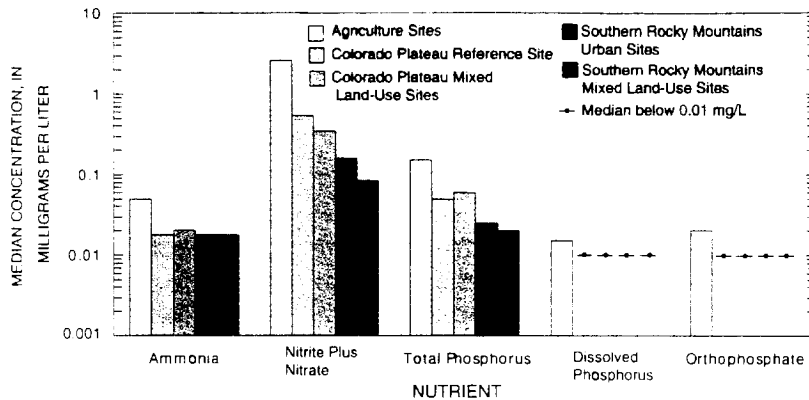


Figure 14. Concentrations of nutrients in agricultural areas of the UCOL were greater than concentrations in other land-use settings. A log scale is used due to the large range of concentrations.

tive of natural conditions but is probably representative of small streams in the Colorado Plateau. Median concentrations of nutrients at agricultural sites also were

greater than median concentrations at mixed land-use and urban sites within the Southern Rocky Mountains. The urban areas shown in the bar chart are downstream from Vail

and Crested Butte, Colorado (fig. 6), and are streams that are not dominated by wastewater effluent such as might be found in large metropolitan areas. Concentrations of nitrite and nitrate were within State standards for agricultural water use. Water from the agricultural streams is not commonly used for domestic supply.

Geology and agriculture contribute to elevated suspended-sediment concentrations in the Colorado Plateau. Suspended-sediment concentrations in the streams and rivers of the Colorado Plateau were much greater than concentrations in the Southern Rocky Mountains (fig. 15), due in large part to differences between the sedimentary geology of the

NITROGEN CONCENTRATIONS AT THE UCOL AGRICULTURAL SITES WERE GREATER THAN CONCENTRATIONS FOR MANY AGRICULTURAL SITES NATIONWIDE

Total nitrogen (the sum of nitrite, nitrate, ammonia, and organic nitrogen) concentrations at the UCOL agricultural monitoring sites (Reed Wash and Dry Creek) were ranked in the group of national agricultural sites with the highest concentrations. The map below shows a comparison of average annual total nitrogen concentrations at the two UCOL agricultural sites with agricultural sites in other NAWQA Study Units. The average concentrations for the UCOL sites were 3.8 milligrams per liter for Reed Wash and 3.2 milligrams per liter for Dry Creek and are less than State instream standards. Average annual total phosphorus concentrations at the two agricultural sites in the UCOL also ranked among the group of national agricultural sites with the highest total phosphorus concentrations.

TOTAL NITROGEN IN STREAMS—AGRICULTURAL AREAS

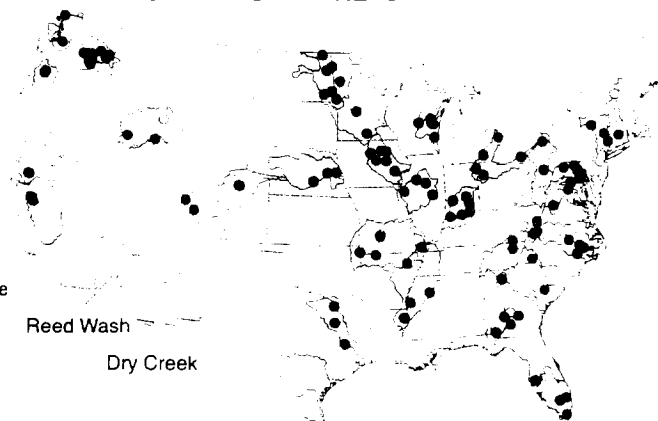
EXPLANATION

Average annual concentration of total nitrogen, in milligrams per liter

- Highest (greater than 2.9)
- Medium (0.64 to 2.9)
- Lowest (less than 0.64)

Average annual total nitrogen input, in pounds per acre, by county, for 1995–98. Inputs are from fertilizer, manure, and the atmosphere

- Greater than 25 pounds per acre
- 6 to 25 pounds per acre
- Less than 6 pounds per acre



Colorado Plateau and the igneous-metamorphic geology of the Southern Rocky Mountains. Agricultural return flows also contribute to the greater sediment concentrations measured at the agricultural sites.

The reference site in the Colorado Plateau also had elevated sediment concentrations. Any rain or snowmelt event in these areas tends to substantially increase the sediment concentrations of streams and rivers because of the high erodibility of the soils.

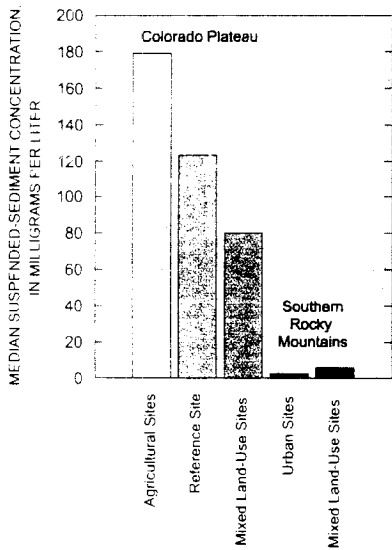


Figure 15. Suspended-sediment concentrations were greatest in the agricultural areas of the Colorado Plateau.



Sediment and water-quality sampling at Reed Wash. (Photograph by Norman Spahr, U.S. Geological Survey.)

BIOLOGICAL MEASURES INDICATE THAT UCOL AGRICULTURAL AND COLORADO PLATEAU MIXED LAND-USE SITES RANK AMONG THE MORE DEGRADED SITES NATIONALLY

The elevated nutrient and sediment concentrations found in the agricultural and surrounding areas of the Colorado Plateau help to explain the prevalence of more tolerant biological species. Algal, invertebrate, and fish communities in the Colorado Plateau generally consist of species more tolerant to nutrients and sediments. These result in higher degradation rankings for all three biological measures. The biological measures in the Colorado Plateau indicate that agricultural and mixed land-use sites in the UCOL are above the national average in terms of degradation. These rankings compare UCOL agricultural sites to other NAWQA agricultural sites and UCOL mixed land-use sites to other NAWQA mixed land-use sites nationally.

Site name	Biological indicator		
	Algal status	Invertebrate status	Fish status
Dry Creek (agricultural site)	□	■	■
Reed Wash (agricultural site)	■	■	■
Colorado River near Cameo (Colorado Plateau mixed land use)	■	□	□
Gunnison River near Grand Junction (Colorado Plateau mixed land use)	■	■	■
Colorado River at State line (Colorado Plateau mixed land use)	■	■	□

EXPLANATION
 □ Middle 50 percent nationally
 ■ Highest 25 percent nationally, most degraded sites

Explanation of Biological Rankings

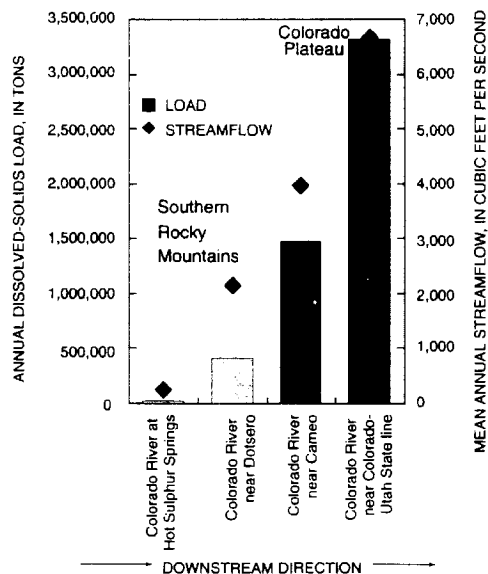
The three selected biological indicators respond to changes in stream degradation. Degradation can result from a variety of factors that modify habitat or other environmental features such as land use, water chemistry, and flow. Algal status focuses on the changes in the percentage of certain algae in response to increasing siltation and often is positively correlated with higher nutrient concentrations in many regions of the Nation. Invertebrate status is the average of 11 invertebrate metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. Fish status focuses on changes in the percentage of tolerant fish species that make up the total number of fish. "Tolerant" fish are reported to thrive in degraded water quality. For all indicators, higher values indicate degraded water quality.

Salinity in the Colorado River Basin

A serious water-quality issue in the Colorado River Basin is salinity defined as the concentration of dissolved mineral salts or total dissolved solids in water. Salinity increases in the Colorado River in a downstream direction; the dissolved-solids concentration is about 50 mg/L in the upstream mountain areas and averages about 850 mg/L at Imperial Dam, Arizona (Kircher, 1984). In the UCOL, annual dissolved-solids loads in the Colorado River ranged from about 17,700 tons at Hot Sulphur Springs in the Southern Rocky Mountains to more than 3,300,000 tons near the Colorado-Utah State line.

Major sources of salinity in the Colorado River Basin are mineral springs and nonpoint-source runoff. The major human influence is irrigated agriculture. About 11 percent of the salt load in the Colorado River near the U.S.–Mexico border is contributed from the Grand Valley and Uncompahgre Valley (lower Gunnison River Basin) agricultural areas (U.S. Department of the Interior, 1999). Much of the soil in these areas is derived from and overlies the Mancos Shale, a saline marine deposit. Deep percolation of irrigation water and seepage losses from irrigation systems leach salt from the soil and shale, increasing the salinity of return flows. Salinity-control projects have been constructed throughout the Colorado River Basin, including two projects in the UCOL—the Grand Valley Unit and the Lower Gunnison Basin Unit.

As part of the study of salinity in the Colorado River Basin, trends in dissolved-solids concentrations in the basin have been investigated in numerous studies (Vaill and Butler, 1999; Bauch and Spahr, 1998; Butler, 1996; Liebermann and others, 1989; Kircher, 1984). The most recent study (Vaill and



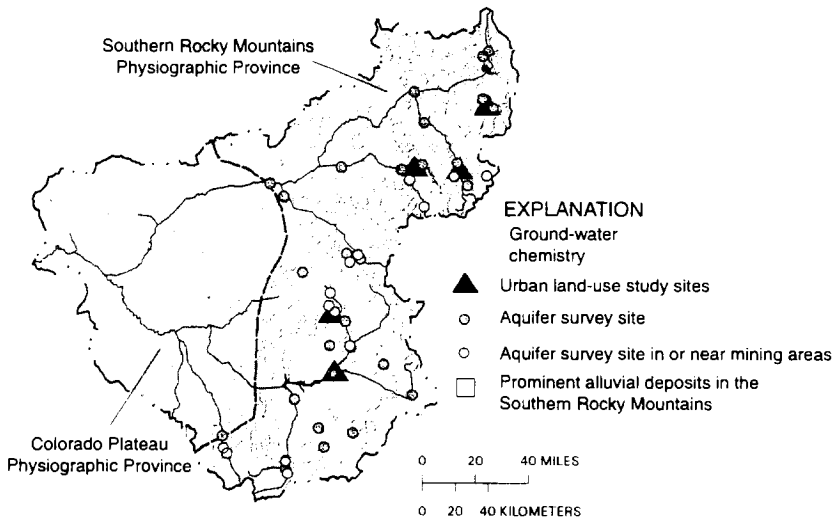
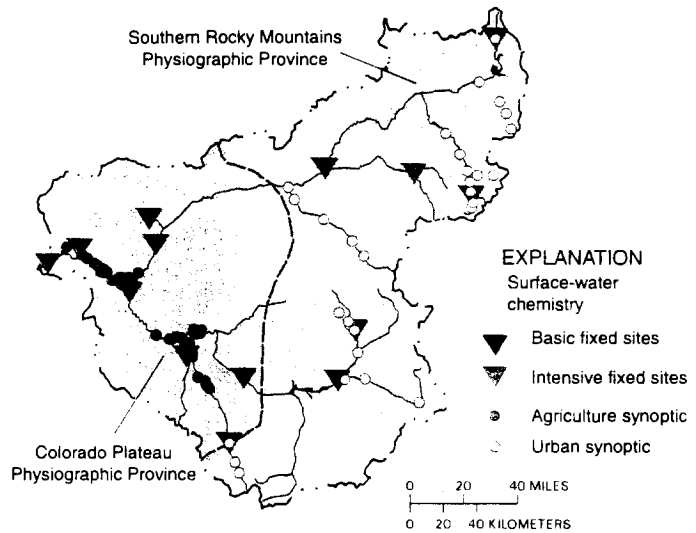
Mean annual dissolved-solids loads increase greatly between sites in the Southern Rocky Mountains and sites in the Colorado Plateau. Data based on water years 1970 to 1993. (Butler, 1996; Bauch and Spahr, 1998)

Butler, 1999) determined that since the 1960s there have been, in general, downward trends in flow-adjusted annual and monthly dissolved-solids concentrations and loads in the Colorado River Basin upstream from Lake Powell in Utah, except in the Yampa River Basin. In the UCOL, there were downward trends both upstream and downstream from the salinity-control projects in the Grand Valley and the lower Gunnison River Basin. With the downward trends as evidence, it appears that both natural processes and human efforts such as salinity-control projects may be decreasing salinity loading in the Colorado River Basin.

STUDY UNIT DESIGN

SURFACE-WATER CHEMISTRY

The basic fixed site network was established to investigate the differences in stream-water quality associated with land use and environmental settings within the UCOL. Basic fixed sites were sampled monthly to determine temporal variability of water-quality constituents. The three intensive fixed sites (a subset of the basic fixed site network) were sampled weekly to monthly to further define temporal variability. Synoptic studies were designed to investigate agriculture and urban development effects on water quality. Synoptic sites were sampled one or two times to determine spatial influences of land use on water chemistry.

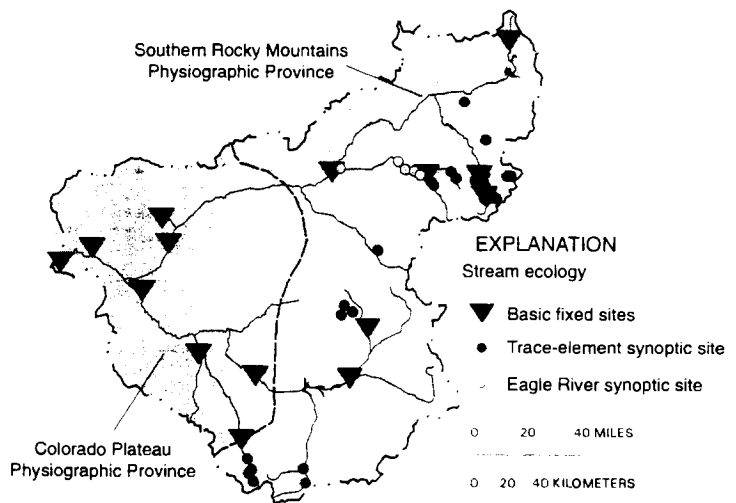


GROUND-WATER CHEMISTRY

To evaluate the effects of urban land use on shallow ground-water quality, five areas within the Southern Rocky Mountains that are undergoing urban development and use ground water as a primary drinking-water source, were studied. An overall assessment of water quality in predominantly alluvial aquifers of the Southern Rocky Mountains was also completed. In addition to the overall assessment, wells were selected in or near mining areas. Alluvial aquifers were selected for these studies because they are generally productive and can be susceptible to land-use practices.

STREAM ECOLOGY

Ecological assessments were done at all basic fixed sites plus one additional site on the Blue River. A synoptic study in mining areas of the Southern Rocky Mountains investigated trace elements in streambed sediment, aquatic moss, and surface water. A characterization of the algal and invertebrate communities and nutrient concentrations in surface water was completed at five sites on the Eagle River.



SUMMARY OF DATA COLLECTION IN THE UPPER COLORADO RIVER BASIN, 1996-98

Study component	What data were collected and why	Types of sites sampled	Number of sites	Sampling frequency and period
Surface-Water Chemistry				
Basic fixed sites—general water chemistry	Monthly samples collected for major ions, nutrients, organic carbon, and suspended sediment to describe concentration variability and loads. Daily streamflow also collected or computed.	Streams and rivers representing urban development, mining, agricultural, and mixed land use.	14	Monthly plus additional high- or low-flow samples for water years 1996-98
Intensive fixed sites	All constituents collected for the fixed sites. Weekly to monthly samples collected to describe temporal variability in pesticide and volatile organic compounds concentrations.	A subset of the fixed site network. One site representing urban development, one site representing agricultural land use, and the outlet of the Study Unit.	3	Weekly to monthly plus additional high- or low-flow samples for water year 1997
Synoptic sites—agriculture	One time sample to describe the spatial variability of pesticides and nutrients in agricultural areas.	Agricultural drains, streams and rivers in the Grand and Uncompahgre Valleys. Outlet of the Study Unit.	43	May 1998
Synoptic sites—urban development	Nutrient and algae samples along an upstream to downstream profile in areas of urban development to investigate influences of urban development.	Upstream to downstream sites along seven tributary rivers.	30	March and September 1998
Ground-Water Chemistry				
Land-use effects—Urban	Samples collected for major ions, nutrients, 18 trace elements, radon, dissolved organic carbon, 87 pesticides, 86 volatile organic compounds, bacteria (total coliform and <i>E. coli</i>), and chlorofluorocarbons to describe the effects of urban development on the shallow ground water in five urban areas of the Southern Rocky Mountains.	Shallow monitoring wells	25	Spring and Fall 1997
Aquifer survey—Southern Rocky Mountains	Samples collected for major ions, nutrients, 18 trace elements, radon, dissolved organic carbon, 47 pesticides, 86 volatile organic compounds, and bacteria (total coliform and <i>E. coli</i>) to describe the water-quality conditions in selected alluvial aquifers throughout the Southern Rocky Mountains Physiographic Province.	Private and municipal drinking-water wells Monitoring well Spring	28 1 1	Once in 1997
Aquifer survey—Southern Rocky Mountains in or near mining areas	Samples collected for same as above (except that 13 of these sites were not sampled for pesticides or volatile organic compounds) to describe the water-quality conditions in selected alluvial aquifers throughout the Southern Rocky Mountains Physiographic Province for wells located in or near mining areas.	Private and municipal drinking-water wells Spring	14 1	Once in 1997
Stream Ecology				
Basic fixed sites	Fish, invertebrate, and algae communities, trace elements, and organics in streambed sediment and fish tissue, and habitat data were collected to assess the occurrence and distribution within the UCOL.	Streams and rivers representing urban development, mining, agricultural, and mixed land use.	15	Fish community, invertebrate community, and algae August, 1996-98; Habitat August 1996; Streambed sediment and fish tissue August-October 1995
Synoptic sites—trace elements	Trace-element concentrations in streambed sediment, aquatic moss, and water; and invertebrate community structure to assess the spatial extent and magnitude of trace-element contamination from mining areas.	Areas of mining land use and background sites in areas of no mining.	32	August 1998
Synoptic sites—Eagle River	Invertebrates, algae, and nutrients in water to characterize current conditions in an area of extensive urban development.	Sites within and downstream from developed areas. Additional sites in areas that may be developed in the future.	5	February and September 1997

GLOSSARY

- Algae**—Chlorophyll-bearing, nonvascular, primarily aquatic species that have no true roots, stems, or leaves; most algae are microscopic, but some species can be as large as vascular plants.
- Alluvium**—Deposits of clay, silt, sand, gravel or other particulate rock material left by a river in a streambed, on a flood plain, delta, or at the base of a mountain.
- Alluvial aquifer**—A water-bearing deposit of unconsolidated material (sand and gravel) left behind by a river or other flowing water.
- Ammonia**—A compound of nitrogen and hydrogen (NH₃) that is a common by-product of animal waste. Ammonia readily converts to nitrate in soils and streams.
- Aquatic-life guideline**—Water-quality guidelines for protection of aquatic life. Often refers to U.S. Environmental Protection Agency water-quality criteria for protection of aquatic organisms. See also Water-quality guidelines.
- Basin**—See Drainage basin.
- Benthic invertebrates**—Insects, mollusks, crustaceans, worms, and other organisms without a backbone that live in, on, or near the bottom of lakes, streams, or oceans.
- Bioaccumulation**—The biological sequestering of a substance at a higher concentration than that at which it occurs in the surrounding environment or medium. Also, the process whereby a substance enters organisms through the gills, epithelial tissues, dietary, or other sources.
- Breakdown product**—A compound derived by chemical, biological, or physical action upon a pesticide. The breakdown is a natural process which may result in a more toxic or a less toxic compound and a more persistent or less persistent compound.
- Chlorofluorocarbons**—A class of volatile compounds consisting of carbon, chlorine, and fluorine. Commonly called freons, which have been used in refrigeration mechanisms, as blowing agents in the fabrication of flexible and rigid foams, and, until several years ago, as propellants in spray cans.
- Community**—In ecology, the species that interact in a common area.
- Concentration**—The amount or mass of a substance present in a given volume or mass of sample. Usually expressed as microgram per liter (water sample) or micrograms per kilogram (sediment or tissue sample).
- Confluence**—The flowing together of two or more streams; the place where a tributary joins the main stream.
- Constituent**—A chemical or biological substance in water, sediment, or biota that can be measured by an analytical method.
- Cubic foot per second (ft³/s, or cfs)**—Rate of water discharge representing a volume of 1 cubic foot passing a given point during 1 second, equivalent to approximately 7.48 gallons per second or 448.8 gallons per minute or 0.02832 cubic meter per second.
- Detect**—To determine the presence of a compound.
- DDT**—Dichloro-diphenyl-trichloroethane. An organochlorine insecticide no longer registered for use in the United States.
- Discharge**—Rate of fluid flow passing a given point at a given moment in time, expressed as volume per unit of time.
- Dissolved constituent**—Operationally defined as a constituent that passes through a 0.45-micrometer filter.
- Drainage basin**—The portion of the surface of the Earth that contributes water to a stream through overland runoff, including tributaries and impoundments.
- Eutrophication**—The process by which water becomes enriched with plant nutrients, most commonly phosphorus and nitrogen.
- Health advisory**—Nonregulatory levels of contaminants in drinking water that may be used as guidance in the absence of regulatory limits. Advisories consist of estimates of concentrations that would result in no known or anticipated health effects (for carcinogens, a specified cancer risk) determined for a child or for an adult for various exposure periods.
- Instream standards**—See Water-quality standards.
- Invertebrate**—An animal having no backbone or spinal column. See also Benthic invertebrates.
- Load**—General term that refers to a material or constituent in solution, in suspension, or in transport; usually expressed in terms of mass or volume.
- Maximum contaminant level (MCL)**—Maximum permissible level of a contaminant in water that is delivered to any user of a public water system. MCLs are enforceable standards established by the U.S. Environmental Protection Agency.
- Median**—The middle or central value in a distribution of data ranked in order of magnitude. The median is also known as the 50th percentile.
- Micrograms per liter (µg/L)**—A unit expressing the concentration of constituents in solution as weight (micrograms) of solute per unit volume (liter) of water; equivalent to one part per billion in most streamwater and ground water. One thousand micrograms per liter equals 1 mg/L.
- Midge**—A small fly in the family Chironomidae. The larval (juvenile) life stages are aquatic.

Milligrams per liter (mg/L)— A unit expressing the concentration of chemical constituents in solution as weight (milligrams) of solute per unit volume (liter) of water; equivalent to one part per million in most stream-water and ground water. One thousand micrograms per liter equals 1 mg/L.

Monitoring well— A well designed for measuring water levels and testing ground-water quality.

Mouth— The place where a stream discharges to a larger stream, a lake, or the sea.

Nitrate— An ion consisting of nitrogen and oxygen (NO_3^-). Nitrate is a plant nutrient and is very mobile in soils.

Nonpoint source— A pollution source that cannot be defined as originating from discrete points such as pipe discharge. Areas of fertilizer and pesticide applications, atmospheric deposition, manure, and natural inputs from plants and trees are types of nonpoint source pollution.

Nutrient— Element or compound essential for animal and plant growth. Common nutrients in fertilizer include nitrogen, phosphorus, and potassium.

Organochlorine pesticide— A class of organic insecticides containing a high percentage of chlorine. Includes dichlorodiphenylethanes (such as DDT), chlorinated cyclodienes (such as chlordane), and chlorinated benzenes (such as lindane). Most organochlorine insecticides were banned because of their carcinogenicity, tendency to bioaccumulate, and toxicity to wildlife.

Pesticide— A chemical applied to crops, rights of way, lawns, or residences to control weeds, insects, fungi, nematodes, rodents or other "pests".

Phosphorus— A nutrient essential for growth that can play a key role in stimulating aquatic growth in lakes and streams.

Picocurie (pCi)— One trillionth (10^{-12}) of the amount of radioactivity represented by a curie (Ci). A curie is the amount of radioactivity that yields 3.7×10^{10} radioactive disintegrations per second (dps). A picocurie yields 2.22 disintegrations per minute (dpm) or 0.037 dps.

Radon— A naturally occurring, colorless, odorless, radioactive gas formed by the disintegration of the element radium; damaging to human lungs when inhaled.

Reference site— A NAWQA sampling site selected for its relatively undisturbed conditions.

Sediment— Particles, derived from rocks or biological materials, that have been transported by a fluid or other natural process, suspended or settled in water.

Species— Populations of organisms that may interbreed and produce fertile offspring having similar structure, habits, and functions.

Streambed sediment— The material that temporarily is stationary in the bottom of a stream or other watercourse.

Study Unit— A major hydrologic system of the United States in which NAWQA studies are focused. Study Units are geographically defined by a combination of ground- and surface-water features and generally encompass more than 4,000 square miles of land area.

Surface water— An open body of water, such as a lake, river, or stream.

Suspended sediment— Particles of rock, sand, soil, and organic detritus carried in suspension in the water column, in contrast to sediment that moves on or near the streambed.

Synoptic sites— Sites sampled during a short-term investigation of specific water-quality conditions during selected seasonal or hydrologic conditions to provide improved spatial resolution for critical water-quality conditions.

Trace element— An element found in only minor amounts (concentrations less than 1.0 milligram per liter) in water or sediment; includes arsenic, cadmium, chromium, copper, lead, mercury, nickel, zinc and selenium.

Un-ionized ammonia— The neutral form of ammonia-nitrogen in water, usually occurring as NH_4OH . Un-ionized ammonia is the principal form of ammonia that is toxic to aquatic life. The relative proportion of un-ionized to ionized ammonia (NH_4^+) is controlled by water temperature and pH. At temperatures and pH values typical of most natural waters, the ionized form is dominant.

Volatile organic compounds (VOCs)— Organic chemicals that have a high vapor pressure relative to their water solubility. VOCs include components of gasoline, fuel oils, and lubricants, as well as organic solvents, fumigants, some inert ingredients in pesticides, and some by-products of chlorine disinfection.

Water-quality guidelines— Specific levels of water quality which, if reached, may adversely affect human health or aquatic life. These are nonenforceable guidelines issued by a governmental agency or other institution.

Water-quality standards— State-adopted and U.S. Environmental Protection Agency-approved ambient standards for water bodies. Standards include the use of the water body and the water-quality criteria that must be met to protect the designated use or uses.

Water year— The continuous 12-month period, October 1 through September 30, in U.S. Geological Survey reports dealing with the surface-water supply. The water year is designated by the calendar year in which it ends and which includes 9 of the 12 months. Thus, the year ending September 30, 1980, is referred to as water year 1980.

REFERENCES

- Bauch, N.J., and Spahr, N.E., 1998. Salinity trends in surface waters of the Upper Colorado River Basin, Colorado: *Journal of Environmental Quality*, v. 27, no. 3, p. 640–655.
- Bauch, N.J., and Spahr, N.E., 2000. Pesticides in surface waters of the Upper Colorado River Basin, Colorado, 1996–98: U.S. Geological Survey Water Resources Investigations Report 00–4005, 46 p.
- Bureau of the Census, 1994, 1992 Census of agriculture, Volume 1 geographic area series Part 6, Colorado State and County Data: U.S. Department of Commerce report AC92–A–6, 400 p. plus appendixes.
- Bureau of the Census, 1999, 1998 Population estimated and demographic components of change, Washington, D.C.: U.S. Department of Commerce, <http://www.census.gov/population/www/estimates>
- Butler, D.L., 1996. Trend analysis of selected water-quality data associated with salinity-control projects in the Grand Valley, in the lower Gunnison River Basin, and at Meeker Dome, western Colorado: U.S. Geological Survey Water-Resources Investigations Report 95–4274, 38 p.
- Butler, D.L., Wright, W.G., Stewart, K.C., Osmundson, B.C., Krueger, R.P. and Crabtree, D.W., 1996, Detailed study of selenium and other constituents in water, bottom sediment, soil, alfalfa, and biota associated with irrigation drainage in the Uncompahgre Project Area and in the Grand Valley, west-central Colorado, 1991–93: U.S. Geological Survey Water-Resources Investigations Report 96–4138, 136 p.
- Canadian Council of Ministers of the Environment, 1999, Canadian sediment quality guidelines for the protection of aquatic life—Summary tables, *in* Canadian environmental quality guidelines: 1999: Winnipeg, Canadian Council of Ministers of the Environment.
- Carter, L.F., and Porter, S.D., 1997, Trace-element accumulation by *Hygrohypnum ochraceum* in the Upper Rio Grande Basin, Colorado and New Mexico, USA: *Journal of Environmental Toxicology and Chemistry*, v. 16, p. 2521–2528.
- Clements, W.H., 1994, Benthic invertebrate community responses to heavy metals in the upper Arkansas River Basin, Colorado: *Journal of the North American Benthological Society* v. 13, p. 30–44.
- Clements, W.H., and Kiffney, P.M., 1995, The influence of elevation on benthic community responses to heavy metals in Rocky Mountain streams: *Canadian Journal of Fisheries and Aquatic Sciences* v. 52, p. 1966–1977.
- Colorado Climate Center, 1984, Colorado average annual precipitation 1951 to 1980: Fort Collins, Colorado Climate Center, scale 1:500,000.
- Colorado Department of Local Affairs, 2000, Preliminary population projections for Colorado Counties 1990–2025: State of Colorado, Colorado Department of Local Affairs, accessed October 4, 2000, at <http://www.dlg.oem2.state.co.us/demog/project.htm>
- Colorado Ski Country USA, 2000, Colorado Ski Country USA skier visits: Colorado Ski Country USA, accessed October 3, 2000, at http://www.skicolorado.org/media_center/skiervisits.htm
- Davis, M.W., and Streufert, R.K., 1990, Gold occurrences of Colorado: Colorado Geological Survey Resources Series 28, 101 p.
- Deacon, J.R., and Driver, N.E., 1999, Distribution of trace elements in streambed sediment associated with mining activities in the Upper Colorado River Basin, Colorado, USA, 1995–96: *Archives of Environmental Contamination and Toxicology*, v. 37, p. 7–18.
- Deacon, J.R., and Stephens, V.C., 1998, Trace elements in streambed sediment and fish liver at selected sites in the Upper Colorado River Basin, Colorado, 1995–1996: U.S. Geological Survey Water-Resources Investigations Report 98–4124, 19 p.
- Denver Water Department, 1999, Comprehensive annual financial report; Denver Water Department: 96 p.
- Engberg, R.A., 1999, Selenium budgets for Lake Powell and the Upper Colorado River Basin: *Journal of the American Water Resources Association*, v. 35, no. 4, p. 771–786.
- Environment Canada, 1999, Canadian water-quality guidelines for the protection of aquatic life, Summary tables: Environment Canada, Environmental Quality Division, Guidelines and Standards Division, accessed October 31, 2000, at [URL http://www.ec.gc.ca/ceqg-rcqe/water.htm](http://www.ec.gc.ca/ceqg-rcqe/water.htm).
- Fegeas, R.G., Claire, R.W., Guptil, S.C., Anderson, K.E., and Hallam, C.A., 1983, U.S. Geological Survey digital cartographic data standards—Land use and land cover digital data: U.S. Geological Survey Circular 895–E, 21 p.
- Fenneman, N.M., 1946, Physical divisions of the United States: Washington, D.C., U.S. Geological Survey special map, scale 1:7,000,000.
- Goodbred, S.L., Gilliom, R.J., Gross, T.S., Denslow, N.P., Bryant, W. L., and Schoeb, T.R., 1997, Reconnaissance of 17 β -estradiol, 11-ketotestosterone, vitellogenin, and gonad histopathology in common carp of United States streams—Potential for contaminant-induced endocrine disruption: U.S. Geological Survey Open-File Report 96–627, 47 p.
- Green, G.N., 1992, The digital geological map of Colorado in Arc/Info format: Denver, U.S. Geological Survey (revised April 12, 1995), Open-File Report 0507, scale 1:500,000.

- Hitt, K.J., 1995, Refining 1970's land-use data with 1990 population data to indicate new residential development: U.S. Geological Survey Water-Resources Investigations Report 94-4250, 15 p.
- Kiffney, P.M., and Clements, W.H., 1994, Structural responses of benthic macroinvertebrate communities from different stream orders to zinc: *Environmental Toxicology and Chemistry* v. 13, p. 389-395.
- Kircher, J.E., 1984, Dissolved solids in the Colorado River Basin, in U.S. Geological Survey, National water summary 1984—Hydrologic events, selected water-quality trends, and ground-water resources: U.S. Geological Survey Water-Supply Paper 2275, p. 74-78.
- Liebermann, T.D., Mueller, D.K., Kircher, J.E., and Choquette, A.F., 1989, Characteristics and trends of streamflow and dissolved solids in the Upper Colorado River Basin, Arizona, Colorado, New Mexico, Utah, and Wyoming: U.S. Geological Survey Water-Supply Paper 2358, 64 p.
- Mayland, H.F., 1994, Selenium in plant and animal nutrition, in Frankenberger, W.T., Jr., and Benson, Salley, eds., Selenium in the environment: New York, Marcel Dekker, Inc. p. 29-45.
- Myers, D.N., and Sylvester, M.A., 1997, Fecal indicator bacteria: U.S. Geological Survey Techniques of Water-Resources Investigations, book 9, chap. A7, 38 p.
- Northern Colorado Water Conservancy District, 2000, Colorado Big Thompson Project: Northern Colorado Water Conservancy District, accessed September 11, 2000, at <http://www.ncwcd.org/cgi-bin/ncwcd?go=CBT>
- Ohlendorf, H.M., Hoffman, D.J., Saiki, M.K., and Aldrich, T.W., 1986, Embryonic mortality and abnormalities of aquatic birds—Apparent impacts of selenium from irrigation drainwater: *Science of the Total Environment*, v. 52, p. 49-63.
- Rice, K.C., 1999, Trace-element concentrations in streambed sediment across the conterminous United States: *Environmental Science and Technology*, v. 33, no. 15, p. 2499-2504.
- Seiler, R. L., Skorupa, J.P., and Peltz, L.A., 1999, Areas susceptible to irrigation-induced selenium contamination of water and biota in the Western United States: U.S. Geological Survey Circular 1180, 36 p.
- Squillace, P.J., Pankow, J.F., Korte, N.E., and Zogorski, J.S., 1996, Environmental behavior and fate of methyl *tert*-butyl ether (MTBE): U.S. Geological Survey Fact Sheet FS-203-96, 6 p.
- Stephens, V.C., and Deacon, J.R., 1998, Distribution and concentrations of selected organochlorine pesticides and PCB's in streambed sediment and whole-body fish in the Upper Colorado River Basin, 1995-96: U.S. Geological Survey Fact Sheet FS-167-97, 6 p.
- Tweto, Ogden, comp., 1979, Geologic map of Colorado: U.S. Geological Survey State Geologic Map, scale 1:500,000 (reprinted).
- U.S. Department of the Interior, 1995, Quality of water, Colorado River Basin—Progress report no. 17: U.S. Department of the Interior, 96 p.
- U.S. Department of the Interior, 1999, Quality of water—Colorado River Basin: Denver, Colo., Bureau of Reclamation Progress Report 19, 93 p. plus appendix.
- U.S. Environmental Protection Agency, 1996, Drinking water regulations and health advisories: Washington, D.C., U.S. Environmental Protection Agency Report 822-B-96-002, Office of Water [pagination unknown].
- U.S. Environmental Protection Agency, 1997, Drinking water advisory: Consumer acceptability and health effects analysis on methyl *tert*-butyl ether (MTBE): Washington D.C., U.S. Environmental Protection Agency Report 822-F-97-008, Office of Water, 4 p.
- U.S. Environmental Protection Agency, rev. October 18, 1999, Proposed radon in drinking water rule: U.S. Environmental Protection Agency Report EPA 815-F-99-006: accessed March 6, 2000, at <http://www.epa.gov/OGWDW/radon/fact.html>
- U.S. Geological Survey, 1995, Water use in the United States: U.S. Geological Survey, accessed March 6, 2000, at <http://water.usgs.gov/wateruse/>
- U.S. Geological Survey, 1999, The quality of our Nation's waters—Nutrients and pesticides: U.S. Geological Survey Circular 1225, 82 p.
- Upper Colorado River Commission, 1999, Fifty-first annual report of the Upper Colorado River Commission, September 30, 1999: Salt Lake City, Utah, 92 p.
- Vaill, J.E., and Butler, D.L., 1999, Streamflow and dissolved-solids trends, through 1996, in the Colorado River Basin upstream from Lake Powell—Colorado, Utah, and Wyoming: U.S. Geological Survey Water-Resources Investigations Report 99-4097, 47 p.

Additional Publications of the UCOL Study Unit:

- Apodaca, L.E., 1997, Analysis of ground-water-quality data of the Upper Colorado River Basin, water years 1972-92: U.S. Geological Survey Water-Resources Investigations Report 97-4240, 31 p.
- Apodaca, L.E., and Bails, J.B. 1999, Fraser River Watershed, Colorado—Assessment of available water-quantity and water-quality data through water year 1997: U.S. Geological Survey Water-Resources Investigations Report 98-4255, 58 p.
- Apodaca, L.E., and Bails, J.B., 2000, Water quality in alluvial aquifers of the Southern Rocky Mountains Physiographic Province, Upper Colorado River Basin, Colorado, 1997: U.S. Geological Survey Water-Resources Investigations Report 99-4222, 68 p.

- Apodaca, L.E., and Bails, J.B., 2000, Ground-water-quality assessment of selected wells in the Fraser River Watershed, Colorado, August 1998: U.S. Geological Survey Water-Resources Investigations Report 99-4243, 8 p.
- Apodaca, L.E., Driver, N.E., and Bails, J.B., 2000, Occurrence, transport, and fate of trace elements, Blue River Basin, Summit County, Colorado—An integrated approach: *Environmental Geology*, v. 39, no.8, p. 901-913.
- Apodaca, L.E., Driver, N.E., Stephens, V.C., and Spahr, N.E., 1996, Environmental setting and implications on water quality, Upper Colorado River Basin, Colorado and Utah: U.S. Geological Survey Water-Resources Investigations Report 95-4263, 33 p.
- Apodaca, L.E., Stephens, V.C., and Driver, N.E., 1996, What affects water quality in the Upper Colorado River Basin: U.S. Geological Survey Fact Sheet FS-109-96, 4 p.
- Bauch, N.J. and Apodaca, L.E., 1995, Bibliography, indices, and data sources of water-related studies, Upper Colorado River Basin, Colorado and Utah, 1972-1995: U.S. Geological Survey Open-File Report 95-450, 282 p.
- Deacon, J.R., 1997, Biological and contaminant investigations, Upper Colorado River Basin—What's been done and how can the information be used?: U.S. Geological Survey Fact Sheet FS-032-97, 2 p.
- Deacon, J.R., and Mize, S.V., 1997, Effects of water quality and habitat on composition of fish communities in the Upper Colorado River Basin: U.S. Geological Survey Fact Sheet FS-122-97, 6 p.
- Deacon, J.R., Mize, S.V., and Spahr, N.E., 1999, Characterization of selected biological, chemical, and physical conditions at fixed sites in the Upper Colorado River Basin, Colorado, 1995-98: U.S. Geological Survey Water-Resources Investigations Report 99-4181, 35 p.
- Deacon, J.R., and Spahr, N.E., 1998, Water-quality and biological community characterization at selected sites on the Eagle River, Colorado, September 1997 and February 1998: U.S. Geological Survey Water-Resources Investigations Report 98-4236, 8 p.
- Deacon, J.R., and Stephens, V.C., 1996, Summary of biological and contaminant investigations related to stream water quality and environmental setting in the Upper Colorado River Basin, 1938-1995: U.S. Geological Survey Water-Resources Investigations Report 96-4172, 37 p.
- Driver, N.E., 1994, National Water-Quality Assessment Program—Upper Colorado River Basin: U.S. Geological Survey Open-File Report 94-102, 2 p.
- Greve, A.I., Spahr, N.E., Van Metre, P.C., and Wilson, J.T., in press, Identification of water-quality trends using sediment cores from Dillon Reservoir, Summit County, Colorado: U.S. Geological Survey Water-Resources Investigations Report 01-4022.
- Spahr, N.E., and Boulger, R.W., 1997, Interim results of quality-control sampling of surface water for the Upper Colorado River National Water-Quality Assessment Study Unit, water years 1995-96: U.S. Geological Survey Water-Resources Investigations Report 97-4227, 34 p.
- Spahr, N.E., Boulger, R.W., and Szmajter, R.J., 2000, Water quality at basic fixed sites in the Upper Colorado River Basin National Water-Quality Assessment Study Unit, October 1995-September 1998: U.S. Geological Survey Water-Resources Investigations Report 99-4223, 63 p.
- Spahr, N.E., and Deacon, J.R., 1998, Water-quality characterization of the Slate and East Rivers during the winter recreational season, December, 1996: U.S. Geological Survey Open-File Report 98-279, 9 p.
- Spahr, N.E., Driver, N.E., and Stephens, V.C., 1996, The Upper Colorado River National Water-Quality Assessment Program Surface-Water Monitoring Network: U.S. Geological Survey Fact Sheet FS-191-96, 4 p.
- Spahr, N.E., and Wynn, K.H., 1997, Nitrogen and phosphorus in surface waters of the Upper Colorado River Basin: *Journal of the American Water Resources Association*, v. 33, no. 3, p. 547-560.

APPENDIX—WATER-QUALITY DATA FROM THE UPPER COLORADO RIVER BASIN IN A NATIONAL CONTEXT

For a complete view of Upper Colorado River Basin data and for additional information about specific benchmarks used, visit our Web site at <http://water.usgs.gov/nawqa/>. Also visit the NAWQA Data Warehouse for access to NAWQA data sets at <http://infotrek.er.usgs.gov/wdbctx/nawqa/nawqa.home>.

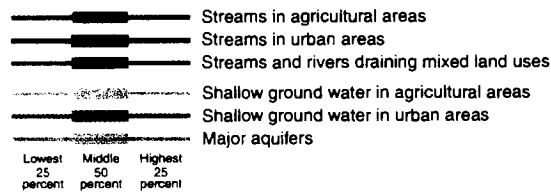
This appendix is a summary of chemical concentrations and biological indicators assessed in the Upper Colorado River Basin. Selected results for this Basin are graphically compared to results from as many as 36 NAWQA Study Units investigated from 1991 to 1998 and to national water-quality benchmarks for human health, aquatic life, or fish-eating wildlife. The chemical and biological indicators shown were selected on the basis of frequent detection, detection at concentrations above a national benchmark, or regulatory or scientific importance. The graphs illustrate how conditions associated with each land use sampled in the Upper Colorado River Basin compare to results from across the Nation, and how conditions compare among the several land uses. Graphs for chemicals show only detected concentrations and, thus, care must be taken to evaluate detection frequencies in addition to concentrations when comparing study-unit and national results. For example, simazine concentrations in Upper Colorado River Basin agricultural streams were similar to the national distribution, but the detection frequency was much higher (72 percent compared to 18 percent).

CHEMICALS IN WATER

Concentrations and detection frequencies, Upper Colorado River Basin, 1996–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals

- ◆ Detected concentration in Study Unit
- 66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- .. Not measured or sample size less than two
- 12 Study-unit sample size. For ground water, the number of samples is equal to the number of wells sampled

National ranges of detected concentrations, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

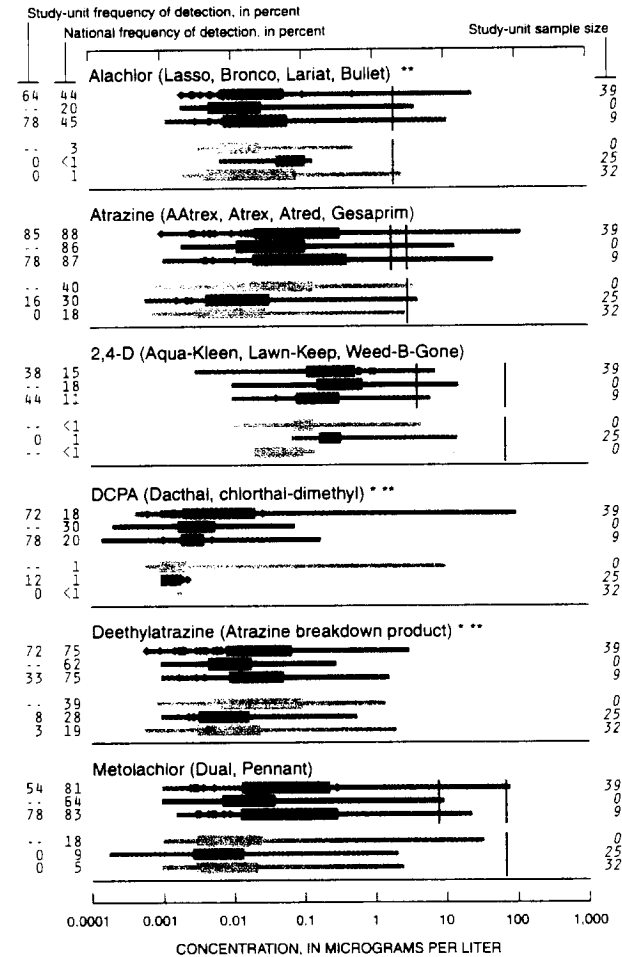


National water-quality benchmarks

National benchmarks include standards and guidelines related to drinking-water quality, criteria for protecting the health of aquatic life, and a goal for preventing stream eutrophication due to phosphorus. Sources include the U.S. Environmental Protection Agency and the Canadian Council of Ministers of the Environment

- | Drinking-water quality (applies to ground water and surface water)
- | Protection of aquatic life (applies to surface water only)
- | Prevention of eutrophication in streams not flowing directly into lakes or impoundments
- No benchmark for drinking-water quality
- .. No benchmark for protection of aquatic life

Pesticides in water—Herbicides



Other herbicides detected

- Acetochlor (Harness Plus, Surpass) * **
- Benfluralin (Balan, Benefin, Bonalan) * **
- Bentazon (Basagran, Bentazone) **
- Bromacil (Hyvar X, Urox B, Bromax)
- Cyanazine (Bladex, Fortrol)
- 2,4-DB (Butyrac, Butoxone, Embutox Plus, Embutone) * **
- Dicamba (Banvel, Dianat, Scotts Proturf)
- EPTC (Eptam, Farmarox, Allrox) * **
- Ethalfuralin (Sonalan, Curbit) * **
- Oryzalin (Surlan, Dirimal) * **
- Prometon (Pramitol, Princep) **
- Simazine (Princep, Caliber 90)
- Tebuthiuron (Spike, Tebusan)
- Trifluralin (Treflan, Gowan, Tri-4, Trific)

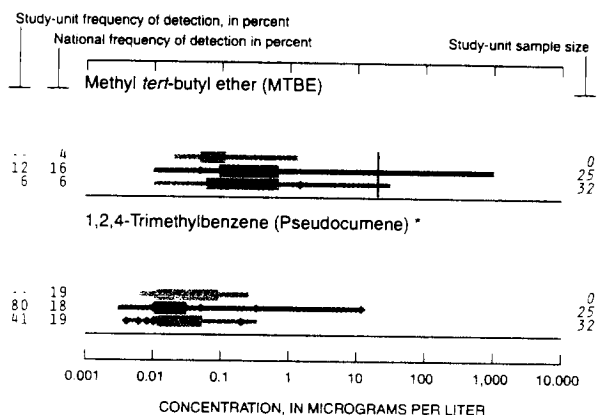
Herbicides not detected

- Acifluorfen (Blazer, Tackle 2S) **
- Bromoxynil (Buctril, Brominal) *
- Butylate (Sutan +, Genate Plus, Butilate) **
- Chloramben (Amiben, Amilon-WP, Vegiben) **
- Clopyralid (Stinger, Lontrel, Transline) * **
- Dacthal mono-acid (Dacthal breakdown product) * **
- Dichlorprop (2,4-DP, Seritox 50, Lentemul) * **
- 2,6-Diethylaniiline (Alachlor breakdown product) * **
- Dinoseb (Dinoseb)
- Diuron (Crisuron, Karmex, Diurex) **

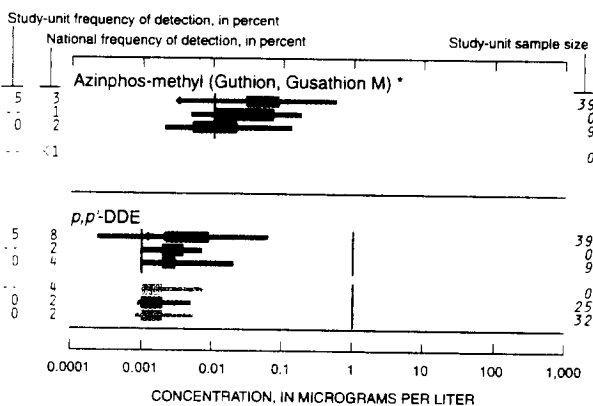
Fenuron (Fenulon, Fenidim) * **
 Fluometuron (Flo-Met, Cotoran) **
 Linuron (Lorox, Linex, Sarclex, Linurex, Afaion) *
 MCPA (Rhomene, Rhonox, Chiptox)
 MCPB (Thistro) * **
 Metribuzin (Lexone, Sencor)
 Molinate (Ordram) * **
 Napropamide (Devrinol) * **
 Neburon (Neburea, Neburyl, Noruben) * **
 Norflurazon (E vital, Predict, Solicam, Zorial) * **
 Pebulate (Tillam, PEBC) * **
 Pendimethalin (Pre-M, Prowl, Stomp) * **
 Picloram (Grazon, Tordon)
 Pronamide (Kerb, Propyzamid) **
 Propachlor (Flamrod, Satecid) **
 Propanil (Stam, Stampede, Wham) * **
 Propham (Tuberite) **
 2,4,5-T **
 2,4,5-TP (Silvex, Fenoprop) **
 Terbacil (Sinbar) **
 Thiobencarb (Bolero, Saturn, Benthicarb) * **
 Triallate (Far-Go, Avadex BW, Tri-allate) *
 Triclopyr (Garlon, Grandstand, Redeem, Remedy) * **

Volatile organic compounds (VOCs) in ground water

These graphs represent data from 16 Study Units, sampled from 1996 to 1998



Pesticides in water—Insecticides



Other insecticides detected

Carbaryl (Carbamine, Denapon, Sevin)
 Carbofuran (Furadan, Curaterr, Yaltox)
 Chlorpyrifos (Brodan, Dursban, Lorsban)
 Diazinon (Basudin, Diazatol, Neocidol, Knox Out)
 gamma-HCH (Lindane, gamma-BHC)
 Malathion (Malathion)
cis-Permethrin (Ambush, Astro, Pounce) * **
 Phorate (Thimet, Granutox, Geomet, Rampart) * **
 Propargite (Comite, Omite, Ornamate) * **
 Terbufos (Contraven, Counter, Pilarfox) **

Insecticides not detected

Aldicarb (Temik, Ambush, Pounce)
 Aldicarb sulfone (Standak, aldoxycarb)
 Aldicarb sulfoxide (Aldicarb breakdown product)
 Dieldrin (Panoram D-31, Octalox, Compound 497)
 Disulfoton (Disyston, Di-Syston) **
 Ethoprop (Mocap, Ethoprophos) * **
 Fonofos (Dyfonate, Capfos, Cudgel, Tycap) **
 alpha-HCH (alpha-BHC, alpha-lindane) **
 3-Hydroxycarbofuran (Carbofuran breakdown product) * **
 Methiocarb (Slug-Geta, Grandslam, Mesuro) * **
 Methomyl (Lanox, Lannate, Acinate) **
 Methyl parathion (Pennacap-M, Folidol-M) **
 Oxamyl (Vydate L, Pratt) **
 Parathion (Roethyl-P, Alkron, Panthion, Phoskii) *
 Propoxur (Baygon, Blattanex, Unden, Proprotol) * **

Other VOCs detected

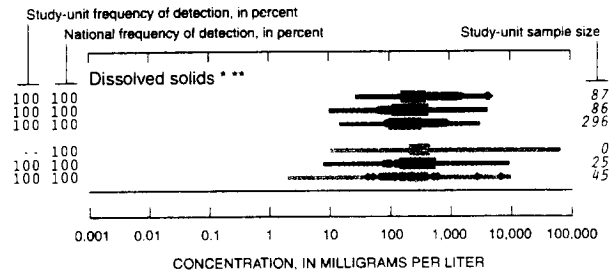
Benzene
 Bromodichloromethane (Dichlorobromomethane)
 2-Butanone (Methyl ethyl ketone (MEK)) *
n-Butylbenzene (1-Phenylbutane) *
sec-Butylbenzene *
 Carbon disulfide *
 Chloroethane (Ethyl chloride) *
 Chloromethane (Methyl chloride)
 1,3-Dichlorobenzene (*m*-Dichlorobenzene)
 Dichlorodifluoromethane (CFC 12, Freon 12)
 Dichloromethane (Methylene chloride)
 Diethyl ether (Ethyl ether) *
 1,2-Dimethylbenzene (*o*-Xylene)
 1,3 & 1,4-Dimethylbenzene (*m*-&*p*-Xylene)
 1-4-Epoxy butane (Tetrahydrofuran, Diethylene oxide) *
 Ethenylbenzene (Styrene)
 1-Ethyl-2-methylbenzene (2-Ethyltoluene) *
 Ethylbenzene (Phenylethane)
 Iodomethane (Methyl iodide) *
 Isopropylbenzene (Cumene) *
p-Isopropyltoluene (*p*-Cymene) *
 4-Methyl-2-pentanone (Methyl isobutyl ketone (MIBK)) *
 Methylbenzene (Toluene)
 Naphthalene
 2-Propanone (Acetone) *
n-Propylbenzene (Isocumene) *
 Tetrachloroethene (Perchloroethene)
 Tetrachloromethane (Carbon tetrachloride)
 1,2,3,5-Tetramethylbenzene (Isodurene) *
 1,1,1-Trichloroethane (Methylchloroform)
 Trichloroethene (TCE)
 Trichloromethane (Chloroform)
 1,2,3-Trimethylbenzene (Hemimellitene) *
 1,3,5-Trimethylbenzene (Mesitylene) *

VOCs not detected

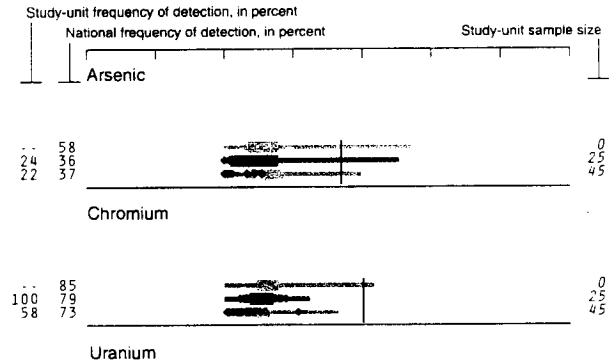
tert-Amyl methyl ether (*tert*-amyl methyl ether (TAME)) *
 Bromobenzene (Phenyl bromide) *
 Bromochloromethane (Methylene chlorobromide)
 Bromoethene (Vinyl bromide) *
 Bromomethane (Methyl bromide)
tert-Butylbenzene *
 3-Chloro-1-propene (3-Chloropropene) *
 1-Chloro-2-methylbenzene (*o*-Chlorotoluene)
 1-Chloro-4-methylbenzene (*p*-Chlorotoluene)
 Chlorobenzene (Monochlorobenzene)
 Chlorodibromomethane (Dibromochloromethane)
 Chloroethene (Vinyl chloride)
 1,2-Dibromo-3-chloropropane (DBCP, Nemagon)
 1,2-Dibromoethane (Ethylene dibromide, EDB)
 Dibromomethane (Methylene dibromide) *
trans-1,4-Dichloro-2-butene ((*Z*)-1,4-Dichloro-2-butene) *
 1,2-Dichlorobenzene (*o*-Dichlorobenzene)
 1,4-Dichlorobenzene (*p*-Dichlorobenzene)

- 1,2-Dichloroethane (Ethylene dichloride)
- 1,1-Dichloroethane (Ethylidene dichloride) *
- 1,1-Dichloroethene (Vinylidene chloride)
- trans*-1,2-Dichloroethene ((E)-1,2-Dichloroethene)
- cis*-1,2-Dichloroethene ((Z)-1,2-Dichloroethene)
- 1,2-Dichloropropane (Propylene dichloride)
- 2,2-Dichloropropane *
- 1,3-Dichloropropane (Trimethylene dichloride) *
- trans*-1,3-Dichloropropene ((E)-1,3-Dichloropropene)
- cis*-1,3-Dichloropropene ((Z)-1,3-Dichloropropene)
- 1,1-Dichloropropene *
- Diisopropyl ether (Diisopropylether (DIPE)) *
- Ethyl methacrylate *
- Ethyl *tert*-butyl ether (Ethyl-*t*-butyl ether (ETBE)) *
- Hexachlorobutadiene
- 1,1,1,2,2,2-Hexachloroethane (Hexachloroethane)
- 2-Hexanone (Methyl butyl ketone (MBK)) *
- Methyl acrylonitrile *
- Methyl-2-methacrylate (Methyl methacrylate) *
- Methyl-2-propenoate (Methyl acrylate) *
- 2-Propenenitrile (Acrylonitrile)
- 1,1,2,2-Tetrachloroethane *
- 1,1,1,2-Tetrachloroethane
- 1,2,3,4-Tetramethylbenzene (Prenhitene) *
- Tribromomethane (Bromofom)
- 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon 113) *
- 1,2,4-Trichlorobenzene
- 1,2,3-Trichlorobenzene *
- 1,1,2-Trichloroethane (Vinyl trichloride)
- Trichlorofluoromethane (CFC 11, Freon 11)
- 1,2,3-Trichloropropane (Allyl trichloride)

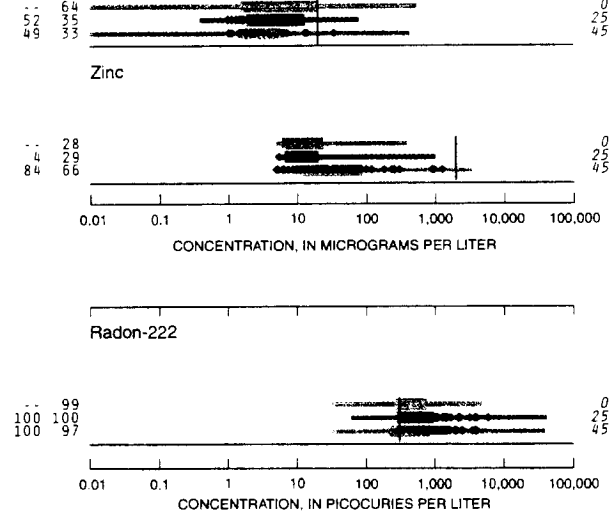
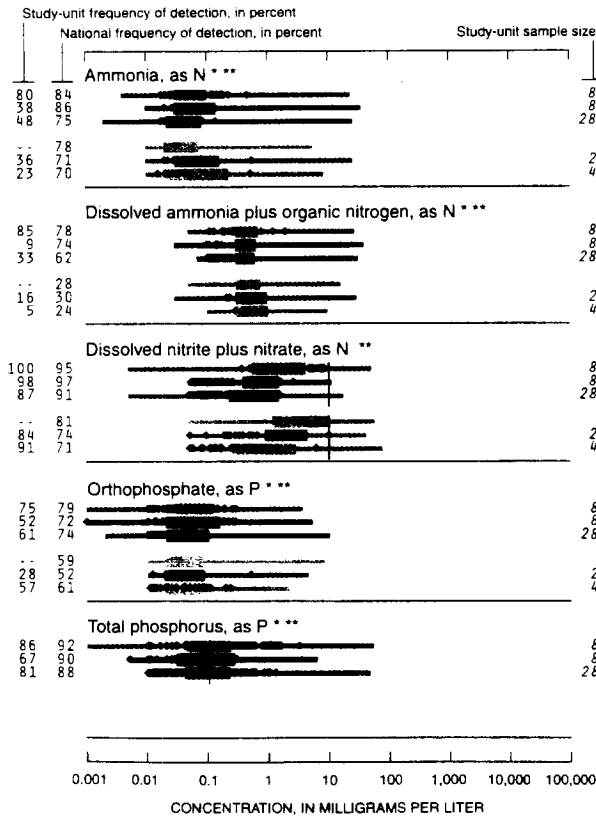
Dissolved solids in water



Trace elements in ground water



Nutrients in water



- Other trace elements detected**
- Lead
- Selenium
- Trace elements not detected**
- Cadmium

CHEMICALS IN FISH TISSUE AND BED SEDIMENT

Concentrations and detection frequencies, Upper Colorado River Basin, 1996-98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals.

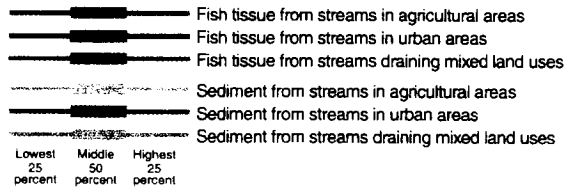
Study-unit frequencies of detection are based on small sample sizes; the applicable sample size is specified in each graph

- ◆ Detected concentration in Study Unit

66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency

- Not measured or sample size less than two
- 12 Study-unit sample size

National ranges of concentrations detected, by land use, in 36 NAWQA Study Units, 1991-98—Ranges include only samples in which a chemical was detected

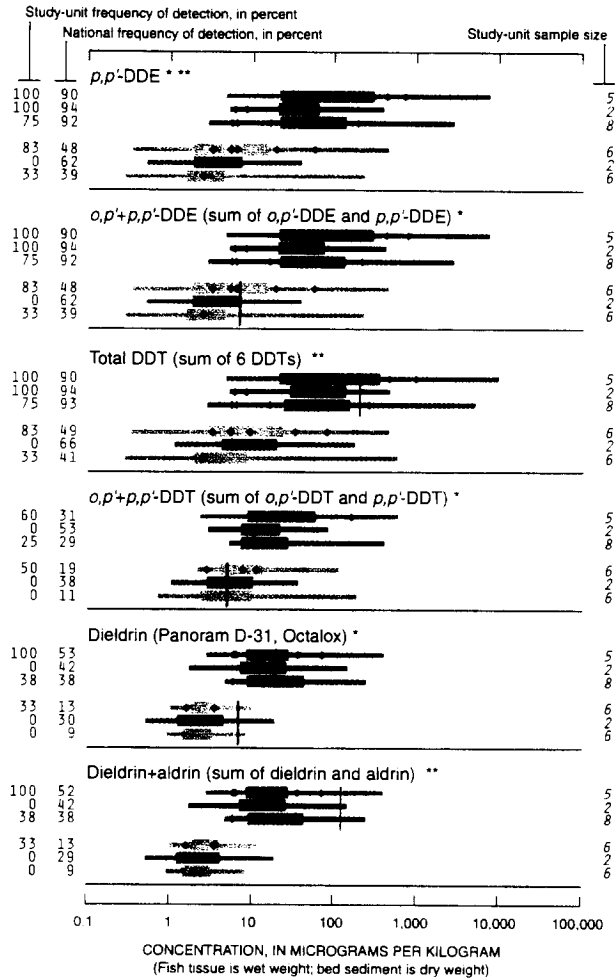
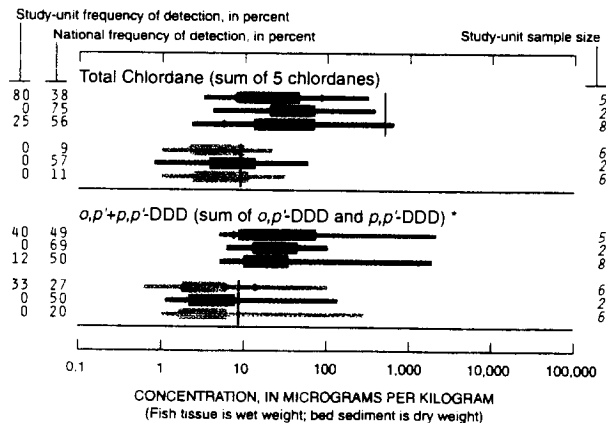


National benchmarks for fish tissue and bed sediment

National benchmarks include standards and guidelines related to criteria for protection of the health of fish-eating wildlife and aquatic organisms. Sources include the U.S. Environmental Protection Agency, other Federal and State agencies, and the Canadian Council of Ministers of the Environment

- | Protection of fish-eating wildlife (applies to fish tissue)
- | Protection of aquatic life (applies to bed sediment)
- No benchmark for protection of fish-eating wildlife
- ** No benchmark for protection of aquatic life

Organochlorines in fish tissue (whole body) and bed sediment



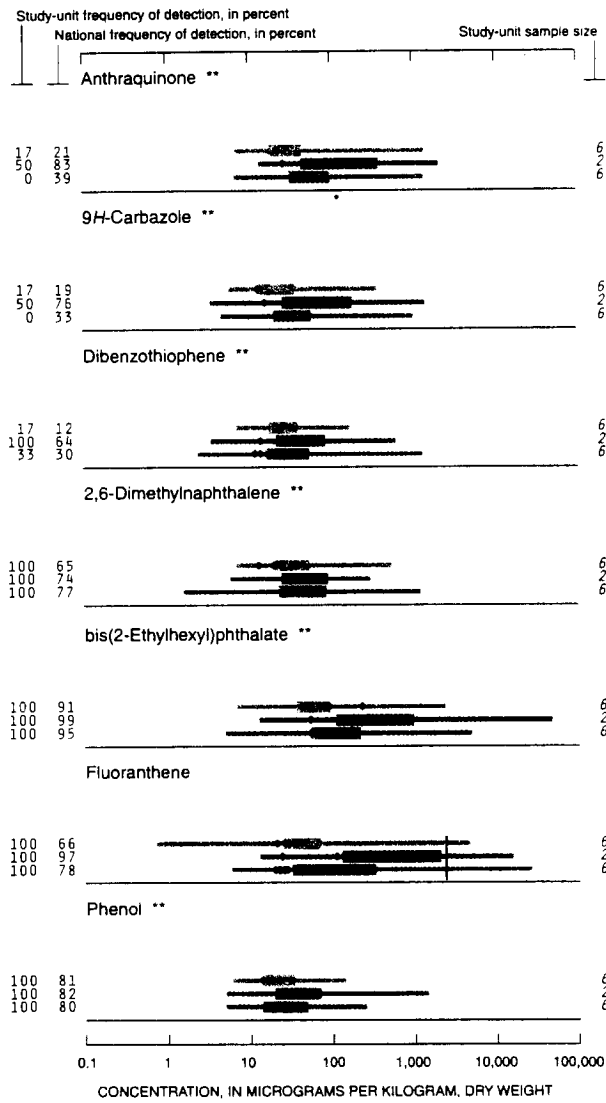
Other organochlorines detected

- DCPA (Dacthal, chlorthal-dimethyl) ***
- Heptachlor epoxide (Heptachlor breakdown product) *
- Heptachlor+heptachlor epoxide (sum of heptachlor and heptachlor epoxide) **
- Hexachlorobenzene (HCB) **
- Toxaphene (Camphechlor, Hercules 3956) ***

Organochlorines not detected

- Chloroneb (Chloronebe, Demosan) ***
- Endosulfan I (alpha-Endosulfan, Thiodan) **
- Endrin (Endrine)
- gamma-HCH (Lindane, gamma-BHC, Gammexane) *
- Total-HCH (sum of alpha-HCH, beta-HCH, gamma-HCH, and delta-HCH) **
- Isodrin (Isodrine, Compound 711) ***
- p,p'-Methoxychlor (Marlate, methoxychlor) **
- o,p'-Methoxychlor ***
- Mirex (Dechlorane) **
- Total PCB
- Pentachloroanisole (PCA) ***
- cis-Permethrin (Ambush, Astro, Pounce) ***
- trans-Permethrin (Ambush, Astro, Pounce) ***

**Semivolatile organic compounds (SVOCs)
in bed sediment**



Other SVOCs detected

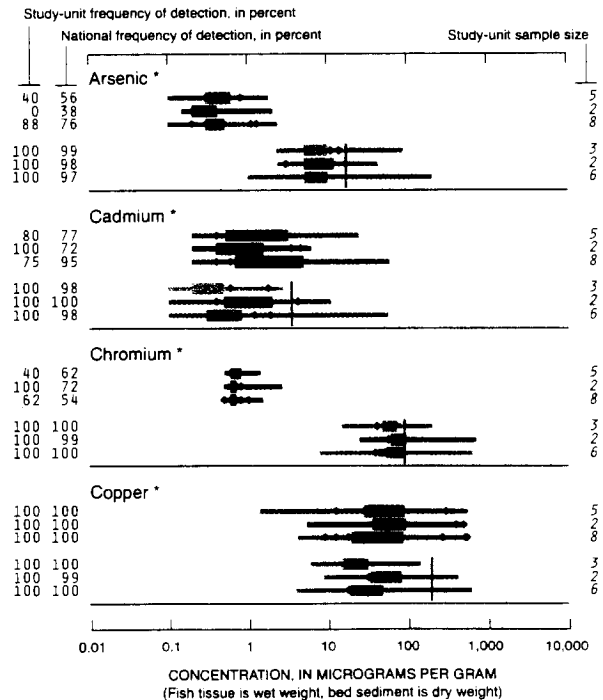
- Acridine **
- Anthracene
- Benzo[a]anthracene
- Benzo[a]pyrene
- Benzo[b]fluoranthene **
- Benzo[ghi]perylene **
- Benzo[k]fluoranthene **
- Butylbenzylphthalate **
- Chrysene
- p-Cresol **
- Di-n-butylphthalate **
- Di-n-octylphthalate **
- Dibenz[a,h]anthracene
- Diethylphthalate **
- 1,2-Dimethylnaphthalene **
- 1,6-Dimethylnaphthalene **
- Dimethylphthalate **
- 2-Ethylphthalate **
- 9H-Fluorene (Fluorene)
- Indeno[1,2,3-cd]pyrene **

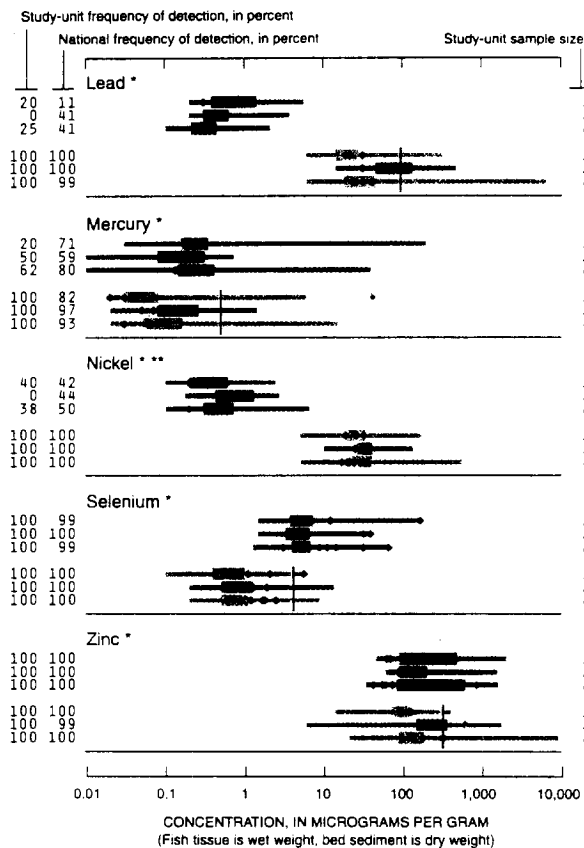
- 1-Methyl-9H-fluorene **
- 2-Methylanthracene **
- 4,5-Methylenepheneanthrene **
- 1-Methylphenanthrene **
- 1-Methylpyrene **
- Naphthalene
- Pentachloronitrobenzene **
- Phenanthrene
- Pyrene
- 2,3,6-Trimethylnaphthalene **

SVOCs not detected

- Acenaphthene
- Acenaphthylene
- C8-Alkylphenol **
- Azobenzene **
- Benzo[c]cinnoline **
- 2,2-Biquinoline **
- 4-Bromophenyl-phenylether **
- 4-Chloro-3-methylphenol **
- bis(2-Chloroethoxy)methane **
- 2-Chloronaphthalene **
- 2-Chlorophenol **
- 4-Chlorophenyl-phenylether **
- 1,2-Dichlorobenzene (o-Dichlorobenzene) **
- 1,3-Dichlorobenzene (m-Dichlorobenzene) **
- 1,4-Dichlorobenzene (p-Dichlorobenzene) **
- 3,5-Dimethylphenol **
- 2,4-Dinitrotoluene **
- Isophorone **
- Isoquinoline **
- Nitrobenzene **
- N-Nitrosodi-n-propylamine **
- N-Nitrosodiphenylamine **
- Phenanthridine **
- Quinoline **
- 1,2,4-Trichlorobenzene **

**Trace elements in fish tissue (livers) and
bed sediment**





BIOLOGICAL INDICATORS

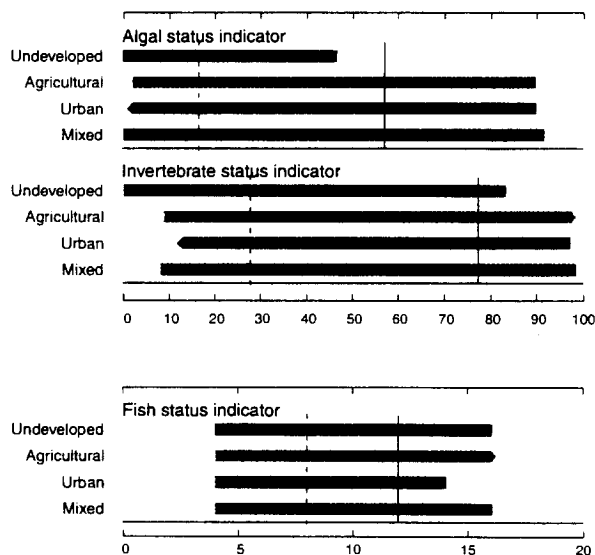
Higher national scores suggest habitat disturbance, water-quality degradation, or naturally harsh conditions. The status of algae, invertebrates (insects, worms, and clams), and fish provide a record of water-quality and stream conditions that water-chemistry indicators may not reveal. **Algal status** focuses on the changes in the percentage of certain algae in response to increasing siltation, and it often correlates with higher nutrient concentrations in some regions. **Invertebrate status** averages 11 metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. **Fish status** sums the scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that increase in association with water-quality degradation

Biological indicator value, Upper Colorado River Basin, by land use, 1996-98

- ◆ Biological status assessed at a site

National ranges of biological indicators, in 16 NAWQA Study Units, 1994-98

- Streams in undeveloped areas
- Streams in agricultural areas
- Streams in urban areas
- Streams in mixed-land-use areas
- 75th percentile
- - - 25th percentile



A COORDINATED EFFORT

Coordination with agencies and organizations in the Upper Colorado River Basin Study Unit was integral to the success of this water-quality assessment. We thank those who served as members of our liaison committee.

Federal Agencies

Bureau of Land Management
Bureau of Reclamation
National Park Service-Curecanti National Recreation Area
National Park Service
Natural Resources Conservation Service
U.S. Bureau of Mines
U.S. Environmental Protection Agency
U.S. Fish and Wildlife Service
U.S. Forest Service

State Agencies

Colorado Department of Public Health and Environment
Colorado Division of Minerals and Geology
Colorado Division of Wildlife
Colorado Geological Survey
Colorado River Basin Salinity Control Forum
Colorado River Water Conservation District
Colorado Water Conservation Board
Eagle County
Eagle River Water and Sanitation District
East Grand Water Quality Board
Grand County Water & Sanitation District
Grand Junction Drainage District
Grand County Commissioners
Denver Water

Local Agencies

City of Aurora
City of Gunnison
City of Grand Junction
Clifton Water District
Crested Butte South Metropolitan District
Gunnison County
Mesa County Water Association
Mt. Crested Butte Water and Sanitation District
Northern Colorado Water Conservancy District
Northwest Colorado Council of Governments
Silvercreek Water and Sanitation District
Summit County Water Quality Committee
Town of Crested Butte
Town of Fraser
Town of Granby
Town of Silverthorne
Town of Winter Park
Town of Vail
Uncompahgre Valley Water Users Association
Upper Gunnison River Water Conservancy District

Universities

Colorado State University Cooperative Extension

Other public and private organizations

Colorado Water Resources Research Institute
Gunnison Basin POWER
Sierra Club
Vail Associates

We thank the following individuals for contributing to this effort.

Bob Boulger, lead technician.

Jeff Foster, Dave Hartle, and Dennis Smits for data collection.

Adrienne Greve, Jason Gurdak, Scott Mize, Cory Stephens, and Rick Szmajter for data collection and compilation.

Earl and Frances Partch, Virgil and Lee Spann Ranches Inc., Dos Rios Golf Course, and Maryvale Associates for allowing us to drill monitoring wells on their property.

Gail Cordy, Tyler Martineau, Sarah Ryker, and Edward Wang for review of manuscript.

Mary Kidd and Betty Palcsak for editorial review.

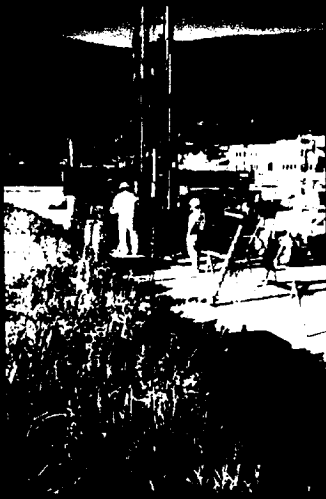
Alene Brogan for manuscript layout.

John Evans and Loretta Ulibarri for illustrations.

NAWQA

National Water-Quality Assessment (NAWQA) Program
Upper Colorado River Basin

Spaht and others—Water Quality in the Upper Colorado River Basin
U.S. Geological Survey Circular 1214



ISBN 0-607-95424-8

9 780607 95424 1

R0024666

Water Quality in the Sacramento River Basin California, 1994-98



U.S. Department of the Interior
U.S. Geological Survey

Circular 1215

R0024667

POINTS OF CONTACT AND ADDITIONAL INFORMATION

The companion Web site for NAWQA summary reports:

<http://water.usgs.gov/nawqa/>

Sacramento River Basin contact and Web site:

Joseph Domagalski
U.S. Geological Survey
Water Resources Division
Placer Hall, 6000 J Street,
Sacramento, CA 95819-6129
e-mail: joed@usgs.gov
http://water.wr.usgs.gov/sac_nawqa/index.html

National NAWQA Program:

Chief, NAWQA Program
U.S. Geological Survey
Water Resources Division
12201 Sunrise Valley Drive, MS 413
Reston, VA 20192
<http://water.usgs.gov/nawqa/>

Other NAWQA summary reports

River Basin Assessments

Albemarle-Pamlico Drainage Basin (Circular 1157)
Allegheny and Monongahela River Basins (Circular 1202)
Apalachicola-Chattahoochee-Flint River Basin (Circular 1164)
Central Arizona Basins (Circular 1213)
Central Columbia Plateau (Circular 1144)
Central Nebraska Basins (Circular 1163)
Connecticut, Housatonic, and Thames River Basins (Circular 1155)
Eastern Iowa Basins (Circular 1210)
Georgia-Florida Coastal Plain (Circular 1151)
Hudson River Basin (Circular 1165)
Kanawha-New River Basins (Circular 1204)
Lake Erie-Lake Saint Clair Drainages (Circular 1203)
Las Vegas Valley Area and the Carson and Truckee River Basins
(Circular 1170)
Lower Illinois River Basin (Circular 1209)
Long Island-New Jersey Coastal Drainages (Circular 1201)
Lower Susquehanna River Basin (Circular 1168)
Mississippi Embayment (Circular 1208)
Ozark Plateaus (Circular 1158)
Potomac River Basin (Circular 1166)
Puget Sound Basin (Circular 1216)

Red River of the North Basin (Circular 1169)
Rio Grande Valley (Circular 1162)
San Joaquin-Tulare Basins (Circular 1159)
Santee River Basin and Coastal Drainages
(Circular 1206)
South-Central Texas (Circular 1212)
South Platte River Basin (Circular 1167)
Southern Florida (Circular 1207)
Trinity River Basin (Circular 1171)
Upper Colorado River Basin (Circular 1214)
Upper Mississippi River Basin (Circular 1211)
Upper Snake River Basin (Circular 1160)
Upper Tennessee River Basin (Circular 1205)
Western Lake Michigan Drainages (Circular 1156)
White River Basin (Circular 1150)
Willamette Basin (Circular 1161)

National Assessments

The Quality of Our Nation's Waters—Nutrients and
Pesticides (Circular 1225)

Front cover: Aerial view of the Sacramento River in the Sacramento Valley (*Rand Schaal, Ph.D.*, pilot and photographer).

Back cover: Left, Collection of water samples during high flow conditions on the Sacramento River; right, Collection of samples of aquatic organisms and assessment of habitat conditions at a northern California headwaters stream.

Water Quality in the Sacramento River Basin, California, 1994–98

By Joseph L. Domagalski,¹ Donna L. Knifong,¹ Peter D. Dileanis,¹ Larry R. Brown,¹ Jason T. May,² Valerie Connor,³ and Charles N. Alpers¹

¹U.S. Geological Survey, Placer Hall, 6000 J Street, Sacramento, CA 95819-6129

²California State University Foundation, 6000 J Street, Sacramento, CA 95819-6063

³Central Valley Regional Water Quality Control Board, 3443 Routier Road, Sacramento, CA 95827-3098

U.S. GEOLOGICAL SURVEY CIRCULAR 1215

R0024669

U.S. DEPARTMENT OF THE INTERIOR
BRUCE BABBITT, Secretary

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

2000

Free on application to the
U.S. Geological Survey
Information Services
Box 25286 Federal Center
Denver, CO 80225

Or call: 1-888-ASK-USGS

Library of Congress Cataloging-in-Publication's Data

Water-quality assessment of the Sacramento River Basin, California, 1994-98 / by Joseph L. Domagalski...[et al.].

p. cm. -- (U.S. Geological Survey Circular ; 1215)

Includes bibliographical references.

ISBN 0-607-95419-1 (alk. paper)

1. Water quality--California--Sacramento River Watershed. I. Domagalski, Joseph L. II. Geological Survey (U.S.) III. Series.

TD224.C3 W393 2000
363.739'42'097945--dc21

00-049453

R0024670

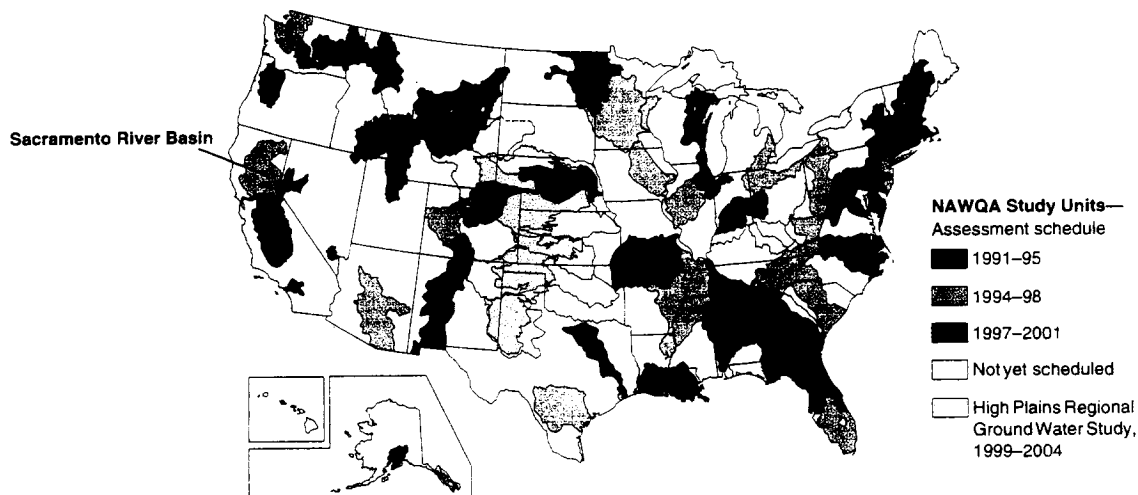
CONTENTS

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM	IV
SUMMARY OF MAJOR FINDINGS	1
Stream and River Highlights	1
Ground-Water Highlights	2
INTRODUCTION TO THE SACRAMENTO RIVER BASIN	3
Land-Use Effects on Water Quality and Stream Habitat	4
Water Use in California	5
Effects of Hydrologic Conditions on Study Results	5
Study Design Focuses on Land Use	6
MAJOR FINDINGS	7
Surface Water	7
NATIONAL PERSPECTIVE—Nutrients in a National Context	8
Pesticides in Surface Water	9
Pesticides of Historical Use	10
NATIONAL PERSPECTIVE—Organophosphate Insecticides in Agricultural and Urban Streams in a National Context	11
Metals in Water and Streambed Sediment	12
Mercury in Water and Streambed Sediment	14
Ambient Toxicity Monitoring by California State Agencies	17
Aquatic Biology	19
NATIONAL PERSPECTIVE—Aquatic Biology in a National Context	21
Ground Water	22
STUDY UNIT DESIGN	25
GLOSSARY	27
REFERENCES	29
APPENDIX—WATER QUALITY DATA FROM THE SACRAMENTO RIVER BASIN IN A NATIONAL CONTEXT	31

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

THIS REPORT summarizes major findings about water quality in the Sacramento River Basin that emerged from an assessment conducted between 1994 and 1998 by the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program. Water quality is discussed in terms of local and regional issues and compared to conditions found in all 36 NAWQA study areas, called Study Units, assessed to date. Findings are also explained in the context of selected national benchmarks, such as those for drinking-water quality and the protection of aquatic organisms. The NAWQA Program was not intended to assess the quality of the Nation's drinking water, such as by monitoring water from household taps. Rather, the assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water monitoring programs. The comparisons made in this report to drinking-water standards and guidelines are only in the context of the available untreated resource. Finally, this report includes information about the status of aquatic communities and the condition of in-stream habitats as elements of a complete water-quality assessment.

Many topics covered in this report reflect the concerns of officials of State and Federal agencies, water-resource managers, and members of stakeholder groups who provided advice and input during the Sacramento River Basin assessment. Basin residents who wish to know more about water quality in the areas where they live will find this report informative as well.



THE NAWQA PROGRAM seeks to improve scientific and public understanding of water quality in the Nation's major river basins and ground-water systems. Better understanding facilitates effective resource management, accurate identification of water-quality priorities, and successful development of strategies that protect and restore water quality. Guided by a nationally consistent study design and shaped by ongoing communication with local, State, and Federal agencies, NAWQA assessments support the investigation of local issues and trends while providing a firm foundation for understanding water quality at regional and national scales. The ability to integrate local and national scales of data collection and analysis is a unique feature of the USGS NAWQA Program.

The Sacramento River Basin is one of 51 water-quality assessments initiated since 1991, when the U.S. Congress appropriated funds for the USGS to begin the NAWQA Program. As indicated on the map, 36 assessments have been completed, and 15 more assessments will conclude in 2001. Collectively, these assessments cover about one-half of the land area of the United States and include water resources that are available to more than 60 percent of the U.S. population.

New reports on water quality

USGS recently released reports on water quality in 16 major river basins and aquifers across the Nation. The assessments indicate that the Nation's waters are generally suitable for irrigation, drinking water supply, and other home and recreational use. Major challenges remain, however, in protecting aquatic resources from nonpoint sources of pesticides, nutrients, metals, petroleum-based compounds, naturally occurring pollutants and other contaminants that continue to enter waterways in every basin.

Because each of the assessments adheres to a nationally consistent design and sampling methods, managers and planners in a specific locality or watershed can compare their water-quality conditions to those in other geographic regions. Collectively the assessments advance an understanding of the status of our Nation's waters and whether it is getting better or worse over time.

Pollution levels vary from season to season and among watersheds because of differences in land and chemical use, land-management practices, degree of watershed development, and natural features, such as soils, geology, hydrology, and climate. Understanding these variations and their causes helps in the implementation of effective management for water-quality improvement in specific geographic areas.

If you are interested in receiving additional information or would like to discuss water-resource topics of interest to you, please contact the NAWQA Program at (703) 648-5716 or through email at nawqa_info@usgs.gov. A map of the NAWQA assessments and additional ordering information, including USGS Circular report numbers, are provided in the enclosed brochure.

Information supports water-resource decisions by communities and states

A sampling of the types of information contained in the newly released assessments is provided, as well as examples of how this information is used to support science-based decisions at all levels for resource management and the protection of drinking water and aquatic ecosystems. For more details, go to the USGS website: <http://water.usgs.gov/nawqa>.

- Increased nutrients from the St. Croix River tributaries are increasing the rate of eutrophication in Lake St. Croix, **Minnesota**. In response to these findings, a multi-agency, cooperative agreement was signed to implement a protection strategy for the St. Croix River.
- **Mississippi Embayment** ground water is good for drinking-water supply; however, concentrations of DDT in streams and rivers are often higher than national norms. These results have contributed to decisions by the State of Mississippi to establish fish-consumption advisories for selected organochlorine compounds and other contaminants.
- Findings in 14 major urban centers across the Nation, including **San Antonio and Dallas, Texas** show that lead, PCBs, and DDT are decreasing since restrictions on their use in the 1970s. In contrast, the studies show that polycyclic aromatic hydrocarbons (PAHs) and zinc are increasing in watersheds experiencing urban growth and increased motor vehicle traffic.
- Fish and aquatic invertebrate communities are commonly impaired in the urbanized parts of **New Jersey**. The findings suggest that the amount of impervious road area and density of development affect the amount of runoff in a watershed and are important considerations in maintaining stream quality. These findings are used by the State of New Jersey in its development of realistic stream management and restoration goals for urbanized streams, to be adopted in the State's Watershed Management Plans.

---SEE BACK FOR MORE---

- Conservation tillage practices by farmers have resulted in reduced sediment erosion to streams in **northwestern Ohio** and **northeastern Indiana**.
- Diazinon, used by homeowners on lawns and gardens, was frequently detected in urban streams in the **Puget Sound** and **Sacramento River Basins**, often exceeding guidelines for protecting aquatic life.
- Nutrients are elevated and have resulted in increased algal growth in areas of urban development in the Southern Rocky Mountains. These findings were used by officials in **Grand County, Colorado** to help support legislation requiring improved septic systems and annual septic system monitoring in an effort to prevent further water-quality degradation.
- Concentrations of two pesticides—alachlor and cyanazine—have decreased in the **Illinois River** since 1991 because of decreased use of these pesticides by farmers. Relations between chemicals used in agricultural and urban settings, and the types and concentrations of contaminants in streams and ground water, are seen in every basin.
- Herbicide degradation products, which are mostly non-regulated, are commonly 10 times the concentration of parent compounds in the Iowa River, Iowa. On the basis of these findings, local officials in the city of **Cedar Rapids, Iowa** require monitoring and analysis of herbicide degradation products in the city water supplies.
- Since the Surface Mining Control and Reclamation Act of 1977, stream quality in the coal regions of the **Allegheny and Monongahela River Basins in Pennsylvania** and **West Virginia** and the **Kanawha River Basin in West Virginia** and **Virginia** has improved because of decreased acidity, but the diversity and abundance of aquatic organisms are reduced compared to unmined areas.
- Findings for Shingle Creek and 13 other major streams in the **Minneapolis metropolitan area** indicate clear relations between chloride concentrations and impervious surfaces and snowmelt. The information, placed in the context of other sampled urban watersheds throughout the Nation, helped to demonstrate to the Shingle Creek Watershed Management Commission that elevated chloride in Shingle Creek was not a local stockpile issue. The Commission has since recommended a metropolitan approach to assess current practices of road salt use, which is leading to improved strategies needed to meet **Minnesota** standards.

“The NAWQA Program is essential to understanding the impact of land-use changes on the quality of water flowing to the Great Lakes...Findings are directly applicable to resource managers and planners and transferable throughout the Great Lakes Basin” (Great Lakes Commission, composed of 8 states).

“The Florida Everglades has the misfortune to exhibit some of the highest concentrations of mercury in its fish and wildlife. The USGS has brought together scientists of many disciplines and enabled the South Florida Mercury Science Program to develop a deeper understanding of the scope, scale, status, and trends of mercury problems in Florida. This has thereby greatly enhanced our ability to model and manage the factors that contribute to this problem” (Dr. Thomas Atkeson, Florida Department of Environmental Protection).

“The NAWQA Program [provides data to] the U.S. Environmental Protection Agency (USEPA) [to help in their] implementation of the new pesticide law...Food Quality Protection Act (FQPA), passed in August 1996. USEPA has been required to factor potential exposures to pesticides through drinking water into already complex procedures used to set pesticide “tolerance levels” in foods...We do not have all the answers yet by any means, but we in USEPA who are charged with implementing this part of the new FQPA are greatly impressed with the knowledge and expertise contributed by NAWQA Program scientists and managers to assist USEPA in addressing the questions” (Joseph Merenda, Office of Pesticide Programs, USEPA).

May 2001

R0024674

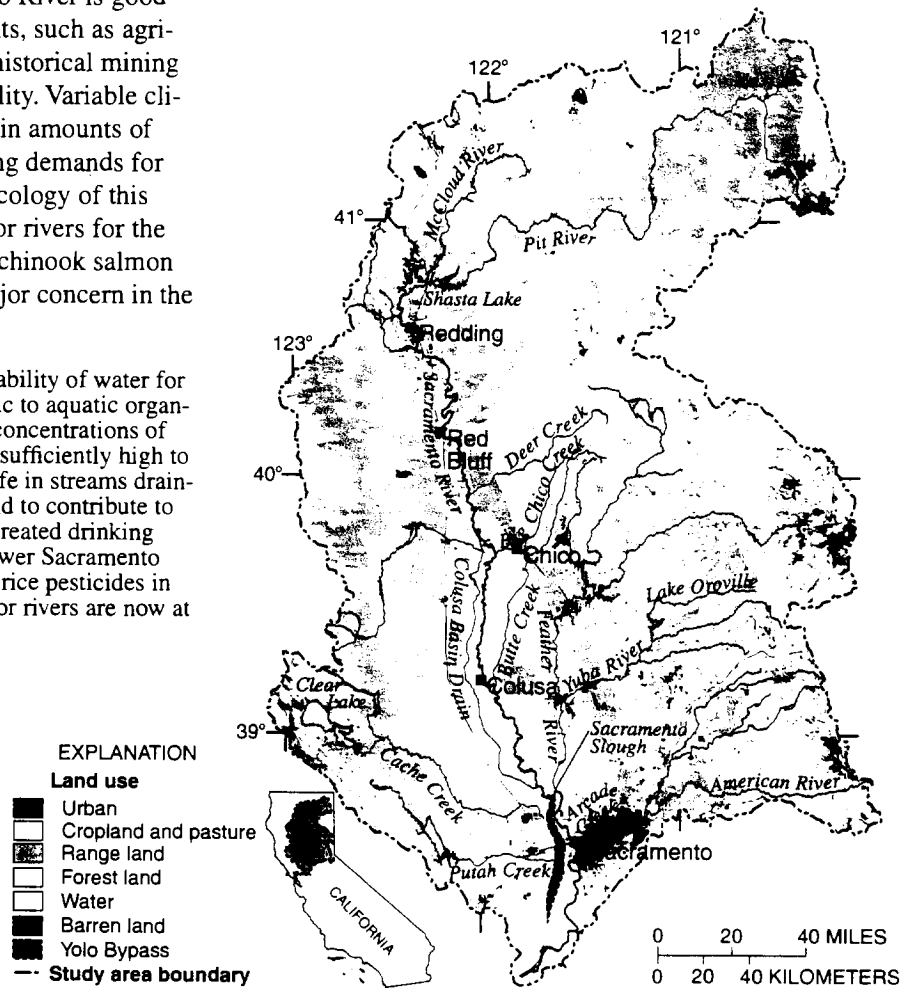
SUMMARY OF MAJOR FINDINGS

Stream and River Highlights

The water quality of the Sacramento River and its major tributaries supports most beneficial uses most of the time, including drinking and irrigation water, recreation, and protection of fish and other aquatic life. Most of the water in the Sacramento River and its major tributaries, such as the Feather and American rivers, is derived from melting snow that enters the rivers by managed discharges of water from reservoirs. Because the snow is pure, much of the Sacramento River and its large tributaries have low concentrations of dissolved minerals. Although water quality of the Sacramento River is good most of the year, seasonal events, such as agricultural runoff or runoff from historical mining operations, may affect this quality. Variable climatic conditions and variation in amounts of rainfall, coupled with competing demands for water uses, affect the aquatic ecology of this basin. Management of the major rivers for the migration and reproduction of chinook salmon and other salmonid fish is a major concern in the Sacramento River Basin.

- Pesticides can affect the suitability of water for drinking and can also be toxic to aquatic organisms. In previous years, the concentrations of pesticides used on rice were sufficiently high to affect the health of aquatic life in streams draining the rice growing areas and to contribute to taste and odor problems for treated drinking water withdrawn from the lower Sacramento River. The concentrations of rice pesticides in agricultural streams and major rivers are now at acceptable levels.
- Organophosphate insecticides, a group of pesticides used in agricultural and urban areas, enter the Sacramento River from multiple sources at concentrations that exceed recommended criteria for protection of aquatic life. Although the concentrations in agricultural and urban streams sometimes exceed amounts that are toxic to zooplankton in laboratory tests, the toxicity is greatly reduced or eliminated when concentrations of these pesticides are diluted by the Sacramento River.

- Phosphorus, a plant nutrient related to algal growth, was elevated in most samples collected in agricultural and urban streams.
- Mercury from historical mining activities has been a pervasive and prevalent problem of the Sacramento River Basin and downstream locations. Mercury concentrations in water exceeded recommended guidelines for the protection of aquatic life during this study.
- Salmonid fish reproduce in mountain streams, with subsequent migration to marine waters and final migration back to the mountain streams for reproduction. Water management projects (reservoirs and dams) have blocked the normal migration routes, forcing fish to move to less desirable habitats, thus affecting their reproduction.



The Sacramento River Basin Study Unit has a wide range of land uses that encompass about 70,000 square kilometers in California. The large cropland and pasture area is known as the Sacramento Valley. The Sacramento River is the largest river in California and supplies drinking and irrigation water to communities and farms in both northern and southern California. In 1995, over 2.2 million people lived within the Study Unit boundary, with more than 1 million in the Sacramento metropolitan area.

- Optimal temperature of rivers for fish migration is maintained most of the time, but temperature management can be difficult during a drought.
- Reservoirs have affected habitats of bottom-dwelling aquatic insect populations downstream from the dams. This may affect the food supply for critical life stages of fish.
- Nonnative fish and other nonnative aquatic species have affected streams in the Sacramento Valley. Nonnative species may outcompete native species, resulting in new aquatic community assemblages, thus creating an imbalance in formerly stable ecosystems.

Major Influences on Streams and Rivers

- Year-to-year variation in precipitation amounts
- Runoff from agricultural, urban, and mining areas
- Existence and maintenance of water-supply and flood-control projects

Selected Indicators of Stream-Water Quality

	Small Streams			Major Rivers
	Urban	Agricultural	Mining	Mixed Land Uses
Pesticides ¹			—	
Nitrate ²				
Phosphorus ³				
Trace elements ⁴				
Mercury ⁵				
Organochlorines ⁶				
Semivolatile organics ⁷				

Percentage of samples with concentrations **greater than or equal to** health-related national guidelines for drinking water, protection of aquatic life, or contact recreation

Percentage of samples with concentrations **less than** health-related national guidelines for drinking water, protection of aquatic life, or contact recreation

Percentage of samples with **no detection**

— Not assessed

¹ Insecticides, herbicides, and pesticide metabolites, sampled in water.

² Nitrate (as nitrogen), sampled in water.

³ Total phosphorus, sampled in water.

⁴ Arsenic, mercury, and metals, sampled in sediment.

⁵ Total mercury in unfiltered water samples.

⁶ Organochlorine compounds including DDT and PCBs, sampled in fish tissue.

⁷ Miscellaneous industrial chemicals and combustion by-products, sampled in sediment.

Ground-Water Highlights

Ground water of the Sacramento Valley accumulated in aquifers from precipitation in low hills surrounding

the valley and from infiltration of rain, rivers, and irrigation on the valley floor. Ground water is affected by agricultural and urban land uses.

- Bentazon, a herbicide applied to rice fields, was detected in 71 percent of shallow wells sampled in the rice-growing area, despite having been suspended from use since 1989. Bentazon concentrations measured in this study did not exceed any existing drinking-water standard. To protect rivers from pesticide contamination, the rice-field water is required, by means of mechanical controls, to remain on the fields for about 1 month. During that time, pesticide levels decrease by various processes, but evaporation of the water may increase the salinity of the shallow ground water by leaving salts behind.
- Urban growth of the Sacramento metropolitan area has affected ground-water quality. Nitrate concentrations are elevated but are below drinking-water standards in most wells.
- Some of the most heavily used portion of the southeastern Sacramento Valley aquifer was shown to generally have good water quality suitable for drinking and other uses. Only about 3 percent of the ground-water samples collected had nitrate or trichloroethene concentrations that exceeded a drinking-water standard. Radon concentrations exceeded guidelines in most of the domestic wells sampled.

Major Influences on Ground Water

- Agricultural and urban land-use practices
- Soil and aquifer properties

Selected Indicators of Ground-Water Quality

	Shallow Ground Water		Supply Wells	
	Urban	Agricultural	Domestic	Public
Pesticides ¹				—
Nitrate ²				—
Radon	—	—		—
Volatile organic compounds ³		—		—

Percentage of samples with concentrations **greater than or equal to** health-related national guidelines for drinking water

Percentage of samples with concentrations **less than** health-related national guidelines for drinking water

Percentage of samples with **no detection**

— Not assessed

¹ Insecticides, herbicides, and pesticide metabolites, sampled in water.

² Nitrate (as nitrogen), sampled in water.

³ Solvents, refrigerants, fumigants, and gasoline compounds, sampled in water.

INTRODUCTION TO THE SACRAMENTO RIVER BASIN

The Sacramento River Basin occupies nearly 70,000 square kilometers (km²) in the north central part of California (fig. 1). The Sacramento River is the largest river in California, with an average annual runoff of 27 billion cubic

meters (m³) (Domagalski and Brown, 1998). The basin includes all or parts of six landforms or physiographic provinces—the Great Basin, the Middle Cascade Mountains, the Sierra Nevada, the Klamath Mountains, the Coast

Ranges, and the Sacramento Valley (fig. 1). The Sacramento Valley is the low-lying province of the basin; the other provinces are mountainous. Land use in the mountainous regions of the basin is principally forest, although forest

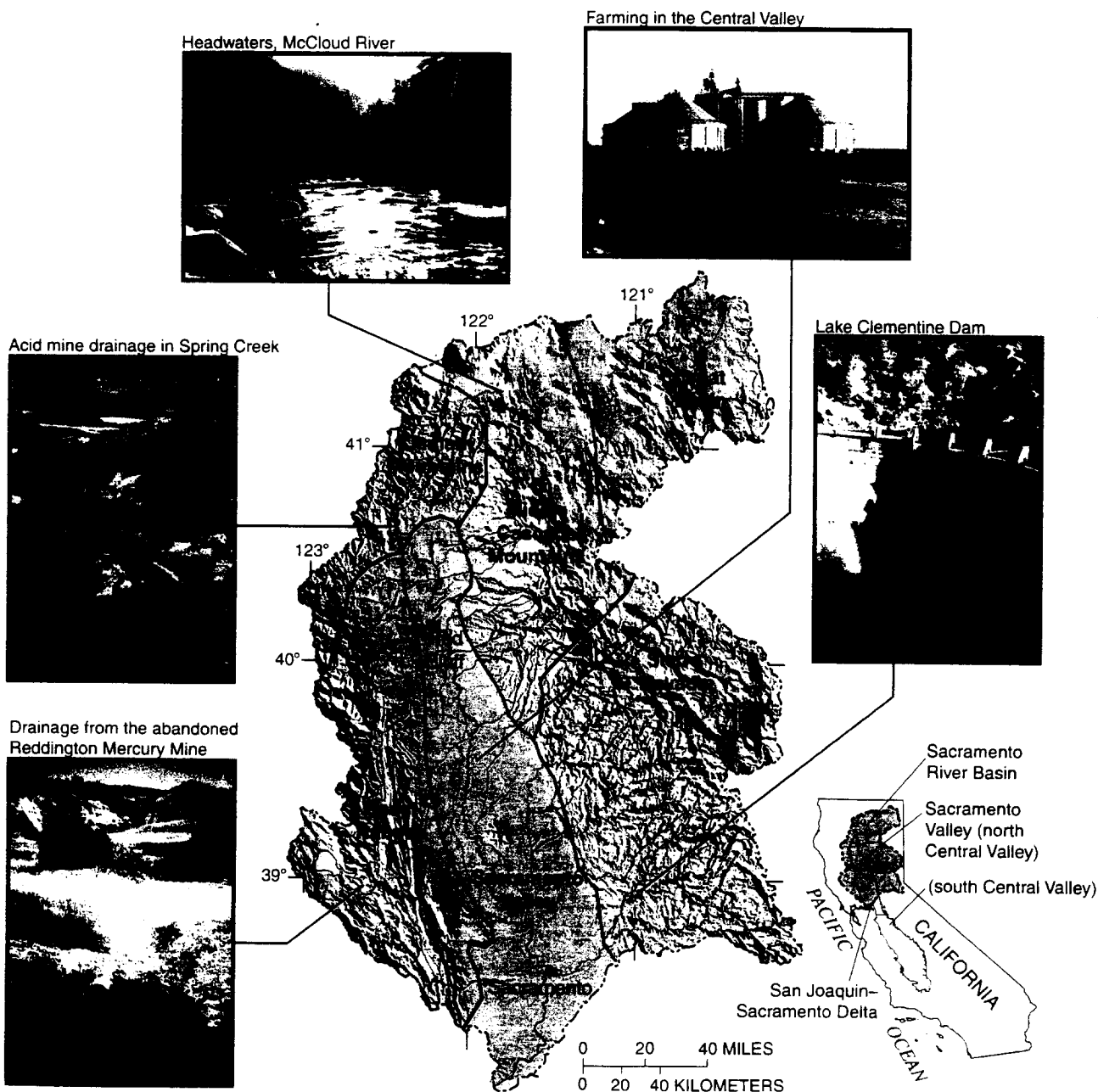


Figure 1. Physiographic provinces in the Sacramento River Basin. Physiographic provinces are regions defined principally by geologic and topographic features.

and rangeland are mixed in regions of the Coast Ranges and the Great Basin. Domagalski and others (1998) have provided more information on the physiographic provinces of the Sacramento River Basin.

The Sacramento Valley is the northern portion of the Central Valley of California and is fully contained in the Study Unit. The Sacramento Valley has the greatest population of any part of the basin, and it is there that the greatest effects or potential effects on surface and ground water are likely to occur from land-use activities. The Sacramento Valley is also the area of greatest water use in the basin.

Land-Use Effects on Water Quality and Stream Habitat

The Sacramento Valley supports a diverse agricultural economy, much of which depends on the availability of irrigation water. Water is collected in reservoirs at several locations within the mountains surrounding the Sacramento Valley and is released according to allocations for agricultural, urban, and environmental needs. The reservoirs also are managed for flood control. The reservoirs provide flood protection and allow the storage of water during dry years, but the placement of dams at the reservoirs has blocked migration routes for salmonid fish.

More than 8,000 km² of the Sacramento Valley are irrigated. The major crops are rice, fruits, nuts, tomatoes, sugar beets, corn, alfalfa, and wheat. Dairy products also are an important agricultural commodity. The land areas adjacent to the Sacramento Valley are mostly forested (fig. 1).

The largest cities of the basin are in the Sacramento Valley and include Chico, Red Bluff, Redding, and Sacramento. The Sacramento metropolitan area is home to more than 1 million people, which is nearly half of the total population (U.S. Department of Commerce, 1992) in the basin.

Previous mining for copper, lead, and zinc (fig. 2) in the Klamath Mountains has resulted in acid mine drainage (drainage of acidic waters from mines) into part of Keswick Reservoir, which is

located immediately downstream from Shasta Lake. The drainage includes both mined metals and nonmined metals such as cadmium. Mercury that was used in previous mining within the Coast Ranges (fig. 3) enters the Sacramento Valley through Cache and Putah creeks. Although neither creek flows directly into the Sacramento River during low-flow conditions, the load of mercury can be transported to downstream receiving waters, including the San Francisco Bay, during stormwater runoff

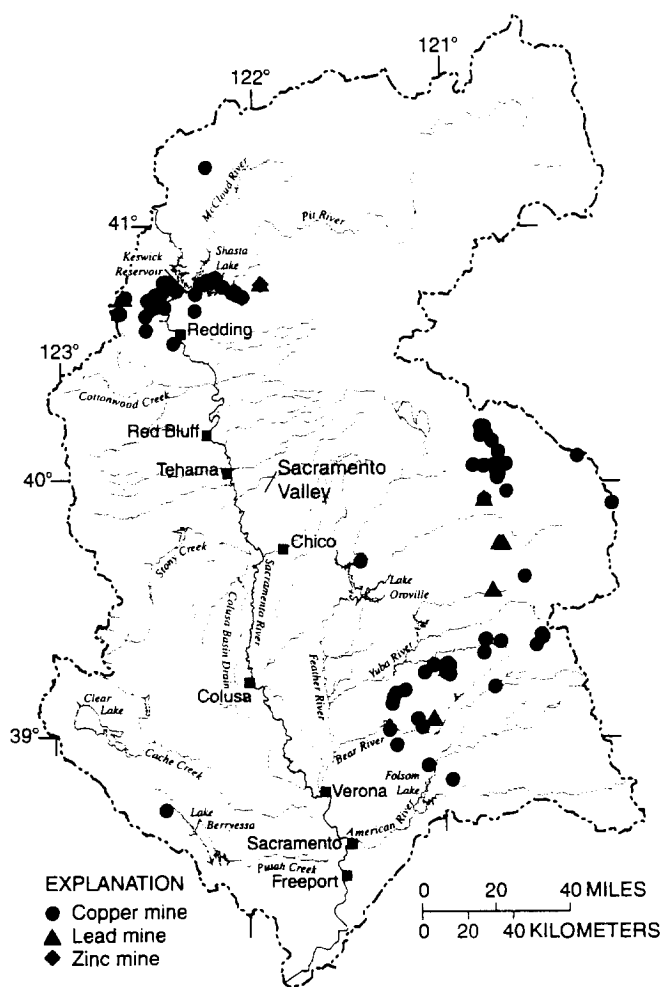


Figure 2. Locations of copper, lead, and zinc mines. The most severe case of drainage of acidic waters from mines has been in the region near Shasta Lake.

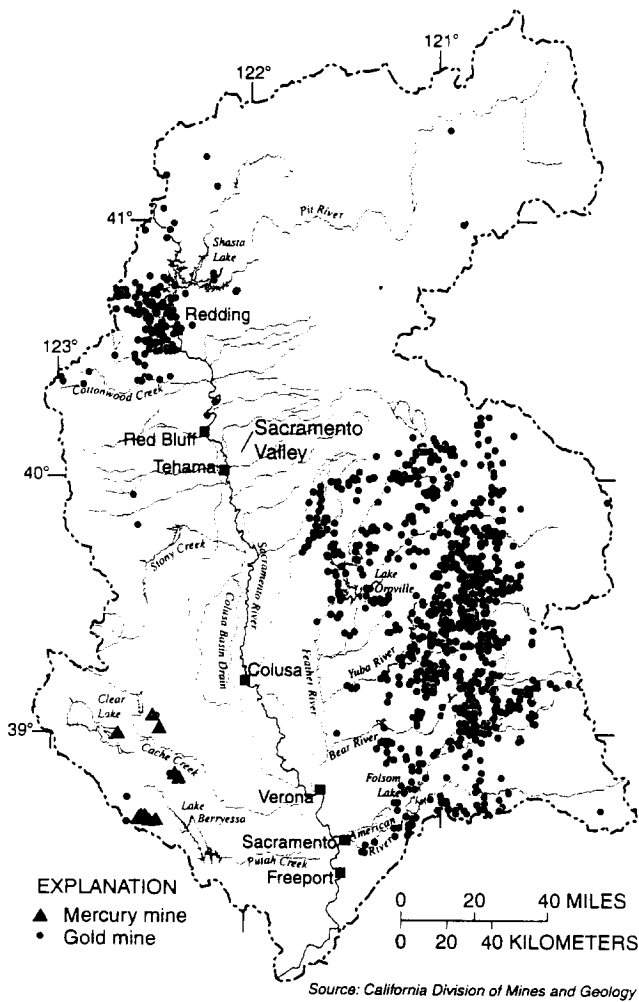


Figure 3. Locations of gold and mercury mines. The era of gold mining began in 1849 after the discovery of placer deposits in the American River.

conditions. Mercury also can enter the Sacramento River from the Sierra Nevada, where it was used in historical gold mining (fig. 3).

Water Use in California

Water storage, transportation, and allocation in California are strategically managed to take into account the wide diversity of the State's geography and physiography. Water is stored in nonpopulated areas of California and then transferred by natural stream channels or man-made canals to areas of demand. Reservoirs within the Sacramento River Basin have been constructed in the mountainous areas just adjacent to the valley. All major rivers of the Sacramento River Basin have one or more reservoirs, which were

constructed during the late 1940s to late 1960s mainly for flood control. Storage capacity of the reservoirs is managed to capture runoff from winter storms. However, stored water is not used solely in the Sacramento River Basin; it is transported to other locations in California and is a major source of supply for Los Angeles and other southern California communities.

Total water use in the Sacramento River Basin is about 18 billion cubic meters per year (m^3/yr). Allocations of water for agricultural, urban, and environmental uses are made according to the California water plan (California Department of Water Resources, 1998) but are modified on the basis of the yearly conditions of reservoir storage. For example, during drought years, total allocations are decreased. Pie charts in figure 4 show the percentage of allocations during average years of rainfall for 1995 and projected for 2020. Of those allocations, surface water provides 82 percent and ground water 18 percent of the total demand. During drought years, surface water drops to about 75 percent and ground water rises to about 25 percent of total demand.

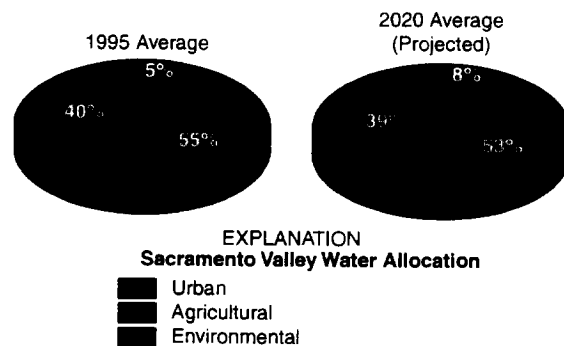


Figure 4. Water allocations for average rainfall years for 1995 and projected for 2020.

Effects of Hydrologic Conditions on Study Results

The average annual precipitation for the entire Sacramento River Basin is 914 millimeters (mm), most of which falls as rain or snow during November through March. Because little or no rain falls during the summer growing season, irrigation is required for successful agriculture. Most of the water-quality samples for the Sacramento River Basin study were collected between the fall of 1995 and the spring of

1998, which covered a series of wet winters. Precipitation amounts in northern California are variable and dependent on the location of the Pacific jet stream. The average annual rainfall at the city of Sacramento is about 460 mm. Since the 1940s, however, as little as 140 mm and as much as 915-mm have been recorded in a year.

Two major hydrological events occurred during the period of this study. The first, a major flood, occurred during the winter of 1996–97. Flooding began on January 1, 1997, and affected a major part of the Sacramento River downstream from Shasta Lake as well as tributaries to the Sacramento River, especially the Feather and Yuba rivers. Some streams, such as the Yuba River, had the highest recorded stream-flow following the rainfall associated with the flood of January 1, 1997 (fig. 5). The

second major hydrological event was the El Niño episode of 1997–98. The term “El Niño” refers to an “ocean–atmosphere phenomenon” during which wind and ocean current in the equatorial Pacific result in warmer-than-normal water along the North and South American Pacific coasts. El Niño winters frequently bring higher-than-normal precipitation in northern California because of a southward shift of the storm tracks and jet stream over North America.

Study Design Focuses on Land Use

Chemical and biological samples were collected from rivers and streams within, or downstream from, forested, urban, agricultural, and mining areas to assess overall quality and effects of specific land-use practices. In most cases, river or stream sampling sites were located in the Sacramento

Valley—the region of both the highest water use and where many potential effects on water quality had occurred and are likely to occur. At some sites, water samples were collected monthly and during storms to assess the effects of storm runoff on contaminant transport. Other sites were sampled only monthly, usually during normal flows. Shallow ground water was sampled from three areas—the highly used part of the southeastern Sacramento Valley aquifer, downstream from rice fields, and downstream from the recently urbanized area of metropolitan Sacramento. Domestic wells (existing wells) were sampled for the southeastern Sacramento Valley aquifer study area, whereas monitoring wells (drilled for this study) were sampled for the rice and urban land-use study areas.

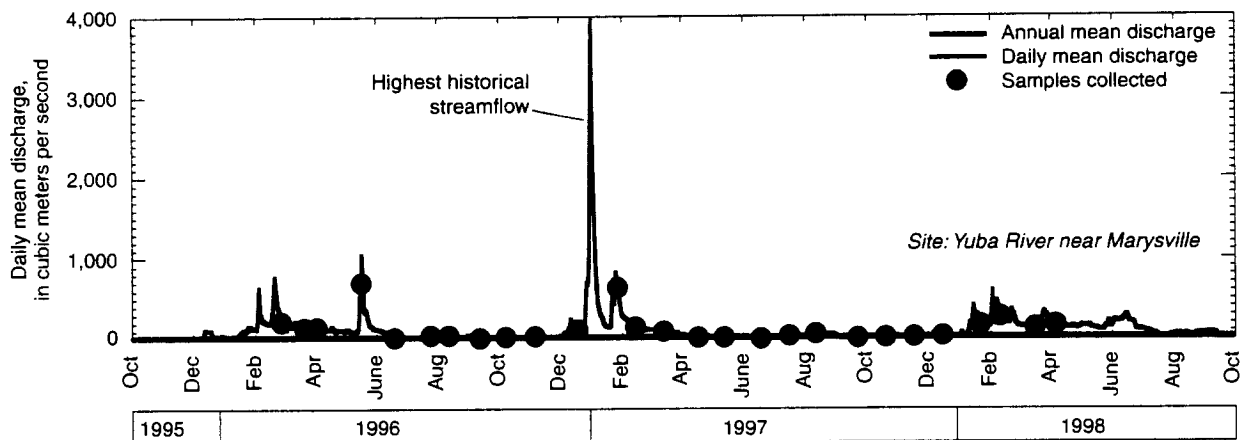


Figure 5. Yuba River hydrograph. The highest recorded discharge for the lower Yuba River occurred shortly after a large rainfall on January 1, 1997.

MAJOR FINDINGS

Surface Water

Water of the Sacramento River and its major tributaries is generally of good quality; the source is snow that melts and collects in upstream reservoirs and is released in response to water needs or flood control. The amount of dissolved solids in the Sacramento River and its major tributaries (Yuba, Feather, and American rivers) was low at all of the sampled locations (fig. 6). Higher median concentrations of dissolved solids occurred at agricultural sites such as the Sacramento Slough and Colusa Basin Drain, but those are diluted upon mixing with Sacramento River water (Domagalski and Dileanis, 2000). Nutrient concentrations such as nitrate also were low throughout the Sacramento River Basin (Domagalski and Dileanis, 2000) (fig. 7), and drinking-water standards for nitrate were not exceeded during the course of this study. At some locations, algae attached to streambed material was abundant, indicating that further investigation of nutrient dynamics and their consequences to the streams of

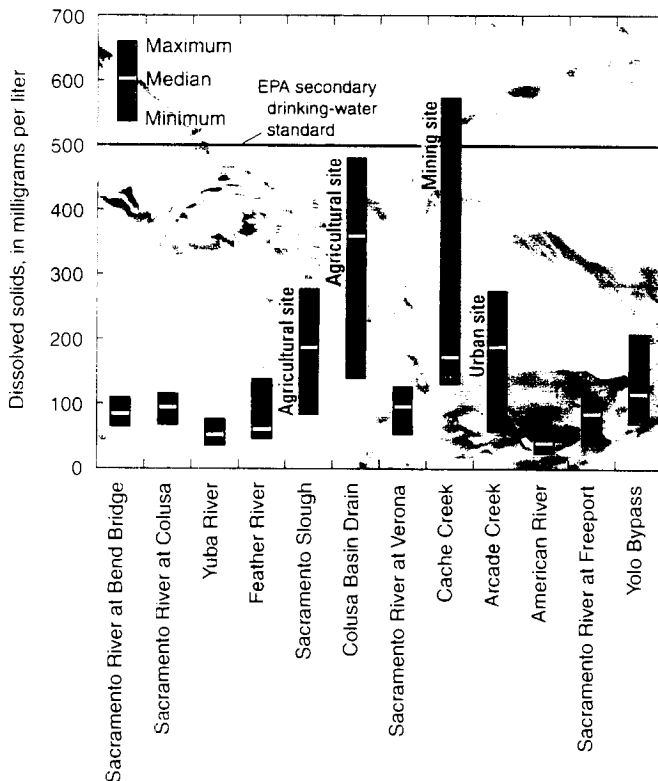


Figure 6. Concentrations of dissolved solids at the fixed sites. The highest concentrations were measured at the agricultural, mining, and urban sites. (EPA, Environmental Protection Agency)

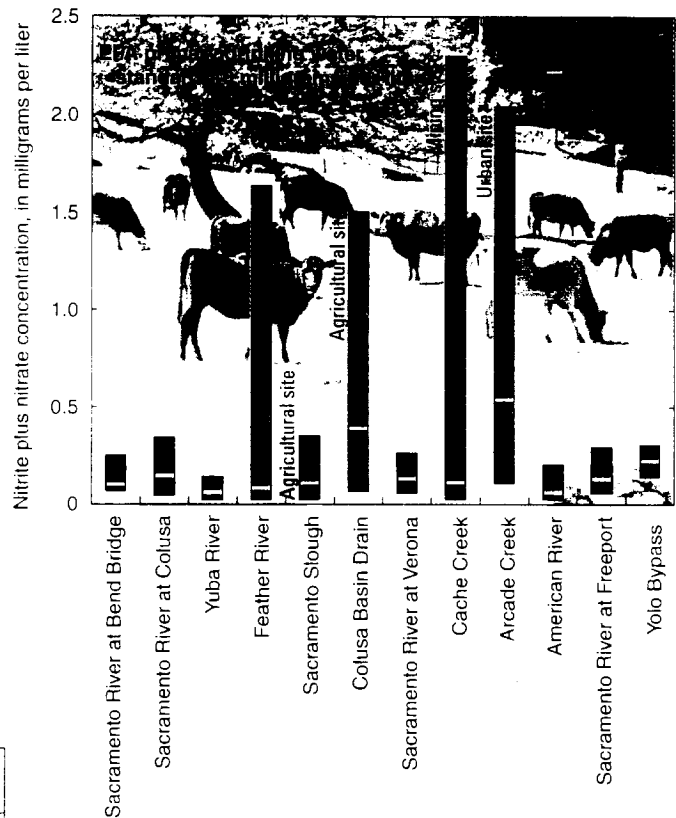


Figure 7. Concentrations of nitrite plus nitrate at the fixed sites. The highest concentrations were measured at the mining and urban sites.

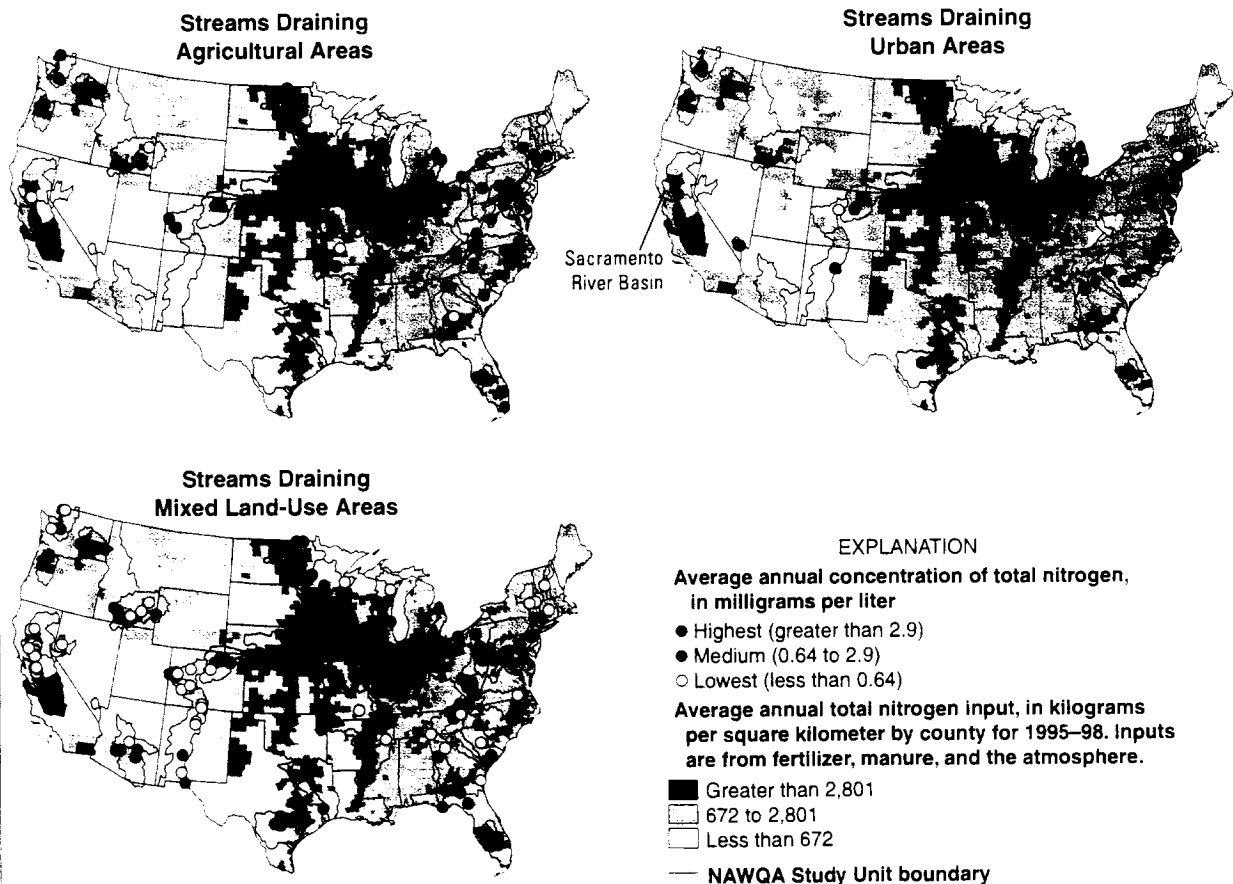
this watershed is warranted. Excess algal growth, which is usually related to higher-than-normal nutrient inputs to streams, is a water-quality concern when the algae affect the aquatic community (because of dissolved oxygen depletion). No such effects were observed in the Sacramento River or its major tributaries. Excess algae also can contribute to taste and odor problems in drinking water.

Some stream segments are listed as “impaired” by various contaminants (U.S. Environmental Protection Agency, accessed January 2, 2000). An impairment means that a standard of water quality for beneficial uses (for example, as a source of drinking water or for recreation or industrial use) is not being met. The impaired water bodies shown in figure 8 are mainly affected by nonpoint sources of contaminants from agriculture or from a combination of point and nonpoint sources from abandoned mines. Water-quality objectives are usually not met only during conditions of stormwater-driven runoff. The Clean Water Act requires States to maintain a listing of



NUTRIENTS IN A NATIONAL CONTEXT

Nutrient concentrations in the streams of mixed land-use and agricultural regions of the Sacramento River Basin tend to be lower relative to those measured in other areas of the United States with similar fertilizer applications within their watersheds. The maps show nitrogen; phosphorus concentrations have a very similar pattern. Elevated concentrations of nitrogen or phosphorus can stimulate nuisance growth of algae. The nutrient concentrations tend to be less than those of adjacent areas in California, agricultural areas of the Pacific Northwest, and large areas of the midcontinent region. In contrast to mixed land-use streams, nutrient concentrations of the urban stream are among the highest of similar urban streams throughout the United States. The lower concentrations in streams of mixed land-use probably can be attributed to dilution by streamflow. The Sacramento River and its major tributaries are derived from melting snow, which has low nutrient concentrations. These rivers tend to dilute the agricultural drainage, and therefore nutrient concentrations remain low in the major rivers. In addition, some instream processes remove nutrients, such as algal growth that incorporates nutrients in algae biomass. The urban stream, Arcade Creek, is entirely within an urbanized area, and all runoff to the stream is affected by that urban land use. The only inputs of water to Arcade Creek are from impacted land. The range in nutrient concentrations for all NAWQA Study Units is shown in the Appendix.



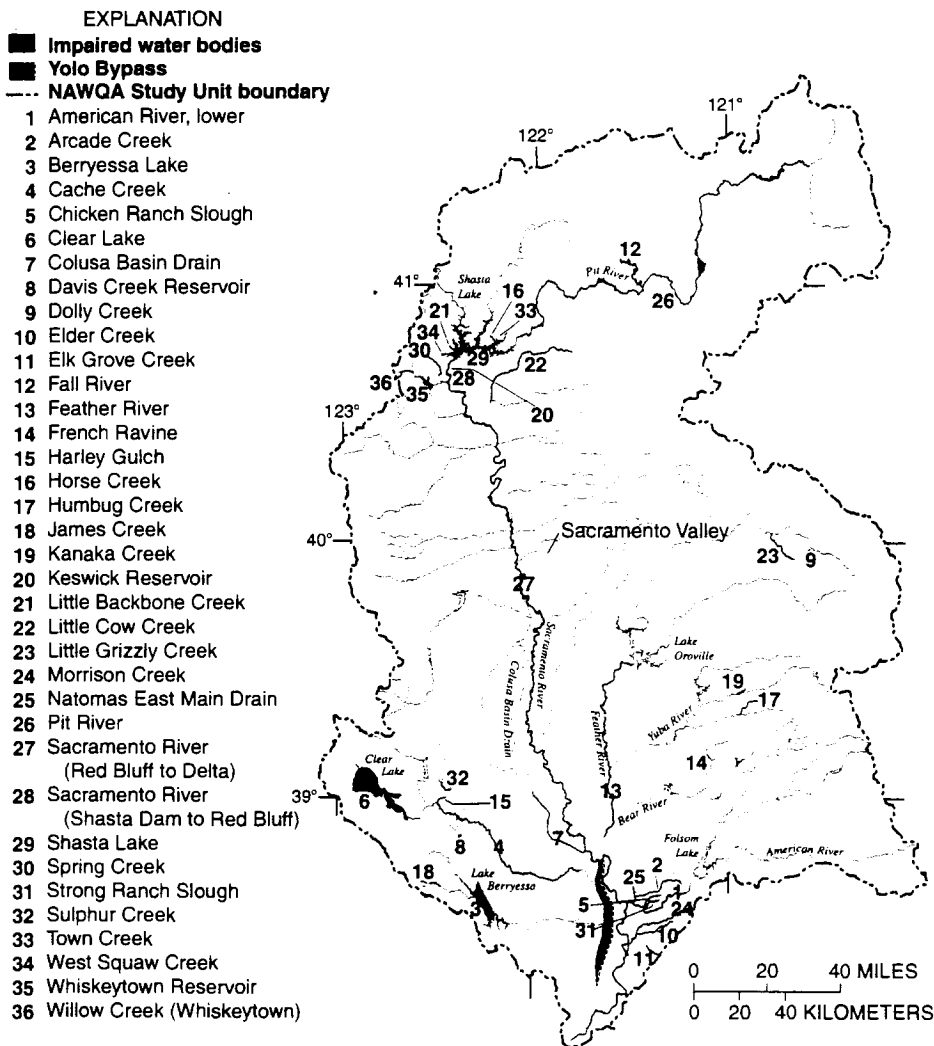


Figure 8. Impaired water bodies of the Sacramento River Basin according to the California 303(d) list (U.S. Environmental Protection Agency, accessed January 2, 2000). Impaired water bodies require the implementation of a management plan called a Total Maximum Daily Load (TMDL) to bring the water body into compliance with existing standards. Most of the impairments are the result of pesticides from agricultural or urban use, or from metals derived from historical mining operations.

impaired water bodies for the purpose of establishing a Total Maximum Daily Load (TMDL). A TMDL is a plan to restore the beneficial uses of the stream or to otherwise correct the impairment. The most prevalent listings in the Sacramento River Basin are for organophosphate pesticides and

mercury, for which TMDLs currently are being considered.

Pesticides in Surface Water

The concentrations of molinate and other pesticides (used in rice farming) measured during this study in the Colusa Basin Drain or in the

Sacramento River, represent a significant improvement over concentrations measured in previous years (Domagalski, 2000). The major pesticides that have been used on rice are molinate, thiobencarb, and carbofuran. Rice farming requires that fields be flooded with water throughout the growing season. Molinate and thiobencarb are applied to control aquatic grasses and weeds, whereas carbofuran is applied to control insects. During the late 1970s, the levels of rice pesticides in the Colusa Basin Drain were sometimes acutely toxic to fish such as carp (*Cyprinus carpio*) (Bennett and others, 1998). The toxicity was attributed to molinate.

In the early 1980s, consumers of drinking water in the city of Sacramento reported an objectionable taste, which was attributed to thiobencarb. A management program was enacted to reduce the levels of these pesticides in streams. The plan requires that rice-field water be retained on fields for 1 month following pesticide application to allow concentrations in water to be reduced through mechanisms such as volatilization, biological processes, or sunlight-induced degradation. Sampling of rice pesticides during this study showed that concentrations occasionally were in excess of management objectives in agricultural streams but always were very low in the Sacramento River (fig. 9). A target concentration

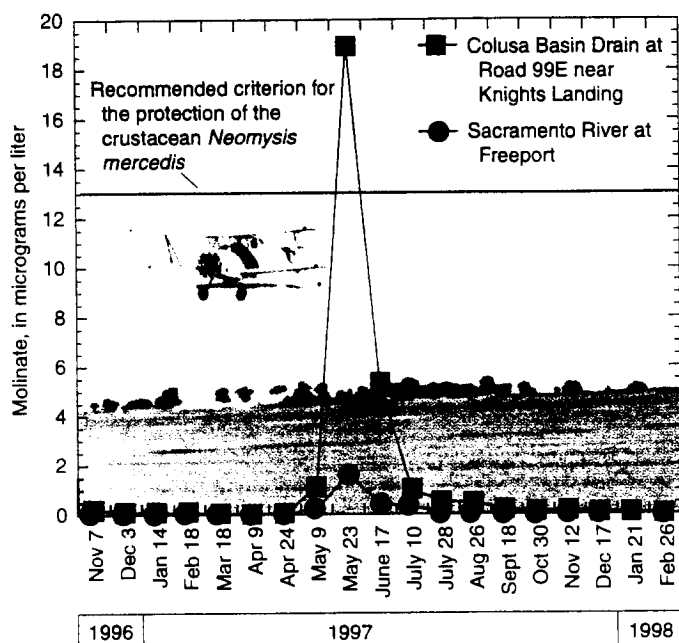


Figure 9. Concentrations of molinate at the Colusa Basin Drain at Road 99E near Knights Landing and Sacramento River at Freeport sites. The water in the Colusa Basin Drain is primarily agricultural drainage.

of 13 micrograms per liter ($\mu\text{g/L}$) of molinate in water was chosen for management of this herbicide. That level was chosen to protect the crustacean *Neomysis mercedis*, an important part of the food chain for young fish (Harrington, 1990). Concentrations in agricultural streams exceeded the target at least during 1 month of the year. Concentrations in the Sacramento River were always below those reported to be harmful to *N. mercedis* (Domagalski, 2000).

Pesticides also are transported to the Sacramento River, its tributaries, and agricultural drainage canals during winter storms (Kuivila and Foe, 1995; MacCoy and others, 1995; Domagalski, 1996). The pesticide that is considered a major problem for stormwater-driven transport is diazinon because of its toxicity to

aquatic organisms and its high detection frequency. Diazinon is toxic to some species of zooplankton, such as *Ceriodaphnia dubia*, at low concentrations ($0.35 \mu\text{g/L}$) (Amato and others, 1992). The zooplankton species *C. dubia*, is used in laboratory assays to test water for toxicity (U.S. Environmental Protection Agency, 1991a,b). Diazinon is applied to orchard crops, especially almonds, prunes, and stone fruits, during December and January to protect trees from insects that lay eggs in the trees during the winter and hatch the following spring. Toxic concentrations in tributaries to the Sacramento River can occur when agricultural areas contribute storm runoff; toxic concentrations rarely occur in the Sacramento River itself (MacCoy and others, 1995). Diazinon was present in

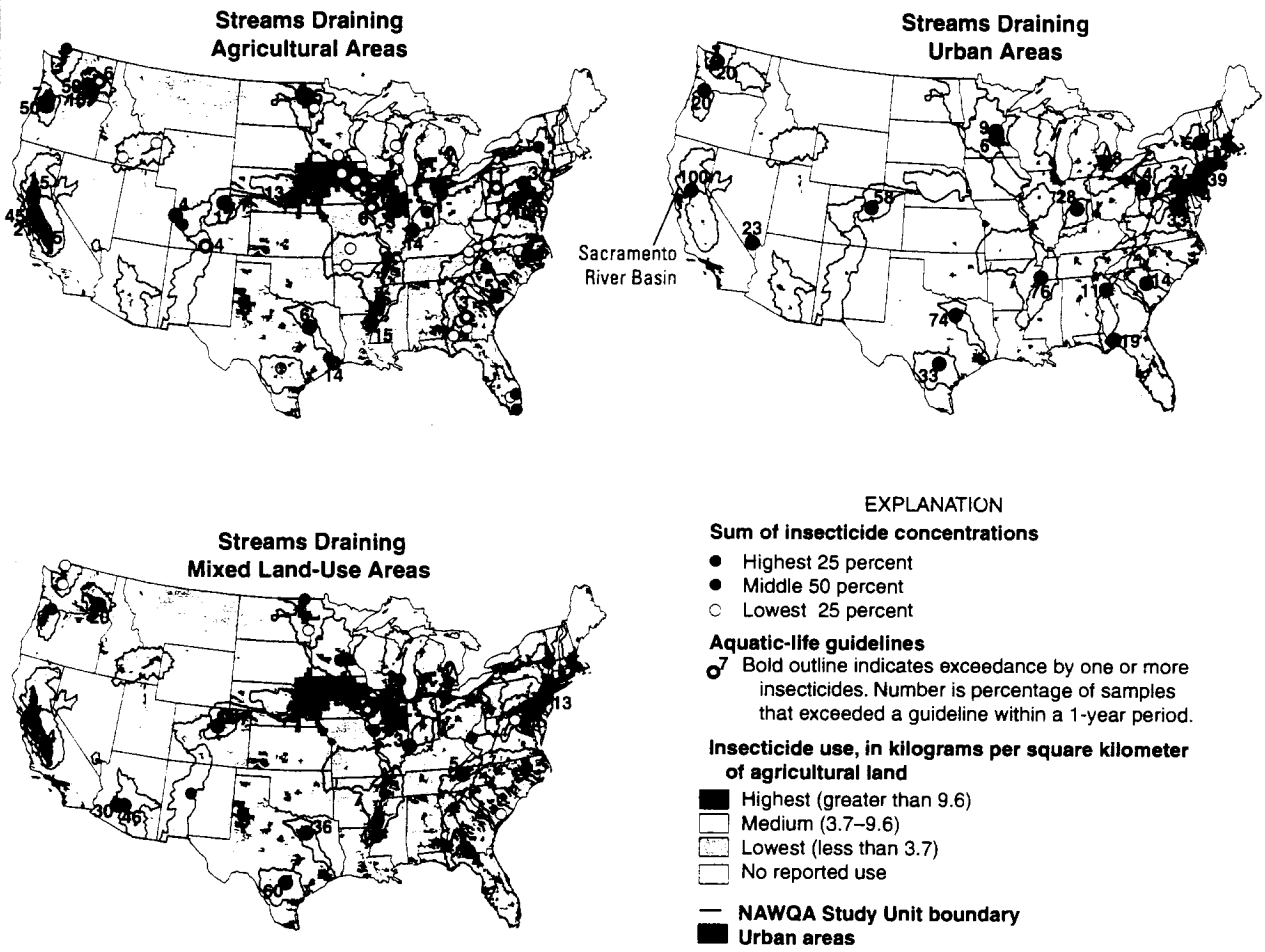
stormwater runoff at a number of sites in 1994 (Domagalski, 1996), and in nonstorm flows during 1996 through 1998. In the 1994 study, the Feather River was shown to be the greatest source of diazinon to the Sacramento River during a single storm, but other streams probably contributed to the diazinon load in the Sacramento River as well. This depended in part on the timing of diazinon applications and the location of greatest rainfall. The results of the routine samplings for diazinon during stable flow conditions at Arcade Creek near Del Paso Heights, Colusa Basin Drain at Road 99E near Knights Landing, and Sacramento River at Freeport are shown in figure 10. No stormwater runoff samples were collected. The highest concentrations during this NAWQA study occurred at Arcade Creek near Del Paso Heights, an urban site. Concentrations of diazinon in Arcade Creek that are toxic to *C. dubia* can occur in any season and result from household pesticide use and urban runoff. A standard for diazinon of $0.08 \mu\text{g/L}$ was proposed by the International Commission for the Great Lakes. As figure 10 shows, that standard was frequently exceeded at the Arcade Creek near Del Paso Heights site.

Pesticides of Historical Use

Pesticides that are no longer used, such as DDT, can still be detected in streambed sediments and the tissues of aquatic organisms because of their persistent chemical characteristics. Concentrations of pesticides such as DDT and its breakdown products tended to be low to nondetectable in the



ORGANOPHOSPHATE INSECTICIDES IN AGRICULTURAL AND URBAN STREAMS IN A NATIONAL CONTEXT



Organophosphate insecticides such as diazinon, chlorpyrifos, and malathion are toxic at low concentrations to some aquatic organisms. Some species of zooplankton are affected by diazinon concentrations as low as 0.35 µg/L. Diazinon levels at the urban stream, Arcade Creek, were elevated at various times of the year and exceeded recommended criteria for the protection of aquatic life for every measurement taken. Those levels were among the highest in the Nation. Most of the diazinon measured at the Arcade Creek site probably originated from household use throughout the watershed. Runoff from yards from either rainwater or irrigation water contributes to the loading of diazinon to stormwater drains that ultimately discharge into the creek. Diazinon enters agricultural drainage mainly in stormwater runoff because it is sprayed on orchards during the rainy winter season. Previous studies have shown that concentrations of diazinon in agricultural streams can be elevated less than a day after rainfall. Although diazinon concentrations of agricultural streams also were among the highest in the Nation, only one sample from those streams taken in this study exceeded recommended criteria. The range in diazinon concentrations for all NAWQA Study Units is shown in the Appendix.

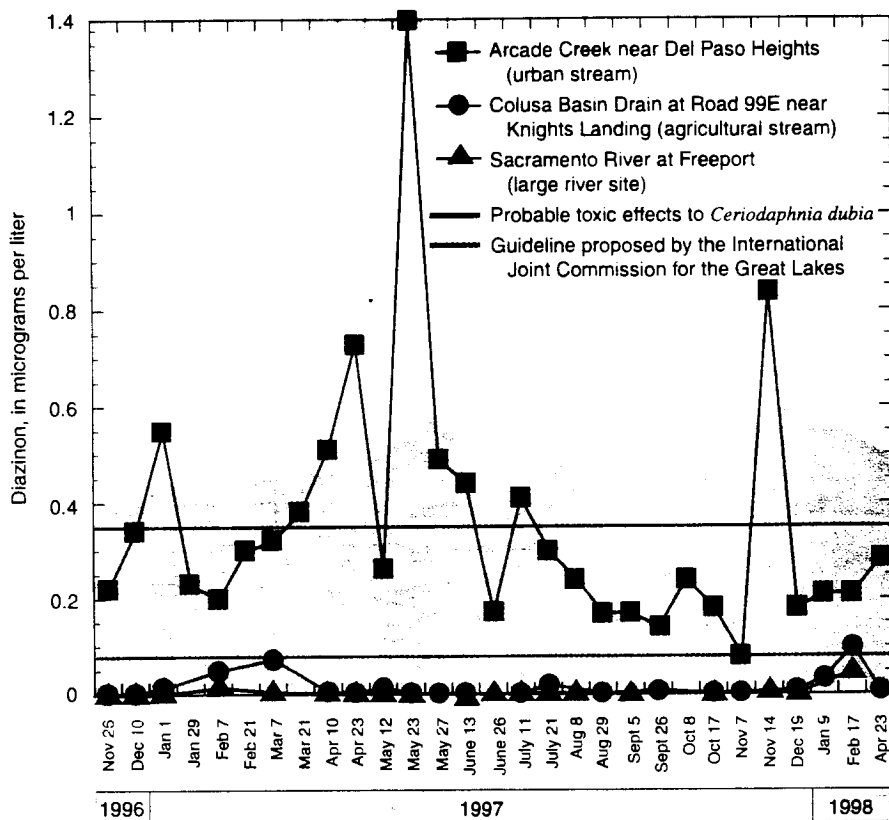


Figure 10. Concentrations of diazinon at the Colusa Basin Drain at Road 99E near Knights Landing, Arcade Creek near Del Paso Heights, and Sacramento River at Freeport sites. The highest concentrations were in the urban stream, Arcade Creek.

streambed sediments of the Sacramento River (MacCoy and Domagalski, 1999). Concentrations were higher in the streambed sediment of agricultural and urban streams. At some agricultural sites, and the urban site on Arcade Creek, the concentrations of DDT or its breakdown products in streambed sediment exceeded the Canadian sediment quality guidelines. The Canadian guidelines are designed to limit the accumulation of specific contaminants in organisms to levels below those that may adversely affect aquatic life (Canadian Council of Ministers of the Environment, 1995).

Concentrations of DDT or its breakdown products such as *p,p'*-DDE, or other organochlorine

insecticide residues, were very low in the tissues of aquatic organisms collected in the Sacramento River and its large tributaries (fig. 11). Concentrations of *p,p'*-DDE in the tissues of aquatic organisms collected from agricultural drainage sites were higher (fig. 11). The levels found in tissues of aquatic organisms from the agricultural drainage sites probably do not pose a health risk for humans but are above criteria developed by the New York State Department of Environmental Conservation (Newell and others, 1987) for protection of fish-eating wildlife such as birds. No national criteria exist to protect fish-eating wildlife from organochlorine compounds

such as DDT or its breakdown products; New York's are the only criteria available for comparison. The levels of *p,p'*-DDE found in the tissue of aquatic organisms tended to be relatively high when compared with other NAWQA Study Units. The range in concentrations for DDT and its breakdown products for all NAWQA Study Units is shown in the Appendix.

Metals in Water and Streambed Sediment

Acid mine drainage has been a serious environmental problem in the northern portion of the Sacramento River Basin (Alpers and others, 2000a,b). Several streams are listed as impaired (fig. 8) because of high concentrations of metals such as cadmium, copper, lead, and zinc. Metals concentrations in previous years have been toxic to fish in the upper Sacramento River near and downstream from Redding (Alpers and others, 2000a,b). Recent mitigation efforts at one of the more contaminated sites in the Spring Creek drainage near Shasta Lake have significantly lowered concentrations of metals in the Sacramento River, and no toxic effects to fish were observed during the course of this investigation (Alpers and others, 2000a,b). However, elevated levels of metals such as copper in streambed sediment can still be measured in the upper Sacramento River Basin downstream from Redding (MacCoy and Domagalski, 1999). Copper and other metals may still affect aquatic organisms.

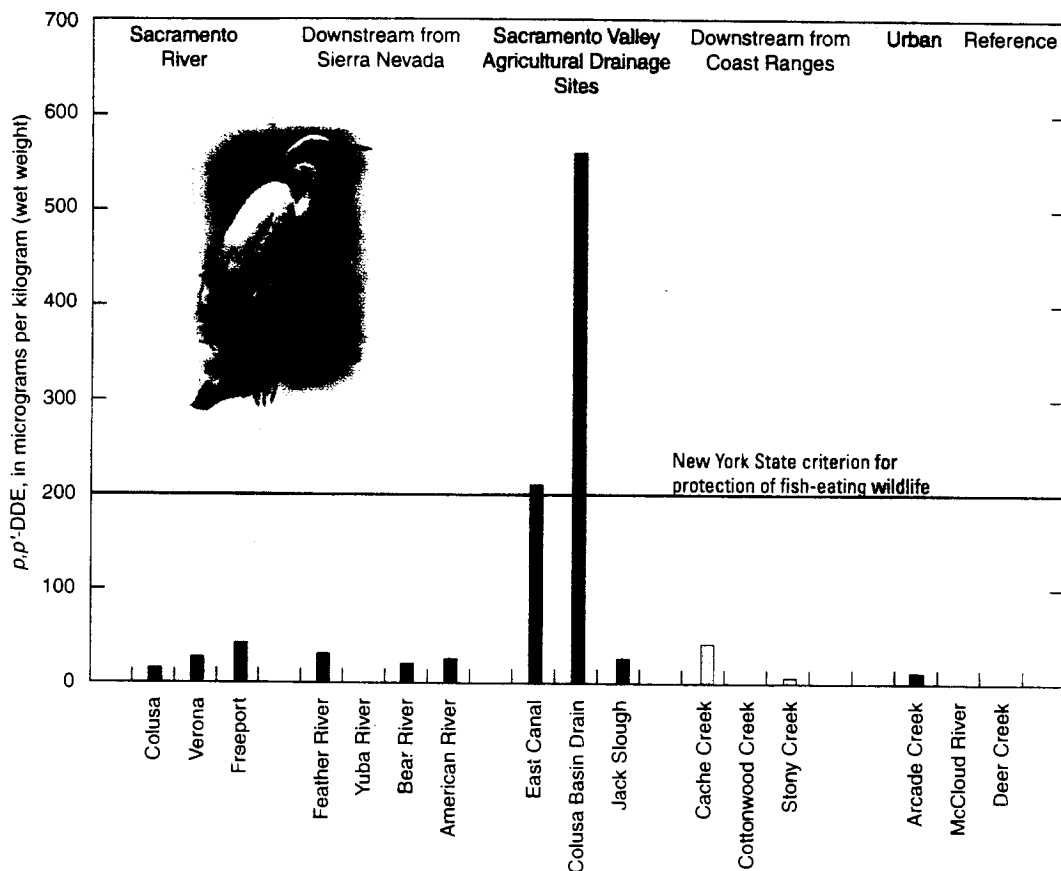


Figure 11. Concentrations of *p,p'*-DDE (a DDT breakdown product) in biota from the Sacramento River Basin. The use of DDT in the United States was terminated in 1972.

Trace Metals in Aquatic Organisms

For the NAWQA Program, looking for trace metals includes sampling streambed sediment and tissues of aquatic organisms. In theory, the transfer of metals from the streambed sediment into aquatic organisms can be understood by knowing the concentrations and geochemical forms of the trace metals in both the sediment and biota, as long as the feeding behaviors of the organisms are also understood. A predictive model of metals in tissue that is based on the amounts in streambed sediment could be developed from these studies. In practice, this becomes difficult because the actual bioavailability of metals in sediment can vary from site to site. For

example, the concentrations measured in aquatic organisms may not be fully assimilated into the cellular material of the organism but rather may be present as undigested material or even attached to external body parts. Because metals contamination from acid mine drainage is an important water-quality issue for the upper Sacramento River Basin, and knowledge of the actual bioavailability of metals in the mine drainage is essential for current and future management of the mine waste, a collaborative study was completed by the Sacramento River Basin NAWQA Program and the National Research Program of the USGS (Cain and others, 2000). In that study, biologists examined streambed sediment and the cytosol from the caddisfly, an aquatic insect widely distributed

in the upper Sacramento River and part of the food chain for a variety of fishes, including salmonid species. Cytosol is cellular material that can be isolated from aquatic insects. Biologists analyzed both whole body samples and cytosol samples for metals. Metal concentrations in cytosol provide a good indication of the potential effects on aquatic organisms from acid mine drainage. Aquatic insects also were sampled in a nearby reference stream that was unaffected by acid mine drainage. Elevated levels of cadmium, copper, lead, and zinc, derived from acid mine drainage, could clearly be distinguished in the cytosol samples, and it was shown that metals from the acid mine drainage were transported at least 120 kilometers downstream from the mine sources.

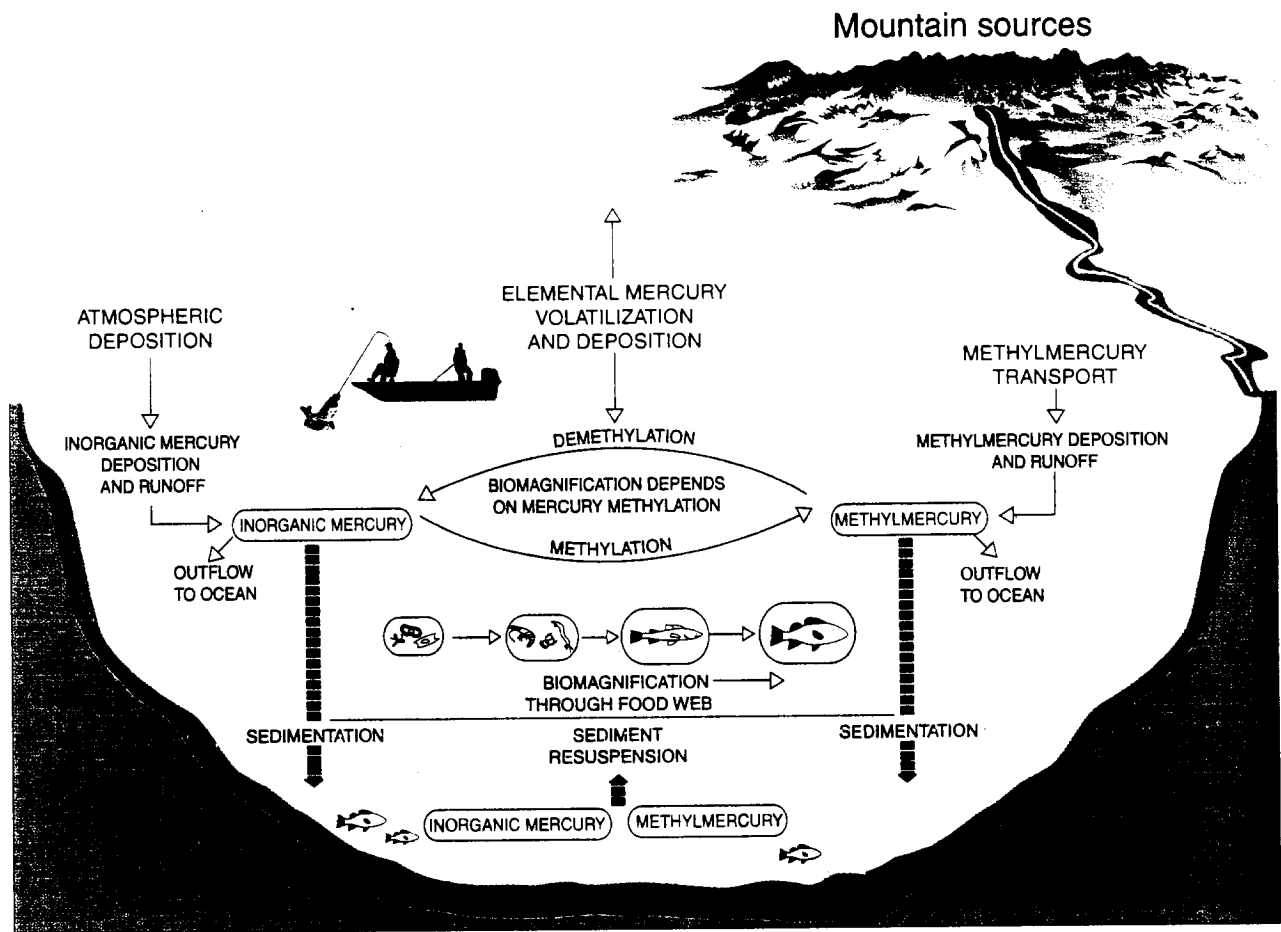


Figure 12. Mercury pathways in aquatic systems.

Mercury in Water and Streambed Sediment

Mercury is currently considered the most serious water-quality problem in the Sacramento River, some tributaries of the Sacramento River, and downstream locations including the San Francisco Bay (Domagalski, 1998). Mercury can enter streams or aquatic systems through either atmospheric deposition or transport from geological or man-made sources (fig. 12). Several processes contribute to the subsequent bioaccumulation of mercury in fish tissue. Because of the presence of mercury in the tissue of certain fish species, advisories have been posted for several

water bodies, and more advisories are planned, both within the Sacramento River Basin and in the San Francisco Bay. Specific advisories for fish species and locations are listed on the California Office of Environmental Health Hazards Assessment Web site at <http://www.oehha.org/fish.html>. A recent study (Davis and others, 2000) documented mercury levels of concern to human health in sport fish collected in the lower Sacramento and Feather rivers.

Although atmospheric mercury is the principal cause of mercury contamination of water bodies in other parts of the United States, especially the midwestern and eastern United States, the cause is

different in California. Geologic and anthropogenic sources, especially from historical mining for both mercury and gold, are the main reasons for mercury problems in the Sacramento River Basin (Domagalski, 1998). Mercury was mined in the Coast Ranges near Clear Lake and at locations east and south of Clear Lake (fig. 3), and it was used in the recovery of gold from ore and stream deposits during the late 19th century (fig. 3).

Decades of gold mining in the Sierra Nevada have resulted in the deposition of mercury in the streambed sediments of the gold mining region. The release of mercury from ore to streambed sediments in the mercury mining regions of the Coast Ranges also

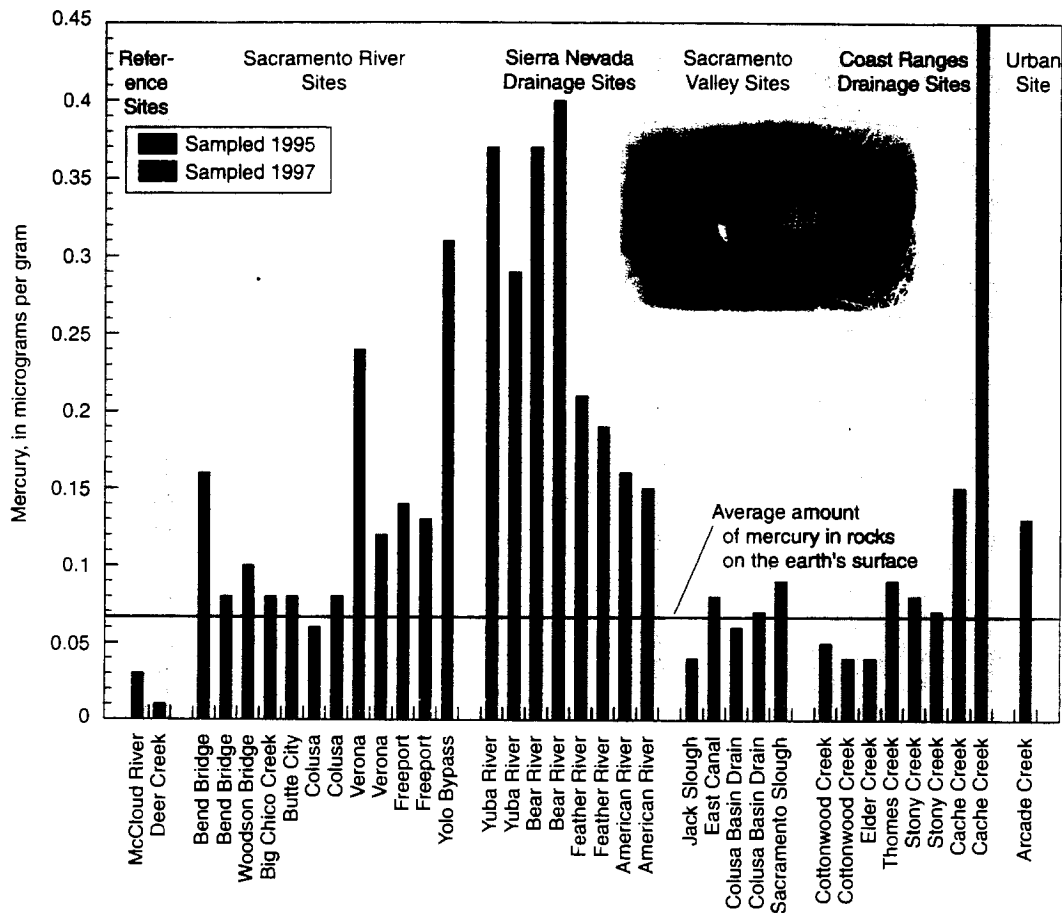


Figure 13. Concentrations of mercury in streambed sediment at select locations of the Sacramento River Basin.

has occurred (Hunerlach and others, 1999). The construction of reservoirs in the lower Sierra Nevada between 1948 and 1968 has had the positive effect of reducing the amount of mercury transported downstream (Slotton and others, 1997). Reservoirs trap mercury because suspended sediment, the principal means by which it is transported, tends to settle to the bottom. This trapping of mercury will have future implications on the management of these reservoirs, including potential dam removal. Some dams are being considered for removal in order to restore habitat for fish. Residual mercury from mining operations is present in the streambed sediments

downstream from the Sierra Nevada reservoirs, as indicated by the NAWQA Program for the Sacramento River Basin. Concentrations of mercury in the streambed sediments of 24 sites sampled during the NAWQA Program are shown in figure 13. The highest concentrations of mercury in streambed sediment were measured in samples collected from sites downstream from the Sierra Nevada and the Coast Ranges. Sites on the Sacramento River downstream from the Feather River tended to have higher mercury concentrations relative to sites sampled upstream from the confluence of these two rivers because of historical gold mining.

The U.S. Environmental Protection Agency (EPA) has recommended water-quality criteria for mercury to protect aquatic life and human health. A recommended criterion of 12 nanograms per liter (ng/L) of total mercury in water was proposed by the EPA in 1985 (Marshack, 1995). That criterion is supposed to limit the amount of mercury accumulation in fish tissue and thereby protect human health. The 12-ng/L criterion was exceeded mainly during runoff conditions at all fixed sites during the timeframe of this investigation. In 1999, the recommended level was revised to 50 ng/L (U.S. Environmental Protection Agency,

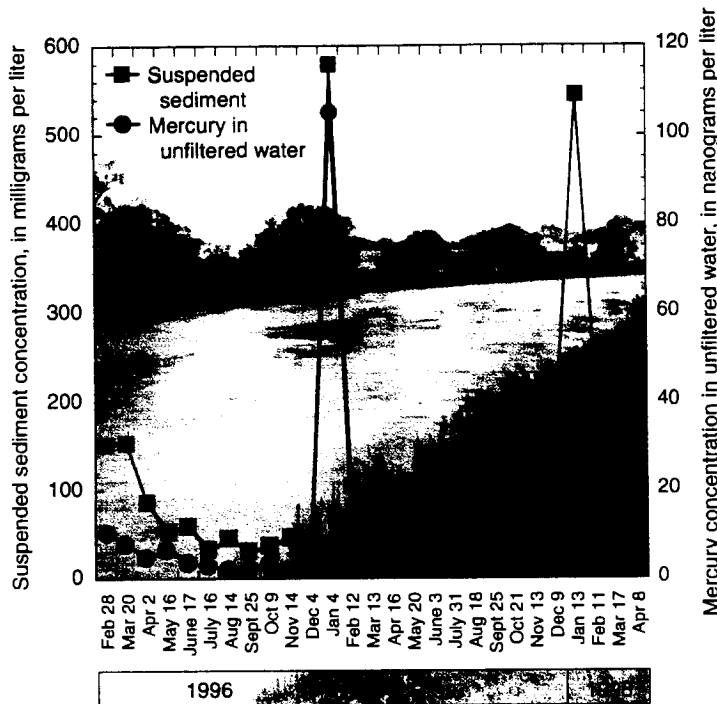


Figure 14. Mercury and suspended sediment concentrations for the Sacramento River at Colusa site. Mercury concentrations increase with sediment concentrations because mercury is attached to sediment particles.

1999). The 50-ng/L level was exceeded only at Cache Creek at Rumsey, Sacramento River at Colusa, and the Yolo Bypass at Interstate 80. Continued monitoring of mercury levels in fish tissue will be required to determine if the 50-ng/L criteria is effective. Time series plots of mercury and suspended sediment for the Sacramento River at Colusa are shown in figure 14. The higher concentrations of mercury correlate well with suspended sediment because much of the load of total mercury is transported with the suspended material (Alpers and others, 2000b).

Methylmercury is the most bioaccumulative form of mercury in the environment because it builds up in organisms more readily than other forms of mercury. It is a toxic form of mercury that can bioaccumulate in

the tissues of aquatic organisms and cause human health problems if fish with high levels are consumed. Methylmercury usually is formed by bacterially mediated reactions in sediments. Concentrations in water were measured at selected sites on the Sacramento River and at sites receiving agricultural drainage to assess the effects of agricultural activities on the production or the levels of methylmercury.

Concentrations of methylmercury in unfiltered water are shown in figure 15. The highest median concentration was 0.19 ng/L, which represents the samples collected at the Colusa Basin Drain at Road 99E near Knights Landing site. There is no water-quality standard in California or in any other State that is based on methylmercury concentrations in water. However,

a concentration at or below 0.1 ng/L of methylmercury has been suggested as being representative of pristine water (Rudd, 1995). That concentration is typical of rivers upstream from wetland environments and away from mercury sources (Rudd, 1995). The median methylmercury concentrations for the Sacramento River sites were slightly above 0.1 ng/L and maximum concentrations approach 2 ng/L. It is not known how or if those levels of methylmercury in water contribute to elevated levels of mercury in fish tissue.

There was a seasonal component to methylmercury concentrations for the sites at which they were measured for this study. The lowest concentrations were measured during middle to late summer (fig. 16). Higher concentrations tended to be measured during the autumn to winter months. The magnitude of the concentrations may also have been related to precipitation and runoff conditions. The highest concentrations for the period of this study were measured during January and February of 1997 and were attributed to the January 1997 flood. During the subsequent El Niño winter of 1997–98, higher-than-normal amounts of rain were recorded for much of the Sacramento River Basin, although there was no single storm of the magnitude of the January 1997 flood. Methylmercury concentrations increased during the El Niño winter, but not to the extent of the flood of the previous year. The effect of these methylmercury concentrations on downstream water bodies, such as the San Francisco Bay, has not been determined.

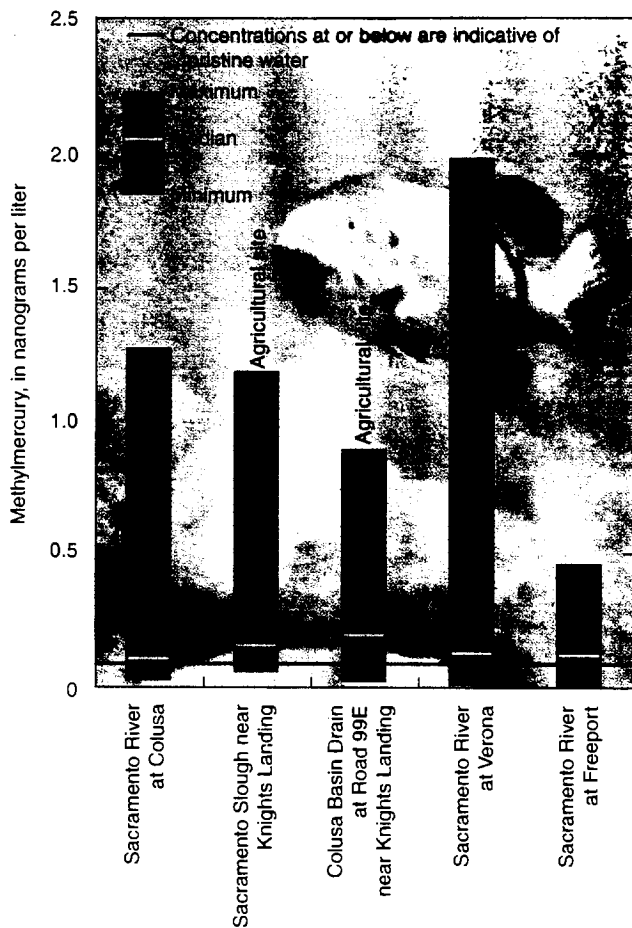


Figure 15. Concentrations of methylmercury at select locations in the Sacramento River Basin. Methylmercury is the form of mercury most likely to accumulate in aquatic species such as fish.

Ambient Toxicity Monitoring by California State Agencies

Ambient toxicity testing uses laboratory bioassays to assess the effect of contaminants on aquatic life. Essentially, the tests answer the question: Can specific types of fish, invertebrates, and algae species continue to live, grow, and reproduce in water samples collected from water bodies? The EPA protocols (U.S. Environmental Protection Agency, 1991a,b) for conducting chronic toxicity tests on freshwater species include representatives from three phyla and trophic levels. The three species are the fathead minnow, a small planktonic crustacean (*C. dubia*), and a planktonic green alga (*Selenastrum capricornutum*). These three species have been used to evaluate ambient water quality in the Sacramento River Basin since

1986. California regulatory agencies rely on the tests for evaluating compliance with narrative toxicity objectives, which state that “all waters shall be maintained free of toxic substances in concentrations that produce detrimental physiological responses in aquatic life” (Marshack, 1995). Testing has changed from conducting broad watershed surveys that determine the spatial and temporal distribution of toxicity to conducting detailed follow-up studies that couple toxicity testing with a Toxicity Identification Evaluation (TIE) to identify specific chemicals and land uses responsible for toxicity. Water-quality data from the NAWQA Program are useful in determining the exposure of aquatic organisms to specific groups of contaminants such as organophosphate pesticides. Results from 13 years of monitoring suggest that the EPA toxicity tests are powerful tools for assessing water quality.

The toxicity testing program in the Sacramento River watershed was the first indicator of the potential water-quality problems currently associated with pesticide runoff from urban areas and orchards. Using *C. dubia* tests, pulses of toxicity have been detected over a 10-year period throughout the Sacramento Valley in waters that receive drainage from orchards (de Vlaming and others, 2000). The toxicity has been linked to diazinon applied to dormant orchards and chlorpyrifos applied to nondormant orchards. *C. dubia* also was effective in identifying toxicity attributed to drainage from rice fields. Rice-field drainage also was toxic to two important local species, larval striped bass and an invertebrate, *N. mercedis*. The invertebrate toxicity was caused by methyl parathion and carbofuran. As mentioned previously, a rice management program has eliminated the toxicity to all three species. *C. dubia* toxicity is detected throughout the year in waters that receive drainage from urban areas (fig. 10). This toxicity is attributed to diazinon and chlorpyrifos. Although most toxicity has been detected with *C. dubia* and linked to insecticides, other examples of ambient water toxicity have been identified. In areas of the Sacramento Valley that receive acid mine drainage, toxicity to *S. capricornutum* and *C. dubia* has been linked to copper and zinc. Mine remediation projects have reduced both metal concentrations and toxicity. Concentrations of copper and zinc measured in this NAWQA Program confirm that metal concentrations are below toxic levels (Alpers and others, 2000b). *S. capricornutum* toxicity has been documented in

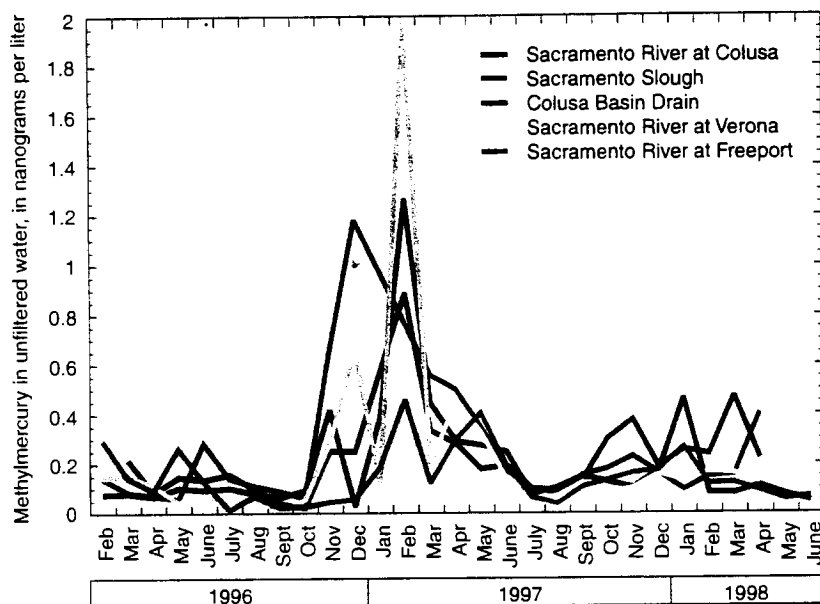


Figure 16. Seasonal changes of methylmercury concentrations. The highest concentrations were measured during high streamflow and following rainfall.

waters that receive agricultural or urban runoff. Some of the toxicity can be attributed to the herbicide diuron, also detected in samples collected by the NAWQA Program; however, additional unidentified toxicants are present. Fathead minnow toxicity has been traced to ammonia originating from dairies and wastewater treatment plants. Taken together, the results of the last decade reveal that all three testing procedures, in association with TIEs and chemical analyses, have been effective for the identification of an array of toxicants originating from various sources. In several cases, alternative land-use practices or management strategies have improved water quality, as demonstrated by toxicity test monitoring. Because resources are not available for monitoring the complete array of potential contaminants, toxicity testing is a useful tool for focusing on chemicals present in a water body at toxic levels.

NAWQA Participation with Local Water-Quality Programs

All NAWQA Study Units maintain communication and program coordination with a liaison committee of outside parties interested in water quality within the respective basins. During the early part of this NAWQA study, two significant programs were taking shape that involved new approaches to understanding and promoting the better management of water quality in the Sacramento River Basin.

In 1994, Congress recognized the need to develop a coordinated, technically sound, adequately funded program that would focus on establishing toxic pollutant standards for the Sacramento River Basin. Congress then appropriated funds for the Sacramento River Toxic Pollutant Control Program (SRTPCP) and has continued to support the program.

The long-term objective of the SRTPCP is to develop and

implement a program that will bring the Sacramento River and its tributaries into compliance with water-quality standards for toxic pollutants and thereby protect beneficial uses. A second objective of the SRTPCP is to help form a viable organization of watershed stakeholders. The stakeholder organization is intended to address not only the related toxic-pollutant issues of the watershed, but also the broader water-quality and watershed issues that must be resolved to protect and enhance surface and ground water throughout the basin.

The broader program to be conducted by this stakeholder organization has been named the Sacramento River Watershed Program (SRWP). The SRWP, although initiated with funding provided under the SRTPCP, is much broader in scope than the SRTPCP. The SRWP is intended to provide a forum to address a broad array of water-quality-related issues within the watershed, not just issues on toxic pollutants. Other issues that may be addressed under the broader watershed program include, but are not limited to, conventional water quality (including sediment, temperature, and dissolved solids), habitat, endangered species, streamflow, and ground-water issues.

The NAWQA Program participates in various committees of the SRWP, including the monitoring committee, to share data. Data from the NAWQA Program are shared with the SRWP and are used to help interpret the current water-quality conditions and to help guide the continued management of the water resources of the Sacramento River Basin.

Aquatic Biology

Changes in land use in the Sacramento River Basin have had major effects on the streams in the basin and on the aquatic communities dependent on them. Riparian forests and wetlands have been removed or degraded. Water development activities, particularly construction of dams and reservoirs, have altered natural flow and water temperature. The seasonal nature of higher temperatures along a stretch of the Sacramento River downstream from Shasta Lake is shown in figure 17. Water diversions for irrigation result in less water in the Sacramento River and rapidly increasing temperatures that are potentially harmful to certain fish in the spring and summer, especially downstream from the site on the Sacramento River above Bend Bridge near Red Bluff.

Collectively, these changes in streams have resulted in corresponding changes in native ecological communities, including species extirpation and population declines in remaining native species (Moyle and Nichols, 1974; Brown and Moyle, 1993; Brown, 2000). These declines have resulted in the listing of a number of animal and plant species as threatened or endangered under State or Federal law.

Native fish species are still common in Sacramento River Basin streams

Thirty-five species of fish were collected, including 12 species native to California. Statistical techniques were used to categorize sites on the basis of similar fish groupings. Four species of fish were collected at mountain

streams—rainbow trout, brown trout, riffle sculpin, and chinook salmon. These species are generally associated with cold, clear water and are considered intolerant of other environmental conditions such as warm water. Only brown trout is an introduced species. Rainbow trout were collected at all sites and were the most abundant species, representing 74 percent of the fish collected.

Fish were also sampled at sites below the mountains but above the valley floor. Those sites are referred to as the foothill sites. Twenty-one species of fish, including 13 native species, were collected at the nine sites in the foothill group. None of the introduced species was abundant, and no introduced species represented more than 3 percent of the fish collected. Native minnows, hardhead, Sacramento pikeminnow, speckled dace, California roach, and Sacramento sucker dominated sites in this group.

Twenty-four species of fish, including nine native species, were collected at the three large river sites, which are located at low elevations on the larger rivers. Native species tended to be more abundant. The most abundant native species were Sacramento pikeminnow, Sacramento sucker, tule perch, and prickly sculpin. No introduced species exceeded 7 percent of the catch.

Agricultural land sites within the Sacramento Valley were dominated by introduced species and included sites on natural and artificial waterways that were heavily influenced by agricultural land uses or water management activities. Twenty species were collected at these sites, including only three native species.

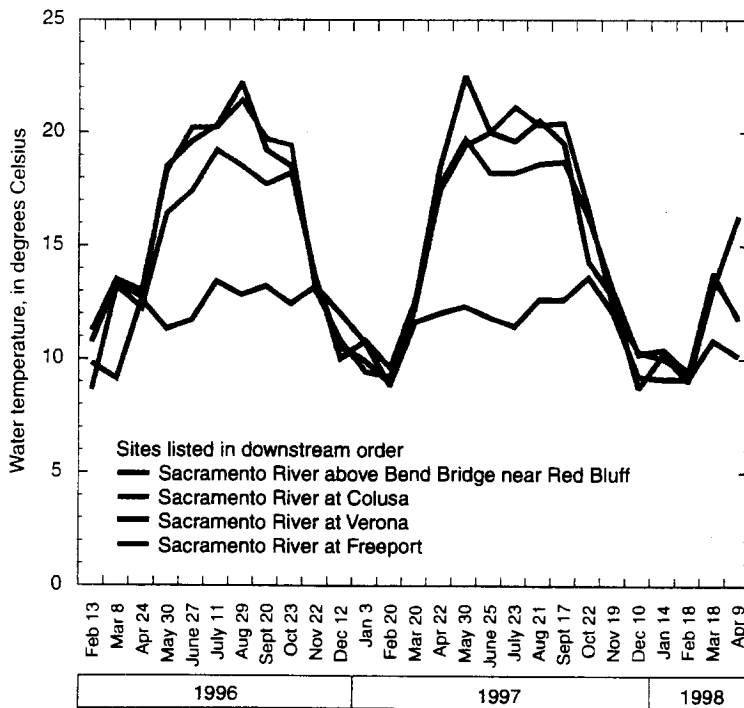


Figure 17. Temperature of the Sacramento River at select locations. Seasonal increases are caused by diversion of river water for irrigation.

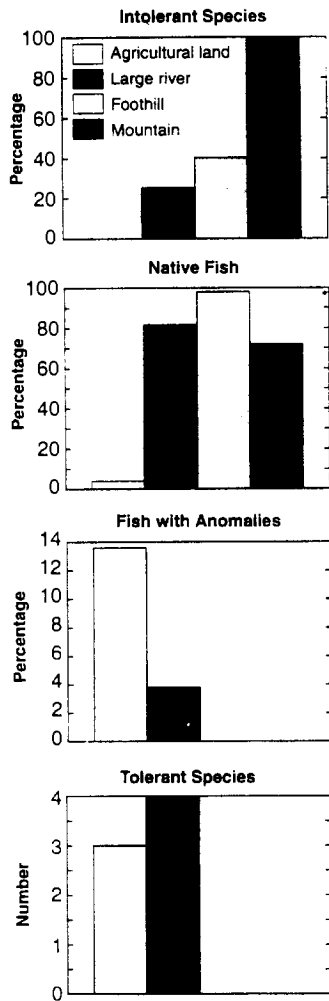


Figure 18. Characteristics of fish from ecological studies.

The group of fish at the agricultural land sites had the lowest percentage of native fish, the lowest percentage of intolerant fish, and the highest percentage of fish with external anomalies (fig. 18), indicating degraded environmental conditions. Intolerant fish are defined as those that are not adaptable to human alterations to the environment and thus decline in numbers when these alterations occur. The agricultural land site group also had a low number of native species

compared with the large river and foothill site groups. In summary, the fish site data indicate that the agricultural land site group had the most degraded environmental conditions, the foothill and mountain site groups had the best conditions, and the large river site group had somewhat intermediate conditions.

Previous studies of fish community structure in California have established correlations between the increasing numbers of introduced fish species in Central Valley streams and the disturbances caused by human activities (Moyle and Nichols, 1974; Brown and Moyle, 1993; Brown, 2000). Environmental disturbances associated with human activities include changes in water quality, streamflow, and habitat. Changes in streamflow, in particular changes in quantity and timing, have been identified as very important determinants of the structure of California fish communities (Baltz and Moyle, 1993; Moyle and Light, 1996a,b; Brown, 2000).

Differences in water quantity and water management have subsequent effects on water quality and habitat. Although these observations do not provide definitive support for the primary importance of streamflow in maintaining populations of native fish species, such relations and their effects on native fishes warrant consideration whenever changes in water management are being considered. Assessments of resident fish communities may be useful in determining the effectiveness of such changes in restoring natural ecological functions.

Streams without large reservoirs supported higher abundance of aquatic insects than streams with large reservoirs

Aquatic insect communities were compared at 23 locations including sites on two streams, Big Chico and Deer creeks, which have no major dams. Most streams sampled as part of this NAWQA Program have at least one major water project that affects a portion of the natural channels. Big Chico and Deer creeks are, therefore, unique in this study in that their flows are largely unregulated. Statistical analyses of the aquatic insect communities show that some sites on Deer and Big Chico creeks have more species and greater abundances of benthic macroinvertebrates compared with sites downstream from dams on the other streams sampled. Further research is needed to verify and evaluate these relations.

Populations of anadromous salmonids, including steelhead, rainbow trout, and chinook salmon, have declined throughout the Central Valley, resulting in protection of the remaining populations under Federal and State endangered species legislation (Yoshiyama and others, 1998). The reasons for the declines are complex and interactive; however, the construction of dams and reservoirs on California streams and rivers is widely recognized as one of the important factors (Yoshiyama and others, 1998). Dams and reservoirs block established migration routes, causing fish to reproduce in less desirable habitats. Another ecological effect from dams and reservoirs can be



AQUATIC BIOLOGY IN A NATIONAL CONTEXT

Biological indicators of water and habitat quality for 6 sites in the Sacramento River Basin were compared with similar data from 140 sites from NAWQA Study Units throughout the Nation. Because the rankings have not been calibrated for Sacramento River Basin streams, they should not be interpreted as designating “good” and “bad” water quality in streams. The sites simply score higher or lower in relation to other NAWQA sites, which represent a wide range of environmental settings.

The Yuba River site ranked among the least degraded sites nationally for all three indicators. The consistently low scores for this site can be attributed to abundant native fishes and the presence of healthy invertebrate and algal communities associated with cool water and abundant riffle habitats. The Arcade Creek and Colusa Basin Drain sites ranked among the most degraded sites nationally for all three indicators. This score may have to do more with the harsh nature of the physical environment than with water quality. Flow at these sites fluctuates widely because of storm runoff and variation in urban runoff. Biological communities in such streams tend to have few species, resulting in low indicator scores. Local determination of habitat and water quality is better accomplished using locally derived data, as has been demonstrated for Central Valley fish (Brown, 2000; May and Brown, 2000) and invertebrate communities (Brown and May, 2000a,b).

Comparisons of biological indicators of water quality from Sacramento River Basin sites with other sites with biological data from the NAWQA Program¹

Site Name	National Land Use Category ¹	Fish Status	Invertebrate Status	Algal Status
Colusa Basin Drain at Road 99E near Knights Landing	Agriculture	■	■	■
Sacramento River at Colusa	Mixed	■	■	■
Yuba River near Marysville	Mixed	■	■	■
Feather River near Nicolaus	Mixed	■	■	■
American River at Sacramento	Mixed	■	■	■
Arcade Creek near Del Paso Heights	Urban	■	■	■

¹Represents 140 sites from NAWQA Study Units throughout the United States with algal, invertebrate (primarily insects, worms, crayfish, clams), and fish data. “Mixed” indicates a combination of agricultural, urban, and other use.

- Highest 25 percent nationally (comparatively more degraded)
- Middle 50 percent nationally
- Lowest 25 percent nationally (comparatively less degraded)

changes in downstream populations of aquatic insects. This is potentially important to anadromous and resident fish populations because aquatic insects are a critical food source for nearly all species of fish at some life stage.

The implication of this study is that construction of dams and reservoirs in the foothills has either submerged the productive habitat in Central Valley streams or altered them indirectly through downstream effects on ecological conditions. The importance of other downstream effects, such as disruption of sediment transport, has been recognized for anadromous salmonids. As a result of this recognition, a variety of projects are underway or are being proposed to improve conditions for anadromous fishes and to restore ecological processes in general.

Ground Water

Ground-water quality was investigated in the portion of the aquifer of the southeastern Sacramento Valley used for domestic purposes or irrigation and in the shallow portion of the aquifer below the valley's two major land uses—rice cultivation and land that had been urbanized from 5 to 25 years ago. The southeastern Sacramento Valley was chosen for investigation because it is in that region that domestic and irrigation usage of ground water is highest. Although ground water is used in other parts of the valley, its usage is not as great as in the southeastern Sacramento Valley.

The ground-water quality of some areas of the Sacramento Valley, such as the southwestern portion, is not entirely suitable for human or agricultural use because of the presence of elevated

concentrations of boron, fluoride, chloride, nitrate, and sulfate (Hull, 1984; Davisson and Criss, 1993). The study conducted in the southeastern Sacramento Valley aquifer was designed to address the suitability of a portion of the aquifer. Specifically, only existing domestic wells were sampled.

Out of 31 wells sampled, only one sample exceeded the drinking water standard for nitrate. The median concentration of nitrate was 1.3 mg/L. Previous NAWQA investigations have found a median nitrate concentration of 0.48 mg/L in major aquifers throughout the United States (Nolan and Stoner, 2000). Therefore, the wells sampled in this study have nitrate values above the national median.

One or more pesticides were detected in 9 of 31, or 29 percent, of the wells of the southeastern Sacramento Valley (fig. 19). Simazine was detected in three wells, but the concentrations were very low and were not close to any drinking-water standard. Bentazon was one of the more frequently detected pesticides. Bentazon is a

herbicide that was used on rice, but its use was suspended in 1989 pending a review; then, in 1992, it was formally banned by the California Department of Pesticide Regulation (Miller-Maes and others, 1993). All bentazon concentrations were below drinking-water standards.

Volatile organic chemicals (VOCs) are not causing any apparent water-quality problems in the shallow aquifer of the southeastern Sacramento Valley. In one of the wells sampled, which is down-gradient from a known point source, eight different VOCs were detected. One of those VOCs (trichloroethene, measured at 5.5 µg/L) exceeded current drinking-water standards (primary MCL is 5 µg/L).

The effects of rice cultivation—one of the most prevalent agricultural practices in the Sacramento Valley—on ground water were examined by drilling and sampling 28 new wells (fig. 20). The wells were drilled to completion near the water table so that the agricultural effects on the most

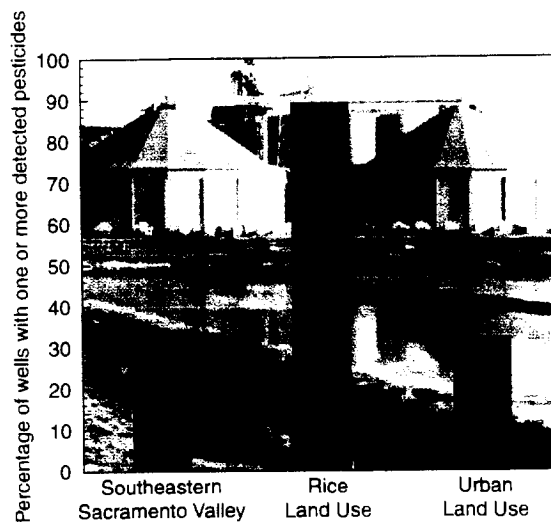


Figure 19. Percentage of wells having one or more detected pesticides for the three ground-water studies.

recent water that entered the ground could be assessed. Rice cultivation requires that fields be flooded for the duration of the growing season, which lasts from May through September.

Pesticides were more frequently detected in the wells of the rice land-use study area compared with other regions outside the rice study area. One or more pesticides were detected in 25 of 28 (89 percent) of the wells sampled (fig. 19). The most frequently detected pesticide of the rice study was bentazon, a herbicide used in rice fields until suspension in 1989 and a formal ban in 1992. Although no observed concentrations exceeded drinking-water standards, the high detection frequency, almost 10 years since the last known use, suggests that bentazon is easily transported to ground water and does not readily degrade in ground water. Molinate,

another herbicide used on rice, was detected in 7 out of 28 wells. Thiobencarb, also a herbicide, was found in three wells; carbofuran, an insecticide used on rice, was detected in four. Herbicides and insecticides are applied to rice at the same time, shortly after planting in May. The most heavily used pesticides on rice are molinate, thiobencarb, and carbofuran. Therefore, it is not surprising to see these compounds in the ground water under the rice land-use region. None of the pesticide concentrations that were measured exceeded any known water-quality standard.

Nutrient concentrations tended to be low in the ground water under the rice fields. The median nitrate concentration was 2 mg/L, and no concentrations exceeded a drinking-water standard. The median nitrate concentration

measured in previous NAWQA investigations in agricultural land use settings was 3.4 mg/L (Nolan and Stoner, 2000).

Dissolved solids were elevated in rice fields relative to the other ground water sampled in this study. The elevated concentrations are most likely related to evaporation of irrigation water, which leaves behind salt. The effects of these increases in dissolved solids on deeper portions of the aquifer are unknown.

The effects of recent urbanization on the quality of shallow ground water were investigated as part of this NAWQA study (fig. 21). The chosen metropolitan area was that of the city and surrounding counties of Sacramento. The part of the metropolitan area developed between 5 and 25 years ago was chosen for the investigation because it was assumed that the water quality of older urban land might have degraded water quality because it was developed prior to the passage of the Clean Water Act and the period of more recent environmental awareness.

Trichloromethane was the most frequently detected volatile organic chemical (16 of 19 wells). The concentrations were always very low and did not exceed any drinking-water standards. The presence of trichloromethane can be attributed to lawn irrigation using water treated by chlorination. One or more pesticides were detected in 6 of 19 (32 percent) of the wells sampled in the urban land-use study (fig. 19). Atrazine, or its degradation product, was the most frequently detected pesticide in the shallow ground water under the recently urbanized area. The



Figure 20. Drilling and installing a new monitoring well in the rice-growing region of the Sacramento Valley.

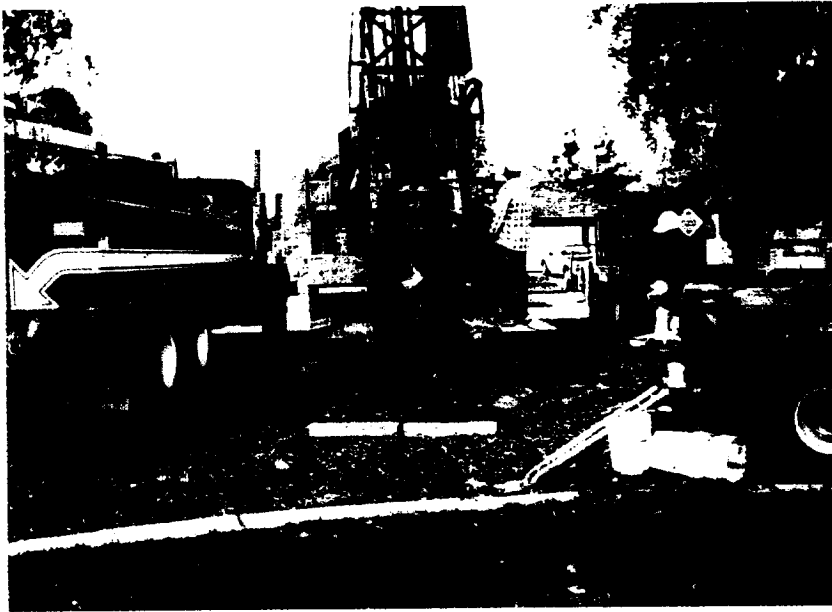


Figure 21. Drilling and installation of a new monitoring well in the Sacramento metropolitan area. This study of shallow ground water was the first of its kind within the Sacramento metropolitan area.

occurrence of atrazine in ground water could not be attributed to either current urban land use or past agricultural production. All atrazine concentrations were below drinking-water standards.

An examination of the nitrate data revealed the highest potential contamination of ground water from recent urban development. Although no wells had concentrations of nitrate above the drinking-water standard, one well had a concentration of 8 mg/L, and 5 of the 19 (26 percent) of the sampled wells had concentrations that exceeded 5 mg/L. The median concentration of nitrate was 2.4 mg/L. The median concentration of nitrate in ground water under urban areas measured in previous NAWQA investigations was 1.6 mg/L (Nolan and Stoner, 2000).

With the exception of arsenic, trace elements generally were not found to be a problem from the

perspective of human toxicity. Some wells did have high concentrations of iron and manganese. The primary drinking-water standard for cadmium (5 $\mu\text{g/L}$) was exceeded in three wells of the rice land-use study. The source of that cadmium is unknown. Arsenic exceeded the current drinking-water standard of 50 $\mu\text{g/L}$ in one urban well and approached the drinking-water standard in three other wells. Arsenic would be more problematic if the drinking-water standard were lowered to 5 $\mu\text{g/L}$, as proposed by the EPA (U.S. Environmental Protection Agency, 2000). At that level, the standard would be exceeded in 53 percent of the urban wells, 39 percent of the rice land-use wells, and 48 percent of the domestic wells sampled in the southeastern Sacramento Valley. Iron and manganese dissolve when oxygen is absent in the ground water. Although generally nontoxic, these two metals can

limit the beneficial uses of the ground water because they may precipitate when the ground water is exposed to air. The precipitation can be severe enough to adversely affect household uses of water as well as plumbing.

Radon concentrations exceeded the EPA proposed drinking-water standard of 300 picocuries per liter in 90 percent of the domestic wells sampled. The median concentration from wells sampled in the southeastern Sacramento Valley was 495 picocuries per liter. Radon is a colorless gas formed from the radioactive decay of radium. Radium is produced by the radioactive decay of uranium, which has a half-life of 4.4 billion years, whereas the half-life of solid radium is 1,620 years. Radon, on the other hand, has a half-life of only 3.8 days. Because radon is a gas, it moves easily in underground geologic environments and readily enters ground water. Regions of the country with geologic formations containing granite, volcanic rocks, certain types of shale known as dark shale, and some sedimentary and metamorphic rocks are more likely to have soil or ground water enriched in radon. Those conditions exist in the Sacramento River Basin. The health effects of consuming water containing radon at levels determined in this study have not been identified.

STUDY UNIT DESIGN

Stream Chemistry

Basic and Intensive Fixed Sites (see Glossary) were selected to assess the occurrence of dissolved compounds and select compounds associated with solid materials in stream water or streambed sediment. Basic Fixed Sites were sampled less frequently and for fewer compounds than Intensive Fixed Sites.

Basic and Intensive Fixed Sites

1. Sacramento River above Bend Bridge near Red Bluff
2. Sacramento River at Colusa
3. Yuba River at Marysville
4. Feather River near Nicolaus
5. Cache Creek at Rumsey
6. Colusa Basin Drain at Road 99E near Knights Landing
7. Sacramento Slough near Knights Landing
8. Sacramento River at Verona
9. Arcade Creek near Del Paso Heights
10. American River at Sacramento
11. Sacramento River at Freeport
12. Yolo Bypass at Interstate 80 near West Sacramento

Special Studies

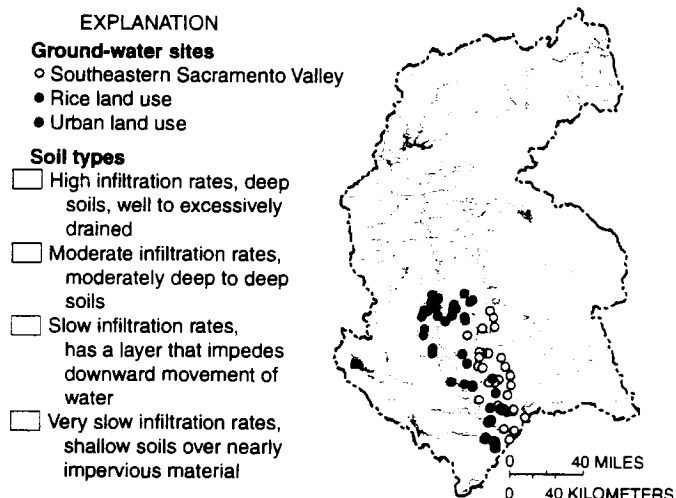
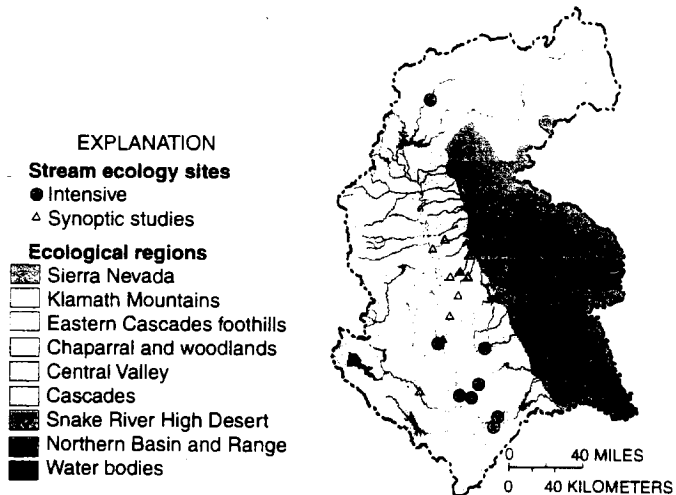
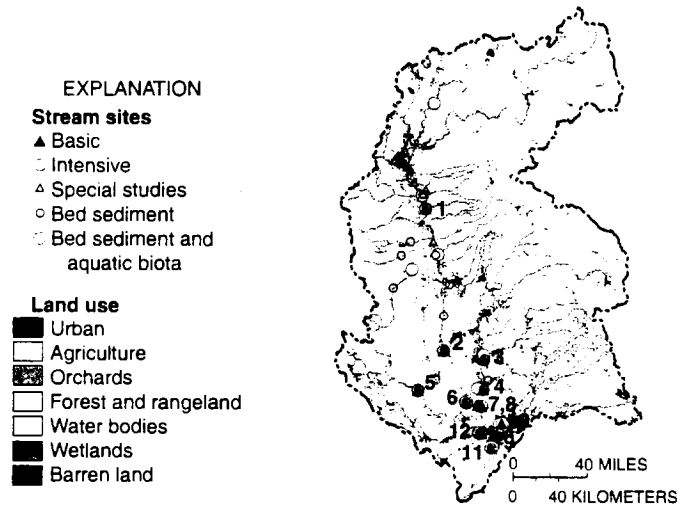
A study of metals transport from an acid mine drainage site and a generalized study of mercury transport along a reach of the Sacramento River downstream from Shasta Lake were completed.

Stream Ecology

Ecological assessments were completed along mountain to valley reaches (synoptic studies) of 3 streams, at 7 of the 12 Basic Fixed Sites, and at 1 reference site to determine variations in the community structure of aquatic biota.

Ground-Water Chemistry

Surveys of water quality in a used portion of the Sacramento Valley aquifer and the effects of agricultural and urban land uses on water quality were completed.



SUMMARY OF DATA COLLECTION IN THE SACRAMENTO RIVER BASIN, 1994-98

Study component	What data were collected and why	Types of sites sampled	Number of sites	Sampling frequency and period
Stream Chemistry				
Basic Fixed Sites— General water chemistry	Streamflow, nutrients, major ions, suspended sediment, water temperature, specific conductance, organic carbon, trace metals, mercury, dissolved oxygen, alkalinity and pH; to describe concentrations, loads, and seasonal variations.	Large rivers, most with continuous streamflow measurements available; streams with continuous streamflow measurements that drain forested, agricultural, and mining areas.	8	Monthly, 02/97-04/98
Intensive Fixed Sites— Agricultural and large river	In addition to the above constituents, 84 pesticides; to describe concentrations and seasonal variations.	One agricultural stream that drains primarily agricultural areas; one large river site near the mouth of the basin.	3	Monthly, 02/96-04/98; Monthly for pesticides, 11/96-03/97 and 08/97-04/98; twice per month for pesticides 04/97-07/97
Intensive Fixed Sites—Urban	The same constituents as Basic Fixed Sites and Intensive Fixed Sites and, in addition, 85 volatile organic compounds; to describe concentrations and seasonal variations.	One stream that drains a primarily urbanized area.	1	Monthly for pesticides and volatile organic chemicals, 11/96-12/96 and 01/98-04/98; twice monthly for pesticides, 01/97-11/97. Storm sampling for volatile organic chemicals, 04/96 and 10/96
Contaminants in streambed sediments	Trace elements and organic compounds; to determine presence of potentially toxic compounds attached to streambed sediments.	Depositional zones of large rivers and select tributaries, including fixed sites.	17 in 1996; 19 in 1997	One sampling for trace elements and organic compounds in 1995; one sampling for trace elements in 1997
Contaminants in tissues of aquatic organisms	Asiatic clams and bottom-feeding fish were collected to determine the presence of contaminants that can accumulate in tissues of aquatic organisms. The tissue samples were analyzed for trace elements and organic compounds.	Fixed sites and other select sites of large rivers and select tributaries.	17	One sampling in 1997
Stream Ecology				
Intensive Assessments	Fish, macroinvertebrates, and algae; to assess biological communities and habitat in streams representing primary ecological regions.	Sites at or near a fixed site or at a pristine or reference location.	9	Three samplings, 1996-98
Synoptic Studies	Fish, macroinvertebrates, and algae; to determine spatial distribution and community structure of aquatic species and habitat.	Sites along an elevation gradient from the Sierra Nevada to the Sacramento Valley.	14	Two samplings, 1997-98
Ground-Water Chemistry				
Aquifer Survey	Major ions, nutrients, pesticides, trace elements, volatile organic compounds, and radon; to describe the overall water quality and natural chemistry in a surficial aquifer.	Domestic wells in the southeastern Sacramento Valley.	31 wells	Once in 1996
Land-Use Effects— Agriculture (rice)	Major ions, nutrients, trace elements, and pesticides; to describe the water quality and natural chemistry in a surficial aquifer in an agricultural setting.	Newly drilled monitoring wells completed near the water table in a surficial aquifer beneath or near rice fields.	28 wells	Once in 1997
Land-Use Effects— Urban	Major ions, nutrients, pesticides, trace elements, volatile organic compounds, and radon; to describe the overall water quality and natural chemistry in a surficial aquifer.	Newly drilled monitoring wells completed near the water table in a surficial aquifer beneath a recently urbanized area.	19 wells	Once in 1998
Special Studies				
Sacramento River Trace Metals Study	Trace elements measured in whole water, filtered water, ultrafiltered water, and on colloids.	Sacramento River and select tributaries including an acid mine drainage site.	19	During selected high flow and low-flow stream conditions, 07/96-06/97

GLOSSARY

- Amalgamation**—The dissolving or blending of a metal (commonly gold and silver) in mercury to separate it from its parent material.
- Aquatic guidelines**—Specific levels of water quality which, if reached, may adversely affect aquatic life. These are nonenforceable guidelines issued by a governmental agency or other institution.
- Aquatic-life criteria**—Water-quality guidelines for protection of aquatic life. Often refers to U.S. Environmental Protection Agency water-quality criteria for protection of aquatic organisms. See also Water-quality guidelines and Water-quality standards.
- Aquifer**—A water-bearing layer of soil, sand, gravel, or rock that will yield usable quantities of water to a well.
- Basic Fixed Sites**—Sites on streams at which streamflow is measured and samples are collected for temperature, salinity, suspended sediment, major ions and metals, nutrients, and organic carbon to assess the broad-scale spatial and temporal character and transport of inorganic constituents of stream water in relation to hydrologic conditions and environmental settings.
- Bed sediment**—The material at the bottom of a stream or other watercourse.
- Benthic invertebrates**—Insects, mollusks, crustaceans, worms, and other organisms without a backbone that live in, on, or near the bottom of lakes, streams, or oceans.
- Bioaccumulation**—The biological sequestering of a substance at a higher concentration than that at which it occurs in the surrounding environment or medium. Also, the process whereby a substance enters organisms through the gills, epithelial tissues, dietary, or other sources.
- Bioavailability**—The capacity of a chemical constituent to be taken up by living organisms either through physical contact or by ingestion.
- Criterion**—A standard rule or test on which a judgment or decision can be based.
- Dissolved solids**—Amount of minerals, such as salt, that are dissolved in water; amount of dissolved solids is an indicator of salinity or hardness.
- Drainage basin**—The portion of the surface of the earth that contributes water to a stream through overland runoff, including tributaries and impoundments.
- Drinking-water standard or guideline**—A threshold concentration in a public drinking-water supply, designed to protect human health. As defined here, standards are U.S. Environmental Protection Agency regulations that specify the maximum contamination levels for public water systems required to protect the public welfare; guidelines have no regulatory status and are issued in an advisory capacity.
- Ecological studies**—Studies of biological communities and habitat characteristics to evaluate the effects of physical and chemical characteristics of water and hydrologic conditions on aquatic biota and to determine how biological and habitat characteristics differ among environmental settings in NAWQA Study Units.
- Ecoregion**—An area of similar climate, landform, soil, potential natural vegetation, hydrology, or other ecologically relevant variables.
- Ecosystem**—The interacting populations of plants, animals, and microorganisms occupying an area, plus their physical environment.
- Ground water**—In general, any water that exists beneath the land surface, but more commonly applied to water in fully saturated soils and geologic formations.
- Habitat**—The part of the physical environment where plants and animals live.
- Hydrograph**—Graph showing variation of water elevation, velocity, streamflow, or other property of water with respect to time.
- Intensive Fixed Sites**—Basic Fixed Sites with increased sampling frequency during selected seasonal periods and analysis of dissolved pesticides for 1 year.
- Maximum contaminant level (MCL)**—Maximum permissible level of a contaminant in water that is delivered to any user of a public water system. MCLs are enforceable standards established by the U.S. Environmental Protection Agency.
- Median**—The middle or central value in a distribution of data ranked in order of magnitude. The median is also known as the 50th percentile.
- Micrograms per liter ($\mu\text{g/L}$)**—A unit expressing the concentration of constituents in solution as weight (micrograms) of solute per unit volume (liter) of water; equivalent to one part per billion in most stream water and ground water. One thousand micrograms per liter equals 1 milligram per liter (mg/L).
- Nitrate**—An ion consisting of nitrogen and oxygen (NO_3^-). Nitrate is a plant nutrient and is very mobile in soils.
- Nonpoint source**—A pollution source that cannot be defined as originating from discrete points such as pipe discharge. Areas of fertilizer and pesticide applications, atmospheric deposition, manure, and natural inputs from plants and trees are types of nonpoint source pollution.
- Nutrient**—An element or compound essential for animal and plant growth. Common nutrients in fertilizer include nitrogen, phosphorus, and potassium.

- Organochlorine compound**—Synthetic organic compounds containing chlorine. As generally used, the term refers to compounds containing mostly or exclusively carbon, hydrogen, and chlorine. Examples include organochlorine insecticides, polychlorinated biphenyls, and some solvents containing chlorine.
- Organophosphate insecticides**—A class of insecticides derived from phosphoric acid. They tend to have high acute toxicity to vertebrates. Although readily metabolized by vertebrates, some metabolic products are more toxic than the parent compound.
- Phosphorus**—A nutrient essential for growth that can play a key role in stimulating aquatic growth in lakes and streams.
- Plankton**—Floating or weakly swimming organisms whose migration is controlled by waves and currents. Animals of the group are called zooplankton and the plants are called phytoplankton.
- Point source**—A source at a discrete location such as a discharge pipe, drainage ditch, tunnel, well, concentrated livestock operation, or floating craft.
- Polychlorinated biphenyls (PCBs)**—A mixture of chlorinated derivatives of biphenyl, marketed under the trade name Aroclor with a number designating the chlorine content (such as Aroclor 1260). PCBs were used in transformers and capacitors for insulating purposes and in gas pipeline systems as a lubricant. Further sale for new use was banned by law in 1979.
- Radon**—A naturally occurring, colorless, odorless, radioactive gas formed by the disintegration of the element radium; damaging to human lungs when inhaled.
- Recharge**—Water that infiltrates the ground and reaches the saturated zone.
- Runoff**—Excess rainwater or snowmelt that is transported to streams by overland flow, tile drains, or ground water.
- Semivolatile organic compound (SVOC)**—Operationally defined as a group of synthetic organic compounds that are solvent-extractable and that can be determined by gas chromatography/mass spectrometry. SVOCs include phenols, phthalates, and polycyclic aromatic hydrocarbons (PAHs).
- Suspended sediment**—Particles of rock, sand, soil, and organic detritus carried in suspension in the water column, in contrast to sediment that moves on or near the streambed.
- Tolerant species**—Those species that are adaptable to (tolerant of) human alterations to the environment and often increase in number when human alterations occur.
- Trace element**—An element found in only minor amounts (concentrations less than 1.0 milligram per liter) in water or sediment; includes arsenic, cadmium, chromium, copper, lead, mercury, nickel, and zinc.
- Volatile organic compounds (VOCs)**—Organic chemicals that have a high vapor pressure relative to their water solubility. VOCs include components of gasoline, fuel oils, and lubricants, as well as organic solvents, fumigants, some inert ingredients in pesticides, and some by-products of chlorine disinfection.
- Water-quality guidelines**—Specific levels of water quality which, if reached, may adversely affect human health or aquatic life. These are nonenforceable guidelines issued by a governmental agency or other institution.
- Water-quality standards**—State-adopted and U.S. Environmental Protection Agency-approved ambient standards for water bodies. Standards include the use of the water body and the water-quality criteria that must be met to protect the designated use or uses.
- Watershed**—See Drainage basin.
- Water table**—The point below the land surface where ground water is first encountered and below which the earth is saturated. Depth to the water table varies widely across the country.
- Wetlands**—Ecosystems whose soil is saturated for long periods seasonally or continuously, including marshes, swamps, and ephemeral ponds.

REFERENCES

- Alpers, C.N., Antweiler, R.C., Taylor, H.E., Dileanis, P.D., and Domagalski, J.L., 2000a. Metals transport in the Sacramento River, California, 1996–1997—Interpretation of metal loads: U.S. Geological Survey Water-Resources Investigations Report 00-4002, v. 2, 120 p.
- Alpers, C.N., Taylor, H.E., and Domagalski, J.L., eds., 2000b. Metals transport in the Sacramento River, California, 1996–1997—Methods and data: U.S. Geological Survey Water-Resources Investigations Report 99-4286, v. 1, 430 p.
- Amato, J.R., Mount, D.I., Durhan, E.J., Lukasewycz, M.T., Ankley, G.T., and Robert, E.D., 1992. An example of the identification of diazinon as a primary toxicant in an effluent: *Environmental Toxicology and Chemistry*, v. 11, p. 209–216.
- Baltz, D.M., and Moyle, P.B., 1993. Invasion resistance to introduced fishes by a native assemblage of California stream fishes: *Ecological Applications*, v. 3, p. 246–255.
- Bennett, K.P., Singhasemanon, N., Miller, N., and Gallavan, R., 1998. Rice pesticides monitoring in the Sacramento Valley, 1995: Department of Pesticide Regulation, Environmental Hazards Assessment Program, Report Number EH 98-03, California Environmental Protection Agency, 45 p., appendices.
- Brown, L.R., 2000. Fish communities and their associations with environmental variables, lower San Joaquin River drainage, California: *Environmental Biology of Fishes*, v. 57, p. 251–269.
- Brown, L.R., and May, J.T., 2000a. Macroinvertebrate assemblages on woody debris and their relations with environmental variables in the lower Sacramento and San Joaquin river drainages, California: *Environmental Monitoring and Assessment*, v. 64, p. 311–329.
- 2000b. Benthic macroinvertebrate assemblages and their relations with environmental variables in the Sacramento and San Joaquin river drainages, California, 1993–1997: U.S. Geological Survey Water-Resources Investigations Report 00-4125, 25 p.
- Brown, L.R., and Moyle, P.B., 1993. Distribution, ecology, and status of fishes of the San Joaquin drainage, California: *California Fish and Game*, v. 79, p. 96–114.
- Cain, D.J., Carter, J.L., Fend, S.V., Luoma, S.N., Alpers, C.N., and Taylor, H.E., 2000. Metal exposures in a benthic macroinvertebrate, *Hydropsyche californica*, related to mine drainage in the Sacramento River: *Canadian Journal of Fisheries and Aquatic Sciences*, v. 57, p. 380–390.
- California Department of Water Resources, 1998. California water plan update: California Department of Water Resources Bulletin 160-98.
- Canadian Council of Ministers of the Environment, 1995. Protocol for the derivation of Canadian sediment quality guidelines for the protection of aquatic life: Evaluation and Interpretation Branch, Environment Canada Report CCME EPC-98E, 38 p.
- Davis, J.A., May, M.D., Ichikawa, G., and Crane, D., 2000. Contaminant concentrations in fish from the Sacramento-San Joaquin Delta and lower San Joaquin River, 1998: Richmond, Calif., San Francisco Estuary Institute, 52 p.
- Davisson, M.L., and Criss, R.E., 1993. Stable isotope imaging of a dynamic groundwater system in the southwestern Sacramento Valley, California: *Journal of Hydrology*, v. 144, p. 213–246.
- de Vlaming, V., Connor, V., DiGiorgio, C., Bailey, H. C., Deanovic, L.A., and Hinton, D.E., 2000. Application of whole effluent toxicity test procedures to ambient water quality assessment: *Environmental Toxicology and Chemistry*, v. 19, p. 42–62.
- Domagalski, J.L., 1996. Pesticides and pesticide degradation products in stormwater runoff—Sacramento River Basin, California: *Journal of the American Water Resources Association*, v. 32, p. 953–964.
- 1998. Occurrence and transport of mercury and methyl mercury in the Sacramento River Basin, California: *Journal of Geochemical Exploration*, v. 64, p. 227–291.
- 2000. Pesticides in surface water measured at selected sites in the Sacramento River Basin, California, 1996–1997: U.S. Geological Survey Water-Resources Investigations Report 00-4203, 24 p.
- Domagalski, J.L., and Brown, L.R., 1998. National Water-Quality Assessment Program—The Sacramento River Basin: U.S. Geological Survey Fact Sheet FS 94-029, 2 p.
- Domagalski, J.L., and Dileanis, P.D., 2000. Water quality assessment of the Sacramento River Basin, California—Water quality of fixed sites, 1996–1998: U.S. Geological Survey Water-Resources Investigations Report 00-4247, 60 p.
- Domagalski, J.L., Knifong, D.L., MacCoy, D.E., Dileanis, P.D., Dawson, B.J., and Majewski, M.S., 1998. Water quality assessment of the Sacramento River Basin, California—Environmental setting and study design: U.S. Geological Survey Water-Resources Investigations Report 97-4254, 31 p.
- Harrington, J.M., 1990. Hazard assessment of the rice herbicides molinate and thiobencarb to aquatic organisms in the Sacramento River system: State of California, The Resources Agency, Department of Fish and Game, Environmental Services Division, Administrative Report 90-1, 97 p.
- Hull, L.C., 1984. Geochemistry of groundwater in the Sacramento Valley, California: U.S. Geological Survey Professional Paper 1401B, 36 p.

- Hunerlach, M.P., Rytuba, J.J., and Alpers, C.N., 1999. Mercury contamination from hydraulic placer-gold mining in the Dutch Flat Mining District, California. *in* Morganwalp, D.W., and Buxton, H.T., eds., U.S. Geological Survey Toxic Substances Hydrology Program—Proceedings of the Technical Meeting, March 8–12, 1999, v. 2: U.S. Geological Survey Water-Resources Investigations Report 99-4018B, p. 179–190.
- Kuivila, K.M., and Foe, C.G., 1995, Concentrations, transport, and biological effects of dormant spray pesticides in the San Francisco Estuary, California: *Environmental Toxicology and Chemistry*, v. 14, p. 1141–1150.
- MacCoy, D.E., Crepeau, K.L., and Kuivila, K.M., 1995, Dissolved pesticide data for the San Joaquin River at Vernalis and the Sacramento River at Sacramento, California, 1991–1994: U.S. Geological Survey Open-File Report 95-110, 27 p.
- MacCoy, D.E., and Domagalski, J.L., 1999, Trace elements and organic compounds in streambed sediment and aquatic biota from the Sacramento River Basin, California, October and November, 1995: U.S. Geological Survey Water-Resources Investigations Report 99-4151, 37 p.
- Marshack, J.B., 1995, A compilation of water quality goals: Sacramento, Calif., California Regional Water Quality Control Board, 51 p.
- May, J.T., and Brown, L.R., 2000, Fish community structure in relation to environmental variables within the Sacramento River Basin and implications for greater Central Valley, California: U.S. Geological Survey Open-File Report 00-247, 19 p.
- Miller-Maes, C., Pepple, M., Troiano, J., and Weaver, D., 1993, Sampling for pesticide residues in California well water, 1993 update—Well inventory database: California Department of Pesticide Regulation Report EH-9306, 167 p.
- Moyle, P.B. and Light, T., 1996a, Biological invasions of freshwater—Empirical rules and assembly theory: *Biological Conservation*, v. 78, p. 149–162.
- 1996b, Fish invasions in California—Do abiotic factors determine success?: *Ecology*, v. 77, p. 1666–1670.
- Moyle, P.B., and Nichols, R.D., 1974, Decline of the native fish fauna of the Sierra Nevada foothills, central California: *The American Midland Naturalist*, v. 92, p. 72–83.
- Newell, A.J., Johnson, D.W., and Allen, L.K., 1987, Niagara River Biota Contamination Project—Fish flesh criteria for piscivorous wildlife: New York State Department of Environmental Conservation, Division of Fish and Wildlife, Bureau of Environmental Protection, Technical Report 87-3.
- Nolan B.T., and Stoner, J.D., 2000, Nutrients in ground waters of the conterminous United States, 1992–1995: *Environmental Science and Technology*, v. 34, p. 1156–1165.
- Rudd, J.W.M., 1995, Sources of methylmercury to freshwater ecosystems—A Review: *Water, Air, and Soil Pollution*, v. 80, p. 697–713.
- Slotton, D.G., Ayers, S.M., Reuter, J.E., Goldman, C.R., 1997, Gold mining impacts on food chain mercury in northwestern Sierra Nevada streams, *in* Larry Walker and Associates, Sacramento River Mercury Control Planning Project: Final Project Report, 71 p.
- U.S. Department of Commerce, 1992, 1990 Census of population and housing—Summary social, economic, and housing characteristics: U.S. Department of Commerce, Economics and Statistics Administration, Bureau of the Census, variously paged.
- U.S. Environmental Protection Agency, California 1993 Section 303(d) list: California State Water Resources Control Board, accessed January 2, 2000, at URL <http://www.epa.gov/region09/water/tmdl/calist/list5.html>.
- 1991a, Technical support document for water quality-based toxics control (2d ed.): U.S. Environmental Protection Agency, Office of Water, EPA/505/2-90001.
- 1991b, Methods for aquatic toxicity characterization procedures (2d ed.): U.S. Environmental Protection Agency, Office of Research and Development, EPA/600/6-91003.
- 1999, National recommended water quality criteria—Correction: U.S. Environmental Protection Agency, Office of Water, EPA 822-Z-99-001, 25 p.
- 2000, Drinking water standards and health advisories: Environmental Protection Agency, Office of Water, EPA/822/B-00001.
- Yoshiyama, R.M., Fisher, F.W., and Moyle, P.B., 1998, Historical abundance and decline of chinook salmon in the Central Valley region of California: *North American Journal of Fisheries Management*, v. 18, p. 487–521.

APPENDIX—WATER-QUALITY DATA FROM THE SACRAMENTO RIVER BASIN IN A NATIONAL CONTEXT

For a complete view of Sacramento River Basin data and for additional information about specific benchmarks used, visit our Web site at <http://water.usgs.gov/nawqa/>. Also visit the NAWQA Data Warehouse for access to NAWQA data sets at <http://infotrek.er.usgs.gov/wdbctc/nawqa/home>.

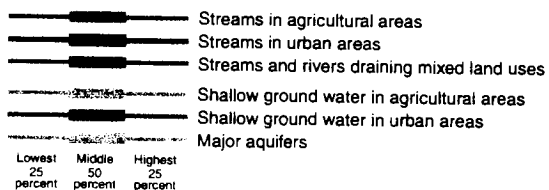
This appendix is a summary of chemical concentrations and biological indicators assessed in the Sacramento River Basin. Selected results for this basin are graphically compared to results from as many as 36 NAWQA Study Units investigated from 1991 to 1998 and to national water-quality benchmarks for human health, aquatic life, or fish-eating wildlife. The chemical and biological indicators shown were selected on the basis of frequent detection, detection at concentrations above a national benchmark, or regulatory or scientific importance. The graphs illustrate how conditions associated with each land use sampled in the Sacramento River Basin compare to results from across the Nation, and how conditions compare among the several land uses. Graphs for chemicals show only detected concentrations and, thus, care must be taken to evaluate detection frequencies in addition to concentrations when comparing study-unit and national results. For example, molinate concentrations in Sacramento River Basin agricultural streams were similar to the national distribution, but the detection frequency was much higher (100 percent compared to 7 percent).

CHEMICALS IN WATER

Concentrations and detection frequencies, Sacramento River Basin, 1994–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals

- ◆ Detected concentration in Study Unit
- 66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- Not measured or sample size less than two
- 12 Study-unit sample size. For ground water, the number of samples is equal to the number of wells sampled

National ranges of detected concentrations, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected



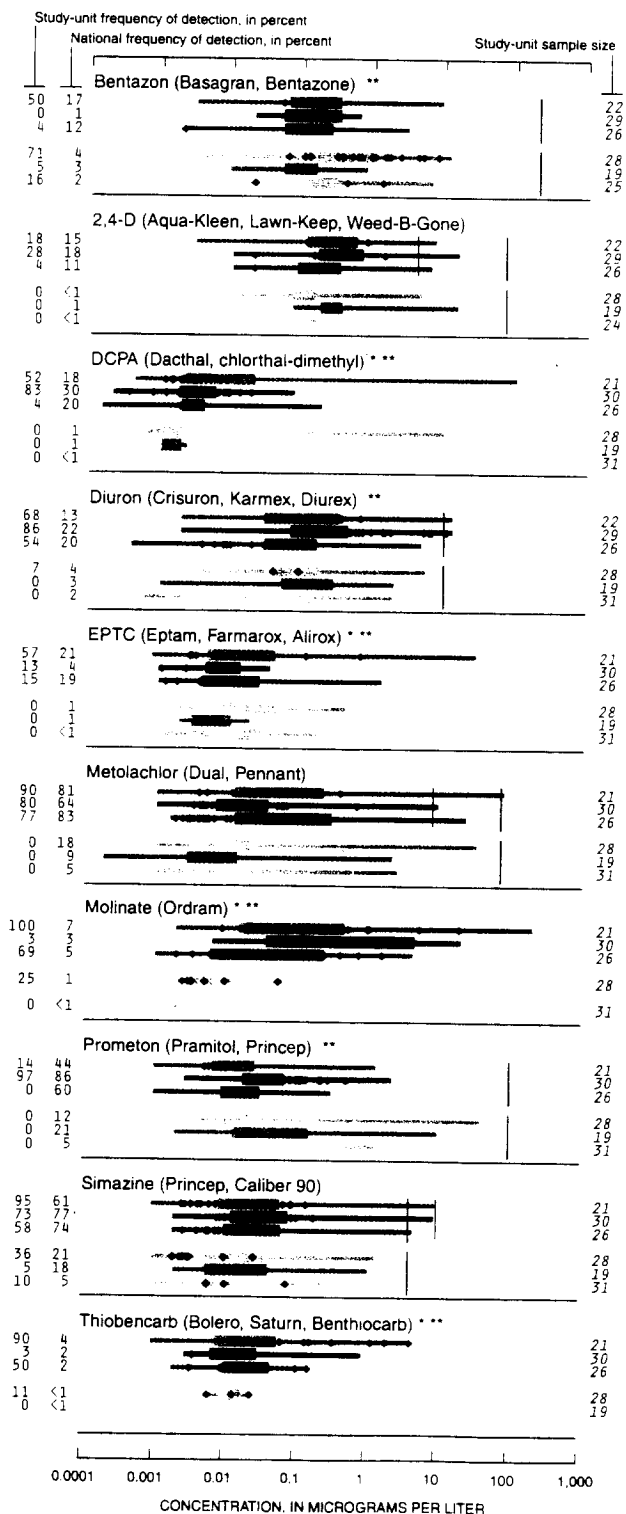
National water-quality benchmarks

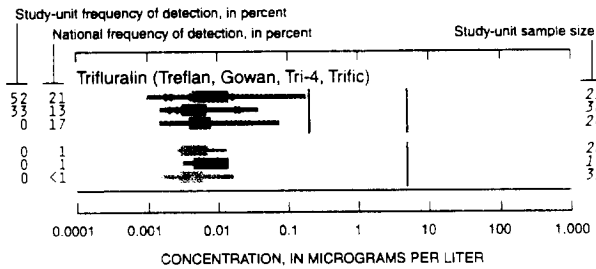
National benchmarks include standards and guidelines related to drinking-water quality, criteria for protecting the health of aquatic life, and a goal for preventing stream eutrophication due to phosphorus. Sources include the U.S. Environmental Protection Agency and the Canadian Council of Ministers of the Environment

- | Drinking-water quality (applies to ground water and surface water)
- | Protection of aquatic life (applies to surface water only)
- | Prevention of eutrophication in streams not flowing directly into lakes or impoundments

- No benchmark for drinking-water quality
- ** No benchmark for protection of aquatic life

Pesticides in water—Herbicides





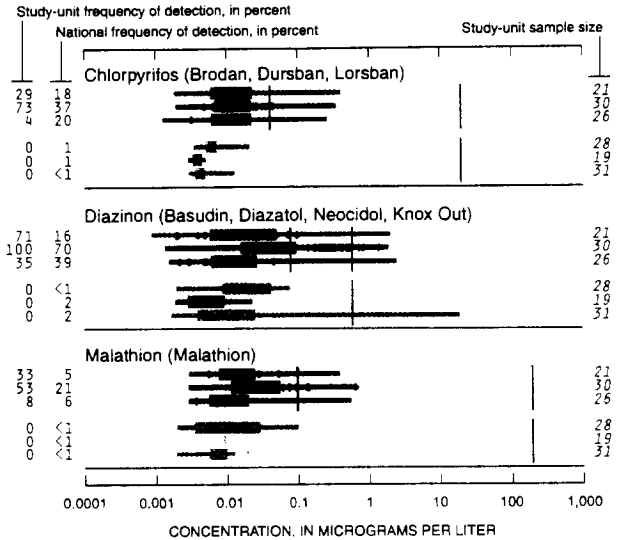
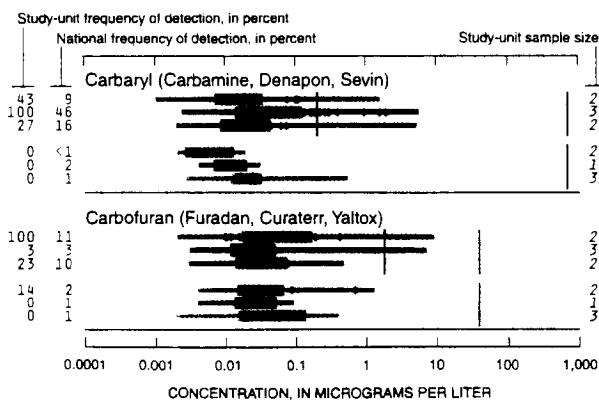
Other herbicides detected

- Alachlor (Lasso, Bronco, Lariat, Bullet) **
- Atrazine (AAtrex, Atrex, Atred, Gesaprim)
- Benfluralin (Balan, Benefin, Bonalan) ***
- Bromacil (Hyvar X, Urox B, Bromax)
- Bromoxynil (Buctril, Brominal) *
- Cyanazine (Bladex, Fortrol)
- Deethylatrazine (Atrazine breakdown product) ***
- Dichlorprop (2,4-DP, Seritox 50, Lentemul) ***
- 2,6-Diethylaniline (Alachlor breakdown product) ***
- MCPA (Rhomene, Rhonox, Chiptox)
- Metribuzin (Lexone, Sencor)
- Napropamide (Devrinol) ***
- Norflurazon (Evital, Predict, Solicam, Zorial) ***
- Oryzalin (Surflan, Dirimal) ***
- Pebulate (Tillam, PEBC) ***
- Pendimethalin (Pre-M, Prowl, Stomp) **
- Pronamide (Kerb, Propyzamid) **
- Propanil (Stam, Stampede, Wham) **
- Tebuthiuron (Spike, Tebusan)
- Triclopyr (Garlon, Grandstand, Redeem, Remedy) ***

Herbicides not detected

- Acetochlor (Harness Plus, Surpass) ***
- Acifluorfen (Blazer, Tackle 2S) **
- Butylate (Sutan +, Genate Plus, Butilate) **
- Chloramben (Amiben, Amilon-WP, Vegiben) **
- Clopyralid (Stinger, Lontrel, Transline) ***
- 2,4-DB (Butyrac, Butoxone, Embutox Plus, Embutone) ***
- Dacthal mono-acid (Dacthal breakdown product) ***
- Dicamba (Banvel, Dianat, Scotts Proturf)
- Dinoseb (Dinosebe)
- Ethalfuralin (Sonaian, Curbit) ***
- Fenuron (Fenuon, Fenidim) ***
- Fluometuron (Flo-Met, Cotoran) **
- Linuron (Lorox, Linex, Sarclax, Linurex, Afalon) *
- MCPB (Thistrol) ***
- Neburon (Neburea, Neburyl, Noruben) ***
- Picloram (Grazone, Tordon)
- Propachlor (Ramrod, Satecid) **
- Propham (Tuberite) **
- 2,4,5-T **
- 2,4,5-TP (Silvex, Fenoprop) **
- Terbacil (Sinbar) **
- Triallate (Far-Go, Avadex BW, Tri-allate) *

Pesticides in water—Insecticides



Other insecticides detected

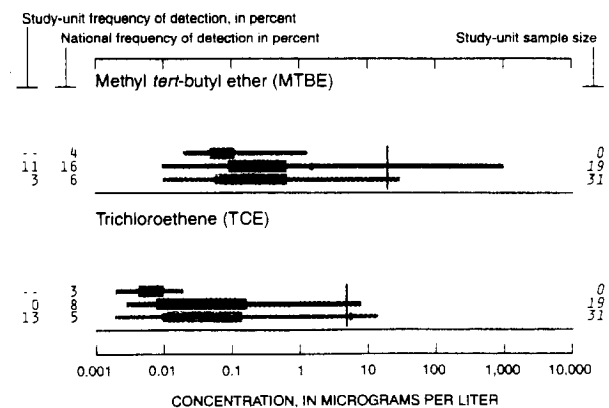
- Azinphos-methyl (Guthion, Gusathion M) *
- p,p'-DDE
- Propargite (Comite, Omite, Ornamate) **

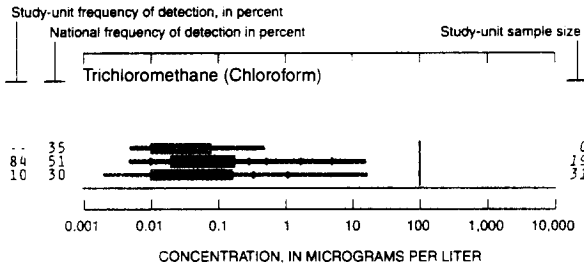
Insecticides not detected

- Aldicarb (Temik, Ambush, Pounce)
- Aldicarb sulfone (Standak, aldoxycarb)
- Aldicarb sulfoxide (Aldicarb breakdown product)
- Dieldrin (Panoram D-31, Octalox, Compound 497)
- Disulfoton (Disyston, Di-Syston) **
- Ethoprop (Mocap, Ethoprophos) ***
- Fonofos (Dyfonate, Capfos, Cudgei, Tycap) **
- alpha-HCH (alpha-BHC, alpha-lindane) **
- gamma-HCH (Lindane, gamma-BHC)
- 3-Hydroxycarbofuran (Carbofuran breakdown product) ***
- Methiocarb (Slug-Geta, Grandslam, Mesurol) ***
- Methomyl (Lanox, Lannate, Acinate) **
- Methyl parathion (Pennacap-M, Folidol-M) **
- Oxamyl (Vydate L, Pratt) **
- Parathion (Roethyl-P, Alkron, Panthion, Phoskil) *
- cis-Permethrin (Ambush, Astro, Pounce) ***
- Phorate (Thimet, Granutox, Geomet, Rampart) ***
- Propoxur (Baygon, Blattanex, Unden, Proprotox) ***
- Terbufos (Conraven, Counter, Pilarfox) **

Volatile organic compounds (VOCs) in ground water

These graphs represent data from 16 Study Units, sampled from 1996 to 1998





Other VOCs detected

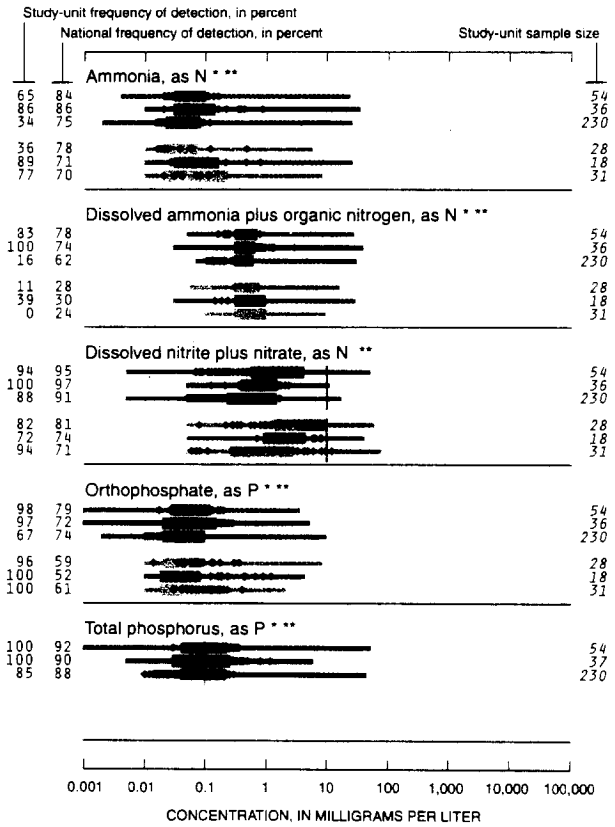
- Bromodichloromethane (Dichlorobromomethane)
- Carbon disulfide *
- Chloromethane (Methyl chloride)
- Dichlorodifluoromethane (CFC 12, Freon 12)
- 1,2-Dichloroethane (Ethylene dichloride)
- 1,1-Dichloroethane (Ethylidene dichloride) *
- cis*-1,2-Dichloroethene ((Z)-1,2-Dichloroethene)
- Ethenylbenzene (Styrene)
- Iodomethane (Methyl iodide) *
- Tetrachloroethene (Perchloroethene)
- Tetrachloromethane (Carbon tetrachloride)
- 1,1,1-Trichloroethane (Methylchloroform)
- Trichlorofluoromethane (CFC 11, Freon 11)
- 1,2,4-Trimethylbenzene (Pseudocumene) *

VOCs not detected

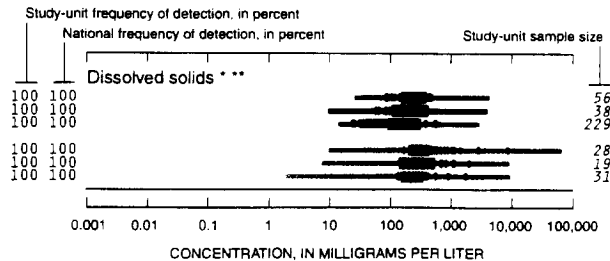
- tert*-Amyl methyl ether (*tert*-amyl methyl ether (TAME)) *
- Benzene
- Bromobenzene (Phenyl bromide) *
- Bromochloromethane (Methylene chlorobromide)
- Bromoethene (Vinyl bromide) *
- Bromomethane (Methyl bromide)
- 2-Butanone (Methyl ethyl ketone (MEK)) *
- n*-Butylbenzene (1-Phenylbutane) *
- sec*-Butylbenzene *
- tert*-Butylbenzene *
- 3-Chloro-1-propene (3-Chloropropene) *
- 1-Chloro-2-methylbenzene (*o*-Chlorotoluene)
- 1-Chloro-4-methylbenzene (*p*-Chlorotoluene)
- Chlorobenzene (Monochlorobenzene)
- Chlorodibromomethane (Dibromochloromethane)
- Chloroethane (Ethyl chloride) *
- Chloroethene (Vinyl chloride)
- 1,2-Dibromo-3-chloropropane (DBCP, Nemagon)
- 1,2-Dibromoethane (Ethylene dibromide, EDB)
- Dibromomethane (Methylene dibromide) *
- trans*-1,4-Dichloro-2-butene ((Z)-1,4-Dichloro-2-butene) *
- 1,2-Dichlorobenzene (*o*-Dichlorobenzene)
- 1,3-Dichlorobenzene (*m*-Dichlorobenzene)
- 1,4-Dichlorobenzene (*p*-Dichlorobenzene)
- 1,1-Dichloroethene (Vinylidene chloride)
- trans*-1,2-Dichloroethene ((E)-1,2-Dichloroethene)
- Dichloromethane (Methylene chloride)
- 1,2-Dichloropropane (Propylene dichloride)
- 2,2-Dichloropropane *
- 1,3-Dichloropropane (Trimethylene dichloride) *
- trans*-1,3-Dichloropropene ((E)-1,3-Dichloropropene)
- cis*-1,3-Dichloropropene ((Z)-1,3-Dichloropropene)
- 1,1-Dichloropropene *
- Diethyl ether (Ethyl ether) *
- Diisopropyl ether (Diisopropylether (DIPE)) *
- 1,2-Dimethylbenzene (*o*-Xylene)
- 1,3 & 1,4-Dimethylbenzene (*m*-&*p*-Xylene)
- 1-4-Epoxy butane (Tetrahydrofuran, Diethylene oxide) *
- Ethyl methacrylate *
- Ethyl *tert*-butyl ether (Ethyl-*t*-butyl ether (ETBE)) *
- 1-Ethyl-2-methylbenzene (2-Ethyltoluene) *
- Ethylbenzene (Phenylethane)
- Hexachlorobutadiene
- 1,1,1,2,2,2-Hexachloroethane (Hexachloroethane)
- 2-Hexanone (Methyl butyl ketone (MBK)) *
- Isopropylbenzene (Cumene) *
- p*-Isopropyltoluene (*p*-Cymene) *
- Methyl acrylonitrile *

- Methyl-2-methacrylate (Methyl methacrylate) *
- 4-Methyl-2-pentanone (Methyl isobutyl ketone (MIBK)) *
- Methyl-2-propenoate (Methyl acrylate) *
- Methylbenzene (Toluene)
- Naphthalene
- 2-Propanone (Acetone) *
- 2-Propenenitrile (Acrylonitrile)
- n*-Propylbenzene (Isocumene) *
- 1,1,2,2-Tetrachloroethane *
- 1,1,1,2-Tetrachloroethane
- 1,2,3,4-Tetramethylbenzene (Prehnitene) *
- 1,2,3,5-Tetramethylbenzene (Isodurene) *
- Tribromomethane (Bromoform)
- 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon 113) *
- 1,2,4-Trichlorobenzene
- 1,2,3-Trichlorobenzene *
- 1,1,2-Trichloroethane (Vinyl trichloride)
- 1,2,3-Trichloropropane (Allyl trichloride)
- 1,2,3-Trimethylbenzene (Hemimellitene) *

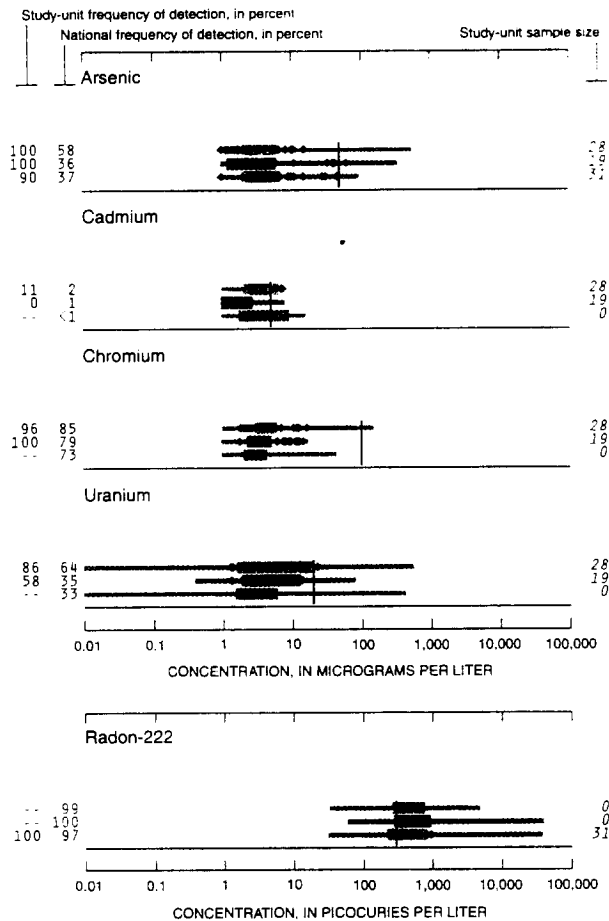
Nutrients in water



Dissolved solids in water



Trace elements in ground water



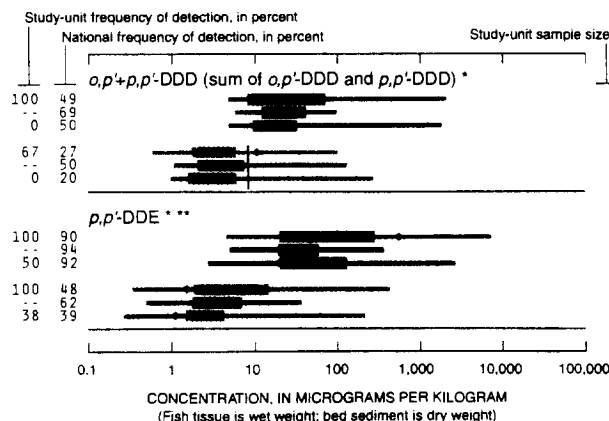
Other trace elements detected

Selenium
Zinc

Trace elements not detected

Lead

Organochlorines in fish tissue (whole body) and bed sediment



CHEMICALS IN FISH TISSUE AND BED SEDIMENT

Concentrations and detection frequencies, Sacramento River Basin, 1994-98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals. Study-unit frequencies of detection are based on small sample sizes; the applicable sample size is specified in each graph

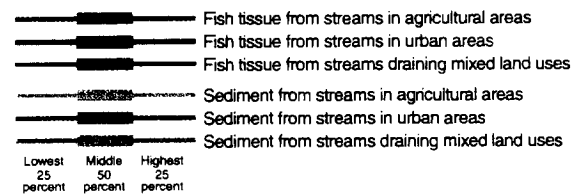
◆ Detected concentration in Study Unit

56 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency

--- Not measured or sample size less than two

12 Study-unit sample size

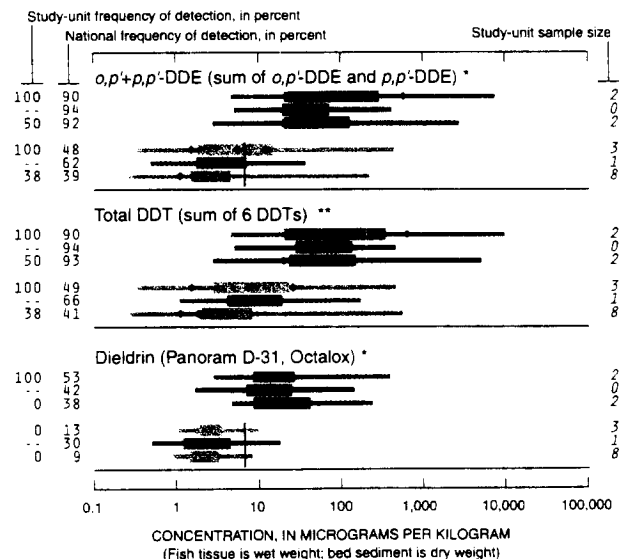
National ranges of concentrations detected, by land use, in 36 NAWQA Study Units, 1991-98—Ranges include only samples in which a chemical was detected

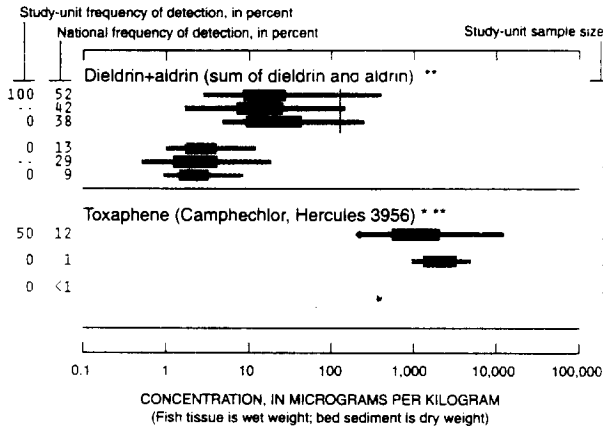


National benchmarks for fish tissue and bed sediment

National benchmarks include standards and guidelines related to criteria for protection of the health of fish-eating wildlife and aquatic organisms. Sources include the U.S. Environmental Protection Agency, other Federal and State agencies, and the Canadian Council of Ministers of the Environment

- | Protection of fish-eating wildlife (applies to fish tissue)
- | Protection of aquatic life (applies to bed sediment)
- No benchmark for protection of fish-eating wildlife
- No benchmark for protection of aquatic life





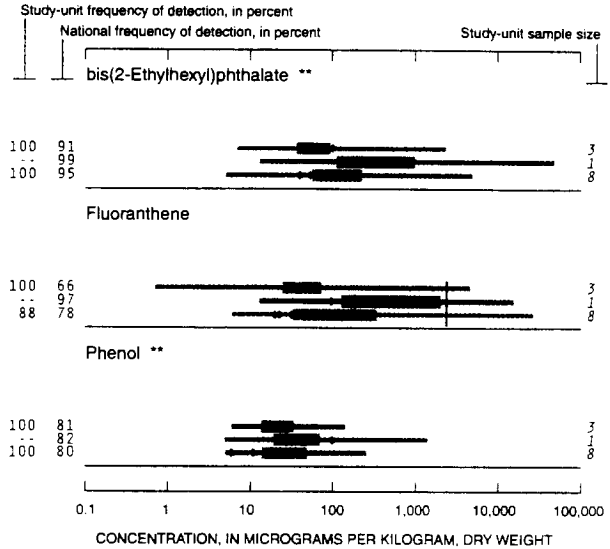
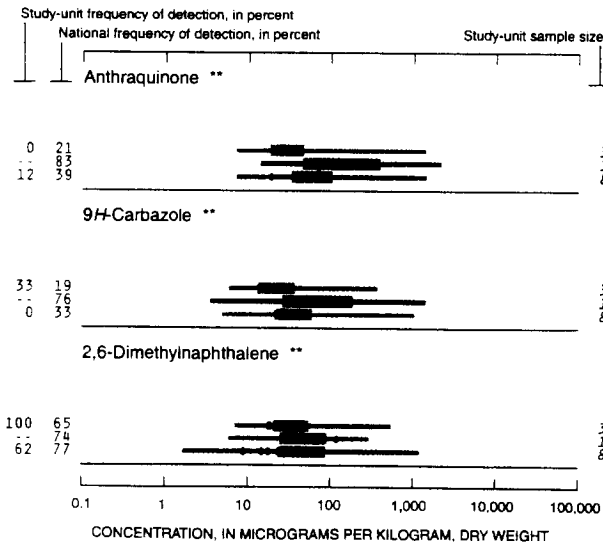
Other organochlorines detected

o,p'+*p,p'*-DDT (sum of *o,p'*-DDT and *p,p'*-DDT) *

Organochlorines not detected

- Chloroneb (Chloronebe, Demosan) * **
- DCPA (Dacthal, chlorthal-dimethyl) * **
- Endosulfan I (alpha-Endosulfan, Thiodan) * **
- Endrin (Endrine)
- gamma-HCH (Lindane, gamma-BHC, Gammexane) *
- Total-HCH (sum of alpha-HCH, beta-HCH, gamma-HCH, and delta-HCH) **
- Heptachlor epoxide (Heptachlor breakdown product) *
- Heptachlor+heptachlor epoxide (sum of heptachlor and heptachlor epoxide) **
- Hexachlorobenzene (HCB) **
- Isodrin (Isodrine, Compound 711) * **
- p,p'*-Methoxychlor (Marlate, methoxychlor) * **
- o,p'*-Methoxychlor * **
- Mirex (Dechlorane) **
- PCB, total
- Pentachloroanisole (PCA) * **
- cis*-Permethrin (Ambush, Astro, Pounce) * **
- trans*-Permethrin (Ambush, Astro, Pounce) * **

Semivolatile organic compounds (SVOCs) in bed sediment



Other SVOCs detected

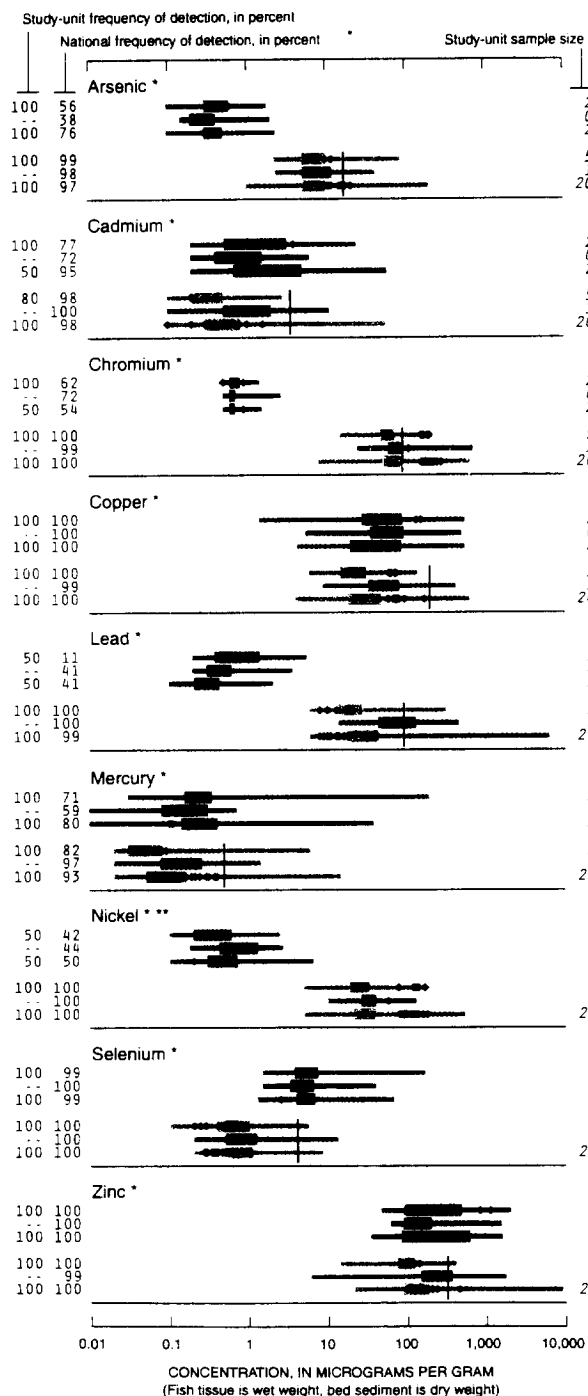
- Acenaphthene
- Acenaphthylene
- Anthracene
- Benzo[a]anthracene
- Benzo[a]pyrene
- Benzo[b]fluoranthene **
- Benzo[g]hperylene **
- Benzo[k]fluoranthene **
- Butylbenzylphthalate **
- Chrysene
- p*-Cresol **
- Di-n*-butylphthalate **
- Di-n*-octylphthalate **
- Diethylphthalate **
- 1,6-Dimethylnaphthalene **
- Dimethylphthalate **
- 9H-Fluorene (Fluorene)
- Indeno[1,2,3-*cd*]pyrene **
- 2-Methylantracene **
- 4,5-Methylenephenanthrene **
- 1-Methylphenanthrene **
- Naphthalene
- Phenanthrene
- Pyrene

SVOCs not detected

- Acridine **
- C8-Alkylphenol **
- Azobenzene **
- Benzo[c]cinnoline **
- 2,2-Biquinoline **
- 4-Bromophenyl-phenylether **
- 4-Chloro-3-methylphenol **
- bis(2-Chloroethoxy)methane **
- 2-Chloronaphthalene **
- 2-Chlorophenol **
- 4-Chlorophenyl-phenylether **
- Dibenz[*a,h*]anthracene
- Dibenzothiophene **
- 1,2-Dichlorobenzene (*o*-Dichlorobenzene) **
- 1,3-Dichlorobenzene (*m*-Dichlorobenzene) **
- 1,4-Dichlorobenzene (*p*-Dichlorobenzene) **
- 1,2-Dimethylnaphthalene **
- 3,5-Dimethylphenol **
- 2,4-Dinitrotoluene **
- 2-Ethyl-naphthalene **
- Isophorone **
- Isoquinoline **
- 1-Methyl-9H-fluorene **
- 1-Methylpyrene **
- Nitrobenzene **
- N*-Nitrosodi-*n*-propylamine **

N-Nitrosodiphenylamine **
 Pentachloronitrobenzene **
 Phenanthridine **
 Quinoline **
 1,2,4-Trichlorobenzene **
 2,3,6-Trimethylnaphthalene **

Trace elements in fish tissue (livers) and bed sediment



BIOLOGICAL INDICATORS

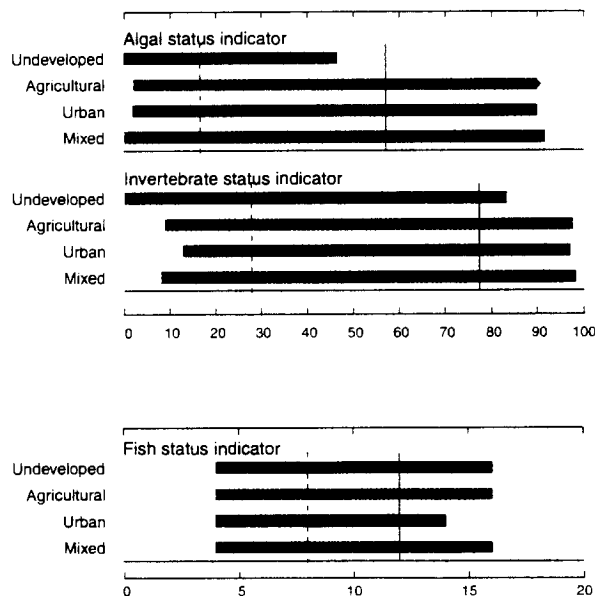
Higher national scores suggest habitat disturbance, water-quality degradation, or naturally harsh conditions. The status of algae, invertebrates (insects, worms, and clams), and fish provide a record of water-quality and stream conditions that water-chemistry indicators may not reveal. **Algal status** focuses on the changes in the percentage of certain algae in response to increasing siltation, and it often correlates with higher nutrient concentrations in some regions. **Invertebrate status** averages 11 metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. **Fish status** sums the scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that increase in association with water-quality degradation.

Biological indicator value, Sacramento River Basin, by land use, 1994-98

◆ Biological status assessed at a site

National ranges of biological indicators, in 16 NAWQA Study Units, 1994-98

- Streams in undeveloped areas
- Streams in agricultural areas
- Streams in urban areas
- Streams in mixed-land-use areas
- 75th percentile
- - - 25th percentile



A COORDINATED EFFORT

Coordination with agencies and organizations in the Sacramento River Basin was integral to the success of this water-quality assessment. We thank those who served as members of our liaison committee.

Federal Agencies

U.S. Environmental Protection Agency
U.S. Fish and Wildlife Service
Bureau of Reclamation
National Marine Fisheries Service

State Agencies

California Regional Water Quality Control Board
California Department of Water Resources
California Department of Fish and Game
California Division of Mines and Geology
California State Water Resources Control Board

Other

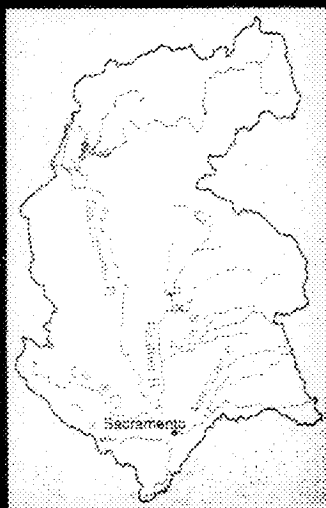
University of California, Davis

We also thank the following organizations for contributing to this effort.

We are grateful to the Sacramento River Watershed Program (SRWP) and the Sacramento Regional County Sanitation District (SRCSD). The SRWP was initiated in 1995, just as the NAWQA Program initiated most of the water-quality sample collection activities in the basin. The SRWP has the following mission statement: "To ensure that current and potential uses of the watershed's resources are sustained, restored, and where possible, enhanced, while promoting the long-term social and economic vitality of the region." The current SRWP membership includes a diverse group of stakeholders interested in water quality of the Sacramento River Basin. The infrastructure of the SRWP provides a unique means of achieving the NAWQA goals of coordination. The SRCSD anticipated the need for a more comprehensive approach to water-quality management of the Sacramento River Basin. As a result, the SRCSD was instrumental in developing the Sacramento River Toxic Pollutant Control Program and was a partner with the Sacramento River Basin NAWQA Program in a detailed study of trace metals in the Sacramento River system. Other agencies contributing to or helping to facilitate funding for that study include the California State Water Resources Control Board, the U.S. Environmental Protection Agency, and the National Marine Fisheries Service.


NAWQA

National Water-Quality Assessment (NAWQA) Program
Sacramento River Basin



Domagalski and others—Water Quality in the Sacramento River Basin
U.S. Geological Survey Circular 1215

ISBN 0-607-95419-1



9 780607 954197

R0024712

Water Quality in the Puget Sound Basin

Washington and British Columbia, 1996–98



POINTS OF CONTACT AND ADDITIONAL INFORMATION

The companion Web site for NAWQA summary reports:

<http://water.usgs.gov/nawqa/>

Puget Sound Basin contact and Web site:

USGS State Representative
U.S. Geological Survey
Water Resources Division
1201 Pacific Avenue
Suite 600
Tacoma, WA 98402
e-mail: gs-w_nawqa_pugt_ch@usgs.gov
<http://wa.water.usgs.gov/ps.nawqa.html>

National NAWQA Program:

Chief, NAWQA Program
U.S. Geological Survey
Water Resources Division
12201 Sunrise Valley Drive, M.S. 413
Reston, VA 20192
<http://water.usgs.gov/nawqa/>

Other NAWQA summary reports

River Basin Assessments

Albemarle-Pamlico Drainage Basin (Circular 1157)
Allegheny and Monongahela River Basins (Circular 1202)
Apalachicola-Chattahoochee-Flint River Basin (Circular 1164)
Central Arizona Basins (Circular 1213)
Central Columbia Plateau (Circular 1144)
Central Nebraska Basins (Circular 1163)
Connecticut, Housatonic and Thames River Basins (Circular 1155)
Eastern Iowa Basins (Circular 1210)
Georgia-Florida Coastal Plain (Circular 1151)
Hudson River Basin (Circular 1165)
Kanawha - New River Basins (Circular 1204)
Lake Erie - Lake Saint Clair Drainages (Circular 1203)
Las Vegas Valley Area and the Carson and Truckee River Basins (Circular 1170)
Lower Illinois River Basin (Circular 1209)
Long Island - New Jersey Coastal Drainages (Circular 1201)
Lower Susquehanna River Basin (Circular 1168)
Mississippi Embayment (Circular 1208)
Ozark Plateaus (Circular 1158)
Potomac River Basin (Circular 1166)

Red River of the North Basin (Circular 1169)
Rio Grande Valley (Circular 1162)
Sacramento River Basin (Circular 1215)
San Joaquin-Tulare Basins (Circular 1159)
Santee River Basin and Coastal Drainages (Circular 1206)
South-Central Texas (Circular 1212)
South Platte River Basin (Circular 1167)
Southern Florida (Circular 1207)
Trinity River Basin (Circular 1171)
Upper Colorado River Basin (Circular 1214)
Upper Mississippi River Basin (Circular 1211)
Upper Snake River Basin (Circular 1160)
Upper Tennessee River Basin (Circular 1205)
Western Lake Michigan Drainages (Circular 1156)
White River Basin (Circular 1150)
Willamette Basin (Circular 1161)

National Assessments

The Quality of Our Nation's Waters—Nutrients and Pesticides (Circular 1225)

Front cover: The Port of Tacoma in foreground looking up the Puyallup River Valley toward Mount Rainier in the background. (Photograph © by Kemer Nelson, Aequalis Aerial Photography, Tacoma, Washington. Used with permission.)

Back cover: The Port of Tacoma in foreground looking northwest toward the Olympic Mountains. (Photograph © by Kemer Nelson, Aequalis Aerial Photography, Tacoma, Washington. Used with permission.)

R0024714

Water Quality in the Puget Sound Basin, Washington and British Columbia, 1996–98

*By James C. Ebbert, Sandra S. Embrey, Robert W. Black, Anthony J. Tesoriero,
and Alan L. Haggland*

U.S. DEPARTMENT OF THE INTERIOR
BRUCE BABBITT, Secretary

U.S. GEOLOGICAL SURVEY
Charles G. Groat, Director

The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

2000

Free on application to the
U.S. Geological Survey
Information Services
Box 25286 Federal Center
Denver, CO 80225

Or call: 1-888-ASK-USGS

Library of Congress Cataloging-in-Publications Data

Water-quality assessment of the Puget Sound basin, Washington and British Columbia, 1996-98 / by James C. Ebbert...[et al.].

p. cm. -- (U.S. Geological Survey Circular ; 1216)

Includes bibliographical references.

ISBN 0-607-95420-5 (alk. paper)

1. Water quality--Washington (State)--Puget Sound Region. 2. Water quality--British Columbia. I. Ebbert, J. C. II. Series.

TD224.C3 W394 2000
363.739'42'097977--dc21

00-049456

R0024716

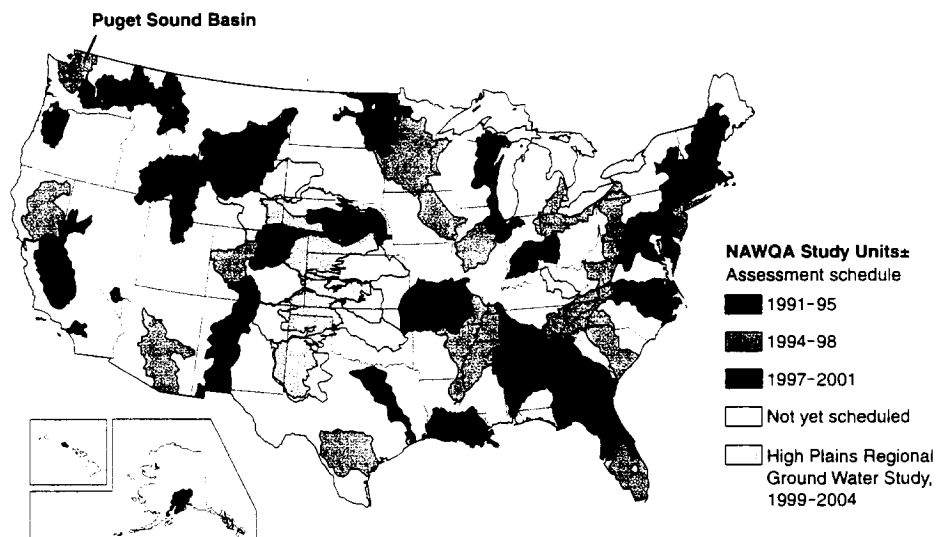
CONTENTS

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM.....	IV
SUMMARY OF MAJOR FINDINGS	1
Stream and river highlights	1
Ground-water highlights	2
INTRODUCTION TO THE PUGET SOUND BASIN	3
Land use affects water quality and stream habitat	3
Water availability is a major issue.....	4
Hydrologic conditions probably affected study results	5
Some ground water is susceptible to contamination	5
Study design focuses on land use	5
MAJOR FINDINGS	6
Pesticides in surface water were indicative of upstream land uses, and more pesticides were detected in streams than in rivers	6
NATIONAL PERSPECTIVE—Insecticides in Thornton Creek were also detected in urban streams throughout the Nation.....	8
Historically used pesticides and PCBs are still detected in streambed sediments and fish tissue.....	8
Streams and rivers in developed areas were enriched with nutrients relative to those in undeveloped areas	11
Runoff during rainstorms contributes contaminants to streams, indicating nonpoint sources.....	12
Bacteria indicate the presence of fecal contamination in many Puget Lowland streams	13
Stream habitat and health are degraded in agricultural and urban streams.....	13
NATIONAL PERSPECTIVE—Puget Sound stream invertebrate status ranged from high to low when compared with national levels.....	15
Ground-water quality is generally good	16
Nitrate concentrations in shallow ground water beneath residential areas were elevated, but concentrations of other compounds were low	17
High nitrate concentrations and fumigant-related compounds were detected in shallow ground water beneath an agricultural area.....	18
NATIONAL PERSPECTIVE—MTBE was rarely detected in ground water in the Puget Sound Basin	19
STUDY UNIT DESIGN	20
GLOSSARY	22
REFERENCES.....	24
APPENDIX—WATER-QUALITY DATA FROM THE PUGET SOUND BASIN IN A NATIONAL CONTEXT	26

NATIONAL WATER-QUALITY ASSESSMENT PROGRAM

THIS REPORT summarizes major findings about water quality in the Puget Sound Basin that emerged from an assessment conducted between 1996 and 1998 by the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program. Water quality is discussed in terms of local and regional issues and compared to conditions found in all 36 NAWQA study areas, called Study Units, assessed to date. Findings are also explained in the context of selected national benchmarks, such as those for drinking-water quality and the protection of aquatic organisms. The NAWQA Program was not intended to assess the quality of the Nation's drinking water, such as by monitoring water from household taps. Rather, the assessments focus on the quality of the resource itself, thereby complementing many ongoing Federal, State, and local drinking-water monitoring programs. The comparisons made in this report to drinking-water standards and guidelines are only in the context of the available untreated resource. Finally, this report includes information about the status of aquatic communities and the condition of in-stream habitats as elements of a complete water-quality assessment.

Many topics covered in this report reflect the concerns of officials in State and Federal agencies, water-resource managers, and members of stakeholder groups who provided advice and input during the Puget Sound River Basin assessment. Basin residents who wish to know more about water quality in the areas where they live will find this report informative as well.



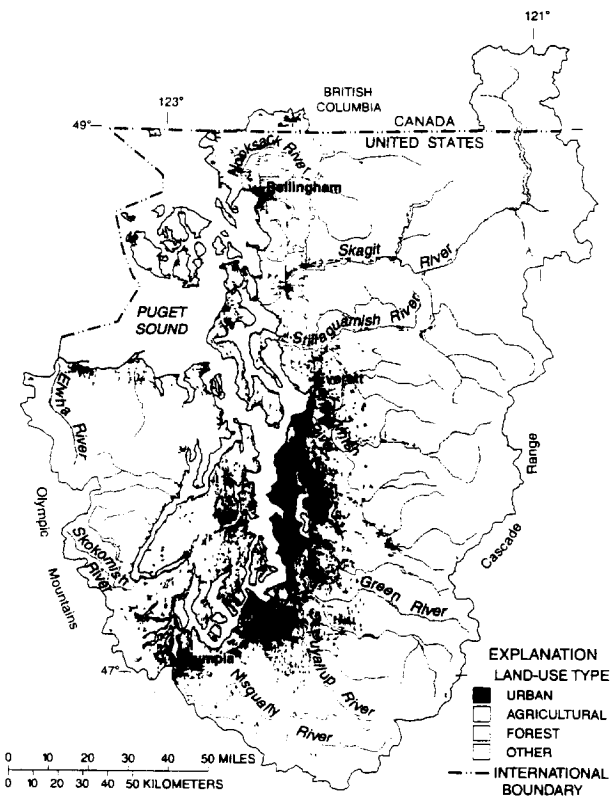
THE NAWQA PROGRAM seeks to improve scientific and public understanding of water quality in the Nation's major river basins and ground-water systems. Better understanding facilitates effective resource management, accurate identification of water-quality priorities, and successful development of strategies that protect and restore water quality. Guided by a nationally consistent study design and shaped by ongoing communication with local, State, and Federal agencies, NAWQA assessments support the investigation of local issues and trends while providing a firm foundation for understanding water quality at regional and national scales. The ability to integrate local and national scales of data collection and analysis is a unique feature of the USGS NAWQA Program.

The Puget Sound Basin is one of 51 water-quality assessments initiated since 1991, when the U.S. Congress appropriated funds for the USGS to begin the NAWQA Program. As indicated on the map, 36 assessments have been completed, and 15 more assessments will conclude in 2001. Collectively, these assessments cover about one-half of the land area of the United States and include water resources that are available to more than 60 percent of the U.S. population.

SUMMARY OF MAJOR FINDINGS

Stream and River Highlights

Streams and rivers in the Puget Sound Basin met most Federal and State water-quality guidelines. In general, large rivers were more likely to meet guidelines than were small streams. Concentrations of fecal bacteria frequently exceeded U.S. Environmental Protection Agency (USEPA) recreational criteria and State standards protecting beneficial uses of surface water, and insecticide concentrations were occasionally higher than guidelines recommended to protect aquatic life. A total of 74 man-made organic chemicals were detected in streams and rivers, with different mixtures of chemicals linked to agricultural and urban settings. Though most chemical concentrations appeared to be low, guidelines for drinking water and aquatic life that



The Puget Sound Basin is a 13,700-square-mile area of mountains and coastal lowlands in western Washington State and portions of British Columbia. About 4 million people live in the basin, mainly in metropolitan areas of Seattle, Tacoma, Everett, Bellingham, and Olympia. Headwaters of major rivers provide much of the drinking water for these metropolitan areas. Ground water is the primary source of drinking water in rural areas and, increasingly, for new suburbs.

	Selected Indicators of Stream-Water Quality			
	Small Streams			Major Rivers
	Urban	Agricultural	Undeveloped and Forest	Mixed Land Uses
Pesticides ¹	●	●	—	◐
Nutrients ²	●	●	◐	●
Organo-chlorines ³	◐	◐	—	◐
Semivolatile organics ⁴	●	●	◐	◐

Percentage of samples with concentrations equal to or greater than a health-related national guideline for drinking water or aquatic life; or above a national goal for preventing excess algal growth
 Percentage of samples with concentrations less than a health-related national guideline for drinking water or aquatic life; or below a national goal for preventing excess algal growth
 — Percentage of samples with no detection
 — Not assessed

are needed to make a full assessment do not exist for more than half the compounds detected.

- The insecticide diazinon, commonly used by homeowners on lawns and gardens, was frequently detected in urban streams at concentrations that exceeded guidelines for protecting aquatic life (p. 6).
- The average concentration of total nitrogen in small streams draining agricultural lands was twice the concentration in streams draining urban areas and over 40 times the concentration in streams draining undeveloped areas. Concentrations of total phosphorus were less dependent on land use, and concentrations above the USEPA desired goal of 0.1 mg/L to prevent excessive plant growth were detected in rivers and streams in all but undeveloped areas (p. 11).
- Concentrations of *E. coli* bacteria exceeded USEPA criteria for moderate water-contact recreation, including swimming, in 15 of 31 small streams. Livestock, pet, and wildlife wastes, and to some extent human sewage, are likely sources of these bacteria (p. 13).

¹ Insecticides, herbicides, and pesticide metabolites, sampled in water.
² Total phosphorus and nitrate (as nitrogen), sampled in water.
³ Organochlorine compounds including DDT and PCBs, sampled in fish tissue.
⁴ Miscellaneous industrial chemicals and combustion by-products, sampled in sediment.

- Urban expansion into forested areas is changing stream habitats, in part because of changes in water quality. Streams in urban and agricultural areas are warmer and support less diverse populations of insects than those in forested areas (p. 14).

Trends in Surface-Water Quality

Concentrations of nitrate and phosphorus in the Nooksack and Green Rivers did not change between 1980 and 1997. During the same period, concentrations of nitrate in the Skokomish River and in Big Soos and Newaukum Creeks increased slightly (p. 12).

Major Influences on Streams and Rivers

- Contaminants in runoff from urban and agricultural land surfaces
- Degraded stream habitat in urban and agricultural areas

Ground-Water Highlights

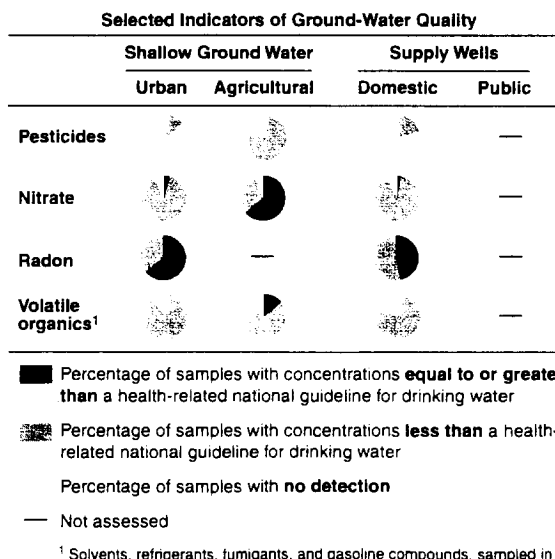
Reliance on ground water as a source of drinking water is increasing with urban and suburban development. With some exceptions, ground water is of high quality. However, as indicated by elevated concentrations of nitrate and the presence of pesticides and other organic compounds, shallow ground water in both urban and agricultural settings is vulnerable to contamination. Monitoring wells in urban residential areas generally contained low-concentration mixtures of chemicals associated with transportation and household activities. Shallow ground water, at depths tapped for domestic supply in agricultural areas, contained fertilizer residues (nitrate) at concentrations that commonly exceeded the drinking-water standard. Other agricultural chemicals were also frequently detected, though mostly at concentrations below current Federal and State drinking-water guidelines.

- Use of fertilizers on urban lawns and gardens and drainage from septic systems have elevated nitrate concentrations in shallow ground water beneath urban residential areas. In most samples, these concentrations were substantially less than the drinking-water standard (p. 17).

- Pesticides were not detected in wells that are more than 120 feet deep, the depth below which most large public-supply wells withdraw water. Pesticides were detected in wells that are less than 100 feet deep, the range of many rural and suburban domestic wells, but concentrations met drinking-water guidelines. Only about half of the detected pesticides have guidelines, and no benchmarks are available to assess the significance of low-concentration mixtures of pesticides (p. 16).
- Applications of fertilizers and dairy and poultry manure to cropland in agricultural areas of the Nooksack River Basin have increased nitrate concentrations above the USEPA drinking-water standard in about 60 percent of the shallow ground water sampled (p. 18).
- Prior to 1977, 1,2-dichloropropane was one of the ingredients in fumigants used on potatoes and berries in the Nooksack River Basin. Currently used fumigants contain only trace amounts of this compound, and their application is not likely contaminating ground water to concentrations exceeding drinking-water standards (p. 19).

Major Influences on Ground Water

- Poultry and dairy waste
- Lawn and garden fertilizers
- Septic systems
- Fumigants no longer in use



INTRODUCTION TO THE PUGET SOUND BASIN

The Puget Sound Basin encompasses the 13,700-square-mile area that drains to Puget Sound and adjacent marine waters. Included are all or part of 13 counties in western Washington, as well as the headwaters of the Skagit River and part of the Nooksack River in British Columbia, Canada. Streams and rivers drain three physiographic provinces—the Olympic Mountains in the west, the Cascade Range in the east, and the Puget Lowlands in the center of the basin (fig. 1).

Land Use Affects Water Quality and Stream Habitat

Nearly 4 million people, or about 70 percent of Washington State's population, live in the Puget Sound Basin. Urban growth is rapid; by 2020, the population is

expected to increase by 1.1 million people, with most growth in urban and suburban areas. Urban and agricultural land uses, which cover about 9 and 6 percent of the basin, respectively (fig. 2), are concentrated in the lowlands. Forest dominates land use and cover in the basin and is concentrated in the foothills and mountains (see map, p. 1).

The quality of water and aquatic biota has been affected by a range of forestry, agricultural, and urban development practices. The chemical quality of surface water in the foothills and mountains is

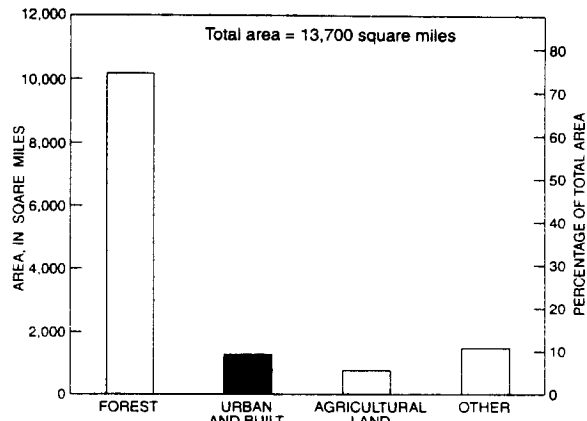
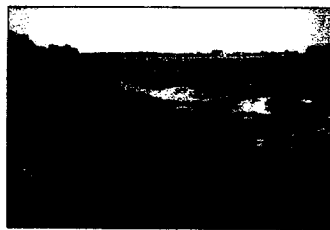


Figure 2. Land use and cover in the basin is predominantly forest.

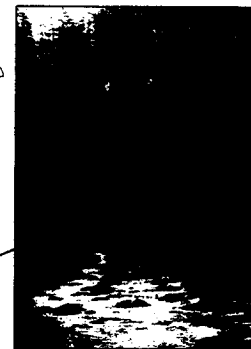
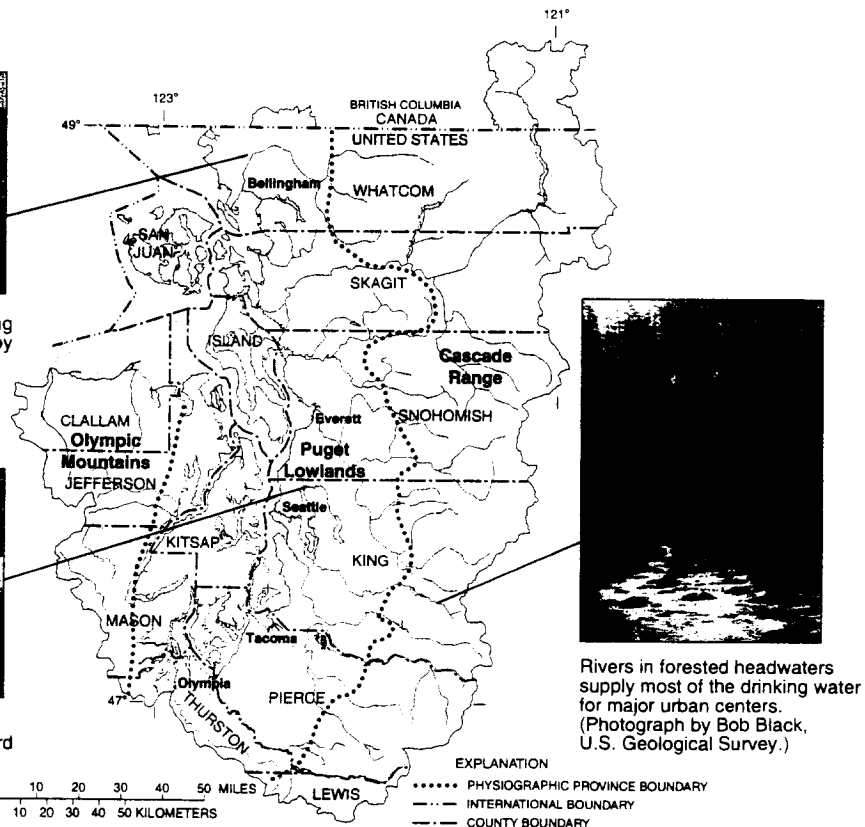
generally suitable for most uses. However, the physical hydrology, water temperature, and biologic integrity of streams have been influenced to varying degrees by logging (Black and Silkey, 1998).



Streambank vegetation is sparse along an agricultural stream. (Photograph by Bob Black, U.S. Geological Survey.)



Urban land use is concentrated in the Puget Lowlands. (Photograph by Ward Staubitz, U.S. Geological Survey.)



Rivers in forested headwaters supply most of the drinking water for major urban centers. (Photograph by Bob Black, U.S. Geological Survey.)

Figure 1. Streams and rivers drain three physiographic provinces in the Puget Sound Basin (Black and Silkey, 1998).

The quality of ground water in the upper watersheds probably differs little from natural conditions.

Because of development, many streams in the Puget Lowlands have undergone changes in structure and function with a trend toward simplification of stream channels and loss of habitat (Black and Silkey, 1998). Sources of contaminants to lowland streams and lower reaches of large rivers are largely nonpoint because most major point sources discharge directly to Puget Sound. Compared with that in small streams in the Puget Lowlands, the quality of water in the lower reaches of large rivers is better because much of the flow is derived from the forested headwaters.

More than half of the agricultural acreage in the basin is located in Whatcom, Skagit, and Snohomish Counties. Agricultural land use consists of about 60 percent cropland and 40 percent pasture. Livestock produce a large amount of manure that is applied as fertilizer to cropland, sometimes in excess amounts, resulting in runoff of nitrogen and phosphorus to surface water and leaching of nitrate to ground water. Runoff from agricultural areas also carries sediment, pesticides, and bacteria to streams (Staubitz and others, 1997). Pesticides and fumigant-related compounds are present, usually at low concentrations, in shallow ground water in agricultural areas.

Heavy industry is generally located on the shores of the urban bays and along the lower reaches of their influent tributaries, such as Commencement Bay and the Puyallup River in Tacoma and

Elliott Bay and the Duwamish Waterway in Seattle. High-density commercial and residential development occurs primarily within and adjacent to the major cities. Development in recent years has continued around the periphery of these urban areas but has trended toward lower density. This trend has resulted in increasing urban sprawl in the central Puget Sound Basin (fig. 3).

Urban land-use activities have had a significant impact on the quality of streams in the Puget Sound Basin (Staubitz and others, 1997). Water-quality concerns related to urban development include providing

adequate sewage treatment and disposal; transport of contaminants to streams by storm runoff, and preservation of stream corridors.

Water Availability Is a Major Issue

Although surface water provides most of the drinking water for the major urban centers (fig. 4), ground water is used in rural areas, and reliance on ground water as a source of drinking water is increasing with urban and suburban development (Staubitz and others, 1997).

Water availability has been and will continue to be a major, long-term issue in the Puget Sound Basin. It is now widely recognized that ground-water withdrawals can

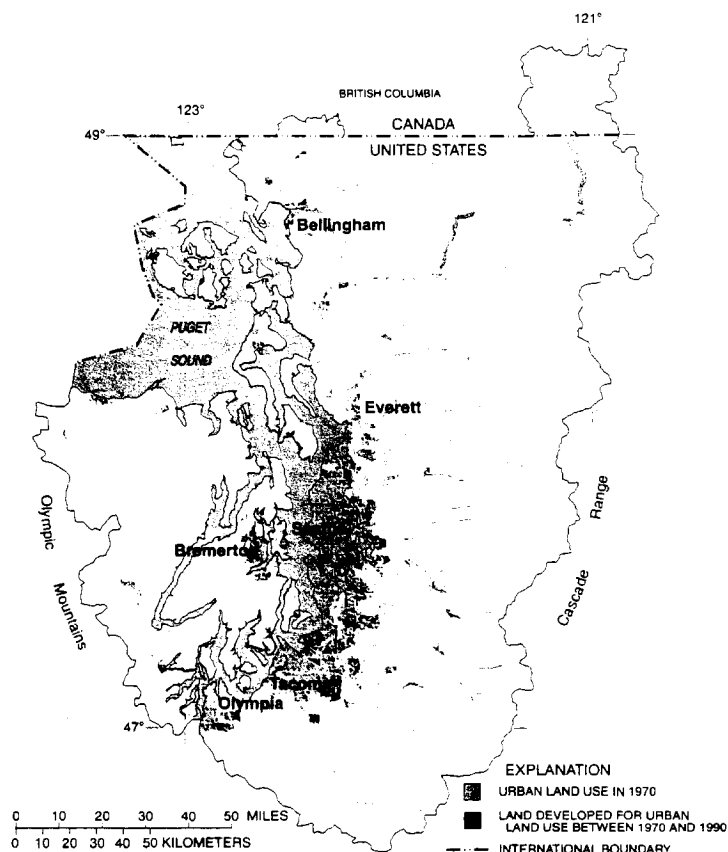


Figure 3. Recent urban development in the Puget Sound Basin has been around the periphery of established urban areas.

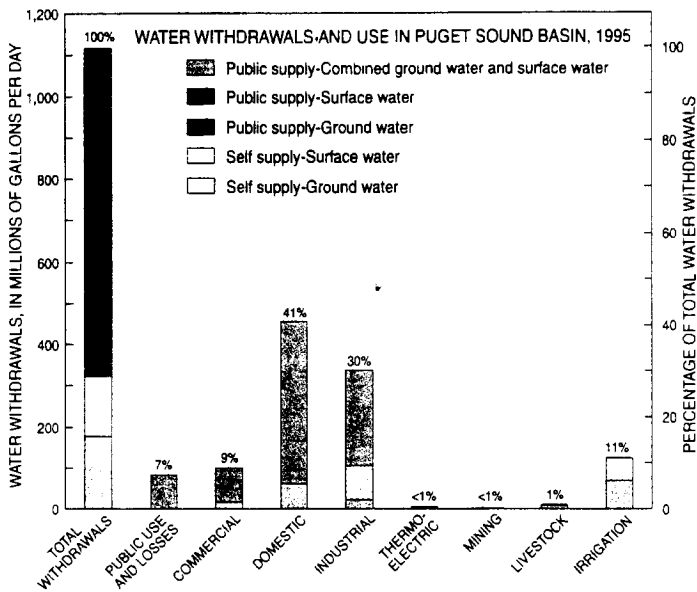


Figure 4. Excluding water used for hydroelectric power, 41 percent of water was used for domestic supply in 1995. Ground water accounted for 47 percent of all withdrawals.

deplete streamflows (Morgan and Jones, 1999), and one of the increasing demands for surface water is the need to maintain instream flows for fish and other aquatic biota.

Hydrologic Conditions Probably Affected Study Results

Surface water was sampled during 1996 and 1997 when rainfall and streamflows generally

were above the 30-year average (fig. 5). Because of increased runoff during periods of rainfall, larger amounts of sediment, nutrients, pesticides, bacteria, and other contaminants may have been transported from the land surface into streams and rivers than during drier years. High flows due to runoff from paved surfaces can also alter stream habitat.

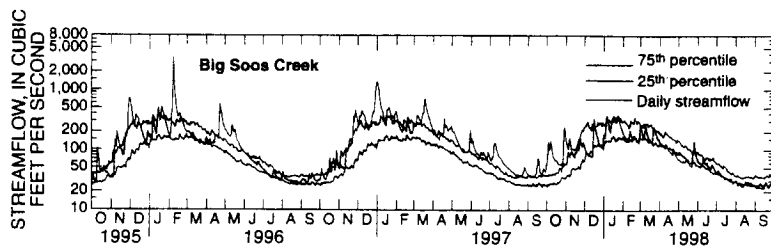


Figure 5. Streams and rivers were sampled during 1996 and 1997 when streamflows generally were above the 30-year average.

Some Ground Water Is Susceptible to Contamination

Shallow ground water (less than 100 feet deep) in the Puget Lowlands is more susceptible to contamination in areas where the overlying sediments are coarse grained than where they are fine grained (see p. 16 and map, p. 20). This is because rainfall, or applied irrigation water, seep relatively easily through coarse-grained sediments and can transport contaminants to ground water.

Study Design Focuses on Land Use

Chemical and biological samples were collected from a mix of rivers and streams in forested, urban, and agricultural areas to assess overall quality as well as the effects of specific land-use practices. At some sites, water samples were collected monthly and during storms to assess the effects of storm runoff on contaminant washoff. Other sites were sampled only once, usually during normal flows.

Shallow ground water was sampled from aquifers in coarse-grained glacial deposits considered to be the most at risk to contamination. Water from these aquifers was sampled to assess general water quality and specifically to assess the quality of water in agricultural and residential areas. Monitoring wells and domestic wells were sampled. Data from a previous sampling of public-supply wells (Ryker and Williamson, 1996) are used to help evaluate the quality of drinking water and the quality of deep ground water. (See table 3, page 21 for details on study design.)

MAJOR FINDINGS

Pesticides in Surface Water Were Indicative of Upstream Land Uses, and More Pesticides Were Detected in Streams Than in Rivers

Twenty-nine pesticides were detected in Fishtrap Creek, which drains an agricultural area in the Nooksack River Basin (see map; p. 20), compared with 21 in Thornton Creek, an urban stream. In Fishtrap Creek, the more frequent detections of the herbicides atrazine and metolachlor, used more in agricultural areas, and less frequent detections of the herbicides dichlobenil and prometon (fig. 6), used more in urban areas, are consistent with the predominant land use in the

drainage basin. Urban-use pesticides in Fishtrap Creek are probably transported from urban areas in the upper and lower parts of the basin.

The two large rivers sampled for pesticides, the Nooksack and Duwamish (see map; p. 20), integrate land-use effects. The herbicides prometon, simazine, and tebutiuron, which are used in urban settings, were detected frequently in the Duwamish River (fig. 6). This is indicative of the urban land use surrounding and

Table 1. Concentrations of three insecticides were sometimes above chronic guidelines for the protection of aquatic life

Insecticide	Aquatic life guideline (micrograms per liter) ¹	Percentage of all samples exceeding guideline	Site in exceedance and (percentage of samples)	Range of concentrations detected at the site (micrograms per liter)
Chlorpyrifos	0.041	1	Thornton (2)	0.006–0.074
Diazinon	0.08	9	Thornton (20) Fishtrap (3) Duwamish (4)	0.003–0.501 0.004–0.113 0.004–0.083
Lindane	0.01	1	Thornton (2)	0.02

¹ U.S. Geological Survey, 1999a.

immediately upstream from the sampling site. Metolachlor, an agricultural herbicide, was detected more often in the Duwamish River than in the Nooksack River, which was sampled in an agricultural setting. Metolachlor in the Duwamish River was likely transported from agricultural areas upstream from the mostly urbanized lower part of the basin, where samples were collected.

Pesticides transported to the Nooksack and Duwamish Rivers are diluted by the volume of high-quality water from forested headwaters, resulting in lower detection frequencies compared with Fishtrap and Thornton Creeks (fig. 6). In addition to lower detection frequencies, fewer pesticides were detected in the large rivers.

Diazinon was the most frequently detected insecticide in both small streams and large rivers (fig. 6). It is used heavily in urban areas of the Puget Sound Basin, and its frequent detection, sometimes at concentrations exceeding the chronic guideline for the protection of aquatic life (table 1), prompted a more focused study of pesticides in urban streams in Seattle and surrounding King County (see page 7).

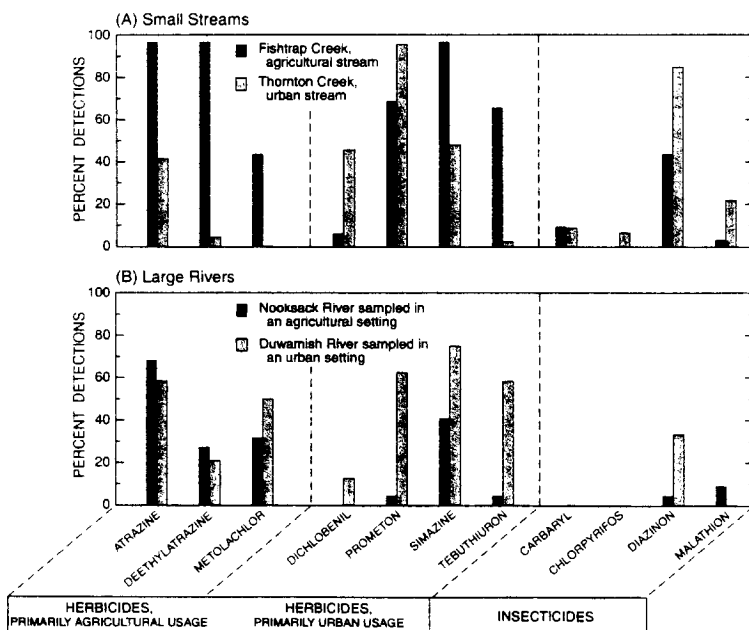


Figure 6. Pesticides detected in small streams (A) and large rivers (B) were indicative of land use, but detection frequencies in large rivers were lower because of dilution by flows from forested headwaters.

Concentrations of Pesticides in Surface Water Draining Urban and Agricultural Areas Sometimes Exceeded Guidelines for the Protection of Aquatic Life

The insecticides chlorpyrifos, diazinon, and lindane were sometimes detected at concentrations above chronic guidelines for the protection of aquatic life (table 1). Concentrations of diazinon exceeded its guide-

line in 9 percent of all samples, and in Thornton Creek, concentrations exceeded the guideline in 20 percent of samples. It is estimated that about half the 83,000 pounds of diazinon applied annually in the Puget Sound Basin are applied in King County (Tetra Tech Incorporated, 1988). Diazinon is the insecticide purchased most frequently by King County residents (fig. 7).

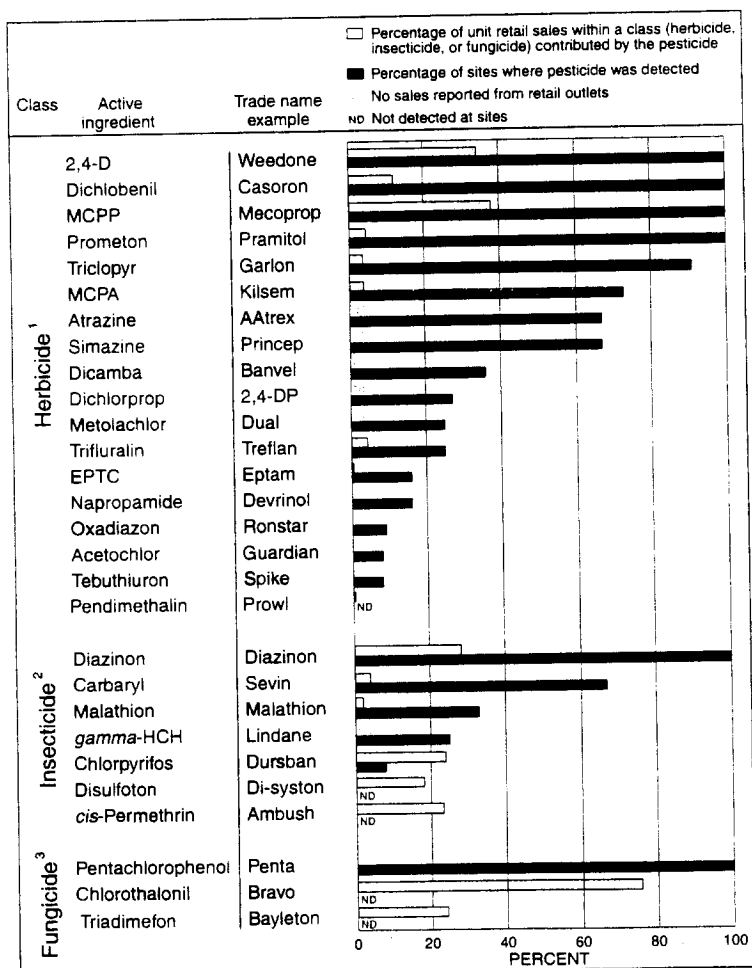
Pesticides Applied to Lawns and Gardens, As Well As Those Applied in Nonresidential Areas, Are Sources of Pesticides in Urban Streams

Because of the prevalence of diazinon and other pesticides in Thornton Creek, the USGS collaborated with the Washington State Department of Ecology and the King County Hazardous Waste Management Program to study pesticides in streams in 10 urban and suburban watersheds in King County (see map, p. 20). Samples were collected during April and May 1998 when retail sales of pesticides are highest (Voss and others, 1999), and they were collected during storms when runoff can transport pesticides to streams.

Because USGS and Washington State Department of Ecology laboratories both participated in the study, additional pesticides not included in the NAWQA study were analyzed. To help determine sources of pesticides detected, King County provided sales data for pesticides sold in 10 large home and garden stores during 1997.

Twenty-three of 98 pesticides analyzed for were detected in the urban streams. Homeowner use as a source of pesticides in streams is indicated for compounds like the herbicide 2,4-D and the insecticide diazinon, which were detected in all streams and were sold frequently in the home and garden stores (fig. 7). Some pesticides sold in stores were not detected in streams, indicating that other factors, like the rate at which a compound breaks down, affect the relation between usage and detection.

Almost half of the pesticides detected in the streams had no retail sales, indicating that they are usually not applied by homeowners. In urban areas, pesticides are also applied in commercial areas, along road rights-of-way, and in parks and recreational areas.



¹ Unit retail sales for these herbicides total to 140,000 units.

² Unit retail sales for these insecticides total to 66,000 units.

³ Unit retail sales for these fungicides total to 1,700 units.

Figure 7. A comparison of pesticides detected in urban streams with retail sales data suggests that homeowner and nonresidential applications both are sources of pesticides in streams. Sales were reported in units, which represent a bag, bottle, or other package containing the pesticide.



Insecticides in Thornton Creek Were Also Detected in Urban Streams Throughout the Nation

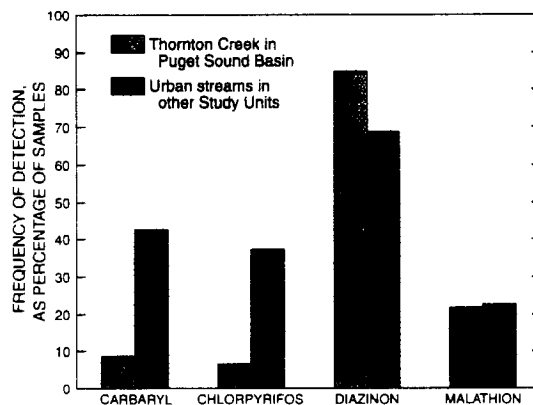
Throughout the Nation, the insecticides diazinon, carbaryl, chlorpyrifos, and malathion were detected much more frequently in streams draining urban basins than in streams draining agricultural basins (U.S. Geological Survey, 1999b). Except for carbaryl, these pesticides also were detected more frequently in Thornton Creek, an urban stream, than in Fishtrap Creek, an agricultural stream (fig. 6).

Detection of these pesticides is related to usage. For example, diazinon and chlorpyrifos, which nationally rank 1 and 4 among insecticides used for homes and gardens, rank 1 and 2 in unit sales of insecticides sold by home improvement stores in urban and suburban areas of King County (fig. 7).

Insecticides in streams are a concern because even at relatively low concentrations they can exceed guidelines for the protection of aquatic life (table 1).



Areas where pesticides are applied in urban settings are often in close proximity to streams. (Photograph by Sandra Embrey, U.S. Geological Survey.)



Although detection frequencies vary, insecticides in Thornton Creek were typical of those detected in urban streams in other Study Units throughout the Nation.

Historically Used Pesticides and PCBs Are Still Detected in Streambed Sediments and Fish Tissue

Streambed sediment and whole fish (sculpin, a bottom fish) tissue were analyzed at sites throughout the basin (see map, p. 20) in 1995 for organochlorine pesticides and PCBs. These compounds have been shown to have negative impacts on the health of aquatic organisms as well as the organisms that consume them. More organochlorine compounds were detected in both streambed sediments and fish tissue at sites surrounded by agricultural and urban land uses than at sites in undeveloped, forest, or mixed-land-use areas (fig. 8).

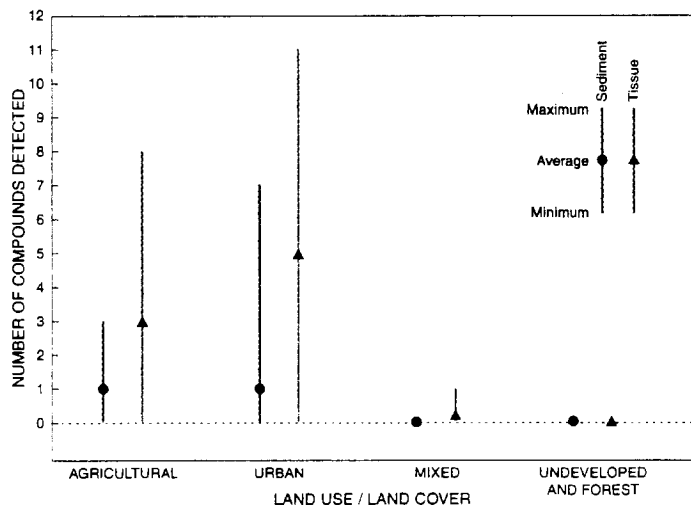


Figure 8. Organochlorine compounds, including the insecticide DDT and PCBs, in streambed sediments and fish tissue were primarily detected in urban and agricultural streams.

Fish Tissue Contamination is Related to Urban and Agricultural Land Uses

The relation between land use and the probability of detecting organochlorine compounds in fish (sculpin) tissue were statistically significant. This relation suggests that urban and agricultural land uses both contribute to the probability that a fish is contaminated with organochlorine pesticides. The probability of detecting total PCBs in fish was significantly related to urban land use only (Black and others, 2000). These relations also suggest that there is a land-use threshold below which the probability of finding organochlorine compounds in fish tissue is unlikely (fig. 9).

Organochlorine Compounds in Streambed Sediment and Fish May Be a Concern for Aquatic Ecosystems

Organochlorine pesticides were detected in streambed sediment at 3 of 19 sites (fig. 10). At Thornton Creek, an urban stream, levels of DDE (a breakdown product of DDT) and DDT exceeded the Canadian Council of Ministers of the Environment's (Canadian Council of Ministers of the Environment, 1995) probable effects level (PEL). Compounds exceeding the PEL are likely to result in adverse effects on aquatic organisms. It is important to note that levels of DDT in Thornton Creek sediment were higher than either one of its breakdown products, DDD and DDE. This may indicate that land disturbances in the basin have reintroduced buried soils contaminated with DDT.

Organochlorine compounds were also detected in the tissue of whole bottom fish (sculpin) collected at 8 of 18 study sites (fig. 11).

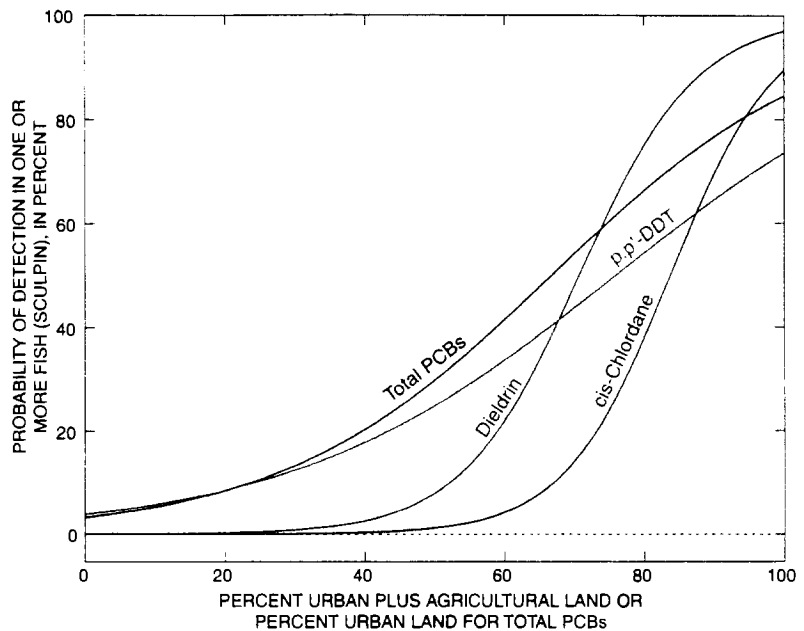


Figure 9. The probability of detecting specific organochlorine compounds in one or more fish (sculpin) at a site increased with the percentage of urban and agricultural land in the drainage basin. Dieldrin, *cis*-Chlordane, and *p,p'*-DDT relations are based on the percentage of agriculture plus urban land upstream from the sampling site. The total PCBs relation is based on the percent of urban land upstream from the sampling site.

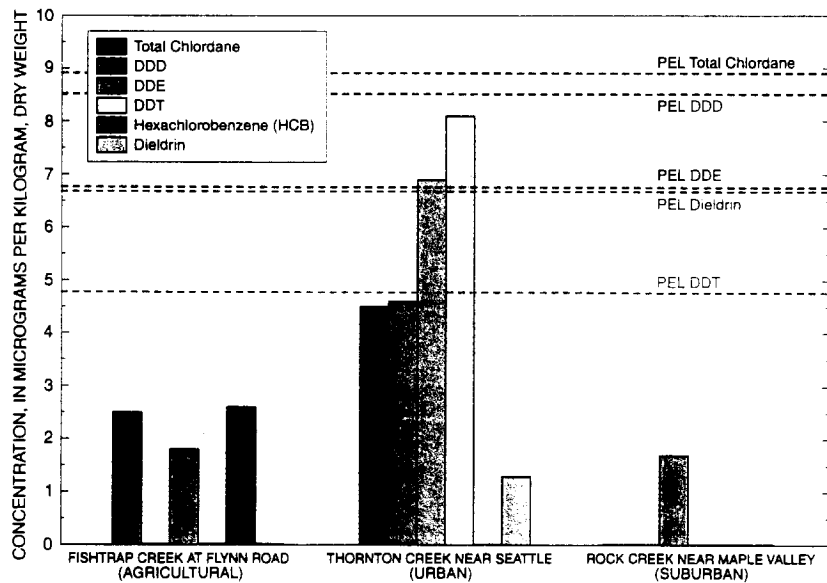


Figure 10. Concentrations of organochlorine compounds in streambed sediment in the Puget Sound Basin sometimes exceeded Canadian probable effects levels (PELs). Compounds above the PELs may have negative impacts on aquatic organisms.

The largest number and highest concentrations of these compounds were in the fish collected from urban sites. At some sites, total PCBs and DDT concentrations were equal to or above the New York State Department of Environmental Conservation (NYSDEC) criteria for the protection of fish-eating wildlife (Newell and others, 1987).

Generally, the concentrations of organochlorine compounds in sediment and fish tissue were in the middle 50 percent nationally (see Appendix). Occasionally, concentrations of DDE, total chlordane, and total DDT were in the upper 25 percent of those reported nationally.

Whole sculpin (*Cottus*) were analyzed for



organochlorine compounds. (Photograph by Ward W. Staubitz, U.S. Geological Survey.)

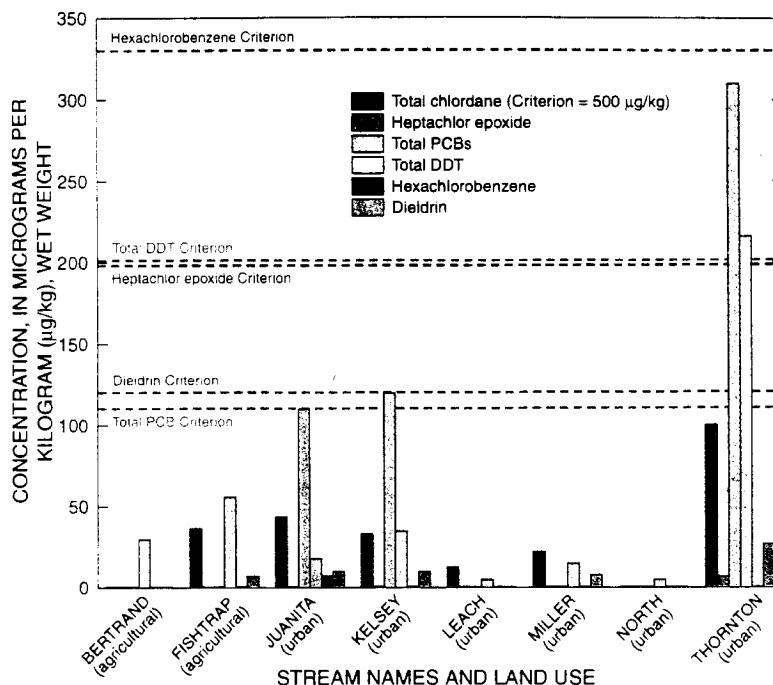


Figure 11. Concentrations of organochlorine compounds in fish (sculpin) tissue from streams in the Puget Sound Basin sometimes exceeded New York State Department of Environmental Conservation (NYSDEC) criteria. Concentrations above criteria may have a detrimental effect on fish-eating organisms.

Characteristics and historical uses of organochlorine compounds detected in Puget Sound Basin fish tissue and streambed sediment

Organochlorine compounds are synthetic organic compounds containing chlorine. As generally used, the term refers to compounds containing mostly or exclusively carbon, hydrogen, and chlorine. Examples include organochlorine insecticides, polychlorinated biphenyls, and some solvents containing chlorine.

Chlordane (Octachloro-4,7-methanotetrahydroindane) is an organochlorine insecticide no longer registered for use in the United States. Technical chlordane is a mixture in which the primary components are

cis- and *trans*-chlordane, *cis*- and *trans*-nonachlor, and heptachlor.

DDT (Dichloro-diphenyl-trichloroethane) is an organochlorine insecticide no longer registered for use in the United States.

Dieldrin is an organochlorine insecticide no longer registered for use in the United States. It is also a breakdown product of the insecticide aldrin.

Heptachlor epoxide is a breakdown product of the organochlorine insecticide heptachlor. It was used in the United States until the 1970s.

Hexachlorobenzene (HCB) is a fungicide used as a seed and soil treatment. It was discontinued from use in the United States in the 1980s.

Polychlorinated biphenyls (PCBs) are a mixture of chlorinated derivatives of biphenyl, marketed under the trade name Aroclor with a number designating the chlorine content (such as Aroclor 1260). PCBs were used in transformers and capacitors for insulating purposes and in gas pipeline systems as a lubricant. Further sale for new use was banned by law in 1979.

Streams and Rivers in Developed Areas Were Enriched with Nutrients Relative to Those in Undeveloped Areas

The highest average concentrations of total nitrogen were in small streams draining agricultural areas. The concentrations were nearly twice those in streams draining urban areas and over 40 times the average concentrations in streams draining undeveloped land (table 2). Drainage basins in agricultural areas also yielded the most nitrogen per square mile (fig. 12). Concentrations and yields of total nitrogen in Puget Sound Basin streams and rivers correlate with usage and atmospheric deposition of nitrogen in drainage basins (Inkpen and Embrey, 1998). Fertilizers used in both agricultural and urban areas, manure associated with dairy farms, and atmospheric deposition are sources of nitrogen in Puget Sound Basin rivers (Embrey and Inkpen, 1998).

Concentrations of Phosphorus in Some Streams and Rivers May Promote Excessive Plant Growth

Average concentrations of total phosphorus exceeding the USEPA desired goal of 0.1 mg/L to prevent excessive plant growth were detected in streams and rivers in all land-use areas except undeveloped land

Table 2. Highest average concentrations of total nitrogen were in streams draining agricultural areas. Average concentrations of total phosphorus exceeding the USEPA desired goal of 0.1 mg/L to prevent excessive plant growth (shown in bold) were present in streams in urban, agricultural, and forested basins

River or stream	Average total nitrogen concentration (milligrams per liter)	Average total phosphorus concentration (milligrams per liter)	Basin area size	Land use/land cover (dominant land use in mixed) ¹
Fishtrap	3.54	0.086	small	agricultural
Newaukum	2.82	0.210	small	agricultural
Thornton	1.73	0.131	small	urban
Big Soos	1.20	0.054	small	urban
Springbrook	1.10	0.165	small	urban
Nooksack, Brennan	0.66	0.301	large	mixed (agricultural)
Duwamish	0.65	0.089	large	mixed (urban)
Nooksack, North Cedarville	0.39	0.152	large	mixed (forest)
Skokomish	0.13	0.078	large	mixed (forest)
North Fork Skokomish	0.07	0.012	small	undeveloped
Green	0.07	0.012	small	forest

¹Forest land that is not logged is designated undeveloped. In basins with mixed land use, the land use in the part of the basin where samples were collected is usually considered the dominant land use.

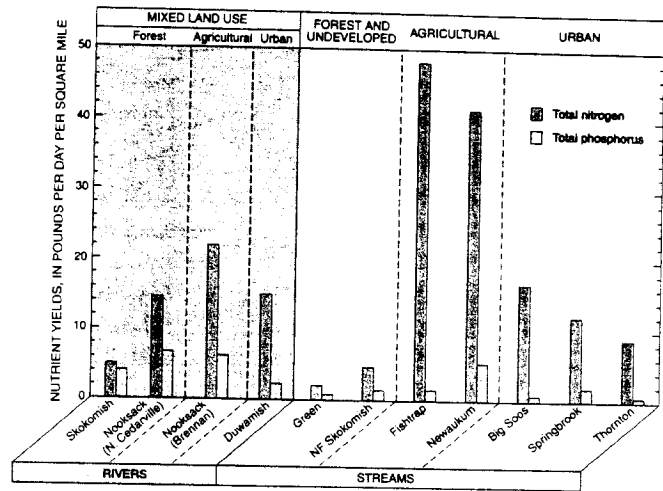


Figure 12. Basins in agricultural areas yielded the most nitrogen per square mile. The forested area upstream from the Nooksack River at North Cedarville yielded the most phosphorus. In basins with mixed land use, the dominant land use is indicated. The Green River above Twin Camp Creek and North Fork Skokomish River fall into the stream category because their drainage basins are relatively small.

(table 2). Unlike total nitrogen, concentrations of total phosphorus in streams and rivers in the Puget Sound Basin do not correlate with usage and atmospheric deposition of phosphorus in drainage basins (Inkpen and Embrey, 1998). This is because phosphorus attaches to soil particles and usually remains close to application areas unless it is transported to rivers by soil erosion.

Because of the importance of erosion in transporting phosphorus to streams, yields of total phosphorus correlate with yields of suspended sediment. The highest yields of both suspended sediment and total phosphorus were in the Nooksack River at North Cedarville, which drains a forested area.

Erosion of unstable streambanks and landslides transport sediment and phosphorus to headwater streams in the Nooksack River Basin. Most of the unstable streambanks are associated with road construction and logging (U.S. Forest Service, 1995).

Concentrations of Nitrate and Phosphorus in Streams and Rivers Have Not Changed or Have Only Slightly Increased with Time

Concentrations of nitrate, one of the major forms of nitrogen in water, and of total phosphorus have not changed much over the period 1980 to 1997 in several Puget Sound Basin rivers and streams (fig. 13). In three streams, nitrate concentrations have increased by a small amount, 0.014 mg/L per year or less. Only in

Site name (land use and cover, dominant in mixed)	Nitrate, as N		Total phosphorus	
	Trend	Rate of change, in mg/L per year	Trend	Rate of change, in mg/L per year
Skokomish River (mixed, forest)	↑	0.004	—	—
Nooksack River at Brennan (mixed, agricultural)	—	—	—	—
Nooksack River at N. Cedarville (mixed, forest)	—	—	—	—
Green River at Renton Junction (mixed, urban)	—	—	—	—
Big Soos Creek (urban)	↑	0.010	—	—
Newaukum Creek (agricultural)	↑	0.014	↑	0.002

— No trend or no value ↑ Significant upward trend mg/L=milligrams per liter

Figure 13. In three of six rivers and streams monitored by the Washington State Department of Ecology and King County, nutrient concentrations have remained stable from 1980 to 1997. In three others, concentrations of nitrate have gradually increased with time.

Newaukum Creek did concentrations of both nitrogen and phosphorus increase over time. Although data were not collected to determine the cause, trends often reflect land-use changes. For example, urban development might be a factor related to increasing nitrate concentrations in Big Soos Creek and in Newaukum Creek, which drains an agricultural basin with hobby farms and residential development.

Runoff During Rainstorms Contributes Contaminants to Streams, Indicating Nonpoint Sources

Washoff of nutrients and pesticides during storms causes both seasonal and short-term temporal variations in concentrations. A typical seasonal pattern for Puget Sound Basin streams and rivers is illustrated by lower concentrations of nitrogen and phosphorus in Newaukum Creek from July

through October when rainfall amounts are lowest (fig. 14).

Commonly used pesticides, such as diazinon, often had higher concentrations during rainstorms when they were washed off from areas of application (fig. 15). Even when daily rainfall amounts were small, less than 0.05 inch, diazinon concentrations increased in Thornton Creek.

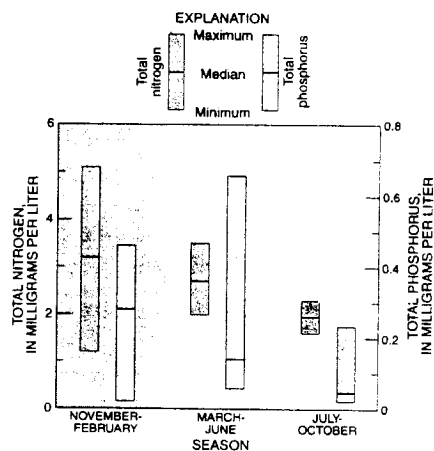


Figure 14. Nutrient concentrations in Newaukum Creek were highest during the rainy winter and spring periods. Concentrations were lowest during late summer-early fall when rainfall amounts are lowest.

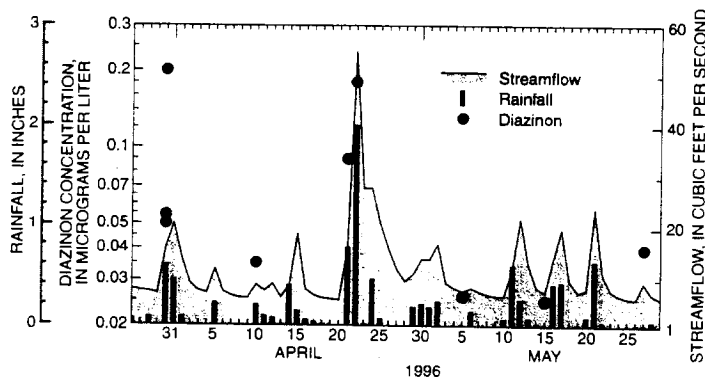


Figure 15. Peak concentrations of diazinon in Thornton Creek often occurred during spring rainstorms, which can produce an inch or more of rain in a day.

Bacteria Indicate the Presence of Fecal Contamination in Many Puget Lowland Streams

A study of 31 small lowland streams showed fecal coliform, *E. coli*, and enterococci bacteria to be present in every stream sampled during base flow. The presence of *E. coli* and enterococci bacteria is evidence that fecal contamination has occurred. During base flow, fecal contamination in urban streams could result from leaky sewer systems, failing septic

systems, and direct fecal inputs from pets and wildlife, including waterfowl. In agricultural and rural streams, fecal contamination is likely to be mostly from animals, including farm animals (dairy and beef cattle, and horses). Other sources could be wildlife, with perhaps some input from onsite septic systems.

Concentrations of all three fecal-indicator bacteria frequently exceeded standards and criteria (fig. 16); 81 percent of all sites had concentrations of fecal coliforms



Concentrations of bacteria in urban streams, which are attractive recreation areas, often exceed recommended concentrations for moderate water-contact recreation. (Photograph by Ward W. Staubitz, U.S. Geological Survey.)

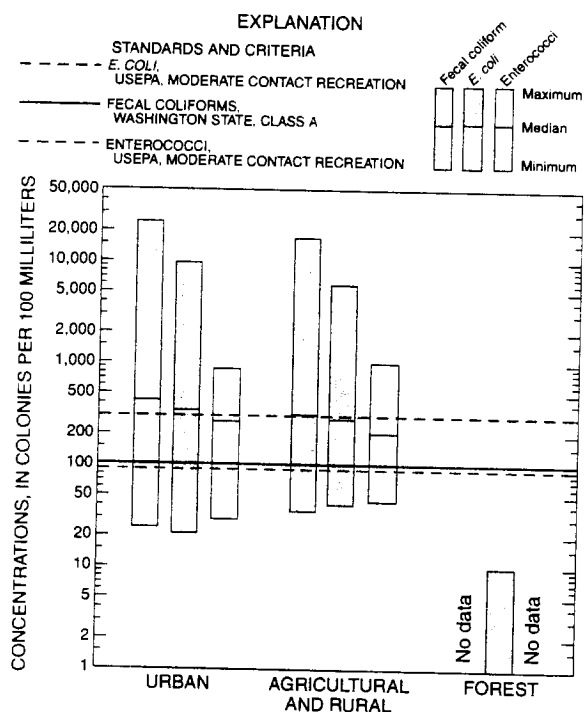


Figure 16. Concentrations of *E. coli* and enterococci bacteria exceeded U.S. Environmental Protection Agency recreational criteria, and fecal coliform bacteria exceeded Washington State standards in many lowland streams. Concentrations of bacteria in single samples were compared with criteria and standards. USEPA criteria for moderate full-body contact recreation apply to a single sample (U.S. Environmental Protection Agency, 1986). Washington State standards for fecal coliform bacteria apply to the geometric mean of concentrations in samples collected during a period of no more than 30 days (State of Washington, 1992). Because no minimum number of samples is specified, the standard is routinely applied to concentrations in single samples (Hallock and others, 1996).

exceeding Washington State standards, and 48 percent of all sites had levels of *E. coli* exceeding USEPA's recommended concentration for moderate water-contact recreation. Because concentrations of *E. coli* and enterococci are related to cases of gastrointestinal illness in swimmers (U.S. Environmental Protection Agency, 1986), there could be some risk of illness to children and adults playing and swimming in these accessible streams.

The types of fecal bacteria and their concentrations in streams were similar among urban, agricultural, and rural areas (fig. 16). However, concentrations of *E. coli* in the urban and agricultural streams were all well above those in the upper Green River in the forested headwaters of the basin.

Stream Habitat and Health Are Degraded in Agricultural and Urban Streams

During the late summer and early fall, between 1995 and 1998, the USGS and the Washington State Department of Ecology evaluated instream and riparian (streamside) habitat conditions at 45 sites. These evaluations indicated that habitat conditions at streams draining urban

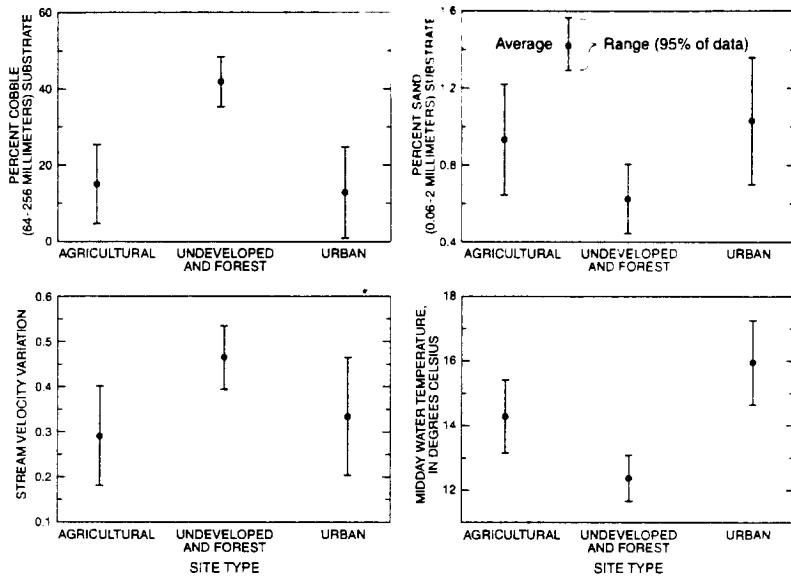


Figure 17. Values of selected habitat variables indicate that habitat conditions are degraded in agricultural and urban streams compared with streams in forest and undeveloped land. For each habitat variable, except temperature, multiple measurements were taken within a 330- to 660-foot-long stream reach at each study site.

and agricultural basins were significantly different from those at streams draining basins with forest and undeveloped land (fig. 17). Compared with either urban or agricultural sites, forest and undeveloped stream sites had a much higher percentage of cobble substrate, ideal for a diverse population of aquatic organisms, and a lower percentage of sand, detrimental to salmon spawning. In addition, forest and undeveloped sites had a much higher variability in water flow velocities than either urban or agricultural sites. More variability in flow velocities increases habitat diversity and wildlife diversity. Midday water temperatures were also much lower in the forested streams. The higher temperatures observed in the agricultural and urban streams are not ideal for native salmon and other aquatic organisms.

Human Activities Have Altered Aquatic Invertebrate Communities

As part of the USGS and Washington State Department of Ecology habitat data collection, aquatic invertebrates (insects and worms) were also collected from multiple riffle habitats at all 45 sites. A number of invertebrate community measures were examined to evaluate the status and quality of the aquatic invertebrate community. For each invertebrate community measure, values were lower, significantly so in some cases, at the urban and agricultural sites (fig. 18).

Urban development and agriculture can reduce habitat quality, alter typical stream flows, and increase chemical contaminants and temperatures, all of which are reflected in invertebrate community measures. A reduction in the total number of different invertebrates, stoneflies, mayflies, intolerant invertebrates, and percentage of predatory invertebrates indicates stream ecosystem degradation. For example, the total number of different invertebrates was much higher at the forest and undeveloped sites. A greater number of different invertebrates indicates a less degraded stream system capable of supporting more numerous desirable species such as salmon.

A summation of many invertebrate community measures is also shown in figure 18. This measure is known as a Benthic Index of Biological Integrity (BIBI) and is an overall indication of the biological integrity or health of a particular stream site (Black and MacCoy, 2000). As seen in figure 18, urban and agricultural systems have lower biological integrity scores than the forest and undeveloped sites.

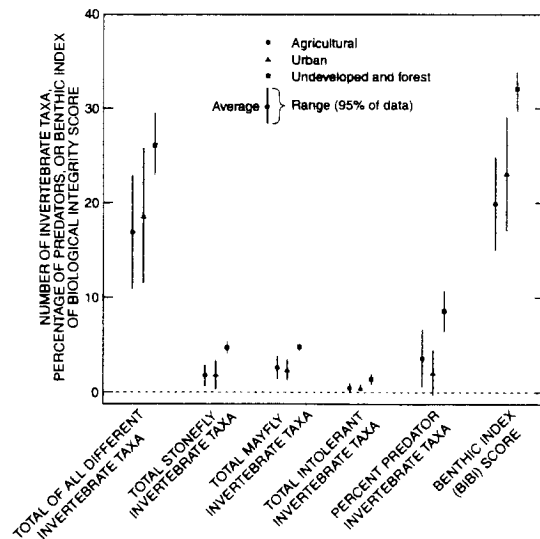
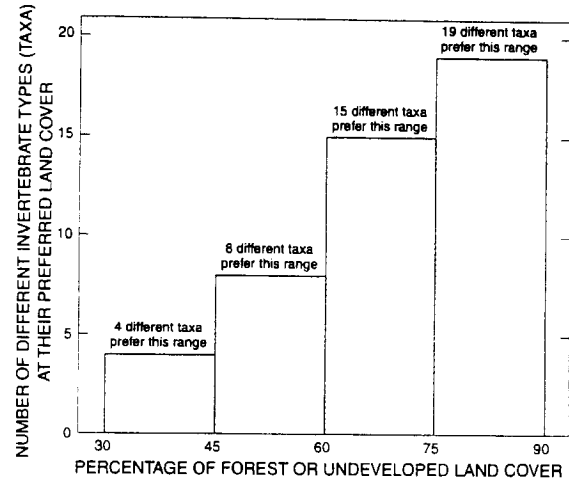


Figure 18. As indicated by the average values of six different invertebrate community indicators of stream ecosystem conditions, conditions were better at sites in forest and undeveloped land than at sites in urban and agricultural areas. Lower values suggest a more degraded stream site. Taxa are different types of invertebrates.

Streams in Urban and Agricultural Areas Have Less Than Optimal Amounts of Forest and Undeveloped Land to Support Healthy Aquatic Invertebrate Communities

A total of 46 different aquatic invertebrate taxa were identified at 45 sites throughout the Puget Sound Basin. Of the 46, only 4 taxa prefer watersheds with low forest/undeveloped land cover, whereas 19 taxa prefer watersheds with high forest/undeveloped land cover (fig. 19). These results indicate that streams in watersheds dominated by urban and agricultural land are optimally suited for only a few of the different taxa and many of these taxa are indicators of degraded stream ecosystems. Watersheds dominated by forest and undeveloped land are preferred by a much larger number of different invertebrate taxa, many of which are indicators of healthy stream systems.

Figure 19. More different aquatic invertebrate types (taxa) prefer streams that have watersheds with greater than 75 percent forest/undeveloped land cover. Each bar represents the preferred range of forest/undeveloped land cover



for the specified number of invertebrate taxa out of the 46 taxa identified in the Puget Sound Basin. An animal is healthiest when it is in its preferred range. This figure indicates that more different invertebrate taxa prefer sites in forested/undeveloped watersheds, but not that more invertebrate taxa or total numbers of invertebrates were found at these sites. The method used to determine preferred land-cover percentages was based on the work of Line and others (1994).

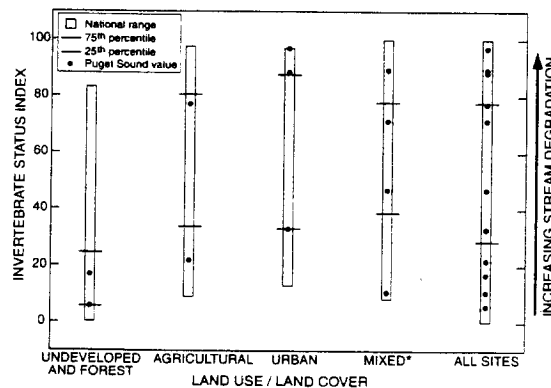


Puget Sound Stream Invertebrate Status Ranged from High to Low When Compared with National Levels

In the Puget Sound Basin, aquatic invertebrates collected at the 11 fixed monitoring sites were compared with those collected at 140 NAWQA sites nationwide. An **Invertebrate status index** (Tom Cuffney, U.S. Geological Survey, written commun., 2000) was developed by averaging 11 invertebrate metrics that summarized changes in richness, tolerance, food preference, and dominance associated with water-quality degradation. Invertebrate status scores in the basin ranged from low (less degraded) for streams in forest and undeveloped areas to high (more degraded) for other streams. As indicated by the invertebrate status scores, some Puget Sound Basin streams are highly degraded. This is the result

of poor habitat conditions and possibly water chemistry. As shown in figure 17, high midday water temperatures and percent sand were observed in urban and agricultural streams, both of which have

negative effects on aquatic organisms. In addition, numerous pesticides, some at concentrations above guidelines set to protect aquatic life, were detected at these sites (see Appendix).



Invertebrate status index by land use and cover for Puget Sound Basin sites in relation to national conditions. A low invertebrate status index score represents a less degraded stream site. *Mixed land uses can have an agricultural, urban, or forestry influence (table 2).

Ground-Water Quality is Generally Good

Prior to conducting field studies, an analysis of existing data was conducted to determine which Puget Sound Basin aquifers are most at risk to contamination. Unconfined aquifers that are overlain by coarse-grained glacial deposits (see map, p. 20) were found to be the most susceptible to contamination (see sidebar). Aquifers in coarse-grained glacial deposits in the Puget Sound Basin are collectively referred to as the Fraser aquifer. Three studies were conducted in the unconfined part of the Fraser aquifer (Inkpen and others, 2000). A study-unit survey was conducted by randomly selecting 30 domestic wells without regard to land use to evaluate the overall quality of shallow ground water. Ground-water-quality studies were also conducted in residential areas in Pierce and Thurston Counties

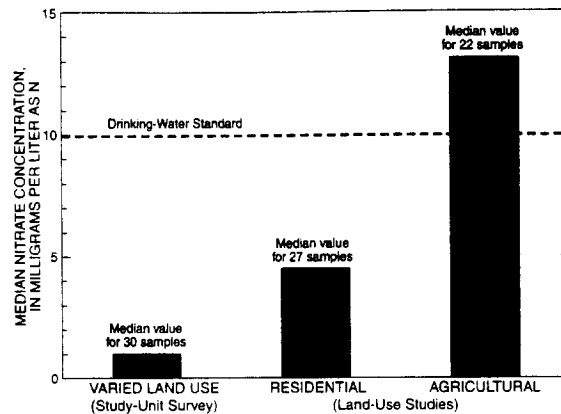


Figure 20. Median nitrate concentrations in shallow ground water sampled for the study-unit survey (varied land use) were low. Higher nitrate concentrations were more common in both the residential land-use and agricultural land-use studies.

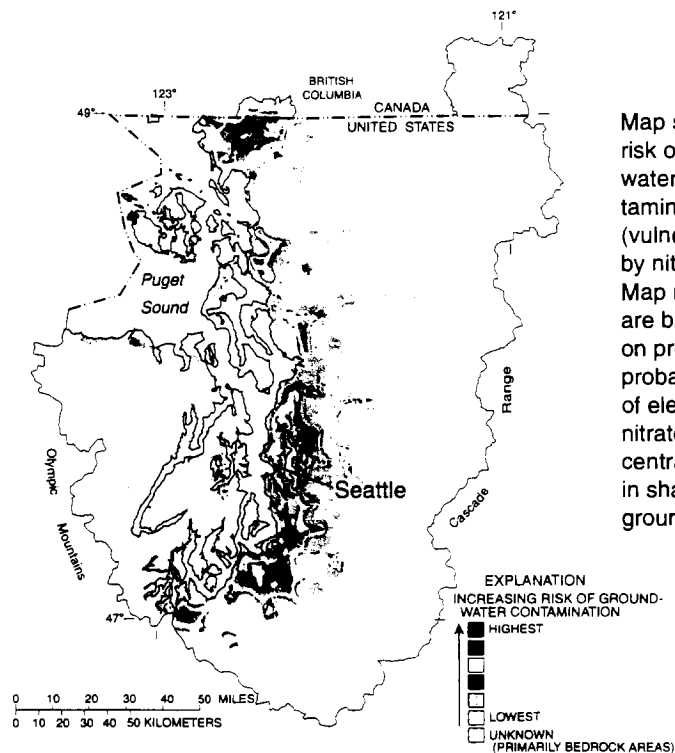
and in an agricultural area in Whatcom County (see map, p. 20) to examine the influence of these important land-use categories on shallow ground water. Shallow ground water in all parts of the Fraser aquifer sampled is used for domestic supplies.

The study-unit survey results indicate that ground water is of generally good quality, as only one well did not meet a drinking-water standard (for nitrate). Nitrate was detected in

most wells, but the concentrations were low, having a median concentration of 1.0 mg/L (fig. 20). Pesticides were detected in 20 percent of study-unit survey wells (fig. 21) but were present at low concentrations (less than 0.2 µg/L) (micrograms per liter) and were well below drinking-water standards or guidelines. Based on data from a previous study (Ryker and Williamson, 1996), pesticides were not detected in wells more than

Predicting Ground-Water Susceptibility and Vulnerability

A statistical model (Tesoriero and Voss, 1997; Erwin and Tesoriero, 1997) was created to predict which areas are (1) most likely to become contaminated if sources of contaminants are present (susceptibility) and (2) at the greatest risk of contamination, based on current land-use practices (vulnerability). Well depth, surficial geology, and land use were the factors that significantly correlated with elevated nitrate concentrations and were used in the models. Shallow ground water in areas with coarse-grained glacial deposits at the surface were the most susceptible to contamination. These areas become increasingly vulnerable to nitrate contamination as the amount of agricultural and urban land use increases. Vulnerable areas include the intensive agricultural areas in the northern part of the Study Unit as well as urban areas extending north and south of Seattle.



Map shows risk of ground-water contamination (vulnerability) by nitrate. Map results are based on predicted probabilities of elevated nitrate concentrations in shallow ground water.

120 feet deep (Bortleson and Ebbert, 2000). Volatile organic compounds were detected frequently (80 percent of wells sampled) but were present at low concentrations (less than 1 µg/L).

Radon concentrations exceeded the proposed drinking-water standard of 300 picocuries per liter (U.S. Environmental Protection Agency, 1999a) in about 50 percent of the domestic and monitoring wells sampled for radon (table 3, p. 21). Radon is a naturally occurring product from the decay of uranium. The median concentration of radon in ground water in the Puget Sound Basin was 320 picocuries per liter. Elevated radon concentrations are by no means unique to the Puget Sound Basin. In fact, radon concentrations in ground water collected for NAWQA studies throughout the Nation exceeded the proposed standard more often and had a higher median value (420 picocuries per liter) than in the Puget Sound Basin (Wentz and others, 1999).

Nitrate Concentrations in Shallow Ground Water Beneath Residential Areas Were Elevated, but Concentrations of Other Compounds Were Low

Although nitrate concentrations in shallow ground water beneath the residential areas in Pierce and Thurston Counties were elevated compared with those in ground water sampled by the study-unit survey (fig. 20), only one sample exceeded the drinking-water standard. Stable isotope measurements of nitrogen in nitrate suggest that septic-system effluent is a significant source of nitrate in unsewered areas. Additional nitrate sources are indicated by the similar concentrations of nitrate in sewerred and unsewered areas (Inkpen and others, 2000). Some nitrate in ground water in unsewered areas and most nitrate in sewerred areas is likely from the application of fertilizers to lawns and gardens.

Pesticides Were Rarely Detected in Ground Water Beneath Residential Areas

In spite of the vulnerability of the unconfined Fraser aquifer, only a few pesticides were detected (fig. 21) and these were at low concentrations (less than 0.2 microgram per liter) beneath residential areas developed since 1970. These results suggest that pesticides currently used in residential areas may pose little risk to ground water. However, it should be noted that several commonly used pesticides (glyphosate, for example) were not analyzed.

Volatile Organic Compounds Were Detected Frequently at Low Concentrations in Ground Water Beneath Residential Areas

Although volatile organic compounds (VOCs) were detected in over 90 percent of shallow ground-water samples in residential areas, concentrations were low (fig. 22). Most detections were less than 0.05 µg/L,

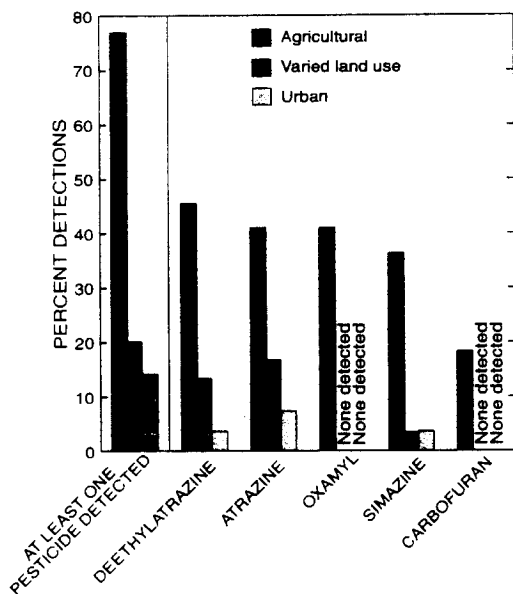


Figure 21. A pesticide was detected in less than 15 percent of shallow ground-water samples beneath residential areas, and the most frequently detected compounds were each detected in less than 10 percent of these samples.

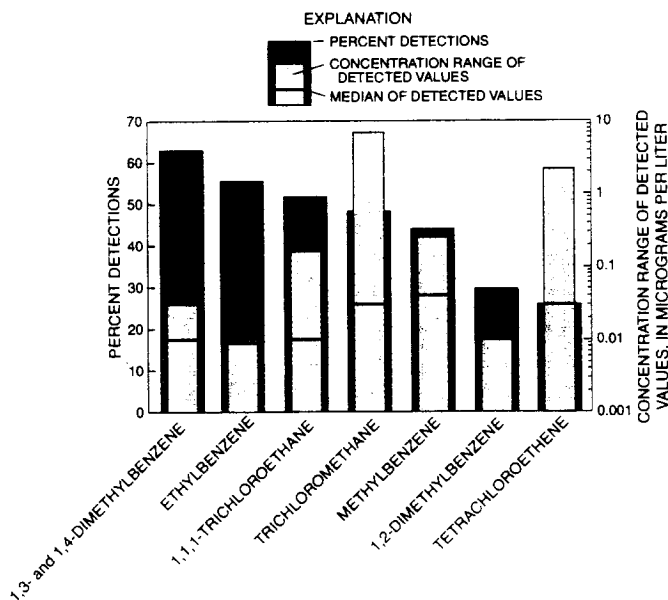


Figure 22. Volatile organic compounds were frequently detected at low concentrations in shallow ground water beneath residential areas.

and none was above drinking-water standards or guidelines. Solvents, chlorinated by-products, and fuel-related compounds were the most commonly detected VOCs.

High Nitrate Concentrations and Fumigant-Related Compounds Were Detected in Shallow Ground Water Beneath an Agricultural Area

Nitrate concentrations in more than half of the shallow ground water sampled for the agricultural land-use study exceeded the drinking-water standard. Major sources of nitrate are animal manure from poultry and dairy operations and fertilizers applied to crops. High concentrations of nitrate have persisted for many years in wells sampled in this aquifer (Hii and others, 1999), suggesting that land applications of manure and inorganic fertilizers are contributing nitrate to this aquifer at a sufficient rate to sustain these levels (fig. 23).

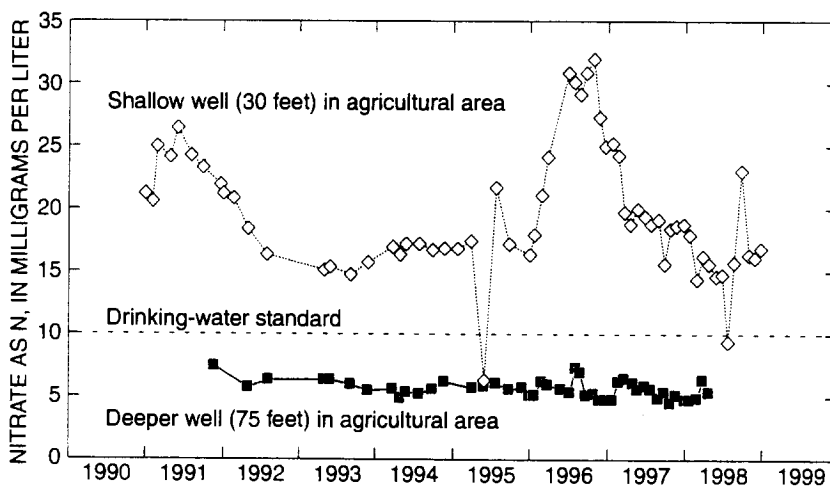


Figure 23. Nitrate concentrations in shallow ground water often exceeded the drinking-water standard (10 milligrams per liter), with high concentrations persisting for many years. Nitrate concentrations in deeper ground water, while elevated, generally met the drinking-water standard. Both the shallow and deeper ground water are used for domestic supplies.

Reactions in the Riparian Zone Remove Nitrate from Ground Water

Nitrate concentrations in ground water in the agricultural land-use study area commonly exceeded 10 mg/L, a concentration that can promote excessive plant growth in streams where this ground water discharges. A detailed analysis of water-quality changes along ground-water flow paths was conducted in this area (flow-path study area, p. 20). High nitrate concentrations persisted throughout much of the shallow aquifer. It was not until ground water reached the riparian zone of Fishtrap Creek that nitrate concentrations decreased to low levels (fig. 24). A decrease in nitrate concentrations in ground water concurrent with an increase in nitrogen gas (N_2) concentrations in this zone is evidence that nitrate is

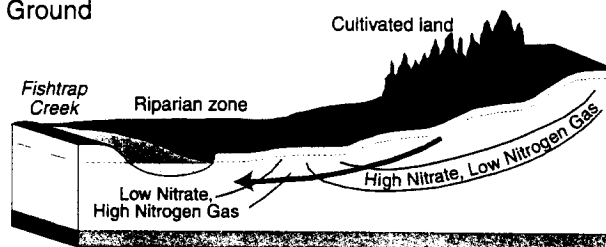


Figure 24. Nitrate concentrations in ground water remain high until water seeps through stream sediments that favor microbial denitrification. This process can effectively transform nitrate to nitrogen gas before the water discharges to the stream.

being converted to N_2 by denitrification (Tesoriero and others, 2000).

High Concentrations of 1,2-Dichloropropane Were Probably Caused by Applications of Older Fumigant Formulations

The most commonly detected volatile organic compounds in the agricultural land-use study area were associated with the application of fumigants to soils prior to planting. One or more fumigant-related compounds (1,2-dichloropropane, 1,2,2-trichloropropane, and 1,2,3-trichloropropane) were detected in over half of the samples. Each of these compounds is present in varying amounts in historically and/or presently used fumigants. Concentrations of 1,2-dichloropropane in water from two wells were above the drinking-water standard for this compound (fig. 25).

The amount of 1,2-dichloropropane and 1,2,3-trichloropropane in fumigant formulations has dropped substantially over the past few decades (Zearth and others, 1998), while the amount of 1,2,2-trichloropropane has decreased only slightly.

The relative amounts of these chloropropanes in ground water were compared with those present in

fumigant formulations to determine the origin of these compounds in ground water (Tesoriero and others,

in press). Results indicate that high concentrations of 1,2-dichloropropane in ground water are largely due to older formulations (fig. 26).

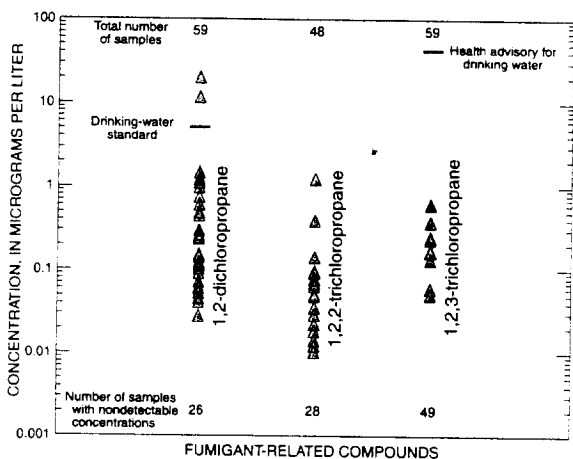


Figure 25. Fumigant-related compounds were detected frequently in shallow ground water in the agricultural land-use study area. Concentrations of 1,2-dichloropropane exceeded the drinking-water standard in two samples.

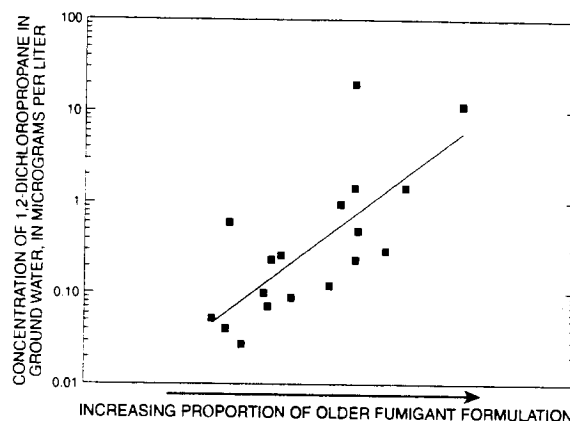

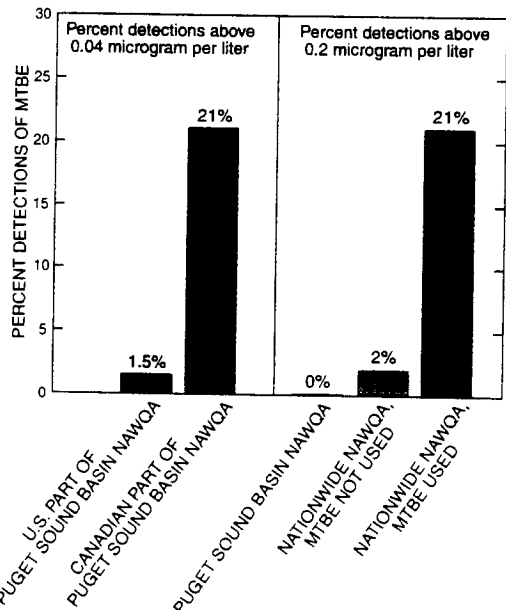


Figure 26. Based on the relative amounts of chloropropanes in samples, higher concentrations of 1,2-dichloropropane are likely derived from older fumigant formulations.



MTBE Was Rarely Detected in Ground Water in the Puget Sound Basin

Areas that exceed air-quality standards for carbon monoxide are required to use a fuel oxygenate to comply with the Clean Air Act of 1990. Methyl *tert*-butyl ether (MTBE) is the most widely used oxygenate in the United States. However, frequent detections of MTBE, sometimes high concentrations, in ground water in areas where this compound is used has caused the USEPA to recommend that it be discontinued or reduced (U.S. Environmental Protection Agency, 1999b). MTBE is used in the Canadian part of the Puget Sound Basin but not in the U.S. part. Not surprisingly, detections of this compound were more frequent in the Canadian part of the Study Unit. When detected, MTBE concentrations were low (less than



Region	Percent detections above 0.04 µg/L	Percent detections above 0.2 µg/L
U.S. PART OF PUGET SOUND BASIN	1.5%	0%
CANADIAN PART OF PUGET SOUND BASIN	21%	21%
NATIONWIDE (MTBE NOT USED)	0%	2%
NATIONWIDE (MTBE USED)	1.5%	21%

The detection frequency of MTBE in shallow ground water in the Puget Sound Basin was less than detection frequencies in other parts of the Nation (Squillace, 1999). Ethanol, not MTBE, is used as a fuel oxygenate in the U.S. part of the Puget Sound Basin.

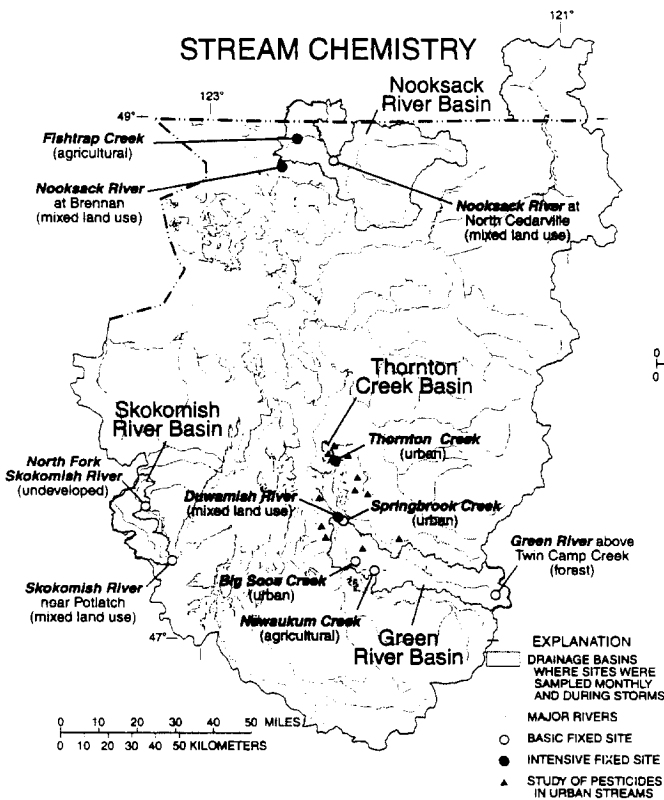
0.2 microgram per liter). Low concentrations of MTBE are likely derived from nonpoint sources, such

as stormwater runoff or atmospheric deposition, or from diluted point sources (Squillace and others, 1996).

STUDY UNIT DESIGN

The Puget Sound Basin study was designed to address local and national goals of providing widely comparable water-quality data focused on stream chemistry, stream ecology, and ground-water chemistry.

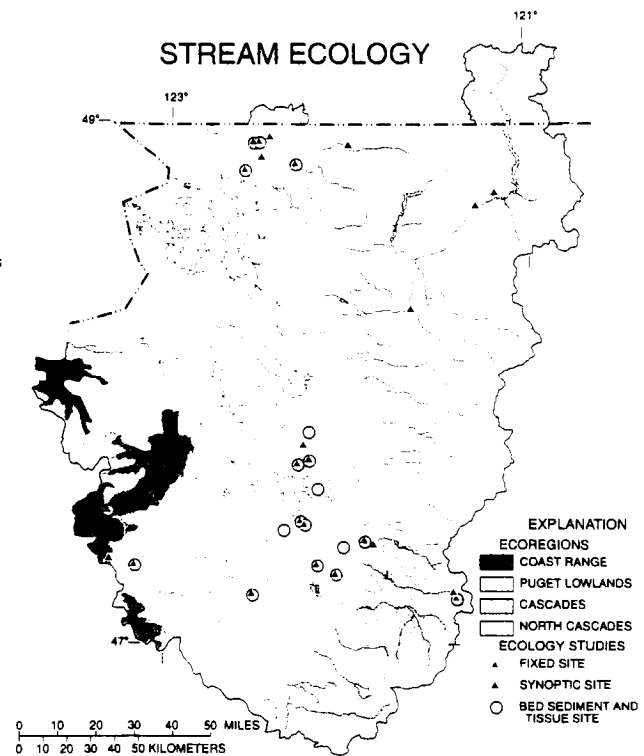
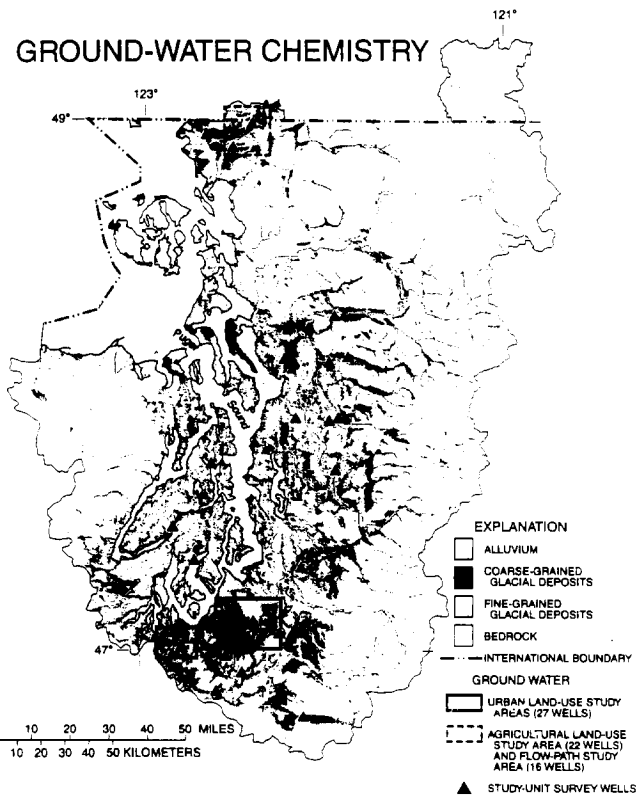
Of the 12 major and numerous minor tributaries to the Puget Sound, sampling was concentrated in four representative drainage basins: the Nooksack and Green River Basins with varied land uses, the Thornton Creek Basin in a totally urban environment, and the Skokomish River Basin, which is mostly forested.



Some sampling was done outside these areas for special studies, such as the study of pesticides in urban streams.

Wells sampled for the survey of ground-water quality in the Study Unit were distributed throughout the Puget Lowlands. Agricultural effects on ground-water quality and changes in quality along flow paths were evaluated using wells located in the lower Nooksack River Basin. Wells sampled in residential areas surrounding Olympia and Tacoma were used for determining urban land-use effects on shallow ground-water quality.

Stream ecology, bed sediment, and aquatic biota sampling was done at all the fixed stream-chemistry sites, and one or more of these types of samples were



collected at 14 other sites. Two-thirds of the sites were within the Puget Lowlands, while the remainder were in other ecoregions (Black and Silkey, 1998).

Table 3. Summary of data collection in the Puget Sound Basin, 1994–98¹

Study component	What data were collected and why	Types of sites sampled	Number of sites	Sampling frequency and period
Stream Chemistry				
Basic fixed sites—general water quality	Streamflow, dissolved oxygen, pH, alkalinity, specific conductance, temperature, nutrients, major ions, trace elements, organic carbon, and suspended sediment were measured to determine occurrence and concentration.	Streams draining basins ranging in size from 12 to 790 square miles, reflecting forest and mixed land use, and widely distributed geographically within the Study Unit.	7	Monthly plus storms Mar. 1996–Apr. 1998
Intensive fixed-sites—pesticides and VOCs	Above constituents plus 87 pesticides and 85 volatile organic compounds.	Sites selected for closer proximity to and more direct influence from agricultural and urban land uses plus integration of mixed-use larger basins.	3 1	Weekly to monthly Mar. 1996–May 1997 Weekly to monthly Mar. 1996–May 1998
Synoptic sites—pesticides	Streamflow, pH, specific conductance, temperature plus pesticides during varying flow conditions to relate occurrences and concentrations to retail sales of pesticides.	Sites predominantly influenced by urban residential land use plus 1 reference site.	13	2 to 4 samples over storm hydrograph Apr.–May 1998
Contaminants in bed sediment	Trace elements and organic compounds to determine occurrence and distribution in streambed sediment.	Depositional zones of all basic and intensive stream-chemistry sites plus additional sites.	19	Once Sept. 1995
Contaminants in fish tissue	Trace elements and organic compounds in the tissue of whole fish.	Same sites from which bed sediment samples were collected.	18	Once Sept. 1995
Stream Ecology				
Fixed sites	Invertebrate, algae, and fish communities, streamflow, basic water chemistry, and riparian habitat conditions surveyed to assess biological communities in the basin.	Sites collected with basic and intensive stream-chemistry sites and having contributing drainage areas from 12 to 790 square miles.	11	Annually 1995–97
Synoptic	Invertebrate, algae, and fish communities, streamflow, nutrients, and habitat conditions surveyed to determine land-use effects on biological communities.	4 bed sediment and tissue sites and 10 other sites influenced by various land uses, with contributing drainage areas ranging from 3 to 48 square miles.	14	Once Sept.–Oct. 1996
Ground-Water Chemistry				
Study Unit—varied land use	Nutrients, major ions, pesticides, volatile organic compounds, and radon in shallow, unconfined glacial outwash aquifer to assess the drinking-water quality of domestic wells in the Fraser aquifer.	Existing domestic supply wells widely distributed through the Puget Lowlands.	30	Once 1996
Land use—residential	Above compounds to determine effects of urban land use on ground-water quality in the Fraser aquifer. One-half of the wells were sampled for radon.	Monitoring wells in urban residential areas with both sewer and private septic systems.	27	Once 1996–97
Land use—agricultural	Above compounds, except radon, to determine effects of agricultural land use on ground-water quality in the Fraser aquifer.	Monitoring wells (18) and existing domestic supply wells (4) in an area of intensive row crops (raspberries, for example).	22	Once 1997–98
Flow path—agricultural	Above compounds, except radon, to determine changes in water quality occurring as water moves from recharge to discharge areas.	Shallow and deep monitoring wells along flow paths in an agricultural watershed.	16	Varied 1997–98
Special Studies				
Synoptic study—microbiology and wastewater chemicals	Turbidity, pH, specific conductance, temperature, wastewater chemicals, fecal-indicator bacteria, coliphage, and coliphage serotypes to determine occurrence and distribution in the Puget Lowlands and infer sources of fecal contamination.	Sites predominantly influenced by urban and agricultural land use.	31	Once Aug. 1998
Land use and scale	Invertebrates, instream habitat, and riparian condition data from USGS and Washington State Department of Ecology sites (common protocol) were combined to evaluate land-use impacts at different spatial scales.	Indicator sites with smaller drainage basins and mixed land uses.	20 USGS 25 WDOE	Annually Aug.–Sept. 1995–97
Drinking-water assessment—pesticides	Pesticides collected in previous study used to estimate detection probability.	Public-supply wells throughout the Study Unit.	78	Once 1994

¹Most data were collected 1996–98.

GLOSSARY

Aquatic guidelines - Specific levels of water quality which, if reached, may adversely affect aquatic life. These are nonenforceable guidelines issued by a governmental agency or other institution.

Aquifer - A water-bearing layer of soil, sand, gravel, or rock that will yield usable quantities of water to a well.

Atmospheric deposition - The transfer of substances from the air to the surface of the Earth, either in wet form (rain, fog, snow, dew, frost, hail) or in dry form (gases, aerosols, particles).

Base flow - Sustained, low flow in a stream; ground-water discharge is the source of base flow in most places.

Basic fixed sites - Sites on streams at which streamflow is measured and samples are collected for temperature, salinity, suspended sediment, major ions and metals, nutrients, and organic carbon to assess the broad-scale spatial and temporal character and transport of inorganic constituents of streamwater in relation to hydrologic conditions and environmental settings.

Benthic invertebrates - Insects, mollusks, crustaceans, worms, and other organisms without a backbone that live in, on, or near the bottom of lakes, streams, or oceans.

Breakdown product - A compound derived by chemical, biological, or physical action upon a pesticide. The breakdown is a natural process that may result in a more toxic or a less toxic compound and a more persistent or less persistent compound.

Chlorinated solvent - A volatile organic compound containing chlorine. Some common solvents are trichloroethylene, tetrachloroethylene, and carbon tetrachloride.

Coliphages - Bacteriophages, a type of virus, that infect and replicate in coliform bacteria and appear to be present wherever total and fecal coliforms are found.

Coliphage serotypes - Groups of coliphages that can be identified and used to infer sources of coliform bacteria.

Community - In ecology, the species that interact in a common area.

Denitrification - A process by which oxidized forms of nitrogen such as nitrate (NO_3^-) are reduced to form nitrites, nitrogen oxides, ammonia, or free nitrogen: commonly brought about by the action of denitrifying bacteria and usually resulting in the release of nitrogen to the air.

Detection limit - The minimum concentration of a substance that can be identified, measured, and reported within 99 percent confidence that the analyte concentration is greater than zero; determined from analysis of a sample in a given matrix containing the analyte.

Drainage basin - The portion of the surface of the Earth that contributes water to a stream through overland runoff, including tributaries and impoundments.

Drinking-water standard or guideline - A threshold concentration in a public drinking-water supply, designed to protect human health. As defined here, standards are U.S. Environmental Protection Agency regulations that specify the maximum contamination levels for public water systems required to protect the public welfare; guidelines have no regulatory status and are issued in an advisory capacity.

Ecoregion - An area of similar climate, landform, soil, potential natural vegetation, hydrology, or other ecologically relevant variables.

Fecal bacteria - Microscopic single-celled organisms (primarily fecal coliforms and fecal streptococci) found in the wastes of warm-blooded animals. Their presence in water is used to assess the sanitary quality of water for body-contact recreation or for consumption. Their presence indicates contamination by the wastes of warm-blooded animals and the possible presence of pathogenic (disease producing) organisms.

Flow-path study - A study to examine the relations of land-use practices, ground-water flow, and contaminant occurrence and transport. A flow-path study is conducted within a land-use study.

Freshwater chronic criteria - The highest concentration of a contaminant that freshwater aquatic organisms can be exposed to for an extended period of time (4 days) without adverse effects. See also Water-quality criteria.

Fumigant - A substance or mixture of substances that produces gas, vapor, fume, or smoke intended to destroy insects, bacteria, or rodents.

Habitat - The part of the physical environment where plants and animals live.

Headwaters - The source and upper part of a stream.

Intensive fixed sites - Basic fixed sites with increased sampling frequency during selected seasonal periods and analysis of dissolved pesticides and volatile organic compounds for 1 year. Most NAWQA Study Units have one to two integrator intensive fixed sites and one to four indicator intensive fixed sites.

Intolerant organisms - Organisms that are not adaptable to human alterations to the environment and thus decline in numbers where human alterations occur. See also Tolerant species.

Land-use study - A study that is a subset of the study-unit survey and has the goal of relating the quality of shallow ground water to land use. See also Study-unit survey.

Load - General term that refers to a material or constituent in solution, in suspension, or in transport; usually expressed in terms of mass or volume.

Median - The middle or central value in a distribution of data ranked in order of magnitude. The median is also known as the 50th percentile.

Metabolite - A substance produced in or by biological processes.

Micrograms per liter ($\mu\text{g/L}$) - A unit expressing the concentration of constituents in solution as weight (micrograms) of solute per unit volume (liter) of water; equivalent to one part per billion in most streamwater and ground water. One thousand micrograms per liter equals 1 mg/L .

Milligrams per liter (mg/L) - A unit expressing the concentration of chemical constituents in solution as weight (milligrams) of solute per unit volume (liter) of water; equivalent to one part per million in most streamwater and ground water. One thousand micrograms per liter equals 1 mg/L .

Nonpoint source - A pollution source that cannot be defined as originating from discrete points such as pipe discharge. Areas of fertilizer and pesticide applications, atmospheric deposition, manure, and natural inputs from plants and trees are types of nonpoint source pollution.

Nutrient - Element or compound essential for animal and plant growth. Common nutrients in fertilizer include nitrogen, phosphorus, and potassium.

Pesticide - A chemical applied to crops, rights of way, lawns, or residences to control weeds, insects, fungi, nematodes, rodents, or other "pests."

Picocurie (pCi) - One trillionth (10^{-12}) of the amount of radioactivity represented by a curie (Ci). A curie is the amount of radioactivity that yields 3.7×10^{10} radioactive disintegrations per second (dps). A picocurie yields 2.22 disintegrations per minute (dpm) or 0.037 dps.

Radon - A naturally occurring, colorless, odorless, radioactive gas formed by the disintegration of the element radium; damaging to human lungs when inhaled.

Riparian - Areas adjacent to rivers and streams with a high density, diversity, and productivity of plant and animal species relative to nearby uplands.

Study Unit - A major hydrologic system of the United States in which NAWQA studies are focused. Study Units are geographically defined by a combination of ground- and surface-water features and generally encompass more than 4,000 square miles of land area.

Study-unit survey - Broad assessment of the water-quality conditions of the major aquifer systems of each Study Unit. The study-unit survey relies primarily on sampling existing wells and, wherever possible, on existing data collected by other agencies and programs. Typically, 20 to 30 wells are sampled in each of three to five aquifer subunits.

Synoptic sites - Sites sampled during a short-term investigation of specific water-quality conditions during selected seasonal or hydrologic conditions to provide improved spatial resolution for critical water-quality conditions.

Taxon (plural taxa) - Any identifiable group of taxonomically related organisms.

Tolerant species - Those species that are adaptable to (tolerant of) human alterations to the environment and often increase in number when human alterations occur.

Unconsolidated deposit - Deposit of loosely bound sediment that typically fills topographically low areas.

Volatile organic compounds (VOCs) - Organic chemicals that have a high vapor pressure relative to their water solubility. VOCs include components of gasoline, fuel oils, and lubricants, as well as organic solvents, fumigants, some inert ingredients in pesticides, and some by-products of chlorine disinfection.

Water-quality criteria - Specific levels of water quality which, if reached, are expected to render a body of water unsuitable for its designated use. Commonly refers to water-quality criteria established by the U.S. Environmental Protection Agency. Water-quality criteria are based on specific levels of pollutants that would make the water harmful if used for drinking, swimming, farming, fish production, or industrial processes.

Yield - The mass of material or constituent transported by a river in a specified period of time divided by the drainage area of the river basin.

REFERENCES

- Black, R.W., Haggland, A.L., and Voss, F.D., 2000, Predicting the probability of detecting organochlorine pesticides and polychlorinated biphenyls in stream systems on the basis of land use in the Pacific Northwest, USA: *Environmental Toxicology and Chemistry*, v. 19, no. 4, p. 1044–1054.
- Black, R.W., and MacCoy, D.E., rev. 2000, The development and evaluation of a benthic index of biological integrity for the Cedar River Watershed, Washington: U.S. Geological Survey Water-Resources Investigations Report 99–4203, 92 p.
- Black, R.W., and Silkey, M., 1998, Water-quality assessment of the Puget Sound Basin, Washington—Summary of stream biological data through 1995: U.S. Geological Survey Water-Resources Investigations Report 97–4164, 78 p.
- Bortleson, G.C., and Ebbert, J.C., 2000, Occurrence of pesticides in streams and ground water in the Puget Sound Basin, Washington and British Columbia, 1996–98: U.S. Geological Survey Water-Resources Investigations Report 00–4118, 14 p.
- CCME (Canadian Council of Ministers of the Environment), 1995, Protocol for the derivation of Canadian sediment quality guidelines for the protection of aquatic life: Winnipeg, Manitoba, report CCME EPC–98E, Task Group on Water Quality Guidelines, 38 p.
- Embrey, S.S., and Inkpen, E.L., 1998, Water-quality assessment of the Puget Sound Basin, Washington, nutrient transport in rivers, 1980–93: U.S. Geological Survey Water-Resources Investigations Report 97–4270, 30 p.
- Erwin, M.L., and Tesoriero, A.J., 1997, Predicting ground-water vulnerability to nitrate in the Puget Sound Basin: U.S. Geological Survey Fact Sheet 061–97, 4 p.
- Hallock, Dave, Ehinger, William, and Hopkins, Brad, 1996, River and stream ambient monitoring report for wateryear 1995: Washington State Department of Ecology publication 96–355, 29 p., 3 appendices.
- Hii, Basil, Liebscher, Hugh, Mazalek, Mike, and Tuominen, Taina, 1999, Ground water quality and flow rates in the Abbotsford aquifer, British Columbia: Environment Canada, 36 p.
- Inkpen, E.L., and Embrey, S.S., 1998, Nutrient transport in the major rivers and streams of the Puget Sound Basin, Washington: U.S. Geological Survey Fact Sheet 009–98, 4 p.
- Inkpen, E.L., Tesoriero, A. J., Ebbert, J.C., Silva, S.R., and Sandstrom, M.W., 2000, Ground-water quality in regional, agricultural, and urban settings in the Puget Sound Basin, Washington and British Columbia, 1996–1998: U.S. Geological Survey Water-Resources Investigations Report 00–4100, 66 p.
- Line, J.M., ter Braak, C.J.F., and Birks, H.J.B., 1994, WACALIB version 3.3—A computer program to reconstruct environmental variables from fossil assemblages by weighted averaging and to derive sample-specific errors of prediction: *Journal of Paleolimnology*, v. 10, p. 147–152.
- Morgan, D.S., and Jones, J.L., 1999, Numerical model analysis of the effects of ground-water withdrawals on discharge to streams and springs in small basins typical of the Puget Sound Lowland, Washington: U.S. Geological Survey Water-Supply Paper 2492, 73 p.
- Newell, Arthur J., Johnson, David W., and Allen, Laurie K., 1987, Niagara River biota contamination project—Fish flesh criteria for piscivorous wildlife: New York State Department of Environmental Conservation, Division of Fish and Wildlife, Bureau of Environmental Protection, Technical Report 87–3, 182 p.
- Ryker, S.J., and Williamson, A.K., 1996, Pesticides in public supply wells in Washington State: U.S. Geological Survey Fact Sheet 122–96, 2 p.
- Squillace, P.J., Zogorski, J.S., Wilber, W.G., and Price, C.V., 1996, Preliminary assessment of the occurrence and possible sources of MTBE in ground-water in the United States, 1993–1994: *Environmental Science and Technology*, v. 30, no. 5, p. 1721–1730.
- Squillace, P.J., 1999, MTBE in the Nation's ground Water: U.S. Geological Survey, accessed August 18, 2000, at URL <http://sd.water.usgs.gov/nawqa/vocns/brp-pjs-handout.html>

- State of Washington, 1992, Water quality standards for surface waters of the State of Washington: Olympia, Washington, Washington Administrative Code, chap. 173-201A, 14 p.
- Staubitz, W.W., Bortleson, G.C., Semans, S.D., Tesoriero, A.J., and Black, R.W., 1997, Water-quality assessment of the Puget Sound Basin, Washington—Environmental setting and its implications for water quality and aquatic biota: U.S. Geological Survey Water-Resources Investigations Report 97-4013, 76 p.
- Tesoriero, A.J., Liebscher, H., and Cox, S.E., 2000, Mechanism and rate of denitrification in an agricultural watershed—Electron and mass balance along ground-water flow paths: *Water Resources Research*, v. 36, no. 6, p. 1545-1559.
- Tesoriero, A.J., Löffler, F.E., and Liebscher, H., in press, The fate and origin of 1,2-dichloropropane in an unconfined shallow aquifer: *Environmental Science and Technology*.
- Tesoriero, A.J., and Voss, F.D., 1997, Predicting the probability of elevated nitrate concentrations in the Puget Sound Basin—Implications for aquifer susceptibility and vulnerability: *Ground Water*, v. 35, no. 6, p. 1029-1039.
- Tetra Tech Incorporated, 1988, Pesticides of concern in the Puget Sound Basin—A review of contemporary pesticide usage: Seattle, Wash., prepared for U.S. Environmental Protection Agency, Region 10, contract TC3338-32, 97 p.
- U.S. Environmental Protection Agency, 1986, Ambient water quality criteria for bacteria, 1986: U.S. Environmental Protection Agency Office of Water, EPA 440/5-84-002, 18 p.
- _____, rev. February 23, 1998, Biological effects of ionizing radiation (BEIR) VI report, "The health effects of exposure to indoor radon"—Executive Summary: accessed November 9, 1999, at URL <http://www.epa.gov/iaq/radon/beirvi.html>
- _____, rev. October 18, 1999a, Proposed radon in drinking water rule: U.S. Environmental Protection Agency report EPA 815-F-99-006: accessed November 11, 1999, at URL <http://www.epa.gov/OGWDW/radon/fact.html>
- _____, 1999b, Achieving clean air and clean water—The report of the blue ribbon panel on oxygenates in gasoline: EPA420-R-99-021 accessed August 18, 2000, at URL <http://www.epa.gov/oms/consumer/fuels/oxypanel/blueribb.htm>
- U.S. Forest Service, 1995, Pilot watershed analysis for the Canyon Creek watershed (draft), Mt. Baker-Snoqualmie National Forest, Pacific Northwest region, 268 p.
- U.S. Geological Survey, 1999a, Pesticides analyzed in NAWQA samples—Use, chemical analyses, and water-quality criteria: U.S. Geological Survey, accessed April 12, 2000, at URL <http://water.wr.usgs.gov/pnsp/>
- _____, 1999b, The quality of our Nation's waters—Nutrients and pesticides: U.S. Geological Survey Circular 1225, 82 p.
- Voss, Frank D., Embrey, Sandra S., Ebbert, James C., Davis, Dale A., Frahm, Annette M., and Perry, George H., 1999, Pesticides detected in urban streams during rainstorms and relations to retail sales of pesticides in King County, Washington: U.S. Geological Survey Fact Sheet 097-99, 4 p.
- Wentz, D.A., Morace, J.L., and Ryker, S.J., 1999, Summary—Radon in ground water from 1991 and 1994 NAWQA study units: accessed April 26, 2000, at URL http://co.water.usgs.gov/trace/usgs/summary/radon_nov99.html
- Zebarth, B.J., Szeto, S.Y., Hii, B., Liebscher, H., and Grove, G., 1998, Groundwater contamination by chlorinated hydrocarbon impurities present in soil fumigant formulations: *Water Quality Research Journal Canada*, v. 33, no. 1, p. 31-50.

R0024743

APPENDIX—WATER-QUALITY DATA FROM THE PUGET SOUND BASIN IN A NATIONAL CONTEXT

For a complete view of Puget Sound Basin data and for additional information about specific benchmarks used, visit our Web site at <http://water.usgs.gov/nawqa/>. Also visit the NAWQA Data Warehouse for access to NAWQA data sets at <http://infotrek.er.usgs.gov/wdbctx/nawqa/nawqa.home>.

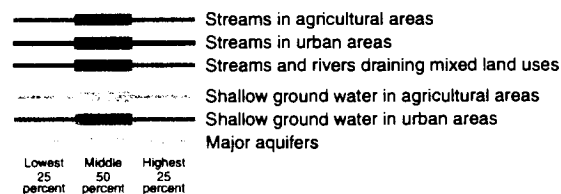
This appendix is a summary of chemical concentrations and biological indicators assessed in the Puget Sound Basin. Selected results for this basin are graphically compared to results from as many as 36 NAWQA Study Units investigated from 1991 to 1998 and to national water-quality benchmarks for human health, aquatic life, or fish-eating wildlife. The chemical and biological indicators shown were selected on the basis of frequent detection, detection at concentrations above a national benchmark, or regulatory or scientific importance. The graphs illustrate how conditions associated with each land use sampled in the Puget Sound Basin compare to results from across the Nation, and how conditions compare among the several land uses. Graphs for chemicals show only detected concentrations and, thus, care must be taken to evaluate detection frequencies in addition to concentrations when comparing study-unit and national results. For example, simazine concentrations in Puget Sound Basin agricultural streams were similar to the national distribution, but the detection frequency was much higher (97 percent compared to 61 percent).

CHEMICALS IN WATER

Concentrations and detection frequencies, Puget Sound Basin, 1996–98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals

- ◆ Detected concentration in Study Unit
- 66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- Not measured or sample size less than two
- 12 Study-unit sample size. For ground water, the number of samples is equal to the number of wells sampled

National ranges of detected concentrations, by land use, in 36 NAWQA Study Units, 1991–98—Ranges include only samples in which a chemical was detected

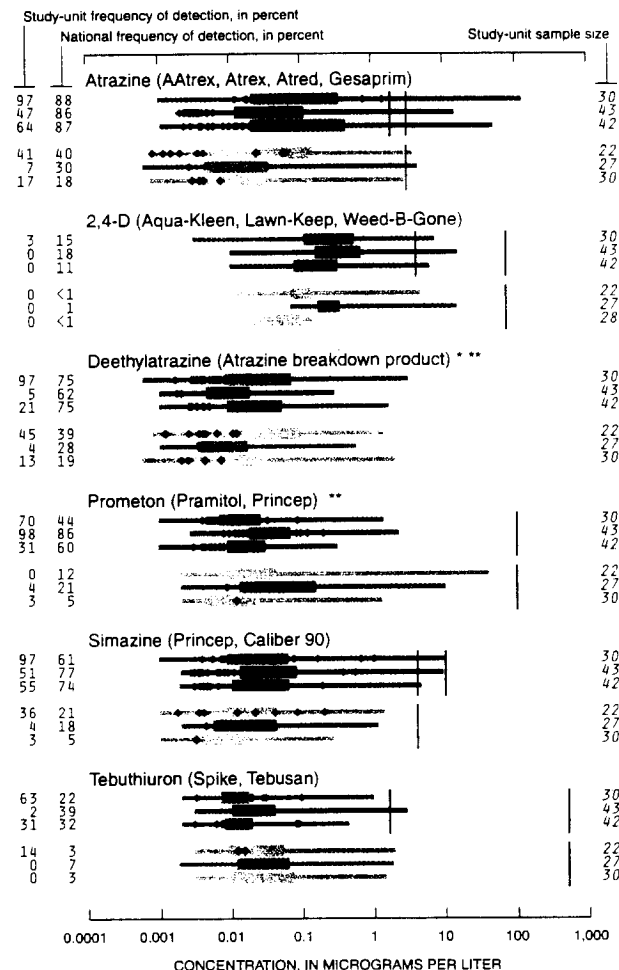


National water-quality benchmarks

National benchmarks include standards and guidelines related to drinking-water quality, criteria for protecting the health of aquatic life, and a goal for preventing stream eutrophication due to phosphorus. Sources include the U.S. Environmental Protection Agency and the Canadian Council of Ministers of the Environment

- | Drinking-water quality (applies to ground water and surface water)
- | Protection of aquatic life (applies to surface water only)
- | Prevention of eutrophication in streams not flowing directly into lakes or impoundments
- No benchmark for drinking-water quality
- ** No benchmark for protection of aquatic life

Pesticides in water—Herbicides



Other herbicides detected

- Acetochlor (Harness Plus, Surpass) **
- Alachlor (Lasso, Bronco, Lariat, Bullet) **
- Bromacil (Hyvar X, Urox B, Bromax)
- DCPA (Dacthal, chlorthal-dimethyl) **
- Dicamba (Banvel, Dianat, Scotts Proturf)
- Dinoseb (Dinoseb)
- Diuron (Crisuron, Karmex, Diurex) **
- EPTC (Eptam, Farmarox, Alirox) **
- Fluometuron (Flo-Met, Cotoran) **
- Linuron (Lorox, Linex, Sarclax, Linurex, Afalon) *
- MCPA (Rhomene, Rhonox, Chiptox)
- Metolachlor (Dual, Pennant)
- Metribuzin (Lexone, Sencor)
- Molinate (Ordran) **
- Napropamide (Devrinol) **
- Norflurazon (Evtal, Predict, Solicam, Zorial) **
- Oryzalin (Surflan, Dirimal) **
- Pebulate (Tillam, PEBC) **
- Pronamide (Kerb, Propyzamid) **
- Terbacil (Sinbar) **
- Triclopyr (Garlon, Grandstand, Redeem, Remedy) **
- Trifluralin (Treflan, Gowan, Tri-4, Trific)

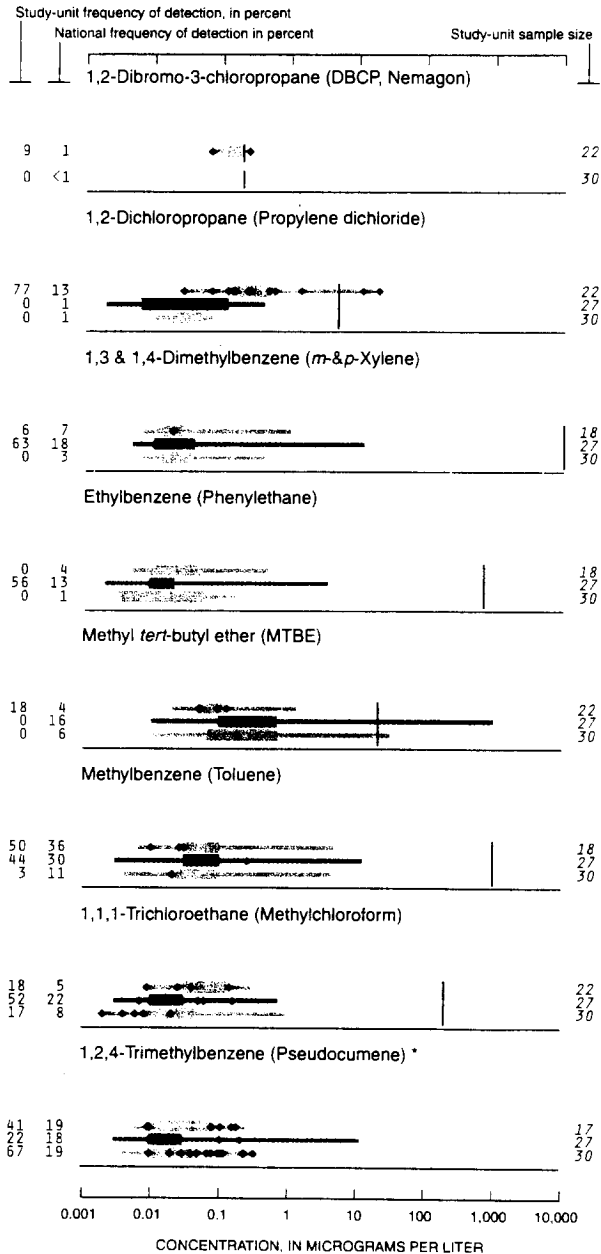
Herbicides not detected

- Acifluorfen (Blazer, Tackle 2S) **
- Benfluralin (Balan, Benefin, Bonalan) * **
- Bentazon (Basagran, Bentazone) **
- Bromoxynil (Buctril, Brominal) *
- Butylate (Sutan +, Genate Plus, Butilate) **
- Chloramben (Amiben, Amilon-WP, Vegiben) **
- Clopyralid (Stinger, Lontrel, Transline) * **
- Cyanazine (Bladex, Fortrol)
- 2,4-DB (Butyrac, Butoxone, Embutox Plus, Embutone) * **
- Dacthal mono-acid (Dacthal breakdown product) * **
- Dichloroprop (2,4-DP, Seritox 50, Lentemul) * **
- 2,6-Diethylaniline (Alachlor breakdown product) * **
- Ethalfuralin (Sonalan, Curbit) * **
- Fenuron (Fenulon, Fenidim) * **
- MCPB (Thistrol) * **
- Neburon (Neburea, Neburyl, Noruben) * **
- Pendimethalin (Pre-M, Prowl, Stomp) * **
- Picloram (Grazon, Tordon)
- Propachlor (Ramrod, Satecid) **
- Propanil (Stam, Stampede, Wham) * **
- Propham (Tuberite) **
- 2,4,5-T **
- 2,4,5-TP (Silvex, Fenoprop) **
- Thiobencarb (Bolero, Saturn, Benthicarb) * **
- Triallate (Far-Go, Avadex BW, Tri-allate) *

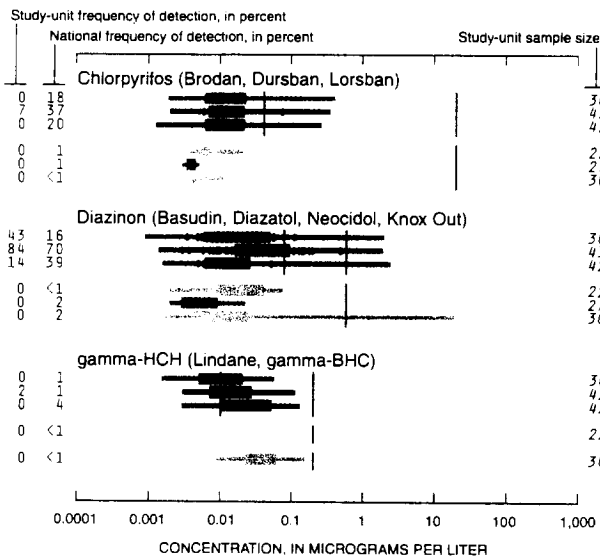
- Methomyl (Lanox, Lannate, Acinate) **
- Methyl parathion (Penncap-M, Folidol-M) **
- Parathion (Roethyl-P, Alkron, Panthion, Phoskil) *
- cis-Permethrin (Ambush, Astro, Pounce) * **
- Phorate (Thimet, Granutox, Geomet, Rampart) * **
- Propoxur (Baygon, Blattanex, Uden, Proprotox) * **
- Terbufos (Contraven, Counter, Pilarfox) **

Volatile organic compounds (VOCs) in ground water

These graphs represent data from 16 Study Units, sampled from 1996 to 1998



Pesticides in water—Insecticides



Other insecticides detected

- Carbaryl (Carbamine, Denapon, Sevin)
- Carbofuran (Furadan, Curaterr, Yaltox)
- Ethoprop (Mocap, Ethoprophos) * **
- Malathion (Malathion)
- Oxamyl (Vydate L, Pratt) **
- Propargite (Comite, Omite, Ornamate) * **

Insecticides not detected

- Aldicarb (Temik, Ambush, Pounce)
- Aldicarb sulfone (Standak, aldoxycarb)
- Aldicarb sulfoxide (Aldicarb breakdown product)
- Azinphos-methyl (Guthion, Gusathion M) *
- p,p'-DDE
- Dieldrin (Panoram D-31, Octalox, Compound 497)
- Disulfoton (Disyston, Di-Syston) **
- Fonofos (Dyfonate, Capfos, Cudgel, Tycap) **
- alpha-HCH (alpha-BHC, alpha-lindane) **
- 3-Hydroxycarbofuran (Carbofuran breakdown product) * **
- Methiocarb (Slug-Geta, Grandslam, Mesuro) * **

Other VOCs detected

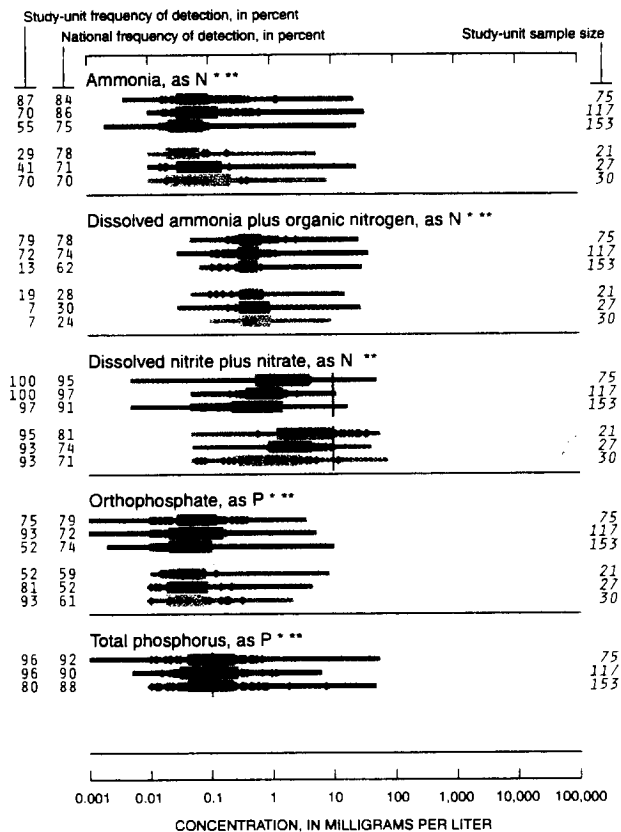
- Benzene
- Bromodichloromethane (Dichlorobromomethane)
- Carbon disulfide *
- 1-Chloro-2-methylbenzene (o-Chlorotoluene)
- Chlorobenzene (Monochlorobenzene)
- Chlorodibromomethane (Dibromochloromethane)
- Chloromethane (Methyl chloride)

R0024745

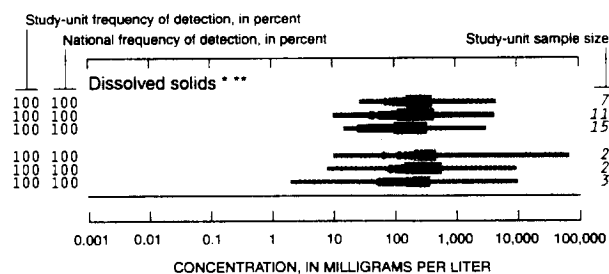
Dichlorodifluoromethane (CFC 12, Freon 12)
 1,2-Dichloroethane (Ethylene dichloride)
 Dichloromethane (Methylene chloride)
 1,3-Dichloropropane (Trimethylene dichloride) *
 1,2-Dimethylbenzene (*o*-Xylene)
 1-4-Epoxy butane (Tetrahydrofuran, Diethylene oxide) *
 1-Ethyl-2-methylbenzene (2-Ethyltoluene) *
 Iodomethane (Methyl iodide) *
 Isopropylbenzene (Cumene) *
 2-Propanone (Acetone) *
 n-Propylbenzene (Isocumene) *
 Tetrachloroethene (Perchloroethene)
 Trichloroethene (TCE) *
 Trichlorofluoromethane (CFC 11, Freon 11)
 Trichloromethane (Chloroform)
 1,2,3-Trichloropropane (Allyl trichloride)
 1,2,3-Trimethylbenzene (Hemimellitene) *
 1,3,5-Trimethylbenzene (Mesitylene) *

VOCs not detected
tert-Amyl methyl ether (*tert*-amyl methyl ether (TAME)) *
 Bromobenzene (Phenyl bromide) *
 Bromochloromethane (Methylene chlorobromide)
 Bromoethene (Vinyl bromide) *
 Bromomethane (Methyl bromide)
 2-Butanone (Methyl ethyl ketone (MEK)) *
n-Butylbenzene (1-Phenylbutane) *
sec-Butylbenzene *
tert-Butylbenzene *
 3-Chloro-1-propene (3-Chloropropene) *
 1-Chloro-4-methylbenzene (*p*-Chlorotoluene)
 Chloroethane (Ethyl chloride) *
 Chloroethene (Vinyl chloride)
 1,2-Dibromoethane (Ethylene dibromide, EDB)
 Dibromomethane (Methylene dibromide) *
trans-1,4-Dichloro-2-butene ((*Z*)-1,4-Dichloro-2-butene) *
 1,2-Dichlorobenzene (*o*-Dichlorobenzene)
 1,3-Dichlorobenzene (*m*-Dichlorobenzene)
 1,4-Dichlorobenzene (*p*-Dichlorobenzene)
 1,1-Dichloroethane (Ethylidene dichloride) *
 1,1-Dichloroethene (Vinylidene chloride)
trans-1,2-Dichloroethene ((*E*)-1,2-Dichloroethene)
cis-1,2-Dichloroethene ((*Z*)-1,2-Dichloroethene)
 2,2-Dichloropropane *
trans-1,3-Dichloropropene ((*E*)-1,3-Dichloropropene)
cis-1,3-Dichloropropene ((*Z*)-1,3-Dichloropropene)
 1,1-Dichloropropene *
 Diethyl ether (Ethyl ether) *
 Diisopropyl ether (Diisopropylether (DIPE)) *
 Ethenylbenzene (Styrene)
 Ethyl methacrylate *
 Ethyl *tert*-butyl ether (Ethyl-*t*-butyl ether (ETBE)) *
 Hexachlorobutadiene
 1,1,1,2,2,2-Hexachloroethane (Hexachloroethane)
 2-Hexanone (Methyl butyl ketone (MBK)) *
p-Isopropyltoluene (*p*-Cymene) *
 Methyl acrylonitrile *
 Methyl-2-methacrylate (Methyl methacrylate) *
 4-Methyl-2-pentanone (Methyl isobutyl ketone (MIBK)) *
 Methyl-2-propenoate (Methyl acrylate) *
 Naphthalene
 2-Propenenitrile (Acrylonitrile)
 1,1,2,2-Tetrachloroethane *
 1,1,1,2-Tetrachloroethane
 Tetrachloromethane (Carbon tetrachloride)
 1,2,3,4-Tetramethylbenzene (Prehnitene) *
 1,2,3,5-Tetramethylbenzene (Isodurene) *
 Tribromomethane (Bromoform)
 1,1,2-Trichloro-1,2,2-trifluoroethane (Freon 113) *
 1,2,4-Trichlorobenzene
 1,2,3-Trichlorobenzene *

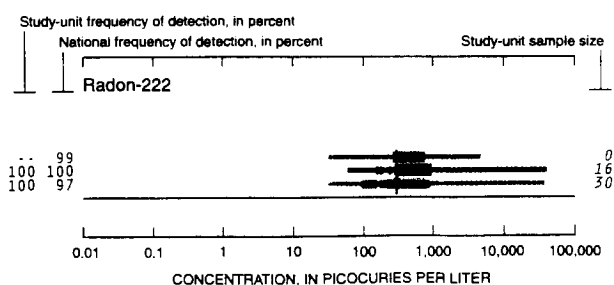
Nutrients in water



Dissolved solids in water



Trace elements in ground water



Other trace elements detected

Chromium

Trace elements not detected

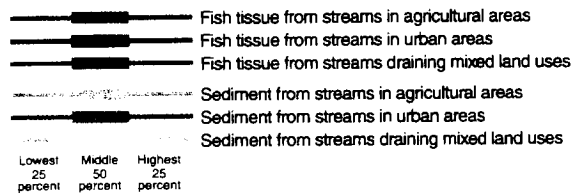
Arsenic
Cadmium
Lead
Selenium
Uranium
Zinc

CHEMICALS IN FISH TISSUE AND BED SEDIMENT

Concentrations and detection frequencies, Puget Sound Basin, 1996-98—Detection sensitivity varies among chemicals and, thus, frequencies are not directly comparable among chemicals. Study-unit frequencies of detection are based on small sample sizes; the applicable sample size is specified in each graph

- ◆ Detected concentration in Study Unit
- 66 38 Frequencies of detection, in percent. Detection frequencies were not censored at any common reporting limit. The left-hand column is the study-unit frequency and the right-hand column is the national frequency
- Not measured or sample size less than two
- 12 Study-unit sample size

National ranges of concentrations detected, by land use, in 36 NAWQA Study Units, 1991-98—Ranges include only samples in which a chemical was detected

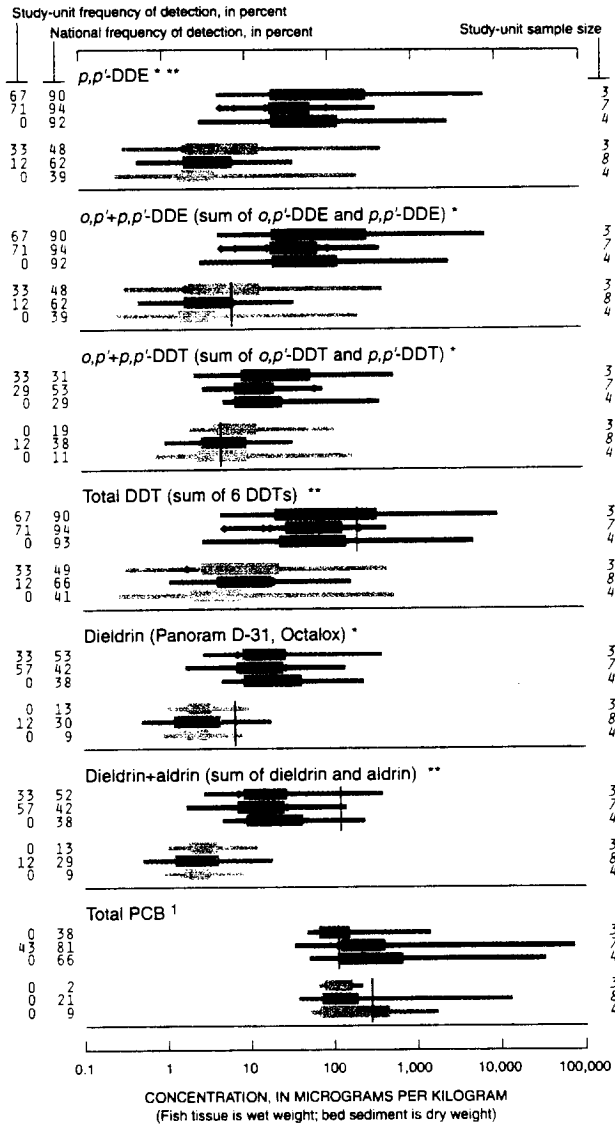
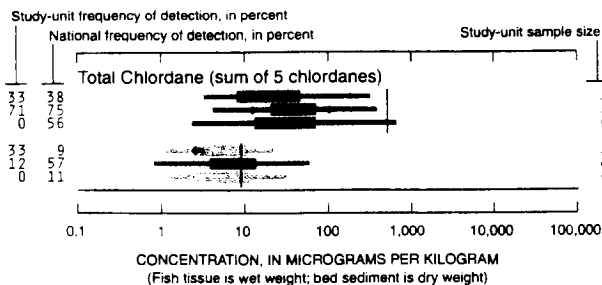


National benchmarks for fish tissue and bed sediment

National benchmarks include standards and guidelines related to criteria for protection of the health of fish-eating wildlife and aquatic organisms. Sources include the U.S. Environmental Protection Agency, other Federal and State agencies, and the Canadian Council of Ministers of the Environment

- | Protection of fish-eating wildlife (applies to fish tissue)
- | Protection of aquatic life (applies to bed sediment)
- No benchmark for protection of fish-eating wildlife
- No benchmark for protection of aquatic life

Organochlorines in fish tissue (whole body) and bed sediment



¹ The national detection frequencies for total PCB in sediment are biased low because about 30 percent of samples nationally had elevated detection levels compared to this Study Unit. See <http://water.usgs.gov/nawqa/> for additional information.

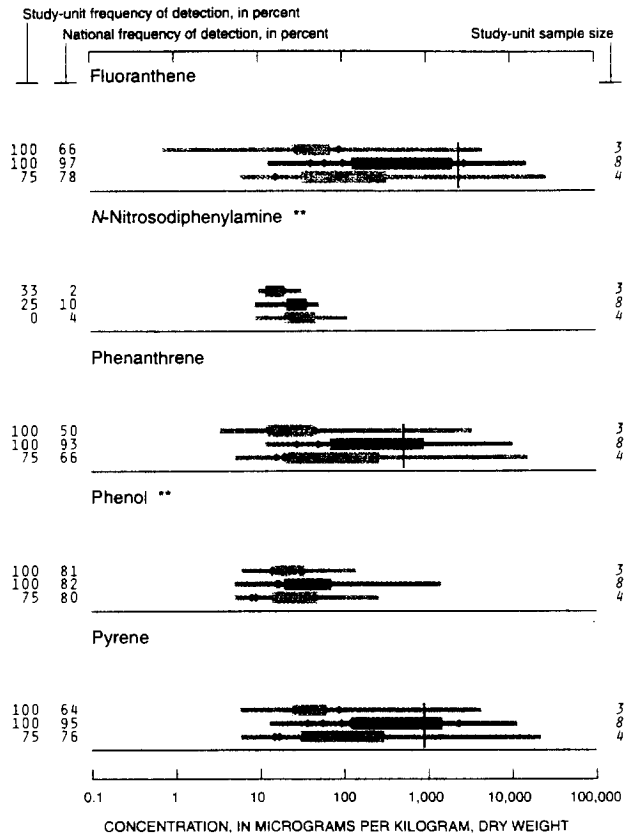
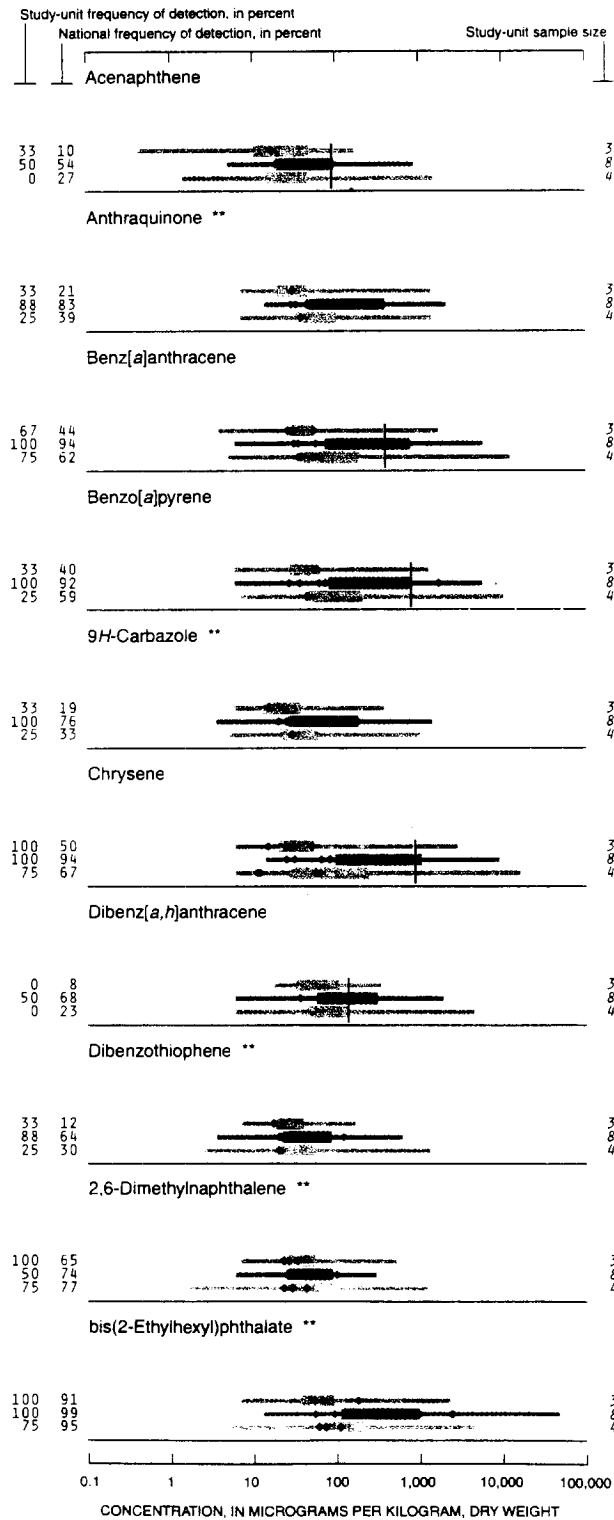
Other organochlorines detected

- o,p'+p,p'-DDD (sum of o,p'-DDD and p,p'-DDD) *
- Heptachlor epoxide (Heptachlor breakdown product) *
- Heptachlor+heptachlor epoxide (sum of heptachlor and heptachlor epoxide) **
- Hexachlorobenzene (HCB) **
- Pentachloroanisole (PCA) ***

Organochlorines not detected

- Chloroneb (Chloronebe, Demosan) ***
- DCPA (Daclhal, chlorthal-dimethyl) ***
- Endosulfan I (alpha-Endosulfan, Thiodan) ***
- Endrin (Endrine)
- gamma-HCH (Lindane, gamma-BHC, Gammexane) *
- Total-HCH (sum of alpha-HCH, beta-HCH, gamma-HCH, and delta-HCH) **
- Isodrin (Isodrine, Compound 711) ***
- p,p'-Methoxychlor (Marlate, methoxychlore) **
- o,p'-Methoxychlor ***
- Mirex (Dechlorane) **
- cis-Permethrin (Ambush, Astro, Pounce) **
- trans-Permethrin (Ambush, Astro, Pounce) **
- Toxaphene (Camphechlor, Hercules 3956) **

Semivolatile organic compounds (SVOCs) in bed sediment



Other SVOCs detected

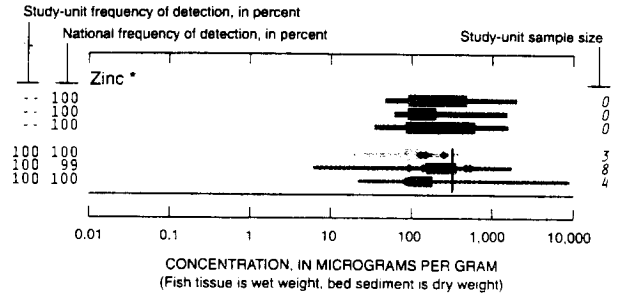
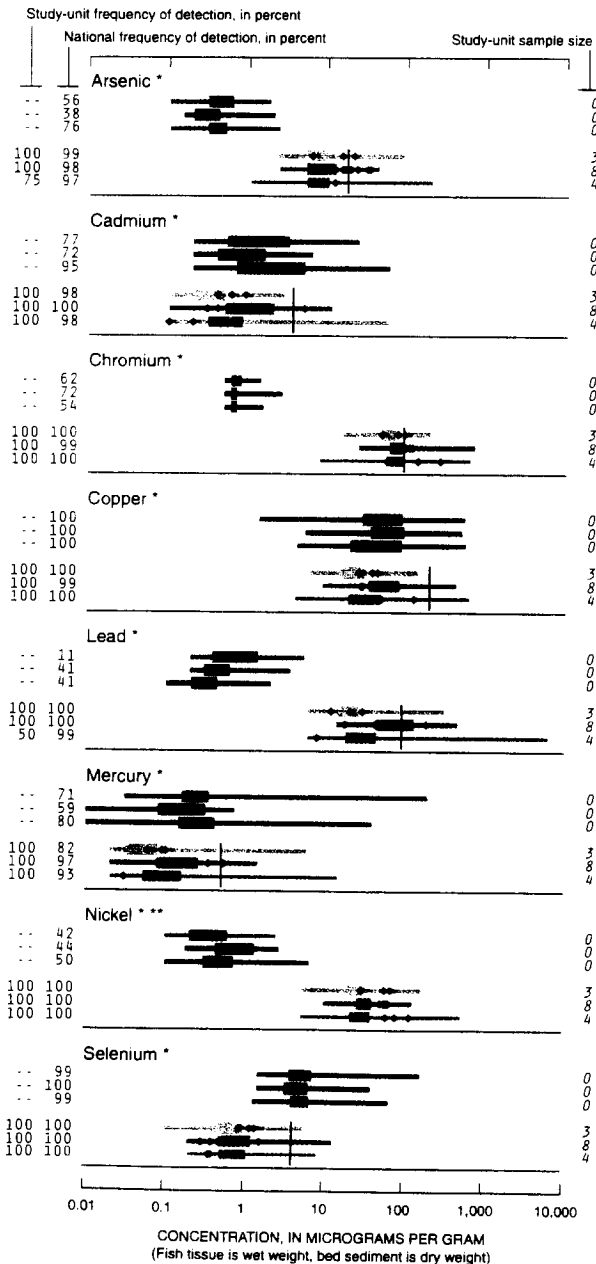
Acenaphthylene
Acridine **
Anthracene
Azobenzene **
Benzo[b]fluoranthene **
Benzo[ghi]perylene **
Benzo[k]fluoranthene **
2,2-Biquinoline **
Butylbenzylphthalate **
p-Cresol **
Di-*n*-butylphthalate **
Di-*n*-octylphthalate **
Diethylphthalate **
1,2-Dimethylnaphthalene **
1,6-Dimethylnaphthalene **
3,5-Dimethylphenol **
Dimethylphthalate **
9H-Fluorene (Fluorene)
Indeno[1,2,3-*cd*]pyrene **
Isoquinoline **
1-Methyl-9H-fluorene **
2-Methylantracene **
4,5-Methylenephenanthrene **
1-Methylphenanthrene **
1-Methylpyrene **
Naphthalene
Phenanthridine **
2,3,6-Trimethylnaphthalene **

SVOCs not detected

C8-Alkylphenol **
Benzo[c]cinnoline **
4-Bromophenyl-phenylether **
4-Chloro-3-methylphenol **
bis(2-Chloroethoxy)methane **

- 2-Chloronaphthalene **
- 2-Chlorophenol **
- 4-Chlorophenyl-phenylether **
- 1,2-Dichlorobenzene (*o*-Dichlorobenzene) **
- 1,3-Dichlorobenzene (*m*-Dichlorobenzene) **
- 1,4-Dichlorobenzene (*p*-Dichlorobenzene) **
- 2,4-Dinitrotoluene **
- Isophorone **
- Nitrobenzene **
- N*-Nitrosodi-*n*-propylamine **
- Pentachloronitrobenzene **
- Quinoline **
- 1,2,4-Trichlorobenzene **

Trace elements in fish tissue (livers) and bed sediment



BIOLOGICAL INDICATORS

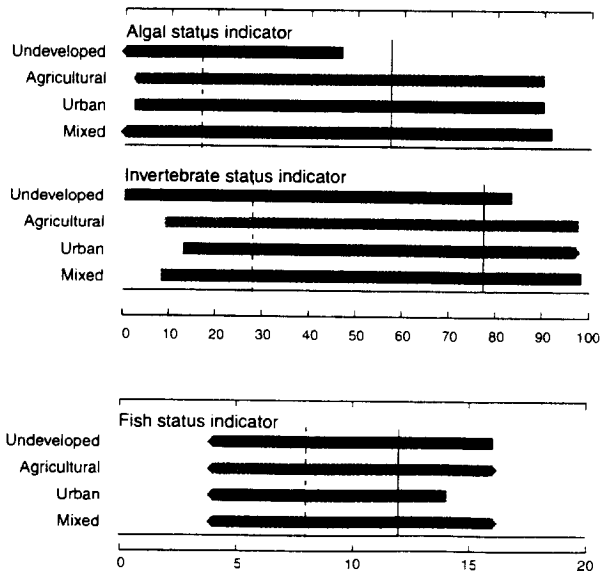
Higher national scores suggest habitat disturbance, water-quality degradation, or naturally harsh conditions. The status of algae, invertebrates (insects, worms, and clams), and fish provide a record of water-quality and stream conditions that water-chemistry indicators may not reveal. **Algal status** focuses on the changes in the percentage of certain algae in response to increasing siltation, and it often correlates with higher nutrient concentrations in some regions. **Invertebrate status** averages 11 metrics that summarize changes in richness, tolerance, trophic conditions, and dominance associated with water-quality degradation. **Fish status** sums the scores of four fish metrics (percent tolerant, omnivorous, non-native individuals, and percent individuals with external anomalies) that increase in association with water-quality degradation.

Biological indicator value, Puget Sound Basin, by land use, 1996-98

- ◆ Biological status assessed at a site

National ranges of biological indicators, in 16 NAWQA Study Units, 1994-98

- Streams in undeveloped areas
- Streams in agricultural areas
- Streams in urban areas
- Streams in mixed-land-use areas
- 75th percentile
- - - 25th percentile





United States Department of the Interior

U.S. GEOLOGICAL SURVEY

Water Resources Division
Washington District
1201 Pacific Avenue, Suite 600
Tacoma, Washington 98402
(253) 428-3600 * FAX (253) 428-3614
<http://wa.water.usgs.gov>

Dear Colleague:

Enclosed is a new report from the U.S. Geological Survey Puget Sound Basin Study Unit of the National Water Quality Assessment (NAWQA) Program.

USGS Circular 1216, "Water Quality in the Puget Sound Basin, Washington and British Columbia, 1996-98," summarizes surface-water quality, ground-water quality, biology, and ecological conditions from data collected during three years of intensive study on selected streams and ground water in the Puget Sound Basin. The report also compares water-quality conditions and biological characteristics observed in Puget Sound Basin streams and ground water with conditions observed in other NAWQA study units across the Nation.

The Puget Sound Basin NAWQA reports are available online, or can be ordered, at:
<http://wa.water.usgs.gov/ps.nawqa.html>

Thank you for your interest,

Sandra S. Embrey
Chief Puget Sound Basin NAWQA

R0024750

Coordination with agencies and organizations in the Puget Sound Basin was integral to the success of this water-quality assessment. We thank those who served as members of our liaison committee.

Federal Agencies

Bureau of Indian Affairs
National Marine Fisheries Service
National Park Service
Northwest Indian Fisheries Commission
U.S. Army Corps of Engineers
U.S. Environmental Protection Agency
U.S. Fish and Wildlife Service
U.S. Forest Service

Canadian Agencies

British Columbia Ministry of
Environment
Environment Canada

State Agencies

Puget Sound Water Quality Action
Team
Washington State Department of
Agriculture
Washington State Department of
Ecology
Washington State Department of
Fish and Wildlife

Washington State Department of
Health
Washington State Department of
Natural Resources
Washington State Department of
Transportation

Local Agencies

City of Auburn
City of Bremerton
City of Lacey
City of Olympia
City of Renton
City of Seattle
City of Tacoma
King County
Kitsap County
Pierce County
Seattle Public Utilities
Snohomish County
Tacoma-Pierce County Health
Department
Thurston County
Whatcom County

Native American Tribes and Nations

Lummi Tribe
Muckleshoot Tribe
Nooksack Tribe
Puyallup Tribe
Skokomish Tribe

Universities

University of Washington
Washington State University

Other Public and Private Organizations

Adopt a Stream Foundation
People for Puget Sound
Thornton Creek Project
Washington Environmental Council
Washington Toxics Coalition

We thank the following individuals for contributing to this effort.

- Gil Bortleson (USGS, Washington District) participated in the design of the study and authored several reports.
- Dale Davis (Washington State Department of Ecology) was a partner and coauthor in the study of pesticides in streams.
- Sue Davis (Thurston County), Mindy Fohn (City of Bremerton), Keith Grellner (Kitsap County), Ray Hanowell (Pierce County), Joe Joy (Washington State Department of Ecology), and Ginny Stern (Washington State Department of Health) helped in the design of the microbiological studies.
- Annette Frahm and George Perry (King County) provided sales data for pesticides, conducted toxicity tests, and coauthored a fact sheet about pesticides in urban streams.
- Kathleen Greene, Joseph Gilbert, Gary Holloway, Brett Smith, and Greg Justin (USGS, Washington District) played a major role in the collection and compilation of water-quality and streamflow data.
- Emily Inkpen (USGS, Washington District) collected and compiled water-quality data and authored a number of reports.
- Dr. James Karr (University of Washington) provided advice on the analysis of macroinvertebrate data.
- Moon Kim, Raegan Huffman, Stephanie Leisle, Mark Munn, Steve Sumioka, and Ann Vanderpool (USGS, Washington District); Steve Goodbred and Terry Short (USGS, Western Region); Mariabeth Silkey and Michael Hummling (University of Washington); and Dave Serdar and Art Johnson (Washington State Department of Ecology) provided field support and assisted in data collection.
- Hugh Liebscher and Basil Hii (Environment Canada) provided water level and chemistry data for wells in Canada.
- Dorene MacCoy (USGS, Idaho District) helped collect and interpret biological data and helped produce a number of reports.
- Rob Plotnikoff (Washington State Department of Ecology) provided macroinvertebrate and habitat data.
- Dr. Mark Sobsey, Fu-Chih Hsu, and Douglas Wait (University of North Carolina, Chapel Hill) provided technical assistance and laboratory analyses for the microbiological study.
- Ward Staubitz (USGS, Virginia District), project chief until April 1997, played a major role in the design of the study.
- Ian Waite and Dennis Wentz (USGS, Oregon District) provided fish tissue data from the Willamette Basin, Oregon.
- Frank Voss (USGS, Washington District) managed the project data base and produced a number of reports.

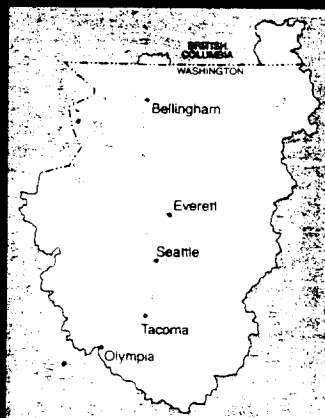
Appreciation is extended to those individuals who reviewed this report or helped prepare it.

Joe Joy, Washington State Department of Ecology
Cindy Moore, Washington State Department of Agriculture
Richard Ramsey, Staff to the Washington State Environmental
Quality and Water Resources Committee

Dennis Wentz, USGS, Oregon District
Ginger Renslow, Editor, USGS, Washington District
Robert Crist, Illustrator, USGS, Washington District

NAWQA

National Water-Quality Assessment (NAWQA) Program Puget Sound Basin



Ehbert and others—Water Quality in the Puget Sound Basin
U.S. Geological Survey Circular 1216

ISBN 0-607-95420-5



9 780607 954203

R0024751