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Nutrients in Estuaries

A SUMMARY REPORT OF THE *NATIONAL ESTUARINE EXPERTS WORKGROUP* *2005–2007*

Cover photos. top: Chesapeake Bay (www.thinkstockphotos.com); bottom: Coastal Bays (photo by J. Thomas, www.ian.umces.edu)

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Residential development, Maryland Coastal Bays. Photo by J. Thomas (www.ian.umces.edu).

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Preface

The Maryland portion of the Coastal Bays. Photo by J. Thomas (www.ian.umces.edu).

This document is a scientific synthesis of the understanding of the effects of nitrogen and phosphorus pollution on estuaries, the underlying causes and co-factors, ecological consequences and scientific pathways to help scientists develop numeric nutrient criteria. It characterizes the biogeochemical complexities inherent in estuaries and, in turn, the complexities associated with developing numeric nutrient criteria protective of designated uses. The document emphasizes methods, models and data that facilitate in either characterizing or constraining complexity, and it identifies characteristics of data useful for criteria development. The document also includes case-studies of estuaries across the United States that illustrate the main principles identified.

This synthesis is the product of the National Estuarine Experts Workgroup. The workgroup effort had two phases: deliberation and compilation. The first phase, from 2005 through 2007, involved multiple modes of communication of the entire workgroup, or subgroups within it. The second phase involved synthesis, writing and administrative and peer review of the results of the deliberation phase.

The synthesis, writing and review of the results of the first phase proved to be a long process. The members of the workgroup recognize that considerable recent efforts have been made in nutrient criteria development, which are not recognized in this document. However, despite some development in the scientific body of knowledge since the deliberation phase, the basic foundational synthesis is still applicable and relevant today, and this document provides scientifically defensible material foundational to the understanding of the ecological response of estuaries to nutrient inputs.

This document complements and does not supersede previously published EPA guidance such as the *Nutrient Criteria Technical Guidance Manual: Estuarine and Coastal Marine Waters* (EPA-822-B-01-003). The practitioner interested in developing numeric nutrient criteria for an estuary or class of estuaries should consult these and other documents.

Nutrients in Estuaries

Executive Summary

The Clean Water Act (CWA) directs states to adopt water quality standards for their navigable waters, which include estuaries. Water quality criteria, a component of water quality standards, are set to protect designated uses and must be based on sound scientific rationale. For nutrients, EPA has published, under the CWA, a series of peer-reviewed, national technical approaches and methods regarding the development of numeric nutrient criteria for lakes and reservoirs, rivers and streams, and estuaries and coastal marine waters. Nutrient criteria are benchmarks that help to establish the level of nutrient pollution below which waterbodies can maintain their designated uses—primarily aquatic life and recreation. Estuaries are very complicated bodies of water, and the task of setting nutrient criteria for them, in turn, is complex.

This document describes some of the current science in estuarine nutrient criteria development. It does not set criteria, nor does it recommend a single methodology for doing so. Criteria can vary from site to site and from estuary type to estuary type, and the methods for setting criteria can vary depending on both the estuary type and availability of data. This document is intended to describe the complexities of nutrients in estuaries and how they vary with estuarine type. Some examples are provided for approaches that can be considered in developing estuarine nutrient criteria. A suite of case studies is also provided to demonstrate that, while patterns emerge in nutrient dynamics based on estuarine characteristics, multiple individual factors contribute to the uniqueness of each estuary.

Estuaries across the country are experiencing threats from nutrient over-enrichment. Nutrient criteria are being developed to help control and reduce nutrient enrichment and their associated effects. Top photo by P. and M. Peck (www.strengthinperspective.com); bottom photo by J. Thomas (www.ian.umces.edu).

Estuarine Nutrient Complexity

Estuarine nutrient complexity arises from the influence of multiple chemical, physical and biological factors interacting in the delivery of nutrients and their transformations within these semi-enclosed waterbodies. There are multiple sources of nutrients to estuaries, from land-based point and nonpoint sources, to atmospheric and groundwater inputs. Each source can vary in the amount of specific nutrients (nitrogen or phosphorus) they contribute and their proportional ratio to other nutrients in that source. Different sources can also vary in the chemical form of these nutrients, inorganic or organic, or in the case of nitrogen, oxidized (NO_3^- or NO_2^-) or reduced (NH_4^+). Each of these different forms can affect the ecosystem differently. Estuarine nutrients are also highly dependent on the physical attributes of the estuary, as residence time determines the amount of time nutrients stay in the system and are available for biological processes. Stratification also plays a role in biological processing, as does turbidity, affecting the light availability for algal and seagrass growth. Climate variability and change also interact with these factors. Nutrient supply is tightly coupled with freshwater input that, in turn, is driven by regional climate variability. Nutrients delivered with freshwater input determine, to a large extent, the spring chlorophyll *a* maximum in many estuaries. The health of estuarine fisheries, particularly those containing filter-feeding fish and shellfish, also plays an important role in determining the nutrient status of an estuary. Benthic filter feeders effectively reduce algal biomass through filter feeding and also affect nutrient recycling processes which, in turn, alter nutrient supply. Moreover, ecosystem response to eutrophication is a continual process rather than a static one, and thus different systems fall on different points along the estuarine continuum, making their response to nutrients variable.

Estuarine Typology

Estuaries can respond to similar nutrient loads in very different ways, but some of those responses can be categorized by estuarine type. An estuarine typology is an organizing framework that groups estuaries on the basis of descriptive and quantitative characterizations. Such an organizing framework can assist in classifying estuaries into a few broad categories with a characteristic dose-response relationship developed for each class, from which similar dose-response relationships can be developed. The typology can be based on physical (e.g., tides), hydrological (e.g., discharge), geomorphological, hydromorphological (e.g., stratified water column) and other properties. A conceptual classification scheme herein recognizes the following four classes of estuaries: riverine, coastal lagoon, coastal embayment and fjord. For each estuarine class, relationships among variables associated with nutrients (e.g., loading rates, chemical form) and the estuarine response variables can be established. Estuarine classification can be especially useful in setting preliminary criteria for estuaries with little or no observational data on past response to nutrient enrichment.

A Framework for Nutrient Criteria Development

The challenge in developing nutrient criteria is to describe, and ultimately set, quantitative values for water quality parameters below which attainable conditions of biotic integrity or a suite of designated uses for that waterbody can be maintained. A common way to establish such quantitative values is to examine the relationships between variables that are thought to be representative of nutrient loading (causal variables) and those that are representative of a biological response (response variables). Causal variables can provide some information regarding the possibility for nutrient impairment, yet they are highly dynamic and rapidly transformed and modified by biological and biogeochemical processes. Causal variables can include nutrient concentrations, nutrient loads, or a proxy for nutrient loads, such as land use. Hydrology, estuarine typology and climate, among other natural factors, can be considered supporting variables and can serve to scale the response to the causal variables. Response variables, in turn, represent a measure of a biotic response. Response variables can consider a single measure, such as the amount of chlorophyll *a*, but when data availability permit, more integrated assessments of the biological community can provide an improved understanding of the responses to nutrients. Such integrated assessment, or biocriteria, can include species, populations or communities of organisms that integrate the aquatic condition and provide information on ecosystem condition, such as algal species composition or submerged aquatic vegetation. Benthic organisms and/or fish can be included in such indices because they represent different responses to stressors. Integrated measures of community response can also include chemical indices, such as dissolved oxygen levels. Integrated response variables, which average a numeric response variable for individual response variables, have shown widespread application, including in the recent U.S. National Estuarine Eutrophication Assessment.

Relationships between causal and response variables vary depending on the temporal and spatial scales being considered. Improved relationships emerge when data are averaged on seasonal, annual or other bases. Furthermore, some estuaries show characteristics of one estuarine type in one segment but another estuarine type in another segment; defining estuarine zones might be appropriate to identify criteria specific to different zones. In effect, that approach extends the concept of estuarine typology.

Once criteria are established, a method to document exceedances is recommended. Various approaches are available, ranging from a fixed percentage to differential response by season. How to establish exceedances and what might be deemed allowable, again, varies by the parameter and the designated use.

Grand Bay, Mississippi. Photo by P. Hoar (NOAA public image library collection).

Case Studies

The case studies described herein include 10 examples representing either river-dominated or lagoonal systems: Narragansett Bay, Delaware Bay, Chesapeake Bay, Neuse River Estuary, San Francisco Bay Estuary, Yaquina Estuary, Barnegat Bay-Little Egg Harbor, Coastal Bays, Florida Bay and Pensacola Bay. The case studies provide examples of systems that vary in typology and in size, depth and watershed area. They also vary considerably in the population size and land use of the watershed, from highly urbanized areas to those that are more agricultural; thus, they vary in their degree of eutrophication. The case studies, furthermore, provide examples of some systems that had major changes in their nutrient status. Several have been affected by hurricanes over the past decade, while others have had improvements in sewage treatment. These examples demonstrate that

- There is order in the complexity of estuarine system responses to nutrients driven by the geomorphology and physical dynamics of the systems.
- Underlying common biogeochemical mechanisms explain some of the patterns in responses, although the dominant biological, chemical or geochemical processes vary in different systems.

In practice, establishing estuarine criteria for each estuary depends on the availability of current and historical data, the capability for monitoring and the types of parameters that can be assessed. As additional information is gathered for each system, particularly about the rates of processing of nutrients in systems of different typology, and as the monitoring data record becomes larger, criteria can be developed with increased certainty and sophistication.

Chapter 1:

Introduction

David Flemer and Patricia M. Glibert

Figure 1.1. Symptoms of eutrophication often include dense algal blooms and hypoxia, which, in turn, can have devastating effects on the biota. Photos by P. Glibert (left and bottom) and A. Jones (upper-right).

Introduction to Estuarine Nutrient Criteria

Nutrient over-enrichment is recognized as a serious threat to estuarine and coastal waters throughout most of the United States, and indeed throughout much of the world (Nixon 1995; Bricker et al. 1999, 2007; Cloern 2001; Rabalais and Turner 2001; Howarth et al. 2002; Wassmann 2005). Eutrophication, the process of increased organic enrichment of an ecosystem through increased nutrient inputs, is perhaps the oldest water quality problem created by humankind (Vollenweider 1992; Nixon 1995). Rapid population growth in recent decades, especially in the coastal zone, has increased the demand for energy, food, fiber and housing. Coincident with this consumption is the increased demand for fertilizer use, increased output of sewage and other waste products, and increased use of freshwater (Smil 2001; Glibert et al. 2005, 2006). In addition, landscape changes, invasive species, alteration to hydrology, overfishing and climate changes have all further altered coastal and estuarine ecosystems. Effects such as harmful algal blooms (HABs), fish kills, marine mortality events, loss of seagrass and bottom habitat, coral reef destruction and development of hypoxia and anoxia are now recognized to be common in response to the over-enrichment of nutrients and are considered characteristic of eutrophic environments (Cloern 2001).

> ***Eutrophication* is the process of increased organic enrichment of an ecosystem, generally through increased nutrient inputs.** Nixon 1995

The development of nutrient criteria for estuaries will aid states' ability to control and reduce nutrient enrichment and its associated effects. Criteria form the scientific basis against which ecological consequences can be measured and are a benchmark for management decision making. Effective nutrient criteria can and should be developed within the framework of existing data and knowledge of responses by ecosystems to nutrients. Estuarine responses to nutrients differ from those of freshwater and wetland ecosystems, and also vary among estuarine types. EPA's National Nutrient Criteria

Program recognizes a process that involves data collection for individual areas, developing criteria for a range of water quality parameters, assessing additional waterbodies of similar characteristics against the established criteria, designing and implementing appropriate management action and evaluating its relative success (USEPA 2001). This document is intended to be an informational resource to promote a better understanding of the complex processes that drive ecosystem responses to nutrients to guide the development of criteria for estuaries.

Water quality standards and criteria were first mandated under the Clean Water Act (CWA) of 1977. That act, which itself was an amendment of the Federal Water Pollution Control Act Amendments of 1972, established the basic structure for regulating discharges of pollutants into the waters of the United States. It gave EPA the authority to implement pollution control programs such as setting wastewater standards for industry. Over the years, many other laws have amended parts of the CWA. Title I of the Great Lakes Critical Programs Act of 1990, for example, required EPA to help the states implement nutrient criteria on a specific schedule.

Beginning in the early 1990s, EPA concluded that eutrophication was a national problem requiring a national strategy. In June 1998, EPA published a policy document titled *National Strategy for the Development of Regional Nutrient Criteria* (National Strategy) (USEPA 1998). That policy document lays out a proposed schedule for deploying technical guidance and case studies, starting with lakes and reservoirs and progressing to rivers and streams, wetlands and then estuaries and coastal waters. Such a progression reflects the recognized increasing complexity of responses to nutrients in such types of systems.

A critically important aspect of the National Strategy document is the recommendation for EPA to develop scientific information on pollutants and to publish ***criteria guidance***. Criteria guidance is often expressed as pollutant concentration levels below which will result in attainment of a ***designated use*** of the waterbody (e.g., aquatic life, recreation; Figure 1.2) .

Criteria may be in narrative or numeric form. Narrative criteria are more general statements, rather than numeric values, about attainable conditions of biotic integrity or

History of federal water legislation before the Clean Water Act

Early Federal Water Pollution Control Acts led up to the 1972 Federal Water Pollution Control Act Amendments (collectively known as the Clean Water Act);
Source: www.epa.gov/lawsregs/laws/cwahistory.html

1899	Refuse Act (River and Harbors Act)	Protected navigable waterways from pollution
1948	Water Pollution Control Act	Required that technical assistance be given to States
1956	Federal Water Pollution Control Act	Added provisions for research, training, collection of basic data and grants for construction of treatment works
1965	Water Quality Act	Mandated water quality standards
1966	Clean Water Restoration Act	Expanded focus to include interstate water quality issues

water quality for a given designated use. An example of narrative criteria is highlighted in the box below for Chesapeake Bay. In addition to nutrient criteria, another metric is a ***total maximum daily load*** (TMDL). A TMDL is the amount of pollutant that can enter a waterbody while still maintaining water quality standards.

Figure 1.2. Estuarine water quality should support all designated uses. Photo by J. Shannahan.

In the case of nutrients it is understood that there is a great deal of variability in inherent nutrient levels and the biotic responses to nutrients. This natural variability is due to differences in geology, climate and waterbody type. Because of that variation, EPA has accepted that various types of waterbodies need to be evaluated differently and that recommended nutrient concentration levels need to reflect such a variation. Thus, nutrient criteria are not typically transferable from lakes to estuaries, nor from one type of estuary to another. In factors that affect system susceptibility to eutrophication, lakes and estuaries differ in water residence time (e.g., flushing), mixing, water chemistry, turbidity, geomorphology, physical energy (e.g., coastal storms and tidal action) and relative importance of nitrogen (N) versus phosphorus (P) as limiting nutrients. Gradients in salinity and sediment types and associated biological species and community distribution and abundance patterns also differ greatly between estuaries and lakes. EPA previously published the *Nutrient Criteria Technical Manual: Estuarine and Coastal Waters* (USEPA 2001), which introduces a sequence of actions leading to the development of recommended nutrient criteria for estuaries and near-coastal waters. In that document, EPA concludes that, “no current classification approach provided all the information that a site may need to make decisions“ (USEPA 2001, p. 3-11), but that “...physical classification of estuaries and coastal waters...can provide improved understanding of the processes that contribute to ecosystem susceptibility and variability in the expression of nutrient effects” (USEPA 2001, p. 7-3). This document is intended as a supplemental resource to that 2001 manual in classifying estuaries and in guiding estuarine criteria development. This document begins with a review of the national and international interest in eutrophication.

Recommended Chesapeake Bay narrative chlorophyll a criteria (from USEPA 2003, p. 149)

Concentrations of chlorophyll *a* in free-floating microscopic plants (algae) [must] not exceed levels that result in ecologically undesirable consequences—such as reduced water clarity, low dissolved oxygen, food supply imbalances, proliferations of species deemed potentially harmful to aquatic life or humans or aesthetically objectionable conditions—or otherwise render tidal waters unsuitable for designated uses.

National and Global Interest in the Problem of Eutrophication

The issue of coastal and estuarine eutrophication is of concern nationally and worldwide. In addition to the efforts under the CWA described earlier, government agencies, commissions and other groups are tackling the issue. The synopses of their efforts described below are not meant to be all-inclusive, but rather to be illustrative of the other efforts and resources that are available on the issue. Such a community of effort underscores the importance of this topic.

Commission Reports

The U.S. Commission on Ocean Policy

The Ocean Act of 2000 represents a significant advancement in understanding the threats to the oceans; it authorized the U.S. Commission on Ocean Policy.

The commission collected expert and public testimony that major changes are needed in the management of the oceans. Of note was the recognition that the issues facing oceans and coastal management are complex and involve interconnections between natural and human systems.

The U.S. Commission on Ocean Policy's report (Figure 1.3, top), delivered to the President and Congress in September 2004, calls for a new governance framework, more investments in marine science, and an ecosystem-based management effort to halt the decline of the quality and resources in the nation's coasts and oceans.

The Pew Oceans Commission

Similar to the U.S. Commission on Ocean Policy report, the Pew Oceans Commission report was released in 2004 (Figure 1.3, bottom) with similar findings. It also outlined the threats to the oceans' living resources and called for more unified management approaches. The report states, "We have reached a crossroads where the cumulative effect of what we take from, and put into, the ocean substantially reduces the ability of marine ecosystems to produce the economic and ecological goods and services that we desire and need" (Pew Oceans Commission, 2004, Executive Summary). It notes that without action and reform, every aspect of the oceanic natural resources will be in jeopardy.

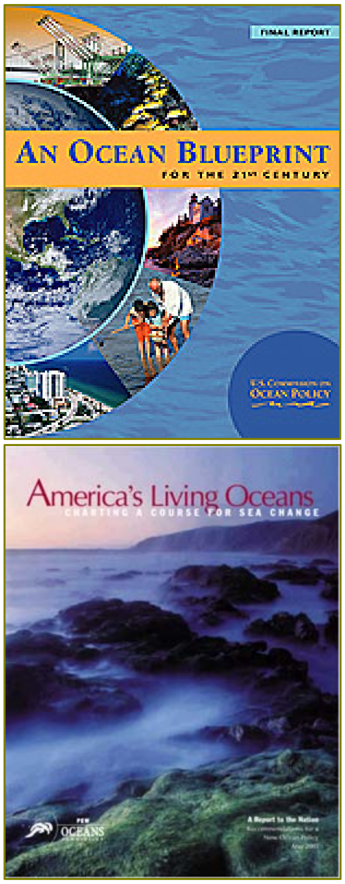


Figure 1.3. The U.S. Commission on Ocean Policy report and the Pew Oceans Commission report are at http://oceancommission.gov/documents/full_color_rpt/000_ocean_ full_report.pdf and www.pewtrusts.org/uploadedFiles/wwwpewtrustsorg/Reports/Protecting_ocean_life/env_pew_oceans_final_report.pdf.

Millennium Ecosystem Assessment

The Millennium Ecosystem Assessment (2005) was prepared at the request of United Nations Secretary, with a goal of determining how ecosystems and their services have changed, the causes of those changes, and the effects of those changes on human well-being.

In the report, the authors recognize that, "Over the past four decades, excessive nutrient loading has emerged as one of the most important direct drivers of ecosystem change in terrestrial, freshwater, and marine ecosystems" (Millennium Ecosystem Assessment 2005, p. 69). Moreover, the report also states that the harmful effects of nutrients are projected to continue to increase and that there is a great need to increase our understanding of critical thresholds of nutrients and other pollutants that alter ecosystems. It also addresses how systems change once those thresholds have been exceeded, or how they can be returned to their prior state.

The U.S. Commission on Ocean Policy report states, "ocean policy should be grounded in an understanding of ecosystems, and our management approach should be able to account for and address the complex interrelationships among the ocean, land, air, and all living creatures, including humans, and consider the interactions among multiple activities that affect entire systems. An ecosystem-based management approach should overcome the challenges inherent in addressing complex issues that cross traditional jurisdictional boundaries, and it must be able to continually adapt to new scientific information and improved management tools" *(U.S. Commission on Ocean Policy 2004, Executive Summary).*

Federal Congressional Acts, Programs and Plans

The Harmful Algal Bloom and Hypoxia Research and Control Act

The Harmful Algal Bloom and Hypoxia Research and Control Act (HABHRCA), originally enacted in 1998, and reauthorized for 2004–2006, recognizes that many of our nation's coastal areas suffer from HABs and hypoxia and that the effects on our waters threaten coastal ecosystems and potentially endanger human health. The legislation calls for stronger interaction between federal and local resource managers in developing plans for assessing and monitoring HABs and hypoxia. Among the legislation's requirements are calls to prepare a number of national assessments, including

- The national status for prediction and response of such events.
- A plan for national scientific research, development, demonstration and technology transfer.
- A scientific assessment of the problems of freshwater HABs.
- A scientific assessment of marine HABs.
- A scientific assessment of the extent of hypoxia in national waters.

The HABHRCA authorizes several research programs that are concerned with HABs and hypoxia. The ***Ecology and Oceanography of Harmful Algal Blooms*** (ECOHAB) program was initiated as a multiagency partnership more than a decade ago as a "scientific program designed to increase our understanding of the fundamental processes underlying the impacts and population dynamics of HABs" (Anderson 1995, p. 3). Similarly, the ***Monitoring and Event Response for Harmful Algal Blooms*** (MERHAB) program, administered through the National Oceanic and Atmospheric Administration (NOAA), seeks to build the capabilities of local, state, tribal and private-sector interests for regular and intensive measurements of HAB parameters, making existing monitoring efforts more efficient while providing for improved coverage in time and space. HABHRCA also authorized the funding of the ***Coastal Hypoxia and Nutrient Pollution*** program, which is aimed at improving the understanding between nutrient over-enrichment, eutrophication and the effects on human health ecosystem health. The ***Hypoxia*** program has focused considerable efforts on understanding the *Dead Zone* of the Gulf of Mexico, including developing models to predict the effects of variable nutrient loads on the extent of the hypoxic zone. Understanding how ecosystem functions are altered with hypoxia, including fisheries impacts, are also under the purview of this program.

The priorities of the ECOHAB, MERHAB and other related programs are also guided by several synthesis and planning documents in addition to the commission reports highlighted above. The following documents exemplify syntheses that were developed by the scientific community in coordination with federal agencies.

Priority Topics for Nutrient Pollution in Coastal Waters: An Integrated National Research Program for the United States

The report, *Nutrient Pollution in Coastal Waters: An Integrated National Research Program for the United States* (Howarth et al. 2003), developed as a consensus of the scientific community, lays out a plan to address nutrient pollution in the United States, recognizing the needs for improved coastal monitoring, periodic and comprehensive assessments of coastal ecosystems and strengthened research programs. The report also states that although general tools for assessing the impacts of nutrient pollution are largely lacking, both physical and biological attributes of a coastal system might either enhance or serve to dampen the response to nutrient pollution. The report identifies 10 research questions of high priority, including 5 that relate to critical uncertainties of nutrient effects within estuarine systems and 5 that relate to critical uncertainties with respect to nutrient delivery to estuaries.

Harmful Algal Research and Response: A National Environmental Science Strategy (HARRNESS) 2005–2015

HARRNESS is a national plan of the research and management needs of the community for actions needed over the next decade to address the increasing proliferations of HABs in our nation's waters. Increasing nutrient enrichment is one of the causes of these events (e.g., Anderson et al. 2002; Glibert et al. 2005).

Those working at HARRNESS identified several priorities for research and, through a national committee, designed a framework to facilitate coordination among research, management and federal agency responses (Figure 1.4). The research priorities for HARRNESS include bloom ecology and dynamics, toxins and their effects, food webs and fisheries, and public health and socioeconomic impacts. Through new initiatives, communication, understanding, mitigation and control of HAB events can be improved.

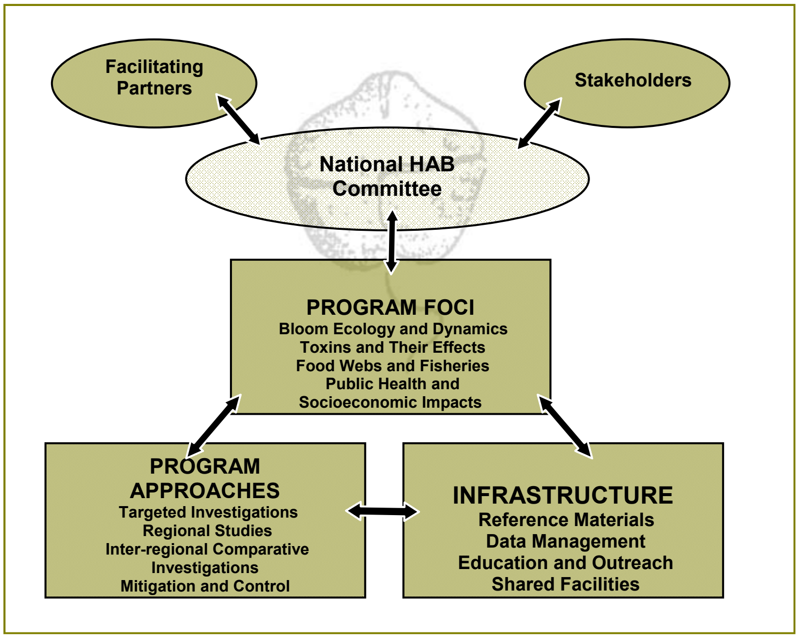


Figure 1.4. Information about the Harmful Algal Research and Response National Environmental Science Strategy and the activities of the National HAB Committee is at http://www.whoi.edu/redtide/.

Regional Plans

Many regions of the country have developed regional plans to reduce nutrients and to improve water quality. ***The Chesapeake Bay Agreement***, first signed in 1983 and reaffirmed in 1987 and again in 2000 by Maryland, Pennsylvania, Virginia, the District of Columbia, the Chesapeake Bay Commission and EPA, sets specific goals for improvements in water quality, living resource and vital habitat protection and restoration, among other goals.

The Gulf Hypoxia Action Plan 2008, an update of a similar plan in 2001, continues to set long term goals to accelerate the reduction of N and P fueling the hypoxic zone of the Gulf of Mexico. It similarly sets goals to advance the science and to raise awareness of the issue and its socioeconomic impacts. The plan embraces an adaptive management approach, recognizing that new science can yield new methods and approaches for nutrient reduction.

Related International Programs

Multiple international research programs are also concerned with nutrient pollution and its effects. Several of those are briefly reviewed below.

Global Ecology and Oceanography of Harmful Algal Blooms (GEOHAB)

GEOHAB is an international, multidisciplinary program that assists investigators from different disciplines and countries to exchange technologies, concepts and findings to address issues related to the global ecology and oceanography of HABs. One of the emphases of GEOHAB is understanding the extent to which eutrophication might be related to the increased proliferation of HABs.

Worldwide, strong relationships have been observed between increases in nutrient loading and proliferations of specific types of HABs. In many locales, HABs have increased in response to alterations in the type of nutrient, not only major nutrient forms such as N and P, but changes in the chemical form of these nutrients. Organic, not just inorganic, nutrient loading is increasing worldwide and has been correlated with many blooms of both dinoflagellates and cyanobacteria. Advancements in our understanding of the physiology of those organisms has yielded important insights as to why the algal classes respond so favorably to the nutrients.

Although there are good quantitative estimates of many sources and forms of nutrient loads, the transformation processes of these nutrients and how they are affected by landscape changes, food web alterations and climatic variations are not well understood.

The GEOHAB Core Research Project on HABs and Eutrophication (Figure 1.5) has identified the following key questions as priority areas for additional study:

- Are there clusters or specific types of HAB species that are indicative of global nutrient increases?
- To what extent do residence time and other physical processes affect the relationship between nutrient loading and HAB proliferation?
- How do feedbacks and interactions between nutrients and the planktonic, microbial food webs affect HABs and their detrimental effects?
- Do anthropogenic alterations of the food webs, including overfishing and aquaculture activities, synergistically interact with nutrients to favor HABs?
- How do anthropogenic changes in land use, agricultural use of fertilizer, NOx emissions from vehicles and global changes in land cover affect the delivery of nutrients to coastal waters and the resulting incidences of HABs?
- How do the stoichiometry and quality of those nutrient sources regulate the biological responses favoring HABs?
- Do climate change and climate variability have effects on ecosystems that augment the effects of eutrophication in the formation of HABs?

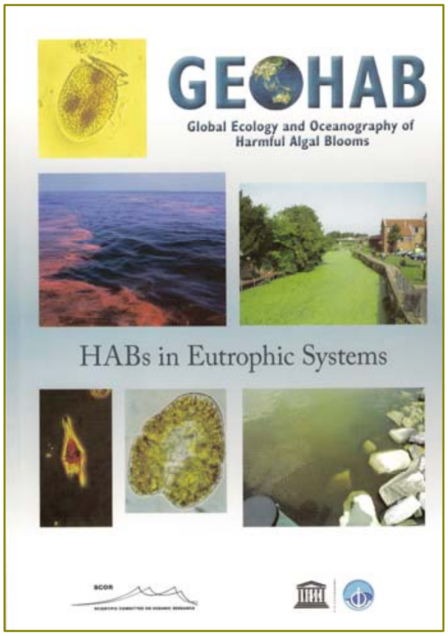


Figure 1.5. Information on the GEOHAB program is at www.geohab.info.

The International Nitrogen Initiative (INI)

The INI, under the auspices of the Scientific Committee on Problems of the Environment (SCOPE) and the International Geosphere-Biosphere Programme (IGBP), is a global effort to optimize nitrogen's beneficial role in sustainable food production and to minimize the negative effects of nitrogen on human health and the environment (Figure 1.6). As part of its objectives, knowledge of the flows of nitrogen and the related problems in several targeted regions of the globe are being developed. They

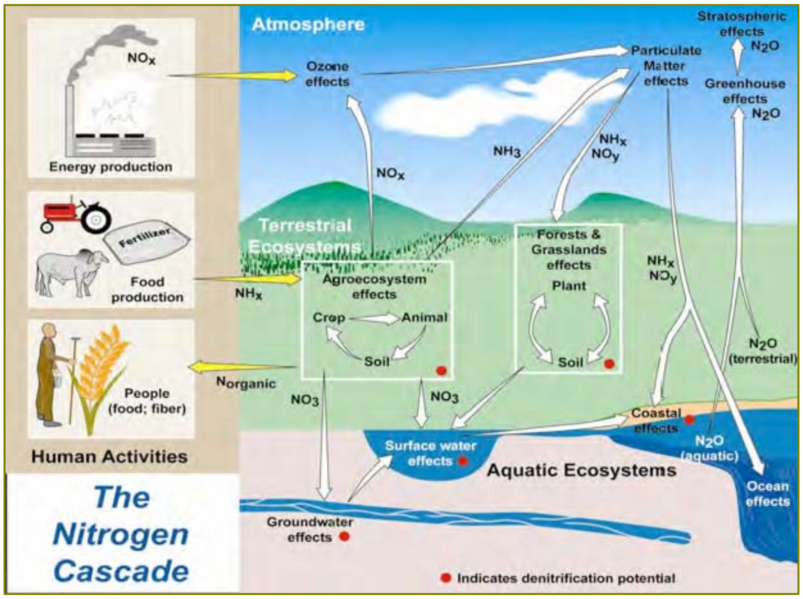
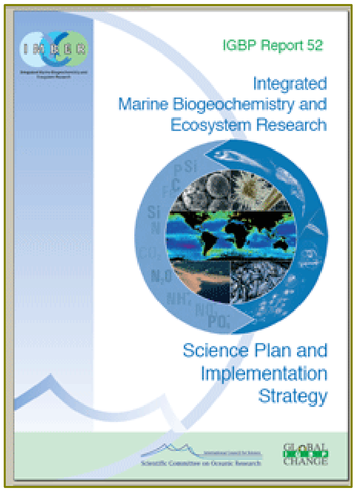


Figure 1.6. Information, including publications, about the International Nitrogen Initiative is at www.initrogen.org.


IGBP Report 52

Integrated
Marine Biogeochemistry and
Ecosystem Research

Science Plan and
Implementation
Strategy

Figure 1.7. Information about the IMBER Project, including the Science Plan and Implementation Strategy, is at www.imber.info.

aim to develop a better scientific understanding of N-related issues, from which a foundation for policy related to N will develop.

The Integrated Marine Biogeochemistry and Ecosystem Research Program (IMBER)

The IMBER program (IMBER; Figure 1.7) is being developed with a goal of providing a comprehensive understanding of ocean biological and chemical responses to accelerating global change and the consequent effects on the earth systems and human society. The goals of IMBER are to identify the key interactions between marine biogeochemical cycles and ecosystems, and to provide assessments of how those interactions respond to complex natural and anthropogenic forcings, changing physical and biological dynamics, changing carbon cycle chemistry and nutrient fluxes, and widespread marine harvesting. Planned observation and process studies related to the effects of nutrient inputs to coastal areas are relevant to the continued understanding of nutrient criteria.

Land-Ocean Interactions in the Coastal Zone (LOICZ)

The LOICZ project aims to understand the dynamics of global change in the coastal zone and to inform the scientific community, stakeholders, policy makers and managers of those changes (Figure 1.8). Specifically, the goals are to determine at both regional and global scales

- The nature of the interactions between land, ocean and atmosphere.
- How changes in various components of the earth system are affecting coastal zones and altering their role in global cycles.
- How future changes in coastal zones will affect their use by people.
- A sound scientific basis for future integrated management of coastal areas sustainably.

Figure 1.8. Information, including the LOICZ Science Plan and Implementation Strategy is at www.loicz.org.

Integrated Ocean Observing System (IOOS)

The IOOS, the United States' contribution to the Global Ocean Observing System, is a network of observational systems that routinely and continuously provide quality controlled data and information on the current state of the oceans and Great Lakes. It is a multidisciplinary system designed to provide data in forms and at rates required by decision makers to address a number of societal goals, among which are the protection and restoration of healthy coastal ecosystems and the sustained use of ocean and coastal resources. The technologies for monitoring nutrients, chlorophyll *a*, dissolved oxygen (DO) and other parameters of interest to nutrient criteria will continue to evolve into increased operational status.

Other Global Initiatives

Throughout the world, efforts are ongoing to either set criteria or develop parameters that will ensure *good water quality status* for waterbodies (e.g., Borja et al. 2006). In Europe, several efforts parallel those ongoing in the United States. The Oslo Paris Convention for the Protection of the North Sea (OSPAR) Comprehensive Procedure (COMPP) is an approach used to evaluate conditions in waters of European countries that are members of OSPAR. The second effort is the European Water Framework Directive, where, much like EPA's effort described herein, efforts are being made to sub-divide waterbodies into those that respond similarly to like pressures, while recognizing the unique responses of individual bodies of water. The European Union Water Framework Directive requires that all waterbodies be evaluated and to have monitoring and assessment schedules as well as management measures to bring impaired waters to a specific water quality condition. The approach is multifaceted, recognizing multiple driving forces (e.g., land use, agriculture, urban development, and so on), the complex ways in which those driving forces are expressed, the current status of the environment, the impacts on human health and the environment, and those policy measures that can be taken in response (Borja et al. 2006). Thresholds are also set against a reference value (Devlin et al. 2007). In all cases, improved understanding of interactions and uncertainty of pressures and impacts will continually be needed. Similar assessments are also in place in Hong Kong, Australia (Xu et al. 2004; Scanes et al. 2007) and elsewhere, and comparative integrated assessments will ultimately be useful.

The Process of the National Estuarine Expert Workgroup to Produce this Report

This report is the product of the National Estuarine Expert Workgroup, which met several times from 2005 to 2006. Subcommittees of the workgroup were formed to focus on estuarine typology, developing case studies and developing an estuarine nutrient database (Appendix I). An editorial workgroup was formed to merge the elements of the subcommittees to produce this report. This document is a summary of current information on estuarine nutrients and the processes that regulate them in various classes of estuaries. This information should prove to be a resource for states and managers charged with developing estuarine nutrient criteria.

The document first introduces the general complexity of nutrient dynamics in estuaries (Chapter 2), then presents a series of approaches for estuarine classification (Chapter 3). A framework for estuarine criteria development, along with supporting examples, is then provided (Chapter 4), and last, a series of case studies from 10 estuaries is presented (Chapter 5).

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Chapter 2:

Impacts of Nutrients in Estuaries

Patricia M. Glibert
David Flemer and
Hans Paerl

Figure 2.1. Runoff from agricultural areas and sewage are major sources of nutrients fueling estuarine eutrophication. Photos by P. Glibert (top) and A. Kana (bottom).

Estuarine Eutrophication: A Brief History

Eutrophication research played a central role in basic and applied limnology during the last century (Hutchinson 1969; Likens 1972; Wetzel 2001). Limnologists working in the north temperate zone, especially northern Europe and the more northern lakes in the United States, began to study the relationship between nutrient supplies and lakes early, before many lakes experienced large nutrient loads. Thus, there was an ability to capture lake ***reference conditions*** that facilitated the ecological assessment of natural versus anthropogenic effects of nutrient supplies. Reference conditions are the natural, or ambient, conditions against which nutrient enrichment can be compared. Nutrient-driven pollution was recognized as a serious threat to many large lakes in Europe and North America in the 1950s and 1960s—Lakes Erie and Washington are well-known examples.

Sewage-based eutrophication emanating from urban areas has long been recognized. Sewage pollution in the Thames River, for example, was recognized to be a problem in the 1800s (Attrill 1998). By the 1940s, wastewater treatment facilities were being developed in many regions to remove visible debris and pathogenic microorganisms from sewage effluents (Nixon 1995). By the 1960s, some municipalities began to remove organic matter (i.e., biochemical oxygen demand) and oxidizable (i.e., consumed oxygen) forms of N (e.g., ammonia).

Recognition of the serious threats to coastal and estuarine waters did not come until recent decades. Eutrophication and excess nutrients in estuaries began to receive

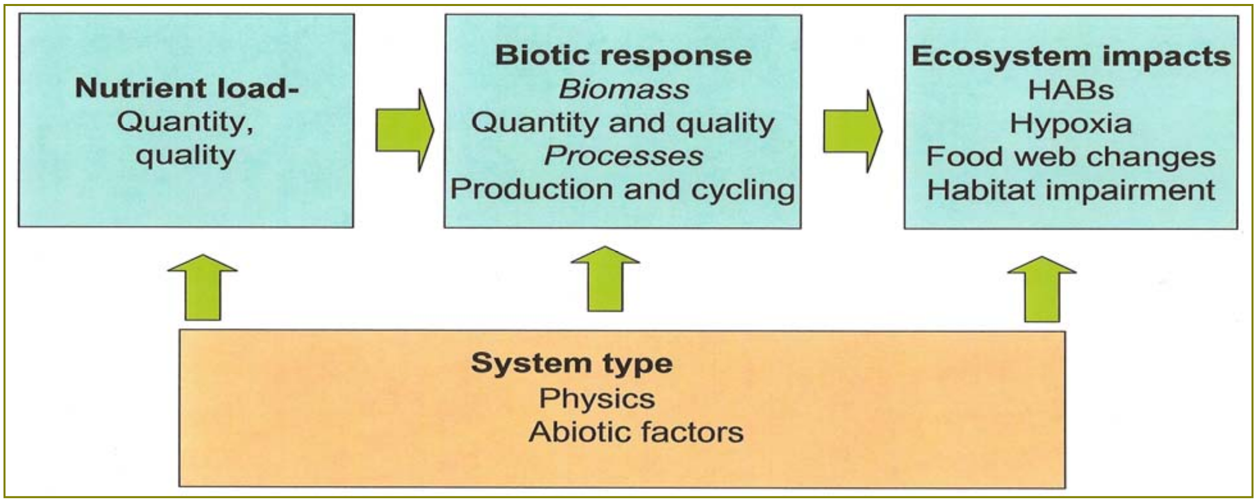


Figure 2.2. Nutrient load, in quantity and quality, affect the biotic response and the ecosystem impacts. These, in turn, are influenced by the type of system, and its physical and abiotic attributes.

attention in the late 1970s when the first international symposium on nutrient enrichment in estuaries was held (Neilson and Cronin 1981). Sewage and phosphates from detergents were the dominant focus.

The Complexity of Nutrient Enrichment in Estuaries—An Overview

Documented impacts of nutrient pollution in the United States and worldwide have included changes in habitat, decreases in biodiversity and increases in hypoxia and HABs (e.g., Nixon 1995; Bricker et al. 1999; NRC 2000; Burkholder 2001; Cloern 2001; Rabalais 2002; Anderson et al. 2002; Breitburg 2002; Glibert et al. 2005a, 2005b). However, the extent to which various symptoms are expressed depends on the rate of nutrient loading, its composition, seasonality of the loads relative to the growth state of the resident organisms, status of higher trophic levels, residence time, stratification and many other abiotic factors, such as suspended sediment load (e.g., Figure 2.2).

One of the important factors determining the expression of eutrophication symptoms is the composition of the nutrient pool. Nutrients can be delivered to an ecosystem from riverine sources, groundwater, atmospheric, marine and other sources. Each source can vary in the amount of specific nutrients they contribute (N, P or Silicon [Si]), as well as their proportional ratio to other nutrients in that source. They can also vary in the chemical form of those nutrients, inorganic or organic, or, in the case of N, oxidized (NO_3^- or NO_2^-) or reduced (NH_4^+) forms.

A broad range of anthropogenic activities contribute to the nutrient load of estuaries. Increasing human population contributes to greater sewage loads (Figure 2.3). In some regions, such as Long Island Sound and Kaneohe Bay, Hawaii, sewage has largely been responsible for the bulk of the N inputs (Nixon and Pilson 1983; NRC 1993). However, for most estuarine and coastal systems, nonpoint source nutrient inputs are of greater concern than point sources. Howarth et al. (1996), for example, estimated that only 12 percent of the N flux to the North American continental shelf is derived from point sources. Land application of fertilizers remains the largest source of nonpoint nutrient pollution, and direct relationships between fertilizer use and riverine nitrogen flux have been established (Vitousek et al. 1997; Smil 2001; Glibert et al. 2005a). Fertilizer use has increased over the past several decades, and its composition has been changing to more organic forms (Glibert et al. 2006). The development of concentrated animal operations near

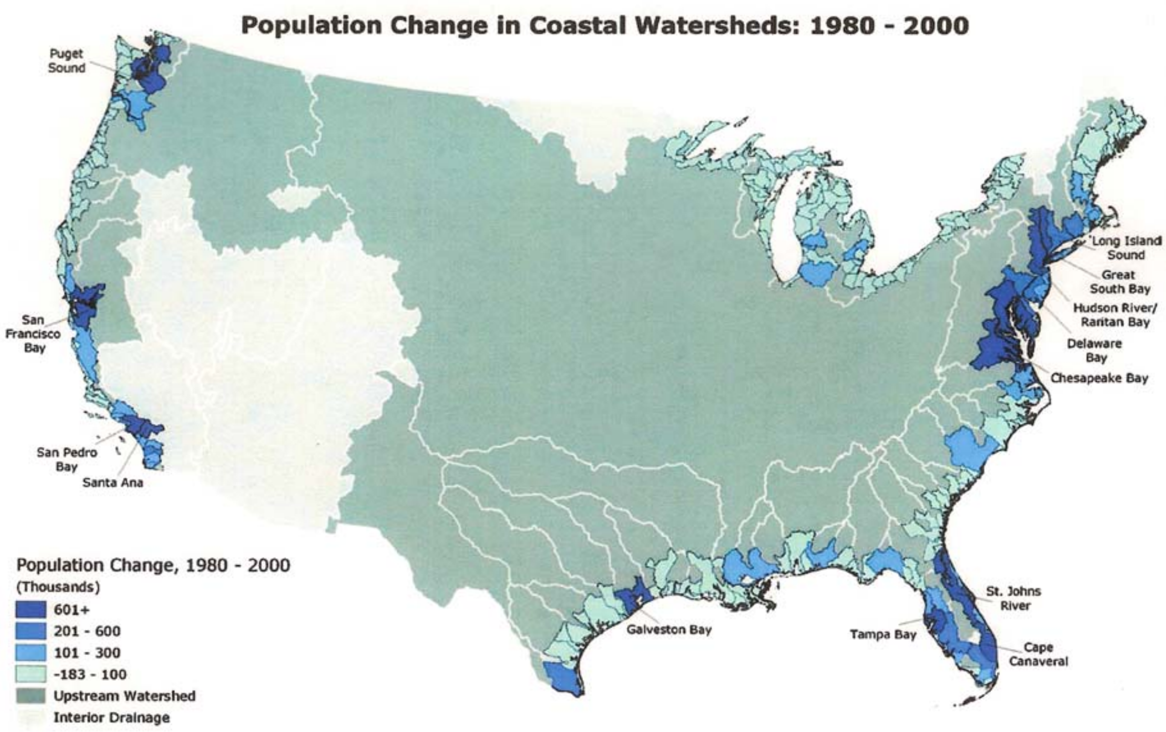


Figure 2.3. More than 50 percent of the nation's population lives in areas that can be called coastal regions. It is estimated that the coastal population is increasing at more than 3,000 persons per day (NRC 2000), yielding a projected 325 people per square mile in the coastal areas by the year 2015. In addition to permanent residents in coastal areas, those regions draw seasonal visitors, further escalating the stresses from human population. From Crossett et al. 2004.

coastal waters is also an increasing source of nutrients (Mallin 2000).

In many regions, atmospheric deposition of N can contribute up to 40 percent of the local nutrient input (Figure 2.4; Paerl 1995, 1997; Howarth et al. 2002). Atmospheric deposition is generally thought to be increasing because of NO_x emissions from fossil fuel burning and from volatilization of animal manures. Direct deposition to estuarine waters is an additionally important new N source that can bypass the terrestrial and in-stream filters that

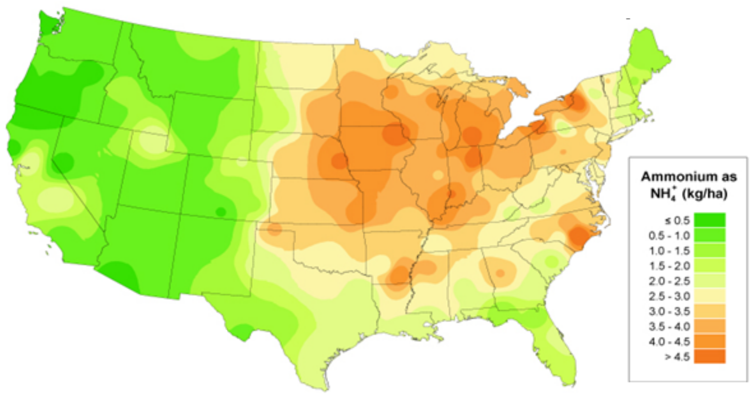


Figure 2.4. Atmospheric deposition of ammonium in 2003. From Paerl 1997.

process N entering via the watershed. In European and U.S. estuarine and coastal waters, downwind of anthropogenic emission sources, atmospheric N deposition contributes from 10 to more than 40 percent of new N loading (Jaworski et al. 1997; Paerl 1997). In some instances, it is the single largest source of new N affecting these waters. In eastern North Carolina, atmospheric N deposition (chiefly NO_x) has at least doubled since the 1970s as a result of urbanization and, more recently, agricultural growth (Mallin 2000). Recent growth and intensification of animal operations in the Midwest and coastal regions (e.g., Mid-Atlantic coastal plain) have been linked to increasing amounts of volatilization of animal manures. Ammonium (NH_4^+) emitted from such operations accounts for approximately half the atmospheric N deposition in nearby estuarine (Neuse River Estuary-Pamlico Sound system) and Atlantic coastal waters (Whitall et al. 2003; Paerl 1997).

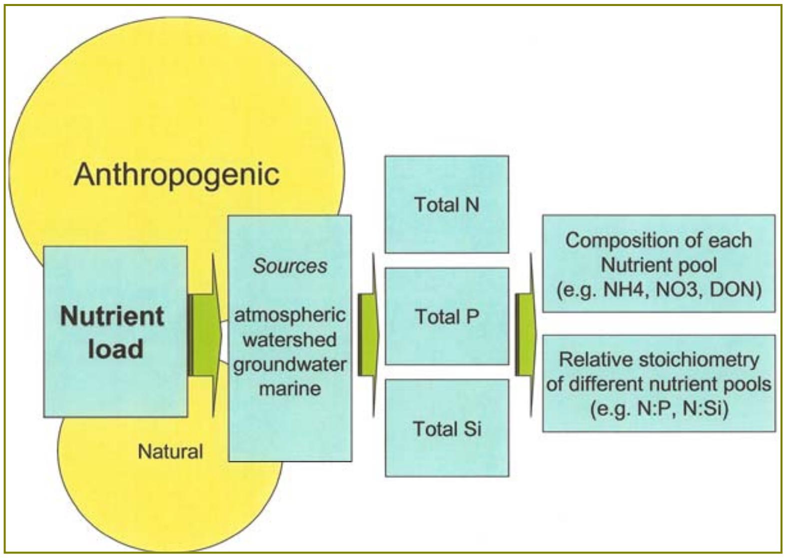


Figure 2.5. The total nutrient load of any estuarine system has both natural and anthropogenic components from many sources. The nutrients are available in various proportions and in various chemical forms, and it is their relative availability that has a major impact on the biotic response.

Groundwater is another important nonpoint nutrient source. However, there can be significant lag periods (up to decades) between the time of the human activities that enrich the groundwater and the effects in estuarine systems. For example, it has been suggested that the HAB events dominated by the brown tide species *Aureococcus anophagefferns* in Long Island embayments could be a reflection of the rapid escalation in population development and fertilizer applications on Long Island of prior decades (LaRoche et al. 1997).

Of particular concern in determining the biotic response to nutrient loading is the form or type of nutrient (Figure 2.5). Not only can the relative stoichiometry of available nutrients determine both the rate of primary production and the composition of the biota, but different primary producers can preferentially use one form of a nutrient over another. About half of the dissolved N transported by rivers to the coastal ocean is now dissolved organic N (DON; Meybeck 1982). Fertilizer N is now proportionately higher in urea than in previous decades (Figure 2.6; Glibert et al. 2006). Fertilizer N is also directly related to riverine N flux (Figure 2.7; Smil 2001). Organic nutrients have been shown to be important in the development of blooms of various HAB species, in particular cyanobacteria and dinoflagellates (e.g., Paerl 1988; Glibert et al. 2001, 2005a, 2005b).

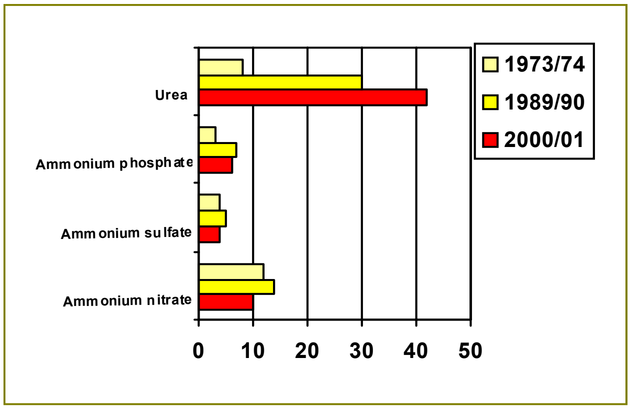


Figure 2.6. Global increase and compositional change in world N fertilizer use. Data are in million metric tons per year for the years indicated. Data are replotted from the International Fertilizer Industry. From Glibert et al. 2005a.

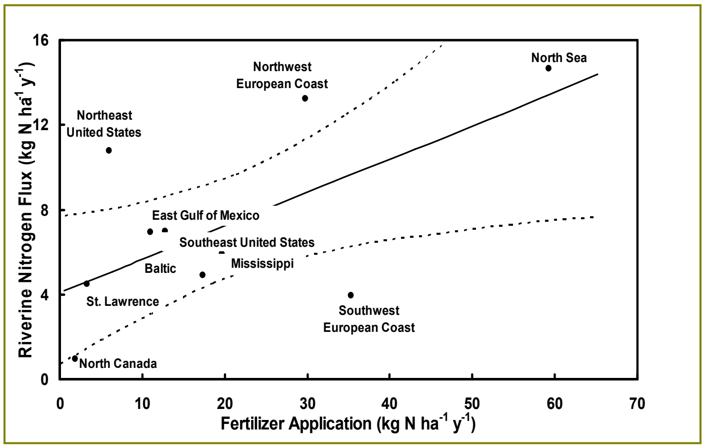
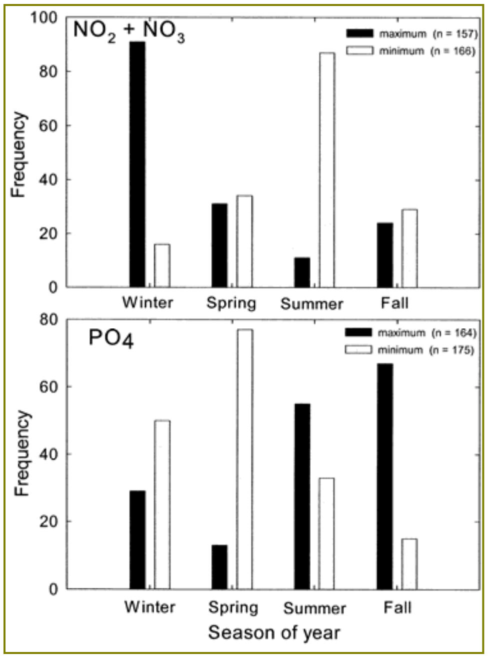


Figure 2.7. The relationship between the rate of fertilizer applications and the flux of riverine N discharge. From Smil 2001.

An example from the Chesapeake Bay illustrates the dynamic variability in the ratios of N and P on both short- and longer-term time scales (Figure 2.8). On a seasonal basis, the maximum input of NO_3^- typically occurs in the winter, whereas the maximum in phosphate PO_4^{-3} typically occurs in summer and fall. Furthermore, even on the scale of days, these ratios can be highly dynamic in response to rainfall or other meteorological events. Those differences create challenges for managers if a single numeric criterion is applied for all seasons of the year, or if decisions are based on a single sample from a single point in time.

Ecological responses are also a function of the physical dynamics, residence times and mixing within a system (Figure 2.9). Estuaries and near-shore environments also vary in the type, abundance and geographic coverage of biological communities at risk to nutrient enrichment. Depending on system typology, responses in the biotia might not be directly related to nutrients, or can be lagged in time or highly complex and nonlinear (Cloern 2001). Nutrient enrichment effects are interactive. Positive feedbacks between biotic changes, habitat impairment and biogeochemistry can occur, which might help to reinforce the impacts of eutrophication (Figure 2.9).

Climate variability and change interact with these important stressors. Nutrient supply is tightly coupled with freshwater input (e.g., Caraco 1995; Vitousek et al. 1997) that, in turn, is driven by regional climate variability (Najjar 1999; Miller et al. 2005; Burkholder et al. 2006). Freshwater input determines, to a large extent, the spring chlorophyll *a* maximum in many coastal systems by delivering nutrients (Harding 1994; Malone et al.



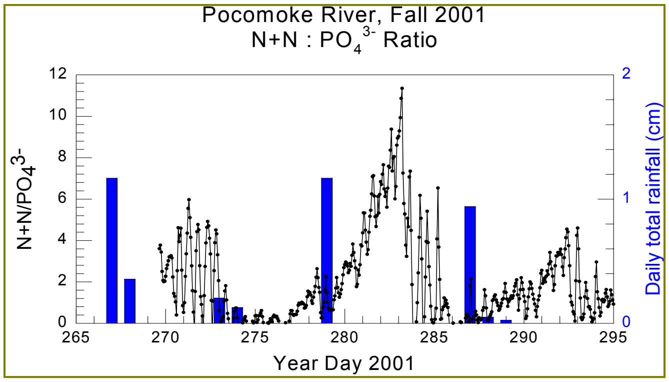


Figure 2.8. The dynamic variability in N and P on both long-term (left) and short-term (above) scales. Left: frequency in which maximum and minimum NO_2^- + NO_3^- and PO_4^{-3} concentrations, by season, were observed in a variety of U.S. estuaries. Data from Frank et al. 2008. Top: daily change in (NO_2^- + NO_3^-):PO_4^{-3} for the Pocomoke River, a tributary of the Chesapeake Bay. From Glibert et al. 2005b.

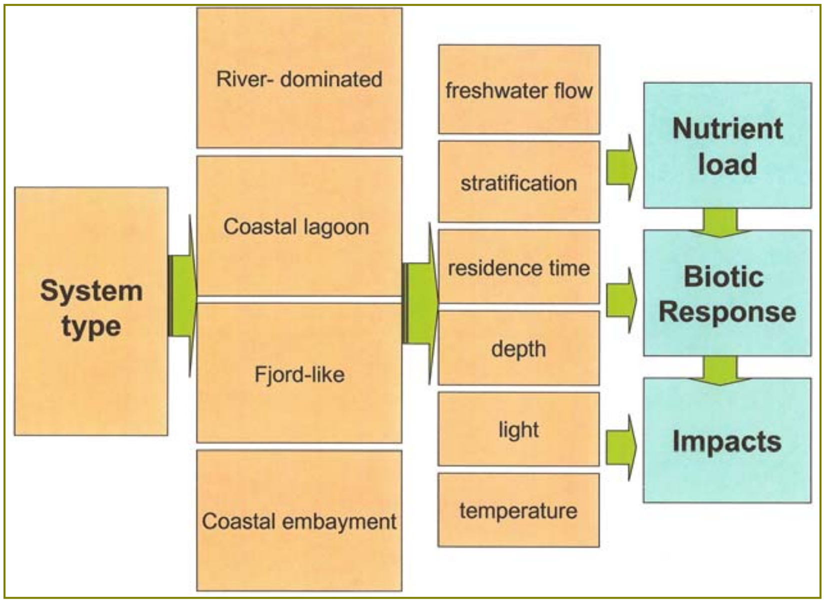


Figure 2.9. Estuarine typology influences the many abiotic factors that, in turn, affect the nutrient load and the expressions of eutrophication.

1996; Kemp et al. 2005). Climate variability affects the timing of freshwater flow, residence times, the magnitude and timing of nutrient pulses and resulting biotic responses (Miller et al. 2006). In coastal lagoons, where riverine input is not the dominant source of nutrient delivery, climate variability can alter the input of groundwater nutrients (e.g., LaRoche et al. 1997).

Major climatological events, such as strong storms and hurricanes, also affect estuarine conditions during short periods. As examples, Hurricane Isabel in 2003 caused a large phytoplankton bloom to develop in the Chesapeake Bay within days (Miller et al. 2005). Extensive hypoxia occurred in the lagoonal Neuse River Estuary after Hurricanes Dennis and Floyd, whereas a storm that caused less flooding, Hurricane Fran, led to more concentrated pollutants, extensive anoxia and massive fish kills (Burkholder et al. 2004). Furthermore, as the global climate warms, many processes within ecosystems are altered. Temperature tolerances can be exceeded for some species, while for others, more subtle changes such as spawning timing or success can be altered. In the plankton, heterotrophy can be favored because of differential temperature effects on phytoplankton production, respiration, and microbial productivity (Lomas et al. 2002). Increasing temperature, too, decreases the amount of oxygen that can be dissolved in water (Millero 2006), thereby increasing the potential for hypoxia.

Overharvesting of shellfish and fish and nutrient enrichment have long been recognized also to be interactive forces on ecosystem functioning. Benthic filter feeders affect nutrient recycling processes, which also effectively reduce algal biomass through filtering (Cloern 1991; Ulanowicz and Tuttle 1992). A recent review of overfishing in coastal ecosystems, based on paleo-ecological and more present ecological data, suggested that enormous losses of vertebrates and shellfish have occurred in virtually all coastal ecosystems with profound implications (Jackson et al. 2001). This review identified several corollaries of overfishing, including that nutrient pollution and eutrophication generally occurred after overfishing in a historical context, that overfishing might be a precondition for eutrophication to occur, and that massive removal of suspension feeders leaves systems more vulnerable to microbial outbreaks and disease (Jackson et al. 2001). Estuaries on the U.S. East Coast now have oyster populations that are ~1 percent of their 19th century levels, with similar statistics for clams and scallops (e.g., Newell 1988; Kirby 2004; Kemp et al. 2005). Nutrient enrichment can also lead to more pelagic rather than demersal fish (because of habitat loss) and a general decline in populations as water quality conditions further deteriorate (Caddy 1993; Breitburg 2002). Decreasing oxygen and other degradations of habitat can also affect fish reproduction and recruitment success (Niklikshek 2001).

Sediment quality also affects nutrient effects in estuaries. Adsorption and desorption rates of P, in particular, by sediments are dependent on their physicochemical quality (e.g., Froelich 1988; Koop et al. 1990; Pant and Reddy 2001). For example, the carbonate nature of subtropical and tropical sediments relates to high rates of P adsorption, and thus a tendency for lower dissolved P levels in the overlying water, which, in turn, causes higher frequency of P limitation (e.g., Berner 1974; Morse and Cook 1978). Interactions with iron and sulfur also affect the influx and efflux of P from sediments. Interactions of N with sediments are also important, and the pathways by which N is transformed in the sediment are controlled in part by the redox chemistry of the sediments (reviewed by Bianchi 2007). Other controls on the rate of exchange of N between sediments and the overlying water include the rate of deposition of organic matter to the sediments, rates of bacterial remineralization of nutrients and abiotic factors such as temperature (e.g., Nixon et al. 1976; Kemp and Boynton 1984; Billen et al. 1991). Collectively, all those factors affect the quality of the biota and their interactions (Figure 2.10).

With increasing nutrient loads to estuaries, total production cannot increase infinitely; a maximum is attained as other factors begin to play a more important controlling role (Cloern 2001). The nutrient in least availability relative to the needs of the organisms (i.e., the nutrient that algae deplete first) will limit total production (e.g., Liebig's Law of the Minimum, reviewed in Wetzel 2001). The concept of limiting nutrients in a eutrophic system actually is very complex. Ecosystem response to nutrient enrichment, or eutrophication, is a continual process rather than a static condition or a trophic state (Cloern 2001; Smayda 2005). As relationships are explored between eutrophication and biotic responses, it is important to recognize that different systems fall on different points along the eutrophication continuum.

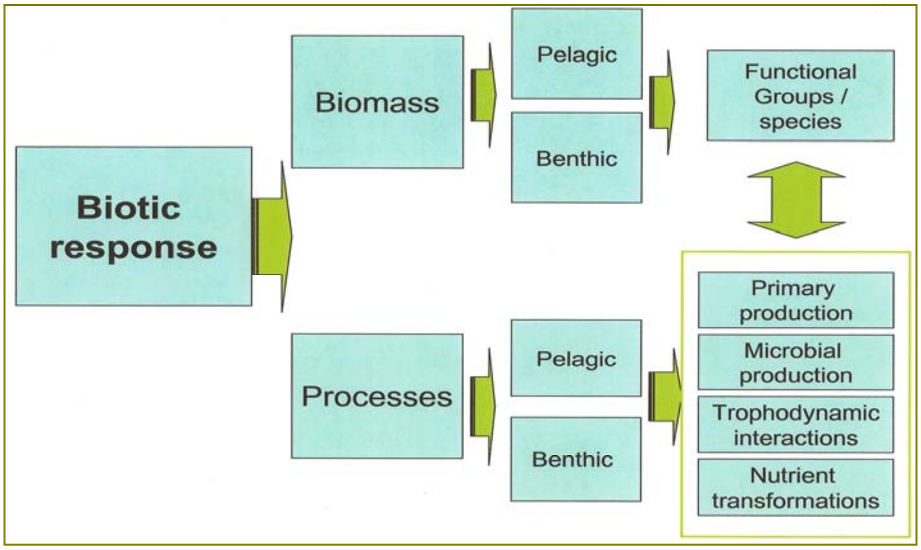


Figure 2.10. One of the expressions of eutrophication is a change in the amount of primary production in the water column versus the benthos. This also alters trophic transfers and biogeochemistry of nutrient cycling.

Conceptual Models of Eutrophication and Applications of Ecological Health Indices

The conceptual relationship between and among system responses and effects of nutrient over-enrichment has advanced considerably in the past decade. Whereas our initial understanding of the relationships between nutrient loading and responses was that they were thought to be direct and proportional (e.g., Cloern 2001, Phase 1 and 2 models), we now know that such relationships are often complex, modulated by interactive effects and dependent on both biological responses as well as physical dynamics. Furthermore, the responses to nutrients are also often synergistic with responses to other drivers of change, such as climate change, habitat change and

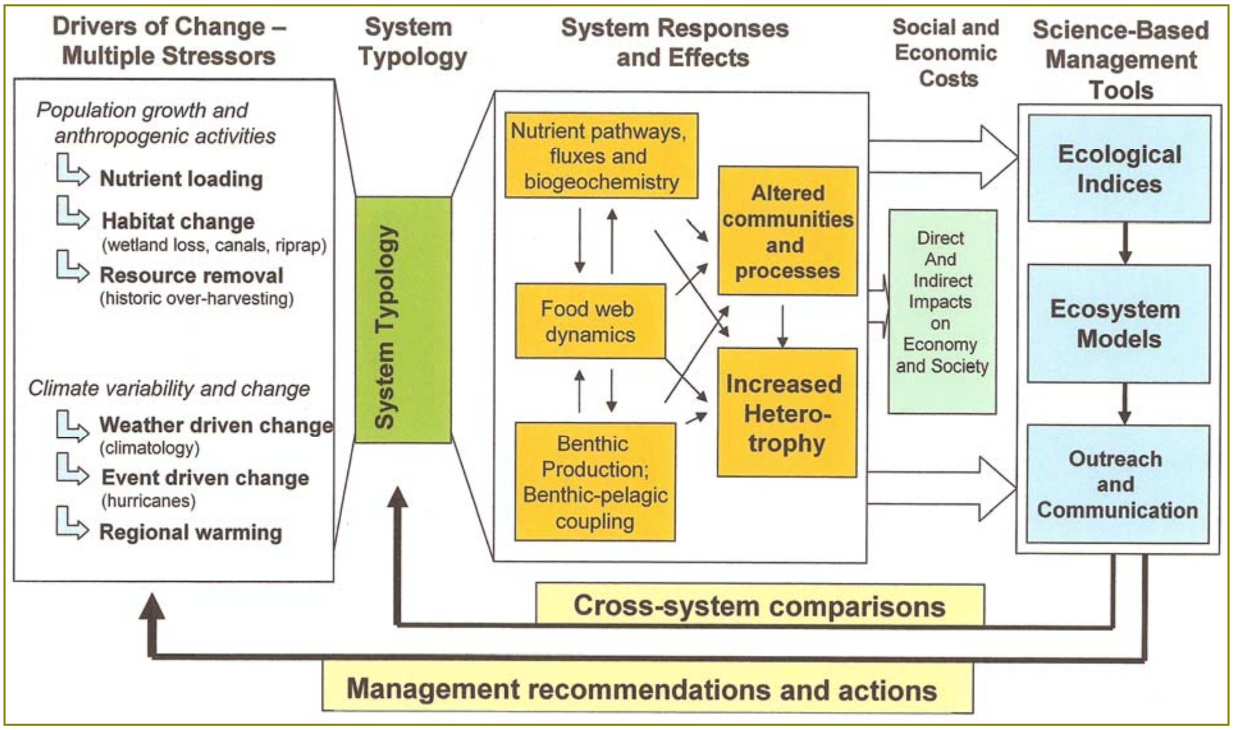


Figure 2.11. Conceptual relationship between the stressors, responses and management tools and actions. This schematic was redrawn and modified from the Phase III eutrophication model of Cloern 2001.

alteration of resources due to overfishing.

Cloern (2001) captured this diversity of effects in a relatively complex conceptual model, noting that estuaries represent a dynamic suite of ecosystems where cause and effect of nutrient pollution is modulated by multiple forms of human disturbance superimposed on highly variable types of natural systems and processes. This *Phase III* model of coastal eutrophication (Figure 2.11) recognizes multiple stressors that have specific measurable responses and that those responses and effects can be quantified and related to system typology, around which science-based management tools can be developed. ***The challenge now is to develop and apply those tools and to understand their application in the context of estuarine criteria.***

Ecological indicators provide integrated assessments of ecological function and, ideally, those metrics should be linked to causes, allow future predictions, and be sensitive enough to assess changes resulting from management actions (Suter 1993). Historical concentrations, narrative statements, or comparisons to reference stations, on their own, are not necessarily effective in providing a solid foundation for assessing the relationships between water quality and biotic responses. Determining how various combinations of parameters are related to ecosystem responses increases our understanding of responses in different estuarine types. Various models and eutrophication indices are now available. Several are highlighted here, not as a thorough review, but as examples of the types of indices that can be developed and applied. More discussion of integrated assessment, or application of biocriteria, is given in Chapter 4.

Phytoplankton response is often measured in chlorophyll *a*. However, an improved assessment of phytoplankton response is ***phytoplankton community composition***. With nutrient enrichment, a shift in plankton community composition is frequently observed, with large diatoms giving way to smaller cyanobacteria and small flagellates (e.g., Smayda 1980; Marshall et al. 2005; MacIntrye et al. 2004). Many of the species can be harmful to higher trophic levels, disrupting normal ecosystem function. The dominance of such species can result in a failure of normal predator-prey interactions, which, in turn, enhances the transfer of nutrients that sustain the blooms at the expense of competing algal species (Irigoien et al. 2005; Mitra and Flynn 2006; Sunda et al. 2006).

Another example is the ***relationship between production by benthic microphytobenthos and macrophytes and phytoplankton***. Particularly in shallow systems, such as coastal lagoons, increased nutrient loading from anthropogenic sources generally leads to a shift in primary producers from benthic microalgae and macrophytes to pelagic phytoplankton (e.g., Cloern 2001). Increased phytoplankton and macroalgal proliferation at high nutrient levels affects seagrasses and benthic microbiota that compete for light (Burkholder et al. 1992; Deegan 2002), in turn altering the food web structure by altering the habitat that supports fish and shellfish. All aspects of metabolism, predator-prey interactions and species success are altered when a system is stressed by nutrient loading (Brietburg et al. 1999; Breitburg 2002).

Multiple, integrated parameters are often better indicators of ecosystem response than single cause and effect relationships. Ultimately, integrated parameters should incorporate system state variables (e.g., nutrients, DO), with those of biotic response (e.g., phytoplankton composition) and secondary ecological responses (e.g., seagrass occurrence or shellfish abundance). Where possible, rates of processes (e.g., primary production, bacterial production, zooplankton grazing) and rates of biogeochemical fluxes (e.g., regeneration rates of benthic fluxes) should also be incorporated. That combination of diverse measures ensures that the approach is broadly applicable along the typical eutrophication transition within a system, from benthic dominated to water-column dominated primary production. Different parameters provide information about changes within systems at different times in the process. It is ideal to include further metrics on seagrass, macroalgae, fisheries and wetlands within a fully integrated assessment for tracking ecosystem status.

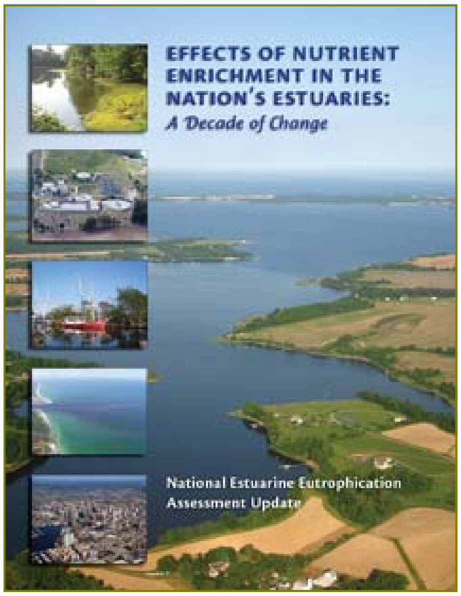


Figure 2.12. The National Estuarine Eutrophication Assessment (Bricker et al. 2007) updated the status of the nation's estuaries.

In the National Estuarine Eutrophication Assessment (Bricker et al. 1999, 2007; Figure 2.12) Bricker and colleagues used an integrated set of parameters to establish the overall expression of eutrophic condition for all the nation's estuaries and also examined factors influencing the development of observed problems (i.e., nutrient inputs and a system's natural susceptibility), and how conditions are expected to change by the year 2020. This eutrophic condition analysis included evaluation of the expression of primary symptoms, such as those that respond directly to nutrients (i.e., chlorophyll *a* and macroalgal abundances), and secondary symptoms, those that are indirect and are indicative of more serious problems (i.e., loss of submerged aquatic vegetation [SAV], occurrences of nuisance and toxic blooms [HABs], and depleted DO). Examples of these include

- Primary:
  – Chlorophyll *a*
  – Macroalgal blooms
- Secondary:
  – Loss of SAV
  – Presence of harmful algae blooms
  – Presence of low DO

In the national assessment, each of the five indicators received a rating, on the basis of expert opinion, that combines the observed concentration or problem occurrence, the spatial coverage of the problem levels and the frequency of occurrences (i.e., periodic or seasonal, episodic, or persistent). The secondary symptoms were given a higher weight than the primary symptoms when the two are combined to give an overall eutrophic condition of a system. The scores were then converted to categorical ratings of high, moderate, or low, using established thresholds. That approach successfully provided an assessment of 138 estuaries, covering 90 percent of the estuarine surface area of the United States and the Mississippi plume. At the time of its first application in 1999, 40 percent of U.S. estuaries were characterized as highly eutrophic, with the most severe expression of eutrophication along the Gulf of Mexico and the mid-Atlantic coasts. The 2007 assessment revealed that while some estuaries have shown improvement over the past decade, a large percentage have significant eutrophication problems and are likely to worsen over time (Figure 2.13).

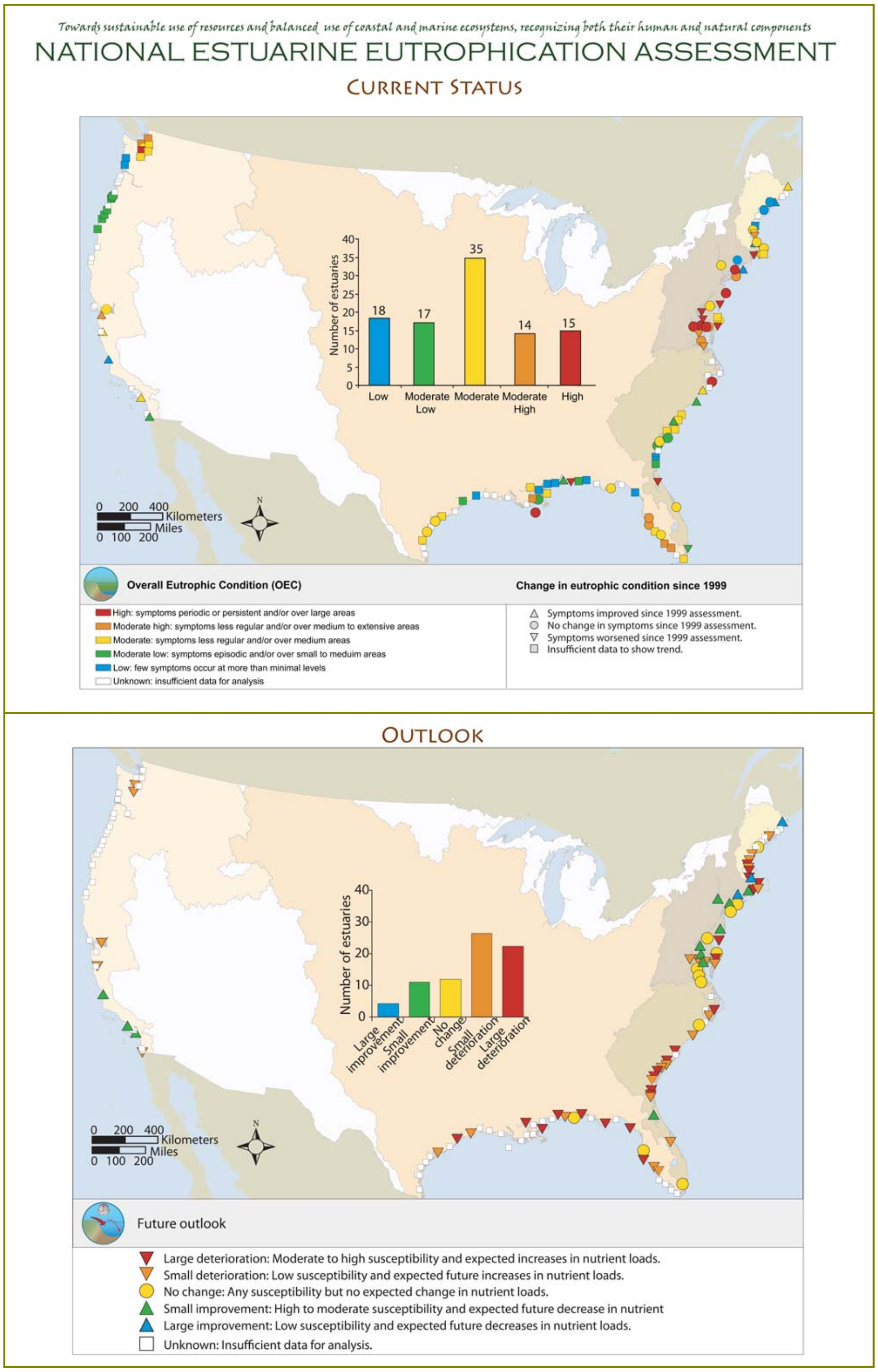


Figure 2.13. The National Estuarine Eutrophication Assessment shows that estuaries of the mid-Atlantic region are the most eutrophic, and that the outlook for many estuaries, particularly on the East Coast, is for increasing eutrophication. From Bricker et al. 2007.

EPA has published its assessment of the nation's estuaries several times over the past decade, the most recent of which was in 2007 (USEPA 2007; Figure 2.14). Those assessments also have multiple benchmarks of ecological health in developing an overall regional score: water quality, sediment quality, benthic quality, coastal habitat and fish tissue contaminants. Each of those five benchmarks were, in turn, composed of various indicators. The water quality index, for example, was made up of measurements of N, P, chlorophyll *a*, water clarity and DO, as specific indices of eutrophication status. Using specific criteria for each of these parameters, waters were classified as *good*, *fair*, or *poor* according to the following guidelines for each site (Figure 2.15). The overall condition for each region was assessed using a weighted averaging of the five benchmarks, such that fish tissue contaminants weighted most highly, followed by water quality and sediment quality, and last—benthic and coastal habitat conditions. Using that approach, the overall quality of the nation's coasts was classified as *fair*, and most regions rated between *poor* and *fair*.

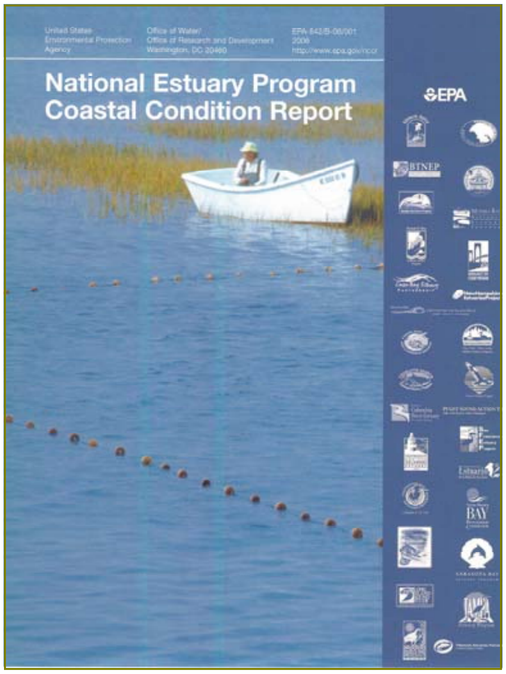


Figure 2.14. The most recent EPA National Coastal Condition Report was published in 2007 and applies multiple ecological health indices.

Rating	Criteria
Good	A maximum of one indicator is fair and no indicators are poor
Fair	One of the indicators is rated poor, and two or more are rated fair
Poor	Two or more of the five indicators are rated poor
Missing	Two components of the indicator are missing

Figure 2.15. EPA's assessment uses multiple indicators to derive a cumulative eutrophication rating.

Applying Ecosystem-Based Management Approaches and Predictive Tools

In moving toward recommendations for numeric criteria, the ultimate goal is to develop a process whereby the complexities of the system responses are recognized, biotic as well as abiotic characteristics are included, and opportunities are provided for continual evaluation and alteration of criteria.

Such an approach, ***Ecosystem-Based Management***, "looks at all the links among living and nonliving resources, rather than considering single issues in isolation" (U.S. Commission on Ocean Policy 2004). The goal of ecosystem-based management is, "to maintain the health of the whole as well as the parts. It acknowledges the connections...." (Pew Oceans Commission 2004). Both the U.S. Commission on Ocean Policy report and the Pew Oceans report call on the United States to adopt an ecosystem-based management approach for all ocean policies (Figure 2.16).

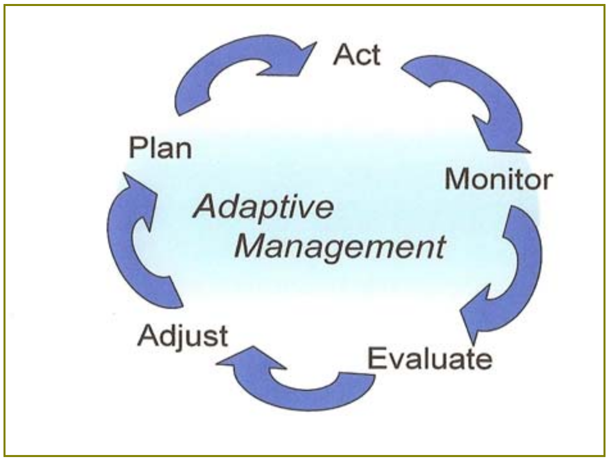


Figure 2.16. Adaptive ecosystem-based management is a continual process of evaluation and adjustment as new knowledge is gained.

Ecosystem-based management:

- Emphasizes the protection of ecosystem structure, functioning and key processes.
- Is place-based in focusing on a specific ecosystem and the range of activities affecting it.
- Explicitly accounts for the interconnectedness within systems, recognizing the importance of interactions between many target species or key services and other non-target species.
- Acknowledges interconnectedness among systems, such as between air, land and seas.
- Integrates ecological, social, economic and institutional perspectives, recognizing their strong interdependencies.

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Chapter 3:

Estuarine Typology Development and Application

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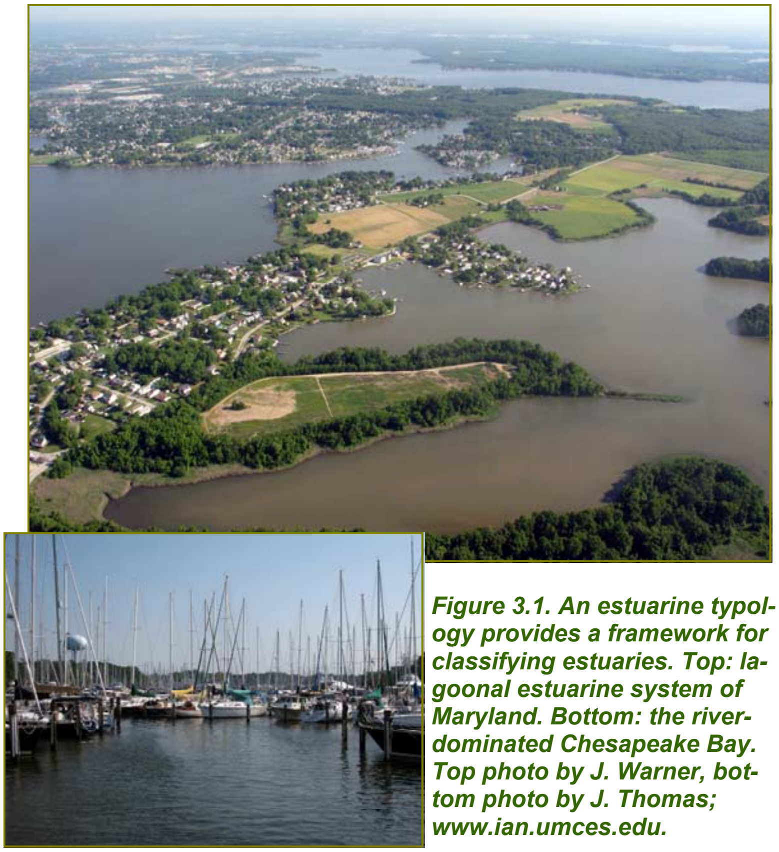

Figure 3.1. An estuarine typology provides a framework for classifying estuaries. Top: lagoonal estuarine system of Maryland. Bottom: the river-dominated Chesapeake Bay. Top photo by J. Warner, bottom photo by J. Thomas; www.ian.umces.edu.

The Need for an Estuarine Classification

Estuaries can respond to similar levels of nutrient loading in very different ways. As described throughout this report, this disparity can be ascribed to fundamental differences in the way the respective waterbodies receive and process inputs. This chapter explores patterns of estuarine behavior that tend to co-occur within a few types of estuaries and assesses their relevance as a means to classify estuaries. Organization of estuaries in a logical classification framework that provides a scientific basis for grouping and analysis of their responses to nutrients may aid in the derivation of numeric nutrient criteria (e.g., Hayden et al. 1984, Kurtz et al. 2006). If those responses can be classified within an established framework, the prediction of how other estuaries of those types would respond to nutrient inputs is achievable.

A classification is an organizational system that groups elements according to a specific typology. A ***typology*** is an organizing framework that groups elements according to a specific underlying model. As described here, there are a number of models that can be used to implement a typology. It is important that the typological model be stated explicitly and exactly in order for the classification to have meaning for its users. Underlying the typology, an ***ontology*** provides the set of specific relationships among the elements that dictates where they fall in the typology. Thus, a typology can be thought of as the methodology that is used to sift through the data that populate a classification. The ontology is the set of rules and definitions for applying the classification, and the classification itself is the compendium of elements, in this case estuaries, and their assigned types.

This chapter describes estuarine typology, the definition of the major classes of estuaries and the identification of the major class attributes that distinguish them. This chapter also introduces guidance for evaluating and assigning estuaries to classes, and for statistically analyzing data for derivation of numeric nutrient criteria, a topic which is further developed in Chapter 4.

Definition of an Estuary

The first step in the task of classifying estuaries is the clear establishment of the definition of an estuary. In *The Nutrient Criteria Technical Guidance Manual: Estuarine and Coastal Marine Waters* (USEPA 2001, p. 1-2) EPA discusses the diversity of definitions of estuaries. It states,

> …a classical definition of estuaries focuses on selected physical features – e.g., "semi-enclosed water bodies which have a free connection to the open sea and within which sea water is measurably diluted with freshwater derived from the land" (Pritchard 1969). This definition is limited because it does not capture the diversity of shallow coastal ecosystems today often lumped under the rubric of estuary. For example, one might include tidal rivers, embayments, lagoons, coastal river plumes, and river-dominated coastal indentions that many consider the archetype of estuary. To accommodate the full range of diversity, the classical definition should be expanded to include the role of tides in mixing, the sporadic freshwater input (e.g., Laguna Madre, TX), coastal mixing near large rivers (e.g., Mississippi and Columbia rivers), and tropical and semitropical estuaries where evaporation may influence circulation.

Using that EPA guidance, the broadest possible definition of estuary was used in this analysis so that the working definition could include open and closed systems, estuarine and hypersaline coastal waters influenced by land drainage, and open coastal marine waters that receive freshwater input. Thus, the preferred definition of *estuary* is, "***a semi-enclosed coastal waterbody with restricted circulation, or coastal marine waters influenced by significant freshwater inflow during at least part of the year***." That definition is purposely lacking in quantitative criteria in defining terms (semi-enclosed, restricted, coastal, marine, influenced, significant, freshwater and part of the year) so as to leave wide latitude for assessing the characteristics in the initial phases of constructing a typology.

Statistical models can yield results that differ depending on initial assumptions and the quality of data. The purpose of the classification process presented here is to sufficiently constrain those models to reduce uncertainty and to increase the predictive power of models that relate nutrient inputs to estuarine response. The intent is to use the constrained models to help develop a semi-quantitative understanding of how nutrients are processed by categories of estuaries. Those relationships then can be refined by accounting for specific processes and characteristics that further influence nutrient dynamics and fluxes through the ecosystem. Several typologies are examined here that demonstrate ways in which estuaries can be effectively grouped. A proposed classification is recommended that provides a simple, coherent framework that identifies four estuary types.

Selecting the Population of Estuaries

Any classification approach should initially determine the population of estuaries to classify. The following factors were used in identifying systems to be included for this analysis:

- Ecosystems on the seacoasts of the U.S. mainland, Alaska and Hawaii that are receiving bodies for freshwater of areal size larger than 2.5 km^2 or possess a drainage area larger than 2.5 km^2. (Future size ranges can be adjusted as needed to accommodate management needs.)
- Coastal waters that are directly influenced or potentially influenced by drainage from land, including those coastal waters receiving channelized surface flow, subsurface flow, groundwater input, nonpoint

source input, precipitation input.

- Open waters adjacent to identified estuaries that are influenced by freshwater outflow (e.g., coastal plumes) and are of areal size larger than 2.5 km^2 or possess a drainage area larger than 2.5 km^2.

Classification Approaches

Over the past several decades scientists have taken many approaches to estuarine classification. Pritchard (1952) classified estuaries on the basis of the dynamic flux of salt. Fairbridge (1980) suggested that estuaries could be classified into roughly eight categories by geomorphology (Bianchi 2007). Another approach (Perillo 1995; Bianchi 2007) classifies estuaries on the basis of shoreline and their morphogenic characteristics, which consider the structure from which the estuary was formed (e.g., former fluvial valleys, former glacial valleys).

The concept of an estuarine type implies that a particular class of estuaries combines common ranges of several physical variables that tend to generally coexist for each member of that class, foster similar responses in estuaries, and by which different types can be distinguished from each other. The understanding of how some general features of an estuary can be important determinants of estuarine response is both intuitive and based on an abundance of data. For example whether an estuary is well-flushed, river-dominated or poorly-circulated conjures an idea of the major characteristics of the light regime, oxygen regime, trophic status and even salinity regime. The concept of a geophysical classification underpins the importance of size, shape and flushing in dictating processes within an estuary.

In general, at least three kinds of approaches can be used for classifying geospatial units. Those are

- A ***Conceptual approach***, a process for distinguishing classes of estuaries on the basis of descriptive and quantitative characteristics, using *a priori* assumptions about critical classifiers.
- A ***Statistical-cluster approach***, a process that uses cluster analysis to assign estuaries to categories with respect to an established threshold of statistical similarity and where the categories are determined by a statistical algorithm.
- A ***Statistical-Bayesian Classification and Regression Tree (B-CART)***, an approach that simultaneously optimizes for class separation and for the fit of dose-response variables.

Those approaches are complementary and were applied here in combination to develop an estuarine typology that might be relevant for estuarine criteria development. In ordering estuaries into a classification, the variables that are not controllable (i.e., the natural characteristics of the environment that influence processing to varying degrees) are applied. Each of the approaches was independently used to group estuaries according to physical (e.g., tide), hydrological (e.g., discharge), and geomorphological (e.g., hypsometry), and hydromorphological (e.g., stratified water column) properties. Each classification approach draws on different kinds of variables from a suite of controlling hydrogeomorphologic/physical characteristics. Each of the approaches has strengths and has a unique role to play in the assessment of estuarine response to nutrient inputs.

For the purposes of nutrient criteria analysis, it was deemed important to classify systems without using information that would later be relevant in defining nutrient inputs or biological responses that would be quantified as criteria. Separating estuaries by using information that includes these responses reduces the ability to interpret why estuaries respond differently (biologically) to similar nutrient inputs.

The conceptual and statistical-cluster approaches are considered indirect approaches because the categorization (grouping of estuaries) and model development activities are two independent steps. The B-CART analysis is considered a direct approach because it specifically and simultaneously optimizes stressor-response models. For the purpose here, the conceptual and statistical cluster approaches were used; the B-CART analysis is still in development and herein described only briefly.

The conceptual classification is drawn from a hierarchical framework for describing North American coastal and marine systems embodied in the *Coastal Marine Ecological Classification Standard* (CMECS, Madden et al. 2005) and a classification previously developed by the National Oceanic and Atmospheric Administration (NOAA) (Alexander and Bricker 2003). The statistical-cluster approach is supported by two initiatives, one developed by EPA (Burgess et al. 2004; Engle et al. 2007) and one developed by NOAA (Bricker et al. 2007). After a description of each type of classification approach, this chapter then describes a multivariate regression analysis that couples physical/hydrological classes with stressor-response models to begin to demonstrate its utility in the development of thresholds protective of designated uses.

From the perspective of threshold development, the importance of classification schemes is the development of class-specific nutrient load-response (***L-r***) models. ***If all estuaries are unique, those models would need to be developed for each estuary and embayment of the United States. It is hoped that a classification scheme that affords an ability to apply categorization and normalization protocols can lead to a small number of models that characterize estuaries.***

Conceptual Classification of Estuaries

The conceptual classification is the simplest kind of classification but can be a very powerful approach because of its simplicity. The approach uses a detailed set of quantitatively defined geomorphological, energy and biogeographic classifiers that fall into multiple categories. The variables considered in this scheme are those thought to be *natural* characteristics of the estuary, in both material and energetic terms, meaning those that influence estuarine processing to varying degrees and are not generally controllable or influenced by either stressor or response variables (Table 3.1). They, and their modifiers, are organized into a simple framework of classes in a way that provides an efficient scheme for storing descriptive data for coastal systems.

The conceptual classification scheme was originally designed to encompass all aquatic habitats in coastal and marine regimes of the United States, from wetlands to the abyssal plains and mid-ocean ridges of the central oceans (Madden et al. 2005). A subset of the original CMECS classification is used here that focuses only on the coastal zone and includes the classes included in near-shore marine, fresh-influenced marine regime, and estuarine regimes. There are 14 types of estuaries in the CMECS classification, but for simplicity they have been grouped into four dominant types: ***Riverine***, ***Coastal Lagoon***, ***Coastal Embayment***, ***and Fjord***, of which all the other types can be considered subsets. From each of those types, individual *L-r* models can be developed and applied to nutrient criteria (Figure 3.2):

Table 3.1. Detailed list of geomorphological, energy and biogeographical descriptors used in classifying estuaries. From Madden et al. 2005.

Bathymetry
Biogeographic region
Biological structure (reef, prop roots)
Circulation
Climate/temperature
Currents
Depth
Energy intensity
Energy type
Estuarine drainage area
Estuary size/area
Estuary volume
Flushing rate
Geomorphology
Habitat components
Hydrology/hydrography
Hypsometry
Salinity regime
Stratification
Substrate
Tides
Turnover time
Water column structure and stability
Wetland/water ratio

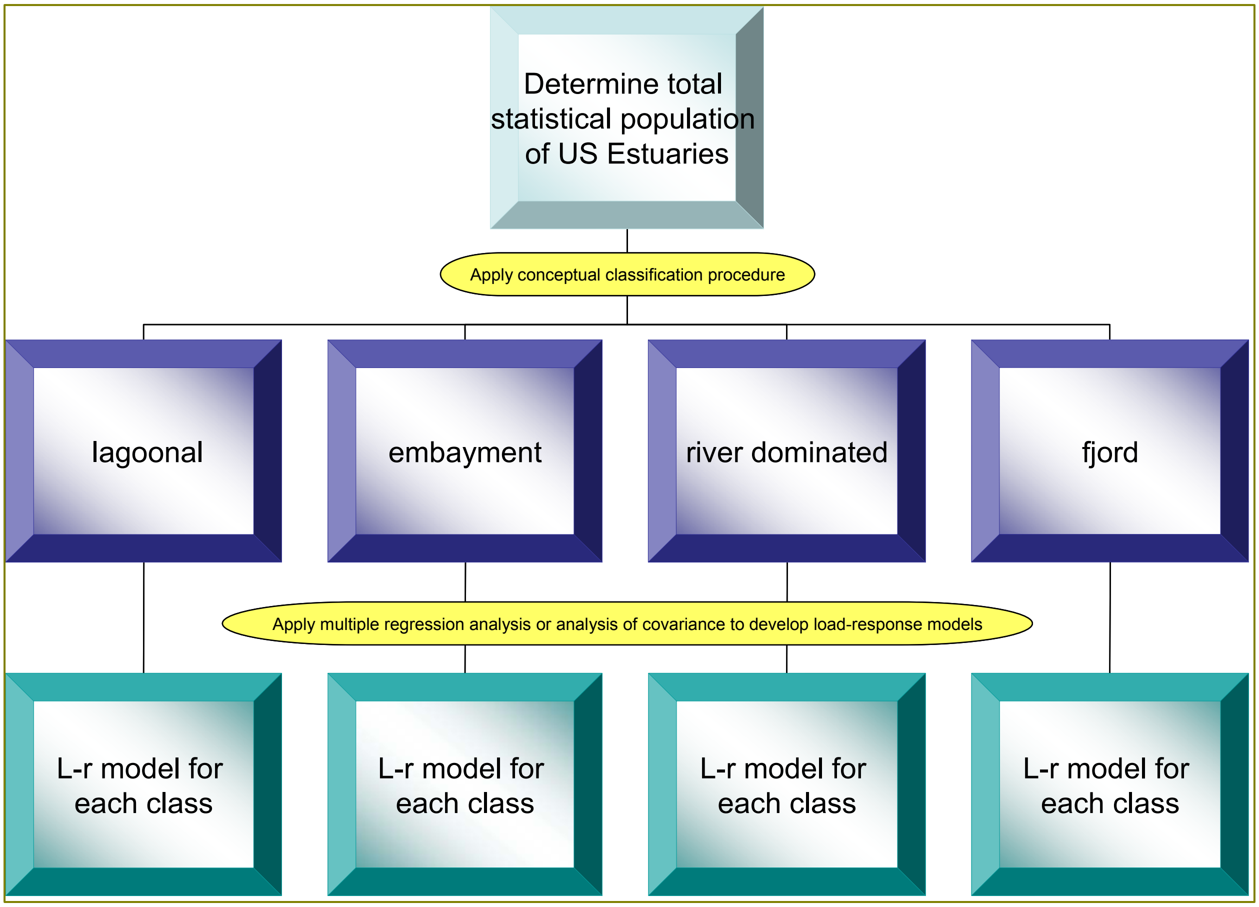

Figure 3.2. Flow chart showing the steps of applying conceptual classification and the development of L-r models for each class of estuary.

1. **Riverine**
    - Drowned river valley
    - Deltaic estuary
    - River channel
    - Salt wedge estuary
    - Tidal fresh marsh

This class of estuary tends to be linear and seasonally turbid, especially in upper reaches, and can be subjected to high current speeds. These estuaries are sedimentary and depositional, and can be associated with a delta, bar or barrier island and other depositional features. These estuaries also tend to be highly flushed with a wide and variable salinity range and seasonally stratified. They have moderate surface to volume ratios with a high watershed to water area ratio and can have very high wetland to water area ratios as well. These estuaries are often characterized by a V-shaped channel configuration and a salt wedge.

2. **Coastal lagoon**
    - Barrier Island estuary
    - Bar-built estuary
    - Lagoon
    - Slough
    - Tidal inlet

This class of estuary tends to be shallow, highly enclosed, with reduced exchange with the ocean, often experiencing high evaporation, and is quiescent in terms of wind, current and wave energy. They tend to have a very high surface to volume ratio, low to moderate watershed to water area ratio, and can have a high wetland to water ratio.

3. **Coastal embayment**
    - Bay
    - Coastal bight
    - Sound

This class of estuary is loosely bounded by landforms, and open to marine exchange, with moderate to high salinities. They are well-flushed, often deep and subject to potentially high-energy input from tides, winds, waves and currents. These estuaries can range from very low to very high in terms of surface area to volume, watershed to water area and wetland to water ratio.

4. **Fjord**

Fjords are deep, seasonally cold-water estuaries with low to moderate riverine inputs and exist at mid to high latitudes. This class of estuary has relatively complex, usually rocky shorelines and bottoms and is partially enclosed sometimes by mountainous landforms, often with a geologic sill formation at the seaward end due to formation by glacial action. The morphology combined with a low exchange of bottom waters with the ocean can result in formation of hypoxic bottom waters. Because of their depth, these estuaries tend to have low surface area to volume ratios. They have moderate watershed to water area ratios and low to moderate wetland to water ratios.

Statistical Clustering Approach to Estuarine Classification

Cluster analysis is a statistical process that identifies classes of objects that are more similar to objects within a single class than to objects in other classes. In the context of classification for this purpose, cluster analysis is performed using independent variables alone, according to the assumption that similarity in driving variables can be used to predict similarity in response. Cluster analysis is an exploratory process. It is not in itself a test of the hypothesis of differences in system response but should be followed by other techniques. For example, discriminant function analysis can be used to determine the subset of independent variables that most reliably predicts membership in each cluster and to define an equation to predict membership class, as well as to estimate a classification error rate. To determine if clusters of systems actually differ statistically in nutrient response relationships, a procedure such as analysis of covariance (ANCOVA) should be performed, with classes included as a categorical factor in the explanatory model. The *L-r* models tested within each class can either be simple (with one independent variable such as total N [TN] concentration) or multiple regression models (with more than one independent variable; Figure 3.3).

Several examples of the statistical clustering approach to estuarine classification are available (e.g., Burgess et al. 2004; Bricker et al. 2007; Engle et al. 2007). Each of the classifications used similar methods to analyze a large number of variables for many estuaries through cluster analysis to arrive at groups of key variables that separate estuaries (Figures 3.4, 3.5). The cluster analysis approaches employed to date (Burgess et al. 2004; Bricker et al. 2007; Engle et al. 2007) used the same spatial units to identify estuaries—the 138 drainage units studied in the National Estuarine Eutrophication Assessment (Bricker et al. 2007). They differ largely in the definition of the variables used and the parameterization of the statistical models. The NOAA approach (Bricker et al. 2007) screened 70 variables and found that 5 were important. EPA's approach (Burgess et al. 2004; Engle et al. 2007), however, used a total of 14 variables, including 2 constructed variables representing combinations of other variables. In both approaches, those variables that had significant relationships with several response variables were grouped to characterize, respectively, 10 and 11 groups representing estuarine types (Table 3.2, Figures 3.4, 3.5).

The variables deemed significant by both groups are informative (Table 3.2). All are physical; some pertain to the relationship of inflow to volume or some proxy of that relationship. They differ in subtle ways, however, on the basis of data availability and ecological assumptions. For example, the approach used by NOAA (Bricker et al. 2007) incorporates data for temperature to give a climat-

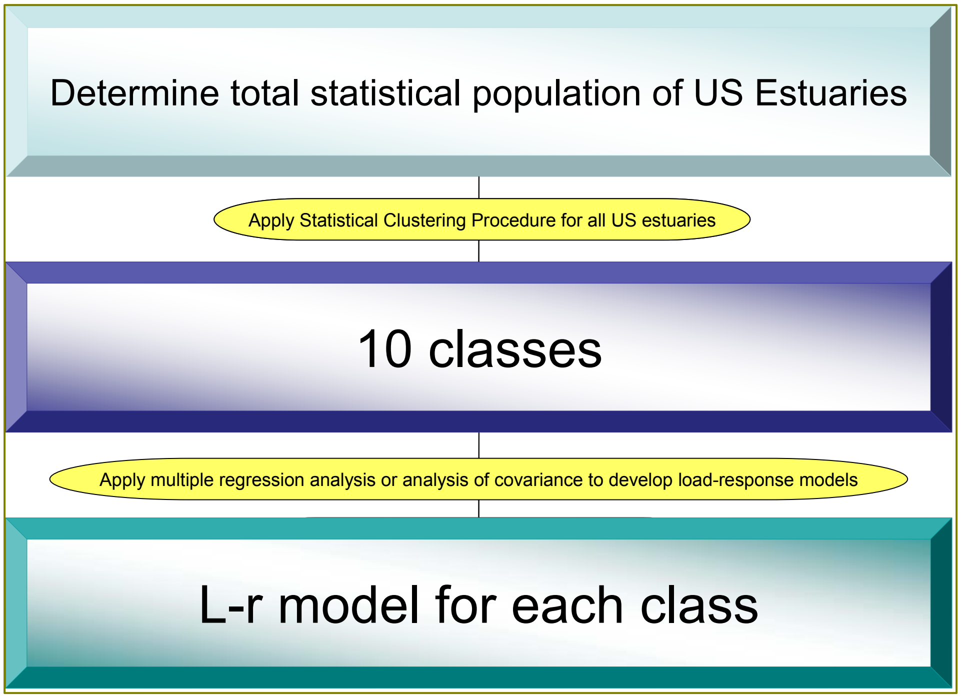

Figure 3.3. Flow chart showing the steps of applying statistical clustering classification and the development of L-r models for each class of estuary.

ic perspective to their classification, whereas that used by EPA (Burgess et al. 2004; Engle et al. 2007) uses data for surface and bottom salinity to provide an indicator of stratification. EPA's latest classification (Engle et al. 2007) incorporates a climatic component, and these two efforts continue to move together to reflect the similar goals of both agencies to classify estuaries on the basis of susceptibility to eutrophication.

The classifying characteristics in the two typologies provide descriptors similar to those in the conceptual classification, in essence statistically confirming the *a priori* assumptions in the conceptual classification approach. For example, the NOAA approach (Bricker et al. 2007) accounted for flow with *fresh inflow*, which corresponds to the Riverine class in the conceptual approach. The EPA approach (Burgess et al. 2004, Engle et al. 2007) used the variable *flow*. For EPA, *size* was important, as was *area*, while for NOAA, the *estuarine area* was used. EPA's classification also used the estuarine drainage area (EDA). It is interesting that EPA found four spatial scaling variables (EDA, size, area, and volume) to be significant. The NOAA approach (Bricker et al. 2007) applied only estuarine area and depth. Depth is accounted for in the conceptual approach by the distinction between lagoon and other types. In NOAA's typology (Bricker et al. 2007), the finding of *openness* to be an

Table 3.2. Comparison of parameters most significant in distinguishing estuary clusters in the EPA and NOAA cluster models.

EPA (Burgess et al. 2004)	NOAA (Bricker et al. 2007)
Estuary area	--
Estuary depth	Estuary depth
Estuary volume	--
Estuary salinity	--
Estuarine drainage area (EDA)	--
Estuary area as fraction of EDA	--
--	Tidal range
--	Ratio of freshwater input to estuary area
--	Mean annual temperature
--	Mouth openness

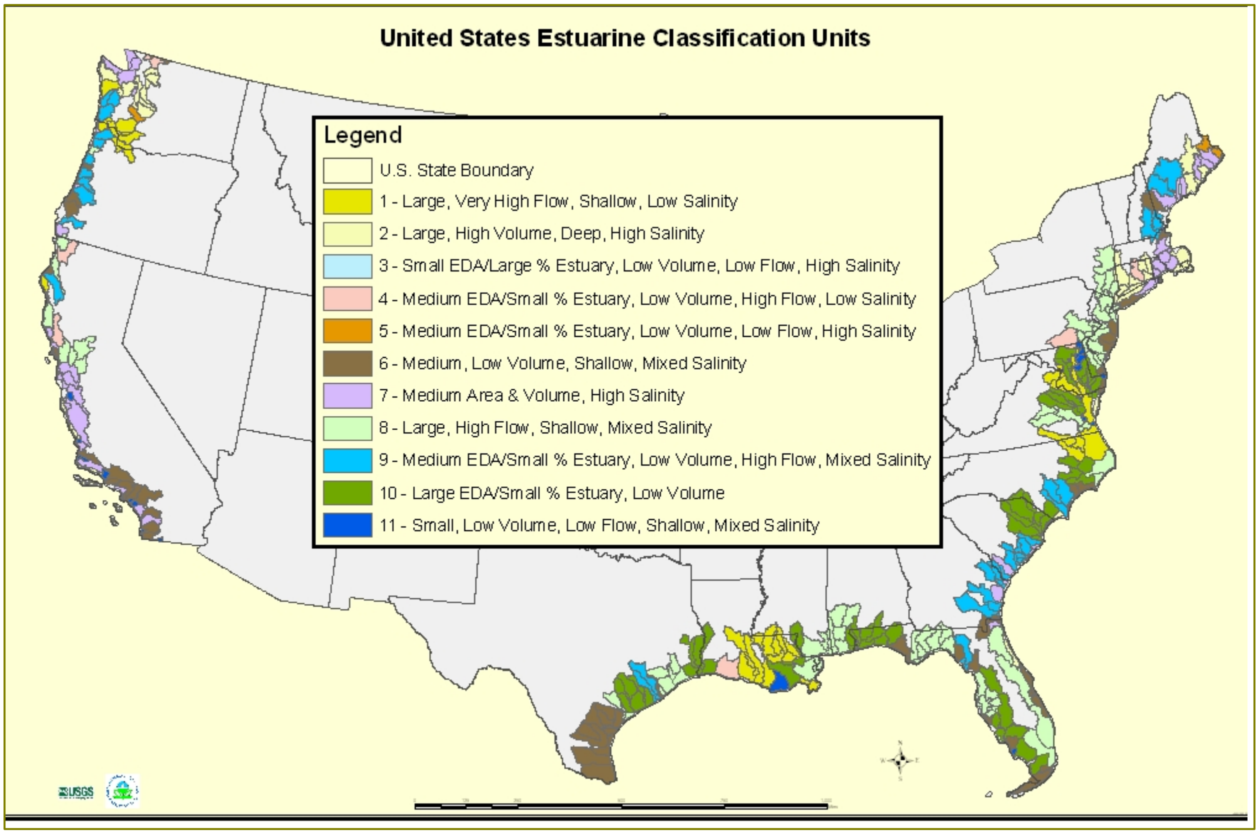

Figure 3.4. Classification of the nation's estuaries based on EPA's typology. From Burgess et al. 2004.

important clustering variable is similar to the conceptual primary type criterion that distinguishes between lagoon and embayment according to degree of openness. Temperature is important in determining rates of biogeochemical processes and is captured in the conceptual approach by the temperature modifier. Interestingly, NOAA (Bricker et al. 2007) found that tidal range was important, which likely acts as a proxy for energy (tide range, frequency) and for flushing (tidal prism).

Statistical B-CART Approach to Estuarine Classification

Statistical B-CART analysis has already been applied to derive lake classes that respond differently to TN and total P (TP) concentrations and is being used to develop a watershed classification system for EPA Region 5 to explain differences in stream and river response to nutrient concentrations (N. Detenbeck, EPA, personal communication). In B-CART classification, both the class membership and model fits are optimized simultaneously. Programs have been written to classify each estuary on the basis of linear regression models (i.e., with a continuous response variable) and on the basis of logistic regression models (i.e., with a categorical response variable; Figure 3.6). Either type of model could be useful. For example, one might want to apply a logistic model to identify probability of impairment on the basis of DO or chlorophyll *a*.

It is clear that the relationship of the volume of the receiving body's waters to the inputs and the retention characteristics of the estuary are key factors in all three classifications examined. Tide, flow and estuarine geomorphology (openness) affect flushing rates and turnover times. Salinity might be related to several effects in the estuary such as flushing, openness, chemistry, stratification and stability. It could be a proxy for biological associations that affect nutrients or chlorophyll *a*, represent a physicochemical control on nutrient processing or be a tracer for the input of fresh/salt water.

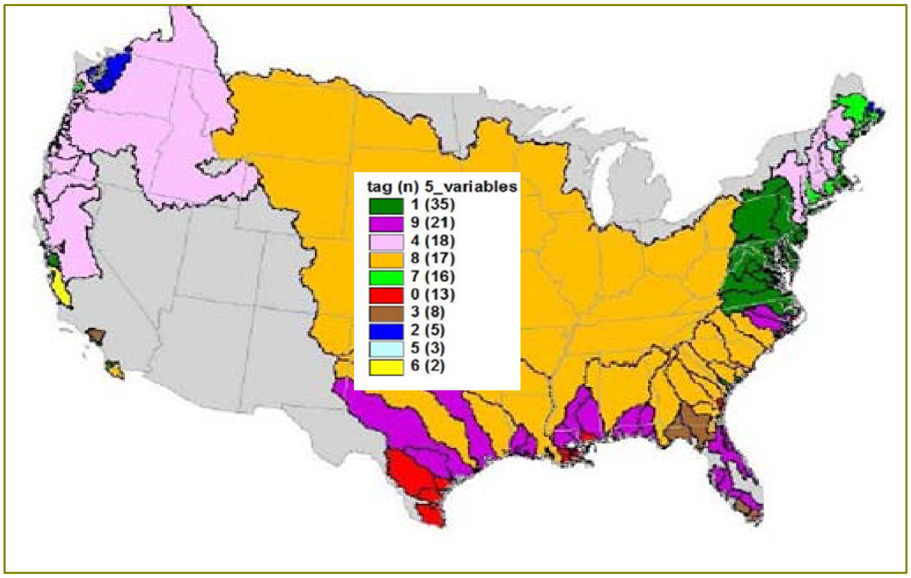

Figure 3.5. Classification of the nation's estuaries based on the NOAA typology. From Bricker et al. 2007.

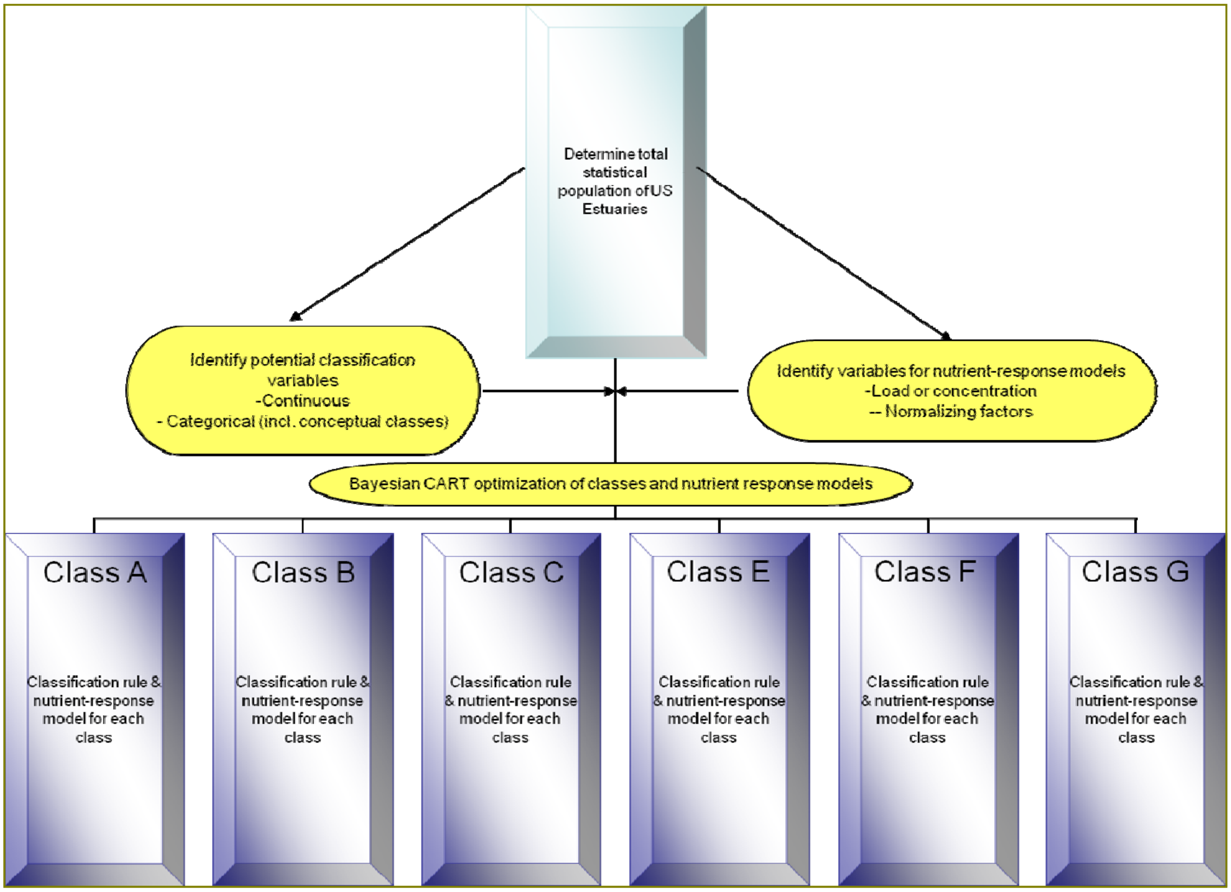

Figure 3.6. Flow chart showing the steps of applying statistical classification (B-CART) and direct testing of different models.

Development, Testing and Application of Load-Response Models from Classification Approaches

Overview

The above sections present three separate classification procedures used to address estuarine classification. The next step is to formulate a process by which those procedures can be combined to create synergy that exceeds the ability of any single analysis to inform us about estuarine response to nutrient inputs.

Dose-response models are a means by which a stressor, or ***causal variable***, can be related to a ***response.*** The examples below compare nutrient loading rates to biological response in terms of chlorophyll *a*. To yield the dose-response relationships, the dose or stressor variables are plotted on the abscissa and are the independent and potentially controllable factors—the allochthonous inputs to and drivers of the system. The selection of the independent (dose) variable will have a strong cascading effect on subsequent decisions about analyses and criteria development. Although TN loading is one factor to use as the independent stressor variable, additional dose and response variables to consider in the regression analysis include

Land use type/intensity
Nutrient concentration
Nutrient species
Nutrient source

The decision about how to apply the dose variable, whether loading or concentration, and how to average it spatially and temporally, will likely bear on all analyses because it needs to be applied uniformly to all estuaries classified.

For some systems, land use may be a useful independent variable. Land use has been used as a proxy for watershed inputs of nutrients to estuaries. For example, it was observed from a study of multiple small estuaries that agricultural land use area and estuarine volume explained nearly 70 percent of the variance in chlorophyll *a* levels (Meeuwig 1999). Land use integrates multiple factors (e.g., population, fertilizer inputs) that determine nutrient loading and can be managed to control these loads. Land use as a causal variable is further discussed in Chapter 4.

Dose-response models can be further enhanced by considering a complex function of nutrient loading as the independent variable, and/or a complex metric or biocriterion as the dependent variable. These dose-response relationships can then be used, along with thresholds for acceptable response, to determine nutrient limits for each estuary. Multivariate models of nutrient stressor variables (e.g., chemical species) versus the estuarine response variables (e.g., algal biomass, DO, SAV abundance) can be performed using the aforementioned conceptual and statistical-cluster classifications.

Water quality and ecological responses are the dependent variables and are influenced by the nutrient input and estuarine processes acting on them. Chlorophyll *a* is probably the most widely sampled response parameter, but other variables to consider include

Autotrophic locus (benthic, water column)
Benthic autotrophy type (macro, micro, mat)
Benthic fauna complex
Benthic productivity
Benthic-pelagic coupling
Hypoxic status
Light characteristics of the water column
Macrophyte composition
Macrophyte type
Net system metabolism
Particulate organic matter
Phytoplankton biomass
Phytoplankton composition
Phytoplankton type
Pigment composition
Total organic carbon
Trophic status

In the application examples described below, the concentration of chlorophyll *a* in the water column is used as the primary response variable. Further analysis of the use of chlorophyll *a* as the response variable, and the application of other response variables in integrated biocriteria, are given in Chapter 4.

Multivariate Regression Analysis of Dose-Response Data

In each case, whether conceptual, statistical-clustering, or statistical B-CART approaches, a dose-response model is built on the null hypothesis that the percent variation in nutrient-response relationships for models incorporating estuarine classes is not greater than the percent variation explained for a nutrient-response relationship with all classes combined. In other words, estuarine classification can have an effect on nutrient-response relationships. In the conceptual and statistical-clustering approaches, the classification is the first step, followed by a separate test of the hypothesis and development of nutrient-response relationships (Figures 3.2 and 3.3). In the third approach (direct statistical classification using B-CART), testing of alternative models is accomplished as part of the classification process (Figure 3.6).

The typical effects of the dependent (dose) variables, as described in a large body of estuarine eutrophication studies in the literature, are nonlinear, and thus a nonlinear form of the regression equation is used to be consistent with mechanistic theory (Thomann and Mueller 1987). The input data for a given indicator variable are assigned values of unity (1) for observations pertaining to that class, and are set to zero for observations pertaining to all other classes. In the regression equation, indicator variables can be associated with any of the continuous variables, or terms, in the equation, and have the effect of allowing those variables or terms to have different values for different classes. A regression coefficient is estimated for each indicator variable, just as for each of the continuous predictor variables in the equation. Thus, for example, a term in the model for the effect of water residence time could take on different values in riverine estuaries as opposed to lagoons or wide coastal embayments.

Indicator variables can also be included to distinguish estuaries in a specific geographical region from those in other regions just as for conceptual classes of estuaries. The addition of indicator variables can only increase the overall goodness-of-fit of the regression (r^2), but there is an important tradeoff involved in adding more indicator variables (i.e., creating a finer classification): the quantity of data used to estimate the coefficients for each class (and associated continuous variable) decreases with the number of classes, and the prediction accuracy of the model (as opposed to r^2) can either increase or decrease as a result. Multiple estuarine classifications can be developed either on the basis of *a priori* concepts or through statistical methods. Alternative classifications then can be evaluated on the basis of their ability to increase accuracy of the dose-response model. Overall, this approach provides flexibility in the way estuarine classification is used in the modeling process.

Preliminary Dose-Response Model Incorporating Estuary Classes

A preliminary regression model was constructed of water column chlorophyll *a* for four classes of estuarine systems as a function of nutrient loading rate and several additional estuarine characteristics. The four estuarine classes were consistent with the conceptual classification scheme and were Riverine, Coastal Lagoons, Coastal Embayments and Fjords. All data for the model were obtained from the National Estuarine Eutrophication Assessment (Bricker et al. 2007), which includes data for 138 estuarine systems. Matched chlorophyll *a* and ancillary data for the preliminary model were available for 108 of the estuarine systems.

The chlorophyll *a* data in the database were derived from satellite-based SeaWiFS (Sea-Viewing Wide Field-of-

Field-of-View Sensor; NASA 2003) imagery rather than direct analysis of water samples. The accuracy of SeaWiFS chlorophyll *a* data as a measure of phytoplankton biomass in estuarine systems is still the focus of research (Harding et al. 2005) and might not be as reliable as that of *in situ* data.

The area-average, maximum monthly mean SeaWiFS chlorophyll *a* value for each estuary was chosen as the measure of estuarine response in this preliminary analysis. Area-average maximum monthly mean chlorophyll *a* values in the database were developed by first averaging chlorophyll *a* observations obtained by the SeaWiFS sensor for each 1.1-km^2 pixel of each estuary during each month over the 7-year period of 1998–2004 (10 to 15 measurements per month), and then averaging over space within each estuary (1 to 7,000 pixels per estuary). Thus, area-average maximum monthly mean values for an estuary were based on approximately 100 to 500,000 point measurements from the sensor, depending on the area of the estuary. A few other response variables, including mean monthly SeaWiFS chlorophyll *a*, were tested in exploratory models but displayed weaker correlation with nutrient loading rate.

TN loading rate (mass per time) is the nutrient *stressor* used in the dose-response model presented here. Other continuous variables included in the model on an *a priori* basis were freshwater residence time, estuarine volume, temperature and algal loss rate. Their formulation was based on Dettmann's (2001) estuarine export and denitrification model.

The regression equation for this analysis was as follows

Equation 1

$$C = \sum_{K=1 \text{ to } K} E_k \left[\beta_k + \beta_{kN} \ln \left\{ [(Nt/V)]\, [1/(1+\alpha t)] \right\} + \beta_{kT} T \right] + E_R \beta_R + \varepsilon$$

where *C* is predicted chlorophyll *a* concentration in mg m^{-3}; *N* is TN loading in g d^{-1}; *t* is freshwater residence time in days; α is an assumed loss rate from settling and denitrification (estimated through trial and error to be .001 per day); *V* is estuarine volume in m^3; *T* is mean annual air temperature in degrees Celsius [°C]; E_k is an indicator variable for a specific class of estuaries; *K* is the number of estuarine classes; β_k, β_{kN}, and β_{kT} are estimated regression coefficients for the effect of estuarine class, the class-specific effect of nutrient loading, and the class-specific effect of temperature, respectively (i.e., a total of 3K coefficients); E_R and β_R are the indicator variable and estimated regression coefficient for the effect of region; and ε is the model error (Table 3.3).

All the regression parameters in Table 3.3 are significant (most are highly so) except for the temperature coefficient for riverine estuaries, which is very small in value. The effect of temperature on chlorophyll *a* in coastal embayments and fjords was found to be insignificant ($p > 0.88$) in several exploratory regressions, and those parameters were eliminated from the model presented in Table 3.3. The r^2 value for the regression was 0.59, and the root mean square error (RMSE) was 4.9, indicating that about two-thirds of the predicted values for the estuaries in the data set were within 4.9 mg per m^3 of their measured values (Figure 3.7).

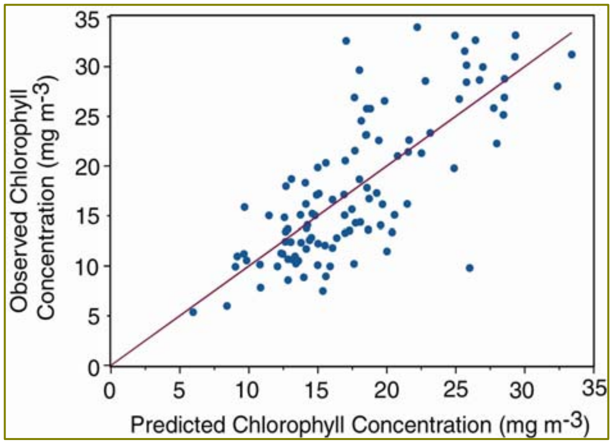

Figure 3.7. Plot of predicted versus measured values for a regression of SeaWiFS maximum monthly chlorophyll a for 108 estuaries. Observed data are from Bricker et al. 2007.

Table 3.3. Parameter estimates and their significance levels for a regression of SeaWiFS chlorophyll a data on estuarine characteristics.

Variable	Parameter estimate	p
Coastal embayment intercept	14.4	< 0.0001
Fjord intercept	25.7	< 0.0001
Lagoon intercept	40.2	< 0.0001
Riverine intercept	22.3	< 0.0001
Effect of N loading in coastal embayments	2.1	0.0036
Effect of N loading in fjords	4.6	0.0002
Effect of N loading in lagoons	1.7	0.0013
Effect of N loading in riverine estuaries	2.0	0.0139
Effect of temperature in coastal embayments	0	NS
Effect of temperature in fjords	0	NS
Effect of temperature in lagoons	–1.3	< 0.0001
Effect of temperature in riverine estuaries	–0.29	0.10
Regional effect (Ches. Bay/Delmarva Peninsula)	9.3	< 0.0001

As a visual guide to the dose-response model described in Equation 1, a plot of predicted chlorophyll *a* concentration versus normalized TN loading rate is presented in Figure 3.8. Normalized loading rate has the units of concentration, and as discussed above, is a function of N mass loading rate, freshwater residence time, estuarine volume, and N loss rate within the estuary.

The accuracy of the model for predicting chlorophyll *a* levels in estuaries that are not in the data set, which is the intended application of the dose-response model, can be somewhat lower than the estimated model value because the uncertainty of the parameter estimates must be accounted for and because the intention is to use the model to estimate chlorophyll *a* levels under different (and changing) nutrient inputs. A useful format for applying the model to make use of the statistical error distribution surrounding model predictions for estuaries having little or no available chlorophyll *a* data and to examine how the probability of exceeding various potential chlorophyll *a* criteria varies with changing hypothetical nutrient input rates. The use of the prediction error allows managers developing criteria to include the reliability level of the model in the decision making on where to set the criteria. For a given loading rate in an estuary with specific characteristics, the predicted likelihood of exceeding the criteria will be lower (and the possibility that the actual chlorophyll *a* lies below the threshold will be higher) if the model prediction error is large compared to a more accurate model.

An example application of the model developed here to a hypothetical fjord with physical characteristics similar to Penobscot Bay is shown in Figure 3.9. The predicted maximum monthly chlorophyll *a* level for that system under the current loading rate is 21 mg m^{-3}. In setting criteria for N loading rate in this example system, water resource managers could use the model to estimate the

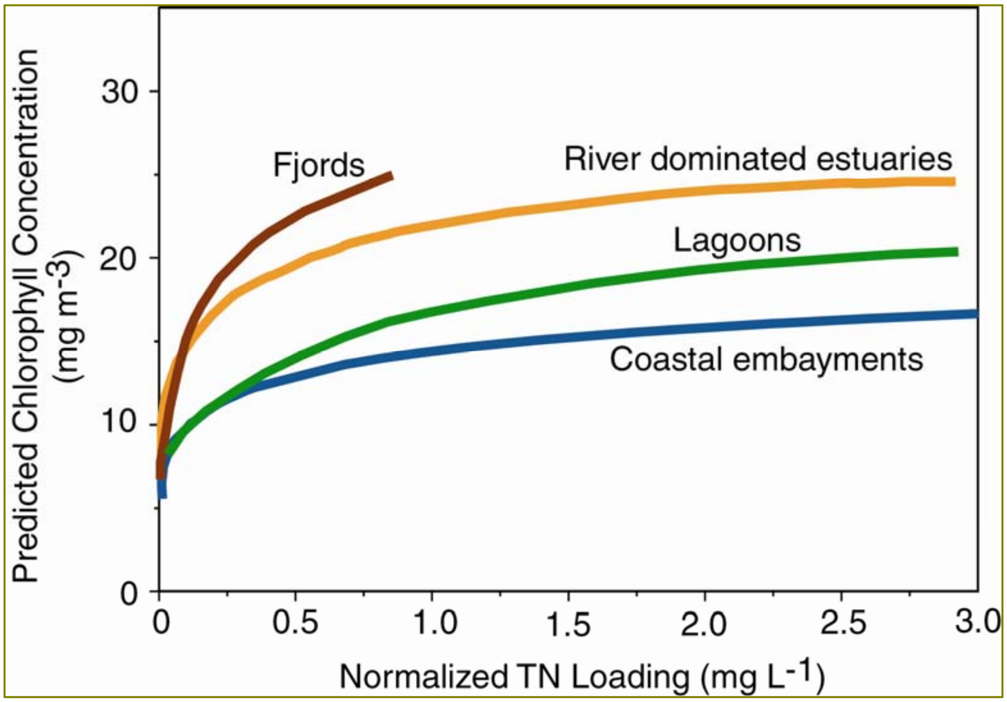

Figure 3.8. Plot of predicted SeaWiFS maximum monthly chlorophyll a versus normalized TN loading rate. The temperature for lagoons and riverine estuaries was arbitrarily set at 20 °C. Estimated temperature coefficients for fjords and coastal embayments did not differ significantly from zero.

likelihood of reducing predicted maximum monthly chlorophyll *a* concentration to selected benchmark levels (shown as C* in Figure 3.9) as a function of alternative limits on loading rate. As a preliminary step, it is worth noting that, because of the error bounds of the dose-response model, there is a 14 percent chance that the true maximum monthly chlorophyll *a* in the system is actually less than 15 mg m^{-3} under the current loading rate (rather than the predicted 21 mg m^{-3}), and a 27 percent chance that the true chlorophyll *a* level is above 25 mg m^{-3}. Next, the model can be used to estimate effects of lowering the N loading rate to, for example, 4.2 x 10^8 moles per year (= 5.9 x 10^6 kg y^{-1}; a 40 percent reduction). Under those conditions, the predicted maximum monthly chlorophyll *a* level decreases to 19 mg m^{-3}, with a 25 percent chance the true value would fall below 15 mg m^{-3}.

Inspection of Figure 3.9 indicates that, in general, the current loading rate is fairly high on the curve of chlorophyll *a* versus loading rate, which suggests that achieving a significant reduction in predicted chlorophyll *a* requires a large reduction in loading. Moreover, because of the model error bounds, large reductions in loading might be required to reduce the *risk* that reductions would fail to meet specific chlorophyll *a* goals. In the present example, the model indicates that a load reduction of 75 percent would be required to achieve a greater than 50 percent chance that the maximum monthly chlorophyll *a* level would decline to less than 15 mg m^{-3}. It is, of course, important to note that these dose-response relationship models are preliminary, and more work is needed on additional systems before they should be applied.

In Summary

As shown above, the regression analysis step in the criteria development process is to develop dose-response relationships between variables associated with nutrients as stressors (e.g., loading rates, chemical form) and the estuarine response variables associated with the underlying objectives of setting nutrient criteria (e.g., algal biomass, DO). Depending on the level of success of the regressions, such models might provide empirical evidence of the value of establishing criteria and could be especially useful in setting preliminary criteria for estuar-

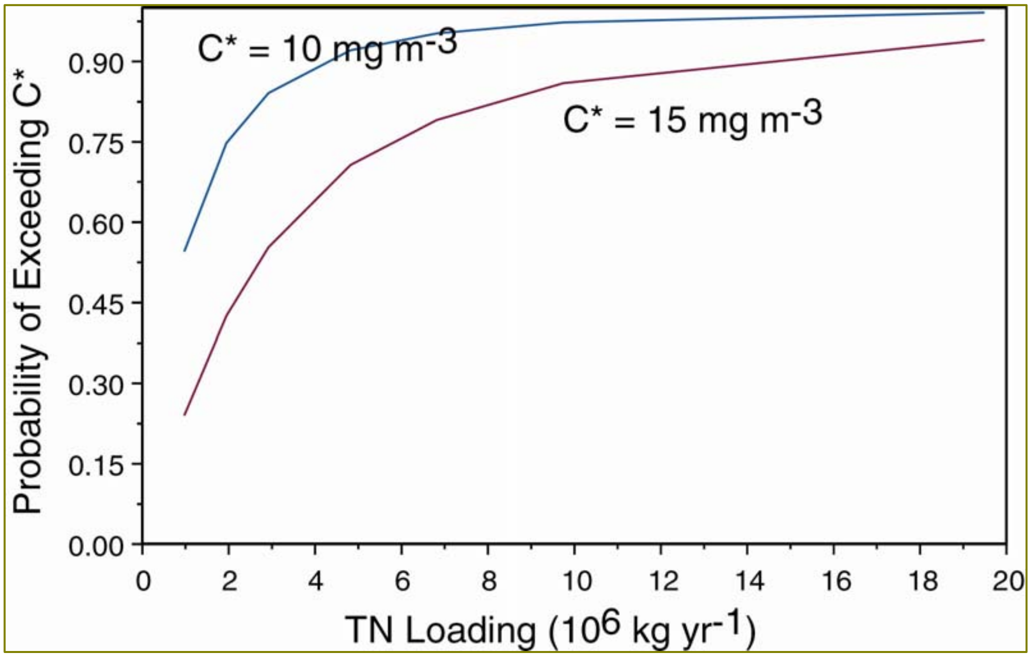

Figure 3.9. An example application of the dose-response model developed here to an estuary similar to Penobscot Bay, a fjord. The estimated probabilities of exceeding two hypothetical summer maximum chlorophyll **a** ***thresholds (C*) are shown as a function of TN loading rate. The current TN loading rate to Penobscot Bay is 9.7 x 10^6 kg y^{-1}. The volume is 2.44x10^{10} m^3; the mean freshwater residence time is 649 days.***

ies with little or no observational data on past response to nutrient enrichment. Confidence in the transfer value of these models to data-poor estuaries might be enhanced by including data from a large number and variety of estuarine systems in the model building. Wide variation in the physical and biogeochemical characteristics of the estuaries represented in the data set, however, will increase the complexity of the model required to predict estuarine response to nutrient enrichment. To some degree, the influence of estuary-to-estuary differences in physical characteristics on biogeochemical response can be accounted for by developing multivariate regression equations containing continuous (normalizing) variables (i.e., volume, tidal and freshwater flushing) known to be relevant to the effects of nutrient loading on estuaries. The effects of those variables, as described in a large body of estuarine eutrophication studies in the literature, are often nonlinear, and nonlinear regression methods might be required to relate the regression equations to mechanistic theory.

Including an estuarine classification step in the criteria development process can be thought of as an extension of the approach described above; that is, it is also likely to be useful to identify and incorporate noncontinuous variables for discrete classes or types of estuaries in the dose-response model to successfully account for the large diversity of estuarine systems in the United States. It might appear that the simplest way of accommodating different estuarine classes would be to develop entirely separate regression models for each class. That has the disadvantage, however, of reducing the amount of data used to calibrate each model compared to a method that incorporates variables representing the different estuarine classes in a single model.

More examples of dose-response relationships and a discussion of their complexities and factors to consider in applying those relationships are given in Chapter 4. A series of case studies is presented in Chapter 5 for further appreciation of the range of responses to nutrients in estuaries and the extent to which the responses vary between and within estuarine types.

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Chapter 4:

A Framework for Developing Nutrient Criteria

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Figure 4.1. One of the objectives in establishing nutrient criteria is to maintain water quality for multiple uses. Top photo by J. Hawkey; bottom photo by H. Lane. www.ian.umces.edu. Right photo: www.inshoreslam.com/.

Nationally, many states are in the process of developing numeric nutrient criteria for their waters; nutrient criteria for estuaries are one part of this process. Few states have criteria in place for their estuarine waters because of the lack of historical data and the wide range of responses in estuaries of differing types or degree of impairment. The topic of the complexity of nutrient enrichment in estuaries is described in Chapter 2. The issue of nutrient enrichment in coastal waters is both simple, in the sense that a range of adverse effects can generally be associated with increased inputs of nutrients, and extremely complex, because the effects that occur and the magnitude of the effects depend on factors that vary significantly among estuaries.

In this chapter, the issues of estuarine complexity are reviewed as they relate to developing estuarine nutrient criteria, and various variables in criteria application are described. This chapter begins with a synthesis of the case studies, which are presented in detail in Chapter 5. This synthesis of these case studies, along with use of a database, developed as part of this workgroup effort (Appendix I), provide support for the notion that much can be learned from comparative analyses and the transfer of information from one estuarine system to others of similar type. This chapter then describes ***causal and response variables*** and how those variables can be applied in nutrient criteria development. While the relationships are initially discussed using nutrients as the causal variable and chlorophyll *a* as the response variable, examples are also given for how integrated biocriteria, which include variables beyond chlorophyll *a,* can be developed. Last, the questions of how to establish ***reference conditions*** and how to evaluate ***criteria exceedances*** are also discussed.

Complexity of Nutrient Enrichment in Estuaries: Synthesis of Detailed Case Studies

For some systems, only rudimentary data on the inputs and pathways of nutrient processes are available. For some others, the complexities of how nutrients are processed within estuaries, and how those responses translate into expressions of eutrophication, have been parameterized into models of ecosystem function that range from simple to sophisticated (e.g., Figure 4.2). Available data could determine whether simplified or sophisticated approaches can be undertaken for individual systems.

Results of the analysis of estuarine typology (Chapter 3) suggest that there are strong commonalities among estuarine systems and that those commonalities could be useful in deriving numeric nutrient criteria for estuaries. Those analyses also underscore that estuarine systems

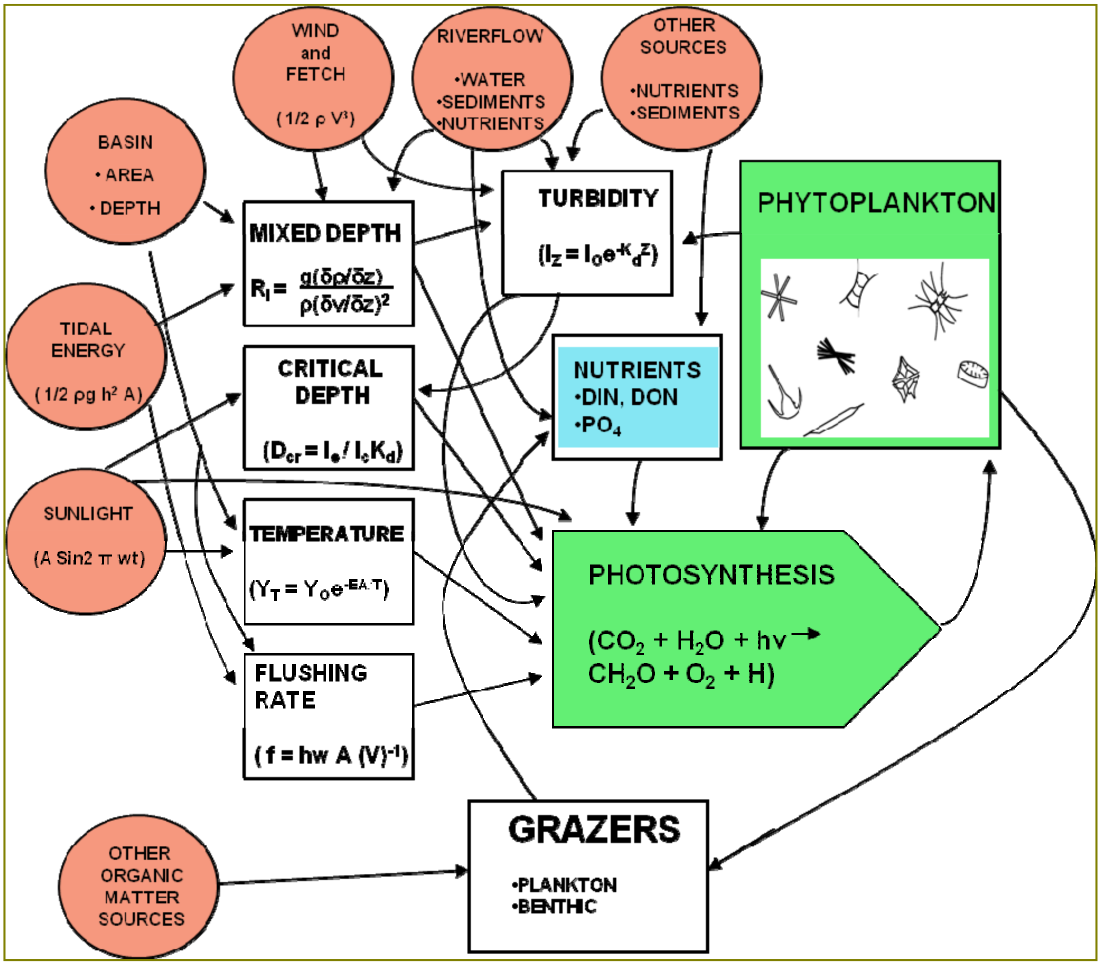


Figure 4.2. A simple box and arrow diagram showing factors (circles) and mechanisms (white boxes) commonly influencing rates (green arrowhead) and biomass (green box) of estuarine phytoplankton. Influencing factors and mechanisms can vary widely among estuarine ecosystems. From W. Boynton (unpublished).

are complex and variable. Conceptually, the classes of estuaries are useful in dividing types. If the relationship between nutrient dose and biotic response were simply a mass balance transfer of one form into the other, predicting the result of loading would require only a simple calculation. However, because there are multiple potential pathways for nutrients and the strength of these alternative paths is dependent on numerous environmental factors, the task of predicting the effects of nutrient loading is more complicated.

The case studies described in detail in Chapter 5 are developed from systems with a wide range of nutrient loadings, sources and resultant concentrations (Table 4.1). The case studies included six that can be classified as ***River-dominated*** (Narragansett Bay, Delaware Bay, Chesapeake Bay, Neuse River Estuary, San Francisco Bay and Yaquina Bay), and four that can be classified as ***Lagoonal*** (Barnegat Bay–Little Egg Harbor, the Coastal Bays, Florida Bay and Pensacola Bay). The Delaware Bay has very high average total dissolved inorganic nitrogen (DIN), with levels in the freshwater upper estuary that exceed 150 μM-N year-round. Both the Chesapeake and San Francisco bays also have very high DIN concentrations in their upper reaches, but the high concentrations tend to be more seasonal than those of the Delaware Bay. At the other extreme, Pensacola Bay and Barnegat Bay–Little Egg Harbor tend to have average DIN concentrations < 5 μM-N, and Florida Bay has levels that generally are even lower. Concentrations of PO_4^{-3} are also quite different among the estuaries, and the patterns are also different from those of DIN. The San Francisco Bay system has the highest average PO_4^{-3} (3 to 5 μM-P), while the Chesapeake Bay, Coastal Bays, and Florida Bay have relative low PO_4^{-3} compared to the San Francisco Bay, Delaware Bay, Narragansett Bay and Neuse River Estuary. Nutrient sources in the Chesapeake Bay, Delaware Bay, San Francisco Bay, Neuse River Estuary and Narragansett Bay have strong influence from anthropogenic sources, and they all have human populations in their drainage basins exceeding one million. On the other hand, Pensacola Bay and Yaquina

Table 4.1. Summary of dominant nutrient form and approximate average chlorophyll a concentration in each of the case studies presented. Bloom chlorophyll a estimates exclude extreme events.

Estuarine system	**Classification**	**Nitrogen load (gN m^{-2} y^{-1})**	**Approx. max NO_3^- (mM N)**	**Dominant form of DIN (surface)**	**Approx. max PO_4^{-3} (mM-P)**	**Avg bloom chlorophyll* (mg L^{-1})**
Narragansett Bay	River-Dominated	28	20	NO_3^- and NH_4^+	4	10
Delaware Bay	River-Dominated	26	175	NO_3^- and NH_4^+	6	30
Chesapeake Bay	River-Dominated	21	100	NO_3^-	1.5	50–60
Neuse River	River-Dominated		300	NO_3^-	2	30
San Francisco Bay	River-Dominated	29	50	NO_3^-	4	6
Yaquina Bay	River-Dominated	100	100	NO_3^-	3	10
Barnegat Bay	Lagoonal	5	20	NH_4^+	< 1	14
Coastal Bays	Lagoonal	2–4	< 5	NH_4^+	< 0.5	20
Florida Bay	Lagoonal	10	10	NH_4^+	< 1	8
Pensacola Bay	Lagoonal	14	14	NO_3^-	< 0.5	10

Bay appear to have minimal nutrient loading. Delaware and San Francisco estuaries, even with their high nutrient enrichment are considered to have little classical symptoms of eutrophication.

Light attenuation on the estuaries also differs, and for various reasons. The Delaware, San Francisco and Chesapeake bays receive considerable suspended sediment loads in the upper reaches of their respective estuaries, leading to light attention and potential light limita-

Table 4.2. Summary of the dominant forms of HABs and their primary mode of impact in the case studies presented.

Estuarine system	Classification	Dominant phytoplankton groups	Common or dominant HAB	Primary mode of HAB impact
Narragansett Bay	River-Dominated	Diatom blooms (spring-summer); Dinoflagellates and raphidophytes (summer)	*Heterosigma akashiwo,* *Prorocentrum minimum,* *Aureococcus anophagefferens*, other diverse dinoflagellates	Variable; Mostly high biomass, Some ichthyotoxic
Delaware Bay	River-Dominated	Diatoms (winter –spring); Mixed (summer)	uncommon	
Chesapeake Bay	River-Dominated	Diatoms (winter-spring); Mixed flagellates (summer)	*Karlodinium veneficum,* *Prorocentrum minimum,* *Pfiesteria* sp., others *Microcystis* in tributaries	High biomass and toxic
Neuse River	River-Dominated	Mixed diatom, dinoflagellate and cyanobacteria community	*Heterosigma akashiwo*	High biomass and toxic
San Francisco Bay	River-Dominated	Diatoms; dinoflagellate and cyanobacteria (summer)	*Heterosigma akashiwo,* *Alexandrium* sp., *Akashiwo sanguinium,* *Myrionecta rubra*	High biomass and toxic
Yaquina Bay	River-Dominated	Diatoms (spring-fall); dinoflagellate and cyanobacteria (summer)	*Myrionecta rubra*	High biomass
Barnegat Bay	Lagoonal	Diatoms (winter); Pelagophytes, cyanobacteria and flagellates (summer)	*Prorocentrum minimum*, *Aureococcus Anophagefferens*	High biomass
Coastal Bays	Lagoonal	Diatoms (winter); Pelagophytes, cyanobacteria and flagellates (summer)	*Aureococcus anophagefferens*, raphidophytes, *Microcystis*	High biomass
Florida Bay	Lagoonal	Diatoms; Picocyanobacteria (summer)	*Synechococcus* sp., *Pyrodinium* sp.	High biomass and toxic
Pensacola Bay	Lagoonal	Diatoms and cyanobacteria	*Myrionecta rubra*	High biomass

tion of algal growth. In Pensacola Bay, on the other hand, colored dissolved organic matter (CDOM) is the major contributor to light attenuation, and it contributes to light attenuation in the upper Neuse River Estuary, Delaware Bay and Coastal Bays. Algal biomass is the major contributor to light attenuation in the mid and lower Chesapeake Bay, Barnegat Bay–Little Egg Harbor, Coastal Bays and the mid and lower Neuse Estuary.

In spite of the wide range of physical attributes, nutrient loading, and highly complex expressions of eutrophication in the case studies presented, a number of commonalities emerge according to estuarine typology (Figures 4.3, 4.4, Tables 4.1, 4.2). Recognizing that, even within classifications, not all regions of each estuary fit the classification and can show characteristics of other classes, the following distinctive attributes appear to differentiate the river dominated from the lagoonal systems: nutrient loading rate, maximum NO_3^- and PO_4^{-3} concentration levels, dominant N form, size spectrum and taxonomic composition of chlorophyll *a* during blooms, time scale of hypoxia and extent of seagrass coverage. Each of these characteristics is summarized below, and, using data from the Estuarine Nutrient Criteria Database Query Tool (Appendix I), another estuary from each class is assessed to determine if the general patterns hold.

Nutrient loading rate. Intuitively, the nutrient loading rate for riverine-dominated systems should be higher than that of the lagoonal systems because rivers are a major source of nutrient loads. Indeed, the data from the case studies support that notion. Of the systems described here, for example, N loading exceeded 20 gN m^{-2} y^{-1} for the river-dominated systems but was < 15 gN m^{-2} y^{-1} for the lagoonal systems, and in the case of the Barnegat Bay–Little Egg Harbor and Coastal Bays N loading was < 5 gN m^{-2} yr^{-1}. Interestingly, Duarte and Agusti (1998) suggested that 5 gN m^{-2} y^{-1} is a *balance point* above which eutrophication responses become more pronounced. Nutrient loading rates for coastal lagoons are also more difficult to quantify because they are generally dominated by nonpoint sources, such as runoff and groundwater input. Thus, they are also more likely to be underestimated.

Maximum levels of NO_3^- and PO_4^{-3}. In all the case studies presented, the maximum nutrient concentrations reported in the synopses for the river-dominated systems were significantly higher than those of the lagoonal systems. Maximum concentrations of NO_3^- were generally in the range of 50–300 μM-N for the river dominated systems, while they were generally in the range of < 10 μM-N for the lagoonal systems. Similarly for PO_4^{-3}, maximum reported PO_4^{-3} concentrations range from ~1.5 to 6.0 μM-P for the riverine systems but were < 1.5 μM-P for the lagoonal systems. High potential for PO_4^{-3} scavenging and adsorption by bottom sediments in the lagoonal systems likely explains the sustained low PO_4^{-3}.

Dominant Inorganic N form. River-dominated systems had, at least on a seasonal basis, higher concentrations of NO_3^- than the lagoonal systems. It is, however, recognized that seasonality is a significant factor when considering the dominant nitrogen form in all these systems. For example, NO_3^- generally tends to become depleted with the progression of a winter/spring bloom, and NH_4^+ increases as these blooms begin to decline and nutrients are regenerated. The Delaware Estuary is an exception in this regard, with higher availability of NH_4^+ in the spring and higher concentrations of NO_3^- in the summer. In the lagoonal systems in all the case studies presented, NH_4^+ was generally the dominant N form relative to NO_3^- throughout the year. In the lagoonal systems, the lack of riverine input, combined with their more shallow nature, likely increases the importance of benthic nutrient regeneration, which in turn can lead to significant accumulations of the reduced form of inorganic nitrogen, NH_4^+.

Size spectrum and taxonomic composition of bloom chlorophyll. Of the case studies presented, the general pattern in phytoplankton community composition was the development of a more significant winter/spring diatom

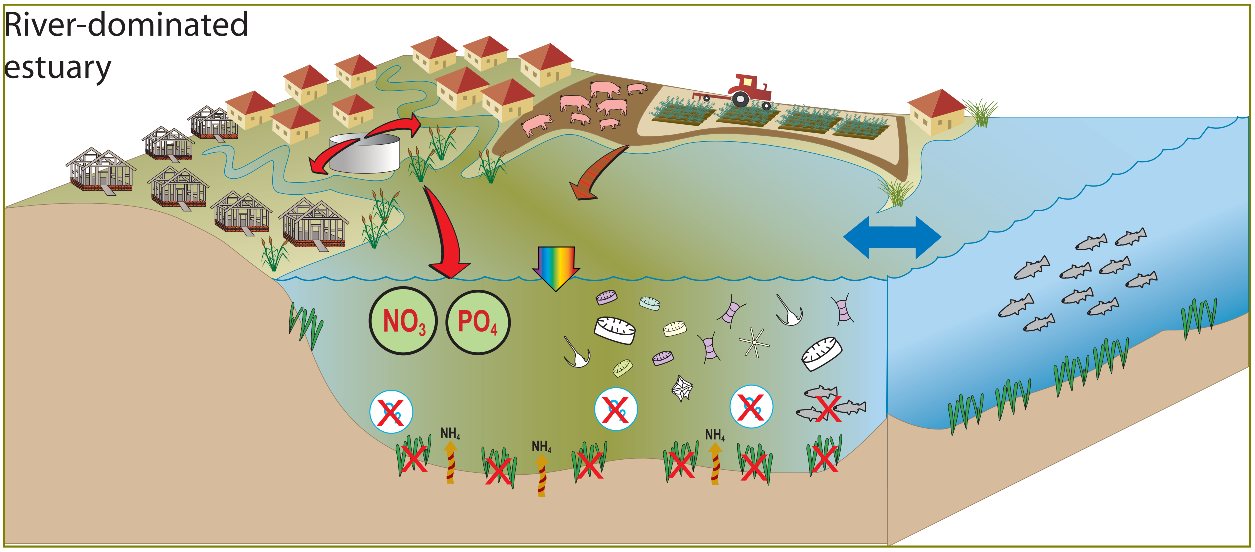


Figure 4.3. In river-dominated systems, the major source of nutrients is from riverine input. Most of the N from this source is in the form of nitrate (NO_3^-). The major sources of nutrients are sewage from the human population and runoff and groundwater input from the agricultural and animal operations in the watershed. Diatoms are common in the resulting phytoplankton community. Nutrient regeneration in the benthos can be significant especially when hypoxia or anoxia develops.

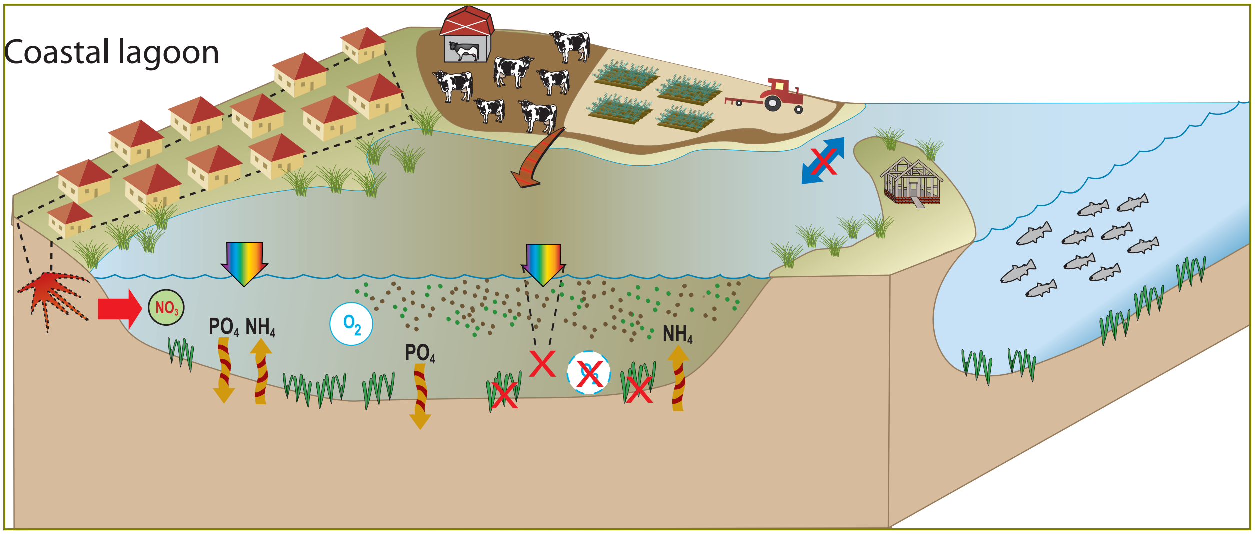


Figure 4.4. In coastal lagoons, the major source of nutrient input is from nonpoint sources. Those include sewage from the human population, often septic rather than treated sewage, and runoff and groundwater input from the agricultural and animal operations in the watershed. Although variable, the dominant N form can be ammonium (NH_4^+). Phytoplankton are often dominated by cells that are small, such as cyanobacteria or brown tide. Nutrient fluxes from the benthos are important in regulating the nutrient availability in these systems.

bloom in the river-dominated systems than in the lagoonal systems. The extent of development of HABs also differs between these systems. HABs can cause harm to the ecosystem either through the development of high biomass, in turn leading to light reduction, and/or hypoxia, or through the production of toxins that can intoxicate shellfish or possibly kill fish (Landsberg 2002; Glibert et al. 2005a, 2005b; Backer and McGillicudy 2006). Some of the estuaries included in the case study analysis, including Delaware Bay, Pensacola Bay, and

Yaquina Estuary, do not have chronic HAB problems. If and when present, such blooms can lead to discolored water, such as in the case of *Myrionecta rubra* blooms that cause red water but do not cause toxicity or significant alterations to the ecosystems (Table 4.2). On the other hand, other systems have chronic HAB problems. The Chesapeake Bay has increasing blooms of HAB species that can develop large biomass accumulations, and, in some cases, toxicity and fish kills. High biomass HAB flagellates, typically > 10 µm, such as *Prorocentrum minimum* and *Heterosigma akashiwo*, are more prevalent in summer in the riverine dominated systems following the collapse of the diatom-dominated spring blooms. Also, in general, the HAB blooms in the lagoons tended to be dominated by phytoplankton that are < 3 µm in size, such as *Synechococcus* (Florida Bay), and *Aureococcus* (Coastal Bays and Barnegat Bay-Little Egg Harbor). *Aureococcus* has been previously observed in Narragansett Bay, but it appears to have been an episodic event and not witnessed for many years. In the Coastal Bays, Barnegat Bay–Little Egg Harbor, and Narragansett Bay during earlier years, blooms of the brown tide species *Aureococcus anophagefferens* are, or were, common and have caused significant negative effects on benthic bivalve communities. In fact, in the Coastal Bays the blooms have increased in magnitude every year for at least a decade (Glibert et al. 2007). Florida Bay has large cyanobacteria blooms. Interestingly, both *Aureococcus* and *Synechococcus* have been found to prefer forms of N that are organic or in reduced form, and generally show negative relationships with availability of the oxidized forms of N, such as NO_3^- (e.g., Berg et al. 1997; Glibert et al. 2004, 2007). Sustained blooms of such organisms that generally are of a size class < 3 µm have been termed *ecosystem disruptive algal blooms, EDABS* and are more common in shallow lagoons where they can be sustained on regenerated nutrients that do not accumulate in the water column (Sunda et al. 2006). Where dinoflagellates are observed in the lagoonal systems, they are either generally associated with the benthos, as in *Pyrocystis* in Florida Bay, or localized to specific subsegments of the estuary, as in the case of *Pfiesteria* sp. in the Coastal Bays. Conversely, *Synechococcus* has been shown to bloom in the riverine-dominated Neuse Estuary and Chesapeake Bay but only during the summer when riverine flow is low and the residence time increases accordingly.

Time scale of hypoxia. In terms of hypoxia, the Chesapeake Bay and Neuse River Estuary have major, sustained, bottom-water hypoxia attributed to excess biomass from seasonal nutrient enrichment. Although not all of the riverine systems described here have significant problems with hypoxia, those that do tend to have sustained seasonal hypoxia/anoxia develop during the summer months following the collapse of the spring diatom bloom. Delaware and San Francisco bays, with their high nutrient loadings, have relatively few current problems with hypoxia. Interestingly, both of those systems had serious hypoxia in the past because of high organic matter loading and resultant BOD from sewage discharges. In contrast, most of the lagoonal systems examined here tend to have far fewer seasonal problems with hypoxia, Pensacola Bay being the exception to that pattern. Hypoxia in the lagoon systems appears to be more variable and episodic and in general appears to develop and dissipate on a diel scale, if and when it occurs. Documentation of hypoxia is not as common in lagoons unless *in situ* instrumentation is available to observe the diel fluctuations; individual measurements might not be sufficient to capture this variability. Thus, in Florida Bay, Coastal Bays and Barnegat Bay–Little Egg Harbor, only limited hypoxia is apparent and generally not sustained on a seasonal basis. Yaquina Estuary has summer hypoxia near its mouth, but it is considered to be the result of upwelled water from offshore.

Benthic primary producers. Because of the generally deeper nature of the riverine systems, the SAV coverage in these systems is significantly less than in the lagoonal systems. Except along the fringes and in specific tributaries, SAV coverage is less than 20 percent of the bottom

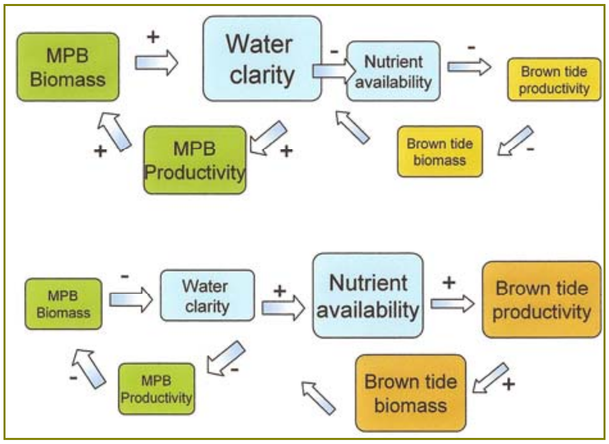


Figure 4.5. Conceptual relationships between microphytobenthos (MPB) and blooms of* Aureococcus anophagefferens *(brown tide). Note that in the upper panel, a benthic- dominated state can lead to low brown tide, while in the lower panel, pelagic nutrient enrichment can lead to high brown tide biomass. Modified and redrawn from MacIntyre et al. 2004.

in riverine-dominated systems, but in lagoons, SAV coverage can exceed 70 percent in some regions. Furthermore, in lagoonal systems, benthic chlorophyll *a* can be a significant contributor to production in the system, and the presence of significant benthic primary producers, in turn, helps to sustain DO levels in the water column. Large decreases in SAV coverage have been documented in Chesapeake Bay, Pensacola Bay and Barnegat Bay–Little Egg Harbor and attributed to general increases in nutrient loading. Furthermore, these changes have often been accompanied by increases in the growth of epiphytic algae and macroalgal proliferation. Yaquina Estuary, on the other hand, has macroalgal proliferations, but they have been attributed to coastal upwelling rather than to eutrophication. Florida Bay also has experienced large die-offs of SAV, and they have been attributed to multiple stressors, such as high temperature, salinity and sulfide, in addition to nutrient increases and to cyanobacterial blooms. In the examples given, Pensacola Bay could be the exception because of its somewhat deeper water column, and progressive nutrient loading has led to declines in SAV abundance over the past several decades.

A conceptual model linking the importance of benthic primary producers to the development of brown tide in coastal lagoons has been proposed for the Long Island embayments (MacIntyre et al. 2004; Figure 4.5). Conditions leading to a benthic dominated state will likely lead to low brown tide biomass, while conditions leading to pelagic nutrient enrichment will be more likely to be suitable to sustain brown tide. The lagoonal systems described here that are affected by brown tide (Barnegat Bay–Little Egg Harbor, Coastal Bays) suggest that similar mechanisms could be operating.

These case studies also provide several examples of ecosystem response to change and underscore that both natural and anthropogenic changes can significantly alter the processing of nutrients. One type of significant perturbation that has substantially altered ecosystems and the expression of eutrophication is hurricanes. The Neuse River Estuary experienced a series of hurricanes from 1996 to 2000, as did Florida in 2005. In the Neuse, those events triggered alterations in phytoplankton production and community structure, leading to a system with increasing numbers of coccoids and flagellates, increased frequencies, expansion and duration of HABs, increased hypoxia and anoxia and stress on finfish and shellfish species among other habitat and water quality changes (Burkholder et al. 2006; Paerl et al. 2006). In the lagoonal Florida Bay, the ecosystem response to the hurricane events and other activities of the summer of 2005 was also an increase in algal blooms, decreases in water clarity, and loss of SAV. Thus, these examples suggest that natural alterations of flow from hurricanes can lead to long-term changes.

The case studies also provide other examples of system alterations that are derived from direct human activities. The Delaware, for example, responded to the ban on PO_4^{-3} in the 1970s, and as a consequence, increasing P limitation was observed and the frequency of algal blooms declined. In the San Francisco Bay, sewage treatment changes have led to an increase in the availability of NH_4^+ relative to NO_3^-, and rates of primary production, and phytoplankton species composition have been altered. Both of those cases are examples where productivity and algal biomass declined in response. However, in San Francisco Bay, the algal biomass declines have been accompanied by shifts to less desirable species in terms of supporting higher trophic food webs. Other significant alterations, such as species introductions, alterations of top-down control, such as increased fishing pressure, and, as in the case of Yaquina Estuary, alterations in use of the river for other purposes, such as logging, are factors that also contribute to long-term ecosystem function.

The estuaries that have been described in detail provide examples of the rich diversity and complexity of nutrient loads and responses in systems that range widely in size, watershed, and in typology. The systems have also, to greater or lesser extents, been affected by natural or anthropogenic alterations that have yielded alterations in the ecosystems. Those seemingly contrasting observations can be summarized as

1. There is order in the complexity of estuarine system responses to nutrients, driven by the geomorphology and physical dynamics of the system.
2. Underlying common biogeochemical mechanisms explain some of the pattern in responses, although the dominant biological, chemical or geochemical processes vary in different systems.

The following sections of this chapter apply those generalizations in the context of nutrient criteria development. The challenge is to describe the relationships between nutrients, habitat alterations and attributes, and to link responses to the causal variables. Although many relationships can be developed, for the purposes of nutrient criteria, the following categorization of variables has proven to be useful: ***causal variables***, those that characterize alterations in habitat or in nutrient loading; ***response variables,*** those that are the direct ecological responses; and ***supporting variables***, those that help to normalize or characterize the causal and response variables (Figure 4.6). Estuarine typology and climate are examples of supporting variables and are well described elsewhere in this document. In the following sections, ***causal and response variables*** are reviewed with the intent of understanding how issues of estuarine complexity relate to the application of these variables in nutrient criteria development. While the relationships are initially discussed using nutrients as the causal variable and chlorophyll *a* as the response variable, examples are also given to demonstrate how integrated biocriteria, which include variables beyond chlorophyll *a,* can be developed.

Causal Variables: Nutrient Concentrations, Forms and Loads

Nutrient concentrations themselves can provide some information regarding the possibility of nutrient impairments. However, nutrient concentrations are also highly dynamic and are rapidly modified and transformed by many biological and biogeochemical processes. A fundamental issue is the relationship between ***nutrient loads*** and ***nutrient concentrations***. Nutrient loads represent a synthetic parameter that accounts for watershed area, use, and other factors. Nutrient loads are inherently much harder to determine than ambient concentration levels. On the other hand, nutrient load data seem intuitively more useful, and limnologists have had great success in relating trophic state of many lakes to appropriately scaled nutrient loads.

There has been some success with relating external nutrient loads to *in situ* nutrient concentrations in estuaries ***when data are averaged over long periods***. In general, variations in N loading rates are reflected in concentrations of N in receiving waterbodies. Although many processes act to modify nutrient concentrations at various rates, mean TN concentrations were found to be significantly correlated to TN loading for five subsystems of Chesapeake Bay ***averaged over a decadal period*** (Figure 4.7; Boynton and Kemp 2008). In addition, Conley et al. (2000) reported that, on ***an annual basis***, about 70 percent on the variation in TN concentration could be explained by variation in TN loads in a large sample of Danish estuaries.

Causal Variables: Human Alteration and Land Use

Land use is another causal variable because it integrates the multiple factors that determine nutrient load. The effect of land use is well illustrated using the 138 estuaries that were studied in the National Estuarine Eutrophication Assessment (Bricker et al. 2007). By comparing the eutrophication status of these estuaries with the percent of land use in agricultural or urban development (Figure 4.8), it can be seen that a greater percent of estuaries were classified as moderately high to highly eutrophic when more land was devoted to these uses. Virtually all estuaries that were categorized as low or moderately low on the eutrophic status scale developed, had < 40 percent of the land in urban or agriculture use.

Response Variables: Phytoplankton Biomass and Production

Obtaining an adequate measure of impairment by nutrients (i.e., a ***response variable***) is a critical first step toward managing nutrient enrichment in coastal waters. A range of variables and indices can be considered according to the information that is available, beginning with simple measures and expanding to more sophisticated, integrated approaches when possible.

Simple measures such as chlorophyll *a* concentration have the advantage of broad application and low cost for implementation. Indeed, in many estuarine ecosystems high levels of chlorophyll *a* associated with phytoplankton are a clear sign of nutrient over-enrichment, and dose-response relationships with nutrients can be robust for some types of systems. In many cases, states have already embraced this simple index of phytoplankton biomass as an adequate basis for including a coastal water body in their list of impaired waters under CWA section 303(d).

Relating nutrients to biomass is not simple, however. Availability of nutrients at a given time is not necessarily related to the biomass at that point in time because of biological uptake, transport, grazing and many other factors. In fact, no *a priori* relationship should be expected

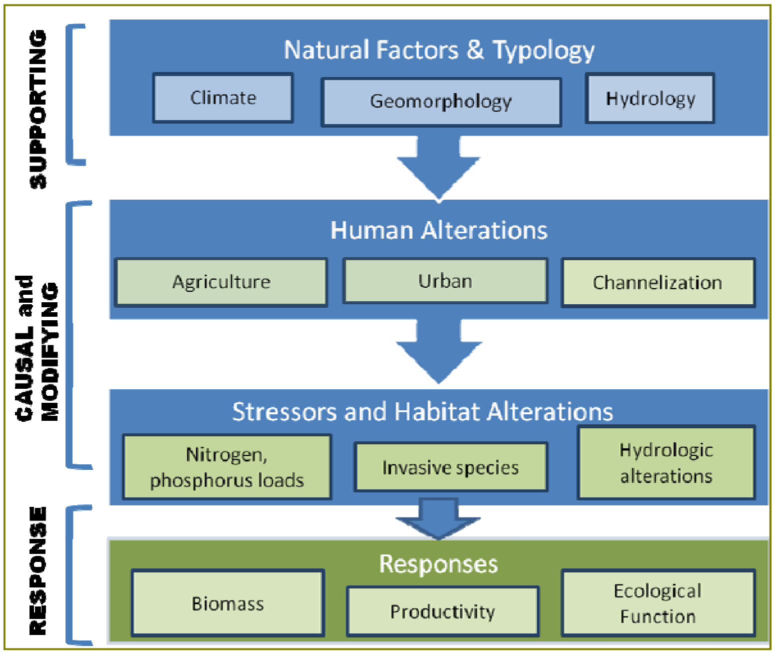


Figure 4.6. Conceptual relationships among supporting and causal variables, and responses. Supporting variables are natural aspects of the landscape and waterbody, as described elsewhere in defining typology. Causal and modifying variables are those that directly cause or modify a response that can be classified in terms of biomass, rate processes or more integrated measures of ecological function. The variables listed are examples only.

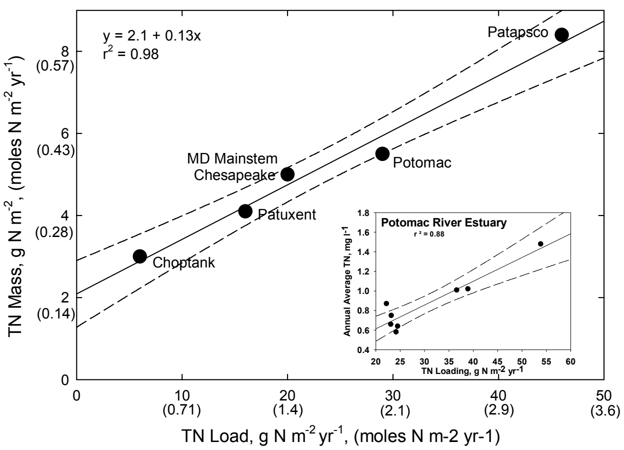


Figure 4.7. Simple regression relating annual TN loading rate to TN mass in a selection of Chesapeake Bay tributaries. A time-series of load-mass data are shown for the Potomac estuary in the inset. Adapted from Boynton and Kemp 2008.

for synoptically collected data. That is exemplified for data from the Delaware Bay in which paired data for ambient DIN and synoptic chlorophyll *a* concentration show no functional relationship (Figure 4.9).

Using a broad data set from 92 estuarine and coastal sites worldwide, Smith (2006) was able to show a strong correspondence between ***annual mean*** concentrations of TN and TP and the standing stock of chlorophyll *a*, although the relationship was considerably stronger for N than for P (Figures 4.10, 4.11). The following section explores those relationships on a seasonal basis for additional estuaries. Additionally, a substantial number of statistical models have been developed to relate primary production or algal biomass to nutrient characteristics in estuarine ecosystems (Table 4.3; Boynton and Kemp 2008). These models take a variety of forms, for example, using either N concentration or N loading rate as an independent variable and rate of primary production or algal biomass (chlorophyll *a*) as a response variable. Most models are computed on annual time scales. Sample sizes used to develop the models varied widely, and some were based on multi-system comparisons while others were based on multiyear data collected for one system. Given the myriad factors known to influence production and algal biomass accumulation, it is noteworthy that N was able to account for a large portion of the variability of phytoplankton production or algal biomass.

Moreover, when dealing with nutrient criteria and specifically measures of nutrient concentrations, the form and relative proportion of each nutrient is also important. The strength of relationships between N, for example, and the chlorophyll *a* depends on whether the data are reported as DIN concentration (NO_3^- + NO_2^- + NH_4^+) or TN (DIN + DON + PON). As will be seen, the strength of these relationships is generally greater with TN. A significant amount of research on the DON pool (which would be included in TN but not DIN) during the last decade has indicated that (1) significant portions of the DON pool are available to plants, including algae; (2) DON often is the dominant N pool in estuarine systems, especially during warm times of the year; and (3) this pool can be dynamic (reviewed by Antia et al. 1991; Seitzinger et al. 2002; Berman and Bronk 2003). Thus, use of just DIN will likely

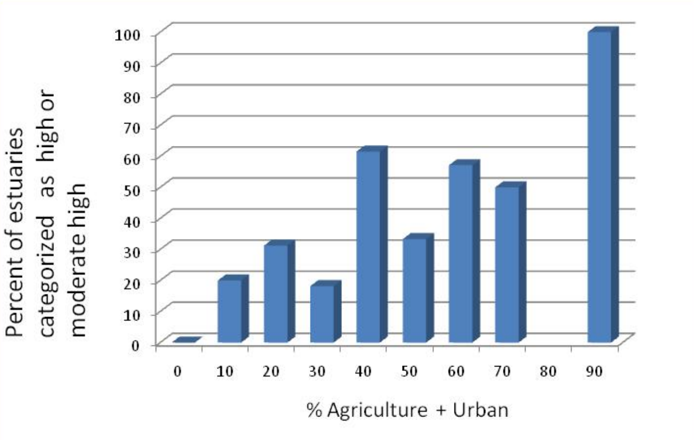


Figure 4.8. Relationship between the percent of estuaries categorized as high or moderately highly eutrophic using the assessment of Bricker et al. 2007 and the percent of watershed area of these estuaries in agricultural or urban use.

underestimate the influence of N in these systems. Similarly, with P, PO_4^{-3}, does not necessarily reflect the availability of the TP pool to organisms.

There are, of course, some limitations with these analyses. They are correlations and, thus, do not demonstrate causation (even though terms such as causal variable are used); results can be related in part to some other covariate. Other possible explanations for production or biomass variability were not always thoroughly examined; in some, it appeared that N was assumed, *a priori*, to be the key explanatory variable, while other factors, such as P or light availability, were not examined with equal rigor. In spite of those limitations, several important points emerge from the Smith (2006) and other similar analyses. First, very simple relationships in the absence of any scaling (e.g., time averaging) are not likely to show useful relationships. Second, picking the appropriate period over which to average is of vital importance. Third, better relationships were found when TN is used as the causal variable than when DIN only is used.

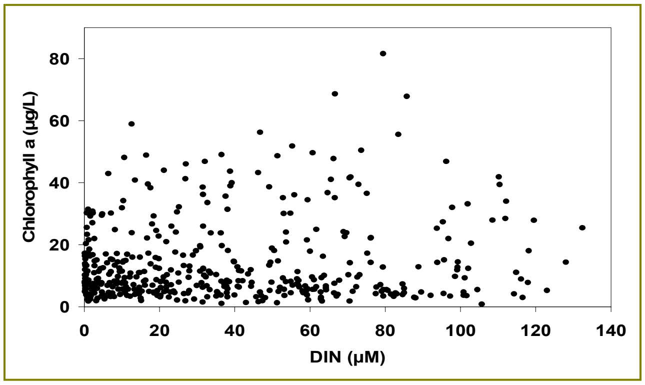
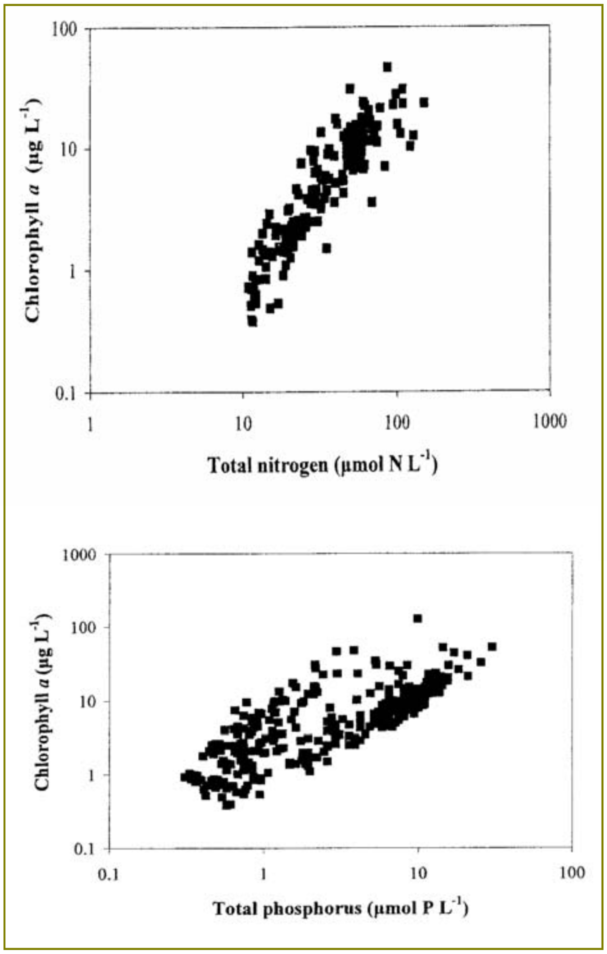


Figure 4.9. Scatter diagrams for the mid and lower Delaware Bay, for data from 1980 to 2003, illustrate that simple, synoptic correlations between nutrients and biomass often do not display characteristics of a dose-response relationship. From J. Sharp, unpublished.

Examples of Relating TN to Chlorophyll a: Cross-Estuary Comparisons

As described throughout this document, estuarine typology serves to scale estuarine responses; typology is a ***supporting variable***. The importance of typology was highlighted in the review of the case studies for ***river-dominated*** versus ***lagoonal*** systems. Here, the role of typology is further described in terms of the relationships between concentrations of planktonic chlorophyll *a* and TN in 10 estuaries in the eastern United States (Dettmann and Kurtz 2006). This analysis includes some of the systems described in the detailed case studies and some additional systems. Four of these (Boston Harbor and nearby regions of Massachusetts Bay, Long Island Sound, the Peconic Estuary and Tampa Bay) are classified as ***coastal embayments***, the other estuaries of that study (Delaware Bay, the mainstem of Chesapeake Bay, and four tributaries of Chesapeake Bay: the Patuxent, Potomac, Rappahannock and James rivers) are ***river-dominated estuaries***. This analysis compared summer (June, July, August) relationships at individual stations that were averaged over multiple years (Figure 4.12). Each estuary is represented by data for a number of stations, typically 9 or 10, but sometimes more (18 for Long Island Sound and 54 for Tampa Bay). The coastal embayments exhibited a strong relationship between chlorophyll *a* and TN. Data for the river-dominated estuaries show considerably more scatter, and many of the data points fall below the general trend defined by the coastal embayments.

Regressions between chlorophyll *a* and TN concentrations were also developed for each individual estuary in this study (Dettmann and Kurtz 2006). Power law relationships ($[\text{Chl } a] = a\,[\text{TN}]^b$) between chlorophyll *a* and TN within each of the ***coastal embayments*** all had statistically identical values of the parameter *b*; that is, all had identical slopes when plotted on log-log plots (Figure 4.13). The values of the intercept parameter *a*, which gives the concentration of chlorophyll *a* for TN = 1 mg L^{-1}, were negatively and linearly correlated with average concentrations of total suspended solids (TSS) within the



Figures 4.10, 4.11. Relationship between annual mean TN concentration (top) and TP (bottom) concentrations and annual mean concentrations of chlorophyll a based on 335 cases from 92 sites worldwide. From Smith 2006.

estuaries (R^2 = 0.95, p = 0.025). Power law relationships for ***river-dominated estuaries*** had highly variable slopes, but when regressions were restricted to stations within estuary zones that had narrow ranges of TSS concentrations, values of the parameter *b* for most systems fell into a relatively narrow range (Figure 4.14). Values of the parameter *a* for most segments were negatively and linearly correlated with average TSS concentrations (R^2 = 0.99, p = 0.0007). Thus, each estuary class had a set of relationships between chlorophyll *a* and TN, with differences among relationships within each class largely explainable by light availability (inversely related to TSS).

Examples of Relating TN to Chlorophyll a: Within Estuary Variability

The issue of how to address intra-estuarine concentration gradients is also complex. Response to nutrients can vary along the longitudinal axis of estuaries because of variation in loading or local retention time. Spatial variation in response can also be a function of geomorphology; for example, the proportion of an estuary shallower than the critical depth for light penetration given background concentrations of color and turbidity, or portions of an estuary intrinsically vulnerable to hypoxia/anoxia in isolated deep channels with limited tidal exchange. Estuarine zonation or segmentation can be considered as a sub-estuarine classification process.

The importance of mixing time scales in regulating eutrophication responses has long been recognized in freshwater systems, and estimates of mixing times are explicitly used in comparative empirical models of phytoplankton response to nutrient inputs in lakes (Vollenweider 1976). Estuarine typologies aimed at classifying estuarine susceptibility to eutrophication include some type of mixing time scale as a primary classification variable (Bricker et al. 1999; Ferreira et al. 2007; Painting et al. 2007). However, despite this conceptual understanding and the work done in freshwater systems, the linkage between estuarine mixing time scales and phytoplankton-based eutrophication has only rarely been empirically described (Lucas et al. 1999a,1999b; Howarth et al. 2002).

Three approaches can be used to identify estuary subsections for deriving numeric nutrient criteria. First, variation in local residence time can be explicitly built into response models as a normalizing factor or used to partition segments into those susceptible to bloom formation versus those that are well flushed. Approaches to segment estuaries on the basis of local residence time or

Table 4.3. A summary of statistical models relating phytoplankton primary productivity or biomass to N (concentration and loading rates) or other variables. Abbreviations B, Zp, Io, Z and R_{time} refer to phytoplankton biomass (as chlorophyll a), euphotic depth, incident irradiance, average system depth, and water residence time, respectively; "na" indicates information not available. Adapted from Boynton and Kemp 2008.

Location	Independent Variable, X (units)	Dependent Variable, Y	r^2 / n	Reference
Multiple estuaries	TN-loading (g N m^{-2} y^{-1})	Phytoplankton Production	0.60 / 14	Boynton et al. 1982
SF Bay and other estuaries	Composite parameter X = f(B, Z_p, I_0)		0.82 / 211	Cole and Cloern 1987
Narragansett Bay	Composite parameter X = f(B, Z_p, I_0)		0.82 / 1010	Keller 1988
Multiple estuaries	DIN-loading (mol N m^{-2} y^{-1})		0.93 / 19	Nixon et al. 1996
Multiple estuaries	TN-loading (g N m^{-2} y^{-1})		0.36 / 51	Borum and Sand-Jensen 1996
Boston Harbor	Composite parameter X = f(B, Z_p, I_0)		0.66 / 12	Kelly and Doering 1997
Waquoit Bay system	Annual average DIN conc (µM)		0.61 / 12	Valiela et al. 2001
Chesapeake Bay	TN(x_1), TP(x_2) load (kg mo^{-1})		0.67 / 11	Harding et al. 2002
Multiple estuaries	DIN (mM m^{-3}); tidal range (m)	Phytoplankton Biomass	na / 163	Monbet 1992
Multiple systems / MERL	DIN input (mmol m^{-3} y^{-1})		na / 34	Nixon 1992
Ches Bay mesohaline	River flow (m^3 d^{-1}); proxy for N-load		0.70 / 34	Harding et al. 1992
Maryland lagoons	TN load (g N m^{-2} y^{-1})		0.96 / 9	Boynton et al. 1996
Danish coastal waters	TN concentration (ug l^{-1})		0.64 / 168	Borum 1996
Canadian estuaries	TN concentration (ug l^{-1})		0.72 / 15	Meeuwig 1999
Ches Bay and Tributaries	TN Load; (mg N m^{-2} yr^{-1}) (R_{time}, yrs$)^{-1}$		0.82 / 17	Boynton and Kemp 2000
Danish estuaries	TN concentration (ug N l^{-1})		0.30 / 1347	Nielsen et al. 2002

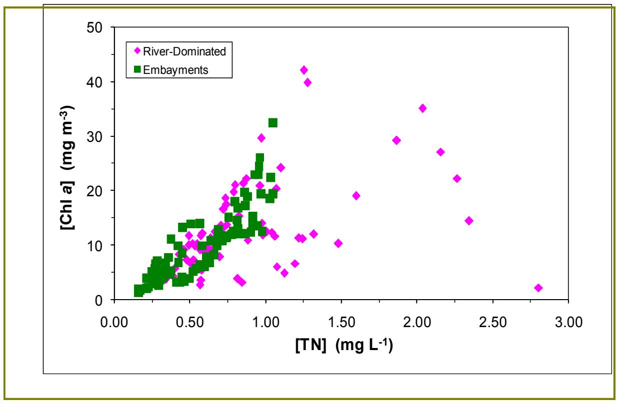


Figure 4.12. Multiyear mean summer concentrations of chlorophyll a versus TN for riverine and lagoonal estuaries as described in text. From E. Dettmann, unpublished data.

pulse residence time have been developed by Abdelrhman (2005), Hagy et al. (2000), and Miller and McPherson (1991). Second, estuary zones can be defined on the basis of potential habitat use or designated uses assigned by the states as has been done for Chesapeake Bay (USEPA 2003). This approach restricts the space within which appropriate nutrient-response models will be applied and allows targets to be identified to support specific designated uses. Third, zonation by salinity can be used to explore variation in response according to differential species sensitivities or to partition the estuaries into zones influenced by nutrient loading from the seaward boundary (upwelling influence) versus loading from the watershed (Lee and Brown 2009).

Figures 4.15 and 4.16 show the variation of concentrations of TN and chlorophyll *a* with distance along the axes of Long Island Sound and the Rappahannock River (a tributary of Chesapeake Bay). Such gradients are present in most estuaries. In estuaries for which the main nutrient source is in the inner estuary, the gradients are qualitatively similar to those shown in Figures 4.15 and 4.16. In regions such as the Pacific Northwest, however, where seasonal upwelling occurs, a major input of nutrients can occur across the seaward boundary (Nelson and Brown 2008; Lee and Brown 2009). In these estuaries, the direction of gradients would be reversed from that shown in Figures 4.15 and 4.16. High salinity areas of estuaries across the Pacific Northwest tend to have higher DIN concentrations than low salinity areas during the summer period of peak biological responses. Low summer rainfall diminishes watershed inputs, while upwelling increases input from the near coastal region. This pattern is also seen within individual estuarine systems such as Yaquina Estuary, Oregon (see Section 5.7).

In a recent study (Lehrter et al. 2006), a comparative empirical analysis was used to relate chlorophyll *a*, on a sampling event basis, to estuarine mixing time scales, estuarine nutrients, and watershed inputs of freshwater and nutrients in seven oligohaline tidal river regions of Mobile Bay, Alabama (Figure 4.17). The riverine discharges and freshwater nutrient concentrations delivered to these tidal river regions varied 10-fold for discharge, 7-fold for TN, and 4-fold for TP from 2000 to 2001 (Lehrter et al. 2006). Estuarine mixing time scales in

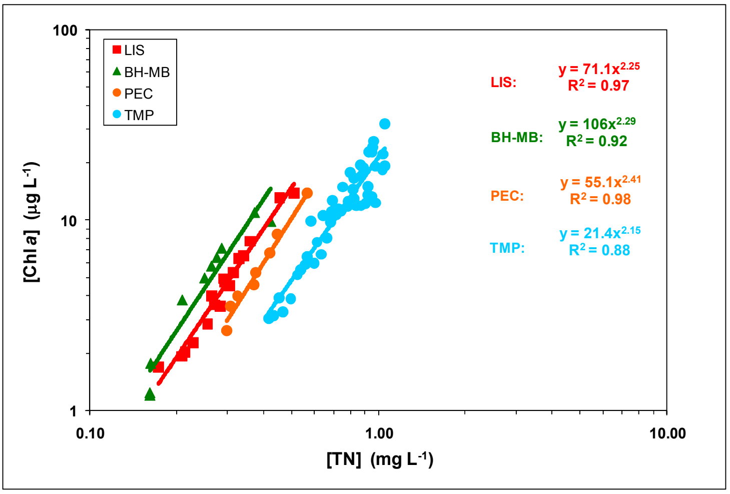


Figure 4.13. Multiyear mean summer concentrations of chlorophyll a versus TN at individual stations in coastal embayments: Boston Harbor–Massachusetts Bay (BH-MB), Long Island Sound (LIS), the Peconic Estuary (PEC) and Tampa Bay (TMP). Also included are regression lines for individual systems. From Dettmann and Kurtz 2006.

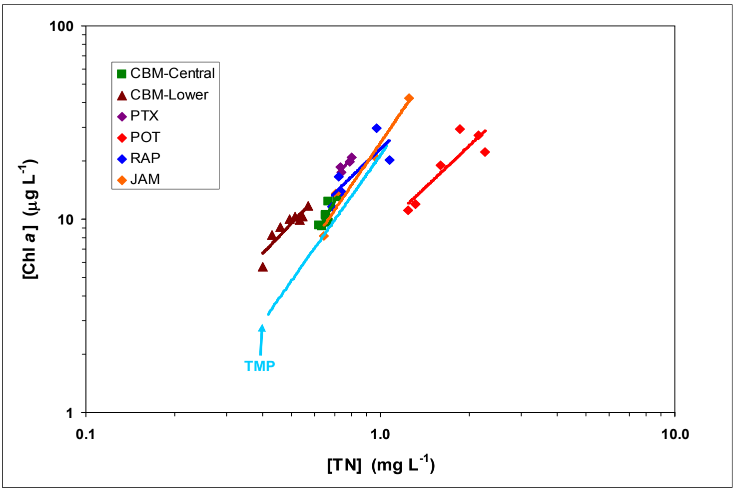


Figure 4.14. Multiyear mean summer concentrations of chlorophyll a versus TN for riverine estuaries: central and lower Chesapeake Bay (CBM), Patuxent River (PTX), Potomac River (POT), Rappahannock River (RAP) and James River (JAM). Data and regressions are for estuary segments having narrow ranges of TSS concentrations. The regression line for Tampa Bay is included for comparison. From Dettmann and Kurtz 2006.

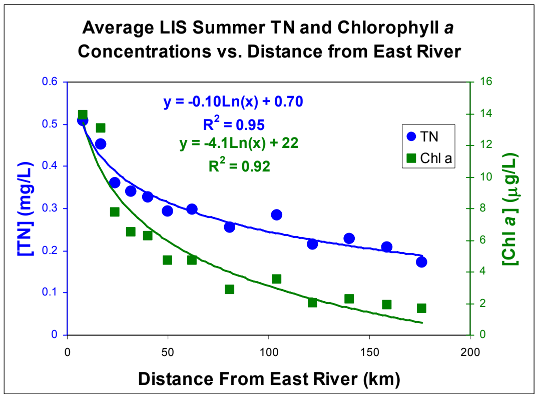


Figure 4.15. Chlorophyll a and TN concentrations show clear gradients in Long Island Sound, with highest concentrations in the inner sound, near New York City. From E. Dettmann, unpublished data.

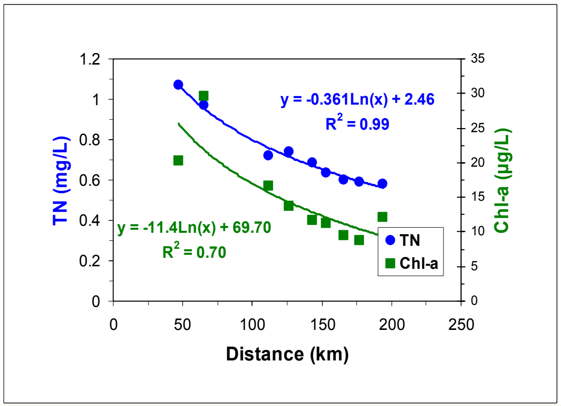


Figure 4.16. Average summer chlorophyll a and TN concentrations show clear gradients along the length of the Rappahannock River, with highest concentrations in the upper reaches (Dettmann and Kurtz 2006). From E. Dettmann, unpublished data.

those tidal rivers varied 20-fold, with median residence times ranging from 0.5 to 20 days and median freshwater flushing times ranging from 1.4 to 19 days. That constituted a natural experiment for examining chlorophyll *a* response to nutrient inputs and estuarine mixing time properties.

Reference Conditions

Nutrient criteria also depend on establishing an appropriate ***reference condition***. A reference condition is the, ***"comprehensive representation of data from several similar, minimally impacted, 'natural' sites on a waterbody or from within a similar class of waterbodies"*** (USEPA 2001). Reference conditions can be defined in terms of TN, TP, chlorophyll *a*, Secchi depth or other metrics of the biota. The reference condition represents the unimpaired or minimally affected state of a waterbody and is important in deriving numeric nutrient criteria for two reasons. As described in Chapter 1, criteria are often expressed as a pollutant concentration at which the ***designated use*** is still supported. Thus, a reference condition relates nutrient-related variables to an ***existing designated use*** in that waterbody. Second, it serves as a ***reference point*** for taking corrective measures to attain the designated use in a compromised and degraded waterbody. The relationships between nutrient criteria and reference conditions are conceptualized in Figure 4.18 (USEPA 2001).

The process of identifying the reference condition for an estuary, regardless of the method used, involves two essential elements, (1) an unimpaired or minimally-impaired reference estuary, and (2) data on nutrient-related variables derived from that reference estuary. The reference estuary need not be another separate estuary but can be the estuary that is already in a degraded state if historical data are available. If data are not available, an outside reference estuary can be used.

The systematic, statistical classification of estuaries according to typology is one way to start to reduce the complexity of estuarine systems and, in turn, the effort needed to identify the appropriate reference condition. However, as underscored throughout this document, while estuaries in each class share similar characteristics, they do not necessarily *behave* or function in the same ways relative to excessive nutrient inputs. Additionally, estuaries within a class are not necessarily unimpaired or pristine, so caution must be taken not to infer that an estuarine class reflects an aggregate reference condition. Rather, once the estuary of concern is placed in its appropriate class, aggregate data within the class

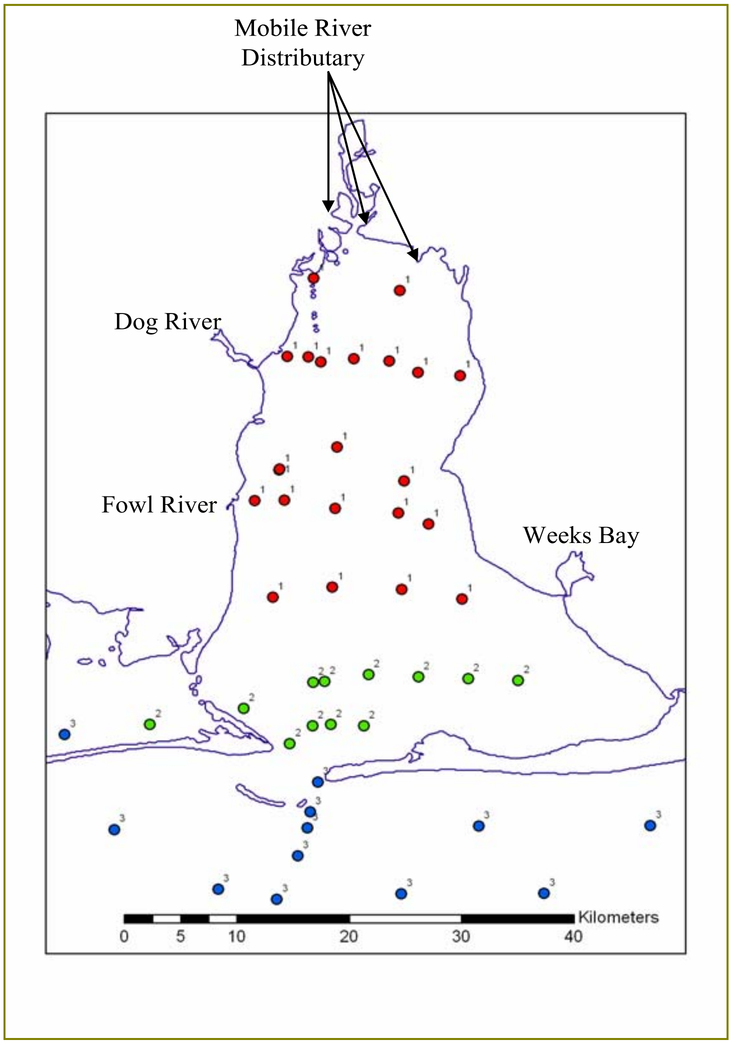


Figure 4.17. Map of Mobile Bay showing the water quality stations grouped into zones of similar influence as denoted by the numbers and colors. The groupings were determined using a multivariate approach on the basis of water quality data from the individual stations. From J. Lehrter, unpublished data.

can be used to extrapolate the reference condition. Careful examination is recommended to assess whether a class of estuaries can be used to derive the reference condition, or whether further subdivision within that class to an individual estuary or estuaries is needed to make the appropriate comparison.

Data resources are another essential element to any effort to establish a reference condition. That is true when relying on historical data from an estuary to determine the reference condition or when aggregated data from estuaries binned together through classification are used. Data should be scientifically reliable in terms of the analytical methods used to generate them and how they are expressed. Data should also be easily accessible. For the purposes of supporting numeric nutrient criteria derivation, estuarine data were collected from a variety of sources and consolidated into a single database. Appendix I describes the database, its sources, the nutrient-related variables, its construction and utility.

There is an additional approach that might be feasible for some estuaries where data exist concerning nutrient concentrations collected during times when the estuary was judged to be in *good condition*. The approach is not generally useful because of a lack of appropriate historical data. However, by way of example, during the 1940s to 1960s, the mainstem of Chesapeake Bay had abundant SAV, almost no anoxia, modest hypoxia, and well-developed benthos. Concentrations of N in the major river entering the bay were on the order of 35 µM-N. During the 1980s and early 1990s, SAV declined, benthic communities were severely affected in portions of the bay, hypoxic volume increased, deep water became anoxic every year and river N concentrations had increased to > 100 µM-N (Hagy et al. 2004). There might be enough estuarine systems through the country where enough historical data are available to make that approach useful as an additional guide to setting criteria. The approach, while simple, does provide some guidance as to nutrient concentrations when an estuary still had favorable characteristics. It does not consider hysteresis effects that could call for even lower nutrient conditions to achieve restoration goals or extended periods under reduced nutrient concentrations before positive responses are observed.

Other Response Variables: Use of Integrated Biocriteria

As illustrated in Chapter 2, and as further described in the case studies (Chapter 5), measures other than chlorophyll *a* can provide useful additional information about the extent of nutrient impairments. The use of biocriteria and bioassessment in conjunction with physical and chemical water quality analyses is an approach that goes beyond basic measurements of chlorophyll *a* in evaluating the condition of estuarine waterbodies. Bioassessment is defined as the “characterization of environmental conditions through the use of biological organisms” (MacDonald et al. 1996). Such data can range from the size spectrum of chlorophyll *a*, to the species composition of the phytoplankton community, including the presence of HABs, the decline in SAV or the extent of development of hypoxia. Knowledge of specific dominant taxa can be a powerful measure of water quality status. Various photopigments can be used as diagnostics of the abundance of diatoms, chlorophytes, cyanobacteria, and even specific HAB species, such as brown tide (Mackey et al. 1996; Pinckney et al. 2001; Trice et al. 2004; Paerl 2006; Glibert et al. 2007). For example, in the case study of the Neuse Estuary, the relative contribution of chlorophytes, cryptophytes and diatoms to the total chlorophyll *a* pool coincided with, and was enhanced by, periods of elevated river flow to the Neuse Estuary. Cyanobacteria, however, demonstrated greater relative biomass when flushing was minimal and residence times were longer, especially during the summer months. Understanding those dynamics yields greater insight into ecosystem responses than chlorophyll *a* alone.

Designed to document organism responses to pollution and other anthropogenic perturbations, bioassessment targets both biotic and habitat components. It is generally not possible to delineate overall ecosystem impact of multiple estuarine stressors without sufficient information on the structure, function, and responses of the constituent biotic communities. By focusing on physical and chemical assessments of estuarine systems and not on integrated bioassessment, the degradation of system attributes can be underestimated, thereby precluding

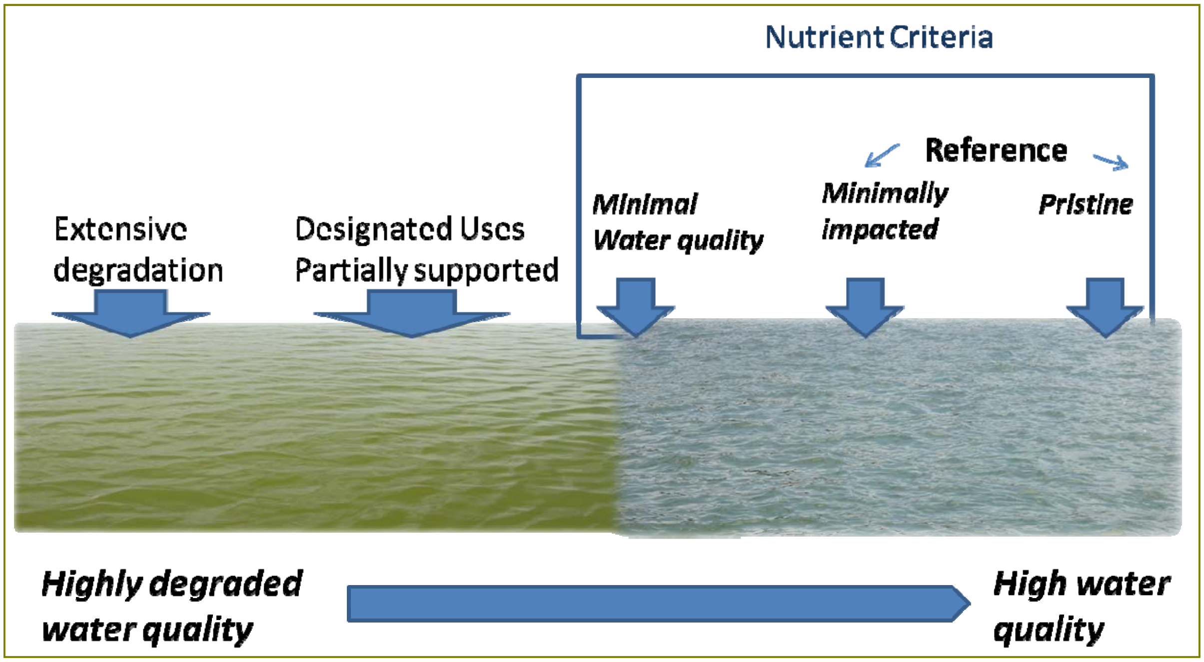


Figure 4.18. Environmental quality scale representing the relationship between reference conditions and potential nutrient criteria relative to designated uses. Modified from USEPA 2001.

accurate characterization of the resource condition in estuaries affected by anthropogenic influences. To date, biocriteria and bioassessments have been most successfully applied to freshwater systems, most notably rivers and streams, where monitoring of fish assemblages, benthic invertebrate communities and algae have been particularly useful in providing data for determining ecological condition and impairment (Gibson et al. 2000).

Biological indicators can be considered as structural entities of ecosystems that serve as sentinels of overall condition. As such, they can consist of a single species, populations, or communities of organisms, and often include the biological responses observed at the organism level or above. Biomarkers, which typically reflect exposure to environmental stressors, are expressed at the suborganismal levels of biological organization, notably biochemical, biomolecular, and physiological levels (Adams 2005). Biological indicators are significant because they represent the integration of aquatic conditions and provide the information necessary to assess abiotic and biotic conditions and cumulative effects in impacted systems (Bilkovic et al. 2005). Examples include key species or indicator taxa that connote specific environmental conditions.

SAV are also excellent bioindicators of estuarine water and sediment quality, as well as overall ecosystem health (Hemminga 1998; Duarte 1999; Corbett et al. 2005; Lamote and Dunton 2006). Assessing the distribution and abundance of SAV in lagoon-type, coastal-bay systems to track escalating eutrophication impacts is a useful indicator. Because changes in SAV distribution and abundance can occur over periods as short as weeks or months, rapid and cost-effective tools should be applied to determine SAV condition.

The development of indices of benthic community condition is another valuable tool in bioassessment of estuarine ecosystems. During the past decade, benthic assemblages have been used to assess water quality and environmental status and trends in regional areas (Van Dolah et al. 1999; Paul et al. 2001; Borja et al. 2003; Llanso et al. 2003; Muniz et al. 2005). Those indices have proven valuable in delineating overall environmental health of estuarine ecosystems (Dauer 1993; Diaz and Rosenberg 1995; Weisberg et al. 1997; Rosenberg et al. 2004). That is so because benthic species are largely sedentary, highly responsive to habitat disturbances, and many of them have long life spans. They are considered to be more reliable indicators than drift macroalgae, plankton, and fish, and provide *in situ* measures of relative biotic integrity and habitat quality (Gibson et al. 2000). In addition, they integrate water and sediment quality conditions and play an important role in the biogeochemical cycling of nutrients and other substances (Dauer 1993; Diaz et al. 2004).

Furthermore, benthic assemblages respond predictably to many natural and anthropogenic stressors, and thus have been used to document the effects of specific stressors including organic enrichment, hypoxia, chemical contaminants, and other factors (Weisberg et al. 1997; Rosenberg et al. 2003). Such disturbances in the benthos are typically manifested by changes in species composition, abundance, biomass and diversity signaling successional shifts in benthic community structure (Rosenberg et al. 2004). The data might also shed light on changes in trophic structure and function that could reflect bottom-up or top-down effects. Several studies have demonstrated the value of benthic communities as indicators of ecosystem health (Dauer 1993; Weisberg et al. 1997; Diaz et al. 2004). Those studies have examined various univariate and multivariate methods or biotic coefficients for assessment of estuarine environmental status. Benthic indices employing species abundance, dominance, diversity and other parameters are useful measures of community composition and function, and they serve as indicators of estuarine condition. Because of their sensitivity to stress-induced changes in benthic communities, benthic indices also have utility in assessing anthropogenic impacts. The development of benthic indices reduces large biotic data sets to values that permit more meaningful statistical assessments.

Various multimetric indicators have been described for biocriteria. All have in common species richness, abundance/biomass, presence of sensitive and tolerant species, and many other parameters. One such metric is a species tolerance index. The observed/expected index identifies taxa that are expected to be present at specific types of habitats. The metric is, thus, easy to describe, and it is habitat independent. The presence/absence of species is a sensitive metric. A major disadvantage of this index, however, is that the baseline data on species tolerances are limited and it requires a large amount of data from clean sites along habitat gradients to establish expected species along numerous habitat gradients.

As with phytoplankton community indicators, benthic and fish community indicators are relatively expensive to derive, and their interpretation can be complex. In particular, biotic measurements provide direct information about the status of the biotic resources to be protected, not just the biochemical environment. The disadvantage of multivariate indices is that they are not necessarily intuitive and can be highly dependent on the particular test data set. The index component can also change when additional data become available.

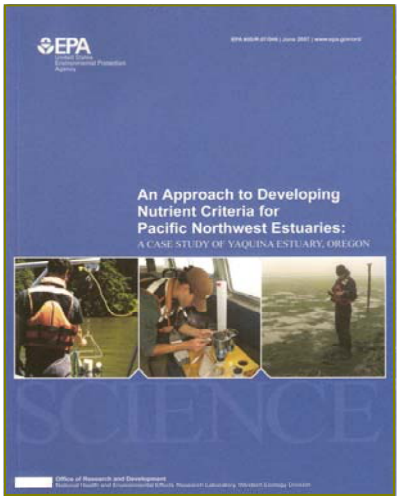
The range of valid indicators of nutrient impairment that have been developed, from simple to complex, represent an opportunity to design criteria suitable to individual waterbodies and the capabilities for monitoring that are available. The following sections describe examples of criteria development, using a range of metrics, from DIN and chlorophyll *a* to integrated biocriteria.

Example of Criteria Development: Yaquina Estuary, Oregon

The Yaquina Estuary is a small estuary on the central Oregon coast; its general environmental conditions and trends are described more fully in Section 5.7. The derivation of numeric nutrient criteria for this system (Figure 4.19) is provided here as an example of application of the principles discussed in this chapter and throughout this document. Note that as is the case with the other examples, this is only provided to illustrate an approach.

The criteria for this estuary were based on identification of the following ***designated uses***, “aquatic life harvesting (shellfish growing and fishing), agriculture (livestock watering), municipal (public water supply), recreation (water contact recreation), ecological (resident fish and aquatic life, salmonid spawning and rearing, anadromous fish passage) and aesthetics” (Brown et al. 2007, p. iii).

To identify ***reference conditions*** for this estuary, *in situ* observations were examined to develop cumulative distributions functions and compared with data from other Oregon estuaries. Key percentiles (25th, 50th and 75th) were identified for various water quality parameters and then applied in a model that was developed to assess the response of SAV to various stressors (Brown et al. 2007).



An Approach to Developing Nutrient Criteria for Pacific Northwest Estuaries:
A CASE STUDY OF YAQUINA ESTUARY, OREGON

Figure 4.19. Specific nutrient criteria have been proposed for Yaquina Estuary, Oregon, on the basis of results from field data from a range of temporal and spatial scales and from reconstruction of historical conditions, as well as various modeling approaches (Brown et al. 2007); EPA report EPA/600/R-07/046.

On the basis of the available data, it was suggested that different criteria be developed for wet and dry seasons and for different zones of the estuary (see Section

Table 4.4. Potential criteria for Yaquina Estuary for the dry season. From Brown et al. 2007.

Parameter	Estuary Zone 1	Estuary Zone 2
DIN (mM-N)	14	14
PO_4^{-3} (mM-P)	1.3	0.6
Chlorophyll *a* (mg L^{-1})	3	5
Water clarity (m^{-1})	0.8	1.5
DO (mg L^{-1})	6.56.5	

5.7). For example, during the wet season, N sources are dominated by riverine inputs, whereas during the dry season, N sources are dominated by oceanic inputs. These different sources in turn drive water quality differentially in different zones of the estuary. Criteria were developed for the dry season and tested with the SAV stressor-response model that confirmed that eelgrass habitat could be maintained with this water quality (Brown et al. 2007; Table 4.4).

Use of Integrated Biocriteria: Examples from the Chesapeake Bay

Previous examples of application of integration of metrics into an eutrophication index were described in Chapter 2, particularly in relation to the National Estuarine Eutrophication Assessment (Bricker et al. 2007). Here, examples of the efforts ongoing in Chesapeake Bay to identify criteria are described (USEPA 2003). Chesapeake Bay has a several-decade-long record of monitoring for many biological, chemical, and physical parameters, and considerable effort has been ongoing to define the best metrics of water and habitat condition. The Chesapeake Bay also provides an interesting example because, while water quality and living resources have declined in recent decades, chlorophyll *a* values were found to be only slightly lower in the 1950s than current averages (Harding and Perry 1997). Thus, this is a clear example where use of chlorophyll *a* alone might not be a sufficient criterion. The example also illustrates how data, collected using a range of protocols, can be applied and integrated.

A Phytoplankton Index of Biotic Integrity (P-IBI) was developed for Chesapeake Bay using several dozen phytoplankton metrics (Table 4.5), and assessed relative to reference conditions that were established from total DIN, PO_4^{-3} and Secchi depth (Buchanan et al. 2005; Lacouture et al. 2006). The methodology for formulating the P-IBI is described in more detail in Gibson et al. (2000). Each metric was subsequently ranked on a scale of 1–5, depending on its degree of deviation from the previously established reference. A ranking of 1 indicates serious impairment, or deviation from reference, while scores of 3 and above are indicative of conditions close to, or at, reference levels (Lacouture et al. 2006). The strength of this approach can be seen in the comparison of the spring P-IBI values in comparison with the reference conditions for each salinity region of the bay (Figure 4.20). Although there was some overlap between the lowest P-IBI and the highest reference conditions, such as in the tidal fresh regions, overall a high degree of discrimination between reference and degraded communities was apparent (Lacouture et al. 2006). Of all the phytoplankton metrics examined, the most useful throughout all regions of Chesapeake Bay were found to be the carbon: chlorophyll *a* ratio, surface chlorophyll *a*, dissolved organic carbon and phaeophytin, and total nano-micro plankton biomass. Other, taxon-specific metrics varied in their usefulness by salinity zone of the bay, such as diatom and dinoflagellate biomass in the meso- and polyhaline regions, where they are normally most abundant (Lacouture et al. 2006).

Table 4.5. Examples of the types of phytoplankton and chemical metrics examined in the analysis of P-IBI for the Chesapeake Bay. Each metric was statistically compared against previously established reference conditions for spring and summer, and for each region of the Chesapeake Bay, tidal fresh, oligohaline, mesohaline, and polyhaline. Both algal class and some algal species are included. Modified from Lacouture et al. 2006.

Chlorophyll *a*, above pycnocline
Chlorophyll *a*, surface only
Phaeophytin
Carbon: chlorophyll *a*
Chlorophyte abundance
Chlorophyte biomass
Chrysophyte abundance
Chrysophyte biomass
Cryptophyte abundance
Cryptophyte biomass
% total biomass composed of cryptophytes
Cyanophyte abundance
Cyanophyte biomass
% total biomass composed of cyanophytes
Diatom abundance
Diatom biomass
% total biomass composed of diatoms
Dinoflagellate abundance
Dinoflagellate biomass
% total biomass composed of dinoflagellates
Prorocentrum minimum abundance
Prorocentrum minimum biomass
Microcystis aeruginosa abundance
Microcystis aeruginosa biomass
Dissolved oxygen
Dissolved organic carbon
Particulate carbon
Total organic carbon
Total suspended solids

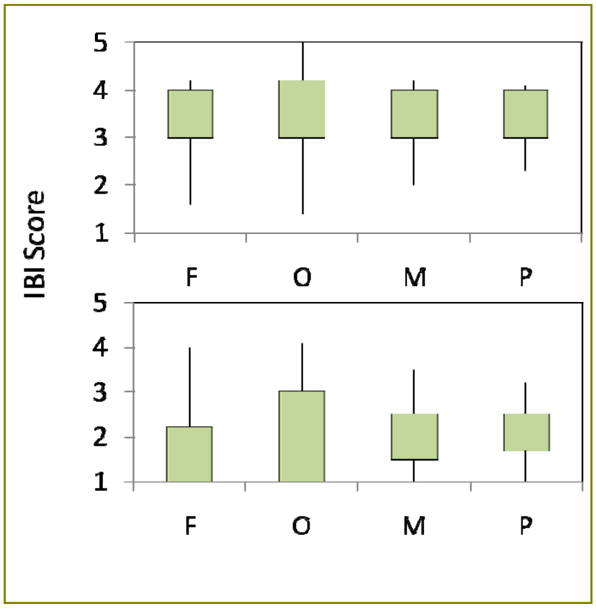


Figure 4.20 The P-IBI for the Chesapeake Bay for degraded (lower panel) and reference (upper panel) regions. The interquartile range, and 5th and 95th percentiles for each salinity zone (fresh (F), oligohaline (O), mesohaline (M) and polyhaline (P) are shown for spring. Redrawn from Lacouture et al. 2006.

Another Chesapeake Bay effort is geared toward providing an index of ecosystem health that can be combined into single scores, or report cards, that are geographically detailed but that can also be visually presented and regularly updated. The effort uses the P-IBI as described above, but further integrates it with other metrics, including area of SAV coverage, and an integrated measure of benthic community (Williams et al. 2007). Each value is compared to a reference value and its deviation scored. An integrated measure is then calculated from five reporting regions of the bay. An average Chesapeake Bay Habitat Health Index is then determined by averaging the various metrics for each bay region, to yield an overarching value. The values are then mapped to give a visual display of the combined index (Figure 4.21).

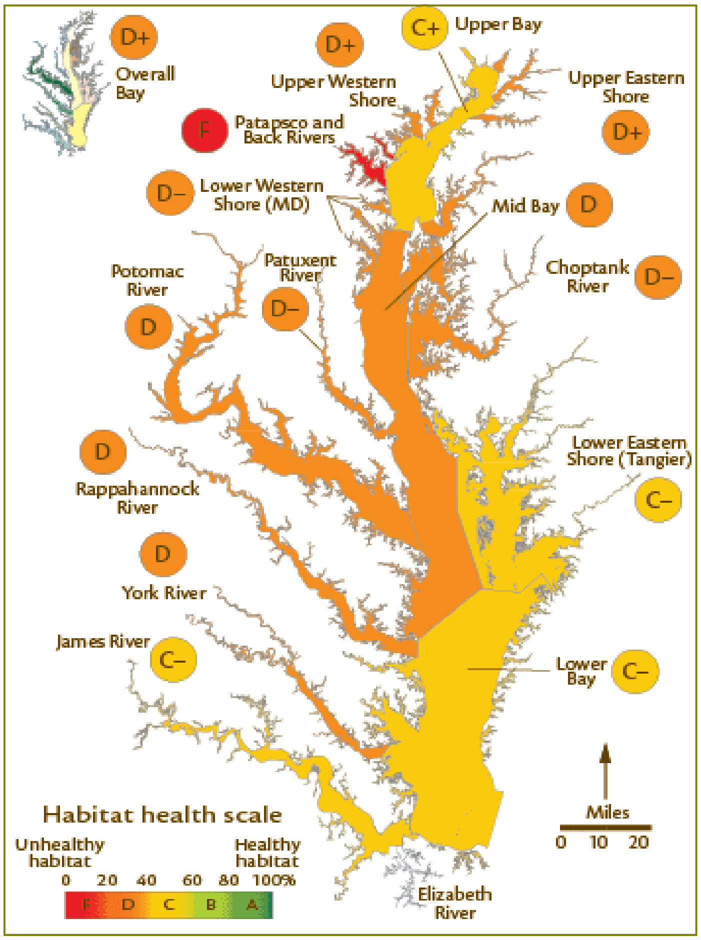


Figure 4.21. The map shows the integrated habitat health index for the Chesapeake Bay for 2006 using the six metrics described in the text. Figure from, and additional details are at www.eco-check.org/reportcard/chesapeake/.

Addressing Exceedances of Criteria

Once criteria are established, the next step is to develop a methodology for determining allowable exceedances of the criteria. EPA (2003) has previously developed such guidance in detail, so only a brief synopsis is provided.

Several approaches can be used to determine allowable exceedances. They vary in how a reference curve is established. First, a fixed percentage of samples (e.g., 10 percent) can be allowed to exceed the established criteria. Such an approach does not take into account differential response to exceedances that might vary by season, for example. Second, laboratory or empirical data can be used to define a biologically based reference curve against which the frequency of exceedances can be compared. Third, a reference curve developed from known statistical or analytical error can be developed (Figure 4.22). In each case, values above these allowable exceedances would be deemed unallowable and would require action.

In some cases, such as those illustrated by Figure 4.22 (top), the area of unallowable exceedance is large, and thus any decision about impairment or nonattainment of designated use is clear. Where the area of unallowable criteria exceedance of the curve is smaller, a decision about impairment might be more difficult.
The percentage of allowable exceedance can also vary with individual parameters and criteria; for example, 10 percent may be acceptable for one parameter, but a more stringent percentage may be appropriate for another parameter, such as a biological parameter that has a more narrow habitat requirement. Moreover, allowable exceedances may differ for different segments of an estuary. Where data permit, the spatial extent of exceedances may be evaluated on the basis of estuarine segments, or even by individual cells. Such a segmentation approach is necessary where spatial variation is large and where data availability is substantial.

In Summary

This document illustrates the various processes by which estuaries can be categorized, the complexity of pathways of nutrient responses, and how various approaches for criteria can be derived and evaluated. Candidate causal and response variables to be used in criteria derivation can vary depending on estuarine type and available data.

Estuaries differ in a number of characteristics that pro-

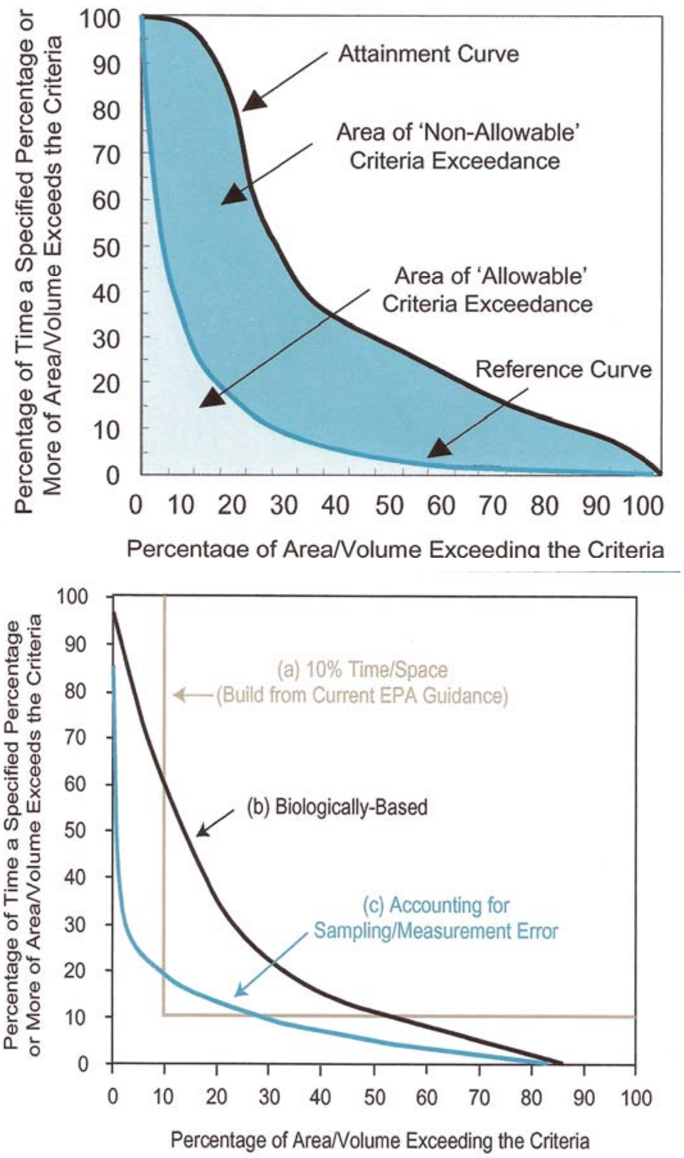


Figure 4.22. Graphical depictions of reference curves and allowance exceedances. The top panel shows how exceedances can be defined relative to a reference curve. The dark area indicates impairment (i.e., exceedances above that which is allowable). The bottom panel shows three different methods for estimating the reference based on (a) a fixed 10 percent allowable exceedance, (b) a biologically determined curve, and (c) an analytical or measurement based curve. Values above those lines would be considered unallowable exceedances. From USEPA 2003.

foundly affect their ecological characteristics. Thus, no single criterion method works across all estuarine systems. These differences influence the natural water quality conditions of estuaries, which water quality problems are most likely to develop, and the relative sensitivity of the ecosystem to these problems. In many or even most estuaries, nutrient enrichment effects occur within the context of other important ecological changes that have been imposed as a result of human activities. From broad land use changes in the watershed to local effects such as shoreline modifications, dredging and fishing activities, human activities have effects on water quality. Fishing activities can affect the ecosystem both via the trophic effects associated with fish and shellfish removals and from the fishing process itself, such as effects of dredging and trawling on benthic communities and SAV habitats. Although estuaries might have unique attributes, many features are also relatively common across groups of estuaries. Recognizing the commonalities through application of an appropriate classification, such as described in Chapter 3, will assist efforts to manage nutrients through a consistent, but not uniform, strategy. Many useful approaches have been developed, each serving different purposes. The approaches are variable because the estuaries are variable. Which metrics are chosen for application in nutrient criteria derivation will also depend on available information and the designated use of the system under consideration.

In practice, the derivation of numeric estuarine criteria for each estuary will depend on the availability of existing and historical data, the capability for monitoring and types of parameters that can be assessed. As additional information is gathered, particularly about the rates of processing of nutrients within systems, and as the monitoring data record becomes larger, criteria can be

Table 4.6. The goals of ecosystem based management, as described by the Ecological Society of America (from Christensen et al. 1996, p. 665).

- **Sustainability:** Ecosystem management does not focus primarily on deliverables but rather regards intergenerational sustainability as a precondition.
- **Goals:** Ecosystem management establishes measurable goals that specify future processes and outcomes necessary for sustainability.
- **Sound Ecological Models and Understanding:** Ecosystem management relies on research performed at all levels of ecological organization.
- **Complexity and Connectedness:** Ecosystem management recognizes that biological diversity and structural complexity strengthen ecosystems against disturbance and supply the genetic resources necessary to adapt to long-term change.
- **The Dynamic Character of Ecosystems:** Recognizing that change and evolution are inherent in ecosystem sustainability, ecosystem management avoids attempts to freeze ecosystems in a particular state of configuration.
- **Context and Scale:** Ecosystem processes operate over a wide range of spatial and temporal scales, and their behavior at any given location is greatly affected by surrounding systems. Thus, there is no single appropriate scale or timeframe for management.
- **Humans as Ecosystem Components:** Ecosystem management values the active role of humans in achieving sustainable management goals.
- **Adaptability and Accountability:** Ecosystem management acknowledges that current knowledge and paradigms of ecosystem functions are provisional, incomplete, and subject to change. Management approaches must be viewed as hypotheses to be tested by research and monitoring programs.

refined. Thus, as described in Chapter 2 and highlighted in Table 4.6, the principles of ***ecosystem-based managment*** should be applied.

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Chapter 5:

1. Overview of the Case Studies

Figure 5.1.1. Some coastal lagoons are particularly susceptible to eutrophication as the following case studies illustrate. Photo by P. Glibert.

Patricia M. Glibert
Jonathan H. Sharp
Christopher J. Madden
Walter Boynton and
Cynthia Heil

In Chapter 3 the classification of estuaries was demonstrated using a number of different approaches. In this chapter, 10 detailed case studies are presented, representing two of these estuarine types, ***river-dominated*** and ***lagoonal*** (Table 5.1.1, Figure 5.1.2). The summaries are specifically focused on the status and trends in DO, turbidity and light, nutrients, chlorophyll *a* and primary production, phytoplankton blooms, benthic primary producers and long-term changes. These case studies serve to illustrate the wide range of responses and demonstrate that there are several commonalities or

Table 5.1.1. Size comparisons of the 10 estuaries considered in this collection of case studies. The data are from Bricker et al. 2007 or from the following individual chapters. For San Francisco Bay, statistics for North Bay only are given here. Shading differentiates river-dominated from lagoonal systems.

Name	Classification	Water surface area (km^2)	Depth (m)	Volume (km^3)	Residence time (days)	Watershed (km^2)	Ratio (watershed: surface area)
Narragansett Bay	River-dominated	416	8.3	3.5	26	4,310	10
Delaware Estuary	River-dominated	2,070	6.1	13	8	33,254	16
Chesapeake Bay	River-dominated	6,974	7.3	51	90–180	79,584	11
Neuse River Estuary	River-dominated	456	2.7	1.3	~70	14,066	31
North San Francisco Bay	River-dominated	837	6.7	5.6	2–29	114,323	137
Yaquina Bay	River-dominated	14	2.1	0.03	1–9	650	45
Barnegat Bay	Lagoonal	280	1.5	0.1	24–74	1,730	7.7
Coastal Bays	Lagoonal	335	0.7–1.2	0.7	10–60	487	1.5
Florida Bay	Lagoonal	2,220	1–2	1.0	90–210	905	0.5
Pensacola Bay	Lagoonal	370	3.0	1.4	~10	17,650	37

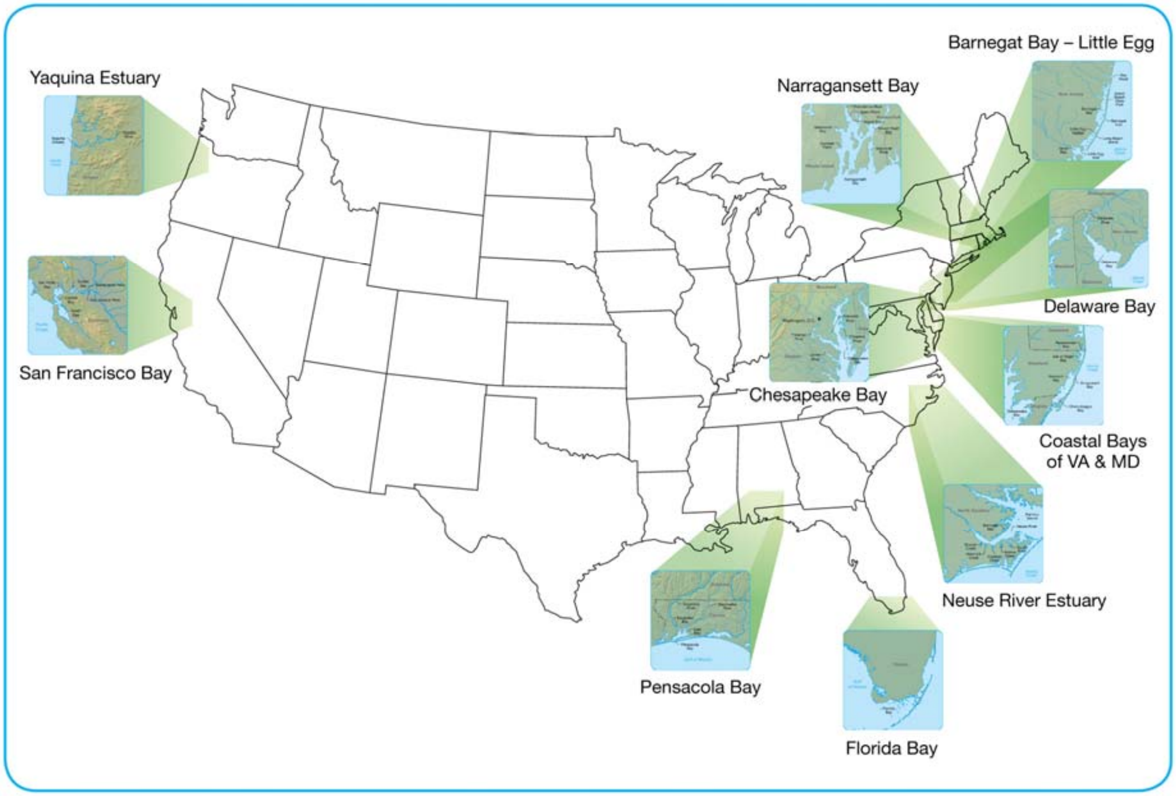


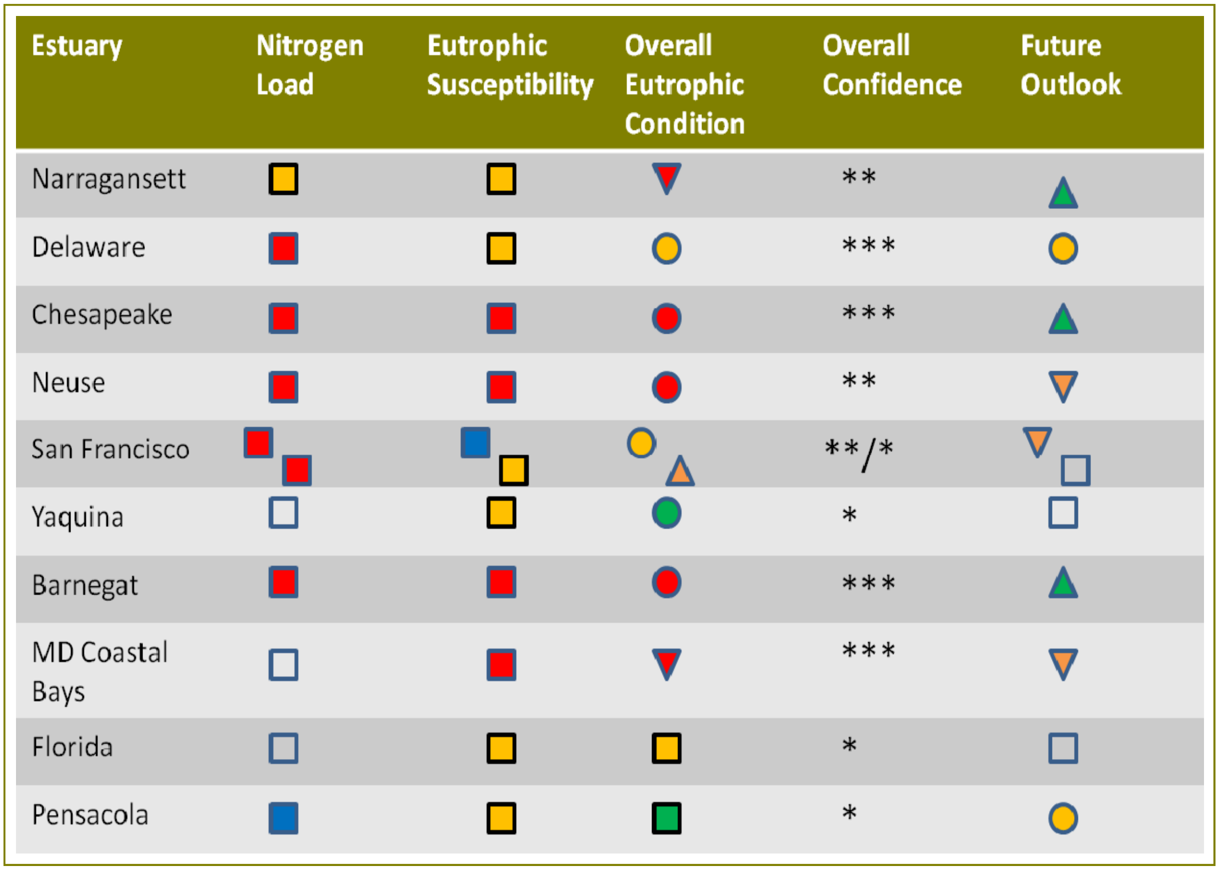
Figure 5.1.2. The 10 estuaries for which detailed case studies are presented in this chapter.

fundamental properties that are specific to each estuarine type. These commonalities are summarized in Chapter 4.

The selection of these case studies was not from a systematic evaluation of all estuaries in the United States, nor were they chosen to represent all types of estuaries. However, they represent many of the major estuaries of the United States and are among those that have extensive databases. They also include those with drainage to the Atlantic and Pacific oceans and the Gulf of Mexico. The case studies include six that can be classified as ***river-dominated***, Narragansett Bay, Delaware Bay, Chesapeake Bay, Neuse River Estuary, San Francisco Bay and Yaquina Bay; and four that can be classified as ***lagoonal***, Barnegat Bay-Little Egg Harbor, the Coastal Bays, Florida Bay and Pensacola Bay (Table 5.1.1); they are presented in that order. The river-dominated systems also illustrate an additional dimension of the complexity of estuarine classification; that is, those systems tend to have characteristics of river-dominated systems at the head of the estuary but to greater or lesser extents, transition to systems that are considered similar to a ***coastal embayment*** in character at the mouth.

The case studies presented here further provide examples of systems that vary widely in size, depth, and watershed area (Table 5.1.1). For example, the drowned river valley or tectonic estuaries range in size from the Chesapeake and Delaware Bay Estuaries, with water surface areas from 2,000 to approximately 7,000 km^2, to the much smaller Yaquina Bay, Narragansett Bay, and Neuse Estuary, at 14 to approximately 450 km^2. Of the lagoonal system examples given here, Barnegat Bay, the Coastal Bays and Pensacola Bay are all of similar size, approximately 400 km^2, whereas Florida Bay is about four times larger. Those systems also have considerable divergences in watershed size. For example, in the river-dominated systems, the watershed area of San Francisco Bay and Chesapeake Bay are approximately 80,000 to 114,000 km^2, while at the other extreme, the Yaquina Estuary drains a very small watershed area, only about 600 km^2. The lagoonal systems also vary in watershed size, with Pensacola Bay draining the largest area, and Florida Bay and Coastal Bays draining areas nearly two orders of magnitude smaller. Those systems also vary considerably in the population or land use of the watershed. Two of the smallest estuaries, San Francisco Bay and Narragansett Bay, have the highest ratio of population to surface water area, > 6,000 and 3,800 people per km^2, respectively, while the two largest estuaries, Chesapeake Bay and Delaware Bay, vary widely in their respective population density, 900 and 3,500 people per km^2, respectively. Yaquina Estuary has the lowest population density, with 12 people per km$^{2.}$ Of the case studies given for the lagoonal systems, the Coastal Bays, Florida Bay, and Pensacola Bay are not heavily urbanized but drain considerable agricultural land, whereas Barnegat Bay–Little Egg Harbor drains a more urbanized watershed.

Table 5.1.2. Comparison of nitrogen load, eutrophic susceptibility, overall eutrophic condition and future outlook for the case studies presented here based on the National Estuarine Eutrophication Assessment (from Bricker et al. 2007). For additional explanation, see the text. Red symbols indicate high load, susceptibility or condition, yellow moderate, green moderate to low, and blue slight. Clear symbols indicate insufficient data to evaluate. For eutrophic condition and outlook, circles indicate no change since the 1999 eutrophication assessment (Bricker et al. 1999), up arrows (green) indicate improvement, and down arrows (orange) indicate worsening conditions, but small change relative to current condition. Overall confidence reflects the availability of data for these assessments. Two entries are given for San Francisco Bay to distinguish north San Francisco Bay (upper entry) and south San Francisco Bay (lower entry).



Estuary	Nitrogen Load	Eutrophic Susceptibility	Overall Eutrophic Condition	Overall Confidence	Future Outlook
Narragansett	🟨	🟨	🔻	**	▲
Delaware	🟥	🟨	🟡	***	🟡
Chesapeake	🟥	🟥	🔴	***	▲
Neuse	🟥	🟥	🔴	**	▽
San Francisco	🟥 🟥	🟦 🟨	🟡 △	**/*	▽ ☐
Yaquina	☐	🟨	🟢	*	☐
Barnegat	🟥	🟥	🔴	***	▲
MD Coastal Bays	☐	🟥	🔻	***	▽
Florida	☐	🟨	🟨	*	☐
Pensacola	🟦	🟨	🟩	*	🟡

In a few cases, the estuarine examples provided here also illustrate the effects of changes to a system. Both the Neuse Estuary and Florida Bay have experienced a series of severe hurricanes in the past decade, and both systems had significant ecosystem responses to these events. In the cases of San Francisco Bay, Delaware Bay and Coastal Bays, nutrient loading was altered over the course of the past one to two decades from improvements in sewage treatment, and that led to ecosystem alterations. Thus, collectively the examples illustrate a wide range of nutrient loading and responses and a wide range of eutrophic status and trends.

The estuaries described here have also been examined in the National Estuarine Eutrophication Assessment (Bricker et al. 2007, for details on assessment methodology, see also Chapter 2). A synopsis of those findings is given in Table 5.1.2. for comparison. In general, N loads were considered a critical factor in the development of eutrophic conditions. N loads were rated as high (> 80 percent from land), medium (20–80 percent from land) or low (< 20 percent from land). Susceptibility also accounted for the natural flushing time of the water. The overall eutrophic condition (Table 5.1.2) was based on evaluation of occurrence, spatial extent and frequency of primary (chlorophyll *a* and macroalgae) and secondary (loss of SAV, increased HABs, and decreased DO) symptoms. Future outlook projections were based on the anticipated human population change or expected change in wastewater treatment, agriculture or urban runoff by the year 2020.

For the case studies described, all have sufficiently long time series records to evaluate the overall condition and changes therein. From the individual following descriptions and from the analyses of the same estuaries by Bricker et al. (2007), it can be seen that all are moderately to highly susceptible to eutrophication (Table 5.1.2), but the degree to which eutrophication is expressed varies widely.

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Chapter 5:

2. Narragansett Bay

Theodore J. Smayda and
David Borkman

General Description

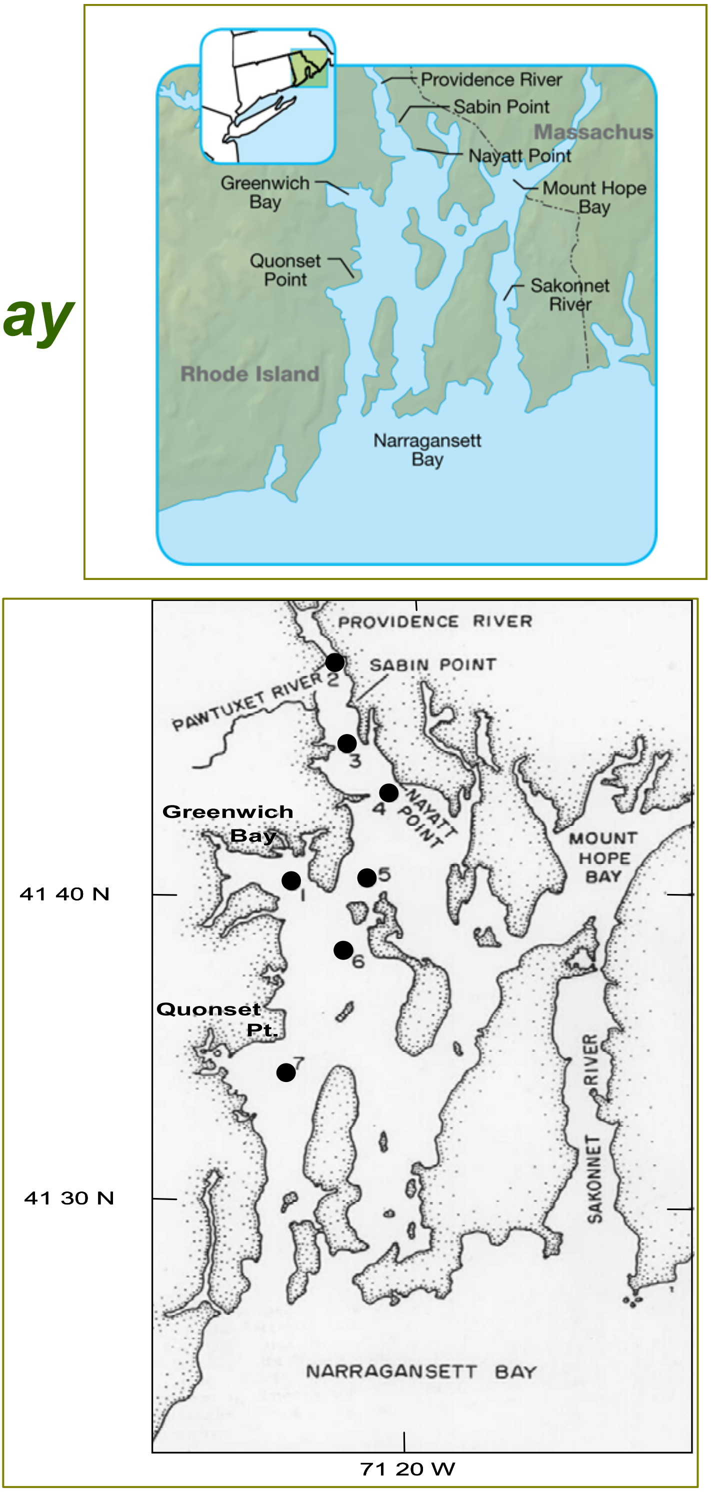

Figure 5.2.1. Two maps of Narragansett Bay. The top map indicates its broader location, and the bottom map shows the monitoring stations, 1–7. Station 7 is the long-term (1959–present) monitoring site.

Narragansett Bay (about 416 km^2) is a well-mixed, relatively shallow (mean depth 8.3 m), estuary southwest of Cape Cod along the eastern U.S. coast (41° 30'N, 71° 20'W) and contiguous with Rhode Island and Long Island sounds (Figure 5.2.1). It is within the coastal region extending from Maine to Virginia characterized by *extensive geographic continuity* in meteorology and climatic trends, including anomalies in air temperature, precipitation, wind stress and irradiance (Ingham 1982). Freshwater flow into upper Narragansett Bay via the Providence River estuary produces a salinity gradient that increases down Narragansett Bay from 20 up-bay to 33 at its entrance (Kremer and Nixon 1978; Smayda and Borkman 2007). The mean residence time of Narragansett Bay water is 26 days, varying from 10 to 40 days dependent on the volume of freshwater input and wind conditions (Pilson 1985). Tidal currents dominate the circulation in which higher salinity water flows into Narragansett Bay from Rhode Island Sound through East Passage, and lower salinity water flows southward exiting through West Passage (Hicks 1959). Recent studies reveal a more complex physical oceanography than reported by those previous investigations (Kincaid and Pockalny 2003).

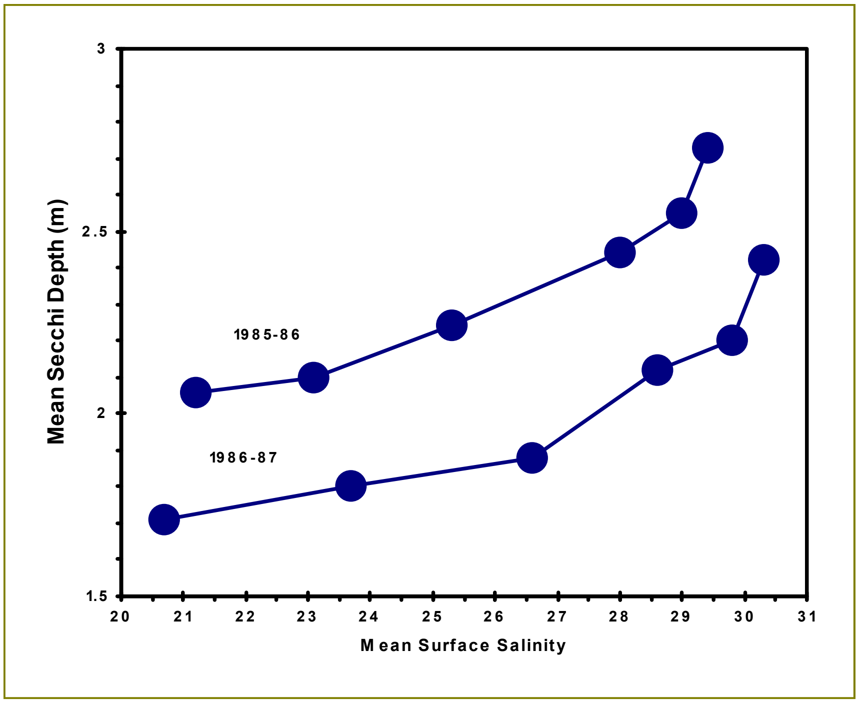

Figure 5.2.2. Secchi disk depth as a function of salinity along a six-station north-south (Providence River to Bay mouth) transect during 1985–1986 and 1986–1987. Each data point is the mean of all observations at each station in each year (n = 34 transect surveys during 1985–1986 and n = 28 transect surveys during 1986–1987).

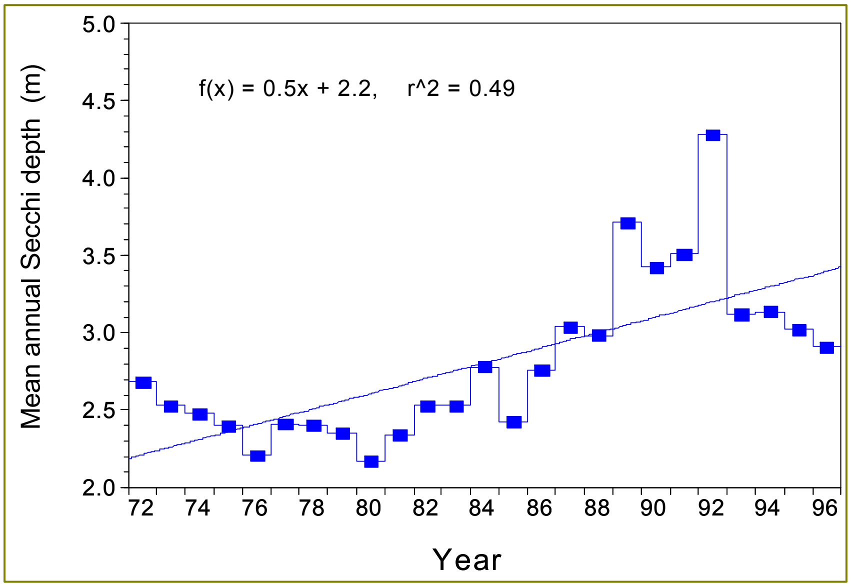

Figure 5.2.3. Mean annual Secchi depth at the long-term monitoring station in lower Narragansett Bay, 1972–1996. From Borkman and Smayda 1998.

Dissolved Oxygen

Low DO levels are primarily a problem in upper Narragansett Bay during the summer months. Most of Narragansett Bay has DO concentrations > 4.8 mg L^{-1} (USEPA 2004), with values falling below that threshold mainly in summer in the upper bay and in semi-enclosed coves (Deacutis et al. 2006). Summer hypoxia in the upper bay is elevated during periods of reduced tidal flushing associated with neap tides (Bergondo et al. 2005). From 18 to 25 percent (based on area) of Narragansett Bay experienced hypoxia during the summers of 2001 and 2002, respectively, with hypoxia greater in the dry summer of 2002 (Deacutis et al. 2006). Reduced estuarine circulation and concomitant increased residence time during the dry summer (2002) might have contributed to increased summer hypoxia that extended southward of Quonset Point in the summer of that year (Deacutis et al. 2006). Such acute low DO events can be detrimental to some organisms, with hypoxia the putative cause of a fish and shellfish die-off event in an enclosed bay (Greenwich Bay) during August 2003 (RI DEM 2003) and a mussel (*Mytilus edulis*) die-off event during summer 2001 (Altieri and Witman 2006). Benthic organism abundance, community composition and sediment redox potential have been established from sediment profile camera images (Valente et al. 1992). They have distributions similar to the distribution of summer hypoxia, which suggests a long-term ecological effect on Narragansett Bay benthos (Deacutis et al. 2006).

Turbidity and Light

Narragansett Bay has clear water relative to other estuaries in the contiguous Virginian province extending from Cape Cod south to Chesapeake Bay (USEPA 2001, 2004). Within Narragansett Bay, there is a spatial gradient in water clarity, which increases by about 40 percent along the salinity gradient from the Providence River (Secchi depth of 1.5 to 2.0 m) southward to near the mouth of the bay where Secchi depth is typically 2.5 to 3.0 m (Smayda and Borkman 2007). That gradient reflects the transition from turbid, nutrient-enriched, low-salinity water under riverine influence in the upper bay (Providence River) to clear, lower nutrient, higher salinity water in the lower bay (Figure 5.2.2).

A long-term increase in water clarity occurred in lower Narragansett Bay between 1972 and 1996. On the basis of weekly observations at the long-term monitoring site (station 7 in Figure 5.2.1), water clarity increased about 30 percent, from a mean Secchi depth of 2.4 m (1972–1983) to 3.1 m in 1984–1994 (Figure 5.2.3; Borkman and Smayda 1998). Some of the increase in water clarity could be related to declining TSS loading from wastewater treatment plants (Borkman and Smayda 1998) and decreased chlorophyll *a* levels during the early 2000s (Li and Smayda 1998, 2001). Water clarity in lower Narragansett Bay has remained similar to that observed in the mid-1990s. In some years, the timing of the winter-spring phytoplankton bloom in Narragansett Bay is regulated by light availability, with the bloom beginning only after attainment of a minimum light threshold (Hitchcock and Smayda 1977). The recent trend of increased water clarity (Borkman and Smayda 1998) suggests an earlier release of light limitation of the winter–spring bloom and of phytoplankton growth generally that can influence both the temporal (timing) and spatial patterns of blooms in Narragansett Bay.

Nutrients

Annual Nutrient Cycles

Annual nutrient (PO_4^{-3}, NO_2^-, NO_3^-, NH_4^+, $Si(OH)_4$) cycles for lower Narragansett Bay are well characterized (Pilson 1985; Smayda unpublished). Pilson (1985) summarizes surface nutrient concentrations in the lower bay (GSO dock) on the basis of 5 years of weekly sampling. Dissolved inorganic P concentrations (PO_4^{-3}) ranged from near 0 (detection limit) to about 2.5 µM-P L^{-1}. Minimal concentrations (~0.5 to 1 µM-P L^{-1}) were observed during the winter-spring bloom (March to April), with levels rapidly rebounding to 1.5 to 2.0 µM-P L^{-1} during the summer (July to August), and then gradually declining to ~1 µM-P by December. Total inorganic nitrogen (DIN) concentrations ranged from near 0 to ~20 µM-N L^{-1}, with a distinct winter-spring to early summer decrease during March through July (total DIN < 4 µM-N L^{-1}), and a late-summer through early autumn increase that peaked (about 10 µM-N L^{-1}) during November through January. There is a seasonal change in the dominant form of DIN; during winter (November to February) NO_2^- + NO_3^- comprise most of the DIN, while during summer (May to September), NH_4^+ contributes a larger proportion.

The $Si(OH)_4$ annual cycle is highly correlated with the seasonal diatom abundance pattern, with diatom growth being limited by $Si(OH)_4$ availability in winter-spring and summer blooms (Pratt 1965; Smayda 1973, 1974). There is a seasonal variation in terrestrial Si-uptake in the Narragansett Bay watershed, which influences the $Si(OH)_4$ annual cycle in Narragansett Bay (Fulweiler and Nixon 2005). The interaction of seasonal variation in watershed delivery of Si to Narragansett Bay and diatom Si uptake results in an annual $Si(OH)_4$ cycle that features elevated levels > 10 µM-Si L^{-1} during autumn and early winter, followed by a decline to about 1 to 2 µM-Si L^{-1}, or less, during the winter-spring diatom bloom in March to

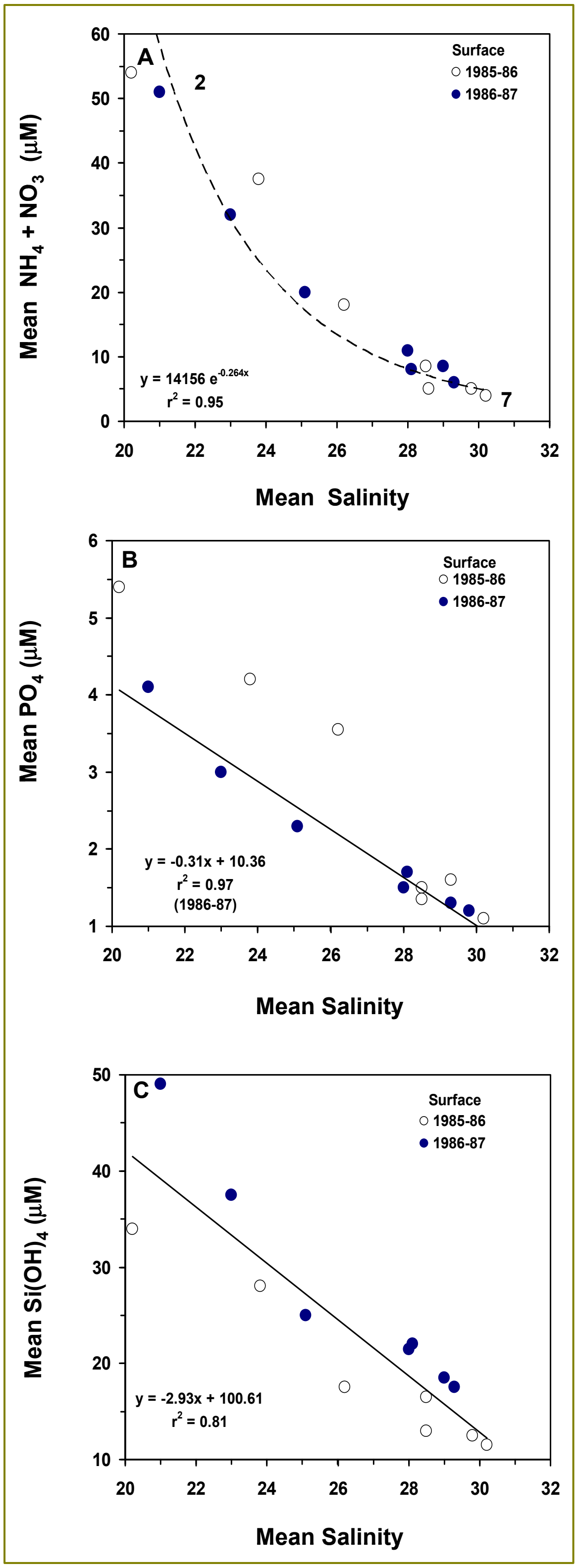

Figure 5.2.4. Nutrient versus salinity relationships for DIN, P and Si along the seven-station transect route in Narragansett Bay. From Smayda and Borkman 2007.

April, and recovery to ~10 µM-Si L^{-1} during summer (Pilson 1985). In some years, summer diatom blooms depress Si levels to < 1 to 2 µM-Si L^{-1} in July to August (Smayda 1974).

Regional Nutrient Gradients

Nutrient annual cycles in upper Narragansett Bay tend to follow the pattern found in lower Narragansett Bay but exhibit a greater seasonal amplitude and elevated nutrient concentrations. For example, the mean annual P, DIN and Si cycles in Mt. Hope Bay (Figure 5.2.1) during 1972–1983 closely tracked the mean patterns seen in lower Narragansett Bay during 1977–1982 (Pilson 1985). The dominant N, P and Si sources (riverine and waste water treatment plant input) are in the upper regions of Narragansett Bay (Nixon et al. 1995). The nutrient distributions exhibit a pronounced down-bay decrease in mean concentration that was strongly coupled to the salinity gradient (Figure 5.2.4). The inverse correlations found between nutrient concentrations and salinity are highly significant statistically (Smayda and Borkman 2007). The nutrient gradient is driven primarily by the copious discharge of riverine and sewage effluent nutrients into the Providence River estuary, with the zone of initial dilution found in the region of stations 2, 3 and 4 (Figure 5.2.1). The region between stations 4 and 5 transitions into the lower nutrient conditions found down-bay, with the gradients set up by the progressive dilution and uptake of nutrients by phytoplankton as the enrichment plume moves down-bay.

While the Providence River estuary (stations 2, 3) functions as a major anthropogenic nutrient pump, delivering N, P and Si that are then transported down-bay, a secondary source of Narragansett Bay *new* nutrients is delivered at the entrance into Narragansett Bay where NO_3^- is advected through inflow of enriched bottom water and transported up-bay from station 7 (Nixon et al. 1995; Culver-Rymsza 1988). The salinity distribution along the horizontal and vertical axes of Narragansett Bay proxies this dual system of nutrient input. Nutrients recycled by food web dynamics along the salinity gradient supplement these two major input mechanisms (Vargo 1976; Verity 1985). The relative importance of the physically and biologically regulated nutrient fluxes along the gradient changes regionally and seasonally. In lower Narragansett Bay, *in situ* biological recycling and offshore input of nutrients become progressively more important along the gradient than the down-bay nutrient flux from the Providence River estuary. The very high nutrient concentrations that persist in upper Narragansett Bay, and the low concentrations near the mouth of Narragansett Bay, with intermediate concentrations along the down-bay salinity gradient, reflect those differing nutrient accretion mechanisms, which Nixon et al. (1995) considered in their assessment of nutrient mass balance in Narragansett Bay.

The down-bay gradient in differential nutrient accretion and use produces a gradient in mean nutrient ratios in Narragansett Bay. Nutrient ratios and their shifts are of interest because they influence phytoplankton *functional group* selection, which can have significant differences in their physiology and ecological effects. The primary nutrient expected to regulate the shift in functional groups from diatoms to flagellates is Si (Officer and Ryther 1980; Smayda 1990). Diatoms are the major phytoplankton component driving productivity in Narragansett Bay (Pratt 1959, 1965), with evidence that long-term shifts in their abundance (Borkman 2002) and altered diatom: flagellate abundance ratios (Smayda and Borkman unpublished) have occurred. Over two annual cycles (1985 –1987) in Narragansett Bay, mean annual N:P and N:Si ratios were strongly and inversely correlated with mean salinity (Figure 5.2.5; Smayda and Borkman, 2007). The mean annual N:P ratio (using $NH_4^+ + NO_3^-$) progressively decreased down-bay, from 10:1 to 12.5:1 (Providence River, station 2) to 3.2:1 to 4.9:1 (lower Narragansett Bay, station 7). Upper Narragansett Bay, on average, is more sensitive to the amount of P available relative to N, while lower Narragansett Bay, particularly the region extending down-bay from station 4, becomes progressively more N-sensitive. The mean annual N:Si ratio

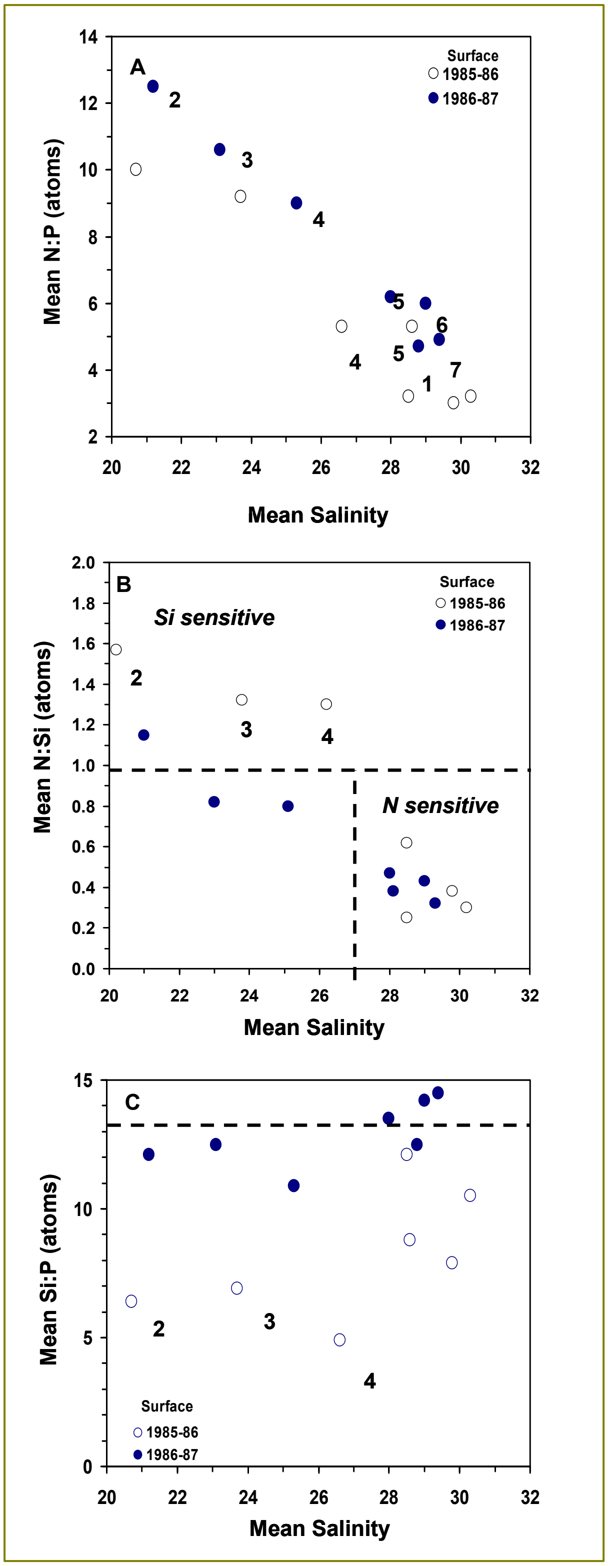

Figure 5.2.5. Mean ratios (by atoms) of (A) N:P, (B) N:Si and (C) Si:P along the salinity gradient in Narragansett Bay.

(using $NH_4^+ + NO_3^-$) progressively decreased along the salinity gradient, and was inversely correlated with mean salinity (Figure 5.2.5). The N:Si ratio progressively decreased from about 1.2:1 to 1.6:1 in the upper bay to about 0.3:1 in the lower bay. Narragansett Bay, on average, is regionally partitioned into an upper, Si-sensitive region that extends from Nayatt Point (station 4; Figure 5.2.1) up into the Providence River estuary, and a strongly N-sensitive mid- and lower-bay region that extends down-bay south from the mid-bay region near Prudence Island. The Si:P gradient displayed greater interannual variability, and unlike the N:Si and N:P ratios, increased down-bay with mean salinity.

Collectively, the three nutrient ratios indicate that, on average, upper Narragansett Bay is more sensitive to P and Si availability relative to N, while mid- to lower-Narragansett Bay is more sensitive to N limitation relative to P and Si.

Chlorophyll and Primary Production

Chlorophyll

Chlorophyll *a* levels are moderate; most areas of Narragansett Bay have chlorophyll *a* concentrations of < 5 µg L^{-1}, which is similar to those observed in other northeastern U.S. estuaries (USEPA 2004). An apparent long-term decline in chlorophyll *a* levels has occurred (Li and Smayda 1998, 2001). Annual mean chlorophyll *a* concentrations at the long-term monitoring station in the lower West Passage of Narragansett Bay fell from about 6 to 7 µg L^{-1} in the early 1970s to about 4 µg L^{-1} during the late 1980s; a decline of about 45 percent during the 18 years of weekly observations (Li and Smayda 1998, 2001). Chlorophyll *a* levels in the lower bay have remained at these lower levels (~3 to 4 µg L^{-1}) during the 1990s and early 2000s, with the exception of 1995 and 1996 (unpublished GSO/URI Narragansett Bay monitoring data). The long-term (1970–2005) decline in chlorophyll *a* (Figure 5.2.6) is shown combining the 1970s to 1990s data of Li and Smayda (1998) and unpublished data from 1999 to 2005. With the exception of the early 1990s when chlorophyll *a* showed a general increase during several unusually cold years, chlorophyll *a* levels

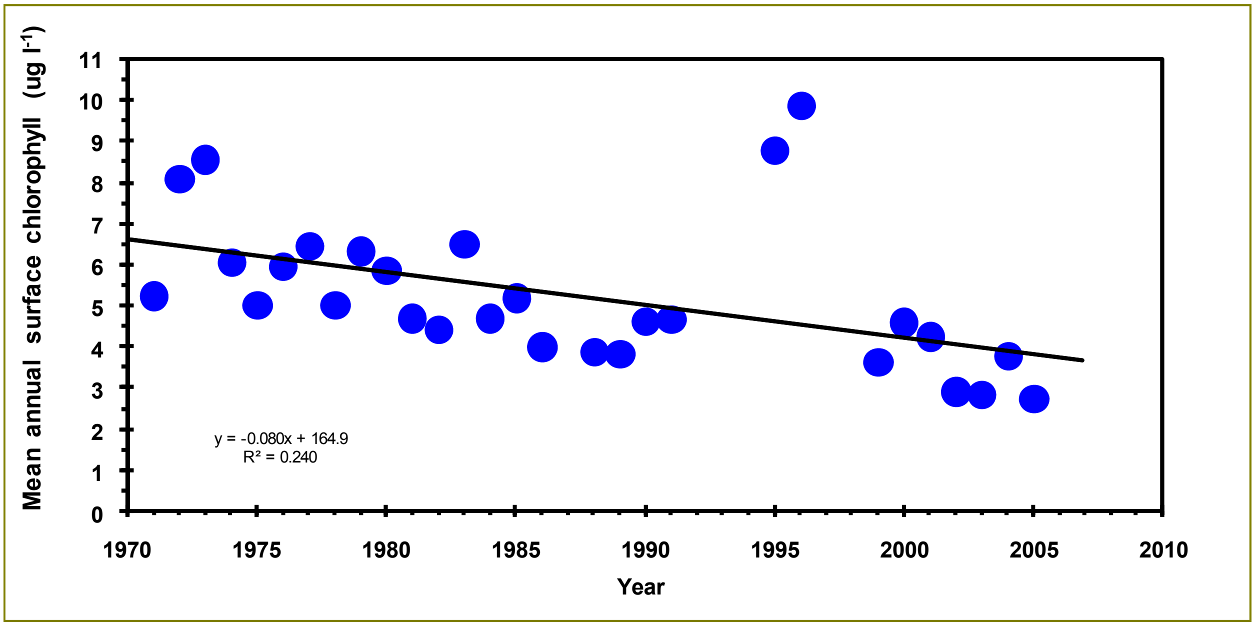

Figure 5.2.6. Long-term decline in mean annual surface chlorophyll a concentration in lower Narragansett Bay, 1970–2005. Annual mean chlorophyll levels calculated from weekly observations using the 1970–1996 data from Li and Smayda 1998 and Smayda unpublished, and 1999–2005 data from URI/GSO monitoring.

have declined linearly at a rate of about 0.1 µg L^{-1} y^{-1} from 1970 to 2005 (Figure 5.2.6). Part of the long-term decline appears to be a decrease in winter-spring bloom chlorophyll *a* levels. Winter-spring chlorophyll *a* concentrations were elevated (peaks > 20 µg L^{-1} were common in the lower bay) and dominated the annual chlorophyll *a* cycle in the 1970s (Pilson 1985) and earlier (Li and Smayda 1998), but appear to have declined in the 1990s (Oviatt et al. 2002).

There is a distinct gradient in chlorophyll *a* concentration in Narragansett Bay, with greatest concentrations in the mid-bay, and declining levels both seaward and up-bay (Figure 5.2.7). Elevated chlorophyll *a* concentrations of > 20 μg L^{-1} are commonly observed in the upper bay. The gradient in nutrient concentration, available light and flushing rate sets up the observed hyperbolic gradient in chlorophyll *a* (Smayda and Borkman 2007). The assimilatory capacity of the upper regions of the Narragansett Bay (salinity < 22; Providence River region) appears to be exceeded and has reduced chlorophyll *a* levels. The mid-bay intermediate salinity region (salinity = 23–28) typically has the maximum chlorophyll *a* levels that are about twice those observed either up-bay or seaward of the mid-bay chlorophyll *a* maximum (Figure 5.2.7).

Phytoplankton Primary Production

Narragansett Bay is a productive estuary—the bay-wide mean annual phytoplankton production rate is about 300 g C m^{-2} y^{-1}. Annual production rate estimates vary with station location and methods, but most cluster around values of 250 to 325 g C m^{-2} y^{-1}. Early estimates, based on O_2 production (light and dark bottles) at several stations, yielded a bay-wide average annual production of 270 g C m^{-2} y^{-1} (Oviatt et al. 1981). Later in the 1970s, an estimate for the lower West Passage of Narragansett Bay, based on weekly ^{14}C measurements, gave an estimate of 310 g C m^{-2} y^{-1} (Furnas et al. 1976). Three annual primary production surveys based on ^{14}C method in the lower West Passage yielded annual estimates of 150 to 250 g C m^{-2} y^{-1} (Keller 1989). More recently (1997 –1998), biweekly measurements at three stations using the ^{14}C method yielded a bay-wide average of 323 g C m^{-2} y^{-1} (Oviatt et al. 2002).

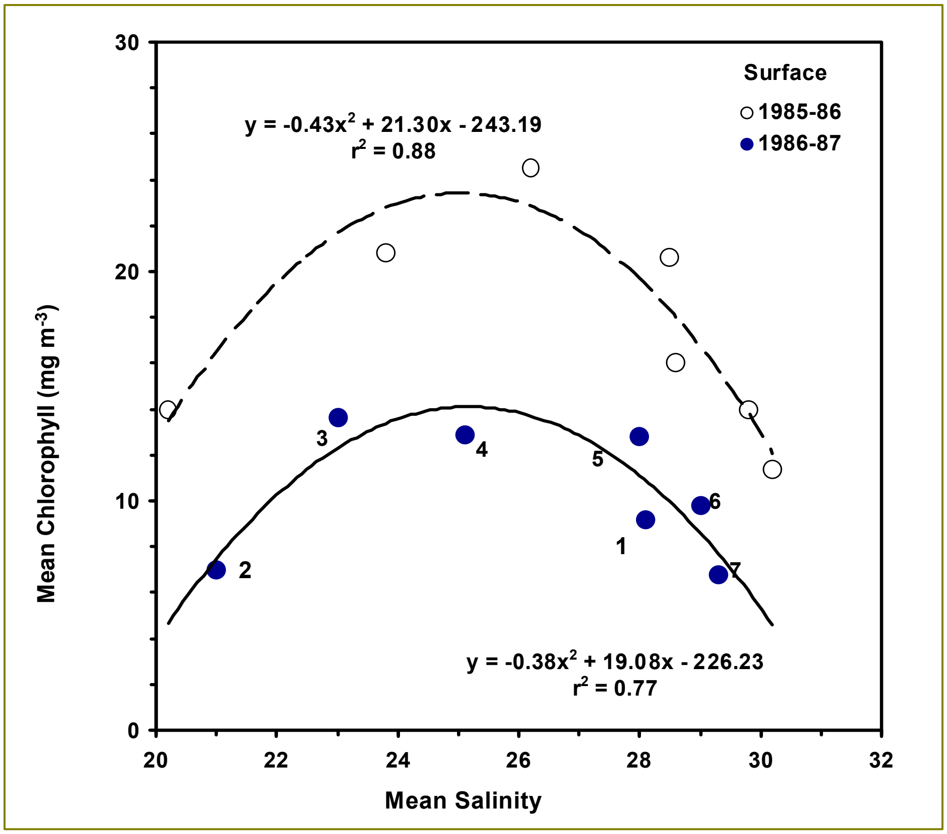

Figure 5.2.7. Gradient in Narragansett Bay chlorophyll a*. Mean annual concentration at seven stations along the salinity gradient from the head of the bay (Providence River, station 2) to near the mouth of the bay (station 7) during two survey years (1985–1986 and 1986–1987). From Smayda and Borkman 2007.*

The seasonal primary production pattern in Narragansett Bay is dominated by elevated summer production (Durbin and Durbin 1981; Oviatt et al. 1986; Smayda unpublished data). Summer (June, July, August) daily production rates during a 1976 survey (Durbin and Durbin 1981) were 2 to 4 g C m^{-2} d^{-1} in the lower bay, compared to 1 to 2 g C m^{-2} d^{-1} observed during winter, spring and autumn. That pattern in which summer production rates were roughly two-fold higher than during the rest of the year was also observed in mid- and upperNarragansett Bay (Durbin and Durbin 1981). Evidence of a summer production peak was also seen in a 28-month mesocosm experiment (Oviatt et al. 1986). However, respiration (R) also increased during the warm summer months, such that the production (P):R ratio was maximized during the cold winter months (February) despite the relatively low rate of P observed during winter (Oviatt et al. 1986). A summer production peak was also found during a bi-weekly survey during 1997–1998 when rates were about two-fold (lower bay) to six-fold (upper bay) greater than observed in the winter (Oviatt et al. 2002).

Phytoplankton Blooms and Species Composition

The composition and dynamics of the phytoplankton and zooplankton in Narragansett Bay, generally, are similar to that reported for contiguous Rhode Island and Long Island sounds and northeastern U.S. coastal waters (Riley 1967; Smayda 1957, 1973, 1980; Malone et al. 1983; Marshall1976, 1978; Turner et al. 1983; Townsend and Cammen 1988). Bloom dynamics in Narragansett Bay primarily are diatom-driven, and largely reflect the bloom and successional behavior of *Skeletonema costatum (*Karentz and Smayda 1984, 1998). The annual bloom cycle of this species has been shifting, with the result that the annual bloom maximum has recently become a summer event, rather than a winter-spring event (Borkman and Smayda 2009). Blooms of HABs (Hargraves and Maranda 2002) occur unpredictably during the period from May through September (Li and Smayda 2000; Smayda and Villareal 1989a; Smayda 1998). This late-spring-summer bloom window has been termed the *open niche* period because it is unpredictable whether a diatom, dinoflagellate, raphidophyte, or some other phylogenetic species will bloom during this period. The duration and magnitude of the bloom are also unpredictable (Smayda and Villareal 1989b). Bloom species selection and bloom dynamics during this period appear to be regulated both (either singly or in combination) by nutrient conditions (Smayda unpublished) and the grazer community, with the ctenophore *Mnemiopsis leidyi*, through its predation on the summer copepod population, an important top-down regulator of bloom dynamics during the open niche period (Deason and Smayda 1982a, 1982b). The variable interannual abundance of *Mnemiopsis* during the open niche period precludes prediction of whether a diatom or flagellate will then bloom and, if a flagellate, which flagellate species will produce a harmful bloom or red tide. Diatom blooms tend to dominate during periods of ctenophore abundance, their blooms then a consequence of the grazing down of the copepod population, which facilitates diatom growth (Deason and Smayda 1982a, 1982b).

Tight benthic-pelagic coupling exists in Narragansett Bay, with clam (*Mercenaria mercenaria*) and mussel (*Mytilus edulis*) populations dominating the benthic filter-feeding community. A significant, temperature-dependent nutrient flux into the water column from benthic habitat processes occurs seasonally (Hale 1975). Hypoxia development in the upper half of Narragansett Bay (Altieri and Witman 2006) and brown tide blooms (Tracey 1988; Tracey et al. 1988) have periodically decimated the benthic bivalve community, particularly, and other benthic components. Comparative, inter-estuarine analyses (Boynton et al.1982; Cloern 1987; Monaco and Ulanowicz 1997) show that Narragansett Bay falls within the response gradient characteristic of coastal estuaries.

Benthic Primary Producers

Narragansett Bay has a diverse macroalgal flora, with at least 94 species present (Villalard-Bohnsack and Harlin 1992). The flora is dominated by red algae, with the Narragansett Bay macroalgal flora at, or near the southern limit of the temperate macroalgal species assemblage distribution (Villalard-Bohnsack and Harlin 1992). Macroalgal community composition might be changing, with recent additions of an introduced species (*Grateloupia doryphora*; Harlin and Villalard-Bohnsack 2001).

Macroalgae (predominantly *Ulva*) form benthic mats in many of the embayments and salt ponds in, and near Narragansett Bay during summer and early autumn (Thorne-Miller et al. 1983; Brush and Nixon 2003). Those layered, benthic *Ulva* mats can reach a biomass of up to 400 g m^{-2} (dry weight), with 100 to 400 g m^{-2} (dry weight) peak weight reported in Greenwich Bay (Brush and Nixon 2003) and a peak of 370 g m^{-2} (dry weight) reported in the coastal salt ponds of southern Rhode Island (Thorne-Miller et al. 1983). N isotope analyses have indicated that much of this biomass in upper Narragansett Bay is supported by anthropogenic N input (Pruell et al. 2006). Temperature-induced die-offs of the *Ulva* mats lead to dumping of organic matter to the bottom, with subsequent bacterial respiration that can contribute to hypoxia in some areas of upper Narragansett Bay (RI DEM 2000).

Long-Term Changes

Narragansett Bay is an ecosystem undergoing change. A long-term increase in water clarity has occurred coincident with a decrease in the discharge of TSS from sewage treatment plants. Between 1972–1996, Secchi disk depth (= water clarity) increased at a linearized rate of 0.05 m y^{-1} (Borkman and Smayda 1998). Most of the increase in transparency occurred during 1984–1994, when TSS loading decreased 75 percent (Borkman and Smayda unpublished data). Nutrient concentrations have also undergone long-term changes, with a 40 percent decline in PO_4^{-3} concentration between the early 1970s and the late 1990s, and a 40 percent increase in $Si(OH)_4$ concentration between the early 1980s and the late 1990s (Smayda unpublished). Annual mean chlorophyll *a* has decreased since the 1970s. That progressive decrease is inversely correlated with the North Atlantic Oscillation Index, which suggests that this response has been, at least partially, climatically driven (Li and Smayda 1998, 2001).

Major finfish and shellfish kills have recently occurred in Narragansett Bay, interpreted by some as responses to increasing nutrient enrichment and evidence of *creeping eutrophication*. Hypoxia has been specifically implicated as the cause of these die-offs, with the DO deficit widely believed to have been caused by the decomposition of poorly grazed phytoplankton blooms that have been stimulated by eutrophication (Deacutis et al. 2006; Deacutis 2008). That contention requires an assessment of the long-term trends in nutrient levels in Narragansett Bay. While hypoxia occurs in upper Narragansett Bay (Bergondo et al. 2005; Altieri and Witman 2006; Deacutis et al. 2006), its relationship to long-term changes in nutrient patterns, and its concentrations and seasonal cycling, remain to be analyzed. Roman et al. (2000) reconstructed the annual concentrations of total N and P entering into Narragansett Bay between 1900 and 1995, and Nixon (1997) has considered the relationship between the historic nutrient inputs and productivity in Narragansett Bay. While there is clear evidence for long-term changes in nutrient delivery into Narragansett Bay, the effect of this on recent ecological disturbances is obscure.

Narragansett Bay is in the transitional region between the Boreal (north of Cape Cod) and Temperate (south of Cape Cod) biogeographical zones. The location ideally positions Narragansett Bay to serve as a sentinel site for the detection of climate change effects on plankton blooms and other ecosystem features. It is precisely at such biogeographical boundaries where the effects of climate change are expected to be most readily detectable, and indeed this appears to be the case (Smayda et al. 2004; Smayda and Borkman 2008). Over a 40-year period, mean winter water temperature in Narragansett Bay has increased about 3 °C, a warming that has led to a major retraction and decrease in the winter-spring bloom behavior of the cryophilic (boreal) diatoms *Detonula confervacea* and *Thalassiosira nordenskioeldii*, previously the major winter-spring bloom species in Narragansett Bay (Smayda et al. 2004). A major shift in the annual phytoplankton maximum from winter-spring to summer has occurred along with reduced abundance of the dominant species in Narragansett Bay, *Skeletonema costatum* (Borkman 2002; Borkman and Smayda 2009). Those shifts correlate with changing temperature and long-term variations in the North Atlantic Oscillation Index (see also Smayda et al. 2004).

Some of the long-term decline in Narragansett Bay phytoplankton biomass (Figure 5.2.6) could be from the decline in the abundance of winter-spring diatom bloom species such as *Detonula confervacea* (Smayda et al. 2004) and changes in the bloom magnitude and timing of persistent (present year-round) diatoms like *Skeletonema costatum* (Borkman 2002). Warming Narragansett Bay temperature (Cook et al. 1998; Keller et al. 1999), especially during winter, with Q_{10} (the lowest one-day average flow that occurs once every 10 years)-related effects on winter zooplankton development and feeding rates (Durbin and Durbin 1992) have been implicated in the decline of the Narragansett Bay winter-spring phytoplankton standing stock (Keller et al. 1999). Narragansett Bay also is at the approximate northern distribution limit of the ctenophore *Mnemiopsis leidyi*, a voracious predator of zooplankton, and that in apparent response to the winter warming trend, might be appearing earlier in the annual plankton cycle and becoming more abundant (Sullivan et al. 2001; Costello et al. 2006). Thus, both top-down and bottom-up responses to climate change are taking place in Narragansett Bay.

Summary of Ecosystem Impacts

Narragansett Bay displays little evidence of severe ecosystem impacts, although current trends indicate that this ecosystem is in a state of change in response to changing climate and altered nutrient loading patterns. Effects of N loading, such as localized excess algal biomass and reduced DO, are evident mainly in the upper bay during the summer months. Nutrient loading appears to be the major anthropogenic modification of the bay, with N loading about 5-fold greater than that expected during *pristine* historical background levels (Nixon 1997). Phytoplankton has responded with an approximate three-fold increase in biomass and an approximate two-fold increase in primary production compared to pristine levels (Nixon 1997). Yet, more recent declines in Narragansett Bay phytoplankton biomass (Li and Smayda 1998) to levels that are now (in early 2000s) about twice those estimated for a pristine Narragansett Bay (Nixon 1992) could be partially in response to a declining N input, especially of DON (Nixon et al. 2005). Despite those anthropogenic N additions, phytoplankton production still appears to be light- and N-limited (Oviatt et al.1995).

Narragansett Bay, and northeast U.S. coastal waters generally, have warmed 1 to 3 °C since the 1960s (Cook et al. 1998; Nixon et al. 2004; Oviatt 2004). The effects of the warming are not fully known, but a zooplankton-mediated decline in winter-spring phytoplankton biomass has been suggested (Keller et al. 1999). Winter-spring warming might also have influenced trophodynamics, with warmer winter temperatures resulting in an increase in predation of a dominant copepod (*Acartia tonsa*) by a gelatinous zooplankton predator (*Mnemiopsis leidyi*) (Costello et al. 2006). Given the declining phytoplankton biomass trend observed in the bay, food limitation of the dominant zooplankton (Durbin et al. 1983) might be increasing.

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Chapter 5:

3. Delaware Estuary

Jonathan H. Sharp

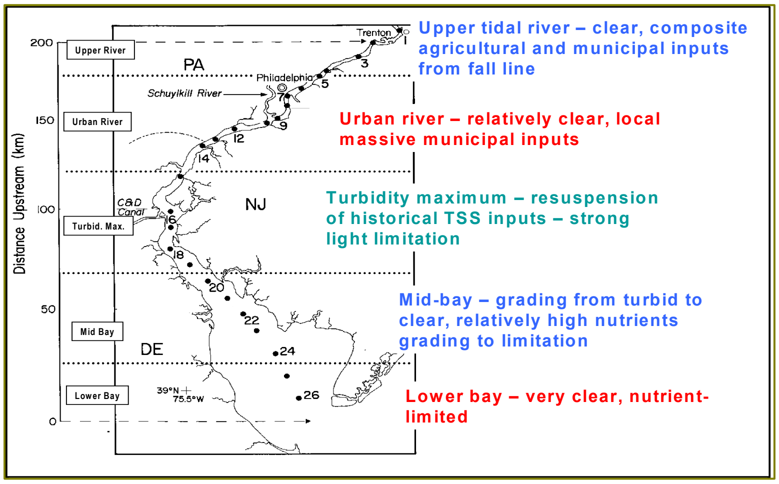

Figure 5.3.1. Two maps of the Delaware Estuary. The top map shows the broader location, and the bottom map details the linear distance up the estuary in km, routinely sampled stations and separation into five regions.

General Description

The Delaware Estuary includes a tidal freshwater river of about 100-km length and the Delaware Bay salinity gradient of about 120-km length, terminating in mid-Atlantic coastal waters (Figure 5.3.1). The tidal fresh portion is less than 5 percent of the overall 2,070 km^2 estuarine surface area (Sharp et al. 1982). The majority of the 33,254-km^2 watershed is in mountains of the upper drainage basin. The Delaware River contributes 58 percent of the water input to the estuary, and the Schuylkill River, entering within the tidal fresh region, contributes about 14 percent; no other single input throughout the length of the estuary exceeds 1.5 percent of the total estuarine flow (Smullen et al. 1984). The head of the tide, near Trenton, New Jersey, is the fall line between the upland Piedmont and the coastal plain. The Delaware Estuary has semi-diurnal tides with relatively high tidal ranges (up to 1.5 m near the mouth of the bay and more than 2 m in the tidal river) and very high tidal current velocities. From the total volume of water in the estuary and the river discharge, the approximate flushing time for the estuary is estimated at about 80 days (Sharp et al. 1986; Cifuentes et al. 1990). A spring freshette usually gives rise to moderate stratification that persists for 2–8 weeks in late February to April. Through the remainder of the year, stratification rarely persists over the tidal cycle with well-mixed waters near slack tides (Pennock 1985; Sharp et al. 1986; Cifuentes et al. 1989).

Figure 5.3.1 shows the estuary with distances down the axis (0 at the mouth of the Delaware Bay to 220 km at the head of the tide) and five estuarine regions. The upper tidal river has relatively significant nutrient inputs from the upper drainage basin (cumulative agriculture inputs and relatively large point source municipal input from Trenton). The urban river region has large nutrient inputs from the greater Philadelphia municipal region. The beginning of the salinity gradient is characterized by a persistent turbidity maximum. As the bay widens, the waters become clearer, and nutrient concentrations are lower from dilution in the mid-bay region. The lower bay has comparatively clear waters and comparatively low nutrients. More detail on the characteristics of the regions are in Yoshiyama and Sharp (2006).

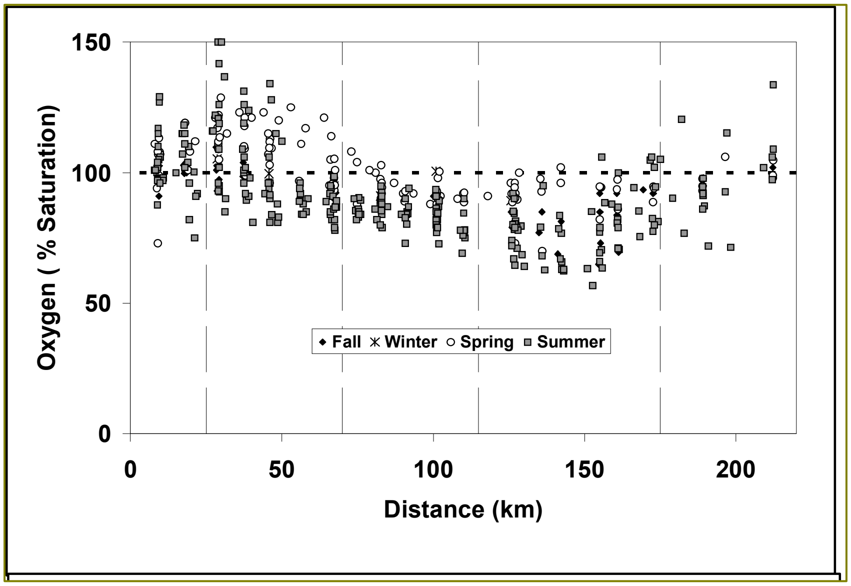

Figure 5.3.2. DO saturation along the axis of the Delaware Estuary separated by season (data from 1990 to 2003). The vertical lines indicate the five regions of the estuary.

Dissolved Oxygen

During the spring stratification, the waters of the estuary are very cold, and there is no measurable depletion of DO in the isolated, bottom waters. During the rest of the year the waters are well mixed, and there is no gradient between surface and bottom waters (Sharp et al. 1982; Culberson 1988). In the past, the urban river region into the turbidity maximum had hypoxia in surface to bottom waters during the warmer months of the year (Albert 1988). Today, some DO depletion occurs in the urban region, but usually the DO concentration in the summer does not drop much below 70 percent saturation even on warm, still days (Figure 5.3.2). The mid-bay region often shows appreciable DO supersaturation during the daytime in the summer.

Turbidity and Light

Compared to many other U.S. estuaries, the Delaware is relatively turbid. Figure 5.3.3 shows TSS concentrations in surface waters of the estuary. A pronounced maximum of TSS can be seen most of the time in the region that we call the turbidity maximum (average of 46 mg L^{-1}). The high concentration is due primarily to resuspension of bottom sediments by tidal currents (Biggs et al. 1983). The TSS concentrations are usually lower in the other four regions (averages of < 10 mg L^{-1} for the lower bay

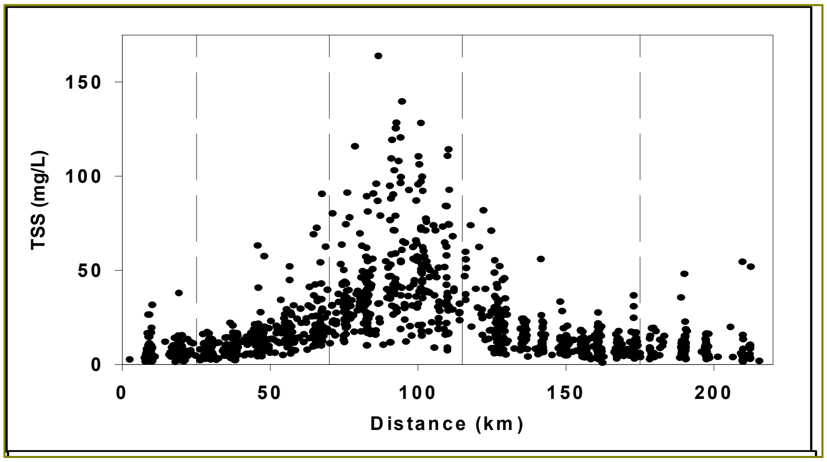

Figure 5.3.3. TSS along the length of the Delaware Estuary (data from 1978 to 2003). Vertical lines indicate the five regions of the estuary.

and upper river); on rare occasions, elevated levels of TSS occur in the upper river because of heavy discharge and in the mid and lower bay from strong wind events. Along the salinity gradient, the correlation to TSS accounts for 88 percent of the light attenuation. In the turbidity maximum region, the 1 percent light level is often as shallow as 0.5 m, contrasted with the lower bay where it is often 6 to 8 m (Yoshiyama and Sharp 2006). Clay minerals dominate the TSS so that phytoplankton is considered insignificant for light attenuation; in the turbidity maximum, the TSS often contains < 1 percent carbon.

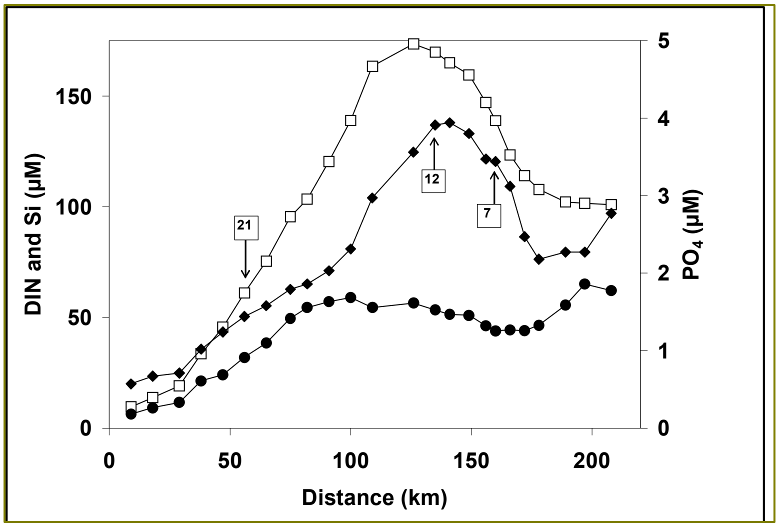

Figure 5.3.4. Nutrient concentrations along the length of the Delaware Estuary. Monthly weighted annual averages from sampling in 1986–1988. Total DIN in open squares; phosphate (PO_4^{-3}) in filled diamonds; and silicate ($Si(OH)_4$) in filled circles. Location of stations 7, 12 and 21 are indicated.

Nutrients

Nutrient loading and nutrient concentrations are very high in the Delaware Estuary (Sharp 1988, 1994). Figure 5.3.4 shows annual average concentrations of total dissolved inorganic N (DIN = NH_4^+ + NO_3^- + NO_2^-), PO_4^{-3}, and $Si(OH)_4$ along the length axis of the estuary. Large, seasonal variations exist in distributions of all nutrients (Pennock and Sharp 1986), but there is a consistent general pattern of the annual averages. The NH_4^+ concentration is higher in winter (up to 50 µM-N in urban river) and lower in summer, with NO_3^- showing a reverse seasonal variation. In the upper river, $Si(OH)_4$ is often 70 to 100 µM-Si in winter, but it drops dramatically in May and remains low throughout the tidal river regions in summer. PO_4^{-3} shows some predictable seasonal pattern but most dramatically shows an inverse relationship in the tidal river to discharge at the fall line; under very low flow conditions, it can reach 6 to 7 µM-P and be as low as 2 µM-P at high flow (Lebo and Sharp 1992, 1993).

All nutrients show considerable dilution going down the salinity gradient from the end of the freshwater stretch at about 120 km. Both N and P are high at the head of the tide, but both show very large increases in the urban river from about 150 km–120 km. That is from very large municipal sewage input. No appreciable input of nutrients is in the saline portion of the estuary. The extensive marsh periphery of the bay helps buffer potential diffuse inputs from agricultural activity in the lower estuary. Estimates of sources of nutrients to support primary production indicate that there is significant advection of the freshwater sources, especially in the spring period. Seasonal and spatial patterns of microbial use of N (Pennock 1987) and P (Lebo 1990) have been well described. At the end of the spring bloom, the lower estuary shows exhaustion of NH_4^+ , PO_4^{-3}, and $Si(OH)_4$ (Sharp et al. 1984; Cifuentes et al. 1989; Pennock and Sharp 1994); all three increase in concentration in the summer from regeneration in the fluid mud layer near the bottom.

Chlorophyll and Primary Production

Compared to many coastal and estuarine waters, the Delaware Estuary has high chlorophyll *a* content in the waters of both the tidal river and the bay (Pennock 1985; Pennock and Sharp 1994). However, because no warm-weather stratification occurs, DO is not depleted because of excess algal biomass. Seasonal average chlorophyll *a* for the length of the estuary (Figure 5.3.5) shows that for most of the year, average concentrations are in the 5 to 10 µg L^{-1} level within the main portion of the estuary. Occasional peaks above 20 µg L^{-1} occur, especially in shallows near the shore. In the spring, chlorophyll *a* reaches an average above 20 µg L^{-1} throughout the mid estuary with peaks as high 100 µg L^{-1}. That high spring biomass is not consumed by grazing and sinks to deeper water, which is cold in the spring. The urban river has higher average chlorophyll *a* than the bay in the summer; that is the only time of year when chlorophyll *a* is high in the river.

The pattern of primary production is significantly different from that of chlorophyll *a*. In spite of lower chlorophyll *a*, the production in the mid and lower estuary is higher in the summer than in the spring bloom (Pennock and Sharp 1986, 1994;Yoshiyama and Sharp 2006). When production is normalized to chlorophyll *a* biomass, it can be seen that this production:biomass (P:B) ratio is higher in all regions in the summer than in other months (Figure 5.3.6). The lower biomass in summer is probably due to grazing (Pennock 1985). In addition, the average P:B is higher in the mid-bay region than the other four regions for each season. The P:B ratio provides some physiological information. Examining Figure 5.3.6, nutrient limitation is probably the cause of lower values near the mouth of the bay; severe light limitation can explain the lower P:B in the turbidity maximum region. The low P:B in the urban river, where nutrients are high and light is available, is a possible indication of contaminant inhibition (Sanders and Riedel 1992; Yoshiyama and Sharp 2006). At different times and places, the estuary shows limitation of primary production by light, N, P and possibly carbon (Pennock and Sharp 1994; Fogel et al. 1992) as well as the probable contaminant inhibition. In the summer, phytoplankton production is tightly linked to bacterial production, but it is not linked in the spring (Coffin and Sharp 1987; Parker 2005).

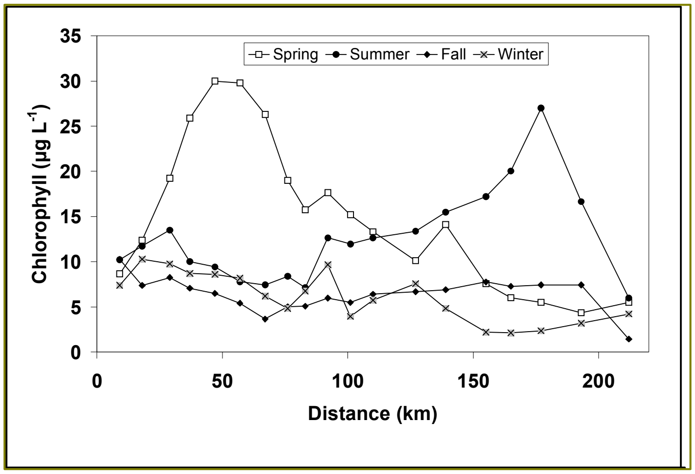

Figure 5.3.5. Chlorophyll a concentration averaged for seasons along the length of the Delaware Estuary. Data from 1980 to 2003.

Phytoplankton Blooms and Species Composition

Little research or monitoring has been done of phytoplankton species in the Delaware Estuary. The only major bloom seen consistently in the Delaware Estuary is the spring bloom, which occurs primarily in the mid estuary. The bloom appears to always be dominated by the ubiquitous coastal diatom, *Skeletonema costatum* (Pennock 1985). When the bloom occurs, minimal zooplankton grazing occurs, and the majority of the biomass is not consumed. It sinks to the bottom, but because the water is cold, little microbial consumption occurs. By sometime in early April to early May, the stratification has completely broken down, and the waters remain well mixed during the warmer months of the year. It appears that in the fall, winter and the spring, diatoms are the most prominent members of the phytoplankton in both the tidal river and the saline bay. This is somewhat surprising because $Si(OH)_4$ is often comparatively low (Si:N ratios are usually < 1). Some dinoflagellates have been noted in the mid and lower estuary and some cyanobacteria in the tidal river, but prominent blooms of those groups have not been recorded. In the summer months, small flagellates and cyanobacteria appear to be the most noted groups, but not usually in bloom proportions.

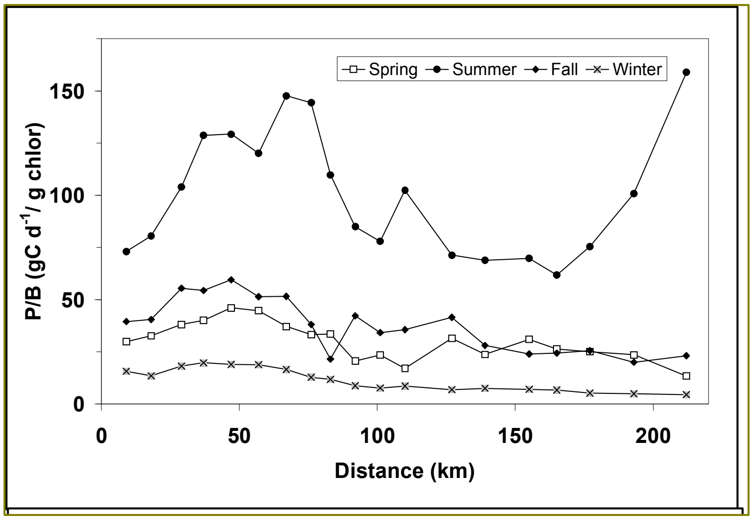

Figure 5.3.6. Primary production from ^{14}C simulated in situ incubations normalized to chlorophyll a biomass averaged for seasons along the length of the Delaware Estuary. Data from 1980 to 2003.

Benthic Primary Producers

Because of the highly turbid nature of the Delaware Estuary, no SAV is in the saline portion of the estuary. Also no major macro-algal population exists in the Delaware Estuary. The majority of the area of the estuary has high TSS from resuspension and marsh input along the edges, so the Delaware Bay can be viewed as having only pelagic primary production. The freshwater upper river has considerable SAV populations and probably extensive benthic algal mats. Those do not continue in the urban river, largely because of extensive bulkheading of the river.

Long-Term Changes

Because of large industrial and municipal inputs, the urban region of the Delaware Estuary had serious impairment for well over a century. By the 1940s, the urban river (with extension down into the bay) showed severe hypoxia in the warmer months of the year (Albert 1988; Sutton et al. 1996). Some controls of sewage treatment plants and industrial inputs yielded slight improvements in the 1950s and 60s, but it was not until major sewage treatment plant upgrades in 1970–1985 that the severe DO sag was alleviated. The improvement is well documented with an increase near station 12 (Figure 5.3.1) equivalent to about 10 μM -O per year from 1970 to 1990 (Sharp 1994). The DO sag extended from the urban river down through the turbidity maximum region; it is illustrated with transect data from summer of 1967 compared to one in 1997 (Figure 5.3.7). A time trend from 1967 to 2004 for station 21, in the mid bay, shows no change; it is close to saturation during the entire period. The strong DO demand in the past was primarily a result of BOD from organic carbon and reduced N in sewage effluents.

Figure 5.3.8 shows N speciation in summer of 1967 contrasted to 1997. No evidence exists of major algal blooms or hypoxia from excess algal production in the past. The hypoxia and high NH_4^+ content of the past extended through the urban river and down into the bay. Also, a much higher TP input into the estuary occurred in

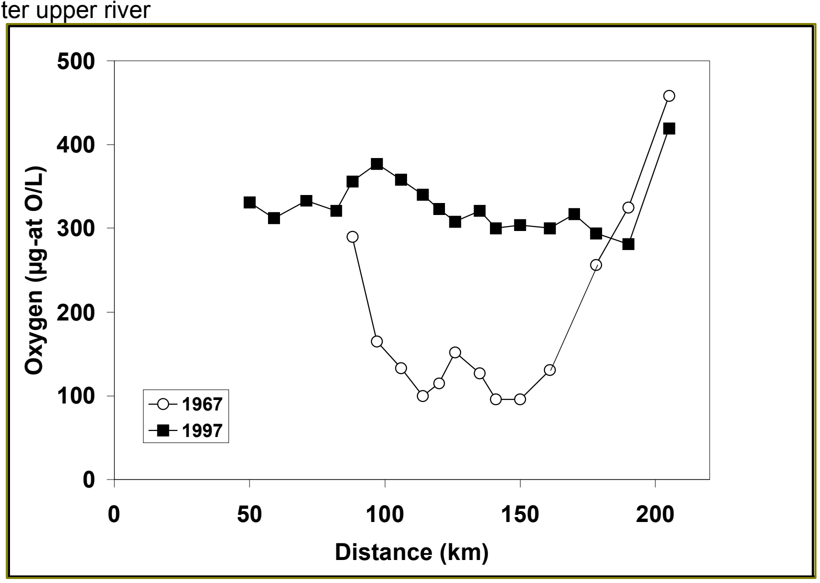

Figure 5.3.7. Summer DO concentrations in µg at O L^{-1} (= µM) in transect along the Delaware Estuary in 1967 and 1997. Data are average values for sampling stations in Delaware River Basin Commission boat run monitoring for summer (July–August).

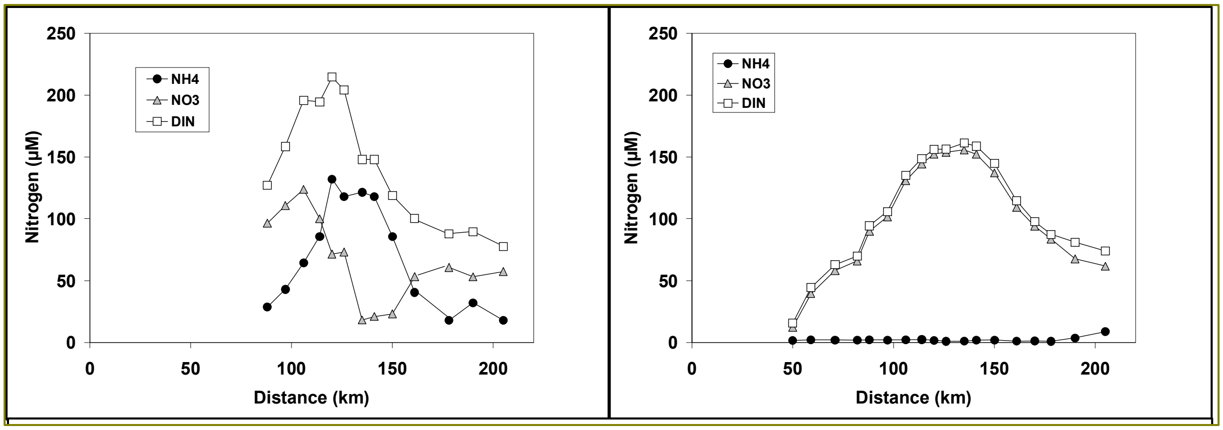

Figure 5.3.8. DIN as a total pool and NO_3^- and NH_4^+ as components for transects in the Delaware Estuary for summer of 1967 (left) and 1997 (right). Data are average values for sampling stations in Delaware River Basin Commission boat run monitoring for summer (July–August).

the past (Sharp 1988). With improved sewage treatment and the PO_4^{-3} detergent ban, the TP in the urban region of the estuary dropped abruptly by about fivefold in the early 1970s. Figure 5.3.9 contrasts the N:P ratio along the estuarine transect between 1967 and 1997. Although no algal or production data exist for the earlier period, the large change in N:P would indicate a shift from P sufficiency to comparatively P deficiency, which should influence algal speciation and primary production. The shift probably is manifest throughout the estuary.

Summary of Ecosystem Impacts

Despite very high nutrient loading and high nutrient concentrations, the Delaware Estuary does not show the classical eutrophication symptoms of hypoxia or HABs. Although surface-bottom hypoxia existed in the urban river in the past, apparently it did not extend to the mid - or lower-bay hypoxia. The lack of bottom water hypoxia is largely because the estuary is rapidly flushed, and the waters are not stratified during the warmer months. The hypoxia that occurred in the past was from BOD from allochthonous inputs, not from autochthonous excess algal production. The lack of HABs in the Delaware Estuary when such blooms have been found in nearby Chesapeake Bay and in the Delaware and Maryland Coastal Bays is not well explained, although high turbidity, high flushing and low light might play important roles. Essentially, no SAV and almost no macro-algal growth exists in the saline portions of the Delaware Estuary. It is thought that any SAV coverage that might have existed in the Delaware Estuary was probably gone more than a century ago, maybe longer.

It appears that the major controlling factor of primary production is light, and to a lesser extent, N and P in the lower estuary. In the summer, grazer control probably is partly responsible for keeping the biomass relatively small. Large changes in nutrients in the past several decades appear to have a major effect on the estuary

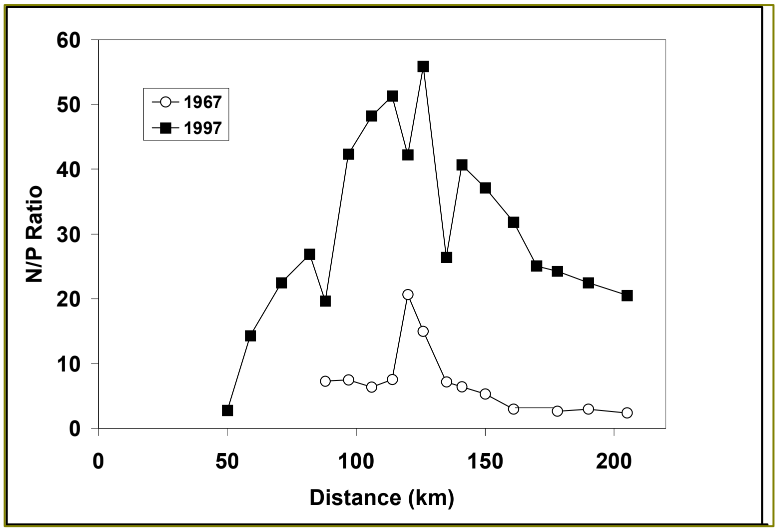

Figure 5.3.9. Ratio of DIN:TP for summer transects in the Delaware Estuary for 1967 and 1997.

water quality (decreased DO demand) but have had little or no demonstrable effect on phytoplankton. No specific nutrient management plans exist for the Delaware Estuary. Nutrient management is probably needed in some of the subtributaries for local impact, and nutrient management of subtributaries flowing into the Chesapeake Bay have been suggested. It is unclear how nutrient management of subtributaries will improve the Delaware Bay's water quality. It is also uncertain how much influence the Delaware Bay outflow to the adjacent coastal waters has on conditions in the coastal waters.

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Chapter 5:

4. Chesapeake Bay

Walter R. Boynton

General Description

The Chesapeake Bay is 300 km long, with a relatively deep (20 to 30 m) and narrow (1 to 4 km) central channel with a sill at its seaward end (Figure 5.4.1). Broad, shallow areas flank the central channel. Depths exceeding 10 m constitute only 24 percent of the bay's surface area; the estuary's mean depth is 7.3 m (Boicourt et al. 1999). An average of 2,300 $m^3 s^{-1}$ of freshwater flows from the watershed into its 51-km^3 water volume. The Susquehanna River provides more than half of the flow (Schubel and Pritchard 1986). Freshwater flow sets up stratification, and winter-spring Susquehanna River flow controls stratification from spring into the fall (Figure 5.4.2). Episodic wind mixing events contribute to periodic de-stratification, but stratification is quickly reestablished within a few days (Goodrich et al. 1987). River flow drives the estuarine circulation, characterized by a lower-layer counter-flow that acts to retain particulate and dissolved materials and creates relatively long residence times (90 to 180 days) for freshwater and associated materials. The combination of long water residence time, narrow central channel isolated by sills, stratified water column and wide flanking shallows make the system highly susceptible to nutrient enrichment effects such as hypoxia (Boicourt 1992).

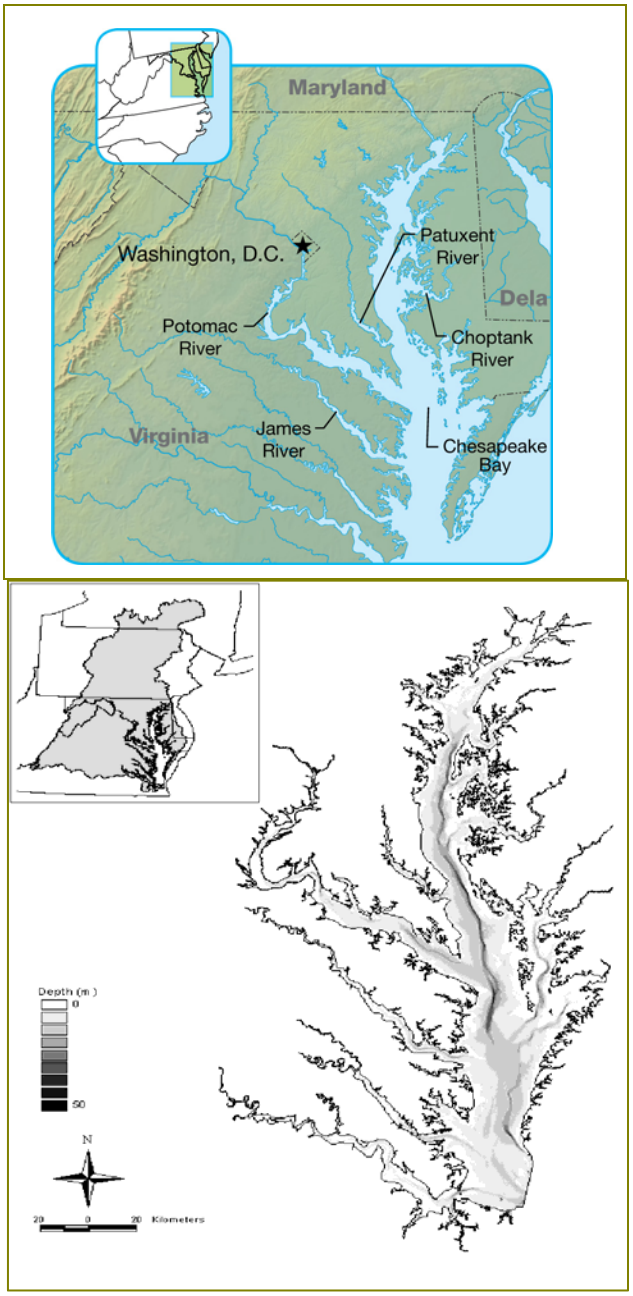

Figure 5.4.1. Two maps of the Chesapeake Bay. The upper map shows the broader location; the lower map shows water depth contours. The inset of the lower map shows the extent of the watershed.

Dissolved Oxygen

The spatial distribution and seasonal development of hypoxia in the bay illustrate that DO depletion arises from interactions between biological and physical processes (Kemp et al. 1992). Hypoxia develops in the bot-

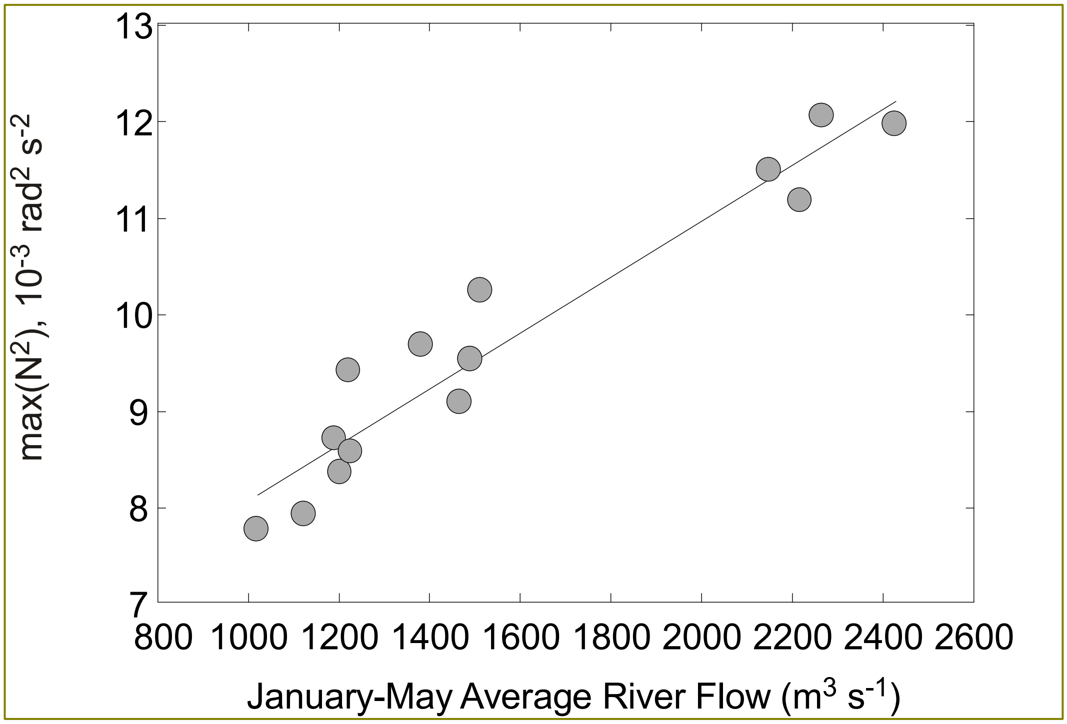

Figure 5.4.2. Relationship between January–May average Susquehanna River flow and April–September average water column stratification in the middle bay (Brunt-Väisälä frequency).

tom mixed layer, appearing first in late spring at the northern limit of the stratified flow and expanding southward during summer. The timing of DO depletion in spring is predicted by freshwater inflow, which regulates water column stratification and associated rates of DO replenishment, and by spring water temperature, which affects respiration rates. High freshwater inflow during spring has also been correlated with increased organic deposition to sediments, which would promote increased DO demand.

Springtime rates of DO decline have been essentially the same from 1938 to the present, despite increases in nutrient loading during the period (Newcombe and Horne 1938; Hagy et al. 2004). That suggests that the initial spring decline in DO is strongly controlled by physical processes, while the late spring DO decline and the extent of summer hypoxia are more closely related to eutrophication.

Turbidity and Light

Light attenuation in the upper Chesapeake Bay is mainly related to suspended sediments, while in the middle and lower bay, it is controlled by interactions between plank-

Table 5.4.1. Estimated changes in euphotic depth, lighted bottom area and benthic gross primary production (GPP) for regions of the Chesapeake Bay during the 1930s and 1990s.

Era	Region	$Z_{1\%}$ (m)	Bottom Area <$Z_{1\%}$ (10^6 m^2)	Benthic GPP (10^9 gC m^{-2} y^{-1})
1930's	Upper Bay	1.7	101	18
	Middle Bay	7.6	828	137
	Lower Bay	8.8	1508	210
	Total Bay	7.6	2508	396
1990's	Upper Bay	1.7	101	17
	Middle Bay	4.5	461	77
	Lower Bay	5.2	649	99
	Total Bay	4.6	1365	217

ton and suspended sediments. Sharp declines in water clarity during spring and summer can be related to algal blooms supported by nutrients from the watershed (Gallegos 2001; Gallegos and Jordan 2002). Historical Secchi depth measurements reveal dramatic increases in turbidity between 1930 and 1990 (Table 5.4.1). The depth to which 1 percent of surface light penetrated in the middle and lower bay has decreased from 7.6 and 8.8 m in 1930 to 4.5 and 5.2 m in 1990. The upper bay was turbid during both periods (D'Elia et al. 2003).

Nutrients

Highest nutrient concentrations tend to occur in river-dominated estuaries, like Chesapeake Bay, having major riverine sources of nutrients. Concentrations are highest in tidal freshwater or oligohaline locations and rapidly decrease seaward. The Chesapeake Bay also has large differences in concentration between dry and wet years (a factor of approximately 10) in the polyhaline and mesohaline regions; concentrations of $NO_3^- + NO_2^-$ are generally high (100 µM-N) in the oligohaline zone because of proximity to the riverine N-source and limited uptake by phytoplankton in this turbid portion of the system. Concentrations of $NO_3^- + NO_2^-$ are highest during winter and early spring and minimum concentrations mainly occur during summer. Seasonal maximum PO_4^{-3} concentrations generally occur during summer or fall, opposite the pattern observed for $NO_3^- + NO_2^-$. Seasonal minimum concentrations of PO_4^{-3} are most frequently observed during spring, the time of the year that P has been found to limit spring diatom blooms (Chesapeake Bay Water Quality Monitoring Program 2001).

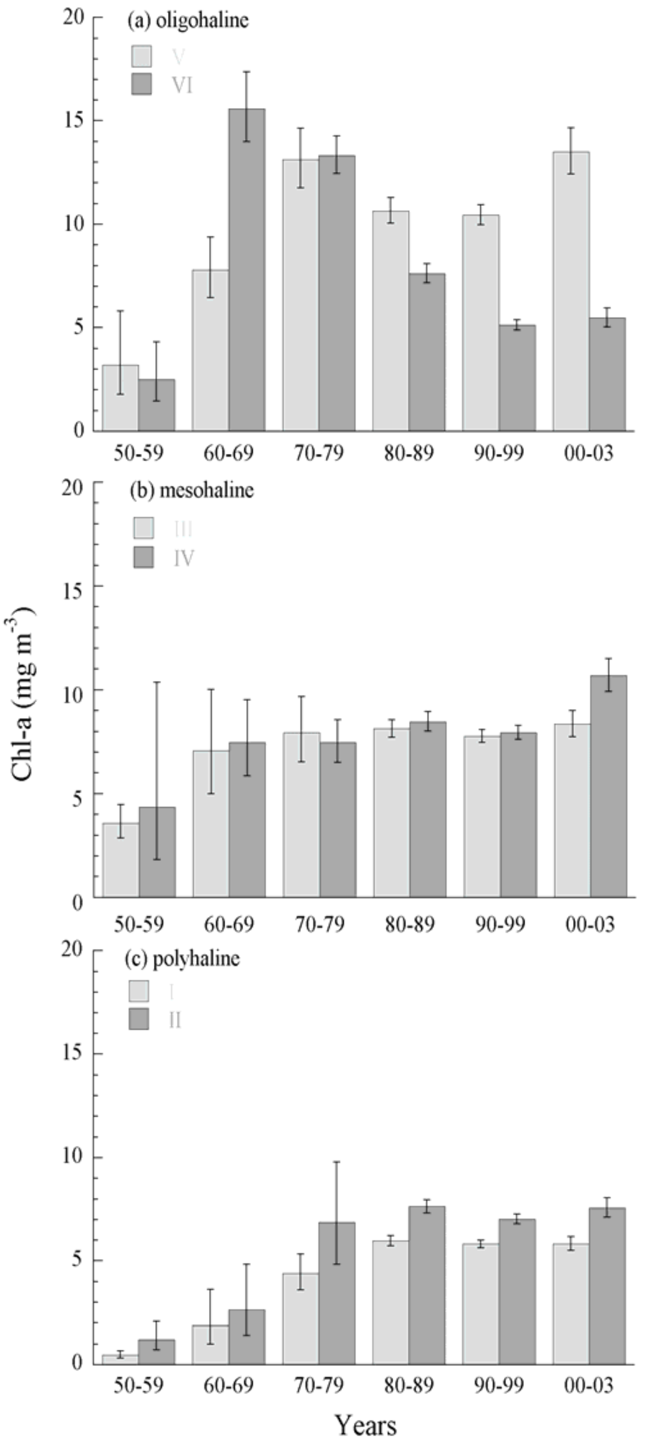

Figure 5.4.3. Multi-decadal patterns in surface water chlorophyll a in oligohaline, mesohaline and polyhaline regions of the Chesapeake Bay. From Kemp et al. 2005.

Chlorophyll and Primary Production

Monitoring of phytoplankton chlorophyll *a* has shown that concentrations have increased since 1950, with largest changes in the polyhaline region (Harding 1994; Harding and Perry 1997). Chlorophyll *a* increased 1.5- to 2-fold in oligohaline and mesohaline regions peaking in the 1960s, and 5- to 13-fold in the polyhaline bay from the 1950s to the 1980s (Figure 5.4.3). Chlorophyll *a* decreased in the upper oligohaline region since the 1970s in response to relatively high turbidity associated with elevated river flow. That pattern—phytoplankton chlorophyll *a* increasing between the 1950s and 1980s and unchanged during the last decade—corresponds to reported trends in N loading during the period (Hagy et al. 2004).

Phytoplankton Blooms and Species Composition

Retrospective analysis indicates increases of phytoplankton chlorophyll *a* during the past half century have been accompanied by shifts of floral composition; the diatom community has experienced increased overall abundance, declining diversity and domination by small pelagic species (Marshall 1994). Abundance of dinoflagellates, cyanobacteria and small flagellates appear to have also increased. In some instances, eutrophication-induced shifts in phytoplankton community involve enhanced growth of algal species that cause direct harmful effects, including production of toxins, noxious discoloration and floating mucilage. While factors causing HABs are complex, many have been associated with nutrient enrichment (e.g., Glibert et al. 2005). In the Chesapeake Bay and its tributaries, several types of HABs appear to be related directly to nutrient inputs. In the Potomac River estuary, frequency of summer blooms of the cyanobacterium, *Microcystis aeruginosa*, declined sharply in the early 1970s when P-removal from sewage was initiated (Sellner et al. 1988). Blooms of both the dinoflagellate, *Prorocentrum minimum*, and the mixotrophic dinoflagellate, *Pfiesteria piscicida*, appear to be stimulated by addition of DON including urea (Glibert et al. 2001). The dinoflagellate *Karlodinium veneficum*, responsible for recurring fish kills, also appears to be stimulated by nutrient inputs.

Benthic Primary Producers

In addition to increases in phytoplankton biomass, other changes in primary producers also occurred in the Chesapeake Bay. The area of lighted bottom has decreased substantially between 1930 and the present time and estimates indicate a reduction in benthic diatom production by about a factor of two (Kemp et al. 1999; Table 5.4.1). Additionally, SAV cover in most portions of the bay system also decreased and the likely cause is nutrient-induced decreases in light available for SAV production (Kemp et al. 2005; Figure 5.4.4). During the last several decades, phytoplankton has become the dominant

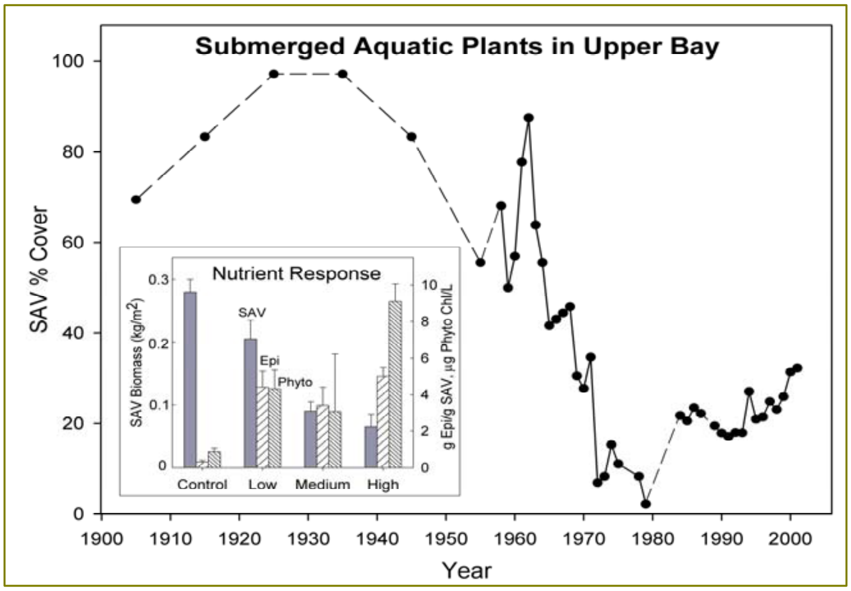

Figure 5.4.4. Temporal pattern of seagrass (SAV) cover in the upper Chesapeake Bay from 1905 to 2004. The inset shows SAV responses to nutrient additions in pond mesocosms. From Kemp et al. 2005.

source of organic matter for all bay food webs.

Long-Term Changes

The Chesapeake watershed covers 164,200 km^2 of diverse ecological and physiographic features and human settlement. The ratios of watershed area to estuarine water area and volume (14.3 and 2.2 m^{-1}, respectively) are large compared to other estuaries. Coupled with the long (18,800 km) shoreline, the estuary is closely connected with its watershed. Population has grown exponentially since colonial times and now numbers 16 million, but population density remains modest (1 person ha^{-1}). Land use change was dominated by land clearing for agriculture until the mid-1800s when about half of the basin was deforested. During the past 200 years, urban land has expanded, agricultural land declined, and forested land gradually increased (Figure 5.4.5). From 1890 to 2005 flow from the Susquehanna River has been highly variable including drought periods (1930s), prolonged droughts (1960s), huge flood events (200-year storm in 1972), prolonged wet periods (1970s), and decades of extreme variability (1990 to present). Use of commercial fertilizers grew dramatically after the 1950s as did large animal agriculture. That, coupled with other human activities, increased nutrient loading by 2.5-fold from 1945 to 1990 (Figure 5.4.6; Kemp et al. 2005).

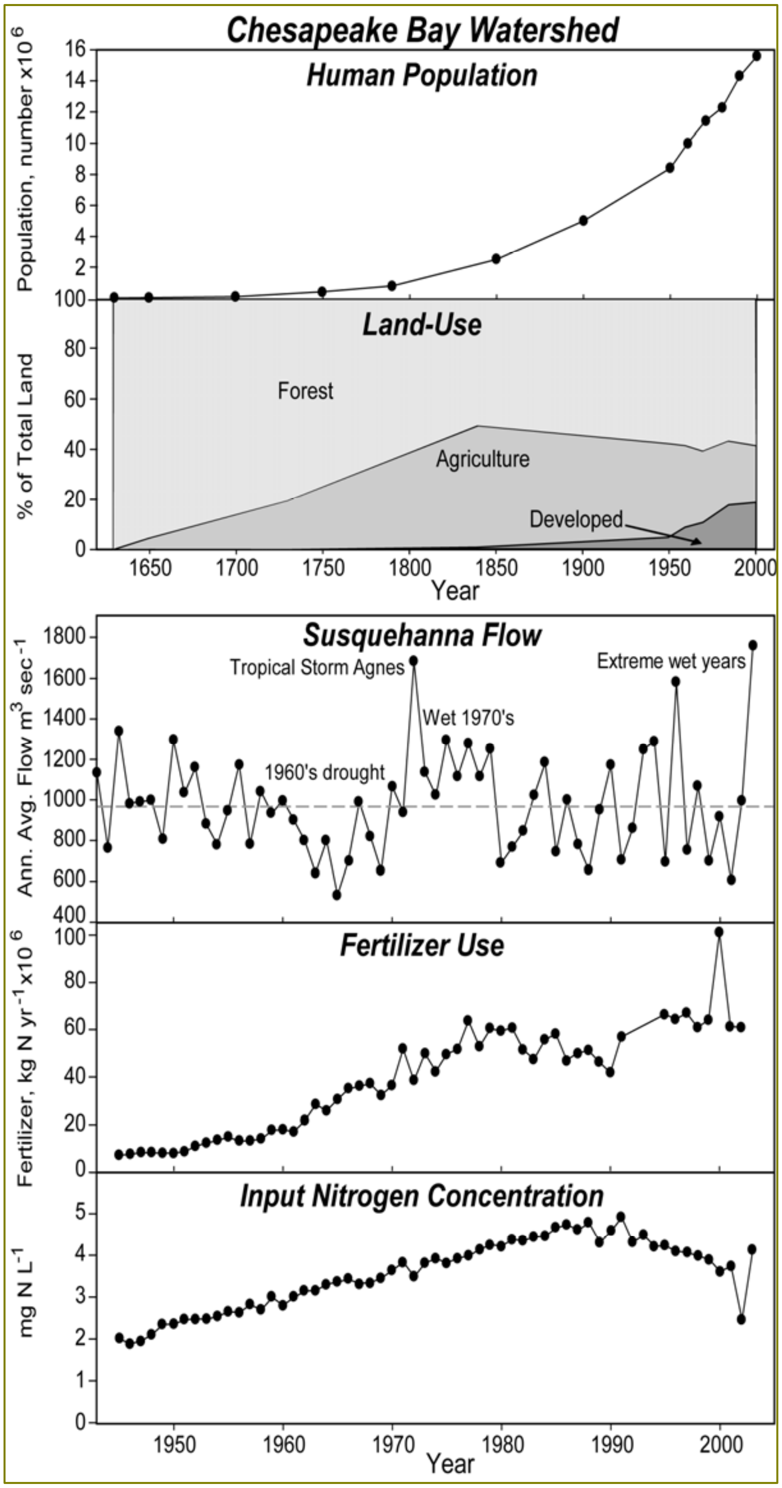

Figure 5.4.5. Temporal changes in land uses, river flow, fertilizer use and river N concentration. From Kemp et al. 2005.

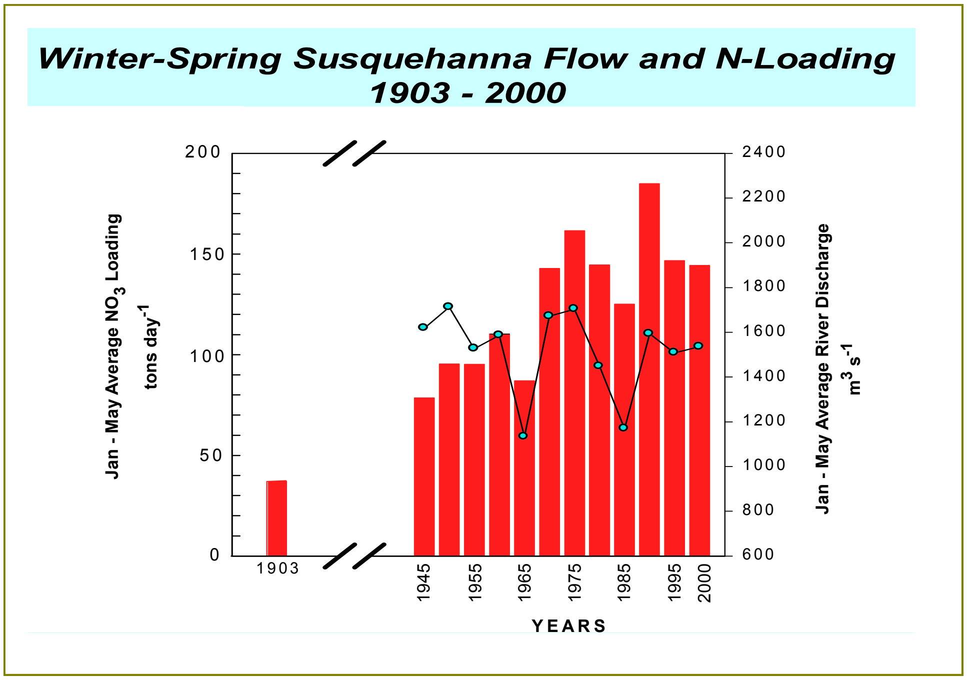

Figure 5.4.6. Winter-spring Susquehanna River flow (red bars) and N load (black line), 1903–2000.

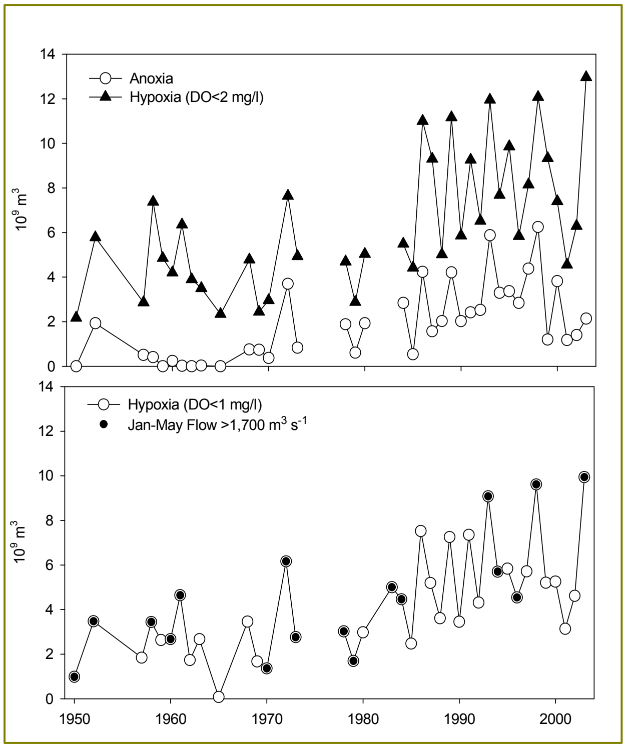

Figure 5.4.7. Volume of water in the mainstem Chesapeake Bay affected by hypoxia (DO < 2 mg L^{-1}) and anoxia (DO ≤ 0.2 mg L^{-1}) during summer, 1950–2003 (upper panel). Volume of water with severe hypoxia (DO < 1 mg L^{1}). Filled circles indicate wet winter-spring years (lower panel).

It now appears that increasing frequency and magnitude of seasonal DO depletion from bottom waters is linked with anthropogenic nutrient enrichment. Direct measurements indicate that hypoxia (< 2.0 mg DO L^{-1}) occurred only occasionally in deep waters in the 1930s. Recent analysis of data collected between 1950 and 2003 indicates significant increases in severity and spatial extent of hypoxia and anoxia since the 1950s (Hagy et al. 2004; Figure 5.4.7). It is clear that human activities from 1950 to now have significantly changed N loading to the bay. Because hypoxia and N loading both increased over time, the two are correlated. However, it appears that hypoxia tends to be more severe in recent years even at equivalent levels of N loading. That implies that the bay has become less able to assimilate N inputs without developing hypoxia, a change that might have arisen from degradation of key ecological processes sensitive to eutrophication effects. Potential mechanisms include loss of benthic photosynthesis and nutrient uptake with declining water clarity and increased efficiency of N and P recycling with decreased denitrification and P precipitation, both in response to decreasing bottom water O_2 (Kemp et al. 2005).

Summary of Ecosystem Impacts

A recent and comprehensive analysis of ecosystem impacts in the Chesapeake Bay has been developed by Kemp et al (2005). This synthesis, of which several findings were highlighted in the previous sections, showed that

- Initial signs of organic enrichment were evident 200 years ago.
- Increased algal production and decreased water clarity appeared about 100 years ago.
- Severe hypoxia and loss of SAV began about 4–5 decades ago.
- Eutrophication-caused changes in fish abundance and production have yet to appear.
- Shallow-water habitats have been radically altered with widespread loss of SAV and large reductions in benthic micro-algal production.
- Marked increases in the relative importance of pelagic versus demersal food chains and declines in trophic transfer efficiency (fisheries harvest per primary production) have emerged during the eutrophication period (1970 to 2005).
- Important benthic habitat has been lost because of hypoxia, and oyster abundance has dramatically declined.
- Abundance and production of bacterial and gelatinous plankton are high compared to other coastal ecosystems.

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Chapter 5:

5. Neuse River Estuary

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Figure 5.5.1. Two maps of the Neuse River Estuary. The top map shows its broader location. The bottom map details the estuary segments, (upper tidal river) stations 0–50; (middle estuary) stations 50–120; and (lower estuary) stations 120–180.

General Description

The Neuse River Estuary is a shallow, microtidal (< 0.3 m) ecosystem, the largest subestuary of the Albemarle-Pamlico Sound system, the second-largest estuarine ecosystem in the United States. The Neuse River Estuary typifies many other poorly flushed, relatively long water residence time estuarine systems worldwide where internal feedback mechanisms have changed because of external perturbations (Paerl et al. 1998, 2001, 2006a, 2006b). The wide (6.5 km) and shallow (2.7 m) basin receives inputs from a large watershed that, while urbanized upstream, has intensive row crop agriculture and hog operations toward the coast (Stow et al. 2001). A near 90 degree bend divides the Neuse into a distinct morphometry that interacts with river discharge and wind events to dominate circulation (Figure 5.5.1; Buzzelli et al. 2002; Reynolds-Fleming and Luettich 2004). Those physical properties create flushing times of weeks to months that greatly facilitate internal biogeochemical processing of materials (Christian et al. 1991; Nixon et al. 1996; Reynolds-Fleming and Luettich 2004).

Generally, the middle segment of the Neuse River Estuary (stations 50–120, see Figure 5.5.1) appears most sensitive to changes in nutrient inputs and transformations (Paerl et al. 1995, 1998; Luettich et al. 2000). The segment is just seaward of the typical location of the isohaline (salinity of ~2) during summer conditions. A majority of the water column organic matter in the segment is autochthonous and reaches the sediment (Matson and Brinson 1990; Rizzo and Christian 1996; Clesceri 2003). Vertical density stratification occurs in the main channel throughout much of the spring through fall because of the low tidal amplitude (< 0.3 m), and promotes the regular formation of bottom water hypoxia (< 2.0 mg L^{-1}) in the middle and lower estuary segments during typical summer conditions of seasonally lower river flows (Stanley and Nixon 1992; Boicourt 1992; Borsuk et al. 2001; Buzzelli et al. 2002). Changes in wind velocity with the passage of weather fronts can redistribute hypoxic water to lateral shoals affecting fauna in those regions (Taylor and Eggleston 2000; Reynolds-Fleming and Luettich 2004; Eby and Crowder 2002).

Seasonal and storm-driven variation in river flow shifts the position of the salinity gradient in the Neuse River Estuary from upstream of New Bern downstream to Pamlico Sound (Figure 5.5.1). Seasonally, the river flow is typically higher in the winter-spring months and lower during summer and fall. Storm events throughout the year tend to push saltwater extent downstream, at least in the surface layer. In cases of extremely large events, saltwater throughout the water column shifts downstream into the lower estuary and can extend downstream into Pamlico Sound. Such large events have occurred several times in recent years (Peierls et al. 2003; Paerl et al. 2006a).

Dissolved Oxygen

The DO concentration in the Neuse River Estuary is highest during November to March, consistent with seasonal minimum temperatures and lowest in bottom waters during June to September (Figure 5.5.2). Values shown in the figure are averages of all stations in the middle segment of the estuary from 2001 to 2005, according to monthly surveys. Low average DO concentrations during summer months along the main channel occur concurrent with stable salinity stratification in the middle and lower estuary segments of the Neuse. In years with prolonged low flows, stable salinity stratification can extend upstream into the upper tidal river segment (Figure 5.5.2). In terms of DO concentrations in the lower segment (stations 120–180; Figure 5.5.1), the seasonal pattern is similar to the one illustrated in Figure 5.5.2, but minimum DO concentrations are approximately 1 mg L^{-1} higher than in the middle segment. In contrast, the vertical DO gradient in the tidal freshwater upper segment is smaller than the downstream segments, reflecting infrequent saltwater intrusion, and average DO in the bottom layer during June to September is 3.7 mg L^{-1}.

DO concentrations in the Neuse Estuary are affected by storm events through watershed drainage and wind-driven physical forcing. Hurricane events over the past decade have shown that large pulses of freshwater, and associated organic matter and often lower DO concentrations than typical conditions, decrease DO concentration in the upper and middle segments of the Neuse (Peierls et al. 2003, Paerl et al. 2006a). Indeed, the DO concentration following Hurricane Fran in September 1996 was < 2 mg L^{-1} throughout the water column for more than a 60-km stretch of the Neuse, beginning upstream of station 0 and extending downstream to station 120. For extreme drainage events (e.g., Hurricane Floyd), the freshet can extend downstream of the lower

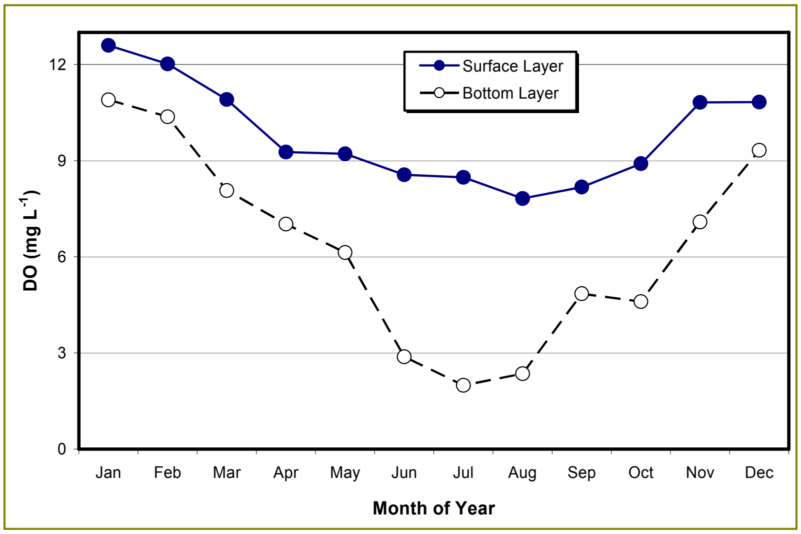

Figure 5.5.2. Average DO concentrations in the Middle Segment of the Neuse Estuary during 2001–2005, by month.

estuary segment into Pamlico Sound. More regularly, storm events and passage of weather fronts enhance physical mixing in the middle and lower segments but can move deeper hypoxic waters from the central channel into shallow regions through wind-driven seiches (e.g., Reynolds-Fleming and Luettich 2004).

Turbidity and Light

Light availability in the Neuse River Estuary is affected by highly colored dissolved organic matter (CDOM), suspended sediment, and planktonic algal biomass. Much of the CDOM is derived from natural watershed sources, including marsh and wetland vegetation, forests and soils. The relatively high phytoplankton biomass accumulation in the Neuse increases light attenuation in bloom patches. That, combined with periodic high sediment load from watershed sources and resuspension in the mesohaline segment, can lead to severely restricted light penetration in the water column (Paerl et al. 1995; Luettich et al. 2000). Seasonally, average TSS is highest (5.9 to 6.2 mg L^{-1}) during winter and spring months and lowest (4.0 mg L^{-1}) during fall. Peak TSS concentrations, exceeding 20 mg L^{-1}, were measured during 2001–2005 during winter, spring and summer months.

In terms of spatial variation in light availability, diffuse light attenuation coefficients (K_d) tend to be highest (median 2.1 m^{-1}) in the upper estuary and decrease downstream as particulate matter and CDOM concentrations also decrease. The photic zone in the Neuse (> 1 percent surface irradiance) is typically 2 to 4 m but can be < 1 m in the tidal river during watershed drainage events or anywhere in the system during algal bloom events.

Nutrients

Nutrients in the Neuse River Estuary are highest in the tidal river upper segment and are typically processed in the middle segment. In terms of N and P limitation of algal growth, the system is primarily N-limited but can experience co-limitation during winter months (e.g., Rudek et al. 1991). NO_3^- + NO_2^- availability is reduced to low levels by the middle segment of the river, where salinity is about 10 (Figure 5.5.3). An important controlling factor on where watershed NO_3^- is used in the estuary is river flow. As river flow increases, the salinity mixing zone in the Neuse River Estuary moves downstream (e.g., Mallin et al. 1993; Lebo et al. 2002; Paerl et al. 2006a).

Typical nutrient fractions by estuary zone, as multiyear median values, are listed in Table 5.5.1. Spatially, NH_4^+ and NO_3^- fractions in river water are transformed into particulate matter in the middle and lower segments. A bulk of the DON essentially passes through the system.

Table 5.5.1. Neuse River Estuary nutrient (µM-L^{-1}) and chlorophyll a median concentrations (µg L^{-1}) by segment for data collected during 2001–2005, with the exception of P data that were from 1999 to 2002 when both PO_4^{-3} and TP data are available.

Parameter	Upper tidal river	Middle segment	Lower estuary
NH_4^+	3.78	0.93	0.86
NO_2^- + NO_3^-	33.57	6.42	< 0.75
DON	25.71	22.14	20.00
Particulate N	7.86	20.00	17.86
PO_4^{-3}	1.88	0.94	0.31
TP	4.06	3.13	2.19
Chlorophyll *a*	4.0	20.0	15.0

Chlorophyll and Primary Production

The accumulation of algal biomass as chlorophyll *a* in the Neuse River Estuary is affected by river flow but is generally highest in the middle segment (Pinckney et al. 1999; Valdes-Weaver et al. 2006). Although large, freshwater algal blooms occurred in the Neuse River through the mid-1980s (see Paerl et al. 2004), chlorophyll *a* con-

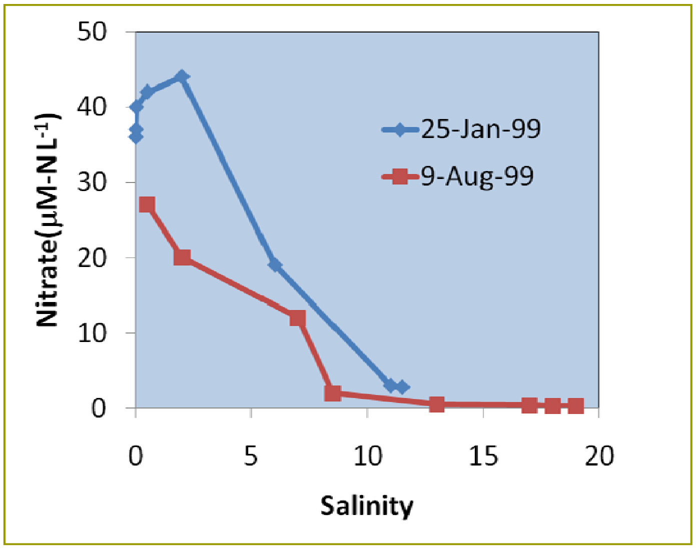

Figure 5.5.3. Relationship between NO_3^- and salinity in the Neuse River Estuary for two dates.

centrations in the tidal river upper segment of the Neuse are typically low (< 4 µg L^{-1}; Table 5.5.1) and inversely related to river flow (Valdes-Weaver et al. 2006). For the middle segment, the chlorophyll *a* concentration typically increases at low salinities to a peak value where the salinity is approximately 3–5, concurrent with a decrease in NO_3^- concentration (Figures 5.5.3, 5.5.4; Lebo et al. 2002). Annually, mean chlorophyll *a* concentration in the middle segment of the Neuse varies by a factor of 2 among years (see Valdes-Weaver et al. 2006). Elevated chlorophyll *a* concentrations also occur in the middle and lower segments (Table 5.5.1, Figure 5.5.4), with peaks often associated with large river pulses of NO_3^- (Paerl et al. 2004). Year-to-year variation in mean chlorophyll *a* in the lower estuary indicates algal biomass can be affected both immediately by river pulses and in the following year, presumably fueled by regenerated nutrients (Christian et al. 1991).

Primary production in the Neuse is highest in the upper to middle segments and decreases in the lower estuary, although data are more limited than for nutrients and general water quality. In terms of spatial pattern, Paerl et al. (1995) report mean annual primary production varied between 281 g C m^{-2} yr^{-1} near station 180 to 703 g C m^{-2} yr^{-1} at station 30, with a production of 583 g C m^{-2} yr^{-1} near station 140. Boyer et al. (1993) report similar levels of primary production for 1985–1988 of 456 g C m^{-2} yr^{-1} averaged over the entire estuary. In fact, the spatial and temporal patterns in primary production reported by Boyer et al. (1993) support the importance of nutrient pulses to the lower estuary; primary production in the lower estuary increased in both 1987 and 1988 from elevated river flow and downstream transport of nutrients in 1987.

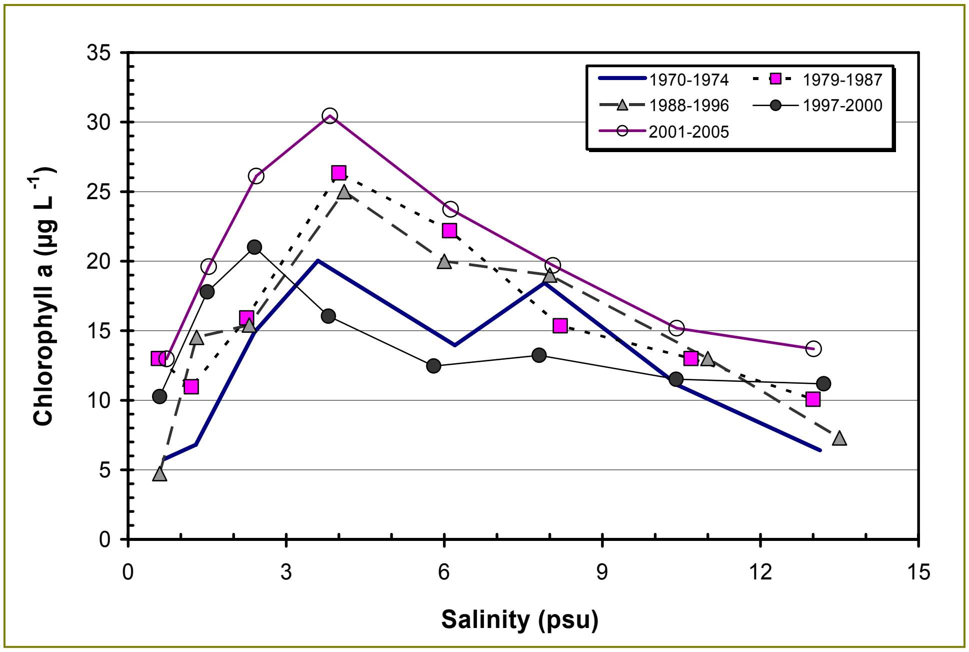

Figure 5.5.4. Multiyear median chlorophyll a concentrations by salinity interval and period. Data were extracted from several monitoring programs, with some variation in sample collection methodology.

On the basis of recent data, it appears that annual primary productivity in the open sound, downstream of the Neuse River Estuary, is on the order of 300 g C m^{-2} yr^{-1}, and mean chlorophyll *a* levels are typically 4 to 20 µg L^{-1} (Table 5.5.1). The seasonal patterns of productivity and chlorophyll *a* are dampened in Pamlico Sound, relative to the Neuse, in large part because the Neuse effectively filters biologically reactive N out of the water column before it enters the Pamlico Sound. In general, the Pamlico Sound shows a strong reliance on regenerated N to support moderate rates of productivity. The exception is when high rainfall events, typically associated with tropical storms and hurricanes, affect the region. Large amounts of N-laden freshwater discharge following these storms can, at times, overwhelm the estuarine filter of the Neuse, leading to the direct input of N (as NH_4^+ or NO_3^-) to Pamlico Sound, supporting elevated levels of primary production and chlorophyll *a* (Paerl et al. 2001; Peierls et al. 2003).

Phytoplankton Blooms and Species Composition

Monitoring in the Neuse River Estuary for chlorophyll *a* and taxonomic composition have shown blooms to occur in all segments of the estuary and at various times of the year (late winter through fall) (e.g., Pinckney et al. 1999; Paerl et al. 2004; Valdes-Weaver et al. 2006). Blooms in the tidal river are generally limited to low-flow periods when saltwater moves upstream of the middle segment (see Paerl et al. 2004; Pinckney et al. 1999). However, large blooms have also appeared during late winter (February to March), in certain years associated with decreased freshwater flow (Paerl et al. 1995).

Valdes-Weaver et al. (2006) showed that species from chlorophytes, cryptophytes, cyanobacteria, diatoms and dinoflagellates are present in the Neuse in all seasons. In general, chlorophytes were positively correlated with river flow, while cyanobacteria were inversely related to flow and more dominant in summer months. Conversely, diatoms and dinoflagellates were more abundant in winter and spring. Before 1996, the typical river flow pattern reflected winter rains followed by early spring drought conditions. That scenario, which is illustrated by 1994 and 1995, includes large blooms of the nontoxic dinoflagellate *Heterocapsa triquetra* and cryptomonads *(Cryptomonas* spp.) in early spring (Figure 5.5.5). Those blooms were greatly reduced during the years in which high rainfall hurricanes affected the Neuse watershed (since 1996), but they could be coming back as a more regular feature in the early 2000s concurrent with a lack of high rainfall tropical storm events since 1999.

During the elevated freshwater discharge hurricane years (1996–2000), fast growing, chlorophytes (coccoids, flagellates and desmids) formed more frequent and extensive blooms throughout the estuary, starting in spring. Cyanobacteria, dominated by picoplanktonic species (*Synechococcus* spp.) including filamentous non-heterocystous species typically are most dominant during slow-flow, long residence time summer months, especially following high-flow spring runoff years (1998, 2000). Diatoms (data not shown) are a common feature of the phytoplankton community. They typically account for about 20 percent of the total phytoplankton community biomass but, interestingly, do not form extensive blooms. That is in sharp contrast to the nearby Chesapeake Bay, where diatom blooms are commonly observed in springtime. The dominant harm exerted by phytoplankton bloom taxa is their periodically high biomass, which is the fuel sustaining summertime bottom water hypoxia (Paerl et al. 1998).

Benthic Primary Producers

High levels of phytoplankton biomass accumulation, CDOM, and episodic sediment loading and internal resuspension can lead to severely restricted light penetration in the water column (Paerl et al. 1995; Luettich et al. 2000). As a result, approximately 80 percent of the Neuse primary production is phytoplanktonic (Paerl et al. 1998). Benthic microalgae are present, but their contribution to primary production is restricted to the shallow (< 2

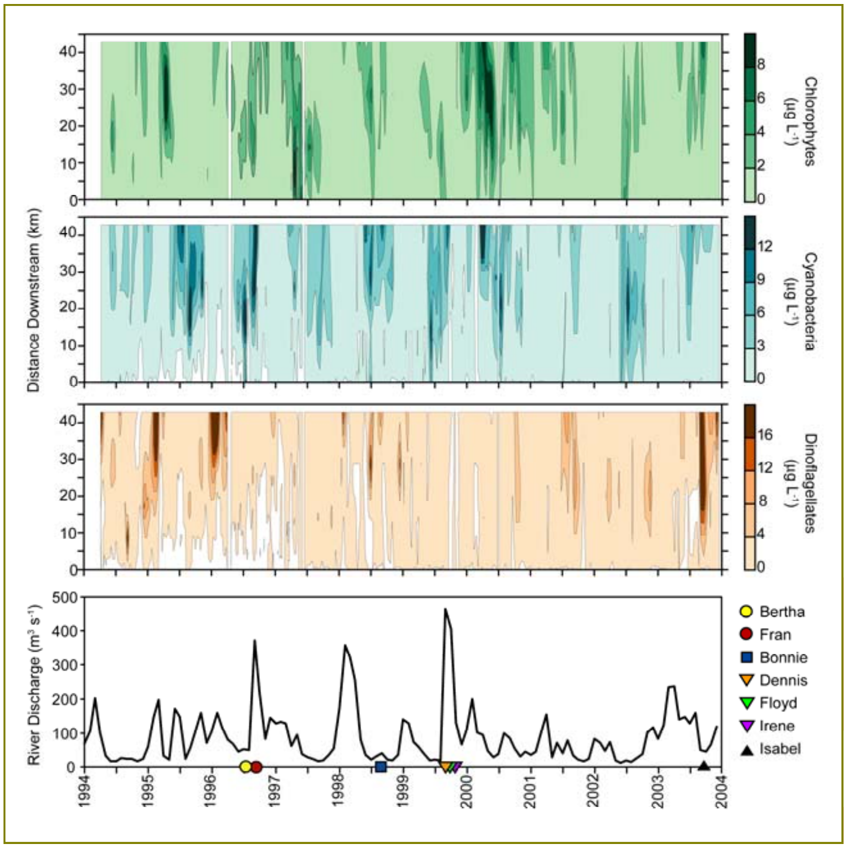

Figure 5.5.5. Phytoplankton community dynamics in the Neuse River Estuary. Shown are pre-hurricane (before 1996) and post-hurricane years (after 1996). HPLC-Chemtax derived biomass data of three major phytoplankton groups: chlorophytes, cyanobacteria and dinoflagellates that are shown for surveys along the length of the estuary. From Paerl et al. 2006a.

m) littoral zone or downstream waters in Pamlico Sound, as are benthic macroalgae and SAV to the best of our knowledge. Because phytoplankton is a major contributor to light attenuation (Woodruff et al. 1999; Gallegos 2001), decreased algal biomass through nutrient controls should substantially increase light penetration (Rizzo and Christian 1996; Darrow et al. 2003; Fear et al. 2004). Because most of the Neuse is historically devoid of SAV, it is hypothesized that higher clarity would promote the proliferation of microphytobenthos and reintegrate the benthic and pelagic ecosystem components (Vadaboncoeur et al. 2003).

Long-Term Changes

Nutrient enrichment of the Neuse has been ongoing since the late 1970s concurrent with a large increase in N and P use in the Neuse River basin (e.g., Stow et al. 2001). Control of eutrophication in the Neuse has focused on freshwater algal blooms through P control in the 1980s and more recently on estuarine blooms since the late 1990s (Paerl et al. 2004). The Neuse had severe cyanobacterial bloom problems (i.e., *Microcystis*) in the freshwater riverine regions during the 1970s and 1980s (Christian et al. 1986; Paerl 1987). Those problems were due to the combined effects of excessive P loading and a protracted number of years of summer drought. Through the initial control efforts, river PO_4^{-3} experienced a threefold decrease in concentration due to a PO_4^{-3} detergent ban and effluent P limits for municipal and industrial dischargers in the basin (e.g., Lebo et al. 2002). Concurrent with the decrease in P levels in the tidal river segment of the Neuse was a large decrease in peak and average chlorophyll *a* concentrations and the general absence of *Microcystis aeruginosa* (Paerl et al. 2004). After successful implementation of P controls in the Neuse River watershed, NO_3^- input to the tidal river segment of the Neuse and algal blooms in the middle segment increased (e.g., Lebo et al. 2002; Paerl et al. 2004). The second phase of nutrient control of inputs to the Neuse, which is being implemented, mandates a 30 percent reduction in TN loading to the system (from the 1991–1995 average loading) to reduce the frequencies and intensities of estuarine blooms (defined as chlorophyll *a* concentrations in excess of the state water quality standard of 40 µg L^{-1} [NC DENR 2002]).

Summary of Ecosystem Impacts

The Neuse River Estuary has experienced at least four decades of nutrient over-enrichment and enhanced sediment loading associated with urbanization and agricultural diversification in its watershed and overall airshed. Resultant eutrophication has led to increases in algal blooms (including some HABs), bottom water hypoxia, and episodes of fish disease and fish kills and to overall habitat degradation. Those changes have been captured by state and university collaborative long-term water quality monitoring programs (e.g., Luettich et al. 2000; NC DENR 2002). While those programs have documented nutrient-driven decline in water quality, they are also being used to gauge the effects of North Carolina state legislatively mandated nutrient input reductions implementing the TMDLs developed for the Neuse River Estuary to address unwanted symptoms of eutrophication. In addition to having been affected by anthropogenic stressors, the Neuse and Pamlico Sound have been affected by tropical storm and hurricane activity. Those events triggered substantial physical and biogeochemical changes lasting from one to several years, including shifts in salinity regimes, vertical stratification, residence time, hypoxia and anoxia. Biotic and ecological responses include changes in phytoplankton production and community structure, increased frequencies, expansion and duration of algal blooms, potential food web changes, habitat alteration and increased stress on finfish and shellfish species, most likely from large-scale habitat and water quality changes (Paerl et al. 2006c). Last, intense fishing pressure in the Neuse and Pamlico Sound has undoubtedly created additional, interactive, top-down stress on the system (Paerl et al. 2006c), the effects of which are under investigation.

As the largest sub-estuary of the Pamlico Sound, the Neuse reflects the range of biogeochemical and ecological impacts of major anthropogenic and natural (climatic) perturbation that affect the country's second largest estuarine complex. The Neuse is also representative of shallow, microtidal estuaries nationwide. High frequency, spatially and temporal-intensive monitoring programs have helped elucidate the complex interplay of human and climatic stressors that have affected this estuary and the downstream Pamlico Sound system. Data from those programs are being coupled to remote sensing of optically active parameters (temperature, turbidity, chlorophyll *a* and other diagnostic photopigments) to scale up to ecosystem-level responses. Mechanistic and probabilistic modeling efforts, based on data from these monitoring programs, also are helping to link land use changes to nutrient, hydrologic and other stressors that are driving water quality and habitat conditions and trends in the Neuse-Pamlico Sound continuum. That integration of watershed-based changes with conditions in the Neuse-Pamlico Sound and other stressors affecting the ecosystem facilitates adaptive nutrient management, policy and decision making for the system.

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Chapter 5:

6. San Francisco Bay

James E. Cloern and Richard Dugdale

General Description

San Francisco Bay is a large (1,240 km^2) estuary with a deep central channel, broad lateral shallow bays and intertidal mudflats (Figure 5.6.1), and overall mean depth of approximately 6 m at mean lower low water (MLLW) (Conomos 1979).

The San Francisco Bay-Delta ecosystem comprises two distinct but connected estuary types: North Bay is the tidal estuary of the Sacramento and San Joaquin rivers that carry runoff produced in a 163,000-km^2 agricultural watershed, and South Bay is a marine lagoon in the densely populated urban watershed between San Francisco, Oakland and Silicon Valley. River discharge is strongly seasonal, with peak discharge from winter storms and spring snowmelt in the Sierra Nevada mountains and low discharge during the dry summer to autumn. The salinity distribution in North Bay changes with seasonal fluctuations in Sacramento–San Joaquin discharge, and

Figure 5.6.1. Two maps of San Francisco Bay. The top map shows the broader location. The bottom map details the estuarine segments. North San Francisco Bay is the estuary between the Sacramento–San Joaquin rivers and the Pacific Ocean at Golden Gate. It comprises lateral shallows in the oligohaline Suisun Bay and mesohaline San Pablo Bay. South Bay is a marine lagoon-type estuary. Central Bay is a deep mixing basin of water originating in the Pacific Ocean, South, and North Bays. Sacramento and San Joaquin River flows are routed through the delta, a complex network of tidal freshwater channels and shallow, open-water habitats. Circles show USGS sampling stations.

salinity in South Bay responds to seasonal fluctuations in river discharge and runoff produced in the local urban watershed (Figure 5.6.2). Wastewater discharge is the primary source of fresh water to South Bay during the dry season.

The tides are mixed semidiurnally with maximum current speed of 1.75 m s^{-1} (Walters et al. 1985) and tidal amplitude of 1.7 m (Conomos 1979) at the Golden Gate. Tidal currents and amplitude are damped as the tidal wave propagates into North Bay, but the tidal range becomes amplified to 2.6 m in the closed South Bay (Conomos 1979). The water column is typically mixed by wind and tidal currents; density (primarily salinity) stratification develops only during neap tides of the wet season, and it breaks down on spring tides, so San Francisco Bay does not experience persistent stratification. Horizontal transports also vary over the neap-spring cycle, with amplification of net residual (baroclinic) currents driven by the horizontal density gradient during neap tides. Residence time varies from days in North Bay during large floods to months in South Bay during the dry season (Walters et al. 1985).

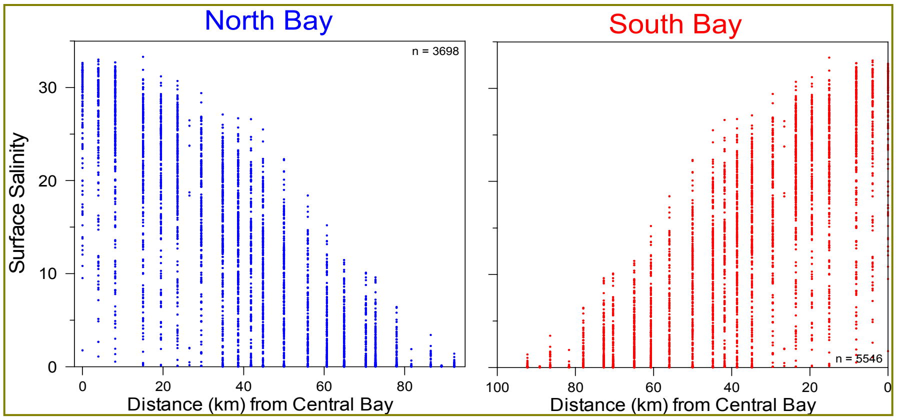

Figure 5.6.2. Surface salinity in North Bay and South Bay vs. distance from Golden Gate. Data shown are all measurements made by USGS from 1969 to 2001.

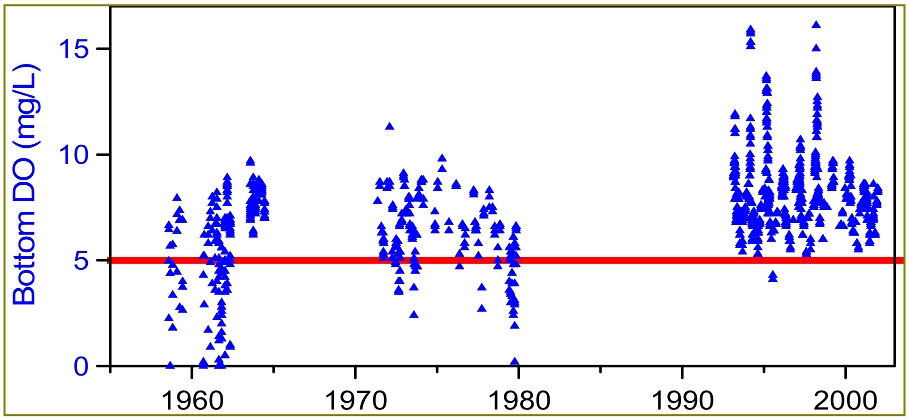

Figure 5.6.3. Near-bottom DO in lower South Bay. Hypoxia disappeared after implementations of advanced wastewater treatment that included nitrification.

Dissolved Oxygen

Through the 1970s, isolated regions of San Francisco Bay experienced seasonal or episodic hypoxia. However, after implementing advanced wastewater treatment mandated by the 1972 CWA, occurrences of hypoxia have been eliminated from South San Francisco Bay (Figure 5.6.3). Seasonal anoxia does develop at one location in the Delta—the deep ship channel at Stockton (Jassby and Van Nieuwenhuyse 2005).

Turbidity and Light

North San Francisco Bay receives > 1 metric ton of sediments yearly from the Sacramento and San Joaquin Rivers (McKee et al. 2006), and South Bay receives sediments from urban runoff delivered by

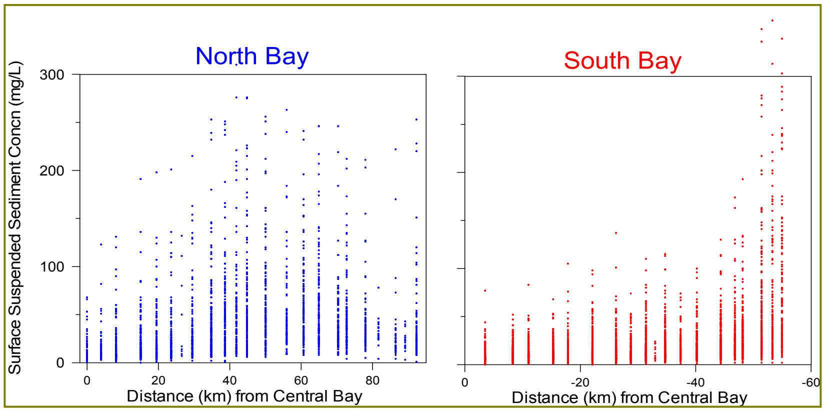

Figure 5.6.4. Surface suspended sediment concentrations in North Bay and South Bay versus distance from Golden Gate. Data shown are all measurements made by USGS 1969–2001.

local streams. Sediments delivered during the wet season are redistributed by tidal and wind-driven currents during the dry season, so there is strong seasonality in the concentration and spatial distribution of suspended sediments. Distributions along the North Bay salinity gradient reflect accumulation of sediments in a mid-estuary turbidity maximum, and distributions along South Bay reflect the large source from urban runoff in the southernmost basin (Figure 5.6.4).

As a result of large river inputs, shallow depth and continual resuspension from winds and tides, San Francisco Bay has high concentrations of suspended sediments and high turbidity that limit phytoplankton photosynthesis and growth rates**.** From the long-term measurements, the mean photic depths in the channels of North and South Bay are 1.4 m and 2.1 m, respectively. Turbidity is substantially higher in the lateral subtidal shallows subjected to wind-wave resuspension (Cloern et al. 1985; May et al. 2003).

Nutrients

San Francisco Bay is highly enriched in N and P, a result of riverine inputs from a large agricultural watershed and wastewater discharge from > 6 million people (Smith and Hollibaugh 2006). Overall distributions of DIN (= NH_4^+ + NO_3^- + NO_2^-) and P (as PO_4^{-3}) do not show consistent strong spatial patterns along the salinity gradient (Figures 5.6.5, 5.6.6). However, distributions of $Si(OH)_4$ reveal the strong freshwater source of this nutrient (Figure 5.6.7).

Figures 5.6.5 to 5.6.7 compare dissolved inorganic nutrient concentrations along the salinity gradient against one index of potential limitation of phytoplankton growth: a value taken as twice the half-saturation constants for algal growth (K_P = 0.15 µM-P, K_N = K_{Si} = 1.5 µM-N or Si). On the basis of this index, dissolved inorganic nutrient concentrations are virtually always well above those that limit algal growth in North Bay. Potential P limitation is extremely rare (only 2 of 1,264 measurements), but potential N and Si limitation occur episodically in South Bay during unusually large spring diatom blooms (Cloern 1996).

Although the 12-year data set shows no obvious pattern for DIN versus salinity, the winter data (when biological activity is low) for NO_3^-, NH_4^+, and $Si(OH)_4$ all show a strong inverse relationship to salinity (Figure 5.6.8). The major source of the nutrients in North San Francisco Bay is in the freshwater inputs at the head of the estuary.

Recent studies show the anthropogenically altered composition of DIN (relative contribution of NH_4^+ and NO_3^-) may have resulted in decreased primary production. Measurements of NO_3^- and NH_4^+ uptake made from 1999 to 2003 with the stable isotopic tracer ^{15}N revealed that NO_3^- uptake rarely occurs in the North Bay, a result of high ambient NH_4^+ concentrations that inhibit uptake of NO_3^- by the phytoplankton (Figure 5.6.9 upper panel).

High NH_4^+ inputs to the estuary resulted from the conversion of sewage treatment from primary to secondary in the early 1980s with the result that now spring phytoplankton blooms occur only in years when NH_4^+ concentrations decline to about 4 µM-N or less (Dugdale et al. 2007) and the available NO_3^- can then be taken up at higher rates than for NH_4^+ (Figure 5.6.9 lower panel) with maximal levels of chlorophyll *a* accumulation. For blooms to occur in North Bay, favorable irradiance conditions are required to reduce NH_4^+ concentrations to non-inhibiting levels and then allow rapid NO_3^- uptake. Because NO_3^- is present in concentrations higher than NH_4^+ by a factor of up to 10, phytoplankton or chlorophyll *a* accumulation on NH_4^+ only will be comparatively weak. Growth rates on NH_4^+ are also only 50 percent of that on NO_3^- in those measurements (Figure 5.6.9). The high NH_4^+ concentrations reduce the frequency and size of blooms and the rate of primary production (Dugdale et al. 2007), a condition that becomes more severe as NH_4^+ increases toward the head of the estuary (Figure 5.6.8).

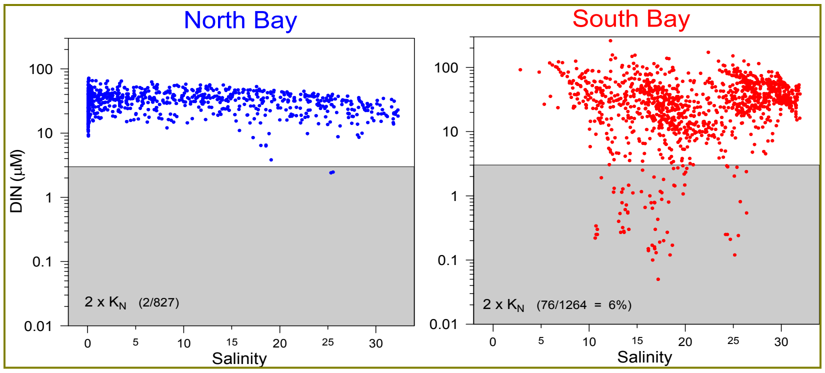

Figure 5.6.5. Near-surface DIN concentration versus salinity in North and South Bay. Data shown are all measurements made by USGS, 1988–2000. The gray rectangle indicates potential N limitation.

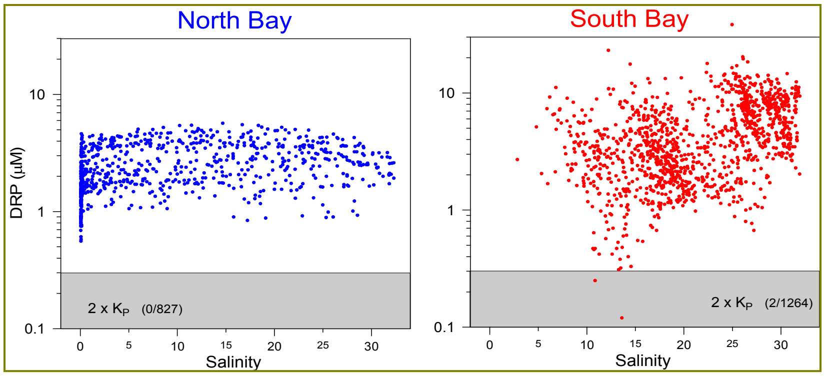

Figure 5.6.6. Near-surface DRP (PO_4^{-3}) concentration versus salinity in North and South Bay. Data shown are all measurements made by USGS, 1988–2000. The gray rectangle indicates potential P limitation.

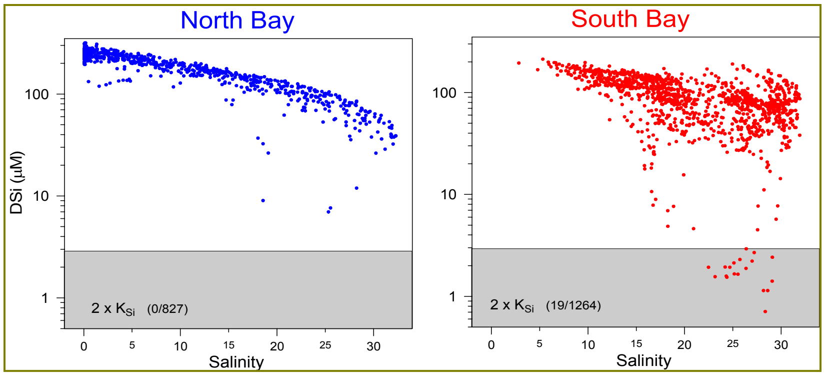

Figure 5.6.7. Near-surface $Si(OH)_4$ (DSi) concentration versus salinity in North and South Bay. Data shown are all measurements made by USGS, 1988–2000. The gray rectangle indicates potential Si limitation.

Chlorophyll and Primary Production

The Sacramento-San Joaquin Delta and North and South San Francisco Bay have each experienced large and ecologically important changes in chlorophyll *a* concentrations and seasonal patterns over the past three decades. As a result of a multidecadal trend of chlorophyll *a* decrease, primary production in the delta is very low (~ 75 g C m^{-2}) and declined > 40 percent between 1975 and 1995 (Jassby et al. 2002). Low primary production is considered a contributing factor to declining stocks of zooplankton and planktivorous fish, including native species listed as threatened or endangered, and the CALFED Ecosystem Restoration Program is considering actions to aug-

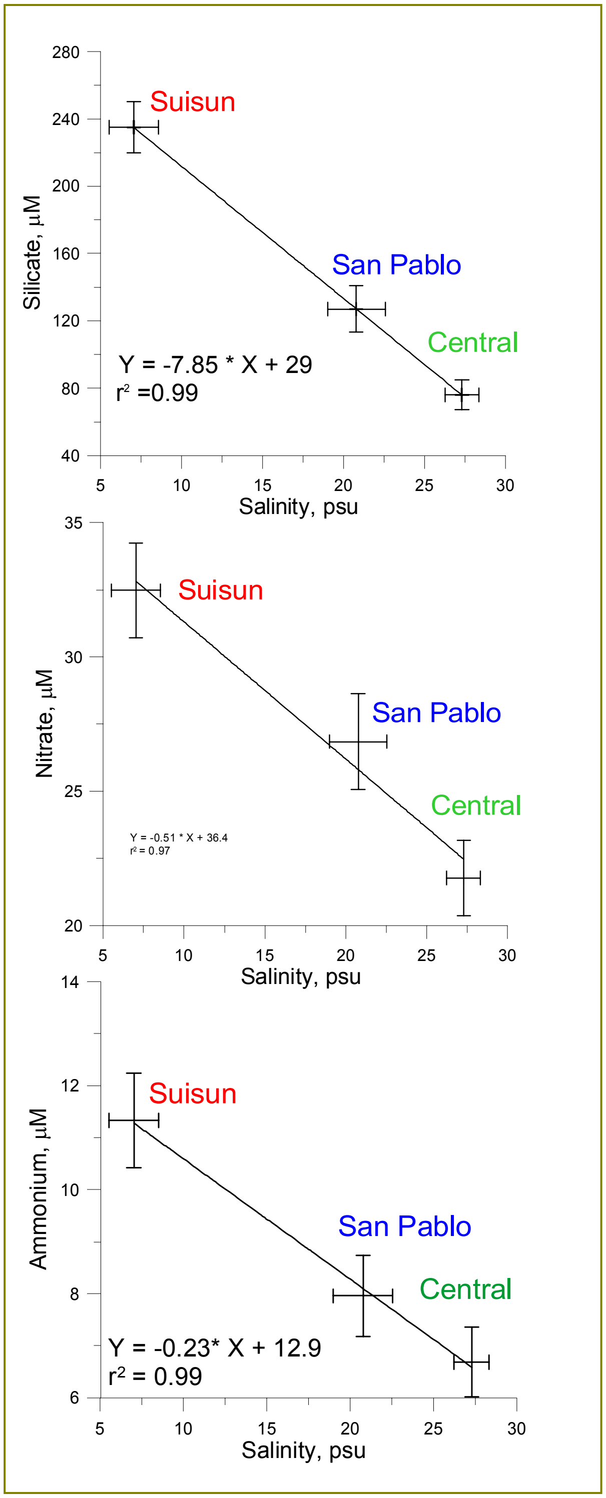

Figure 5.6.8. Mean nutrient concentrations, µM (± standard error of mean) for Suisun, San Pablo and Central bays versus mean salinity for 1999–2003 for the winter months (December, January, February): (top panel) $Si(OH)_4$, (center) NO_3^-, (bottom) NH_4^+ showing higher concentrations at the head of the bay. From Wilkerson et al. 2006.

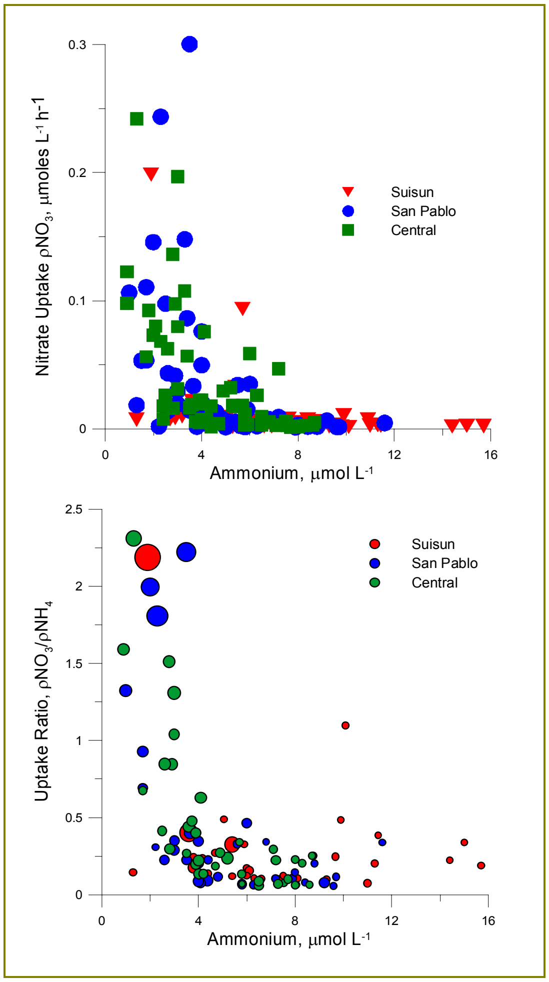

Figure 5.6.9. (Top panel) Saturated NO_3^- uptake, ρNO_3^-, µM h^{-1} versus NH_4^+ concentration, µM-N for Suisun, San Pablo and Central Bays and (bottom) ratio of saturated NO_3^- to NH_4^+ uptake, ρNO_3^- :ρNH_4^+ versus NH_4^+. The points in the graph are shown as bubbles that indicate higher chlorophyll a concentrations with high NO_3^- uptake and low NH_4^+ values. From Dugdale et al. 2007.

ment the phytoplankton food supply to consumers (Lopez et al. 2006). Management goals to amplify phytoplankton production in the nutrient-rich delta provide an instructive contrast to management goals of reducing phytoplankton biomass in estuaries where nutrient enrichment has stimulated excessive algal production.

The phytoplankton seasonal cycle in the oligohaline North Bay (Suisun Bay) was historically characterized by a large summer bloom, with variations of the annual cycle by hydrologic extremes such as the record high discharge of 1983 and record drought of 1977 (Figure 5.6.10). That pattern ended abruptly in 1987 as Suisun Bay became rapidly and densely colonized by the non-native clam *Corbula amurensis*. Phytoplankton primary production was low (~100 g C m^{-2}) in Suisun Bay before that invasion, but it was reduced five-fold after the non-native filter feeder prevented development of the summer bloom (Alpine and Cloern 1992). Reduced primary production has led to collapse of the pelagic food web, with near extinction of the estuarine copepod *Eurytemora affinis* and native mysid *Neomysis mercedis* (Orsi and Mecum 1996).

Uptake rates of NH_4^+ in Suisun Bay are lower than in the other two northern embayments. The presence of a chemical toxin has been suggested by experiments in which inoculated phytoplankton failed to grow in Suisun water but grew in San Pablo and Central Bay water. The low NH_4^+ uptake rates combined with the highest NH_4^+ concentrations in the North Bay, indicate a low probability of a bloom in Suisun Bay.

The seasonal pattern in South Bay has historically (1977 to 1998) been characterized by a spring (March to April) bloom followed by persistent low chlorophyll *a* and high nutrient concentrations. This pattern changed in 1999 when new autumn-winter blooms occurred and a trend of increasing autumn-winter chlorophyll *a* began (Figure 5.6.11). The underlying cause(s) of the regime change in South Bay is unknown, but the contemporaneous trends of phytoplankton decrease in North Bay and increase in South Bay illustrate the complexity of estuarine phytoplankton dynamics and their nonuniform response to nutrient enrichment.

Phytoplankton Blooms and Species Composition

Phytoplankton biomass in San Francisco Bay is overwhelmingly (> 80 percent) dominated by diatoms, primarily because of their importance during spring blooms (Cloern and Dufford 2005). Spring blooms are episodes of population growth by marine taxa, many of which develop in the adjacent California Current upwelling system, such as *Coscinodiscus* spp., *Thalassiosira* spp., *Chaetoceros* spp. and *Skeletonema* spp. Other common diatom taxa include benthic forms, such as *Paralia sulcata*, *Gyrosigma* spp. and *Pleurosigma* spp., indicating suspension into the plankton of cells produced on the mudflats. Dinoflagellates (e.g., *Akashiwo sanguinea, Heterocapsa triquetra*, *Prorocentrum minimum*, *Alexan-*

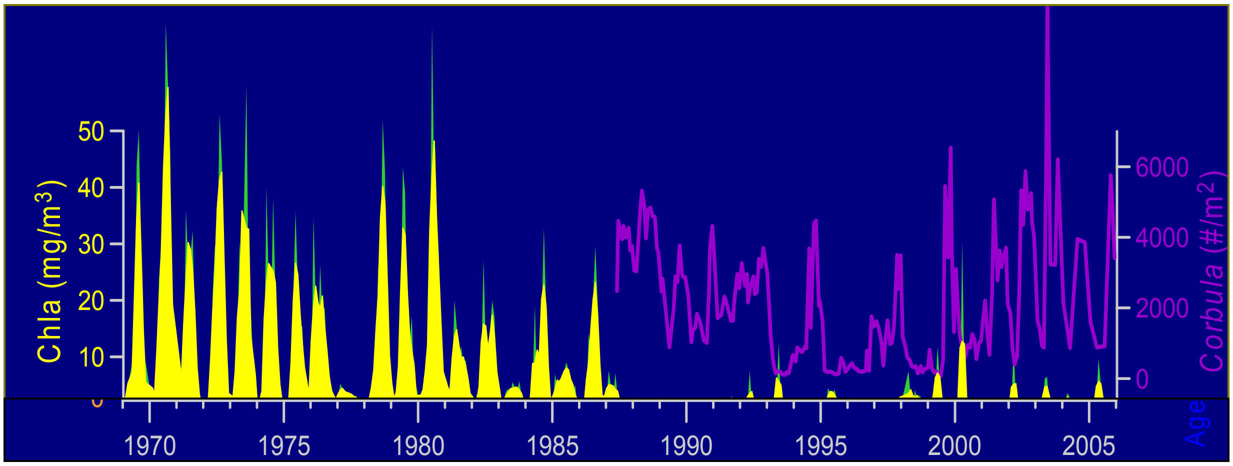

Figure 5.6.10. Time series of chlorophyll a concentration (yellow) and abundance of the alien clam Corbula amurensis (purple) in Suisun Bay. Data are from the Interagency Ecological Program (http://bdat.ca.gov/).

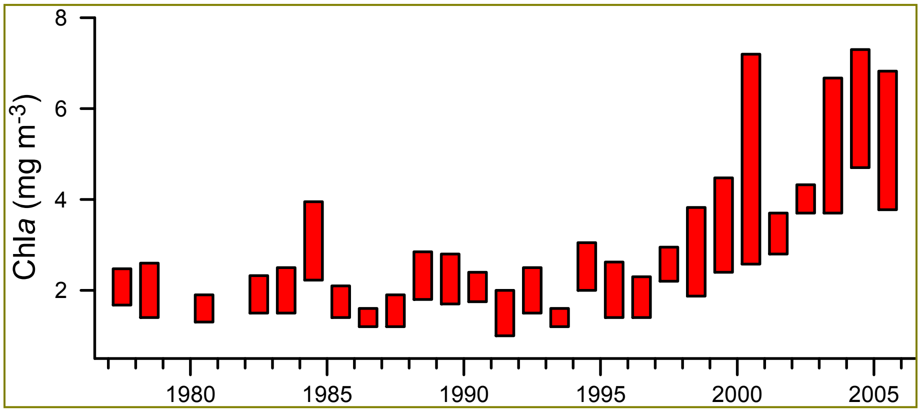

Figure 5.6.11. Interquartile ranges of surface chlorophyll a in South Bay during August–December, showing significant trends of increase after 1998.

drium spp. and *Protoperidinium* spp.) compose 11 percent of overall phytoplankton biomass, and some of those taxa are also commonly found in the California current following relaxation of upwelling. Cryptophytes, which compose 5 percent of overall biomass, are widely distributed in time and space but do not dominate blooms. Picocyanobacteria, primarily *Synechococcus*, constitute a small component of phytoplankton biomass in San Francisco Bay, consistent with the generality that phytoplankton biomass is dominated by large cells in high-nutrient pelagic habitats (Ning et al. 2000). Cole et al. (1986) measured size-fractionated biomass and primary production every 2 weeks at a shallow and deep site in Suisun, San Pablo and South bays and determined large cells to be the most important fraction of phytoplankton across the estuary. Wilkerson et al. (2006) measured fractionated chlorophyll *a* and N productivity and similarly found phytoplankton in San Francisco Bay to be mostly cells > 5 μm and > 10 μm in diameter.

The plankton of San Francisco Bay includes many species that have produced toxic or harmful blooms elsewhere, but there are no documented occurrences of HAB events causing human illness or mortality of fish or shellfish. Visible blooms of *Mesodinium rubrum* occur during years of high runoff (Cloern et al. 1994), but they are benign and perhaps beneficial to consumers. Episodic red tides of *Heterosigma akashwio*, *Alexandrium* spp., and *Akashiwo sanguinea* have been observed in recent years, but each seems to have been triggered by events in the coastal Pacific that propagated into the estuary. Evidence suggests that water and habitat quality in the freshwater delta have been impaired in recent years by blooms of the toxic cyanobacterium *Microcystis aerugenisa* (Lehman et al. 2005).

Benthic Primary Producers

As a contrast to many estuaries of the U.S. East Coast, San Francisco Bay does not provide habitat for SAV because of its high turbidity and muddy, mobile bottom. However, intertidal mudflats provide large habitat surface for benthic microalgae. Primary production by that community has not been measured systematically, but estimates based on habitat area and seasonal rate measurements indicate that the benthic microalgae contribute about one-third of total primary production in San Francisco Bay (Jassby et al. 1993; Guarini et al. 2002).

Long-Term Changes

San Francisco Bay has been described as the world's *most invaded estuary* with more than 230 nonnative species (Cohen and Carlton 1998), many of which have caused major ecological disturbance. The biomass of benthic invertebrates is dominated by nonnative species (Nichols et al. 1986), the mesozooplankton community has been transformed by nonnative species several times over the past three decades (Kimmerer and Orsi 1996; Hooff and Bollens 2004), and competition/predation by introduced freshwater fish is a contributing factor to the long-term declines of native species in the delta and Suisun Bay (http://science.calwater.ca.gov/pod/pod_synthesis.html). Sustainability of native fauna has also been compromised by habitat loss, inputs of toxic contaminants (emerging pesticides such as pyrethroids, selenium, heavy metals), legacy contaminants (PCBs, mercury), and water diversions that export up to 80 percent of the freshwater inflow to the Delta and San Francisco Bay. Those stressors and their interactions have caused decreased abundance and complexity of indigenous biological communities in San Francisco Bay and its river delta. Although nutrient enrichment has clearly caused comparable disturbance of other estuaries around the world, and San Francisco Bay has clearly become enriched with N and P as a result of human activities, the ecological consequences of enriching this estuary are not well established. However, a correlation appears between increased NH_4^+ concentration with reduced primary productivity, observed also in the Delaware Estuary (Yoshiyama and Sharp 2006). The recent occurrences of dinoflagellate red tides, altered seasonality of phytoplankton and significant trends of chlorophyll *a* increase in South Bay suggest the possibility that the estuary's apparent resistance to the harmful consequences of enrichment might be changing.

Summary of Ecosystem Impacts

San Francisco Bay presents a valuable contrast to other nutrient-enriched estuaries because its phytoplankton biomass is highly variable in time and space, but only some of that variability can be attributed to changing nutrient availability. The persistence of high nutrient concentrations during the past decades of phytoplankton increase in South Bay and decrease in North Bay implies that phytoplankton dynamics in this estuary are driven by processes other than simple nutrient regulation of growth rate. However, the persistent high nutrient concentrations are in part the result of increased NH_4^+ concentrations, reducing phytoplankton use of NO_3^- and reducing phytoplankton growth rates. In considering anthropogenic impacts on estuaries, it is important to distinguish between what might be called intrinsic limitations, i.e., that which would be in place and controlling ecosystem characteristics historically, and anthropogenic factors superimposed on intrinsic limitations. For example, the well-demonstrated intrinsic limitation in San Francisco Bay is the ambient turbidity and its effect on the irradiance field, relegating the estuary to a low position in the productivity hierarchy of estuaries. The high NH_4^+ concentrations that probably resulted from changes in treatment practices impose a further reduction in primary productivity and increased variability.

Sustained research in San Francisco Bay has documented the importance of (1) top-down control by benthic suspension feeders (Cloern 1982; Lopez et al. 2006); (2) salinity stratification on neap tides that induces blooms by retaining phytoplankton cells in a high-light, high-nutrient, surface layer and isolating them from benthic consumers (Cloern 1991); (3) net, tidal-residual currents that transport phytoplankton between habitats that function as net sources and sinks of algal biomass (Lucas et al. 1999); (4) connectivity to the Pacific Ocean as a source of phytoplankton cells that can seed blooms within the estuary (Cloern et al. 2005); (5) high turbidity as a constraint on phytoplankton growth rate such that the large pool of dissolved inorganic nutrients is not fully assimilated into biomass (Alpine and Cloern 1988; Cloern 1999); and (6) interactions between the diel light and semidiurnal tidal cycles that determine whether light exposure is sufficient to sustain blooms (Lucas and Cloern 2002). Recent studies have documented (1) the importance of high NH_4^+ inputs in reducing the frequency and intensity of spring blooms through inhibition of NO_3^- uptake; (2) the necessity for periods of favorable irradiance conditions about 5 days for bloom initiation; and (3) the suggestion that in Suisun Bay, primary productivity is additionally impaired by undetermined chemical inhibitors.

San Francisco Bay receives comparable areal loadings of N and even larger loadings of P than the Chesapeake Bay, but it has much lower phytoplankton biomass and primary production. That contrast exemplifies the variability across estuaries in their response to nutrient enrichment (Cloern 2001), it also highlights the importance of studying and managing eutrophication in the context of all processes that regulate the efficiency with which estuarine ecosystems convert exogenous nutrients into algal biomass.

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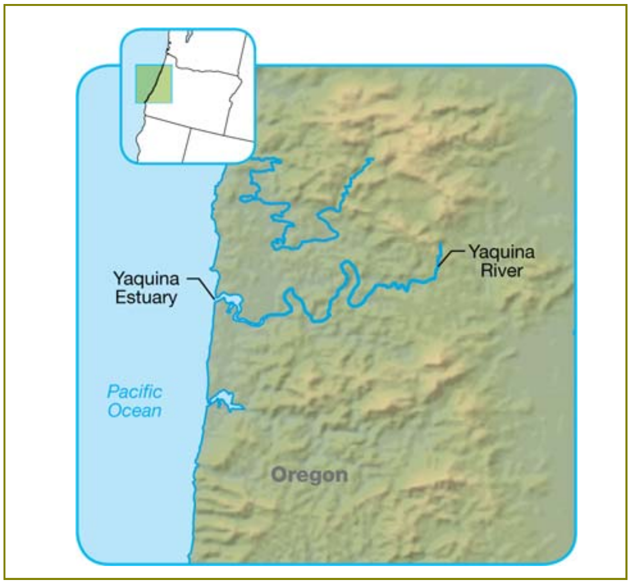
Chapter 5:

7. Yaquina Estuary

Cheryl A. Brown and
Walter G. Nelson



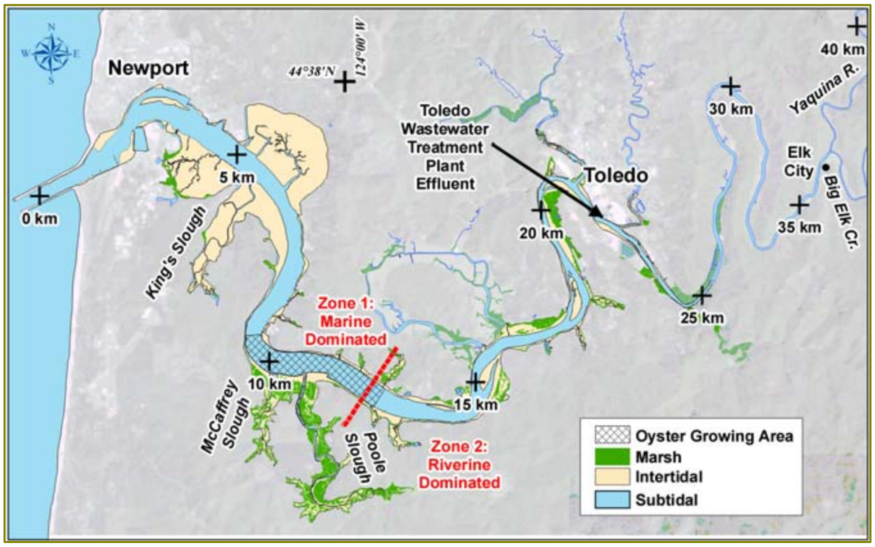


Figure 5.7.1. Two maps of the Yaquina Estuary. The top map shows the broader location; the bottom map shows the estuarine zones.

General Description

Yaquina Estuary is a small, drowned-river valley estuary along the central Oregon coast (44.62°N, 124.02°W; Figure 5.7.1) with a surface area of 14 km^2 and a watershed surface area of 650 km^2 (Lee et al. 2006). The Yaquina watershed is heavily forested with deciduous, evergreen and shrub land use classes constituting 85 percent of the watershed (Lee et al. 2006). Grasslands constitute about 6 percent of the watershed, while high- and low-intensity development combined constitute only about 0.5 percent of the watershed. The population in the Yaquina watershed is low (12.3 persons per km^2; Lee et al. 2006 based on U.S. Census 2000 data). The estuary experiences mixed semidiurnal tides with mean tidal range of approximately 1.9 m and a tidal prism volume of about 2.4 x 10^7 m^3 (Shirzad et al. 1988).

Because of the small volume of the estuary (2.5 x 10^7 m^3 at MLLW) and the strong tidal forcing, the estuary and the coastal ocean are closely coupled. Approximately 70 percent of the volume of the estuary is exchanged with the coastal ocean during each tidal cycle (Karentz and McIntire 1977). Yaquina Estuary receives freshwater inflow primarily from two tributaries, the Yaquina River and Elk Creek, which have similarly sized drainage areas and contribute approximately equally to the freshwater inflow (Oregon State Water Resources Board 1965). In November through April, the Oregon coast receives high precipitation, and the estuary is river-dominated. Beginning in May and continuing through October, the riverine freshwater inflow declines, and the estuary switches from riverine to marine dominance. The estuary is classified as well-mixed under low-flow conditions and as partially mixed during winter, high riverine inflow conditions (Burt and McAlister 1959). The flushing time of the estuary during the dry season varies from one day near the mouth to 9 days in the upstream portions (Choi 1975). In May to October, winds from the north drive coastal upwelling on the shelf adjacent to the estuary, which brings cold, nutrient (NO_3^- and PO_4^{-3})-rich waters to the surface that enter the estuary during flood tides. In addition to the riverine and oceanic nutrient inputs to the system, Toledo, Oregon, population of 3,400 (U.S. Census Bureau 2004) discharges wastewater treatment facility effluent into the Yaquina Estuary about 22 km upstream of the estuary mouth.

The wet season (November to April) is defined as when the monthly median discharge of Yaquina River and Elk Creek (computed using data from 1972 to 2002) exceeds the 30-year median discharge of Yaquina River and Elk Creek of 7.5 $m^3 s^{-1}$, while the dry season (May to October) is defined as when the monthly median discharge is less than the long-term median. An approximately fivefold difference exists in the average daily wet season (26.7 $m^3 s^{-1}$) and dry season (5.1 $m^3 s^{-1}$) riverine discharge into the estuary and about 84 percent of the annual freshwater inflow enters the estuary during the wet season.

On the basis of analysis of water quality and isotopic data and hydrodynamic modeling, the estuary can be divided into two zones, a marine dominated zone (zone 1), in which the water properties are primarily determined by ocean conditions, and a riverine-dominated zone (zone 2), in which watershed inputs primarily determine water properties (Brown et al. 2007; Figure 5.7.1). The division between the two zones occurs approximately at a median dry season salinity of 26.

Dissolved Oxygen

Seasonal differences exist in DO with median wet and dry season DO levels of 10.4 mg L^{-1} and 7.4 mg L^{-1}, respectively. Since 2002, the incidence of hypoxic events on the Oregon shelf have increased (Grantham et al. 2004); those have the potential to influence DO levels within the estuary. Recent DO data collected near the mouth of the estuary demonstrates that hypoxic shelf water is imported into the estuary during flood tides (Brown et al. 2007; Figure 5.7.2). Minimum DO levels occur during maximum salinities, demonstrating that the hypoxic water is imported into the estuary during flood tides. The intervals of low DO conditions are relatively short with DO levels increasing to 6 to 8 mg L^{-1} during

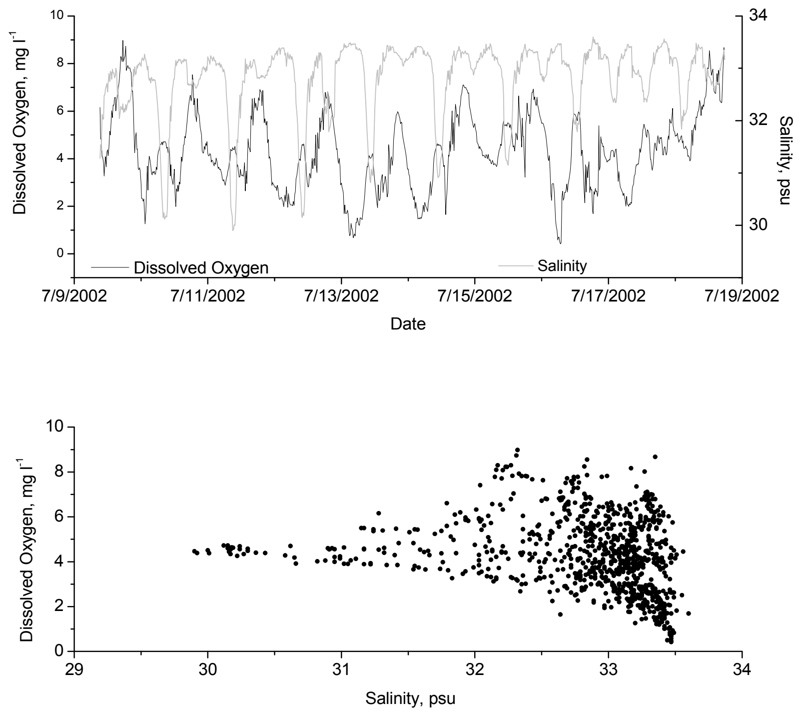


Figure 5.7.2. Top: time series of DO and salinity. Bottom: salinity versus DO showing import of hypoxic ocean water at a station 4 km from the mouth of the Yaquina Estuary.

ebb tides. Oregon has set a DO criterion for estuarine waters to be 6.5 mg L^{-1}. Using data from datasondes deployed at two locations during the dry season of 2006, one 3.7 km from the mouth and the other 18 km from the mouth, the frequency with which Oregon DO criteria were not met during May to October of 2006 were examined. DO levels fell below the 6.5 mg L^{-1} standard 37 percent and 28 percent in these areas (Brown et al. 2007). The more frequent occurrence of low DO levels near the mouth is probably related to the oceanic import of hypoxic water.

Turbidity and Light

Concentrations of TSS are about 10 mg L^{-1} in the marine dominated section of the estuary and increase to about 30 mg L^{-1} in the freshwater portion (Callaway et al. 1988). Wet and dry season TSS levels are similar with median values of about 8.3 mg L^{-1} (Brown et al. 2007). A statistically significant difference exists in TSS between zone 1 and zone 2 during both seasons with median zonal values of 6.5 and 11.7 mg L^{-1}, respectively (Brown et al. 2007). A turbidity maximum occurs about 15 km from the mouth of the estuary (Figure 5.7.3). The water is relatively clear throughout the year in the lower estuary because of the input of ocean water, which is also evident in the low turbidities near the mouth (Figure 5.7.3). The months of highest light attenuation approximately coincide with periods of maximal water column chlorophyll *a*.

Nutrients

Oceanic and riverine inputs are the major sources of nutrients to the estuary, with oceanic sources dominating during the dry season and riverine sources dominating

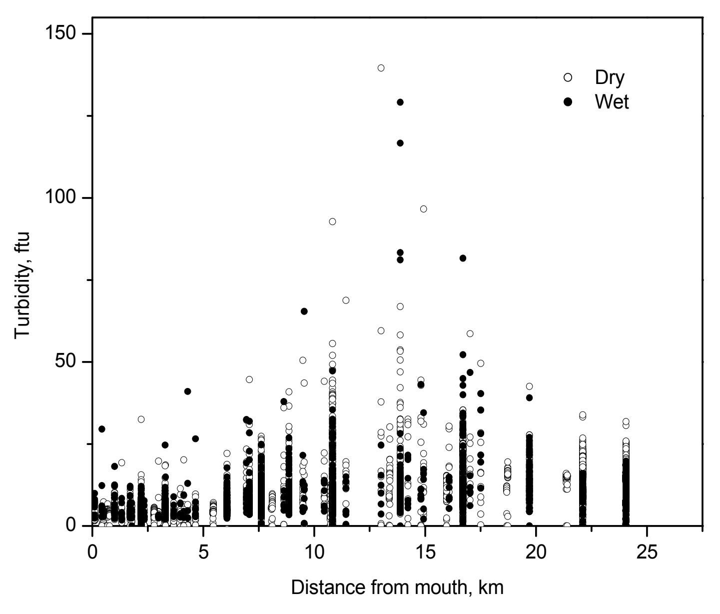


Figure 5.7.3. Spatial variation in turbidity for wet and dry seasons.

Table 5.7.1. Comparison of N sources during wet and dry seasons for Yaquina Estuary, Oregon (from Brown and Ozretich 2009; Brown et al. 2007).

Source	Nitrogen input: Wet season, mol DIN d^{-1}	Nitrogen input: Dry season, mol DIN d^{-1}	Nitrogen input: Annual average, mol DIN d^{-1}
River	2.7×10^5	2.5×10^4	1.4×10^5
Ocean	3.0×10^4	$3.7–4.7 \times 10^5$	2.3×10^5
Wastewater	1.7×10^3	1.5×10^3	1.6×10^3
Benthic flux[a]	--	4.3×10^4	--
Atmospheric deposition On estuary On watershed	 2.2×10^2 1.1×10^4	 1.2×10^2 6.0×10^3	 1.7×10^2 8.5×10^3

Note:
a. DeWitt et al. 2004

during the wet season (Table 5.7.1). Atmospheric deposition and wastewater treatment facility effluent are minor sources of N to Yaquina Estuary (Brown and Ozretich 2009). During the wet season, riverine input is the largest source of DIN to the estuary, composing approximately 78 percent of the input and about 91 percent of the annual riverine N input delivered during the wet season. The amount of DIN in the riverine water is related to the discharge, with wet season DIN levels averaging 95 µM-N and dry season averaging about 42 µM-N.

Compton et al. (2003) found that the presence of N-fixing red alder (*Alnus rubra*) in Pacific Northwest watersheds influences the N export from the watershed into streams. Alder is a native species in the Pacific Northwest that colonizes areas disturbed by fires, logging and landslides. Compton et al. (2003) found a significant relationship between alder cover in the watershed and NO_3^- concentration in the streams in the Salmon River watershed, which is 45 km north of Yaquina Estuary. Using 1996 vegetation data obtained from the Coastal Landscape Analysis and Modeling Study (http://www.fsl.orst.edu/clams), it is estimated that approximately 23 percent of the Yaquina watershed is vegetated with red alder (assuming that the broadleaf cover is primarily alder). Using published N-fixation rates for red alder (Boring et al. 1988; Binkley et al. 1994), stream N export rates (Compton et al. 2003) and the coverage of alder in the Yaquina watershed, it is estimated that 80 to 100 percent of the annual riverine N loading to Yaquina Estuary is related to the presence of red alder in the watershed. Thus, riverine nutrient loading is strongly influenced by forest species composition.

During the wet season, NO_3^- is the primary form of DIN in the estuary (median = 88 percent), and the river is the dominant source (Figure 5.7.4). Phytoplankton uses little of the dissolved inorganic nutrients in the estuary during the wet season because of short residence time (from high freshwater inflow) and low solar irradiance. The average incident photosynthetically active radiation (PAR) varies from 15 mol quanta m^{-2} d^{-1} during the wet season to 38 mol quanta m^{-2} d^{-1} during the dry season.

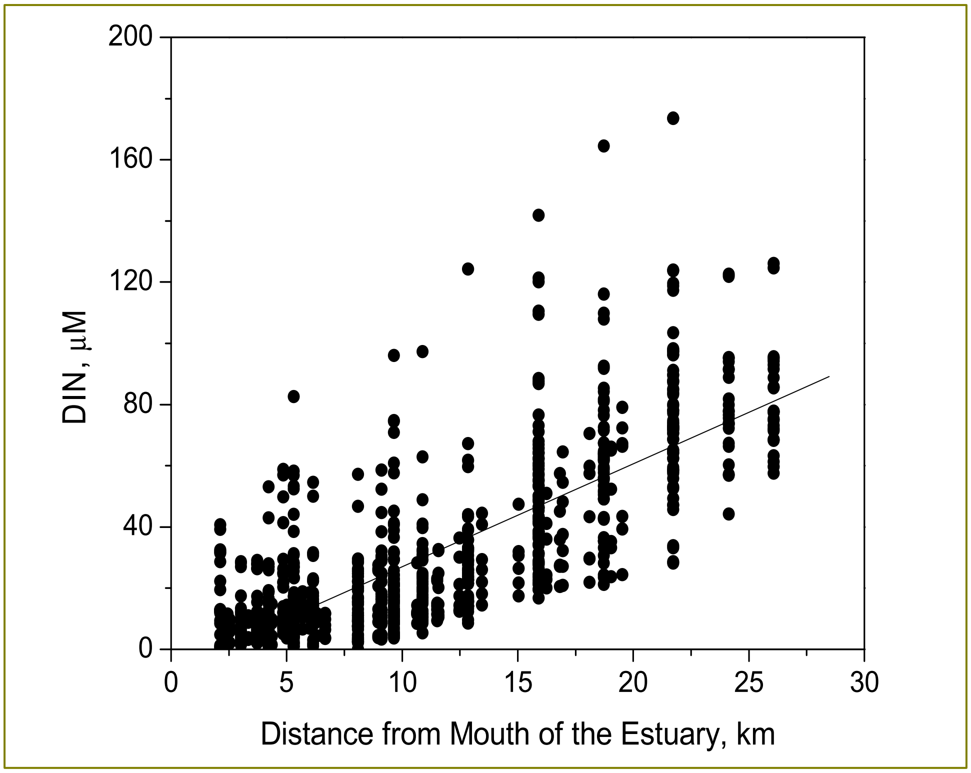


Figure 5.7.4. Spatial variation in wet season DIN. The line indicates linear regression (DIN = –6.7 + 3.4 × distance, r^2 = 0.58, p < 0.001).

During the dry season, NO_3^- is the primary form of DIN (median of 75 percent), while NO_2^- is a minor component contributing only about 2 percent. A mid-estuary minima is in the mean dry season NO_3^- + NO_2^-, suggesting that the estuary receives NO_3^- from both the ocean and the river

(Figure 5.7.5). Nutrient-rich water associated with coastal upwelling is advected into the Yaquina Estuary during flood tides. During the dry season, high levels of DIN and PO_4^{-3} entering the estuary lag upwelling favorable winds by about 2 days (Brown and Ozretich 2009). Median inputs of oceanic NO_3^- and PO_4^{-3} to the estuary during the dry season are 8.6 µM-N and 1.3 µM-P, respectively (n = 830). The primary source of PO_4^{-3} to the system is the ocean, and a steady decline in PO_4^{-3} occurs with distance into the estuary (Figure 5.7.6). The oceanic signal in NO_3^- and PO_4^{-3} propagates approximately 13 km up the estuary (Brown and Ozretich 2009).

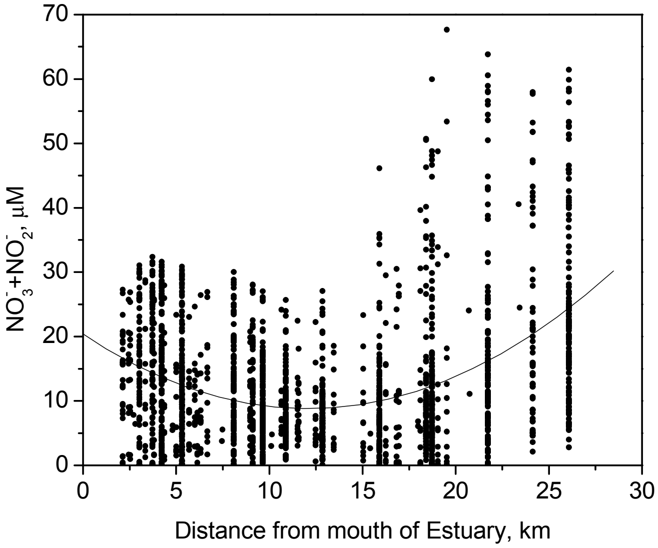


Figure 5.7.5. Spatial variation in dry season $NO_3^- + NO_2^-$. The line indicates a second-order polynomial fit to the data ($NO_3^- + NO_2^- = 20.4 - 1.9 \times \text{distance} + 0.079 \times \text{distance}^2$, $r^2 = 0.15$, $p < 0.001$).

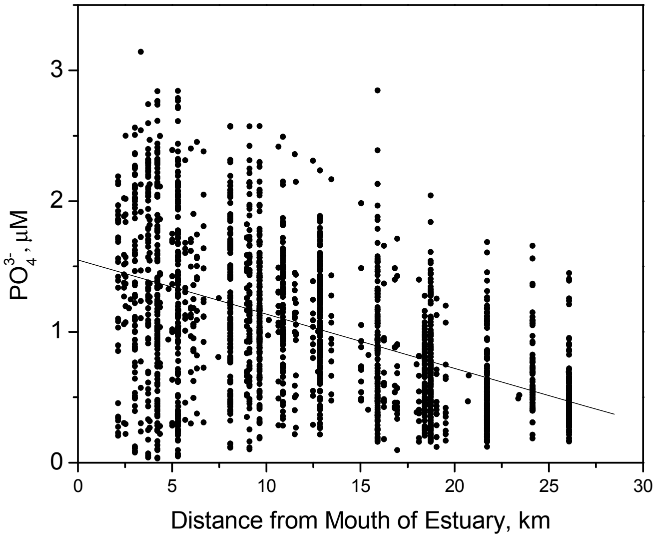


Figure 5.7.6. Spatial variation in dry season PO_4^{-3}. The line indicates linear regression ($PO_4^{-3} = 1.55 - 0.041 \times \text{distance}$, $r^2 = 0.24$, $p < 0.0001$).

The median N:P ratio during the dry season is approximately 12:1, suggesting that N will be depleted before P for the majority of the estuary. Evidence of P limitation in the upper portions of the estuary exists (17 to 27 km from the mouth) with the N:P ratio reaching as high as 260:1. In only 12 percent of the estuarine sampling events were the N:P ratios greater than 20 and PO_4^{-3} less than 0.5 µM-P, suggesting the potential for P limitation. During the dry season, the median DIN concentration is 14 µM-N, and 95 percent of the time the DIN concentration is greater than 2 µM-N (typical half saturation constant for phytoplankton). In only 5 percent of the estuarine sampling events were the N:P ratios < 10 and DIN < 2 µM. That suggests that although the N:P ratio often falls below 16:1, the estuary is not usually limited by either N or P. That is supported by assimilation ratio data (primary production: chlorophyll *a;* P:B) of Johnson (1981) collected during the dry season at a station about 16 km from the mouth of the estuary (Figure 5.7.1), which suggests that 77 percent of the time, there were sufficient nutrients for planktonic primary production; 15 percent of the time, there was borderline nutrient deficiency; and 8 percent of the time, there was evidence of nutrient depletion. Specht (1975) conducted algal bioassays at six locations in Yaquina Estuary during 1972–1975 to examine potential for N and P limitation. The experiment suggests that the upper portion of the estuary (26 km from mouth to tidal fresh) was predominantly P limited, while in the lower estuary, the system is N limited during the dry season and P limited during the wet season.

Chlorophyll and Primary Production

Previous studies have demonstrated that chlorophyll *a* is advected into estuaries along the Oregon and Washington coasts from the coastal ocean during the dry season (Roegner and Shanks 2001; Roegner et al. 2002). Chlorophyll *a* is imported into the Yaquina Estuary from the coastal ocean as evident by high chlorophyll *a* levels at high salinities (Figure 5.7.7). Peak chlorophyll *a* concentrations imported from the coastal ocean into the estuary during the dry season reach 50 µg L^{-1} and median values are about 4 µg L^{-1} (n = 181). The oceanic signal attenuates more rapidly for chlorophyll *a* compared to NO_3^- and PO_4^{-3} with a statistically significant relationship between oceanic chlorophyll *a* concentrations and within estuary chlorophyll *a* only evident about 11 km into the estuary (Brown and Ozretich 2009). The more rapid decline in the ocean signal in chlorophyll *a* is probably the result of benthic grazing on oceanic phytoplankton. Oyster aquaculture is present in Yaquina Estuary in the region 10 to 15 km from the mouth, and in the lower estuary tidal flats exist that have high densities of filter-feeding burrowing shrimp (DeWitt et al. 2004; Griffen et al. 2004).

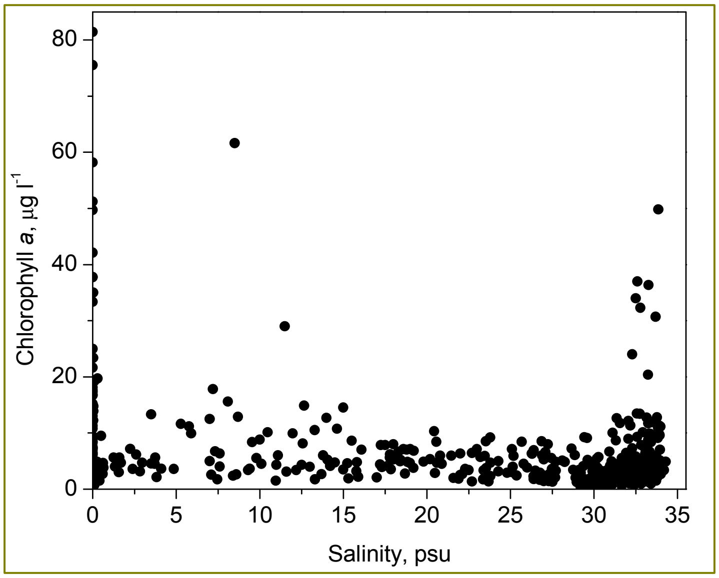


Figure 5.7.7 Dry season chlorophyll a versus salinity (all stations from 1974 to 2006) showing high chlorophyll a at high salinities, demonstrating the oceanic import of chlorophyll a from the coastal ocean into the Yaquina Estuary. The plot also shows the high chlorophyll a in the tidal fresh portion of the estuary.

The estuary shows seasonal differences in water column chlorophyll *a*. The median wet season chlorophyll *a* is 1.6 µg L^{-1} (n = 293), while during the dry season, the median increases to 4.9 µg L^{-1} (n = 1205). Peak chlorophyll *a* levels occur from June to August (Figure 5.7.8).

Limited water column primary productivity data exists for Yaquina Estuary. Primary production (at a station 14 km from the mouth of the estuary) during the dry season ranges from 0.25 to 2.8 g C m^{-2} d^{-1} with mean of 0.9 g C m^{-2} d^{-1} (Johnson 1981).

Phytoplankton Blooms and Species Composition

The import of chlorophyll *a* to zone 1 is consistent with the findings of Karentz and McIntire (1977) that in the spring to fall, marine genera of diatoms dominate in the lower estuary (stations 3.4 and 6.7 km from the mouth of the estuary), while freshwater and brackish taxa dominate in the upper estuary (stations 12.3 and 18.8 km from the mouth). In the tidal fresh portion of the estuary, recurrent algal blooms occur during June and July, with chlorophyll *a* concentrations reaching 80 µg L^{-1}. During the late spring, non-toxic red tide blooms of *Myrionecta rubra* recur in the vicinity of Toledo. During the dry season, chlorophyll *a* concentrations occasionally reach 20 µg L^{-1} in the vicinity of Toledo (8 percent of recent observations).

Benthic Primary Producers

In Yaquina Estuary, two species of seagrasses occur—the native eelgrass *Zostera marina* and the non-indigenous dwarf eelgrass *Z. japonica*. *Z. marina* is the dominant marine angiosperm encompassing approximately 1 km^2 in Yaquina Estuary (Brown et al. 2007). The spatial extent of

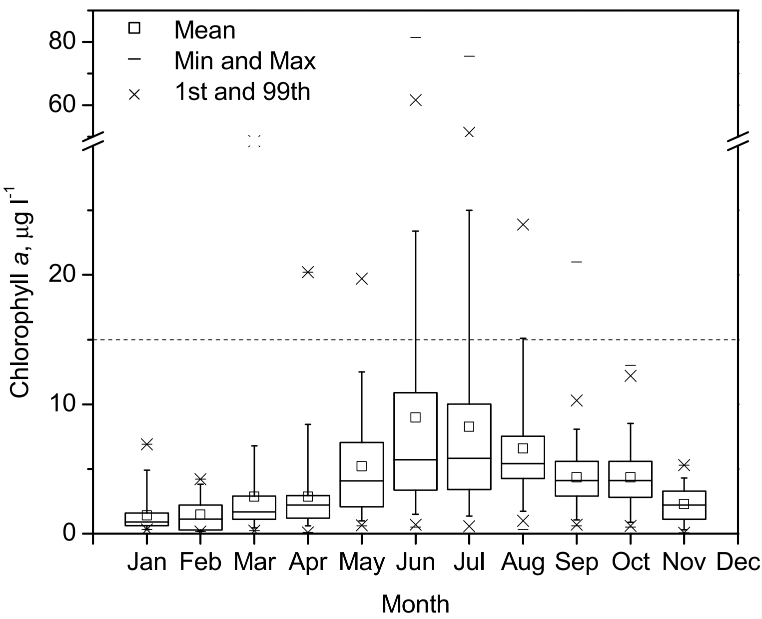


Figure 5.7.8 Box plot of monthly chlorophyll a data from the Yaquina Estuary (all stations, 1973–2006). The dashed line indicates the Oregon estuarine chlorophyll a criterion. The boxes represent the 25th and 75th percentiles, the whiskers represent the 5th and 95th percentiles, and the horizontal line is the median.

Z. japonica is limited, with an areal distribution of ~1,600 m^{-2} (Kaldy 2006b); however, anecdotal observations suggest that this might be increasing.

Macroalgal blooms (*Ulva* and *Entermorpha* spp.) occur during the dry season with peak biomasses typically occurring in September on the tidal flats near the mouth of the estuary. During those blooms, mean macroalgal biomass reach 200 gdw m^{-2}. Natural abundance stable isotope data ($\delta^{15}N$) combined with modeling suggests that the N source for the blooms is primarily from oceanic upwelling (Lee and Brown 2009).

In the Yaquina Estuary, benthic primary producers dominate the total estuary primary productivity. Primary productivity of benthic microalgae in the lower portion of the estuary (zone 1) ranges from 125 to 325 g C m^{-2} y^{-1} (depending on the location and elevation; Riznyk and Phinney 1972). Davis (1981) measured net primary production during the dry season in the lower portion of the estuary of 46 g C m^{-2} d^{-1} and 0.26 g C m^{-2} d^{-1} for green macroalgae and benthic microalgae, respectively. Net primary production for *Z. marina* and *Z. japonica* in the lower portion of the estuary was 181 and 130 g C m^{-2} y^{-1}, respectively (Kaldy 2006a, 2006b).

Long-Term Changes

Although limited historical water quality data exist, trend analysis reveals that there have not been any major changes in water column nutrients or chlorophyll *a* in the estuary (Brown et al. 2007). In addition, comparisons of recent and historic data suggest that there have been no changes in frequency or intensity of macroalgal blooms or in spatial distribution of *Z. marina* during the past 20–30 years (Brown et al. 2007).

From 1960 to 1984, a significant trend of increasing DO is apparent in zone 2 during both the dry and wet seasons, suggesting that water quality has improved. Recent (2002–2006) DO levels in zone 2 are similar to DO levels during the mid-1980s, suggesting no recent changes in DO levels. In contrast, no significant trends in dry or wet season DO in zone 1 are apparent, suggesting that the trend in historic DO levels in zone 2 was not a result of differences in ocean conditions.

A report by the Federal Water Pollution Control Administration (1966) states that the water quality in the lower portion of the Yaquina basin was "adversely affected by existing and man-made conditions," including "inadequately treated wastes from municipalities and industries," that placed "an excessive demand on oxygen resources of Yaquina Bay during annual periods of low streamflow." In 1956, Toledo upgraded its wastewater treatment facility to primary treatment, and in 1981 it upgraded it to secondary treatment. During the early 1900s until the 1980s, the estuaries and streams of the Pacific Northwest were used for the transport and storage of logs (Sedell and Duval 1985). A decline of log rafting in Yaquina occurred from 1962 through the 1980s (Sedell and Duval 1985). One effect of bark debris associated with log rafts is increased BOD (Sedell and Duval 1985). Because of the multiple stressors on the Yaquina Estuary during that period, there is no way to determine the cause of the observed trend in DO levels in zone 2.

Summary of Ecosystem Impacts

The Yaquina Estuary does not appear to be suffering from symptoms of anthropogenic eutrophication. Nutrient budgets combined with $\delta^{15}N$ of green macroalgae reveal that most of the N and P loading is associated with natural sources (ocean and red alder) and that the nutrient loading from natural sources is high. The close coupling between the coastal ocean and Yaquina Estuary strongly influences nutrient, chlorophyll *a* and DO conditions within the estuary. Nutrient standards developed for Yaquina Estuary should incorporate natural variability associated with ocean conditions and longer-term changes in ocean conditions (e.g., El Niño, La Niña, Pacific Decadal Oscillations, climate change). The relatively high natural nutrient loading and the import of hypoxic ocean water could result in the Yaquina Estuary being susceptible to future changes in anthropogenic nutrient loading (Figure 5.7.9). Historically lower DO levels suggests zone 2 might be susceptible to anthropogenic activities. For example, during minimal riverine inflow (August to September) wastewater treatment facility effluent increases in importance, particularly in zone 2. It is estimated that during those low-flow periods, the wastewater treatment facilities might contribute 30–60 percent of the DIN in the vicinity of Toledo (Brown et al. 2007). During the dry season, chlorophyll *a* values reach 20 µg L^{-1} and HABs have been observed in this region. Additional research is recommended to determine if physical flushing and benthic grazing is strong enough to ameliorate potential eutrophication symptoms.

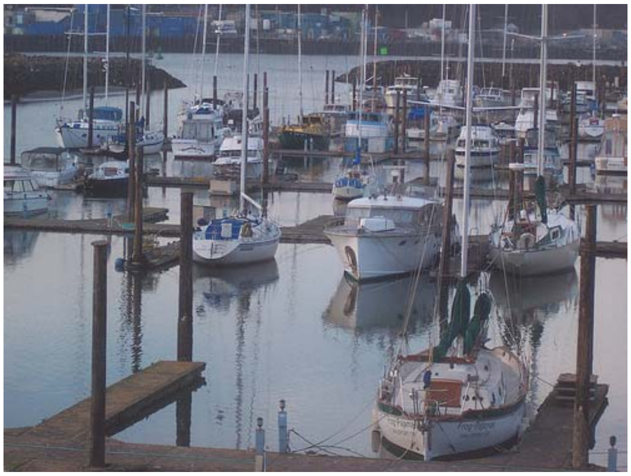


Figure 5.7.9. While the overall eutrophic condition of Yaquina Bay appears to be good, it has a moderately high susceptibility for increased eutrophication in the future (NOAA 2007; Brown et al. 2007). Photo www.flickr.com/photos/koocheekoo/.

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Chapter 5:

8. Barnegat Bay–Little Egg Harbor

Michael J. Kennish

Figure 5.8.1. The Barnegat Bay–Little Egg Harbor Estuary.

General Description

The Barnegat Bay–Little Egg Harbor Estuary and the shallow inland bays from Tuckerton to Cape May compose the New Jersey coastal bay system. Extending for 70 km from Bay Head in the north, to Little Egg Inlet in the south, the Barnegat Bay–Little Egg Harbor Estuary (39°31'N, 74°02'W; 40°06'N, 74°20'W) is the most intensely studied waterbody in this coastal bay system (Figure 5.8.1). It ranges from ~2 to 6 km in width and has an average depth of ~1.5 m (range = < 1 to 7 m). With a surface area of ~280 km^2, the estuary forms an irregular tidal basin separated from the Atlantic Ocean by a narrow barrier island complex (i.e., Island Beach and Long Beach Island) that is breached at Barnegat Inlet, ~35 km south of Bay Head, and at Little Egg Inlet in the southern perimeter. Barnegat Bay has a volume of ~2.38 x 10^8 m^3, and Little Egg Harbor, a volume of ~1.16 x 10^8 m^3 (Kennish 2001a). The watershed covers an area of ~1,730 km^2. *Spartina alterniflora* marshes cover extensive areas along the estuarine shoreline, and these marshes reach their greatest expanse at Little Egg Harbor and the bays to the south.

Winds, tides, salinity gradients, bathymetry and basin morphology contribute to complex circulation patterns in the estuary. Subtidal motion in the lagoonal system is primarily forced by coastal sea level (Chant 2001). Because of the extreme enclosure, shallowness of the system and relatively low freshwater inflow, the winds and tidal currents (< 0.5 to 1.5 m s^{-1}) largely control hydrological conditions. Southerly winds (south-southwest) predominate in the summer and westerly winds (west-northwest) in the fall, winter and spring at a velocity of < 15 km h^{-1}. The tidal range is < 0.5 to 1.5 m and, because of the morphological structure and frictional effects of the estuarine floor, the tide rises faster than it falls. The water column is vertically well mixed; however, there is a tendency for weak stratification in the deepest waters of the Intracoastal Waterway. The flushing time exceeds 70 days in summer (Kennish 2001a).

The inland coastal bays have a more discontinuous distribution than the estuarine waters to the north, being bounded by drumstick barrier islands. They communicate with the open ocean through a series of inlets (i.e., Brigantine, Absecon, Great Egg Harbor, Corson's, Townsend's, Hereford, and Cape May inlets). Although the total surface area of the inland bays (~278 km^2) is nearly the same as that of the Barnegat Bay–Little Egg Harbor Estuary, the watershed area (3,431 km^2) is much greater. In the coastal watersheds to the south, the resident population is nearly 40 percent less than in the Barnegat Bay watershed. The depth of the inland bays averages 1.1 m, and the tidal ranges are typically < 1 m. The flushing times are ~25–30 days. Temperatures and salinities of the New Jersey coastal bays generally range from –1.5 to 30 °C and 10 ‰–32 ‰ (Figure 5.8.2), respectively. The physicochemical conditions of the inland coastal bays are similar to those of the Barnegat Bay–Little Egg Harbor Estuary to the north (Table 5.8.1). However, fewer investigations have been conducted on biotic communities in these southern systems, which make comparisons difficult.

Dissolved Oxygen

Hypoxia is not commonly observed in the coastal bays because of a well-mixed water column generated by tidal currents and winds. Mean DO levels in the estuary generally exceed 6 mg L^{-1}, with the measurements being relatively consistent throughout the system (Figure 5.8.3, Table 5.8.1). When low DO levels are observed, they usually occur in embayments and man-made lagoons with restricted circulation or near tidal creeks with areas of stagnant conditions (e.g., Tuckerton Creek).

Turbidity and Light

Winds and tidal currents cause resuspension of bottom sediments and increased turbidity, with values ranging

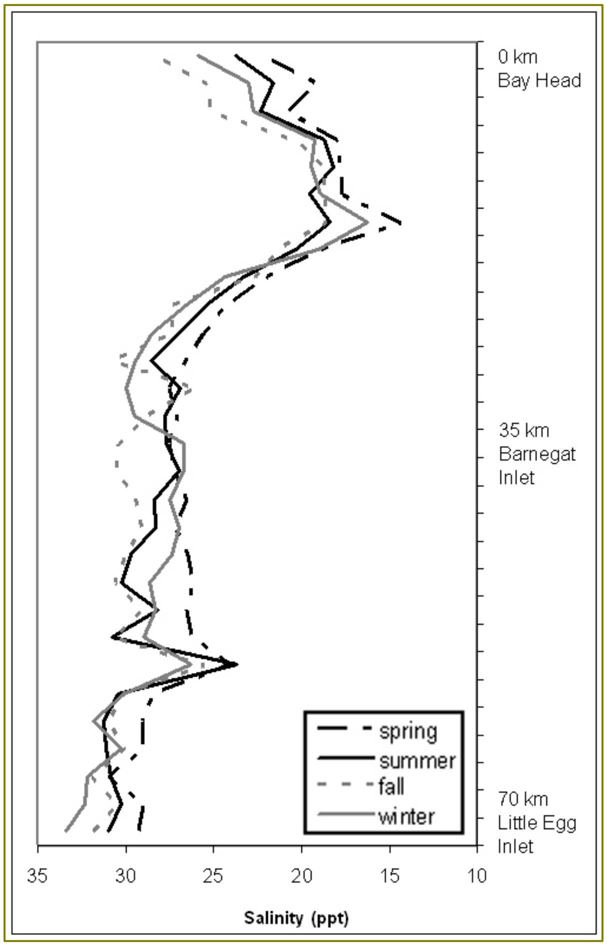


Figure 5.8.2. Seasonal salinities in the Barnegat Bay–Little Egg Harbor Estuary.

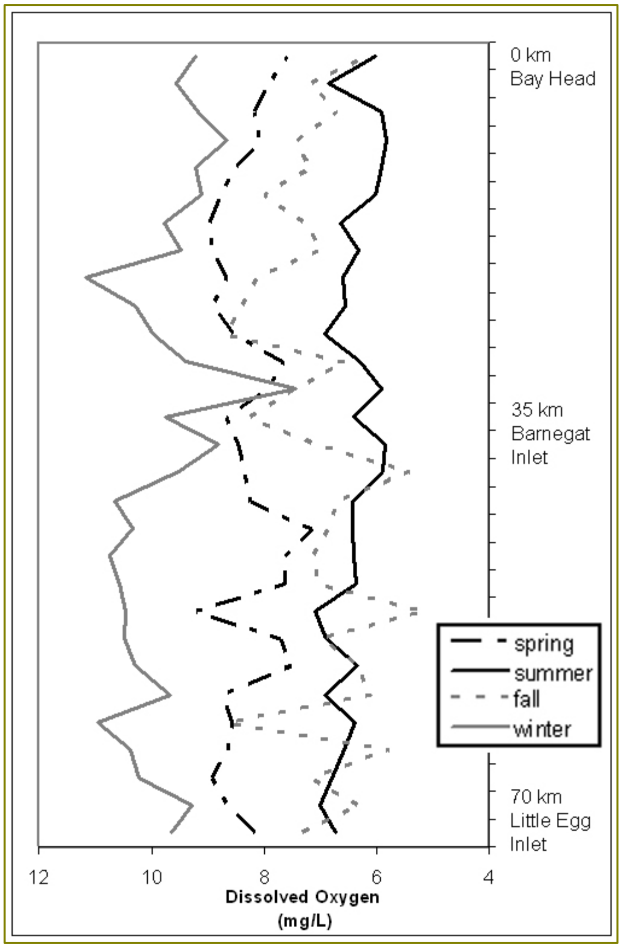


Figure 5.8.3. DO concentrations in the Barnegat Bay –Little Egg Harbor Estuary.

from ~1.0 to 2.5 nephelometric turbidity units (NTUs). Heavy boat use in summer exacerbates those conditions. Escalating nutrient inputs in coastal bays are also correlated with increased turbidity caused by phytoplankton blooms, as well as accelerated epiphytic and macroalgal growth, which reduce underwater light availability (Kennish 2001b; Brush and Nixon 2002; Lamote and Dunton 2006). Secchi disk measurements have been compiled to estimate light penetration in the estuary (Figure 5.8.4). Mean values generally range from 2 to 4 m in the system, and unlimited visibility occurs most frequently from late fall to early spring. However, the estuary is relatively turbid during the summer growing season when the Secchi depth is often less than 1 m (Lathrop et al. 2006).

Nutrients

Increased nutrient loading in coastal watersheds and airsheds of New Jersey associated with accelerating population growth and development over the past several decades has been linked to an array of adverse environmental impacts in coastal bay waters, such as rapid growth of micro- and macroalgae, HABs, increased pathogens, altered benthic communities, affected harvestable fisheries and loss of essential habitat (e.g., shellfish and seagrass beds). The Barnegat Bay–Little Egg Harbor Estuary is particularly susceptible to nutrient loading problems because it is shallow, poorly flushed and bordered by highly developed watershed areas (~525 km^2). Most of the N load (~50 percent) derives from surface

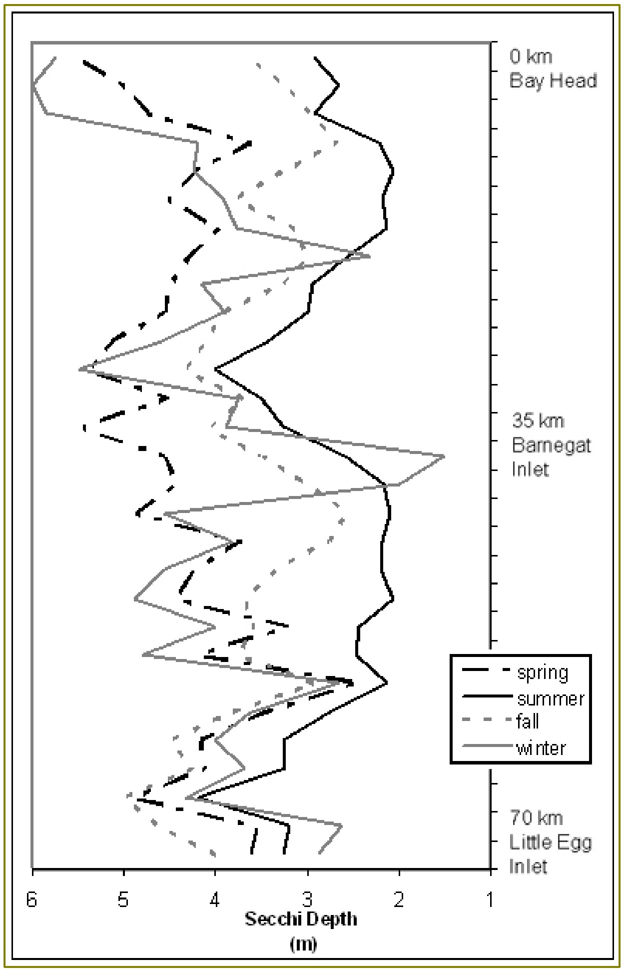


Figure 5.8.4. Secchi depth measurements in the Barnegat Bay–Little Egg Harbor Estuary.

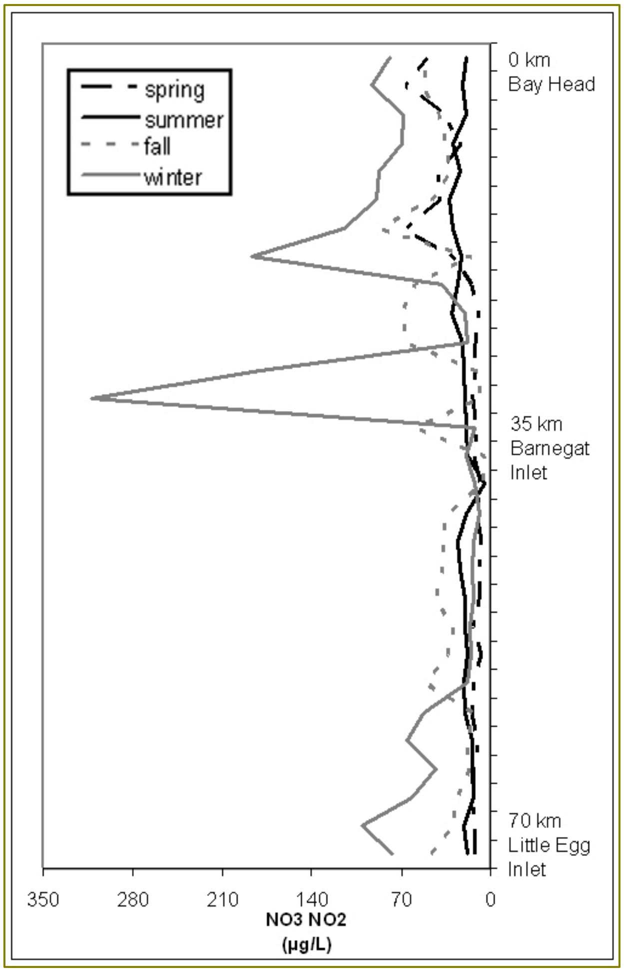


Figure 5.8.5. NO$_3^-$ + NO$_2^-$ levels in the Barnegat Bay–Little Egg Harbor Estuary.

Table 5.8.1. Selected water quality measurements for Barnegat Bay–Little Egg Harbor, New Jersey, from sampling over the past several years.

Variable	Mean	Min	Max	n
Salinity	30.6	17.6	36.3	210
DO (mg L^{-1})	6.2	2.8	9.9	213
$NO_3^- + NO_2^-$ (μM-N)	1.81	0.14	20.7	217
NH_4^+ (μM-N)	4.8	0.15	35.8	216
PO_4^{-3} (μM-P)	1.19	0.03	4.27	217

water inflow, but substantial fractions also originate from atmospheric deposition (~39 percent), and direct groundwater discharges (~11 percent) (Hunchak-Kariouk and Nicholson 2001). As a result, the estuary is classified as a highly eutrophic system threatened by ongoing environmental degradation (Kennish et al. 2007). The total annual N load to the Barnegat Bay–Little Egg Harbor Estuary amounts to ~1.2 million kg yr^{-1} compared to ~1.9 million kg yr^{-1} for the inland coastal bays. The TN loading rate calculated for Barnegat Bay is 340 mmol N m^{-2} yr^{-1}, and the DIN loading rate is 240 mmol N m^{-2} yr^{-1} (Seitzinger et al. 2001).

TN concentrations in the Barnegat Bay–Little Egg Harbor Estuary typically range from ~20 to 80 μM-N. Organic N levels in the estuary are about 10 times higher than the DIN concentrations, and they might play an important role in the genesis of brown tide blooms. The mean concentrations of NO_3^- + NO_2^- are < 4 μM-N (=< 56 μg L^{-1}) during phytoplankton and benthic macroalgal blooms, with highest levels observed in winter (Seitzinger et al. 2001; Figure 5.8.5). Highest concentrations occur in the upper estuary north of Barnegat Inlet (Figure 5.8.5). Mean NH_4^+ concentrations amount to < 2.5 μM-N, and highest levels occur in summer. PO_4^{-3} exhibits similar patterns as those of NH_4^+. The mean concentrations of PO_4^{-3} are less than those of the DIN forms. Highest levels repeatedly develop in summer, a seasonal peak documented in many other mid-Atlantic estuaries.

Chlorophyll and Primary Production

Phytoplankton production and mean chlorophyll *a* values in the Barnegat Bay–Little Egg Harbor Estuary are ~480 gC m^{-2} $yr^{-1}$and ~10 μg L^{-1}, respectively. The highest phytoplankton production and chlorophyll *a* values, as well as turbidity, occur in the northern estuary during the summer because of greater nutrient inputs from the more developed areas of the watershed. High phytoplankton cell counts in summer, particularly associated with blooms of dinoflagellates, microflagellates and pelagophytes, cause shading effects that can be detrimental to SAV beds.

Phytoplankton Blooms and Species Composition

A total of 242 phytoplankton species have been identified in the estuary. A detailed taxonomic investigation indicates that dinoflagellates comprise the largest number of species (~41 percent of the total), followed by diatoms (31 percent) and phytoflagellates (24 percent) (Olsen and Mahoney 2001). The most seasonally abundant forms belong to the phytoflagellate group (Cryptophyceae, Euglenophyceae, Prasinophyceae, Chlorophyceae, Chrysophyceae, Haptophyceae and Raphidophyceae). Nanoplankton (5 to 15 μm) and picoplankton (1 to 5 μm) are the numerically dominant forms. Coccoid algae in the picoplankton size range, notably *Nannochloris atomus* (Chlorophyceae) and *Aureococcus anophagefferens* (Pelagophyceae), can attain extremely high numbers (> 10^6 cells mL^{-1}) in the late spring and summer months. *N. atomus* appears to be the most widespread species, both spatially and temporally. During blooms, the brown tide species *A. anophagefferens* discolors the water a yellowish-brown hue and can exceed *N. atomus* in total numbers. Brown tides have been observed in the estuary since only 1995, with intense blooms being recorded in 1997 and 1999–2002. The blooms have been widespread and prolonged in the estuary. The years of significant brown tide blooms in the estuary were characterized by the occurrence of extended drought conditions, corresponding low freshwater inputs and elevated bay salinity. The peak numbers of *A. anophagefferens* declined from 2000 (2.155 x 10^6 cells mL^{-1}) and 2001 (1.883 x 10^6 cells mL^{-1}) to 2002 (1.561 x 10^6 cells mL^{-1}) during wetter flow years (Olsen and Mahoney 2001).

Chlorophyll *a* levels recorded during summer phytoplankton blooms range from ~10 to 30 µg L^{-1}, but the levels are generally < 10 µg L^{-1} during non-bloom conditions (Figure 5.8.6). Diatom blooms also occur in the estuary during late winter and early spring (March). *Skeletonema costatum*, *Thalassiosira* spp., and *Nitzschia* spp. are important components of such events. The HAB dinoflagellate *Prorocentrum minimum* has consistently been reported in the estuary in recent years, although localized blooms of *Gonyaulax spinifera* have been reported in the past. Other HAB species found in the estuary, but not at bloom concentrations, are *Katodinium rotundatum*, *Scripsiella trochoidea* (= *Perdinium trochoideum*), *Protoperidinium brevipes*, and the raphidophycean *Heterosigma carterae*.

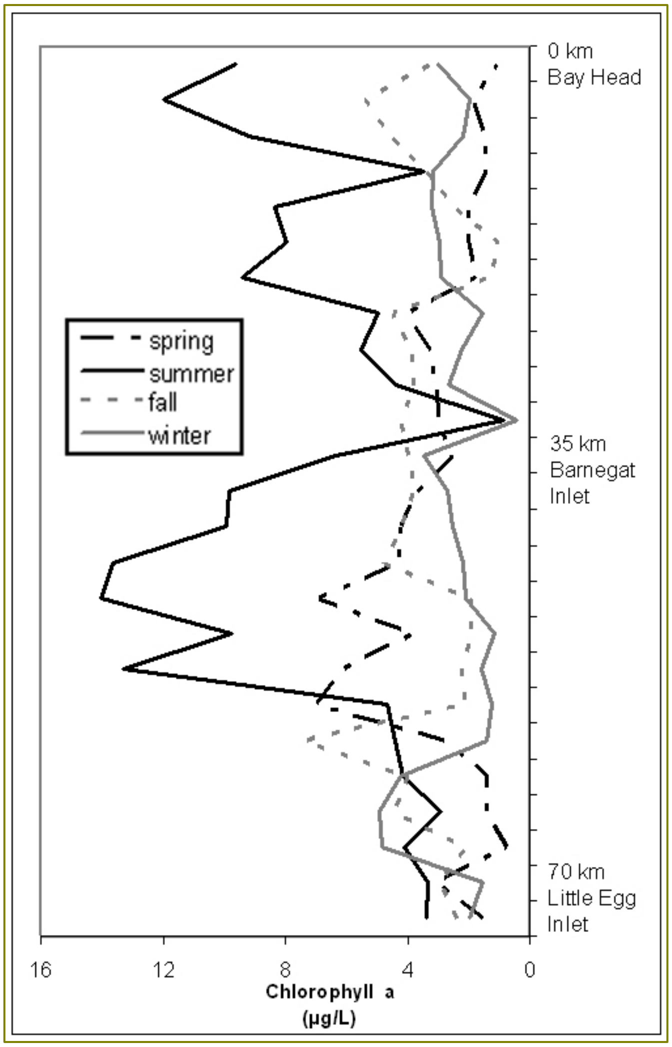
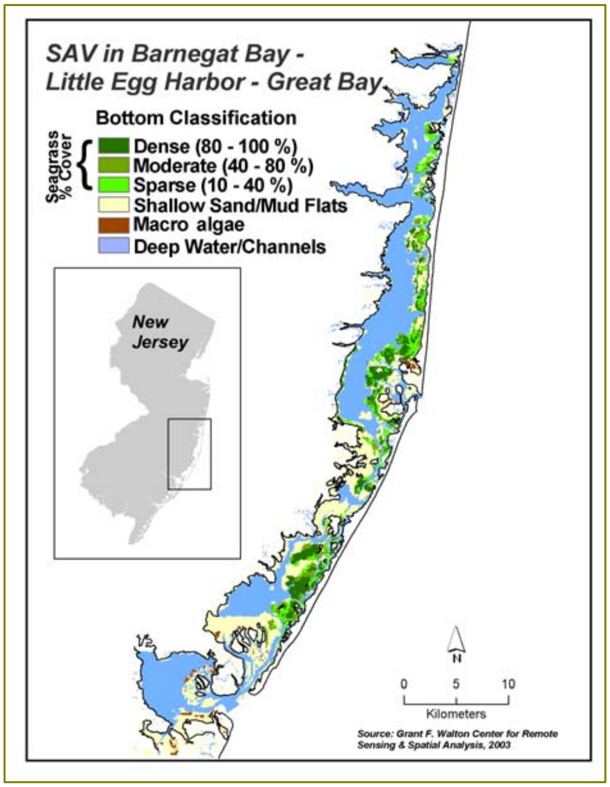


Figure 5.8.6. Chlorophyll a levels in the Barnegat Bay–Little Egg Harbor Estuary.

Benthic Primary Producers

N loading has also contributed to benthic macroalgal blooms and accelerated growth of benthic microalgae. Macroalgal overburden can affect SAV by smothering the beds or altering the sediment geochemistry, promoting hypoxic conditions in bottom sediments. Sheet-like masses of drifting algae (e.g., *Ulva lactuca* and *Enteromorpha* spp.) are especially problematic because they grow rapidly when light and nutrient conditions are favorable, and their high biomasses can seriously damage SAV habitat and associated benthic faunal communities within one growing season. Sea lettuce (*Ulva lactuca*) appears to be the most serious problem. *Ceramium* spp., *Champia parvula* and *Spyridia filamentosa* are also relatively abundant forms. The loss of *Z. marina* habitat in Little Egg Harbor has been noted as a consequence of macroalgal (e.g., *Ulva, Codium*, *Gracilaria*) loading effects. Macroalgal biomass generally ranges from 1 to 30 g dry weight m^{-2} during the July–October period but exceeds 400 g dry weight m^{-2} during bloom events when the complete elimination of the aboveground biomass of *Z. marina* occurs in some affected areas of Little Egg Harbor. Rapid growth of algal epiphytes on SAV leaves has exacerbated the effects.

Benthic microalgal (gross) photosynthesis typically ranges from 75 to 3,300 μM C m^{-2} h^{-1}, with the highest rates recorded in sandy sediments in early summer (Seitzinger et al. 2001). Light intensity is a major factor controlling benthic microalgal production in the estuary. Higher phytoplankton abundance and sediment suspension cause shading problems that reduce benthic microalgal production. Peak production values are recorded when bottom light exceeds 50 μE m^{-2} s^{-1} during the June to October period (Seitzinger et al. 2001).



Source: Grant F. Walton Center for Remote Sensing & Spatial Analysis, 2003

Figure 5.8.7. SAV beds in the Barnegat Bay–Little Egg Harbor Estuary.

SAV beds form a major benthic habitat in the Barnegat Bay–Little Egg Harbor Estuary, composing ~75 percent (> 6,000 ha) of New Jersey's SAV (Figure 5.8.7). Eelgrass (*Z. marina*) is by far the dominant species, mainly inhabiting sandy sediments along the shallow eastern part of the estuary. The density of Z. *marina* typically ranges from 300 to 1,200 shoots m^{-2}, and the biomass, from 50 to 200 g dry weight m^{-2}. Widgeon grass (*Ruppia maritima*) also occurs in the beds but at much lower abundance. North of Toms River, sago pondweed (*Potamogeton pectinatus*) and horned pondweed (*Zannichellia palustris*) have been found in lower salinity waters.

Long-Term Changes

During the past 30 years, a significant decline in SAV abundance appears to have occurred in the coastal bays, resulting in the reduction of essential fish habitat and the potential loss of commercially and recreationally important species. As much as 25 percent of the SAV in Barnegat Bay and 60 percent of the SAV in Little Egg Harbor might have been lost since the mid-1970s, concomitant with increased nutrient loading in the system. In 2006, the biomass of SAV beds in Barnegat Bay decreased by ~50 percent and in Little Egg Harbor by 88 percent compared to that of the 2004–2005 period (Kennish et al. 2007). Heavy epiphytic growth on seagrass shoots and blades has also been linked to ongoing nutrient enrichment, a condition observed in other mid-Atlantic estuaries (Moore and Wetzel 2000).

Shellfish resources have also decreased substantially in the coastal bays from fishing pressure and development and eutrophication impacts. The loss of the bay scallop (*Argopecten irradians*) fishery during the 1950s and 1960s, the limited abundance of the soft clam (*Mya arenaria*) and the rapidly diminishing stock of the hard clam (*Mercenaria mercenaria*) since the mid-1980s have severely curtailed shellfishing activity in the bays. Two-thirds of the hard clam standing stock in Little Egg Harbor was lost between 1986 and 2001, as revealed by detailed field surveys.

Summary of Ecosystem Impacts

The New Jersey coastal bays are subject to multiple anthropogenic impacts from an expanding population in adjoining coastal watersheds. Eutrophication poses the most serious threat to the long-term health and function of the bays, affecting essential habitats (e.g., seagrass and shellfish beds) and finfish nursery areas. Nutrient and organic carbon loading in the shallow, lagoon-type estuaries has been linked to an array of cascading environmental problems such as increased micro- and macroalgal growth, HABs, bacterial and viral pathogens, high turbidity, altered benthic invertebrate communities, and affected harvestable fisheries. The Barnegat Bay–Little Egg Harbor Estuary is classified as a highly eutrophic estuary after applying the National Estuarine Eutrophication Assessment model (Bricker et al. 2007). Because it is shallow, poorly flushed, and bordered by highly developed watershed areas, the estuary is particularly susceptible to nutrient loading. Other adverse effects on the bays include nonpoint source inputs of pathogens and chemical contaminants, as well as the physical alteration of habitat from bulkheading, diking and ditching, dredging, and lagoon construction. Point source effects of power plants (i.e., biocidal releases, thermal discharges, impingement, and entrainment) increase mortality of estuarine and marine organisms in Barnegat Bay. Human activities in watershed areas, notably deforestation and infrastructure development, partition and disrupt habitats while also degrading water quality and altering biotic communities. Ongoing land development raises turbidity and siltation levels in tributaries of the estuary, creating benthic shading problems. Management actions, including the purchase of open space, improved stormwater controls, and smart development, are underway to remediate some of the aforementioned effects and restore vital estuary functions.

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Chapter 5:

9. Coastal Bays

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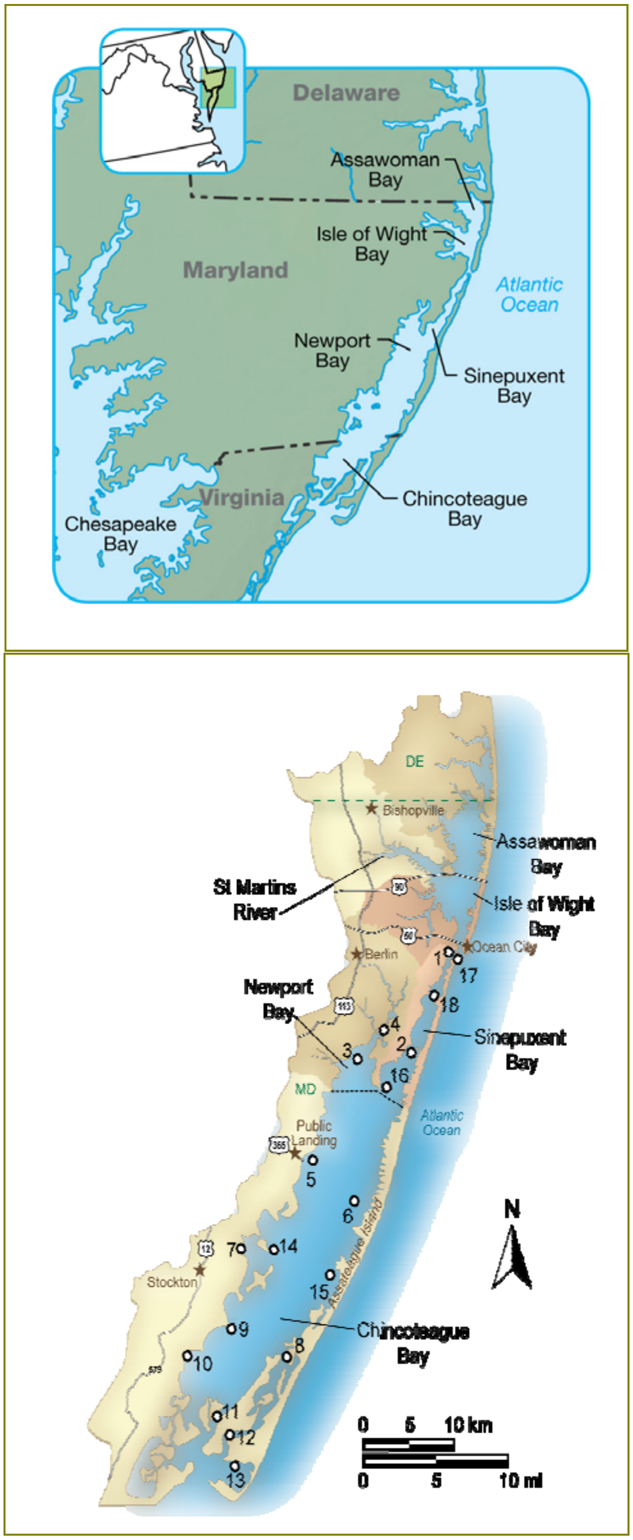

Figure 5.9.1. Two maps of the Coastal Bays. The top map shows the broader location. The bottom map shows stations that have been maintained for monthly water quality sampling for at least a decade.

General Description

The Coastal Bays of Maryland and Virginia, a network of multiple bays—including Assawoman, Isle of Wight Bay, Newport and Sinepuxent Bays, in Maryland, and Chincoteague Bay, in Maryland and Virginia (Figure 5.9.1)—are shallow lagoons at the interface of freshwater and saltwater behind the barrier island, Assateague (Boynton et al. 1996; Wazniak et al. 2004, 2007). These embayments are connected to the Atlantic Ocean by two inlets at the northern and southern ends of Assateague Island. Salinities range from near 0 at the headwaters of Trappe Creek that feeds Newport Bay, to > 32 in areas in Chincoteague Bay. The tidal range exceeds 1 m at the oceanic outlet, but in the northern bays it is < 0.3 m (Boynton et al. 1993). The bays have average depths ranging between 0.67 and 1.22 m and are poorly flushed (Boynton et al. 1996), with generally non-stratified waters. The flushing rate has been estimated to be on the order of 7 percent day^{-1} (Pritchard 1969), which approximates 10–20 days in the northern segments and > 60 days in Chincoteague Bay (Pritchard 1969; Lung 1994). Thus, contaminants and nutrients that enter the bays tend to stay in the bays.

The regional watershed is ~450 km^2 and has traditionally been dominated by farming and forestry. Land use in the

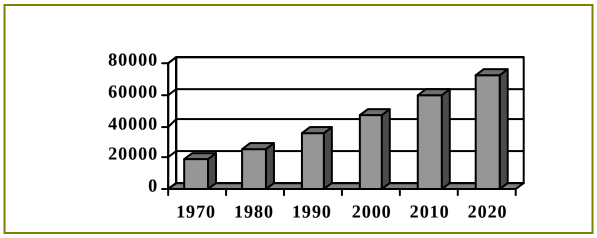

Figure 5.9.2. Change in human population in the watershed of Maryland Coastal Bays and the projected trend for the next 15 years.

relatively small watersheds is a mix of agriculture (including intensive poultry growing operations), forests, extensively ditched wetland systems, a National Park barrier island system, and rapidly increasing residential development (Boynton et al. 1996; Wazniak et al. 2007). These coastal lagoons have been classified as highly susceptible to eutrophication (Bricker et al. 1999, 2007; Wazniak et al. 2007) and are showing multiple signs of nutrient over-enrichment in recent years. A significant amount of loss of wetlands has also occurred through construction of canals and bulkhead (Wazniak et al. 2007). The recent growth in human population is of particular concern. The regional population doubled from 1980-2000 and is expected to double again by 2020 (Figure 5.9.2). This region is the fastest growing in the state of Maryland, and one of the fastest growing regions in the country (Crosset et al. 2004). Seasonal tourism has also swelled from nearly 8 million in the 1990s (Bohlen et al. 1997) to over 11 million today. The health of the environment is critical in the economic viability of the region (www.mdcoastalbays.org).

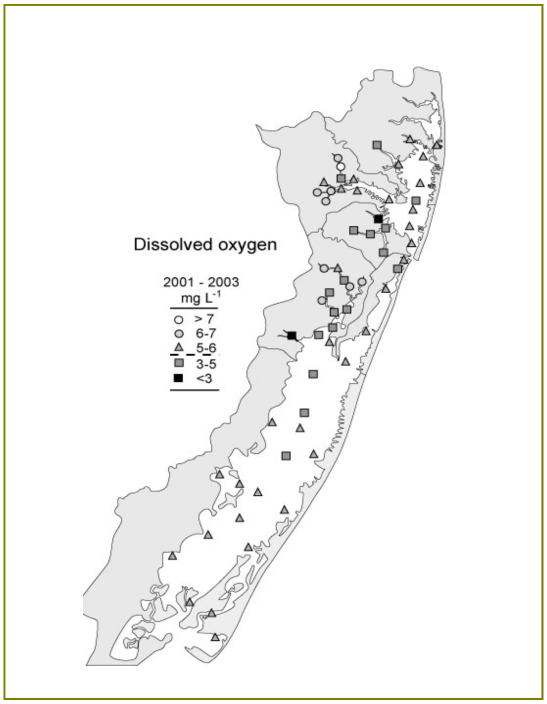

Figure 5.9.3. Status of mean DO for the sites indicated for 2001–2003, for June, July, and August only. From Wazniak et al. 2007.

In presenting the data that follow, many of the examples given are from station 5 at Public Landing, a site that has traditionally been considered pristine, but that has experienced multiple stresses, including outbreaks of HABs, in recent years.

Dissolved Oxygen

Both long-term monthly data collection at 18 stations (Wazniak et al. 2007), combined with deployments of *in situ* oxygen sensors in targeted northern bay stations, have permitted both long- and short-term scale analysis of DO. Over the period from 2001 to 2003, during which only daytime data are available, 40 percent of the stations did not meet the level of 5 mg L^{-1} (Wazniak et al. 2007). The stations where low DO was most common included the Coastal Bays tributaries, such as the St. Martin's River and the Isle of Wight tributaries, as well as most of Newport Bay (Figure 5.9.1., 5.9.3). *In situ* monitoring data reveal that values of DO oxygen fall well below 5 mg L^{-1} during the summer months even in Chincoteague Bay.

Turbidity and Light

Secchi depths average 0.5 to 1.0 m throughout the length of the estuary. Mean Secchi depths drop in midsummer, coincident with the development of algal biomass. Turbidity, however, is highly variable from wind and storm events, as also illustrated by continuous monitoring of turbidity at Public Landing (station 5) in 2006 (Figure 5.9.4). High turbidity develops during the summer months, coincident with the summer blooms, but turbidity

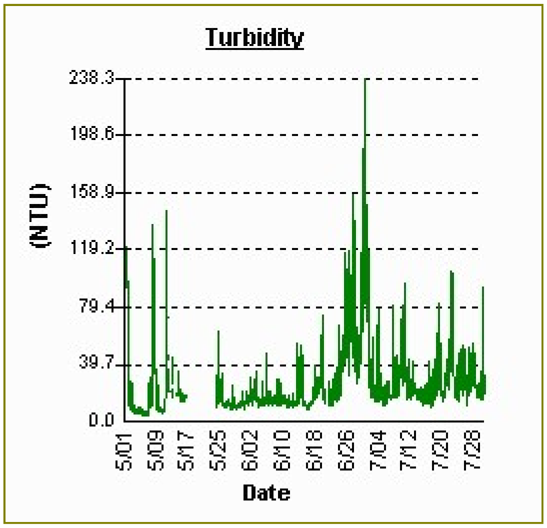

Figure 5.9.4. In situ *measurements of turbidity from summer 2006 in the Coastal Bays at Public Landing (station 5). From www.eyesonthebay.net*.

is highest in the early fall. In the northern bays, turbidity and water transparency might not be a good measure of water quality; however, because considerable inputs of tannins and humics lead to natural reductions in light penetration (Wazniak et al. 2007).

Nutrients

Nutrient loads to the Coastal Bays are dominated by nonpoint sources (e.g., surface runoff, groundwater, atmospheric deposition, shoreline erosion) (Boynton et al. 1993; Wells et al. 2004; Wazniak et al. 2007; Glibert et al. 2007), with estimates suggesting one-half to two-thirds of nutrients entering the bays coming from agricultural sources, the dominant land use in the area (Jacobs et al. 1993; Bohlen et al. 1997). The highest concentrations of TN and TP (dissolved and particulate) are in Assawoman Bay and in the small tributaries that discharge into Assawoman Bay and Newport Bay (Figure 5.9.1, Figure 5.9.5).

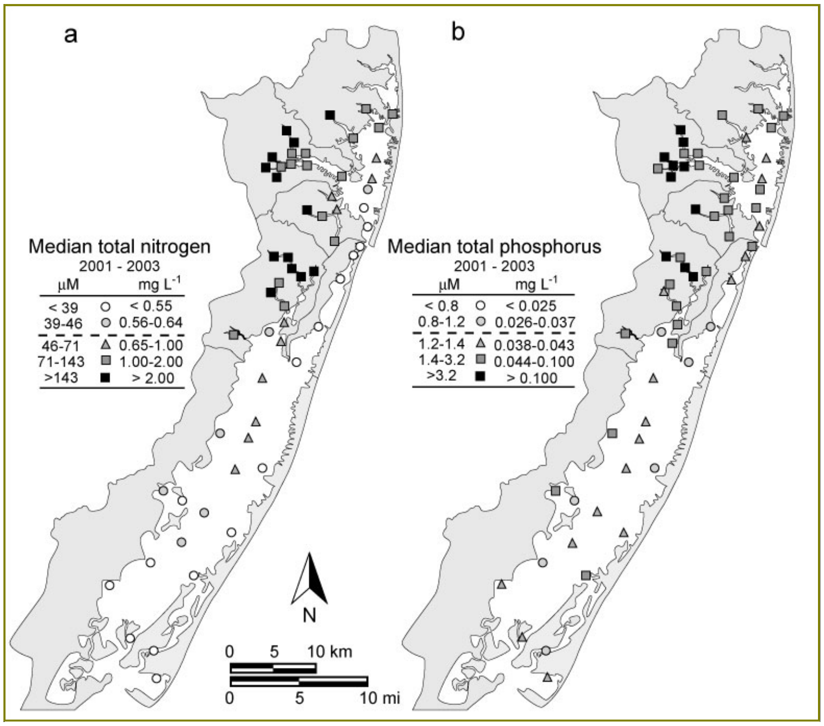

Figure 5.9.5. Median concentration of TN (a) and TP (b) for 2001–2003 for all stations indicated in the Coastal Bays. From Wazniak et al. 2007.

Dissolved inorganic nutrient concentrations are generally low relative to its larger neighboring estuaries, the Chesapeake and the Delaware. Although variable by site within the Coastal Bays, concentrations of $NO_3^- + NO_2^-$ and NH_4^+ typically remain <5 µM-N throughout the year in Chincoteague Bay, as shown for the Public Landing site, station 5 (Figure 5.9.6; Glibert et al. 2007). Organic forms of N help to compensate for the deficit in DIN. Concentrations of DON range from 10 to 30 µM-N and display a drawdown in midsummer, followed by a late summer increase (Figure 5.9.6; Glibert et al. 2007). Annual concentrations of PO_4^{-3} tend to remain low, < 1 µM-P, throughout the year, declining in the spring, but increasing again by midsummer (Figure 5.9.7). Concentrations of $Si(OH)_4$ range from ~20 µM-Si during spring to > 80 µM-Si during summer (not shown).

Inorganic nutrient ratios are indicative of inorganic N limitation throughout most of the year, with DIN:DIP well below Redfield proportions (16:1) except during spring (Figure 5.9.8). Similarly, the ratio of DIN:Si is well below

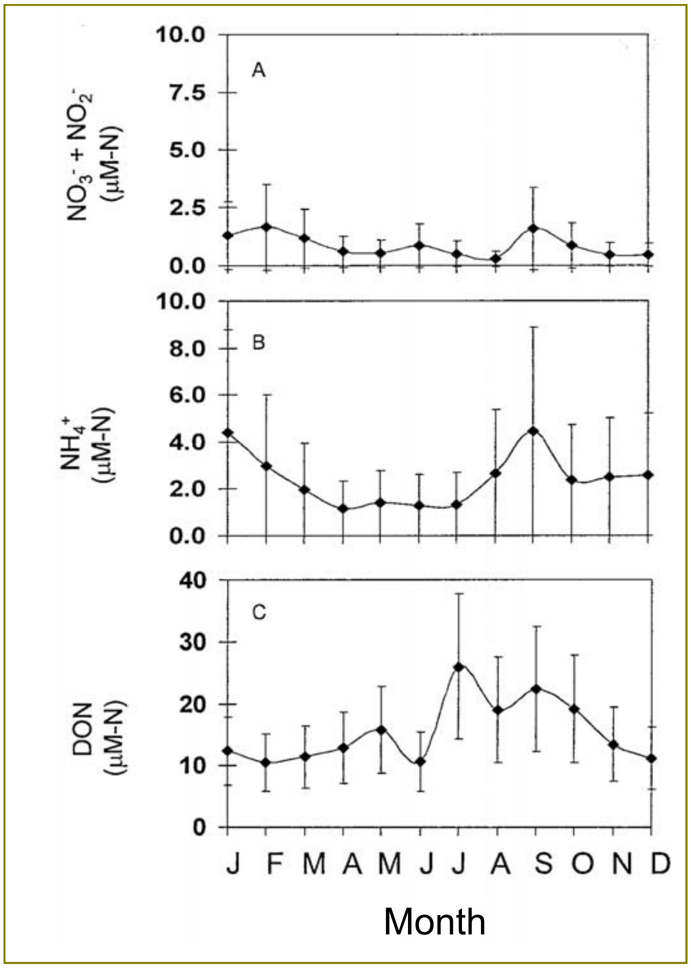

Figure 5.9.6. Mean annual concentrations of NO_3^- + NO_2^- (top) and NH_4^+ (middle) and DON (bottom) for station 5 in the Coastal Bays. From Glibert et al. 2007.

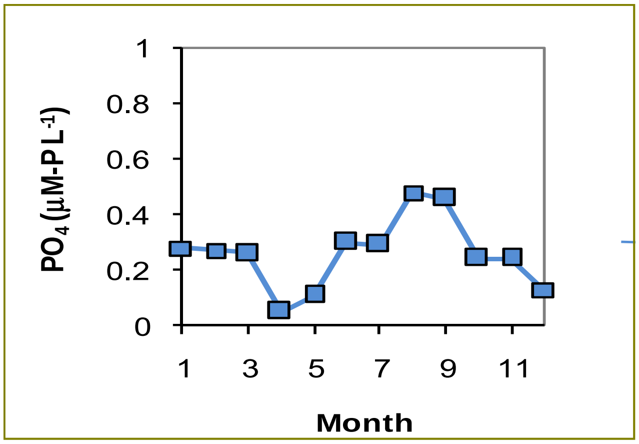

Figure 5.9.7. Mean annual concentrations of PO_4^{-3} for station 5 in the Coastal Bays based on averages of the years 1999-2004. From C. Wazniak unpublished data.

the Redfield proportion (1:1) for the entire year, also indicative of DIN limitation (Figure 5.9.8). However, when viewed as ratios using TN, instead of DIN, those ratios are considerably higher. Furthermore, the ratios are exclusive of the most recent years during which significant elevations in concentrations of NH_4^+ were observed. As a consequence, there could be shifts in limiting nutrients in years to come.

Chlorophyll and Primary Productivity

The annual maximum in chlorophyll *a* occurs in summer; there is no significant spring bloom. The average annual peak in chlorophyll *a* occurs in June, reaching concentrations ~20 µg L^{-1} (Figure 5.9.9), while specific sites, such as in Newport Bay can exceed > 100 µg L^{-1} on occasion. Average annual concentrations of chlorophyll *a* decline slightly in July and August but remain elevated through the summer before declining to < 5 µg L^{-1} for the remainder of the year (Figure 5.9.9). Thus, compared to many eutrophic estuaries, the concentrations of chlorophyll *a* in most of Chincoteague Bay remain fairly low, even during maximum biomass.

Rate measurements of phytoplankton productivity in the Coastal Bays show maximum rates in the summer and appear to be temperature dependent. Rates in Chincoteague Bay are much less than in some of the more degraded sub-embayments (not shown).

Phytoplankton Blooms and Species Composition

For the Coastal Bays overall, phytoflagellates, diatoms, and dinoflagellates dominate spring and summer seasons according to data from 2001 to 2003. The fall is strongly dominated by phytoflagel-

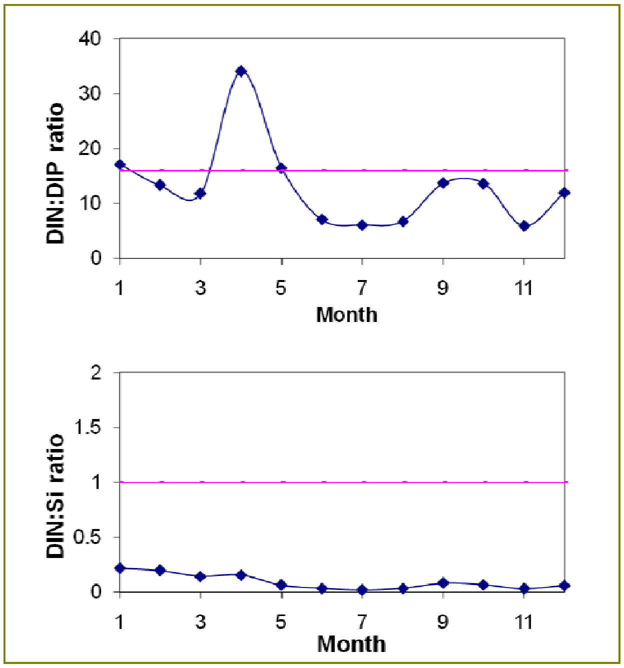

Figure 5.9.8. Annual variation in the ratio of DIN:DIP (top) and DIN:Si (bottom) for station 5 in the Coastal Bays, according to averages from 1999 to 2002. The horizontal line is the ratio normally taken as balanced for phytoplankton growth. From C. Wazniak, unpublished data.

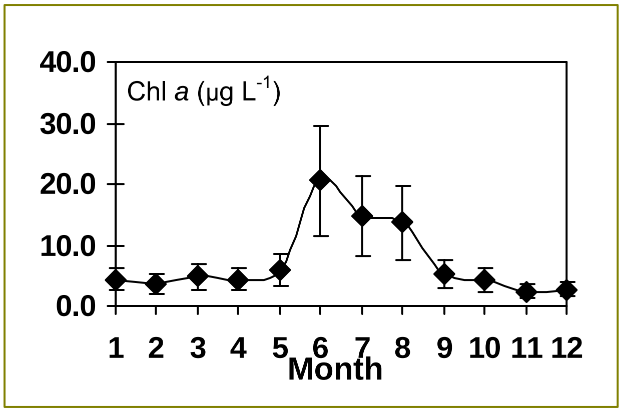

Figure 5.9.9. Mean concentration of chlorophyll a (µg L^{-1}) for June through September at station 5 in the Coastal Bays. From Glibert et al. 2007.

lates, with diatoms and cryptophytes also appearing at relatively high levels. Highest diversity is observed during winter when samples are dominated by phytoflagellates and diatoms (Tango et al. 2004).

Although phytoplankton blooms in Chincoteague Bay do not reach biomass levels normally indicative of eutrophic conditions, blooms are a major concern because a significant fraction of algal biomass can be composed of HAB species (Wazniak and Glibert 2004). In particular, the brown tide species *Aureococcus anophagefferens* has bloomed in the Coastal Bays every year for at least the past decade, the period over which such data are available (Trice et al. 2004; Gobler et al. 2005; Glibert et al. 2007). Blooms of brown tide annually exceed the threshold for blooms that are known to cause severe effects or mortality on shellfish and reduction in SAV (Gastrich and Wazniak 2002; Glibert et al. 2007).

Although brown tide is the predominant harmful species in the Coastal Bays, there are many other harmful algal species that are present in the embayments, including *Dinophysis* sp., *Heterosigma akashiwo, Chattonella* sp., *Karlodinium veneficum* (=*K. micrum*), *Pfiesteria* sp., *Pseudo-nitzschia* spp. and *Prorocentrum minimum* (Tango et al. 2004). The greatest number of species has been found in the more nutrient-polluted tributaries, such as the St. Martin's River and Newport Bay. Several of these species have been documented to be toxic and/or to have caused large scale fish kills over the past several years (Tango et al. 2004). The toxic cyanobacterium *Microcystis aeruginosa* is also present in the northern bays but has recently been declining in abundance relative to pre-2000 levels (Tango et al. 2004).

Benthic Primary Producers

SAV coverage is estimated to occupy 67 percent of the potential habitat in the Maryland portion of the bays with the greatest percentage of SAV habitat occupied in Sinepuxent (77 percent) and Chincoteague bays (76 percent) (Figure 5.9.10; Wazniak et al. 2004, 2007). Macroalgae are also abundant and well distributed throughout the Coastal Bays (Figure 5.9.11, McGinty et al. 2004). Microphytobenthos are also an important part of this system and, as typical of a shallow, lagoonal system, in some regions more chlorophyll *a* is in the benthic microalgae than in the phytoplankton.

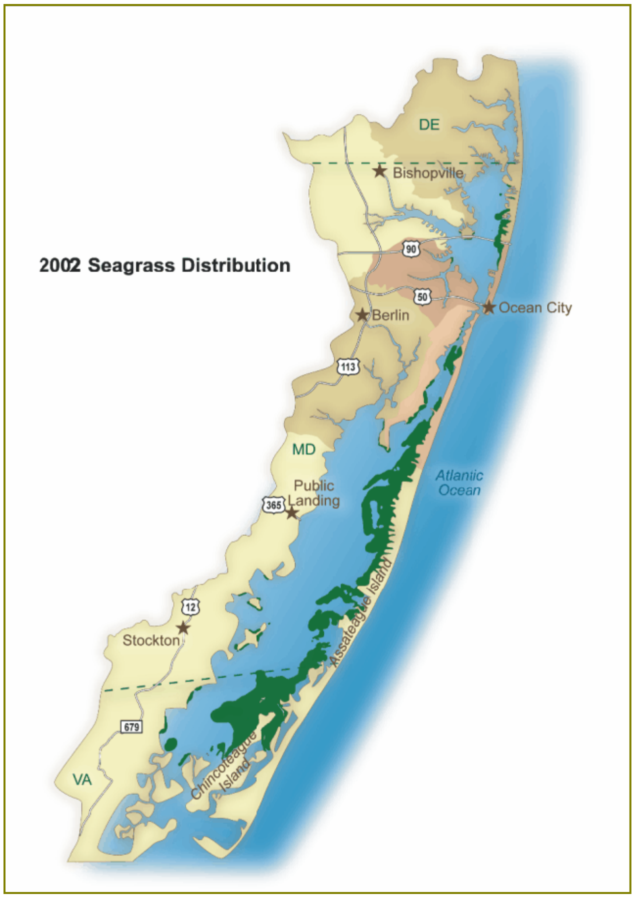

Figure 5.9.10. Total seagrass coverage in the Coastal Bays as determined by a 2002 Virginia Institute of Marine Science survey. From Wazniak et al. 2004.

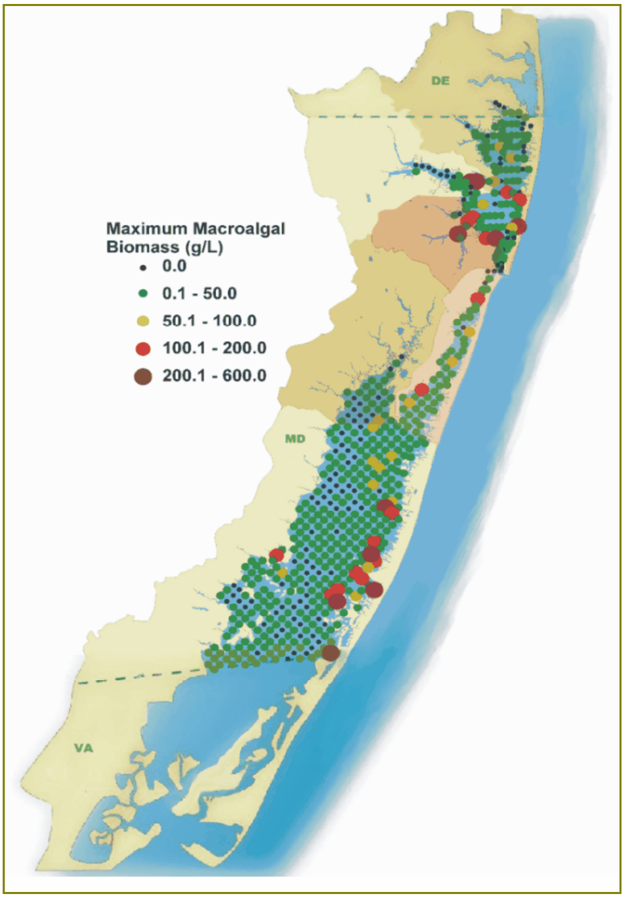

Figure 5.9.11. Maximum total macroalgal biomass per station for all seasons for three survey years (1999–2000, 2001–2002, and 2003). From McGinty et al. 2004.

Long-Term Changes

Significant long-term changes in nutrient loading and ecosystem response have occurred over the past several decades. As noted above, the resident and tourist populations in the watershed are rapidly increasing (Hager 1996; Wazniak et al. 2007). Additionally, anthropogenic changes in hydrodynamics (through the long-term stabilization of inlets, dredging and development on the barrier islands themselves) have altered the natural resilience of many of the systems.

In the past few years, accumulation of very high levels of NH_4^+ is increasing, with averages nearing 10 µM-N for many stations (Figure 5.9.12), with some instances of individual concentrations exceeding 50 µM-N. This trend is of significant concern for the ecological health of the Coastal Bays because such high concentrations can be stressful for many organisms, including plankton and seagrasses, but even at lower levels negatively affect physiology.

When nutrient trends over the past two decades are examined, the relationships are complicated by apparent improvements (decreases) in nutrient concentrations during the 1980s, followed by increases in nutrient concentrations beginning in the late 1990s (Wazniak et al. 2007). Thus, long- term trends appear to differ whether the change in direction is taken into consideration. When linear trends from the late 1980s through 2003 were examined, there appeared to be some increases in water

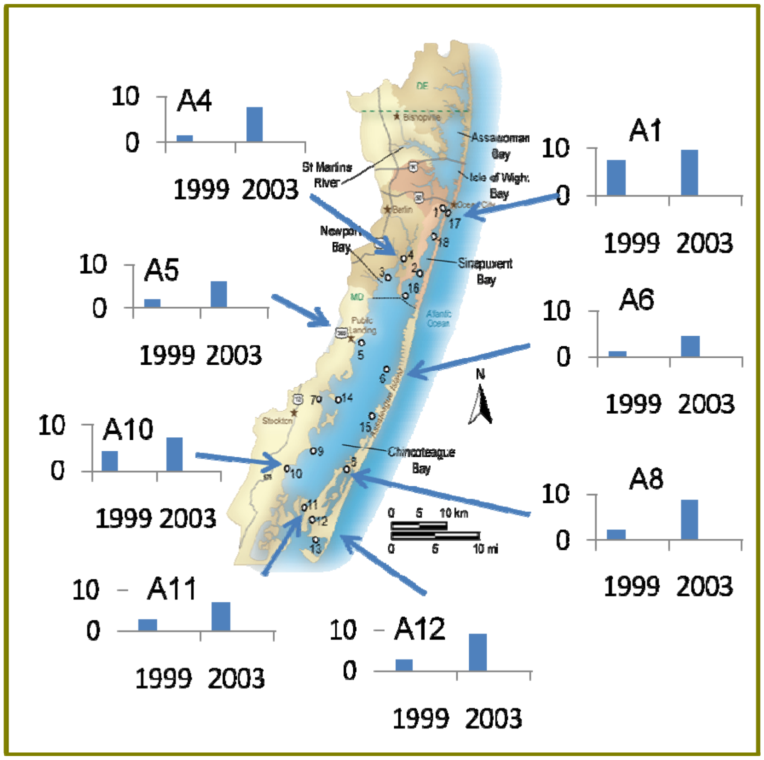

Figure 5.9.12. Change in annual average concentration of NH_4^+ (µM-N) between 1999 and 2003 for the stations indicated. From Glibert and Wazniak, unpublished data.

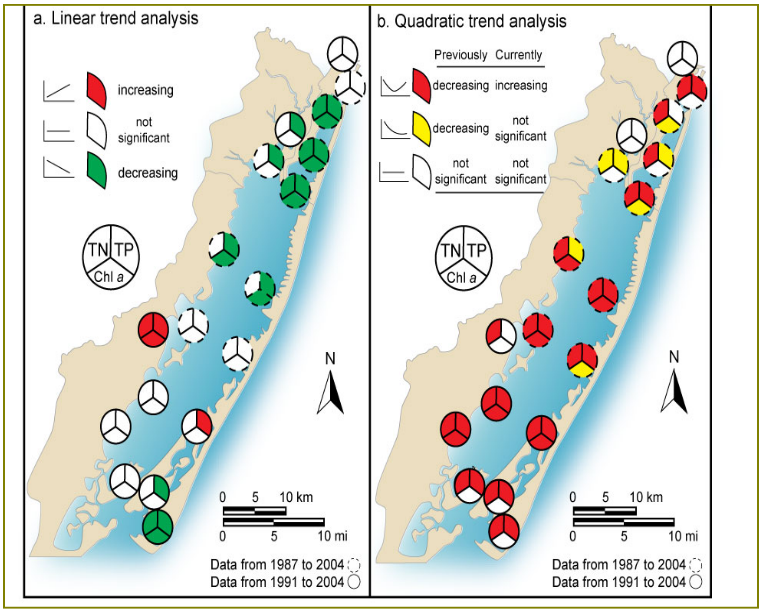

Figure 5.9.13. Trend analyses for stations indicated for the Coastal Bays according to data from the late 1980s to the mid 2000s. Data are shown for calculations based on linear analyses (a) and quadratic analyses (b), which account for a change in direction of the trend over time. From Wazniak et al. 2007.

quality-based on TP and chlorophyll *a* (Figure 5.9.13). However, when the trends are examined using a quadratic model that accounts for an improvement in the 1990s, followed by a reversal in water quality, significant deterioration in water quality based on TN, TP and chlorophyll *a* can be seen (Figure 5.9.13, Wazniak et al. 2007).

As specific examples of these changes, mean summer concentrations of chlorophyll *a* (June to September) increased nearly two-fold from 1996 to 2004, following a linear trend (R^2= 0.76; Figure 5.9.14) at station 5. The strength of the brown tide blooms, as indicated by the maximum concentration of one of its pigments, *but-fuco*, also increased significantly in Chincoteague Bay from 1999 to 2004: that increase was approximately four-fold (R^2 = 0.86; Figure 5.9.14; Glibert et al. 2007). Mean concentrations of DON also appear to be increasing, according to trends of the past decade (Figure 5.9.15).

Throughout the Coastal Bays, SAV coverage increased steadily since monitoring began, with an approximate three-fold increase since 1986 (Figure 5.9.16). However, those increases have leveled over the past several years (Orth et al. 2004, 2006).

Summary of Ecosystem Impacts

Overall water quality status in the Coastal Bays was recently assessed using a water quality index that incorporated values of TN, TP, DO and chloro-

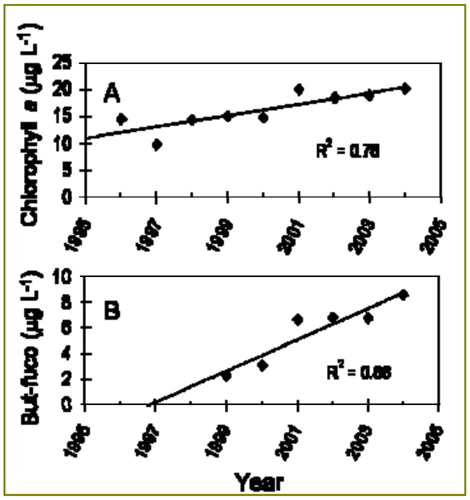

Figure 5.9.14. Mean concentrations of chlorophyll a (µg L^{-1}; A) and 19'-butanoyloxyfucoxanthin (but-fuco, µg L^{-1}; B) of all samples collected during June to September at station 5 in the Coastal Bays from 1996 to 2004. Lines represent linear regressions. From Glibert et al. 2007.

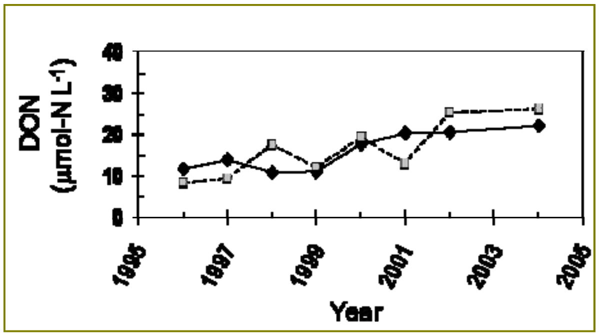

Figure 5.9.15. Concentration of DON (µmol-N) for the year indicated. Values indicated by diamonds are annual averages; squares are averages of April and May only. From Glibert et al. 2007.

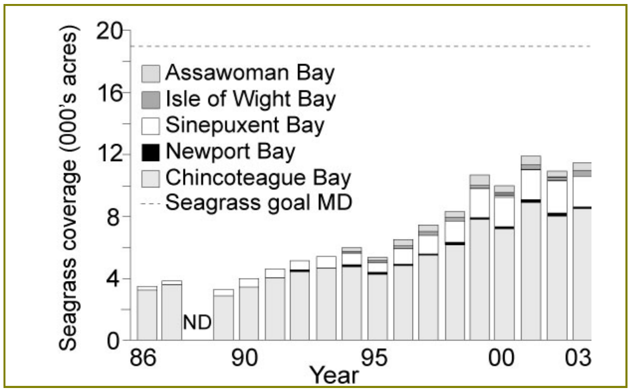

Figure 5.9.16. Change in seagrass coverage in each of the embayments from 1986 to 2003. From Wazniak et al. 2007.

phyll *a* for all stations with at least 10 records for each variable for the years 2001–2003 (Wazniak et al. 2007). Those values were then compared to previously established thresholds, and stations were subsequently rated as excellent, very good, poor, degraded or very degraded. Using that approach, the Coastal Bays had generally poor or degraded water quality in or close to the tributaries and good or excellent water quality in the better flushed, open bay regions (Figure 5.9.17).

The Coastal Bays are showing signs of increasing eutrophication effects in many areas. Concentrations of NH_4^+ and DON appear to have increased substantially over the past several years. The signs of eutrophication impact include seasonal hypoxic events, increases in macroalgae biomass in areas, and annual blooms of *Aureococcus anophagefferens*, the phytoplankton species that causes brown tide and the presence of numerous other HABs. Blooms of brown tide also appear to be increasing annually in intensity (Trice et al. 2004; Glibert et al. 2007). This is of concern as aquaculture is developing in this region. The effects of brown tide on the scallop and clam industries of Long Island have been well established (reviewed by Bricelj and Lonsdale 1997; Gobler et al. 2005); however, clam populations in the Coastal Bays have been generally stable over the past decade (Tarnowski et al. 2004), albeit at low numbers compared to historical abundances. Clam growth in the Coastal Bays aquaculture facilities have been shown to be retarded during periods of brown tide abundance, although recovery in clam growth does occur when brown tide densities begin to subside (Wazniak and Glibert 2004).

Even though there has been extensive expansion of seagrass acreage over the past three decades, that trend has leveled off during recent years (Orth et al. 2006). In relation to that, SAV-dependent bay scallops (*Argopecten irradians*), while present, are found only in low densities, suggesting that the long-term viability of the population is in question (Tarnowski et al. 2004).

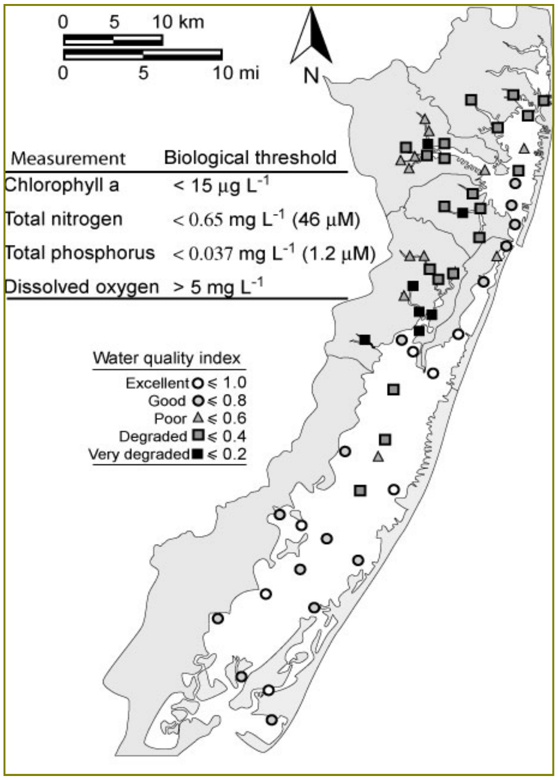

Figure 5.9.17 Summary of water quality status according to a synthetic water quality index based on chlorophyll a, TN, TP and DO. The index compares the average values to the biological thresholds indicated in the figure. From Wazniak et al. 2007.

Such varied observations suggest that the Coastal Bays are undergoing ecosystem change, consistent with increasing anthropogenic nutrient loading.

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Chapter 5:

10. Florida Bay

Christopher J. Madden

General Description

Florida Bay is a shallow (1 to 2 m), wedge-shaped, subtropical lagoon, with an area of about 2,200 km^2 at the southern end of the Florida peninsula, bounded by the Atlantic Ocean and the Gulf of Mexico (Figure 5.10.1). The northern boundary of the bay is formed by the Everglades wetland system on the Florida mainland, and the eastern and southern boundaries are formed by the arc of the Florida Keys and the associated reef tract (McIvor et al. 1994). Limited exchange with the Atlantic occurs through tidal passes between the Keys. On its western border, the bay exchanges freely with the Gulf of Mexico, where a small diurnal tide (amplitude of about 0.5 m) and westerly currents circulate Gulf water into the bay (Smith 1998). The subtropical ecosystem has an average annual temperature of 25 °C and two distinct meteorological seasons: a November–April dry season and a May–October rainy season when 75 percent of the average 152 cm annual precipitation occurs (Duever et al. 1994).

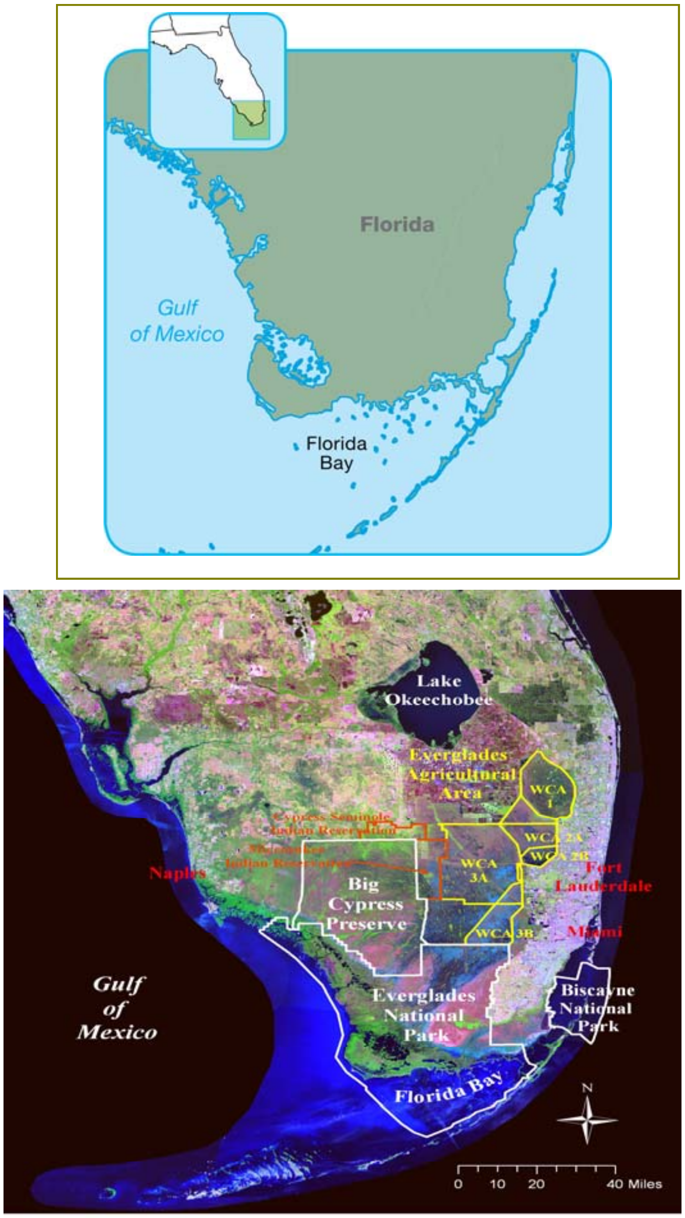

Figure 5.10.1. Two maps of Florida Bay. The top map shows its broader location; the bottom map shows the Everglades-Florida Bay watershed in south Florida, Lake Okeechobee headwaters, agriculture-conversion areas and semi-natural areas of the Everglades flow-way, urban development along the eastern boundary and Florida Bay receiving waters.

The region sporadically experiences climatic extremes, including occasional frost, drought and intense windstorms. Tropical storms and hurricanes have the potential to radically affect the shallow system (Davis et al. 2004) by extirpating benthic macrophytes and mats, resuspending sediments and interstitial nutrients into the water column, redistributing sediments that alter bathymetry and circulation (Nuttle et al. 2003) and laying thick mud deposits on the surrounding wetlands (Davis et al. 2004). Eight intense storms passed directly through or near the Everglades-Florida Bay complex in 2004

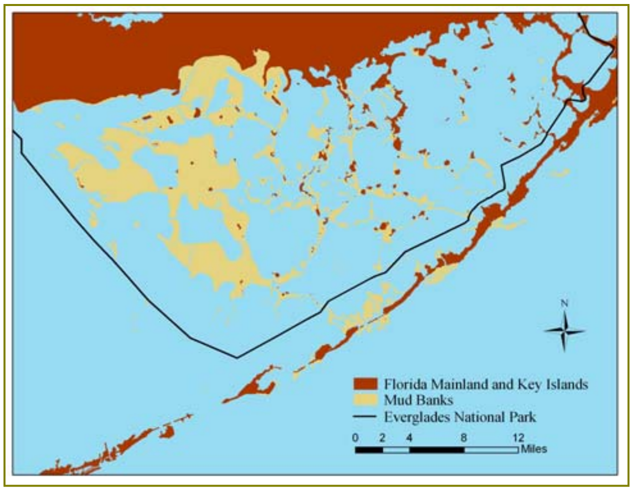

Figure 5.10.2. Mud banks and Key Islands land masses in Florida Bay creating hydrologically quasi-isolated basins.

(Charley, Frances, Ivan, Jeanne) and 2005 (Dennis, Katrina, Rita, Wilma).

Hydrology dominates the ecological dynamics in the Florida Bay estuary and its Everglades watershed. Slowly flowing water, originating in Lake Okeechobee, traverses the Everglades wetlands and mangrove ecotone at the northern boundary of Florida Bay, mobilizing and transforming nutrients and delivering them to the bay. In Florida Bay, nutrient transformations and distributions are dependent on salinity regime and hydrologic transport and biological processes. When the rainy season begins, Florida Bay is a true estuarine system, receiving considerable freshwater flow from the Everglades. However, in the dry season, the bay can resemble a marine lagoon with marine salinities and frequent hypersalinity events. The hydrology of Florida Bay is particularly complex compared to other estuaries. It is strongly controlled by multiple significant inputs including rainfall, freshwater runoff from the Everglades, groundwater input, exchange with the Atlantic Ocean through tidal passes and with the Gulf of Mexico through the western boundary (Wanless et al. 1994; Brewster-Wingard et al. 1999). The water budget has been altered significantly over the past century by filling Atlantic tidal passes and reducing freshwater flow, dramatically changing water transport, salinity regimes and nutrient distributions throughout the bay (Wanless et al. 1994; Rudnick et al. 2006).

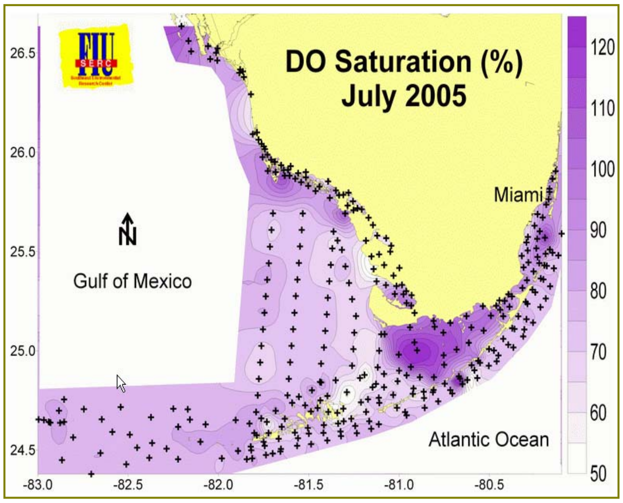

Figure 5.10.3. Example of distribution of DO % saturation in Florida Bay and surrounding waters. From Boyer, unpublished data.

Florida Bay's unique geomorphology includes a system of banks and shoals that create barriers to hydrologic circulation (Figure 5.10.2). The banks form about 40 distinct quasi-isolated basins (Nuttle et al. 2003) where water transport is often via cross-bank flow or through narrow inter-basin channels (Wanless et al. 1994). The banks restrict the tidal exchange and penetration of marine waters into the interior bay. Thus, despite its interposition between two oceans, almost all the tide and hydrologic circulation in eastern and central Florida Bay is wind-driven and fairly limited. The net effect of restricted circulation in the estuary is that high evaporation and long residence times can concentrate salts, particularly in the central bay, which periodically (often annually) experiences hypersaline conditions, with salinities as high as 70 (Smith and Pitts 2001).

Dissolved Oxygen

DO is monitored regularly in Florida Bay, and hypoxia has not yet been shown to be a problem in the water column; although, evidence exists that anoxic sediments can be harmful to rooted macrophytes at certain times and might be related to SAV die-off. Because of the shallow nature of the system and the well-mixed water column, stratification in Florida Bay is rare and transient. Saturation of DO rarely falls below 70 percent (Boyer 2005). The highly productive vegetative benthic community supplies DO to the water column throughout the year. Figure 5.10.3 presents an example of the DO distribution observed in the summer when percent saturation values are generally lowest, showing the high saturation levels in the bay and somewhat lower saturation on the shelf in the Gulf of Mexico.

Turbidity and Light

After decades of extremely clear waters, areas of Florida Bay became increasingly turbid in the early 1990s. Incidence of turbidity, measured as NTUs, increased between 1989 and 1992 by factors of 2, 4 and 20 in the eastern, western and central bays, respectively (Boyer et al. 1999). From a comparative study, Stumpf et al. (1999) reported the water column to be relatively clear in 1987, with a baywide mean downwelling light attenuation parameter (Kd) of 0.51 m^{-1} while in 1995, the mean attenuation was 2.82 m^{-1}. Much of the increased turbidity was from increased phytoplankton concentration and increased resuspension of carbonate bottom sediments (Boyer et al. 1999). The abrupt changes are likely associated with a significant loss of SAV in the late 1980s, leading to reduced sediment binding, increased sediment resuspension, increased nutrient availability and release

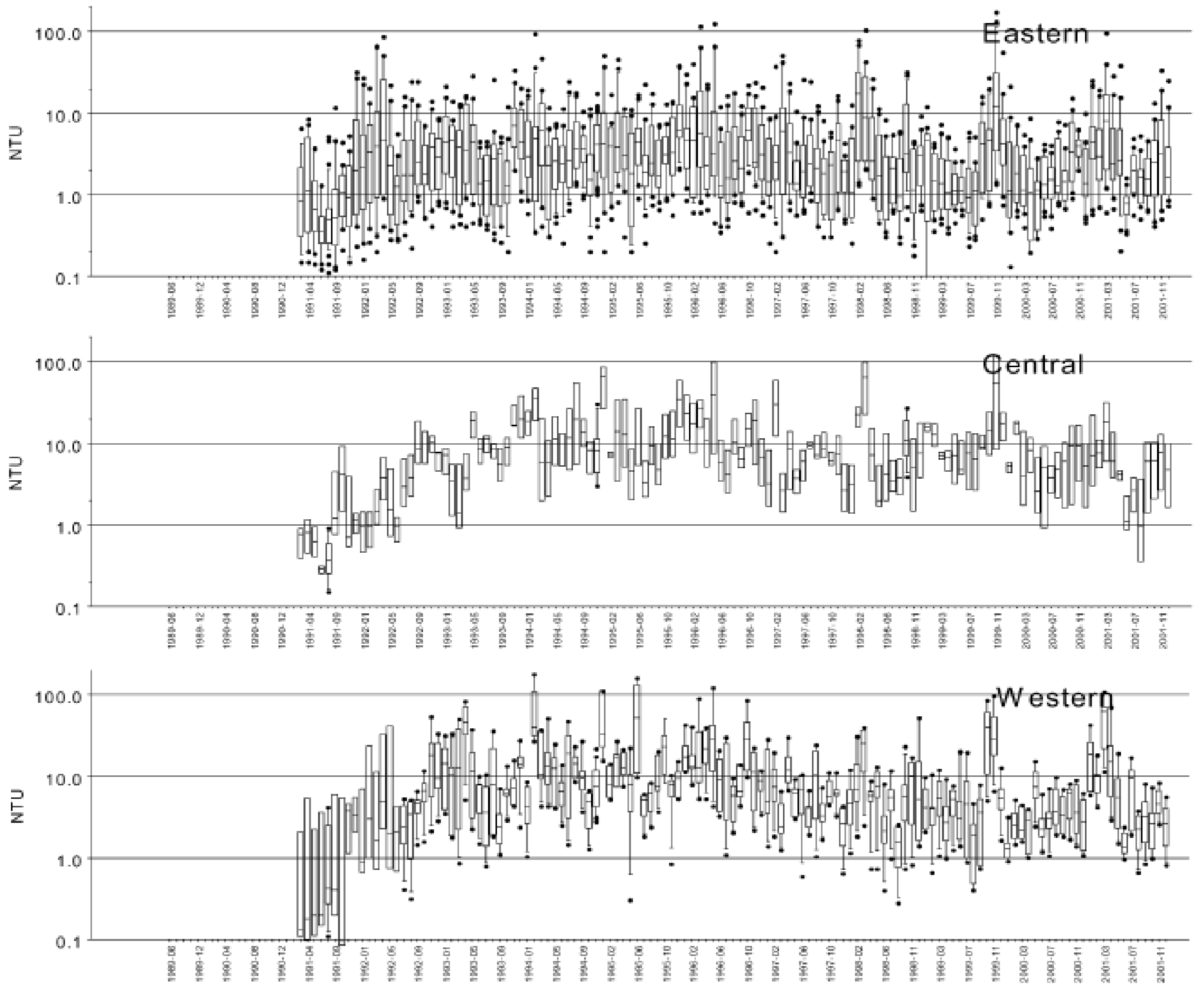

Figure 5.10.4. Long-term monitoring record of monthly turbidity (in NTUs) 1989–2003 at 24 stations in eastern (top), central (middle) and eastern (bottom) Florida Bay. From Boyer et al 1999.

from the benthos (Hunt and Nuttle 2007). Turbidity has somewhat declined through the late 1990s and into the 2000s, although it seems to be stabilizing at a higher level than in previous decades of very clear water, probably a consequence of continuing phytoplankton blooms and more resuspendable sediments. The transition of the bay from clear to less clear is captured in the early monitoring program data shown in Figure 5.10.4.

Table 5.10.1. Parameters sampled in the Florida Bay monitoring program and long-term averages for the entire bay (all) and for the central, eastern, and western regions of the bay.

Variable	Zone	Median	Min	Max	n
Chlorophyll *a* (mg L^{-1})	All	0.84	< 0.03	35.61	3,612
	Central	1.79	0.11	35.61	542
	East	0.55	< 0.03	11.35	2,284
	West	1.55	0.14	22.08	786
DO - surface (mg L^{-1})	All	6.6	0.4	12.3	3,633
	Central	6.4	2.8	12.3	545
	East	6.7	0.4	11.7	2,289
	West	6.3	3.0	11.5	799
DO - bottom (mg L^{-1})	All	6.5	1.4	13.4	3,414
	Central	6.3	1.5	12.2	514
	East	6.7	1.4	13.4	2,174
	West	6.2	3.0	11.1	726
Salinity-surface	All	31.9	0.2	63.0	3,691
	Central	34.0	8.7	63.0	554
	East	28.9	0.2	54.3	2,324
	West	35.0	16.5	52.0	813
Salinity-bottom	All	31.3	0.2	63.0	3,376
	Central	33.2	11.9	63.0	510
	East	28.4	0.2	54.3	2,140
	West	34.7	16.6	51.0	72
NO_3^- (mM-N)	All	0.36	< 0.03	11.0	3,580
	Central	0.21	< 0.03	5.71	537
	East	0.64	< 0.03	11.0	2,268
	West	0.14	< 0.03	7.21	775
NH_4^+ (mM-N)	All	2.28	< 0.03	120	3,592
	Central	3.64	< 0.03	120	535
	East	2.78	< 0.03	82.1	2,277
	West	0.78	< 0.03	24.4	780
PO_4^{-3} (mM-P)	All	0.03	< 0.03	0.8	3,570
	Central	0.03	< 0.03	0.8	537
	East	0.03	< 0.03	0.5	2,260
	West	0.03	< 0.03	0.3	773

Nutrients

The heterogeneous patterns of freshwater input and circulation create four distinct zones in the bay, each exhibiting coherent patterns of salinity and water quality (Table 5.10.1, Figure 5.10.5, Boyer et al. 1997). Freshwater inflow from the Everglades most influences the eastern bay. The central bay is the most hydrologically isolated region, receiving low freshwater input and having a high water residence time. The western bay is characterized by marine inflows from the Gulf of Mexico and is indirectly affected by freshwater transported from Shark River Slough as it discharges onto the western Florida Shelf. A fourth water quality sector characterizes the mangrove transition zone at the Everglades-Florida Bay interface, which is typified by seasonal freshwater inundation, and extensive mangrove wetlands interspersed with shallow ponds and small channels.

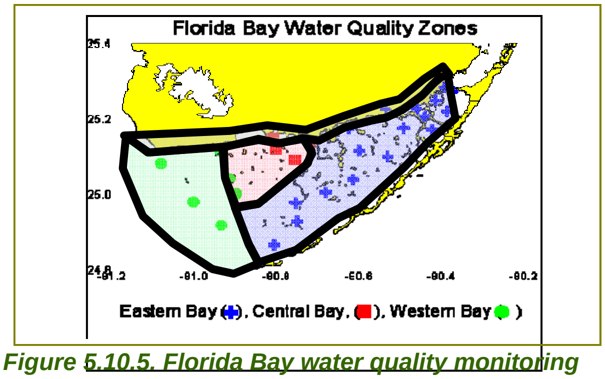

Figure 5.10.5. Florida Bay water quality monitoring program stations grouped into zones of similar water quality, eastern, central and western bay plus mangrove transition zone at the Everglades-Florida Bay interface. Adapted from Boyer et al. 1997.

A water quality monitoring program has tracked chlorophyll *a,* inorganic and total nutrients, turbidity and hydrographic parameters monthly at up to 28 stations in Florida Bay beginning as early as 1989 for some stations (Table 5.10.1). Everglades discharge to Florida Bay has been measured at several sites since 1995. Inputs to the eastern bay from the Everglades panhandle are characteristically very low in P, as the calcium carbonate substrate of the Everglades marl and Florida Bay sediments effectively scavenges P from the water column, binding and sequestering it in a variety of forms (Nielsen et al. 2006). Most of the P available to autotrophs in the eastern bay is found in the sediments either in the solid phase as loosely bound oxy-hydroxides or as apatite, or in the interstitial porewaters. Water column productivity is thus generally very low in the east, while benthic plants have better access to this limiting nutrient in the sediments (Fourqurean et al. 2002; Nielsen et al. 2006).

In the central bay, inflows from Taylor Slough tend also to be low in P and relatively higher in N compounds, particularly DON. The influence of the Gulf of Mexico can be seen in this region in the delivery of marine P. Combined with N from Everglades runoff, the convergence of nutrients can lead to the highest chlorophyll *a* concentrations and water column productivity in the system. Seagrasses and other benthic macrophytes and microphytobenthos are also much more productive in this region than in the eastern bay. N, mostly in the form of NH_4^+, is generally readily available to benthic plants in sediment pools (Jackson and Burd 2002). Both P and N have decreased in all regions of the bay since the mid-1990s and are at their lowest levels in a decade despite the transient effect of Hurricane Irene in 1999, tropical storms in 2002 and several hurricanes in 2004 and 2005. Florida Bay waters transition from P limitation in the east to N limitation in the west in the central bay region, and autotrophs are generally most nutrient sufficient in that area.

In the western area of the bay, being farther from major Everglades inputs, available N is in somewhat lower supply, while P forms are in greatest supply from across the Gulf boundary. Some blooms occur here, especially when favorable currents bring Everglades discharge and associated N into the bay from the Florida Shelf. The P load of the Gulf waters is enhanced by the discharge of terrestrial P into the western Florida coastal waters.

Nutrients from the Everglades (Sutula et al. 2003; Davis et al. 2004) and diffuse sources flow through the mangrove transition zone at the bay-wetland interface, and they are related to seasonal patterns of freshwater discharge. Nutrient loading and nutrient concentration increase with increasing water discharge during the wet season, although not linearly. Output of P from the Everglades to Florida Bay occurs only during the wet season mostly as DOP, in very low concentrations. During the dry season, P is imported by the mangrove ecotone from Florida Bay. The output of N to the bay from the Everglades is mostly as DON, in high concentrations, resulting in a significant N loading and very high molar TN:TP ratios, near 200, in the export.

A nutrient budget for Florida Bay was calculated by Rudnick et al. (1999, 2006). The major source of both P (490 MT y^{-1}) and N (11,500 MT y^{-1}) to the bay is from the Gulf of Mexico (Figures 5.10.6, 5.10.7). P also enters the system from groundwater (38 MT y^{-1}), atmospheric deposition (16 MT y^{-1}; a highly uncertain term), and the Atlantic Ocean (16 MT y^{-1}). The smallest source of TP is from the Everglades (< 10 MT y^{-1}). Significant P is exported to the Atlantic (180 MT y^{-1}) and to the Gulf (87 MT y^{-1}).

For N inputs, following Gulf input, groundwater supplies the next highest amount of N to the bay (8,800 MT y^{-1}), which is a bit over half of the Gulf contribution, although that estimate might be high. Atmospheric (1,400 MT y^{-1}) and Atlantic (87 MT y^{-1}) inputs are minor and mostly in inorganic form, while the input from the oligotrophic southern Everglades is 290 MT y^{-1}, just over 1 percent of TN inputs, is mostly in organic form. The contribution from Shark Slough of P (< 10 MT y^{-1}) and N (1,240 MT y^{-1}) was not figured in these calculations because the hydrologic path is uncertain—depending on current direction some, all or none of that export can be transported into Florida Bay.

Chlorophyll and Primary Production

Compared to other estuaries, Florida Bay has low chlorophyll *a* in general and, in particularly in the eastern bay, where P is severely limiting (Figure 5.10.8), according to water column stoichiometry (Phlips et al. 1999). Eastern bay chlorophyll *a* has averaged 0.85 µg L^{-1} since 1989

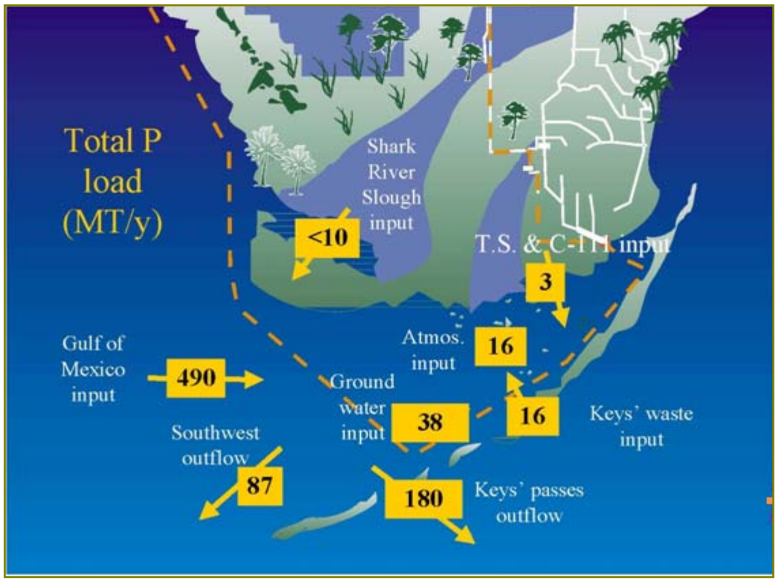

Figure 5.10.6. Annual budget for TP (in MT y^{-1}) for Florida Bay. From Hunt and Nuttle 2007. Note that estimated fluxes from groundwater and at the Gulf of Mexico boundary have high uncertainty.

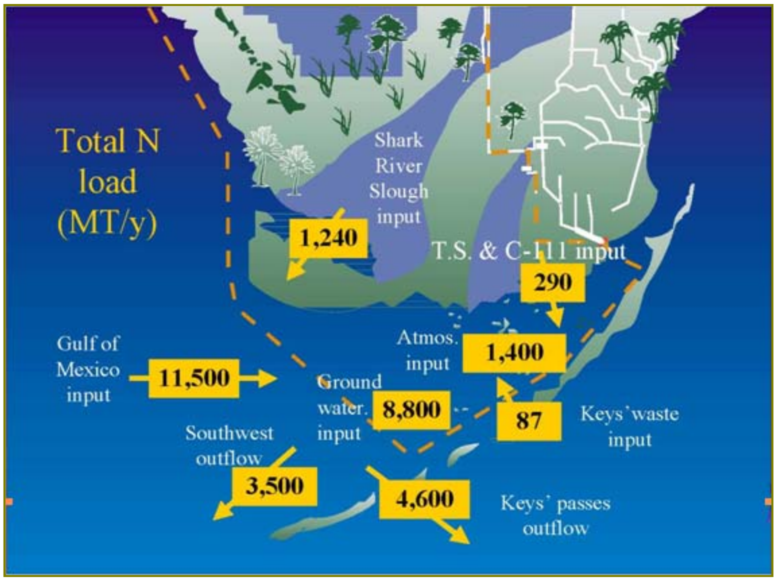

Figure 5.10.7. Annual budget for TN (in MT y^{-1}) for Florida Bay. From Hunt and Nuttle 2007. Note that estimated fluxes from groundwater and at the Gulf of Mexico boundary have high uncertainty.

(Boyer et al. 1999). Bioassay studies confirmed that phytoplankton are P-limited there (Tomas et al. 1999), although more recent bioassays (2006–2007) suggest that N limitation can develop during some blooms in the east (Heil and Glibert, unpublished). In the central bay, where N and P are more balanced because of the convergence of Gulf and Everglades nutrient inputs, chlorophyll *a* concentrations are highest, averaging 2.34 µg L^{-1}, but regularly exceeding 10 µg L^{-1} and frequently exceeding 20 µg L^{-1} during blooms (Phlips et al. 1999).

Chlorophyll *a* concentrations in the western bay are more moderate, averaging 1.93 µg L^{-1}. Phytoplankton in this area responds to additions of N and Si, indicating that both nutrients limit production (Tomas et al. 1999).

Phytoplankton Blooms and Species Composition

Phytoplankton community composition within Florida Bay varies within the major regions of the bay. In the north-central region, diatom blooms (eg. *Thalassiosira* spp.) often occur in bays adjacent to the mangrove fringe. Blooms in the western bay exhibit a seasonal variation, peaking in late summer to winter and tending to be dominated by centric (*Rhizosolenia* spp.) and pennate diatoms (*Cocconeis*, *Navicula*, and *Surirella* sp.; Phlips and Badylak 1996). High biomass blooms of both the toxic dinoflagellate *Karenia brevis* and the N_2-fixing cyanobacteria *Trichodesmium* are occasionally transported into western Florida Bay from the Gulf of Mexico. Blooms in the eastern and central bays tend to occur in late summer and fall (Phlips et al. 1999) and are dominated by cyanobacteria (*Synechococcus* sp).

Before 2005, the northeastern bay phytoplankton community was a diverse mixture of non-bloom forming dia-

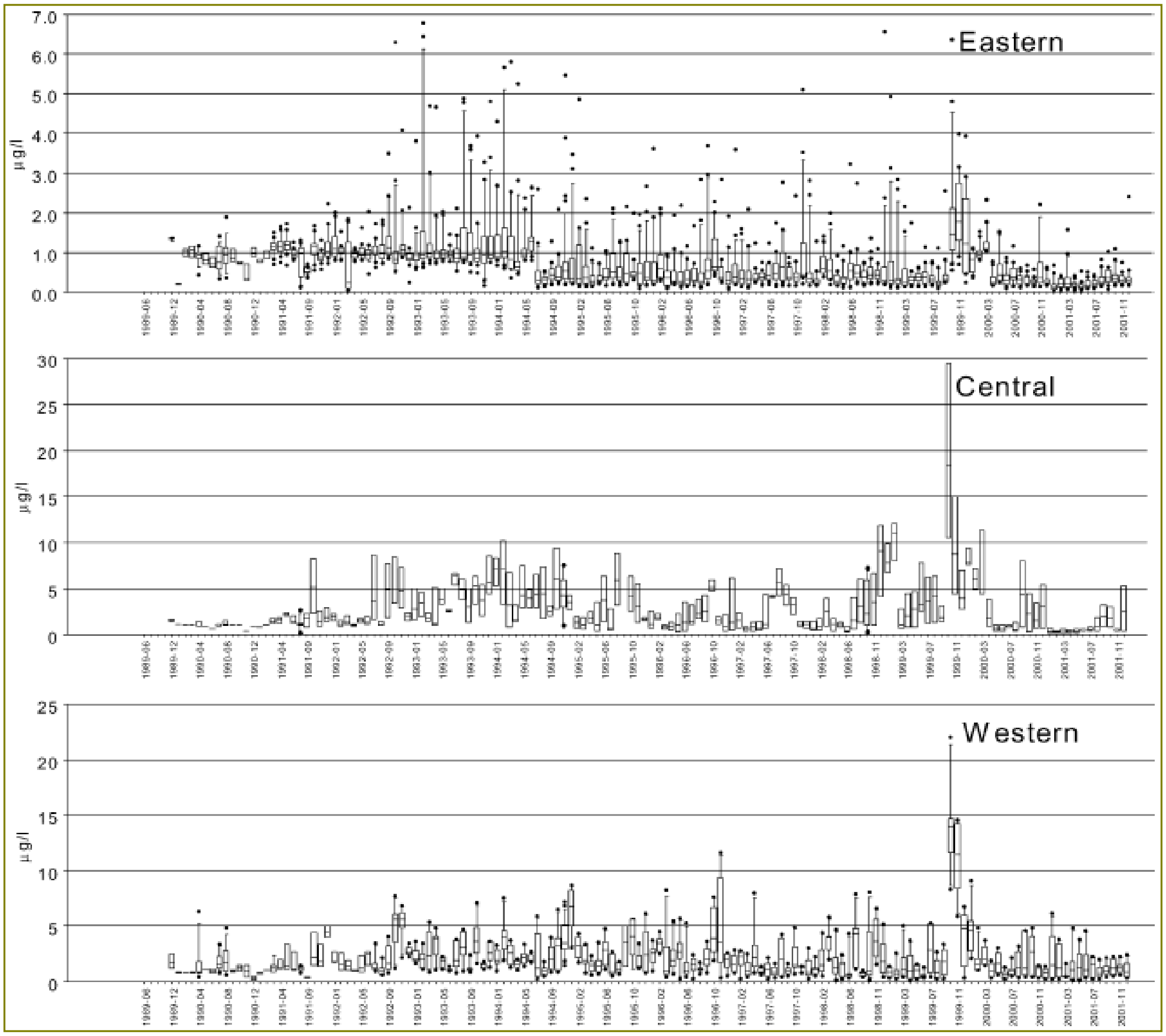

Figure 5.10.8. Long-term monitoring record (1989–2003) of monthly chlorophyll **a** ***(in µg L^{-1}) at 24 stations in eastern (top), central (middle) and western (bottom) Florida Bay. From Boyer et al. 1999.***

toms, cyanobacteria, microflagellates and dinoflagellates, including ciguatera associated species and *Pyrodinium bahamense* (Hunt and Nuttle 2007).

Following Hurricane Katrina in 2005, however, a bloom of *Synechococcus* spp. developed and persisted through at least 2007. Similar blooms of *Synechococcus* spp. have become a common feature of the central bay region in late summer and fall since about 1992 (Phlips et al. 1999; Glibert et al. 2004).

Benthic Primary Producers

SAV cover an estimated 5,500 km^2 of the greater Florida Bay and Keys area (Zieman 1982) in meadows dominated by turtle grass (*Thalassia testudinum*), often mixed with shoal grass (*Halodule wrightii*). Manatee grass (*Syringodium filiforme)* is found in generally deeper waters nearer the Gulf of Mexico, and widgeon grass *(Ruppia maritima)* occurs in the fringes of the mangrove transition zone near fresher water. SAV are the keystone community of the Florida Bay ecosystem, playing roles in many important physicochemical (Stumpf et al. 1999;

Matheson et al. 1999), autotrophic (Fourqurean et al. 2002) and higher trophic (Ley and McIvor 2002; Lorenz et al. 2002) functions of the bay's ecology. They are the dominant primary producers in the bay and have a high standing biomass, increasing along a nutrient gradient from east to west. The sediment-binding capacity of the SAV serves to reduce turbid resuspension events and bottom scouring, promote water clarity and thereby enhance benthic primary production (Zieman 1982). SAV remove nutrients from the water column, reducing nutrients available to phytoplankton.

Long-Term Changes

In fall 1987, the SAV community underwent a catastrophic mortality event (Robblee et al. 1991), which destroyed 4,000 ha of *Thalassia* (Figures 5.10.9, 5.10.10), and thinned an additional 23,000 ha (Robblee et al. 1991), resulting in the loss of 30 percent of the community (Hall et al. 1999; Durako et al. 2002). The mortality is hypothesized to be the result of multiple stresses (high temperature, salinity and sulfide; Zieman et al. 1999). Maximum loss of *Thalassia* occurred in the highest density beds (Figure 5.10.10) and loss of this keystone species caused a cascade of ecological effects. Within 3 years of the die-off, beginning in 1991, phytoplankton blooms began to appear in the central and western bay (Boyer et al. 1999; Stumpf et al. 1999). A 100 percent mortality of sponges ensued, and several genera of sponges permanently disappeared from the bay (Fourqurean and Robblee 1999). The nursery function of Florida Bay was affected as landings of spiny lobster (Butler et al. 1995) and pink shrimp at Tortugas Banks plunged in 1988 to their lowest levels in decades (Robblee et al. 1991). Game fish landings also declined as SAV community composition shifted.

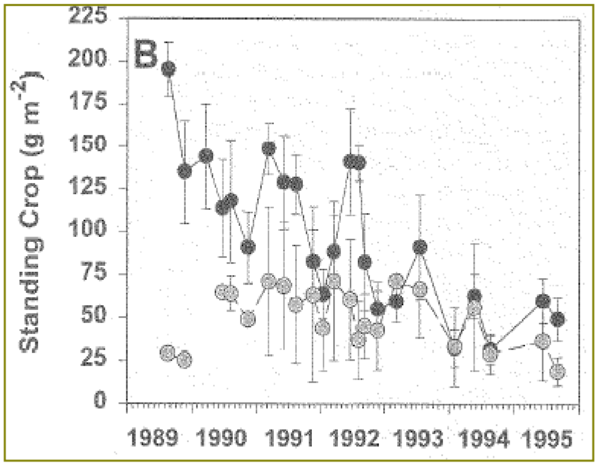

Figure 5.10.9. Long-term standing crop of* Thalassia *at multiple stations in both die-off (dark circles) and non-die-off (light circles) areas. Plants in areas that experienced die-off converged toward the (possible) equilibrium biomass level of non-dieoff areas. From Zieman et al. 1999.

Bloom events varied spatially, with large blooms dominated by diatoms occurring in the western bay and cyanobacteria blooms in the central bay (Boyer et al. 1999). The blooms continued through the 1990s and into the mid-2000s (Richardson and Zimba 2002; Glibert et al. 2004) with a recent expansion of bloom activity to the eastern bay. Low-level SAV die-offs continue today as *Thalassia* ebbs and increases in the central bay region (Figure 5.10.11).

In addition to the central bay algal blooms that seem to have been a secondary effect of SAV mortality, a new bloom event has taken hold in an area where no SAV die-off has occurred. Beginning in October 2005, an unprecedented phytoplankton bloom developed and has persisted through at least 2007 in oligotrophic eastern

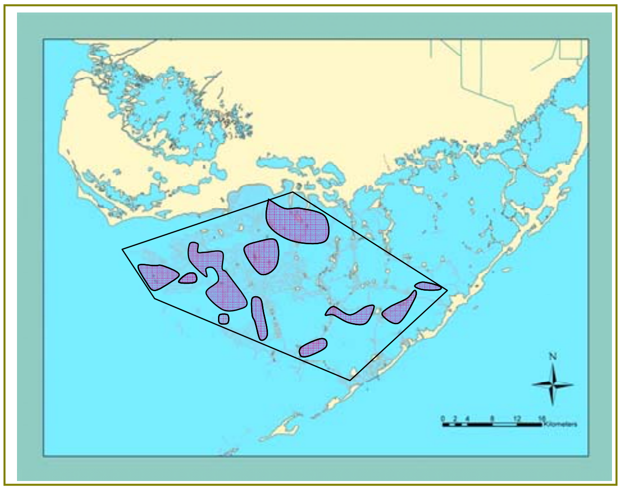

Figure 5.10.10. Patches indicate approximate areas of major* Thalassia *die-off in 1987. Adapted from Robblee et al. 1991.

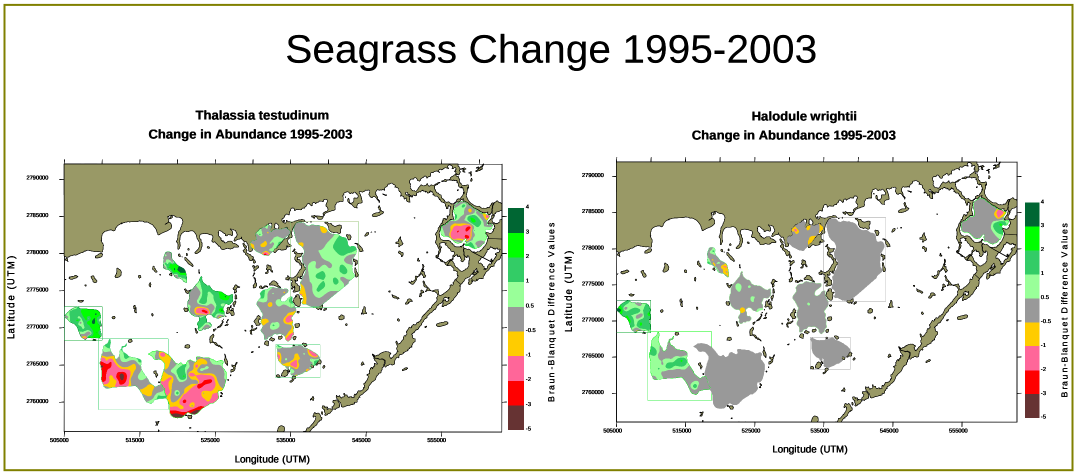

Figure 5.10.11. Change maps of SAV in Florida Bay during the post-1987 die-off period. Red areas are SAV loss during the interval, and green areas are gains for **Thalassia** ***(left) and*** **Halodule** ***(right). From Durako et al. 2005.***

Florida Bay (Figure 5.10.12). Peak chlorophyll *a* concentrations near 30 µg L^{-1} greatly exceeded values ever recorded in this area through 17 years of coastal water quality monitoring (Madden 2007). The dominant species in the bloom is *Synechococcus* sp., the same species implicated in previous central bay blooms, but the trigger for the eastern bloom is likely to have been unrelated to blooms in other parts of Florida Bay. The timing of the eastern bay bloom onset in 2005 roughly coincided with, and might be related to, the confluence of several hurricane events combined with the beginning of construction activities on U.S. Highway 1 along a narrow 20-mile land-bridge connecting the mainland to the Keys and bisecting eastern Florida Bay. Road construction required cutting and mulching mangrove trees in place and extensive soil excavation and tilling. That activity might have released an unknown amount of nutrients into the system (Rudnick et al. 2006). In addition, during the construction, three hurricanes (Katrina, Rita, Wilma) affected south Florida in 2005, resulting in wind disturbance of plants, soils and sediments and a large discharge of freshwater and P from the drainage canal network to the eastern bay. Water quality monitoring indicates that the bloom coincided with a large increase in TP in the area (Figure 5.10.13). The proximity of the blooms to U.S. Highway 1, where blooms had never been previously observed, might thus implicate the unique disturbance of construction of a major causeway as a cause of the bloom, possibly in concert with the hurricane effects and water releases. Such construction resulted in destruction of a significant number of mangroves, with much of the organic matter being deposited in the bay. The timing of the P peak and the subsequent bloom indicates that a synergy of anthropogenic and natural disturbance might be the likely causal agent.

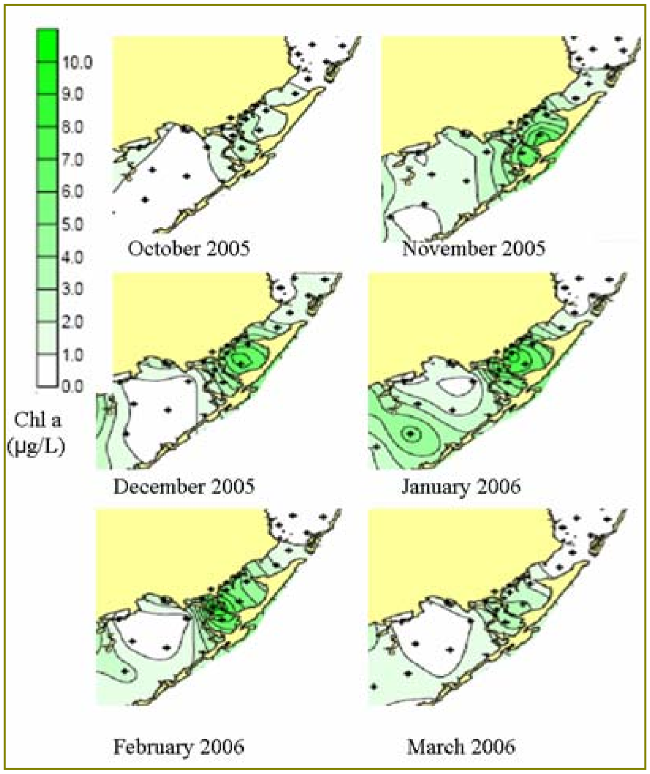

Figure 5.10.12. Monthly monitoring of the phytoplankton bloom development (as chlorophyll **a** ***in µg L^{-1}) in Eastern Florida Bay from October 2005 to March 2006.***

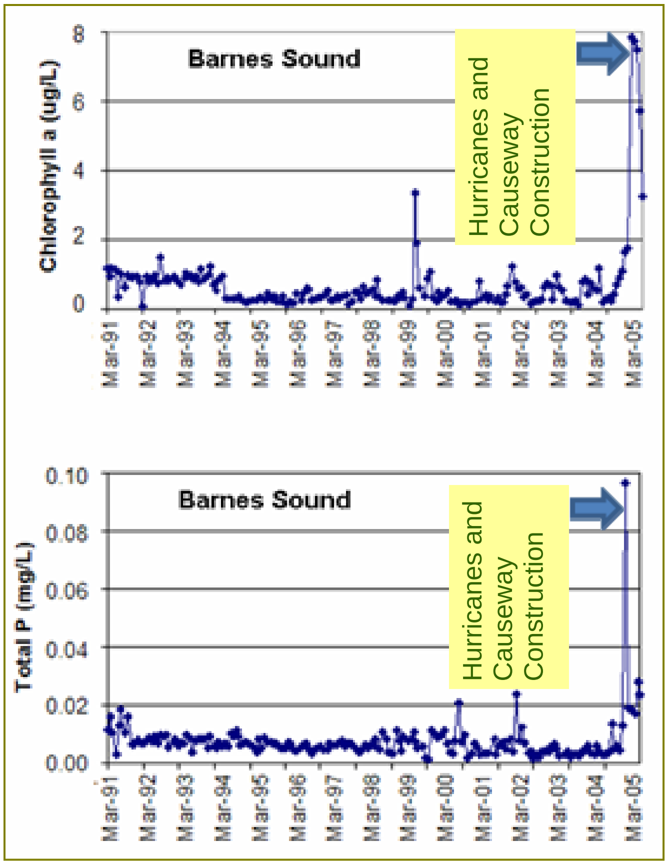

Figure 5.10.13. Long-term monthly monitoring record of chlorophyll* a *and TP in Barnes Sound, locus of the recent phytoplankton bloom in Eastern Florida Bay.

The management strategy for Florida Bay seeks ecological restoration on an ecosystem scale. It involves increasing Everglades freshwater flows downstream by implementation of massive water diversion projects (CERP 2005). It is important that the additional water be of sufficient quality and be below critical nutrient thresholds such that the Florida Bay system does not respond negatively. As seen by the ongoing persistent phytoplankton bloom in the eastern part of the generally oligotrophic system, even small or short-term nutrient increases can have wide-ranging and long-term effects. The bay's trophic status could be poised between alternate stable states of benthic and planktonic dominance, and care must be taken to prevent crossing critical nutrient thresholds that might favor a pelagic-based algal system at the expense of the important benthic community.

Summary of Ecosystem Impacts

Historically, the gradual seasonal pulsing of freshwater entering Florida Bay from Everglades surface runoff, groundwater and precipitation, created a broad salinity gradient ranging from the fresh northern bay to the marine southwestern bay (Zieman 1982; Wanless et al. 1994). Today's freshwater inputs to Florida Bay equal about 30 percent of historical flows. Human effects in the upstream watershed, primarily the increased consumption of water and the drainage of Everglades water directly to the oceans, bypassing Florida Bay, have changed the natural hydrology of the bay. Flows into the northeast and central bay are now flashy, influenced by flood protection criteria upstream, leading to wide variations in salinity in this region. The hydrological changes have certainly altered the functioning of the estuary.

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Chapter 5:

11. Pensacola Bay

James Hagy

Figure 5.11.1. Map of Pensacola Bay.

General Description

The Pensacola Bay system in Florida is a complex of estuaries that includes Escambia Bay, Blackwater Bay, East Bay, and Pensacola Bay (Figure 5.11.1). Compared to well-studied estuaries, relatively few published studies address the condition of the Pensacola Bay ecosystem or important features of its ecological functioning. A 1975 report (Olinger et al. 1975) provides the best, albeit dated, overview of ecological features, condition, and function of the Bay. EPA (2005) also reviewed major ecological features of the system and provided a new assessment of ecological condition, in particular water and sediment quality. Other recent and important publications examine both specific aspects of the ecology of the system and, more broadly, the role of nutrients in the ecosystem (Murrell et al. 2002; Murrell and Lores 2004; Murrell and Caffrey 2005; DiDonato et al. 2006; Hagy and Murrell 2007; Murrell et al. 2007; Hagy et al. 2008).

The combined Pensacola Bay system is medium-sized (370 km^2) and shallow (mean depth = 3.0 m). A substantial portion of the estuary is usually well-stratified at a depth of 1 to 2 m (Hagy and Murrell 2007). Bottom salinity can be up to 20 higher than surface salinity (Figure 5.11.2). Tides are diurnal and have low amplitude, ranging from 15 to 65 cm. Tidal currents are weak and there is generally low vertical diffusive exchange (Hagy and Murrell 2007). Very stagnant conditions within the lower layer can be created (Gallagher et al. 1999). Freshwater residence time is approximately 10 days on average (Cherry and Hagy 2006), but whether the average has any useful application in this case is unclear; flushing time is usually much longer, with periods of rapid flushing associated with episodic peak flows. Nutrients and organic inputs are flushed from the system much more rapidly during peak flow (Murrell et al. 2007), whereas residence time is much longer at other times. The rate of mixing of the lower layer, which influences development of hypoxia, is entirely unrelated to freshwater residence time (Hagy and Murrell 2007).

Dissolved Oxygen

Hypoxia ($O_2 < 2.0$ mg L^{-1}) has been observed in Pensacola Bay in every month except January, with dramatically higher frequency and extent in June through October. Hypoxia affected a relatively substantial 24 percent (16 to 36 percent) of the bay bottom during summer probabilistic surveys in 1996–2000 (USEPA 2005), making it one of the most important water quality issues for this system. Anoxia is much less prevalent: only 17 percent of DO observations less than 2.0 mg L^{-1} were also less than 0.2 mg L^{-1}. The presence of hydrogen sulfide in bottom waters was never noted during surveys conducted by EPA in 2002–2004. The location of hypoxic waters is strongly associated with water column stratification and the two-layer residual transport regime in both branches of the system (Figure 5.11.2; Hagy and Murrell 2007).

Recent measurements indicate that metabolic rates remain low compared to many estuaries (Murrell et al. 2009). The extent of hypoxia, therefore, appears to result principally from extreme sensitivity of the system to developing hypoxia, rather than high DO demand.

Turbidity and Light

Light attenuation is relatively low in Pensacola Bay (USEPA 2005), especially in relation to the mean depth of the system (3.0 m). Nearly 70 percent of Secchi depth observations during monthly surveys in 2000–2004 were 1 to 2 m, with values at times as high as 6 m (Hagy et al. 2008). Secchi depth was typically ~1 m near the river, where the bay is shallowest, and 2 m at locations downbay, where deeper depths occur. Euphotic zone depth commonly exceeds the pycnocline depth and sometimes includes the entire water column. Throughout most of the bay, TSS concentrations are < 10 mg L^{-1}, of which ~50 percent is organic particles (USEPA, unpublished data). Those observations suggest that light attenuation is dominated by CDOM or plankton, but generally not suspended mineral sediments.

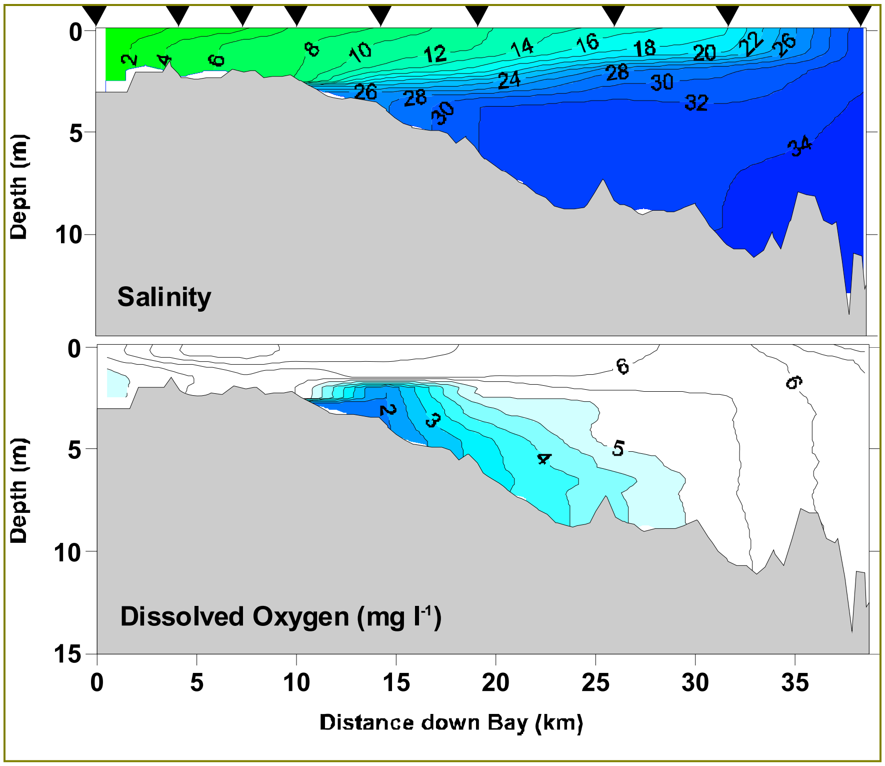

Figure 5.11.2. Cross-sectional profiles of salinity and DO on June 4, 2003. The transect runs from the mouth of the Escambia River to Pensacola Pass. Arrows indicate the locations of sampling stations at which CTD casts were conducted. Adapted from Hagy and Murrell 2007.

Nutrients

TN and TP concentrations at the mouth of the Escambia River, the largest river source to the bay, are 40 to 80 µM -N and 0.4 to 1.5 µM-P, respectively. TN is slightly lower (~40 µM-N) in the Blackwater River, whereas TP is somewhat higher, ~2.5 µM-P. Annual average loading of TN and DIN is estimated to be 1000 and 465 mmol N m^{-2} y^{-1}, respectively. TP and DIP loading rates are 24 and 4.6 mmol P m^{-2} y^{-1}. The loading ratio for N:P is ~100 (Cherry and Hagy 2006; Hagy et al. 2008).

NO_3^- concentrations decrease strongly along the salinity gradient from Escambia River into Pensacola Bay, from 14 µM -N in freshwater to undetectable levels (< 0.5 µM-N) at the seaward endpoint (Figure 5.11.3). Nearly conservative mixing profiles occurred in winter, but strong removal occurred in warmer months. In some instances, NO_3^-

was not detectable throughout much of the estuary. NH_4^+ concentrations decrease with increasing salinity in both surface and bottom water. In bottom water, however, positive departures from conservative mixing bottom layer suggest net NH_4^+ regeneration rates that are substantial relative to the turnover time of bottom layer waters. Direct measurements of benthic NH_4^+ fluxes were relatively low (DiDonato et al. 2006; Murrell et al. 2009).

Surface water PO_4^{-3} decreases from 0.35 µM-P in Escambia river water to 0.1 µM-P just outside the river and remains low along the transect from Escambia River to Pensacola Pass. In bottom water, apparent strong summertime regeneration (Figure 5.11.3) coincides with the region of hypoxia. A local increase in surface P is observed where the landward flowing bottom layer terminates and upwells into the surface layer (Figure 5.11.3). Application of biogeochemical budgeting to Pensacola Bay indicates that, on average, P uptake in the bay is balanced by P regeneration, implying balanced ecosystem metabolism (Cherry and Hagy 2006). As a consequence, one can infer that the net NO_3^- removal can be attributed largely to denitrification, which is estimated to have a modest annual mean system-wide rate of 17 µmol m^{-2} h^{-1} (Cherry and Hagy 2006).

Chlorophyll and Primary Production

Phytoplankton production and biomass is low in Pensacola Bay relative to many estuaries (Hagy et al. 2008). Surface chlorophyll *a* is highest in mesohaline (salinity 5 to 18) waters, where the median is 6.5 µg L^{-1} (Figure 5.11.4). Median surface chlorophyll *a* in the oligohaline and polyhaline regions of the estuary is 3.5 and 4.4 µg L^{-1}, respectively. Surface chlorophyll *a* peaks in summer (May through August) when the median is ~8 to 10 µg L^{-1}. Winter concentrations in the oligohaline reaches of the bay are as low as 1 µg L^{-1} (Figure 5.11.4).

Phytoplankton production from Pensacola Bay is proportional to the product of biomass, photic depth, and incident PAR irradiance (Murrell et al. 2007). Accordingly,

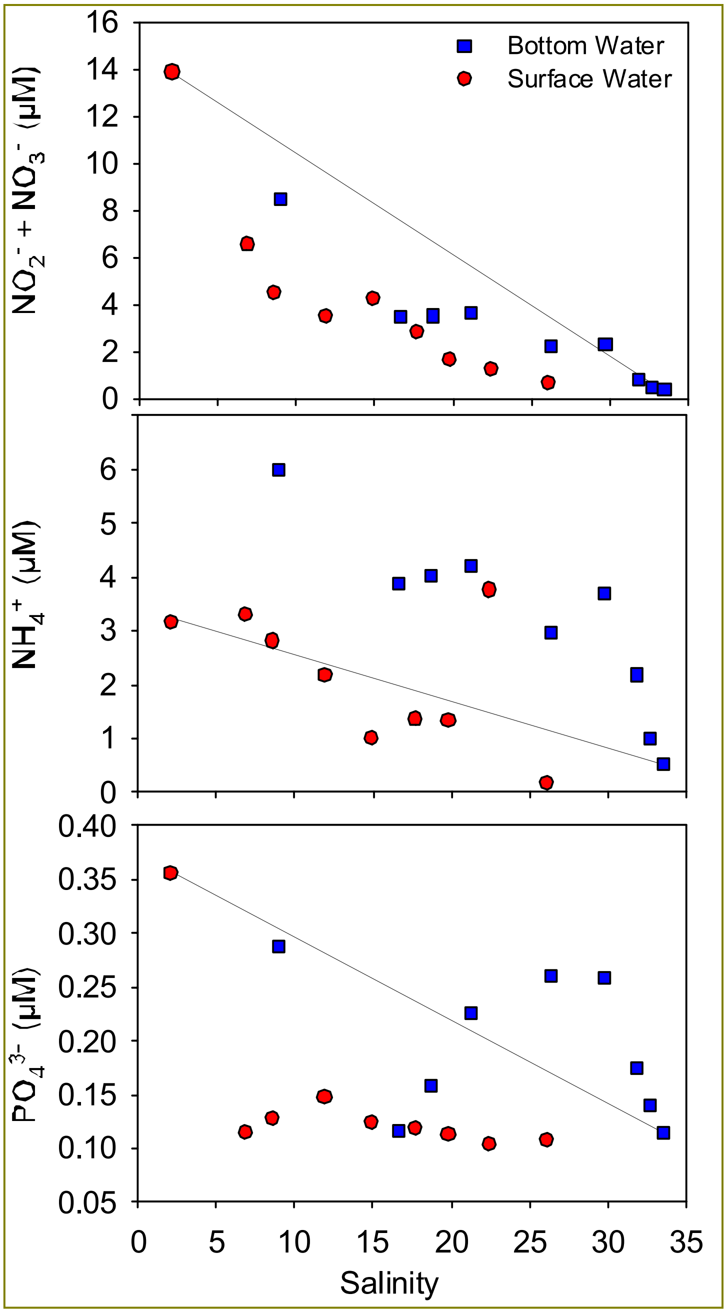

Figure 5.11.3. Mixing diagrams for average salinity and nutrient concentrations along the Escambia River–Pensacola Pass transect during May–September.

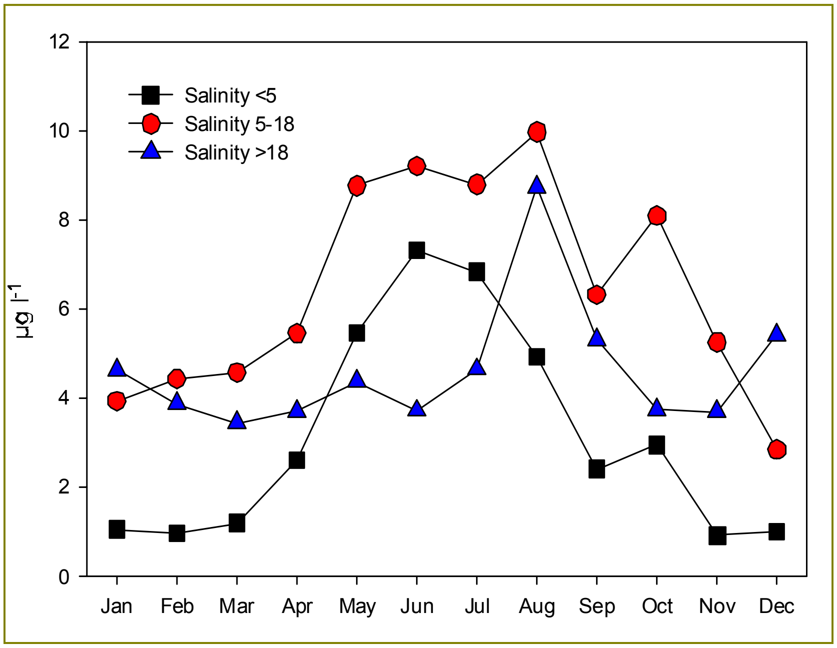

Figure 5.11.4. Monthly median surface water chlorophyll a in three salinity zones of Pensacola Bay during 2002–2004.

one can infer that seasonal maximum production occurs during summer. Annual integrated phytoplankton production has been estimated to be ~230 g C m^{-2} y^{-1}. Bioassay experiments from Pensacola Bay show that phytoplankton growth is usually nutrient limited. P is often the most limiting nutrient for phytoplankton in Escambia Bay, where salinity is low (Murrell et al. 2002). N limitation and co-limitation (N and P) have also been observed, with N limitation more prevalent in the seaward portions of the system (Juhl and Murrell 2008).

Phytoplankton Blooms and Species Composition

A temperature-dependent summer increase in cyanobacterial abundance drives the seasonal distribution of phytoplankton biomass and is the most important feature of phytoplankton community dynamics that emerges from species composition data (Murrell and Lores 2004; Murrell and Caffrey 2005). Among the eukaryotic plankton, diatoms generally account for > 50 percent of total abundance. The most abundant genera are *Thalassiosira* and *Cyclotella*, with unclassified pennate diatoms also very abundant. On one winter cruise, when dinoflagellates accounted for > 70 percent of abundance, the most abundant taxa was *Prorocentrum minimum* (Murrell and Lores 2004). HABs have not emerged as a significant issue in Pensacola Bay. Whereas blooms of the red tide species *Karenia brevis* have occurred almost annually along the central west coast of the Florida peninsula during the past 30 years, blooms have occurred only sporadically along the northern Gulf coast (Tester and Steidinger 1997; Steidinger et al. 1998). Moreover, even though *K. brevis* can be transported into estuaries, blooms do not initiate in inshore waters.

Benthic Primary Producers

The distribution of SAV decreased ~50 percent between the 1960s and 1980s and has remained at relatively low levels ever since (Hagy et al. 2008). Marine SAV, dominated by *Thalassia testudinum* (turtle grass), is extensive only in Santa Rosa Sound, a bar-built embayment adjacent to the Pensacola Bay system. *Vallisneria americana* was abundant near the river mouths in a 1992 survey, but it was found to have declined substantially by 2003. In remaining SAV beds in Pensacola Bay, plants are stunted and sparse compared to the most healthy beds in the region (e.g., St. Joseph's Bay, Florida). Declines in SAV coverage between 1960 and 1980 could be attributed to poor water quality resulting from extreme industrial pollution during that time (Olinger et al. 1975). The reasons for the failure of SAV to recover in recent years have not been determined. Water clarity and nutrient concentrations appear generally favorable for SAV growth. Metabolic stress associated with high sulfide in the sediment pore waters has been implicated as a possible cause for the decline, because sulfide concentrations as high as 5 mM have been measured (USEPA unpublished data) and are especially harmful to plants in combination with high salinity and water temperature (Koch and Erskine 2001). Attached and free-floating macroalgae are present as are microphytobenthos; however, biomass and production is poorly quantified. On the basis of light distributions, it is clear that extensive areas of the sediments frequently receive adequate light to support benthic primary production.

Long-Term Changes

The condition of the Pensacola Bay system, and particularly Escambia Bay, became a matter of public concern as early as the late 1960s (Olinger et al. 1975), earlier than in many estuaries. Initial evidence for a pristine condition was based on biological surveys of stream fauna. By the early 1960s, after significant industrial point-source discharges (especially NH_4^+ and organic matter) were established, similar biological surveys indicated declining health in the river. Reports of fish kills and declining fisheries outputs ultimately led to actions that, by the mid-1970s, greatly reduced the point sources, and in some cases eliminated them completely. Olinger et al. (1975) provides a remarkable early compilation of ecological conditions in the bay, intended principally to document the recovery of the system following reductions in industrial waste loads. No comparable data were collected after 1975 until EPA began bay-wide water quality surveys in 1996 (USEPA 2005). Whereas differences in survey methodology and data reporting mostly preclude quantitative analysis of ecological changes, comparing early data and the conclusions of the early investigators with more recent studies suggests that neither the ecological conditions nor the nature of the major ecological concerns have changed dramatically in the past 30 years. The major ecological concerns in 1975 were (1) bottom-water hypoxia, (2) loss of SAV habitats, (3) toxic contamination, and (4) degradation of biotic communities, including fisheries. Bottom-water hypoxia and loss of SAV habitats, both of which are likely to be related to nutrient enrichment, continue to be concerns for Pensacola Bay (USEPA 2005). Recent studies have shown that toxic contamination (PCBs, for example), once a high-profile issue in Pensacola Bay, remain an important concern (Karouna-Renier et al. 2007).

Summary of Ecosystem Impacts

The health of biological communities in Pensacola Bay, particularly as it relates to nutrient and eutrophication effects, is not well characterized. The condition of benthic macrofaunal communities, which might provide a good indication of the impact of seasonal hypoxia, is not characterized adequately for this purpose. Engle and Summers (1998) evaluated benthic condition from samples collected in April 1992, before the seasonal onset of hypoxia. Livingston (1999) reported benthic biomass and abundance data for Pensacola Bay but averaged data from both summer and other times of the year. Nonetheless, those data show evidence that macrobenthic biomass was reduced in the area affected by hypoxia (Livingston 1999).

Overall, the data suggest that Pensacola Bay is mesotrophic. Nutrient concentrations in the major rivers entering the system are moderate, reflecting high forest cover in most of the watershed. The system is very vulnerable, however, to harmful effects resulting from nutrient enrichment and eutrophication. Therefore, management of both N and P is important to at least maintain current water quality and trophic conditions. Ongoing monitoring of water quality can help to track any changes in eutrophic conditions. Once nutrient management is established to protect the bay as a whole, additional work can assist in better understanding the causes and consequences of the impaired condition of the bay's SAV beds and evaluate options for effective habitat restoration.

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Appendix I:

An Estuarine Nutrient Database

Cynthia Heil and Greg Colianni

The Value of Long-Term Databases

Long-term databases are often essential for identifying trends and environmental change in ecological systems, including estuaries. Such system changes can occur over a variety of time and space scales and can vary with both natural and anthropogenic influences. Despite an essential need for the information, very few long-term databases are available for estuarine and marine ecosystems. Thus, management and policy decisions are frequently based on existing short-term data or models, or both. Long-term monitoring and generation of the associated databases are conducted for many purposes. Four major purposes are

- Characterizing waters and identifying changes or trends in water quality over time.
- Identifying specific existing or emerging water quality problems.
- Gathering information to design specific pollution-prevention or remediation programs.
- Determining whether management actions such as compliance with pollution regulations or implementation of effective pollution control actions are being met.

Long-term monitoring activities might meet several of those purposes at once. Long-term data sets are invaluable for determining the influence of changes in climate, hydrology, land use, and point source loading on associated changes in nutrient levels and primary production in estuaries. Thus, they provide a powerful tool that can be used to separate the influence of natural factors from anthropogenic influences including management actions. In addition, long-term data sets help characterize the range of possible environmental states that might exist in an estuary both spatially and temporally. The data also provide critical information needed to develop potential dose-response relationships that exist between drivers and response variables.

Because estuaries are large, complex systems, it is very difficult if not impossible to conduct controlled experiments to test the response of those systems to increased nutrient loading. As discussed earlier in this document, that complexity is a function of both estuarine typology, land uses, nutrient loading and anthropogenic alterations. One of the few tools that can be used to assist in isolating the influence of anthropogenic changes from natural variability is the use of long-term data sets. The data sets can incorporate periods of minimal man-made disturbance and natural variation in rainfall and river discharge. Hence, the relative influence of river discharge and human sources of nutrients can be separated.

The maintenance of long-term monitoring and associated databases is also essential to inform water quality management. In estuaries experiencing increased urban and industrial development it is essential that management agencies have access to long-term data sets to detect trends in water quality indicators that might require management action. In addition, continued long-term monitoring can assess the effectiveness of various management practices as they are implemented. However, some of the practices are often costly. Long-term monitoring data are one of the few tools that allow organizations to assess the effectiveness of the measures. Finally, long-term monitoring and associated databases can provide for the early detection of problems in an estuary. Long-term monitoring allows for the early detection of problems before they become more significant and costly.

Some examples of long-term databases providing useful information for determining the relative influence of human actions and natural variability on nutrient levels and eutrophication include programs conducted in Chesapeake Bay and Narragansett Bay. Some of the data records for those systems extend back to the 1950s or earlier (Sections 5.2 and 5.4). Long-term studies of the Chesapeake Bay chronicle how changes in land use, fertilizer use and human population have influenced nitrogen concentrations. The studies also document the decline of SAV and the increase of hypoxia. Studies in Narragansett Bay document the long-term increase in water clarity coincident with the decrease of discharge of TSS from sewage plants. The data also document long-term changes in chlorophyll *a* levels associated with climatic fluctuations (Section 5.2).

A database containing existing and historical information can be of considerable use to local and state regulators. Although states and tribes are responsible for developing and implementing water quality standards and criteria, estuaries are not confined to political boundaries. In the cases where an area spans multiple political regions, data from multiple sources (e.g., federal agencies, state agencies, universities) might need to be combined for a holistic view of the estuary in question. When combined, data from multiple sources can fill in data gaps that are lacking in a single monitoring program. To establish trends and patterns that might be useful for identifying affected or reference sites or both, all available data should be considered for an analyses specific to the area (or, in the case where data are not available for an area, specific to that estuary classification) where the criteria are to be set. This report and the accompanying database are valuable as a tool for states to manage their own estuary systems. It might be particularly useful for identifying the parameters, the timespan and the data quality necessary for criteria development by estuary type.

Common Limitations of Available Databases

Only a limited number of the larger, well-studied estuaries have longer-term nutrient databases. In modern (post-1999) databases, metadata are commonly available, and the quality of the data is readily assessable. In historical databases, however, the quality of the data is often difficult to assess. Supplementing a larger database with available data from other smaller estuaries can result in additional limitations, including spatial and temporal data gaps, inconsistency in the parameters collected, differences in collection methods, laboratory methods and unknown quality of the data.

Those caveats should not always preclude the use of the data for criteria development. The data included in the estuary nutrient criteria database has been solicited from reliable sources, such as the Chesapeake Bay Program and Modernized STORET. Discrete variables have been identified, and the units of measure for each have been standardized.

Long-term data sets are lacking or unavailable for many estuarine systems. In many cases, that could be because of the observed lack of routine monitoring by state or federal agencies before a given period. However, in some cases, the data exist but might not be electronically available in a regional, state or national database. For example, data generated by many fisheries agencies, local agencies and university researchers are often not readily or easily available. In some cases, electronic versions of the data do not exist. In addition, the data are available in formats that are not easily imported into standard national database formats. Consequently, major reformatting or data entry efforts might be needed to incorporate the data.

One issue that often arises is that data might have been collected or generated using various collection and analysis techniques and under various degrees of quality assurance and quality control. In some cases, those differences are dealt with implicitly by assigning unique parameter codes to each variable. A good example is chlorophyll *a* measurements conducted by fluorometry or spectrophotometry. In STORET, data generated by the two techniques are given unique parameter codes. In some cases, however, this might not be the case if the data were collected by organizations that do not include that information either implicitly or in their metadata.

Various levels of review are needed to evaluate the appropriateness of data that will be used for evaluating relationships of management actions and associated water quality variables. The validity and usefulness of data depend on the care with which they were collected, analyzed and documented. The five factors that can affect the usefulness of the data for nutrient criteria development and for evaluation of eutrophication include the following:

- **Representativeness**: The sampling program design (when, where and how sampled) should produce samples that are representative or typical of the environment being described.
- **Completeness:** Data sets are often incomplete because of practical problems (e.g., spilled samples, faulty equipment or lost field notebooks). A quality assurance/quality control plan should describe how complete the data set must be to answer the questions posed (with a statistical test of given power and confidence) and the precautions being taken to ensure that completeness. Data collection procedures should document the extent to which those conditions have been met. Incomplete data sets might not invalidate the collected data, but they can reduce the rigor of statistical analyses.
- **Comparability:** To compare data collected under different sampling programs or by different agencies, sampling protocols and analytical methods must demonstrate comparable data. The most efficient way to produce comparable data is to use sampling designs and analytical methods that are widely used and accepted (e.g., Standard Methods for the Examination of Water and Wastewater (APHA, AWWA, WEF, 1998) and EPA methods manuals).
- **Accuracy and Precision**.

Estuarine Nutrient Database Development and Parameter Structure

The database developed for this purpose was based on the Modernized STORET, EPA's National Nutrient Database (NUTDB), and available databases from NOAA and the Gulf of Mexico Program Office. A quality assurance project plan was also developed especially for handling the secondary data sets involved in this process.

Identifying Database Variables and Criteria of Interest

Common variables of interest included variables commonly measured and associated variables required for interpretation. Additional variables of interest were identified, but their availability is unknown because of measurement complexities.

Specific estuarine databases vary greatly in both temporal and spatial coverage, parameterization and quality control. Some of the issues have been directly addressed by governmental agencies when incorporating regional or system data into larger regional or federal databases such as Modernized STORET or NUTDB and are included in associated metadata files. Older data such as is available in Legacy STORET often do not have associated metadata, making evaluation of data quality difficult. Nonetheless, historical data are of immense value in determining long-term trends in systems associated with altered nutrient inputs. Additional complications that could compromise or limit the use of local databases include the electronic platform used and its availability, database quality assurances and metadata, data transferability and the purpose of the specific databases.

The Estuarine Database

The estuary database is in ORACLE and has data from NUTDB, Modernized STORET and other sources. Some querying abilities have been included in the database, searches can be performed on multiple variables, and limited statistical manipulation (e.g., average, mean, minimum, maximum values) are available. Data can be downloaded as comma separated value (CSV) files for further data manipulation, statistical analysis or graphing. ***Information on accessing the database is available from EPA's Office of Water*** (see http://water.epa.gov/scitech/swguidance/waterquality/standards/criteria/aqlife/pollutants/nutrient/database_index.cfm).

Table I.1. Discrete variables for data characterizations and additional variables of interest.

- Chlorophyll *a*, macroalgal biomass
- Color
- Date, time and hour
- Depth
- Dissolved oxygen
- Freshwater discharge
  - Patterns
  - USGS gauging stations
- Harmful algal bloom (HAB) history
  - HPLC
  - Phytoplankton community composition
- Latitude, longitude
- Loadings
  - C, N and P
  - Concentrations, flow
  - Total and relative contributions of tributary loadings
- Particulate C, N, P
- pH
- Salinity
- SiO_4
- Submerged aquatic vegetation (SAV)
  - macroalgae
  - macrobenthos
  - seagrass
- Surface radiation
- Temperature
- TKN, TN, DON, NH_4^+, NO_2^-, NO_3^-
- TOC, DOC
- TP, DOP, TDP, PO_4^{-3}
- Transparency (Secchi), light attenuation
- Turbidity (optical, gravimetric)

Additional variables of interest if available

- Food Web responses
- General description and history of estuary
- Phytoplankton/biodiversity: counts and community composition
- Primary Production
- SAV loss
- Sediment: bulk sediment, pore water profiles
- Watershed features and characterization
  - Information on status of higher trophic levels
- Zooplankton abundance (as dry weight) and composition

Appendix II:

Abbreviations and Glossary

Abbreviations

ANCOVA analysis of covariance

APHA American Public Health Association

B-CART Bayesian Classification and Regression Tree

BOD biological (or biochemical) oxygen demand

CBPO Chesapeake Bay Program Office

CDOM colored dissolved organic matter

CMECS Coastal Marine Ecological Classification Standard

CSV comma separated value

CWA Clean Water Act

DFA discriminant function analysis

DIN dissolved inorganic nitrogen

DO dissolved oxygen

DON dissolved organic nitrogen

DSi dissolved silicate

ECOHAB Ecology and Oceanography of Harmful Algal Blooms

EDA estuarine drainage area

EPA U.S. Environmental Protection Agency

EUNIS European Nature Information System

GEOHAB Global Ecology and Oceanography of Harmful Algal Blooms

GMPO Gulf of Mexico Program Office

HAB harmful algal bloom

HABHRCA Harmful Algal Bloom and Hypoxia Research and Control Act

HPLC high-performance liquid chromatography

IBI index of biotic integrity

IGBP International Geosphere–Biosphere Programme

IMBER Integrated Marine Biogeochemistry and Ecosystem Research Program

IMCRA Interim Marine and Coastal Regionalisation of Australia

INI International Nitrogen Initiative

IOOS Integrated Ocean Observing System

LOICZ Land-Ocean Interaction in the Coastal Zone international project

MERHAB Monitoring and Event Response of Harmful Algal Blooms

MLLW mean lower low water

N nitrogen

NAO North Atlantic Oscillation

NASA National Aeronautics and Space Administration

NEEA National Estuarine Eutrophication Assessment

NEP National Estuary Program

NH_3 ammonia

NH_4^+ ammonium

NO_2^- nitrite

NO_3^- nitrate

NOAA National Oceanic and Atmospheric Administration

NTU nephelometric turbidity units

NUTDB Nutrient Database (EPA)

P phosphorus

PAR photosynthetically active radiation

PO_4^{-3} phosphate

RMSE root mean square error

QAP quality assurance plan

QAPP quality assurance project plan

SAV submerged aquatic vegetation

SCOPE Scientific Committee on Problems of the Environment

SeaWiFS Sea-Viewing Wide Field-of-View Sensor

Si silicon

$Si(OH_4)$ dissolved silica

STORET STorgage and RETrieval

TN total nitrogen

TP total phosphorus

TSS total suspended solids

WEF Water Environment Federation

Glossary

allochthonous. Nutrients or materials whose source is away from the site of interest.

analysis of covariance. A statistical method to examine the effect of a set of variables on a response.

anoxia. A condition in which no oxygen is present.

anthropogenic. Caused or influenced by the actions of humans.

autochthonous. Nutrients of materials whose source is local to the site of interest; generated *in situ*.

autotrophic. An organism or organisms capable of making their own food through photosynthesis.

benthos. A group of organisms, other invertebrates, that live in or on the bottom in aquatic habitats.

bioavailabiliity. Degree of ability to be absorbed and ready to interact in organism metabolism.

biocriteria (biological criteria). Narrative or numeric expressions that describe the desired biological condition of aquatic communities inhabiting particular types of waterbodies and serve as an index of aquatic community health.

biodiversity. The variety and variability among living organisms and the ecological complexes in which they occur. Diversity can be defined as the number of different items and their relative frequencies. For biological diversity, these items are organized at many levels, ranging from complete ecosystems to the biochemical structures that are the molecular basis of heredity. Thus, the term encompasses different ecosystems, species, and genes.

biological (biochemical) oxygen demand (BOD). A measure of the amount of oxygen consumed in the biological processes that break down organic matter in water. Generally, the greater the BOD, the greater the degree of pollution.

biomass. The quantity of living matter, expressed as a concentration or weight per unit area.

biota. The animal and plant life of a given region.

causal variable. When used in relation to development of nutrient criteria, it is those variables that characterize alterations in habitat or in nutrient loading.

chlorophyll *a*. A pigment contained in plants that converts light energy into food. The pigment is often used as a measure of phytoplankton biomass.

chlorophyte. Green algae.

cluster analysis. An exploratory multivariate statistical technique that groups similar entities in an hierarchical structure.

criteria. Descriptive factors that EPA takes into account in setting standards for various pollutants. When issued by EPA, the criteria provide guidance to the states on how to establish water quality standards. Section 304(a)(1) of the Clean Water Act requires criteria for water quality that accurately reflect the latest scientific knowledge. Such criteria are to be based solely on data and scientific judgments on pollutant concentrations and environmental or human health effects. Criteria are developed for the protection of aquatic life and for human health.

criteria exceedance. A measure of the deviation of a specific criterion from the established reference value.

cryptophyte. A class of flagellated algae.

cyanobacteria. Blue-green algae.

dead zone. A layer of water having hypoxic or anoxic conditions.

designated use. An element of a water quality standard that describes an appropriate intended human or aquatic life objective for a body of water. Designated uses may refer to recreation, fishing, water supply and aquatic life habitat

diatom. A class of algae that are silica requiring.

dinoflagellate. A class of flagellated algae, of which most are autotrophic, but many are mixotrophic, and a few are exclusively heterotrophic.

discriminant function analysis. A statistical process used to determine which variables discriminate between two or more naturally occurring groups.

dissolved inorganic nitrogen (DIN). The sum of the concentrations of nitrate, nitrite and ammonium in water.

dissolved inorganic phosphorus. The sum of the concentrations of phosphorus in water; phosphate is the major inorganic form.

dissolved oxygen. A measure of the amount of gaseous oxygen dissolved in water.

dose-response curve. A graphical representation of the relationship between the dose of a stressor and the biological response thereto.

ecosystem based management. A process that integrates ecological, social, and economic goals and recognizes humans as key components of the ecosystem; that considers ecological—not just political—boundaries; that addresses the complexity of natural processes and social systems and uses an adaptive management approach in the face of resulting uncertainties; that engages multiple stakeholders in a collaborative process to define problems and find solutions; that incorporates understanding of ecosystems, processes and how ecosystems respond to environmental perturbations; that is concerned with the ecological integrity of ecosystems and the sustainability of both human and ecological systems (Source: EBM tools network).

environmental sustainability. Long-term maintenance of ecosystem components and functions for future generations.

estuary. A semi-enclosed coastal waterbody with restricted circulation, or coastal marine waters influenced by significant freshwater inflow during at least part of the year.

eutrophic. A condition of an aquatic system in which increased nutrient loading leads to progressively increasing amounts of algal growth and biomass accumulation. When the algae die off and decompose, the amount of dissolved oxygen in the water becomes reduced.

geomorphology. Land forms, their origin and their influence on other processes.

habitat. The place where a population (e.g., human, animal, plant, microorganism) lives and its surroundings, both living and non-living.

habitat indicator. A physical attribute of the environment measured to characterize conditions necessary to support an organism, population, or community in the absence of pollutants; e.g., salinity of estuarine waters or substrate type in streams or lakes.

harmful algae (harmful algal blooms) (HAB). Proliferations of algae that can cause fish kills or seafood contamination through toxins, alter ecosystems in detrimental ways through their biomass accumulation or cause human health problems through toxins that can be carried through the air.

heterotrophic. Referring to organisms that are dependent on organic matter for food.

hypoxic/hypoxia. Waters with dissolved oxygen concentrations of less than 2 parts per million, the level normally taken for organisms to survive and reproduce.

index of biotic integrity (IBI). An integrative expression of the biological condition that is composed of multiple metrics.

irradiance. The amount of light energy received on a unit area per unit time.

light attenuation. The absorption, scattering or reflection of light by water, chlorophyll *a*, dissolved substances or particulate matter.

mesohaline. Pertaining to moderately brackish water with low to middle range salinities (~5 to18).

mesotrophic. A condition of an aquatic system in which the system is in between eutrophic (nutrient enriched) and oligotrophic (nutrient poor) conditions.

metadata. Descriptive information about data in a database, for example, the details on the methods used for quantification of a chemical parameter.

mixotrophy (mixotrophic). Referring to organisms that have the capability to be both autotrophic and heterotrophic, either at the same time or at different times in their life cycle.

National Estuary Program. A program established under the Clean Water Act Amendments of 1987 to develop and implement conservation and management plans for protecting estuaries and restoring and maintaining their chemical, physical, and biological integrity, as well as controlling point and nonpoint pollution sources.

nutrient. Compound dissolved in water that is essential to the growth of plants and animals. The nutrients of major concern for water quality are nitrogen and phosphorus.

nutrient concentration. The concentration of a nutrient form in a waterbody.

nutrient load. The amount of nutrient delivered to a waterbody, accounting for watershed area, land use and other factors.

nutrient pollution. Contamination of water resources by excessive inputs of nutrients. In surface waters, excess algal production is a major concern.

oligohaline. Pertaining to moderately brackish water with low range of salinity (~5).

ontology. The set of specific relationships among elements of a model that dictate where they fall in a typology.

pelagic. Organisms that live in the water column.

pelagophyte. A class of algae that causes brown tides.

photic zone. The layer of water that receives sufficient sunlight to drive photosynthesis, typically taken as > 1 percent of surface irradiance.

photosynthesis. The manufacture by plants of carbohydrates and oxygen from carbon dioxide mediated by chlorophyll in the presence of sunlight.

phytoflagellate. Flagellated algae.

phytoplankton. Microscopic plants, algae, capable of making their own food through photosynthesis.

point source. A stationary location or fixed facility from which pollutants are discharged; any single identifiable source of pollution.

polyhaline. Pertaining to waters with a high salinity range (~18 to 30).

pycnocline. The portion of a water column where density changes rapidly because of salinity and temperature.

raphidophyte. A class of algae.

reference condition. When used in the context of water quality criteria, it is the comprehensive representation of data from several similar, minimally affected *natural* sites on a waterbody or from a similar class of waterbodies.

residence time. The amount of time it takes a parcel of water to move through a system such as an estuary.

response variable. In the context of nutrient criteria, it is those variables that are the direct biological or ecological responses to alterations in habitat or nutrient loading.

quality assurance/quality control. A system of procedures, checks, audits, and corrective actions to ensure that all research design and performance, environmental monitoring and sampling, and other technical and reporting activities are of the highest achievable quality.

salinity. A measure of the salt content of water

Secchi depth. A measure of the turbidity of surface water determined by the depth at which a Secchi disk—a flat black and white disk—cannot be seen from the surface.

stoichiometry. The quantitative relationships between chemical reactants; the ratio of the availability of various nutrients in a water sample.

STORET. EPA's computerized water quality database that includes physical, chemical and biological data measured in waterbodies throughout the United States.

stressors. Physical, chemical, or biological entities that can induce adverse effects on ecosystems or human health.

submerged aquatic vegetation. Rooted vegetation that grows under water in shallow zones where light penetrates.

total suspended solids. Solids in water that can be trapped by a filter (usually with a pore size > 0.45 mm).

trophic level. The level in the food chain in which one group of organisms serves as a source of nutrition for another group of organisms.

turbidity. A measure of the cloudiness of water as a result of suspended sediments, algae or other particles.

typology. An organizing framework that groups elements according to a specific model.

water clarity. Measurement of how far one can see through the water.

water column. Depth integrated open-water environment.

water quality criteria. Numeric or narrative description of a water quality parameter that represents a quality of water that supports a designated use.

water quality standard. A provision of a state or federal law consisting of a designated use or uses for a waterbody and a narrative or quantifiable criterion supportive of the use(s) describing the desired condition of the subject waters to which they apply.

watershed. A region bounded at the periphery by physical barriers that cause water to part and ultimately drain to a body of water.

zooplankton. A community of floating, often microscopic animals that inhabit aquatic environments.

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Figure 2.2. More than 50 percent of the nation's population lives in areas that can be called coastal regions. It is estimated, furthermore, that the coastal population is increasing at more than 3,000 persons per day (NRC 2000), yielding a projected 325 people per square mile in the coastal areas by the year 2015. In addition to permanent residents in coastal areas, these regions draw seasonal visitors, further escalating the stresses from human population.	Reprinted from Crosset, K.M., T.J. Culliton, P.C. Wiley and T.R. Goodspeed. 2004. *Population Trends Along the Coastal United States: 1980-2008*. National Oceanic and Atmospheric Administration. Management and Budget Office, Special Projects. Government document.
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Figure 2.8. The dynamic variability in N and P on both long term (left) and short term (above) scales. Left panel: frequency in which maximum and minimum $NO_2^- + NO_3^-$ and PO_4^{-3} concentrations, by season, were observed in a variety of U.S. estuaries. Data from Frank et al. (2008). Top panel: daily change in $(NO_2^- + NO_3^-):PO_4^{-3}$ for the Pocomoke River, tributary of Chesapeake Bay.	Lower panel reprinted from Glibert, P.M. S. Seitzinger, C.A. Heil, J.M. Burkholder, M.W. Parrow, L.A. Codispoti, and V. Kelly. 2005b. The role of eutrophication in the global proliferation of harmful algal blooms: new perspectives and new approaches *Oceanography* 18 (2): 198-209. With permission of The Oceanography Society.
Figure 2.13. The NOAA 2007 Eutrophication Assessment shows that estuaries of the Mid-Atlantic region are the most eutrophic, and that the outlook for many estuaries, particularly on the east coast is for increasing eutrophication.	Reprinted from Bricker, S.B., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks, and J. Woerner. 2007. *Effects of nutrient enrichment in the nation's estuaries: A decade of change*. NOAA Coastal Ocean Program Decision Analysis Series No. 26. National Center for Coastal Ocean Science, Silver Spring, MD 328 pp. With permission of the authors.

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Figure 4.7. Simple regression relating TN loading rate to TN mass in a selection of Chesapeake Bay tributaries on a yearly basis. A time-series of load-mass data are shown for the Potomac estuary in the inset.	Reprinted from Boynton, W.R., W. M. Kemp. 2008. Nitrogen in estuaries. In D. Capone, E. Carpenter, D. Bronk, M. Mulholland (eds.) *Nitrogen in the Marine Environment*. Academic Press, New York. With permission of the publisher.
Figures 4.10, 4.11. Relationship between annual mean TN concentration (top) and TP (bottom) concentrations and annual mean concentrations of chlorophyll a based on 335 cases from 92 sites worldwide.	Reprinted from Smith, V.H. 2007. Responses of estuarine and coastal marine phytoplankton to nitrogen and phosphorus enrichment. *Limnology and Oceanography* 51:377-384. With permission of American Society of Limnology and Oceanography.
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Figure 4.22. This map shows the integrated habitat health index for the Chesapeake Bay for 2006 using the six metrics described in the text.	Reprinted from Williams, M., R. Llanso, B. Longstaff, and W.C. Dennison. 2007. *Calculating the 2006 Chesapeake Bay Report Card Series*. From www.eco-check.org/reportcard/chesapeake/. With permission of the authors.
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Figure 5.2.7. Gradient in Narragansett Bay chlorophyll *a*. Mean annual concentration at seven stations along the salinity gradient from the head of the bay (Providence River, station 2) to near the mouth of the bay (station 7) during two survey years (1985–1986 and 1986–1987).	Reprinted from Smayda, T.J., and D.G. Borkman. 2007. Nutrients and phytoplankton gradients in Narragansett Bay. In A. Desbonnet and B. Costa-Pierce (eds.) Ecosystem-Sullivan, B.L., D. Van Kueren, and M. Clancy. 2001. Timing and size of blooms of the ctenophore *Mnemiopsis leidyi* in relation to temperature in Narragansett Bay, RI. *Hydrobiologia* 451:113-120. With permission of the publisher.
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Figure 5.5.5. Phytoplankton community dynamics in the Neuse River Estuary. Shown are pre-hurricane (before 1996) and post-hurricane years (after 1996). HPLC-Chemtax derived biomass data of three major phytoplankton groups, chlorophytes, cyanobacteria, and dinoflagellates are shown for ModMon surveys along the length of the estuary.	Reprinted from Paerl, H.W., L.M. Valdes, J.E, Adolf, B.M. Peierls, and L.W. Harding Jr. 2006a. Anthropogenic and climatic influences on the eutrophication of large estuarine ecosystems. *Limnology and Oceanography* 51:448-462. With permission of the American Society of Limnology and Oceanography.

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Figure 5.6.8. Mean nutrient concentrations, μM (± standard error of mean) for Suisun, San Pablo and Central bays versus mean salinity for 1999–2003 for the winter months (December, January, February): (top panel) $Si(OH)_4$, (center) NO_3^-, (bottom) NH_4^+ showing higher concentrations at the head of the bay	Reprinted from Wilkerson, F.P., R.C. Dugdale, V.E. Hogue, and A. Marchi. 2006. Phytoplankton blooms and nitrogen productivity in San Francisco estuary. *Estuaries and Coasts* 29:401-416. With permission of the Coastal and Estuarine Research Federation.
Figure 5.6.9. (Top) Saturated NO_3^- uptake, ρNO_3^-, $\mu M\ h^{-1}$ versus NH_4^+ concentration, μM-N for Suisun, San Pablo and Central Bays and (bottom) ratio of saturated NO_3^- to NH_4^+ uptake, $\rho NO_3^- : \rho NH_4^+$ versus NH_4^+. The points in the graph are shown as bubbles that indicate higher chlorophyll a concentrations with high NO_3^- uptake and low NH_4^+ values.	Reprinted from Dugdale, R.C., F.P. Wilkerson, V.E. Hogue, and A. Marchi. 2007. The role of ammonium and nitrate in spring diatom bloom development in San Francisco Bay. *Estuarine, Coastal and Shelf Science.* 73: 17-29. With permission of the publisher.
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Figure 5.9.6. Mean annual concentrations of NO_3^- + NO_2^- (top) and NH_4^+ (middle) and DON (bottom) for station 5 in the Coastal Bays.	Reprinted from Glibert, P.M., C.E. Wazniak, M. Hall, and B. Sturgis. 2007. Seasonal and interannual trends in nitrogen in Maryland's Coastal Bays and relationships with brown tide. *Ecological Applications* 17(5):S79-S87. With permission of the Ecological Society of America.
Figure 5.9.9. Mean concentration of chlorophyll *a* (μg L^{-1}; panel A) for June through September at station 5 in the Coastal Bays.	Reprinted from Glibert, P.M., C.E. Wazniak, M. Hall, and B. Sturgis. 2007. Seasonal and interannual trends in nitrogen in Maryland's Coastal Bays and relationships with brown tide. *Ecological Applications* 17(5):S79-S87. With permission of the Ecological Society of America.
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Figure 5.9.13. Trend analyses for stations indicated for the Coastal Bays according to data from the late 1980s to the mid 2000s. Data are shown for calculations based on linear analyses (a) and quadratic analyses (b), which account for a change in direction of the trend over time.	Reprinted from Wazniak, C.E., M.R. Hall, T. Carruthers, and R. Sturgis. 2007. Linking water quality to living resources in a mid-Atlantic lagoon system, USA. *Ecological Applications.* 17(5):S64-S78. With permission of the Ecological Society of America.
Figure 5.9.14. Mean concentrations of chlorophyll a (µg L^{-1}; A) and 19'-butanoyloxyfucoxanthin (but-fuco, µg L^{-1}; B) of all samples collected during June to September at station 5 in the Coastal Bays from 1996 to 2004. Lines represent linear regressions.	Reprinted from Glibert, P.M., C.E. Wazniak, M. Hall, and B. Sturgis. 2007. Seasonal and interannual trends in nitrogen in Maryland's Coastal Bays and relationships with brown tide. *Ecological Applications 17(5):*S79-S87. With permission of the Ecological Society of America.
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Figure 5.9.17 Summary of water quality status as based on a synthetic water quality index based on chlorophyll a, total N, total P and DO. The index compares the average values to the biological thresholds indicated on the figure.	Reprinted from Wazniak, C.E., M.R. Hall, T. Carruthers, and R. Sturgis. 2007. Linking water quality to living resources in a mid-Atlantic lagoon system, USA *Ecological Applications*. 17(5):S64-S78. With permission of the Ecological Society of America.
Figure 5.10.4. Long-term monitoring record of monthly turbidity (in NTUs) 1989–2003 at 24 stations in eastern (top), central (middle) and western (bottom) Florida Bay.	Reprinted from Boyer, J.N., J.W. Fourqurean, and R.D. Jones. 1999. Seasonal and long-term trends in the water quality of Florida Bay (1989-1997). *Estuaries* 22(2B):417-430. With permission of the Coastal and Estuarine Research Federation.
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Figure 5.10.9. Long-term standing crop of *Thalassia* at multiple stations in both die-off (dark circles) and non-die-off (light circles) areas. Plants in areas that experienced die-off converged toward the (possible) equilibrium biomass level of non-dieoff areas.	Reprinted from Zieman, J.C., J.W. Fourqurean, and T.A. Frankovich. 1999. Seagrass eie-off in Florida Bay: long-term trends in abundance and growth of turtle grass, *Thalassia testudinum*. *Estuaries* 22:460-470. With permission of the Coastal and Estuarine Research Federation.
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