

# Evaluation of Biocriteria as a Concept, Approach and Tool for Assessing Impacts of Entrainment and Impingement under § 316(b) of the Clean Water Act

*Technical Report*

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# **Evaluation of Biocriteria as a Concept, Approach, and Tool for Assessing Impacts of Impingement and Entrainment Under § 316(b) of the Clean Water Act**

**TR-114007**

Topical Report, August 2000

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This report describes research sponsored by EPRI.

The report is a corporate document that should be cited in the literature in the following manner:

*Evaluation of Biocriteria as a Concept, Approach, and Tool for Assessing Impacts of Impingement and Entrainment Under § 316(b) of the Clean Water Act*, EPRI, Palo Alto, CA: 2000. TR-114007.



# REPORT SUMMARY

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This report documents the current state of development of multimetric bioassessment and biocriteria for assessing the biological integrity of aquatic ecosystems. The report also examines the suitability of multimetric bioassessment for regulating cooling water intake structures (CWIS) under § 316(b) of the Clean Water Act (CWA). This report will be valuable to industry, resource agencies, non-governmental environmental organizations, and universities involved in research, management, and protection of aquatic resources.

## **Background**

Over the past two decades, multimetric indices of biological condition have been widely adopted as part of a suite of tools for comprehensive monitoring of ambient water quality. Increasingly, these indices are being incorporated into regulations in the form of numeric, biological criteria. Forty-two states now use multimetric assessments of biological condition, and an additional six states are developing biocriteria programs. Presently, biocriteria are central to a proposed draft regulatory framework, which the U.S. Environmental Protection Agency (EPA) is developing under a consent decree to implement § 316(b) of the CWA. The proposed framework will address the potential adverse environmental impacts from CWIS through the development of biocriteria programs involving use of multimetric indices and other bioassessment methods. While biocriteria and their assessment tools have been embraced by the regulatory agencies and incorporated within many of their water programs over the past decade, EPA's tentative decision to make biocriteria an integral part of its framework for regulating CWIS under § 316(b) places biocriteria in a new regulatory context. That is, it extends the application of biocriteria principles and methods from small-scale systems, for which biocriteria were originally developed, to larger scale, more open systems where power plants are typically located. Given this proposed application of biocriteria to § 316(b) and the trend towards integration of biocriteria within environmental regulation and management, it is important to critically review the performance of biocriteria. Do the concepts, methods and process underlying biocriteria ensure they are robust and reliable indicators of water body impairment generally and within the context of § 316(b)?

## **Objectives**

To provide an evaluation of strengths, weaknesses, and research needs surrounding multimetric bioassessment and biocriteria, both generally as an assessment approach and specifically in the context of regulation of cooling water intake structures under § 316(b) of the CWA.

## **Approach**

For the first part of the report, the project team developed a primer on biocriteria that provides the reader with an overview of the biocriteria process by outlining and defining key concepts and reviewing EPA's guidance on the steps for implementing a biocriteria program for water

resource management. For the second part of the report, the team identified strengths that can be exploited, potential weaknesses that should be addressed through targeted research, and inherent limitations that must be acknowledged and accommodated as biocriteria and other components of the regulatory framework are developed to implement § 316(b). In separate sections, the team also examined several general issues and other more specific concerns that may arise in each type of water body for which EPA has, or is planning to develop, guidance.

## **Results**

The suitability of biocriteria, as biological benchmarks based on multimetric indices, for regulation of CWIS under § 316(b) depends largely on the specific roles biocriteria will be assigned in the final regulatory framework. Those potential roles range from a general assessment of water body integrity without regard to the source or magnitude of adverse effects to identification of the source of impairment, assessment of the magnitude of impairment attributable to entrainment and impingement, and evaluation of technology options to minimize adverse effects. The effectiveness of a biocriteria approach ultimately depends on an ability to define reference conditions that characterize a state of health for the system. Generic issues arising from this dependence on definable reference conditions are described in detail in the report, as are more specific issues that arise in the context of specific types of water bodies. The report also identifies specific research that should be conducted to address weaknesses of the multimetric approach and more clearly describe inherent limitations that must be acknowledged as regulations are developed.

## **EPRI Perspective**

This report will provide utility managers with an improved understanding of the current strengths, weaknesses, and inherent limitations of the multimetric approach to biological assessment and water quality management. It will be useful to researchers, regulators, and the regulated community by identifying issues that must be addressed as biocriteria programs are developed for application to regulation of cooling water intakes and power plants in general.

## **TR-114007**

### **Keywords**

316 Issues

Ecology

Fisheries

Water Quality Criteria

Environmental Impact assessment



## ACKNOWLEDGMENTS

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This report was prepared under the direction of EPRI Project Manager, Dr. Douglas Dixon. The authors thank him for his support and guidance as well as his thorough review of the report. Thanks also go to the following individuals who reviewed the draft document and provided constructive comments: David Bailey and Jules Loos, Potomac Electric Power Company; Kristy Bulleit, Hunton & Williams (Counsel to UWAG); James Wright and Gary Hickman, Tennessee Valley Authority; James Karr, University of Washington; Larry Olmsted, Duke Power; Rob Reash, American Electric Power; Greg Seegert, EA Engineering, Science & Technology; and Chris Yoder, Ohio EPA.



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# 1

## BIOCRITERIA AND § 316(B): BACKGROUND

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### Introduction

Over the past two decades, multimetric indices of biological condition have been widely adopted as part of a suite of tools for comprehensive monitoring of ambient water quality. Increasingly, these indices are being incorporated into regulations in the form of numeric biological criteria. Forty-two states now use multimetric assessments of biological condition, and an additional six states are developing biocriteria programs (Karr and Chu 1999).

Presently, biocriteria play an important role in a draft regulatory framework that the U.S. Environmental Protection Agency (“EPA” or “Agency”) is developing under a consent decree to implement § 316(b) of the Clean Water Act (the “CWA” or the “Act”). Section 316(b) requires that the location, design, construction, and capacity of cooling water intake structures reflect the best technology available for minimizing adverse environmental impact (33 U.S.C. § 1326(b)).<sup>1</sup> The current draft framework uses biocriteria, including multimetric indices and other bioassessment methods, as a means of assessing the potential for adverse environmental impacts from cooling water intake structures (CWIS).

While biocriteria and related bioassessment tools have been embraced by a number of regulatory agencies and incorporated within many water resource protection programs over the past decade, EPA’s inclusion of biocriteria as an integral part of its draft framework for regulating CWIS under § 316(b) of the CWA places biocriteria in a new regulatory context. That is, EPA’s proposal has the potential to extend the application of multimetric biocriteria principles and methods from small scale systems, for which biocriteria were originally developed, to larger scale, more open systems where power plants typically are located. Furthermore, it implies an ability to assess a stressor that differs from those typically assessed in existing applications.

Given the possible application of biocriteria to the § 316(b) decision-making process, and the general trend towards integration of biocriteria within environmental regulation and management, it is important to critically review the performance of biocriteria, as well as the underlying concepts, methods and processes, to ensure they yield robust and reliable indicators of water body impairment generally and within the context of § 316(b).

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<sup>1</sup> In 1976, EPA promulgated regulations implementing § 316(b). Those regulations were declared invalid on procedural grounds in 1977, and formally withdrawn in 1979 (Dey et al. 2000).

## **Regulatory Background**

The term “biocriteria” refers to the characterization of biological condition or “health” of an ecosystem through use of narrative or numeric standards based on reference conditions of preferred biological condition (Gibson et al. 1996). The regulatory basis for incorporating this concept into federal water programs can be traced to the water quality standards program in § 303(c) of the CWA.

A stated goal of the CWA is to “restore and maintain the chemical, physical, and biological integrity of the nation’s waters” (CWA § 101(a), as amended by the Water Quality Act of 1987). Section 303(c) of the Act charges states with responsibility for establishing ambient water quality standards, including designated uses and criteria, which take into account the use and value of state water for a variety of purposes, and which serve this and other statutory goals. In other words, § 303(c) requires states to set water quality standards that reflect the degree of biological integrity that is desirable and attainable. Biocriteria use “reference conditions” to detect and assess any threat to the designated uses of sites under review (typically sites within the same water body or region). In EPA’s guidance document for application of biocriteria to streams and small rivers, the Agency refers to biocriteria as “the benchmarks for water resource protection and management: they reflect the closest possible attainment of biological integrity” (Gibson et al. 1996).

EPA’s current draft framework for § 316(b), which incorporates biocriteria, represents the Agency’s most recent efforts towards development of regulations for implementing § 316(b). This action was prompted by a consent decree reached in the settlement of a lawsuit brought by a coalition of environmental groups against the Agency. The consent decree established a seven-year schedule during which EPA was required to propose and take final action to address the impacts from CWISs (Nagle and Morgan 1999).

As set forth in the most recent Agency draft, biocriteria standards and assessment techniques would play an integral role in evaluation of potential adverse environmental impacts.<sup>2</sup> Specifically, biocriteria would be used to evaluate the overall condition of the water body and establish the level of intensity of further studies (Nagle and Morgan 1999). Additionally, EPA personnel have suggested in informal discussions that biocriteria might be used in some cases to evaluate potential for impairment of biological condition associated with entrainment and impingement at specific CWISs.

## **Rulemaking Schedule**

EPA, per the consent decree, was originally scheduled to release a draft 316(b) rule in July 1999 with a final rule promulgated in August 2001. EPA, however, brought a motion in federal court seeking an extension of time for issuing § 316(b) regulations under the consent decree. Among the factors cited by EPA as necessitating the extension were: the “extremely complicated”

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<sup>2</sup>“Some of the tools and approaches of particular interest include recent advances in fish population models and new environmental assessment techniques such as biocriteria which the Agency believes will play an increasingly valuable role in evaluating the impacts of CWISs” (Nagle and Morgan 1999).

technical nature of regulating the intake rather than discharge of wastewater and the broad spectrum of affected industries, the environmental impacts of entrainment and impingement associated with CWISs and the “highly site-specific” nature of the environmental impacts and, finally, the “potentially high cost of § 316(b) regulation to the regulated community” (“Declaration of J. Charles Fox [EPA Assistant Administrator for Water] in Support of EPA’s Motion to Modify Consent Decree”, July 29, 1999; hereafter “Fox Declaration”).

In its motion for extension of time, EPA proposed to bifurcate the rulemaking process into two phases. Phase I would address newly constructed facilities employing CWISs and Phase II would address existing facilities already using CWISs. Under this revised schedule, the EPA Administrator would sign the Phase I proposal addressing new facilities on October 5, 2000 and take final action on the Phase I proposal on May 16, 2002. EPA would also propose the Phase II Regulation for existing facilities on May 16, 2002, with final action on that rule to occur on April 1, 2004 (Fox Declaration; Memorandum of Law in Support of EPA’s Motion to Modify the Consent Decree to Extend the Time to Issue Regulations).

On March 27, 2000, the presiding judge issued an opinion and order responding to EPA’s motion to modify the consent decree (Opinion and Order in Cronin v. Browner [S.D.N.Y., No. 93 Civ. 0314, March 27, 2000]). While the Court agreed with EPA’s proposed bifurcation method, it did not find that the proposed modifications to the deadlines were justified when consideration was given to the interest of the public in the prompt issuance of the Regulation. The Court found that the public interest requires that the proposed Phase I Regulation be issued sooner than the deadline proposed by EPA and ordered EPA to promulgate the Phase I proposal by July 20, 2000. The Court concluded that EPA’s proposed schedule for the Phase II Regulation should also be shortened and ordered EPA to issue the proposed Phase II Regulation by July 20, 2001. Either of these deadlines may be modified by the parties as part of a further settlement. The Court refused to specify the dates for final action on the Phase I and Phase II Regulations, citing the complexity of the issues involved. The Court opined that the parties should continue to negotiate with the purpose of reaching settlement on those deadlines before July 20, 2000.

In its Order, the Court states that it is prepared to appoint a special master if the parties have not agreed on a schedule for final action on the Phase I and Phase II Regulations by July 20, 2000. The special master would have a mandate to: (1) enforce and monitor compliance with the consent decree, and (2) provide a forum for discussion between the parties regarding settlement of the deadlines for promulgation of the Phase I and Phase II Regulations.

### **EPA Draft Framework for § 316(b)**

EPA’s draft regulatory framework for implementing § 316(b) sets forth a three-tiered decision process for evaluating the potential adverse environmental impacts from the operation of CWISs. At present, the application of biocriteria is identified as a critical component of Tier 2 of this framework.

In the first tier, information is collected on facility performance and site characteristics to determine if the CWIS meets operational criteria designated by EPA for minimizing the potential for adverse environmental impacts. If the CWIS meets Tier 1 criteria, it is removed from further consideration. Those CWIS not satisfying these criteria will proceed to a Tier 2 analysis.

In Tier 2, the site is evaluated on the basis of the source water body's designated uses and biological status. Biocriteria play a role here. Specifically, biocriteria are intended to aid in the assessment of impairment for the source water body, provide the basis for determining the scope of study needed in Tier 3, and contribute data required for Tier 3 investigations.

Tier 3 involves the further investigation and analysis of a particular CWIS's contribution to existing impairment at the site. Biological data collected as part of the bioassessment process in Tier 2 are expected to contribute to the Tier 3 data requirements.

This constitutes a brief overview of the draft framework. The requirements of the three-tiered decision framework and the specific role of biocriteria are discussed in detail in Chapter 3 of this report.

## **Rationale and Scope of This Study**

When environmental regulation falls short of its intended purpose, significant ecological and economic consequences can result. As has occurred since enactment of the CWA, ecological entities and processes not covered by narrowly focused chemical or effluent toxicity assessments can ultimately experience overall degradation. Such negative results for non-assessed components of ecosystems and the resource as a whole can occur even as large sums of money are spent to address factors, such as chemical concentrations, that are within the assessment framework (Karr and Chu 1999). Avoiding this undesirable outcome provides a strong motivation for more comprehensive water quality assessment (Karr 1991, Karr and Chu 1999), but should also serve as a constant reminder of the need for careful consideration of potential shortcomings in any assessment frameworks used for regulation.

Theoretically, biocriteria provide a more comprehensive approach to water quality assessment by evaluating impairment of aquatic systems using multiple measures of biological condition. Yet, as in any assessment framework used for regulation, potential limitations do exist. Given the expanding role of biocriteria in environmental management and regulation, and the likelihood of their application to new categories of water bodies which present different spatial scales and ecological relationships, it is prudent to critically examine this approach.

The primary purpose of this report is to provide an evaluation of biocriteria, both conceptually and as assessment tools. The first part of the report presents a primer on biocriteria. The primer provides the reader with an overview of the biocriteria process by outlining and defining key concepts and reviewing EPA's guidance on the steps for implementing a biocriteria program for water resource management. The second part of the report highlights potential weaknesses that should be addressed through targeted research, and inherent limitations that must be acknowledged and accommodated as biocriteria and other components of the regulatory framework are developed to implement § 316(b).

Development and application of biocriteria for water resource protection encompasses science, regulation, and policy. While the technical aspects of the biocriteria approach are the principal subject of this report, they cannot be considered in isolation. Because multimetric bioassessment constitutes an approach for meeting regulatory requirements and achieving policy goals, biocriteria must also be evaluated against these objectives. Consequently, regulation and policy

are addressed in this report to the extent they have technical implications. This review is based on existing EPA guidance, experience with biocriteria on the part of various states and other governmental organizations, and multimetric bioassessment methods published in the peer reviewed literature.





# 2

## PRIMER ON BIOCRITERIA APPROACH

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This chapter provides a primer on the biocriteria approach. Specifically, this chapter:

1. Introduces key concepts relevant to biocriteria, including biological integrity, reference conditions, bioassessment, and the multimetric approach,
2. Presents an historical and regulatory context for the adoption of biocriteria in EPA's water programs, and
3. Provides an overview of EPA's guidance on biocriteria by outlining the basic steps for developing and applying biocriteria for water resource management.

This primer is based on insights and successes associated with developing biocriteria for streams and small rivers; however, translation of the approach to larger water bodies where power plants more typically are located will present many new challenges. Technical issues relevant to application of the biocriteria approach to CWISs and § 316(b) are discussed in Chapter 3.

The terms biocriteria, bioassessment, and biosurvey are frequently interchanged and often confused in definition and use. This report focuses on definitions and use that are consistent with EPA guidelines (Gibson et al. 1996):

- ***Biocriteria:*** Numeric values or narrative expressions that describe the preferred biological condition of aquatic communities based on designated reference sites. The criteria act as thresholds or endpoints for judging whether a waterbody's designated use, as established by the states and tribes, is impaired.
- ***Bioassessment:*** An evaluation of the biological condition of a waterbody that uses biosurveys and other direct measurements of resident biota in surface water, including benthic environments. Bioassessment includes the process of collecting (biosurvey) and processing representative samples of a resident aquatic community to determine the community structure and function. Multiple field measures designed to determine community structure and function are formally combined as a multimetric index. Such measures are usually derived from a single sample collected in the field. A multimetric index value (numeric or narrative) that is determined by the states and tribes as the preferred biological condition is formalized as the biocriterion.

The ***biocriteria approach*** incorporates biocriteria development and use, bioassessment and its inclusive biosurveys and multimetric indices, determination of reference conditions, and selection of reference sites. Determining biologically meaningful and technically/socially defensible regulatory criteria and proper use and integration of the numerous biological survey

tools, each with different strengths, weaknesses, and underlying assumptions, is the resource management challenge of the biocriteria approach.

## Key Concepts

### ***Biological Integrity as an Organizing Principle***

Maintaining and protecting the biological integrity of the nation's waters is an explicit goal of the CWA. Regulatory agencies have responded to this goal by making biological integrity an integral component of federal and state water programs.

Understanding the concept of biological integrity and being able to define it for the regulated system is critical to the success of any biocriteria program. Attempts to define the concept of biological integrity, or *biotic integrity*, have yet to produce one universally accepted definition (Karr and Chu 1999). EPA (USEPA 1990) defines biotic integrity as "the condition of the aquatic community inhabiting unimpaired water bodies of a specified habitat as measured by community structure and function." As more broadly defined by Karr and Dudley (1981), biotic integrity is "the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region."

Based on these definitions, Karr and Chu (1999) suggest that biotic integrity is present when evolutionary and biogeographic processes alone structure biological communities, that is, where there is no anthropogenic influence. However, this definition seems unworkable in that it is difficult to find a biological community that is free from human influence. For this reason, sites exhibiting high biological integrity are often described as "minimally impaired" (Gibson et al. 1996).

An effective biocriteria program is designed to both characterize and protect the designated use of a given water body or site. This is achieved, in part, through the establishment of reference conditions that characterize the minimally impaired or attainable biological condition for representative sites within an aquatic system. The reference conditions, then, operationally define biological integrity for those sites and become the standard against which other similar sites are compared and regulated. Consequently, biological integrity becomes both an objective and organizing principle for structuring a biocriteria approach. Biological integrity defines one end of the scale used for assessment; however, actual numeric biological criteria often reflect conditions more altered by human activity and more consistent with society's near-term goals (i.e., designated uses) for a given waterbody.

In order to develop biocriteria based on the biological integrity of a minimally impaired system, the biological integrity of that system must first be defined. This is achieved by identifying the condition of the biota for a given water body, which requires an understanding of elements found at the molecular (e.g., genetics), individual (e.g., physiology), population (e.g., functional role of a species, abundance of a species), community (e.g., competition, predation) and landscape (e.g., emigration, juvenile recruitment from source populations) levels of biological organization (Karr 1990).

If these elements are not well described, a preliminary study of them may be required in order to measure the attributes of the minimally impaired system and develop metrics for assessing impairment in similar systems. Once delineated, these measures of biological integrity are used, in conjunction with other regulatory objectives and expectations for the water resource, to devise biocriteria standards that will protect the designated uses of the resource. Biocriteria, however, do not directly protect biological integrity. Rather, they serve as benchmarks for determining whether corrective action is needed. Technical issues associated with biological integrity and “minimally impaired” systems and reference sites are discussed in detail in Chapter 3.

### **Reference Conditions**

Biocriteria are based on the premise that habitats minimally affected by human activities have biological structure and function that is characteristic of a healthy ecosystem. Reference conditions define the “healthy” system by characterizing, through measurement, the condition of biological communities at minimally impaired site(s) within a water body or region that reflect the desired ecological condition (Davis et al. 1996, Gibson et al. 1996). These characterizations must be both representative and attainable for that type of aquatic system because they will serve as a basis for judging the degree of impairment at other sites within the same region; however, actual biocriteria for specific water bodies won’t necessarily conform to the reference condition. Increasing degrees of human impact usually result in a corresponding increase in departure from reference conditions.

Reference conditions are defined using metrics, which are measurable attributes of biological condition that change in predictable ways in response to human influence (Gibson et al. 1996). Several methods and information sources can be used singly or in combination to establish reference conditions. These include use of reference sites, historical data, simulation models, and expert consensus. EPA biocriteria guidance documents (e.g., Gibson et al. 1996, USEPA 1998a), discuss establishment of reference conditions and the strengths and weaknesses of each of these methods. The preferred method for establishing reference conditions is through the use of data collected at reference sites (i.e., minimally impaired sites).

### **Bioassessment and a Multimetric Approach**

The use of living organisms to evaluate the health of an aquatic system is known as biological assessment or *bioassessment*. The evaluation of aquatic “health” through the use of biological indicators or conditions reflects a break with past regulatory approaches, where assessment of aquatic systems was based primarily on physical and chemical information. Physical and chemical attributes of aquatic systems were measured and compared to pre-determined standards. If one or more parameters failed to meet or exceeded these standards, the water body was considered impaired.

Scientists and regulatory agencies have begun to place greater emphasis on developing biocriteria and the use of bioassessment in water resource management programs. Bioassessment works in similar fashion to physical and chemical protocols, in that measurements of biological attributes of a particular system are compared to pre-determined standards (biocriteria) to

evaluate the condition of a water body. Bioassessment methods, then, are the “tools” for assessing aquatic system health.

One of the bioassessment methods most commonly used in biocriteria programs is that of multiple metrics or *multimetrics*. A multimetric approach involves the measurement of multiple biological metrics, as opposed to a single metric, of an aquatic system to assess its health. The perceived advantage of a multimetric approach to bioassessment is that it produces a more robust and representative measure of system condition because it not only relies on an array of metrics, but also uses this array to compensate for the relative strengths and weaknesses of individual metrics in responding to different stressors.

The selection of metrics will be driven by such considerations as program objectives, geographic region, biological characteristics within the reference area (the type of system being assessed and its biota), and potential anthropogenic influences that must be detected. Candidate metrics are identified, evaluated and calibrated. Those that respond in a predictable, consistent fashion are aggregated into a *multimetric index*. This index is calculated by scoring each metric according to its deviation from its expected value, which is defined by the metric value of the reference condition. The sum of the metric scores (the index) is compared to the index derived from reference conditions to evaluate the degree of impairment.

## **The Basis for Adoption of a Biocriteria Approach in Water Resource Programs**

EPA has broadly embraced a biocriteria approach within its water programs. Not only have biocriteria been recommended for use by states for protection of streams, small rivers, lakes, and reservoirs, but EPA is now developing guidance to promote their use in other, larger systems such as large rivers, estuaries, and coastal marine waters. Many state agencies have responded by incorporating biocriteria language and bioassessment techniques in their water quality standards programs (Figure 2-1; Davis et al. 1996).

This trend reflects the Agency view that biological criteria provide a more accurate and robust evaluation of aquatic system health than does sole reliance on physical and chemical water quality criteria. Examination of the historical context from which biocriteria emerged and the perceived advantages that a multimetric bioassessment approach presents provide an understanding of EPA’s aggressive promotion of this approach.

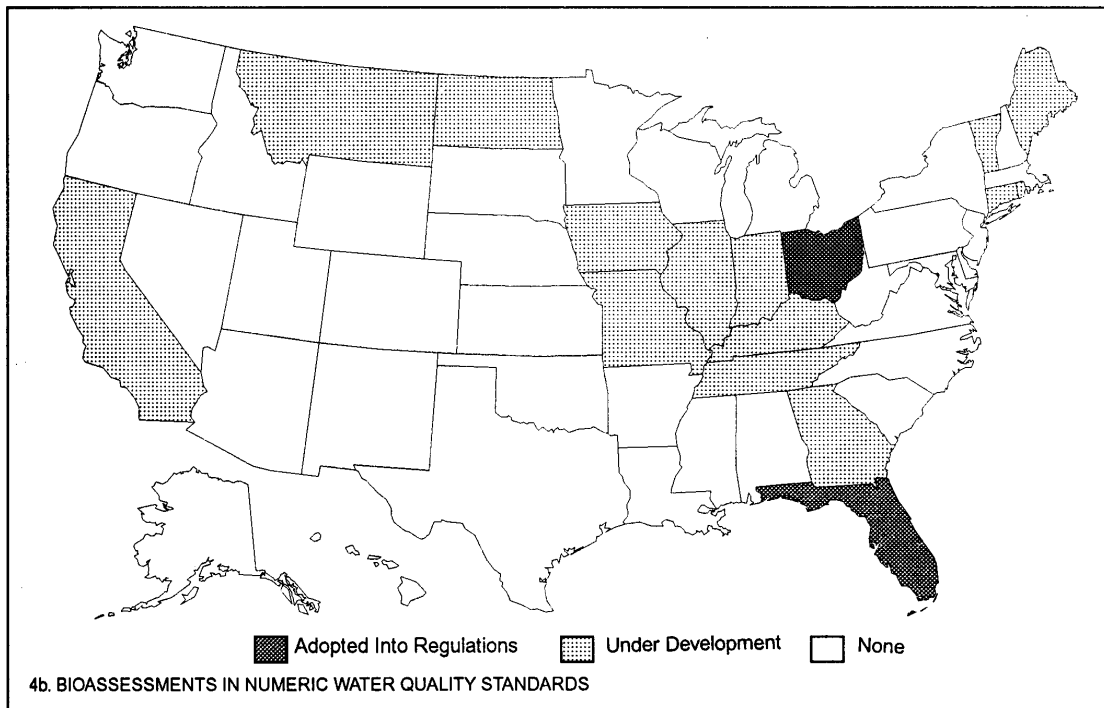
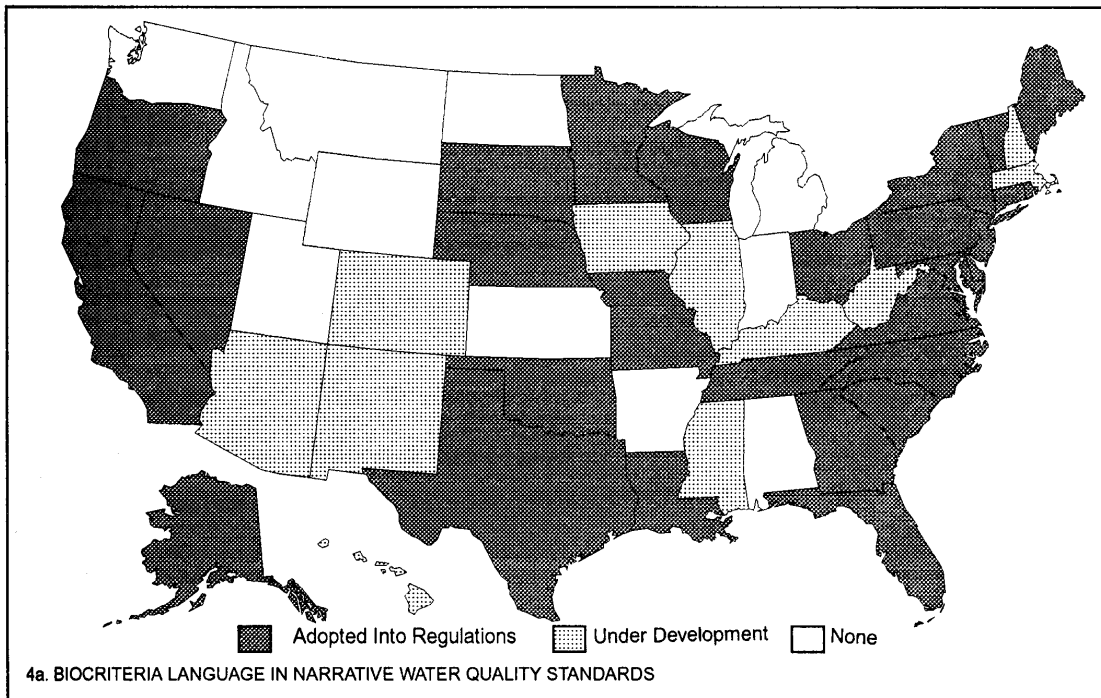


Figure 2-1  
State Biocriteria and Bioassessment Programs. Source: Davis et al. 1996

## **The Historical Perspective**

Principles of biological criteria have been applied, in an informal way, for centuries. In medieval times, kings relied on slaves to taste food and wine for poisons. If the slave did not become sick or die, the food and drink were considered safe for consumption. Similarly, during the 19<sup>th</sup> century, the coal mining industry relied on canaries placed in coal mines to monitor air quality. When a canary became ill or died, the conditions were considered unsafe and the mine was evacuated (Cairns and Pratt 1993).

The application of bioassessment to aquatic habitats can be traced to the development of the Saprobic System in early 20<sup>th</sup> century Europe (Kolkwitz and Marsson 1908a,b). This system relied on the presence or absence of microorganisms associated with plankton and periphyton (algae attached to bottom substrate) communities to indicate the presence of untreated effluents in urban areas (Karr 1991, Metcalfe 1989). From the Saprobic System, the concept of “indicator” organisms was created. **Indicator organisms** are organisms whose presence, absence, abundance, condition, or behavior provides information on the health of an ecological system.

The development of bioassessment protocols continued through the middle and latter half of the century. One of the earliest and most widely adopted protocols was Patrick’s (1949) use of diatoms (a ubiquitous group of algae) as indicators of water quality.

Over time, greater emphasis was placed on the development and use of diversity indices in bioassessment. These indices are designed to characterize the number and relative abundance of species. While diversity indices provide useful insight into some aspects of system health, many scientists questioned their predictive capability when used as the sole basis for bioassessment. Consequently, diversity indices were replaced with a broader based, integrative approach consisting of the contemporary **multimetric indices**. Multimetric indices combine multiple attributes (metrics) of the resident biota, such as the number of species (species richness), relative abundance of organisms that are pollution tolerant or intolerant, and incidence of external anomalies such as tumors or lesions, to provide a more comprehensive and robust characterization of the biotic integrity or health of the system (Tolkamp 1985). The development of biotic indices has been part of a larger movement toward more comprehensive assessment of water body conditions. A critical advance in bioassessment methods occurred with the publication of the “Index of Biotic Integrity” or the IBI (Karr 1981). The IBI formalized the concept of using multiple metrics or a multimetric approach in bioassessment. The original IBI was based on twelve biological metrics, which were grouped into such categories as trophic structure and species richness, in an effort to convey the breadth of the biological integrity of a particular water body.

Many of the advances in bioassessment methodology that have occurred during most of the past century have not yet been incorporated into water quality regulation. Water quality regulation has generally focused on point source pollution (i.e., easily identified stationary sources of pollution), human health, and clean drinking water (Karr 1991, Karr and Chu 1999). This emphasis, in turn, led to the dominance of chemical rather than biological or ecological assessments of aquatic systems in North America for much of this century (Cairns and Pratt 1993).

Yet, a lack of improvement in biological communities within some regions, even though chemical standards have been met, underscores the inadequacy of relying solely on chemical and physical assessments of water resources (Angermeier and Karr 1986, Karr 1991). This lack of improvement, in conjunction with increased environmental awareness and changing attitudes about water quality as reflected in the Water Pollution Control Act of 1972 and recent amendments, has helped redirect the focus of regulatory agencies to the biological health of aquatic systems. The result has been a large increase in the use of biocriteria to assess aquatic system health.

### ***The Advantages of a Biocriteria Approach***

The advantages of biocriteria compared to present assessment approaches have made them attractive assessment tools for EPA. Relative to physical and chemical criteria, biocriteria potentially provide a more comprehensive and sensitive set of indicators of water quality and ecological integrity (Karr 1991, Yoder 1995). Chemical and physical assessment techniques provide “snapshots” of a given system, as they only describe conditions at the time of sampling. In contrast, the biological communities on which biocriteria are based may integrate transient physical and chemical conditions that are often difficult to characterize with chemical and physical measurements.

Assessment methods based on *in situ* biological assemblages are well suited to detect cumulative effects and non-point source problems and other subtle or diffuse, but poorly defined impacts. By contrast, intermittent physical and chemical evaluations do not account for many stressors (e.g. nutrient enrichment, introduced species, and sedimentation) and may not detect acute disturbances that are not present at the time of sampling (Resh et al. 1996).

Biocriteria also provide a more direct evaluation of the biological integrity of a water body, and, thus, have the potential to yield more robust assessments for management and regulation. Other methods used to protect water quality under the CWA require extrapolation from effects on individuals (often in the laboratory) to effects on populations and ecosystems. For example, chemical water quality criteria generally rely on extrapolation from laboratory toxicity studies to population- and community-level effects in the field. Since bioassessment methods do not rely on conservative assumptions to extrapolate from individuals to higher levels of organization, they have the potential to provide a more realistic assessment of the ecological integrity of the water body.

Biocriteria tend to be more easily understood by the public than chemical or physical criteria (e.g., relative abundance of trout in a mountain stream versus the acid-neutralizing capacity of the same mountain stream). This type of understanding facilitates public awareness and involvement in water resource management plans. Finally, the addition of biological criteria to current surface water management plans is viewed by some (Karr and Chu 1999) as progress towards the legislative goal of restoring and maintaining “the chemical, physical and *biological* integrity of the nation’s waters.”

## **The Development and Application of Biocriteria**

This section provides an overview of EPA's guidance, outlining the key components necessary for developing and implementing a biocriteria program. EPA's policy has been to provide to the states both general guidance and guidance specific to various water body types. This guidance is in various stages of development and is expected to be completed within the next several years. Once equipped with this guidance, it is incumbent upon the states to develop and implement biocriteria programs for use within their borders. The states, in turn, may call upon the regulated community to provide information needed to develop and implement biocriteria relevant to their permits.

The following discussion is based on three EPA guidance documents: 1) "Biological Criteria: National Program Guidance for Surface Waters" (USEPA 1990), 2) "Biological Criteria: Technical Guidance for Streams and Small Rivers" (Gibson et al. 1996) and 3) "Lake and Reservoir Bioassessment and Biocriteria" (USEPA 1998a). Because the biocriteria approach was initially developed for use in streams and small rivers, the biocriteria framework for those water body types represents the core approach to biocriteria development. This overview of the biocriteria process is based largely on that framework. It does not reflect the challenges posed by extension of the approach to larger water bodies. The challenges posed by application of the approach to larger water bodies and regulation of CWISs are discussed in subsequent chapters.

Discussion of the development of a biocriteria program has been distilled here to its most fundamental elements. For more thorough analysis the reader is referred directly to the guidance documents.

A biocriteria program may be broken down into several fundamental steps:

1. Formulate Objectives
2. Develop the Biocriteria Program: Define Biological Integrity Objectives for the Subject System
3. Establish and Validate Reference Conditions
4. Evaluate Metrics and Develop a Multimetric Index
5. Develop Sampling Protocols: Biosurveys of Target Organisms/Assemblages and General Sampling Considerations
6. Develop and Implement Quality Assurance Plans for Bioassessment Program
7. Establish Biocriteria
8. Assess Water Body Impairment
9. Diagnose Cause(s) of Impairment and Remediate



While the discussion of biocriteria components occurs sequentially as steps, in practice many of these components may occur concurrently or in a varied sequence within a particular biocriteria program.

### **1. Formulating Objectives**

The first phase of a biocriteria program is the identification of the general program objectives. These objectives usually pertain to determining and assessing aquatic life use guidelines, identifying high quality systems in need of protection, determining sources of impairment, monitoring the success of water management programs, and promoting an antidegradation policy (Gibson et al. 1996). All aspects of biocriteria development and application, including protocols and decisions on technical issues, will be based on these objectives.

### **2. Developing the Biocriteria Program: Defining the Biological Integrity Objectives for the Subject System**

After identifying these objectives, it is necessary to define the biological integrity of the given site or water body to be protected. This definition of biological integrity, in conjunction with the objectives formulated above, provides the basis for narrative or numerical biocriteria standards (USEPA 1998a). Because the biological integrity of a given system is most commonly defined through the designation of reference conditions, this step also initiates the process for establishing reference conditions.

If the components of a system's biotic integrity are not well understood, then biosurveys must be designed and sampling protocols developed and validated so that the necessary information may be collected on the relevant molecular, individual, population, community, and landscape levels of biological organization (Karr 1990).

It may also be useful at this time to investigate other state's biocriteria programs. While these programs may not share the same objectives, the underlying characteristics of an effective biocriteria framework will be similar (Gibson et al. 1996).

### **3. Establishing Reference Conditions**

Establishing reference conditions is a critical part of the development of a biocriteria program. It not only defines the biotic integrity for that system but provides the basis for making comparisons to potentially affected sites and, ultimately, for detecting impairment. Because of the impossibility of finding pristine sites, managers must instead rely on the use of representative conditions at sites that exhibit minimal disturbance (i.e., human interference) relative to the overall region of study (Gibson et al. 1996).

Reference conditions may be established by using historical data, predictive models, minimally impaired reference sites, and expert consensus. These approaches can be used in combination. Criteria for establishing reference conditions outlined below reflect the dominance of streams and small rivers in existing biocriteria programs and EPA guidance.

Historical data, that is data collected from past surveys of a particular region, can be useful in establishing reference conditions by providing insight into the type of biological community that can be attained for a particular water body type. These kinds of data may be found at museums, universities, and state agencies. Paleolimnological methods, involving use of archeological information such as sediment records, may also be used to determine the composition of past communities (Charles et al. 1994). When using historical data, though, extreme caution must be exercised to ensure that the data were not collected from an impaired site, or involved the use of different sampling protocols, equipment, and survey objectives (Gibson et al. 1996). Care must also be exercised to ensure that reference conditions derived from historical information are, in fact, attainable.

In recent years, predictive or “simulation” models have become popular assessment tools. A well-built model of an aquatic system can enable scientists to simulate components of the natural biological community based on the natural characteristics of an area (e.g., topography, soils, and climate). Unfortunately, the use of predictive models for development of biocriteria in aquatic systems is limited at this time by the ongoing process of model development and validation (Gibson et al. 1996). The successful use of these models in other areas of ecology, however, suggests that they may be an important component of future biocriteria development.

The most widely used method for establishing reference conditions relies upon reference sites. Reference sites are minimally impaired sites (i.e., sites that experience minimal human interference), similar in location and habitat characteristics to the area of interest (Gibson et al. 1996). The site of interest, or test site, is the location within a water body where biological condition is being investigated for potential adverse impact from human activity or influence. Use of reference sites is based on the premise that similarity in location and habitat characteristics of the reference and test sites should result in similar biological communities.

The two primary considerations in selecting reference sites are representativeness and minimal level of disturbance (i.e., human interference). Determining representativeness, that is whether the reference sites exhibit conditions similar to those of other sites in the same region, requires examination of habitat characteristics. These characteristics for ideal reference sites include representative: riparian vegetation, diversity of substrate materials, channel structures, natural hydrograph, natural color and odor of water, and animals that have some dependence on the aquatic system(s) (Gibson et al. 1996).

The location or geographic scope of reference sites must also be well defined in order to permit valid comparisons to test sites. Managers will need to conduct a preliminary resource assessment to determine the feasibility of using reference sites. If no acceptable sites are found, then other methods for determining reference conditions must be used.

Once candidate reference sites are identified, resource managers must determine which ones will constitute the “minimally impaired” reference sites. This task must be done carefully, because these site conditions will become the biocriteria benchmark for determining impairment. Gibson et al. (1996) suggests the use of several criteria for streams and small rivers. These criteria are, in order of importance:

1. All drainage within the ecoregion of interest
2. No upstream impoundments
3. No known discharge or contaminants in place
4. No known spills or pollution incidents
5. Low human population density
6. Low agricultural activity
7. Low road density
8. Drainage on public lands
9. Minimal non-point source pollution problems
10. No known intensive fish stocking or other intrusive management activity.

The increasing difficulty in finding “minimally impaired” reference sites in close proximity to test sites has spurred the development of more spatially robust site-classification schemes. These classification schemes attempt to group sites based on environmental and ecological similarities (Karr and Chu 1999). This is done *a priori* based on ecological theory and known environmental conditions, and *a posteriori* based on collected data (Gibson et al. 1996). Reference sites may then be designated for sites within a specific grouping.

One of the more common *a priori* classification schemes used in aquatic bioassessment is that of *ecoregions* (Figure 2-2). In theory, ecoregions are ecologically homogenous geographic regions with similar terrestrial vegetation, topography, climate, and geology (Gibson et al. 1996, Warry and Hanau 1993). The justification for using ecoregions is analogous to that of reference sites—similar water bodies within ecoregions should have similar biological communities (Hughes et al. 1986). In fact, assemblages of aquatic organisms in streams and small rivers have been shown to be closely associated with ecoregions (Gibson et al. 1996).

This grouping of ecologically similar sites within a prescribed geographic region permits the designation of *regional reference sites*. Use of regional reference sites allows the establishment of reference conditions for those test sites that lack more local comparisons of attainable biological condition. Regional reference conditions also provide for more efficient management and regulation by encouraging the pooling of resources between states which share the same ecoregion (Gibson et al. 1996).



**Figure 2-2**  
**Level III Ecoregions of the Coterminous United States. Source: USEPA 1995**

Regional reference sites also may be established for *watersheds*. Watersheds comprise hydrologically linked networks of stream reaches that form a nested hierarchy. Stream networks provide means of dispersal and colonization by aquatic organisms. Because the distribution of many fish species follows major drainage boundaries, watersheds are ideally suited for developing regional reference sites. When one considers that a regional-scale watershed may span several ecoregions, it appears that ecoregions, themselves, may also be appropriate units for selecting reference sites and developing regional reference conditions.

Not only may reference sites be regional in scope, but they may also be *site-specific*. Use of a site-specific reference condition requires the availability of comparable habitat within the same water body for both the reference and impacted area. The need for site-specific reference conditions arises where regional and more local reference conditions cannot be developed because adequate reference sites are lacking. This occurs most frequently when evaluating the impacts from a point of influence on a water body (e.g., point discharge) and either gradients in contaminant concentration occur (*the near field-far field approach*) or the particular water body has a strong directional flow (*the upstream-downstream approach*) (USEPA 1990).

The near field-far field approach is based on the dose-response principle of toxicology in which the magnitude of the biological effect is positively related to the magnitude of the dose. Organisms in the far field are presumably exposed to lower doses than organisms in the near field because of dilution effects. Near-field and far-field sample sites are chosen so that they are similar in all other ecologically important respects. Relatively poor biological status in the near field is presumed indicative of an adverse effect attributable to the near-field source of stress. Gibson (1995) demonstrated the utility of the near field-far field approach in a biological assessment of a near-coastal wastewater outfall.

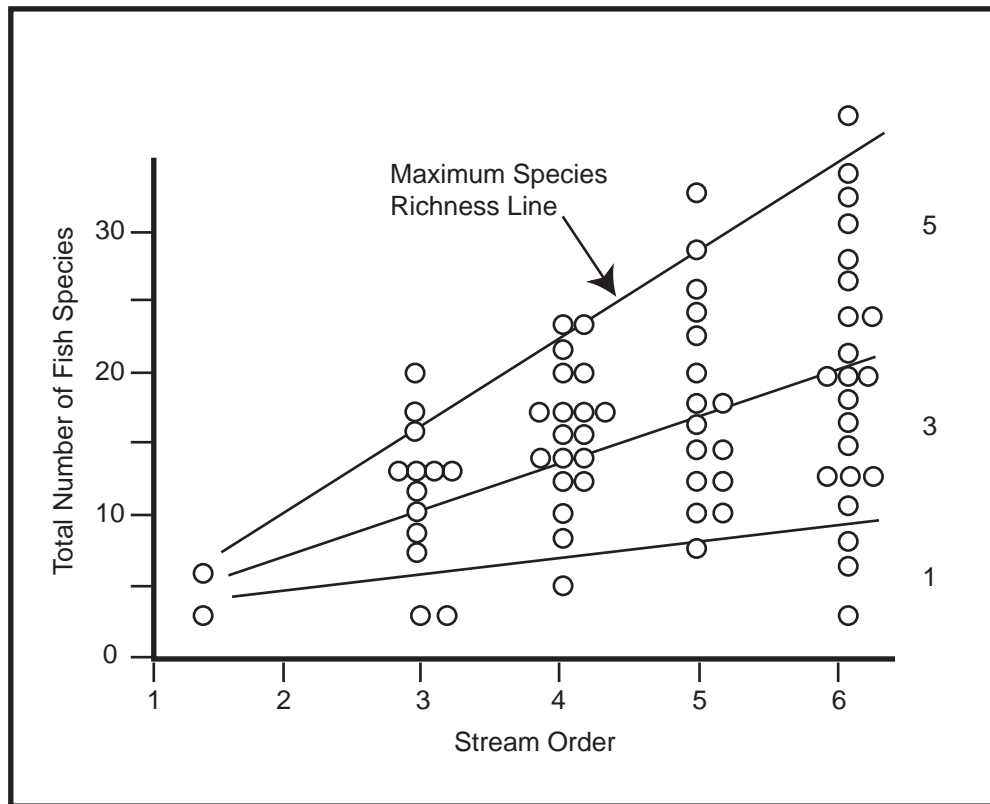
The upstream-downstream approach follows a similar analytical framework. The downstream site is analogous to the near field and the upstream site is analogous to the far field site in the near field-far field approach.

Following the classification of reference sites by ecoregion (or an alternate *a priori* method), the biota of the sites are surveyed according to standard protocol. The data collected during these surveys are used to determine those factors (e.g. elevation, salinity, and stream size) which explain biological variability among the reference sites. These factors are then used to develop a final (*a posteriori*) classification of reference sites (USEPA 1998a).

In an effort to address sources of variability among reference sites, and reduce uncertainty in reference conditions, reference sites may be classified according to common physical attributes. For streams and small rivers, these classification schemes are typically based on local factors of water body size and instream physical habitat characteristics (e.g. gradient and substrate type).

The effectiveness of a classification scheme lies in its ability to partition variation (Gibson et al. 1996). For example, in most regions of the country there is a direct relationship between stream size and the number of fish species found in the stream (see Figure 2-3). If a survey of reference sites confirms this relationship in a particular ecoregion, streams should then be classified by size. If this relationship is not taken into account, small streams (with naturally low numbers of fish species) may incorrectly be categorized as impaired. In some cases, further classification

(e.g. categorizing small streams by pH) may explain additional variation among reference sites, but too much classification may unnecessarily complicate biocriteria development and should be avoided (Karr and Chu 1999).



**Figure 2-3**  
**Correlation Between Stream Order and Total Number of Fish Species. Source: Fausch et al. 1984**

The “ideal” determination of reference conditions involves the synthesis of historical data, simulation models, reference sites, and expertise of biologists with relevant knowledge of the region (Bailey et al. 1998, Gibson et al. 1996). According to EPA guidance, use of actual reference sites to establish reference conditions is important in two respects. First, it reflects real, attainable biological goals. Second, reference sites can be easily monitored. Where there are no historical data, appropriate models, or representative reference sites, reference conditions should be determined by a panel of regional experts. Furthermore, according to EPA guidance, the panel should determine, based on their collective expertise, what the biological community would look like in the absence of human influence. The reference conditions derived by these experts should be only temporary until better methods of establishing reference conditions are found (Gibson et al. 1996). In the case of artificial systems such as reservoirs, alternative methods must be used to identify appropriate reference conditions (USEPA 1998a).

#### 4. Evaluating Metrics and Developing a Multimetric Index

Bioassessment is used to evaluate or compare biological conditions of a water body using measurements of biological indicators, or *metrics*. A metric is a measurable characteristic of the biological community that responds in a predictable way to increased human influence (Gibson et al. 1996). Selection and evaluation of candidate metrics is an important component of biocriteria development.

Each biological metric used to detect anthropogenic stress has associated strengths and weaknesses. To account for a metric's weaknesses and incorporate its strengths, many water management programs typically use a *multimetric* approach to bioassessment. By incorporating more than one metric, bioassessment protocols are more likely to detect impairment over a greater range of stressors (Gibson et al. 1996). Furthermore, the use of multiple metrics sensitive to different types of stress facilitates the determination of the cause of impairment (Karr and Chu 1999).

##### a. Multimetric Approach

The core of the multimetric approach is the *multimetric index*. A multimetric index is obtained by scoring metric values according to their deviation from expected values (i.e., the metric values for the reference condition). The sum of these scores (the index) is compared to the index obtained from reference conditions in order to evaluate the degree of impairment.

Several types of metrics can be used to construct a multimetric index. The metrics commonly used to evaluate the attributes of biological communities as part of a biocriteria framework can be classified into three broad categories: organismal response, indicator organisms, and community response.

*Organismal response* measures the response of organisms at the individual level to particular stressors. This category of metrics can be broken down into biochemical indicators, physiological indicators, morphological abnormalities, behavioral responses, life-history responses, and sentinel organisms. Biochemical indicators (e.g., enzyme activity, ion regulation) and physiological indicators (e.g., heart rate) are more difficult to apply because they usually require extensive expertise and the use of special equipment. Furthermore, in many organisms the range of normal functions is not understood well enough to determine whether the survivability of the organism in question is actually compromised. By contrast, morphological abnormalities, behavioral responses (e.g., dormancy, emigration), life-history responses (e.g., mortality, growth), and sentinel organisms are much easier to recognize and have been used successfully in bioassessment (Johnson et al. 1993).

The most common method for determining organismal response is through the use of a *sentinel organism*. A sentinel organism is an organism that can concentrate pollutants from ambient water into its body. Bioassessment programs use these organisms as indirect measures of pollutants in a specific area. The "ideal" sentinel organism has the following features (Johnson et al. 1993):

1. Concentration of pollutants within the organism reflecting that of the surrounding environment at all locations and under all conditions,
2. Lack of impairment of reproductive capability or survival at maximum pollutant levels,
3. Sedentary nature,
4. Sufficiently large and abundant for laboratory analysis,
5. Broad geographic distribution to allow for comparison,
6. Long-lived to permit study of long-term effects,
7. Easily sampled,
8. Resiliency to survive transport and laboratory handling.

The second category of metrics used to assess aquatic system health is that of **indicator organisms**. Indicator organisms aggregate responses to a particular stressor or class of stressors within a species or other taxonomic group. These organisms have “particular requirements with regard to a known set of physical or chemical variables such that changes in presence/absence, numbers, morphology, physiology or behavior of that [taxon] indicate that the given physical or chemical variables are outside its preferred limits” (Johnson et al. 1993).

Rosenberg and Wiens (1976) identify the following characteristics of the “ideal” indicator taxon:

1. Sound taxonomy and ease of recognition by non-specialists,
2. Cosmopolitan distribution,
3. Naturally high abundances,
4. Low genetic and ecological variability,
5. Large size,
6. Well understood ecology,
7. Suitability for laboratory studies.

Indicator organisms may also be grouped according to one or more shared functional attributes. This type of grouping of similar species is referred to as a **guild**. The guild concept has been used in ecology for more than 30 years (Root 1967), and is especially useful in bioassessment.

One of the more widely applied “indicator” guilds in aquatic bioassessment is the functional feeding group (Cummins 1973). Functional feeding groups are groups of organisms with similar feeding morphology and behavior. The abundance or presence/absence of certain functional feeding groups is believed by some to be a reliable indicator of stream conditions (Vannote et al. 1980). Landres (1983) reviews the use of the guild concept in environmental impact assessments.



Others (Karr and Chu 1999) believe that functional feeding groups are not especially useful as components of a multimetric index, arguing that the construction of ratios of functional feeding groups (e.g., scrapers/filter feeders) is complicated and may lack biological meaning.

The third type of metric for assessing aquatic system health is *community response*. Since human-induced alterations usually affect multiple taxa, many bioassessment programs rely on this type of metric to measure the response of entire assemblages (e.g. fish or benthic macroinvertebrates). The most common measures of community response are species richness (i.e. the number of species) and various measures of the abundance of species relative to each other, such as dominance (the percent composition of the dominant taxon) and evenness (the degree to which the relative abundance of organisms are equal). These measures are often combined into indices.

Diversity indices combine information on the number and relative abundance of species into a single value (e.g. the Shannon-Weiner Index combines measures of evenness and species richness). Diversity indices have been often used as direct indicators of an area's ecological health because the number and relative abundance of taxa tend to respond directly to environmental change (Pratt and Coler 1976).

Diversity indices have also been heavily criticized, however, for their tendency to ignore the ecological sensitivities of individual taxa and disregard natural diversity. Lenat and Barbour (1994) support this view and have demonstrated that some aquatic systems (e.g., western streams) have naturally low diversities irrespective of the introduction of anthropogenic stress. At present, the consensus among the scientific community is that diversity indices must be used with great caution, if at all. The trend has been to use the components of diversity indices as individual metrics in a multimetric index.

Based on the shortcomings of diversity indices, many scientists have endorsed a more integrative approach to bioassessment—*multimetric indices*. Multimetric indices combine assemblage/community attributes such as taxonomic diversity with information on the ecological sensitivity of individual taxa. Multimetric indices are based on the idea that tolerance to anthropogenic stress is different among taxa. In some indices, such as the Hilsenhoff Index (Hilsenhoff 1987), these differences are quantified by tolerance values being assigned to different taxa depending on the particular index and the stressor(s) being investigated.

One of the most comprehensive indices of biological integrity is the IBI (Index of Biotic Integrity). As originally developed, it consisted of 12 biological metrics which were grouped into categories of species richness and composition, trophic structure, and fish abundance and condition (Karr 1981, Table 2-1). Although originally designed for Midwestern stream fish assemblages, the IBI has been modified for use in diverse geographic areas and with a broader range of organisms (Davis and Simon 1995, Simon 1999, Karr and Chu 1999).

**Table 2-1**  
**IBI Scoring Criteria. Source: Karr 1991**

Metrics	Rating Of Metric <sup>1</sup>		
	5	3	1
<b>Species richness and composition</b>			
1. Total number of fish species <sup>1</sup> (native fish species) <sup>2</sup>	Expectations for metrics 1-5 vary with stream size and region.		
2. Number and identity of darter species (benthic species)			
3. Number and identity of sunfish species (water-column species)			
4. Number and identity of sucker species (long-lived species)			
5. Number and identity of intolerant species			
6. Percentage of individuals as green sunfish (tolerant species)	< 5	5-20	> 20
<b>Trophic composition</b>			
7. Percentage of individuals as omnivores	< 20	20-45	> 45
8. Percentage of individuals as insectivorous cyprinids (insectivores)	> 45	45-20	< 20
9. Percentage of individuals as piscivores (top carnivores)	> 5	5-1	< 1
<b>Fish abundance and condition</b>			
10. Number of individuals in sample	Expectations for metric 10 vary with stream size and other factors.		
11. Percentage of individuals as hybrids (exotics, or simple lithophils)	0	> 0-1	> 1
12. Percentage of individuals with disease, tumors, fin damage, and skeletal anomalies	0-2	> 2-5	> 5

<sup>1</sup> Original IBI metrics for Midwest United States.

<sup>2</sup> Generalized IBI metrics (see Miller et al. 1988).

## b. Metrics Evaluation and Scoring

The development of a multimetric index requires the selection, evaluation, scoring, and calibration of metrics based on sound ecological principles. Although the selection of metrics will vary according to program objectives, geographic region, and the type of system being assessed, the general criteria used to select metrics are relatively consistent (Gibson et al. 1996).

Because components of biotic integrity include patterns and processes from the individual to the ecosystem level, an accurate assessment of biotic integrity requires the use of metrics that span multiple levels of biological organization (Karr et al. 1986). The chosen metrics should have low variability with regard to their expected range of responses among reference sites and be capable of discriminating between impaired and unimpaired systems (Gibson et al. 1996). Finally, metrics should change quantitatively along a gradient of human-induced stresses (Karr and Chu 1999). This allows investigators to detect not only if a system is impaired, but also the degree of impairment.

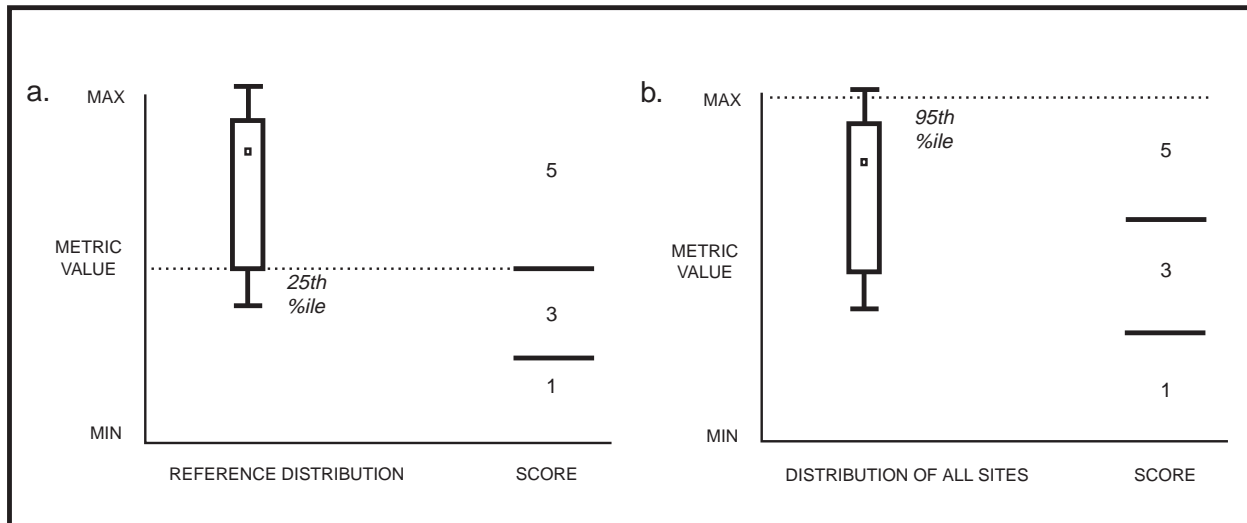
Once candidate metrics have been selected, they are evaluated to determine whether they meet the criteria of low variability and responsiveness. EPA (Gibson et al. 1996) recommends the use of percentiles to evaluate the variability of a particular metric. The metric values of the reference conditions are plotted along an axis and divided into four equal-sized groups (quartiles). An “interquartile coefficient” is then calculated as the ratio of the interquartile range (distance between 25<sup>th</sup> and 75<sup>th</sup> percentile) to the scope for detection (distance between 25<sup>th</sup> percentile and minimum possible value of the metric). A coefficient greater than 1 generally indicates large variation among metric values, in which case the particular metric should be used cautiously (USEPA 1998a).

In order to determine whether a metric responds quantitatively to varying degrees of human influence, it must be measured at multiple sites along a continuum of stressor intensities. Since it is difficult to find sites affected by only one type of stress, Karr and Chu (1999) recommend ranking sites according to the overall severity of their exposure to anthropogenic stress.

Metrics that do not meet the criteria previously discussed (responsiveness and low variability) may be calibrated and reevaluated, or eliminated. When a metric does meet the appropriate criteria, it should be tested against other selected metrics for redundancy (i.e., the response of one metric to impairment should be independent of the responses of other metrics). When two metrics are correlated (resulting in redundancy) along the entire continuum of stressor intensities, one should be eliminated (Karr and Chu 1999); however, a certain degree of redundancy is desirable to ensure the index is robust.

Following the evaluation and selection of metrics, metric values are converted to scores that are used to calculate the final index. The scoring system used most frequently is the assignment of ordinal scores of 5, 3, and 1, where 5 represents a metric value similar to reference conditions, 3 represents a value somewhat different from reference conditions, and 1 represents a value very different from the value of reference conditions.

Two methods commonly used for scoring metrics are *bisection* scoring and *trisection* scoring. The selection of a method is determined by the quality of the reference sites. When reference sites are sufficient in number and quality, the bisection method is used to assign ordinal scores. As before, the range of metric values is plotted. Those values above the 25<sup>th</sup> percentile of the distribution are considered similar to reference conditions and assigned a score of 5. Values falling below the 25<sup>th</sup> percentile are considered to have some degree of impairment. The range of the 0 to 25<sup>th</sup> percentile is then bisected with the upper half of the range receiving a score of 3 and the lower half receiving a score of 1 (Figure 2-4; USEPA 1998a). It appears that the states do not generally adhere to this aspect of EPA guidance.



**Figure 2-4**  
**Basis of Bioassessment Scores—**a. Bisection Method Based on Unimpaired Reference Sites; b. Trisection Method Based on Population Distribution. Source: USEPA 1998a

EPA guidance (Gibson et al. 1996, USEPA 1998a) recommends use of the trisection method when appropriate reference sites are not available. Metric values for all the sites are plotted and the range of values from 0 to the 95<sup>th</sup> percentile is trisected. A score of 5 is given to values in the top third, a score of 3 is given to values in the middle third, and a score of 1 is given to values in the bottom third (Figure 2-4; USEPA 1998a).

The exact use of these scoring protocols varies among programs. The division of the percentiles is arbitrary and is dependent upon the amount of uncertainty a program is willing to tolerate (USEPA 1998a). Furthermore, the scores assigned by these methods are often adjusted based on site-specific circumstances. For example, if there are 47 small lakes in a particular ecoregion and only two are naturally acidic, further classification by acidity is not necessarily needed. Instead, the scoring of metrics such as the percent acid-tolerant taxa can be set separately for the two acidic lakes. Others (e.g. Minns et al. 1994, Hughes et al. 1998) have advocated alternative scoring protocols.

## **5. Developing Sampling Protocols: Biosurveys of Target Assemblages and General Sampling Considerations**

To adequately characterize the biological communities for a particular water body, field biological surveys (biosurveys) and other measurements of the resident biota are conducted. Biosurveys provide the only direct method for measuring the structure and function of an aquatic community (USEPA 1990).

Two key components of biosurveys for development of biocriteria are: the selection of target organisms or assemblages representative of the biological integrity of the water body, and the use of sampling protocols which will ensure the best representation of the aquatic community (USEPA 1990). The following discussion is based on EPA guidance for streams and small rivers and, to a lesser degree, lakes and reservoirs.

### **a. Selection of Target Organisms/Assemblages**

It is impossible to rigorously assess the entire flora and fauna at a bioassessment site. Consequently, biocriteria programs typically focus on one to several biological assemblages. The choice of assemblage(s) may be made on the basis of relevance to existing narrative criteria in the state's water quality standards regulation, sensitivity to stressors that are of concern, compatibility with preexisting sampling programs, or because use of a particular assemblage is generally accepted.

Organisms typically used in aquatic bioassessment programs include fish, benthic macroinvertebrates, zooplankton, and phytoplankton. The relative strengths and weaknesses of these assemblages for use in a biocriteria assessment program are evaluated in Table 2-2.

**Macroinvertebrates** are the most frequently used assemblage in stream bioassessment programs. The assemblage comprises the visibly distinguishable crustaceans, mollusks, insects, and other fairly large aquatic invertebrates (Gibson et al. 1996). In addition to the advantages described in Table 2-2, benthic invertebrate assemblages:

- Are able to integrate local environmental conditions (Plafkin et al. 1989),
- Tend to react quickly to perturbation (Cook 1976),
- Are usually heterogeneous (Metcalf 1989), and
- Are useful bioindicators of episodic discharges of trace metals (Lynch et al. 1988).

**Fish** also are a common target assemblage for biological assessments. The fish assemblage is particularly attractive because of:

- The longevity of fish and their ability to integrate environmental conditions over long periods of time (Ohio EPA 1987b),
- The importance of fish in structuring lotic food webs (Power 1990),
- The public's awareness of and concern for this assemblage, and

- The success of the original IBI (Karr 1981) and regional modifications of the index.

Other commonly used assemblages include *periphyton* and *macrophytes*. Periphyton have been broadly defined to comprise benthic algae, bacteria and related waste products, as well as various species of microinvertebrates (Gibson et al. 1996, Lamberti and Moore 1984). Plafkin et al. (1989) identify the following advantages of using the periphyton assemblage as targets for bioassessment in streams and small rivers:

- Rapid reproduction rates and short life cycles responsive to short-term impacts,
- Structure and function directly affected by physical and chemical factors,
- Straightforward and rigorous sampling methods,
- Standardized methods for quantifying non-taxonomic attributes such as biomass and chlorophyll, and
- Algal sensitivity to stressors for which other assemblages may be relatively insensitive.

Similarly, macrophytes (large aquatic plants) offer several distinct advantages as target assemblages in a bioassessment program (Gibson et al. 1996):

- Relatively straightforward genus-level taxonomy,
- Partial dependence on local environmental conditions,
- Potentially high densities,
- Growth patterns sensitive to herbivory,
- Longevity, distribution, and rate of population growth reflective of prevailing environmental conditions.

The EPA guidance also includes wildlife as a possible target assemblage that can be useful in bioassessment of streams and small rivers. This group includes mammals, birds, reptiles, and amphibians. Specific advantages cited for this group are:

- Longer life spans suitable for evaluation of cumulative effects,
- Relatively large body size and behavior characteristics permitting visual and auditory observation,
- Sensitivity (birds) to riparian alteration,
- Well understood taxonomy,
- Numerous biomarkers (e.g. physical and biochemical alterations of individuals as well as other sublethal effects in response to contamination),
- Straightforward trapping techniques for small mammals and availability of information derived from tracks and feces,
- Ease of public involvement in assisting in wildlife assessments.

**Table 2-2**  
**Relative Strengths and Weaknesses of Four Animal Assemblages Often Used in Bioassessment**

	Targeted Assemblage			
	Phytoplankton	Zooplankton	Benthic Invertebrates	Fish
<b>Precedent and indicator availability</b> Indicators have already been developed for the targeted assemblage for many waterbody types and geographic regions.	◐	◐	●	●
<b>Availability of baseline data</b> Biosurveys of the targeted assemblage are commonly conducted by state or federal monitoring programs.	●	◐	●	●
<b>Lower trophic level condition</b> Targeted assemblage reflects conditions of primary and secondary trophic levels, the food base upon which higher levels depend.	●	●	●	◐
<b>Upper trophic level condition</b> Targeted assemblage reflects conditions of tertiary and higher trophic levels, such as top predators.	○	○	◐	●
<b>Short term impacts</b> The targeted assemblage responds quickly to stresses and is a good indicator of short-term impacts.	●	●	◐	◐
<b>Long term impacts</b> The targeted assemblage responds more slowly to stresses and integrates cumulative or chronic impacts over longer time periods.	○	○	●	●
<b>Sampling methods</b> Sampling methods for the targeted assemblage are well developed and can be standardized across programs.	●	●	●	●
<b>Cost of sampling</b> Personnel and equipment costs needed to conduct sampling for the targeted assemblage are relatively low (excluding boat costs).	●	●	●	◐
<b>Skill level for sampling</b> Sampling does not require special skills beyond those obtainable in a short training course (excluding boat operation).	●	●	●	◐
<b>Cost of sample processing</b> Post-collection sample processing costs, including sorting, identifying, and enumerating specimens, are relatively low.	○	○	○	●
<b>Skill level for sample processing</b> A high degree of skill is not required for processing samples, including taxonomic identification.	○	○	◐	●

**KEY:**

● = the targeted assemblage has the described characteristic

◐ = the targeted assemblage partially fulfills the described characteristic

○ = the targeted assemblage does not have the described characteristic

Overall, solid black is positive and reflects strengths of the targeted assemblage, while solid white is negative and reflects weaknesses of the targeted assemblage. A half circle indicates an intermediate rating.

Scientists have identified disadvantages associated with the use of each type of assemblage. For example, the mobility of fish (Moyle 1994) and many benthic macroinvertebrate taxa (Rosenberg and Resh 1993) decreases their value as bioindicators of transient conditions. Likewise, wildlife are also dependent on terrestrial habitat conditions that may confound attempts to use wildlife as indicators of aquatic conditions. The advantages and disadvantages of using each assemblage vary according to region, program objectives, and the choice of metrics.

Because certain assemblages are better indicators of certain types of stress, bioassessment programs will benefit from the use of multiple assemblages. At present, most agencies determine which organisms to use based on available taxonomic skills, financial resources, and comparability of existing data bases (Lenat and Barbour 1994). This trend is changing, though, and many states and Native American tribes are beginning to incorporate multiple assemblages into their bioassessment protocols (Davis et al. 1996).

## b. Timing and Location of Sampling

Once target assemblages have been identified, sampling issues must be addressed. Biological communities are temporally and spatially variable. Consequently, collection and interpretation of data on biological communities must account for changes over time and space. Because it is unrealistic to obtain continuous samples from all areas of a system, resource managers must choose the time and place of sampling based on the ability of a sample to capture representative conditions and fulfill program objectives. This requires an extensive knowledge of seasonal habitat variation and the life history characteristics of the resident biota.

### (1) Choosing Sampling Periods

Collecting representative samples throughout the year is the best way to account for temporal variability in biological communities. However, due to budget constraints and/or program objectives, it is not always possible, or necessary. Gibson et al. (1996) recommend that if one sampling period—an *index period*—is chosen, it be designed to minimize among-year variability and maximize gear efficiency and accessibility of the target assemblage.

Minimizing among-year variability requires that sampling periods be chosen based on seasonal changes rather than calendar dates. This requires that the choice of sampling periods take into consideration annual variability in seasonal effects. For example, in some years spring begins earlier than in other years. One common method used to account for temporal variation in seasonal characteristics is the use of degree-days, that is, the summation of mean daily temperatures above a threshold. For some seasonal phenomena that are strongly influenced by temperature, such as aquatic insect emergence and aquatic vegetation growth, the number of degree-days is a more consistent predictor of seasonal effects than is calendar date.

Seasonal hydrology may also be an appropriate consideration in selecting sampling periods. In reservoirs with large seasonal fluctuations in water level, it is often appropriate to sample during a particular season when water levels are within a relatively narrow range.



A disadvantage of using an index period is that there is no collection of baseline data with which to assess an “off-season” event. Moreover, in some situations an index period sampling approach would not permit assessment of the effects until long after an event has occurred. Thus, to adequately assess “off-season” impacts, multiple sampling windows would be necessary.

In order to maximize target assemblage accessibility, sampling must be based on reproductive, migratory, and recruitment cycles. Gibson et al. (1996) recommend that sampling occur when target assemblages have stabilized after larval recruitment, and “subsequent mortality and the use of their niche space is at its fullest.” However, sampling during this time may not always be realistic, and inclusion of young of year fish introduces additional variability in assemblage structure (Yant et al. 1984, Schlosser 1985, Angermeier and Karr 1986). Inefficiency of collecting gear may preclude effective sampling under certain climatic conditions such as high flows.

## *(2) Choosing Habitats to Sample*

Aquatic systems are heterogenous environments consisting of a multitude of macrohabitats (e.g. shallow shoreline of a lake) and microhabitats (e.g. bed of aquatic vegetation within the shallow shoreline habitat). Since it is usually impossible for surveys to include all habitat types, resource managers are faced with the task of selecting one or more habitats whose biota will provide enough information to assess the health of the entire system (Gibson et al. 1996). This makes the selection of the habitat(s) to be sampled crucial to the success of the bioassessment program. Considerations in choosing a habitat type include selecting productive habitats (more taxa and individual organisms) and dominant habitats (most representative of the entire system) (Plafkin et al. 1989).

Based on program objectives and logistical constraints, resource managers may choose to sample a single habitat type or multiple habitat types. The advantage of selecting one representative habitat type is that it minimizes between-site variability. However, the problem with single habitat sampling is that the most productive or dominant habitat at one site may not be the most productive or dominant habitat at another site, making comparisons difficult (Gibson et al. 1996).

For this reason, many investigators advocate sampling multiple habitats. Although this requires additional resources, it allows for the sampling of habitats in the proportions that they occur. The shortcoming of multiple habitat assessment is that as the number of habitats sampled increases, so does the likelihood that a habitat will be absent from a test site or that there will be differences in the quantity and quality of the habitat (Gibson et al. 1996).

Choice of the number and types of habitats to be sampled is also influenced by the assemblage and water body being surveyed. When fish assemblages are sampled in smaller systems such as streams, small rivers, and shallow lakes, a multitude of habitats can usually be sampled with a single technique such as electrofishing or seining (Gibson et al. 1996). However, when fish assemblages are sampled in larger habitats, such as estuaries, coastal waters, deep lakes, and large rivers, sampling multiple habitats requires multiple sampling techniques. For example, obtaining a representative sample of a large lake’s fish assemblage may require the use of otter trawls to collect benthic fishes, midwater trawls or experimental gill nets to collect pelagic fishes, and beach seining and electrofishing to collect fishes inhabiting shallow waters.

Obtaining a representative sample of benthic macroinvertebrates would require sampling an even greater number of habitats, since invertebrate taxa are distributed heterogeneously based on microhabitat characteristics such as substrate composition and flow (Rabeni and Minshall 1977). Sampling macrophytes may also require a multihabitat approach, since macrophyte distribution is affected by microhabitat characteristics such as light, temperature, water velocity, and nutrient availability (Carr et al. 1997). Although periphyton assemblages are affected by a multitude of habitat characteristics, most periphyton sampling is done in a single habitat (Gibson et al. 1996).

### c. Habitat Measurement

Habitat measurement allows an assessment of the effect of habitat on biota and aids in the interpretation of changes in the biota. This information is important for evaluating biological integrity because biological expectations at a site are not only influenced by physical environmental characteristics at the regional scale, but they are also influenced by physical habitat characteristics at a local, and even micro-scale (especially for macroinvertebrates).

Habitat measurement provides crucial information on whether habitat may be a cause of perceived impairment of biological conditions (Gibson et al. 1996, Karr et al. 1986, Plafkin et al. 1989, USEPA 1998a). In some instances, the habitat may have been altered by anthropogenic influences. In other cases, the habitat condition is of natural origin and the resultant biological condition is attributable to natural variation.

Physical habitat conditions at the local scale result from the complex interaction of regional climatic, geological, and biogeographic factors with local geology, topography, and hydrology. Human factors such as patterns and practices of land and water use also affect physical habitat.

As defined in EPA's guidance on lakes and reservoirs, habitat measurement "seeks to identify the physical and chemical characteristics of the [lake] habitat—both natural and anthropogenic—that affect the biota" (USEPA 1998a). This evaluation, in part, involves the use of a classification scheme based on the intrinsic physical and chemical attributes of the system which are minimally affected by human activity. Under this scheme, a particular water body or site is placed in a category based on such variables as geology, morphology, and soils. Reference conditions for each category are then compared to test site conditions within the same category to determine any deviation for both habitat and biological indicators (USEPA 1998a).

Habitat measurement also relies on direct assessment, through sampling, of relevant human activity (e.g., land use and discharges) and water quality variables. Gibson et al. (1996) recommend sampling of natural substrates because the resident biota reflect the biological potential of the habitat, including the physical habitat, which can be altered by human activities. Artificial substrate may also be used for benthic macroinvertebrate samples if it can be matched to the natural substrate (e.g., a rock basket in a cobble substrate stream (SAB 1993)).

## **6. Assuring the Quality of the Bioassessment Program**

Biocriteria are only as good as the quality of the program used to develop and apply them. For this reason, quality assurance plans are a necessary part of all bioassessment programs. Quality

assurance plans are designed to ensure the integrity and usefulness of data collected during all stages of biocriteria development and application. The plan should include a statement of data quality objectives, personnel training, instrument maintenance, and standard ecological quality control measures such as peer review, voucher specimens, double data entry, and protocols designed to ensure the consistency of data gathering and processing. This last element, consistency of the data collection process, is essential to ensure the comparability of spatially and temporally discrete samples (Gibson et al. 1996).

Part of the quality assurance process for a bioassessment program entails the modification and refinement of protocols. Prior to initiating large-scale surveys, the bioassessment protocol should be tested to determine its ability to detect human impacts. Ideally, the protocol should be tested on multiple systems known to have varying degrees of human influence (e.g., for streams, no logging, some logging, heavy logging, clear cuts). This allows investigators to determine the sensitivity of metrics, assemblages and other aspects of the protocol to various degrees of environmental alteration (Karr and Chu 1999). Opportunities to examine the responsiveness of the metrics in multiple systems over a range of degrees of human influence will be relatively limited in larger, open systems such as large estuaries, coastal marine waters, and the Great Lakes. The protocol should also be tested to ensure that human-induced changes can be discerned from natural variation. Any deviations from these objectives require modification of the protocol. Testing and modification of protocols should be repeated as often as necessary (Gibson et al. 1996).

## **7. Establishing Biocriteria Standards**

Once sampling and bioassessment protocols have been validated, reference conditions are established by surveying reference sites and/or using one or more of the alternative methods previously discussed (historical data, simulation models, and expert consensus). The biological data collected are used to establish the range of values (e.g. among the reference sites) that will operationally define biological integrity. These values are then translated into biocriteria.

The translation of reference condition values into biocriteria is based largely on the specific objectives of a bioassessment program and the designated uses of water bodies in the region. For example, if a lake contains a large number of endemic species, resource managers may establish biocriteria as follows: *Narrative*—“There should be no change in species richness or trophic composition of the lake from one sampling period to the next.”, *Numeric*—“The IBI (multimetric index) of the lake must be above the 90<sup>th</sup> percentile of the IBI values for reference sites.” Although these biocriteria are stringent, their attainment may be considered necessary to protect the endemic species in the lake. If the lake contained no endemic or endangered species, and received heavy recreational use, biocriteria likely would be set much lower.

Regardless of program objectives, biocriteria should not be set so high that they are unattainable or so low that sites not attaining designated uses are judged unimpaired. Finally, biocriteria should be revised as new information becomes available, natural conditions change, or reference conditions improve (Gibson et al. 1996).

## **8. Assessing Water Body Impairment**

The establishment of biocriteria allows for implementation of a biomonitoring program that focuses on systems that may potentially be impaired. Using the established protocol, the biological community of a system is surveyed and the data analyzed to determine if biocriteria are met. If the biocriteria are not met, the system is characterized as impaired. Investigators should then determine the relative degree of impairment, based on the biological data collected and any additional chemical, physical, or other relevant data available (Gibson et al. 1996).

## **9. Diagnosing Cause(s) of Impairment and Undertaking Remediation**

Restoring the biological integrity of impaired systems requires identification of the biological properties and/or processes that have been altered, and the cause of their alteration (Gibson et al. 1996). This requires an extensive evaluation of the biological, chemical, physical, geographical, and historical data. In some cases, a synthesis of the available data may point directly to the cause of impairment. In most cases, though, the cause of impairment may be complex. Impairment caused by non-point source pollution, cumulative stressors, or multiple stressors requires a much more extensive analysis of available data.

Once a potential cause of impairment is identified, remedial action can be taken. For example, if sedimentation due to loss of riparian vegetation is suspected of degrading a small creek, a plan to revegetate the riparian zone might be implemented. Continued biomonitoring of the creek could then continue to determine the effectiveness of the remediation plan. Even if a site's biotic integrity is eventually restored or a site is initially classified as unimpaired, biomonitoring should continue on an appropriate schedule so that any future impairment may be detected (Gibson et al. 1996).

Diagnosing cause(s) of impairment will be particularly challenging in large, open systems that encompass multiple stressors within relatively large geographic areas. This issue is further discussed in Chapter 3.

## **Other Considerations**

As the number of states implementing biocriteria continues to increase, more public and private organizations will be required to comply with bioassessment protocols. Compliance with these protocols requires the acquisition of the necessary resources.

Basic resources needed to implement bioassessment programs for streams, large rivers, lakes, estuaries, and coastal waters include taxonomists, statisticians, field technicians, data entry/analysis personnel, technical writers, project coordinators, vehicles, collecting permits, specimen reference collections, computers, and lab equipment (including microscopes, preservatives, taxonomic keys, stains, balances, and assorted containers and glassware). The type of sampling gear required will depend on the habitat and assemblage being sampled (Table 2-3).

While implementation of a biocriteria program is resource intensive, use of a rapid assessment approach may be, in some instances, an adequate and economical alternative. The goal of "*rapid*

**bioassessment**” (sensu Plafkin et al. 1989) is to expend minimum effort at test sites, in terms of time and resources, to obtain scientifically valid results that can be used to assess water quality and make management decisions. Most rapid bioassessments are designed to go from the field to a report in 3-5 working days (Lenat and Barbour 1994, Resh and Jackson 1993). Even “rapid bioassessment protocols”, however, require substantial investments of time and resources to develop protocols appropriate for a given ecological setting.

Rapid bioassessment programs use many techniques to decrease resource expenditure associated with implementation of the protocols. In most rapid assessment protocols, one large sample is typically taken instead of several individual replicates. Collections are then subsampled to obtain a representative sample of the community at hand (Resh et al. 1995). There are opposing views of the advisability of this practice (e.g. Barbour and Gerritsen 1996, Doberstein et al. 2000).

The use of binary (presence/absence) data in place of quantitative data can also be used to minimize resource expenditure. Finally, many rapid bioassessment programs rely on reduced taxonomic resolution to limit the time spent on the labor-intensive task of organism identification; however, reduced taxonomic resolution can compromise responsiveness to some stressors and potentially decrease one’s ability to diagnose the cause of effects that are detectable with coarser taxonomic resolution.

## **Remarks**

EPA’s biocriteria framework is designed to provide a more comprehensive and sensitive set of indicators of water quality and ecological integrity relative to other water quality criteria (Karr 1991, Yoder 1995). However, no single assessment tool is ideally suited for all tasks, and a multi-purpose tool is rarely the best choice for a narrowly defined task. Thus, biocriteria, and a multimetric approach to water quality management in particular, should be evaluated against the suite of tasks to which they will be applied. One of those tasks demanding such scrutiny will be the application of biocriteria and its tools to the regulation of power plant cooling water intake structures under § 316(b) of the CWA.

Much of what is known about the multimetric approach to bioassessment derives from research and applications in streams and small rivers; however, power plants typically are located on much larger water bodies. Subsequent chapters of this report examine issues associated with application of multimetric bioassessment to other water body types in general and specifically for regulation of cooling water intake structures (CWISs).

**Table 2-3  
Sampling Gear Requirements**

<b>Assemblage/ Habitat</b>	<b>Stream/Small River</b>	<b>Large River</b>	<b>Lake/Reservoir</b>	<b>Estuary/Coastal Waters</b>
<b>Fish</b>	<ul style="list-style-type: none"> <li>• Backpack Electrofisher</li> <li>• Block and Dip Nets</li> <li>• Live Buckets</li> <li>• Bag Seine</li> <li>• Chest Waders</li> </ul>	<ul style="list-style-type: none"> <li>• Boat</li> <li>• Boat-Mounted Electrofisher</li> <li>• Dip Nets</li> <li>• Live Buckets</li> </ul>	<ul style="list-style-type: none"> <li>• Backpack Electrofisher</li> <li>• Boat</li> <li>• Boat-Mounted Electrofisher</li> <li>• Trawl (Bottom and Water)</li> <li>• Gill Net</li> <li>• Bag Seine</li> <li>• Dip Nets</li> <li>• Live Buckets</li> </ul>	<ul style="list-style-type: none"> <li>• Boat</li> <li>• Trawl (Bottom and Water)</li> <li>• Gill Net</li> <li>• Bag Seine</li> <li>• Dip Nets</li> <li>• Live Buckets</li> </ul>
<b>Benthic Macroinvertebrates</b>	<ul style="list-style-type: none"> <li>• Surber Sampler</li> <li>• Portable Invertebrate Box Sampler</li> <li>• D-Frame Kick Net</li> <li>• Chest Waders</li> </ul>	<ul style="list-style-type: none"> <li>• Ekman Grab</li> <li>• Petersen Grab</li> <li>• Boat</li> </ul>	<ul style="list-style-type: none"> <li>• Ekman Grab</li> <li>• Petersen Grab</li> <li>• Boat</li> </ul>	<ul style="list-style-type: none"> <li>• Ekman Grab</li> <li>• Petersen Grab</li> <li>• Boat</li> </ul>
<b>Periphyton</b>	<ul style="list-style-type: none"> <li>• PVC pipe w/ neoprene collar</li> <li>• Brush/scrapper</li> <li>• Bulb pipettes</li> </ul>	<ul style="list-style-type: none"> <li>• PVC pipe w/ neoprene collar</li> <li>• Brush/scrapper</li> <li>• Bulb pipettes</li> <li>• S.C.U.B.A.</li> <li>• Ekman Grab</li> </ul>	<ul style="list-style-type: none"> <li>• PVC pipe w/ neoprene collar</li> <li>• Brush/scrapper</li> <li>• Bulb pipettes</li> <li>• S.C.U.B.A.</li> <li>• Ekman Grab</li> </ul>	<ul style="list-style-type: none"> <li>• PVC pipe w/ neoprene collar</li> <li>• Brush/scrapper</li> <li>• Bulb pipettes</li> <li>• S.C.U.B.A.</li> <li>• Ekman Grab</li> </ul>
<b>Macrophytes</b>	<ul style="list-style-type: none"> <li>• Collection Bags</li> </ul>	<ul style="list-style-type: none"> <li>• Boat</li> <li>• Collection Bags</li> <li>• S.C.U.B.A.</li> </ul>	<ul style="list-style-type: none"> <li>• Boat</li> <li>• Collection Bags</li> <li>• S.C.U.B.A.</li> </ul>	<ul style="list-style-type: none"> <li>• Boat</li> <li>• Collection Bags</li> <li>• S.C.U.B.A.</li> </ul>
<b>Wildlife</b>	<ul style="list-style-type: none"> <li>• Varies by Species</li> </ul>	<ul style="list-style-type: none"> <li>• Varies by Species</li> </ul>	<ul style="list-style-type: none"> <li>• Varies by Species</li> </ul>	<ul style="list-style-type: none"> <li>• Varies by Species</li> </ul>
<b>Zooplankton</b>	<ul style="list-style-type: none"> <li>• Plankton Sweep Net</li> </ul>	<ul style="list-style-type: none"> <li>• Plankton Tow Net</li> <li>• Boat</li> </ul>	<ul style="list-style-type: none"> <li>• Plankton Tow Net</li> <li>• Boat</li> </ul>	<ul style="list-style-type: none"> <li>• Plankton Tow Net</li> <li>• Boat</li> </ul>
<b>Sediment Diatoms</b>	<ul style="list-style-type: none"> <li>• Ekman Grab</li> </ul>	<ul style="list-style-type: none"> <li>• Ekman Grab</li> <li>• Boat</li> </ul>	<ul style="list-style-type: none"> <li>• Ekman Grab</li> <li>• Boat</li> </ul>	<ul style="list-style-type: none"> <li>• Ekman Grab</li> <li>• Boat</li> </ul>

# 3

## EVALUATION OF BIOCRITERIA IN RELATION TO § 316(B): GENERAL TECHNICAL CONSIDERATIONS

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The EPA's proposal to incorporate a biocriteria program within § 316(b) raises several technical considerations which present conceptual and practical challenges to use of biocriteria. This chapter addresses technical considerations surrounding the use of biocriteria and, where relevant, addresses accompanying policy and regulatory objectives. The purpose is to critically evaluate biocriteria, by identifying strengths to be exploited, potential weaknesses to be addressed through targeted research and inherent limitations to be accommodated, as biocriteria and other components of the regulatory framework are developed to implement § 316(b) of the CWA.

Technical considerations associated with the use of biocriteria and multimetric bioassessment methods under § 316(b) fall into two broad categories: general considerations relating to the generic application of biocriteria, and specific considerations arising in the context of biocriteria application to particular water body types. The general technical issues are addressed in this chapter. Technical issues relevant to specific water body types are discussed in Chapter 4. Technical considerations are discussed within the following organizational framework:

### General Technical Considerations (Chapter 3)

1. Integration of biocriteria with other EPA programs
2. Issues surrounding biocriteria development and application:
  - a. reference conditions
  - b. choice of target assemblages
  - c. multimetric index development
  - d. antidegradation policy
3. Research needs

### Technical Considerations Specific to Water Body Type (Chapter 4)

1. Streams and small rivers
2. Large rivers
3. Lakes and reservoirs
4. Estuaries and coastal marine waters

Prior to addressing the general technical issues, the role of biocriteria within the EPA draft § 316(b) regulatory framework is initially reviewed.

## **The Role of Biocriteria in EPA'S Draft Framework for § 316(b)**

EPA's draft regulatory framework sets forth a process for implementing § 316(b) on a consistent, nationwide basis (Nagle and Morgan 1999). This framework is intended to serve as a basis for development of § 316(b) implementing rules. Section 316(b) requires that *the location, design, construction, and capacity of cooling water intake structures reflect the best technology available for minimizing adverse environmental impact*. Impacts from cooling water intake structures (CWIS) result from two processes: impingement and entrainment.

Entrainment occurs when aquatic organisms, eggs, or larvae are taken into a facility's cooling system, passed through its heat exchanger, and then discharged from the facility. Impingement occurs when aquatic organisms are trapped, "impinged," against the intake screens or other technologies at the entrance of a facility's CWIS by the velocity of the intake flow. While CWIS impacts result from impingement and entrainment, EPA has not yet defined what level of impact would constitute "adverse environmental impact" for purposes of § 316(b). This is a key issue for the current rulemaking effort.

EPA's pre-decisional, draft regulatory framework consists of a 3-tiered decisional process, with each tier requiring a different set of information and level of assessment (Figure 3-1). The primary objectives of this process are to:

- Determine whether a facility's CWIS satisfies the requirement of minimizing adverse environmental impacts through application of best technology available (BTA),
- Identify the specific actions that must be taken in cases where insufficient information exists to make a determination, and
- Prescribe additional actions that must be taken in order to diagnose and minimize the CWIS's effects in cases where adverse environmental impacts are found.



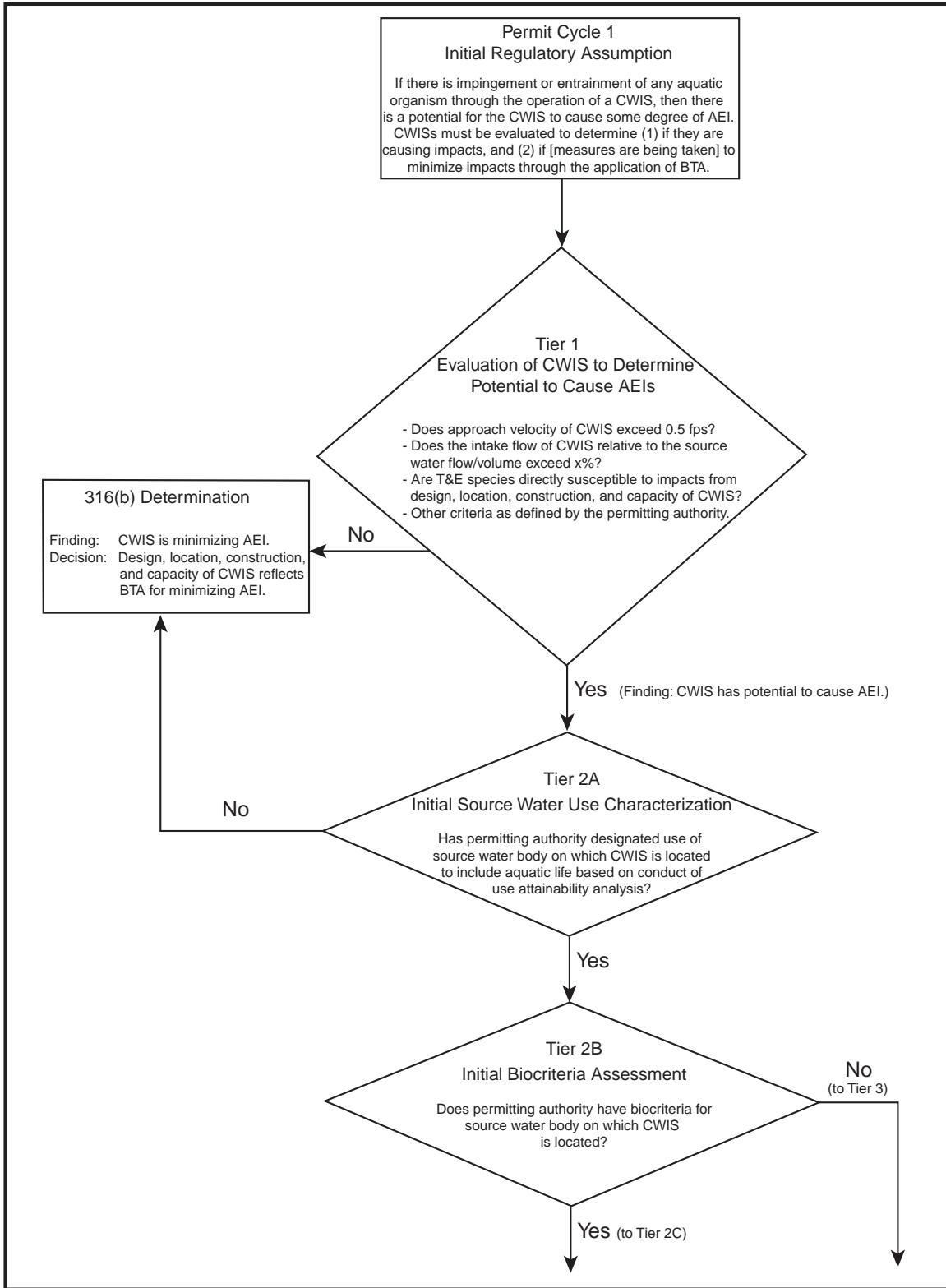


Figure 3-1  
Section 316(b) Draft Regulatory Flow Chart. Source: Nagle and Morgan (1999)

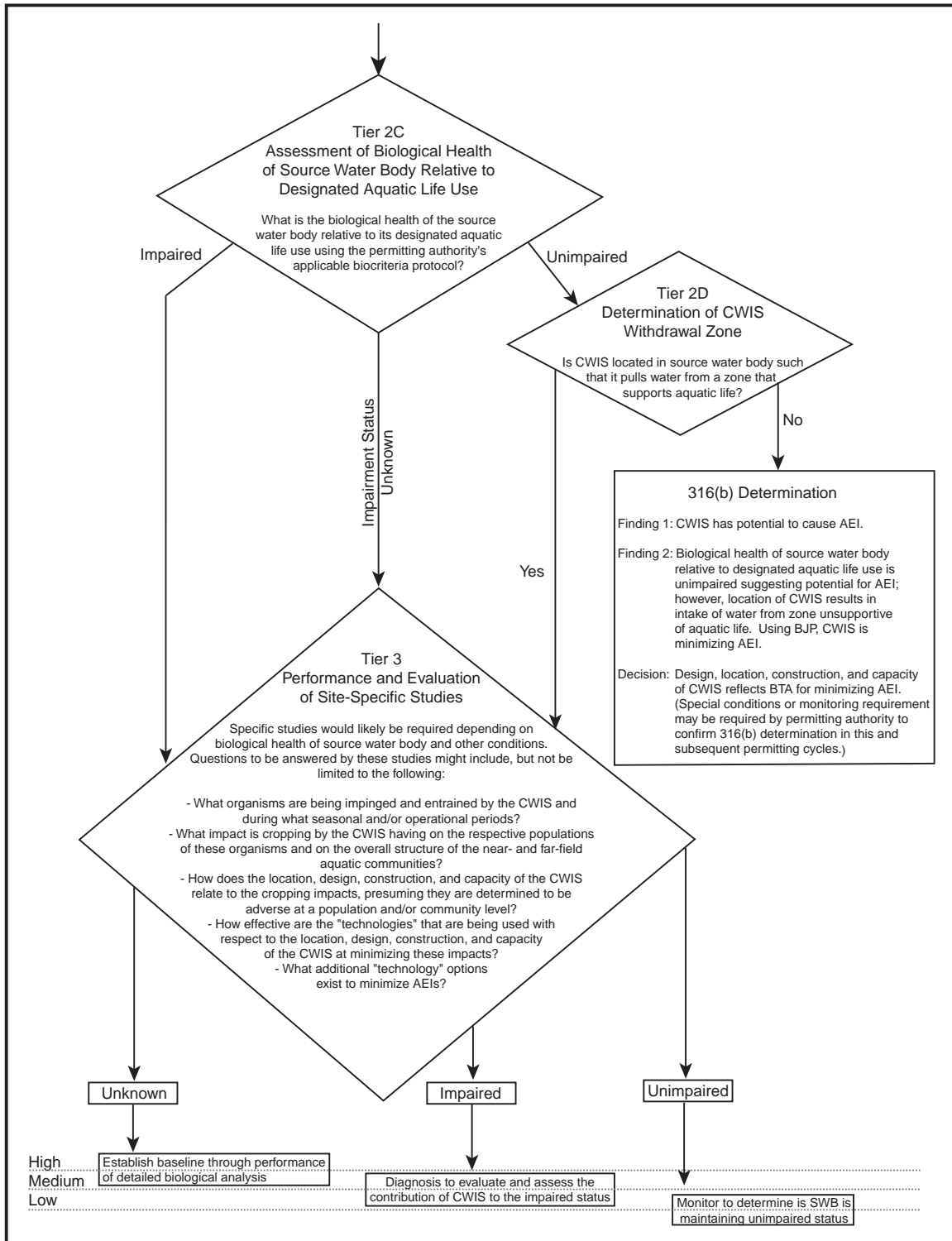


Figure 3-1 (continued)  
Section 316(b) Draft Regulatory Flow Chart. Source: Nagle and Morgan (1999)

EPA's **Tier 1** involves a screening level of analysis, and requires information on certain performance characteristics of the CWIS (e.g., approach velocity, intake flow) as well as environmental characteristics of the site (e.g., presence of threatened or endangered species, and physical features that could concentrate organisms near the CWIS). Based on this facility performance and environmental information, a determination is made that either the intake structure: 1) meets Tier 1 criteria for minimizing adverse environmental impact (AEI) and can be removed from further consideration, or 2) has the potential to cause AEI and a Tier 2 analysis is needed. EPA is currently developing Tier 1 criteria.

EPA's **Tier 2** consists of an evaluation based on the source water body's designated uses and biological status. Currently, the two possible outcomes of this evaluation are: 1) the intake structure is considered to meet the requirement of minimizing AEI because there is no aquatic life use designation for the source water body or because the intake structure only withdraws water from a zone unsupportive of aquatic life within a biologically unimpaired source water body, or 2) more information is required and a Tier 3 analysis is needed.

Bioassessment methods are an integral part of EPA's process for characterizing the biological status of the source water body. Therefore, it is in Tier 2 that biocriteria play an important role in § 316(b) implementation. In addition to facilitating the characterization of the source water body as impaired, unimpaired, or unknown, biocriteria also help identify the appropriate intensity and nature of studies to be conducted under Tier 3 when such additional investigations are required.

EPA's **Tier 3** involves further investigations and analyses which must be tailored to assess the intake structure's potential to cause AEI, so that appropriate options to minimize those impacts can be identified. The amount of additional data needed to make a § 316(b) determination at the Tier 3 level is viewed by EPA as being dependent on the source water body's biological status. Therefore, the intensity of studies called for in Tier 3 can be low, medium, or high, depending upon whether the biological status of the source water body was classified in Tier 2 as unimpaired, impaired, or unknown. The nature of Tier 3 studies will vary on a case-by-case basis according to the types of organisms likely to be impacted by impingement and entrainment. Available bioassessment data and any established biocriteria standards are expected to play an important role in Tier 3 study design.

EPA hopes to further its goals of providing broader protection of aquatic ecosystems and of basing that protection on meaningful measures of environmental condition through incorporation of biocriteria into this evolving § 316(b) decisional framework. As previously discussed, biological criteria and bioassessment methods provide a more direct and comprehensive evaluation of the biological integrity of a water body than do chemical or physical criteria. Thus, biocriteria have the potential to yield more robust assessments for management and regulation.

For example, bioassessment techniques rely on direct assessment of biological assemblages rather than on extrapolation from effects on individuals (often in the laboratory) to effects on populations, communities, and ecosystems, as is typical of chemical water quality criteria. Section 316(b) demonstrations often project effects on higher levels of biological organization (e.g., populations and ecosystems) from estimates or counts of entrained and impinged individuals. Since direct assessment of biological assemblages does not rely on conservative assumptions to extrapolate from individuals to higher levels of organization, bioassessment has

the potential to provide more realistic assessments of the ecological integrity of a water body for § 316(b) analyses.

At present, the role of biocriteria in EPA's draft decisional framework is confined to Tier 2 analyses. In Tier 2, biocriteria will be used to assess the "biological integrity" of the source water body. This application of biocriteria within the decisional framework is consistent with the primary purpose for which biocriteria and bioassessment were developed.

If biocriteria were additionally applied in Tier 3 assessments, their usefulness may be more limited. In Tier 3, the potential magnitude and ecological significance of entrainment and impingement impacts are assessed. By design, biocriteria and multimetric bioassessment are not stressor- or regulation-specific. A premise of EPA's biocriteria program is that multimetric indices such as those described in EPA guidance documents yield reliable assessments of biological integrity of water bodies without regard to the stressor(s) involved. In fact, one of the major criticisms of multimetric indices is that such indices have limited diagnostic ability (Suter 1993).

Nonetheless, stressor identification using the multimetric approach is an area of active research. EPA plans to publish biocriteria guidance on stressor identification by 2003 (USEPA 1998b), and more limited biocriteria guidance addressing stressor identification in the context of the TMDL program will be available by the summer of 2000 (D. Reid-Judkins, USEPA, pers. com.). Furthermore, relatively recent guidance for lakes and reservoirs (USEPA 1998a) and unpublished draft guidance for estuaries and coastal marine waters (Gibson et al. 1997) place a greater emphasis on collection of ancillary information (e.g., nutrients and contaminants) that may be used to help identify the source(s) of impairment than have previous EPA biocriteria guidance documents.

Specific evaluation of the contribution of impingement and entrainment to water body impairment would represent a significant expansion of biocriteria's role beyond that of detecting water body impairment in Tier 2. Such a role also raises a number of regulatory and technical questions which are addressed in subsequent sections. At the present time, EPA's partitioning of assessment questions in Tiers 2 and 3 and use of biocriteria in Tier 2 alone (and for low intensity monitoring following Tier 3 assessment), may be an implicit acknowledgment of the strengths and limitations of the current biocriteria approach.

Biocriteria may be of help, however, in identifying a level of analysis that is appropriate for Tier 3 and in focusing Tier 3 investigations on the components of the aquatic ecosystem most likely to be adversely affected by impingement and entrainment. Multimetric bioassessment conducted in Tier 2 could also provide information useful for designing more detailed analyses conducted in Tier 3. ***The manner in which this information will be used, though, has not been specified in the framework.***

Individual metrics of multimetric indices provide information on specific components of aquatic communities, and the raw data supporting those metrics provide quantitative, species-specific information that has been collected using well-defined protocols. In some cases, careful examination of the information used in the Tier 2 assessment may be adequate for use in the Tier 3 assessment. In other cases, additional data may be collected using the same sampling protocols to supplement other data collected in a Tier 3 assessment.

Tier 3 assessments will undoubtedly focus on population-specific and population-level questions. Multimetric indices, however, are designed to minimize reliance on population-level information by examining individual and community level attributes. This is done, in part, because the resources available in many situations for sampling and analysis are not adequate to detect population-level effects (Karr and Chu 1999). Consequently, multimetric bioassessment data may contribute to population-level Tier 3 studies, but this information is probably insufficient to rigorously answer many population-level questions.

Based on the above issues surrounding the use of biocriteria in EPA's three-tiered decisional framework for § 316(b), it is unlikely that biocriteria will eliminate the need for other, traditional assessment tools. In the context of § 316(b), then, bioassessment should be viewed as a complement to, rather than a replacement for, other methods of assessment.

## **General Technical Considerations**

Understanding EPA's proposed decisional framework for § 316(b) implementation and the specific role biocriteria will play in this framework is just the first step in the critical examination of biocriteria. General strengths, weaknesses, and research needs related to the use of biocriteria in § 316(b) regulation also derive from:

- The specific ways biocriteria investigations conducted under § 316(b) will be integrated with other CWA programs and initiatives, and
- The fundamentals of biocriteria development and application: designation of reference conditions, choice of target assemblages, index development, and promotion of an antidegradation policy.

Other general issues will undoubtedly arise as specific § 316(b) regulations and additional guidance are developed by EPA.

### ***1. Integration With Other EPA Programs and Initiatives***

Consistency within and among EPA's regulatory programs improves both the efficiency and effectiveness of environmental management. EPA has worked to achieve this result in its water programs through the adoption of the Watershed Protection Approach (WPA).

The WPA is an administrative and technical framework for enhancing the management of aquatic resources by basing management on hydrologic units such as watersheds and river basins. One of the objectives of the WPA is to maximize the effectiveness of monitoring and permitting activities under water programs through coordination of data collection and assessment and synchronization of permitting cycles. One of the ways this can be accomplished is by conducting joint assessments for the same water resource under different water programs, thereby eliminating potential duplication of effort.

In principle, much of the information collected for bioassessment under a § 316(b) biocriteria program for a particular water body will also be applicable to multiple CWA assessment requirements for that same resource. Specifically, bioassessment data requirements, protocols,

sampling locations, metric selection, calibration, scoring, and water body status determination for § 316(b) should be applicable to other assessment requirements. In fact, a single determination of impairment/non-impairment for a given water body could potentially be used to support the Water Quality Standards (WQS) program, the NPDES program (National Pollution Discharge Elimination System, § 402 of the CWA), § 305(b) reporting, and Tier 2 of the § 316(b) framework (see Table 3-1). This is possible, because in each case the assessment question is the same: is the biological integrity of the water body impaired?

Given this degree of overlap between assessment requirements, and the use of biocriteria within § 316(b) and EPA's other water programs, requiring a separate § 316(b) assessment of water body integrity using biocriteria would be duplicative. With regard to implementation of Tier 2 of EPA's draft framework, then, it would be unnecessary for EPA to issue separate biocriteria guidance specific for § 316(b) and to require separate assessment of water body integrity using biocriteria. Moreover, because the opportunity exists for joint assessments, a requirement of separate implementation of biocriteria under § 316(b) would appear to be inconsistent with the WPA objective of coordinating assessment efforts to avoid duplication.

**Table 3-1**  
**Applications of Lake Biological Monitoring Protocols and Biocriteria. Source: USEPA**  
**(1998)**

Program	Biological Monitoring and Assessment	Biological Criteria
<b>Section 305(b) Reporting</b>	<ul style="list-style-type: none"> <li>• Improving data for beneficial use assessment.</li> <li>• Improving water quality reporting.</li> </ul>	<ul style="list-style-type: none"> <li>• Identifying waters that are not achieving their aquatic life use support.</li> <li>• Defining an understandable endpoint in terms of “biological health” or “biological integrity” of waterbodies.</li> </ul>
<b>Section 314/Clean Lakes Program</b>	<ul style="list-style-type: none"> <li>• Assessing status of biological components of lake systems.</li> <li>• Measuring effects of ongoing restoration projects.</li> <li>• Measuring success of lake clean-up efforts and other mitigation activities.</li> <li>• Assessing lake trophic status and trends, assessing biological trends.</li> </ul> <p>[Monitoring and sampling needs vary for each lake.]</p> <p>[Clean Lakes Program Regulations monitoring components: algal pigments, algal genera, cell densities, algal cell volumes, limiting nutrients, macrophyte coverage, bacteria, and fish flesh analysis.]</p>	<ul style="list-style-type: none"> <li>• Identifying lakes that are not attaining designated use (including aquatic life use) support.</li> <li>• Defining lake biological integrity based on a reference condition.</li> <li>• Identifying impairments due to toxic substances.</li> </ul>
<b>Section 319/ Nonpoint Source Program</b>	<ul style="list-style-type: none"> <li>• Evaluating nonpoint source impacts and sources.</li> <li>• Measuring site-specific ecosystem response to remediation or mitigation activities.</li> <li>• Assessing biological resource trends within watersheds.</li> </ul>	<ul style="list-style-type: none"> <li>• Determining effectiveness of nonpoint source controls.</li> </ul>
<b>Watershed Protection Approach</b>	<ul style="list-style-type: none"> <li>• Assessing biological resource trends within watersheds.</li> </ul>	<ul style="list-style-type: none"> <li>• Setting goals for watershed and regional planning.</li> </ul>
<b>TMDLs</b>	<ul style="list-style-type: none"> <li>• Identifying biological assemblage and habitat impairments that indicate nonattainment of water quality standards.</li> <li>• Documenting ecological/water quality response as a result of TMDL implementation.</li> <li>• Priority ranking waterbodies.</li> </ul>	<ul style="list-style-type: none"> <li>• Identifying water quality-limited waters that require TMDLs.</li> <li>• Establishing endpoints for TMDL development, i.e., measuring success.</li> </ul>
<b>NPDES Permitting</b>	<ul style="list-style-type: none"> <li>• Measuring improvement or lack of improvement of mitigation efforts.</li> <li>• Developing protocols that demonstrate relationship of biological metrics to effluent characteristics.</li> </ul>	<ul style="list-style-type: none"> <li>• Performing aquatic life use compliance monitoring.</li> <li>• Helping to verify that NPDES permit limits are resulting in achievement of state water quality standard.</li> </ul>

<b>Program</b>	<b>Biological Monitoring and Assessment</b>	<b>Biological Criteria</b>
<b>State Monitoring Programs</b>	<ul style="list-style-type: none"> <li>Improving water quality reporting.</li> <li>Documenting improvement or lack of improvement of mitigation efforts including lake clean-up efforts, TMDL application, NPDES efforts, nonpoint source pollution controls, etc.</li> <li>Problem identification and trend assessment.</li> <li>Prioritizing waterbodies.</li> </ul>	<ul style="list-style-type: none"> <li>Measuring effectiveness of controls.</li> <li>Performing watershed planning.</li> <li>Performing regional planning.</li> </ul>
<b>Risk Assessment</b>	<ul style="list-style-type: none"> <li>Providing data needed to estimate ecological risk to assessment endpoints.</li> </ul>	<ul style="list-style-type: none"> <li>Development of an assessment or measurement endpoint.</li> </ul>
<b>Water Quality Criteria and Standards</b>	<ul style="list-style-type: none"> <li>Developing data bases for lake phytoplankton, macroinvertebrates, fish, plants, and other assemblages.</li> <li>Developing indices that assess lake biota compared to reference.</li> </ul>	<ul style="list-style-type: none"> <li>Identifying waterbodies that are not attaining aquatic life use support.</li> <li>Refining aquatic life use classifications.</li> <li>Developing site-specific standards.</li> </ul>

## **2. Issues Surrounding Biocriteria Development and Application**

In addition to issues raised by biocriteria’s role in § 316(b) and integration with other water programs, general concerns about biocriteria exist with respect to fundamental concepts of reference conditions, choice of assemblages, construction of the multimetric index, and EPA’s antidegradation policy for biocriteria.

### **a. Reference Conditions**

The concept of reference conditions is central to the entire multimetric bioassessment approach. It derives from the notion that each ecosystem has an ideal state it can be expected to achieve if not disturbed by human activity. This concept is at odds with a significant amount of current ecological understanding.

In the decades since enactment of the Clean Water Act and publication of the seminal works on multimetric bioassessment, there have been substantial changes in ecologists’ conceptions of ecosystem structure and function. However, these concepts have not yet been incorporated into the tools employed in aquatic ecosystem management (e.g., bioassessment). To the extent that these changes represent true advances in ecological understanding, they need to be integrated into the biocriteria approach.

Two critically important insights into the structure and function of ecosystems are:

(1) recognition of the dynamic character of ecosystems, and (2) awareness of the significance of context and scale (Christensen et al. 1996, Levin 1992, Wu and Loucks 1995). Wu and Loucks (1995) consider these concepts, which are discussed in detail below, part of a paradigm shift in ecology—from “balance of nature” to “hierarchical patch dynamics.”



The relevance of these concepts to effective management of ecosystems has been acknowledged by an increasing number of scientific bodies and by the scientific community at large. Among others, the Ecological Society of America's Committee on the Scientific Basis for Ecosystem Management, which comprises a broad group of government and academic scientists, has recognized the need to incorporate the elements of this new paradigm into environmental management (Christensen et al. 1996). Consequently, these concepts merit attention and are valid criteria for identifying strengths and weaknesses of current approaches to bioassessment, and for identifying research needs.

### *(1) Ecological Dynamics*

Within the hierarchical patch dynamics (HPD) paradigm, ecosystems are viewed as dynamic. Natural variability is a hallmark of a dynamic ecosystem. Thus, spatial heterogeneity, natural disturbance, stochastic recruitment, and within-patch biotic interactions generate a shifting mosaic of internally dynamic patches. This shifting mosaic varies over multiple time-space scales and levels of biological organization (Levin 1992, Pickett and White 1985, Reice 1994, Wilson 1992).

Based on this dynamic view of ecosystems, it follows that most ecosystems do not exhibit an ideal, equilibrium state that can be maintained indefinitely in the absence of disturbance; rather, ecological processes yield multiple equilibria and absence of equilibria, which creates and maintains diversity (Holling 1996, Wu and Loucks 1995).

This perspective differs sharply from the balance of nature paradigm. The latter derives from the turn-of-the-century view, promoted by Frederick Clements (1904), that communities and ecosystems are "superorganisms" that tend toward a fixed, normative state (De Leo and Levin 1997). The notion of a normative or ideal ecosystem state, captured by the modern analog of "ecosystem health" is the basis for public perceptions of nature and currently provides the conceptual framework for most environmental management and regulation (Botkin 1990). This conception of ecosystems is also central to multimetric bioassessment and biocriteria, which focus on the characterization of the normative state or ecological health of the ecosystem through the designation of reference conditions. As Karr and Chu (1999) describe:

*In essence, understanding baseline, or reference, conditions in different places is analogous to veterinarians' learning what indicates health in different kinds of animals.*

From an ecosystem health perspective, biological assemblages in naturally functioning ecosystems are at or near ecological equilibrium. Such ecosystems are considered to be tightly integrated and controlled at local scales by large-scale, long-term biogeographic and evolutionary processes (Angermeier 1997, Angermeier and Karr 1994, Karr and Chu 1999, Karr and Dudley 1981).

By contrast, HPD perceives these same ecosystems (that are minimally affected by human activities) as loosely integrated, non-equilibrium systems which, therefore, do not tend toward an ideal state. Instead, ecosystems are strongly influenced at local scales by stochastic processes, which are a source of natural variability within the system. Consequently, from an HPD perspective, these stochastic processes deserve protection rather than reduction. In fact,

according to some researchers, planned or unplanned reductions in natural disturbance or spatio-temporal variability often lead to loss of resilience and biodiversity, and invasion of non-native species (De Leo and Levin 1997, Holling 1996, Holling and Meffe 1996, Reice 1994, Reice et al. 1990, Vitousek et al. 1996).

These divergent conceptions of the dominant factors structuring biological communities at local scales have significant implications for biocriteria—specifically, they determine how variability among sites least influenced by human activities (reference sites) should be interpreted. From the HPD perspective, variability among reference sites is a manifestation of natural processes that need to be accommodated by the assessment tools and protected through management.

By contrast, from an ecosystem health perspective, this variability is noise in the assessment process that can be assumed to derive—at least in part—from unidentified anthropogenic stress. This latter characterization of natural variability is problematic. If natural variability among reference sites is not accommodated within an assessment framework, it could result in a site's misclassification as impaired. While this may appear to be protective of the resource, an assessment framework cannot protect components of biological integrity that it does not recognize. Assessment criteria become *de facto* management objectives and, in this case, potentially lead to anthropogenic reduction in natural variability. As noted above, reductions in variability may have undesirable long-term consequences for an ecosystem in the form of reduced resilience and biological diversity.

The preferred approach of establishing reference conditions (i.e., biological expectations), as discussed in Chapter 2, is to do so on a regional, geographic basis. Biological expectations are established for areas that are relatively homogeneous with respect to certain ecological attributes. Within these ecologically homogeneous regions, biological expectations are modified by local factors such as the size of the water body and naturally occurring habitat characteristics such as substrate type and salinity. These factors are used to classify reference sites in an effort to reduce variability among sites arising from these attributes.

This method of partitioning variation, however, does not currently address natural variability associated with ecological dynamics or processes. The presence of this underlying natural variability among reference sites can make the establishment of robust reference conditions a challenging, if not impossible, task.

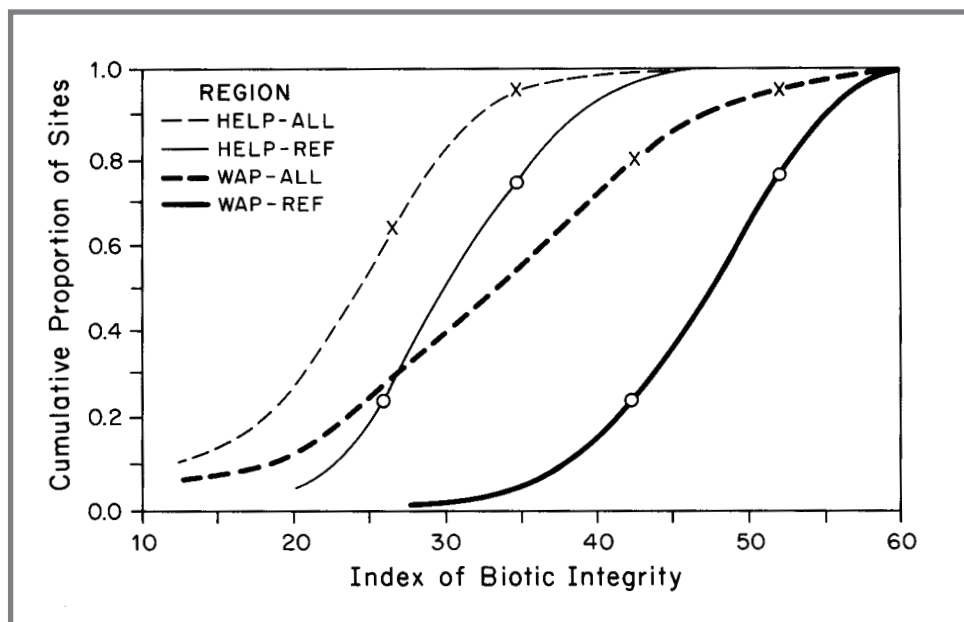
Reference conditions in a biocriteria framework typically are based on multiple reference sites within a region. These sites, in turn, are portrayed using a dozen or so biological metrics, which is the foundation of a multimetric approach. While these sites share the characteristic of being representative of the biota for that region, the sites themselves are not identical. The presence of this variability among sites introduces uncertainty into the reference conditions.

Uncertainty in reference conditions affects the utility of the indices for classifying sites as impaired or unimpaired. Specifically, it creates a trade-off between Type I and Type II error. A low rate of Type I error (false positives) is desirable because it indicates the bioassessment method will yield few false alarms; likewise, a low rate of Type II error (i.e. false negatives) is desirable because it indicates that the bioassessment method is sensitive to anthropogenic effects. The more overlap in index scores between impaired and unimpaired sites, the more significant the tradeoff between Type I and Type II error becomes. The practical consequence of this result

is that the classification of sites within a broad range of index scores is highly uncertain and any criterion within that range will yield both false positives and false negatives.

Evaluation of the statistical properties of the IBI (Index of Biotic Integrity) and similar indices has been limited. Fore et al. (1994) assessed the ability to discriminate individual sites based on their IBI scores using data from Ohio EPA's bioassessment program. They found that the IBI has high power to discriminate among fish assemblages at different stream sites. Precision is an important characteristic of biological indices such as the IBI, but power to distinguish among sites, per se, is not relevant. Of direct relevance is an index's ability to discriminate degrees of anthropogenic effect, or, in a regulatory context, to discriminate sites with anthropogenic effects from unimpaired sites. Fore et al. (1994) did not examine this issue when evaluating IBI.

The earlier work of Hughes and Noss (1992), while not evaluating the statistical properties of the IBI, used the index to characterize the proportion of sampled sites in Ohio that were impaired. They presented data similar to those in Fore et al. (1994) showing substantial overlap of cumulative distributions of IBI scores for all sites within two regions and corresponding distributions of scores for regional reference sites (Figure 3-2). Because of the presence of variability among the reference site IBI scores, the 25<sup>th</sup> percentile of reference site scores was used to delineate impaired from unimpaired conditions. This approach is consistent with EPA guidance (Barbour et al. 1996a, Gibson et al. 1996, Ohio EPA 1990, USEPA 1998a). As a consequence of adopting a 25<sup>th</sup> percentile threshold, 64-80% of all sites were classified as impaired. In their example, however, only 0-30% of the sites had IBI scores below the range exhibited by reference sites. Thus, 50-64% of all sites fell within the range of reference site scores but below the arbitrary threshold set at the 25<sup>th</sup> percentile. This example highlights the importance of properly interpreting variability among reference sites.

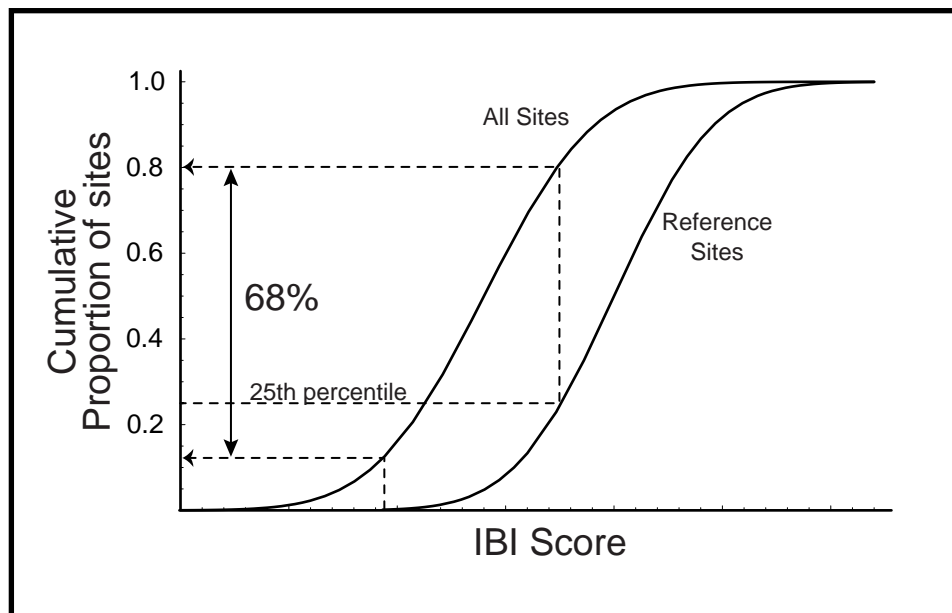


HELP = Huron-Erie Lake Plain, WAP = Western Allegheny Plateau.

**Figure 3-2**  
**Cumulative Distributions of IBI Scores for All Sites and Reference Sites in Two Ecoregions of Ohio. Source: Reproduced With Permission From Hughes and Noss (1992)**

Constraining the range of reference conditions by use of a 25<sup>th</sup> percentile is “conservative” only to the extent that the IBI is a monotonic function of biological integrity. The use of a 25<sup>th</sup> percentile threshold value for reference site scores represents an attempt to cope with the uncertainty of reference conditions. As illustrated in the work of Hughes and Noss, however, this may produce inappropriate results when evaluating impairment. That is because the use of a threshold value does not discriminate between natural variation and anthropogenic influence but rather treats all sites falling below the threshold value the same.

Figure 3-3 further illustrates this phenomenon using a hypothetical case in which 68% of all sites are within the range of IBI scores for reference sites but below an arbitrary threshold set at the 25<sup>th</sup> percentile. On this basis alone, these sites are technically unimpaired. Yet, because these same sites also fall below an arbitrary threshold at the 25<sup>th</sup> percentile, they will be designated as impaired. This example demonstrates that the range of uncertainty for reference sites (attributable in part to natural variability) can encompass a large portion of the test sites with the result that a significant portion of the test sites may be inappropriately classified as impaired. The causes and implications of this phenomenon and related ones need to be rigorously examined, and appropriate methodological changes should be made to ensure that biocriteria achieve an appropriate balance of Type I and Type II error, and natural processes that generate variability are protected so as to maintain biological integrity in the long term.



**Figure 3-3**  
**Hypothetical Case in Which 68% of All Sites Have IBI Scores Within the Range of Scores Observed at Reference Sites, but Below an Arbitrary Threshold Set at the 25th Percentile Score for Reference Sites**

Use of the 25<sup>th</sup> percentile of reference site index scores as a threshold creates the paradoxical situation in which a significant portion (i.e. 25%) of sites classified *a priori* as reference sites are subsequently classified as impaired based solely on their index scores. Implicit in this reclassification is the notion that selection of reference sites is not sufficiently rigorous and that “reference sites” in the bottom 25% of the reference site distribution are, in fact, impaired. Ohio

EPA (1987) explicitly acknowledges this assumption as its rationale for using the 25<sup>th</sup> percentile as a threshold. The 25<sup>th</sup> percentile constitutes a safety factor in the derivation of numeric biocriteria and is intended to represent the “typical performance” of the ecoregion and site type (Ohio EPA 1990). Regardless of the merits of this assumption, metric selection and calibration may have incorporated conditions at sites subsequently classified as impaired, further complicating interpretation of the index. Contrary to Hughes and Noss (1992), Ohio EPA applies a downward adjustment of four IBI units to account for “non-significant departure” from the 25<sup>th</sup> percentile. The result is that the range of IBI scores for unimpaired sites encompasses 90-95% of the reference sites (C. Yoder, Ohio EPA, pers. comm.).

Because of natural variability, classification of sites as impaired or unimpaired based on biological characteristics will always have some error. One means of addressing this problem within the current framework is to explicitly acknowledge that index scores for impaired and unimpaired sites overlap and assign probability of membership in impairment classes based on metric scores as the state of Maine does in its water quality standards program (Davies et al. 1993). Maine is unique among states in employing a probabilistic framework to its biocriteria program.

## *(2) Scale Considerations*

A second and related aspect of current ecological knowledge that is relevant to biocriteria and development of reference conditions is recognition of the significance of context and scale.

Ecological systems are structured on multiple space-time scales by a variety of biotic and abiotic factors and processes, such as climate, geology, topography, hydrology, succession, disturbance, predation, competition, migration, and dispersal. Each of these factors and processes produces variability at characteristic temporal and spatial scales. Furthermore, as stated by Levin (1992) “each individual and each species experiences the environment on a unique range of scales, and thus responds to variability individualistically. Consequently, no description of the variability and predictability of the environment makes sense without reference to the particular range of scales that are relevant to the organisms or processes that are being examined.”

Biological integrity comprises biodiversity and the processes that create and sustain biodiversity (Angermeier and Karr 1994). Since biodiversity and its sustaining ecological processes are multi-scaled, it follows that biological integrity must be assessed at multiple spatial and temporal scales (De Leo and Levin 1997). Over the last twenty years, some leaders in multimetric index development have made this point repeatedly with respect to space (Karr and Chu 1999, Karr and Dudley 1981). Current methods, however, generally impose a single scale of human observation, analysis, and interpretation—that of the sample site at a point in time. A notable exception is the recent work of Moyle and Randall (1998) that evaluated biological integrity at the scale of whole watersheds based on watershed-scale metrics.

While a biocriteria program often relies on use of regional reference conditions (based on multiple reference sites) to assess impairment, this approach does not address the problem of multiple scale and biodiversity. That is because regional reference conditions, like individual reference sites, identify species richness or diversity at the site level only and do not incorporate cross-site diversity. This point is illustrated by the fact that two different reference sites can have

the same IBI score yet different community compositions. Based on this scoring alone, there is no way to discern whether, over time and space, there has been an overall reduction in species diversity across sites in the region.

Even if the perspective is expanded to encompass multiple sites over a larger geographic area, one's ability to observe phenomena at the larger scale is compromised by site-level data reduction. For example, sites within a watershed or region often have similar species richness, but the species lists only partially overlap. Homogenization of the biota among sites—a common ecological response to anthropogenic alterations of the environment—can offset loss of species found at only some sites. The net result is greater overlap in species lists and no change—or even an increase—in species richness at the scale of the site, but a decrease in species richness at the scale of the watershed or region.

Creation and maintenance of spatial heterogeneity is a hallmark of biological integrity; however, current biotic indices used in a biocriteria approach have limited ability to quantify such diversity. While these indices capture site-specific information on biological assemblages, they do not consider between-site differences in species composition, except among a few relatively broad taxonomic, functional, or other groupings such as tolerants, intolerants, top carnivores, and omnivores. Explicit consideration of observational scale and its implications will help ensure that indices are appropriate for assessing the range of ecological structures and processes and anthropogenic stresses that are of concern.

Scale considerations will be especially important as assessment methods developed in one type of environment are modified for a variety of water body types and sizes (see Addicott et al. 1987). In particular, application of the reference site concept for larger systems where power plants are located will prove to be more difficult for at least two reasons. First, the affected systems present a much larger scale of analysis and, thus, greater opportunity for effects associated with spatial heterogeneity. Second, there are fewer affected systems on this scale and still fewer that are sufficiently unaffected by anthropogenic stress to serve as reference sites for evaluating effects on spatial heterogeneity and species composition (see Chapter 4).

### *(3) Existing Conditions and Designated Uses*

Incorporation of biocriteria into state Water Quality Standards (WQS) programs requires that an explicit linkage be made between biocriteria and designated uses. Designated uses are water body-specific uses such as drinking water supply, shipping, primary contact recreation (i.e., swimming), secondary contact recreation (i.e., boating), and various forms of aquatic life support (e.g., shellfish production) that are specified in a state's water quality standards.

Designated uses reflect society's goals for a water body and what is reasonably attainable given historical modifications to the water body and its surroundings. Many water bodies have been permanently altered by major human activities such as urban development, channelization, impoundment, and intentional and unintentional introduction of non-indigenous species. These essentially irreversible alterations are reflected in designated uses.

Many aquatic life designated uses are tiered to reflect differing societal goals and degrees of habitat quality. For example, within a given ecoregion a cold water fishery designation generally

reflects better habitat quality than does a warmwater fishery designation. A stream capable of supporting cold water fish such as trout two centuries ago when the watershed was entirely forested may support only warmwater fish following deforestation and conversion of the landscape to agricultural use. The aquatic life use designation for such a stream would be warmwater fishery habitat rather than cold water fishery habitat in acknowledgement of the practical irreversibility of the modifications that have occurred. Biological criteria for such a stream would be based on the designated aquatic life use for the water body.

Specification of biocriteria for these types of water bodies presents additional challenges to the multimetric index approach. While index-based biocriteria are related to both reference conditions and designated uses, the relationship between designated uses and reference conditions is not straightforward. EPA guidance (Gibson et al. 1996, USEPA 1998a) is explicit that biocriteria must be attainable; however, that guidance also consistently maintains that reference conditions should approximate conditions expected at minimally impaired sites that approximate natural conditions. The only exceptions to this are for reservoirs, which are inherently artificial systems, and for highly modified regions where undisturbed reference sites are completely lacking.

Implicitly, the reference conditions upon which biocriteria are based will not necessarily be attainable in all cases. According to EPA guidance (Gibson et al. 1996), biocriteria for water bodies that are not expected to conform to the standard of minimally impaired conditions should be set through downward adjustment of biocriteria (e.g., the threshold of acceptable index values) to conform with designated uses for the water body. EPA's guidance for streams and small rivers (Gibson et al. 1996) states, "... these decisions should not compromise the objective of defining the natural state. Biocriteria can be qualified by the assignment of designated uses, but the reference condition should describe the site as one would expect to find it under natural or minimally impaired conditions." Thus, the biocriteria are derived from an index that remains tied to the minimally impaired reference condition rather than through specification of a less stringent, attainable reference condition and development of an index with that attainable condition as the endpoint.

In varying degrees, states that have incorporated numeric biocriteria into their WQS programs have departed from this approach. Ohio EPA sets criteria based on percentiles of index values at reference sites. For exceptional warmwater fishery habitat (the highest warmwater aquatic life use designation in Ohio's WQS program), the biocriterion is the 75<sup>th</sup> percentile index value of all reference sites combined. For warmwater fishery habitat waters, the criterion is set at the 25<sup>th</sup> percentile index value of the applicable reference sites in the corresponding ecoregion. This is problematic for the reasons described in the preceding section on ecological dynamics.

Ohio also has biocriteria for warmwater habitats not expected to meet CWA goals. These water bodies are designated modified warmwater habitat following a Use Attainability Analysis documenting an inability to meet minimum CWA goals because of extreme, irretrievable habitat modification (Ohio EPA 1990). Biocriteria for these water bodies are derived using "habitat modified reference sites" exhibiting similarly severe habitat modifications but lacking other sources of stress such as point source discharges (Ohio EPA 1987c). This approach for developing biocriteria for highly modified water bodies appears to depart significantly from USEPA guidance.

The state of Maine does not rely on minimally impaired reference conditions and IBI-like multimetric indexes in its biocriteria program. Rather, tiered aquatic life use designations are the starting point and primary focus for biocriteria development. Sites are assigned to one of three narrative aquatic life use classes (or to a class constituting non-attainment of the lowest use designation) based solely on an evaluation of biological data by a panel of biologists. A discriminant function comprising a suite of biological metrics is then derived that classifies the sites so as to maximize consistency with the classification provided by the panel of biologists. The resulting criteria comprised by the discriminant function are direct, numeric representations of the aquatic life use designations and narrative biological criteria. When applied to test sites, the method allows for statements of probability of membership in alternative classes based on the full, potentially overlapping ranges of biological conditions observed within each class. The method, however, relies on a database for criteria development that adequately represents biological variability within each class, as well as appropriate classification of sites in the database by the panel of biologists.

Both of these examples are for streams and small rivers. Consideration of designated uses reflecting significant departures from minimally impaired, natural conditions will be particularly important as biocriteria are developed for larger water bodies that have been physically and biologically altered to support fisheries, navigation, flood control, power production, municipal and industrial water supply, and irrigation. Deliberate and accidental introductions of exotic species (e.g., salmon, trout, carp, zebra mussels) further complicate the “minimally impaired, natural condition” aspiration for reference condition identification. Power plants with CWIS typically are located on large water bodies.

#### *(4) Near Field-Far Field Approach*

A final issue for consideration concerning the reference condition concept arises from the potential application of the *near field-far field* and *upstream-downstream* approaches to § 316(b) biocriteria assessments. These approaches are used to establish site-specific reference conditions when regional and local reference sites cannot be developed due to a lack of adequate reference locations.

Both the near field-far field approach and the upstream-downstream approach are of potential relevance to § 316(b) assessments because of the difficulty that may arise in finding regional reference sites for water bodies with CWISs sited on them. The § 316(b) context, however, differs from the point-source discharge context in which these approaches are usually applied. As discussed below, an important difference in the case of § 316(b) assessments is that the adverse environmental impacts of impingement and entrainment primarily affect mobile aquatic species whose impacts are less detectable under these approaches. The implications of this difference need to be considered before these approaches are adopted for use in a § 316(b) regulatory context.

In order for the near field to reflect adverse effects of the CWIS, the replacement of entrained individuals by redistribution or dispersal into the near field must be low relative to mortality induced by the CWIS. Thus, the near field-far field approach is most defensible for organisms that are sedentary or capable of sensing and avoiding adverse conditions in the near field. In an investigation of the utility of the near field-far field approach at a near-coastal wastewater



outfall, Gibson (1995) found that results were far more promising for benthic invertebrates than for the more mobile benthic fish. Additionally, factors controlling the local distribution of benthic macroinvertebrates are better understood than they are for fish populations.

Species that are vulnerable to significant levels of impingement and entrainment are vulnerable precisely because they are not sedentary and cannot actively avoid the intake. Furthermore, impingement and entrainment mortality at a CWIS cannot be sustained for any period of time without a flow of organisms into the near field to replace those that are killed. The effects of impingement and entrainment on the abundances of those species in the near field are, thus, confounded by redistribution of organisms. That is, organisms killed by impingement or entrainment can be quickly replaced by others that disperse into the intake area. In this case, comparison of the biota in the near field and far field is not a valid indicator of effect.

Redefinition of the near field and far field may seem like an obvious solution; however, that also is problematic because the spatial scale over which dispersal occurs is large relative to the other constraints on the size and location of the near and far fields. The near and far fields must be close enough together that they are similar in all respects except for entrainment. They must be far enough apart that the effect of entrainment in the far field is attenuated. Determination of the appropriate spatial scales for near field-far field studies will be species-specific and site-specific. For assemblage-level assessments, consideration must be made of all of the species comprised by the index. In the case of highly mobile species such as many fishes, near field and far field may not be definable in any way that is useful for assessing adverse environmental impacts of impingement and entrainment.

## **b. Choice of Assemblages**

EPA guidance identifies several assemblages that may be used in biocriteria programs and recommends use of multiple indices based on different assemblages. Recommended assemblages include fish, zooplankton, phytoplankton, benthos, sedimented diatoms, and aquatic plants (USEPA 1998a). Choice of assemblage(s) raises several interrelated technical, regulatory, and policy questions.

Use of multiple assemblages is consistent with the goal of obtaining a comprehensive assessment of biological integrity of the water body. Assemblages differ in how they respond to specific anthropogenic stresses and how they respond to various forms of natural environmental variability in both time and space (Karr and Chu 1999). The multimetric index approach relies on differential response to anthropogenic stresses by biota, observed as the departure from expected abundance of one taxonomic or functional group relative to another. Some differences, however, appear at broader taxonomic levels represented by different assemblages.

Use of information on multiple assemblages may assist in the process of identification of stressors. Yoder and Rankin (Yoder 1991, Yoder and Rankin 1995b) combined multimetric information from both fish and macroinvertebrate assemblages at multiple sites and developed “biological response signatures”. They were able to identify severe biological degradation attributable to toxics, but otherwise were not able to identify the type or source of stress using the biological metrics. Several different stressor types and sources were considered, but

impingement and entrainment were not among the stressor types examined and no reason was given for not including them in the analysis.

A more challenging task is to discriminate among stressors when degradation is less severe and to assess the contribution that impingement and entrainment make to impairment when multiple factors are operating. Within EPA's draft regulatory framework, assessment of the contribution of impingement and entrainment to water body impairment is a Tier 3 question and biocriteria have no clearly specified role (except for possible low-intensity monitoring to ensure continued attainment of water body integrity). The technical basis does not yet exist to support use of multimetric biocriteria for diagnostic purposes in Tier 3, and it is questionable whether a technical foundation supporting that role can be developed on a time scale relevant to the emerging § 316(b) framework and regulation.

The appropriateness of various assemblages for § 316(b) biocriteria evaluations is highly dependent on the role of biocriteria in the § 316(b) decision-making process. Once a role has been finalized, a clear framework is needed for integrating the results from assemblage-specific indices into the decision-making process. Which assemblage or assemblages are judged relevant to § 316(b) evaluations? Many states have only benthic macroinvertebrate programs in place. If multiple assemblages are used, how are conflicting assessments of impairment resolved? Is a weight-of-evidence approach used, in which all assemblages are considered, and if so, are they weighted equally, or does a particular assemblage—such as fish—trump all others? Policy and regulatory decisions that have not yet been made may blur the distinction between Tier 2 (assessment of water body condition) and Tier 3 (evaluation of the magnitude and ecological significance of entrainment and impingement) determinations.

Two assemblages recommended by EPA for use in all water body types are benthic macroinvertebrates and fish. These two assemblages are the ones most commonly used in existing bioassessment programs. However, their use in § 316(b) biocriteria assessments presents many challenges.

Clearly, impingement and entrainment have their greatest potential effects on fish. Yet, fish have several attributes that are problematic for assessment. Fish exhibit migratory behavior and changes in habitat use with age. They also have the ability to alter their distribution over broad areas in response to natural variation in environmental conditions such as salinity, water temperature, and availability of prey. Smaller-scale patchiness is often superimposed on a larger-scale pattern.

The factors responsible for short and long term changes in fish distribution are not always easily discerned and they may differ among species. Individuals are entrained during their early life stages but assessment occurs at later stages. In the case of anadromous fishes, they may not return to the area of entrainment for several years. During the intervening developmental period, the fish may be exposed to stressors that are displaced in space as well as time. These factors may have an overriding influence on abundance. Many species exhibit ontogenetic shifts in habitat use and migratory behavior. Apart from their use by certain fish species, some of these habitats may have little functional connection to the water body where the CWIS is sited and the bioassessment data are collected. The "ecological neighborhood" (i.e., the spatial region that is relevant to an organism or assemblage and ecological process, *sensu* Addicott et al. 1987) of an anadromous fish may stretch from the upper reaches of a river to distant coastal areas. For

commercially and recreationally exploited species, harvesting of adults often has an overriding influence on population dynamics, with community- and ecosystem-level implications (Hall 1999, NRC 1999).

While benthic macroinvertebrates are, by far, the most commonly used assemblage in biocriteria programs across the country, they are generally poor candidates for assessing impacts under § 316(b). Even though benthic invertebrates are tied more closely to the sample site, with some exceptions (e.g., crabs and shrimps, early life pelagic stage of some molluscs such as oysters) they are less directly affected by impingement and entrainment.

The use of other assemblages besides fish does not necessarily enhance the assessment of potential entrainment and impingement impacts. Other assemblages are far more likely to reveal effects of stressors unrelated to § 316(b) than they are to reveal impingement and entrainment effects. They may indicate an absence of water body impairment when a fish index indicates a problem, or one or more of them may indicate a problem when a fish index provides no indication of a problem. In either case, decisions must be made about how inconsistent results among the various indices in large systems will be interpreted in the context of § 316(b) regulations. Future research will need to determine why inconsistencies among indices exist and what those inconsistencies tell us about a given site.

Several factors are known to produce inconsistent results among indices for different assemblages. Research on lotic systems in Ohio showed that consistency between fish and invertebrate indices declined with increasing water body size (Yoder and Rankin 1995a). Karr and Chu (1999) suggest such inconsistencies reflect differences in sensitivities or differences in sampling effectiveness among the various assemblages.

Other potential causes of inconsistent results among indices include differences in ecological neighborhoods (Addicott et al. 1987) and differences in source and sink habitats among the populations constituting the respective assemblages. Pulliam (1988) argues that for many populations a large fraction of the individuals may occur in “sink” habitats, where local reproduction may occur but is insufficient to balance local mortality. Populations can persist in such habitats only because they are sustained by continual immigration from more productive “source” areas nearby. Thus, effects of a localized stress in a sink habitat can be ameliorated by nearby source areas. Conversely, localized stress in a source habitat would first be manifested in sink habitats rather than the site where the biota are exposed to the stress.

Such source-sink relationships are species-specific and can confound site-level investigations of anthropogenic stress and biological effect. In extreme cases, the assemblage of species at a site may be an artifact of the type and proximity of neighboring habitats rather than local conditions (Pulliam 1988). When selecting assemblages and interpreting results, ecological neighborhoods and the potential influence of source-sink relationships on local assemblages should be considered.

## c. Multimetric Index Development

### *(1) Metric Selection and Calibration*

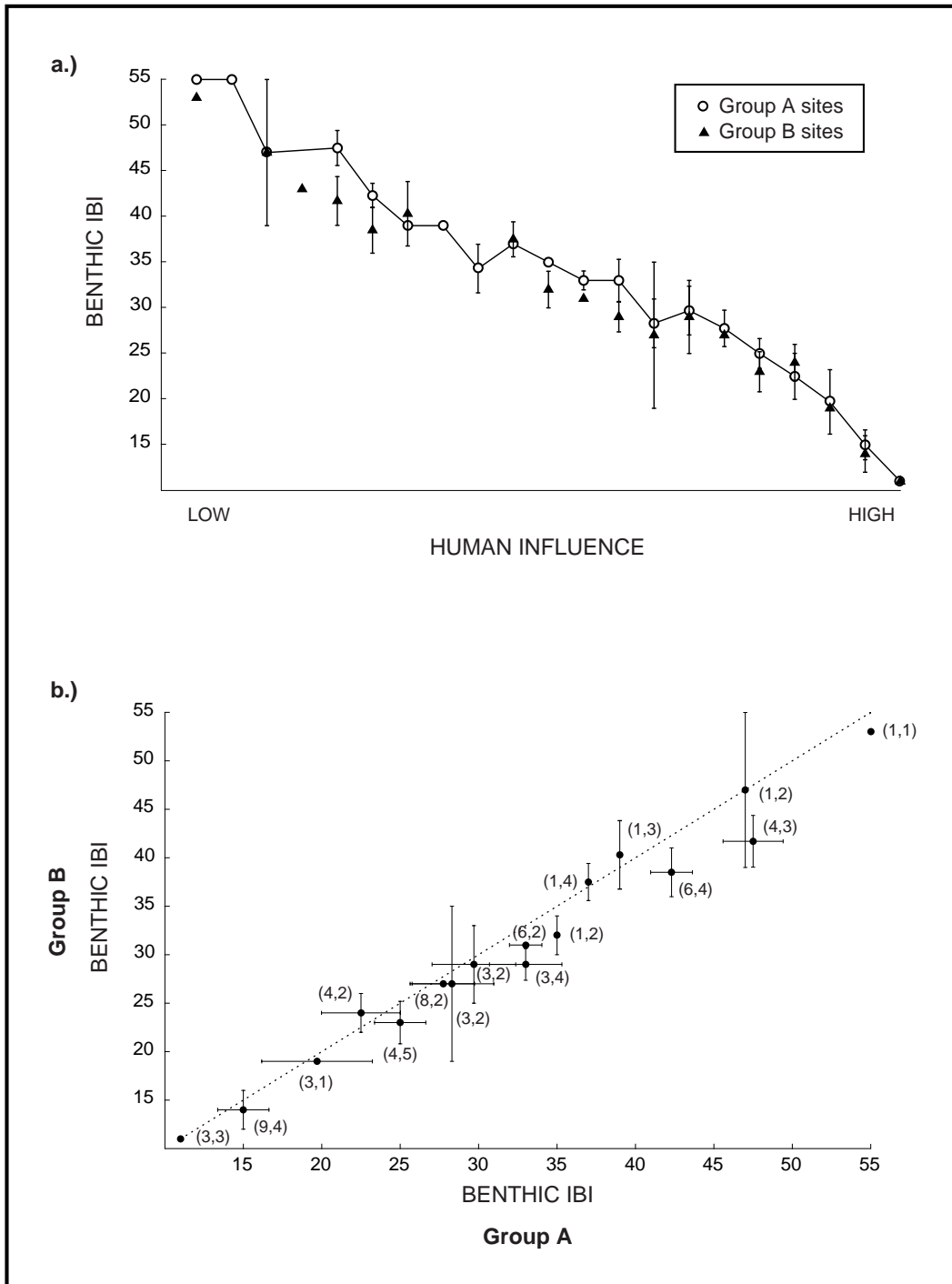
Karr and Chu (1999) assert that no metric should become part of a multi-metric index before it has been thoroughly and systematically tested and its response has been validated across a gradient of human influence. Rossano (1995, 1996) applied that relatively rigorous approach in developing an Index of Biotic Integrity for Japanese streams (IBI-J) based on benthic macroinvertebrates. She split her data, developed the index using one part of the data, and evaluated the performance of the index using the other part.

She classified 113 sites according to their relative degree of human influence and split the sites into two groups. Sixty-six sites (Group A) were used to select and calibrate 11 biological metrics that showed a strong relationship to human influence. The remaining 47 sites (Group B) were scored following index development, for the purpose of validating the index. IBI-J scores for the two groups showed a strong and similar response to human influence (Figure 3-4a); however, sites used to validate the index (Group B sites) tended to have slightly lower IBI scores than Group A sites in the corresponding human influence class (Figure 3-4b).

This bias could be an indication of the IBI-J's ability to respond to anthropogenic stressors that were not explicitly included in the analysis. Such a finding would reinforce the perception of multi-metric bioassessment as a comprehensive assessment tool.

Alternatively, the bias could be an artifact of the methodology for selecting and calibrating metrics, and a manifestation of limited ability of the IBI-J to accommodate natural variability. In that case, the bias is an indicator of potential for Type I error (false positives) when the index is used to classify sites. The magnitude of the bias could be expected to increase as the index is applied over larger temporal and spatial scales, because larger temporal and spatial scales will encompass a greater range of natural variability. In their bootstrap analysis of fish IBI data from Ohio, Fore et al. (1994) also found that IBI scores were negatively biased (i.e., on average below the true value); however, they concluded that the bias was small.

This phenomenon should be examined closely to gain an understanding of the cause of the bias, its statistical significance, its potential magnitude, and factors that tend to make it larger or smaller. Understanding this bias is especially important because many implementations of biocriteria will not be as rigorous as those developed by Rossano and Ohio EPA. The most straightforward way to examine the phenomenon is through Monte Carlo simulation of the index development and application process. Additional description of the approach is provided in the section on research needs.



**Figure 3-4**  
**Benthic IBI Scores by Human Influence Class for Japanese Streams. Error Bars Indicate Plus and Minus 2 Standard Errors. Panel a Shows IBI Scores for 66 Sites Used to Develop the IBI (Group A Sites) and for 47 Sites Scored Following IBI Development (Group B Sites). Panel b Shows the Mean IBI Scores for 17 Human Influence Classes Represented in Both Group A and Group B. Numbers in Parentheses Indicate the Number of Sites in Group A and Group B, Respectively. Data Are From Rossano (1995).**

## *(2) Metric Scoring and Index Construction*

Recall from Chapter 2 (Primer on Biocriteria), that EPA guidance describes construction of an index from individual metrics as a two-step process. First, metric values (e.g., total number of fish species) are assigned a score (e.g., 1, 3, or 5). This is done one of two ways, depending on the availability of reference sites (Figure 2-4). If reference sites are of sufficient number and quality, the “bisection method” is used. The 25<sup>th</sup> percentile of the values for a given metric is chosen as the threshold for a score of 5, and the range of metric values below that threshold is bisected. Sites with metric scores in the upper half of the bisected interval are assigned a score of 3 for that metric, and sites with metric values in the lower half of the bisected interval are assigned a score of 1.

The alternate, “trisection”, method is applied in cases where the number and quality of reference sites is judged to be insufficient to apply the bisection method, but the set of all sites covers a wide range of biological integrity, to include some minimally impaired sites. The range of metric values extending from the minimum value to the 95<sup>th</sup> percentile value is trisected. Sites with metric values in the upper third of the interval are assigned a score of 5 for this metric, sites with metric values in the middle third of the range are assigned a score of 3, and sites with metric values in the lower third are assigned a score of 1 for this metric.

This procedure transforms metric values to a uniform, unit-less scale. The index value is computed as the sum of all metric scores. Thus, the minimum value of the index equals the number of metrics constituting the index, and the maximum value of the index equals the number of metrics multiplied by the maximum value of each metric (i.e., 5). For a twelve metric index such as the fish IBI, the index ranges from 12 to 60. Other scoring systems exist, but the system described here was used in the original IBI and remains the most popular.

Each stage of the process for scoring metrics and constructing the index transforms the data in arbitrary and variable ways. For example, according to EPA guidance, the basis for selecting a scoring method (i.e. bisection or trisection method) is based on some level of “confidence” that the reference sites adequately represent unimpaired conditions (USEPA 1998a). Additionally, the lower end of the interval has the poor statistical properties of the minimum value, while the upper end of the interval has statistical properties related to the upper 95<sup>th</sup> percentile (of the same sample) or the 25<sup>th</sup> percentile (of a subset of the sample). The resulting intervals, metric scores, and index value have no straightforward statistical interpretation.

Furthermore, if scoring criteria are set based on the bisection method, 25% of the reference sites are guaranteed to receive scores of 1 or 3 for each metric, but they will not necessarily be the same sites for each metric. The potential consequence of this result is the creation of novel or “biologically infeasible” reference conditions because it may not be biologically possible to have high metric scores for many attributes within the same location, especially where the attributes are inversely related to one another. An index value is the sum of a dozen or so metrics whose correlation structure is complex and dependent on factors that vary as a function of region, “stream quality”, and stream size (Angermeier and Karr 1986, Steedman 1988). Several studies have examined some of the statistical attributes of the IBI (e.g. Angermeier and Karr 1986, Steedman 1988, Fore et al. 1994, Hughes et al. 1998) and the benthic IBI (e.g., Doberstein et al. 2000); however, the statistical implications of the metric scoring and index construction

protocols have not been rigorously examined, and further investigation of the statistical properties of the IBI are warranted.

#### d. Antidegradation Policy

EPA's antidegradation policy for biocriteria states that biocriteria shall not be relaxed in response to deterioration of conditions at reference sites (Gibson et al. 1996). This policy is simple in concept; however, application of the policy may prove to be less straightforward.

If reduction in index values at reference sites is taken as *prima facie* evidence of deterioration at those sites without consideration of other factors such as natural temporal variability, the tendency to define reference conditions in circular terms is reinforced. Failure to account for natural sources of within-site variability precludes the assessment process from protecting the natural processes that generated that variability, resulting in reduced resilience and biodiversity over the long term. While several studies (e.g., Fore et al. 1994) have examined temporal variability in IBI scores at individual sites, they provide no assessment of long term variability than can be expected in the absence of changes in human influence.

Drawing scientifically defensible distinctions between anthropogenic effects and background change will be difficult in the face of poorly understood emerging environmental threats, natural spatio-temporal variability, and region-wide environmental stresses that may or may not be responsible for observed changes at reference sites. This issue is relevant to both biocriteria in general as well as in § 316(b) assessments.

## Research Needs

### 1. Examination of Existing Methods

The most fundamental issue that needs to be addressed through additional research is the paradox of impaired reference sites. A key to addressing this paradox is improved understanding of the sources of variability in individual metrics and in the composite index values among reference sites. This should be done using two approaches that can be implemented independently of one another.

#### a. Reexamination of Reference Conditions

One approach is to examine the reference sites in more detail to determine if sites that were found to have index scores in the lower end of the reference site distribution (e.g., below the 25<sup>th</sup> percentile) are distinguishable from the other reference sites by their exposure to anthropogenic stressors. This examination should consider the magnitude, timing, and location of stressors previously examined, as well as additional stressors that were not previously considered in selecting reference sites. It is also possible that sites at the lower end of the reference site distribution do not belong in the group because they are dissimilar with respect to some stable ecological factor not affected by human influence. For example, the set of reference

sites might include a few sand bottom streams in a group of reference sites otherwise composed of streams with cobble and boulder substrate.

If reclassification of reference sites at the lower end of the distribution cannot be justified based on a reexamination of the selection criteria, then the sites should not be reclassified or it should be acknowledged when designating water bodies as impaired based on their index values that sites having similar scores are within the range of conditions exhibited by unimpaired sites. Furthermore, the scoring, or even the validity of the metrics that contributed to the relatively low index score should be reexamined.

If reclassification can be justified following reexamination of the reference site selection criteria, then metric selection and calibration should be repeated after excluding sites that don't meet the revised, more rigorous reference site selection criteria. Once this has been done, the unimpaired condition defined by biocriteria should encompass the entire range of index scores exhibited by reference sites. The results of this analysis should be compared with the original results so that the source(s) of variance and its implications may be better understood.

#### **b. Monte Carlo Investigation of the IBI**

Another approach to understanding the sources of variability among reference sites for individual metrics and the composite index is through Monte Carlo simulation of the biocriteria development and implementation process. This approach would allow examination of the implications of the *ad hoc* operational rules common in the approach and the propagation of uncertainty in each stage of the assessment process. The approach could be implemented using both actual data sets (using re-sampling techniques) and synthetic data sets with known statistical properties. It would also allow examination of the implications of important assumptions that the limited statistical studies have relied on when examining a multimetric approach (e.g., Fore et al. 1994). The Monte Carlo approach proposed here differs from resampling procedures employed by Fore et al. (1994) and Doberstein et al. (2000) in that it allows explicit examination of uncertainty associated with the entire index development and application process as opposed to a narrow focus on sampling error at individual sites.

#### **c. Responsiveness to Impingement and Entrainment**

An empirical relationship has been established between multimetric index value and several sources of anthropogenic stress; however, entrainment and impingement are not among the stressors that have been examined. Research is needed to document a relationship between target assemblages for § 316(b) assessments and impingement and entrainment.

## **2. Development of Improved Methods**

### **Multi-Scaled Assessment Methods**

The multimetric approach needs to take greater account of the multi-scaled nature of biological integrity. The implications of site-by-site data reduction when constructing and calculating



metrics and indices need to be examined. Methods need to be developed and tested for assessing biological integrity on multiple spatial scales. The approaches developed must explicitly include spatial heterogeneity as an important component of biological integrity. This type of research could be conducted through analysis of existing data, such as that collected by Ohio EPA, Maryland DNR, Tennessee Valley Authority, or the Ohio River Ecological Research Program.



# 4

## EVALUATION BY WATER BODY TYPE

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This chapter reviews the strengths, weaknesses, developmental status, and research needs for applying biocriteria for § 316(b) implementation on specific water body types. The water body classification used here reflects the classification developed by EPA for its biocriteria program. The classes are: streams and small rivers, large rivers, lakes and reservoirs, and estuaries and coastal marine waters. Where there are important distinctions among water body types within one of EPA's classes (e.g. lakes vs. reservoirs), the distinctions are discussed within the relevant section.

Water body-specific evaluations are presented for a number of reasons. While there are many general attributes of the biocriteria approach which were discussed in preceding chapters, there are also many attributes that arise from the specific application of biocriteria to a particular water body type. Significant differences derive from a water body type's flora and fauna, physical dimensions, degree of connection to other water bodies of various types, physical and chemical factors such as stratification and tides, and many other factors. The biocriteria approach needs to be adapted for each type of water body to reflect these differences.

Furthermore, the scope of EPA guidance has evolved as the approach has been modified for different types of water bodies. While diagnosis of the causes of water body impairment was not prominent in early implementation and guidance for wadeable streams and rivers, it has become increasingly so in more recent guidance (i.e. lakes and reservoirs, estuaries and coastal marine waters).

### Streams and Small Rivers

Few power plant CWISs within the United States are located on streams or small rivers.<sup>3</sup> Thus, at first glance it may seem that methods for streams and small rivers are of little direct relevance to § 316(b) implementation. Bioassessment in streams and small rivers is relevant, however, for several reasons.

First, the theoretical framework for multimetric indices was originally developed and tested in the context of wadeable streams. Techniques developed for streams have been adapted for application in multiple geographic regions, and EPA guidance for streams and small rivers is more highly developed than that for any other water body type. Virtually all operational state biocriteria programs are for streams and small rivers. Thus, streams have yielded most of the practical experience with the multimetric approach and they provide the only model for how

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<sup>3</sup> Small rivers, for the purposes of this discussion, are considered to be those that are wadeable during appropriate sampling periods.

biocriteria programs for larger aquatic systems might develop on a national basis. As a result, many of the methods for larger water body types are adaptations of methods developed in streams and small rivers.

Second, while the classification of water bodies into different types is useful for developing EPA guidance, individuals and populations of some species potentially impacted by entrainment or impingement have life histories or migratory behaviors which cause them to cross boundaries between streams and other water body types (e.g., Fraser et al. 1999) where power plants are located. Consequently, assessment of the biological integrity of streams may be relevant to § 316(b) in some circumstances, even though a CWIS is not sited directly on a stream water body.

Finally, the data base that is being accumulated to support biocriteria programs for streams provides important opportunities to examine the statistical properties of metrics and indices. For all these reasons, discussion of wadeable streams and rivers is relevant to the broader discussion of biocriteria and regulation of CWIS under § 316(b) of the CWA. Nonetheless, it is important to keep in mind that streams are a distinct water body type, and, therefore, some underlying fundamental theories and assumptions of biocriteria methodologies for streams may not apply in other systems.

### ***Status of State Implementation***

Nearly all states have some type of biocriteria program for streams and small rivers (Figure 4-1, Table 4-1). Many states use biocriteria for aquatic life use determinations and 31 states have incorporated biocriteria into their water quality standards program, however, only two states have numeric biocriteria for streams and small rivers (Davis et al. 1996). The state of Ohio has a particularly comprehensive biocriteria program for streams and rivers (Ohio EPA 1987a). Ohio has used numerical and narrative biological criteria in its Water Quality Standards (WQS) program since 1980 (Ohio EPA 1987b). Other well established programs for streams and small rivers exist in Maine and North Carolina. The Tennessee Valley Authority (TVA) also has an established biocriteria program for assessing streams within its jurisdiction.

Table 4-1 depicts the status of biocriteria development and implementation by states (including the District of Columbia and the Ohio River Valley Sanitation Commission). Figure 4-2 shows the geographic distribution of bioassessment programs across the nation. Forty-nine of the 52 entities have programs in place or under development that are based on benthic macroinvertebrates. Thirty-four states have programs based on fish, and a few states have programs using periphyton.

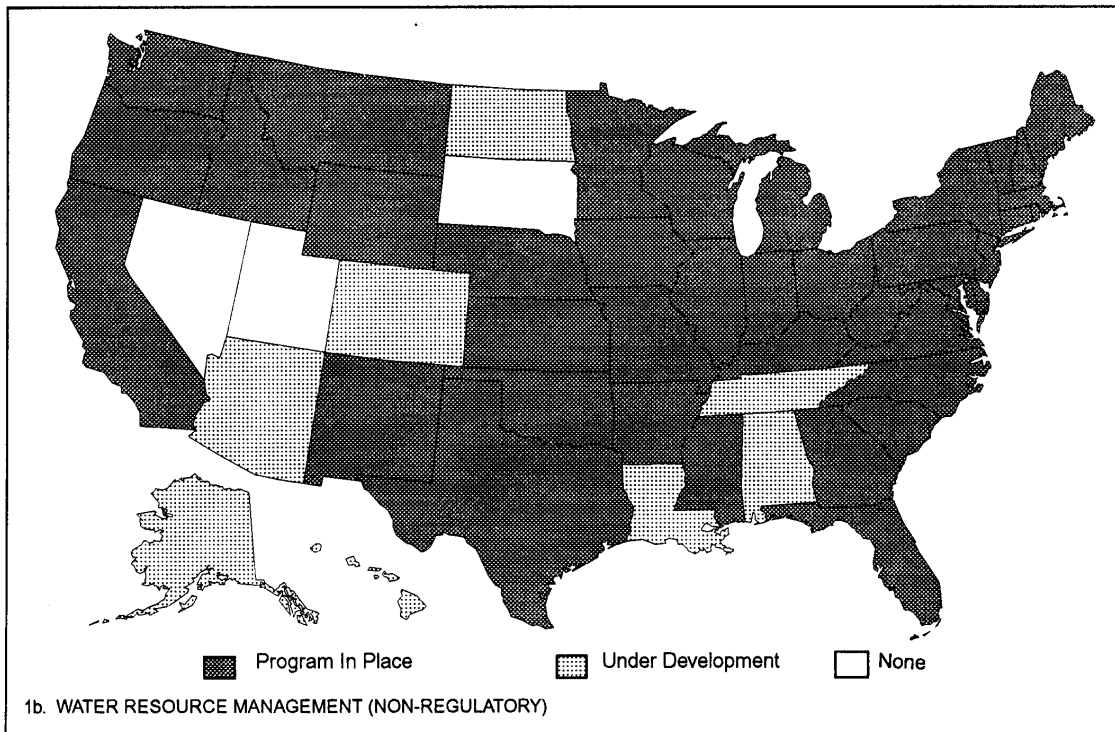
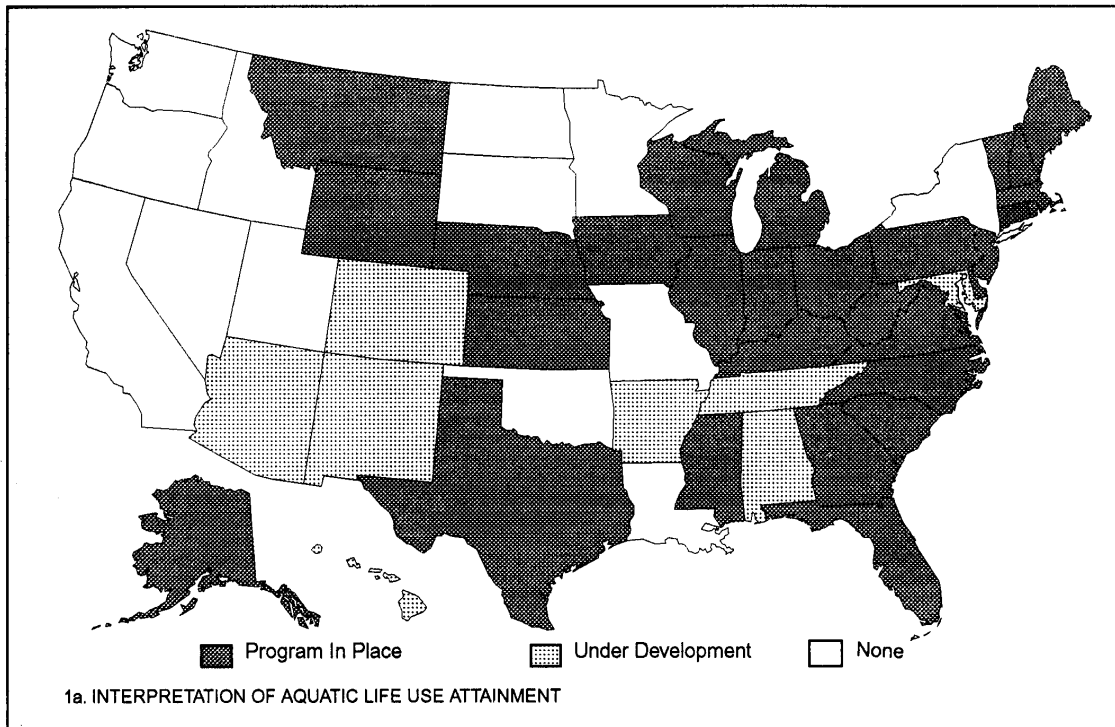


Figure 4-1  
State Biocriteria Programs—Applications. Source: Davis et al. (1996)

**Table 4-1**  
**National Summary of State Bioassessment Programs for Streams and Rivers in 1995**  
**(50 States, the District of Columbia, and the Ohio River Valley Sanitation Commission).**  
**Source: Davis et al. (1996)**

State Program (1995)	In-Place	Under Development	None
<b>Use of bioassessments</b>			
Water resource management (non-regulatory)	41	8	3
Interpret aquatic life use attainment	31	8	13
Narrative water quality standard	29	11	12
Numeric water quality standard	2	15	35
<b>Organism group used</b>			
Fish	29	5	18
Benthic macroinvertebrates	44	5	3
Algae (periphyton, diatoms)	4	3	45
More than one assemblage	26	10	16
<b>Reference conditions</b>			
Ecoregional	15	26	11
Site-specific	31	0	21
State-wide or basin-specific	6	0	46
<b>Multiple Metrics for data analysis</b>			
Biology	42	6	4
Habitat	33	6	13

### **Methods and Guidance**

The status of EPA guidance for biocriteria in streams and small rivers reflects the relatively advanced state of the science for this type of water body. The state of Ohio (Ohio EPA) published thorough descriptions of its biocriteria program in the late 1980's (Ohio EPA 1987b, Ohio EPA 1987c, Ohio EPA 1989). The EPA published its Rapid Bioassessment Protocols for use in streams and rivers about the same time (Plafkin et al. 1989). EPA's technical guidance for streams and small rivers was first published in 1994 (Gibson et al. 1994), and a revised version of that document was published in 1996 (Gibson et al. 1996).

Since the multimetric approach to biocriteria was first developed in streams and small rivers, the framework used for streams and small rivers represents the core of the approach. Consequently, there is a substantial degree of overlap between the descriptions of the biocriteria approach contained in Chapter 2 of this report (Primer on Biocriteria) and the guidance for streams and small rivers. This section focuses on some aspects of the guidance that are specific to, or of particular importance, in streams and small rivers.

### Geographic Classification

Current EPA guidance for streams and small rivers identifies ecoregions as the preferred geographic classification for establishing reference conditions (Gibson et al. 1996). In some circumstances, drainage basins may be biogeographically or evolutionarily significant units as is seen in streams of the Ozarks. In such circumstances, hydrologic units may be an appropriate adjunct or alternative to ecoregions (D. Diamond, USGS, pers. com.).

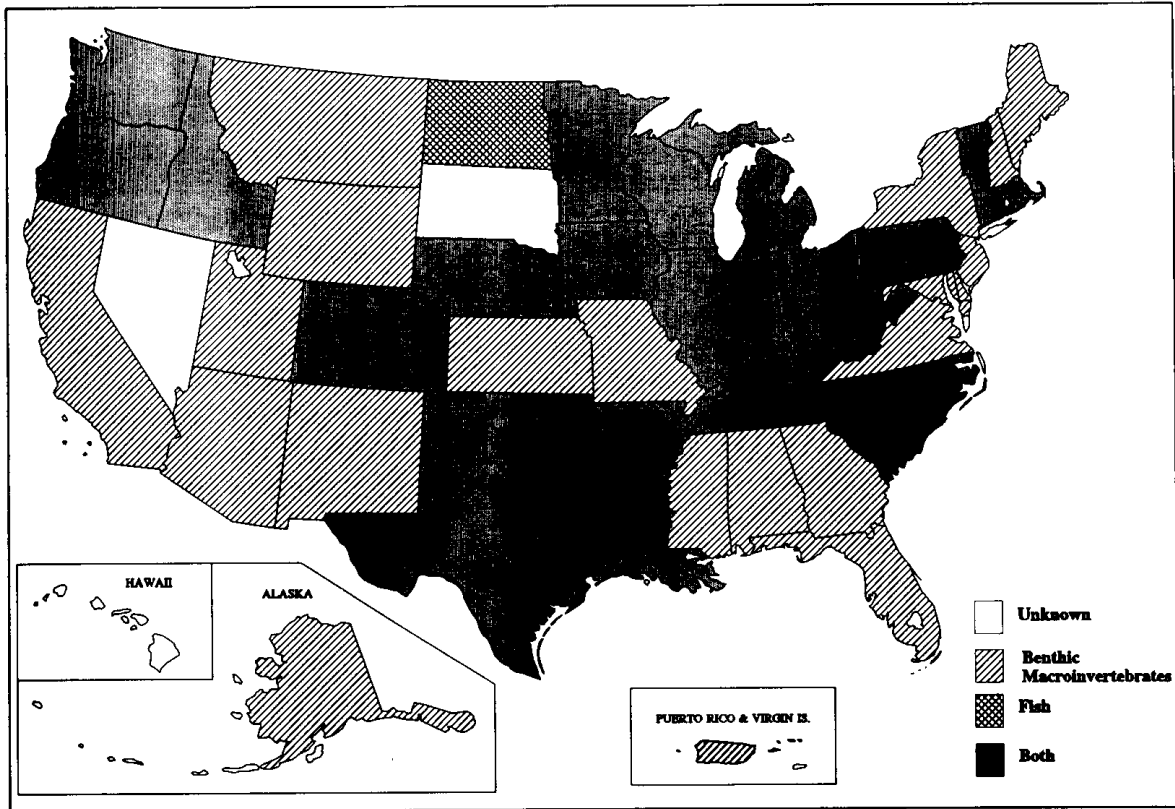


Figure 4-2  
Assemblages Used in State Biocriteria Programs. Source: Davis et al. (1996)

### **Recommended Assemblages**

Target assemblages recommended in EPA guidance for streams and small rivers (Gibson et al. 1996) include:

- Periphyton
- Macrophytes
- Benthic macroinvertebrates
- Fish
- Wildlife

Of these stream and small river assemblages, benthic macroinvertebrates and fish have the most relevance to § 316(b) implementation.

Benthic macroinvertebrates are the universal choice for inclusion in biocriteria programs, as evidenced by the fact that the three states that do not use benthic macroinvertebrates have no biocriteria programs at all. Given the biological expertise that exists in macroinvertebrate-based programs across the country for streams, and the tendency of most organizations to apply existing expertise to new problems, it is likely that benthic macroinvertebrates will be prominent in future state biocriteria programs developed for larger water bodies.

The dominance of benthic macroinvertebrate programs nationally reflects their utility as biological indicators of chemical water quality and quality of the substrate. Most species in this assemblage have relatively limited mobility and provide good site-specific information on a local scale. Unlike fish, benthic macroinvertebrate assemblages in streams are not likely to be affected by intakes in nearby, larger water bodies. This is because benthic macroinvertebrates are less mobile than fish, and the direction of water flow is from streams and small rivers to the larger water bodies.

Biocriteria programs for fish in streams and small rivers may be relevant to regulation of CWIS under § 316(b), because many species of fish are highly mobile and utilize multiple habitats. The importance of spatial scale was highlighted in the preceding chapter. Relevant spatial scales for fish species potentially affected by CWIS may span multiple types of water bodies, including streams and small rivers. Habitats in relatively large water bodies with CWIS, such as lakes, reservoirs, large rivers, or estuaries, are linked to tributary streams by movement of fish between these habitat types. Thus, degradation of spawning habitat in streams may affect biological integrity as assessed in adjacent water bodies with power plants. Conversely, in some circumstances effects of impingement and entrainment may be more readily assessed in tributary streams than in an adjacent water body where the intake structure is located. This can occur if species vulnerable to impingement and entrainment in the larger water body congregate at certain times of the year in streams and small rivers where they are more easily sampled.

Indices such as the fish IBI have been shown to respond to cumulative impacts, presence or absence of refuges, and localized stresses imposed in areas removed from the sample site (Karr 1987). Increasingly, it is being recognized that assessment of biological integrity in streams must



incorporate landscape-scale phenomena (Karr and Chu 1999). This broader perspective will provide for more comprehensive evaluations of biological integrity and support interpretation of local-scale information, including biological assessments in the vicinity of CWIS.

### ***Selection of Reference Conditions***

In addition to the general considerations in selecting reference sites (outlined in preceding chapters), EPA guidance (Gibson et al. 1996) identifies the following characteristics of ideal regional reference sites in streams and small rivers (adapted from Hughes et al. 1986):

- Extensive, natural riparian vegetation representative of the region;
- Representative diversity of substrate materials (fines, gravel, cobbles, boulders) appropriate to the region;
- Natural channel structures typical of the region (e.g., Pools, riffles, runs, backwaters, and glides);
- Natural hydrograph;
- Banks representative of streams in the region that are undisturbed by human influences (e.g., Covered by riparian vegetation with little evidence of bank erosion, or undercut banks stabilized by root wads);
- Natural color and odor—in some regions, clear cold water is typical; in others, the water is turbid or stained;
- Presence of animals, such as piscivorous birds, mammals, amphibians, and reptiles, that are representative of the region and derive some support from aquatic ecosystems.

Many studies have investigated the relationship between riparian zone vegetation and stream conditions (see Correll 1999); and it is generally recognized that extensive natural vegetation, especially in the riparian corridor, is important to the maintenance of biological integrity of streams. However, it is far less clear how extensive the vegetation must be.

The relationship between the areal extent and spatial arrangement of riparian vegetation and biological integrity of aquatic ecosystems is an area of active research. Preliminary results of research in the Clinch River watershed of southwestern Virginia indicates that a swath 200m wide and 1500m long upstream of an in-stream sample site must be examined to yield a good relationship between riparian land use and fish IBI score (Diamond 2000).

Other researchers, however, have found that whole watershed land use in the 1950s was the best predictor of present-day diversity (Harding et al. 1998). Present-day riparian land use and watershed land use in the current decade were found to be poor predictors of diversity in streams. These results suggest that historical land use should be an important consideration in selecting reference sites, and that land use in the entire catchment affects in-stream biota.

Most applications of biocriteria have considered effects of historical land use on fauna at the regional scale, but have tended to focus on current or recent land use when selecting reference sites and conducting analyses. This may account for some of the variability observed among

reference sites within an ecoregion (Harding et al. 1998). Investigation of the sources of variability among reference sites was identified as a critically important research need in Chapter 3. For streams and small rivers, historical land use may be a good place to start.

### ***Habitat Measurement***

Physical habitat strongly influences the structure and function of local instream biological communities. With appropriately designed metrics for habitat quality, habitat measurements can be used to adjust biological expectations at a site, select appropriate sampling sites for biological assessments, and interpret biological assessment results (Gibson et al. 1996). These applications are critical for achieving an accurate assessment of overall ecological integrity and for diagnosing the causes and nature of stressors related to observed biological impacts.

Natural variability in physical habitats dictates that sites will present intrinsically different opportunities to biota. As a result, biological assemblages are expected to vary from site to site, and it is unrealistic to expect every site to have the same potential to achieve a given level of biological condition. This is true at broad geographic scales (e.g., among different physiographic provinces, ecoregions, or drainage basins), as well as at local scales (e.g., among different stream reaches within the same stream). Therefore, designing useful habitat assessment metrics requires that the full range of natural variability be accounted for and distinguished from habitat conditions due to human-induced change.

Table 4-2 lists physical habitat characteristics that are frequently included in habitat assessments for streams and small rivers. These habitat metrics have been chosen because of their relevance to fish and benthic macroinvertebrates. To obtain a habitat quality score for a site, each metric is assigned a numeric score based on qualitative criteria (see Table 4-3) and scores for each metric are summed to yield an overall habitat quality score.

Because biological condition is at least partly dependent upon physical habitat, there is an expected relationship between biological and habitat metrics (Figure 4-3). For example, Figure 4-4 shows the relationship between Ohio's Qualitative Habitat Evaluation Index (QHEI) and fish IBI score for 465 relatively unimpacted and habitat modified stream sites in Ohio. While these data approximately follow the expected relationship between biology and habitat, the scatter in the relationship indicates that factors not captured by the habitat index also contribute to variation in IBI scores.

Physical habitat quality can be defined either by known habitat requirements of species within the biological assemblage of interest, or by habitat attributes that correlate with varying levels of human disturbance. In comparing biological condition and habitat quality metrics, it is important to avoid the circular reasoning that would result from habitat metrics designed on the basis of species habitat requirements. Although it is appropriate to choose habitat parameters that are of ecological importance to the biological assemblages of interest, the scoring of habitat metrics must be referenced to expectations for streams in a given physical setting that have not been adversely affected by human activity. For streams and small rivers, historical land use may be important in understanding the relationships among current land use, instream physical habitat, and biological condition. Therefore, as mentioned previously, historical land use and past human-induced disturbances must be considered in selecting reference conditions.

## Research Needs

1. Historical land use as a cause of reference site variability. Improved understanding of the sources of variability of index scores at reference sites was identified as a major research need in Chapter 3. Two approaches to understanding and reducing that variability were outlined in the preceding section on *Habitat Measurement*. Streams and small rivers provide the best model for conducting this research, because of the large existing database. Furthermore, detailed, historical land use information for stream catchments can be gathered to examine the effects of historical land use on current biological communities and current measures of biological integrity.
2. Partitioning natural versus anthropogenic sources of variability. Differences in physical habitat quality undoubtedly contribute to variability among reference sites in streams. Sources of variability in physical habitat at reference sites should be partitioned between natural and anthropogenic sources, and methods should be developed to remove the effects of measurable natural variability on measures of biological integrity. This is frequently done for the effects of stream size on total species richness (Figure 2-2); however, many studies do not account for other sources of variation that potentially affect the relationship, including anthropogenic factors (Smogor and Angermeier 1999). The feasibility of extending the approach to other aspects of physical habitat and other biological metrics needs to be examined. Such research will require detailed reexamination of habitat conditions in stream segments and development of indices of habitat quality that discriminate natural and anthropogenic variability in habitat quality. Here, also, consideration must be given to the effects of historical land use.
3. Effects of landscape-level patterns and phenomena on local fish assessments. Given the complex ways in which fish utilize and move among different habitats and water bodies, improved understanding is needed of how fish IBIs and other fish-based indices respond to conditions elsewhere in the drainage network. Cumulative effects within a drainage, blockages to fish movement, proximity to refugia from various stressors, and sources of impairment remote from the assessment site can all affect the results of a local stream assessment. Additional research is needed to investigate the effects of proximity to other water body types on multimetric indices in both streams and adjacent water body types. This research should also develop methods for assessing attributes of biological integrity that are manifested at broader spatial scales, as opposed to aggregating small scale assessments over space to characterize larger areas.
4. Statistical properties of fish IBIs. There have been relatively few investigations of the statistical properties of the IBI published in the peer-reviewed literature, and these have been limited in scope (e.g. Angermeier and Karr 1986, Karr et al. 1987, Steedman 1988, Fore et al. 1994, Doberstein et al. 2000). A notable exception is Hughes et al. (1998). As concluded in Chapter 3, additional studies of the statistical properties of multi-metric indices are warranted. Such studies should focus on additional sources of variation other than sampling error at individual sites.

**Table 4-2**  
**Habitat Measurement Variables. Source: Gibson et al. (1996)**

Category by Geographic Scale	Parameter
Watershed	Land use <sup>f</sup> Flow stability <sup>f</sup>
Riparian and bank structure	Upper bank stability <sup>a, f, h</sup> Bank vegetative stability <sup>a, f, h</sup> Woody riparian vegetation <sup>h</sup> - species identity - number of species Grazing or other disruptive pressures <sup>a, f</sup> Streamside cover (% vegetation) <sup>a, f</sup> Riparian vegetative zone width <sup>a, f</sup> Streambank erosion <sup>f</sup>
Channel morphology	Channel alteration <sup>a, d, f</sup> Bottom scouring <sup>a</sup> Deposition <sup>a</sup> Pool/riffle, run/bend ratio <sup>a, c</sup> Lower bank channel capacity <sup>a</sup> Channel sinuosity <sup>a, f, h</sup> Channel gradient <sup>f, h</sup> Bank form/bend morphology <sup>h</sup>
In-stream	Substrate composition/size; % rubble, gravel, submerged logs, undercut banks, or other stable habitat <sup>a, c, d, e, f</sup> % pools <sup>f</sup> Pool substrate characterization <sup>a</sup> Pool variability <sup>a</sup> % embeddedness of gravel, cobble, and boulder particles by fine sediment; sedimentation <sup>a, c, f</sup> Rate of sedimentation Flow rate <sup>a, d</sup> Velocity/depth <sup>a, d, e</sup> Canopy cover (shading) <sup>a, f</sup> Stream surface shading (vegetation, cliffs, mountains, undercut banks, logs) <sup>b, d, f</sup> Stream width <sup>c, h</sup> Water temperature <sup>c</sup>

**References:**

<sup>a</sup> Plafkin et al. 1989

<sup>b</sup> Platts et al. 1987

<sup>c</sup> Platts et al. 1983, Armour et al. 1983

<sup>d</sup> Rankin 1991

<sup>e</sup> Gorman, 1988

<sup>f</sup> Osborne et al. 1991

<sup>g</sup> Barton et al. 1985

<sup>h</sup> Hupp and Simon, 1986, 1991

**Table 4-3  
Habitat Assessment Field Data Sheet, Riffle/Run Prevalence. Source: Barbour and  
Stribling (1990)**

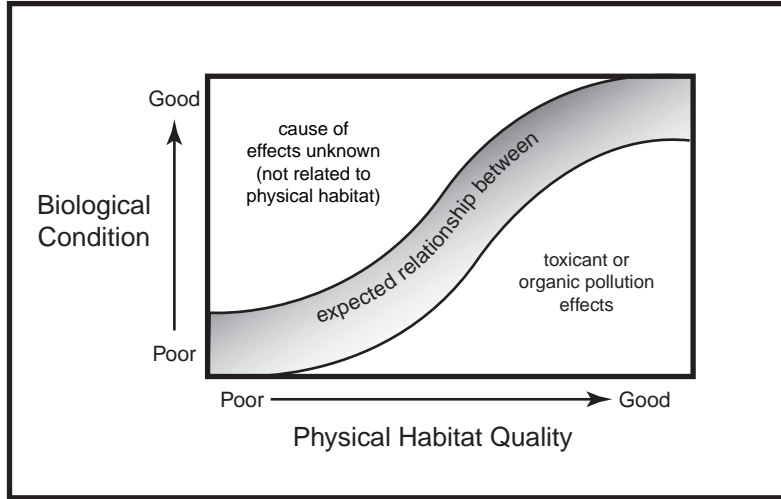
Habitat Parameter	Category			
	Optimal	Sub-Optimal	Marginal	Poor
1. Bottom substrate/ instream cover (a)	Greater than 50% mix of rubble, gravel, submerged logs, undercut banks, or other stable habitat. 16-20	30-50% mix of rubble, gravel, or other stable habitat. Adequate habitat. 11-15	10-30% mix of rubble, gravel, or other stable habitat. Habitat availability less than desirable. 6-10	Less than 10% rubble, gravel, or other stable habitat. Lack of habitat is obvious. 0-5
2. Embeddedness (b)	Gravel, cobble, and boulder particles are between 0-25% surrounded by fine sediment 16-20	Gravel, cobble, and boulder particles are between 25-50% surrounded by fine sediment. 11-15	Gravel, cobble, and boulder particles are between 50-75% surrounded by fine sediment. 6-10	Gravel, cobble, and boulder particles are over 75% surrounded by fine sediment. 0-5
3. $\leq 0.15$ cms (5cfs) → Flow at rep. low  OR  $> 0.15$ cms (5cfs) → velocity/depth	Cold $> 0.05$ cms (2 cfs) Warm $> 0.15$ cms (5 cfs) 16-20  Slow ( $< 0.3$ m/s), deep ( $> 0.5$ m); slow, shallow ( $< 0.5$ m); fast ( $> 0.3$ m/s), deep; fast, shallow habitats all present. 16-20	0.03-0.05 cms (1-2 cfs) 0.05-0.15 cms (2-5 cfs) 11-15  Only 3 of the 4 habitat categories present (missing riffles or runs receive lower score than missing pools). 11-15	0.01-0.03 cms (.5-1 cfs) 0.03-0.05 cms (1-2 cfs) 6-10  Only 2 of the 4 habitat categories present (missing riffles or runs receive lower score). 6-10	$< 0.01$ cms (.5 cfs) $< 0.03$ cms (1 cfs) 0-5  Dominated by 1 velocity/depth category (usually pools). 0-5
4. Canopy cover (shading) (c) (d) (g)	A mixture of conditions where some areas of water surface fully exposed to sunlight, and others receiving various degrees of filtered light. 16-20	Covered by sparse canopy; entire water surface receiving filtered light. 11-15	Completely covered by dense canopy; water surface completely shaded OR nearly full sunlight reaching water surface. Shading limited to $< 3$ hours per day. 6-10	Lack of canopy, full sunlight reaching water surface. 0-5
5. Channel alteration (a)	Little or no enlargement of islands or point bars and/or no channelization. 12-15	Some new increase in bar formation, mostly from coarse gravel; and/ or some channelization present. 8-11	Moderate deposition of new gravel, coarse sand on old and new bars; and/or embankments on both banks. 4-7	Heavy deposits of fine material, increased bar development; and/or extensive channelization. 0-3
6. Bottom scouring and deposition (a)	Less than 5% of the bottom affected by scouring and/or deposition. 12-15	5-30% affected. Scour at constrictions and where grades steepen. Some deposition in pools. 8-11	30-50% affected. Deposits and/or scour at obstructions, constrictions, and bends. Filling of pools prevalent. 4-7	More than 50% of the bottom changing frequently. Pools almost absent due to deposition. Only large rocks in riffle exposed. 0-3
7. Pool/riffle, run/bend ratio (a) (distance between riffles divided by stream width)	Ratio: 5-7. Variety of habitat. Repeat pattern of sequence relatively frequent. 12-15	7-15. Infrequent repeat pattern. Variety of macrohabitat less than optimal. 8-11	15-25. Occasional riffle or bend. Bottom contours provide some habitat. 4-7	$> 25$ . Essentially a straight stream. Generally all flat water or shallow riffle. Poor habitat. 0-3
8. Lower bank channel capacity (b)	Overbank (lower) flows rare. Lower bank W/D ratio $< 7$ . (Channel width divided by depth or height of lower bank.) 12-15	Overbank (lower) flows occasional. W/D ratio 8-15 8-11	Overbank (lower) flows common. W/D ratio 15-25. 4-7	Peak flows not contained or contained through channelization. W/D ratio $> 25$ . 0-3

Evaluation by Water Body Type

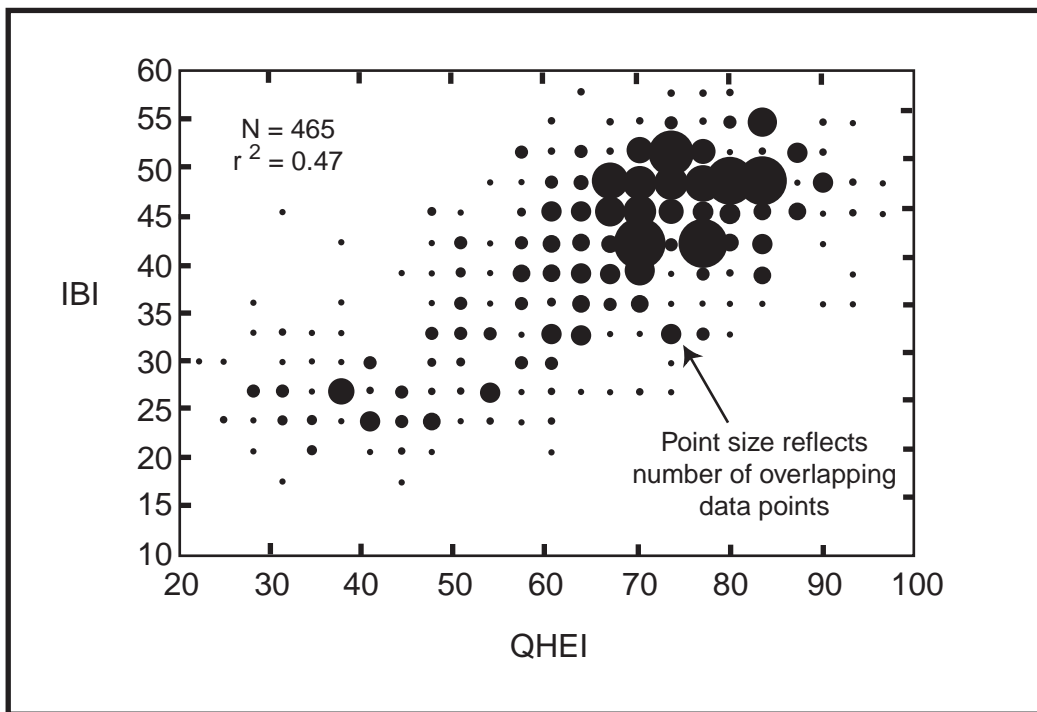
Habitat Parameter	Category			
	Optimal	Sub-Optimal	Marginal	Poor
9. Upper bank stability (a)	Upper bank stable. No evidence of erosion or bank failure. Side slopes generally <30°. Little potential for future problems.  9-10	Moderately stable. Infrequent, small areas of erosion mostly healed over. Side slopes up to 40° on one bank. Slight potential in extreme floods.  6-8	Moderately unstable. Moderate frequency and size of erosional areas. Side of slopes up to 60° on some banks. High erosion potential during extreme high flow.  3-5	Unstable. Many eroded areas. "Raw" areas frequent along straight sections and bends. Side slopes >60° common.  0-2
10. Bank vegetative protection (d) OR Grazing or other disruptive pressure (b)	Over 90% of the streambank surfaces covered by vegetation. 9-10  Vegetative disruption minimal or not evident. Almost all potential plant biomass at present stage of development remains.  9-10	70-89% of the streambank surfaces covered by vegetation. 6-8  Disruption evident but not affecting community vigor. Vegetative use is moderate, and at least on-half of the potential plant biomass remains.  6-8	50-79% of the streambank surfaces covered by vegetation. 3-5  Disruption obvious; some patches of bare soil or closely cropped vegetation present. Less than one-half of the potential plant biomass remains.  3-5	Less than 50% of the streambank surfaces covered by vegetation. 0-2  Disruption of streambank vegetation is very high. Vegetation has been removed to 2 inches or less in average stubble height.  0-2
11. Streamside cover (b)	Dominant vegetation is shrub.  9-10	Dominant vegetation is of tree form.  6-8	Dominant vegetation is grass or forbes.  3-5	Over 50% of the streambank has no vegetation and Dominant material is soil, rock, bridge materials, culverts, or mine tailings.  0-2
12. Riparian vegetative zone width (least buffered side) (e) (f) (g)	> 18 meters.  9-10	Between 12 and 18 meters.  6-8	Between 6 and 12 meters.  3-5	< 6 meters.  0-2
Column Totals				
	Score			

References:

- (a) From Ball 1982.
- (b) From Platts et al. 1983.
- (c) From EPA 1983.
- (d) From Hamilton and Bergersen 1984.
- (e) From Lafferty 1987.
- (f) From Schueler 1987.
- (g) From Bartholow 1989.



**Figure 4-3**  
**Relationship Between Biological Condition and Physical Habitat Quality. Adapted From Barbour and Stribling (1990)**



**Figure 4-4**  
**Qualitative Habitat Evaluation Index (QHEI) Versus the Index of Biotic Integrity (IBI) for 465 Relatively Unimpacted and Habitat Modified Ohio Stream Sites. Source: Gibson et al. (1996)**

## **Large Rivers**

Large rivers, generally defined as those rivers with drainage areas greater than 1,000 mi<sup>2</sup> (2,600 km<sup>2</sup>) (Ohio EPA 1989, Simon and Lyons 1995, Simon and Sanders 1999), are of direct relevance to § 316(b). Rivers (excluding estuaries) are the source of cooling water for approximately 47% of the installed steam-electric utility generating capacity in the U.S. (EEI 1996).

### ***Status of State Implementation***

At this time, Ohio is the only state to have adopted biocriteria in its Water Quality Standards (WQS) program for large rivers. For the purposes of its fish biocriteria program for inland waters, Ohio EPA classifies streams and rivers as headwaters, wadeable, or boatable. Boatable waters in Ohio are generally those with a drainage area greater than 600 mi<sup>2</sup>. This classification encompasses all flowing waters of the state except the Ohio River and portions of tributaries to Lake Erie that are under hydrologic influence of the lake (termed lacustuaries). Ohio has protocols and standards in place for each class of flowing water.

The Ohio River Valley Water Sanitation Commission (ORSANCO) began a concerted effort to develop numeric biological criteria in 1993; however, it will be another 3-5 years before these criteria are incorporated into state water quality standards (Erich Emery, ORSANCO, pers. comm.). Several other mid-western states (e.g., Wisconsin and Indiana) are in the process of developing biological indices for use in large rivers.

### ***Methods and Guidance***

Methods and guidance for developing biocriteria for large rivers are very limited. EPA does not have technical guidance for large rivers at this time. Publication of a guidance document is planned for 2002 (Swietlik 1999).

Simon and Lyons (1995) identified published fish IBIs for only a small number of large rivers, including:

- The Willamette River in northwestern Oregon (Hughes and Gammon 1987),
- The large rivers of Ohio (Ohio EPA 1987b, Ohio EPA 1987c),
- The Seine River in north-central France (Oberdorff and Hughes 1992), and
- The Current River in southeastern Missouri (Hoefs and Boyle 1992).

An IBI has also been developed for the Wabash River in Indiana based on more than 30 years of work by Gammon (1998).

The undeveloped state of biocriteria for large rivers in general is a reflection, at least in part, of the limited amount of more general scientific research that has been conducted on large rivers. Johnson et al. (1995) attribute the limited nature of research in large rivers to the challenges



posed by sampling in these systems and the absence of a clear, theoretical understanding of how large river ecosystems operate.

Interest in large rivers on the part of scientists has increased in recent years, and a substantial amount of information has become available over the last decade. Biocriteria development reflects this growing interest and accumulation of scientific information.

Ongoing developmental projects include:

- The Wabash River in Indiana (Simon and Stahl 1998; T. Simon, USFWS, pers. Comm.),
- The Ohio River (Emery et al. 1999, Simon and Sanders 1999),
- Large rivers in Wisconsin (J. Lyons, Wisconsin DNR, pers. Comm.), and
- The Red River of the North in Minnesota (Niemela et al. 1999).

The Ohio River has perhaps the most extensive data base relevant to biocriteria development of any Great River (Simon and Sanders 1999). ORSANCO has been collecting biological information on the river for more than half a century. The Ohio River Ecological Research Program, sponsored by electric power generation companies with plants on the river, was initiated in 1970 to gather information on the impacts of power plant operation on biota in the Ohio River (Lohner et al. 1999). Ohio EPA has also sampled extensively throughout the upper Ohio River. Ohio EPA's index developed for boatable inland streams has been used to assess impacts of entrainment and impingement near Ohio River power plants (Lohner et al. 1999).

Development of bioassessment methods and biocriteria for large rivers presents numerous challenges. These challenges consist of establishing reference conditions, developing and interpreting IBIs, sampling limitations and data quality, and coping with habitat variability. Issues related to each of these areas are discussed below.

### ***Reference Conditions***

Specification of reference conditions for large rivers raises problems similar to those in streams; however, many of these problems are exacerbated in the large river environment.

Identification of reference sites, from which reference conditions can be derived for large rivers, is especially problematic for several reasons. Work in streams and small rivers has shown the importance of developing reference conditions and biological metrics and indices based on ecoregions or some other ecologically meaningful geographic unit. The number of rivers in any geographic area is a decreasing function of river size. Thus, the number of potential reference sites for large rivers is reduced simply due to the larger size of the systems involved. In fact, large rivers typically traverse multiple ecoregions and form the border between others.

Reash (1999) identified two factors that further confound the establishment of appropriate reference conditions and development of biological criteria for large rivers:

- Significant, essentially irreversible anthropogenic alteration of large river ecosystems, and
- Sparse records of faunal composition prior to anthropogenic change.

Large rivers have been substantially altered by human activities on the landscape (e.g. urban, suburban, agricultural development), and human uses on and adjacent to the water (e.g. damming, water withdrawal, barge and commercial traffic, channelization). These changes, which are widespread and essentially irreversible, have further reduced the potential number of reference sites. Direct biological modifications are also superimposed on this widespread modification of the physical habitat of large rivers. These modifications include intentional and unintentional introduction of non-indigenous species and fishery management practices.

Areas adjacent to large rivers have always been preferred sites for human developments, and anthropogenic change often preceded thorough recording of the local fauna; thus, historical information is of limited value for deriving reference conditions. These factors suggest that specification of reference conditions for large rivers based on reference sites or historical information will be problematic.

Much of the recent research on large rivers (e.g. Gore and Shields 1995, Peterson and Kwak 1999) has focused on characterizing the physical *processes* required to restore and sustain the ecological integrity of those systems. Biocriteria, however, rely on characterization of the biological *status* of an aquatic ecosystem. Development of robust biocriteria for use in large rivers will require clear linkages between the important ecological processes in large rivers and the composition and structure of large-river biological communities. Given the practical constraints on restoration of important physical processes in large rivers, development of biocriteria that are also feasible will require even greater understanding of the relationships between biophysical processes and biological community structure. Reference conditions for large rivers may need to be based on criteria that derive from explicit societal goals rather than on biological conditions that would be expected in the absence of human disturbance, as is the case for reservoirs.

While all ecological systems are unique to some degree, the relative rarity of large, and especially “great” rivers on the landscape highlights their uniqueness. Just as the number of potential reference sites is smaller for larger systems, the number of places where a specific index can be applied will decrease with increasing river size. In many cases, biological indices for large rivers, and especially for great rivers, will be river-specific.

### ***IBI Development and Interpretation***

The IBI for streams relies on an array of broad assumptions supported by empirical research (Fausch et al. 1990; Table 4-4). Large rivers are sufficiently distinct from streams and small rivers that the assumptions underlying multi-metric indices developed in streams must be reexamined before they are assumed to apply in large rivers (Reash 1999). Reash (1995) suggested ways in which Ohio EPA’s IBI for boatable inland waters could be modified for application to the Ohio River (Table 4-5); however, the assumptions underlying these suggested modifications will also require validation.

Existing metrics for assessing the biological health of fish communities are not adequate for use in large river assessments. These metrics, which were originally developed for use in wadeable streams, are not relevant to rivers because of differences between fish communities of streams and rivers. Thus, the identification of tolerant/intolerant species for metrics application must be

tailored to rivers (Seegert 2000). When classifying species according to tolerance, it is important to evaluate the species in the context of the relevant biogeographic range. For example, a particular variety of northern fish may exhibit a different range of tolerances than its southern counterparts. Moreover, as is true for other water body types, metrics developed for use in a particular geographic region may not be applied to another area, without some recalibration to account for differences in assemblages.

**Table 4-4**  
**Underlying Assumptions of the IBI Concerning How Stream Fish Communities Change With Environmental Degradation. Source: Fausch et al. (1990)**

<ol style="list-style-type: none"><li>1. The number of all native species and of those in specific taxa or habitat guilds declines.</li><li>2. The number of intolerant species declines.</li><li>3. The proportion of individuals that are members of tolerant species increases.</li><li>4. The proportion of trophic specialists such as insectivores and top carnivores declines.</li><li>5. The proportion of trophic generalists, especially omnivores, increases.</li><li>6. Fish abundance generally declines.</li><li>7. The proportion of individuals in reproductive guilds requiring silt-free coarse spawning substrate declines, and the incidence of hybrids may increase<sup>1</sup>.</li><li>8. The incidence of externally evident disease, parasites, and morphological anomalies increases.</li><li>9. The proportion of individuals that are members of introduced species increases.</li></ol>
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<sup>1</sup> The incidence of hybrids was originally proposed by Karr (1981) to assess the loss of reproductive isolation due to degradation, but hybrids are difficult to identify and incidence may vary due to other factors. Recently, other investigators such as the Ohio EPA have followed Karr's (1981) suggestion of using metrics based on reproductive guilds to measure the loss of forms requiring clean spawning substrate.

**Table 4-5**  
**Recommended Modifications to Ohio EPA's Fish IBI for Application in the Ohio River.**  
**Source: Reash (1995)**

IBI Metric	Current Inland <sup>a</sup> Cutoffs	Suggested Modification
Number of species	>20, 10-20, <10	Cutoffs may need revision; sites with sparse habitat often yield <20 species.
Percent round-bodied suckers	>38, 19-38, <19	Acceptable for upper river, possibly acceptable for middle river. Cutoffs will need modification.
Sunfish species	>3, 2-3, <2	May require a lower expectation in middle river, where rock bass and pumpkinseed are rare.
Sucker species	>5, 3-5, <3	Scoring cutoffs will require modification; hog sucker, white sucker, and black redhorse are rare in middle river.
Intolerant species	>3, 2-3, <2	A new list of "Ohio River intolerant species" will be needed. Many intolerant species for inland metric are small stream forms.
Percent tolerant	<15, 15-27, >27	Same comment as above. Scoring cutoffs will need modification.
Percent omnivores	<16, 16-28, >28	Scoring cutoffs will need modification. A greater number of omnivore species would be expected in large, impounded rivers.
Percent insectivores	>54, 27-54, <27	This metric is questionable for the Ohio River. Fewer insectivorous species present due to lentic-like hydrology. Benthic production of food organisms much less than in free-flowing systems.
Percent top carnivores	>10, 5-10, <5	May require modification for upper and middle reaches.
Percent simple lithophils	Varies w/drainage area	Questionable for use in Ohio River due to limited area with hard substrate.
Percent DELT anomalies	<0.5, 0.5-3.0, >3.0	Scoring cutoffs will need modification based on Ohio River samples. A greater abundance of carp, catfish, and Ictiobinae suckers in impounded rivers may inflate the prevalence of DELT anomalies.
Fish numbers	<200, 200-450, >450	Scoring cutoffs will need modification. Dense clusters of forage species will cause wide variation in total numbers.

<sup>a</sup> Indicated cutoffs correspond to metric scores of 5, 3, and 1, respectively.

Development and validation of river-specific indices is also problematic. Because anthropogenic influences are linked by river hydrology, there is not an independent system in which the index can be tested. Without independent sites for index development and validation of the relationship between anthropogenic stress and biological condition, inferences of human-induced change are based on circular logic.

Relationships between anthropogenic stressors and biological condition inferred from river-specific metrics are subject to further challenge because anthropogenic gradients are generally confounded with numerous other chemical, physical, and biological gradients. In rivers with a series of dams, reaches between dams are sometimes viewed as independent units. Such river reaches may be useful study units and contribute insights to large river ecology. However, because of larger-scale gradients that exist along the river, differences between these study units must be interpreted with great care.

Virtually all large rivers contain dams. In many cases, the head of one impoundment abuts the tail of the next, such that typical riverine habitat is nonexistent. In cases where indices of biological integrity developed for free-flowing waters have been applied to impounded waters (Reash 1995) and to reservoir tailwaters (Scott 1999) they have failed to produce consistent results, suggesting that indices will need to be modified for use in these environments. In other places where riverine habitat exists, fragmentation and isolation by dams and impoundments can be expected to alter the functioning of those habitats. Thus, river segments may need to be classified by habitat type, in a manner similar to what is done within large reservoirs by the Tennessee Valley Authority (see following section on Lakes and Reservoirs).

Virtually all of the biocriteria development work that has been done on large rivers has been performed in the Midwest. Simon and Sanders (1999) provide a summary of conclusions from that research. Additional research needs to be conducted in other regions of the country. Of particular importance is improved understanding of the habitat requirements of riverine fishes in riverine habitats. Habitat requirements need to be assessed on a spatial scale commensurate with the scale at which the fish assemblage will be assessed.

Lohner et al. (1999) used an IBI to evaluate the condition of the fish assemblage in the vicinity of power plants on the Ohio River, and identified a number of issues that apply in a § 316(b) context. Fish IBIs, such as the one used on the Ohio River, typically include a metric based on the relative abundance of pollution tolerant species. High abundance of tolerant fish is considered an indication of degraded biological integrity, and the site is given a low score for that metric. Lohner et al. (1999) note that if such species are susceptible to impingement and entrainment, these processes could reduce the abundance of tolerant individuals, resulting in an increase in the score for that metric. In other words, biological integrity, as measured by that metric, would increase in response to population-level effects of impingement and entrainment. Thus, a single metric can respond in opposing ways depending on the type of stress. This phenomenon highlights the importance of documenting the relationships among individual biological metrics, the aggregate index, and specific stressors.

## **Sampling and Data Quality**

As noted above, development of IBIs for large rivers has lagged relative to that for other types of water bodies. This situation is due, in large part, to the difficulties of effectively sampling these water bodies. Seegert (2000) has identified several factors that pose obstacles for effective sampling of large rivers: river size, fish movement, variation in sampling effort/efficiency, irruptive species, and sampling variability.

The size of large rivers alone presents several impediments to their effective sampling. First, and most important, one cannot collect a representative sample of all species present. This is problematic because it conflicts with one of the central assumptions of the IBI: the entire fish fauna has been sampled in its true relative abundances without bias toward taxa or size of fish (Karr 1981, Karr et al. 1986). The morphology of the river, specifically its depth and width, permit the sampling of only a small percentage of the area. Moreover, each of the main habitat types has a distinctive fauna, and the habitats most conducive for sampling are not sufficiently representative (Seegert 2000).

Fish movement also may limit the thoroughness of sampling and quality of the resultant data because many riverine species routinely undertake long distance movement (e.g. spawning or overwintering migrations). Seegert (2000) recommends that the sampling index period be adjusted to avoid migratory periods to the greatest extent possible.

The sampling effort required for large rivers is also greater than that for most other water body types due to the longitudinal and horizontal migration of fish and the inaccessibility of species or individuals at any given time due to depth. The use of one type of sampling gear will often be inadequate to characterize the fauna of the river. Seegert (2000) reported that, even with repeated sampling, electrofishing or seining alone collected on average about 2/3 of the species actually present in large rivers with diverse fish communities. To address this incomplete sampling of the fish fauna using a single gear type, he proposed two approaches: 1) continue to sample with one gear and assume that the gear is equally effective at high and low quality sample sites, so that there is no bias with regard to taxa or size; or 2) use more than one gear to sample and combine the data through development of conversion factors that standardize the level of effort. If limited resources require use of a single gear for river sampling, electrofishing is the preferred gear (Ohio EPA 1987a, Seegert 2000). Future research could potentially document the sufficiency of sampling only a portion of the fish fauna and only some habitat types, as has been demonstrated for macroinvertebrates in small streams (Kerans et al. 1992).

Habitat quality is an important factor affecting sampling efficiency for large rivers. It is important to measure habitat quality (e.g. through use of qualitative habitat indices) so that differences in biological results may be appropriately attributed to either water quality or differences in physical habitat quality (Rankin 1989, Seegert 2000). Currently, however, no standard habitat assessment procedures have been developed for large rivers. The influence of habitat effects on the comparability may be minimized by sampling from zones that are of similar habitat quality or by sampling one basic habitat type (Kerans et al. 1992, Karr and Chu 1999).

Irruptive species, that is those species that regularly account for more than 25% of the catch, pose special problems in the context of river sampling. Because of their dominance in the catch,

which may exceed 50% of the catch on occasion, irruptive species may overwhelm such metrics as the Catch Per Unit Effort (CPUE) or other proportional metrics, and effectively skew the data (Seegert 2000). Where irruptive species have a substantial effect on the expected results, they may be removed from the affected metrics as was done by Jennings et al. (1995). However, exclusion of irruptive species is problematic when attempting to assess impacts of impingement and entrainment because eruptive species dominate impingement and entrainment in large rivers.

All of these outlined factors that affect sampling efficiency on large rivers contribute to variability in the IBI values for large river sites. Using data from several different rivers, Seegert (2000) explicitly considered the sample sizes and approaches needed to obtain reliable results. His research showed that changes of ten or more IBI units are common among sampling events in medium to large rivers. To reduce this variability, Seegert recommends the sampling of sites at least two to three times prior to assessment.

Another source of variability in the IBI values for large rivers derives from the use of different scoring criteria when developing the IBI for a particular area. Seegert found that investigators developing an IBI for the same area would commonly use substantially different cut-offs for metric scoring, resulting in considerably different IBI scores. The differences appear to derive from differing interpretations of the data by the researchers involved (Seegert 2000).

Given the above considerations, it is clear that multimetric methods will require substantial modification and development if they are to be effectively applied in large rivers.

### ***Research Needs***

1. Biophysical processes and community structure. Biological communities in large river ecosystems are strongly influenced by a number of biophysical processes. Modification of these processes through damming, channelization, and hydrologic changes has altered virtually all large rivers. Since full restoration of the natural processes will not be feasible in many cases, biocriteria must be based on a complex interplay of societal goals for each river and the feasibility of reestablishing the physical conditions supportive of those goals. This requires a thorough understanding of the cause-effect relationships between biological communities and the biophysical processes that sustain those biological communities. Before appropriate biocriteria can be specified, research is needed to acquire the requisite understanding of the relevant cause-effect relationships.
2. Multiple sources of variability. Research is needed to separate the effects of natural habitat variability, anthropogenic habitat variability, and sampling variability on measures of biological integrity. This will pose major challenges in large rivers because of the ubiquity of anthropogenic effects and the difficulties of sampling effectively in rivers.
3. Application of metrics to large rivers. The assumptions implicit in biological metrics proposed for large rivers must be documented. The validity of those assumptions should be examined through literature review and specific research should be identified for resolving major uncertainties surrounding the application of metrics to large rivers. This evaluation should explicitly examine the possibility that some metrics may not be consistent indicators

of anthropogenic influence when impingement and entrainment are among the stressors of interest.

4. Evaluation of scale considerations. Research is needed to examine the influence of assessment scale on results. Rivers exhibit patchiness and physical gradients on a wide range of spatial scales. Some of this variability is natural and some is of human origin. Organisms respond to this variability on a species-specific basis. Research should explicitly examine the interaction of natural spatial variability, human induced spatial variability, the spatial characteristics of potential CWIS impacts, and the scale of assessment.

## **Lakes and Reservoirs**

Many cooling water intake structures are located on lakes and reservoirs, some of which were constructed for the express purpose of providing cooling water for power plants. Steam electric power plants sited on lakes and reservoirs provide 27% of the nation's steam electric utility generating capacity (EEI 1996).

### ***Status of State Implementation***

Several states have some form of biological monitoring and assessment program in place for lakes and reservoirs. Biocriteria are in use or under development in Maine, Vermont, Wisconsin, Florida, and Minnesota, and are currently being used by the Tennessee Valley Authority (TVA). These programs involve the use of one or more of the following assemblages: benthic macroinvertebrates, phytoplankton, zooplankton, and macrophytes. At present, only TVA uses fish assemblages.

### ***Methods and Guidance***

EPA published technical guidance for lakes and reservoirs in August of 1998 (USEPA 1998a). That guidance document draws upon the extensive body of scientific literature that exists for these water bodies. While lakes are connected to the landscapes that surround them (Hasler 1975), they are also relatively closed systems when compared to lotic and marine systems. Their relatively closed nature makes lakes a useful unit of study (Forbes 1887). Research on lakes in this country dates back more than one hundred years (e.g. Birge 1897, Forel 1874). Within this body of research is a long history of studies of the relationship between environmental conditions and biological communities, beginning with E.A. Birge and his colleagues in Wisconsin (e.g. Birge and Juday 1911). Consequently, developers of biocriteria for lakes have a rich body of scientific research to draw upon. The surface water component of EPA's Environmental Monitoring and Assessment Program (EMAP-SW) is also conducting relevant research (e.g. Stemberger and Lazorchak 1994, Stemberger et al. 1996, Allen et al. 1999).

Scientific study of reservoirs is relatively recent by comparison to lakes, but many of the techniques developed in lakes are also applicable to reservoirs. Furthermore, because they are intensively managed for human uses, reservoirs receive significant scientific attention that supports, either directly or indirectly, development of biocriteria for these systems. Lakes and reservoirs are treated together here, because they are treated together in EPA guidance (i.e.,



USEPA 1998a). The EPA guidance, however, acknowledges that there are significant differences in the limnology of lakes and reservoirs. This section will first focus on important considerations for lakes and subsequently discuss the implications of these considerations for biocriteria in reservoirs.

Size is an important characteristic of lakes that can affect the type of, and potential for, adverse environmental impact. Lake size can also affect biocriteria development. Very large lakes (i.e., the Laurentian Great Lakes) and their coastal zones in some ways resemble marine systems more closely than they resemble their smaller freshwater counterparts. Biocriteria development for lakes will need to account for differences in character and scale associated with very large lakes. Because of their very large size, the Great Lakes are qualitatively different from other inland lakes and merit specific guidance for biocriteria development. EPA's guidance for lakes and reservoirs does not address the Great Lakes, and EPA has not given any indication that separate guidance for biocriteria development in the Great Lakes is forthcoming.

Great Lakes ecosystems have experienced continual anthropogenic change over the last 200 years (Ryder 1990). Ryder and Edwards (1985) characterized this change as a downward trajectory of relative health. Most recently, such change continues in response to introduction of nonindigenous species such as the zebra mussel (*Dreissena polymorpha*), predatory Cladoceran, *Bythotrephes cederstroemii*, round goby (*Neogobius melanostomus*), and ruffe (*Gymnocephalus cernuus*). These and other species have substantially, and irreversibly, altered the structure and function of the Great Lakes. A significant challenge in the Great Lakes has been to define biological integrity for these systems (see Ryder and Edwards 1985, Ryder 1990, Hodson 1990, Koonce 1990, Willford 1990, Evans et al. 1990). Perhaps because of the challenges they pose, the Great Lakes provide very few examples of biocriteria development (e.g. Thoma 1999, Minns et al. 1994). This review focuses on the application of the existing EPA guidance to lakes other than the Great Lakes.

While there is a substantial body of scientific information available for lakes, relatively little has been done to develop numeric biocriteria of the kind discussed for streams and rivers. The broad framework of EPA's technical guidance for lakes and reservoirs is quite different from the existing guidance for other water body types (i.e. streams and small rivers). The lakes and reservoirs guidance takes a multi-faceted, multi-tiered approach, rather than emphasizing multi-metric indices of a few select assemblages (Figure 4-5).

EPA's guidance describes two tiers, with the tiers representing different levels of sampling effort. Tier 2 assessments incorporate the information requirements of Tier 1. Sampling effort is also classified by the number of visits made to the site during the index period. Type A sampling involves a single visit during the index period, while Type B sampling requires multiple visits during a single index period.

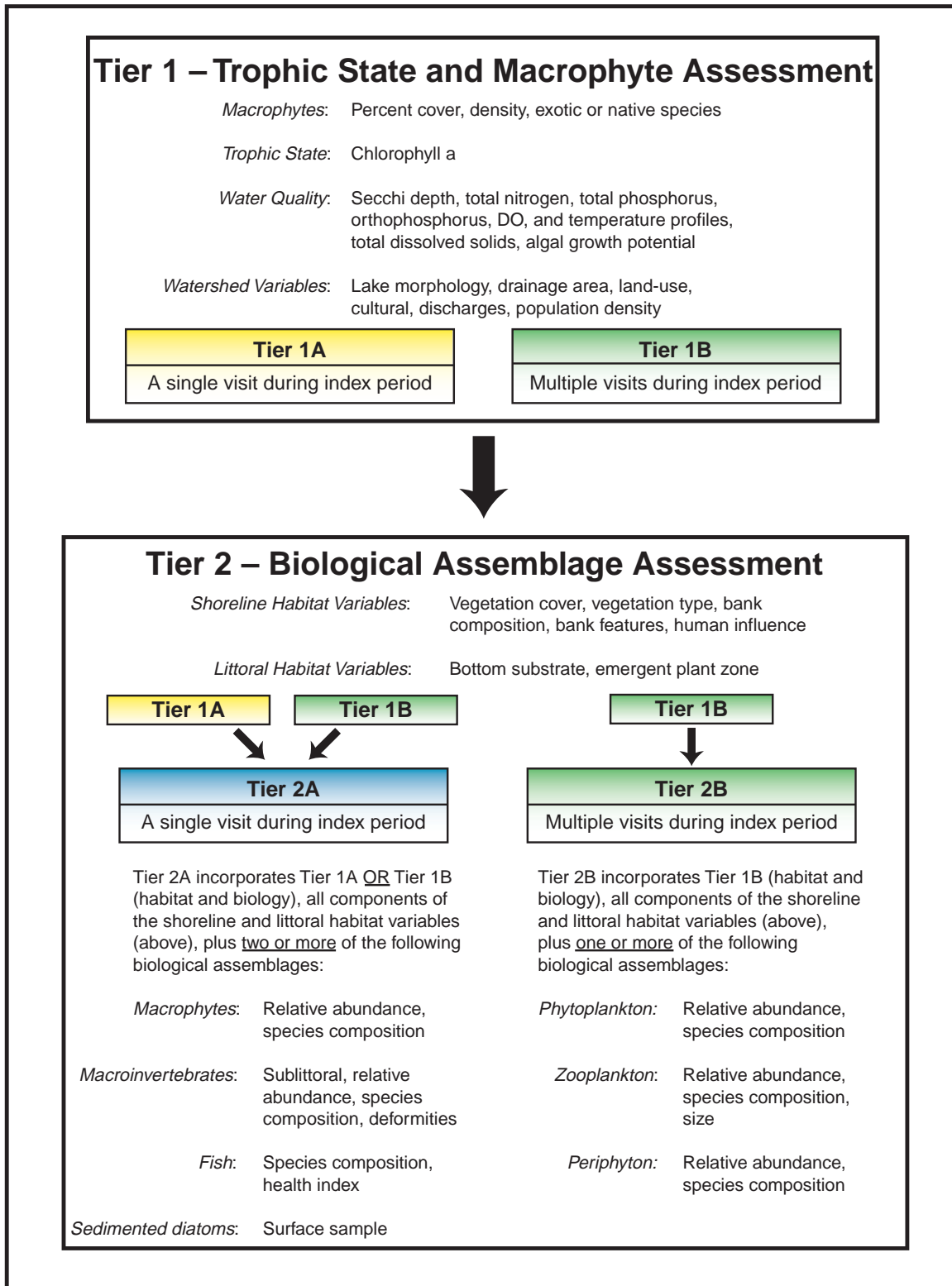
The guidance for lakes and reservoirs also includes two types of study, one being a field study and the other being a desktop study. The desktop study, which precedes the field effort, documents existing information useful in planning field studies and identifies issues or areas of concern to be addressed by subsequent sampling. The components of the desktop study, along with the sources and uses of the information are listed in Table 4-6. One source of information used in the desktop study is a questionnaire, an example of which is attached in Appendix A. Impingement and entrainment are not among the choices of possible limiting factors listed in the

example questionnaire. However, the questionnaire is where potential adverse effects of impingement and entrainment would be identified as an issue to be addressed by an assessment.

Tier 1 addresses habitat components and primary production (Table 4-7). Tier 1A (one site visit) cannot be used to assess single lakes and is only appropriate for regional assessments (USEPA 1998a). Thus, Tier 1A assessments are clearly not appropriate in the context of § 316(b). The utility of Tier 1B (multiple site visits) is also questionable, because the assemblages assessed are not directly affected by entrainment and impingement.

Tier 2 incorporates the information requirements of Tier 1, plus more detailed analysis of at least two biological assemblages (Tables 4-8 and 4-9). Recommended assemblages within Tier 2 include: macrophytes, macrobenthos, fish, sediment diatoms, phytoplankton, zooplankton, and periphyton. Note that the set of recommended assemblages depends on whether the site will be visited once each year (Type A assessment) or multiple times each year (Type B assessment).

As in other types of water bodies, habitat measurement is an important component of the bioassessment process. Habitat must be taken into account to make accurate comparisons with reference conditions and to interpret biological data. Habitat measurement comprises both watershed and in-lake observations and serves two purposes: 1) it provides the information needed to place lakes in an appropriate category for determination of reference conditions, and 2) it helps detect and identify anthropogenic disturbances potentially affecting the biota. Tables 4-10 and 4-11 list recommended habitat measurements and metrics. Note that nutrient concentration and certain other components of water quality are considered a component of habitat.



**Figure 4-5**  
**Tiered Sampling Structure for Lakes and Reservoirs. Source: USEPA (1998)**

**Table 4-6  
Desktop Screening Assessment. Source: USEPA (1998)**

	<b>Component</b>	<b>Data Collection</b>	<b>Responds To or Indicator Of</b>
<b>Watershed Components</b>	1. Watershed land use, NPDES	Maps, existing database, questionnaire. GIS databases, e.g., EPA Reach File; EPA BASINS; Census Bureau TIGER; USGS Land Use, Land Cover.	Identification of potential point and nonpoint source eutrophication, toxicity problems.
<b>Biological Components</b>	1. Algal production - Bloom history	Questionnaire	Identification of perceived problems (eutrophication).
	2. Plant assemblage - Macrophyte cover - Extent (% available habitat) - Density (% cover) - Known weed problems	Questionnaire	Identification of perceived problems (weeds, exotic plants, loss of native plants).
	3. Fish assemblage - Fishery problems	Questionnaire	Identification of perceived problems (species imbalance, exotic species, overfishing, overstocking, disease).

**Table 4-7**  
**Tier 1: Trophic State and Macrophyte Sampling. Source: USEPA (1998)**

	Component	Data Collection		Responds To or Indicator Of
		Tier 1A	Tier 2B	
<b>Habitat Components</b>	1. Watershed land use, population, NPDES	Maps, existing database, questionnaire. GIS databases, e.g., EPA Reach File; EPA BASINS; Census Bureau TIGER; USGS Land Use, Land Cover. Desktop screening habitat.		Potential causes
	2. In-lake physical habitat maximum depth area inflow	Maps or survey (single visit)		Potential causes
	3. Water Quality - DO, temperature profile - pH, alkalinity, conductivity - Secchi depth - Total dissolved solids - Nutrient concentration - Algal growth potential	Single index period  Surface or integrated	Multiple visits  Water column sample	DO problems, eutrophication, stratification, acidification, turbidity.
<b>Biological Components</b>	4. Algal chlorophyll a concentration.	Single visit chlorophyll sample from 0.5 m. Surface integrated water sample.	Multiple visits.	Eutrophication.
	5. Submerged macrophytes - % of available habitat with macrophytes - dominant species	Single visit, aerial photos if possible; otherwise, estimate from shorezone survey. Identify dominant species.	Multiple visits.	Eutrophication, herbicides, exotics.

**Table 4-8**  
**Tier 2A: Routine Biological Sampling. Source: USEPA (1998)**

	Component	Data Collection	Responds To or Indicator Of
<b>Habitat Components</b>	1. Watershed land use, population, NPDES	Desktop screening habitat	
	2. Lake physical	Tier 1 habitat	
	3. Shorezone habitat assessment	3-10 transects: - land use - bank stability - riparian vegetation - emergent vegetation	
	4. Water quality DO seasonal or annual mean, % depth-time; mean pH, alkalinity, Secchi depth.	Tier 1 water quality (1A or 1B)	Trophic state, turbidity
<b>Biological Components</b>	5. Algal chlorophyll a	Tier 1 chlorophyll (1A or 1B)	Trophic state
	6, 7. Assemblages (minimum 2)		
	a. Macrophyte species	2-3 samples from transects; identify plants to species and weigh cumulative sample of each species, or count stems.	Trophic state, exotics, herbicides
	b. Macrobenthos	Sublittoral surface sediment grab at end of each transect; identify to lowest practical level, 100-200 organisms.	DO, siltation, toxicity, productivity
	c. Fish assemblages	Littoral electrofishing sample at the end of each transect; sublittoral netting; identify to species, enumerate, weigh, and record incidence of external anomalies.	DO, toxicity, productivity
	d. Sediment diatoms	Surface sediment grab in deepest part of lake; identify to species and variety.	Nutrient enrichment, toxicity

**Table 4-9**  
**Tier 2B: Water Column Biological Sampling. Source: USEPA (1998)**

	Component	Data Collection	Responds To or Indicator Of
<b>Habitat Components</b>	1. Watershed land use, population, NPDES	Tier 0 habitat	
	2. Lake physical	Tier 1 habitat	
	3. Shorezone habitat assessment	3-10 transects: - land use - bank stability - riparian vegetation - emergent vegetation	
	4. Water quality DO seasonal or annual mean, % depth-time; mean pH, alkalinity, Secchi depth.	Tier 1B water quality (seasonal average)	Trophic state, turbidity
<b>Biological Components</b>	5. Algal chlorophyll a	Tier 1B chlorophyll (seasonal average)	Trophic state
	6, 7. Assemblages (minimum 2)		
	a. Phytoplankton	Surface samples (0.5 m) or integrated samples (hose) identify to genus; count 100-500 cells	Trophic state, acidity, metals, water column toxicity
	b. Zooplankton	Vertical tows; identify to genus; count 100-200 organisms, measure cladocerans	Trophic state, contamination, trophic imbalance
	c. Periphyton		

**Table 4-10**  
**Watershed and Basin Habitat Measurements and Metrics. Source: USEPA (1998)**

	Measurements	Additional Metrics	Calculation	Indicator
<b>Lake and Basin Morphology</b>	Watershed drainage area		Estimated from map contours	Hydrology
	Lake surface area		Map	
		Watershed : Lake area ratio	Watershed area/lake area	Sediment, nutrients
	Shoreline length	Shoreline development ratio		Effect of riparian zone
	Lake volume		Estimated from basin contours	
	Maximum depth		Measurement	Stratification potential
		Mean depth	Volume/surface area	
		Mean basin slope		
		Lake outflow	Retention time	Volume/outflow
<b>Land Use</b>	% forest or natural vegetation			Sediment, nutrients, hydrology
	% agriculture		GIS data base	Sediment, nutrients, contaminants
	% urban and residential			Sediment, nutrients, contaminants, hydrology
		Watershed impervious surface	Estimate from land use	Sediment, contaminants, hydrology
	Population density		U.S. Census, state or county	Sediment, nutrients, contaminants, hydrology
	Discharges		USEPA NPDES data base	Nutrients, contaminants
	Road density		Maps, GIS	Sediment, contaminants, hydrology



**Table 4-11**  
**Physical and Chemical Measurements and Metrics. Source: USEPA (1998)**

Measurements	Metrics	Calculation	Indicator
<b>T Profile</b>	Epilimnion temperature	Mean from temperature profile	
	Hypolimnion temperature	Mean from temperature profile	
	Metalimnion depth	Inflection point of temperature profile	
<b>DO Profile</b>	Epilimnion DO	Mean from DO profile	
	Hypolimnion DO	Mean from DO profile	
	Oxycline depth	Depth at which DO falls below 2 mg/L	DO problems
	Hypoxic volume	Volume of water with DO < 2 mg/L; annual or seasonal mean	DO problems
<b>Secchi Depth (SD)</b>	$TSI (SD) = 60 - 14.41 \ln (SD)$		Transparency
<b>Total N</b>	$TSI (N) = 54.45 + 14.43 \ln (TN)$		N enrichment
<b>Total P</b>	$TSI (P) = 4.15 + 14.42 \ln (TP)$		P enrichment
	N:P ratio	N concentration/P concentration (molar)	Enrichment
<b>Silica</b>			Depletion
<b>Acid Neutralizing Capacity (ANC)</b>	ANC		Sensitivity to acidification
<b>PH</b>	PH		Acidity
<b>Total Dissolved Solids (TDS)</b>	TDS		Dissolved minerals

The guidance identifies a hierarchy of five types of variables that can be used to classify lakes for the purpose of establishing reference conditions (Table 4-12), and advocates ecoregions as a classification scheme that incorporates many of the classification variables in lower levels of the hierarchy. Additional support for use of ecoregions comes from the National Research Council's Committee on Aquatic Ecosystem Restoration, which concluded that "goals for restoration of lakes need to be based on the concept of expected conditions for individual ecoregions" (NRC 1992). The guidance cites several studies that have shown how ecoregions can account for variability of water quality and biota in several areas of the United States (i.e., Barbour et al. 1996a, Barbour et al. 1996b, Heiskary et al. 1987, Hughes et al. 1994, Ohio EPA 1987a).

Most of these studies, however, considered flowing waters rather than lakes. Streams and rivers are hydrologically linked across the landscape, providing a greater degree of biological linkage within ecoregions than exists for lakes. An evaluation of the correspondence between ecoregions and fish assemblages (Heiskary et al. 1987) focused on fish species of interest to anglers and fisheries managers rather than nongame fish species. Nongame species are generally considered to be equally, if not more, important for assessing anthropogenic impacts. Also, fish managers' perceptions of what fish distributions would be in the absence of management are influenced by the same considerations used to define ecoregions (e.g., land form, vegetation, climate). Consequently, correlation of fish managers' expectations of species assemblages with ecoregions is tautological (i.e., the correlation would exist whether a relationship between ecoregions and fish distributions exists or not).

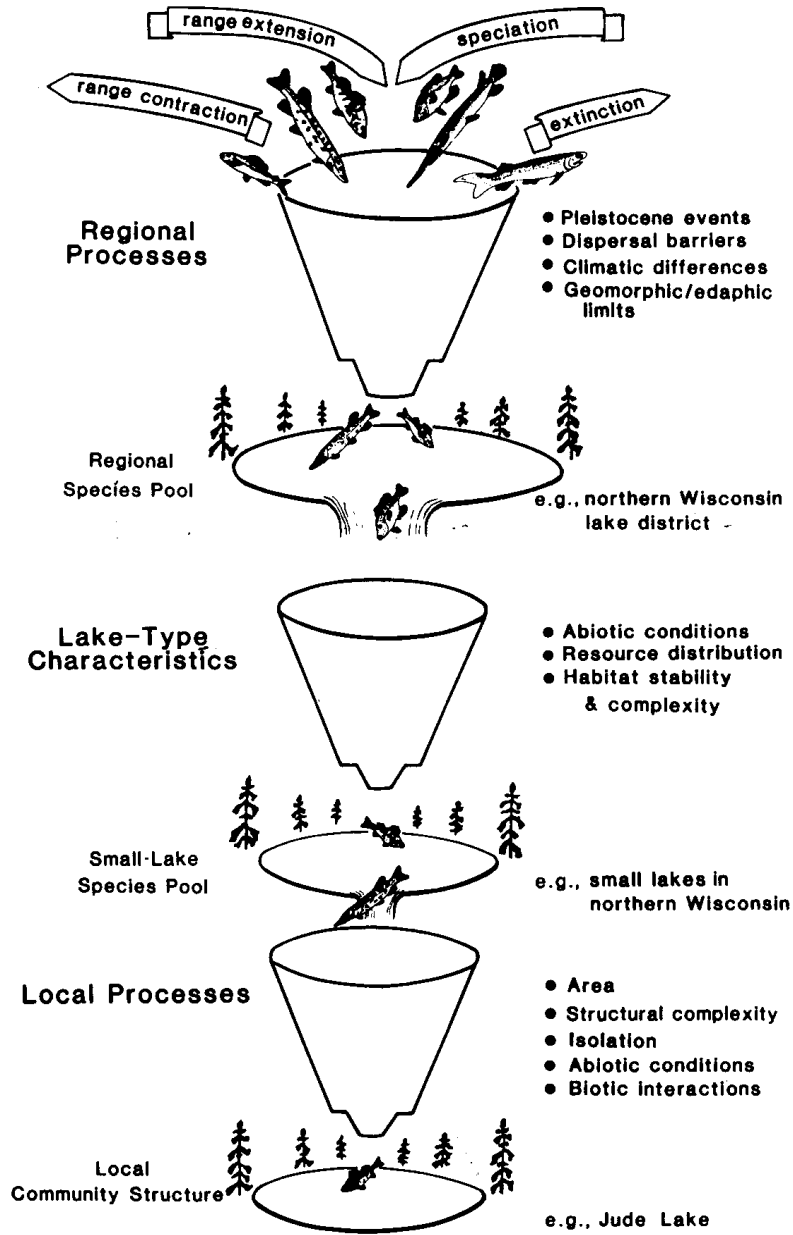
### **Reference Conditions**

Determination of reference conditions is the most important issue in developing multimetric biocriteria in lakes. Community structure at the scale of an individual lake is affected by factors and processes that are manifested at multiple temporal and spatial scales. These factors and processes include regional processes, lake-type characteristics, and local processes, which function like a set of sieves or filters that progressively constrain the potential community structure (Figure 4-6; Magnuson et al. 1994, Tonn 1999, Tonn et al. 1990). In many cases, local factors may dominate to such a degree that classification at the ecoregion scale may not adequately control for natural variation.

While lakes may be relatively isolated as compared to streams and rivers—and differences in isolation can be important in explaining fish community structure (Tonn and Magnuson 1982, Tonn et al. 1990)—fish assemblages in lakes are not stable. In an examination of eleven years of fish assemblage data from seven lakes, Magnuson et al. (1994) found that cumulative species richness within lakes was 1.5 times the mean annual richness. They noted that species turnover was overestimated because of sampling error (i.e., not all species present in a given year were collected). Given the same number of survey years, however, cumulative richness increased with the number of years between observations. They concluded that extinctions and invasions probably occurred within just eleven years, but uncertainty remains because of sampling variability. They warn that the combined effects of extinction, invasion, and sampling variability associated with rare taxa need to be considered when interpreting fish assemblage data. This is particularly important given the role of species composition and species richness in multimetric indices. It also points to difficulties in developing expectations for assemblages based on historical information spanning many years. This concern is also relevant to reservoirs, where cumulative species lists compiled over time are sometimes used to establish reference conditions (e.g., Hickman and McDonough 1996).

**Table 4-12**  
**Hierarchy of Five Types of Variables for Classifying Lakes. Source: USEPA (1998)**

Classification Level	Classification Variables
Geographic Region	Ecoregion (e.g., geology, soils, geomorphology, dominant land uses, natural vegetation) Physiographic province
Watershed Characteristics	Lake drainage type (e.g., flowage, drainage, seepage, reservoir type) Land use Watershed-to-lake area ratio (especially for reservoirs) Slope (especially for reservoirs) Soils and geology (erosiveness of soils)
Lake Basin Characteristics	Depth (mean, maximum) Surface area Bottom type and sediments Shoreline development ratio (shoreline length: circumference of equal area circle) Age (of reservoirs) Epilimnetic / hypolimnetic discharge (reservoirs)
Lake Hydrology	Retention time Stratification and mixing Circulation Water level fluctuation and drawdown
Characteristic Water Quality	Alkalinity Salinity Conductivity Turbidity (Secchi depth, clarity, etc.) Color Dissolved organic carbon (DOC) Dissolved inorganic carbon (DIC)



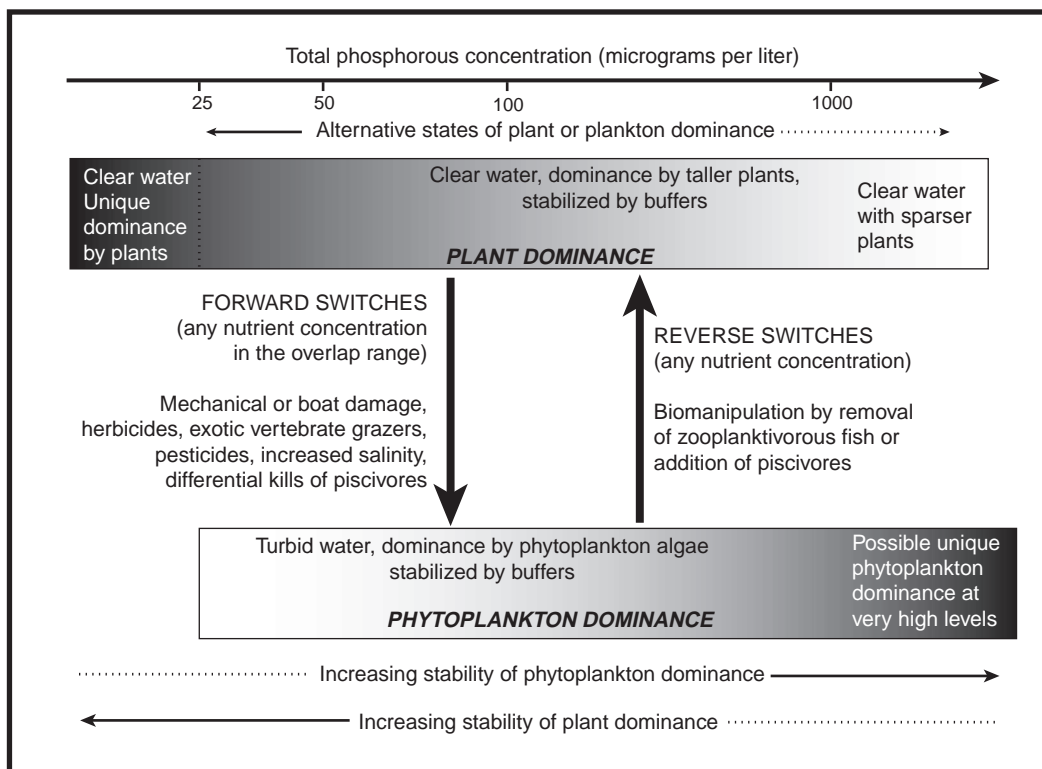
**Figure 4-6**  
**Conceptual Model of the Origin and Maintenance of Fish Assemblages Illustrating the Effects of Filters Operating on Faunal Characteristics and Community Structure on Different Spatial and Temporal Scales. Source: Reproduced With Permission From Tonn et al. (1990)**

A great deal of lake research has examined the relationship among nutrients, primary production, and food web structure. This research has demonstrated that productivity in lakes is controlled by both bottom-up processes (i.e., those that derive from nutrient supply) and top-down processes (i.e., those that derive from abundance and species composition of the food web, particularly at the uppermost trophic levels). Each perspective alone leaves unexplained a large portion of the

observed variation in lake productivity—a major factor in water body integrity as perceived by both the public and water resource managers.

Paleolimnology can be a useful tool for establishing reference conditions with respect to trophic state, because it allows for measurement of trophic state prior to human impacts (usually taken to begin at the time of settlement by European immigrants). Furthermore, paleolimnological methods can provide information on the variability in trophic state exhibited by a specific lake prior to the onset of anthropogenic influence. Still, trophic state is just one aspect of biological integrity.

Shallow lakes can be considered to exist in alternative stable states, in which plants dominate in clear water and phytoplankton dominate in turbid water (Figure 4-7; Moss et al. 1996). While nutrient concentration affects the relative stability of each state, forward and reverse switches may operate at any nutrient concentration to move the lake between states of plant and phytoplankton dominance. Forward switches include mechanical or boat damage, herbicides, exotic vertebrate grazers, pesticides, increased salinity, and differential kills of piscivores. Reverse switches include biomanipulation by removal of zooplanktivorous fish or addition of piscivores (see text box: Lake Mendota Case Study) (Moss et al. 1996). The switch may be due to a transient event (e.g., a fish kill) that has lasting effects on the plant/phytoplankton dominance state of the lake.



**Figure 4-7**  
**General Theory of Alternative Stable States in Shallow Lake Systems. Adapted From Moss et al. (1996)**

Lake Mendota, a large, eutrophic, urban lake in south central Wisconsin, is a well studied system and serves as an excellent case study of the complexity of interacting processes that can be encountered in lakes. That example (see text box) highlights how interaction of short-term and long-term processes, stochastic events, and ecosystem-level effects of an “intolerant” species can confound assessment of anthropogenic change based on assemblage data and regional expectations.

Research on biocriteria development for reservoirs has been spearheaded by the TVA. TVA has a substantial and long-standing research program that has developed multimetric indices for fish and benthic macroinvertebrates in reservoirs (Dycus and Meinert 1991, Hickman and McDonough 1996, Jennings et al. 1995, McDonough and Hickman 1999). The TVA’s Reservoir Fish Assemblage Index (RFAI) is the first and only fish index that has been developed for use in lakes or reservoirs (USEPA 1998a). In many respects, it follows the model of the original Index of Biotic Integrity (IBI; Karr 1981), but includes modifications to reflect the geographic region encompassed by TVA’s reservoirs as well as important differences between streams and reservoirs. Thus, it is a good example of how the IBI developed for Midwestern streams can be adapted for use in other regions and water body types. Metrics used in the RFAI are listed in Table 4-13. The RFAI is also informative regarding the effectiveness and appropriateness of sampling gears commonly used in lake and reservoirs. Statistical analysis of data collected over many years using standard protocols also provides insight into the level of sampling effort required to yield statistically meaningful results.

With respect to reference conditions, reservoirs are fundamentally different from lakes. A reservoir is a man-made water body that eliminates the natural habitat characteristic of any preexisting water body type. Consequently, application of the concepts of biological integrity and reference condition, as they are applied in natural water bodies, is both infeasible and inappropriate. The name given to TVA’s index was precisely chosen for this reason (Jennings et al. 1995). Given the unnatural state of reservoirs, the nebulous and problematic criterion of naturalness must be replaced with explicit human objectives.

Reference conditions for the RFAI are derived from the “best” conditions observed among all sites sampled using species lists that are cumulative over time and multiple reservoirs in the same reservoir classification (McDonough and Hickman 1999). While this approach to defining reference conditions is arguably very stringent (McDonough and Hickman 1999) and may not reflect attainable conditions, it avoids many of the technical problems that arise in lakes and other natural water bodies.

Development and testing of the RFAI continues in reservoirs on the Catawba River (North Carolina and South Carolina) and the Cumberland River (Tennessee and Kentucky) in a collaborative effort among several organizations including EPRI, Duke Power, and TVA (Olmsted and Hickman 1999).

## Case Study of Lake Mendota

Recent events in Lake Mendota, a eutrophic, urban lake in south central Wisconsin, provide an illustration of many of the factors that can affect lake ecosystems and their biotic communities. The trophic cascade hypothesis provided the model for an applied research project that attempted to assess the feasibility of reducing algal biomass and improving water clarity in Lake Mendota through intensive stocking of game fish. The trophic cascade hypothesis predicts that greater numbers of piscivorous game fish would consume larger numbers of zooplanktivorous fish thereby reducing their numbers. Fewer zooplanktivorous fish would exert less predation pressure, leading to an increase in the abundance of large, herbivorous zooplankton. Greater abundance of zooplankton, in turn, would lead to reduced phytoplankton biomass and improved water clarity. Experiments in other settings have demonstrated that manipulation of the food web (biomanipulation) can reduce algal biomass and primary production.

In order to increase the level of piscivory, large numbers of young walleye and northern pike were stocked in the lake. There was an immediate and drastic increase in angling pressure on the lake which was directly attributable to the publicity generated by the project. This was despite the fact that the stocked fish were not of catchable size and would not be of a size that could be legally harvested for several years. In the absence of adjustments in harvest regulations, this sociological response to an anticipated change in the fish community would have completely overwhelmed the initial management action. Prior to the stocked fish reaching a size at which they would have had a significant predatory impact, a massive die-off of the dominant zooplanktivore, cisco (*Coregonus artedii*), occurred. More than 95% of the cisco died over a short period of time, apparently due to a hypolimnetic wave (internal seiche) that enveloped the fish in anoxic water at a time of the year when temperatures in the surface waters prevented their escape. The effect of the die off on zooplanktivory exceeded the effect of predator stocking that was predicted to occur several years hence, and there was a nearly immediate effect on phytoplankton biomass and water clarity (Luecke et al. 1990). The cisco, which is near the edge of its distribution in Lake Mendota, has remained at very low abundance in the lake since that time.

The seiche was triggered by an unusual meteorological event that occurred at the worst possible time for cisco. It followed an extended stagnant period in which an unusually severe algal bloom developed on the lake. Thus, weather conditions, a short-term phenomenon, contributed to the severity of anoxia in the lake and triggered the proximate cause of the die off, while excessive nutrient loading, a long-term process, set the stage.

Cisco were vulnerable to elevated temperatures and low dissolved oxygen because they were near the southern edge of their range. Other species, more characteristic of the region, were more tolerant of conditions in the lake at the time of the cisco die off and were unaffected by the anoxic wave. The fish community has been in an alternative state for more than a decade.

**Table 4-13**  
**Metrics Used in TVA’s Reservoir Fish Assemblage Index (RFAI). Source: Hickman and McDonough (1996)**

Metric	Expected Response to Degradation
Taxon richness and composition Total number of species Number of Lepomis sunfish species Number of sucker species Number of intolerant species Percent individuals as tolerant species Percent dominance (numerical percentage of most common species)	Decrease Decrease Decrease Decrease Increase Increase
Trophic composition Number of piscivorous species Percent individuals as omnivores Percent individuals as invertivores	Decrease Increase Decrease
Reproductive composition Lithophilic spawning species	Decrease
Abundance Total number of individuals	Decrease
Fish Health Percent with diseases, parasites, or anomalies (including hybrids)	Increase

**Research Needs**

1. Biocriteria for Great Lakes. Considerations in developing biocriteria for the Laurentian Great Lakes need to be identified and explored. Similarities and important distinctions with inland lakes and with coastal marine waters should be identified.
2. Lake Classification. Existing lake classification schemes need to be evaluated for their ability to control for natural variability in ways that are useful for developing and applying multimetric indices for fish. In conducting this evaluation, the objective is not to simply reduce within-class natural variation, but to do so in ways that simple metrics and indices can distinguish anthropogenic effects from remaining natural variation in individual lakes. The preferred lake classification scheme will depend, in part, on the biological indices used, because there is little reason to classify lakes by factors to which the selected indices are unresponsive. Classification schemes accounting for within-lake variability should also be investigated.
3. Index Development. Despite the vast body of research on lakes and the extensive development of multimetric indices in other systems, very little index development work has been done for fish in lakes. Index development will be facilitated by more effective lake classification methods. Development of fish indices for lakes may draw upon work that has



been done in reservoirs, however, alternative methods for establishing reference conditions may be required in many cases.

## **Estuaries and Coastal Marine Waters**

Estuaries and coastal marine water bodies provide cooling water for 20% of the U.S. stream-electric utility generating capacity (EEI 1996). Over the last 25 years, licensing and permitting of these plants has largely defined the entrainment and impingement issue under § 316(b) (see Barnthouse et al. 1988). It is in this environment that evaluation of entrainment and impingement impacts will receive the greatest scrutiny and where biocriteria will face the greatest challenge as an assessment tool under § 316(b).

### ***Status of State Implementation***

Developmental work for biocriteria in estuaries and coastal marine waters has been underway on each coast of the country for many years, and biological indices, including multimetric indices, exist for several regions. Florida, however, is the only state that has incorporated biocriteria into its water quality standards for estuaries and coastal marine waters. Florida's biocriteria are based on benthic macroinvertebrates.

### ***Methods and Guidance***

EPA has not yet published draft technical guidance for biocriteria in estuarine and coastal marine waters. EPA currently plans to publish draft guidance in the spring of 2000. This review is based on an advance copy of the draft guidance dated March 30, 1997 (Gibson et al. 1997).

EPA's pre-publication draft guidance describes three tiers for biological surveys in estuaries and coastal marine waters, plus a desktop screening assessment. Each tier represents a more extensive level of sampling and assessment. Table 4-14 summarizes the components of each tier and the progression of the biocriteria process.

The desktop screening assessment (Tier 0) involves compilation of existing information and responses to questionnaires sent to local experts. Components of the desktop screening and recommended data sources are listed in Table 4-15. Desktop screening is expected to precede any of the other tiers and provide the information needed for planning of those surveys.

For Tier 1, a single visit is made to the sample sites during a predetermined index period. The purpose of a Tier 1 assessment is to develop an initial site classification scheme and identify candidate reference sites for each class. A Tier 1 site visit includes sampling of one or more biological assemblages and collection of data to characterize conditions in the water column and on the bottom (Table 4-14). These data are used to identify the nutrient state of the water body and identify point or nonpoint source causes of impairment, if possible. Additionally, the information collected in this tier is intended to provide a snapshot of the general condition of the sampled assemblages and habitat that can be used to detect:

**Table 4-14**  
**Summary of Tiered Sampling and Progression of the Biocriteria Process. Source: Gibson et al. (1997)**

Tier	Assemblage						Water Column Characteristics	Bottom Characteristics
	Proven				Developmental			
	Benthos	Fish	Macrophytes	Phytoplankton	Zooplankton	Epibenthos		
0 Compile existing information. Outline proposed classifications and biocriteria.	Compile documented information on: - estuary area - geomorphometric classification - habitat types present - basin & sub-basin land use				Literature search and survey questions of local experts - human population density - NPDES discharges - biological assemblages - water column & bottom characteristics			
1 Single visit. Preliminary classification and candidate biocriteria development.	- 3 replicate grabs - x-section Smith-McIntyre or Young grab - measure RPD depth - brief description of class & family - record presence/absence below 5 cm	- 3 trawls - 3 seines - species counts - measure standard lengths - record external abnormalities	- % cover estimate - record dominant taxa	- chlorophyll a - record blooms - identify dominant species	n.a.	- 3 trawls (can be concomitant with the fish trawls, if done)	- salinity/ conductivity - temperature - DO - pH - Secchi depth - depth	- grain size estimate & description - RPD layer depth - TVS - sediment toxicity
2 Multiple visits (2 or more) seasonally. Habitat classification and biocriteria established.	- 3 replicate grabs - seasonality covered - taxa & individuals for entire grab for each rep (split top vs bottom) - ID to genus/sp - possible biomass - multiple metrics	- 3 trawls - 3 or more replicates - species counts - measure standard lengths - record external abnormalities - biomass by species	- % cover - % area - maximum depth - taxa ID & wet weight of 2-3 samples each transect	- add dominant species including "nuisance taxa" on a seasonal basis	n.a.	- mid-summer or growing season average	- add nutrients (NH <sub>4</sub> , NO <sub>3</sub> , NO <sub>2</sub> , total & reactive P)	- add grain size (% sand, silt, clay) measurements - add TOC
3 Multiple visits (3 or more) to include seasons plus index period. More detailed studies – diagnostic. Management response possible.		- same as above - biomass by species - replicate 5x - possible stomach analysis - histopathology on representative subsample of catch	- add stem counts & biomass - record pathology	- characterize full community to species	- ID samples or subsamples to species	- same	- add pesticides, metals	- grain size characteristics - add AVS - sediment contaminants (organics, metals)

n.a. = Not applicable

**Table 4-15**  
**Desktop Screening Assessment for Estuaries and Coastal Marine Waters. Source: Gibson et al. (1997)**

Component	Information Source
Estuary area	USGS quad maps, GIS
Geomorphometric classification (coastal plain, estuary, lagoon, fjord, tectonically-caused estuary)	USGS quad maps, GIS
Habitat type (open water, soft bottom, hard bottom, macrophytes, high/low energy beach, sandflat, mudflat, emergent marsh)	NOAA bathymetry charts; historic surveys by federal, state agencies and universities
Biological assemblages (benthos, fish, macrophytes, phytoplankton, zooplankton, epibenthos). Marine mammal tissue contaminants.	Historic data from federal, state agencies and universities. NMFS for marine mammal data.
Watershed land use	USGS land use maps; state and county planning agencies; local zoning agencies; USDA CES
Population density	US census data
NPDES discharges	State water quality agency and regional EPA office; PCS database
Water column and bottom characteristics (salinity/conductivity, temperature, DO pH, Secchi depth, depth, grain size, RPD layer depth, nutrients (N&P), pesticides, metals, sediment contaminants)	Historic data from federal, state agencies and universities. STORET, NODC databases.

- Wetland and shorezone fish habitat loss
- Loss of aquatic macrophytes
- Potential impairment of benthic macroinvertebrate and fish assemblages
- Oxygen stress

Tier 2 assessment involves two or more sampling episodes each year to encompass seasonal variation. It requires sampling of at least two biological assemblages and more detailed investigation of conditions in the water column and substrate (Table 4-14). This level is intended to be sufficiently rigorous to allow the state to develop biocriteria (i.e., numeric benchmarks), detect impairment, and evaluate potential causes of the impairment. Tier 2 also supports assessment of trophic state, extent of macrophyte coverage, and, potentially, identification of phytoplankton taxa responsible for blooms.

Tier 3 assessment is intended to provide the information needed for diagnosis of the sources and causes of physical, chemical, and biological impairment, and for evaluating success of

management actions. This tier requires sampling at least once each season during each year. Data are collected for a minimum of three assemblages, and measurements of chemical and physical conditions in the water column and bottom include the parameters required for Tier 2 assessments, plus additional parameters listed in Table 4-14.

### ***Geographic Classification***

The pre-publication draft USEPA guidance on estuaries and coastal marine waters recommends multiple levels of classification, including geographic region, watershed characteristics, and waterbody characteristics. The guidance specifies that the minimal number of levels required to meaningfully classify a given estuary or coastal area should be used.

### ***Recommended Assemblages***

The pre-publication draft guidance uses the following criteria in identifying target assemblages appropriate for bioassessment in estuaries and coastal marine waters:

- Unambiguous utility for biological assessment,
- Cost-effective data collection and interpretation, and
- Easily calculated metrics for use alone or in a multimetric index.

Based on these criteria, the pre-publication draft guidance recommends the use of the following assemblages:

- Benthic macroinvertebrates,
- Fish,
- Aquatic macrophytes, and
- Phytoplankton (measured as chlorophyll *a*)

EPA pre-publication draft guidance explicitly identifies zooplankton and epibenthos as assemblages that, while promising, do not yet meet the criteria for recommended assemblages.

Indices of biological integrity for estuaries and near coastal waters have been developed for fish (e.g., Deegan et al. 1997, Guillen 1995, Jordan et al. 1992, Thompson and Fitzhugh 1986) and benthic macroinvertebrates (e.g., Chapman 1989, Engle et al. 1994, Ranasinghe et al. 1994, Weisberg et al. 1993). Biotic indices for benthos in estuaries (e.g., Ranasinghe et al. 1994) are more highly developed than those for fish; benthos are sedentary making them good indicators of local environmental conditions (Engle et al. 1994).

### ***Sensitivity of Assemblages to Impacts From Entrainment and Impingement***

Research on various assemblages' response to point source pollution effects in coastal marine areas has revealed that less mobile species, such as benthic macroinvertebrates, tend to be more sensitive indicators than are more mobile species such as fish. In a biological investigation of the outfall of a coastal wastewater treatment plant, Gibson (1995) found that benthic macroinvertebrates exhibited effects while benthic fish did not. Gibson attributed the results to mobility of the fish in open coastal waters, seasonal migrations, and potential sport and commercial fishing pressure. In contrast, benthic invertebrates typically constituting benthic indices have very limited movement and a closer association with environmental conditions at the point of sample collection.

While benthic macroinvertebrate indices have been proven useful for detecting impacts of water quality degradation and sediment contamination, the same cannot be said for their detection of impacts due to power plant intakes. This is to be expected, because the relationship of benthic community structure and function to entrainment and impingement at CWIS is indirect and attenuated at best. Possible exceptions to this statement are crabs, shrimp and oyster spat; however, they also differ in that they are much more mobile than other components of the benthic community and (except for settled oysters) are not captured with the gears typically used for collection of benthic macroinvertebrates.

Phytoplankton are another assemblage recommended by EPA for possible inclusion in biocriteria programs for estuarine and coastal marine waters. It is unlikely that phytoplankton will be useful for assessing impacts of CWIS because of the rapid turnover and mixing of phytoplankton that occurs in estuaries and coastal marine waters.

A case study in Galveston Bay, Texas (Gibson et al. 1997), developed a prototype multimetric fish index for the northwestern Gulf of Mexico. The project examined seines, gillnets, and trawls. Seines and trawls were chosen for inclusion in future studies, because metrics derived from the two gears were not strongly correlated (Tables 4-16 and 4-17). Proposed seine metrics included proportion of bay anchovy in the catch. Scores assigned for this metric were inversely related to the proportion of bay anchovy in the catch, which would appear to be at cross purposes with assessing impacts of entrainment and impingement. This potential problem—entrainment and impingement producing effects counter to those expected for other classes of stressors—may be a common phenomenon as indices are developed for estuarine and coastal fishes. Impingement and entrainment could potentially produce a higher score for any metric that increases with decreasing abundance.

**Table 4-16**  
**Proposed Seine Metrics for Use in an Estuarine IBI Along the Texas Coast. Source: Gibson et al. (1997)**

Metric	Summer Value	Fall Value	Winter Value	Spring Value	Score
<b>Category A</b>					
(A) Total Catch	<200	<50	NA	NA	1
	200-450	50-400	≤900	≤700	2
	>450	>400	>900	>700	3
<sup>1</sup> (A) Log Catch	<4.5	<3.9	<4.2	<1.5	1
	4.5-6	3.9-5.8	4.2-6.4	1.5-6.3	2
	>6	>5.8	>6.4	>6.3	3
<sup>1</sup> Proportion Penaeid Shrimp	<.01	<.25	NA	NA	1
	.01-.3	.25-.56	NA	≤.04	2
	>.3	>.56	NA	>.04	3
<b>Category B</b>					
(B) Proportion Shad	>.83	>.60	>.59	>.78	1
	NA	.04-.60	NA	NA	2
	≤.83	≤.04	≤.59	≤.78	3
<sup>1</sup> (B) Proportion Bay Anchovy If Bay A. = 0 then use "shad" metric	>.8	>.52	>0.13	>.34	1
	NA	.04-.52	NA	NA	2
	≤.8	<.04	≤0.13	≤.34	3
<sup>1</sup> Dominance Ratio	>.88	>.65	>.82	>.78	1
	.44-.88	.40-.65	.26-.82	.27-.78	2
	<.44	<.40	<.26	<.27	3
<b>Category C</b>					
<sup>1</sup> (C) Mean # Taxa	<6	<6	<6	<5	1
	6-11	6-10	6-10	5-10	2
	>11	>10	>10	>10	3
(C) Cum. # Taxa	<10	<6	<11	<11	1
	10-19	6-11	11-18	11-19	2
	>19	>11	>18	>19	3
(C) Mean # Fish Taxa	<3	<3	<3.5	<4	1
	3-7	3-7	3.5-7	4-8	2
	>7	>7	>7	>8	3
<b>Total IBI Score</b>					
Concern	5-7	5-7	4-5	7-9	
Normal	8-12	8-12	6-10	10-12	
Excellent	13-15	13-15	11-12	13-15	
<b>Total IBI Score (WHEN INVERTEBRATES ARE NOT USED)</b>					
Concern	4-5	4-5	4-5	5-6	
Normal	6-10	6-10	6-10	7-10	
Excellent	11-12	11-12	11-12	11-12	

<sup>1</sup> Recommended metric; if mean total or log total catch = 0 then score = high concern.

**Table 4-17**  
**Proposed Trawl Metrics for Use in an Estuarine IBI Along the Texas Coast. Source: Gibson et al. (1997)**

Metric	Summer Value	Fall Value	Winter Value	Spring Value	Score
<sup>a</sup> Proportion Total Catch as P. Shrimp	a	<.42	*	A	1
	≤.45	.42-.83	*	≤.08	2
	>.45	>.83	*	>.08	3
<sup>a</sup> Proportion Total Catch as P?. Shad	>.06	>.08	*	>.03	1
	NA	NA	*	NA	2
	≤.06	≤.08	*	≤.03	3
<b>Category A</b>					
<sup>a</sup> Mean # Nekton Taxa	<1.8	<4.3	<4.4	<4.1	1
	1.8-9.3	4.3-9.9	4.4-8.8	4.1-7.7	2
	>9.3	>9.9	>8.8	>7.7	3
Mean # Fish Taxa	<2.2	<4.2	<2.1	<1.6	1
	2.2-6.3	4.2-6.6	2.1-6.8	1.6-4.9	2
	>6.3	>6.6	>6.8	>4.9	3
<b>Total IBI Score</b>					
Concern	4	3	1	4	
Normal	5-8	4-8	2	5-8	
Excellent	9	9	3	9	

<sup>a</sup> Recommended metric; if mean log total catch or total catch = 0, then score = high concern.

\* Avoidance of winter sampling is recommended due to lack of suitable metrics.

Recommend transformed formula for future applications.

NOTE: To avoid problems caused by division by zero use the following formulas:  
 For shrimp and shad proportions, let metric value = tax group catch/(total catch +1)  
 Alternately if any one replicate total catch = 0, then an IBI score of 'concern' can be given.

Metrics based on species richness and other taxonomic presence-absence information are a prominent feature of most multimetric indices, but they can be problematic when applied to § 316(b). Presence-absence metrics respond to stressors that reduce, either directly or indirectly, the abundance of individuals in various taxa to such low numbers that none are captured at a site. However, impingement and entrainment could significantly reduce the abundance of recreationally or commercially important fishes without reducing their abundance to the point that they are absent from samples. In this case, metrics based on presence-absence information would only be responsive to impingement and entrainment of abundant species via strong, indirect effects on other, less abundant species. Thus, these metrics may not be useful for assessing the potential for adverse effects of impingement and entrainment.

Simple metrics based on relative abundance are also problematic. In fact, Karr and Chu (1999:125) state, without qualifying the type of aquatic ecosystem, “abundance, density, and production vary too much to use in multimetric biological indexes.” Efforts to link variability in

single fish populations to natural variables (e.g., hydrographic) and anthropogenic factors (e.g., gross pollution indicators) have been extensive (e.g., see Summers and Rose 1987, Pepin 1991, Pepin and Myers 1991, Rose and Summers 1992), however, the efforts have had only limited success. Fish distribution is highly variable, at scales ranging from meters (schools) to 1000's of kilometers. Natural temporal variation in abundance can also occur at intermediate time scales that are too long to average over for the purposes of sampling and assessment, but too short to allow adjustment of reference conditions. Monitoring and attempting to explain recruitment variability in marine fish stocks, in fact, has been a challenge to marine fisheries scientists for nearly 100 years (Hjort 1914, Beverton and Holt 1957, Sissenwine 1984, Rothschild 1986, Houde 1987 and 1989, Rothschild and DiNardo 1987, Cushing 1995). Fluctuation in abundance of both west and east coast fish stocks in relation to fluctuations in the Southern Ocean Oscillation (El Nino) demonstrates the broad geographic scale natural variability that can occur in marine and estuarine fish (Rothschild and Fogarty 1998).

A major challenge in constructing multimetric indices for fish in estuaries and near coastal waters will be to develop methods of data collection and analysis that accommodate the multitude of relevant temporal and spatial scales and stressors (Livingston 1987, Wolfe et al. 1987) while remaining responsive to any adverse effects of entrainment and impingement. Estuaries and coastal habitats are large, open systems. Many estuarine and coastal species, particularly those that are of commercial and recreational importance, are wide ranging and integrate conditions over a broad geographic area. For these species, local abundance does not necessarily reflect local conditions. In fact, local conditions may be insignificant compared to factors or processes operating at other points in time and space.

Coastal fisheries are an example of a factor that is displaced in time and space. Harvest is often an overriding factor in the population dynamics of many estuarine and coastal species (Hall 1999, NRC 1999). Striped bass in the Hudson River is a case in point. In the 1970's and 1980's there were major concerns about entrainment and impingement of striped bass by power plant intakes on the river. More recently, striped bass have shown a resurgence. This recovery of striped bass may be due more to coast-wide reductions in harvest and regional fishery closures related to concerns associated with PCB contamination of the fish than to decreased impacts from entrainment or impingement. Confounding of effects of impingement and entrainment with those of commercial and recreational exploitation has been and will continue to be a significant and technically challenging issue.

The issue of stressor identification is important more generally in the context of estuaries and coastal waters, because estuarine and coastal fishes are wide-ranging and potentially respond to a broad array of stressors. Factors other than entrainment and impingement may be responsible for impairment when it exists. The guidance indicates that Tier 3 assessments are intended to provide a diagnosis of the source(s) of impairment. While this may be feasible for some stressors in some situations (e.g., excessive nonpoint source nutrient loading), assessments of the kind described in the draft guidance will often be inadequate to diagnose the causes of impairment of the fish community. This is especially true in cases involving species that are exposed to significant commercial and recreational harvest.

These characteristics of estuarine and coastal marine systems suggest that development of a meaningful multimetric index for marine and coastal fish will be extremely challenging if not infeasible in the near-term, especially for application to § 316(b).



## **Research Needs**

Development of multimetric indices will require careful consideration of the processes controlling the relative abundances and presence/absence of estuarine and coastal fish and of the temporal and spatial scales at which these processes are manifested. This is critically important to developing meaningful reference conditions and indices that are reliably and exclusively responsive to anthropogenic effects. These considerations are not likely to be adequately addressed within a time frame relevant to new 316(b) rules. The biocriteria approach for assessing potential impacts of impingement and entrainment at estuarine and coastal power plants, therefore, is problematic. Long-term research needs toward *future* development of biocriteria for fish in estuarine and coastal systems include:

1. Temporal and spatial scaling. Extensive efforts have been made to describe the life history characteristics of estuarine and coastal fishes; however, this has been done largely on a species-by-species basis. A subset of this information must be compiled and evaluated to identify the temporal and spatial scales at which important parts of the life cycle and the life cycle as a whole unfold. The information can then be synthesized in a community context to identify feasible spatial and temporal scales at which to assess and detect anthropogenic impairment based on assemblage data. Variability of the data used to assess estuarine and coastal fishes also needs to be characterized.
2. Responsiveness of metrics to entrainment and impingement. Individual metrics under consideration for inclusion in fish indices need to be examined for their responsiveness to impingement and entrainment. Given the information identified in item 1 above, computer simulation could be used to examine the implications of intrinsic temporal-spatial variability and sampling variability, and for focusing index development efforts and sampling programs.



# 5

## SUMMARY AND CONCLUSIONS

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The growing emphasis on comprehensive biological assessment in water resource management represents an important advance over more traditional, narrowly focused approaches to assessment. Biocriteria place the emphasis where it belongs, on biological conditions in the waterbody. Furthermore, biological criteria represent the net results of complex and sometimes poorly understood ecological processes, rather than intermediate endpoints whose ecological significance must be further interpreted.

The suitability of biocriteria, as biological benchmarks based on multimetric indices, for regulation of CWISs under § 316(b) of the CWA depends largely on the specific roles biocriteria will be assigned in the final regulatory framework. Multimetric indices are designed to be responsive to a broad array of stressors. Their primary function is one of detecting impairment. However, evaluation of the magnitude of a specific effect and the cause of an effect are questions which multimetric indices are not primarily designed to address. Furthermore, while the multimetric approach is well suited for assessing community-level effects, it is not designed to detect population-level effects. Consequently, other methods are better suited to assessing the population-level effects of impingement and entrainment, especially in situations where other factors, such as commercial and recreational fishing, may have substantial effects on fish populations.

Population-level assessments will be required in many applications of § 316(b). Thus, biocriteria will not eliminate the need for this and other forms of assessment. However, information derived from biocriteria programs will support population-level studies and will be useful in planning those studies. Biocriteria also will help place the results of more narrowly focused assessments in a broader ecological context.

The effectiveness of a biocriteria approach ultimately depends on the definition of reference conditions which characterize a state of health for the system. Yet, assumptions about the structure and function of ecosystems embedded in the concept of ecosystem health as applied to biocriteria appear to conflict with current understanding of ecosystems as dynamic, non-equilibrium systems that are structured on multiple time and space scales.

The implications of these alternative paradigms for biocriteria need to be rigorously examined. As identified in this report, these implications include the potential misclassification of test sites and significant uncertainty in reference conditions due to unaddressed sources of natural variability among reference sites. These issues will be of particular importance for large, open systems such as estuaries and coastal marine waters.

While this study addresses many of the biocriteria-related issues from a conceptual standpoint and considers some of their practical implications, further investigation is needed through

computer- and field-based experimentation to assess the panoply of management consequences for these different types of water resources.

The multimetric approach was developed in response to weaknesses of the earlier assessment approaches. Specifically, resource degradation continued in some cases, even as large sums of money were spent to meet objectives defined by assessment criteria. Only through thorough understanding of the strengths, weaknesses, and pitfalls of the biocriteria approach can water resource managers prevent this recurrence.

In addition to the broad conceptual issues, improved understanding is needed of the theoretical and statistical implications of the entire biocriteria development process, including the fundamentals of multimetric index development. Investigation of these issues requires an expanded cadre of researchers from the one that has developed the approach to this point. More extensive collaboration between theoreticians, statisticians, and field biologists is required to produce bioassessment methods that are practical, effective over the long term, and defensible in the highly adversarial § 316(b) arena.

The most difficult work remains to be done in the very systems that will be most challenging and contentious in the context of § 316(b) regulations. Moreover, this work is of a time-sensitive nature. Over the coming months, EPA will take its § 316(b) regulatory framework from a pre-decisional state to a draft regulatory state and, ultimately, to final regulation. Research undertaken now can benefit this rulemaking process by identifying more appropriate limits and uses for a biocriteria approach under § 316(b). Such research will also help ensure that the approaches employed are technically sound and defensible.

# 6

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## Section A – Overall Assessment

(Instruction: Answer questions 1 - 4 using the following scale. Answer by circling only **one** score for each question.)

<u>Score</u>	<u>Description</u>
5	Species composition, age classes, and trophic structure comparable to non (or minimally) impacted sites of similar waterbody size in that ecoregion.
4	Species richness somewhat reduced by loss of some intolerant species; young of the year of top carnivores rare; less than optimal abundances, age distributions, and trophic structure for waterbody size and ecoregion.
3	Intolerant species absent, considerably fewer species and individuals than expected for that waterbody size and ecoregion, older age classes of top carnivores rare, trophic structure skewed toward omnivory.
2	Dominated by highly tolerant species, omnivores, and habitat generalists; top carnivores rare or absent; older age classes of all but tolerant species rare; diseased fish and anomalies relatively common for that waterbody size and ecoregion.
1	Few individuals and species present, mostly tolerant species and small individuals, diseased fish and anomalies abundant compared to other similar-sized waterbodies in ecoregion.
0	No fish.

1. Circle the score that best describes your impression of the current condition of the waterbody.

5      4      3      2      1      0

2. Classify the condition of the lake 10 years ago.

5      4      3      2      1      0

3. Given present trends, what score will be representative of lake conditions 10 years from now?

5      4      3      2      1      0

4. If the major human-caused limiting factors were eliminated, how would the lake be rated 10 years from now?

5      4      3      2      1      0

**Subsection A.1 – Water Quality**

(Instructions: Complete subsection A.1 – A.4 by **circling** the single most appropriate limiting factor and probable cause. If there is more than one limiting factor and cause, please **rank** them accordingly (by assigning a “1” for the primary factor and cause, “2” for the secondary factor and cause, etc.).)

<u>Limiting Factor</u>	<u>Rank</u>	<u>Probably Cause</u>	<u>Rank</u>
Temperature too high	_____	Quality of tributaries	_____
Temperature too low	_____	In-lake processes	_____
Turbidity	_____	Point source discharge	_____
Salinity	_____	Industrial	_____
Dissolved oxygen	_____	Municipal	_____
Gas supersaturation	_____	Combined sewer	_____
pH too acidic	_____	Mining	_____
pH too basic	_____	Upstream dam release	_____
Nutrient deficiency	_____	Nonpoint source discharge	_____
Nutrient surplus	_____	Individual sewage	_____
Toxic substances	_____	Urban runoff	_____
Excessive water level fluctuation	_____	Landfill leachate	_____
Other (specify below)	_____	Construction	_____
_____		Agriculture	_____
Not limiting	_____	Feedlot	_____
		Grazing	_____
		Silviculture	_____
		Mining	_____
		Dam surface release	_____
		Shorezone disturbance	_____
		Natural	_____
		Unknown	_____
		Other (specify below)	_____
		_____	

**Subsection A.2 – Habitat Structure**

<u>Limiting Factor</u>	<u>Rank</u>	<u>Probable Cause</u>	<u>Rank</u>
Excessive siltation	_____	Agriculture	_____
Insufficient structure	_____	Silviculture	_____
Insufficient shallows	_____	Mining	_____
Insufficient macrophytes	_____	Grazing	_____
Excessive macrophytes	_____	Dam	_____
Insufficient concealment	_____	Diversion	_____
Insufficient reproductive habitat	_____	Channelization	_____
Other (specify below)	_____	Snagging	_____
_____		Natural	_____
Not limiting	_____	Aquatic weed management	_____
		Unknown	_____
		Other (specify below)	_____
		_____	

**Subsection A.3 – Fish Community**

<u>Limiting Factor</u>	<u>Rank</u>	<u>Probable Cause</u>	<u>Rank</u>
Overharvest	_____	Aquarists	_____
Underharvest	_____	Point source	_____
Fish stocking	_____	Nonpoint source	_____
Non-native species	_____	Natural	_____
Migration barrier	_____	Unknown	_____
Tainting	_____	Management	_____
Food limited	_____	State agency	_____
Habitat	_____	Federal agency	_____
Fish kills	_____	Weed control	_____
Other (specify below)	_____	Other (specify below)	_____
_____		_____	
Not limiting	_____		

**Subsection A.4 – Major Limiting Factor**

<u>Limiting Factor</u>	<u>Rank</u>
Water quality	_____
Water quantity	_____
Habitat structure	_____
Fish Community	_____
Other (specify below)	_____

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**Subsection A.5 – Algae**

(Instructions: Please provide short answers to questions 1 – 7 as appropriate.)

1. Is there a presence and history of nuisance algae blooms? \_\_\_\_\_
2. Have algae blooms resulted in fish kills or other adverse changes to the fish community?  
\_\_\_\_\_
3. Has algae caused odor problems or taste problems in drinking water?  
\_\_\_\_\_
4. Have algae blooms deterred swimmers or affected other forms of contact recreation?  
\_\_\_\_\_
5. Are there other problems caused by algae blooms; and if so, what are they?  
\_\_\_\_\_
6. What is the source of your information?  
\_\_\_\_\_
7. Are there other sources of information that the agency should be aware of such as fishery records and gray literature studies? \_\_\_\_\_

## Section B – Aquatic Macrophyte Community

(Instructions: Answer questions 1 – 3 using the following scale. Circle only one score for each question.)

<u>Score</u>	<u>Description</u>
3	Extent and cover are comparable to non (or minimally) impacted sites of similar waterbody size in that ecoregion.
2	Macrophyte beds appear weedy. The extent and/or cover are greater than non (or minimally) impacted sites. The dominant species are those found in highly eutrophic waters.
1	Few macrophytes found compared to non (or minimally) impacted sites. Macrophytes that are found are usually exotics and are tolerant of a wide range of water quality conditions and/or fluctuations.
0	No macrophytes.

1. Circle the score that best describes your impression of the current macrophyte conditions of the lake.

3      2      1      0

2. Classify the macrophyte conditions of the lake 10 years ago.

3      2      1      0

3. Given the present trends, what score will be representative of lake conditions 10 years from now?

3      2      1      0



**Subsection B.1 – Factors Affecting Macrophytes**

(Instructions: Complete subsection by **circling** the single most appropriate limiting factor and probable cause. If there is more than one limiting factor and cause, please rank them accordingly (by assigning a “1” for the primary factor and cause, “2” for the secondary factor and cause, etc.).

<u>Limiting Factor</u>	<u>Rank</u>	<u>Probable Cause</u>	<u>Rank</u>
Grass carp introduction	_____	Aquarists	_____
Exotic species	_____	Point source	_____
Excessive siltation	_____	Nonpoint source	_____
Drawdowns	_____	Natural	_____
Weed control	_____	Unknown	_____
Shoreline cleanup	_____	Management	_____
Excessive epiphytes	_____	State agency	_____
Excessive turbidity	_____	Federal agency	_____
Insufficient shallows	_____	Fishermen	_____
Elevation or latitude	_____	Other (specify below)	_____
Macrophyte beds are expanding	_____	_____	
Other (specify below)	_____		
<hr/>			
Not limiting	_____		

**Subsection B.2 – Macrophyte Extent and Species**

(Instruction: Please provide short answers to questions 1 – 4, as appropriate.)

1. What is the extent of macrophyte coverage in the photic zone? \_\_\_\_\_  
\_\_\_\_\_
2. What are the dominant species? \_\_\_\_\_  
\_\_\_\_\_
3. What is the source of your information on macrophytes? \_\_\_\_\_  
\_\_\_\_\_
4. Are there other sources of information on the macrophyte community in this waterbody that the agency should be aware of such as management reports of gray literature studies?  
\_\_\_\_\_

**Section C – Watershed Characteristics and Land Use**

(Instructions: Please provide short answers to questions 1 – 10, as appropriate.)

1. Watershed size \_\_\_\_\_ acres
2. Elevation difference \_\_\_\_\_ ft  
[watershed divide to lake surface]
3. Forest or natural vegetation \_\_\_\_\_
4. Urban \_\_\_\_\_ %
5. Agricultural \_\_\_\_\_ %
6. Suburban/residential \_\_\_\_\_ %
7. Human population density in lake watershed \_\_\_\_\_
8. Number of dischargers within the watershed (e.g., NPDES permits) \_\_\_\_\_
9. What is the source of your information on the watershed? \_\_\_\_\_
10. Are there other sources of information on the watershed and surrounding land use that the agency should be aware of such as gray literature or land use planning documents?  
\_\_\_\_\_

*Target:*


Section 316 (a) and (b) Fish Protection Issues

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 Printed on recycled paper in the United States of America

TR-114007