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BIOLOGICAL CRITERIA

Technical Guidance for Streams and Small Rivers

Revised Edition

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CHAPTER 1.

Introduction

The goal of this document is to help states develop and use biocriteria for streams and small rivers. The document includes a general strategy for biocriteria development, identifies steps in the process, and provides technical guidance on how to complete each step, using the experience and knowledge of existing state, regional, and national surface water programs.

This guidance document is designed primarily for water resource managers and biologists familiar with standard biological survey techniques and similarly familiar with the EPA guidance document "Rapid Bioassessment Protocols for Use in Streams and Rivers: Benthic Macroinvertebrates and Fish" (Plafkin et al. 1989). It should be used in conjunction with that earlier text.

The biosurvey-biocriteria process provides a way to measure the condition of a water resource, that is, its attainment or nonattainment of biological integrity. In turn, biological integrity is a conceptual definition of the most robust aquatic community to be expected in a natural condition — in a water resource unimpaired by human activities. Thus, biological criteria are the benchmarks for water resource protection and management; they reflect the closest possible attainment of biological integrity. It follows that any criterion representing less than achievable biological integrity is an interim criterion only, since the use of biocriteria are intended to improve the nation's water resources.

The guidance in this document is designed so that users may tailor the methods to their particular biocriteria development needs. Chapters 1 and 8 are inclusive of the methodology — at different levels of complexity — while chapters 2 through 7 explore the process step by step. Thus, the document is organized as follows:

- **Chapter 1:** Introduction. An overview of the process.
- **Chapter 2:** Components of Biocriteria. An exploration of the basic relationship between biological integrity and biocriteria, the complex nature of human disturbances, and the definition of biological expectations.
- **Chapter 3:** The Reference Condition. Selection of reference sites and the role of the reference condition in biocriteria development.

Purpose:
To provide conceptual guidance on how and when to use the biosurvey-biocriteria process to evaluate streams and small rivers.

- **Chapter 4:** Conducting the Biosurvey. An investigation of the design, management, and technical issues related to biocriteria-bioassessment programs, the various biosurvey methods and their standardization.
- **Chapter 5:** Evaluating Environmental Effects. Factors that affect water resource integrity.
- **Chapter 6:** Multimetric Assessment Approaches for Biocriteria Development. Emphasis on the community composition element of biological surveys.
- **Chapter 7:** Biocriteria Development and Implementation. Designing and developing biocriteria from the data and precautions for some site selections.
- **Chapter 8:** Applications of the Biosurvey-Biocriteria Process. Case Studies from North Carolina, Ohio, Delaware, and Maine.

Each chapter concludes with a list of readings containing supplemental information on the specific topic treated in that chapter. An extensive glossary and full reference list appear at the end of the document. Future documents will be oriented to other waterbody types: lakes and reservoirs, rivers, estuaries near coastal marine waters, and wetlands.

The Concept of Biocriteria

Biocriteria are developed from expectations for the region or watershed, site-specific applications, and consensus definitions by regional experts. The biological sampling for this process requires minimally impaired reference sites against which the study area may be compared.

Early efforts to monitor human effects on waterbodies in the 18th century were limited to physical observations of sediment and debris movement resulting from land settlement, and commercial activities (Caper et al. 1983). Later, as analytical methods became increasingly available for measuring microchemical conditions in the waterbody (Gibson, 1992), chemical measurements became the most commonly employed source of water quality criteria. However, investigators and resource managers have long recognized that such water column measurements reflect conditions only at the time of sampling.

To understand fully the effects of human activities on water resources, biological sampling is an important supplement to chemical sampling. Biological measurements reflect current conditions as well as temporal changes in waterbodies, including the cumulative effects of successive disturbances.

Three aspects of water resource management (chemical, physical, and biological) are recognized in the National Clean Water Act as amended by the Water Quality Act of 1987 (U.S. Gov. Print. Off. 1988). Section 101a states that the Act's primary objective is to "restore and maintain the chemical, physical, and biological integrity of the nation's waters."

The development and widespread use of formal biological criteria (biocriteria) has lagged behind chemical-specific, in-stream flow, or toxicity-based water quality criteria in waterbody management (U.S. Environ. Prot. Agency, 1985a,b; 1986). Biological criteria are numeric values or narrative expressions that describe the preferred biological condition of aquatic communities based on designated reference sites. The conditions of aquatic life found at these reference sites are used to help detect both the causes and levels of risk to biological integrity at other sites in the

same region. In keeping with the policy of not degrading the resource, the reference conditions — like the criteria — are expected to be upgraded with each improvement to the water resource. Thus, biocriteria contribute directly to water management programs, and recent recommendations (U.S. Environ. Prot. Agency, 1987a,b) on monitoring strategies for aquatic resources have emphasized the need to accelerate the development of biological sampling as a regular part of surface water programs.

Biocriteria are developed from expectations for the region or watershed, site-specific applications, and consensus definitions by regional authorities. The biological sampling for this process requires minimally impaired reference sites against which the study area may be compared. Minimally impaired sites are not necessarily pristine; they must, however, exhibit minimal disturbance (i.e., human interference) relative to the overall region of study.

Applications of Biocriteria

Biocriteria applications are presented in some detail in chapter eight. Here, a brief description of these applications is sufficient to demonstrate the usefulness of the concept.

■ **Aquatic Life Designated Uses.** The States and Tribes together with EPA identify the most appropriate uses of our water resources and then manage or restore these waters accordingly. Some aquatic life uses are cold water fisheries, warm water fisheries, unique natural systems, and systems including rare or endangered species. Biological assessments and subsequent criteria are essential to the development and refinement of these designations and the management necessary to support them.

■ **Problem Identification.** Biological surveys and their comparison to established biological criteria, in addition to traditional chemical and physical investigations, often provide insights into problems not otherwise identifiable. For example, new compounds or synergistic reactions between existing waterborne chemicals may affect the biota even though individual chemical tests show no rise in historic concentrations; hydrologic modifications such as installed impoundments may restrict species distribution and recruitment; increased watershed sealed surfaces may change flow regimes, cause more scouring, and destroy habitat for essential community assemblages.

■ **Regulatory Assessments.** Much of the work done by EPA is regulatory in nature and involves the use of permits to regulate the discharge of various substances into the waters. The Agency does not require the use of biocriteria as numeric regulatory limits in National Pollution Discharge Elimination system (NPDES) permits. It does, however, strongly recommend that states develop and use biocriteria as a permit assessment tool and as a mechanism for evaluating the success of pollution control efforts. Concurrence of biotic data with established biocriteria can be a key measure of permit effectiveness and of regulatory compliance.

■ **Management Planning.** Water resource managers can use the relative relationships of a series of similar streams, as ranked by their compliance with biocriteria, as a means of assigning priorities to their management ef-

forts. In this way budgets and manpower can be applied most effectively because the manager is better informed about the most pressing problems and about those streams most likely to respond to restorative efforts.

■ **Water Quality Project Evaluations.** The measurement of the resident stream biota before, during, and after implementation of pollution management efforts is an excellent way to evaluate the success or failure of those techniques.

■ **Status and Trends of Water Resources.** As states and tribes gather more biological data in support of their biocriteria, their knowledge of the waters becomes more refined. The condition of the nation's waters will be better understood and the direction of change in the various regions will be more evident and better addressed.

To achieve these objectives for the use of biocriteria, EPA is evaluating not only the role of biocriteria in the permit process but also the independent application of various criteria to determine water resource quality. Presently chemical, physical, and biological criteria — when used in a regulatory context — are applied to a waterbody independently. Compliance or lack of compliance with one criterion does not influence the application of another. As biological and other types of criteria, such as sediment criteria (now being investigated) are more widely implemented in state programs, the Agency will continue to investigate the usefulness of weight of evidence approaches as an alternative.

Thus, biocriteria expand aquatic life use designations and improve water quality standards, help identify impairment of beneficial uses, and help set program priorities. Biological surveys (or biosurveys) in conjunction with biocriteria are valuable because they provide

- a direct measure of the condition of the water resource at the site,
- early detection of problems that other methods may miss or underestimate,
- a systematic process for measuring the effectiveness of water resource management programs,
- an evaluation of the adequacy of permits, and
- a measurement of the status and trends of streams over time and space.

Biocriteria expand aquatic life use designations and improve water quality standards, help identify impairment of beneficial uses and help set program priorities.

The Development, Validation, and Implementation Process for Biocriteria

Three processes are part of the overall implementation plan to incorporate biocriteria into the surface water programs of regulatory agencies: the development of biocriteria and associated biological survey methods, the validation of the reference condition and survey techniques, and the implementation of the program at various sites within watersheds with subsequent determinations of impairment.

The development of biocriteria by regulatory agencies partly depends on bioassessment to evaluate or compare ecosystem conditions. Bioassess-

ment contains two types of data: toxicity tests and field biological surveys of surface waters. Toxicity tests are described elsewhere (U.S. Environ. Prot. Agency, 1985a,b; 1988; 1989) and are not the subject of this document.

The use of bioassessments to investigate potential impairment, evaluate the severity of problems, ascertain the causes of the problems, and determine appropriate remedial action is a step-by-step process.

Inherent in the process for implementation of biocriteria is the assumption that bioassessment methods have been developed. However, the actual development of biocriteria is the most difficult step in the whole process. A conceptual model for biocriteria development was presented by the U.S. Environmental Protection Agency (1990) to streamline the major elements in the process. This model has been refined for presentation here (Fig. 1-1).

Each component of the model is numbered so that it can be identified and discussed more easily as an important part of the biocriteria development process. Nevertheless, these steps are not sequential. The following paragraphs describe the model process in more detail and identify areas of simultaneous development.

Components 1 through 8 describe the development of biocriteria, prior to their use in regulatory programs.

1. **Investigate the Biocriteria Program Concept.** The biocriteria process involves the selection of several program elements that contribute to effective biocriteria. Each state agency will have its own program objectives and agenda for establishing biocriteria; however, the underlying characteristics for effective biocriteria will be the same in all states.

2. **Formulate the Biocriteria Approach.** Defining biological integrity is the first step in the formulation of a biocriteria program. The activities important to this step are planning the biocriteria process; designating the reference condition; performing the biosurveys; and establishing the biocriteria.

3. **Select Reference Sites or Conditions.** The attainable biological status of an aquatic system is primarily described by the reference condition. If we understand the water resources's biological potential, we can judge the quality of communities at various sites relative to their potential quality. Natural environmental variation contributes to a range in expected conditions; deviations from this range help to distinguish perturbation effects.

   Historical datasets existing from previous studies are also an element of the derived biocriterion. These data range from handwritten field notes to published journal articles; however, biological surveys of present reference sites that are minimally impaired is key to the defined reference condition.

   The selection of reference sites is key to the success of biocriteria development. Various spatial scales can be used, but reference conditions must be representative of the resource at risk and must, therefore, be of the same or similar ecological realm or biogeographic region (i.e., an area characterized by a distinctive flora or fauna).

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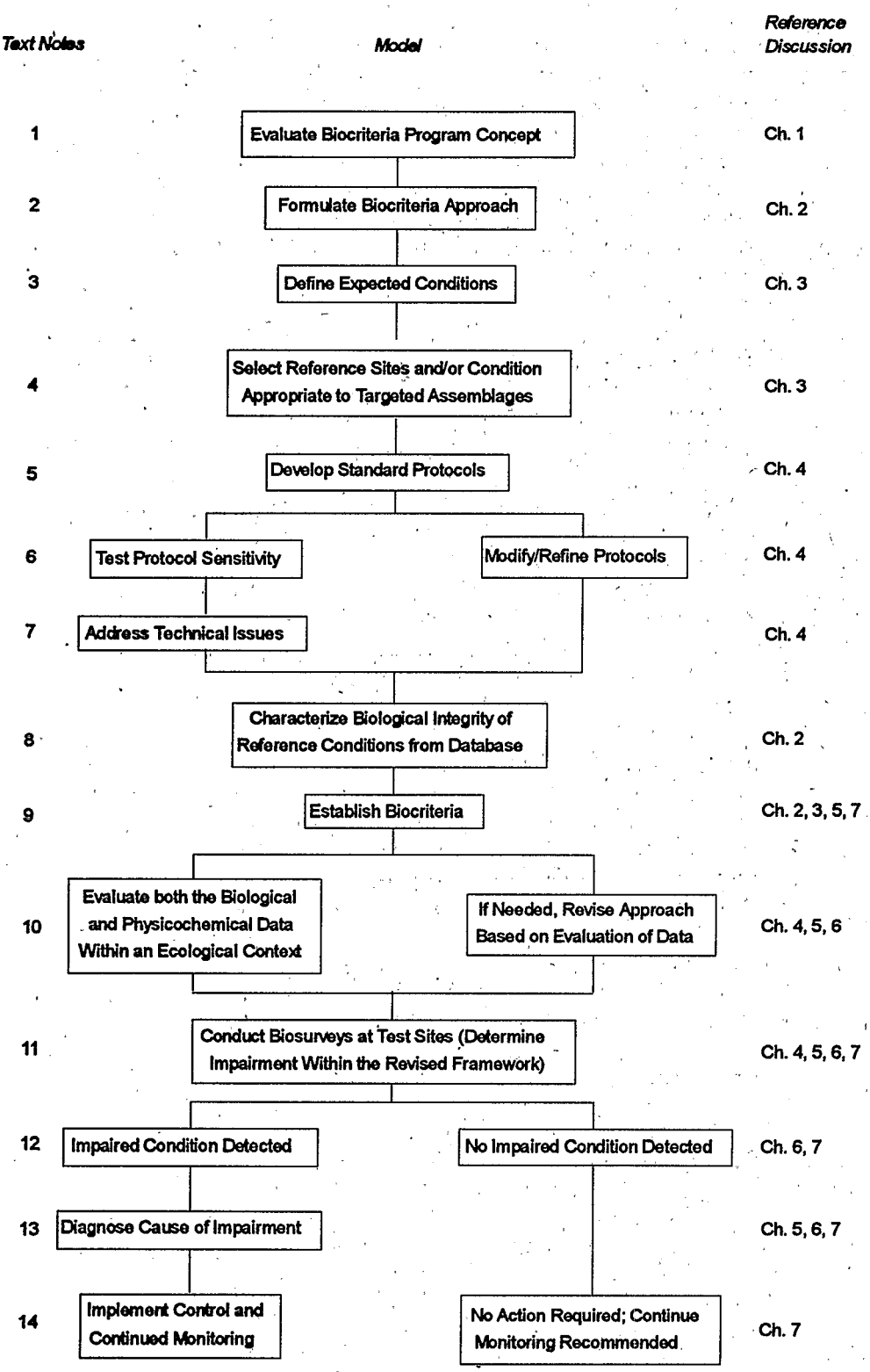

Figure 1-1.—Model for biocriteria development and application.

Candidate reference sites can be selected in a number of ways, but must meet some requirements established on the basis of overall habitat and minimally impaired status in a given region. The reference condition is best described by including data collected from several reference sites representing undisturbed watersheds. Such biological information can be combined for a more accurate assessment of the reference condition and its natural variability. The reference condition approximates the definition of biological integrity unless the reference sites were selected in significantly altered systems.

4. **Select Standard Protocols.** The development of standard protocols requires consensus building relative to the biological and ecological endpoints of interest. The primary goal is to develop measures to assess the biological integrity of aquatic communities in specified habitats, that is, to assess the integrity of the aquatic community as measured by the activities that maintain communities in equilibrium with the environment. There is no correct method to use or biological assemblage to sample; rather, a number of possibilities exist, including the Index of Biotic Integrity (IBI) for fish, and the Rapid Bioassessment Protocols (RBPs) for benthos.

   The process of applying these and other indices across widely differing systems is not a straightforward process and best professional judgment should be exercised before applying them to specific problems. For example, the IBI must be modified for northwestern assemblages since it was developed in the Midwest for midwestern assemblages. These indices measure a structural or functional attribute of the biological assemblage that changes in some predictable way with increased human influence. Combinations of these attributes or metrics provide valuable synthetic assessments of the status of water resources. As the basic theoretical framework and approach should remain consistent, the use of these indices should occur only after rigorous review and evaluation of their documentation. Such reviews are available in a variety of peer-reviewed publications.

The process of applying indices across widely differing systems is not a straightforward process and best professional judgment should be exercised before applying them to specific problems.

5. **Modification and Refinement of the Protocols.** The refinement process is an important step before large-scale biosurveys are conducted. The sensitivity of the protocols should be tested to determine whether differences in community health resulting from anthropogenic activities are discernible from changes caused by other impacts or natural variation. An impact is any change in the chemical, physical, or biological quality or condition of a waterbody caused by external sources. This process applies to all aspects of the protocol from sampling to data analysis and may be repeated as often as necessary.

6. **Address Technical Issues.** Certain technical issues — for example, natural seasonal variability, the aquatic assemblages selected for evaluation, the procedure for selecting sampling sites, and the type of sampling gear or equipment — affect the derivation of biocriteria.

7. **Characterize Biological Integrity.** Analyze biological databases to establish the range of values within the reference condition that will characterize biological integrity. Characterization depends on the use of biological surveys in concert with measurements of habitat structure.

8. **Establish Biocriteria and a Biological Monitoring Program.** Once biological integrity has been characterized and the geographic area regionalized, biological information can be equated to the water quality expectations of the state, and biocriteria can be established for these regions. Biocriteria may vary within a state depending on the region's ecological structure and the type of monitoring used in its water quality programs. Sources for the derived biocriteria are reference sites, historical records, in some instances empirical models of the systems (especially if significantly altered), and the consensus of a representative panel of regional experts evaluating this information.

Step 9 describes the validation of the biocriteria developed in the previous components.

9. **Evaluate and Revise as Needed.** Biocriteria are revised whenever better information is available, natural conditions have changed, and/or the waters of interest have improved. This process includes statistical analyses of biological, physical, and chemical data to establish natural variability and the validity of existing biocriteria. Regional frameworks should be adjusted if biological and geographical data support the need to do so. Reasons for these adjustments and the data used to determine them should be clearly documented.

Steps 10 through 14 describe the use of biocriteria for water resource management, that is, for the assessment, protection, remediation, and regulation of water quality.

10. **Conduct Biosurveys.** Biosurveys conducted at test sites help to determine whether and to what extent a site deviates from the normal range of values observed for the reference condition and from the regional biocriteria. Candidate test sites are any locations along the stream or river in which the conditions are not known but are suspected of being adversely affected by anthropogenic influence.

11. **Detect Impaired and Nonimpaired Conditions.** Decisions on whether adverse or impaired conditions exist must be made, but whether these conditions are socially tolerable may be beyond science. Scientists and resource managers are, however, obliged to determine the relative impairment of the water resource as a precondition for any subsequent decisions.

12. **Review Other Data Sources for Additional Information.** The use of additional data to complement the biological assessment is important in the decision-making process. As part of an integrated approach, whole effluent toxicity (WET) testing, chemical-specific analyses, and physical characteristic measurements can be used to make a comprehensive evaluation.

13. **Diagnose Causes of Impairment.** Once impairment has been determined, its probable causes must be identified before remedial action can be considered and implemented. Probable "causes" may include alteration of habitat structure, energy source, biological interactions, flow characteristics, or water quality. The "source" of the disturbance may be point or nonpoint source contamination or other human activities. Thus, if impairment is detected, the data should be evaluated to determine its probable causes; the site and surrounding area should be investigated for other probable causes; additional data should be collected; and either remedial action should be formulated (if the actual causes have been determined) or the investigation should be continued.

14. **Implement Remedial Actions and Continue Monitoring.** If probable causes have been identified so that an action plan can be developed, the last step is to begin remedial measures and continue monitoring to assess the stream's recovery. This step can be used to evaluate management programs and to determine cost-effective methods. The relative success of the measures depends on the selection of appropriate remedial actions to reduce or eliminate impairments and to attain the designated uses that the biocriteria protect.

    If no impairment is found, no action is necessary except continued monitoring at some interval to ensure that the condition does not change adversely.

Characteristics of Effective Biocriteria

Generally, effective biocriteria share several common characteristics. Well-written biocriteria

- provide for scientifically sound evaluations,
- protect the most sensitive biota and habitats,
- protect healthy, natural aquatic communities,
- support and strive for protection of chemical, physical, and biological integrity,
- include specific assemblage characteristics required for attainment of designated uses,
- are clearly written and easily understood,
- adhere to the philosophy and policy of nondegradation of water resource quality, and
- are defensible in a court of law.

In addition, well-written biocriteria are set at levels sensitive to anthropogenic impacts; they are not set so high that sites that have reached their full potential cannot be rated as attaining, or so low that unacceptably impaired sites receive passing scores. The establishment of formal biocriteria warrants careful consideration of planning, management, and regulatory goals and the best attainable condition at a site. Stringent crite-

ria that are unlikely to be achieved serve little purpose. Similarly, biocriteria that support a degraded biological condition defeat the intentions of biocriteria development and the Clean Water Act. Balanced biocriteria will incorporate multiple uses so that any conflicting uses are evaluated at the outset. The best balance is achieved by developing biocriteria that closely represent the natural biota, protect against further degradation, and stimulate restoration of degraded sites.

Additional general guidance regarding the writing of biocriteria is provided in U.S. Environ. Prot. Agency (1990). Several kinds of biocriteria are possible and vary among state programs. Both narrative and numeric biocriteria have been effectively implemented. Both should be supported by effective operational guidelines and adequate state resources, including people, materials, methods, historical data, and management support.

The best balance is achieved by developing biocriteria that closely represent the natural biota, protect against further degradation, and stimulate restoration of degraded sites.

Narrative biocriteria consist of statements such as "aquatic life as it should naturally occur" or "changes in species composition may occur, but structure and function of the aquatic community must be maintained." An aquatic community, the association of interacting assemblages in a given waterbody, is the biotic component of an ecosystem. Numeric values, such as measurements of community structure and function, can also serve as biocriteria. The numeric criterion should be a defined range rather than a single number to account for a measure's natural variability in a healthy environment. It may also combine several such values in an index. General examples of actual narrative and numeric biocriteria from selected state programs are presented in the following section; the information was taken from *Biological Criteria: State Development and Implementation Efforts* (U.S. Environ. Prot. Agency, 1991a).

Examples of Biocriteria

Five states have adopted definitive biocriteria for water quality management. Maine and North Carolina use narrative criteria; Ohio and Florida have implemented combined narrative and numeric criteria. Delaware has defined biocriteria for estuarine waters, and most other states have programs in various stages of development.

Narrative Biological Criteria

States may draft general narrative biological criteria early in their program — even before they have designated reference sites or refined their approach to biological surveys. This haste does not mean that having reference sites and a refined system for conducting surveys is unimportant; it means that a biocriteria program begins with writing into law a statement of intent to protect and manage the water resources predicated on an objective or benchmark, for example, "aquatic life shall be as naturally occurs."

States may draft general narrative biological criteria early in their program — even before they have designated reference sites or refined their approach to biological surveys.

When the objective to restore and protect the biological integrity of the water resources has been formally mandated, then the operational meaning of the statement and the identification of the agency responsible for developing the necessary procedures and regulations can be stipulated as the state's first steps toward the development of narrative and numeric biological criteria. The key point is that natural or minimally impaired water resource conditions become the criteria for judgment and manage-

ment. For more specific information on this concept and its implementation, see the EPA guidance document "Procedures for Initiating Narrative Biological Criteria" (Gibson, 1992).

Narrative biological criteria form the legal and programmatic basis for expanding biological surveys and assessments and for developing subsequent numeric biological criteria.

Maine and North Carolina are examples of the practical development and use of narrative biological criteria. Maine incorporated the general statement "as naturally occurs" into its biocriteria, but also developed supporting statements that specified collection methods to survey aquatic life. Maine uses narrative biocriteria defined by specific ecological attributes, such as measures of taxonomic equality, numeric equality, and the presence of specific pollution tolerant or intolerant species.

North Carolina uses narrative criteria to evaluate point and nonpoint source pollution and to identify and protect aquatic use classifications. In North Carolina, macroinvertebrate community attributes are used to help define use classifications. These attributes include taxonomic richness and the biotic indices of community functions and numbers of individuals. They are also used in conjunction with narrative criteria to determine "poor," "fair," "good–fair," "good," and "excellent" ratings for the designated uses.

Narrative biological criteria specify the use designations established by the state and describe the type of water resource condition that represents the fulfillment of each use. Conversely, when adopted by the state and approved by EPA, they become one of the standards by which water resource violations are determined.

Nevertheless, narrative biological criteria cannot be fully implemented without a quantitative database to support them. Quantitative data provide a responsible rationale for decision making and assure resource managers a degree of confidence in their determinations. In fact, some states have elected to develop narrative biocriteria and to use this legislative mandate to establish administrative authority for their quantitative implementation in a state natural resources agency. In this manner, future improvements in scientific methods and indicators can be accommodated through the administraive process rather than the more cumbersome and expensive method of amending state laws.

These data are similar to the data used to formulate numeric biological criteria; they can and should include the determination of reference conditions and sites. Thus, when the survey process for narrative biocriteria is well developed and refined, the program can easily begin the development of numeric biocriteria. While not an essential precursor, the narrative process is an excellent way for states to begin expanding their stream resource evaluation and management procedures to include more definitive numeric biocriteria.

Narrative biological criteria cannot be fully implemented without a quantitative database to support them.

Numeric biocriteria include discrete quantitative values that summarize the status of the biological community and describe the expected condition of this system for different designated water resource uses.

Numeric Biological Criteria

Although based on the same concept as narrative biocriteria, numeric biocriteria include discrete quantitative values that summarize the status of the biological community and describe the expected condition of this system for different designated water resource uses.

The key distinction between narrative biocriteria supported by a quantitative database and numeric biocriteria is the direct inclusion of a specific value or index in the numeric criteria. This index allows a level of specification to water resource evaluations and regulations not common to narrative criteria.

To develop numeric criteria, the resident biota are sampled at minimally impaired sites to establish reference conditions. Attributes of the biota, such as species richness, presence or absence of indicator taxa, and distribution of trophic groups, help establish the normal range of the biological community as it would exist in unimpaired systems.

Ohio combines narrative and numeric biocriteria and uses fish and invertebrates in its stream and river evaluation programs. Its numeric biocriteria are defined by fish community measurements, such as the Index of Well-Being (IWB) and the Index of Biotic Integrity (IBI). Ohio also employs an Invertebrate Community Index (ICI). All three measures provide discrete numeric values that can be used as biocriteria.

Ohio's numeric criteria for use designations in warmwater habitats are based on multiple measures of fish and benthic macroinvertebrates in different reference sites within the same ecoregion. Macroinvertebrates are animals without backbones that are large enough to by seen by the unaided eye and caught in a U.S. Standard No. 30 sieve. Criteria for this use designation are set at the 25th percentile of each biological index score recorded from the established reference sites within the ecoregion. Exceptional warmwater habitat criteria are set at the 75th percentile from the statewide set of reference sites (Ohio Environ. Prot. Agency, 1987). Use of the 25th and 75th percentiles, respectively, portrays the minimum biological community performance described by the narrative use designations. Such applications require an extensive database and multiple reference areas across the stream and river sizes represented within each ecoregion.

To develop the most broadly applicable numeric biological criteria, careful assessments of biota in multiple reference sites should be conducted (Hughes et al. 1986). The status of the biota in surface waters may be assessed in numerous ways. No single index or measure is universally recognized as free from bias. Evaluating the strengths and weaknesses of different assessment approaches is important, and a multimetric approach that incorporates information on species richness, trophic composition, abundance or biomass, and organism condition is recommended (see Chapter 6).

Other Biocriteria Reference Documents

EPA has developed several program and technical guidance documents for implementing biocriteria beginning with a preliminary discussion of biocriteria program development issues: legislative authority, steps in developing biocriteria, and the application of biocriteria to surface water management (U.S. Environ. Prot. Agency, 1990).

A survey of existing state programs was conducted in 1990 to delineate the status of bioassessment implementation on a national basis (U.S. Environ. Prot. Agency, 1991a). In addition, a reference guide to the technical literature pertaining to biocriteria has been developed (U.S. Environ.

Prot. Agency, 1991b). The latter contains cross-references to technical papers that develop the concepts, approaches, and procedures necessary to implement habitat assessment and biological surveys in the development and use of biocriteria. In December 1990, a symposium on biological criteria provided a forum for discussing technical issues and guidance for the various surface waterbody types. The proceedings from this conference are presented in U.S. Environ. Prot. Agency (1991d). Most recently, the agency has developed guidance to help states initiate narrative biological criteria (Gibson, 1992).

To develop numeric biocriteria, the resident biota are sampled at minimally impaired sites to establish reference conditions. Attributes of the biota such as species richness, presence or absence of indicator taxa, and distribution of trophic groups are useful for establishing the normal range of biological community components as they would exist in unimpaired systems.

Suggested Readings

Gibson, George. 1992. Procedures for Initiating Narrative Biological Criteria. EPA-822-B-92-002. U.S. Environ. Prot. Agency, Washington, DC.

U.S. Environmental Protection Agency. 1987a. Report of the National Workshop on In-stream Biological Monitoring and Criteria. In-stream Biol. Criteria Comm. Reg. 5, Environ. Res. Lab., Off. Water Reg. Stand., Corvallis, OR.

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———. 1991c. Technical Support Document for Water Quality-based Toxics Control. EPA 505/2-90-001. Off. Water, Washington, DC.

———. 1991d. Biological Criteria: Research and Regulation. EPA 440/5-91-005. Off. Water, Washington, DC.

CHAPTER 2.

Components of Biocriteria

Water resource legislation is usually designed to protect the resource and to ensure its availability to present and future generations. Over the past two decades, legislative and regulatory programs have established goals such as "fishable and swimmable, antidegradation, no net loss, and zero discharge of pollutants." However, actions to meet those goals do not always accomplish the mandate of the Clean Water Act, which is to restore and maintain biological integrity. The purpose of this chapter is to provide managers with a basic conceptual understanding of the relationship between biological integrity and biocriteria and to describe more fully the biocriteria process.

Purpose: To provide managers with a basic conceptual understanding of the relationship between biological integrity and biocriteria, and to describe more fully the biocriteria process.

Conceptual Framework and Theory

Biological integrity was first explicitly included in water resource legislation in the Water Pollution Control Act Amendments of 1972 (Pub. L. 92-500); and the concept, which was retained in subsequent revisions of that act, is now an integral component of water resource programs at state and federal levels (U.S. Environ. Prot. Agency, 1990).

The goal of biological integrity, unlike fishable and swimmable goals, encompasses all factors affecting the ecosystem. Karr and Dudley (1981; following Frey [1975]) define biological integrity as "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." That is, a site with high biological integrity will have had little or no influence from human society.

Edwards and Ryder (1990) recently used the phrase "harmonic community" in a similar context to describe the goal of restoring ecological health to the Laurentian Great Lakes. The sum of balanced, integrated, and adaptive chemical, physical, and biological data can be equated with ecological integrity (Karr and Dudley, 1981). Such healthy ecological systems are more likely to withstand disturbances imposed by natural environmental phenomena and the many disruptions induced by human society. These systems require minimal external support from management (Karr et al. 1986).

The adjective "pristine" is often invoked in such discussions; however, in the late 20th century, it is almost impossible to find an area that is completely untouched by human actions. Thus, the phrase "minimally impaired" is more appropriate than the word "pristine" for describing conditions expected at sites exhibiting high biological integrity.

Degradation of water resources comes from pollution, which is defined in the Clean Water Act of 1987 as "manmade or man-induced alteration of the chemical, physical, biological, or radiological integrity of water" (U.S. Gov. Print. Off. 1988). This comprehensive definition does not limit societal concern to chemical contamination. It includes any human action or result of human action that degrades water resources. Humans may degrade or pollute water resources by chemical contamination or by altering aquatic habitats; they may pollute by withdrawing water for irrigation, by overharvesting fish, or by introducing exotic species that alter the resident aquatic biota. The biota of streams, rivers, lakes, and estuaries, unlike other attributes of the water resource (e.g., water chemistry or flow characteristics), are sensitive to all forms of pollution. Thus, the development of biological criteria is essential to protect the integrity of water resources.

Components of Biological Integrity

While these definitions of integrity establish broad biological goals to supplement more narrowly defined chemical criteria, their use depends on the development of rigorous biological criteria. The challenge is to define biological integrity clearly, identify its components, and develop methods to evaluate a water resource and its surrounding environment based on these conditions.

It is important to distinguish between the attributes of natural systems that we intend to protect (assessment endpoints) and the attributes that we can measure (measurement endpoints). Success in protecting biological integrity depends on the development of measurement endpoints that are highly correlated with assessment endpoints.

Evaluating the elements or components of biological integrity will involve direct or indirect evaluations of biotic attributes. Indirect evaluations are appropriate if direct approaches are prohibitively expensive or in other ways difficult to implement. It is important to distinguish between assessment and measurement endpoints. Attributes of natural systems that we intend to protect, for example, the health of a fish population, are assessment endpoints; and attributes that we can measure, for example, age and size classes of the fish population, are measurement endpoints. Success in protecting biological integrity depends on the development of measurement endpoints that are highly correlated with assessment endpoints.

Important components of biotic integrity have been measured before. Toxicologists have long recognized the importance of individual health in evaluating the extent to which human actions have degraded a water resource, and ecologists have long used the kinds and relative abundances of species as indicators of condition. More recently, and in many ways less insightfully, theoretical measures of diversity have been used to assess species richness, that is, to determine if the number of species or relative abundances of species have been altered. Fish biologists, for example, use a variety of measures to assess the health of populations of targeted species, such as game fish. However, none of the attributes used in the past are comprehensive enough to cover all components of biological integrity.

In recent years, a broader conceptual foundation has been developed to convey the breadth of biotic integrity. The original Index of Biotic Integ-

rity (IBI) consisted of 12 metrics or attributes in three major groups: species richness and composition, trophic structure, fish abundance and condition. Another way of describing biotic integrity contrasts the elements of the biosphere with the processes but argues that both are essential to the protection of biological integrity (Table 2-1). The most obvious elements are the species of the biota, but additional critical elements include the gene pool among those species, the assemblages, and landscapes.

Table 2-1.—Components of biological integrity.

ELEMENTS	PROCESSES
Genetics	Mutation, recombination
Individual	Metabolism, growth, reproduction
Population/species	Age specific birth and death rates Evolution/speciation
Assemblage (community and ecosystem)	Interspecific interactions Energy flow
Landscape	Water cycle Nutrient cycles Population sources and sinks Migration and dispersal

Modified from Karr, 1990.

Processes (or functional relationships) span the hierarchy of biological organization from individuals (metabolism) to populations (reproduction, recruitment, dispersal, speciation) and communities or ecosystems (nutrient cycling, interspecific interactions, energy flow). For example, an important process in streams is an interaction of fish and mussels in which larval stages of the mussel (glochidia) attach to fish gills, presumably to enhance dispersal and to avoid predation.

Other approaches are available, but the important issue is not which classification is the best approach. Rather, efforts to assess biological integrity must be broadly based to cover as many components as possible.

Efforts to assess biological integrity must be broadly based to cover as many components as possible.

The challenge in implementing biocriteria is to develop reliable and cost-effective ways to exploit the insight available through biological analyses. It is not necessary to sample the entire biota. Rather, carefully selected representative taxa should be sampled. The selection should combine as many attributes as possible with precision and sampling efficiency, but all elements and processes are not necessarily covered in standard biological sampling.

Recent efforts to develop such integrative approaches include Karr's IBI later expanded to apply to a wide geographic area (Ohio Environ. Prot. Agency, 1987; Lyons, 1992; Oberdorff and Hughes, 1992), and to taxa other than fish, for example, benthic invertebrate assemblages (Ohio Environ. Prot. Agency, 1987; Plafkin et al. 1989). The Nebraska Department of Environmental Control (Bazata, 1991) has proposed indices that combine fish and invertebrate metrics, and the Ohio Environ. Prot. Agency (1987) has calculated several indices separately (fish and invertebrates) but uses them in combination to determine use attainment status.

Assessing Biological Integrity

A sound monitoring program designed to assess biological integrity should have several attributes. A firm conceptual foundation in ecological principles is essential to a multidimensional assessment that incorporates the several elements and processes of biotic integrity. The use of the concept of a reference condition, a condition against which a site is evaluated, is also important.

In addition, the general principles of sound project management or Total Quality Management (TQM), such as Quality Assurance and Quality Control, are as critical as the use of standard sampling protocols. *Quality assurance* (QA) includes quality control functions and involves a totally integrated program for ensuring the reliability of monitoring and measurement data; it is the process of reviewing and overseeing the planning, implementation, and completion of environmental data collection activities. Its goal is to assure that the data provided are of the quality needed and claimed.

The choice of attributes to be assessed and measured is critical to the success of any biological monitoring and criteria program.

Quality control (QC) refers to the routine application of procedures for obtaining prescribed standards of performance during the monitoring and measurements process; it focuses on the detailed technical activities needed to achieve data of the quality specified by the Data Quality Objectives (DQOs). Quality control is implemented at the laboratory or field level. Finally, biological monitoring must go beyond the collection and tabulation of high quality data to the creative analysis and synthesis of information about relevant biological attributes.

Numerous attributes of the biota have been used to assess the condition of water resources. Some are difficult and expensive to measure while others are not. Some provide reliable evaluations of biological conditions while others, perhaps because they are highly variable, are more difficult to interpret. Thus, the choice of attributes to be measured and assessed is critical to the success of any biological monitoring and criteria program.

The best approach to assessing biological integrity seems to be an integrative one that combines assessment of the extent to which either the elements or the processes of biological integrity have been altered; that is, efforts to protect biotic integrity should include evaluation of a broad diversity of biological attributes.

Historically, most biological evaluations were designed to detect a narrow range of factors degrading water resources. For example, the biotic index (Chutter, 1972; Hilsenhoff, 1987) is designed to detect the influence of oxygen demanding wastes ("organic pollution") or sedimentation, as is the Saprobic Index developed early in this century (Kolkwitz and Marsson, 1908).

With increased understanding of the complexity of biological systems and the complex influences of human society on those systems, more integrative approaches for assessing biological integrity have been developed. Some (Ulanowicz, 1990; Kay, 1990; Kay and Schneider, in press) advocate the use of thermodynamics, while others concentrate on richness or diversity (Wilhm and Dorris, 1968). The best approach seems to be an integrative assessment of the extent to which either the elements or the processes of biological integrity have been altered; that is, efforts to protect biotic integrity should include evaluation of a broad diversity of biological attributes.

Because the goal of biocriteria-bioassessment programs is to evaluate water resource systems stressed by or potentially destroyed by human action, the selection of the monitoring approach is critical. Indicators and monitoring design should be structured so that the same monitoring data

can serve a multitude of needs. This openness requires a reasonable level of sophistication for long-term status and trends monitoring. The more complicated the water resource problem, the larger the number of attributes that should be measured. Finally, programs to monitor the effects of human activity on the environment should have especially broad perspectives to ensure sensitivity to all forms of degradation.

Complex Nature of Anthropogenic Impacts

A number of human activities strain the integrity of water resource systems and the cumulative impacts of these actions create even greater complexity. Thus, it is useful, perhaps even necessary, to develop an organizational framework within which factors responsible for degradation in biotic integrity can be evaluated.

A major weakness of past approaches to protect water resources has been a narrow focus on the factors responsible for degradation. Specifically, past approaches focused on reducing the chemical contamination of the water on the assumption that clean water would produce high quality water resources. Overall, the determinants of the biological integrity of the water resource are complex, and the simplistic approach of making water cleaner, though important, is inadequate.

Biological monitoring and the use of biocriteria to assess biotic integrity provides a more comprehensive evaluation of the status of the resource. Such evaluations enhance our ability to identify the factors responsible for degradation and to treat the problem in the most cost-effective manner. Monitoring specific and ambient (background) conditions offers unique opportunities to detect, analyze, and plan the treatment of degraded resources.

Because human actions may impact a wider range of water resource attributes than water chemistry alone, a broader framework is necessary to identify and reverse the specific factors responsible for the degradation of biotic integrity. Degradation may begin in an area of the watershed or catchment that is external to the reference or test site simply because it is often the result of human actions that alter the vegetative cover of the land surface. These changes combined with the alteration of stream corridors degrade the quality of water delivered to the stream channels and attack the structure and dynamics of those channels and their adjacent riparian environments.

Human activities at the site affect five primary classes of variables — all of which may result in further degradation of water resources (Karr, 1991). These five internal variables should be placed in a larger context as illustrated in Figure 2-1:

1. **Water Quality:** Temperature, turbidity, dissolved oxygen, acidity, alkalinity, organic and inorganic chemicals, heavy metals, toxic substances.

2. **Habitat Structure:** Substrate type, water depth and current velocity, spatial and temporal complexity of physical habitat.

3. **Flow Regime:** Water volume, temporal distribution of flows.

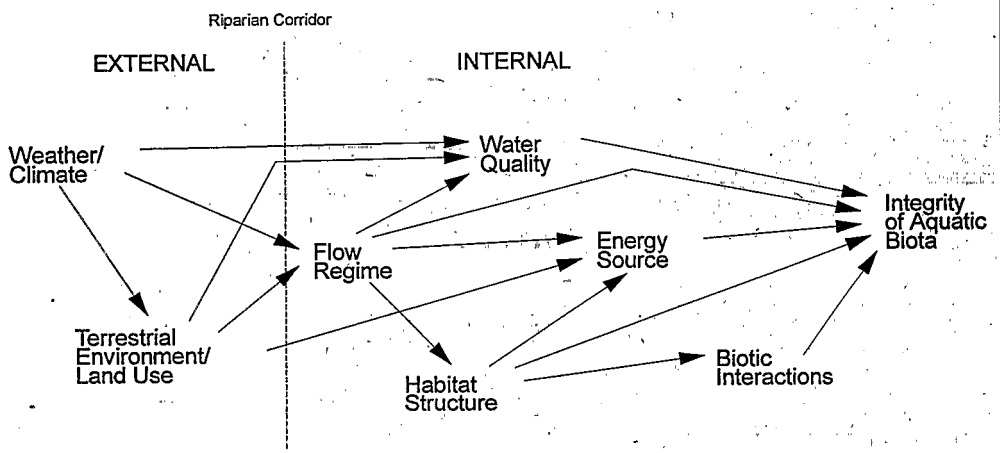

Figure 2-1.—Conceptual model showing the interrelationships of the primary variables relative to the integrity of aquatic biota. External refers to features outside the stream system; internal to in-stream features (Karr, 1991). Terrestrial environment includes factors such as geology, topography, soil, and vegetation.

4. **Energy Source:** Type, amount, and particle size of organic material entering stream, seasonal pattern of energy availability.

5. **Biotic Interactions:** Competition, predation, disease, parasitism, and mutualism.

From this conceptual framework, at least four components of the biota should be evaluated: structure, composition, individual conditions, and biological processes (Fig. 2-2). Sample attributes for each component include the following:

- **Community Structure:** Species richness, relative abundances, including the extent to which one or a few species dominates.
- **Taxonomic Composition:** Identity of the species that make up the biota.
- **Individual Condition:** Health status of individuals in selected species.
- **Biological Processes:** Rates of biological activities across the biological hierarchy (from genes to landscapes).

Comprehensive assessments of these attributes ensure that all the components of biotic integrity are protected. For each component, one or more attributes should be assessed.

Successful metrics represent the expression of the influence of human activities on the resident biota. For example, the presence of a few hardy species of fish in abundance may be a response to sewage in the waters. As human disturbance increases, total species richness, the number of intolerant species, and the number of trophic specialists usually decline, while the number of trophic generalists increases. *Generalists* are organisms that can use a broad range of habitat or food types. Exceptions exist: for example, when coldwater streams are warmed, species richness increases, although this process must be viewed as a degradation of the biotic integrity of a coldwater system.

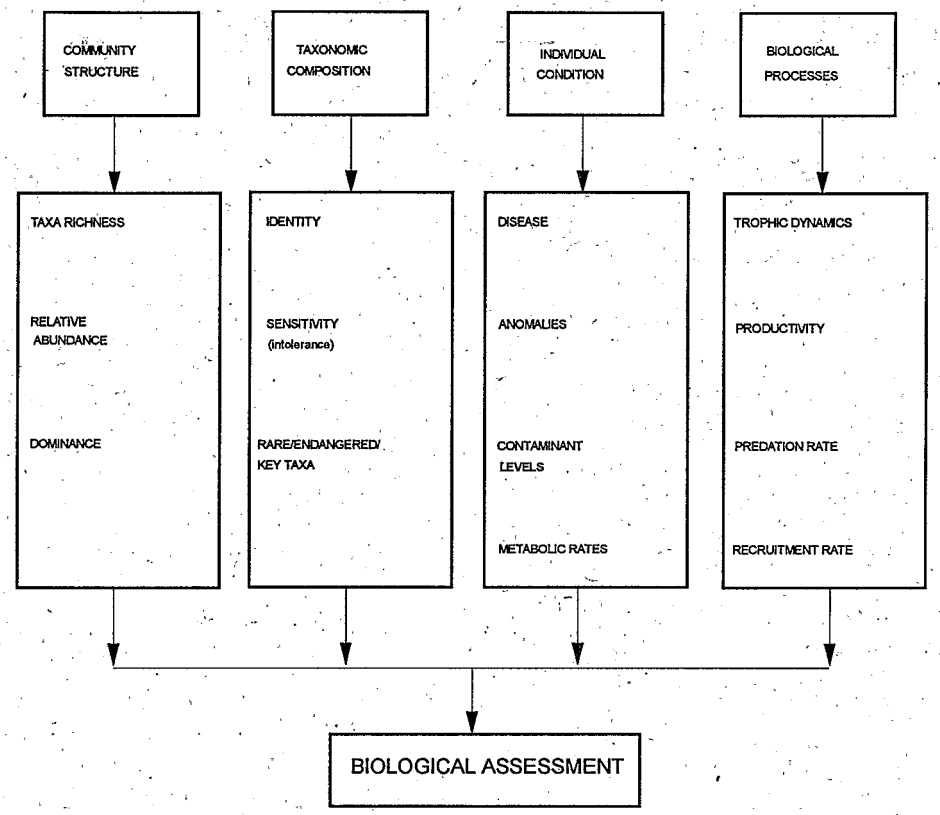

Figure 2-2.—Organizational structure of the attributes that should be incorporated into biological assessments.

Use of biocriteria to evaluate and protect biotic integrity focuses directly on the condition of the resource. The development of biological monitoring is driven by the need for rigorous standardized evaluations of point and nonpoint source pollution and other circumstances in which up- and downstream evaluations may be inappropriate. In short, development of biocriteria is driven by the need for a comprehensive approach to the study and remediation of human effects on water quality.

The Biocriteria Development Process

Biocriteria must be developed with a clear understanding of several important concepts. Foremost is the basic premise underlying biocriteria development: understanding the condition of the biota in a given waterbody provides a baseline for an integrative and sensitive measure of water quality. Biocriteria are operational narrative or numeric expressions that characterize and, if properly used, protect biological integrity.

Understanding the condition of the biota in a given waterbody provides a baseline for an integrative and sensitive measure of water quality.

Biocriteria can be used to protect biological integrity and to establish an aquatic life use classification. Following the definition of biocriteria, field surveys are conducted to determine whether particular sites meet the biocriteria or whether they have been affected by human activity. This determination is made by comparing the aquatic biota at potentially disturbed sites with minimally impaired reference conditions. Natural events

not initiated by or exacerbated by human actions (e.g., fire, beavers) are not considered disturbances in this sense.

The basic premise, that biota provide a sensitive screening tool for measuring the condition of a water resource, depends on the assumption that the greater the anthropogenic impact in a watershed, the greater the impairment of the water resource. A corollary is that streams and rivers not subject to anthropogenic impact contain natural communities of aquatic organisms that reflect unimpaired conditions. These assumptions provide the scientific basis for formulating hypotheses about impairments — departures from the natural condition result from human disturbances.

The basic premise, that biota provide a sensitive screening tool for measuring the condition of a water resource, depends on the assumption that the greater the anthropogenic impact in a watershed, the greater the impairment of the water resource.

Natural disturbances, such as floods or drought, may also affect the aquatic biota as part of normal ecological processes, and these responses vary among ecoregions and stream sizes. For example, relatively stable structure is characteristic of fish communities in the eastern United States but stable fish communities in the Great Plains streams may reflect human disturbance (Bramblett and Fausch, 1991). Molles and Dahm (1991) provide additional cautions on the need to consider natural events in interpreting data from biological systems. Thus, natural disturbances must be considered, but they are not considered as impairments because they are not the result of human activity.

Ideally, biocriteria are reflective of the natural biological integrity of the particular region under study, that is, of the region as it would be had it not become impaired. Depending on the refinement of the biosurvey method, the degree of impairment can often be established as part of the biocriteria development process. Once defined, biocriteria for a stream or river will describe the best attainable condition. The best attainable conditions represent expected conditions and are directly compared to the observed conditions. Each state needs to formulate appropriate definitive descriptors (i.e., biocriteria) for the aquatic organisms in its streams, and these descriptors or biocriteria should support the state's designated use classifications or other resource protection and management objectives.

Once defined, biocriteria for a stream or river will describe the best attainable condition.

Successful implementation of biocriteria requires a systematic program to collect and evaluate complex scientific information and translate that information into an effective planning tool to protect water resources. This effort must be systematic as well as conceptually and scientifically rigorous; it must also be logical and easily understood. The components of a program to implement biocriteria may be divided in a variety of ways.

The four primary steps to develop and implement biocriteria are introduced here and will be discussed in greater detail in later sections of this document. The four steps are

1. planning the biocriteria development process,
2. designating the reference condition for biosurvey sites,
3. performing the biosurveys to characterize reference condition, and
4. establishing biocriteria based on reference biosurvey results.

Each step must be considered in the context of regulatory policy, the scientific method, and the practical aspects of fieldwork involving biosurveys. Further, acceptable biocriteria for streams and rivers can be devel-

oped in various ways. Therefore, biocriteria development should be based on a set of flexible procedures derived from management, the regulatory process, or both. When properly implemented, the procedures lead to self-defined biocriteria that will protect the unique characteristics of streams and rivers. When not properly implemented, water resources continue to be degraded. Although the general concepts and procedures of biocriteria development can be adapted to any stream or river, the development of useful biocriteria requires individual planning for different waterbodies.

The development of useful biocriteria requires individual planning for different waterbodies.

■ **Planning Biocriteria.** Planning includes the classification of surface water types and the definition of designated uses; however, the planning process necessarily extends beyond stream and river use classification. To be effective, planning must ensure that program objectives are clearly defined and that the scientific information generated to meet program objectives is appropriate for making environmental management decisions.

The planning phase assumes the interaction of environmental managers (staff involved in policy, budgeting, and resource management) and technical staff (those involved in data collection and interpretation) to ensure that the environmental data to be collected are acceptable and meet the state's needs. To facilitate interaction, a formal quality assurance and quality control plan that includes the formulation of data quality objectives should be included in the biocriteria development process. Complete data quality objectives describe the decisions to be made, the data required and why, the calculations in which the data will be used, and time and resource constraints. They are used to design data collection plans and to specify levels of uncertainty. Levels of uncertainty pertain to the confidence that decision makers can realistically have that collected data will actually support particular conclusions.

Planning must ensure that program objectives are clearly defined and that the scientific information generated to meet program objectives is appropriate for making environmental management decisions.

Finally, interagency cooperation (within and among states) should be a critical component of the planning process. Time spent on developing good relations with other groups improves biocriteria and their use.

Interagency cooperation should be a critical component of the planning process.

■ **Designating Reference Condition.** Designating the reference condition for biosurvey sites is the second major activity in biocriteria development. This continuation of the planning process shifts attention to the specific data needed to define the biotic conditions that would be expected to occur in the study stream in the absence of human impact. Issues requiring consideration at this stage of the process include

- the database to be formed and evaluated (e.g., the taxonomic assemblages or other biological attributes to be used to describe biological condition);
- the habitat types to be included in the survey (e.g., runs, riffles, pools, and snags);
- the type of reference conditions needed for the program or study being formulated (e.g., regional, ecoregional, or site-specific);
- the geographical scale to which the biocriteria are applicable (e.g., specific river reach, watershed, ecoregion, or other parameters);
- the temporal scale for which biocriteria are being considered (e.g., seasonal, annual, or multiyear);

- how habitat will be assessed to ensure comparability between the reference condition and the habitat at the biosurvey site before human impacts;
- parameters and methods of measurement; and
- how data from the biosurvey are to be evaluated.

Data management, analysis, and reporting requirements should also be determined before any fieldwork is begun. Specific information dealing with the designation of reference condition and biosurvey sites is provided in Chapter 3.

Because knowledge of biological communities and habitats surrounding the surface waters of the study region is essential to effective biological monitoring, definition of the reference condition is a critical step in the process. Careful designation of the reference condition can reduce the likelihood of problems and minimize the costs associated with fieldwork.

Definition of the reference condition is a critical step in the process.

Knowledge of the reference condition may derive from historical data or from pilot studies of local or regional sites that are relatively undisturbed. Macroinvertebrate and fish assemblage data have often been routinely collected by state fish and wildlife agencies, water quality agencies, universities, and others responsible for stream management. Although these historical databases are often overlooked in environmental evaluations, they can be valuable sources of information. An estimation of biological integrity at a minimally impaired site may be accomplished by reviewing existing data and publications for specific streams and rivers. Fausch et al. (1984) developed fish species richness expectations for several midwestern streams based on historical data sets. Obviously, the usefulness of historical data for establishing reference condition is dependent on the original objective of the data collection effort, the collection methods, and the quality of the data. Even if historical data are inadequate for direct use in designating the reference condition, they may provide substantial insight about preexisting conditions at the test or study sites.

■ **Performing Biosurveys.** Performance of the actual biosurvey to characterize the reference condition entails several activities. Often, a presurvey (pilot study) is necessary to finalize the study plan and the actual logistics of the fieldwork. Upon completion of the study plan, technical staff must be fully briefed regarding the study's objectives, quality assurance and quality control operations, and methods of data collection and summarization. At this point, the actual biosurvey may be performed. Biosurveys may include routine local monitoring, sampling over wide geographic areas, or special case evaluations at one or a few sites.

■ **Establishing Biocriteria.** After the biosurveys have been completed or the historical data evaluated, the biological status of the reference condition is used to help define the biocriteria. Based on the results of the surveys, some refinement of aquatic life use designations may be needed for particular streams or rivers. After writing the biocriteria, they must undergo final review and approval by each state and the EPA.

Certain attributes should be considered when drafting formal biocriteria. Ideally, biocriteria should be readily understandable and scientifically

and legally defensible. Further, they should be protective of the most sensitive element of the biota included in the designated aquatic life use of the stream or river and yet express an attainable condition.

Thus, biocriteria should be used in decision making, not only for routine management procedures but also for guiding resource policy determinations. For those decisions to be robust, quality assurance programs must ensure long-term database management, including data entry, manipulation, and analysis.

Biocriteria provide an initial determination of impairment or attainment. Their use may also help to determine sources and causes of degradation when combined with survey information and knowledge of how organisms react to different stresses (e.g., sight-feeding fish decline when turbidity increases; tolerant species increase with nutrient enrichment; anomalies of 40 to 60 percent occur only in the presence of complex toxic effluents and impacts). These response signatures are vital to the successful use of biocriteria to attain water resource protection.

The endpoint of water resource protection using biocriteria is broader than clean water. The endpoint of biocriteria and water resource legislation is "to restore and maintain the physical, chemical, and biological integrity of the nation's waters."

Biocriteria should be readily understandable and scientifically and legally defensible. Further, they should be protective of the most sensitive designated aquatic life use of streams and rivers and yet express an attainable condition.

The endpoint of biocriteria and water resource legislation is "to restore and maintain the physical, chemical, and biological integrity of the nation's waters."

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CHAPTER 3.

The Reference Condition

The term *biocriteria* implies the notion of comparison to the highest attainable condition. The reference condition establishes the basis for making comparisons and for detecting use impairment; it should be applicable to an individual waterbody, such as a stream segment, but also to similar waterbodies on a regional scale. The reference condition is a critical element in the development of a biocriteria program.

Purpose:
To provide guidance for defining biological expectations based on a reference condition, and for making comparisons to test sites.

The reference condition establishes the basis for making comparisons and for detecting use impairment; it should be applicable to an individual waterbody, such as a stream segment, but also to similar waterbodies on a regional scale.

Establishing the Reference Condition

Recognizing that absolutely pristine habitats do not exist (even the most remote lakes and streams are subject to atmospheric deposition), resource managers must agree to accept sites at which minimal impacts exist or have been achieved as the reference condition for a given region. Acceptable reference conditions will differ among geographic regions and states because soil conditions, stream morphology, vegetation, and dominant land use differ between regions. In heavily agricultural, industrial-commercial, or urbanized regions, undisturbed streams or reaches may not exist, and reference conditions may need to be determined based on that which is likely attainable, the historical record, or other methods of estimation.

Reference conditions can be established using a combination of methods — reference sites, historical data, simulation models, and expert consensus.

■ **Historical Data.** In some cases, data are available that describe past biological conditions in the region. Careful scrutiny and evaluation of these data can be an important initial phase in the biocriteria development process because they provide insight about the communities that have been or can be achieved in various waterbody types. These records are usually available in natural history museums, university collections, and some agencies, such as state water resource and fish and wildlife departments; however, some historical biological surveys were conducted at impaired sites, used different sampling methods, were insufficiently documented, or had objectives markedly different from biocriteria determination. Such data would be of questionable value for establishing precise reference conditions and should be used advisedly.

■ **Reference Sites.** Reference sites refer to locations in similar waterbodies and habitat types at which data can be collected for comparison with test sites. Typical reference sites include sites that are upstream of point

Reference conditions can be established using a combination of methods — reference sites, historical data, simulation models, and expert consensus.

sources; sites in nearby watersheds; sites that occur along gradients of impact (near field/far field); and regional reference sites that may be applied to a variety of test sites in a given area. Sites upstream of point sources may or may not exhibit the quality of the overall reference condition. However, their proximity to the site in question makes them a useful qualifier for regional references, specifically in controversial situations.

Achieving biological conditions may be described through a statistical evaluation that integrates biological attributes from a group of sites that have the same characteristics and expectations. This approach can be used to establish biological criteria for aquatic life uses and to test the probability that a particular test site has a biological community comparable to that established group (Maine Dep. Environ. Prot. Agency, 1993).

■ **Simulation Models.** Simulation models include mathematical models (logical constructs following from first principles and assumptions), statistical models (built from observed relationships between variables), or a combination of the two. The complexity of mathematical models that can predict reference conditions is potentially unlimited, but as complexity increases, the costs will be higher and some of the model's predictive ability will be lost (Peters, 1991). Thus, models that predict biological reference conditions should only be used as a last resort and with great caution because they may involve complex and untestable hypotheses (Peters, 1991; Oreskes et al. 1994). Nevertheless, several models that predict water quality in rivers and reservoirs from first principles of physics and chemistry have been quite successful (e.g., Kennedy and Walker, 1990). Mathematical models to predict biological conditions have been less successful and, so far, not very useful in an assessment or management context.

Statistical models can be fairly simple in formulation, such as the Vollenweider model and the morphoedaptic index to predict trophic status (Vollenweider, 1975; Vighi and Chiaudani, 1985). These models require a sufficiently large database to develop predictive relationships and, in their current state of development, predict only nutrient conditions, not the structure of biological communities

Hybrid models use both first principles and statistical relationships between variables. Hybrids are typically large simulation models intended to predict the behavior of a stream over time; they are commonly used to predict water quality for management (Kennedy and Walker, 1990). Most existing models predict water quality variables such as chlorophyll *a*, nutrient concentrations, Secchi depth, and oxygen demand. Inferring the composition of biological assemblages from predicted water quality would require another model relating assemblages to stream water quality.

Model development for biological criteria is still rudimentary. However, as state databases expand, this tool will become more important and will likely assume a growing role in establishing reference conditions.

■ **Expert Opinion/Consensus.** When no candidate reference sites are acceptable, and models are deemed unreliable, then expert consensus is a necessary alternative to establish reference expectations. Under such circumstances, the reference condition may be defined using expert opinion based on sound ecological principles applicable to the region of interest. Several skilled biologists and natural resource managers should be convened for the assessment. Each of these experts should be familiar with

the streams and aquatic biota of the region as they will be asked to develop a description of the assemblage in relatively unimpacted streams based on their collective experience. The description developed by consensus may therefore be more qualitative than quantitative. Even when reference sites are available and models may also be useful, this panel of specialists should be convened to evaluate all the data and help develop the biocriteria.

In sum, investigators will incorporate any or all of these usually interdependent techniques in the effort to establish reference conditions. That is, historical data, reference sites, simulation models, and expert opinion/consensus can and should be used mutually to support reference condition decisions; however, the use of actual reference sites to establish reference conditions is always important. Such sites represent achievable goals, and they can be regularly monitored. Historical data and expert opinion should also be used to make decisions regarding the selection of these reference sites. Such a panel of experts can be reconvened to help establish the subsequent, and related, biological criteria. Simulation models that incorporate historical data combined with expert opinion are the primary alternative to reference sites and may be most useful in the assessment of significantly altered sites or waterbodies unique to the region under study.

The most appropriate approach to establishing reference conditions is to conduct a preliminary resource assessment to determine the feasibility of using reference sites (Fig. 3-1). If reference sites are not acceptable, then even greater reliance must be placed on the other elements, and some form of simulation modeling may be the next best alternative. This situation would occur if no "natural" sites exist and if "minimally impaired sites" are unacceptable. Biological attributes can be modeled from neighboring regional site classes, expert consensus, and/or a composite of "best" ecological (historical) data. Such models may be the only viable means of examining significantly altered systems. The expectations derived from these models may be regarded as hypothetical or temporary until more realistic attainment goals can be developed.

Thus, the use of reference sites remains the best data source to estimate present-day attainment conditions and is the basis for the emphasis on reference sites that follows. The selection of minimally disturbed sites from a site class provides the most realistic basis for expecting that biological integrity can be attained. In this situation, the central tendency of the biological measure is a conservative estimate of the expected biological condition. Some states, for example, Ohio and Florida, use a lower percentile (25th percentile) as their threshold for attainment. When relatively few sites are unimpaired and the sites are more than minimally disturbed, an upper percentile from the range of biological values from all sites may have to be used instead. An interim expected biological condition can be developed from this approach that can be revisited after restoration efforts have been initiated and evaluated by the specialists.

The most appropriate approach to establishing reference conditions is to conduct a preliminary resource assessment to determine the feasibility of using reference sites.

The determination of the reference condition from reference sites is based on the premise that streams minimally affected by human activity will exhibit biological conditions most natural and attainable for streams in the region.

The Use of Reference Sites

The determination of the reference condition primarily from reference sites is based on the premise that streams minimally affected by human activity will exhibit biological conditions most natural and attainable for

Two primary considerations guide the selection of reference sites: minimal impairment and representativeness.

Sites that are undisturbed by human activities are ideal reference sites. However, land use practices and atmospheric pollution have so altered the landscape and quality of water resources nationally that truly undisturbed sites are rarely available.

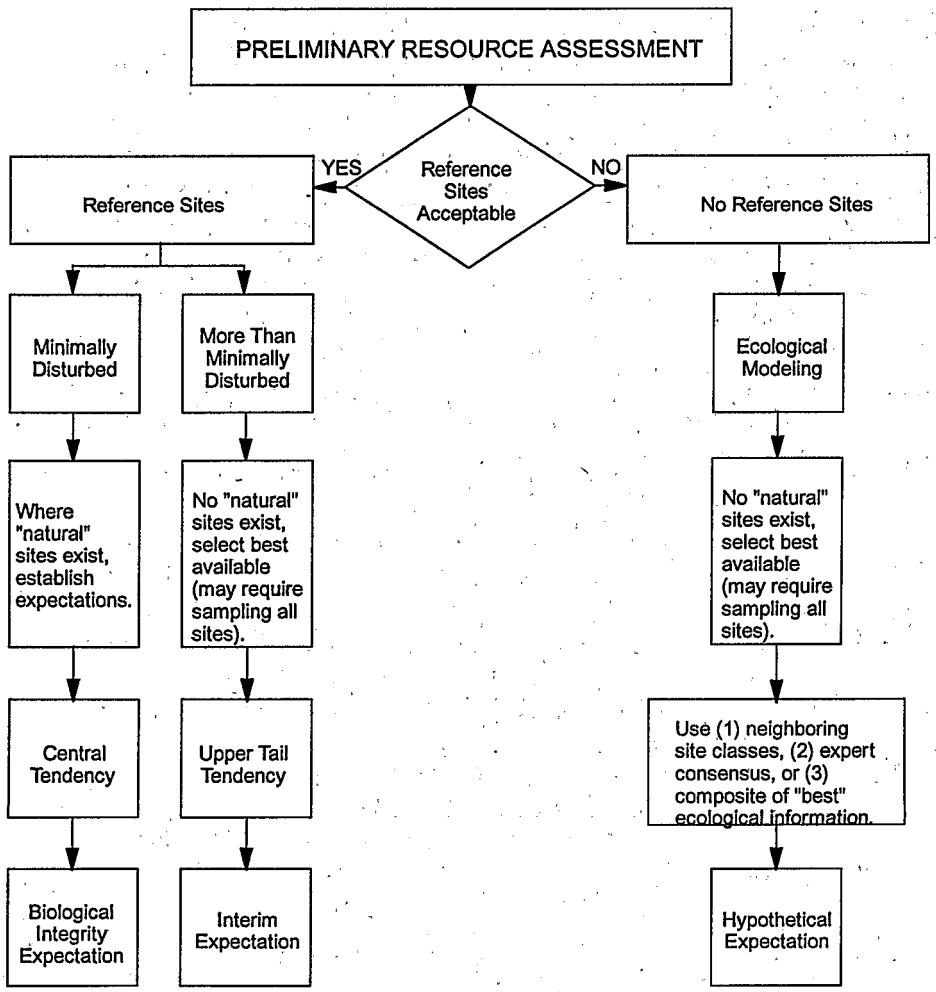


Figure 3-1.—Approach to establishing reference conditions.

streams in the region. Anthropogenic effects include all possible human influences, for example, watershed disturbances, habitat alteration, non-point source runoff, point source discharges, atmospheric deposition, and angling pressure. The premise does not consider any human activities as improvements; for example, planting non-native riparian vegetation or stocking with artificially high abundances of game or non-native fish are not improvements relative to biological integrity. In practice, most reference sites will have some of these impacts; however, the selection of reference sites is made from those with the least anthropogenic influences.

Reference sites must be carefully selected because they will be used as sources for the biocriteria benchmarks against which test sites will be compared. The conditions at reference sites should represent the best range of conditions that can be achieved by similar streams within a particular ecological region. The key to making such biocriteria benchmarks protective is to organize sites into classes so that the minimum acceptable performance is commensurate with the capability of the resource. Therefore, two primary considerations guide the selection of reference sites within each class: minimal impairment and representativeness.

■ **Minimal Impairment.** Sites that are undisturbed by human activities are ideal reference sites. However, land use practices and atmospheric pollution have so altered the landscape and quality of water resources nationally, that truly undisturbed sites are rarely available. In fact, it can be argued that no unimpaired sites exist. Therefore, a criterion of "minimally impaired" must be used to determine the selection of reference sites. In regions where even such minimally impaired sites are significantly degraded, the search for suitable sites should be extended over a wider area, and multistate cooperation may be essential. The purpose of selecting minimally impaired sites to represent reference conditions is primarily goal-setting. Once attainment of these conditions is achieved on a large scale, a higher criterion is possible. In no instance should any notably degraded condition be accepted as the reference for criteria development.

A criterion of "minimally impaired" must be used to determine the selection of reference sites.

■ **Representativeness.** Reference sites must be representative of the waterbodies under investigation; that is, they must exhibit conditions similar to those of other sites in the same region. Sites that contain locally unusual environmental factors will result in uncharacteristic biological conditions and should be avoided.

The overall goal in the establishment of the reference condition from carefully selected reference sites is to describe the biota that investigators can expect to find at sites of interest. These "test or assessment sites" will be compared to the reference sites to determine whether impairment exists. The characteristics of appropriate reference sites vary among regions of the country and for different waterbody and habitat types. In general, the following characteristics (modified from Hughes et al. 1986) are typical of ideal reference sites:

The overall goal in the establishment of the reference condition from carefully selected reference sites is to describe the biota that investigators may expect to find at sites of interest.

- 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 — in some cases, the flow patterns display large seasonal differences in response to rainfall and snowmelt; in other cases, stable discharges are typical of water that originates from underground sources. Biota evolve in the face of natural discharge patterns.
- Banks representative of undisturbed streams in the region (generally covered by riparian vegetation with little evidence of bank erosion, or undercut banks stabilized by root wads). Banks should provide cover for aquatic biota.
- Natural color and odor — in some regions, clear, cold water is typical of the waterbody types in the region; 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.

A single minimally impaired site cannot be truly representative of an entire region or population of sites, and a frequent difficulty is matching upstream

and downstream habitats for valid comparison. For example, if habitat is degraded upstream but not downstream, the effects of a discharge may be masked. Reference conditions based on multiple sites are more representative and form a valid basis for establishing quantitative biocriteria.

In developing and adjusting the biocriteria, managers must strike a balance between the ideal restoration of the water resource and the fact that human activity affects the environment.

One problem in the use of minimally impaired sites as references is what to do if an area is extensively degraded so that even these sites indicate significant deterioration. Many systems are altered through channelization, urbanization, construction of dams and highways, or management for certain sport fisheries or reservoirs (Karr and Dionne, 1991). The condition of these systems is a result of societal decisions that have to be taken into account in the development of biocriteria, but 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.

Although the biocriteria established for altered systems serve as a baseline for judging further degradation, their ultimate goal is to achieve the sites' recovery to the best attainable condition — as represented by conditions at "minimally impaired" sites. Consensus of expert opinion and historical data play an important role in characterizing the reference condition for these systems, as does the application of innovative management practices to obtain improvement.

In developing and adjusting the biocriteria, managers must strike a balance between the ideal restoration of the water resource and the fact that human activity affects the environment. The most appropriate course of action will use minimally impaired sites as the maximum amount of degradation that will be tolerated, thereby ensuring adherence to the antidegradation policy of the Clean Water Act. Continual monitoring should provide the feedback necessary to make reference site and criteria adjustments as warranted during the restoration process.

Characterizing Reference Conditions

Characterization of regional reference conditions for biocriteria development consists of the following steps:

1. Classification of the resource. All streams are not alike; therefore, reference conditions (expectations) will differ among geographic regions and stream types.
2. Selection of the best available sites in each resource class as candidate references.
3. Characterization — including confirmation and refinement of the reference conditions — based on a biological survey of reference sites.

Classification

The purpose of classification is to group similar things together, that is, to prevent the comparison of apples and oranges. Meaningful classification is not arbitrary (an apple is not an orange); professional judgment is usually necessary to arrive at a workable system that recognizes different conditions,

without considering each waterbody or watershed a special case. By classifying, we reduce the complexity of biological information. Classification improves the resolution or sensitivity of biological surveys to detect impairment by partitioning or accounting for variation among sites.

There are two fundamental approaches to classification: a priori and a posteriori (Conquest et al. 1994). A priori classification is a system based on preconceived information and theories, for example, using physiographic provinces to classify streams. The a posteriori approach bases the classification solely on the data collected and finds classes (e.g., using cluster analysis) within these data.

In operational assessment and management of streams, an assessment site is assigned to a class (e.g., mountain headwater streams) before it is actually surveyed and biological data are collected. Ideally, sites should be assigned to a class from mapped information before any sampling is done. Therefore, an a priori classification based on maps or other easily obtainable secondary information is often developed for characterizing reference conditions. The biosurvey data are subsequently used to test that classification.

Stream characteristics that are readily affected by human activities or occur as a biological response to physical or chemical conditions should not be used as classification variables. Such responses may include land use, habitat condition, or nutrient concentrations. For example, in the southern Rockies ecoregion, riparian zones are heavily forested; and in the neighboring Arizona/New Mexico Plateau ecoregion, riparian zones are relatively unvegetated. The classification variable in this case is ecoregion, and riparian vegetation is a response to ecoregion. If dense riparian vegetation were used as a classification variable, we would run the risk of misclassifying an unimpaired, unvegetated stream in the Arizona/New Mexico Plateau as impaired by comparison to natural streams in the southern Rockies. This example shows that the best classification variables are those that are readily obtained from maps or regional water characteristics such as ecoregion, gradient, alkalinity, and hardness.

The intent of this protocol is not to develop a classification scheme applicable to the entire United States. Classification must be regional in scope and use regional expertise to determine which variables to use in a given region.

Framework for Preliminary Classification

The intent of this protocol is not to develop a classification scheme applicable to the entire United States. Classification must be regional in scope and use regional expertise to determine which variables to use in a given region. Further, classification should be parsimonious to avoid proliferation of classes that do not contribute to assessment.

Ecoregions

Biologists have long noted that assemblages and communities can be classified according to distinct geographical patterns (e.g., Wallace, 1869; MacArthur, 1972). We observe areas of the country within which types of resources and their attributes are ecologically consistent and similar when compared to those of other areas. The recognition of such patterns occurs at various levels: global, continental, regional, and local.

Regionalization identifies these natural spatial patterns. It accounts for spatial variation by partitioning the landscape into smaller areas of greater homogeneity. Ecological regionalization (as one type of regionalization) results in a map of ecological regions, or ecoregions. Such maps bring spatial organization to ecological variability. They are useful in a variety of ways.

The basic goal of regionalization is to depict areas of ecological homogeneity relative to other areas.

for example, to summarize the condition of resources in a particular area, to identify potential or achievable ecological conditions (e.g., regionally achievable biocriteria), to characterize typical impact types and impairments, to develop protective and remedial procedures that are tailored to unique regional characteristics, and to present scenarios of realistically achievable ecological conditions in particular regions (Gallant et al. 1989; Hughes et al. 1990; Omernik and Gallant, 1990).

The basic goal of regionalization is to depict areas of ecological homogeneity relative to other areas. Fenneman (1946) defined physiographic provinces within which the physical characteristics of the landscape, for example, surface relief and slope, were homogenous relative to other areas. Kuchler (1964) identified regions of similar potential natural vegetation.

Ecological regionalization should take into account all pertinent available information in the depiction of regions, at whatever scale the regions are to be defined (Omernik, 1987). Primary categories of information used in the process are (1) factors that control spatial patterns, such as climate, topography, and mineral availability (soils, geology); and (2) factors that respond to or integrate these controlling factors, such as vegetation and land use. Both sets of categories and each factor within them must be judged for their usefulness in depicting regions. In some areas, one combination of factors may be more useful than another for detecting regional patterns, and care must be taken to select the right combination each time. The complex interplay among the various factors must also be considered.

Omernik's approach to defining ecoregions grew out of an effort to classify streams for more effective water quality management. Thus, it is one of the few ecological frameworks expressly intended for water quality assessment. In examining spatial patterns of stream quality data, it became clear that neither major land resource areas nor Bailey's ecoregions were adequate (Hughes and Omernik, 1981; Omernik, 1987; Omernik et al. 1982). Hydrologic unit classifications have also been used as a framework for water quality assessments, and drainage basins influence fish distributions, but the spatial differences in the quantity and quality of aquatic resources usually correspond more to ecoregions than to topographic divides (Omernik and Griffith, 1991).

One advantage of having a consistent framework is that states that share the same ecoregion can cooperate across political boundaries. In times of limited resources, such cooperation makes financial as well as scientific sense.

Ecoregions have been used successfully to stratify the biotic characteristics of streams in Arkansas (Rohm et al. 1987), Nebraska (Bazata, 1991), Ohio (Larsen et al. 1986), Oregon (Hughes et al. 1987; Whittier et al. 1988), Wisconsin (Lyons, 1989), and the region of the Appalachians (Gerritsen et al. 1993). Arkansas, Minnesota, and Ohio use the ecoregion/biocriteria approach in their standards program; and several other states, such as Florida, Mississippi, Alabama, Idaho, Montana, Oregon, Washington, and Iowa, are evaluating the advantages of using ecoregions for biological assessments.

One advantage of having a consistent framework is that states that share the same ecoregion can cooperate across political boundaries. In times of limited resources, such cooperation makes financial as well as scientific sense. Where ecoregional biological criteria and use designations have been tested, they have proven to be cost-effective and protective tools (Hughes, 1989). EPA's Science Advisory Board (SAB) has concluded that the ecoregion concept "is superior to the classification methods that are currently used by most environmental managers" (U.S. Environ. Prot. Agency, 1991e).

Careful review of the purposes of regionalization and selection of the appropriate regional framework is an important part of the development of biocriteria. It may also be necessary to increase the resolution of existing regional frameworks by defining separate regions or subregions. Techniques for this process are described in the references listed in this document, particularly in Omernik's studies and Iffrig and Bowles's compendium of regional frameworks (1993).

Watersheds

Watersheds are a spatial organizing unit that can be used to develop biocriteria; however, watershed boundaries are not inconsistent with ecoregions. Increasing attention has been focused on reorienting water quality management programs to operate basinwide on a more comprehensive, coordinated basis than is possible within strict programmatic boundaries. EPA's Watershed Protection Approach (U.S. Environ. Prot. Agency, 1991f; 1993) encourages states to move in the direction of basinwide water quality management. The basinwide approach provides a framework within which to design an optimal mix of water quality management strategies. By integrating and coordinating across program and agency boundaries, basinwide management teams can achieve integrated solutions using limited resources. Thus, they can address the most significant water quality problems without losing sight of other factors contributing to the degradation of the resource. The basinwide approach helps managers achieve their short- and long-term goals for the basin by allowing the application of resources in a timely and geographically targeted manner.

Basinwide management as designed and implemented by states and EPA contains certain features that make it a fitting element of the biocriteria process:

■ **River Basin Management Units.** The state is divided into large-scale basins that provide unique units for management. All program activities that can be facilitated by or that affect basinwide management are coordinated. For instance, data requirements are aggregated and incorporated within monitoring plans, interpretations are pooled to arrive at overall assessments, and management recommendations are the result of collaboration (e.g., teams of modelers, permit writers, biologists, hydrologists, planners, engineers).

■ **Geographic Risk-based Targeting.** Because all states have limited resources and are not able to assess and solve every problem in a watershed, basin management frameworks establish a set of criteria for giving priority to the most important problems in a given area. These problems may include risks to water quality, aquatic life, or human health. While every basin in a state is visited during a basin management cycle, some waters within and across basins receive a great deal more attention than others.

■ **Direct Link to Regionalization.** An important feature of the basin management approach is its ability to incorporate a nested hierarchy of hydrologic units. Minshall (1993) discusses the need to assess ecological conditions in streams and rivers within a hierarchical landscape-scale approach. Frissell et al. (1986) present a hierarchical framework for classifying stream habitat within an overall watershed perspective. Their

Careful review of the purposes for regionalization and selection of the appropriate regional framework is an important part of the development of biocriteria. It may also be necessary to increase the resolution of existing regional frameworks by defining separate regions or subregions.

Table 3-1.— Hierarchical classification of stream riparian habitats (from Minshall, 1993; after Frissell et al. 1986).

STREAM HABITAT (LINEAR SPATIAL SCALE)	DEFINING MEASURES	BOUNDARIES: LONGITUDINAL	BOUNDARIES: LATERAL	APPLICATION	SOURCE OF INFORMATION	PROCEDURE/GUIDELINES REFERENCES
Biogeoclimatic region (10^5 m)	Regional climate Regional geology Regional topography Regional terrestrial vegetation Flow regime			Region; State; Forest; District	Topographic maps (15′) Geologic maps (15′) Landset photos Annual discharge records	Omernik, 1987 Poff and Ward, 1989
Stream system ($10^4 - 10^5$ m)	Local climate Local geology Local topography Local terrestrial vegetation Thermal regime	Drainage divides, and seacoast, or catchment, area	Drainage divides, bedrock faults, joints controlling ridge valley development	Basinwide surveys; Cumulative impacts; Integration of sites within watersheds	Topographic maps (7.5′) Geologic maps Vegetation maps Aerial photos Annual temperature records	Omernik and Gallant, 1986 Vannote and Sweeney 1980 Chorley et al. 1984 Gregory and Walling, 1973
Segment system ($10^2 - 10^2$ m)	Tributary junctions Major geologic discontinuities	Tributary junctions; major falls; bedrock lithologic or structural discontinuities	Valley sideslopes or bedrock outcrops controlling lateral migration	Paried watersheds; Segment classes (e.g., uplands vs lowlands)	Topographic maps (7.5′) Ground reconnaissance Low level aerial photos	
Reach system, ($10^1 - 10^2$ m)	Channel slope Valley form Bed material Riparian vegetation	Slope breaks: structures capable of withstanding < 50-year flood	Local sideslopes or erosion-resistant banks; 50-year floodplain margins	Local effects; Grazing allotments; Dredging	Group survey/mapping	Frissell et al. 1986 Rosgen, 1985; 1933 Minshall et al. 1989 Minshall, 1984 McDonald et al. 1991 Plafkin et al. 1989 Platts et al. 1983, 1987 Petersen, 1992
Pool/riffle system ($10^0 - 10^1$ m)	Bedform and material origin Persistence Mean depth and velocity	Water surface and bed profile slope breaks; location of genetic structures	Mean annual flood channel; midchannel bars; other flow-splitting obstructions	Aquatic habitat inventories; Fisheries Censuses	Group survey/mapping	Frissell et al. 1986 Bissonet al. 1981 McCain et al. 1990
Microhabitat system ($10^1 - 10^0$ m)	Surface particle size; underlying particle size; water depth; velocity; overhead cover (type)	Zones; differing substrate type; size arrangement	Same as longitudinal	Characterization of local spatial heterogeneity and effects (e.g., wading by fishers)	Direct measurement	

framework is designed so that the class of any particular system is partially determined by the class of the higher-level system to which it belongs.

At the broadest scale of organization, Frissell et al. (1986) recognized stream systems (i.e., watersheds), followed in order of increasing spatial resolution (and decreasing spatial extent) by segment, reach, pool or riffle, and microhabitat systems. Minshall (1993) extends the upper end of this classification scheme to include biogeoclimatic regions, thus providing a direct connection to ecoregions; and Gregory et al. (1991) similarly discuss the ecosystem attributes of riparian zones.

Table 3-1 summarizes the Frissell et al. (1986) classification framework as modified by Minshall (1993). Initial stratification of sites by biogeoclimatic regions can be performed using ecoregion delineation (Omernik, 1987). Incorporation of flow information using procedures of Poff and Ward (1989) provides further refinement of this scale of stratification and includes explicit recognition of flow as a major environmental determinant of stream and river ecosystems (Minshall, 1993; Rabeni and Jacobsen, 1993).

Ecoregions are the preferred classification for establishing reference expectations in watersheds because biota and biotic metrics respond to ecoregional differences.

Ecoregions are the preferred classification for establishing reference expectations in watersheds because biota and biotic metrics respond to ecoregional differences. Ecoregional stream systems are defined primarily by local conditions of climate, geology, topography, and terrestrial vegetation. Three examples of ecoregions are sufficient to illustrate biological variability:

1. The Calapooia River watershed (Fig. 3-2) in western Oregon crosses three ecoregions: the Willamette Valley plains; the transitional foothills region; and the Western Cascades (Omernik and Griffith, 1991). Fish, benthic macroinvertebrates, and chemical and physical habitat from 17 sites along the length of the watershed were sampled to assess changes in the river as it passed through these ecoregions. The presumption was that similar biological com-

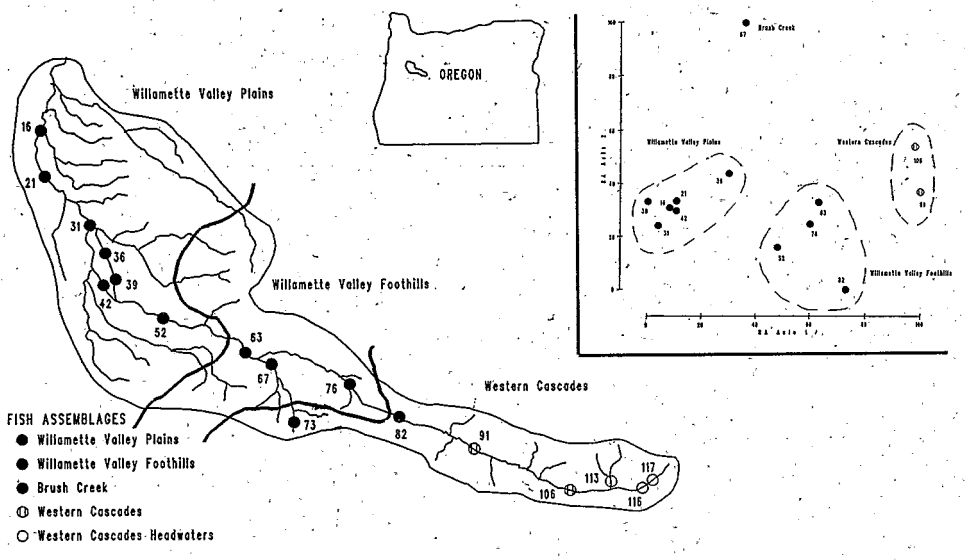


Figure 3-2.—Reciprocal averaging ordination of sites by fish species in the Calapooia River watershed, Oregon. The inset shows the correspondence between fish assemblages in the rivers and ecoregions.

munities would be found in areas of similar habitat, and that variation would correspond to observable patterns of change in the terrestrial features of the watershed.

The study results indicate that imposing an ecoregions framework on the watershed delineation is a useful predictor of stream reaches having similar biological communities. Although there was change in the communities along the watershed, distinct assemblages could be identified corresponding to the separate ecoregions within the Calapooia River watershed.

Acceptable reference sites will differ among geographic regions and states because soil conditions, stream morphology, physiography, vegetation, and dominant land uses differ among regions.

2. Ohio consists of two hydrographic basins, a Lake Erie drainage and an Ohio River drainage. Hydrographic boundaries restrict fish dispersal, and there are minor faunal differences between the two basins (Ohio Environ. Prot. Agency, 1987; Yoder, 1991). Ohio also includes parts of five ecoregions, and ecoregional differences account for a substantial amount of the variance in fish metrics and in the index of biotic integrity (IBI). Two ecoregions straddle the divide between the basins, one is entirely in the Lake Erie drainage, and two are entirely in the Ohio River drainage. If there are major differences between drainage basins, then the ecoregions that straddle the basins should be more variable. However, variability of IBI scores in all five ecoregions is similar, showing that drainage basins are negligible compared to ecoregions for explaining biological variability.

3. Florida comprises two major drainages, the Gulf of Mexico and the Atlantic Ocean. Examination of invertebrate metrics at reference sites in Florida reveal three ecoregional classes: northwest Florida (the Florida panhandle); peninsular Florida, and northeast Florida (EA, Inc., and Tetra Tech, Inc., 1994). Peninsular and northeast Florida both straddle the divide between the Atlantic and Gulf drainages; yet there are no major differences in metric values between Atlantic and Gulf basin sites on the Florida peninsula, and the peninsula differs markedly from the panhandle region, which is in the Gulf drainage.

Biogeographic differences between watersheds can be important when the watersheds are separated by a major, largely impenetrable barrier, such as the Continental Divide. Drainage dividers in more level terrain apparently do not cause significant differences in reference expectations.

Thus, implementation of biocriteria, as noted earlier, is best accomplished through an ecoregionalization approach. The implications of this with respect to states that are developing basinwide management approaches is that there may be a set of reference conditions and biocriteria established for each of the separate ecoregion areas within a given basin. Ecoregional reference conditions and biocriteria will likely be transferable across basins in a given state and — to the extent that ecoregions cross state boundaries — across states. This transferability enhances the ability of adjacent states to develop coordinated basinwide management plans for interstate basins by providing a common set of reference conditions and data to be applied in the corresponding ecoregions.

Site Selection

Because absolutely pristine habitats do not exist, resource managers must, as previously noted, decide what level of disturbance is acceptable in the area that represents the reference condition. That is, a critical element in establishing reference conditions is deciding how to determine that a site is only "minimally impaired." How much degradation can be allowed? Acceptable reference sites will differ among geographic regions and states because soil conditions, stream morphology, physiography, vegetation, and dominant land uses differ among regions.

The selection of representative and minimally impaired reference sites involves qualitative and quantitative information based on past experience and potential disturbances in regional streams. Factors that should be considered in a preliminary selection, in approximate order of importance, include the following:

1. All drainage within the ecoregion of interest.
2. No upstream impoundments.
3. No known discharges (NPDES) or contaminants in place.
4. No known spills or other pollution incidents.
5. Low human population density.
6. Low agricultural activity.
7. Low road and highway density.
8. Drainage on public lands.
9. Minimal nonpoint source problems (agriculture, urban, logging, mining, feedlots, acidic deposition).
10. No known intensive fish stocking (e.g., put-and-take stocking) or other management activities that would substantially shift the community composition.

In most settled regions of the country, reference sites will be selected by searching topographic maps for streams with the least human impacts. If candidate reference sites are more numerous than can be sampled, they should be selected randomly. Random selection will be especially important in regions with large undeveloped or undisturbed areas (e.g., mountainous regions, federal lands). Agricultural and heavily populated regions — including most of the East, Midwest, and California — will require subjective (nonrandom) reference site selection.

In most settled regions of the country, reference sites will be selected by searching topographic maps for streams with the least human impacts.

Montana Reference Conditions

The Montana Department of Health and Environmental Sciences (1990) has compiled data that describe reference conditions. Thirty-eight streams were proportionally allocated among six ecoregions in Montana, and the following criteria were used to determine a set of candidate reference streams.

1. Most or all of the drainage basin of candidate streams is in the "most typical" area of the ecoregion.

2. Each ecoregion includes at least two second-order streams, two third-order streams, and two fourth- or fifth-order streams.
3. Reference streams are not water quality limited.
4. The same streams serve as references for proposed Montana nonpoint source demonstration projects.
5. Reference streams adequately represent the major water use classifications in each ecoregion.
6. Information is available on the kinds and abundances of fish species present in the streams.
7. Sampling sites have comparable habitat from stream to stream and are located to minimize human impacts and access problems.

Site selection in the Appalachian Ridge and Valley

Because of differences in dominant land use and amounts of degradation, neighboring ecoregions may have widely different reference sites and conditions. For example, in the Central Appalachian Ridge and Valley ecoregion, criteria for selecting reference sites differ between the region's agricultural valley subecoregions and its forested ridge subecoregions (Gerritsen et al. 1993; Omernik et al. 1992).

The Ridge and Valley region of the Appalachians consists of sharply folded sedimentary strata that have eroded, resulting in a washboard-like relief of resistant ridges alternating with valleys of less-resistant rocks. The region has been divided into four subecoregions corresponding to ridges and valleys of different parent material (Omernik et al. 1992):

- Limestone valleys are characterized by calcareous bedrock and predominantly agricultural land use.
- Shale valleys are characterized by noncalcareous bedrock, primarily shale; and lower intensity agricultural land use.
- Sandstone ridges are characterized by highly resistant sandstones and forested land use.
- Shale ridges are characterized by shale bedrock and forested land use.

Each subecoregion imparts characteristic topography, hydrology, and water chemistry to streams and thus influences biota. The subecoregions are not continuous but interdigitate throughout the Ridge and Valley.

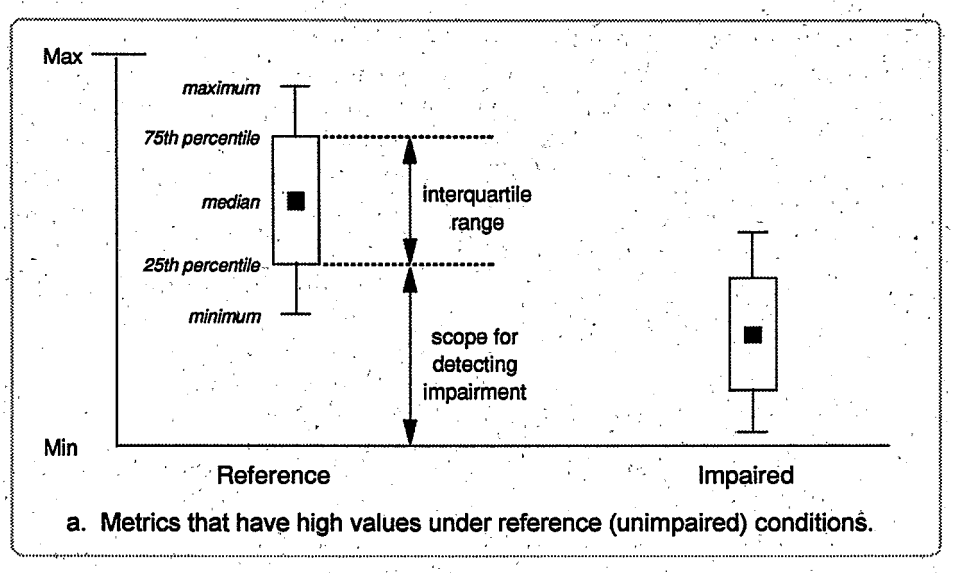
The least impacted sites occur on the ridges, where land use is predominantly forested, and where protected lands (e.g., national forests, recreation areas) are common. In contrast, nearly all streams in the valleys, and especially in the limestone valleys, are impacted by agriculture, habitat modification, and other nonpoint sources. "Minimally impaired" is, therefore, interpreted on a relative, sliding scale in each subecoregion. Reference sites for the ridges are strictly defined: they are unimpacted except by atmospheric sources. They have no discharges, nearly complete forest cover in the drainage, and no recent construction or clearcutting in the drainage. Reference sites in the valley subecoregions are less strictly defined; that is, the interpretation of minimally impaired is flexible enough to allow a sufficient number of reference sites to be selected.

Confirming Reference Conditions — Successful Classifications

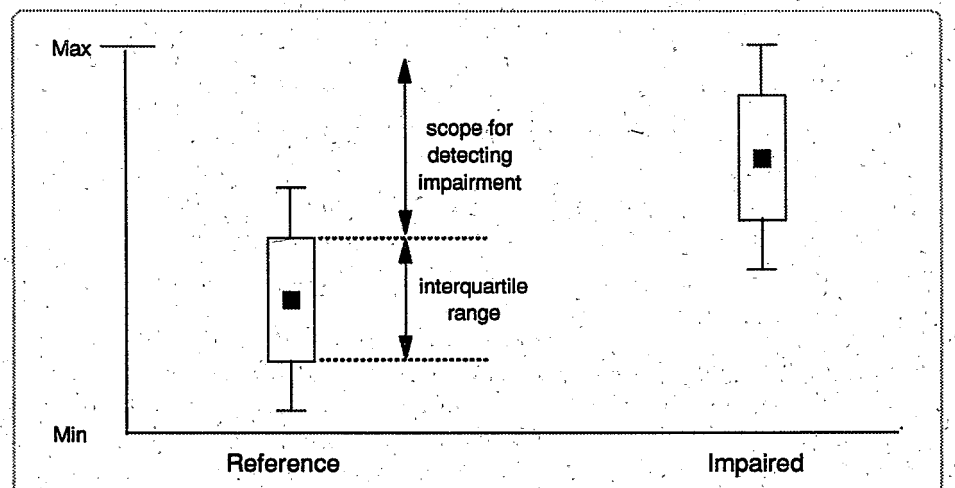
Following site selection, reference sites are surveyed (see Chapter 4) to collect biological and physical data. The data are used to confirm and refine the a priori classification, to characterize reference conditions, and to establish biocriteria (see Chapter 6). Classification is a general guide for confirming reference conditions; its effectiveness is its ability to partition variation. If a classification does not account for variability, it is of little use; the greater the amount of variance accounted for by classification, the more effective the classification.

A key analysis method for evaluating the strength of metrics to detect impairment is a graphic display using box-and-whisker plots (Fig. 3-3). In

Classification is a general guide for confirming reference conditions; its effectiveness is its ability to partition variation.



a. Metrics that have high values under reference (unimpaired) conditions.



b. Metrics that have low values under reference conditions.

Figure 3-3.—Generalized box-and-whisker plots illustrating percentiles and the detec-

The fundamental problem of biological assessment is not whether two populations (or samples) have a different mean, but whether an individual site is a member of the least-impaired reference population.

the display shown here, the central point is the median value of the variable; the box shows the 25th and 75th percentiles (interquartile range); and the whiskers show the minimum to the maximum values (range). Box-and-whisker plots are simple, straightforward, and powerful; the interquartile ranges are used to evaluate real differences between two areas and to determine whether a particular metric is a good candidate for use in the assessment.

Statistical methods used by biologists to determine whether two or more populations have different means using *t* tests include the analysis of variance and various nonparametric methods. However, the fundamental problem of biological assessment is not whether two populations (or samples) have a different mean, but whether an individual site is a member of the least-impaired reference population. If it is not, then the second question is, how far has it deviated from that reference? Such biological assessment requires the entire distribution of a metric, which is easily shown with a box-and-whisker plot.

In operational bioassessment, metric values below the lower quartile of reference conditions are typically judged impaired to some degree (e.g., Ohio Environ. Prot. Agency, 1990). The actual percentile chosen (25, 10, or 5) is arbitrary and reflects the amount of uncertainly a monitoring program can tolerate. The distance from the lower quartile can be termed a "scope for detection" (Fig. 3-3). The larger this distance is compared to the interquartile range, the easier it is to detect deviations from the reference condition. Thus, we define a "detection coefficient" as the ratio of the interquartile range to the scope for detection. This coefficient is analogous to the coefficient of variation (CV); the smaller the value, the easier it is to detect impairment.

Since assessment is based on multiple metrics or species composition, multivariate tests may be more convenient than a succession of individual tests.

Univariate tests of classifications include all the standard statistical tests for comparing two or more groups: *t* test, analysis of variance, sign test, Wilcoxon rank test, Mann-Whitney U-test (Ludwig and Reynolds, 1988). These methods are used to test for significant differences between groups (or classes) and to confirm or reject the classes. However, failure to confirm the classification for any single response variable does not mean that it will fail for other response variables.

Since assessment is based on multiple metrics or species composition, multivariate tests may be more convenient than a succession of individual tests. Discriminant analysis is a multivariate test included in many statistical software packages. It is a one-way analysis of variance that tests differences between a set of groups based on several response variables; and it can be used as a test of classifications (Conquest et al. 1994), provided that the assumptions of linearity and normality are met.

A satisfactory analysis is to develop quantitative, predictive models of biological response to habitat variables. Using a defined population of reference sites that are relatively undisturbed, investigators can develop an empirical (statistical) model that predicts biological communities based on the habitat variables (e.g., Wright et al. 1984; Moss et al. 1987). Univariate models, such as multiple regression or analysis of covariance, are linear and require appropriately transformed linear variables. Community metrics tend to respond linearly, or can be readily transformed to linearly responding variables. Species abundances are typically nonlinear (usually unimodal) in response to environmental variables and require nonlinear models.

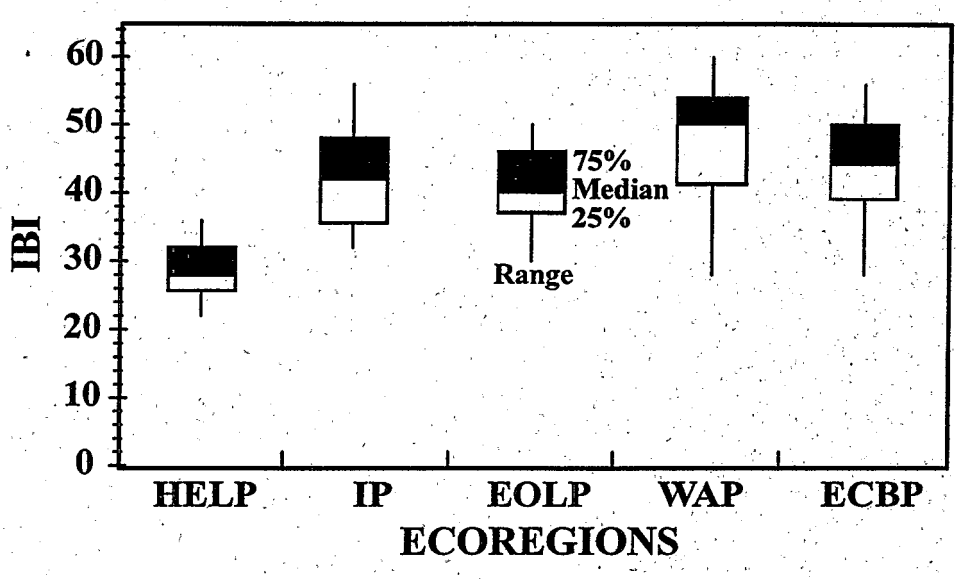


Figure 3-4.—Index of Biotic Integrity at Ohio reference sites.

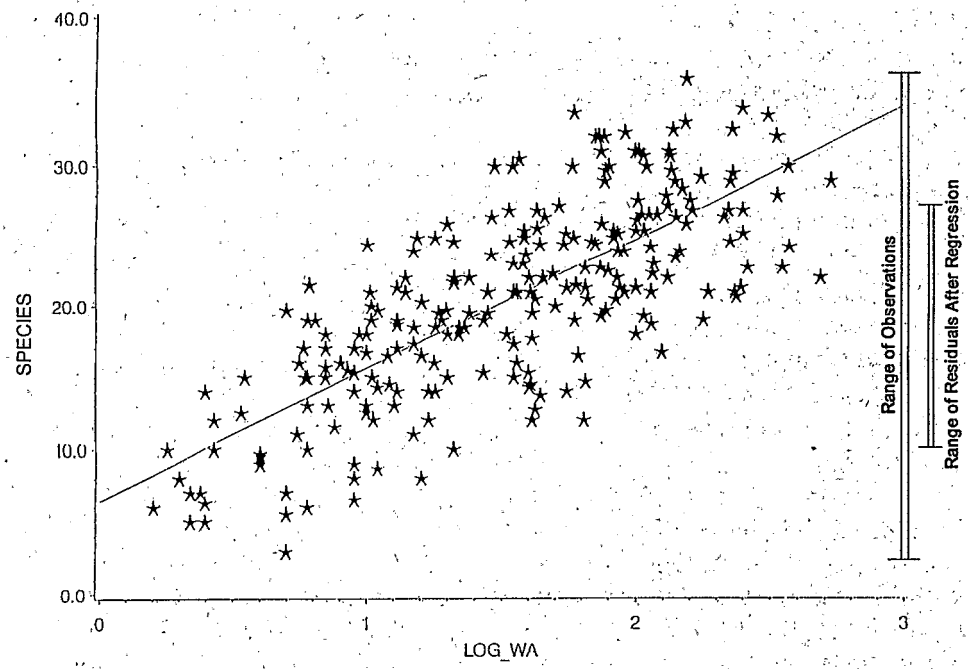


Figure 3-5.—Fish species richness as a function of the log of watershed area. Bars to right indicate range of observations before regression and range of residuals after regression. Residuals have smaller variance than the original observations.

Careful classification contributes significantly to the refinement and use of reference conditions for establishing biocriteria.

The role classification plays in partitioning variation can be illustrated using an example drawn from an extensive biosurvey database developed by the Ohio EPA. A national map of ecoregions (Omernik, 1987) indicates that parts of five ecoregions fall within Ohio. Comparison of the range of IBI, a measure of fish assemblage condition, illustrates that one ecoregion, the Huron/Erie Lake Plain, is characterized by substantially lower values than that observed in the other ecoregions (Fig. 3-4). The IBI was highest in the Western Allegheny Plateau ecoregion.

In this example, classification is used iteratively, that is, decisions for successive classifications are based on their ability to partition variation from that which would be present on a statewide basis.

One way to partition variance is by examining possible gradients to which the indicators of biotic condition may be related. Some possible gradients are stream size, physical habitat condition, and stream gradient. In Figure 3-5, species richness is plotted against a log of watershed area; the watershed area is used as a surrogate measure of stream size. The relationship is clear: increasing species richness in the reference site occurs as stream size (watershed area) increases. In this case, watershed size is used as a covariate to provide adjustments in the expected number of species associated with the drainage area within each class size.

In summary, careful classification contributes significantly to the refinement and use of reference conditions for establishing biocriteria. An iterative process is envisioned by which various classifications of regions and subregions are proposed and evaluated against partitioning of variance: successful classifications partition variance effectively; ineffective classifications provide little improvement beyond no classification. This evaluation process should generally involve multiple metrics to judge the success of multiple purpose ecoregion classifications.

Suggested Readings

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CHAPTER 4.

Conducting the Biosurvey

Purpose:
To provide guidance to technical staff for developing cost-effective biosurvey methods with appropriate resources, expertise, and technical considerations.

The primary goals of a bioassessment-biocriteria program are to evaluate water resource integrity, to provide information on the attainability and appropriateness of existing uses, and to determine the extent and degree of water resource impairment.

State bioassessment-biocriteria programs are usually designed to address one or more of four water resource management objectives:

1. **Aquatic Life Use Designation.** Determine and assess aquatic life uses that should be attained in streams and rivers. Helping to designate and assess aquatic life uses is a major function of biological criteria.

2. **Sensitive Waters Identification.** Characterize high quality waters for protection. High quality waters may become part of the reference database or be classified separately as unique waters.

3. **Diagnostics.** Determine sources of impairment and potential stressors. Biological response signatures are used in conjunction with chemical, toxicological, and physical data to identify causes of impairment.

4. **Program Evaluation.** Monitor effectiveness of pollution abatement programs, including wastewater treatment, watershed restoration, and other water resource quality improvement programs. Biosurveys and the biocriteria benchmarks are used to assess the recovery of the aquatic community.

Detailed multidisciplinary ecological studies are often designed to examine aquatic systems by measuring the elements and processes of biological communities and by describing the physical and chemical characteristics of the waterbody. Biological attributes that may be included in such studies are individual health, trophic organization, measures of primary, secondary, and tertiary production (bodily growth and reproduction), recruitment of key species, predator-prey relationships, population dynamics, and taxonomic structure of assemblages.

While seasonal accommodation is preferable for most bioassessment programs, a single annual sample at a carefully-selected time is sufficient

Quality assurance and control should be a continuous process throughout the development and operation of the biocriteria program, including all aspects of the study.

to characterize biological conditions accurately. Selection of the sampling period should be based on efforts to minimize variability and maximize the efficiency of the equipment and the accessibility of the biota being sampled. Minimal between-year variability is partially addressed by sampling at the same time each year to correct for the natural variability in seasonal cycles.

Water quantity, quality, and climatic conditions should help rather than hinder the efficiency of the sampling gear. For example, if certain flow conditions are necessary for the equipment's performance, sampling schedules should coincide with those conditions. Above all, sampling should occur when the targeted assemblage or assemblages are accessible. For fish, the optimal sampling period in most parts of the country is likely to be from June through September; in general, these months avoid high and low flows, spawning periods, and migration activity. Sampling should be timed to avoid extremes in environmental and biological conditions.

Quality Assurance Planning

A major consideration when designing bioassessment studies is not whether a particular biosurvey approach is more refined than another, but whether the selected approach will achieve the objectives defined in the management plan. A clear definition of management responsibilities and effective quality assurance and quality control procedures (see Chapter 2) are essential to ensure the usefulness of monitoring data (Plafkin et al. 1989).

Quality assurance plans have two primary functions (Klemm et al. 1990). The first function is to ensure that the survey process reliably meets program objectives; the second is to monitor the reliability of the survey data to determine their accuracy, precision, completeness, comparability, and representativeness.

Quality assurance and control pervade all aspects of an ecological study:

- *Study design*
- *Field operations*
- *Laboratory activities*
- *Data analysis*
- *Reporting*

A quality assurance plan should be developed at the onset of an ecological study to delineate responsibility, establish accountability, and ensure the reliability of the data (Stribling and Barbour, 1991). The quality assurance plan should answer three questions:

- What kind of data or information is needed?
- Why is the information or data needed?
- What level of quality is needed to ensure the reliability of decisions based on these data?

Quality assurance for a biocriteria program is concerned with the integrity of the data used to establish biocriteria limits and thresholds along with the documentation that supports the derivation and maintenance of the biocriteria. Quality assurance for specific studies pertains to the data acquisition, their application to established biocriteria, and the validity of associated judgments.

Quality assurance and control should be a continuous process throughout the development and operation of the program, including all aspects of the study: design, field collection, habitat assessment, laboratory processing of samples, database management, analysis, and report-

ing. The appropriateness of the investigator's methods and procedures and the quality of the data to be obtained must be assured before the results can be accepted and used in decision making. Quality assurance is accomplished through data quality objectives, investigator training, standardized data gathering and processing procedures, verification of data reproducibility, and instrument calibration and maintenance.

The use of data quality objectives in field studies (Klemm et al. 1990; Plafkin et al. 1989; U.S. Environ. Prot. Agency, 1984b, 1986) has much to offer the biocriteria development and implementation process. Data quality objectives are qualitative and quantitative statements within the quality assurance plan that address specific decisions or regulatory actions. Generally, data quality objectives consist of a priori statements about the level of uncertainty a decision maker will accept in environmental data. Once the objectives are stated, the quality of particular data can be measured using predetermined types and amounts of error associated with their collection and interpretation.

Quality Management

The implementation of a biocriteria program requires quality management or the proper combination of resources and expertise. State agencies will differ in levels of biological expertise, facilities, and quality of equipment. States already having well-developed bioassessment programs generally have experienced and well-trained biologists, appropriately equipped facilities, and properly maintained sampling gear. A successful biocriteria program depends on (1) a clear definition of goals, (2) the active use of biomonitoring data in decision making, and (3) the allocation of adequate resources to ensure a high quality program.

Biocriteria Program Structure, Personnel, and Resources

Monitoring agencies can and should enhance their program by cooperation with others. For example, they should seek coordination with, and staff assistance from, state fishery, land management, geology, agriculture, and water quality agencies. If federally employed aquatic biologists are stationed in a state or if the state has substantial federal lands, cooperative bioassessments and biocriteria development programs should be initiated. Scientists at state universities should also be included in the planning and monitoring phases of the program; their students make excellent field assistants and future state ecologists.

Monitoring agencies can and should enhance their program by coordination with, and staff assistance from, state fishery, land management, geology, agriculture, and water quality agencies.

■ **Personnel.** Several trained and experienced biologists should be available to provide more thorough evaluations, support for various activities, and serve as quality control checks. They should have training and experience commensurate with the needs of the program. At least one staff member should be familiar with establishing a quality assurance framework. The program should have at least one biologist for every 4,000 miles of stream in the state (C. Yoder and R. Thoma, personal communication).

■ **Resources.** Laboratory and field facilities and services should be in place and operationally consistent with the designed purposes of the program so that high quality environmental data may be generated and processed in an efficient and cost-effective manner (Klemm et al. 1990)

Adequate taxonomic references and scientific literature should support data processing and interpretation. The following program and technical considerations should guide the design and implementation of the biocriteria program.

■ **Program Elements**

1. Quality assurance and quality control (e.g., standard operating procedures, training)
2. Delineated reference conditions with annual updates corresponding to seasons of sampling
3. Multiple assemblage biosurvey
4. Habitat assessment
5. Documentation of program and study plans

■ **Technical Considerations**

1. Assign taxonomy to the lowest possible level based on published keys and descriptions; maintain voucher collections
2. Schedule multiple season sampling if warranted by type of impact and life strategy of assemblage
3. Use multiple metrics to refine the assessment
4. Initiate detailed quality assurance and quality control procedures in the field, laboratory, and taxonomy
5. Provide computer hardware and software (database management, data analysis) with computer training of staff

Different levels of training and experience are necessary for the personnel involved in biocriteria programs. The qualifications and general job descriptions of four levels of professional staff are presented here. Also described are suitable substitutions for these prerequisites and experience.

■ **Professional Staff**

1. **Level 4** — Plans, conducts, and supervises projects of major significance, necessitating advanced knowledge and the ability to originate and apply new and unique methods and procedures. Supplies technical advice and counsel to other professionals. Generally operates with wide latitude for unreviewed action.

   Typical Title: Project Manager, Chief Biologist.

   Normal Qualifications: Ph.D. or M.S. and equivalent experience.

   Experience: Ten or more years, at least three years in a leadership or managerial position.

2. **Level 3** — Under general supervision of project manager, plans, conducts and supervises bioassessment tasks such as trend monitoring or special studies. Estimates and schedules work to meet completion dates. Directs support assistance, reviews progress, and evaluates results; makes changes in methods, design, or equipment as necessary. Operates with some latitude for unreviewed action or decision.

Typical Title: Project Biologist, Group Leader, Crew Leader.

Normal Qualifications: M.S., B.S., or equivalent experience.

Experience: Six or more years in or related to bioassessment, two to three years in a supervisory capacity.

3. **Level 2** — Under supervision of a chief biologist or project manager, carries out assignments associated with projects. Translates technical guidance received from supervisor into usable data applicable to the particular assignment; coordinates the activities of juniors or technicians. Work assignments are varied and require some originality and ingenuity.

   Typical Title: Associate Biologist, Environmental Scientist.

   Normal Qualifications: B.S. or equivalent experience

   Experience: Three to eight years in or related to freshwater biology.

4. **Level 1** — Lowest or entering classification. Works under close supervision of a group or crew leader. Gathers and correlates basic data and performs routine analyses. Works on less complicated assignments that require little evaluation.

   Typical Title: Field Technician.

   Normal Qualifications: B.S. or Associate Degree and equivalent experience.

   Experience: zero to three years.

Experience/Qualifications Substitutions

1. Any combination of additional years of experience in the proposed field of expertise and full-time college-level study in the particular field totaling four years of structured, directed education may be substituted for a B.S.

2. A B.S. and any combination of additional years of experience and graduate-level study in the proposed field of expertise totaling two years may be substituted for the M.S.

3. A B.S. and any combination of additional years of experience and graduate study in the proposed field of expertise totaling four years; or an M.S. and two years of either additional experience or graduate-level study in the proposed field may be an acceptable substitute for the Ph.D.

4. Additional years of graduate-level study in an appropriate field will be considered equal to years of experience on a one-for-one basis.

The quality manager will identify project responsibilities and accountabilities for the bioassessment program. In states with limited resources, the basic responsibilities for all levels will rest with relatively few individuals; however, the accountability of each position will be quite distinct.

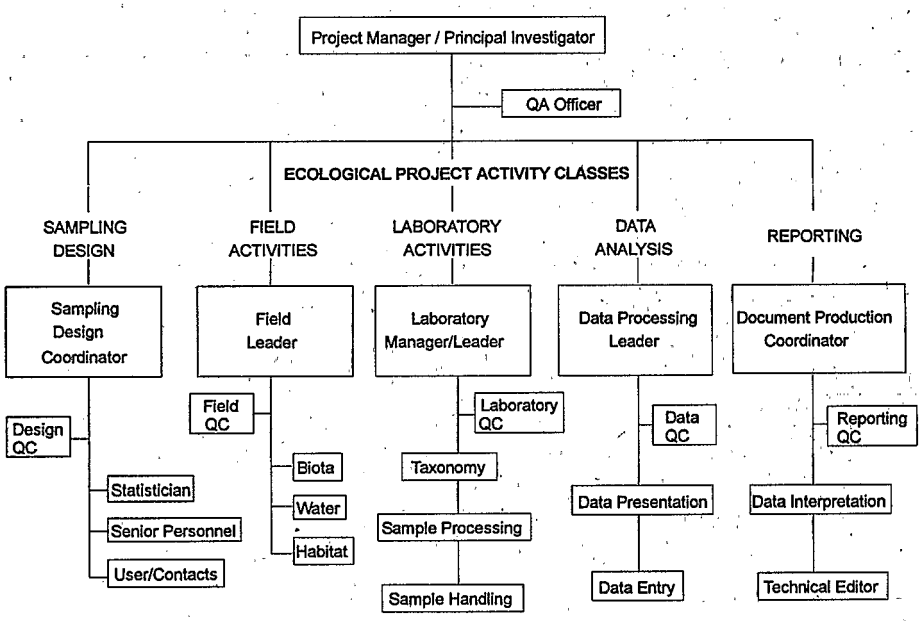

Figure 4-1.—Organizational chart illustrating project organization and lines of responsibility.

Quality management is an important planning aspect of the biocriteria development process that focuses attention on establishing and improving quality in all aspects of a project. Quality management requires that all personnel involved in a biocriteria project (from senior management to field and laboratory technicians) be aware of and responsive to data needs and expectations. The surest way to achieve total quality management (TQM) in an environmental program is to implement an achievable quality assurance program.

Effective quality control procedures are essential to insure the usefulness of the data for biocriteria development and environmental decision making, and to maintain the bioassessment program.

Quality Control Elements in an Ecological Study

Effective quality control procedures are essential to insure the usefulness of the data for biocriteria development and environmental decision making, and to maintain the bioassessment program. The organizational chart in Figure 4-1 identifies the major activity classes in an ecological project; Table 4-1 outlines the quality control elements that are integral to those activities.

All activity classes or phases of field ecological studies have potential error sources associated with them (Barbour and Thornley, 1990). Some general quality control elements for reducing error are discussed here; for more specific approaches, the investigator should refer to Klemm et al. (1990) for benthic macroinvertebrates; and to Karr et al. (1986), Lyons (1992), and Ohio Environ. Prot. Agency (1987) for fish.

■ **Study Design.** Considerations relating to potential error in the study design range from limited resources to insufficient sample replication to selection of inappropriate variables. Two important considerations for developing a study design are interrelated: the availability of baseline data in historical information or pilot studies and the capacity to identify poten-

Table 4-1.—Quality control elements integral to the activities in an ecological study.

A. Quality Management
1. Delineate responsibilities
2. List accountabilities
3. Identify quality assurance officer
4. Develop quality assurance plan
5. Use bioassessment information in decision making

B. Study Design
1. Pilot study or site reconnaissance
2. Account for environmental strata
3. Incorporate historical data
   a. Attempt to duplicate regimes
   b. Attempt to use similar equipment (if appropriate to current objectives)
4. Termination of control point
5. Areas of potential error
   a. Available resources
   b. Logistics
   c. Response variables
   d. Weather
   e. Seasonality
   f. Site selection
   g. Habitat variability
   h. Population variability
   i. Equipment
6. Additional performance effect criteria

C. Sample Collection
1. Instrument calibration and maintenance
2. Field crew
   a. Training
   b. Evaluation
3. Field equipment
4. Sample handling
5. Effort checks
6. Field crew efficiency
7. Areas of potential error
   a. Climate
   b. Site selection
   c. Sampling efficiency of equipment
   d. Equipment operation: human error
   e. Field notes
   f. Samples
      i. Processing
      ii. Transportation
      iii. Tracking
8. Additional performance effect criteria

D. Sample Processing
1. Sorting and verification
2. Taxonomy
3. Duplicate processing
4. Archival procedures
5. Training
6. Data handling
7. Interlaboratory training and collaboration
8. Areas of potential concern
   a. Sample tracking
   b. Improper storage
   c. Sample preparation
   d. Reference error (taxonomy)
   e. Taxonomic error (human)

(continued on next page)

Two of the most important considerations in developing a study design are the availability of baseline data in historical information or pilot studies and the identification of potential sources of error.

Table 4-1.— Continued.

	f. Counting error
	g. Sorting efficiency
	9. Additional performance effect criteria
E.	Data Analysis
	1. Training
	2. Data
	a. Handling
	b. Reporting
	3. Standardized database
	4. Standardized analyses
	5. Peer Review
	6. Range control
	7. Statistical power analysis
	8. Areas of potential error
	a. Inappropriate statistics
	b. Errors in database
	c. Database management
	d. Programming errors
	e. Analytical misinterpretation
	9. Additional performance effect criteria
F.	Report Preparation
	1. Training
	2. Peer review
	3. Technical editor
	4. Standard format
	5. Areas of potential error
	a. Transcription
	b. Poor presentation
	c. Obscure language
	d. Addressing performance effect criteria
	6. Additional performance effect criteria

tial sources of error. In fact, having adequate baseline information may be the only way to identify sources of error. As more than one quality control element may be used to reduce potential error, the interaction among quality control elements must be considered to ensure the overall quality of the plan.

Six qualitative and quantitative characteristics are usually employed to describe data quality:

- **Precision.** The level of agreement among repeated measurements of the same characteristic.
- **Accuracy.** The level of agreement between the true and the measured value; the divergence between the two is referred to as bias.
- **Representativeness.** The degree to which the collected data accurately and precisely reflect the frequency distribution of a specific variable in the population.
- **Completeness.** The amount of data collected compared to the planned amount.
- **Comparability.** The degree to which data from one source can be compared to other sources.

- **Measurability.** The degree to which measured data remain within the detection limits of the analysis — often a function of the sensitivity of instrumentation.

These characteristics should be considered and defined before the data collection begins. Taken collectively, they provide a summary characterization of the data quality needed for a particular environmental decision.

■ **Field Operations.** The major quality control elements in field operations are instrument calibration and maintenance, crew training and evaluation, field equipment, sample handling, and additional effort checks. The potential errors in field operations range from personnel deficiencies to equipment problems. Training is the most important quality control element for field operations. Establishing and maintaining a voucher specimen collection is also important. Vouchers are a mechanism for achieving the source of the data, particularly for benthos. Use of a protocol for double data entry and comparison can also increase the quality of a database.

■ **Laboratory Operations.** The quality control elements in laboratory operations are classified as sorting and verification, taxonomy, duplicate processing, archival procedures, training, and data handling. Potential error sources associated with sample processing are best controlled by staff training. Controlling taxonomic error requires well-trained staff with expertise to verify identifications. Counting error and sorting efficiency are usually the most prominent error considerations; they may be controlled by duplicate processing, sorting, and verification procedures. Errors associated with transcription during the data entry process can be significant. In the laboratory, as in the field, the use of a protocol for double data entry and comparison can increase the quality of a database, and the establishment and maintenance of a voucher specimen collection should be considered.

■ **Data Analysis.** Peer review and range of values are the important quality control elements for data analysis. Peer review helps control operator variability, and measurement values must be kept within the range of natural or normal variability. Further, if inappropriate statistics are used to analyze the data, erroneous conclusions may be drawn regarding trends. Undetected errors in the database or programming can be disastrous, and unless steps are taken to oversee data handling and analysis, problems related to database management will arise. The use of standardized computer software for database management and analysis can minimize errors associated with tabulation and statistics. A final consideration is the possible misinterpretation of the findings. These potential errors are best controlled by qualified staff and adequate training.

■ **Reporting.** The quality control elements in the reporting activity include training, peer review, and the use of a technical editor and standard formats. The use of obscure language can often mislead the reader. Peer review and review by a technical editor are essential to the development of a scientific document. If the primary objective or central question of the study is not specifically addressed in the report or the report is ambivalent, then an error in the reporting process has occurred.

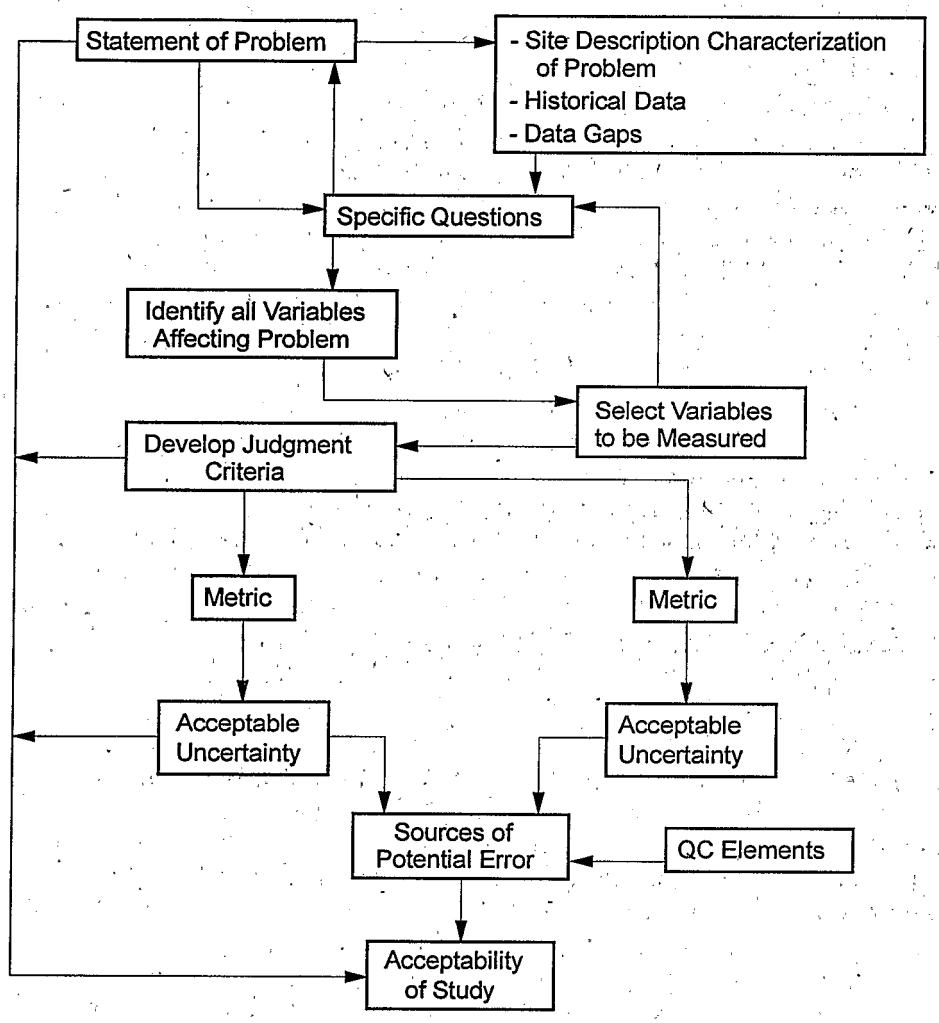

Figure 4-2.—Summary of Data Quality Objective (DQO) process for ecological studies (taken from Barbour and Thornley, 1990).

Data Quality Objectives

The data quality objectives process occurs during the final creation of the research design. Although its aspects are inherently interrelated, the development of data quality objectives is not directly linear. Rather, this development is an iterative or circular process, as shown in Figure 4-2. The initial statement of the problem evolves from specific questions about existing data; then comes the identification and selection of the variables to be measured, which influence the further refinement of the questions; and, finally, judgment criteria are developed for each variable, acceptable uncertainty levels are established, and sources of potential error are identified.

The result of the data quality objectives process is a formal document that can be separate from or part of a formal quality assurance plan. It may also be included in narrative form in a project workplan. The data quality objectives document should state the study's primary objectives, specific questions, and rationale; it should also justify the selection of variables, establish judgment criteria (by developing a logic statement for each

variable), and specify acceptable levels of uncertainty. This information does not have to be presented in a stepwise fashion, but it should be readily available.

All staff involved in the biocriteria development process — senior management, program staff, and all technical staff — should be included in formulating data quality objectives. In fact, quality management in ecological studies requires that all personnel involved in a project be aware of and responsive to detailed needs and expectations. If appropriately executed, data quality objectives will formalize and document all management decision points, the necessary data collection and analysis procedures, the data interpretation steps, and the potential consequences of making an incorrect decision.

Further details of quality assurance and control programs specific to fish and macroinvertebrate field surveys, and methods for determining biological condition, are provided in Klemm et al. (1990) and Plafkin et al. (1989). General guidance for developing comprehensive quality assurance and control plans are discussed in the Code of Federal Regulations (40 CFR Part 30), and U.S. Environ. Prot. Agency (1980a,b; 1984a,c). For information and guidance specific to data quality objectives, see Klemm et al. (1990), Plafkin et al. (1989), and U.S. Environ. Prot. Agency (1984b, 1986).

The primary focus of the study design is to establish objectives, and the statement of the problem to be resolved is the central theme of the objectives.

Study Design

The primary focus of the study design is to establish objectives, and the statement of the problem to be resolved is the central theme of the objectives. For instance, the central problem or question may be, "Is the biological integrity of a specified area of a particular watershed impaired by the operation of a wastewater facility?" This question has several features that, in turn, provide a foundation for more specific questions. The first feature is the concept of biological integrity, which implies that a measurable reference condition exists for the aquatic assemblages being studied. The second feature delineates the spatial area to be evaluated in the watershed; the third determines whether or not a problem is attributable to the operation of the facility. Still more specific questions, or testable hypotheses, related to the central problem may be constructed.

1. Is impairment of the biological condition detectable in the algae, fish, or macroinvertebrate assemblages?
2. Is degradation altering the energy base, water quality, flow regime, habitat structure, or other aspect of the environment?
3. Is there a history of problems in this area of the watershed?
4. What was the historical condition of the aquatic community?

Based on these questions, it is possible to select the biotic and abiotic variables to be measured. For each variable, an acceptable level of degradation should be identified before conducting the biosurvey. Thus, the study design includes selecting the aquatic assemblages, resolving the technical issues associated with their ecology and proper sampling, establishing standard operating procedures, and beginning the biosurvey program.

Biosurveys of Targeted Assemblages

A critical decision in the design of biocriteria programs is how to select appropriate indicators of biotic condition.

A critical decision in the design of biocriteria programs is how to select appropriate indicators of biotic condition. Biosurvey of the targeted assemblages is the most widely employed approach to biocriteria development. This approach, which has been used by Ohio, Illinois, North Carolina, Maine, Arkansas, New York, and Vermont, focuses on a selected component of the biological community; it samples one or several specific aquatic community segments to measure biological condition. Monitoring the specific characteristics of these assemblages helps assess the effects of a variety of environmental conditions (Ohio Environ. Prot. Agency, 1987).

A number of different organisms associated with lotic systems (i.e., streams and rivers) lend themselves to bioassessment procedures. Commonly measured assemblages include, but are not restricted to, macrophytes, algae, macroinvertebrates, and fish. The targeted assemblage approach to bioassessment can also focus on a single assemblage (e.g., periphyton) or several assemblages (e.g., periphyton, macroinvertebrates, and fish). The attributes measured may be functional parameters, such as photosynthesis or respiration, or other attributes, such as individual health. Examples of widely used methods and techniques for targeted assemblages are found in Karr (1981), Karr et al. (1986), Ohio Environ. Prot. Agency (1987), Plafkin et al. (1989), Standard Methods (1989), U.S. Environ. Prot. Agency (1990), and Weber (1973). The primary advantages of this approach are its flexibility, practicality, cost-effectiveness, and relative scientific rigor.

The importance of the periphyton assemblage within most stream ecosystems makes it a prime candidate for consideration as a bioassessment–biosurvey target.

Attributes of Selected Assemblages

■ **Periphyton.** The periphyton assemblage is composed of benthic algae, bacteria, their secretions, associated detritus, and various species of microinvertebrates (Lamberti and Moore, 1984). Periphyton are an important energy base in many lotic situations (Dudley et al. 1986; Minshall, 1978; Steinman and Parker, 1990) and serve as the primary nutrient source for many stream organisms (Lamberti and Moore, 1984). The capacity of benthic assemblages to colonize and increase in biomass is influenced by variability in stream channel geomorphology, flow rates, herbivore grazing pressure, light intensity, seasonality, and random processes (Coleman and Dahm, 1990; Grimm and Fisher, 1989; Hamilton and Duthie, 1984; Korte and Blinn, 1983; Lamberti et al. 1987; Patrick, 1949; Poff et al. 1990; Steinman and McIntire, 1986, 1987; Steinman et al. 1987; and Stevenson, 1990).

The importance of the periphyton assemblage within most stream ecosystems makes it a prime candidate for consideration as a bioassessment-biosurvey target. More specific advantages are outlined by Plafkin et al. (1989):

- The rapid algal reproduction rates and short life cycles of periphyton make them valuable indicators of short-term impacts.
- Physical and chemical factors have direct effects on the structure and functions of periphyton and on their production.
- Periphyton sampling methods are straightforward, and the samples are easily quantified and standardized.

- Methods have also been standardized for recording functional and nontaxonomic characteristics of periphyton communities, such as biomass and chlorophyll measurements.
- Algal components of periphyton are sensitive to some pollutants to which other organisms may be relatively tolerant.

■ **Macrophytes.** The macrophyte assemblage consists of large aquatic plants that may be rooted, unrooted, vascular, or algiforms. Both emergent and submergent macrophytes provide numerous benefits to streams and small rivers thus helping them to support healthy, dynamic, biological communities (Campbell and Clark, 1983; Hurley, 1990; and Miller et al. 1989). Some understanding of the distributional characteristics and environmental conditions affecting macrophytes (Hynes, 1970) enhance their use in bioassessment strategies. Hynes (1970) and Westlake (1975) discuss differences in lotic macrophyte assemblages based on habitat factors such as water hardness, pH, gradient, and propensity for siltation.

Some investigators have emphasized the influence of macrophytes on habitat structure (Carpenter and Lodge, 1986; Gregg and Rose, 1982, 1985; McDermid and Naiman, 1983; Miller et al. 1989; Pandit, 1984); others have studied water chemistry, nutrient cycling, and macroinvertebrate colonization (McDermid and Naiman, 1983; Miller et al. 1989). Pandit (1984), Seddon (1972), and Westlake (1975) pointed to the use of macrophytes as an indicator assemblage in lotic situations.

Aquatic macrophytes are an important food source for birds and mammals. Fassett (1957) lists 36 species of waterfowl, nine marshbirds, four shorebirds, and nine upland game birds that feed on these plants. He also lists beaver, deer, moose, muskrat, and porcupines as aquatic macrophyte herbivores. The use of macrophytes in bioassessment programs has numerous advantages:

- Macrophyte taxonomy to the generic level is relatively straightforward.
- Because the establishment of macrophyte populations in a specific habitat depends partly on local environmental conditions, they are potentially very useful as site-specific indicators.
- Because their specific microhabitat structure does not limit germination, macrophytes are potentially found in high population densities.
- The growth patterns of individual macrophytes are directly influenced by herbivore activity.
- The longevity, distribution, and rate of their population growth may directly reflect prevailing conditions.

Benthic macroinvertebrate assemblages are important indicators of localized environmental conditions.

■ **Macroinvertebrates.** Macroinvertebrates are the visibly distinguishable crustaceans, molluscs, insects, and other fairly large aquatic invertebrates. Benthic macroinvertebrate assemblages are important indicators of localized environmental conditions because they inhabit the degraded or contaminated resources and can be exposed to degradation directly throughout their life history. Their characteristics can be regarded as a reflection of the integration of short-term environmental variability (Plafkin et al. 1989). At sensitive life stages, they respond quickly to stress; how-

Fish assemblages are well suited to help define environmental conditions because fish inhabit the receiving waters continuously, and with lifespans up to 10 years, they can easily represent the integrated historical effects of chemical, physical, and biological habitat factors.

ever, the overall assemblage responds more slowly. Other advantages of using macroinvertebrates include the following:

- Sampling methods are well developed and require minimal personnel and inexpensive gear.
- Macroinvertebrates play a major role in the nutritional ecology of commercial and sport fisheries.
- Most streams support sufficient abundance levels for assessment.
- Molluscs, many species of crustacea, and some insects are largely immobile. As residential organisms, they are particularly valuable indicators of site conditions over time.
- Many states have already performed background benthic surveys, have personnel trained in benthic biology, and can often get assistance in sampling from lay groups.

■ **Fish.** Fish assemblages are well suited to help define environmental conditions — either natural or impaired. Fish are long-lived and inhabit the receiving waters continuously. With lifespans up to 10 years, they can easily represent the integrated historical effects of chemical, physical, and biological habitat factors (Ohio Environ. Prot. Agency, 1987). Power (1990) found that fish exert significant influence on the food chain in lotic systems. More specific advantages of using the fish assemblage for bioassessment (Karr et al. 1986; Plafkin et al. 1989) include the following:

- Fish are usually present in lotic systems except for some headwaters.
- Their populations generally include species that feed at a variety of trophic levels.
- Species composition and dominants are relatively stable in most areas.
- The migration patterns and wide-ranging foraging behavior of some fish allow investigators to accumulate effects from relatively large-scale habitats.
- In comparison to other potential bioassessment groups, fish are relatively easy to identify.
- Autecological studies for many freshwater species are extensive, so their life histories are relatively well known.
- Public, and therefore, legislative appreciation for fish is apparent in the fishable goal of the Clean Water Act, the Endangered Species Act (50 percent of "endangered" vertebrate species are fish), and in more specific commercial and sport fisheries legislation.
- Historical survey data are probably best documented for fish.
- Investigators can often get assistance from lay groups.

■ **Wildlife.** Mammals, birds, reptiles, and amphibians can also provide valuable information for bioassessment decisions. Croonquist and Brooks (1991), applying the concept of response guilds, found that bird species with high habitat specificity decrease with increasing habitat alteration.

This approach has considerable potential for development of an avian index of biotic integrity. Birds have been shown to reflect the condition of riparian systems.

Because amphibians live part of their life cycle in an aqueous or damp environment, they are a link between the aquatic and terrestrial environments. They are also sensitive to littoral zone and riparian disturbances and to changes in their food resources (macroinvertebrates and periphyton). The latter may affect their fitness or force them to emigrate from the home range to another foraging zone. Other advantages of including a biosurvey of mammals, birds, and amphibians in biomonitoring programs are the following:

- Their longer life spans make them well suited for evaluation of cumulative effects.
- The relatively large body size of birds and their behaviors (e.g., singing) allow visual and auditory observation to supply most of the necessary information.
- Birds are sensitive to riparian alteration.
- Wildlife taxonomy is well understood.
- Many biomarkers — physical and chemical alterations in the species in response to contamination — appear in these organisms, and an increased likelihood for sublethal effects in non-emigrating individuals.
- Trapping techniques for small mammals are relatively straightforward, and their tracks and droppings also provide easily attainable survey data.
- The public is usually able to assist in conducting wildlife assessments.

Biological programs that use two or three assemblages and include different trophic levels within each group will provide a more rigorous and ecologically meaningful evaluation of a system's biological integrity and a greater range of temporal responsiveness.

Synthesis

Many bioassessment programs focus on a single assemblage for reasons of regulatory focus or mandate, available expertise, resource limitations, or public awareness and interest. However, state agencies are encouraged to incorporate more than one assemblage (e.g., fish and benthic macroinvertebrates) into their assessment programs. Biological programs that use two or three assemblages and include different trophic levels within each group (e.g., primary, secondary, and tertiary consumers) will provide a more rigorous and ecologically meaningful evaluation of a system's biological integrity (U.S. Environ. Prot. Agency, 1990) and a greater range of temporal responsiveness.

Impairments that are difficult to detect because of the temporal or spatial habits or the pollution tolerances of one group may be revealed through impairments in different species or assemblages (Ohio Environ. Prot. Agency, 1987). Mount et al. (1984) found that benthic and fish assemblages responded differently to the same inputs in the Ottawa River in Ohio. Benthic diversity and abundance responded negatively to organic loading from a sewage treatment plant and exhibited no observable response to chemical input from industrial effluent. Fish exhibited no response to the organic inputs and a negative response to metals. In a more

recent assessment, the Ohio EPA found that distinct response signatures (Yoder, 1991) in both fish and macroinvertebrate assemblages indicated an adverse effect from the sewage treatment plant. Selection of aquatic community components that show different sensitivities and responses to the same disturbance will help identify the nature of a problem (U.S. Environ. Prot. Agency, 1990).

Selecting a single assemblage for assessment may provide inadequate resolution for certain impacts that are highly seasonal in occurrence. Organisms having short life cycles may not reflect direct exposure to highly variable impacts at critical times or when complex cumulative impacts are present. Depending on the collection period, those organisms may provide a false sense of ecosystem health if other assemblages of longer-lived populations are under stress. In cases in which periodic pulses of contaminants may occur, long-lived populations may be slow to exhibit response, whereas short-lived organisms may be severely affected.

Aquatic organisms respond to stress in a variety of ways ranging from alterations in community composition and structure to increases or decreases in the biomass of a single or multiple species, or mortality.

The occurrence of multiple stressors and seasonal variation in the intensity of stressors require that more than one assemblage be incorporated into biocriteria programs whenever practical. Not all assemblages discussed here are in constant contact with the aquatic habitat component. Those that are — the macroinvertebrates, macrophytes, fish, and periphyton — will exhibit direct, and potentially more rapid, responses to water resource degradation. The assemblage comprising mammals, birds, and amphibians indicates the quality of the riparian corridor and can reflect local land use impacts on the water resource.

Aquatic organisms respond to stress in a variety of ways ranging from alterations in community composition and structure to increases or decreases in the biomass of a single or multiple species, or mortality. Fish and drifting macroinvertebrates also exhibit avoidance behavior by seeking refugia from short- and long-term disturbances.

Careful selection of taxonomic groups can provide a balanced assessment that is sufficiently broad to describe the composition and condition of an aquatic ecosystem, yet practical enough for use on a routine basis (Karr et al. 1986; Lenat, 1988; Plafkin et al. 1989). When selecting community components to include in a biological assessment, primary emphasis should be given to including species or taxa that (1) serve as effective indicators of high biological integrity, that is, those likely to live in unimpaired waters, (2) represent a range of pollution tolerances, (3) provide predictable, repeatable results from consistent sampling, (4) can be readily identified by trained state personnel (U.S. Environ. Prot. Agency, 1990), (5) show a consistent response to pollution stress, and (6) closely represent local, indigenous biota.

Technical Issues

The methods and procedures used in bioassessment programs should be based on the study objectives and associated technical issues, including the selection of the proper sampling period, sites, and sampling regime; and the determination of the appropriate habitats to be sampled.

Selection of the Proper Sampling Periods

The ideal sampling procedure is to survey the biological community with each change of season, then select the appropriate sampling periods that accommodate seasonal variation. Such indexing makes the best use of the biological data. It ensures that the sources of ecological disturbance will be monitored and trends documented, and that additional information will be available in the event of spills or other unanticipated events.

The ideal sampling procedure is to survey the biological community with each change of season, then select the appropriate sampling periods that accommodate seasonal variation.

In this way, the response of the community to episodic events (e.g., chemical spills) can be assessed throughout the year. Seasonal impacts, which may be highly variable, can be more effectively characterized through more frequent sampling. Impacts from certain stresses may occur or be "worst-case" at specific times of the year, and it may be important to provide adequate documentation of the biological condition during these times. EPA's Science Advisory Board (SAB) suggests that sampling should — at a minimum — include the major components of the fall-winter and spring-summer (or wet season-dry season) community structure. The Florida Department of Environmental Protection has instituted a program that encompasses sampling during two index periods that correspond to this approach.

If some fish and invertebrate life cycles (e.g., spawning, growth, migration, and emergence) cause marked seasonal changes in stream assemblages, then each sampling season will require a separate reference database, metrics, and biocriteria. When such multiple index periods are used, the operational costs, at least initially, may be considerably higher than if surveys were conducted only once a year. Therefore, states must weigh their needs and the long-term value of this information against these costs. Seasonality must always be considered, and where possible, year-round data should be developed even if it has to be phased in slowly over time and as budgets allow.

The alternative, a single index period, will be deficient; it will not document spills or other single episode or transitory events including stresses that take place in other seasons. It should be selected only if seasonality is not a factor in the program objectives. Still, the major or initial applications of state biocriteria are likely to be assessment and management planning related to chronic habitat alteration and point and nonpoint sources. Such chronic stress impacts are more efficiently assessed with a single index period approach. Resident fish and benthic invertebrate assemblages integrate stress effects over the course of a year, and their seasonal cycles of abundance and taxa composition are fairly predictable within the limits of interannual variability. Single season indexing also represents a cost savings compared to seasonal or more frequent sampling.

State managers must choose the approach most appropriate to their needs and budgets.

Given these considerations, state managers must choose the approach most appropriate to their needs and budgets. They must avoid the temptation to spread multiseason sampling so thin that neither seasonal measurements nor indexing are properly achieved. It is better to do a single index period well than to do two poorly. Presuming, therefore, that most states will initially design their biological criteria programs around single season surveys, the following discussion emphasizes index period designs.

The optimal biological sampling period will be consistent with recruitment cycles of the organisms from reproduction to emergence and migra-

The optimal biological sampling period will be consistent with recruitment cycles of the organisms from reproduction to emergence and migration, such that the maximum amount of information can be derived from the data.

tion, such that the maximum amount of information can be derived from the data. Optimal conditions for biological sampling can be defined as that period of time during which the target assemblages have stabilized after larval recruitment and subsequent mortality and the use of their niche space is at its fullest. Where necessary, a compromise between biologically optimal conditions and water and flow conditions appropriate for the sampling gear must be made. Therefore, selection of the sampling period should be based on efforts to

- minimize between-year variability resulting from natural events,
- maximize gear efficiency, and
- maximize target assemblage accessibility.

Field collections scheduled to correspond to the optimal biological sampling period provide the most accurate assessment of community response to adverse conditions over an annual cycle. Sampling during these periods may not be logistically feasible, however, as a result of adverse weather conditions, staff availability, scheduling constraints, or other factors. The nature of the suspected stressor is an especially important consideration. An agency may be required to perform biological sampling during periods of greatest environmental stress, such as low flow and high temperature periods for point source discharges or high flow and runoff periods for nonpoint source discharges.

Although an estimate of aquatic community structure during optimal biological conditions should reflect the effect of, or recovery from, environmental stress periods (Ohio Environ. Prot. Agency, 1987), assessment of worst-case conditions may be needed under certain permitting regulations or as a follow-up to sampling during biologically optimal periods in which impairment was detected.

Ecological conditions and, thus, optimal sampling periods, vary seasonally as a result of regional climate patterns and the life cycles of the biota. Seven major climatological regions are represented within the contiguous United States (Fig. 4-3). The primary influence of seasonal changes in temperature and rainfall on stream biota is on biological processes (e.g., production, growth, reproduction, distribution, and locomotion). The level of biodiversity may also change seasonally. Even within an ecological region, some scaling of the optimal collection period may be necessary, depending on the elevation of the site, the habitat type, and other broad environmental variables.

Temperature and rainfall are the principal weather factors influencing the selection of sampling protocols and timing. Sampling will be impossible in frozen streams or during extreme high flows. Even subtle changes in temperature and flow may preclude certain kinds of sampling by affecting the equipment or the distribution of target assemblages.

The purpose of the biological sampling program (trend monitoring, special studies) also influences the sampling protocol. Special studies may be conducted at any time depending on need; but trend monitoring studies will focus on annual sampling events with varying sampling frequencies. The most appropriate season for such collections is determined by considering all technical and nontechnical factors. Technical factors include the selected assemblage, recruitment cycles, and severity of degra-

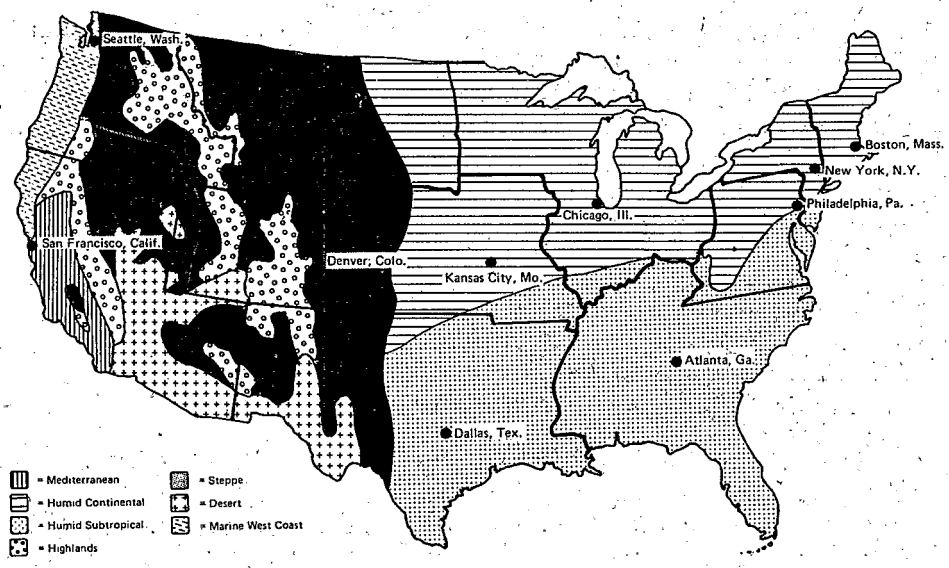

Figure 4-3.—Classification of U.S. climatological regions.

dation or contamination; nontechnical factors include such matters as logistics and personnel. From a practical standpoint, many states may select a sampling period that includes the summer and early fall months.

The investigator must carefully define the objectives of a monitoring program before these design issues can be resolved. Will specific questions be answered by sampling during periods of optimal biological condition or during periods of maximum impact? (These two periods may coincide.) Seasonal considerations are important because community taxonomic structure and the functional composition of some assemblages undergo natural changes in each season and annual cycle.

Natural cycles may also be influenced by chemical or physical alterations. From the traditional perspective of evaluating pollution impacts, summertime low flow conditions are often chosen to assess effects from point source discharges. Low flow conditions capture the effects of minimal effluent dilution in combination with the natural stressors of low water velocity and high temperature. Minimal effluent dilution occurs in summer because the lower quantity of water decreases the ability of the receiving waters to reduce the concentration levels of discharged compounds.

The effects of nonpoint source pollution on the aquatic community are evaluated during the recovery period following high flow because these effects are largely driven by runoff in the watershed. Nonpoint source loadings are estimated using samples collected during periods of high flow. Their actual effects, however, should be based on sampling outside the flow extremes. The effect of regulated and minimum flows are a particular problem during the winter season in the western United States. Regulated flows are a function of anthropogenic activity, usually associated with dams and reservoirs. Sampling activities should be avoided during high and low extremes.

Special studies conducted by state agencies in response to specific regulatory requirements or catastrophic events (e.g., oil spills) may not occur in an optimal season. In these situations, the data should be inter-

Special studies may be conducted at any time depending on need; but trend monitoring studies will focus on annual sampling events with varying sampling frequencies.

preted through concurrent reference data or through a seasonal adjustment to established reference data. If base biocriteria are established for a reference database for a single season, then data collected from the test sites during this season are directly comparable.

Two options are available for collections at test sites during seasons other than that used for base criteria. First, selected reference stations can be sampled concurrently with the test sites to provide baseline comparisons for data interpretation. Criteria established during the optimal season represent a range of values that can be extrapolated to other seasons. In this manner, a percentage of the reference may be acceptable as an alternate criterion.

The selection of an appropriate sampling season depends on the seasonal attributes of the aquatic community, but the administrative issues of sampling efficiency, safety, regulatory requirements, and appropriate metrics for data analysis are equally significant.

The second option may be to develop adjustments for an annual cycle. This can be done through seasonal collections of the reference database to document natural seasonal variation. Alternatively, a knowledge of seasonal appearance and disappearance of particular forms can be used to develop adjustments.

This discussion has focused on the seasonal attributes of the aquatic community. The administrative issues of sampling efficiency, safety, regulatory requirements, and appropriate metrics for data analysis are equally significant and must also be considered in light of the sampling objectives. The following paragraphs consider the sampling protocol in relation to the seasonal attributes of benthic, periphyton, and fish assemblages.

Benthos

Maximum information for a benthic community is obtained when most of its populations are within a size range (later instars) that can be retained during standard sieving and sorting and be identified with the most confidence. Reproductive periods and different life stages of aquatic insects are related to the abundance of particular food supplies (Cummins and Klug, 1979). Peak emergence and reproduction typically occur in the spring and fall, although onset and duration vary somewhat across the United States. During peak recruitment of the young, approximately 80 percent are too small to be captured in sufficient numbers to characterize the community accurately, and the food source requirements for early instars may be different from those for later instars. Therefore, the biologically optimal sampling season occurs following the period of initial recruitment and high mortality of young, and when the food resource has stabilized to support a balanced indigenous community.

The comparative time frames for sampling the benthic community are illustrated in Figure 4-4. The seasonal timetable shows annual high and low flow periods, emergence peaks for aquatic insect communities, and biologically optimal sampling periods (BOSP) for a stream in the New England region. High and low flow correspond to periods of high and low rainfall and associated runoff. Emergence is triggered by average daily temperature and photoperiod and usually occurs at peak intervals in spring and fall. The biologically optimal sampling period falls between the peaks in late winter and late summer and occurs after the population has been exposed to two-thirds of the aquatic phase of the organism's life cycle measured in degree days (that is, in units calculated as the product of time and temperature over a specified interval).

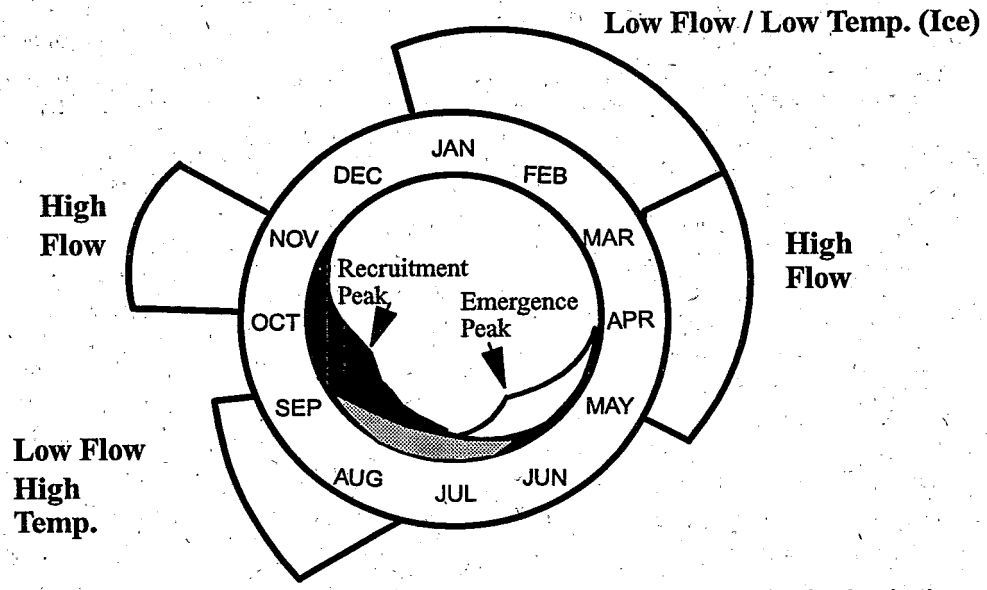

Figure 4-4.—Biological and hydrological factors for sampling period selection in the Northeast (macroinvertebrates). The gray area is the overlap between emergence and recuitment.

In this example (Fig.4-4), sampling in July and early August satisfies most of the criteria for collecting a representative sample at a time of significant chemical contaminant stress. It should be noted that chronic nonpoint source impacts such as sedimentation will be reflected in the quality of the benthic community after flow has returned to near normal following high flow conditions.

In the context of a single population, seasonality may be a significant factor. The early instars are small and difficult to identify, and the young nymphs have a generalized feeding strategy of collecting and scavenging. Only in later instars does feeding specialization occur and the quality of the food source become reflected in the condition of the population. In the case of *Stenonema,* the middle and late instars specialize as scrapers. Scrapers are often considered a pollution sensitive functional feeding group because their food source — diatom algae — responds to the early effects of pollution within the stream.

Periphyton

Periphyton assemblages are associations of algae, bacteria, and fungi that colonize the substrates in a stream. For purposes of bioassessment, most periphyton evaluations focus on diatom algae. The periphyton assemblage exhibits different seasonal abundance patterns than fish or benthos. The key difference is that periphyton assemblages are sufficiently abundant to be collected year-round from streams in temperate zones. Their biologically optimal sampling period may be based on relatively stable conditions but must also account for the comparison of diatom assemblages within similar stages of seasonal succession.

The limiting factors for diatoms are light, temperature, nutrients, water velocity, grazing, and interactions among algae via metabolites. Obviously, the abiotic factors go through an annual cycle of change and, like benthos, the assemblage composition shifts as the changing conditions fa-

vor new species. This process of seasonal succession creates significant seasonal differences in periphyton assemblages that must be considered in developing a study design. Besides changes in periphyton species composition, additional seasonal issues must be controlled to compare collections among sites and annual trends. Two major considerations are (1) the differences in biomass related to light and temperature regimes and (2) the comparisons of periphyton assemblages that have been subjected to heavy rains and scour with those that have matured under more stable hydrological conditions. Differences in light and temperature regimes may reflect human influences, for example, alterations of the stream channel and removal of riparian vegetation.

Fish

Like periphyton and benthic invertebrates, the fish fauna at a site is likely to vary seasonally. In the Northwest, for example, annual spawning migrations of anadromous salmonids set in motion a seasonal cycle of major importance to the biota. Seasonal migrations of fish are less striking but common in other areas as well. Most frequently, fish movements involve upstream movements in search of spawning areas to serve as nesting and nursery areas for young fish. Upstream areas often provide richer food supplies and lower predation rates than downstream areas.

Because of geographic variation in flows and temperatures, no general pattern occurs across all regions. A seasonal timetable representative of physical conditions and fish assemblage activities in the New England region is illustrated in Figure 4-5. Unless the sampling objective includes the study of unusual flow conditions and concurrent biotic responses, field sampling protocols should avoid extreme flow conditions (low or high) that may represent unusual stress, assemblage instability, or result in danger to field crews.

Sampling in several regions of the country has demonstrated that optimal fish sampling periods can be defined with relative ease. Generally, sampling periods should follow the spring spawning migrations that coin-

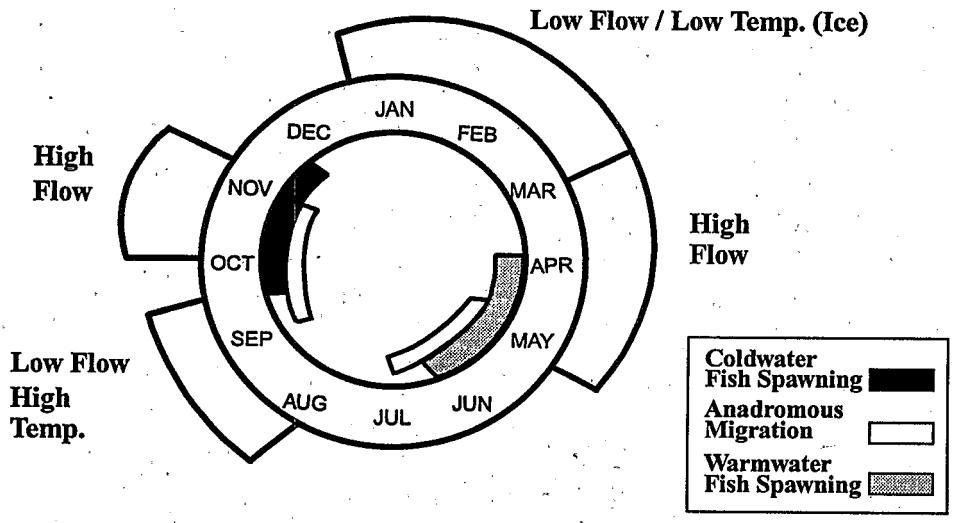

Figure 4-5.—Biological and hydrological factors for sampling period selection in the Northeast (fish).

cide with periods of high flow. Most states in eastern North America select the summer period for sampling (June through August) to coincide with periods of low to moderate stream flow and avoid the variable flow conditions of early spring and autumn (Karr et al. 1986). Fish assemblages during summer are relatively stable and contain the full range of resident species, including all major components of age-structured populations. Angermeier and Karr (1986) have outlined sampling rationale, including the merit of excluding young-of-the-year (YOY) from spring and late summer samples. This exclusion reduces variability and the problem of identifying and sampling very small fry. Excluding YOY from most analyses improves reliability and does not weaken the interpretation of the system's condition.

The scenario presented in Figure 4-5 identifies high and low flow periods in early spring and late summer for streams in the northeastern United States. The number of species is likely to peak in the spring with the spawning migration; the number of individuals will peak in the early autumn with the addition of YOY. The biologically optimal sampling period (BOSP) corresponds to seasonal effects within the fish assemblage and to the flow dynamics that influence sampling efficiency. Because the physical condition of the streams affects the efficiency of fish sampling gear, it also affects the nature or quality of the resulting data. For example, the effectiveness of passive equipment (e.g., trap nets) can be substantially reduced during periods of high or low flow, and the efficiency of active equipment (e.g., electrofishing gear) is reduced by turbidity, water temperature, and conductivity.

Sampling can typically begin in May or June in most areas and proceed into September unless unusually low flow periods occur during late summer drought. The probability that low flow periods will occur in late summer increases in watersheds that have been severely modified by urbanization or agricultural land use, in which case low flow sampling should be avoided.

Decisions about which habitats to sample are critical to the success of a biocriteria program.

Selection of Habitat for Aquatic Assemblage Evaluations

Stream environments contain a number of macro-and microhabitat types, including pools, riffles, and raceways, or surface and hyporheic zones. The latter refers to regions of saturated sediment beneath or beside the stream (Lincoln et al. 1982). Larger rivers have even more complex habitat configurations. Because no single sampling protocol can provide accurate samples of the resident biota in all habitats, decisions about habitats are critical to the success of a biocriteria program. These decisions are usually made in concert with the decision about the assemblages to be sampled, the sampling methods to be used, and the seasonal pattern of sampling.

Selection of habitats for sampling may be influenced by institutional requirements, such as sampling and analysis protocols that are part of an existing monitoring program, or the need to develop data that are consistent with a historical database; however, historical approaches should not be retained without careful evaluation of their ability to provide the data necessary to make informed resource decisions in future years.

Periphyton, invertebrates, and fish species in a stream vary in their distribution among major habitats. Depending on the data quality objectives established for the specific project or program, one or more assem-

blages may be targeted for inclusion in biosurvey activities. Attributes of several potential assemblages and their several advantages were described earlier in this chapter.

A major consideration in the development of bioassessment procedures is whether sampling all habitats is necessary to evaluate biological integrity or whether selected habitats can provide sufficient information. The selection of single habitat over multiple habitat, or vice versa, influences study design and may influence selection of the biotic assemblage to be sampled. Some taxa include individuals whose mobility or natural spatial distribution requires multiple habitat sampling.

A major consideration in the development of bioassessment procedures is whether sampling all habitats is necessary to evaluate biological integrity or whether selected habitats can provide sufficient information.

Generally, fish sampling reduces the need to make more detailed habitat decisions because most fish in small to medium rivers can be sampled using seines or electrofishing methods that efficiently sample all major surface water habitats except hyporheic zones and bank burrows. By sampling the full diversity of stream habitats for fish, the importance of fish movements among microhabitats for resting and foraging is reduced. Efficient sampling of all local habitats limits the problem of correcting evaluations of taxa in case the intensity of sampling varies among the range of available habitats.

Habitats to be sampled for periphyton require different analytical approaches. For example, periphyton assemblages may develop more easily on rigid or hard substrates. Though periphyton can grow on the leaves and stems of macrophytes, more prolific growths are generally seen on the hard surfaces of large substrate particles (e.g., cobble or small boulders). Steinman and McIntire (1986) found that substrate type is one of several characteristics that affect the taxonomic structure of lotic periphyton assemblages. Other factors are the dispersal and colonization rates of taxa in the species pool, competitive interactions, herbivory, chemical composition of the environment, and the character of ecological disturbances. Because it is difficult to remove or collect periphyton from natural substrates (Austin et al. 1981), hard surfaces (either natural or artificial) are usually the focus of sampling efforts. Most strategies for sampling periphyton assemblages are single habitat though other variables introduce additional complexity.

Benthic macroinvertebrates inhabit various habitats in lotic situations, for example, riffles, pools, snags, or macrophyte beds. Complete characterization of the assemblage requires a multihabitat and multisampling protocol such as that advocated by Lenat (1988). The benthic macroinvertebrate protocols for rapid bioassessment advocated by Plafkin et al. (1989) were developed for sampling the most productive and dominant benthic habitat in wadable streams. Consequently, riffles and cobble substrate were the primary focus of the rapid bioassessment protocols because that habitat is predominant across the country.

This approach works for small streams and streams that are dominated by riffles; however, it requires additional evaluation and technical development for use in other habitats. Plafkin et al. (1989) argue that the habitat where riffles predominate, will often be the most productive and stable habitat for the benthic community. The production of the habitat is related to provision of refugia, food resources, and necessary community interactions. It may be necessary to document the extent and character of the habitat because streams differ in these qualities, which differences may

be related to natural and anthropogenic causes. In some streams, riffles are not a dominant feature, and the emphasis on them may be misleading.

Since the issuance of the Rapid Assessment Protocols (RBPs) in 1989, rapid assessment techniques have evolved to focus on sampling of more than one habitat type, usually in the proportion of their representation at the sites of interest. These techniques have been primarily designed for low gradient streams (Mid-Atlantic Coastal Streams Workgroup, 1993; Florida Dep. Environ. Prot. 1994) and encompass the sampling of four or five habitat categories.

The sampling of a single habitat type (e.g., riffles or runs) is intended to limit the variability inherent in sampling natural substrates and to enhance the evaluation of attributes in an assemblage that will vary substantially in various habitats. Double, composited square meter kick net samples (2 m^2) are used in RBPs to collect large representative samples from riffle or run areas. Other gear can also be used to collect such composite samples.

Multihabitat sampling allows the evaluation of a broad range of effects on the benthic assemblage. However, it may also introduce variability into comparisons of the benthic assemblage among sites. Multihabitat investigations of water resource integrity are potentially confounded by (1) the absence of a particular habitat at a station, and (2) the potential differences in the quality and quantity of a habitat. As more habitats are sampled, the more difficult it is to control for comparable habitat among sites; and the absence of a habitat type at one or more stations exacerbates the problem. However, some states, such as North Carolina, have been successful in using a multihabitat sampling approach and advocate this technique as being more appropriate than simply sampling the riffle or run (Lenat, 1988).

A case study in association with the North Carolina Department of Environmental Management addressed the issue of sampling strategy and indicated that the riffle assemblage and the multihabitat assemblage responded similarly to differences among stations (Plafkin et al. 1989). For example, under stress, taxa richness was reduced by the same proportion in both the riffle and the multihabitat assemblage samples at a given station. These responses suggest that either the riffle assemblage or the multihabitat assemblage can be used to assess biotic integrity in streams in which riffles are prevalent.

Kerans et al. (1992) examined patterns of variability and the contribution of pool versus riffle invertebrate samples to the evaluation of biotic integrity and the detection of different kinds of degradation. They evaluated over a dozen attributes of the invertebrate assemblages including numbers of species (total and for a number of taxa) as well as several ecological classifications. At least eight attributes exhibited spatial or temporal trends, or both, depending on whether the habitat was pools or riffles. Attributes that were temporally and spatially unpredictable included some that are most commonly used in stream bioassessment. Kerans et al. conclude that measures of human impact on biotic integrity may be biased if sampling is restricted to only one habitat.

The choice of sampling habitats also entails a choice of sampling methods.

The choice of sampling habitats also entails a choice of sampling methods because conventional sampling methods for invertebrates vary in their efficiency among habitats. Surber and Hess samplers are used for riffles, while grab samplers are used most efficiently in the soft substrate of

pool habitats. Several forms of net samplers have been developed for various stream habitats: kick nets or seines (Plafkin et at. 1989; Lenat, 1988), D-frame nets (Montana Dep. Health Environ. Sci., 1990), and slack (rectangular frame) samplers (Cuffney et al. 1993). Passive colonization-dependent samplers (e.g., Hester-Dendy samplers) may also be used for evaluation of invertebrate assemblages (Ohio Environ. Prot. Agency, 1987).

In either the single habitat or multihabitat approach, the most prevalent and physically stable habitat that is likely to reflect anthropogenic disturbance in the watershed should be chosen.

Substrate Choices

In either the single habitat or multihabitat approach, the most prevalent and physically stable habitat that is likely to reflect anthropogenic disturbance in the watershed should be chosen. These habitats will vary regionally because of differences in topography, geology, and climate. The biological community in a particular stream may also change in response to increasing stream size (Vannote et al. 1980). The key to sampling, pertinent to benthic invertebrate surveys, is to select the habitats that support a similar assemblage of benthos within a range of stream sizes. Habitats that have been used for benthos are riffles, snags, downed trees, submerged aquatic vegetation, shorezone vegetation, and sediments, such as sand, silt, or clay (Table 4-2).

The habitat with the most diverse fauna is emphasized by most investigators because it offers the highest probability of sampling the most sensitive taxa. Riffles usually fit this criterion, and when present, are preferred. This habitat type is followed by hard, coarse substrates, snags, aquatic vegetation, and soft substrates. If multiple habitats are selected, similarity in habitat quality and comparable levels of effort among sampling sites must be considered.

The habitat with the most diverse fauna is preferred — riffles followed by hard, coarse substrates, snags, aquatic vegetation, and soft substrates.

Natural and Artificial Substrates

Most benthic surveys employ direct sampling of natural substrates. This method is particularly important if habitat alteration is suspected as the cause of impairment. A major assumption is that every habitat has a biological potential, which is reflected in the resident biotic community. Be-

Table 4-2.—Common benthic habitats.

SNAGS/DOWNED TREES	SHOREZONE VEGETATION
• Productive in blackwater streams (Benke et al. 1984)	• Present in most streams
• Diversity of epifauna	• Measures riparian impacts
• Community dependent on well-prepared substrate	• Dominated by shredders and collectors
	• May be seasonal
SUBMERGED AQUATIC VEGETATION	**SILT/MUD**
• Productive in coastal zones	• Pool communities
• High standing crop	• Dominated by fauna
• Seasonal habitat	• Sediment quality and water quality effects
• Snails usually abundant	• Fauna usually tolerant to low oxygen
SHIFTING SAND	**LEAF LITTER/DEBRIS**
• Prevalent in erosional areas	• Prevalent in forested streams
• Dominated by opportunistic infauna	• Measures riparian impacts
• Sediment quality and water quality effects	• Dominated by shredders
• High dominance by monotypic fauna	• Microbial preparation of substrate

cause interpretation depends on the level of assemblage development within the existing habitat, sampling natural substrates is recommended. If, however, an artificial substrate can be matched to the natural substrate (e.g., using a rock basket sampler in a cobble substrate stream), then such artificial substrates may also be used (Sci. Advis. Board, 1993). Maine uses this rock basket approach. The Ohio EPA biocriteria program (Ohio Environ. Prot. Agency, 1987) has successfully used Hester-Dendy multiplate artificial substrate samplers supplemented by qualitative, natural substrate samples to assess biological integrity using benthic assemblages.

The advantages and disadvantages of artificial substrates (Cairns, 1982) relative to natural substrates are the following:

■ **Advantages of Sampling with Artificial Substrates**

1. Enhances sampling opportunities in locations that are difficult to sample effectively.
2. Permits standardized sampling by eliminating subjectivity in sample collection technique.
3. Minimizes confounding effects of habitat differences by providing a standardized microhabitat.
4. Directs the interpretation to specific water quality questions without interference of habitat variability.
5. Increases the ease of placing samplers in discrete areas to discriminate impacts associated with multiple dischargers.

■ **Disadvantages of Sampling with Artificial Substrates**

1. Requires the investigator to make two trips for each artificial substrate sample (one to set and one to retrieve).
2. Measures colonization potential rather than resident community structure.
3. Allows problems such as sampler disturbance and loss to occur.
4. Complicates interpretation of the effects of habitat structure.

If artificial substrates are selected, the surface area of the materials should be standardized among units. Introduced substrates, in the context of biological monitoring, are artificial substrates that are constructed to match natural bottom materials at the site of the survey. An example of introduced substrates are rock baskets, such as those used by Maine (Davies et al. 1991), in which baskets that contain rocks native to the region of known surface area are partially buried in the bottom sediment. Where possible, the use of introduced substrate is preferable to other types of artificial substrate as recommended by the SAB (1993). Rock baskets or other substrates should be placed in waters of similar depths, velocities, and daily sun and shade regimes.

Standardization of Techniques

Standard operating procedures should be adhered to in all phases of fieldwork, data analysis, and evaluation. Such standards are essential for maintaining consistency and comparability among data sets and for appropriate quality assurance and control (Kent and Payne, 1988; Klemm et al. 1990; Smith et al. 1988). Without standard operating procedures to mimic previous studies, the difficulties encountered in comparing temporal and spatial data or analytic results may be substantial. The inherent variability of the sampling process (Cairns and Pratt, 1986) can be reduced through standardization of sampling gear, gear efficiency, level of effort, subsampling methods, handling and processing procedures, and computer software. Standardization of project activities provides considerable strength in reducing, controlling, and understanding variability.

Standard operating procedures should be adhered to in all phases of fieldwork, data analysis, and evaluation. Such standards are essential for maintaining consistency and comparability among data sets and for appropriate quality assurance and control.

Sample Collection

A major influence on the comparability of field ecological projects is the type and intensity of appropriate training and professional experience for all personnel (Barbour and Thornley, 1990). Similar exposure to sampling methods and standard operating procedures can reduce the amount of variation from one sampling event or project to the next. Standardizing the equipment relative to operator efficiency, sampling effort, and the area to be sampled greatly affects data quality. Operator efficiency depends on the operator's experience, dexterity, stamina, and adherence to specified survey requirements. Physical habitat conditions at the time of sampling (e.g., flow levels, current velocity, and temperature) also influence efficiency. Active sampling efforts (e.g., using net samples or electrofishing) may be standardized as a function of person-hours spent at each sampling station and by tracking the physical area or volume sampled. Passive methods (e.g., artificial substrates, trap nets) may be standardized by tracking the person-hours and the exposure time. This choice is often dictated by the earlier selection of the assemblage to be sampled; for some, a relatively small selection of sampling techniques may be available. A certain sampling area or volume may be required to obtain an appropriate sample size from a particular community and to estimate the natural variability of that community at the sampling station.

Once the assemblage, sampling equipment, and method have been chosen, standard operating procedures can be written for field operations, including a clear description of the sampling effort to be applied during each sampling event. All employees should have this documentation, and new employees should be accompanied in the field by experienced staff until they are thoroughly familiar with all procedures (Ohio Environ. Prot. Agency, 1987).

Processing samples in the field requires several critical steps. Sample containers for benthic invertebrates and voucher fish should be marked with appropriate and complete information on internal and external labels. Other identifying information and descriptions of visual observations should be recorded in a field notebook.

Data on birds and mammals, which consist primarily of visual observations and for which accurate field taxonomy is possible, will not require subsequent processing in the laboratory. However, the details of each ob-

servation should be carefully recorded so that they may be checked later. Most fish sampling requires sorting, recording, and releasing the fish at the site of capture. Fish sampling crews should have a reference collection available in the field, and specimens should be collected and accurately labeled so that identifications can be confirmed.

Sample containers with preserved specimens should be assigned unique serial or identification numbers. These numbers should be recorded in a logbook along with the appropriate labeling information. All sample containers or specimens should be appropriately packaged for transportation and continued processing in the laboratory.

For assemblages in which extremely large numbers of individuals or associated substrate are obtained in each sample as is often the case with small fish, benthic macroinvertebrates, periphyton, or planktonic organisms, it may be impractical and costly to process an entire sample. In such cases, standardized random subsampling, similar to that recommended by Plafkin et al. (1989), is a valid and cost-effective alternative.

As a subsampling method is developed, every attempt must be made to reduce bias. Therefore, guidelines are needed to standardize the effort and to eliminate investigator subjectivity. Rapid bioassessment protocols, for example, maintain subsampling consistency by defining the mode (a gridded pan), by placing limitations on the mechanics of subsampling and the subsample size, and by assuring that the subsampling technique is consistently random.

Standardized random subsampling is a valid and cost-effective alternative to processing an entire sample. As a subsampling method is developed, every attempt must be made to reduce bias.

Sample Processing

The need for specialized training and expertise is most necessary during the identification of organisms. Unless the project objectives direct otherwise, each specimen should be identified to the most specific taxonomic level possible using current literature. Some techniques may require identification only to the ordinal, familial, or generic level (Ohio Environ. Prot. Agency, 1987; Plafkin et al. 1989), but the most accurate information on tolerances and sensitivities is found at the species level.

Nevertheless, taxonomic resolution should be set at a level achievable by appropriately trained state personnel. State water resource agencies should find it beneficial to establish collaborative working arrangements with local and regional experts who can provide training, technical support, and quality assurance and control. Stream ecology research over the last decade indicates that a specific minimal level of resolution should be set (i.e., the "lowest achievable taxonomic level" is not a helpful criterion) and that additional refinement should be left to individual state groups as their capabilities permit (Sci. Advis. Board, 1993).

The SAB further states that proposed levels of intensity and taxonomic resolution must receive a thorough evaluation by the scientific research community. For example, adult and juvenile fish should usually be identifiable by species (Sci. Advis. Board, 1993). The identification of larval fish may provide useful information; however, it may only be feasible to identify them to the generic or familial levels. Reasonable candidate levels for stream macroinvertebrates are given in Table 4-3.

Once the samples have been analyzed (identified, enumerated, and measured), reference (voucher) material should be placed in the well-estab-

Table 4-3.— Proposed minimal levels of taxonomic resolution for stream macroinvertebrates (taken from Sci. Advis. Board, 1993).

TAXONOMIC LEVEL	GROUPS
Genus	Plecoptera (in part), Ephemeroptera, Odonata, Trichoptera, Megaloptera, Neuroptera, Lepidoptera, Coleoptera (in part, larvae and adults), Hemiptera, Diptera (Tipulidae and Simulidae), Crustacea, Mollusca
Tribe	Chironominae
Subfamily	Chironomidae
Family	Diptera (other than Chironomidae, Tipulidae and Simulidae), Oligochaeta, Plecoptera (in part), Coleoptera (in part)
Order	Other noninsect groups

lished network of federal, state, and university museums for *regionally centralized curation* (Sci. Advis. Board, 1993). This action ensures a second level of quality control for specimen identification. Preferably, collection and identification of voucher specimens will be coordinated with taxonomic experts in regional museums. These repositories, which have always been the centers for systematics, should continue to be used for this function (Sci. Advis. Board, 1993). The SAB recommends that once the information on the samples has been entered into a database and verified, the repository institutions should be encouraged to conduct additional systematic studies on the material. Information from these additional analyses can then be made available to state biocriteria programs.

All identifications should be made using the most up-to-date and appropriate taxonomic keys. Verification should be done in one of two ways: (1) by comparison with a preestablished reference or research specimen collection, or (2) by having specimens confirmed by taxonomic experts familiar with the group in question (Borror et al. 1989). A regional consensus of taxonomic certainty is critical to ensure that the results are comparable both spatially and temporally. The taxonomists should always be contacted by telephone or mail before any specimens are sent to their attention. It is also important to follow their advice on the proper methods for packing and shipping samples. Damaged specimens may be useless and impossible to identify.

Suggested Readings

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CHAPTER 5.

Evaluating Environmental Effects

Purpose:
To provide managers with an understanding of the factors that affect and determine water resource integrity.

Should a biological survey reveal a significant departure from reference conditions or criteria, the next step is to seek diagnostic information leading to remedial action. This action entails the investigation of an array of physical, chemical, and biological factors to determine the likely source of degradation in the water resource.

Five major environmental factors affect and determine water resource integrity (Karr and Dudley, 1981; Karr et al. 1986). These factors are water quality, habitat structure, flow regime, energy source, and biotic interactions. Monitoring programs must integrate, measure, and evaluate the influences of these factors (Fig. 5-1). A comprehensive discussion of all five and the enormous variety of human actions that alter them is beyond the scope of this document. We can, however, present a conceptual sketch of each one and how it influences the integrity of the water resource. Several considerations are involved in evaluating these complex factors.

Human actions often alter one or more of those factors and thus alter the resident biota. Alterations may be obvious, such as the extinction of species or the introduction of exotics, or they may be more subtle, such as altered survival rates, reproductive success, or predation intensity. Protection or restoration of biotic integrity requires identification of the processes that have been altered by human actions. Careful evaluation of the conditions in a watershed can play a critical role in identifying the potential causes of degradation. That identification process is essential to develop the most cost-effective approaches to improving the quality of water resources.

Water Quality

The physical and chemical attributes of water are critical components of the quality of a water resource. Because the earliest water resource legislation (e.g., the Refuse Act of 1899) dealt with disease and oil pollution in navigable waters, emphasis has traditionally been on the physical and chemical properties of water. Physical and chemical attributes of special concern include but are not limited to temperature, dissolved oxygen, pH, hardness, turbidity, concentrations of soluble and insoluble organics and inorganics, alkalinity, nutrients, heavy metals, and an array of toxic substances. These substances may have simple chemical properties, or their

ECOLOGICAL IMPACT OF HUMAN-INDUCED ALTERATIONS

1. Energy Source

- Type, amount, and particle size of organic material entering a stream from the riparian zone versus primary production in the stream
- Seasonal pattern of available energy

→ *Decreased coarse particulate organic matter*
Increased fine particulate organic matter
Increased algal production

2. Water Quality

- Temperature
- Turbidity
- Dissolved oxygen
- Nutrients (primarily nitrogen and phosphorus
- Organic and inorganic chemicals, natural and synthetic
- Heavy metals and toxic substances
- pH

→ *Expanded temperature extremes*
Increased turbidity
Altered diurnal cycle of dissolved oxygen
Increased nutrients (especially soluble nitrogen and phosphorus)
Increased suspended solids

3. Habitat Structure and Quality

- Substrate type and quantity
- Water depth and current velocity
- Spawning, nursery, and hiding places
- Diversity (pools, riffles, woody debris)

→ *Decreased stability of substrate and banks due to erosion and sedimentation*
More uniform water depth
Reduced habitat heterogeneity
Decreased channel sinuosity
Reduced habitat area due to shortened channel
Decreased instream cover and riparian vegetation

4. Flow Regime

- Water volume
- Temporal distribution of floods and low flows
- Flow regulation

→ *Altered flow extremes (both magnitude and frequency of high and low flows)*
Increased maximum flow velocity
Decreased minimum flow velocity
Reduced diversity of microhabitat velocities
Fewer protected sites

5. Biotic Interactions

- Competition
- Predation
- Disease
- Parasitism

→ *Increased frequency of diseased fish*
Altered primary and secondary production
Altered trophic structure
Altered decomposition rates and timing
Disruption of seasonal rhythms
Shifts in species composition and relative abundance
Shifts in invertebrate functional groups (increased scrapers and decreased shredders)
Shifts in trophic guilds (increased omnivores and decreased piscivores)
Increased frequency of fish hybridization

Figure 5-1.—Five major classes of environmental factors that affect aquatic biota in lotic systems. Right column lists selected expected results of anthropogenic perturbation (Karr et al. 1986).

dynamics may be complex and changing, depending on other constituents in a particular situation including the geological strata, soils, and land use in the region. The number of elements and compounds that influence water quality is very large without human influences; with them, the complexity of the problem is even greater. The human effects on biological processes may be direct (i.e., they may cause mortality), or they may shift the balance among species as a result of subtle effects, such as reduced reproductive rates or changing competitive ability. Aquatic life use designations provide protection at various levels from the multitude of anthropogenic effects.

The EPA encourages states to fully integrate biological surveys, whole-effluent and ambient toxicity testing, and chemical-specific analyses to assess attainment or nonattainment of designated aquatic life uses in state water quality standards (U.S. Environ. Prot. Agency, 1991c). Ohio EPA used numeric biological criteria within an existing framework of tiered aquatic life uses to establish attainable, baseline expectations on a regional basis (Yoder, 1991). Use attainment status in the Ohio water quality standards results in a classification of "full attainment," if all applicable numeric biocriteria are met; "partial attainment," if at least one aquatic assemblage exhibits nonattainment but no lower than a "fair" narrative rating; and "nonattainment," if none of the applicable biocriteria are met, or if one assemblage reflects a "poor" or "very poor" narrative rating.

The EPA encourages states to fully integrate biological surveys, whole-effluent and ambient toxicity testing, and chemical-specific analyses to assess attainment or nonattainment of designated aquatic life uses in state water quality standards.

North Carolina's Department of Environment, Health, and Natural Resources has used in-stream biota to assess water quality since the mid-1970s (Overton, 1991), and the water quality regulations in the North Carolina code have been revised to take biological impairment into account. In addition, when fiscal realities in North Carolina required a more efficient water quality program, all NPDES permits within a given river basin were scheduled to be issued within the same year (Overton, 1991). The same strategy makes biological assessment more efficient because the department can focus the assessment on specific river basins coincident with the renewal permits. Other states may have to consider similar strategies to conserve resources.

The Maryland Department of the Environment, Water Quality Monitoring Division, uses biological assessment as part of a statewide water quality monitoring network (Primrose et al. 1991). Using biological assessment, Maryland has been able to differentiate among various degrees of impairment and unimpairment, and to distinguish particular water quality impacts.

The Arkansas Department of Pollution Control and Ecology developed a bioassessment technique in the mid-1980s to assess the impact on receiving waters of discharges exceeding water quality-based limits (Shackleford, 1988). Using its bioassessment approach as a screening tool, Arkansas follows a formal decision tree for assessing compliance with established water quality limits (Fig. 5-2). The initial bioassessment screen may result in the application of other biological, toxicological, or chemical methods. After completion of screening, an on-site decision can be made for subsequent action. In situations where "no impairment" or "minimal impairment" classifications are obtained, field efforts are reduced in frequency or intensity until further information indicates a problem. Streams classified as "substantially" or "excessively" impaired trigger additional

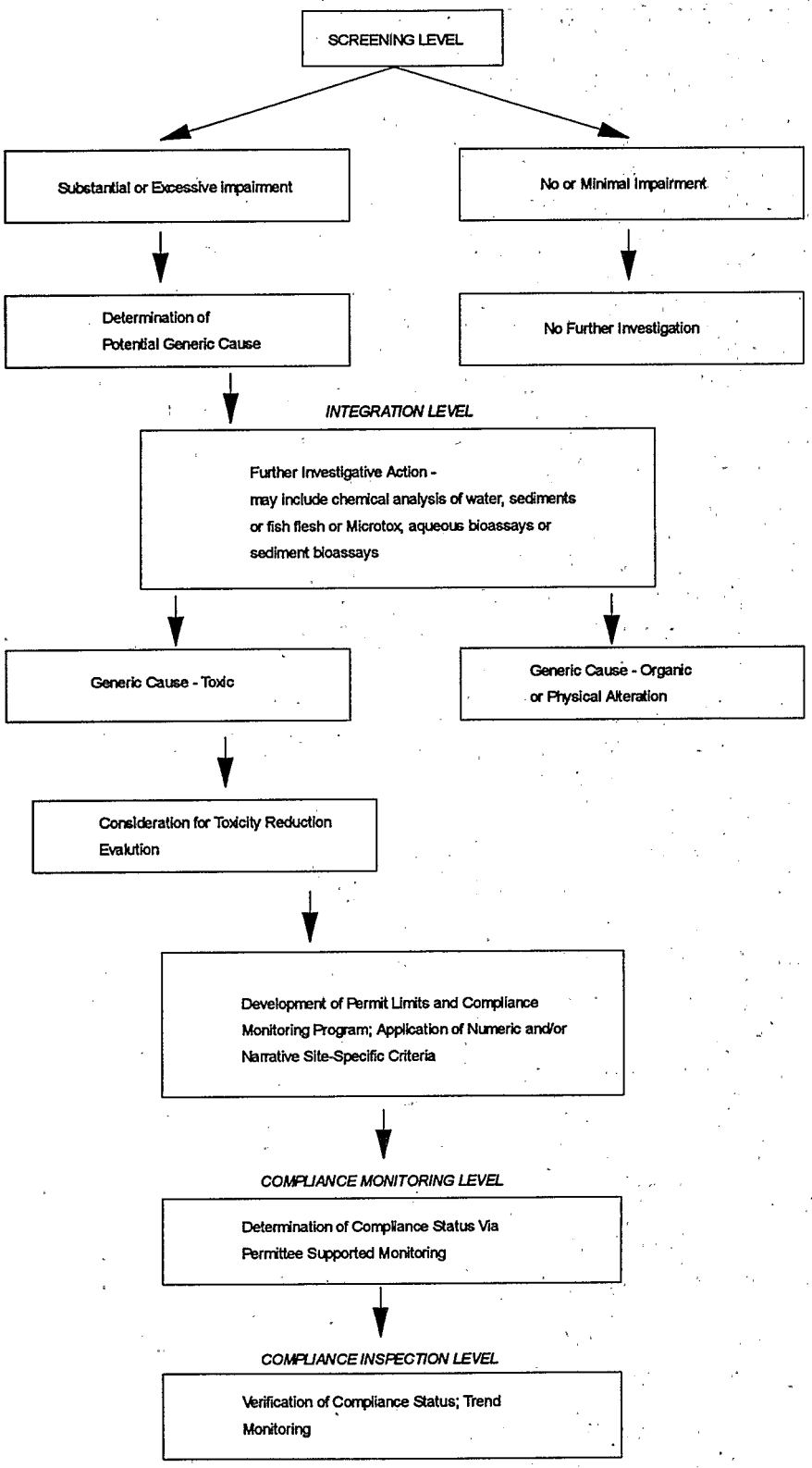

Figure 5-2.—Decision matrix for application of rapid bioassessments in Arkansas for permitted point source discharges (Shackleford, 1988).

investigative steps that employ an integration of methods (Shackleford, 1988).

The definitive evaluation of water quality impacts often requires expensive laboratory analyses. However, careful review of conditions in the watershed can provide early warning signals about the potential for water resource degradation. For example, the presence of industrial, domestic, and agricultural sources of chemical contaminants may be indicated by odors, froth, or colors in the water. These conditions should be noted during field surveys for their potential diagnostic value.

Careful review of conditions in the watershed can provide early warning signals about the potential for water resource degradation.

Habitat Structure

The physical structure of stream environments is critical to the ecological health or integrity of lotic water resources. Attributes of significance to organisms in streams are channel morphology including width, depth, and sinuosity; floodplain shape and size; channel gradient; in-stream cover such as presence of boulders and woody debris; substrate type and the diversity of substrates within a stream reach; riparian vegetation and the canopy cover that it provides; and bank stability.

Channel morphology in natural watersheds is typically meandering with substrate diversity created by varying velocities along and across the channel. As a result, substrates are sorted to form pools and riffles that create horizontal variation in the physical environment. If a channel has been artificially straightened and dredged (channelized), temporal recovery will recreate substrate diversity through vertical and lateral meandering processes (Hupp, 1992; Hupp and Simon, 1986). Because no stream channel is stable, a temporal dimension of diversity also exists. These physical attributes are closely tied to other environmental conditions and impairments (Table 5-1).

The influence of habitat structure spans the range from regional geography to the pattern of interstitial spaces between rocks in the river substrate. Habitat structure on all scales is critical to the biology of most stream organisms, and subtle or massive habitat alteration on any scale may influence the quality of the water resource.

The influence of habitat structure on the aquatic community causes natural variability even in undisturbed communities. Understanding the relationship of expected trends in biological condition as a result of changes in habitat structure is an important feature of biological assessments. Ohio EPA found that their measurement of habitat quality, the Qualitative Habitat Evaluation Index (QHEI), was significantly correlated with the Index of Biotic Integrity (IBI) in Ohio streams (Fig. 5-3) with $r = 0.47$ (Rankin, 1991) on a broad scale over the state. Rankin also found that stream habitat quality and land use at various geographic scales are important influences on fish assemblages and that relatively intact stream habitat throughout the drainage can compensate for short stretches of poor habitat. In contrast, however, habitat-sensitive species may be reduced or destroyed in stream basins with extensive degraded conditions, even if short stretches of good habitat exist. The Maryland Department of the Environment, using the relationship between habitat structure and biological condition, demonstrated effects from various influences (Fig. 5-4) including agricultural runoff, treatment plant effluent, channelization, and landfill operations (Primrose et al. 1991).

Table 5-1.— Parameters that may be useful in evaluating environmental conditions and their relationship to geographic scales and the environmental factors influenced by human actions.

CATEGORY BY GEOGRAPHIC SCALE	PARAMETER	ENVIRONMENTAL FACTORS[i,j]
1. Watershed	Land use[f] Flow stability[f]	Flow regime Physical habitat
2. Riparian and bank structure	Upper bank stability[a,f,h] Bank vegetative stability[a,f,h] Woody riparian vegetation[h] — species identity — number of species Grazing or other disruptive pressures[a,f] Streamside cover (% vegetation)[a,f] Riparian vegetative zone width[a,f] Streambank erosion[f]	Flow regime Energy base Physical habitat
3. Channel morphology	Channel alteration[a,d,f] Bottom scouring[a] Deposition[a] Pool/riffle, run/bend ratio[a,c] Lower bank channel capacity[a] Channel sinuosity[a,f,h] Channel gradient[f,h] Bank form/bend morphology[h]	Flow regime Energy base Biotic interactions Water quality Physical habitat
4. In-stream	Substrate composition/size; % rubble, gravel, submerged logs, undercut banks, or other stable habitat[a,c,d,e,f] % pools[f] Pool substrate characterization[a] Pool variability[a] % embeddedness of gravel, cobble, and boulder particles by fine sediment; sedimentation[a,c,f] Rate of sedimentation Flow rate[a,d] Velocity/depth[a,d,e] Canopy cover (shading)[a,f] Stream surface shading (vegetation, cliffs, mountains, undercut banks, logs)[b,d,f] Stream width[c,h] Water temperature[c]	Flow regime Energy base Biotic interactions Water quality Physical habitat

REFERENCES:

[a]Plafkin et al. 1989
[b]Platts et al. 1987
[c]Platts et al. 1983; Armour et al. 1983
[d]Rankin, 1991
[e]Gorman, 1988
[f]Osborne et al. 1991
[g]Barton et al. 1985
[h]Hupp and Simon, 1986; 1991
[i]Karr and Dionne, 1991
[j]Karr, 1991

An assessment of habitat structure is critical to any evaluation of ecological integrity. Habitat assessment provides information on habitat quality; it also identifies obvious constraints on the site's potential to achieve attainment, assists in the selection of appropriate sampling stations, and provides basic information for interpreting biosurvey results.

Habitat Quality and Biological Condition

The variability of environmental conditions directly affects patterns of life, population, and the micro- and macrogeographic distribution of organisms (Cooper, 1984; Price, 1975; Smith, 1974). An assessment of habitat structure is therefore critical to any evaluation of ecological integrity (Karr et al. 1986; Plafkin et al. 1989). Habitat assessment provides information on habitat quality; it also identifies obvious constraints on the site's potential to achieve attainment, assists in the selection of appropriate sampling sta-

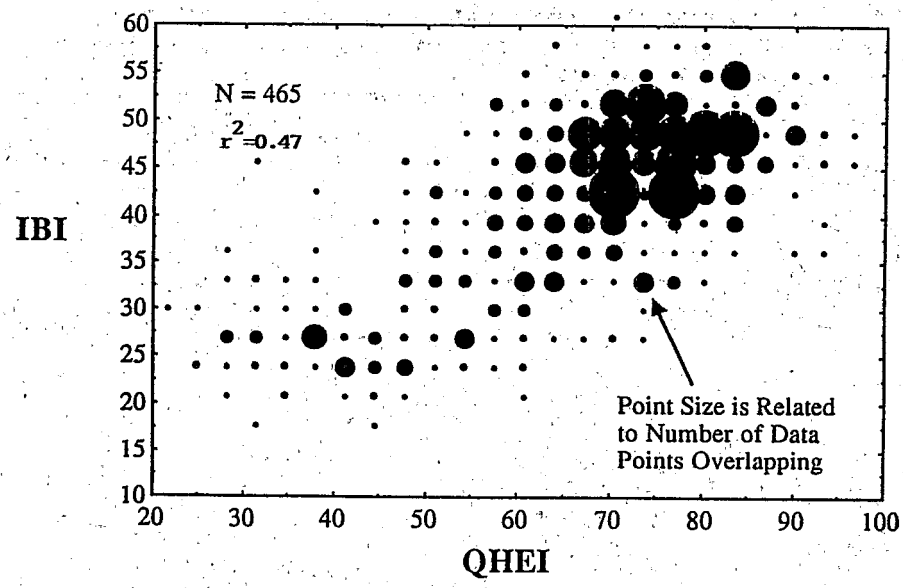

Figure 5-3.—Qualitative Habitat Evaluation Index (QHEI) versus the Index of Biotic Integrity (IBI) for 465 relatively unimpacted and habitat modified Ohio stream sites (Rankin, 1991).

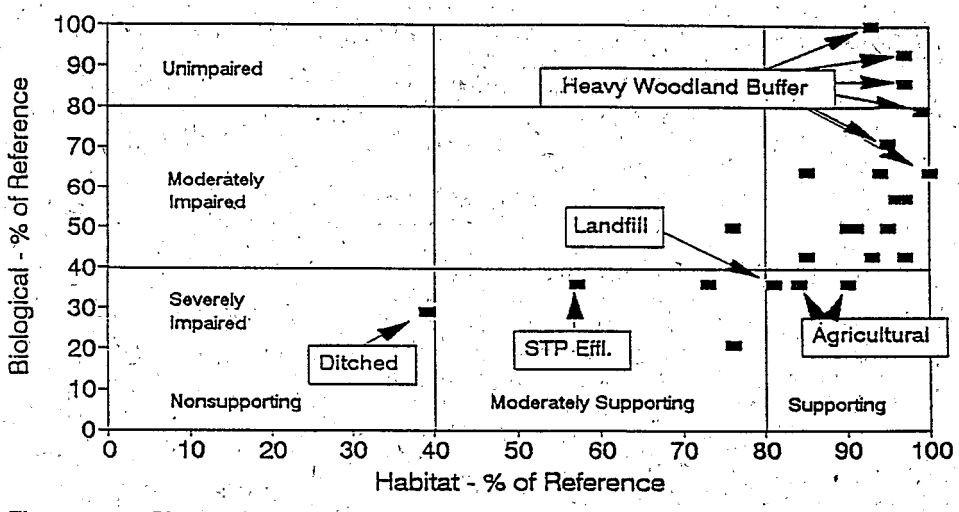

Figure 5-4.—Choptank and Chester rivers tributaries (Primrose et al. 1991).

tions, and provides basic information for interpreting biosurvey results (Atkinson, 1985; Osborne et al. 1991). A carefully conducted habitat evaluation is essential for distinguishing cause and effect elements from among the five environmental factors influenced by human activity.

Development of a Habitat Assessment Approach

The development of a stream habitat assessment approach follows a logical sequence beginning with the characterization of the waterbody. Only similar aquatic systems may be compared; habitat structural parameters applicable to one part of the country may not be applicable in another. For instance, the extent of canopy cover differs between forested mountain streams and open prairie streams found in the southwest. Thus, the absence of canopy cover is a more important habitat influence in a forested

Only similar aquatic systems may be compared; habitat structural parameters applicable to one part of the country may not be applicable in another.

The development of a stream habitat assessment follows a logical sequence.

Waterbody Characteristics
↓
Selection of the taxa (Benthic Macroinvertebrates, Fish)
↓
Influential Habitat Variables (Flow, Shade, Substrate, Buffer Zone)
↓
Judgment Criteria (Optimal, Suboptimal, Marginal, Poor)

stream than in open streams (Barbour and Stribling, 1991). Another consideration would be broad physiographic characteristics, for example, elevation, general topography and gradient, and predominant soil types. Finally, the biogeographic distribution of species and assemblages of organisms varies regionally.

Selection of the taxa, that is, the biological community to be studied, is the important next step. Ideally, this selection is based on the best approach to a comprehensive water resource assessment. However, the availability of resources and the training of available staff will have significant influence.

The selection of one or more assemblages is important for determining which habitat variables are most influential for community development. For each parameter, the range of conditions to be expected is determined and divided into scoring categories. These scoring categories (optimal, suboptimal, marginal, and poor) form the basis of criteria that allow habitats to be judged during on-site evaluation. An important call must then be made. If habitat structure is degraded relative to the expectations provided by the appropriate reference condition, some inference must be drawn about the nature and cause of the difference. If the study site is degraded relative to the reference, then habitat structure has been identified as a potential cause of reduced biotic condition. If habitat structural differences result from the natural landscape rather than human interference, then the possibility that an inappropriate reference condition was used must be considered.

Gradient is perhaps the most influential factor for segregating a lotic waterbody because it is related to topography and landform, geological formations, and elevation, which in turn influence vegetation patterns.

The habitat assessment approach outlined here (following Barbour and Stribling, 1991; Plafkin et al. 1989) is applicable to wadable streams and rivers. Because fish and benthic macroinvertebrates are the focal points of these recommended bioassessment procedures, habitat structural parameters were chosen that influence the development of these communities. Although streams across the country exhibit a wide range of variability, some generalizations can be made. Gradient is perhaps the most influential factor for distinguishing lotic waterbodies because it is related to topography and landform, geological formations, and elevation, which in turn influence vegetation patterns. Four generic stream categories related to gradient can be identified: mountain, piedmont, valley plains, and coastal plains. Several habitat attributes serve as a framework for assessing habitat quality:

- Substrate variety/in-stream cover
- Bottom substrate characterization/embeddedness
- Flow or velocity/depth
- Canopy cover (shading)
- Channel alteration
- Bottom scouring and deposition
- Pool to riffle and run to bend ratios, channel sinuosity
- Lower bank channel capacity
- Upper bank stability
- Bank vegetative stability (grazing or other disruptive pressure)
- Streamside cover
- Riparian vegetative zone width

While the investigator is on-site, the quality of each parameter can be assessed. First, numeric value from a scale based on a gradient of conditions is assigned to assess the quality of each parameter. Then, a composite of information from each parameter is compared to a reference condition. Such a quantified assessment of habitat structure provides a more meaningful interpretation of biological condition. Habitat assessment incorporates information on stream segments or reaches. However, a linear relationship between site-specific quality of habitat and community performance may not exist to the point that habitat structural condition can be used to "predict" biological performance with accuracy.

If habitat degradation has occurred, mitigation or improvement of the habitat through stream restoration activities should be evaluated. Implementation of water quality improvements can be independent of habitat quality, but judgment of the improvement in biological integrity cannot.

Implementation of water quality improvements can be independent of habitat quality, but judgment of the improvement in biological integrity cannot.

Flow Regime

Fluctuating water levels are an integral part of the stream ecosystem, and the biota are dependent on seasonal flow variation. High flow events are especially important in maintaining the habitat complexity of pools, riffles, clean substrates, and bars (Hill et al. 1991). Aquatic organisms have evolved to compensate for changing flow regimes, even periodic catastrophic flow conditions. High water periods are determined by the frequency, occurrence, and type of precipitation event as well as antecedent conditions such as soil moisture, time since last rain, and amount and type of soil cover. Dewatering the channel for major periods as a result of human actions is clearly a degradation of the water resource, but more subtle changes in the volume and periods of flow may have equally devastating effects on the resident biota.

Jones and Clark (1987) discuss the effects of urbanization on the fundamental hydrology of watersheds and the natural flow regime. Increases in impervious surface area (e.g., roads, parking lots) result in a substantial increase in the proportion of rainfall that is rapidly discharged from the watershed as direct runoff and streamflow. Such runoff increases the volume of flood flows and instances of channel instability. Leonard and Orth (1986) developed a cultural pollution index to evaluate the health of the fish community subject to the effects of road density, population encroachment, mining, and organic pollution. These effects have substantial influence on flow regime. Steedman (1988) also evaluated the condition of fish communities in heavily urbanized areas of Ontario. He found that certain attributes that are relatively sensitive to urbanization effects can serve as pertinent response signatures.

Fluctuating water levels are an integral part of the stream ecosystem, and the biota are dependent on seasonal flow variation.

Ohio EPA found that the presence or absence of channelization influenced the relationship between the quality of habitat structure and the condition of the fish community (Ohio Environ. Prot. Agency, 1990). In the absence of channelization, for example, Twin Creek and Kokosing River (Fig. 5-5) had high IBI values, even in the presence of sporadic degraded habitat. In these instances, the relatively good habitat quality throughout the watershed supported the fish community in short reaches of degraded, habitat (Rankin, 1991). In channelized lotic systems, for example, Tiffin River and Little Auglaize River (Fig. 5-5), the best habitats were de-

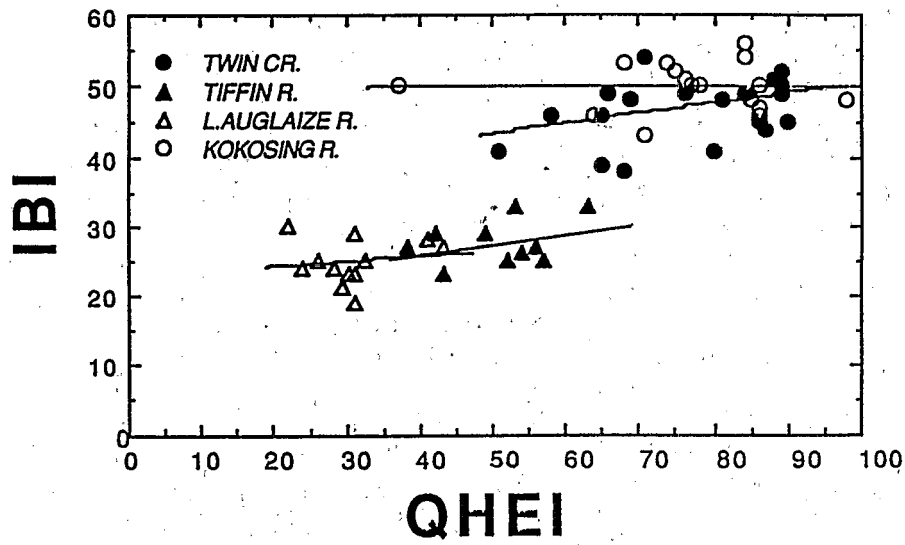

Figure 5-5.—Relationship of the Index of Biotic Integrity (IBI) to changes in the quality of habitat structure through the Qualitative Habitat Evaluation Index (QHEI) in channelized (triangles) and unchannelized (circles) (Ohio Environ. Prot. Agency, 1990).

graded and IBI scores remained essentially unchanged as the habitat was degraded further. The quality of habitat structure and the flow regime are intricately associated. In areas of extensive channelization, communities may consist only of generalists and opportunists able to withstand harsh flow conditions directly, or the secondary effects of those flow conditions (e.g., reduced abundance of food or presence of habitat refuges).

■ **Effects of Channelization.** Unchannelized or otherwise unmodified streams have normal, low-level, and mostly consistent rates of sediment deposition on the bed and low, convex banks. The channel usually has some degree of meandering, and the banks lose very little mass during either low or high flows.

Efforts to control flooding and to drain wetlands often involve channelization of streams to provide more rapid removal of water. Unfortunately, these activities create unstable channels with higher gradients and without meanders. Hydrogeomorphic processes tend to restore the dynamic stability of these systems over time (Hupp and Simon, 1991). The stream continuum hypothesis (Vannote et al. 1980) depicts the stream as an upstream-downstream gradient of gradually changing physical conditions and associated adjustments in functional attributes of the biota.

Biological processes in downstream areas are linked to those in upstream areas by the flow of water, nutrients, and organic materials. Because channelization produces an increase in flow velocity or scour, active bed degradation occurs, causing the movement of substrate particles downstream. As bed degradation continues, degradation of lower streambanks begins, eventually producing bank failure and concave upward banks. During this period of severe instability, the channel is rapidly (in a geologic sense) becoming wider and the water level shallower, sometimes producing a braided flow pattern. Channel widening causes persistent

bank failure in the downstream areas and results in losses of canopy cover and detrital input. These degradation processes move upstream, reducing the rate of channel widening and providing depositional sediment in downstream areas.

Hydrological processes in channelized streams have direct effects on the substrate (embeddedness, scour, and particle size distribution). Transported sediment causes aggradation to occur downstream with deposition on the bed and at the bases of banks. Accretion occurs on the banks with the beginning of the stabilization processes, and seed supplies from riparian vegetation or windblown from other areas settle on these deposits. As vegetation, particularly woody species, becomes established on bank depositional surfaces, stability increases. During this phase of the channel recovery process, meandering features develop through deposition and vegetative stabilization of point bars (inside bend). The return of disturbed stream channels to a dynamically stable, meandering morphology results primarily from the aggradation of banks and beds and the establishment of riparian stands of woody vegetation (Hupp, 1992; Hupp and Simon, 1986, 1991; Simon and Hupp, 1987). Hupp (1992) has estimated that an average of 65 years is needed for this recovery process in nonbedrock controlled, channelized streams in west Tennessee.

A complete concrete lining of natural waterways in western states has long been used to control wet weather flooding. Low flows of reclaimed water are the only source of water for most of the year in these "streams." Wet weather flows are commonly enormous and rapid. Though technically listed as streams and rivers, these engineered channels do not clearly fit definitions commonly understood for either "aquatic habitat" or "streams."

Comparison of historical and current flow conditions can provide valuable information about the extent to which flow alteration is responsible for degradation in biological integrity.

■ **Effects of Flow Regulation.** Many streams are characterized by highly variable and unpredictable flow regimes (Bain et al. 1988). Aquatic macrophyte stands have been shown to be affected by current velocity, but the degree and manner varies with the size of the channel (Chambers et al. 1991). In regulated streams, the importance of a bank-to-midstream habitat orientation becomes magnified. Flow changes displace the shallow shoreline zones, forcing fish restricted to these areas (small fish that use shallow, slow microhabitats) to relocate to maintain their specific set of habitat conditions (Bain et al. 1988). Therefore, if shallow-water habitats are unstable and unable to sustain a well-balanced assemblage, then the functional value of the assemblage is lost and a reduction in organismal population density may follow.

Gislason (1985) illustrates a similar pattern for aquatic insect distribution in fluctuating flows. Bain et al. (1988) also suggest that without the functional availability of shallow, slow, shoreline areas, the stream environment becomes one general type of unstable habitat, dominated by a few habitat generalists and those species using mostly mid-stream habitats. In these cases, the dominance of generalists confounds the assessment of contiguous impact types such as nonpoint source runoff and point source discharges. Comparison of historical and current flow conditions can provide valuable information about the extent to which flow alteration is responsible for degradation in biological integrity.

Energy Source

Stream organisms have evolved to accept and use the energy available to them in natural watersheds. For most small or headwater streams in forested areas of North America, a period of major leaf fall occurs in the autumn. Leaves, in a form referred to as coarse particulate organic matter (CPOM), reach the water and are quickly colonized by bacteria and fungi. The organisms then provide food for invertebrates, which are in turn eaten by fish and other vertebrates. The relative balance of production and respiration varies as a function of stream size, according to the stream continuum hypothesis (Vannote et al. 1980).

Human alteration of the source, type, and quantity of organic material entering streams can affect biological integrity in many ways. Natural shifts in the energy base occur along stream and river gradients, thus providing a major dimension of resource partitioning for the aquatic community. The stream continuum concept (Vannote et al. 1980) outlines different attributes of communities as the energy base shifts from heterotrophic (external) to autotrophic (internal) inputs. These shifts are generally related to increases in drainage area catchments, but exceptions do occur that are related to localized conditions.

Along the stream/river gradient (Fig. 5-6), Cummins (1983) describes the measurement of this shift as a photosynthesis/respiration (P/R) ratio. This P/R ratio is less than 1 in the headwater areas of streams and large rivers. Therefore, these reaches are heterotrophic because in-stream photosynthesis is not a primary energy source. The P/R ratio is greater than 1 in the mid-sized rivers where in-stream photosynthesis is a major contributor to the energy base; the latter are autotrophic. The removal of riparian vegetation for agriculture, channelization, or strip mining, or the shift from natural riparian flora to introduced species for urbanization projects alters the energy base of the aquatic system. Although the stream continuum is thought to no longer hold true for the majority of watersheds, it does exemplify the important considerations in energy base and aquatic ecosystem interaction.

Alterations to the energy base are not independent of alterations to habitat structure. In many instances, assessment of habitat quality is an assessment of impacts to the energy base.

Alterations to the energy base are not independent of alterations to habitat structure. In many instances, assessment of habitat quality is an assessment of impacts to the energy base. However, the evaluation of changes in the energy base can be strengthened by a systematic riparian assessment based on a delineation of natural flora. Alterations in the species of riparian plants influence the functional representation of the aquatic trophic structure biota.

Wilhelm and Ladd (1988) developed a basic tool for conducting natural area assessments in the Chicago region. They presented a checklist of vascular plants of the Chicago region and assigned each species a coefficient of conservatism. This measure expresses the value of the species relative to all other elements in the flora and its particular tie with ancestral vegetation. Low scores are given to native species that are relatively ubiquitous under a broad set of disturbance conditions; high scores are given to species that are sensitive to disturbance; and no scores are assigned to non-native species. In this manner, vegetation can be assessed as representing natural or disturbance conditions.

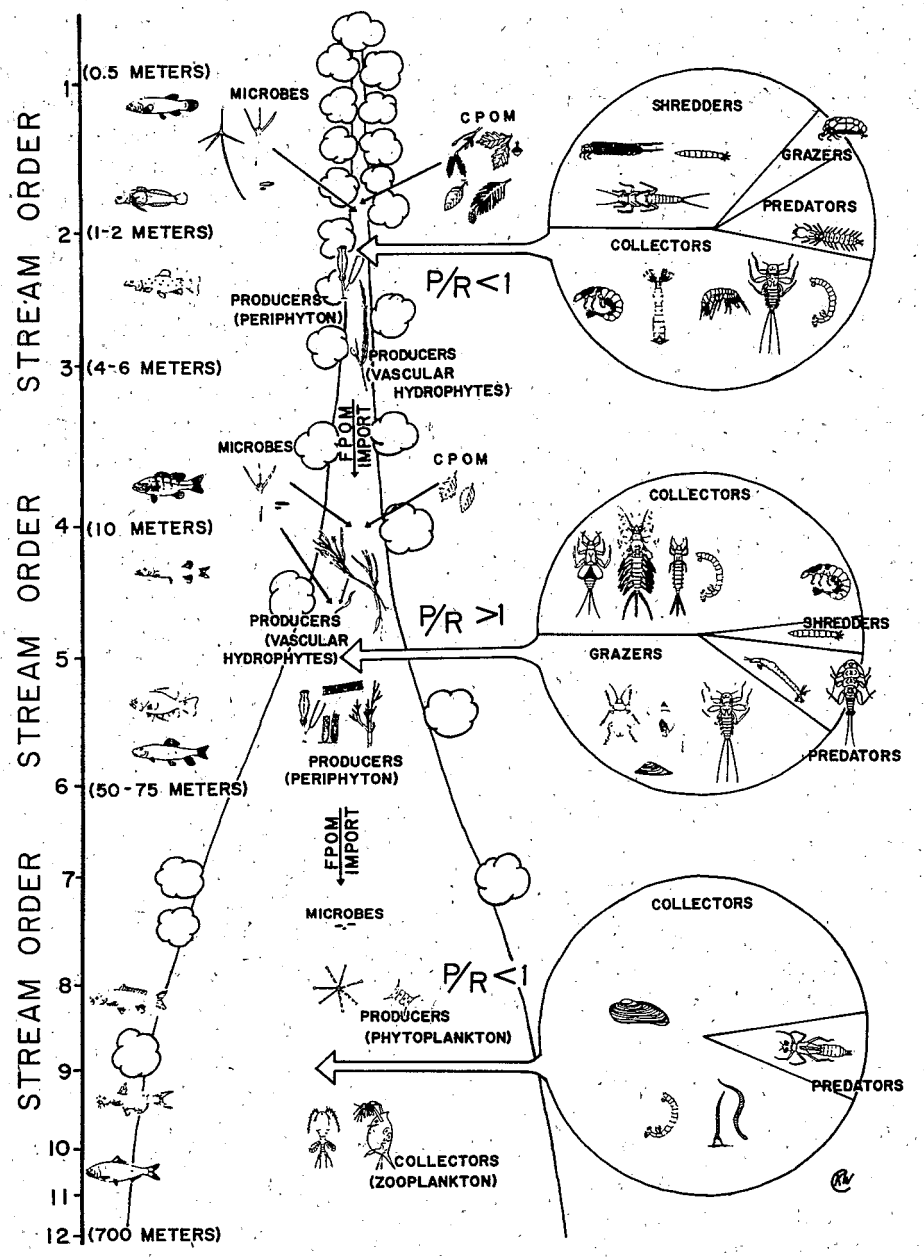

Figure 5-6.—Diagrammatic representation of the stream continuum to illustrate variation in trophic structure of benthic invertebrates (adapted from Cummins, 1983).

Applying this method to riparian corridors would require a similar classification of vegetation. However, much literature is available to aid in classifying riparian flora. The U.S. Forest Service has compiled an extensive database on riparian systems that has been published in several reports (e.g., Platts et al. 1983). Hupp and Simon (1991) recognize early successional species of woody vegetation in riparian zones of disturbed and recovering stream channels in western Tennessee. Padgett et al. (1989) provide a substantial list of references documenting vegetation classification in many of the western states.

Biotic Interactions

Predation, competition, disease, and mutualistic interactions influence where and when species occur within streams. Larval stages of mussels, for example, must attach to the gills of specific fish species to complete their life cycles. Stream communities are often dominated by a few "strongly interacting" species that may have disproportionate effects on the other members of the community (Hart, 1992; Power, 1990). The addition of human influences may alter the integrity of these interactions in ways that alter the abundances of local species and may even cause their demise. Additional human influences are harvests for sport and commercial purposes and the introduction of exotic species, sometimes intentionally but often inadvertently. The practice of stocking fish can be an ecological or genetic disturbance, especially if naturally occurring populations are replaced or infiltrated by stocked individuals. However, the acceptance of this practice is an important societal decision; its advantages and disadvantages must be carefully weighed.

Cumulative Impacts

Even when human actions have an influence on only one of these factors, the effect may cascade through several others. For example, clearing land for agriculture alters the erosion rate and thus the extent to which sedimentation may alter the regional biota. Removal of natural vegetation reduces shading, water infiltration, and groundwater recharge, thereby increasing water temperatures, insolation, and the frequency of flood and drought flows. The resultant agricultural activities may change the stream through channelization, and thus further influence habitat structure. Alterations in the land cover and the channel often have major impacts on water quality (e.g., increased amounts of nitrogen and phosphorus in the runoff from agricultural fields or pesticides in the water). Excess nutrients in modified channels exposed to ample sunlight will enhance the growth of nuisance algae, especially during summer's low flow periods.

Unfortunately, human influences on stream ecosystems cannot be easily categorized (Karr, 1991). The close association between alteration of habitat structure and other impact types complicates the determination of "cause and effect." However, this dimension becomes paramount when mitigative measures are crucial to the attainment of designated uses or biocriteria. In many cases, deductive reasoning, thorough review of the biological data, and use of biological response signatures supported by other environmental data (i.e., physical characterization, toxicity testing, and chemical analyses) aid the assessment of impairment.

The implications of significantly altered systems, for example, channelized streams in urban areas or stream flows regulated by hydroelectric dams, are that reference conditions different from the natural system may have to be established to represent these systems and to evaluate other impact types (Karr and Dionne, 1991). When major impacts (i.e., significant habitat alterations) are present, it is difficult to adequately evaluate changes in community elements and processes that may be attributable to other impacts.

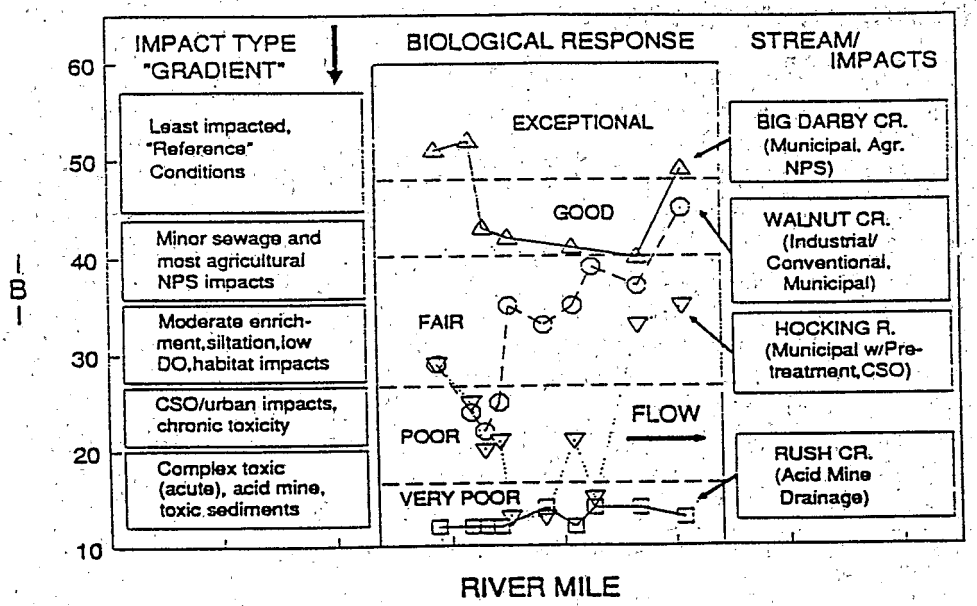

Figure 5-7.—Biological community response as portrayed by the Index of Biotic Integrity (IBI) in four similarly sized Ohio rivers with different types of point and nonpoint source impacts (Yoder, 1991).

The diversity of influences on the quality of water resources requires the kind of multiple attribute approach common to recent biocriteria program efforts. The use of a multiple attribute approach enables the development of biological response signatures to assess probable "causes and effects."

Using biological response signatures, Ohio EPA (Yoder, 1991) was able to assign each of their more severely degraded situations to one of six groups:

- complex municipal and industrial wastes,
- conventional municipal and industrial wastes,
- combined sewer overflow and urbanization,
- channelization,
- agricultural nonpoint source, or
- other, often complex, impacts.

The Ohio EPA also found that various impact types may have one or two biological response characteristics in common. In rare cases, they have three in common. Therefore, only a multiple assemblage, multimetric approach enables a differentiation among impact types. In certain cases, the severity of the impact is related to the type of impact. The IBI has been used by Ohio EPA to characterize these impact types (Fig. 5-7).

The diversity of influences on the quality of water resources requires the kind of multiple attribute approach common to recent biocriteria program efforts. The use of a multiple attribute approach enables the development of biological response signatures to assess probable "causes and effects."

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CHAPTER 6.

Multimetric Approaches for Biocriteria Development

Classical approaches to the assessment of biological integrity have usually selected a single biological attribute that refers to a narrow range of perturbations or conditions (Karr et al. 1986). Likewise, many ecological studies have focused on a limited number of parameters, such as species distributions, abundance trends, standing crops, or production estimates, which are interpreted separately, then used to provide a summary statement about the system's overall health. These approaches are limited because a single attribute may not reflect the overall ecological health of the stream or region. An accurate assessment of biological integrity requires a method that examines the pattern and processes of biotic responses from individual to ecosystem levels (Karr et al. 1986).

Purpose: To describe a multimetric approach for analyzing biological data and to provide guidance for regional selection of metrics.

An alternative approach is to define an array of metrics, each of which provides information on a biological assemblage and, when integrated, functions as an overall indicator of the stream or river's biological condition. The strength of a multimetric assessment is its ability to integrate information from individual, population, community, and ecosystem levels and evaluate this information, with reference to biogeography, as a single, ecologically based index of water resource quality (Karr, 1991; Karr et al. 1986; Plafkin et al. 1989). Multimetric assessments provide detection capability over a broad range and nature of stressors. The Ohio EPA (1987) suggests that the strengths of individual metrics taken in combination minimize any weaknesses they may have individually.

An accurate assessment of biological integrity requires a method that examines the patterns and processes of biotic responses from individual to ecosystem levels.

Abel (1989), LaPoint and Fairchild (1989), and Karr (1991) do not recommend using a single metric. For the broad range of human impacts, a comprehensive, multiple metric approach is more appropriate. Similarly, each of the assemblages discussed in Chapter 4 has a response range to disturbing events and impairments (degraded conditions). Therefore, biosurveys that target multiple assemblages provide the detection capability that is needed to accomplish assessment objectives.

Karr (1991), Karr et al. (1986), Ohio EPA (1987), and Plafkin et al. (1989) recommend use of a number of biological assemblages and metrics that can, when combined and compared with expected conditions, give a more complete picture of the relative biological condition of the study site.

Metric Evaluation and Calibration

Core metrics should represent diverse aspects of structure, composition, individual health, or processes of the aquatic biota.

Core metrics should represent diverse aspects of structure, composition, individual health, or processes of the aquatic biota. Together they form the foundation for a sound integrated analysis of the biotic condition and estimate of the system's biological integrity. Thus, metrics reflecting community characteristics are appropriate in biocriteria programs if their relevance can be demonstrated, their response range verified and documented, and the potential for program application exists. Regional variation in metric details are expected; nevertheless, the general principles used to define metrics seem consistent over wide geographic areas (Miller et al. 1988).

Candidate metrics are determined from the biological data. Good metrics have low variability with respect to the expected range and response of the metrics: it must be possible to discriminate between impaired and unimpaired sites from the metric values. The use of percentiles is a useful technique to evaluate variability of metric performance within stream classes. In operational bioassessment, metric values below the lower quartile of reference conditions are typically judged impaired to some degree (e.g., Ohio Environ. Prot. Agency, 1990). The distance from the lower quartile can be termed a "scope for detection" (Fig. 6-1a). The larger this distance, compared to the interquartile range, the easier it is to detect deviation from the reference condition. Thus, we can define a "detection coefficient" as the ratio of the interquartile range to the scope for detection (Gerritsen and Bowman, 1994). This coefficient is analogous to the coefficient of variation (CV), and the smaller the value, the easier it is to detect the impairment.

Metrics with high variability, or scope for detection, compared to the range of response should be used with caution. Many metrics (e.g., number of taxa) decrease in value with impairment and the detection coefficient for reference sites is thus a good measure of the metrics' potential discrimination ability. Some metric values (e.g., HBI, percent omnivores,

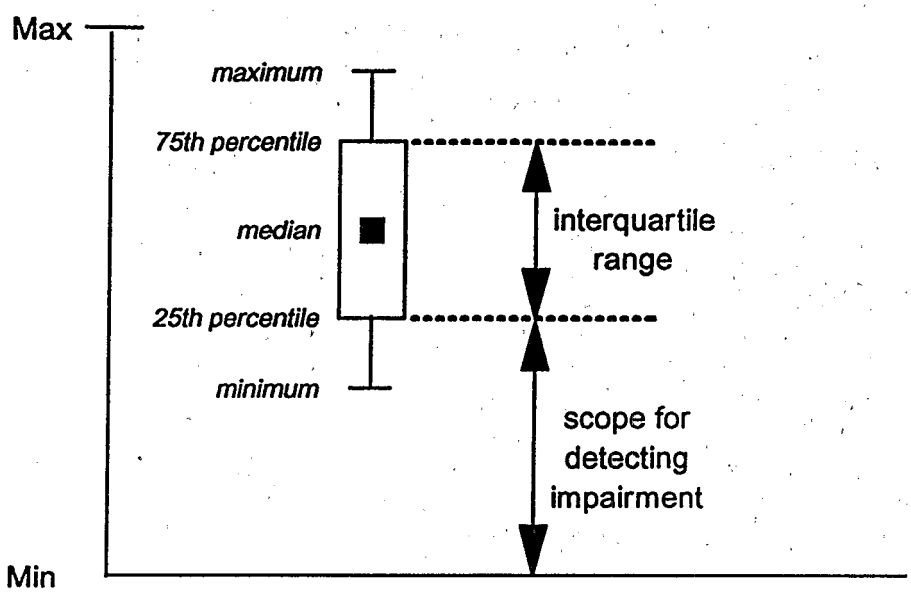

Figure 6-1a.—Metrics that decrease with impairment.

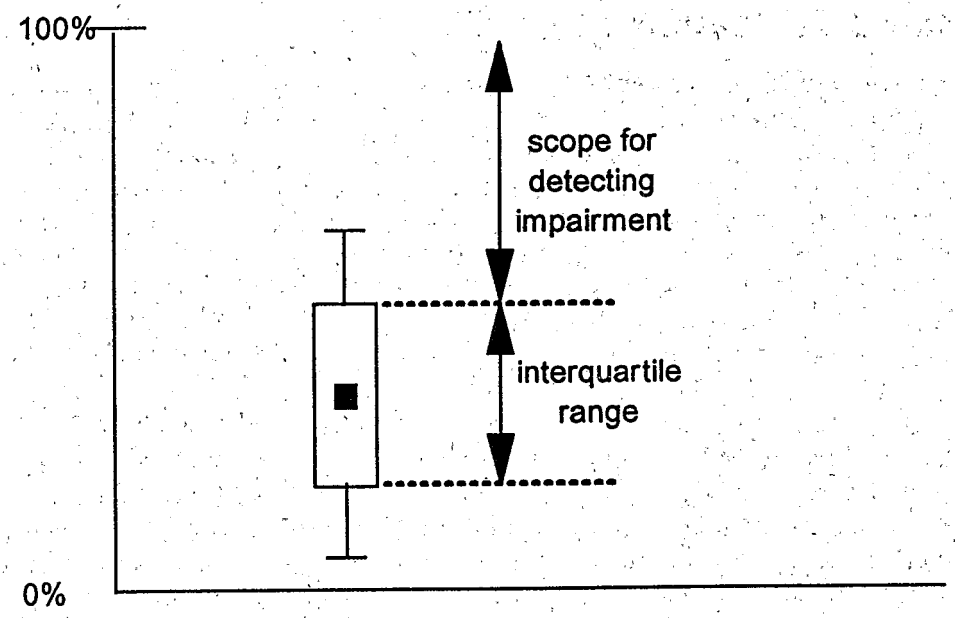

Figure 6-1b.— Metrics that increase with impairment.

percent filterers) may increase under impaired conditions, and the scope for detection would be from the 75th percentile to the maximum value (Fig. 6-1b). The detection coefficient would be calculated the same way and used to judge the discriminatory power of the metrics.

Certain metrics may exhibit a continuum of expectations dependent on specific physical attributes of the reference streams. For example, Fausch et al. (1984) determined that the total number of fish species changes as a function of stream size estimated by stream order or watershed area (Fig. 6-2). They showed that when these data are plotted, the points produce a distinct right triangle, the hypotenuse of which approximates the upper limit of species richness. Thus, a line with a slope fitted to include about 95 percent of the sites is an appropriate approximation of a maximum line of expectations for the metric in question and identifies the upper limit of the reference condition. The area on the graph beneath the maximum line can then be trisected or quadrisected to assign scores to a range of metric values as illustrated in Figure 6-2. The scores provide the transformation of values to a consistent measurement scale to group information from several metrics for analysis.

When different stream classes have different expectations in metric values and a covariate that produces a monotonic response in a metric, a plot of survey data for each stream class may be useful (Fig. 6-3). For each metric, the sites are sorted by stream class (e.g., ecoregion, stream type) and plotted to ascertain the spread in data and the ability to discriminate among classes (Fig. 6-4). If such a representation of the data does not allow discrimination of the classes, then it will not be necessary to develop a separate biocriterion for each class. That is, a single criterion will be applicable to a set of sites that represent different physical classes. Conversely, if differences in the biological attribute are apparent and appear to correspond to the classification, then separate criteria are necessary. This technique is especially useful if the covariates are unknown or do not exist, but a difference in stream class is apparent (Fig. 6-4).

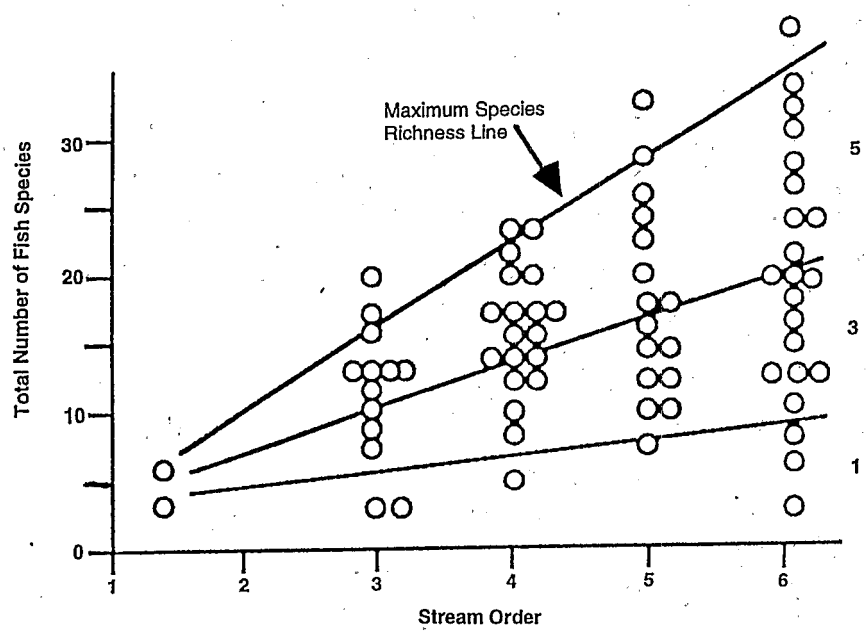

Figure 6-2.—Total number of fish species versus stream order for 72 sites along the Embarras River in Illinois (Fausch et al. 1984).

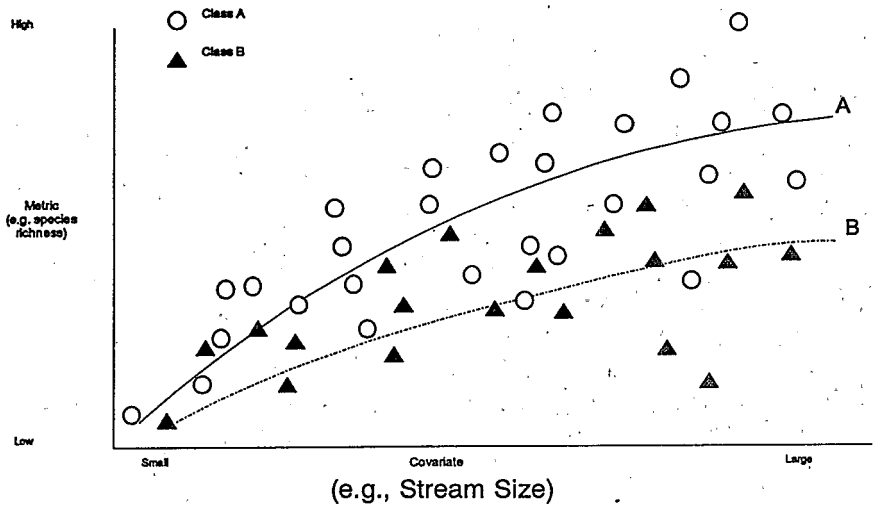

Figure 6-3.—Metrics plotted with a continuous covariate (hypothetical example).

Pilot studies or small-scale research may be needed to define, evaluate, and calibrate metrics. Past efforts to evaluate the use of individual metrics illustrate procedural approaches to this task (Angermeier and Karr, 1986; Barbour et al. 1992; Boyle et al. 1990; Davis and Lubin, 1991; Karr and Kerans, 1992; Karr et al. 1986; Kerans et al. 1992; Lyons, 1992; Resh and Jackson, 1993). Metrics can be calibrated by evaluating the response of metric values to varying levels of stressors.

Sites must be carefully selected to cover the widest possible range of suspected stressors. In general, impaired sites are selected that have impacts from stressors singly and in combination. The selected impaired sites

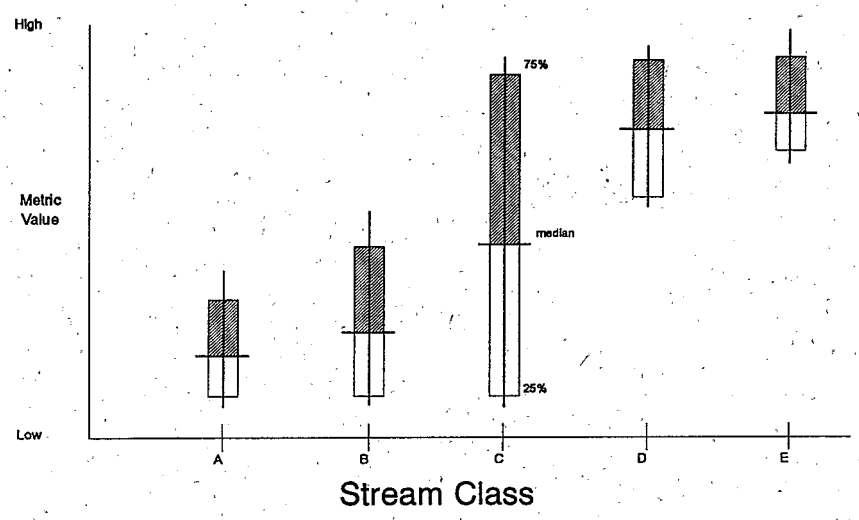

Figure 6-4.—Box and whisker plots of metric values from hypothetical stream classes. Shaded portions are above the median for each class. The box represents a percentile, the vertical line is 1.5 times the interquartile range, and the horizontal line is the median of each distribution.

and the reference sites together are the basis for developing an empirical model of metric response to stressors. Categories of land uses equated with potential impairment are listed in Chapter 7. Candidate metrics that do not respond to any of the stressors expected to occur in a region may be eliminated.

As an example, the discriminatory power of macroinvertebrate metrics was evaluated for Florida streams. The judgment criteria for discrimination were based on the degree of interquartile overlap between the least impaired site category and the impaired site category for each metric. A metric was judged excellent if no overlap existed in the interquartile range (Fig. 6-5a); poor if the overlap was considerable, and no distinction between the impairment categories could be made (Fig. 6-5b). An analysis of a metric's performance among all of the site classes indicated the metric's strength in discriminating between "good" and "bad" conditions.

Additional research is needed to demonstrate the responses of metrics to different stressors in different ecoregions or stream systems. However, once these factors have been considered and demonstrated, the metrics can be incorporated into localized biocriteria programs. It is also important that the metrics and necessary survey methods be appropriate to the logistical and budgetary resources of the investigating agency. Practical application is the penultimate step in metric development. Continued evaluation of metrics and indices is an essential feature of the use of biocriteria.

A biological attribute or metric is some feature or characteristic of the biotic assemblage that reflects ambient condition, especially the influence of human actions.

Biocriteria Based on a Multimetric Approach

The validity of an integrated assessment using multiple metrics is supported by the use of metrics firmly rooted in sound ecological principles (Fausch et al. 1990; Karr et al. 1986; Lyons, 1992). For biocriteria, a biological attribute or metric is some feature or characteristic of the biotic assemblage that changes in a predictable way with increased human influence.

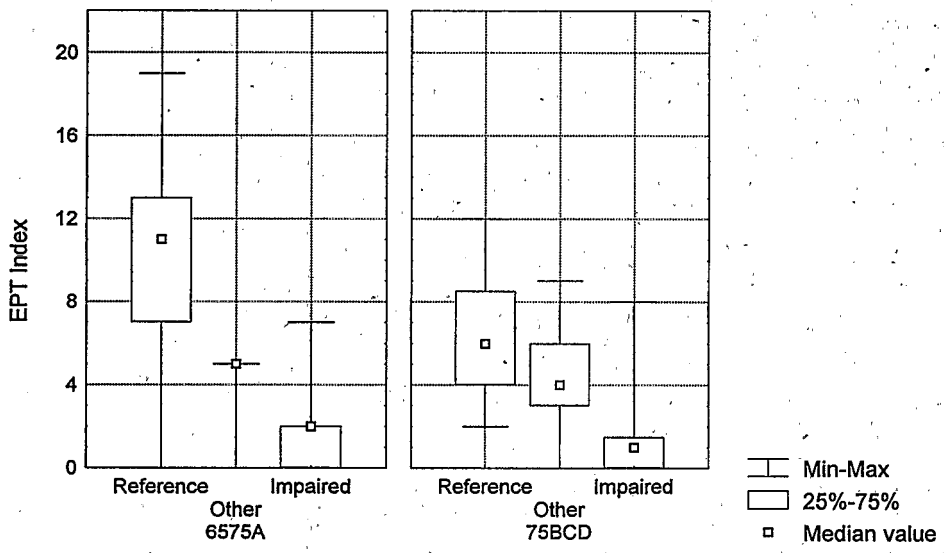

Figure 6-5a.— Site discrimination for the number of Ephemeroptera, Plecoptera, and Trichoptera (EPT index) in Florida streams. (Reference = least impaired, other = unknown, impaired = determined impaired a priori.)

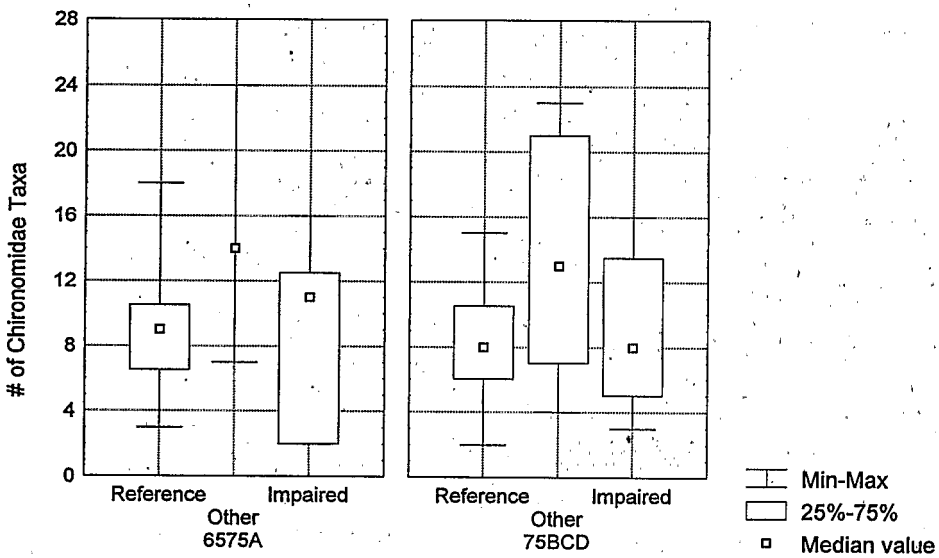

Figure 6-5b.— Site discrimination for the number of Chironomidae taxa in Florida streams. (Reference = least impaired, other = unknown, impaired = determined impaired a priori.)

The status of the biota as indicated by a composite of appropriate attributes (metrics) provides an accurate reflection of the biological condition at a study site. A large number of attributes have been used (e.g., see Fausch et al. 1990; Karr, 1991; Karr et al. 1986; Kay, 1990; Noss, 1990), and each is essentially a hypothesis about the relationship between in-stream condition and human influence (Fausch et al. 1990). Gray (1989) states that the three best-documented responses to environmental stressors are reduction in species richness, change in species composition to dominance by opportunistic species, and reduction in mean size of organisms. But because

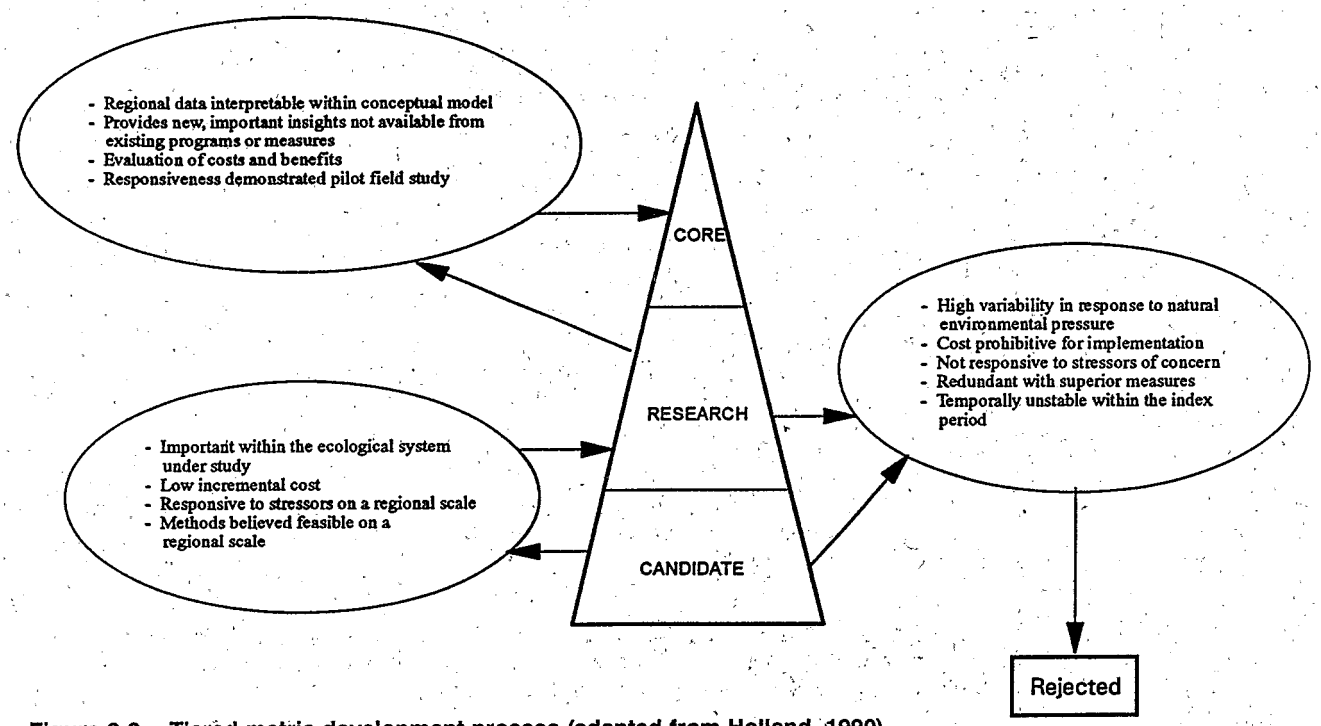

Figure 6-6.—Tiered metric development process (adapted from Holland, 1990).

each feature responds to different stressors, the best approach to assessment is the incorporation of many attributes into the assessment process.

The development of appropriate metrics is dependent on the taxa to be sampled, the biological characteristics at reference conditions, and to a certain extent, the anthropogenic influences being assessed. They must be pertinent to the management objectives to which the biocriteria will be applied. In many situations, multiple stressors impact ecological resources, and specific "cause and effect" assessment may be difficult. However, change over sets of metrics in response to perturbation by certain stressors (or sets thereof) may be used as response signatures.

The development of appropriate metrics is dependent on the taxa to be sampled, the biological characteristics at reference conditions, and to a certain extent, the anthropogenic influences being assessed.

A broad approach for program-directed development of metrics may be modeled after Barbour et al. (1992), Fausch et al. (1990), Holland (1990), or Karr and Kerans (1992). Candidate metrics are selected based on knowledge of aquatic systems, flora and fauna, literature reviews, and historical data (Fig. 6-6). During the research process, these metrics are evaluated for efficacy and validity. Only after careful evaluation should the metrics be introduced into the biocriteria program. Less robust metrics or those not well-founded in ecological principles are weeded out in this research process. Metrics with little or no relationship to stressors are rejected. The remaining, or core, metrics are those that provide useful information in differentiating among sites having good and poor quality biotic characteristics.

The use of multiple metrics to develop a framework for biocriteria is a systematic process involving discrete steps. The process includes site classification (Chapter 3), conduct of a biosurvey and determination of metrics, aggregation into indices, and the formulation of biocriteria. The conceptual model for processing biological data into a biocriteria framework is adapted

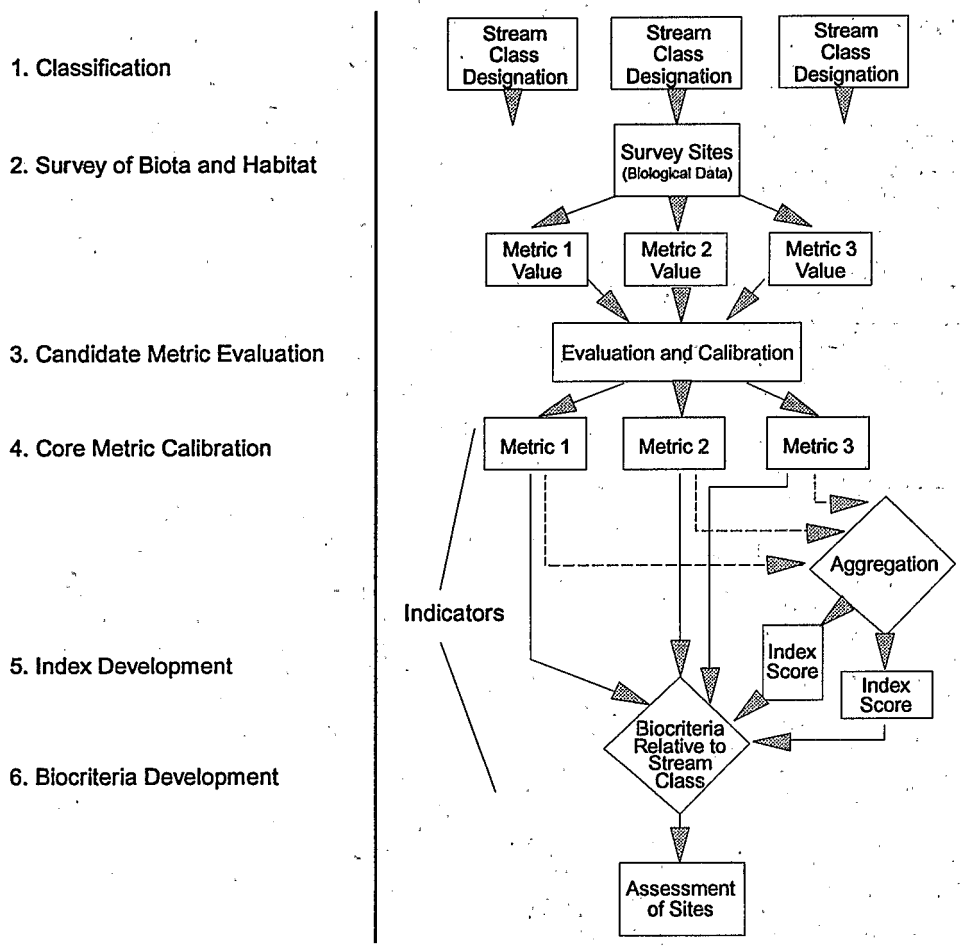

Figure 6-7.—The conceptual process for proceeding from measurements to indicators to assessment condition (modified from Paulsen et al. 1991).

from Paulsen et al. (1991) and illustrated in Figure 6-7. A description of the process is summarized in Table 6-1 and described as follows:

■ **Step 1 — Classification.** Sites are classified as described in Chapter 3 to determine the stream class designation and to ascertain the best and most representative sites for each stream class. The reference condition will be established from this step. Site classification is necessary to reduce and partition variability in the biological data. Multistate collaboration is encouraged in the development of these calibration regions; a benefit is that common methods and metrics can be established among states and cross-state comparisons are enhanced.

■ **Step 2 — Biosurvey.** Surveys of the best sites and those known to be impaired are made for both biota and physical habitat to determine the discriminatory power of the metrics using the impaired and best sites within the stream class. The use of standardized methods (Chapter 4) provides a better interpretation of the raw data than does a conglomeration of techniques. The raw data from a collection of measurements must be evaluated within the ecological context that defines what is expected for similar waterbodies (by reference to waterbody type and size, season, geographic location, and other elements).

Table 6-1.— Sequential progression of the biocriteria process.

BIOCRITERIA PROCESS
Step 1. Classification to Determine Reference Conditions and Regional Ecological Expectations • stream class designation • best and representative sites (reference sites representative of class categories and natural background physical integrity)
Step 2. Survey Best Sites (reference sites) • biota and physical habitat • database consists of raw data (taxonomic lists, abundance levels, and other direct measures and observations)
Step 3. Candidate Metric Evaluation • data analysis (data summaries) of biological attributes • calculation of candidate metrics
Step 4. Core Metric Calibration • testing and validation of metrics by stream class • calibration of metrics to discriminate impairment
Step 5. Index Development • determination of biological endpoints • aggregation of metrics
Step 6. Biocriteria Development • adjustment by physiochemical covariates • adjustment by designated aquatic life use

■ **Step 3 — Candidate Metrics Evaluation and Calibration.** Analysis of the biological data emphasizes the evaluation of biological attributes that represent the elements and processes of the community. All potential metrics having ecological relevance are identified in this step.

■ **Step 4 — Core Metric Calibration.** From the data analysis, metrics are evaluated for relevance to the biological community and validated by stream classes. Calibration of the metrics must address the ability to differentiate between impaired and nonimpaired sites.

■ **Step 5 — Index Development.** For aggregation purposes, transformation to scores from values of various scales of measurement relevant to individual metrics must be done. These scores are normally incorporated into an index, such as the IBI, which, in turn, becomes part of the final assessment process. The individual metrics may also be used as indicators of biological condition in the overall assessment of those endpoints — to support the aggregated index or as individual endpoints.

■ **Step 6 — Biocriteria Development.** Biocriteria are formulated from the indices (Chapter 7) for the stream classes and adjusted by physical and chemical covariates and designated aquatic life uses. The biocriteria may be based on a single aggregated index or established for several biological endpoints.

Potential Metrics for Fish and Macroinvertebrates

A number of attributes can be characterized by metrics within five general classes: community structure, taxa richness, variety, dominance, and relative abundance.

A number of metrics have been developed and subsequently tested in field surveys of benthic macroinvertebrate and fish assemblages (Karr, 1991). Because metrics have been recommended for fish assemblages (Karr, 1981; Karr et al. 1986) and for benthic macroinvertebrates (Barbour et al. 1992; Kerans et al. 1992; Ohio Environ. Prot. Agency, 1987; Plafkin et al. 1989), they will not be reviewed extensively here. A list of the fish assemblage metrics used in the Index of Biotic Integrity (IBI) is presented in Table 6-2, which represents variations in regional fauna. Karr (1991) separates these metrics into three classes: (1) species richness and composition, (2) trophic composition, and (3) abundance and condition. These classes of characteristics generally agree with the areas of assemblage response described as being technically supported (Gray, 1989): reduction in species richness, shift to numerical dominance by a small number of opportunistic species, and reduction in the mean body size of individuals.

Benthic metrics have undergone similar evolutionary developments and are documented in the Invertebrate Community Index (ICI) (Ohio Environ. Prot. Agency, 1987), Rapid Bioassessment Protocols (RBPs) (Barbour et al. 1992; Hayslip, 1992; Plafkin et al. 1989; Shackleford, 1988) and the benthic IBI (Kerans and Karr, in press). Metrics used in these indices are surrogate measures of elements and processes of the macroinvertebrate assemblage. Although several of these indices are regionally developed, some are more broadly based; and individual metrics may be appropriate in various regions of the country (Table 6-3).

Figure 2-2 (see chapter 2) illustrates a conceptual structure for attributes of a biotic assemblage in an integrated assessment that reflects overall biological condition. A number of these attributes can be characterized by metrics within five general classes: community structure, taxa richness, variety, dominance, and relative abundance. Community structure can be measured by variety and distribution of individuals among taxa. Taxa richness, or the number of distinct taxa, reflects the diversity within an assemblage. Multimetric uses of taxa richness as a key metric include the Invertebrate Community Index (Ohio Environ. Prot. Agency, 1987), the Fish Index of Biotic Integrity (Karr et al. 1986), the Benthic Index of Biotic Integrity (Kerans and Karr, in press), and Rapid Bioassessment Protocols (Plafkin et al. 1989). Taxonomic richness is also recommended as critical information in assays of natural phytoplankton assemblages (Schelske, 1984). Taxa richness is usually species level but can also be evaluated as designated groupings of taxa, often as higher taxonomic groups (e.g., family and order, among others) in assessments of invertebrate assemblages.

Relative abundance of taxa refers to the number of individuals of one taxon as compared to that of the whole community. Abundance estimates are surrogate measures of standing crop and density that can relate to both contaminant and enrichment problems. Dominance (e.g., "measured as percent composition of dominant taxon" [Barbour et al. 1992]) or dominants-in-common (Shackleford, 1988) is an indicator of community balance or lack thereof. Dominance roughly equates to redundancy and is an important indicator when the most significant taxa are eliminated from the assemblage or if the food source is altered, thus allowing a few species

Table 6-2.—Index of Biotic Integrity metrics used in various regions of North America.

ALTERNATIVE IBI METRICS	MIDWEST	NEW ENGLAND	ONTARIO	CENTRAL APPALACHIA	COLORADO FRONT RANGE	WESTERN OREGON	SACRAMENTO/ SAN JAQUIN	WISCONSIN
1. Total number of species	X	X		X	X		X	
# native fish species	X					X		X
# salmonid age classes[a]			X			X	X	
2. Number of darter species	X			X	X			
# sculpin species						X		
# benthicinsectivore species		X						
# darter and sculpin species	X	X						
# salmonid yearlings (individuals)[a]						X	X	
% round-bodied suckers	X							
# sculpins (individuals)							X	
3. Number of sunfish species	X				X			X
# cyprinid species						X		
# water column species		X						
# sunfish and trout species			X					
# salmonid species							X	
# headwater species	X							
4. Number of sucker species	X	X				X		X
# adult trout species[a]						X	X	
# minnow species	X				X			
# sucker and catfish species			X					
5. Number of intolerant species	X	X			X	X		X
# sensitive species	X							
# amphibian species								
Presence of brook trout			X				X	
6. Percent green sunfish	X					X		
% common carp								
% white sucker		X			X			
% tolerant species	X							X
% creek chub				X				
% dace species			X					
7. Percent omnivores	X	X	X	X	X			X
% yearling salmonids[a]					X	X		
8. Percent insectivorous cyprinids	X							
% insectivores		X				X		X
% specialized insectivores				X	X			
# juvenile trout								
% insectivorous species	X						X	
9. Percent top carnivores	X	X	X					X
% catchable salmonids						X		
% catchable trout							X	
% pioneering species	X							
Density catchable trout							X	
10. Number of individuals	X		X	X	X	X	X	X*
Density of individuals		X						
11. Percent hybrids	X	X						
% introducted species					X	X		
% simple lithophills	X							X
# simple lithophills species	X							
% native species							X	
% native wild individuals							X	
12. Percent diseased individuals	X	X	X	X	X	X		X

[a]Metric suggested by Moyle or Hughes as a provisional replacement metric in small western salmonid streams.
X = metric used in region. Many of these variables are applicable elsewhere.
*Excluding individuals of tolerant species.
Taken from Karr et al. (1986), Hughes and Gammon (1987), Miller et al. (1988), Ohio EPA (1987), Steedham (1988), Lyons (1992).

Table 6-3.—Examples of metric suites used for analysis of macroinvertebrate assemblages.

ALTERNATIVE BENTHIC METRICS	ICI[a]	RBP[b]	RBP[c]	RBP[d] ID	RBP[d] OR	RBP[d] WA	BIBI[e]
1. Total number taxa	X	X	X	X	X	X	X
% change in total taxa richness				X	X	X	
2. Number EPT taxa	X	X		X	X	X	
# mayfly taxa	X						X
# caddisfly taxa	X						X
# stonefly taxa							X
Missing taxa (EPT)			X				
3. Number diptera taxa	X						
# chironomidae taxa				X		X	
4. Number intolerant snail and mussel species							X
5. Ratio EPT/chironomidae abundance				X	X	X	
Indicator assemblage index			X	X	X		
% EPT taxa				X			
% mayfly composition	X						
% caddisfly composition	X						
6. Percent Tribe Tanytarsini	X						
7. Percent other diptera and noninsect composition	X						
8. Percent tolerant organisms	X						
% corbicula composition							X
% oligochaete composition							X
Ratio hydropsychidae/tricoptera		X			X		
9. Percent individual dominant taxa		X		X	X	X	
% individual two dominant taxa							X
Five dominant taxa in common		X	X		X		
Common taxa index			X			X	
10. Indicator groups				X		X	
11. Percent individual omnivores and scavengers							X
12. Percent individual collector gatherers and filterers							X
% individual filterers				X		X	
13. Percent individual grazers and scrapers				X			X
Ratio scrapers/filterer collectors				X	X	X	
Ratio scrapers/(scrapers + filterer collectors)		X					
14. Percent individual strict predators							X
15. Ratio shredders/total ind. (+ % shredders)		X		X		X	
16. Percent similarity functional feeding groups (QSI)		X	X				
17. Total abundance				X			X
18. Pinkham-Pearson Community Similarity Index		X					
Community Loss Index					X	X	
Jaccard Similarity Index				X			
19. Quantitative Similarity Index (taxa)		X	X				
20. Hilsenhoff Biotic Index		X		X	X	X	
Chandler Biotic Index				X			
21. Shannon-Weiner Diversity Index					X		
Equitability				X			
Index of Community Integrity				X			

[a]Ohio EPA (1987)
[b]Barbour et al. (1992) revised from Plafkin et al. (1989)
[c]Shackelford (1988)
[d]Hayslip (1992); ID = Idaho, OR = Oregon, WA = Washington (Note: These metrics in ID, OR, and WA are currently under evaluation.)
[e]Kerans and Karr (in press)

that are characterized as opportunists to become substantially more abundant than the rest of the assemblage. As a general rule, dominance of one or a few species increasing at a site indicates that the influence of human activities has increased. Comparison to reference conditions provides an important tool to evaluate the extent to which dominance may reflect human activities.

Taxonomic composition can be characterized by several classes of information, including identity and sensitivity. Identity is the knowledge of individual taxa and associated ecological principles and environmental requirements. Key taxa (i.e., those that are of special interest or ecologically important) provide information important to the identity of the targeted assemblages. The presence of exotics or nuisance species may be an important aspect of biotic interactions that relates to both identity and sensitivity. Sensitivity refers to the numbers of pollutant tolerant and intolerant species in the sample. The ICI and RBPs use a metric based on species tolerance values. A similar metric for fish assemblages is included in the IBI (Table 6-2). Recognition of rare, endangered, or important taxa provides additional legal support for remediation activities or recommendations. Species status for response guilds of bird assemblages — for example, whether they are threatened or endangered, their endemicity, or some commercial or recreational value — also relates to the composition class of metrics (Brooks et al. 1991).

Individual condition metrics characterize assemblage features that result from sublethal or avoidance response to contaminants. These metrics focus on low-level chronic exposure to chemical contamination. The condition of individuals can be rated by observation of their physical (anatomical) or behavioral characteristics. Physical characteristics that can be useful for assessing habitat contaminations result from microbial or viral infection, teratogenic or carcinogenic effects arising during development of that individual, or from a maternal effect. These characteristics are categorized as diseases, anomalies, or metabolic processes (biomarkers).

The underlying concept of the biomarkers approach in biomonitoring is that contaminant effects occur at the lower levels of biological organization (i.e., at the genetic, cell, and tissue level) before more severe disturbances are manifested at the population or ecosystem level (Adams et al. 1990). Biomarkers may provide a valuable complement to ecological metrics if they are pollutant specific and if the time and financial costs can be reduced. Unusual behaviors regarding motion, reproduction, or eating habits are often an indication of physiological or biochemical stress. Often behavior measures are difficult to assess in the field.

A metric of individual condition is used for fish in the IBI as "percent diseased individuals" (Table 6-2). The potential for development of biomarkers in biological monitoring exists. McCarthy (1990) briefly discussed several studies that have shown biomarker responses to correlate with predicted levels of contamination and site rankings based on community level measures of ecosystem integrity.

Assemblage processes can be divided into several categories as potential metrics. Trophic dynamics encompass functional feeding groups and relate to the energy source for the system, the identity of the herbivores and carnivores, the presence of detritivores in the system, and the relative representation of the functional groups. Inferences on biological condition

can often be drawn from a knowledge of the capacity of the system to support the survival and propagation of the top carnivore. This attribute can be a surrogate measure for predation rate. Without relatively stable food dynamics, populations of the top carnivore reflect stressed conditions. Likewise, if production at a site is considered high based on organism abundance or biomass, and high production is natural for the habitat type under study (as per reference conditions), biological condition would be considered good.

Process metrics have been developed for a number of different assemblages. For example, Table 6-2 indicates at least seven IBI metrics dealing with trophic status or feeding behavior in fish, focusing on insectivores, omnivores, or herbivores. Also, number or density of individuals of fish in a sample (or an estimate of standing crop) may be considered a measure of production and, thus, in the process class of metrics. Additional information is gained from density measures when considered relative to size or age distribution. Three RBP metrics for benthic macroinvertebrates focus on functional feeding groups (Table 6-3; Barbour et al. 1992; Plafkin et al. 1989). Brooks et al. (1991) use trophic level as one category for rating avian assemblages.

It may not be necessary to establish metrics for every attribute of the targeted assemblage. However, the integration of information from several attributes, especially a grouping of metrics representative of the four major classes of attributes (Fig. 2-2), would improve and strengthen the overall bioassessment. These metrics can be surrogate measures of more complicated elements and processes, as long as they have a strong ecological foundation and allow biologists to ascertain the attainment or nonattainment of biological integrity.

Index Development

Some investigators have suggested that the Index of Biotic Integrity and similar multimetric indices have several problems, particularly the oversimplification of decisions about impairment (Suter, 1993). It is, however, important to consider how these indices are to be employed. Final decisions on the causes of impairment or management actions are not made on the single aggregated number alone; rather, if comparisons to established reference values indicate an impairment in biological condition, then the component parameters (or metrics) are examined for their individual effects on the aggregated value. For each metric, a statement is made describing (1) the derivation of the metric value, (2) the range of possible values, and (3) the ecological implications and relevance of metric values (either absolutely or relative to expectations based on defined reference conditions).

The effects of various stressors on the behavior of specific metrics must be understood. An often-stated concern is that IBI values will be misleading unless the relative sensitivity of the monitored populations to specific pollutants is well known. These concerns are often directed at the use of subjective tolerance values. In fact, field biologists who have extensive experience in local fisheries do know the distribution and ecological requirements of resident fish species. The general concept of integrating tolerance information with distributional data has been used successfully in a vari-

ety of situations (Karr et al. 1986; Ohio EPA, 1987; Hilsenhoff, 1987; Plafkin et al. 1989).

Normalization — and additive aggregation assumes — that each metric has the same meaning (is weighted the same). It also assumes that a 50 percent change in one metric is of equal value to assessment as a 50 percent change in another. Aggregation simplifies management and decision making so that a single index value is used to determine whether action is needed. The exact nature of the action needed (e.g., restoration, mitigation, pollution enforcement) is not determined by the index value, but by analysis of the component metrics.

The stream invertebrate index for Florida was developed by aggregating the metrics that proved responsive to independent (but imprecise) measures of impacts. The approach was to develop expectations for the values of each of the metrics from the reference data set, and to score metrics according to whether they are within the range of reference expectations. Metric values were normalized into unitless scores. Metrics have different numerical scales (e.g., percent Diptera; Shannon-Wiener Index) and must be normalized as unitless values to be aggregated. Metrics within the range received a high score; those outside received a low score. The index value was then the same as the metric scores. The index was further normalized to reference condition, such that the distribution of index values in the reference sites formed the expectations for the region.

Table 6-4.— Index of Biotic Integrity metrics and scoring criteria based on fish community data from more than 300 reference sites throughout Ohio applicable only to boat (i.e., nonwadable) sites. Table modified from Ohio EPA (1987). For further information on metrics see Ohio EPA (1987).

	SCORING DIVISIONS		
	5	3	1
IBI Metric	METRIC VALUE RANGES		
Total no. species	> 20	10 – 20	< 10
% round-bodied suckers	> 38	19 – 38	< 19
No. sunfish species	> 3	2 – 3	< 2
No. sucker species	> 5	3 – 5	< 3
No. intolerant species	> 3	2 – 3	< 2
% tolerant species	< 15	15 – 27	> 27
% omnivores	< 16	16 – 28	> 28
% insectivores	> 54	27 – 54	< 27
% top carnivores	> 10	5 – 10	< 5
% simple lithophils*	> 50	25 – 50	< 25
% DELT anomalies	< 0.5	0.5 – 3.0	> 3
Fish numbers	< 200	200 – 450	> 450

* For sites of a drainage area ≤ 600 miles^2; for sites of of an area > 600 miles^2, scoring categories vary with drainage area

Table 6-5.— Ranges for Index of Biological Integrity values representing different narrative descriptions of fish assemblage condition in Ohio streams. Site category descriptions — wading, boat, and headwaters — indicate the type of site and style of sampling done at those sites. Modified from Ohio EPA (1987).

SITE CATEGORY	EXCEPTIONAL	GOOD	FAIR	POOR	VERY POOR
Wading	50–60	36–48	28–34	18–26	< 18
Boat	50–60	36–48	26–34	16–24	< 16
Headwaters	50–60	40–48	26–38	16–24	< 16

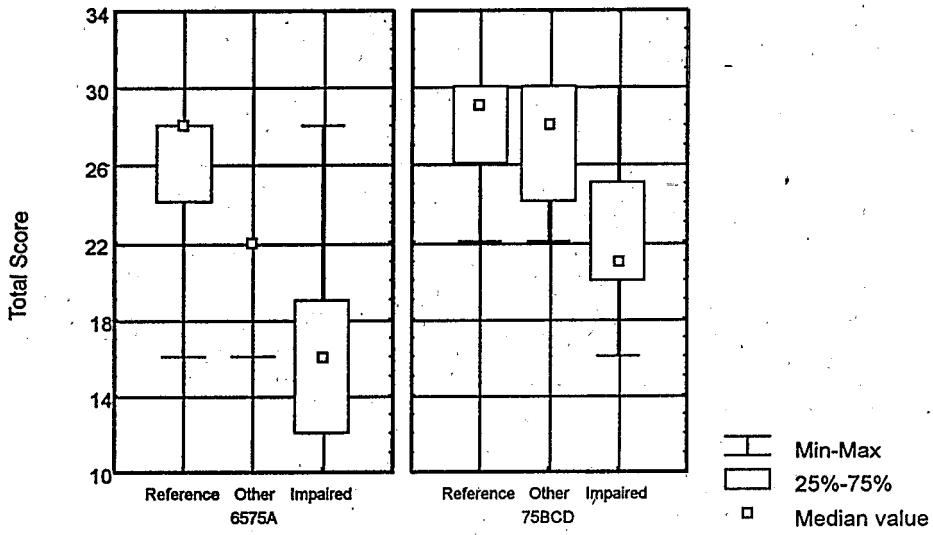

Figure 6-8.—Invertebrate stream index scores for Florida streams.

Ohio EPA (1987) establishes tables based on some predetermined percentiles as discussed above. They recognize three categories of metric scoring ranges for fish assemblage data collected at nonwadable sites (boat sites) (Table 6-4). Ohio EPA (1987) compared individual metric values from sites constituting the reference database to Table 6-4 or similar tables to develop total site scores (aggregated values from 12 normalized metrics) for each of three different types of sites: (1) wadable, nonheadwater streams; (2) nonwadable channels requiring boats for sampling; and (3) headwater streams. These total scores were then used to establish assessment categories (Table 6-5), which are the quantitative basis of biological criteria.

The test of the aggregated index is in the ability to strengthen the discrimination between least impaired and impaired conditions beyond that of the individual metrics. This concept is illustrated in Figure 6-8, as it was done for Florida streams. In some state programs, e.g., Maine and North Carolina, the metrics are treated as individual measures and are not aggregated to form a composite index. For instance, Maine DEP uses as many as 30 biological metrics (macroinvertebrates) to assess attainment of its aquatic life use classes. A threshold coefficient has been established for each metric to be used in a linear discriminant model to test for class attainment. In North Carolina, macroinvertebrate metrics of Taxonomic

Richness, Biotic Index, and EPT Index are the primary metrics of concern in evaluating attainment of their bioclassification criteria for North Carolina's three physiographic provinces.

Multivariate Approaches

An alternative approach to multimetric indices is multivariate analysis of species composition (e.g., Wright et al. 1984; Moss et al. 1987; Furse et al. 1987). The approach consists of developing a model that predicts the expected species composition for sites given their physical and chemical characteristics. Then the observed species composition at a site is compared to the expected species composition predicted by the model. The model characterizes reference conditions, and assessment sites are compared to model-predicted reference conditions.

In the first step of this approach, a classification is developed from species abundance data at reference sites using one or more multivariate clustering or ordination techniques (Ludwig and Reynolds, 1988). Discriminant analysis is then applied to the class assignments and the corresponding physical-chemical data to develop the model for predicting class membership from subsequent physical-chemical site data (Wright et al. 1984). An assessment site is then assigned to a class using the discriminant functions, and its observed species composition is compared to the expected species composition (Moss et al. 1987; Furse et al. 1987). An alternative to discriminant analysis is direct analysis of associations between species composition and environmental variables using methods such as canonical correlation analysis, canonical correspondence analysis, or multidimensional scaling.

Such multivariate approaches for bioassessment are still under development. A predictive model requires extensive physical-chemical data on the reference sites, and there is no assurance that a discriminant model will work well and produce a minimum of misclassifications. The better discriminant models using the above approach misclassify in the range of 25 to 34 percent (Moss et al. 1987). Assessment thresholds and standard procedures are not yet well developed for multivariate assessment, other than professional judgment on missing taxa, similarity indices, or metrics. Nonetheless, as this approach becomes more refined, it may prove to be a viable option to multimetric indexing. In fact, Maine is presently using a combination of the two with promising results.

Multivariate approaches for bioassessment are still under development. . . . Nonetheless, as this approach becomes more refined, it may prove to be a viable option.

Suggested Readings

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CHAPTER 7.

Biocriteria Development and Implementation

Purpose:
To provide water resource agencies with guidance for biocriteria development and implementation.

The first phase in a biocriteria program is the development of "narrative biological criteria" (Gibson, 1992). These criteria are essentially statements of intent incorporated in state water laws to formally consider the fate and status of aquatic biological communities. As stated in that guidance, attributes of sound biological criteria include the following objectives:

1. Support the goals of the Clean Water Act to provide for the protection and propagation of fish, shellfish, and wildlife, and to restore and maintain the chemical, physical, and biological integrity of the nation's waters.
2. Protect the most natural biological community possible by emphasizing the protection of its most sensitive components.
3. Refer to specific aquatic, marine, and estuarine community characteristics that must be present for the waterbody to meet a particular designated use, for example, natural diverse systems with their respective communities or taxa indicated.
4. Include measures of community characteristics, based on sound scientific principles, that are quantifiable and written to protect and/or enhance the designated use.
5. In no case should impacts degrading existing uses or the biological integrity of the waters be authorized.

Establishing Regional Biocriteria

The first decision that a resource agency must make is to determine the set of sites or class to which a biocriterion applies. Site classification (Chapter 3) permits more refined characterization of the reference condition and therefore better resolution in detecting impairment. Any characterization of a reference condition should account for the variability in the biological data used to establish the biocriteria. Thus, the reference condition can be characterized by measures of central tendency (mean, median, trimmed mean) and by variability (standard deviation, quartiles, ranges).

Statewide characterization of reference condition can be expected to exhibit high variance; however, successive intrastate classification will partition the variance from within a large class to among several different component classes. The goal of classification is to minimize within-class variability by allocating the variability to among-class differences. When this goal is achieved, it results in less variation per class and greater resolution of the criteria.

Classification into aquatic types (regional or specific habitat types) should partition overall variance (to achieve lower variability within each class than among classes). The central tendency of each class may be expected to differ (otherwise variability would not be reduced within classes as compared to all classes combined). Investigators for Ohio EPA chose to classify by ecoregion and by aquatic life use. Thus, for each ecoregion and for each aquatic life use within that region, they can characterize a central tendency and variability for the reference condition (from their reference sites).

The more refined the classification, the more precisely the reference condition can be defined; however, an agency also needs to decide when enough classification is enough. Classification can be discrete, as in ecoregions, or continuous, as along a gradient where, for example, expected species richness is a function of stream size.

Biocriteria programs can use discrete and continuous classifications simultaneously; Ohio EPA (1987) has biocriteria that vary by stream size and drainage area within its established ecoregions. The agency's calibration procedures allow investigators to normalize the effects of stream size so that index scores, such as the IBI, can be compared among all streams of a region. For example, the ratio of fish species richness to stream size is an empirical model that accounts for overall variation in species, regardless of stream size. In evaluating whether a test site achieves its species richness potential (a possible biological criterion), one would surely like to take into account the stream size factor. It would be unfair to expect a small stream (with a limited capacity to support a species-rich fish biota) to achieve a high species richness (relative to all streams). By the same token, it would not be good stewardship to allow a large stream (with expected high species richness) to meet attainment merely because its size achieves the statewide criterion.

Designing the Actual Criterion

Having selected its classification scheme, reference sites, and metrics, the agency now has the basic material needed to design the actual criterion. What statistic should be used? A variety of choices are available for measuring central tendency and variability. Two general approaches have evolved, however, for the selection of a quantitative regional biocriterion: the first uses an aggregate or index of metric values, each of which has been assigned a percentile along the distribution of represented minimally impaired sites (Ohio and Florida); the second, a multivariate analysis of metrics or other basic biological data to develop expected thresholds or attainment (Maine).

The percentile that is established for each metric in the first approach is a threshold from which quartiles can be determined for a score ranking system (see chapter 6). The aggregation of these scores for the reference condition functions as the basis for biocriteria.

An example of the second approach is the hierarchical decision-making technique used by Maine. It begins with statistical models (linear discriminant analysis) to make an initial prediction of the classification of an unknown sample by comparing it to characteristics of each class identified in the baseline database (Davies et al. 1991). The output from analysis by the primary statistical model is a list of probabilities of membership for each of four classes (A, B, C, and nonattainment of Class C). Subsequent models are designed to distinguish between a given class and any higher classes as one group, and any lower classes as a second group (Fig. 7-1).

An important consideration is how conservative or protective the agency wants to be. The more conservative the resource agency, the more likely it is that the criterion will be set at the upper end of the condition spectrum. The more liberal the agency is in assessing impairment and maintaining the aquatic life use, the more liberal the criterion will be. Examining the variance structure in a manner similar to that described earlier helps validate the extent to which particular biocriteria apply. If there is little biotic variation evident among the initial regions, or if their differences can be associated with management practices that can be altered, it seems wise to combine those regions to adhere to the same biocriteria.

In the absence of a strong case for subregional biocriteria, it is probably better to overprotect by setting high biocriteria over broad regions than to underprotect by using too low a threshold. Procedures can then be developed that allow for both regional and subregional deviations from the broadly established biocriteria if, and only if, the deviation is justified by natural anomalies.

In these instances, some site-specific rules of exception to regional biocriteria are necessary to accommodate natural limitations. For example, certain natural channel configurations, such as those flowing through bedrock or those that have natural barriers to dispersal, do not offer the habitat diversity of other channel configurations. They cannot, therefore, support the richness

Some site-specific rules of exception to regional biocriteria are necessary to accommodate natural limitations.

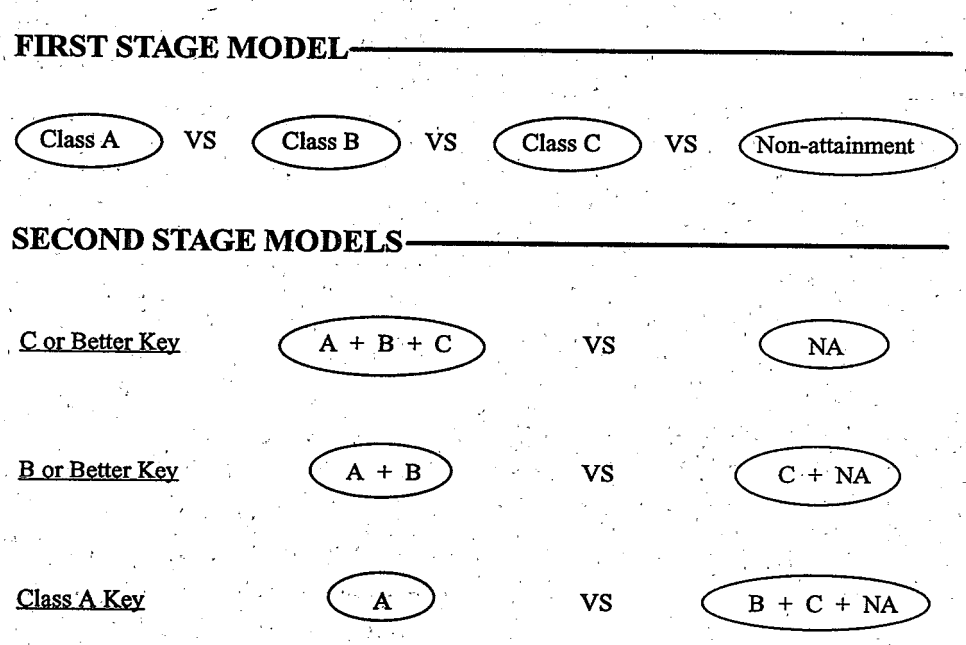


Figure 7-1.—Hierarchy of statistical models used in Maine's biological criteria program (taken from Davies et al. 1993).

and diversity of other nearby channel types. Other natural restrictions to achievement can also be identified, but care must be taken that culturally degraded conditions are not included as evidence for regional biocriteria modification.

The objective in setting biocriteria is to improve the quality of our water resources. Therefore, criteria must not be predicated on accepting the existing, degraded conditions as a matter of course.

Biocriteria for Significantly Impacted Areas

A key element in setting biological criteria is to avoid establishing unduly low thresholds. The objective is to improve the quality of our water resources; therefore, criteria must not be predicated on accepting the existing degraded conditions as a matter of course. In significantly impaired areas, the lowest potentially acceptable criterion is the "best, most natural condition remaining in the region" as defined by a review of the classification data. The upper range for such criteria should be the best condition that is physically and economically achievable by restoration management activities.

This determination is best made by an objective and balanced panel of experts representing the research community, industry, and local, state, and federal water resources specialists using information developed from current and historical data. The actual selection, that is, the point within this range that will become the criterion, should also be established by this panel. This criterion is expected to move upward periodically as management efforts improve the resource condition. A review process should be keyed to the periodic calibrations of regional reference conditions conducted by the states.

In significantly impaired areas, the lowest potentially acceptable criterion is the "best, most natural condition remaining in the region."

There may be no acceptable reference sites in significantly impaired regions. In these areas, an ecological model based on (1) neighboring site classes, (2) expert consensus, and (3) composite of "best" ecological information, may be used (Fig. 3-1). The resultant biocriteria may be an interim or hypothetical expectation that will improve with restoration and mitigation.

Selecting the Assessment Site

Assessment sites should be established to evaluate the effects of human activities on water resources. Potential assessment sites can be identified from land use and topographic maps; specific information can be provided by state and county personnel familiar with the areas. Such sites are generally selected to reflect the influence of known or suspected point and nonpoint source pollution loadings. Final selection should be made only after field reconnaissance by qualified staff at the site verifies that the documented conditions are accurate.

For discrimination of sources and causes of impairment, an agency may need to establish an "impaired" sites database with similar impairments to compare with information at aquatic community test sites. These comparisons can be made using biological response signatures (Yoder, 1991). A *biological response signature* is a unique combination of biological attributes that identify individual impact types or the cumulative impacts of several related human influences. For best results, this process requires the development of an extensive database.

■ **National Pollutant Discharge Elimination System (NPDES) Permit Requests or Renewals.** Public or private wastewater treatment plant administrators and industrial dischargers must apply for NPDES permits. If the number of test sites prohibits annual or more frequent monitoring surveys, a percentage can be surveyed on a rotational basis each year. Priorities can be assigned to permits requiring the earliest renewal or permit award and those in the same geographical area or watershed. Other permitting programs include hazardous waste site regulation, Clean Water Act, section 404/401, dredge and fill certification programs, and construction sites.

■ **Locations of Concentrated Commercial or Industrial Discharges.** In addition to specified permit locations, states may find it appropriate to establish nonspecific monitoring stations along the stream system. These stations can be particularly helpful if located between clusters of commercial, industrial, or municipal operations to help distinguish among potential sources and between groups of users. In addition, the use of nonspecific monitoring stations will help to distinguish discharge effects from preexisting upstream impacts, a distinction particularly helpful given the typical sequential placement of textile or lumber mill operations along small river courses.

■ **Agricultural Concentrations.** Areas of intensive and extensive farming activities are appropriate for the placement of test sites because they can help isolate potential nonpoint source loadings or impairments. Such areas of interest include croplands, rangelands, clearcuts, feedlots, animal holding facilities, manure holding systems, convergent field drainings, contiguous farms, and fertilizer, feed, and pesticide storage facilities. County agricultural extension agents can help determine site placements. They can also identify high risk localities and farms engaged in cooperative conservation programs and suggest appropriate remedial land use practices and programs if and when problems are identified.

■ **Urban Centers.** The locations of shopping centers, commercial districts, and residential areas that include stormwater runoff concentrations are a source of impact to watersheds. Also of interest are urban developments in riparian zones (areas bordering waterbodies), whether or not they contain wastewater treatment plants. On-site wastewater disposal is common in older communities on small lots concentrated near the waterway. The potential septic system problem in these communities can be compounded by an overburdened stormwater drainage network.

■ **Transportation Services.** Vehicle and other traffic modes also affect water resources: major highway interchanges near a watercourse; streams paralleled by extensive, heavily traveled roads or railroads; heavily traveled bridge or overpass systems; pipelines; and maintenance facilities including stockpiles of deicing salt located near a stream system. Airports and railroad or truck marshaling yards may also generate surface runoff problems for nearby stream systems.

■ **Mining and Logging Activities.** Any area affected by cumulative and sequential mining activities and effects including road construction, drilling wells, logging prior to mineral extraction, and acid mine drainage should be evaluated for test site placement. The basis for such decisions will be state mining permit records and associated maps because the areas

For discrimination of sources and causes of impairment, an agency may need to establish an "impaired" sites database with similar impairments to compare with information at aquatic community test sites.

of potential impact, especially from subsurface mining and abandoned mines, may not be self-evident.

■ **Forest Management Activities.** Any areas affected by logging and sawmill activities should be evaluated for test site placement. Instability created by road construction in timber areas is especially damaging to water resources. Effective forestry best management practices (BMPs) will be important influences in these areas. Protection of these areas is critical because many of the representative reference sites will be located in forested lands. Federal and state foresters need to interact with state water quality agencies for identification of sensitive areas.

■ **Disruptive Land Use Activities.** This category will include a variety of planned or existing construction projects: landfills; channelization or other in-stream projects such as dams and flood control structures, fish hatcheries, or aquaculture. Any of these activities on a significant scale or near streams should be monitored and evaluated. If advance notice of these activities is provided, states should establish both spatial and temporal monitoring before, during, and after the activities for biological assessments.

■ **Land Use Activities in Unsurveyed or Remote Areas.** This category includes regions not previously surveyed for which no preexisting information would be available in the event of a spill or major hydrological calamity and remote sites for which development is planned in the near or distant future. Long-term antecedent biological information should be a component in new development planning.

Evaluating the Assessment Site

Assessment sites are points or reaches on a stream at which disturbance is suspected or from which information about the location's relative quality is desired.

Statistically evaluating the test site(s) against the reference condition to assess the extent and degree of impairment is the focus of another document (Reckhow, in review); however, the basic question is this: What evidence do we have that indicates impairment (or absence of impairment)? If the assessment is based on a reference condition determined from a composite of sites, the manager's confidence in the judgment is improved over that from use of a single reference site — notwithstanding that some level of precision may be lost (see Chapter 3).

The simultaneous comparison of an assessment site to a site-specific reference condition is an alternative that is generally undertaken as an upstream/downstream or paired watershed approach. Presumably the site-specific reference condition represents the best attainable condition of the assessment site(s). In this approach, the percent-of-reference may be the most appropriate criterion from which to assess impairment. States that have limited resources may wish to implement this approach as an interim until a larger database is developed. The assessment of sites follows the same guidelines whether reference data are site-specific or regional (Table 7-1).

Assessment sites are points or reaches on a stream at which disturbance is suspected or from which information about the location's relative quality is desired. In selecting assessment sites, the latitude of selection compared to the choice of reference sites may be considerably reduced. If the area is suspect, it must be investigated regardless of its stream charac-

Table 7-1.—Sequential process for assessment of test sites and determination of their relationship to established biocriteria. Refer to Chapter 6 for an explanation of biocriteria establishment.

ASSESSMENT PROCESS	
Step 1.	Determine Class • same classification scheme as for reference sites
Step 2.	Survey Assessment Sites • biota and physical habitat
Step 3.	Calculate Metrics • convert raw data to metric values
Step 4.	Aggregate Metrics to Form Indices • use scoring rules established for metrics • sum normalized metric values
Step 5.	Compare to Reference (Biocriteria) • use established regional biocriteria for assessment
Step 6.	Statement of Condition • characterize existence and extent of impairment • diagnostics as to stressors

teristics or channel configuration. Thus, regionalized reference conditions, while necessary for criteria development, may not always be sufficient to serve as a foundation for expecting a specific biological condition. The investigator facing a potentially contentious situation may find it prudent to augment the regional reference data with results of locally matched reference sites, such as upstream sites or sites in similar, nearby streams.

The assessment process is essentially a replication of the procedure described earlier to develop multiple metrics (see Chapter 6 and Fig. 6-2). Note, however, that the move from the development of metrics and indices to their use in the assessment process leads directly to the development and implementation of biocriteria. The assessment process, summarized in Table 7-1 and illustrated in Figure 7-2, is described as follows:

Step 1 — Classification of Assessment Sites. Sites selected for assessment are assigned to the appropriate classification derived from the initial reference classification scheme. The assessment site is classified according to the stream class designations, not the nature of a suspected land use or point-source discharge impact. In other words, similar receiving waters should be in the same classification whether or not there are similar discharges to those waters.

Step 2 — Biosurvey. Stream or small river biological communities and habitat characteristics should be measured using the same techniques and equipment as were used at designated reference site(s). It will also be necessary to gather data during the same time frame. This schedule may not coincide with a predetermined indexing period. For example, if a construction site is scheduled to open on a particular date or if a critical period of operation is approaching, both the test and reference site(s) will have to be surveyed accordingly.

Step 3 — Calculate Metrics. Many of the intermediate steps used in the criteria development process become unnecessary at this point. Investigators can simply enter the appropriate raw data from the refer-

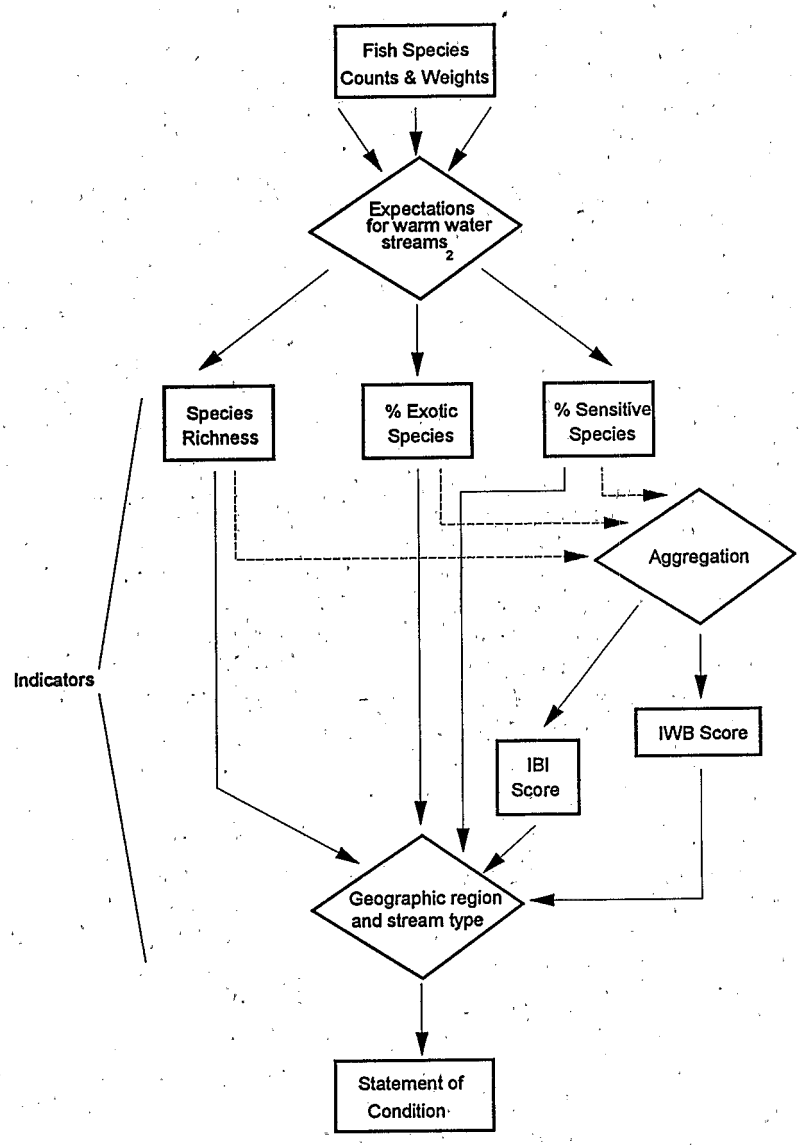


Figure 7-2.—The process for proceeding from measurements of fish assemblage to indicators such as the Index of Biotic Integrity (IBI) or Index of Well Being (IWB) — as used to develop criteria and apply those criteria to streams (modified from Paulsen et al. 1991).

ence and test sites into a preselected format to generate current metrics. In all cases, the integrity of the raw data should be presumed for support and as additional information for more definitive assessment.

Step 4 — Calculate Indices. Where indicated, these metrics are similarly summarized in indices of relative biological condition and habitat description. Some states do not use indices but evaluate the information from the individual metrics as independent measures of biological condition.

Step 5 — Compare to Appropriate Biological Criteria. The biological data from the site under assessment are compared to established criteria to ascertain the status. Both the indices (aggregation of metrics) and

the individual metrics are evaluated as part of the assessment. All available information must be used to confirm the status of the biological condition and to diagnose the cause and effect relationship if impairment is detected.

Step 6 — Statement of Condition. At this point, the assessment sites are evaluated to determine whether they do or do not meet the criteria. The sites can also be placed in priority order using the details of this evaluation to support management plans and resource allocations. Further refinement of the data collected and additional investigations can help determine cause and effect relationships among the stresses identified by this process. Such information will be essential to successful remedial management.

Overview of Selected State Biocriteria Programs

■ **Maine.** In 1986, the State of Maine enacted legislation that mandated an objective "to restore and maintain the chemical, physical, and biological integrity" of Maine waters. In addition, a legislative water quality classification system was established to manage and protect the quality of Maine waters. The classification system established minimum standards for designated uses of water and related characteristics of those uses (Table 7-2). Within each use-attainability class, the minimum condition of aquatic life necessary to attain that class is described.

The descriptions or narrative standards in this legislation range from statements such as "Change in community composition may occur" (Class C) to "Aquatic life as naturally occurs" (Class A and AA). The designated use classes were recombined into four biologically discernible classes (Table 7-2): Classes A and AA were combined, and a fourth class, nonattainment of Class C, was added.

The Maine Department of Environmental Protection has assessed a large, standardized macroinvertebrate community database from samples taken above and below all major point-source discharges, as well as samples from relatively undisturbed areas. Maine used this database as a calibration dataset to develop discriminant functions for classifying sites among the four analytical classes.

The calibration data set consisted of the general level of abundances from 145 rock basket samples collected from first to seventh order streams throughout Maine, and covering a wide range of relatively unimpacted and impacted streams. General abundances were reduced to approximately 30 quantitative metrics.

The calibration data set was given to five stream biologists to assign the 145 sites to the four classes (A, B, C, and NA) using professional judgment. The biologists used only the biological data; they did not see locations, names, habitat, or site chemistry. Disagreements on class assignments were resolved in conference.

The resultant metrics and class assignments were then used to develop linear discriminant models to predict class membership of unknown assessment sites. Two stages of discriminant models were developed from the calibration data set: the first stage estimates the probability that a site belongs to one of the four classes (A, B, C, or NA); the second stage esti-

Table 7-2.—Maine's water quality classification system for rivers and streams, with associated biological standards (taken from Davies et al. 1993).

AQUATIC LIFE USE CLASS	MANAGEMENT	BIOLOGICAL STANDARD	DISCRIMINANT CLASS
AA	High quality water for recreation and ecological interests. No discharges or impoundments permitted.	Habitat natural and free flowing. Aquatic life as naturally occurs.	A
A	High quality water with limited human interference. Discharges restricted to noncontact process water or highly treated wastewater equal to or better than the receiving water. Impoundments allowed.	Habitat natural. Aquatic life as naturally occurs.	A and AA are indistinguishable because biota are "as naturally occurs."
B	Good quality water. Discharge of well treated effluent with ample dilution permitted.	Habitat unimpaired. Ambient water quality sufficient to support life stages of all indigenous aquatic species. Only nondetrimental changes in community composition allowed.	B
C	Lowest water quality. Maintains the interim goals of the Federal Water Quality Act (fishable/swimmable). Discharge of well-treated effluent permitted.	Ambient water quality sufficient to support life stages of all indigenous fish species. Change in community composition may occur but structure and function of the community must be maintained.	C
NA			Not attaining Class C

mates two-way probabilities that a site belongs to higher or lower classes (i.e., A, B, C. *vs.* NA; A, B, *vs.* C, NA; and A *vs.* B, C, NA). Each model uses different metrics.

In operational assessment, sites are evaluated with the two-step hierarchical models. The first stage linear discriminant model is applied to estimate the probability of membership of sites into one of four classes (A, B, C, or NA). Second, the series of two-way models are applied to distinguish the membership between a given class and any higher classes, as one group (Fig. 7-1). Monitored test sites are then assigned to one of the four classes based on the probability of that result, and uncertainty is expressed for intermediate sites. The classification can be the basis for management action if a site has gone down in class, or for reclassification to a higher class if the site has improved.

Maine biocriteria thus establish a direct relationship between management objectives (the three aquatic life use classes and nonattainment) and biological measurements. The relationship is immediately viable for management and enforcement as long as the aquatic life use classes remain the same. If the classes are redefined, a complete reassignment of streams and a review of the calibration procedure will be necessary.

■ **North Carolina.** The North Carolina Department of Environment, Health and Natural Resources, Division of Environmental Management, Water Quality Section has written Standard Operating Procedures for the collection of biological data and the bioclassification of each station sampled. Biological criteria have been included in the North Carolina water quality standards as written narratives. Narrative standards have been in place since 1983. They support the use of biological assessments in point and nonpoint source evaluation, and help identify and protect the best uses of North Carolina waters. High Quality Waters, Outstanding Resource Waters and Nutrient Sensitive Waters are assessed using biocriteria.

Phytoplankton, aquatic macrophytes, benthic macroinvertebrates, and fish are routinely collected as part of North Carolina's biosurvey effort. Only the macroinvertebrate biosurvey data and the associated bioclassification system are summarized here.

Macroinvertebrates are sampled qualitatively by one of two methods: a Standard Qualitative Method or the Ephemeroptera, Plecoptera, and Trichoptera (EPT) Survey Method. When following the Standard Qualitative Method, two kick net samples from cobble substrate, three dip-net samples (sweeps) from vegetation and shore zones, one leaf pack sample, two fine-mesh rock and/or log wash samples, one fine-mesh sand sample, and visual inspection samples are taken.

The EPT survey method focuses on qualitative collection of Ephemeroptera, Plecoptera, and Trichoptera, by collecting one kick sample, one sweep sample, one leaf-pack sample and visual collections. With both methods, invertebrates are sorted in the field using forceps and white plastic trays, and preserved in glass vials containing 5 percent ethanol. Organisms are sorted in approximate proportion to their relative abundance.

Currently, site-specific reference conditions are typically used when conducting surveys. However, where site-specific reference sites are not available, ecoregional reference conditions are used to define unimpaired conditions. North Carolina is developing ecoregional reference conditions based on the available land use information. The three major ecoregions identified in North Carolina are Mountain, Piedmont, and Coastal Plain.

Specific macroinvertebrate metrics, including taxonomic richness, biotic indices, an Indicator Assemblage Index (IAI), diversity indices (Shannon's Index), and the Index of Community Integrity (ICI) are used to rate sites as poor, fair, good/fair, good, and excellent. The ratings are conducted in addition to the narrative descriptions for biocriteria. These metrics are used as independent measures rather than aggregated into an overall index.

Bioclassification criteria for the Mountain, Piedmont, and Coastal Plain ecoregions in North Carolina have been developed for EPT taxa richness values. This community metric has been developed using both the Standard Qualitative Method and the EPT Survey Method. The bioclassification ratings for the number of EPT taxa in each ecoregion for both the Standard Qualitative Method and the EPT method are summarized in Table 7-3. Note that the rating system has been developed solely on summer (June-September) collections. Samples collected in other seasons, therefore, must be seasonally corrected before a bioclassification can be assigned.

The North Carolina classification system was developed for chemical impact assessment and does not address sedimentation or other habitat alteration effects. A special bioclassification rating has also been developed

Table 7-3.—Bioclassification criteria scores for EPT taxa richness values for three North Carolina ecoregions based on two sampling methods.

	STANDARD QUALITATIVE METHOD		
BIOCLASSIFICATION	MOUNTAIN	PIEDMONT	COASTAL PLAIN
Excellent	>41	>31	>27
Good	32-41	24-31	21-27
Fair	12-21	8-15	7-13
Poor	0-11	0-7	0-6
	EPT QUALITATIVE METHOD		
BIOCLASSIFICATION	MOUNTAIN	PIEDMONT	COASTAL PLAIN
Excellent	>35	>27	>23
Good	28-35	21-27	18-23
Good-Fair	19-27	14-20	12-17
Fair	11-18	7-13	6-11
Poor	0-10	0-6	0-5

for small, high quality mountain streams which naturally exhibit a reduced macroinvertebrate taxa number. Streams possessing these particular characteristics, having EPT taxa of ≥ 29 (Standard Qualitative Method) or ≥ 26 (EPT Survey Method) are considered excellent.

■ **Ohio.** Ohio's biological criteria program was developed for complete integration with state water quality standard regulations. As such, biocriteria in Ohio are fully integrated with typical water quality measures, and address three key strategic goals:

- The protection of aquatic life in all Ohio waterways capable of supporting aquatic life is an immediate goal of the Ohio EPA to be accomplished, wherever possible, through a "systems" (biological community response) approach.
- Short- and long-range goals must be established for the control of toxic substances in Ohio's surface waters.
- The protection of human health through the assurance of a "safe" level of exposure to toxic substances in water and fish is an immediate goal of the Ohio EPA.

To accomplish these goals, the Ohio EPA program combines biocriteria, effluent toxicity, and water chemistry. This integrated approach has significantly increased Ohio EPA's ability to detect degradation, particularly in streams receiving point and nonpoint sources and both toxic and conventional pollutants.

The Ohio EPA has employed the concept of tiered aquatic life uses in the Ohio Water Quality Standards (WQS) since 1978. Aquatic life uses in Ohio include the Warmwater Habitat (WWH), Exceptional Warmwater Habitat (EWH), Cold-water Habitat (CWH), Seasonal Salmonid Habitat (SSH), Modified Warmwater Habitat (three subcategories: channel-modified, MWH-C; affected by mines, MWH-A; and impounded, MWH-I), Limited Resource Water (LRW) (Ohio EPA 1992). Each of these use designations are defined in the Ohio WQS.

Water quality standards constitute the numerical and narrative criteria that, when achieved, will presumably protect a given designated use (Ohio EPA 1992). Chemical-specific criteria serve as the "targets" for wasteload allocations conducted under the TMDL (Total Maximum Daily Load) process, which is used to determine water quality-based effluent limits for point source discharges and, theoretically, load allocations for nonpoint sources (in connection with best management practices). Whole effluent toxicity limits consist of acute and chronic endpoints (based on laboratory toxicity tests) and a dilution method similar to that used to calculate chemical-specific limits. The biological criteria are used to directly determine aquatic life use attainment status for the EWH, WWH, and MWH use designations as is stated under the definition of each in the Ohio WQS.

The biological criteria designed for Ohio's rivers and streams incorporate the ecoregional reference approach. Within each of the State's five ecoregions, criteria for three biological indices have been derived. The indices include two measures of fish community structure and one measure of the benthic macroinvertebrate community. The combined indices provide a quantitative measure that can be compared to regional reference indices to assess use attainment.

The two fish community measures include the Index of Biotic Integrity (IBI) and the modified Index of Well Being (IWB). Both indices incorporate structural attributes of the fish community, while the IBI additionally incorporates functional (trophic) characteristics. The two indices incorporate a range of fish community attributes much broader than only species richness and relative abundance. For macroinvertebrate community measurements, Ohio EPA uses the Invertebrate Community Index (ICI). The ICI is a modification of the IBI concept, but has been adapted for use with macroinvertebrates. Like the IBI, ICI values incorporate functional aspects of the community.

Derivation of the above indices requires extensive sampling to provide the quantitative data necessary for analysis. The IBI and IWB require sampling of approximately 500 meters of a river or stream by electroshocking to characterize the community of fish. Data recording is extensive, and includes fish species, number of individuals per species, and various observations of fish condition. The ICI requires that quantitative (Hester-Dendy) and qualitative macroinvertebrate samples be collected. Laboratory analysis of these samples includes taxon determination to genus or species, and quantification of the organisms collected.

The Exceptional Warmwater Habitat (EWH) is the most protective use assigned to warmwater streams in Ohio. Ohio's biological criteria for EWH applies uniformly statewide and is set at the 75th percentile index values of all reference sites combined. The Warmwater Habitat (WWH) is the most widely applied use designation assigned to warmwater streams in Ohio. The biological criteria for fish vary by ecoregion and site type and are set at the 25th percentile index values of the applicable reference sites in each ecoregion (Fig. 7-3a). A modified procedure was used in the extensively modified Huron Erie Lake Plain (HELP) ecoregion.

The Modified Warmwater Habitat (MWH), first adopted in 1990, is assigned to streams that have had *extensive* and irretrievable physical habitat modifications. The MWH use does not meet the Clean Water Act goals and therefore requires a Use Attainability Analysis. There are three sub-

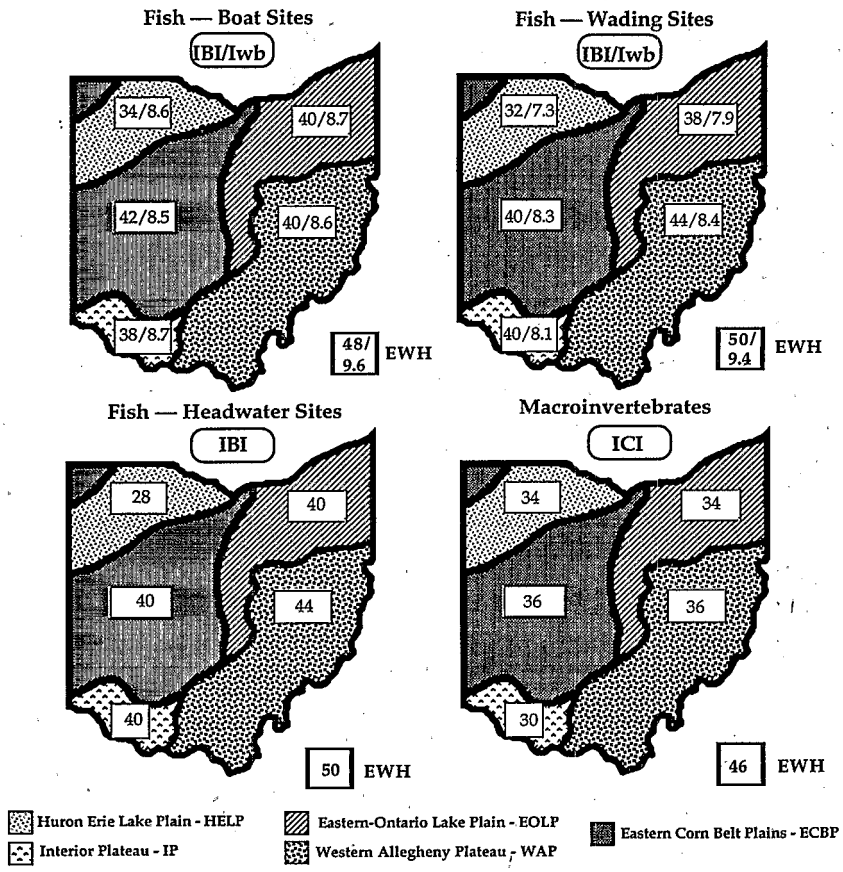


Figure 7-3a.— Biological criteria in the Ohio WQS for the Warmwater Habitat (WWH) and Exceptional Warmwater Habitat (EWH) use designations arranged by biological index, site type for fish, and ecoregion. Index values in the boxes on each map are the WWH biocriteria that vary by ecoregion as follows: IBI/MIwb for Boat Sites (upper left), IBI/MIwb for Wading Sites (upper right), IBI for Headwaters Sites (lower left), and the ICI (lower right). The EWH criteria for each index and site type are located in the boxes just outside each map (Ohio EPA, 1992).

categories: MWH-A, non-acidic mine runoff affected habitats; MWH-C, channel modified habitats; and MWH-I, extensively impounded habitats. Biological criteria were derived from a separate set of modified reference sites. The biocriteria were set separately for each of three categories of habitat impact (Fig. 7-3b). The MWH-C and MWH-I subcategory biocriteria were also derived separately for the HELP ecoregion. The MWH-A applies only within the Western Allegheny Plateau (WAP) ecoregion.

Costs for State Programs Developing Bioassessments and Biocriteria

Biocriteria programs begin with the development of a bioassessment framework. Expertise in ecological principles and resource investment by the agency is required to develop this framework and to implement biocriteria. State agencies will vary in their investment of resources and effort in this process.

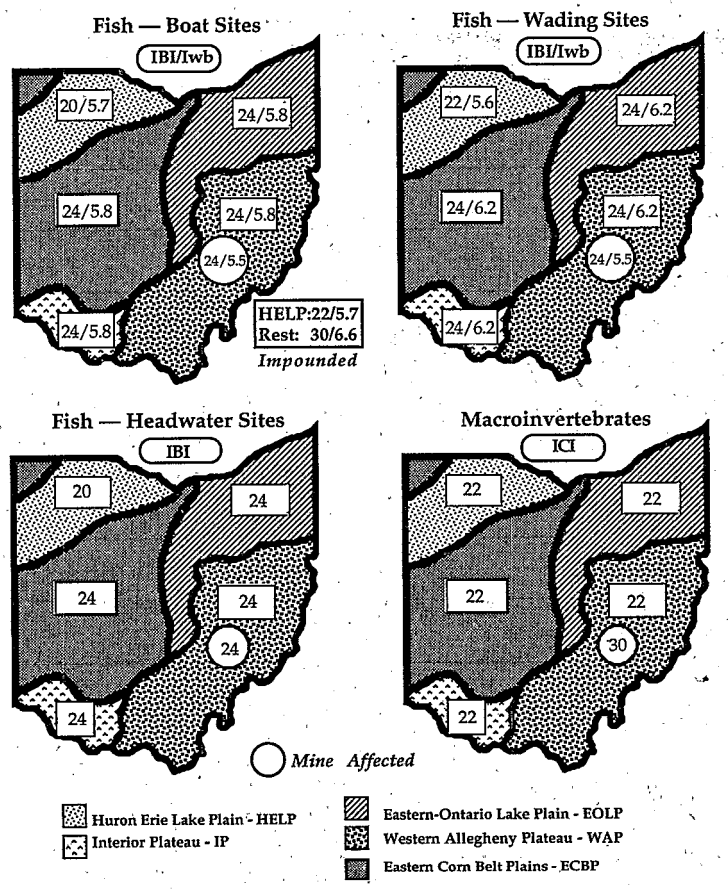


Figure 7-3b.—Biological criteria in the Ohio WQS for the Modified Warmwater Habitat (MWH) use designation arranged by biological index, site type for fish, modification type, and ecoregion. Index values in the boxes on each map are the MWH biocriteria for the channelized modification type that vary by ecoregion as follows: IBI/MIwb for Boat Sites (upper left), IBI/MIwb for Wading Sites (upper right), IBI for Headwaters Sites (lower left), and the ICI (lower right). The MWH criteria for the impounded modification type is located in the box just outside the Boat Sites map. The biocriteria for the mine-affected modification type is represented by the circled value located in the WAP ecoregion on each map (Ohio EPA, 1992).

Several states that have initiated biocriteria programs were polled to obtain estimates of their cost and resource needs. These cost estimates represent a range of program elements including assemblage selection (benthic macroinvertebrates and fish) and geographical coverage (statewide or targeted regions of the state). The following paragraphs briefly characterize each of the state programs included in the poll before extrapolating cost estimates in terms of funding and personnel.

■ **Delaware.** The nontidal streams in Delaware are mostly low-gradient coastal streams that drain agricultural lands. Delaware Department of Natural Resources and Environmental Control (DNREC) developed a modification of the EPA's rapid bioassessment protocols to sample benthic macroinvertebrate from multihabitats in these streams. Technical issues addressed in developing their bioassessment included standardized methods, level of subsampling, taxonomic level (family or genus), and the se-

lection of appropriate metrics. Samples are collected during a specified index period that extends from late summer through the fall season. Biosurveys done by department biologists include survey planning, collection, processing, and data analysis. Consultants are used to assist in processing benthic samples for large projects.

■ **Florida.** Florida Department of Environmental Protection (DEP; formerly the Department of Environmental Regulation) used a combination of in-house biologists, scientists from the EPA's Environmental Research Laboratory in Corvallis, and consultants to develop a statewide stream bioassessment program based on thorough site regionalization and methods development projects. Florida DEP samples benthic macroinvertebrates from multiple stream habitats using a modified RBP method, and assesses biological condition using a suite of metrics. The sampling sites are classified into aggregated subecoregions for determination of appropriate reference conditions. Currently, the portions of Florida that are not adequately delineated are south Florida, south of Lake Okeechobee, and northeastern Florida around Jacksonville. Two index periods are used to assess biological condition—August through September, and January through February. Florida DEP biologists collect and process all samples. Outside consultants are used to analyze the data and develop taxonomic keys.

■ **Idaho.** Both fish and benthic macroinvertebrates are surveyed by Idaho Department of Environmental Quality (DEQ) as part of Idaho's monitoring program. Their biological program is a relatively intense part of a multiyear monitoring effort to assess nonpoint source impacts. Idaho DEQ is now evaluating their current program and refining their biological methods. Consultants are used to assist in this process. The field sampling and sample analysis are conducted by Idaho DEQ regional staff.

■ **Maine.** Maine Department of Environmental Protection (DEP) uses rock-filled baskets as introduced substrate for macroinvertebrate colonization. The statewide program uses aquatic life use designations to establish reference conditions. Numeric biocriteria have recently been incorporated in Maine's rules. Analysis is done using a tiered multivariate procedure that incorporates information from up to 35 metrics. Maine's index period is in the summer. Virtually all of its bioassessment is accomplished by Maine DEP biologists.

■ **Nebraska.** Both fish and benthic macroinvertebrates are sampled in Nebraska by the Department of Environmental Quality (DEQ). A multimetric approach is used for both assemblages, based on the IBI for fish and EPA's RBPs for benthos. Reference conditions have been determined for each ecoregion in Nebraska and a summer index period is used to sample streams. Nebraska's biological monitoring program was developed and is maintained by DEQ biologists.

■ **North Carolina.** The Department of Environment, Health, and Natural Resources (DEHNR) of North Carolina has had an effective bioassessment program in place for several years. A standardized macroinvertebrate sampling procedure is used to sample multiple habitats in North Carolina streams; metrics are used to assess biological condition, and judgment criteria are based on the ecoregion level of site classification. The design and

development of the program as well as all aspects of monitoring are conducted by DEHNR biologists.

■ **Ohio.** Ohio EPA has developed both a fish and benthic macroinvertebrate protocol for conducting bioassessments in Ohio's streams and rivers. A multimetric approach is used in both protocols that focuses on a summer index period. Site classification is by ecoregion with a given percentage of the sites monitored on an annual basis. Numeric biocriteria are included in Ohio's water resource program. They were developed in a hierarchical manner by aquatic life use and ecoregion. Ohio EPA staff designed and developed the bioassessment program, and conducts the annual sampling with in-house staff and summer interns.

■ **Oklahoma.** The Oklahoma Conservation Commission (OCC) has developed a biological assessment program that includes benthic macroinvertebrate, fish, and periphyton sampling to evaluate nonpoint source effects. However, the benthic program is central and reflects the cost of developing the program which is statewide and loosely based on ecoregions. The index period is summer, and monitoring during other seasons is dependent on the case study. Technical consultants were used to help establish the reference condition.

■ **Oregon.** Oregon Department of Environmental Quality (DEQ) has developed a modified RBP approach for surveying benthic macroinvertebrates and fish in streams in the Coastal Range. The other five ecoregions have not been extensively sampled to date. Multiple metrics are calculated and used to assess biological condition. A single fall index period (September, October, November) is emphasized. However, monitoring is done in other seasons to evaluate specific impacts, for example, forest insecticide application. The majority of the biosurvey and assessment is done by DEQ biologists.

Turning now to costs: it is apparent from the states polled that a minimum of two full-time equivalent staff are needed for the development of an effective biological assessment program. The states of Ohio, Maine, North Carolina, and Florida have invested the equivalent of 12 staff (or more) to develop their programs (Table 7-4). However, Ohio EPA points out that only 19 percent of their surface water monitoring program is devoted to biological monitoring (Yoder and Rankin, 1994). When considered on the basis of agencywide water programs, Ohio EPA allocates 6 percent to biological monitoring.

Cost investment will vary depending on the geographical coverage (number of stream miles), the extent of coverage, biological approach and targeted assemblages, and the extent of shared resources (e.g., other state and federal agency assistance, and shared reference conditions). Nebraska and Ohio have developed their program statewide for fish and benthos, whereas other states polled emphasized only benthos and some have not covered the whole state (Table 7-5). Although Delaware and Florida have only partial coverage to date, their programs are relatively complete and are pertinent for the majority of their state streams. A few of the states have used contractor support, which ranged from $10,000 to $350,000.

Though self-reported, the costs reviewed here are typical costs incurred by state bioassessment programs.

Value of Biocriteria in Assessing Impairment

Water resource agencies currently use several tools to assess impairment and monitor changes. However, these tools can be separated into three distinct categories: chemical analysis of water samples, toxicity testing of selected species, and biosurveys. These tools, though not interchangeable in all cases, are most effective when used in conjunction with each other. Chemical and toxicity criteria, however, are only useful for assessing adverse impacts from chemical discharges. Biosurveys and biocriteria are more appropriate than other tools for measuring cumulative or synergistic impacts, the status of the resources, and impairment from stressors other than chemical contamination, such as habitat degradation.

Table 7-4.— The investment of state water resource agency staff needed to develop bioassessment programs as a framework for biocriteria.

	FULL-TIME EQUIVALENT (FTE) STAFF					
STATES	STANDARDIZE METHODS	SITE CLASSIFICATION	FIELD SURVEY	REFERENCE CONDITION	METRICS AND INDICES	DEVELOPMENT TOTAL
Benthos and Fish [Statewide]						
Nebraska	0.04	0.73	0.88	0.28	0.49	2.4
Ohio	2.0	1.0	2.7	2.5	3.0	11.2
Benthos [Statewide]						
Maine	1.0	8.0	1.5	—	3.0	13.5
N. Carolina	8.0	1.0	4.0	2.0	1.0	16.0
Oklahoma	0.05	0.5	0.25	0.75	0.25	1.8
Benthos [Partial Coverage]						
Delaware	0.4	0.1	0.3	0.6	0.6	2.0
Florida	2.6	2.0	5.7	1.0	1.0	12.3
Oregon	0.25	0.25	1.0	1.0	0.5	3.0

Table 7-5.— Costs associated with retaining consultants to develop bioassessment programs as a framework for biocriteria. Dash indicates work done by state employees or information not available; FTE costs for contractors and state employees are not equivalent.

	FULL-TIME EQUIVALENT (FTE) STAFF					
STATES	STANDARDIZE METHODS	SITE CLASSIFICATION	FIELD SURVEY	REFERENCE CONDITION	METRICS AND INDICES	DEVELOPMENT TOTAL
Benthos and Fish [Statewide]						
	—	—	—	—	—	—
Nebraska	—	—	—	—	—	—
Ohio						
Benthos [Statewide]						
Maine	—	8	36	—	13	57
N. Carolina	—	—	—	—	—	—
Oklahoma	—	—	—	25	—	25
Benthos [Partial Coverage]						
Delaware	55	5	—	—	40	100
Florida	100	210	—	75	75	350
Oregon	—	—	10	—	—	10

Several comparison studies were conducted and documented in the Technical Support Document for Water Quality-based Toxics Control (U.S. Environ. Prot. Agency, 1991). These studies used biosurvey results to calibrate the judgment of impairment using toxicity testing.

The Agency conducted studies at eight freshwater sites in which ambient toxicity was compared to the biological impact on the receiving water. These site studies were a part of the Complex Effluent Toxicity Testing Program (CETTP). Testing was performed on-site concurrent with the field surveys. Sites exhibiting biological impacts were included from Oklahoma, Alabama, Maryland, West Virginia, Ohio, and Connecticut. Organisms were exposed to samples of water from various stations and tested for toxicity. Biological surveys (quantitative field sampling of fish, invertebrate, zooplankton, and periphyton communities in the receiving water areas upstream and downstream of the discharge points) were made at these stations at the same time the toxicity was tested to see how well the measured toxicity correlated to the health of the community. These studies have been reviewed and published in an EPA publication series (Mount et al. 1984; 1985; 1986; 1986a; 1986b; Mount and Norberg-King 1985; 1986; Norberg-King and Mount 1986).

A robust canonical correlation analysis was performed to determine whether or not statistically significant relationships existed between the ambient toxicity tests and in-stream biological response variables and to identify which variables play an important role in that relationship (Dickson et al. 1992). Influential variables were then used to classify stations as either impacted or not. *Ceriodaphnia dubia* productivity and/or *Pimephales promelas* weight were used as the basis for predicting impact (U.S. Environ. Prot. Agency, 1991). Fish richness was used to classify streams as impact observed or impact not observed.

In this set of studies, agreement was obtained between the prediction of in-stream toxicity using ambient toxicity testing and the observed biological impairment from the biosurvey results (Fig. 7-4). However, at 10 percent of the sampling stations, agreement was not reached. EPA (1991) has said that this small difference in results would not significantly affect the diagnosis of impairment.

Another study conducted by the North Carolina Division of Environmental Management indicated the high accuracy of predicting receiving water impacts from whole effluent toxicity tests. Forty-three comparisons were made between freshwater flowing streams using the *Ceriodaphnia du-*

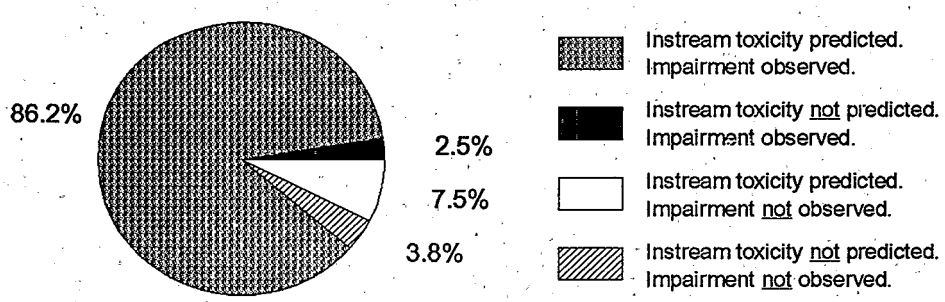


Figure 7-4.—Comparison of ambient toxicity and fish richness surveys at eight sites in various parts of the United States (taken from U.S. EPA, 1991)

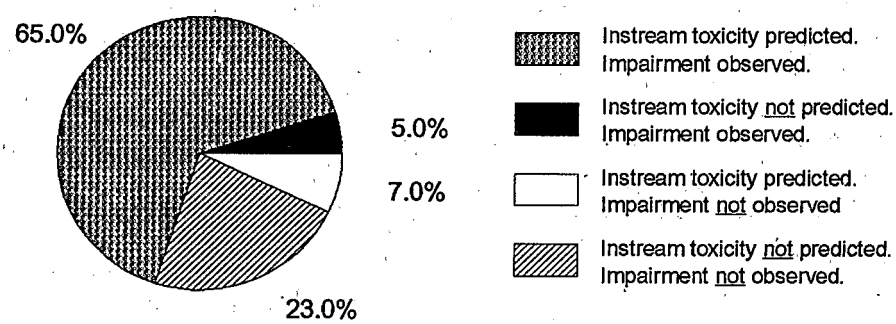


Figure 7-5.—Comparison of effluent toxicity of receiving water impact using *Ceriodaphnia dubia* chronic toxicity tests and freshwater receiving stream benthic invertebrates at 43 point source discharging sites in North Carolina (taken from U.S. EPA, 1991).

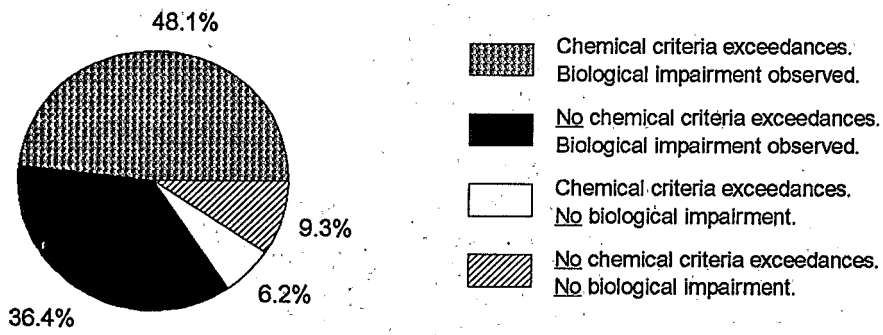


Figure 7-6.—Comparison of chemical criteria exceedances and biosurvey results at 645 stream segments in Ohio.

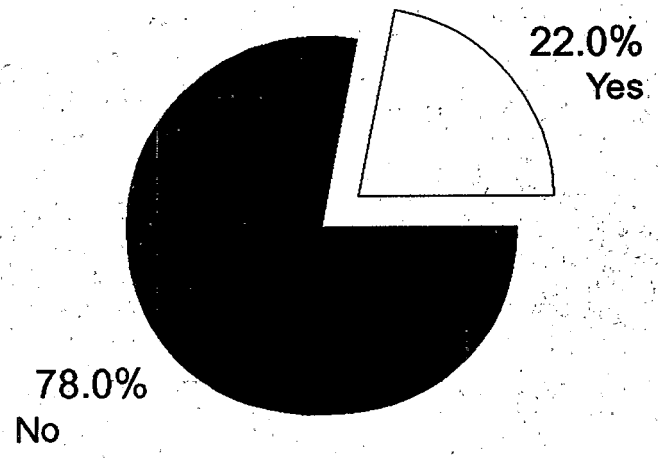
bia chronic test and a qualitative macroinvertebrate sampling. The result was an overall 88 percent accuracy of prediction (Fig. 7-5). However, in 12 percent of the cases, agreement was not reached. Both of these studies indicate that some risk of error exists if impairment is predicted using toxicity tests alone.

Chemical analyses are less accurate in predicting biological impairment. In a study conducted by Ohio EPA, the prediction of impairment from chemical analyses agreed with the biological survey results in only 47 percent of the cases (Fig. 7-6). Chemical analyses were unable to detect the impairment measured by biocriteria at 50 percent of the sites. Ohio EPA (1990) stated that the absence of detected chemical criteria exceedances when biological criteria impairment was indicated may result from several possibilities: (1) chemical parameters other than those sampled have been exceeded, (2) impairments of a nontoxic nature exist, (3) impairments stemming from physical impacts (e.g., habitat modification, flow alteration) exist, and/or (4) impairments related to biological interactions (e.g., exotics, disease) exist. None of these scenarios would be detected or fully understood using chemical criteria assessments alone.

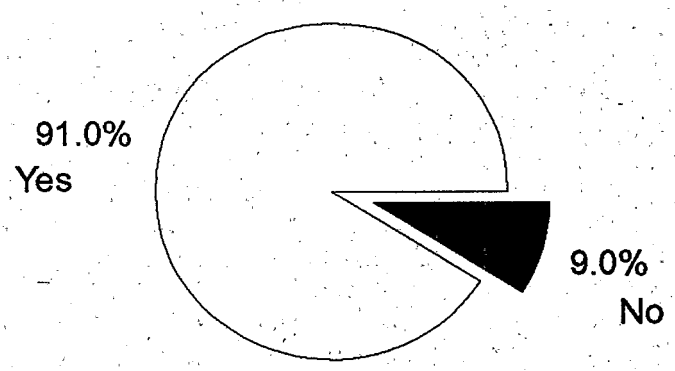
The Delaware Department of Natural Resources and Environmental Control assessed the attainment of their aquatic life use class for nontidal streams in 1994 using both their dissolved oxygen criteria and a biological endpoint. Results indicated that the use of the dissolved oxygen criteria

was inadequate to detect impairment to the aquatic life. Documentation of exceedances to the dissolved oxygen criteria suggested that only 9 percent of Delaware's nontidal streams failed to meet attainment (Fig. 7-7). Whereas the habitat and biological assessment approach indicated that 78 percent of the nontidal streams were not attaining their designated use.

These experiences support the observation that biological criteria are an excellent assessment tool and one that covers environmental variables not necessarily addressed by other chemical, physical, or effluent toxicity studies. While not yet advocated as a method for setting regulatory NPDES permit limits, the biocriteria process is clearly an essential means of environmental assessment and has in fact been used to review these permits and other management efforts in several states including Ohio, Maine, and North Carolina.



Fixed Stations - Dissolved Oxygen
(No statistical confidence)



Probabilistic - Habitat/Biology
(95% Confidence Interval +//- 5-6%)

Figure 7-7.—Assessment of nontidal stream aquatic life use attainment in Delaware. (taken from the state's 305[b] report,1994).

Suggested Readings

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CHAPTER 8.

Applications of the Biocriteria Process

Biocriteria, a critical tool for state agencies to use in protecting the quality of water resources, serve several important purposes: they help (1) characterize and classify aquatic resources, (2) refine aquatic life use categories, and (3) judge use impairment (i.e., they help determine attainment and nonattainment of designated uses). Additionally, biocriteria are used for (4) identifying possible sources of impairment (e.g., habitat degradation, flow regime changes, chemical contamination, energy alterations, or biological imbalance); (5) problem screening; (6) ranking and establishing priorities for needed remedial actions; and (7) assessing the results of new management practices. Other applications of the process include evaluating the adequacy of NPDES permits, and trend reporting for 305(b) reports.

Purpose:
To illustrate the importance of biocriteria in various areas of water resource management.

Stream Characterization and Classification

The process of biocriteria development requires that streams be classified according to type to determine which reference conditions and criteria are required. This classification must be done in each of the nation's ecoregions — as defined by climate, geographic, and geologic characteristics. Then, within these regions, the streams should be further categorized and their classes either combined or subdivided depending on whether they have similar or distinctive biotic compositions.

Initial classifications can be confirmed, refined, or revised on the basis of subsequent biological data. This continued monitoring makes the reference sites and derived biological criteria more certain, and helps the resource managers and biologists identify unique or particularly sensitive streams for special attention or protection. The following case study from North Carolina illustrates this point.

CASE STUDY — North Carolina

STATE	LOCATION	DATES
North Carolina	South Fork of New River	March–August 1990

The South Fork of New River forms the headwaters of the New River in North Carolina. The entire South Fork New River catchment is mountainous with generally steep, forested slopes. The floodplain is broad with

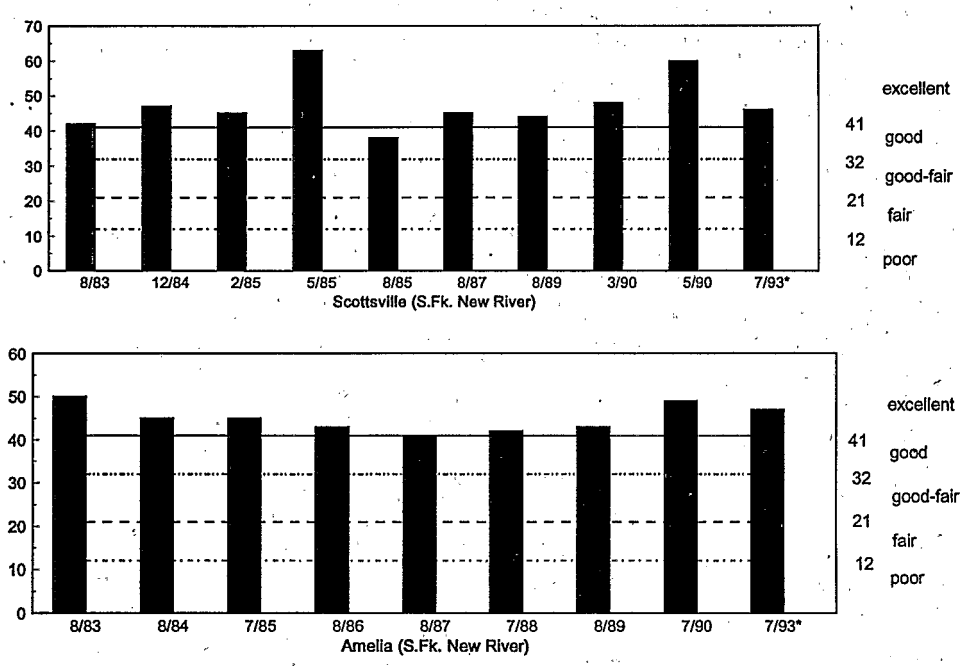
The classification and definition of designated uses of streams and rivers are important in developing and using biocriteria. Similarly, as biocriteria become established, the expanded database helps refine these classifications.

rolling hills; and land uses in the area are primarily rural and agricultural, including crop and dairy pasture production. Nonpoint source runoff from these uses has a high potential for water quality problems (NC Dep. Environ. Manage. 1978).

The North Carolina Environmental Management Commission classifies certain waters of the state as "outstanding resource waters" (ORW) if such waters have an exceptional recreational significance and exceptional water quality. Determining whether a North Carolina stream qualifies for reclassification as an ORW depends primarily on data collected by the Biological Assessment Group, which is part of North Carolina's biocriteria program.

To evaluate an ORW request for the New River, the Biological Assessment Group collected benthic macroinvertebrate samples from 21 riverine and tributary locations within the New River catchment. Main-stem river locations (the South and North Forks of the New River) were sampled using the Group's standardized qualitative collection method, which uses a wide variety of collection techniques (and 10 samples) to inventory the aquatic fauna. The primary output is a taxa list with some indication of relative abundance for each taxon (i.e., abundant, common, or rare). The combined number of species in the pollution-intolerant insect orders of Ephemeroptera, Plecoptera, and Trichoptera (EPT Index) is used with department criteria to assign water quality ratings. Unimpaired or minimally impaired streams and rivers have many species, while polluted areas have fewer species.

Based on analyses of the biological data (Fig. 8-1), excellent water quality was found at the ambient monitoring location on the South Fork New River near Scottsville and Old Field Creek, a tributary of the South Fork New River. Prior data have also consistently shown excellent water



*Seasonal adjustment factor for winter and spring developed for EPT Index after 1990

Figure 8-1.—EPT Index (number of taxa of Ephemeroptera, Plecoptera, and Trichoptera) for two locations on the South Fork of the New River, North Carolina.

quality at the South Fork New River near Jefferson and for the New River itself, below the confluence of the North and South Forks. A site on the North Fork New River also had excellent water quality, but repeated sampling at this site revealed that its samples fluctuate between good and excellent quality on a temporal basis. Until it achieves a more consistent water quality rating, this site on the North Fork will not be recommended for an ORW classification.

Old Field Creek has an outstanding brook trout resource. The South Fork of the New River has been designated as a Natural and Scenic River from the confluence of Dog Creek in the documented excellent reach of the river to its confluence with the New River. The New River — according to information provided by local canoeing outfitters — supports an unusually high level of water-based recreation.

It was, therefore, recommended that the South Fork New River from the confluence of Dog Creek to the New River, and the New River itself, to the last point at which it crosses the North Carolina-Virginia state line be designated ORW. The west prong of Old Field Creek (Call Creek) from its source to Old Field Creek, and Old Field Creek below its confluence with the west prong to the South Fork New River was also designated ORW. On the basis of biological data, the recommendation was accepted. The Commission reclassified these streams in December 1992, thereby ensuring that stricter point and nonpoint source regulations would be enforced in this region.

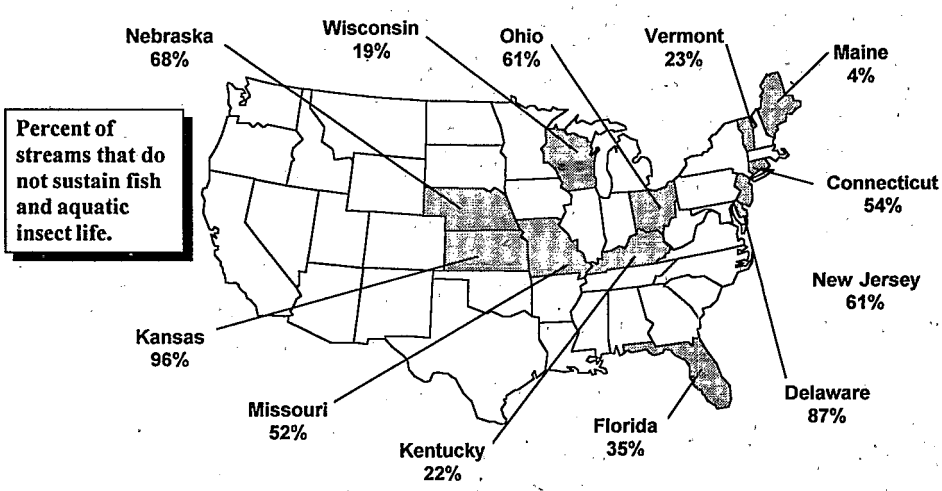
Refining Aquatic Life Uses

As a biocriteria program grows, the accumulated information helps state or tribal biologists refine the aquatic life use categories initially developed. That is, the additional information about the distribution and status of biota helps resource managers refine their categories of aquatic life use. The development of the "outstanding resource waters" category in North Carolina is an illustration of this process in which a less natural and diverse community characterizes the aquatic life use. Information obtained through biological surveys is used to explicitly characterize each aquatic life use. Other examples follow.

Oregon is presently developing state surface water categories based on aquatic life classifications. The proposed language for biological criteria in Oregon separates water resources into two categories. The first classification ("Outstanding Resource Waters") is for waters that shall be managed so that "resident biological communities . . . remain as they naturally occur and all indigenous aquatic species are protected and preserved."

The second category is for all other waters of Oregon. Waters in this class meet their use requirement if and when the following statement is applicable: "other waters of the state, including waters outside designated mixing zones, shall be of sufficient quality to support aquatic species without detrimental changes in the resident biological communities" (Oregon Dep. Environ. Qual. 1991).

Maine has established four classes of water quality for streams and rivers (Table 7-2). The "high quality waters" of Maine are separated into two categories: one category contains waters meeting the highest goal of



Source: State 305(b) Reports, 1992-1994

Figure 8-2.—Examples from some states using biological assessments to determine aquatic life use support in rivers and streams. Failure to sustain fish and aquatic life is defined with respect to the reference condition in that state.

the Water Quality Act (no discharge, Class AA); the other contains waters of high integrity but minimally impaired by human activity (Class A). "Good quality water" is assigned to the second category: Class B. Waters in Class B meet their aquatic life use requirement if and when all indigenous aquatic species are supported and only nondetrimental changes in community composition occur. The fourth category Class C, is reserved for the lowest quality waters. Waters in this class also meet their use requirement if and when all indigenous aquatic species are supported. However, changes in species composition may occur in Class C waters, even though the structure and function of the aquatic community must be maintained (Davies et al. 1991).

These classifications and their refinement depend on a well-established biocriteria program supported by regular, representative biosurveys. In fact, the procedure has been so successful that some states are shifting from only chemical sampling to an emphasis on biological monitoring for their 305(b) assessments. In their water quality assessment reports to Congress in 1992 and 1994, several states used biological assessments to determine the extent of attainment or nonattainment of the aquatic life use designations for their streams (Fig. 8-2). These data should not be used for comparing one state to another because the data — and hence the figures listed in Figure 8-2 — refer to assessed waters only, not to all waters in a given state.

The biocriteria process is a fundamental tool for assessing aquatic life use impairment.

Judging Use Impairment

A key element of water resource management under the Clean Water Act is the establishment and enforcement of standards to protect the nation's surface waters. If these state-developed standards are not met, legal action may be taken against dischargers to protect or restore the water resource. Criteria are scientifically based benchmarks upon which the standards are based, and biological criteria are benchmarks arrived at from direct meas-

urements of the responses of resident fish and other organisms to conditions in the water. Chemical, physical, and whole effluent criteria are indirect or surrogate measurements of degradation based on the amount of pollutant present in the waters, not the actual condition of the biota.

Biocriteria are designed to reflect the designated use of the water resource selected by the state so failure to meet these criteria is a violation of the standards derived from them. Thus, the biocriteria process is a fundamental tool for directly assessing aquatic life use impairment.

In Ohio, use attainment or nonattainment is determined using biocriteria based on both macroinvertebrates and fish. Full use attainment occurs if all criteria are met. Partial use attainment occurs if one assemblage meets its criteria though the other does not. The status is nonattainment if none of the biocriteria are met, or if one assemblage indicates poor or very poor performance, even though the other indicates attainment.

CASE STUDY — Ohio

STATE	LOCATION	DATES
Ohio	Upper Hocking River	1982–1991

The Hocking River basin covers 1,197 square miles in southeast Ohio, and flows through the cities of Lancaster, Logan, Nelsonville, and Athens; each city maintains wastewater treatment facilities (WWTPs) that discharge into the river (Clayton Environmental Consultant, 1992). Historically, the upper Hocking River near Lancaster has been one of the most severely degraded river segments in the state (Ohio Environ. Prot. Agency, 1982). Throughout the 1970s and early 1980s, the river was severely impacted by industrial effluent, combined sewer overflows (CSOs) and inadequate treatment at the Lancaster WWTP (Ohio Environ. Prot. Agency, 1985). The severe chemical impacts — low dissolved oxygen, and high levels of ammonia, lead, cyanide, cadmium, and phenolics — resulted in gross organic enrichment, heavy metal contamination, significant levels of in-stream toxicity, and periodic fish kills. Invertebrate studies of this portion of the river revealed a severely degraded biological condition with little downstream recovery (Fig. 8-3).

Consequently, the city of Lancaster began upgrading its WWTP in 1986 and reached full operation in 1989. The upgrades, sewer rehabilitation, elimination of bypasses, and the addition of a pretreatment program to remove metals, substantially improved both the water quality and the resident aquatic communities.

The Upper Hocking River has since exhibited the greatest improvement in biological performance of any river system in the state, although its recovery is not yet complete. In 1982, the biological communities downstream of the Lancaster WWTP and CSOs reflected the grossly polluted and acutely toxic conditions. None of the 20.5 miles from Lancaster to Logan attained their WWH standard, and 75 percent of them were in poor or very poor condition. In 1990, only 8.7 miles were still in the nonattainment category, while the rest achieved partial or full attainment and the average ICI score for that portion of the river rose from 6.9 to 42, a sevenfold improvement in the invertebrate community index (ICI).

Macroinvertebrate community performance (as measured by the ICI) improved dramatically, largely in response to the improved water quality. The fish community has substantially improved as well, although serious

Biocriteria establish conditions based on attributes of the resident biota which protect the level of aquatic life designated for the water resource by a state or tribe. Failure to meet the biocriteria is evidence of an impaired water resource.

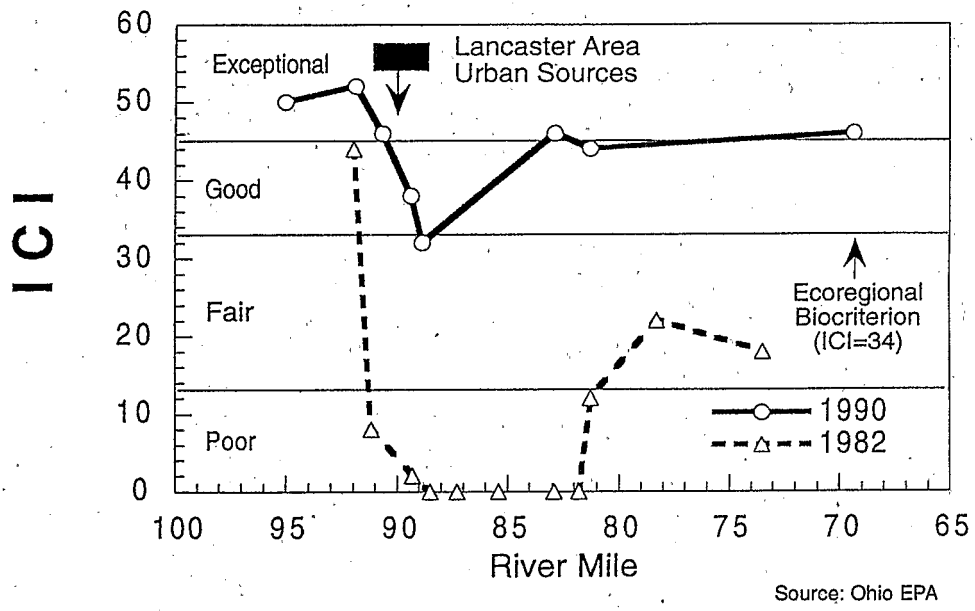


Figure 8-3.—Temporal trends in the improvement of the Upper Hocking River, 1982 - 1990 (adapted from Ohio EPA).

habitat alterations (e.g., channelization, bank erosion, and siltation) continue to inhibit silt-sensitive species. As seen in Figure 8-3, the biocriteria process with its well-defined criterion, careful surveys, and documented biotic indices clearly reveals not only impairment, but management response efforts and the magnitude of the subsequent recovery.

Diagnosing Impairment Causes

An underlying theme of biosurveys and biocriteria is to demonstrate the type and extent of impairment at the sites being evaluated so that proper management can be initiated. This demonstration can be done by comparing the attributes of aquatic communities at these sites with those found at sites that are unimpaired or minimally impaired. All human-induced alterations affect biological integrity simply by impacting the five environmental factors that affect and determine water resource quality. As discussed in chapter 5, the environmental factors of importance to the stream biota are the site's

- energy base
- chemical constituents
- habitat structure
- flow regime, and
- biotic interactions.

These factors not only influence the aquatic biota; they also affect other elements and processes that normally occur along the stream or river gradient.

Their identification provides an important indicator of the type, locale, and extent of remedial or protective management efforts that should be

An underlying theme of biosurveys and biocriteria is to demonstrate the type and extent of impairment at study sites so proper management can be initiated.

taken. For example, anthropogenic impairment may result from nutrient runoff of fertilizers; improper use or disposal of chemical toxins; conversion to cropland or other land use modifications; flow alterations; or overfishing. The evaluation of biological and habitat data collected in the biosurvey-biocriteria process can help reveal these causative elements. For example, the biological data will suggest whether overfishing or stocking are factors, or whether disease (which is not strictly anthropogenic) may also be a contributing factor. The habitat data will divulge any structural or sedimentation rate changes, and attendant or subsequent water quality tests will further define toxic or other problems of chemical origin.

An example in West Virginia involved stream degradation resulting from sewage, mining, and urbanization (Leonard and Orth, 1986). Here fish assemblage measurements were indexed in a "cultural pollution index" or CPI (derived from the IBI) to assess watershed and stream quality based on the assumption that assemblage features change consistently with stream degradation. Some fish community attributes respond more quickly than others to stream degradation (Angermeier and Karr, 1986; Karr et al. 1986). However, each metric of the index is sensitive within a different range of stream degradation. In these small coolwater streams of West Virginia, the CPI was sufficiently broad to rank the degree of degradation variously caused by mining, sewage, and urbanization. This study indicates that biotic indexes and criteria can be developed to reflect both the characteristics of regional fish populations and the particular forms of pollution or disruption they encounter.

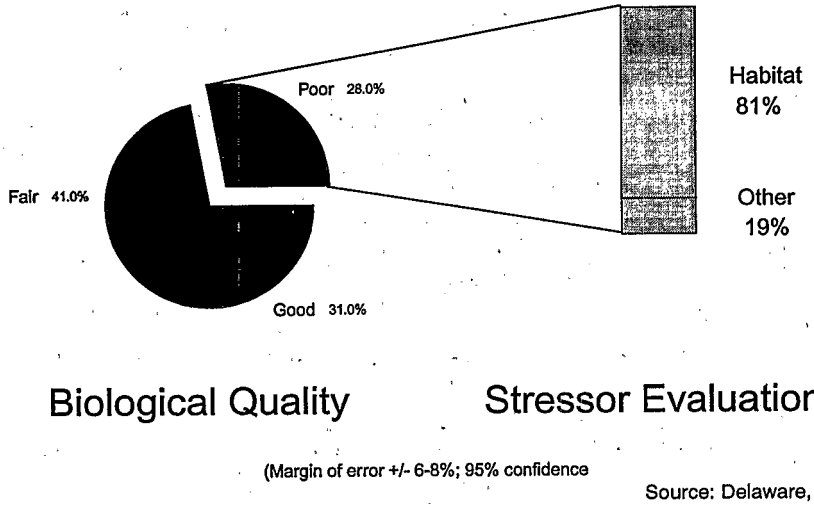
Human-induced alterations may occur as chemical contamination (point or nonpoint) or as a variety of other effects such as flow alteration or habitat modification.

CASE STUDY — Delaware

STATE	LOCATION	DATES
Delaware	Statewide	1991–1994

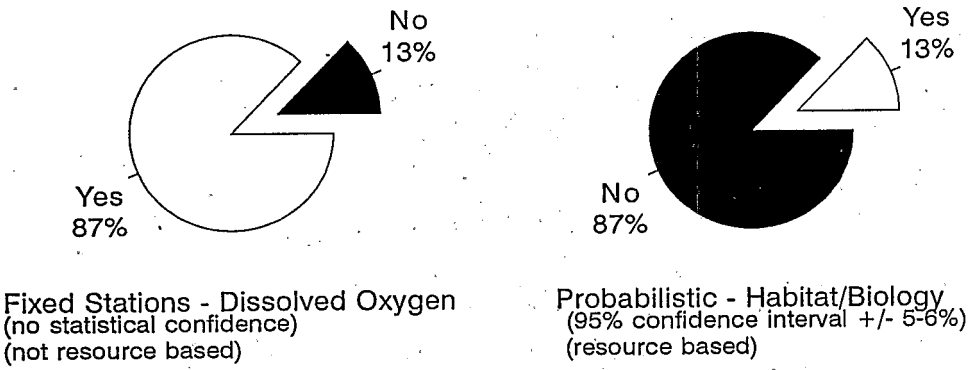
In 1994, the Delaware Department of Natural Resources and Environmental Control (DNREC) completed an assessment of the physical habitat conditions of nontidal streams throughout the state. Based on a sampling of 189 sites, only 13 percent were found to be in "good" condition while 87 percent were found to be in either "fair" or "poor" condition. "Good" conditions were defined as comparable to reference conditions. These results have a 95 percent confidence interval of plus or minus 6 to 8 percent. Results were also reported separately for each of the three Delaware counties and for the Piedmont and Coastal Plain ecoregions. The impairment in the Piedmont ecoregion was caused by urbanization and stormwater while the impairment in the Coastal Plain was caused by agriculture and channelization. This assessment is published as Appendix D of the state's 1994 305(b) report.

This information builds on biological data collected at the sites in the Coastal Plain in 1991 and published in the state's 1992 305(b) report. This report concluded that 72 percent of the nontidal streams in Kent and Sussex Counties (Coastal Plain ecoregion) had "good" macroinvertebrate communities compared to 28 percent that were determined to be in "fair" or "poor" condition. Further analysis has shown that degraded physical habitat was the principle cause of the biological impairment; 81 percent of the sites with "poor" biology had "poor" physical habitat (Fig. 8-4). Further water quality studies have implicated the loss of shade and its effects on dissolved oxygen and temperature as key factors that contributed to



Source: Delaware, 1992

Figure 8-4.—Assessment summary, Kent and Sussex Counties, Delaware, 1991.



Source: Delaware, 1994

Figure 8-5.—State of Delaware 1994 305(b) report, aquatic life use attainment — all nontidal streams.

the biological impairment. A statewide survey of the biological condition of nontidal streams is currently under development.

Prior to the use of biological and physical habitat measures, Delaware used dissolved oxygen (DO) to judge attainment or nonattainment of aquatic life uses. In the 1994 305(b) report, the state reported that 13 percent of its streams were not attaining aquatic life uses based on DO data. However, 87 percent were found to be impaired based on biological and physical habitat measures (Fig. 8-5). The lower estimate of impairment using DO results from (1) sampling during the day when DO levels are the highest, (2) disproportionate sampling of larger streams with better habitat and more assimilative capacity than smaller streams, and (3) a focus on point sources many of which are meeting permit limitations. The higher estimate of impairment using biological criteria and supporting biological community measurements helped reveal a cause of degradation that might not have been identified by other methods. It reflects the impact of nonpoint source activities, primarily urbanization (stormwater) and agriculture, on the state's nontidal streams.

Problem Identification

Monitoring the status and condition of resident communities over time is important to assess trends in the quality of the biota, whether to guard against further degradation or to measure improvement. In the course of such routine monitoring, new problems or conditions are often discovered. In fact, the Florida Department of Environmental Regulation has a specific (unpublished) program underway to determine the environmental damage (or lack thereof) caused by all significant point source discharges in the state. When the Florida DER began permitting point source discharges, staff relied mainly on compliance with numerical chemical standards. Over time, the need to evaluate the effects of these discharges on receiving waters has increased, both to ensure adequate environmental protection and to set priorities for enforcement or remedial action. Emphasis will be placed on detecting losses of biotic integrity through measures of imbalance in the flora and fauna, effects of toxic materials, dominance of nuisance species, and high populations of microbiological indicators.

Monitoring the status and condition of resident communities over time is important to assess trends in the quality of the biota, whether to guard against further degradation or to measure improvement.

A two-tiered approach is being used in the Florida program to detect environmental disturbances in receiving waters. Preliminary investigations (screening phase) involve qualitative sampling and analysis of benthic macroinvertebrate assemblages. A reference or background station is established for comparison with an area downstream of a discharge. Using the results of this relatively low intensity investigation, site impairment is ranked from "no" to "moderate" to "severe." If necessary, subsequent studies on dischargers (definitive phase) will use a more quantitative, multiparameter sampling regime. According to the Florida Department of Environmental Regulation, study parameters (such as macroinvertebrates, periphyton, macrophytes, bacteria, bioassays, sediment analysis, and physical and chemical analyses) are well suited for detection of violations.

The Arkansas Department of Pollution Control and Ecology addresses screening level monitoring using rapid bioassessment at paired stations that bracket pollutant sources for impact identification. As was shown in Figure 5-2, the initial rapid bioassessment screening may result in the application of other biological and chemical methods, after which an on-site decision can be made for subsequent action. In situations where "no impairment" or "minimal impairment" classifications are met, field efforts are discontinued until further information indicates a problem. Streams classified as "substantially" or "excessively" impaired trigger additional investigative steps that employ a variety of methods (Shackleford, 1988).

CASE STUDY — Maine

STATE	LOCATION	DATES
Maine	Piscataquis River	1984-1990

The Piscataquis River, with a drainage area of about 250 square miles northwest of Bangor, runs near the town of Guilford (Clayton Environmental Consultants, 1992). For many years, untreated manufacturing water from a textile mill and untreated domestic sewage from Guilford significantly impacted the river. In an attempt to improve the quality of the waterbody, the town of Guilford constructed a publicly owned treatment works (POTW), which was completed in June 1988. The POTW has aerated lagoons (detention time of 50 days) and a flow of 0.75 million gal-

lons per day (mgd). Seventy-five percent of the total inflow into the plant comes from textile mill waste; the remaining 25 percent from domestic sewage.

Maine's water quality standards designate a specific level of biological integrity that each class of water must maintain. To meet the standards for a Class A water, the aquatic community must be "as naturally occurs" and specific definitions are used to identify ecological attributes that may be tested to determine if the standards are being achieved.

Maine's Department of Environmental Protection uses a multivariate statistical model to predict the probability of attaining each classification. The model uses 31 quantitative measures of community structure, including the Hilsenhoff Biotic Index, Generic Species Richness, EPT, and EP values.

Monitoring of the Piscataquis River occurred at sites upstream and downstream of the textile mill in 1984, 1989, and 1990, and at a site downstream from the POTW in 1989 and 1990. Before 1988, benthic macroinvertebrate samples collected downstream of the mill revealed a severely degraded community consisting primarily of pollutant tolerant organisms. The macroinvertebrate samples indicated that the waterbody failed to meet the lowest aquatic life standards allowed by the state, although chemical water quality parameters (e.g., biochemical oxygen demand) collected at the site were meeting standards. Chemical parameters alone are insufficient to detect every water quality impairment.

Following the rerouting of the textile mill waste and the completion of the POTW in 1988, the river recovered quickly. Monitoring data, collected during the summer of 1989, revealed a substantially improved macroinvertebrate community (Fig. 8-6). Pollution-sensitive organisms were abundant and EPT values had increased from 1 in 1984 to 17 to 20 in 1989 and 1990. The generic richness improved from 6.35 in 1984 to 38 in 1990. The site now fully supports the aquatic life standards of Class A waters.

Other Applications of the Process

■ **Regulatory Assessments.** The biocriteria process is excellent for assessing the adequacy of NPDES permits to accomplish their intended purpose. As indicated earlier in this text, biological parameters are not recommended as permit limits at this time. But an ideal way to evaluate the success of the permit is to compare downstream biota to upstream or regional reference conditions and biological criteria. If the biota are not sufficiently protected as indicated by a downstream survey, the permit should be reviewed and perhaps revised. This biological review should be scheduled each time a permit is due for renewal.

■ **Management Planning.** This application was implied in several of the examples used in this chapter. Streams in a particular ecoregion can be ranked on the basis of their index scores and relative compliance with biocriteria. The natural resource manager can then assign priorities to individual streams or groups of streams for protection, further investigations, or remedial management depending on the availability of personnel and funding resources. That is, a rational decision with a reasonable ex-

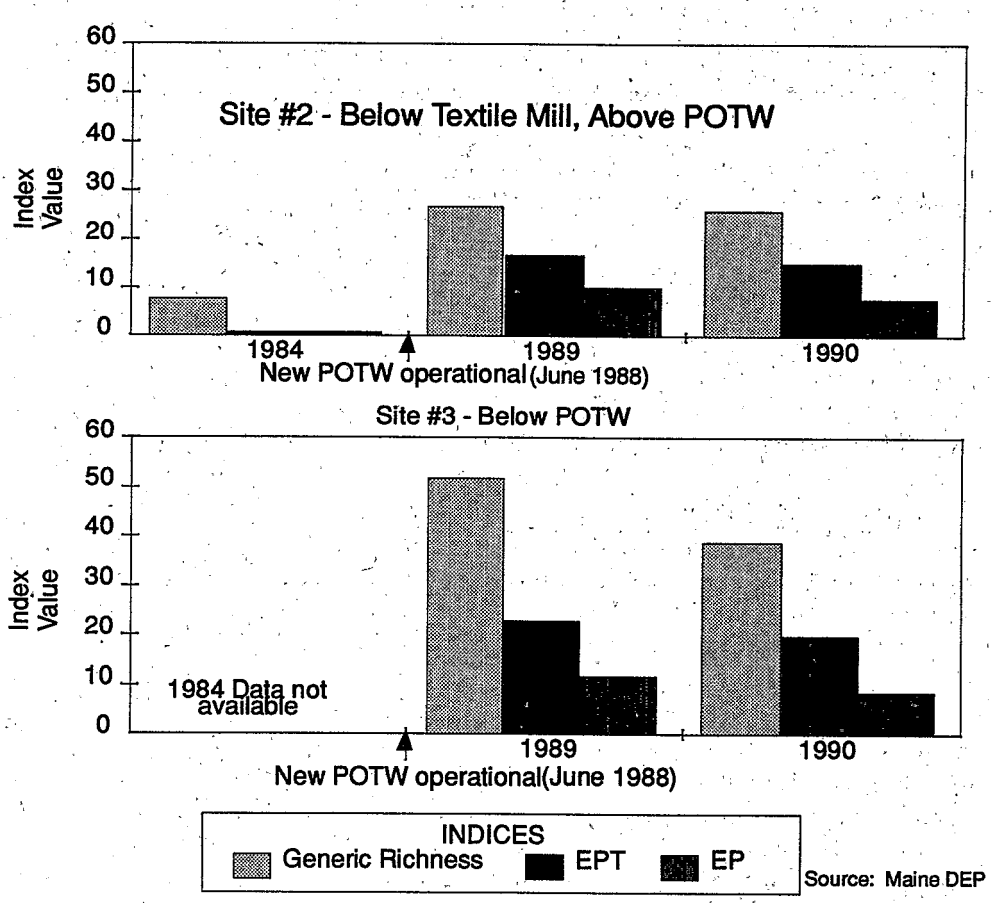


Figure 8-6.—Macroinvertebrates in the Piscataquis River, Maine, 1984-1990. New sewage treatment plant became operational in June 1988 (arrow).

pectation of results can be used to determine which streams will receive attention in any given year.

■ **Water Quality Project and Techniques Evaluation.** When a management plan is implemented, the changed land use practices, bank erosion control structures, and effluent diversion or treatment practices applied can be evaluated for effectiveness by applying the biocriteria process as a "before," "during," and "after" monitoring scheme. If results are as hoped for — as they were, for example, in the Maine case study — the manager can apply the technique to similar problems on other streams. If there is little or no change in the biota, more work is indicated and the technique obviously is not ready for application elsewhere.

■ **Status and Trends Documentation.** This task is one of the primary functions of the biocriteria process and should not be overlooked in discussing other uses of the approach. As an ongoing program, the biosurvey-biocriteria process provides perhaps the best, most direct and comprehensive assessment of water resource condition available to us. Annual surveys of the biota not only refine the biocriteria, but are the basis of state and EPA reports to the nation on the status of surface waters and on our relative success or failure to protect these valuable resources.

Suggested Readings

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Glossary

The development of water quality standards and criteria requires clear understanding of key terms and concepts. Foremost is the differentiation between water quality standards and criteria. A *standard* is a legally established state regulation consisting of two parts: (a) designated uses and (b) criteria. A *designated use* is a classification designated in water quality standards for each waterbody or segment that defines the optimal purpose for that waterbody. Examples of designated uses for particular waterbodies are drinking water use and aquatic life use. *Criteria* are statements of the conditions presumed to support or protect the designated use or uses. In practice, if the conditions specified by the criteria are met, the designated use should be supported.

Biocriteria require additional understanding and a common frame of reference for effective development and use in a water quality standards framework. The following definitions provide this frame of reference, and should be carefully considered to ensure consistent interpretation of concepts and terminology.

An *acceptable/unacceptable threshold* is the minimum measured level at which some condition can be differentiated such that the target location is or is not considered reasonable for maintenance of the designated use. The magnitude of impairment is not addressed with a threshold determination.

Ambient monitoring is sampling and evaluation of receiving waters not necessarily associated with episodic perturbations.

An *aquatic assemblage* is an association of interacting populations of organisms in a given waterbody, for example, fish assemblage or a benthic macroinvertebrate assemblage.

Aquatic biota is the collective term describing the organisms living in or depending on the aquatic environment.

An *aquatic community* is an association of interacting assemblages in a given waterbody, the biotic component of an ecosystem (see also *aquatic assemblage*).

Assemblage structure is the make-up or composition of the taxonomic grouping such as fish, algae, or macroinvertebrates relating primarily to the kinds and number of organisms in the group.

Autotrophic refers to the trophic status, the balance between production and consumption where production within the system exceeds respiration.

Autotrophic systems are those systems for which the primary nutrient source of fixed carbon is intrinsic, such as streams in which there is abundant growth of algae or macrophytes.

A *biogeographic region* is any geographical region characterized by a distinctive flora and/or fauna (see also *ecoregion*).

A *bioindicator* is an organism, species, assemblage, or community characteristic of a particular habitat, or indicative of a particular set of environmental conditions.

Biological assessment is an evaluation of the condition of a waterbody using biological surveys and other direct measurements of the resident biota in surface waters.

Biological criteria, or *biocriteria*, are numerical values or narrative expressions that describe the reference biological condition of aquatic communities inhabiting waters of a given designated aquatic life use. Biocriteria are benchmarks for water resources evaluation and management decision making.

Biological integrity is functionally defined as the condition of an aquatic community inhabiting unimpaired waterbodies of a specified habitat as measured by an evaluation of multiple attributes of the aquatic biota. Three critical components of biological integrity are that the biota is (1) the product of the evolutionary process for that locality, or site, (2) inclusive of a broad range of biological and ecological characteristics such as taxonomic richness and composition, trophic structure, and (3) is found in the study biogeographic region.

Biological monitoring, or *biomonitoring*, is the use of a biological entity as a detector and its response as a measure to determine environmental conditions. Toxicity tests and ambient biological surveys are common biomonitoring methods.

A *biological response signature* is a unique combination of biological attributes that identify individual impact types or the cumulative impacts of several human influences.

A *biological survey*, or *biosurvey*, consists of collecting, processing, and analyzing representative portions of a resident biotic community.

A *biomarker* is any contaminant-induced physiological or biochemical change in an organism that leads to the formation of an altered structure (a lesion) in the cells, tissue, or organs of that individual or change in genetic characteristics.

Channelization is the procedure of deepening and straightening stream or river channels through dredging. In some states, channelization includes complete concrete lining of channel bottom, sides, and easements.

A *community component* is any portion of a biological community. The community component may pertain to the taxonomic group (fish, invertebrates, algae), the taxonomic category (phylum, order, family, genus, species, stock), the feeding strategy (herbivore, omnivore, predator), or the organizational level (individual, population, assemblage) of a biological entity within the aquatic community.

A *confidence interval* is an interval that has the stated probability (e.g., 95 percent) of containing the true value of a fixed (but unknown) parameter.

Data quality objectives (DQOs) are qualitative and quantitative statements developed by data users to specify the quality of data needed to support specific decisions; statements about the level of uncertainty that a decisionmaker is willing to accept in data used to support a particular decision. Complete DQOs describe the decision to be made; what data are required, why they are needed, the calculations in which they will be used; and time and resource constraints. DQOs are used to design data collection plans.

Degradation is any alteration of ecosystems such that chemical, physical, or biological attributes are adversely affected.

Degree days are units used in measuring the duration of a life cycle or growth stage of an organism; they are calculated as the product of time and temperature averaged over a specified interval.

A designated use is a classification specified in water quality standards for each waterbody or segment relating to the level of protection from perturbation afforded by the regulatory agency.

Diversity is the absolute number of species in an assemblage, community, or sample; species richness (see also *taxa richness*).

Ecological assessment is a detailed and comprehensive evaluation of the status of a water resource system designed to detect degradation and, if possible, identify the causes of that degradation.

Ecological health is the degree to which the inherent potential of a biological system is realized, the dynamic equilibrium of system processes is maintained, and a minimal amount of external support for management is needed.

Ecological integrity is the condition of an unimpaired ecosystem as measured by combined chemical, physical (including habitat), and biological attributes.

Ecoregions, or *regions of ecological similarity*, are defined by similarity of climate, landform, soil, potential natural vegetation, hydrology, or other ecologically relevant variables.

Ecoregionalization — See *regionalization*.

Elements are the richness of items that make up biological systems, measured as number of kinds.

Generalists are organisms that can utilize a broad range of habitat or food types.

Heterotrophic input refers to the trophic status, the balance between production and consumption where respiration within the system exceeds production.

Heterotrophic systems are those systems for which the primary nutrient source of fixed carbon is extrinsic, such as streams for which the main source of organic input is from riparian vegetation in the form of leaf litter and woody material.

Historical data are datasets existing from previous studies, which can range from handwritten field notes to published journal articles.

Hyporheic pertains to saturated sediments beneath or beside streams and rivers.

An *impact* is a change in the chemical, physical (including habitat), or biological quality or condition of a waterbody caused by external sources.

An *impairment* is a detrimental effect on the biological integrity of a waterbody caused by an impact that prevents attainment of the designated use.

Level of uncertainty pertains to the confidence, or lack thereof, that data from an assessment will support the conclusions.

Macroinvertebrates are animals without backbones of a size large enough to be seen by the unaided eye and which can be retained by a U.S. Standard No. 30 sieve (28 meshes per inch, 0.595 mm openings).

Macrophytes are large aquatic plants that may be rooted, unrooted, vascular, or algiform (such as kelp); includes submerged aquatic vegetation, emergent aquatic vegetation, and floating aquatic vegetation.

A *metric* is a calculated term or enumeration representing some aspect of biological assemblage structure, function, or other measurable aspect; a characteristic of the biota that changes in some predictable way with increased human influence; combinations of these attributes or metrics provide valuable synthetic assessments of the status of water resources.

Minimal effluent dilution occurs in low flow conditions in which there is a lower quantity of water and thus a decreased ability for receiving waters to lower concentration levels of discharged compounds.

Minimally impaired is a term used to describe sites with slight anthropogenic perturbation relative to the overall region of study.

Mutualism is a form of symbiotic relationship in which both organisms benefit, frequently entailing complete interdependence.

Narrative biocriteria are general statements of attainable or attained conditions of biological integrity and water quality for a given use designation (see also *biocriteria*).

Nonpoint source is the origin of pollution in diffuse sources such as agriculture, forestry, and urbanization. Such pollution is transported by rainfall or snowmelt runoff carrying pollutants overland or through the soil.

Numeric biocriteria are numerical indices that describe expected attainable community attributes for different designated uses (see also *biocriteria*).

Organic pollution results from the presence of living substances in a stream or other waterbody at higher than natural background levels because of anthropogenic activities.

Paleoecological data are records derived from ancient or fossil remains discovered in lake sediments, including, for example, the fossilized remains of diatoms, pollen, seeds, or arthropod exoskeletal fragments. (Arthropoda are the phylum of invertebrate animals with jointed limbs, such as crustaceans and spiders.)

Performance effect criteria are judgment criteria that weigh the effectiveness of a project activity or function; determination of proper functioning.

Periphyton is a broad organismal assemblage composed of attached algae, bacteria, their secretions, associated detritus, and various species of microinvertebrates.

Processes (or biotic processes) pertain to ecological and evolutionary activities that naturally organize and regulate biological systems at all levels from genetic to landscape; examples are production, food acquisition, biotic interactions, and recruitment.

Production is the increase in biomass (somatic growth plus reproduction) of an individual, population, or assemblage.

Point source is the origin of pollutant discharge that is known and specific, usually thought of as effluent from the end of a pipe.

A *population* is an aggregate of individuals of a biological species that are geographically isolated from other members of the species and are actually or potentially interbreeding.

Quality assurance (QA) includes quality control functions and involves a totally integrated program for ensuring the reliability of monitoring and measurement data; the process of management review and oversight at the planning, implementation, and completion stages of environmental data collection activities. Its goal is to assure that the data provided are of the quality needed and claimed.

Quality control (QC) refers to the routine application of procedures for obtaining prescribed standards of performance in the monitoring and measurements process; focuses on the detailed technical activities needed to achieve data of the quality specified by data quality objectives. Quality control is implemented at the bench or field level.

Range control refers to quality control activity through which measurement values are kept within the range of natural or normal variability; control of operator variability.

Reasonably attainable refers to the ability of an aquatic resource to attain its expected potential.

A *reference condition* is the set of selected measurements or conditions of minimally impaired waterbodies characteristic of a waterbody type in a region.

A *reference site* is a specific locality on a waterbody which is minimally impaired and is representative of the expected ecological integrity of other localities on the same waterbody or nearby waterbodies.

Regionalization or *ecoregionalization* is a procedure for subdividing a geographic area into regions of relative homogeneity in ecological systems or in relationship between organisms and their environment.

Regulated flow of a stream or river is that for which the quantity of water moving within its banks is a function of anthropogenic activity, usually associated with dams and reservoirs.

Residuals are the differences between a value predicted by regression and an observed value.

Respiration is the energy expenditure for all metabolic processes. Matter and energy are returned to the environment by respiration; matter as CO_2 and water, and energy as heat.

A *riparian zone* is an area that borders a waterbody.

Streams, as defined for the purpose of this document, are small lotic systems that can be waded by field investigators.

Targeted assemblage approach refers to an assessment procedure that has as its focus of sampling a selected component of the biological community.

A *targeted community segment* is the component of the community, such as a taxonomic category, trophic level, guild, or other designation, that is the focus of a bioassessment.

Taxa richness refers to the number of distinct species or kinds (taxa) that are found in an assemblage, community, or sample (see also *diversity*).

Termination control points are quality control elements that indicate when and where nonvalid procedures are being used or data are being collected and indicate necessary changes in procedures.

A *test site* is the location under study of which the condition is unknown and suspect of being adversely affected by anthropogenic influence.

A *vegetated buffer zone* is a planted or naturally vegetated strip of land between some feature (usually a waterbody) and another landform or habitat that has been altered by human activity (e.g., agricultural fields, roadways, asphalt parking lots, residential areas).

A *water resource assessment* is an evaluation of the condition of a waterbody using biological surveys, habitat quality assessments, chemical-specific analyses of pollutants in waterbodies, and toxicity tests. These environmental assessments may be diverse or narrowly focused depending on the needs of the evaluation, and the probable sources of degradation.

Zooplankton refers to animals which are unable to maintain their position or distribution independent of the movement of water or air.

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