Use of Biological Information to Better Define Designated Aquatic Life Uses in State and Tribal Water Quality Standards: Tiered Aquatic Life Uses

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Preface

Our Nation's waters are a valuable ecological resource. Protecting them begins with State and authorized Tribal adoption of water quality standards. This draft document, the *Use of Biological Information to Better Define Designated Aquatic Life Uses in State and Tribal Water Quality Standards: Tiered Aquatic Life Uses*, provides up-to-date information on practical, defensible approaches to help States and Tribes more precisely define designated aquatic life uses in their water quality standards. Biologically-based tiered aquatic life uses, based on the scientific model presented in this document, can help States and Tribes develop aquatic life uses that more precisely describe the existing and potential uses of a waterbody and then use bioassessments to help measure attainment of the uses.

Biologically-based tiered aquatic life uses coupled with numeric biological criteria provide a direct measure of the aquatic resource that is being protected. The condition of the biota reflects the cumulative response of the aquatic community to individual or multiple sources of stress – an environmental outcome measure. The technical approaches described in this document support U.S. EPA's Environmental Indicators Initiative to move the Agency closer to a performance-based rather than process-based environmental protection system (http://www.epa.gov/indicators). Launched in November 2001, the Environmental Indicators Initiative responds to the President's call to have agencies and departments manage for results by measuring environmental outcomes.

This document is a compilation of the tools, practices, and experiences of State and Tribal scientists who have used biological information to more precisely define their aquatic life uses. The presented model brings biological condition and stressor information together to inform decisions on use designation. The document fulfills a commitment in the U.S. EPA Water Quality Standards Strategy to provide technical support, outreach, training, and workshops to assist States and Tribes with designated uses, including use attainability analyses and tiered aquatic life uses (EPA-823-R-03-010, Strategic Action #7, Milestone #2).

U.S. EPA encourages States and Tribes to incorporate biological information into their decisions. U.S. EPA believes the use of bioassessments will help improve water quality protection. The information in this document can help States and Tribes use bioassessments to more precisely define their aquatic life uses and communicate this information to the public. U.S. EPA is making this document available so States and Tribes can pilot a bioassessment-based tiered approach to defining their designated aquatic life uses. If you choose to undertake a pilot, U.S. EPA would appreciate hearing about your experience. We are interested in feedback on the following questions:

- Is this document helpful in addressing current issues in your program?
- Does this document address the technical challenges in your Region, State, or Tribe?
- How can this document be improved to help you develop tiered aquatic life uses in your program?
- What additional information would be helpful to you?

Should you have any questions or wish to provide feedback, please contact Susan K. Jackson via email at Jackson.Susank@epa.gov or at the following address:

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Executive Summary

This document provides up-to-date information on how States and Tribes can use biological information to more precisely define designated aquatic life uses for their waters. Thirty years ago, under the Clean Water Act (CWA), States and Tribes were required to adopt in their water quality standards, where attainable, designated uses that included the protection and propagation of fish, shellfish, and wildlife. During the 1970s, the biological goals adopted into State or Tribal water quality standards as designated aquatic life uses may have been appropriately general (e.g., "aquatic life as naturally occurs") given the limited data available and the state of the science. However, while such general use classifications meet the requirements of the Clean Water Act and the implementing federal regulations, they may constitute the beginning, rather than the end, of appropriate use designations. Improved precision may result in more efficient and effective evaluation of attainment of condition and utilization of restoration resources. Finally, improved precision in uses can enhance demonstrating progress towards management goals. In the years since the CWA was passed, considerable advancements have been made in the science of aquatic ecology and in biological monitoring and assessment methods. This document summarizes these advancements and provides a scientific model that States and Tribes can use to refine their designated uses in a manner that can improve their water quality assessment and management.

This document was developed based on the technical expertise and practical experience of State and Tribal scientists. In 2000, the U.S. EPA convened a technical expert workgroup, including State and Tribal scientists, to identify scientifically sound and practical approaches to help States and Tribes provide more specificity in their designated aquatic life uses. The workgroup developed a scientific model, the Biological Condition Gradient (BCG), which describes biological response to increasing levels of stressors. The model describes how ten attributes of aquatic ecosystems change in response to increasing levels of stressors. The attributes include several aspects of community structure, organism condition, ecosystem function, and spatial and temporal attributes of stream size and connectivity. The gradient can be considered analogous to a field-based dose-response curve where dose (x-axis) = increasing levels of stressors and response (y-axis) = biological condition. The BCG differs from the standard dose-response curve, in that the BCG does not represent the laboratory response of a single species to a specified dose of a known chemical, but rather the *in situ* response of the biota to the sum of stresses it is exposed to. The BCG is divided into six tiers of biological condition along the stressor-response curve, ranging from observable biological conditions found at no or low levels of stress to those found at high levels of stressors. The model provides a common framework for interpreting biological information regardless of methodology or geography. When calibrated to a regional or state scale, States and Tribes can use this model to more precisely evaluate the current and potential biological condition of their waters and use that information to inform their decisions on aquatic life designations. Additionally, States and Tribes can use this interpretative model to more clearly and consistently communicate these decisions to the public.

Maine and Ohio have adopted biologically-based tiered aquatic life uses in their WQS and have over twenty years experience implementing this type of use designation approach. Both Maine and Ohio developed and adopted tiered aquatic life uses for similar reasons: 1) to incorporate ecologically relevant endpoints into decisions; 2) to inform water quality management decisions; 3) to quantify water quality improvements; and 4) to merge the design and practice of monitoring and assessment with the development and implementation of their water quality standards. Maine and Ohio scientists have identified a sequence of steps and milestones that U.S. EPA has compiled as a template that other States and Tribes may use to develop biologically-based tiered uses. Examples from Maine and Ohio are included in this document to illustrate how they used biological data to establish tiered uses and the programmatic gains from having done so.

The U.S. EPA encourages States and Tribes to incorporate biological information into their decisions. The U.S. EPA believes that the use of biological information can help improve water quality protection. Currently, States and Tribes that use biological data as part of their assessment program apply some type of tiered aquatic life use to guide their interpretation of their biological data. States and Tribes have either explicitly adopted tiers directly into their water quality standards as designated uses, or used tiers in monitoring and assessment of their surface waters. This document provides examples of practical and scientifically sound approaches to using biological information to tier designated aquatic life uses.

FOREWORD
Why is U.S. EPA publishing this document?

In the more than 30 years since the Clean Water Act (CWA) was passed, there has been considerable progress in the science of aquatic ecology and in the development of biological monitoring and assessment techniques. During the 1970s, the biological goals adopted into State or Tribal water quality standards as designated aquatic life uses may have been appropriately general (e.g., "aquatic life as naturally occurs") given the limited data available and the state of the science. However, while such general use classifications meet the requirements of the Clean Water Act and the implementing federal regulations, they may constitute the beginning, rather than the end, of appropriate use designations. Improved precision may result in more efficient and effective evaluation of attainment of condition and utilization of restoration resources. Finally, improved precision in uses can enhance demonstrating progress towards management goals. Tiered aquatic life uses, based on the biological condition gradient model presented in this document, can help States and Tribes to better define and develop more precise, scientifically defensible aquatic life uses that account for the natural differences between waterbodies and should result in more appropriate levels of protection for specific waterbodies.

States and Tribes have created different use classification systems ranging from a straightforward replication of the general uses identified in the CWA (e.g., protection and propagation of fish, shellfish, and wildlife; recreation; agriculture; industrial and other purposes, including navigation) to more complex systems that express designated uses in more specific terms or establish classifications which identify different levels of protection. For example, some States designate general "aquatic life" uses, while others subcategorize waters based on the expected biological assemblage. Some have established tiers representing different levels of biological condition (e.g., excellent, good, fair). Although a variety of defensible approaches have evolved and become established in State and Tribal programs, current U.S. EPA regulations are not specific about the level of precision States or Tribes must achieve in designating uses. This document is designed to help inform States and Tribes how to better define and improve the precision of their designated uses.

Over the past thirty years, both the state of aquatic science and the application of the science in State and Tribal water programs have advanced. Major areas of uncertainty in water management, such as distinguishing between natural variability and effects of stressors on aquatic systems as well as determining the appropriate level of protection for individual waterbodies, are being addressed. Many States and Tribes now use biological information to directly assess the biological condition of their aquatic resources (U.S. EPA 2002a). Three States have formally adopted biologically-based tiered aquatic life uses in their water quality standards. "Lessons learned" from two of these States indicate that implementation of tiered aquatic life uses supports more appropriate levels of protection for individual waters by promoting uses and criteria that are neither over- nor under-protective. U.S. EPA now recognizes that the States having implemented tiered aquatic life uses have significantly benefited from the approach. The use designation process needs to clearly articulate and differentiate intended levels of protection with enough specificity so that 1) decision makers can appropriately develop and implement their water quality standards on a site, reach, or watershed specific basis and 2) the public can understand, identify with, and influence the goals set for waters.

In 2001, the National Research Council (NRC) published its report on *Assessing the TMDL Approach to Water Quality Management* (NRC 2001). In the report, the NRC recommended tiering designated uses as an essential step in setting water quality standards and improving decision-making. The NRC, finding that the Clean Water Act's goals (i.e., "fishable," "swimmable") are too broad to serve as operational statements of designated use, recommended greater specificity in defining such uses. For example, rather than stating that a waterbody needs to be "fishable," the designated use would ideally describe the expected fish assemblage or population (e.g., cold water fishery, warm water fishery, or salmon, trout, bass, etc.) as well as the other biological assemblages necessary to support that fish population.

Additionally, the NRC recommended that biological criteria should be used in conjunction with physical and chemical criteria to determine whether a waterbody is meeting its designated use. The NRC described a "position of the criterion" framework, which reflects how representative a criterion is of a designated use according to its position along a conceptual causal pathway (Figure F-1). This alignment is comparable to that of performance (indicators of point source quality) versus impact standards (indicators of resource condition) (Courtemanch et al. 1989), or of stressor and exposure (effluent, chemical, and physical parameters) in contrast to response indicators (biological) (Yoder and Rankin 1998). In Figure F-1, stressor indicators correspond to box 1 and were termed effluent standards by the NRC. Pollutant-specific indicators that function as indicators of exposure and stress correspond to box 2. Biological indicators show responses to stress and exposure and correspond to box 3. Because designated uses are written in qualitative, narrative terminology, the challenge is to relate a criterion to the designated use. Establishing this relationship is easier as the criterion is positioned closer to the designated use, thus the NRC recommendation on the use of biological information to help determine more appropriate aquatic life uses and to couple the narrative use statements with quantitative methods. The "position of criterion" concept provides a useful construct for considering the relationship of water quality criteria (biological, chemical, and physical) to the designated uses they are intended to protect.

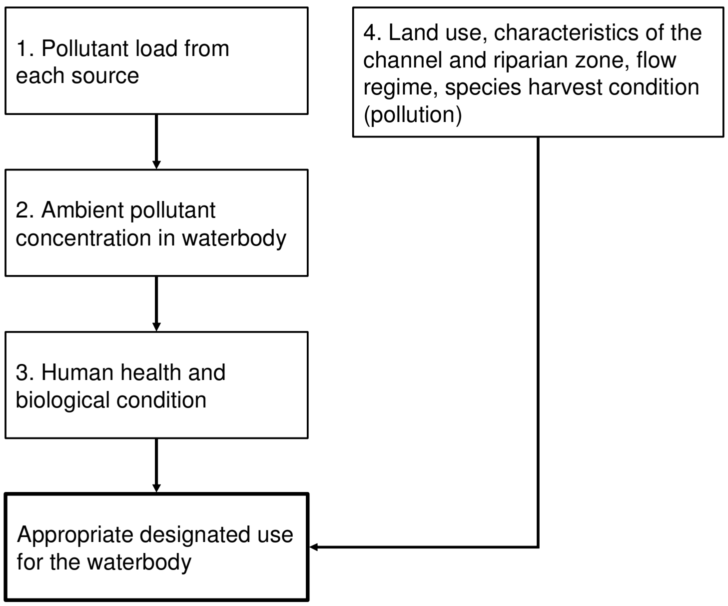

FIGURE F-1. Types of water quality criteria and their position relative to designated uses (after NRC 2001).

To help States and Tribes more precisely define use descriptions, there is a need to incorporate current scientific understanding of aquatic ecology and the appropriate use of monitoring data. To this end, the U.S. EPA convened a technical expert workgroup to identify scientifically sound and practical approaches that would help States and Tribes provide more specificity in their designated aquatic life uses. The workgroup met four times between 2000 and 2003. The workgroup, composed primarily of U.S. EPA, State, and Tribal scientists, also included research scientists from the U.S Geological Survey (USGS), the academic community, and the private sector. The workgroup was asked to base their recommendations on "lessons learned" from State and Tribal water programs in the development and the application of biologically-based aquatic life uses, bioassessments, and biocriteria. The workgroup developed a scientific model, the Biological Condition Gradient (BCG), which describes graduated tiers of biological response to increasing levels of stressors. This model was developed and tested through a series of data exercises using a diverse array of data sets. States and Tribes can use the BCG to more precisely define and set appropriate designated aquatic life uses for their waters.

During the final workgroup meeting in 2003, State and Tribal members discussed their current thinking on how using biological information to tier designated aquatic life uses could benefit their water quality management programs. The main reasons discussed included biologically-based tiered uses could help:

- set ecologically-based aquatic life goals for waterbodies;
- establish a consistent approach for identifying attainable, incremental restoration goals that are grounded in the concept of biological integrity;
- provide a framework that better relates traditional water quality criteria (stressor and exposure variables) and biological criteria (response variables) in determining use attainment, thus strengthening stressor/response models implicit in designated uses and criteria in water quality standards;
- better link monitoring and assessment with water quality standards; and
- prioritize management actions that result in the more effective use of resources.

When asked about the significant value-added outcomes of these benefits to their water programs, States and Tribes workgroup members anticipated being able to make more scientifically defensible listings of impaired waters as well as enhance identifying and protecting high quality waters. For several States, biologically-based tiered uses may help in the transition from reliance on current conditions in developing designated uses to being able to better consider the potential for improvement. Another important added value anticipated by all State and Tribal representatives was the ability to communicate more effectively with program managers, the public, and key stakeholders. Workgroup members expressed the opinion that biologically-based aquatic life uses could help maximize the return on their monitoring and assessment efforts by eliminating a major source of uncertainty in water quality management by 1) accounting for natural variability in aquatic systems and 2) helping to specify an appropriate level of protection for a waterbody that includes consideration of the system's potential for improvement.

Biologically-based aquatic life uses, as described in this document, are a natural evolution that reflects an improved understanding of surface waters resulting from more than 20 years of assessment data. The proposed approach will help better integrate the science of aquatic ecology into Water Quality Standards. This document represents the culmination of four years of workgroup deliberations, including four workgroup meetings and two workshops to "road test" the BCG model. Based on the collective experience of the workgroup members, the science and methods in the fields of biological assessments and criteria have progressed sufficiently over the past thirty-five years to support the use of biological information to tier designated aquatic life uses in State and Tribal water quality standards.

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Introduction

This chapter provides the background and rationale for using biological information to tier designated aquatic life uses and better define them in State and Tribal water quality standards. Ideally, the use designation process clearly articulates and differentiates intended levels of protection with enough specificity so that 1) decision makers can appropriately develop and implement their water quality standards on a reach or watershed specific basis; and 2) the public can understand, identify with, and influence the goals set for waters. In 2000, the U.S. EPA convened a technical expert workgroup, including State and Tribal scientists, to identify existing scientifically sound and practical approaches using biological information to better define aquatic life uses. The workgroup produced a scientific model, the Biological Condition Gradient (BCG), for interpreting biological response to increasing levels of stressors. The workgroup's findings are consistent with The National Research Council's call for greater specificity in water quality standards that can result in improved decision-making (NRC 2001). The BCG is intended to help States and Tribes develop more precise aquatic life uses that should result in more appropriate levels of protection for their surface waters.

CHAPTER 1. WHAT ARE TIERED AQUATIC LIFE USES?

Designated aquatic life uses are State or Tribal descriptions of the biological goals for their waterbodies. Tiered aquatic life uses (TALUs) use biological information to more precisely define these goals relative to natural conditions. Bioassessments can then be used to measure attainment of the goals. U.S. EPA's current thinking is that a system of tiered uses could:

- accommodate observable differences in expected biological condition in waterbodies in different ecological regions;
- provide an objective means of describing the biological potential for a specific waterbody;
- recognize and accommodate observable differences in biological potential among waters with different types and levels of stressors;
- reflect an understanding of the relationship between stressors and biological community response;
- guide selection of environmental indicators for monitoring and assessment and make full use of available biological data; and
- articulate a stressor-response model that maximizes the likelihood of success of water quality management actions based on water quality standards (assessment, 303(d) listings/TMDLS, NPDES permits).

Tiered aquatic life uses are based on general observations about aquatic communities that have become central to aquatic ecology and consistent with 30 years of empirical observations. These are:

- surface waters and the biological communities they support are predictably and consistently different in different parts of the country *(classification along a natural gradient, ecological region concept)*;
- within the same ecological regions, different types of waterbodies (e.g., headwaters, streams, rivers, wetlands) support predictably and consistently different biological communities *(waterbody classification)*;
- within a given class of waterbodies, observed biological condition in a specific waterbody is a function of the level of stress (natural and anthropogenic) that the waterbody has experienced *(the biological condition gradient discussed in this document)*;

- similar stressors at similar intensities produce predictable and consistent biological responses in waters within a class, and those responses can be detected and quantified in terms of deviation from an expected condition *(reference condition)*; and
- waterbodies exposed to higher levels of stressors will have lower biological performance compared to the reference condition than those waters experiencing lower levels of stress (*the biological condition and stressor gradients discussed in this document).*

The first three sections of this chapter provide the statutory and regulatory background of water quality standards, emphasizing the role of designated aquatic life uses. Section 1.4 explores how tiered biologically-based definitions can help set more appropriate and precise designated aquatic life uses in State and Tribal water quality standards. The next two sections discuss the primary products of the technical workgroup charged with identifying existing scientifically sound and practical approaches to help States and Tribes to better define and provide more precision in their designated aquatic life uses. Chapter 1 concludes with a summary of key points, organization of the document, and related technical support documents.

1.1 The CWA goals and objectives for aquatic life

One objective of the 1972 Clean Water Act (CWA) is to restore and maintain the chemical, physical, and biological integrity of the Nation's waters (CWA sec 101a). In the scientific literature, an aquatic system with chemical, physical, and biological integrity has been described as being capable of "supporting and maintaining a balanced, integrated, adaptive community of organisms having a composition and diversity comparable to that of the natural habitats of the region" (Frey 1977). Over the intervening years, our understanding of how to define and measure the integrity of aquatic systems has advanced. The term *integrity* has been further refined in the literature to mean a balanced, integrated, adaptive system having a full range of ecosystem elements (genes, species, assemblages) and processes (mutation, demographics, biotic interactions, nutrient and energy dynamics, metapopulation dynamics) expected in areas with no or minimal human influence (Karr 2000). The aquatic biota residing in a waterbody are the result of complex and interrelated chemical, physical, and biological processes that act over time and on multiple scales (e.g., instream, riparian, landscape) (Karr et al. l986, Yoder l995). By directly measuring the condition of the aquatic biota, we are able to more accurately define the aquatic community that is the outcome of all these factors.

To help achieve the integrity objective, the CWA also established an interim goal for the protection and propagation of fish, shellfish, and wildlife and recreation in and on the water. The protection and propagation interim goal for aquatic life has been interpreted by U.S. EPA to include the protection of the full complement of aquatic organisms residing in or migrating through a waterbody. As explained in U.S. EPA's *Questions and Answers on Antidegradation*, the protection afforded by water quality standards includes the representative aquatic community (e.g., fish, benthic macroinvertebrates, and periphyton):

> "The fact that sport or commercial fish are not present does not mean that the water may not be supporting an aquatic life protection function. An existing aquatic community composed entirely of invertebrates and plants, such as may be found in a pristine tributary alpine stream, should be protected whether or not such a stream supports a fishery. Even though the shorthand expression 'fishable/swimmable' is often used, the actual objective of the Act is to restore the chemical, physical and biological integrity of our Nation's waters (Section 101(a)). The term 'aquatic life' would more accurately reflect the protection of the aquatic community that was intended in Section 101(a)(2) of the Act." (Appendix G, EPA-823-B-94-005)

The representative community of aquatic organisms residing in, or migrating through, a waterbody will vary depending on the waterbody type. For example, fish, benthic macroinvertebrates, and, increasingly, periphyton are aquatic assemblages typically measured by States and Tribes when assessing streams and rivers. In headwater streams and many wetlands, amphibians are an important component of the biotic community and fish may be absent.

1.2 WQS statutory and regulatory background

Section 101(a) of the CWA establishes broad national goals and objectives such as the chemical, physical, and biological integrity objective. Other sections of the CWA establish the programs and authorities for implementation of those goals and objectives. Section 303(c) sets up the basis of the current water quality standards program. Water quality *standards* (WQS) are parts of State (or, in certain instances, federal) law that define the water quality goals of a waterbody, or parts of a waterbody, by designating the use or uses of the waterbody and by setting criteria necessary to protect the uses. The standards also include an antidegradation policy consistent with 40 CFR Part 131.12.

Although the CWA gives the U.S. EPA an important role in determining appropriate minimum levels of protection and providing national oversight, it also gives considerable flexibility and discretion to States and Tribes to design their own programs and establish levels of protection beyond the national minimums. Section 303 directs States and authorized Tribes to adopt water quality standards to protect public health or welfare, enhance the quality of water, and serve the purposes of the Clean Water Act. "Serve the purposes of the Act" (as defined in Sections 101(a), 101(a)(2), and 303(c) of the CWA) means that water quality standards should 1) include provisions for restoring and maintaining chemical, physical, and biological integrity of State and Tribal waters, 2) provide, wherever attainable, water quality for the protection and propagation of fish, shellfish, and wildlife and recreation in and on the water (i.e., "fishable/swimmable"), and 3) consider the use and value of State and Tribal waters for public water supplies, propagation of fish and wildlife, recreation, agricultural and industrial purposes, and navigation. Further requirements for water quality standards are at 40 CFR Part 131.

State WQS provide the foundation for water quality-based pollution control programs. With the public participating in their adoption (see 40 CFR 131.20), such standards serve the dual purposes of establishing the water quality goals for a specific waterbody, and serving as the regulatory basis for the establishment of water quality-based treatment controls and strategies beyond the technology-based levels of treatment required by Sections 301(b) and 306 of the CWA.

A waterbody's *designated use(s)* are those uses specified in water quality standards, whether or not they are being attained (40 CFR 131.3(f)). The "use" of a waterbody is the most fundamental description of its role in the aquatic and human environments. All of the water quality protections established by the CWA follow from the waterbody's designated use. As designated uses are critical in determining the water quality criteria that apply to a given waterbody, determining the appropriate designated use is of paramount importance in establishing criteria that are appropriately protective of that designated use.

Section 131.10 of the regulation describes States' and authorized Tribes' responsibilities for designating and protecting uses. The regulation:

- requires that States and Tribes specify the water uses to be achieved and protected,
- requires protection of downstream uses,
- allows for sub-category and seasonal uses,
- sets out minimum attainability criteria,
- lists six factors of, which at least one must be satisfied to justify removal of designated uses that are not existing uses,
- prohibits removal of existing uses,
- requires upgrading of uses that are presently being attained but not designated, and
- establishes conditions and requirements for conducting use attainability analyses.

In addition, the regulations effectively establish a "rebuttable presumption" that the uses of protection and propagation of fish, shellfish, and wildlife and recreation in and on the water are attainable and should apply to a waterbody, unless it has been affirmatively demonstrated that such uses are not attainable.

40 CFR 131.10(a) requires that States specify appropriate water uses to be achieved and protected. The classification of the waters of the State must take into consideration the use and value of water for public water supplies, protection and propagation of fish, shellfish, and wildlife, recreation in and on the water, and agricultural, industrial, and other purposes, including navigation. Changing designated uses for a specific waterbody requires a change in the water quality standards. Like all new and revised State and Tribal water quality standards, these changes are subject to U.S. EPA review and approval (see 40 CFR 131.21).

Where appropriate, a State may subcategorize or refine the aquatic life use designations for the receiving water. States may adopt subcategories of a use and set the appropriate criteria to reflect varying needs of such subcategories of uses, for instance, to differentiate between coldwater and warmwater fisheries (see 40 CFR 131.10(c)). States may also adopt seasonal uses (40 CFR 131.10(f)). If seasonal uses are adopted, water quality criteria should reflect the seasonal uses; however, such criteria shall not preclude the attainment and maintenance of a more protective use in another season.

Water quality *criteria* are elements of State WQS expressed as constituent concentrations, levels, or narrative statements representing a quality of water that supports a particular use. When criteria are met, water quality will generally protect the designated use (40 CFR 131.3). While some States have adopted a variety of criteria expressed as constituent concentration levels (or *numeric* criteria) for various pollutants for the protection of aquatic life, all States have adopted criteria expressed as narrative statements (or *narrative* criteria). Once adopted into standards, criteria can serve as the basis for 1) regulatory controls on point sources, 2) measuring attainment of standards and the effectiveness or programs, and 3) watershed planning.

Section 304(a) criteria are developed by the U.S. EPA under authority of section 304(a) of the CWA based on the latest scientific information on the relationship that a constituent concentration, level, or measure has on a particular aquatic species and/or human health. This information is issued periodically to the States as guidance for use in developing criteria. In adopting criteria to protect their designated uses, States may establish criteria based on 1) section 304(a) guidance, 2) section 304(a) guidance modified to reflect site-specific conditions, or 3) other scientifically defensible methods.

1.3 The role of designated aquatic life uses in Water Quality Standards

It is in designating uses that States and Tribes establish the environmental goals for their water resources and then measure attainment of these goals. In designating uses, a State or Tribe weighs the environmental, social, and economic consequences of its decisions. The regulation allows the State or Tribe, with public participation, some flexibility in weighing these considerations and adjusting these goals over time. However, reaching a conclusion on the uses that appropriately reflect the current and potential future uses for a waterbody, determining the attainability of those goals, and appropriately evaluating the consequences of a designation can be a difficult and controversial task.

A principal function of designated uses in water quality standards is to communicate the desired state of surface waters to water quality managers, the regulated community, and the interested public. An effective designated use system is one that translates readily into indicators (e.g., numeric water quality criteria, biological indexes) that respond in predictable ways to stress and can be evaluated using data collected from the waterbody. Experience with implementation of various State designated use systems suggests that, regardless of the system selected, States that use biological data as part of their assessment program apply some type of refined, or tiered, aquatic life use approach to guide interpretation of their biological data. States have either made this *explicit* by adopting the tiers directly into their water quality standards as designated uses or *implicit* by using tiers in their monitoring and assessment protocols.

Although the benefits of more specificity may apply to any of the designated uses described in CWA section 303, it may be most relevant for aquatic life uses. Aquatic communities can vary significantly from waterbody to waterbody. One major challenge in assigning designated uses for aquatic life to surface waters is separating the natural variability that is a function of stream type (e.g., naturally coldwater vs. warmwater stream) and location (ecoregion) from the variability that results from exposure to stressors. By accounting for natural variability in aquatic systems, biologically-based tiered aquatic life uses eliminate a major source of uncertainty and error in water quality management efforts.

1.4 State and Tribal experiences with tiered aquatic life uses

Over the years, States and Tribes have created many different use classification systems ranging from a straightforward replication of the uses specifically listed in section 303 of the CWA, to more complex systems that express designated uses in very specific terms or that establish subclassifications identifying different levels of protection. Some States designate general "aquatic life" uses while others list a variety of subcategories based on a range of aquatic community types, including descriptions of core aquatic species representative of each subcategory (e.g., coldwater and warmwater fisheries). Many States also have narrative biological criteria, which is often a general statement such as "aquatic life communities shall be maintained similar to aquatic life as naturally occurs." Single thresholds for attainment of these general uses and narrative biological criteria are established with numeric biological criteria. For example, many State water quality agencies interpret narrative general use statements using an index (e.g., Index of Biotic Integrity (IBI)) (Karr et al. 1987, Karr 1990, Gibson et al. 1996, U.S. EPA 2002a). The index is standardized to regional reference conditions, and the biological criteria threshold is often established as a percentile of the distribution of reference site scores. The index is the basis for numeric biological criteria in many States and Tribes (U.S. EPA 2002a).

The alternative to a single broad use is to divide the continuum of biological condition (the BCG) into several tiers for more precise management. As mentioned earlier, tiered aquatic life uses couple narrative descriptions of the use with criteria for measuring attainment of the use. Ideally, the narrative descriptions should incorporate biologically meaningful differences among tiers. The BCG provides an interpretative framework for defining reference conditions and articulating the biological condition that is being protected or restored in the water of interest.

Several States and Tribes have adopted tiered aquatic life use statements in their water quality standards and some are developing the technical program and further tightening the linkage between their narrative use statements and numeric biological criteria (U.S. EPA 2002a). For example, Texas has had tiered aquatic life uses identified in their water quality standards for surface waters since 1984 (Table 1-1). Texas' current WQS identify numeric dissolved oxygen criteria and include narrative aquatic life attributes. Numeric biological criteria have been developed for assessing both fish and benthic macroinvertebrate communities in wadeable streams. If site-specific conditions do not meet criteria for "High" use category as determined by receiving water assessment, a use attainability analysis will be conducted. Texas continues to evaluate the application of biological criteria for other aquatic systems, but at this point does not have a specific action plan to adopt numeric biological criteria for those systems. Other States cited elsewhere in this document, e.g., Maine, New Jersey, Ohio, and Vermont, have either developed or are considering developing tiered aquatic life uses. Though these approaches for tiering aquatic life uses may differ in detail and assessment methods, their uses share the same core elements:

- Biological information is the basis for the use designations.
- Numeric biological indicators or biocriteria are developed for each use.
- Development of tiers based on data from comprehensive, robust monitoring program.

TABLE 1-1. Aquatic Life Subcategories in Texas WQS (Figure: 30 TAC §307.7(b)(3)(A)(i)).

Aquatic Life Use Subcategory	Dissolved Oxygen Criteria, mg/L			Aquatic Life Attributes					
	Freshwater mean/ minimum	Freshwater in Spring mean/ minimum	Saltwater mean/ minimum	Habitat Character-istics	Species Assemblage	Sensitive species	Diversity	Species Richness	Trophic Structure
Exceptional	6.0/4.0	6.0/5.0	5.0/4.0	Outstanding natural variability	Exceptional or unusual	Abundant	Exceptionally high	Exceptionally high	Balanced
High	5.0/3.0	5.5/4.5	4.0/3.0	Highly diverse	Usual asso-ciation of regionally expected species	Present	High	High	Balanced to slightly imbalanced
Intermediate	4.0/3.0	5.0/4.0	3.0/2.0	Moderately diverse	Some expected species	Very low in abundance	Moderate	Moderate	Moderately imbalanced
Limited	3.0/2.0	4.0/3.0		Uniform	Most regionally expected species absent	Absent	Low	Low	Severely imbalanced

- Dissolved oxygen means are applied as a minimum average over a 24-hour period.
- Daily minima are not to extend beyond 8 hours per 24-hour day. Lower dissolved oxygen minima may apply on a site-specific basis, when natural daily fluctuations below the mean are greater than the difference between the mean and minima of the appropriate criteria.
- Spring criteria to protect fish spawning periods are applied during that portion of the first half of the year when water temperatures are 63.0°F to 73.0°F.
- Quantitative criteria to support aquatic life attributes are described in the standards implementation procedures.
- Dissolved oxygen analyses and computer models to establish effluent limits for permitted discharges will normally be applied to mean criteria at steady-state, critical conditions.
- Determination of standards attainment for dissolved oxygen criteria is specified in §307.9(d)(6) (relating to Determination of Standards Attainment).

The insights and experiences from States and Tribes that have adopted tiered aquatic life uses and numeric biocriteria in their water quality standards, as well as from those currently developing biological assessment and criteria programs, reveal the values of tiered aquatic life uses implemented in State and Tribal WQS (Table 1-2).

TABLE 1-2. The benefits and WQS regulation context for TALUs.

Value-added	Explanation	Supporting WQS Regulation
Set more appropriate designated ALUs	Define ALUs in a more precise way that is neither under-protective of existing high-quality resources nor overprotective for waters that have been extensively and irretrievably altered	40CFR131.10 40CFR131.12 (Protect High Quality Waters) 40CFR130.23 (Support attainment decisions and diagnose causes)
Strengthen the linkage between designated ALUs and how attainment is assessed	TALUs help to clarify and refine water quality goal statements so numeric biological, chemical and physical criteria can be adopted to protect the use	40CFR131.10(c) 40CFR131.12 (Protect High Quality Waters) 40CFR130.23 (Support attainment decisions and diagnose causes)
Enhance public understanding and participation in setting water quality goals	TALUs provide a common frame of reference or generic yardstick to more clearly recognize common ground and differences in desired environmental goals of various stakeholders as designated uses are adopted	40 CFR131.20 (a)(b)

Building on these "lessons learned," the U.S. EPA convened a technical workgroup in 2000 to identify existing scientifically sound and practical approaches that would help States and Tribes provide more precision, or specificity, in their designated aquatic life uses. The workgroup included biologists and aquatic ecologists from States, Tribes, U.S. EPA, USGS, the academic research community, and the private sector. The workgroup was asked to address the following questions:

- What are effective technical approaches using biological information to provide more specificity in their designated aquatic life uses?

- What are the "lessons learned" that can be capitalized on and shared with other States and Tribes?

The workgroup was charged with developing a scientific framework using biological information to better define designated aquatic life uses, enabling more precise use descriptions. Their product is a narrative model describing graduated tiers of biological response to increasing levels of stressors, the *Biological Condition Gradient (BCG).* The model is founded on peer-reviewed work in the field of bioassessments over the past thirty years (Fausch et al. 1984, Karr et al. 1986, Cairns and Pratt 1993, Barbour et al. 1999) and on the experiences and empirical observations of States and Tribes that have developed tiered aquatic life uses and biological criteria for use in their water programs (Courtemanch et al. 1989, Courtemanch 1995, Yoder 1995, Yoder and Rankin 1995b).

1.5 The Biological Condition Gradient: A tool for better defining and developing more precise aquatic life uses

The Biological Condition Gradient (BCG) is a scientific model for interpreting biological response to increasing effects of stressors on aquatic ecosystems (Figure 1-1). The model describes how ten attributes of aquatic ecosystems change in response to the increasing levels of stressors. The attributes include several aspects of community structure, organism condition, ecosystem function, and spatial and temporal attributes of stream size and connectivity. The gradient can be considered analogous to a field-based dose-response curve where dose (x-axis) = increasing levels of stressors and response (y-axis) = biological condition (see figure below). The BCG differs from the standard dose-response curve, in that the BCG does not represent the laboratory response of a single species to a specified dose of a known chemical, but rather the *in-situ* response of the biota to the sum of stresses it is exposed to. The BCG is divided into six tiers of biological condition along the stressor-response curve, ranging from observable biological conditions found at no or low levels (Tier 1) to those found at high levels of stressors (Tier 6). The BCG model was developed to provide a common framework for interpreting biological information regardless of methodology and geography. When calibrated to a regional or state scale, States and Tribes can use the model to more precisely evaluate the current and potential biological condition of their waters and use that information to better define their aquatic life uses. Additionally, States and Tribes can use this interpretative model to more clearly communicate the condition of their aquatic resources to the public.

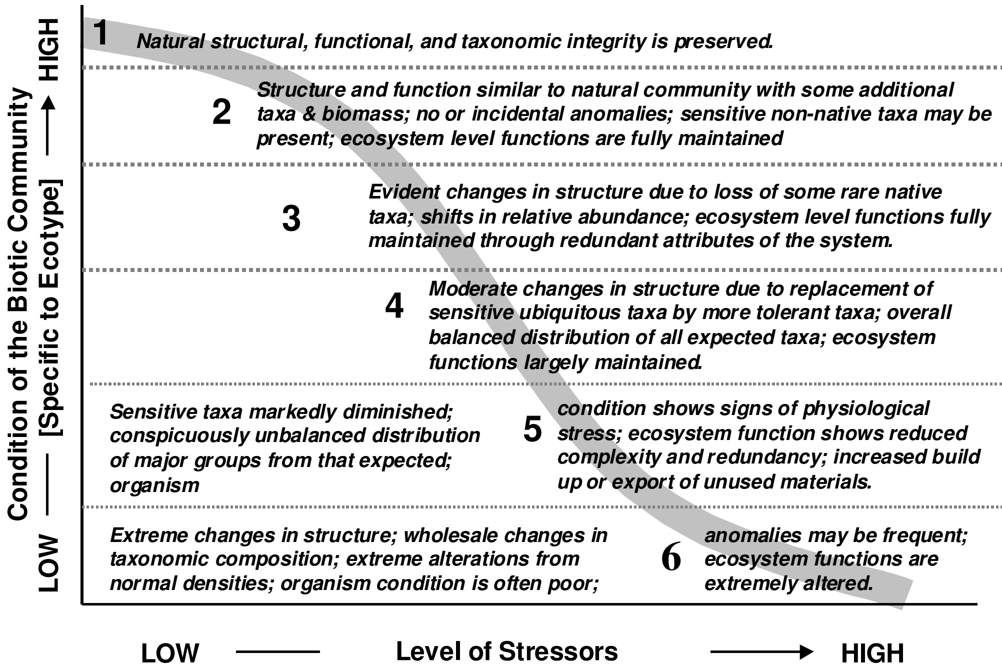


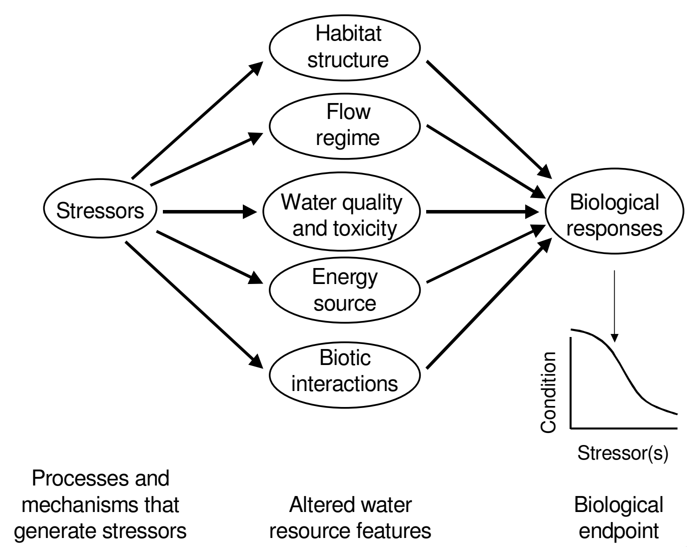
FIGURE 1-1. Conceptual model of the Biological Condition Gradient.

The BCG model was developed based on common patterns of biological response to stressors observed empirically by aquatic biologists and ecologists from different geographic areas of the U.S. Once a draft model was constructed, it was tested at a workgroup meeting and then at two regional workshops. The model was tested by determining how consistently the scientists assigned samples of macroinvertebrates or fish to the different tiers of biological condition. Workgroup members identified similar sequences of biological response to increasing levels of stressors regardless of geographic area. These results support the use of the BCG as a nationally applicable model for interpreting the biological condition of aquatic systems. Chapter 2 discusses the development and makeup of the conceptual BCG and Chapter 3 explores strategies for regionally modifying, or calibrating, the conceptual model. Chapter 4 describes how the x-axis of the BCG model, the stressor gradient, can be characterized and explains how the effects of stressors on biological condition play a role in constructing and using a BCG. Chapter 5 discusses the underlying principles and processes States have learned in using biological information to develop tiered aquatic life uses, and examples of how States have applied tiered uses in water quality management are presented in Chapter 6.

Integral to the development of the BCG is characterizing the model's x-axis, the stressor gradient (Figure 1-1). **Stressors** are physical, chemical or biological factors that induce an adverse response from aquatic biota (U.S. EPA 2000b; EPA/822/B-00/025). For example, high concentrations of certain metals, nutrients, or sediment can adversely impact aquatic biota. Loss of aquatic habitat or presence of aquatic invasive species can also adversely impact, or stress, the aquatic biota expected for a specific waterbody. These stressors can cause aquatic ecosystems to change from natural conditions, exhibiting altered compositional, structural, and functional characteristics. The degree to which stressors affect the biota depends on the magnitude, frequency, and duration of the exposure of the biota to the stressors. Developing a BCG for a given system characterizes the general relationship between its stressors in total (the model's x-axis) and its overall biological condition (the y-axis). Multiple stressors are usually present, and thus the stressor x-axis of the BCG seeks to represent their cumulative influence as a **Generalized Stressor Gradient** (GSG), much as the y-axis generalizes biological condition.

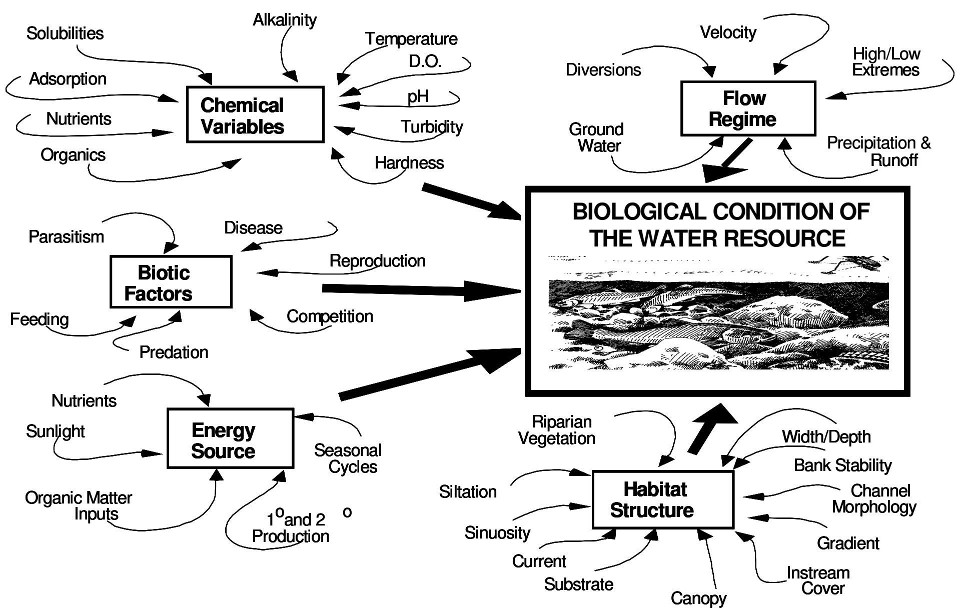
Understanding the links between stressors and their sources and the response of the aquatic biota will help to more accurately determine the existing and potential condition of the aquatic biota (Figure 1-2). There are different approaches and emerging science to define and quantify the causal sequence between stressors and their sources and biological responses. Building on current State and Tribal approaches, a framework for characterizing stressors, the processes and mechanisms that generate them, and the resulting biological response is presented. This framework may not only help State and Tribal managers more precisely define designated uses, including potential future uses, but may support diagnosis of use impairment and help prioritize management decision making.

FIGURE 1-2. The causal sequence from stressors and their sources through the five major water resource features to the biological responses, i.e., the biological endpoints. This model illustrates the multiple pathways that stressors and their sources can affect aquatic biota. Insert illustrates the relationship between stressor dose and the gradient of biological responses (after Karr and Yoder 2004; used by permission of J.D. Allan, originally presented at the 2002 TALU Workgroup Meeting).



1.6 Conceptual basis for the Biological Condition Gradient

The five factors that determine the integrity of a water resource, which were originally described by Karr and Dudley (1981; Figure 1-3), have been consistently used as the conceptual basis for biological assessment and tiered aquatic life uses. In the context of the TALU approach, consideration of the five factors in Figure 1-3 are components of the stressor axis of the BCG model, while the condition of the water resource is accounted for by the response of the biological community to the stressors, the Biological Condition Gradient (BCG). The health and well-being of the aquatic biota is an important barometer to measure progress towards achieving Clean Water Act goals. Biological integrity has been defined as the combined result of chemical, physical, and biological processes in the aquatic environment (Karr and Dudley 1981, Karr et al. 1986). Biological criteria help reconcile the mosaic of factors and interactions that exist, parts of which may be characterized and measured using chemical and physical indicators.



(Modified from Karr et al. 1986).

FIGURE 1-3. The five major factors that determine the biological condition of aquatic resources (modified from Karr et al. 1986).

An important conceptual foundation of tiered aquatic life uses is the "position of the standard" that was described by the National Research Council Committee on Science in TMDLs (NRC 2001; Figure F-1). This concept describes the "position" of different types of criteria with respect to their position along a causal chain of indicators beginning with sources (stressor indicators), to changes in pollutant contributions or attributes of landscape and/or hydrology that emanate from those sources (exposure indicators), to instream exposures (pollutants, attributes of habitat), to indicators of biological condition (response indicators) that directly assess the designated use. Because designated uses are written in qualitative, narrative terminology, the challenge is to relate a criterion to the designated use. In general, establishing this relationship becomes easier as the criterion is positioned closer to the designated use, hence the NRC recommendation on the use of biological information to help determine more appropriate aquatic life uses and to couple the narrative use statements with quantitative methods. Thus biological criteria can fill a gap along this position spectrum and serve a useful role in the expression and implementation of water quality standards.

Karr and Yoder (2004) further elaborated upon this concept by adding the interactive relationships between pollution and pollutants from both point and nonpoint sources (Figure 1-4). It also relates different types of indicators in the causal sequence of events and exemplifies the appropriate roles of chemical, physical, and biological parameters as stressor, exposure, and response indicators (Yoder and Rankin 1998). In this scheme, attainment of a designated use is the desired result of the management of stressors (chemical, biological, physical) and is explained by how stressors influence and change the five factors that determine the integrity of an aquatic resource (Karr and Yoder 2004). In each of these process descriptions, the end outcome of water quality management is reflected in the status of a designated use. Attainment of the designated use confirms the effectiveness of the sequence of management actions; non-attainment is evidence of an incomplete process and a prompt to re-examine the management strategy. Each provides important feedback about the effectiveness of management strategies. Therefore, how designated uses are developed, assigned, and measured is key to the outcomes derived from water quality management.

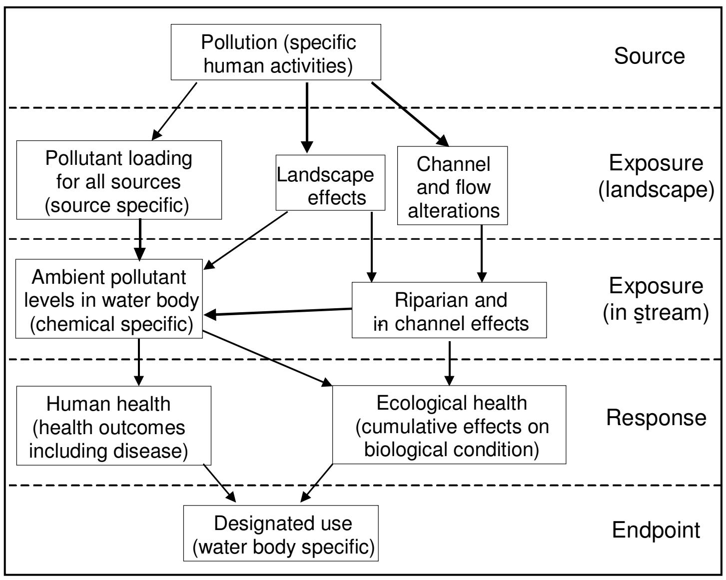


FIGURE 1-4. Modification of the NRC "position of the criterion" concept (Figure F-1) showing the causal sequence from *indicators* of stress, exposure, and response in relation to point and nonpoint source impacts, specific types of criteria, and designated uses that define the endpoints of interest to society (after Karr and Yoder 2004).

1.7 Key points from Chapter 1

1. Section 101(a) of the CWA establishes broad national goals and objectives such as the chemical, physical, and biological integrity objective. To help achieve the integrity objective, the CWA also established, among other things, an interim goal for the protection and propagation of fish, shellfish, and wildlife. The protection and propagation interim goal has been interpreted by U.S. EPA to include the protection of the full complement of aquatic organisms residing or migrating through a waterbody. The health and well-being, or condition, of the aquatic biota is an important barometer to measure progress towards achieving Clean Water Act goals and objectives.

2. State water quality standards provide the foundation for water quality-based pollution control programs. With the public participating in their adoption (see 40 CFR 131.20), such standards serve the dual purposes of establishing the water quality goals for a specific waterbody *(designated uses)* and serve as the regulatory basis for the establishment of water quality-based

treatment controls and strategies beyond the technology-based levels of treatment required by Sections 301(b) and 306 of the CWA.

3. A waterbody's *designated use(s)* are those uses specified in water quality standards, whether or not they are being attained (40 CFR 131.3(f)). The "use" of a waterbody is the most fundamental articulation of its role in the aquatic and human environments. All of the water quality protections established by the CWA follow from the waterbody's designated use.

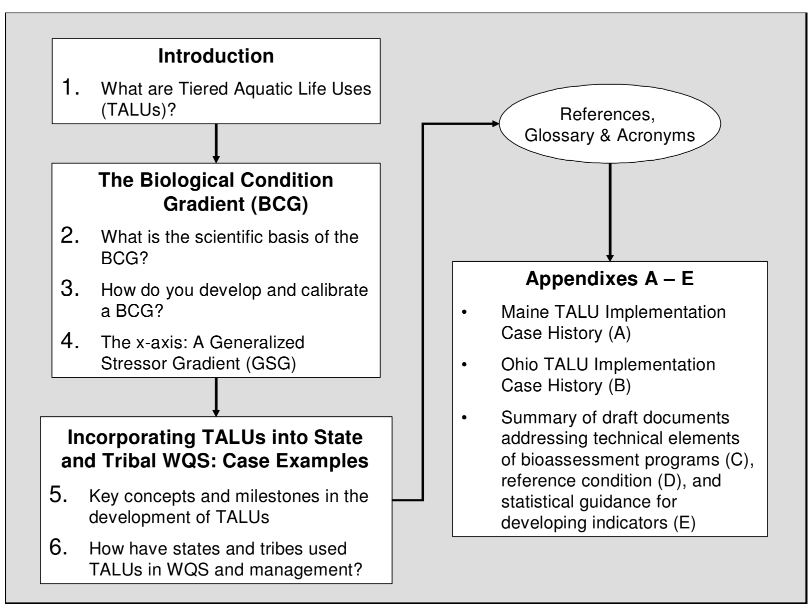
4. *Tiered aquatic life uses* are bioassessment-based statements of expected biological condition in specific waterbodies. Tiered uses allow more precise and measurable definitions of *designated aquatic life uses*.

5. Several States and Tribes have adopted tiered aquatic life uses in their water quality standards. This document is based on the "lessons learned" from their experiences and the recommendations from a technical workgroup charged with integrating existing scientifically sound and practical approaches to 1) tier designated aquatic life uses using biological information, and 2) incorporate information on sources of stress as drivers of biological condition.

1.8 Organization of the document

This chapter provided the background and rationale for using biological information to designate aquatic life uses in tiers that more specifically differentiate the characteristics of the biological community currently present or desired in a waterbody. The following chapters are based on the recommendations of the TALU technical workgroup tasked with identifying existing scientifically sound and practical approaches that would help States and Tribes provide more precision, or specificity, in their designated aquatic life uses (Figure 1-5). Chapters 2 and 3 discuss the Biological Condition Gradient (BCG) – what it is, how the national conceptual model was developed and tested, and how to calibrate the conceptual model to a region. Chapter 4 describes how the x-axis of the BCG model, the stressor gradient, can be characterized and explains how the effects of stressors on biological condition play a role in constructing and using a BCG. Chapter 5 provides examples on how States have developed tiered aquatic life uses. The experiences of Maine and Ohio, two States that have completed this process, serve as comprehensive case histories that are found in Appendixes A and B. Chapter 6 details how Maine and Ohio have used tiered aquatic life uses in assessment and management as examples that might guide future implementation guidance.

FIGURE 1-5. Roadmap to the document.



Related Technical Support Documents:

Appendixes C, D & E contain summaries of three "companion" documents that are under development. Each contains detailed information relevant to developing tiered aquatic life uses, including components of State and Tribal bioassessment programs, statistical methods that use biological data, and best practices for developing reference conditions. Following is a brief description of each document.

Technical Guidelines: Technical Elements of a Bioassessment Program – DRAFT

This document is intended primarily for use by State and Tribal program managers and staff who are responsible for monitoring and assessment and water quality standards programs. The document describes the technical attributes of biological assessment programs, and can thus be used by States and Tribes to 1) determine where they are in the biological assessment and criteria development processes, and 2) develop, structure, and, if necessary, modify their programs and refine designated aquatic life uses.

U.S. EPA project leads: Susan Jackson, Office of Water; Ed Hammer, Region 5; Tina Laidlaw, Region 8; and Gretchen Hayslip, Region 10

The Role of Reference Condition in Biological Assessment and Criteria – DRAFT DOCUMENT ON DEVELOPMENT AND APPLICATION OF THE REFERENCE CONDITION CONCEPT

This document will provide States, Tribes, and other practitioners with guidelines on using reference conditions in their water management programs, particularly for ecological assessments. The guidelines described are intended to facilitate greater implementation of best practices for reference condition, thereby improving the success of individual programs and leading to greater consistency among States and Tribes.

U.S. EPA project leads: Evan Hornig, Office of Water; Phil Larsen, Office of Research and Development; and Wayne Davis, Office of Environmental Information

Statistical Guidance for Developing Indicators for Rivers and Streams: A Guide for Constructing Multimetric and Multivariate Predictive Bioassessment Models – DRAFT

This document will provide methods and outlines the steps required to complete multimetric and multivariate predictive assessment models, two methods for analyzing and assessing waterbody condition from assemblage and community-level biological information.

U.S. EPA project lead: Florence Fulk, Office of Research and Development

The Biological Condition Gradient

The Biological Condition Gradient (BCG) is a scientific model that allows consistent interpretation of biological condition although assessment approaches may differ. The BCG combines scientific knowledge with the practical experience and needs of resource managers and can assist environmental practitioners in the U.S. to better:

- define aquatic resources
- establish direct relationships between biological condition and stressors
- communicate clearly to the public both the existing and potential uses of a waterbody

> *The BCG is consistent with ecological theory and is a means for standardizing interpretations of the response of aquatic biota to stressors. The model should facilitate communication among scientists, managers, and the public on the current conditions and ecological potential for specific waterbodies.*

Chapter 2 outlines the development and makeup of the BCG model. The BCG describes changes in ten ecological attributes across a gradient of biological condition caused by increasing stressors (Table 2-1). It is divided into six condition tiers, Tier 1 representing natural, or undisturbed, conditions through Tier 6 representing severely altered conditions.

TALU Workgroup biologists from across the U.S. agreed that a similar sequence of biological alterations occur in streams in response to stressors, strengthening the feasibility of using the BCG as a common framework to guide management decisions that protect and restore aquatic systems in the U.S. (Davies and Jackson in press). The model is consistent with ecological theory and can be adapted or calibrated to reflect specific geographic regions. Scientific knowledge can be reviewed and consolidated and research needs can be expressed in a context relevant to management. Thus, the model also serves as a framework that 1) synthesizes what has been observed into testable hypotheses, and 2) identifies knowledge gaps in need of further research.

Chapter 3 explores strategies for regionally modifying, or calibrating, the BCG including approaches for recalibrating existing indexes. Three States (Maine, Ohio, and Vermont) have incorporated a BCG into their water quality standards as well as numeric criteria. Several other States (e.g., New Jersey, Texas, and a consortium of New England states) have begun the process of evaluating the potential use of a BCG. Each of these States is following basically the same approach used by the national TALU Workgroup to develop the BCG model, reaching consensus among regional biological experts familiar with natural aquatic communities and their responses to stress.

Chapter 4 describes the model's x-axis, the stressor gradient that illustrates alteration in biological condition. The degree to which stressors affect the biota depends on the magnitude, frequency, and duration of the exposure of the biota to the stressors. Developing a BCG for a given system characterizes the general relationship between its stressors in total (the model's x-axis) and its overall biological condition (the y-axis). Multiple stressors are usually present, and thus the stressor x-axis of the BCG seeks to represent their cumulative influence as a **Generalized Stressor Gradient** (GSG), much as the y-axis generalizes biological condition. Chapter 4 explains how stressors can be characterized and describes how the influence of stressors on biological condition plays a role in constructing and using a BCG.

CHAPTER 2. WHAT IS THE SCIENTIFIC BASIS OF THE BIOLOGICAL CONDITION GRADIENT?

The Biological Condition Gradient (BCG) extends the empirical work of earlier researchers and practitioners to create a nationally consistent model that links management goals for resource condition with the quantitative measures used in biological assessments. The BCG was designed to describe ecological response to stressors in sufficient detail so that a site can be placed into a tier along the BCG continuum through use of the core data elements collected by most State or Tribal monitoring programs.

The practice of using biological indicators to assess water quality is over a century old. The Saprobien System, a concept proposed by Lauterborn in 1901and further developed the following year by Kolkwitz and Marsson (Davis 1995), uses benthic macroinvertebrates and planktonic plants and animals as indicators of organic loading and low dissolved oxygen, and has been updated and is currently used in several European countries. Concurrently, the limnologists Thienemann and Naumann developed the concept of trophic state classification for lakes in the 1920s (Carlson 1992, Cairns and Pratt 1993). These early indexes described a response gradient (or response classes for lakes) to enrichment. The Saprobien System was explicitly developed to assess human pollution in rivers, but the trophic state concept was originally developed to describe natural conditions in lakes and only later became a concept to describe pollution-caused eutrophication (e.g., Vollenweider 1968). The 1950s marked the development of Beck's biotic index in the U.S. and Pantle and Buck's Saprobic Index in Europe, which were directly based on the Saprobien System (cited in Davis 1995). The Saprobic Index, which led to the development of the widely used Hilsenhoff Index (e.g., Hilsenhoff 1987) in the U.S., could be considered the predecessor of today's biotic indexes (Davis 1995).

The conceptual foundation of the BCG is based on many decades of biologists' accumulated experience with biological assessment and monitoring. Biological information from monitoring programs has been frequently synthesized by constructing biotic indexes, such as the Index of Biotic Integrity (IBI) (Karr 1981, Karr et al. 1986). The IBI integrated the concept of anchoring the measurement system in undisturbed reference conditions with the measurement of several indicators intended to reflect ecological components of composition, diversity, and ecosystem processes. It thus combined a conceptual model of ecosystem change in response to increasing levels of stressors with a practical measurement system for fish. The BCG is also grounded in the concepts in Cairns et al. (1993) describing "natural" conditions and the change in biological condition caused by stressors. To achieve maximum potential application nationwide, the BCG tiers were developed based on States' various experiences designing and implementing tiered aquatic life use and management goals as well as the practical experience of aquatic scientists from different bio-geographic areas, each of whom had fifteen to thirty years of experience in the field. The BCG:

1. Describes a complete scale of condition from natural (Tier 1) to severely altered (Tier 6);
2. Synthesizes existing field observations and generally accepted interpretations of patterns of biological change within a common framework; and
3. Helps determine the degree to which a system may have departed from natural condition, based on measurable, ecologically important attributes.

At present, the description of biological attributes that make up the model applies best to permanent, hard-bottom streams that are exposed to increases in temperature, nutrients, and fine sediments because this is the stream-type and stressor regime originally described by the model. The model has been further tested with States and Tribes in different parts of the country (e.g., arid west and great plains) to evaluate the national applicability of the model. Results have been successful with some necessary refinement of the model attributes to accommodate regional differences. For example, during a workshop in Texas where the BCG was being evaluated using Texas data, Attribute II (sensitive-rare taxa) was redefined as *highly sensitive taxa* because rarity of a taxon in the region was not deemed to be associated with sensitivity to stress. In arid streams, many rare, native taxa are highly tolerant to stressors such as low dissolved oxygen and high temperature. Thus, the BCG can be applicable to other aquatic ecosystems and stressors with appropriate modifications. The BCG should be viewed as an evolving model that must be responsive to changes in scientific understanding resulting from the analysis of empirical data.

The value of a heuristic model such as the BCG is not only that it documents experimentally established knowledge, but also that it promotes a more rigorous testing of empirical observations by clearly stating them in a provisional model. Conceptual models formalize the state of knowledge and guide research. Empirically based generalizations have led to conceptual models that describe the behavior of biological systems under stress (Brinkhurst 1993; Margalef 1963, 1981; Odum, et al. 1979; Rapport et al. 1985; Schindler 1987; Fausch et al. 1990; Karr and Dudley 1981). For example, Brinkhurst observed that "Everyone knew [in 1929] that increases in numbers and species could be related to mild pollution, that moderate pollution could produce changes in taxa so that diversity remained similar but species composition shifted, and that eventually species richness declined abruptly and numbers of some tolerant forms increased dramatically." Such ecosystem responses to stressor gradients have been portrayed as a progression of stages that occur in a generally consistent pattern (Odum et al. 1979, Odum 1985, Rapport et al. 1985, Cairns and Pratt 1993). Establishing and validating quantifiable thresholds along that progression with empirical data is a priority need for resource managers (Cairns l981).

2.1 What the BCG model looks like

The BCG model depicts ecological condition in terms of ten system attributes expressed at different spatial scales (Table 2-1). In biological assessments, most information is collected at the spatial scale of a site or reach and the temporal scale of a single sampling event. Many of the attributes that make up the BCG are based on these scales. Site scale attributes include aspects of taxonomic composition and community structure (Attributes I-VI) and organism and system performance (Attributes VII and VIII). At larger temporal and spatial scales, physical-biotic interactions (Attributes IX and X) were also included because of their importance in evaluating the longer term impacts, restoration potential and recoveries.

TABLE 2-1. Biological Condition Gradient matrix.

Ecological Attributes	Biological Condition Gradient Tiers					
	1 Natural or native condition	2 Minimal changes in the structure of the biotic community and minimal changes in ecosystem function	3 Evident changes in structure of the biotic community and minimal changes in ecosystem function	4 Moderate changes in structure of the biotic community and minimal changes in ecosystem function	5 Major changes in structure of the biotic community and moderate changes in ecosystem function	6 Severe changes in structure of the biotic community and major loss of ecosystem function
	Native structural, functional and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability	Virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability	Some changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa but Sensitive-ubiquitous taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system	Moderate changes in structure due to replacement of some Sensitive-ubiquitous taxa by more tolerant taxa, but reproducing populations of some Sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes	Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from that expected; organism condition shows signs of physiological stress; system function shows reduced complexity and redundancy; increased build-up or export of unused materials	Extreme changes in structure; wholesale changes in taxonomic composition; extreme alterations from normal densities and distributions; organism condition is often poor; ecosystem functions are severely altered
I Historically documented, sensitive, long-lived or regionally endemic taxa	As predicted for natural occurrence except for global extinctions	As predicted for natural occurrence except for global extinctions	Some may be absent due to global extinction or local extirpation	Some may be absent due to global, regional or local extirpation	Usually absent	Absent
II Sensitive-rare taxa	As predicted for natural occurrence, with at most minor changes from natural densities	Virtually all are maintained with some changes in densities	Some loss, with replacement by functionally equivalent Sensitive-ubiquitous taxa	May be markedly diminished	Absent	Absent
III Sensitive-ubiquitous taxa	As predicted for natural occurrence, with at most minor changes from natural densities	Present and may be increasingly abundant	Common and abundant; relative abundance greater than Sensitive-rare, taxa	Present with reproducing populations maintained; some replacement by functionally equivalent taxa of intermediate tolerance.	Frequently absent or markedly diminished	Absent
IV Taxa of intermediate tolerance	As predicted for natural occurrence, with at most minor changes from natural densities	As naturally present with slight increases in abundance	Often evident increases in abundance	Common and often abundant; relative abundance may be greater than Sensitive-ubiquitous taxa	Often exhibit excessive dominance	May occur in extremely high OR extremely low densities; richness of all taxa is low
V Tolerant taxa	As naturally occur, with at most minor changes from natural densities	As naturally present with slight increases in abundance	May be increases in abundance of functionally diverse tolerant taxa	May be common but do not exhibit significant dominance	Often occur in high densities and may be dominant	Usually comprise the majority of the assemblage; often extreme departures from normal densities (high or low)

TABLE 2-1. Biological Condition Gradient matrix.

	Biological Condition Gradient Tiers					
	1 Natural or native condition	2 Minimal changes in the structure of the biotic community and minimal changes in ecosystem function	3 Evident changes in structure of the biotic community and minimal changes in ecosystem function	4 Moderate changes in structure of the biotic community and minimal changes in ecosystem function	5 Major changes in structure of the biotic community and moderate changes in ecosystem function	6 Severe changes in structure of the biotic community and major loss of ecosystem function
VI Non-native or intentionally introduced taxa	Non-native taxa, if present, do not displace native taxa or alter native structural or functional integrity	Non-native taxa may be present, but occurrence has a non-detrimental effect on native taxa	Sensitive or intentionally introduced non-native taxa may dominate some assemblages (e.g. fish or macrophytes)	Some replacement of sensitive non-native taxa with functionally diverse assemblage of non-native taxa of intermediate tolerance	Some assemblages (e.g., fish or macrophytes) are dominated by tolerant non-native taxa	Often dominant; may be the only representative of some assemblages (e.g., plants, fish, bivalves)
VII Organism Condition (especially of long-lived organisms)	Any anomalies are consistent with naturally occurring incidence and characteristics	Any anomalies are consistent with naturally occurring incidence and characteristics	Anomalies are infrequent	Incidence of anomalies may be slightly higher than expected	Biomass may be reduced; anomalies increasingly common	Long-lived taxa may be absent; Biomass reduced; anomalies common and serious; minimal reproduction except for extremely tolerant groups
VIII Ecosystem Functions	All are maintained within the natural range of variability	All are maintained within the natural range of variability	Virtually all are maintained through functionally redundant system attributes; minimal increase in export except at high storm flows	Virtually all are maintained through functionally redundant system attributes though there is evidence of loss of efficiency (e.g., increased export or decreased import)	There is apparent loss of some ecosystem functions manifested as increased export or decreased import of some resources, and changes in energy exchange rates (e.g., P/R; decomposition)	Most functions show extensive and persistent disruption
IX Spatial and temporal extent of detrimental effects	N/A A natural disturbance regime is maintained	Limited to small pockets and short duration	Limited to the reach scale and/or limited to within a season	Mild detrimental effects may be detectable beyond the reach scale and may include more than one season	Detrimental effects extend far beyond the reach scale leaving only a few islands of adequate conditions; effect extends across multiple seasons	Detrimental effects may eliminate all refugia and colonization sources within the catchment and affect multiple seasons
X Ecosystem connectance	System is highly connected in space and time, at least annually	Ecosystem connectance is not impacted	Slight loss of connectance but there are adequate local recolonization sources	Some loss of connectance but colonization sources and refugia exist within the catchment	Significant loss of ecosystem connectance is evident; recolonization sources do not exist for some taxa	Complete loss of ecosystem connectance in at least one dimension (i.e., longitudinal, lateral, vertical, or temporal) lowers reproductive success of most groups; frequent failures in reproduction & recruitment

2.1.1 The BCG Attributes

Taxonomic Composition and Structure: Attributes I – VI

Attribute I: Historically documented, sensitive, long-lived or regionally endemic taxa.
"Historically documented" refers to taxa known to have been supported in a waterbody or region prior to enactment of the 1972 Clean Water Act, according to historical records compiled by State or federal agencies or published scientific literature.

"Sensitive or regionally endemic taxa" have restricted, geographically isolated distribution patterns (occurring only in a locale as opposed to a region), often due to unique life history requirements. They may be long-lived, late maturing, low fecundity, limited mobility, or require a mutualist relation with other species. They may be among listed Endangered or Threatened (E/T) or special concern species. Predictability of occurrence is often low, and therefore requires documented observation. Recorded occurrence may be highly dependent on sample methods, site selection, and level of effort.

Attribute II: Sensitive-rare taxa.
These are taxa that naturally occur in low numbers relative to total population density but may make up a large relative proportion of richness. They may be ubiquitous in occurrence or restricted to certain micro-habitats, but because of low density, recorded occurrence is dependent on sample effort. Often stenothermic (having a narrow range of thermal tolerance) or cold-water obligates; commonly k-strategists (populations maintained at a fairly constant level; slower development; longer life-span). May have specialized food resource needs or feeding strategies. Generally intolerant to significant alteration of the physical or chemical environment; are often the first taxa observed to be lost from a community.

Attribute III: Sensitive ubiquitous taxa.
"Sensitive" taxa from Attributes II and III are taxa that are intolerant to a given stress; they are the first species affected by the specific stressor to which they are "sensitive" and the last to recover following restoration. Sensitive ubiquitous taxa are ordinarily common and abundant in natural communities when conventional sampling methods are used. They often have a broader range of thermal tolerance than Sensitive-rare taxa and comprise a substantial portion of natural communities and often exhibit negative response (loss of population, richness) at mild pollution loads or habitat alteration.

Attribute IV: Taxa of intermediate tolerance.
Taxa that comprise a substantial portion of natural communities; may be r-strategists (early colonizers with rapid turn-over times; e.g.," boom/bust population characteristics). May be eurythermal (having a broad thermal tolerance range). May have generalist or facultative feeding strategies enabling utilization of relatively more diversified food types. Readily collected with conventional sample methods. May increase in number in waters with moderately increased organic resources and reduced competition but are intolerant of excessive pollution loads or habitat alteration.

Attribute V: Tolerant taxa.
Taxa that comprise a low proportion of natural communities. Taxa often are tolerant of a broader range of environmental conditions and are thus resistant to a variety of pollution or habitat induced stress. They may increase in number (sometimes greatly) in the absence of competition. Commonly r-strategists (early colonizers with rapid turn-over times: e.g., "boom/bust" population characteristics), able to capitalize when stress conditions occur. These taxa are the last survivors in highly disturbed systems.

***Taxa tolerance to stressors* (ATTRIBUTES I-V).**
Taxa differ in their sensitivities to stressors. Changes in the numbers, kinds and relative abundance of taxa across stressor gradients are important and useful indicators of adverse

effects (Cairns 1977, Karr 1981). Sensitivity of taxa to stress can vary among species, as well as with stressor. Shifts in taxa as a function of differing sensitivities to aquatic and riparian disturbance are well documented (Table 2-2). For perennial streams in temperate zones, disturbance tends to select for short-lived, tolerant species and against longer-lived, less tolerant species (Pianka 1970, Odum 1985, Rapport et al. 1985). In the highest quality tiers of the BCG, locally endemic taxa that are long-lived and ecologically specialized are well represented. With increasing stress, assemblage composition shifts towards tolerant species or short-lived taxa that can rapidly colonize disturbed environments. Assemblages in the lower tiers are dominated by eurytopic taxa (those with wide environmental ranges) with generalist or facultative feeding strategies.

TABLE 2-2. Evidence in support of the depicted changes in ecological attributes in the BCG.

BCG Attribute	Response	Case-specific documentation	Reference
I-V	Shifts in the numbers and kinds of species present, and in the number of individuals per species, as a function of varying tolerances to different kinds of aquatic and riparian disturbance.	changes in lake diatom species composition in response to intentional fertilization	Zeeb et al. 1974; Yang et al. 1996
		loss of sculpins downstream of metal mines	Mebane et al. 2003
		changes in algal species across a nutrient gradient in the Florida Everglades	Stevenson et al. 2002
		changes in diatom assemblages with increased acidification and eutrophication of lakes	Dixit et al. 1999
		shifts in species composition along a gradient of pulp and paper mill effluent concentration in a Maine river	Rabeni et al. 1988
		shifts in damselfly species from specialist species to generalist species along a gradient of organic pollution in an Italian river	Solimini et al. 1997
		variable sensitivities of benthic macroinvertebrate species to acidic conditions	Courtney and Clements 2000
		changes in fish species composition in an Oregon river with increased nutrients and temperature	Hughes and Gammon 1987
		differentially tolerant fish species in response to heavy metal and dissolved oxygen gradients in two Indian rivers	Ganasan and Hughes 1998
		decline in darters, sunfish, and suckers as well as other intolerant fishes and increase in tolerant fishes in the Midwest	Karr et al 1986; Yoder and DeShon 2003
		variable responses of stream amphibians to severe siltation	Welsh and Ollivier 1998
	Shifts from K-selected strategists to r-selected strategists following disturbance or in response to pollution	shifts from fragmentation-sensitive to fragmentation-tolerant bird species in relation to disturbed riparian habitats	Croonquist and Brooks 1993; Allen and O'Connor 2000; Bryce et al. 2002
		higher proportion of r-selected species in a flow regulated river as compared to a natural flow regime river	Nilsson et al. 1991
		shift to r-selected, generalist damselfly species along a gradient of increasing pollution	Solimini et al. 1997
		water-level fluctuation in a mesocosm resulted in increased proportion of r-strategist species	Troelstrup and Hengenrader 1990
		high pollutional stress correlated with increase in r-selected strategists in the same river 21 years apart	Richardson et al. 2000
	Regional and national species attribute lists and taxonomic tolerance values	compendium of pollution tolerance, habitat preferences, feeding guilds for fish species of the northeastern U.S.	Halliwell et al. 1998
		compendium of pollution tolerance, habitat preferences, feeding guilds for fish species of the Pacific northwest, U.S.	Zaroban et al. 1999
		organic pollution tolerance ranks for Wisconsin stream insect taxa	Hilsenhoff 1987
		compendium of pollution tolerance, habitat preferences, feeding guilds of North American fish and aquatic macroinvertebrate taxa	Barbour et al. 1999

TABLE 2-2. Evidence in support of the depicted changes in ecological attributes in the BCG.

BCG Attribute	Response	Case-specific documentation	Reference
VI	Detrimental effects of non-native taxa	loss of 150-200 endemic species in Lake Victoria following intentional introduction of Nile perch (*Lates niloticus*) and Nile tilapia (*Oreochromis niloticus*)	Witte et al. 1992
		dominance of many lowland rivers in the western USA by non-native fishes and invertebrates	Moyle 1986, Karr et al 1986, Miller et al. 1989
		food web disruption and loss of native mussels from zebra mussel invasion	Whittier et al. 1995
		detrimental changes in non-native taxa in TVA rivers where Corbicula is present	Kerans and Karr 1994
		loss of small, soft-finned fish species from Northeast USA lakes following predator introductions	Whittier and Kincaid 1999
		mid-twentieth century collapse of native salmonid fisheries following colonization of the Laurentian Great Lakes by sea lamprey (*Petromyzon marinus*) and alewife (*Alosa pseudoharengus*)	Smith 1972
VII	Changes in organism condition or increase in anomalies in response to pollution gradients	increased fish anomalies in the vicinity of toxic outfalls	Hughes and Gammon 1987, Yoder and Rankin 1995b
		altered blood chemistry and mortality in fish associated with wetlands that received oil sands effluent	Bendellyoung et al. 2000
		changes in growth, organism condition, fecundity, and feeding strategies for creek chub (*Semotilus atromaculatus*) across a variety of pressure gradients (urbanization, agriculture, temperature)	Fitzgerald et al. 1999
		the presence of tumors, deformities, lesions, etc. in the fish from highly disturbed streams	Karr et al. 1986, Yoder and DeShon 2003
VIII	ecosystem-level disruptions of functional integrity	extinction and succession of littoral lake invertebrate species secondary to lake acidification; initially detected by temporal changes in taxonomic and density measures but followed by top-down and bottom up effects at all trophic levels, caused by reduced nutrient cycling. A trophic cascade ultimately involved loss of fish and increased biomass of primary producers.	Appelberg et al. 1993
		simplification of global coastal ocean ecosystems to microbial domination due to combined effects of historical and current overfishing and pollution	Jackson et al. 2001
IX	influence of spatial and temporal scale of pressures on biological effects and recovery potential	large-scale, multi-state status and trends assessments of Pacific salmon influenced the listing of the species under the Endangered Species Act	Nehlsen et al. 1991
		environmental factors operating at different temporal and spatial scales influence the production and survivorship of juvenile Atlantic salmon	Poff and Huryn 1998
		past land use activity has long-term effects on aquatic bio-diversity	Harding et al. 1998
		assessments of stream fish and benthic macroinvertebrate assemblages at state and regional scales reveal serious alterations in indicators of biological integrity	U.S. EPA 2000a
		Ocean-wide ecological extinction of large predators from historical and current overfishing	Myers and Worm 2003
X	ecosystem connectance	replacement of 4 native freshwater fish species by 37 marine species in the lower Rio Grande following flow diversions that caused the lower river to cease flowing and become tidal salt water	Contreras-Balderas et al. 2002
		decreased fish species and guilds with decreased riverine connectivity with floodplain water bodies	Aarts et al. 2004
		5 federally listed headwater fish species have had their ranges restricted and isolated by mainstem impoundments, increasing their susceptibility to local physical and chemical habitat degradation	Freeman et al. 2005

TABLE 2-2. Evidence in support of the depicted changes in ecological attributes in the BCG.

BCG Attribute	Response	Case-specific documentation	Reference
		alteration of natural flow regimes result in changes in biological assemblage structure	Poff et al. 1997, Bunn and Arthington 2002
		extirpation of Pacific Northwest salmon following construction of impassable dams	Frissell 1993
		extirpation of Colorado River fishes following dam construction	Holden and Stalnaker 1975

Attribute VI: Non-native or intentionally introduced taxa.
With respect to a particular ecosystem, any species that is not found in that ecosystem. Species introduced or spread from one region of the U.S. to another outside their normal range are non-native or non-indigenous, as are species introduced from other countries.

This attribute represents both an effect of human activities and a stressor in the form of biological pollution. Although some intentionally introduced species are valued by large segments of society (e.g., gamefish), these species may be just as disruptive to native species as undesirable opportunistic invaders (e.g., zebra mussels). Many rivers in the U.S. are now dominated by non-native fishes and invertebrates (Moyle 1986), and introductions of alien species are the second most important factor contributing to fish extinctions in North America (Miller et al. 1989). The BCG identifies maintenance of native taxa as an essential characteristic of Tier 1 and 2 conditions. The model only allows for the occurrence of non-native taxa in these tiers if those taxa do not displace native taxa and do not have a detrimental effect on native structure and function. Tiers 3 and 4 depict increasing occurrence of non-native taxa. Extensive replacement of native taxa by tolerant or invasive, non-native taxa can occur in Tiers 5 and 6.

Organism Condition and System Performance: Attributes VII and VIII

Attribute VII: Organism condition.
Organism condition is an element of ecosystem function, expressed at the level of anatomical or physiological characteristics of individual organisms.

Organism condition includes direct and indirect indicators such as fecundity, morbidity, mortality, growth rates, and anomalies such as lesions, tumors, and deformities and for purposes of the BCG, primarily applies to fish and amphibians. Some of these indicators are readily observed in the field and laboratory, whereas the assessment of others requires specialized expertise and much greater effort. The most common approach for State and Tribal programs is to forego complex and demanding direct measures of organism condition (e.g., fecundity, morbidity, mortality, growth rates) in favor of indirect or surrogate measures (e.g., % of organisms with anomalies, age or size class distributions) (Simon (ed.) 2003). Organism anomalies in the BCG vary from naturally occurring incidence in Tiers 1 and 2 to higher than expected incidence in Tiers 3 and 4. In Tiers 5 and 6, biomass is reduced, the age structure of populations indicates premature mortality or unsuccessful reproduction, and the incidence of serious anomalies is high.

Attribute VIII: Ecosystem function.
"Function" refers to any processes required for normal performance of a biological system. The term may be applied to any level of biological organization. Immigration and emigration are functional processes at the population level. Examples of ecosystem functional processes are primary and secondary production, respiration, nutrient cycling, and decomposition.

The "functional integrity" of an ecosystem refers to the aggregate performance of dynamic interactions among an ecosystem's biological parts (Cairns 1977). The term "ecosystem function" includes measures of both the interactions among taxa (food web dynamics) and energy and nutrient processing rates (energy and nutrient dynamics). These attributes are included in the BCG because ecologists universally recognize their fundamental importance. At this time, the level of effort required to directly assess ecosystem function is beyond the means of most State and Tribal monitoring programs. Instead, most programs rely on taxonomic and structural indicators to make inferences about functional status (Karr et al. 1986). For example, shifts in the primary source of food may cause changes in trophic guild indexes or indicator species. Although direct measures of ecosystem function are currently difficult or time consuming, they may become practical in the future (Gessner and Chauvet 2002).

Attribute VIII also includes aspects of individual, population, and community condition. Altered interactions between individual organisms and their abiotic and biotic environments may generate changes in growth rates, reproductive success, movement, or mortality. These altered interactions are ultimately expressed at ecosystem-levels of organization (e.g., shifts from heterotrophy to autotrophy, onset of eutrophic conditions) and as changes in ecosystem process rates (e.g., photosynthesis, respiration, production, decomposition). Maine's example scenario (Table 2-3, located at the end of this chapter) describes a progression of functional changes. It depicts a naturally oligotrophic and heterotrophic system with P/R <1 in Tiers 1 and 2. Tiers 3 and 4 depict functional changes commonly associated with the effects of increased temperature and nutrient enrichment (P/R > 1, diurnal sags in dissolved oxygen, changes in taxonomic composition and relative abundance, increased algal biomass). Tier 5 depicts an autotrophic system impacted by excessive algal biomass.

Scale-dependent Factors: Attributes IX and X

Attribute IX: Spatial and temporal extent of stressor effects.
The spatial and temporal extent of stressor effects includes the near-field to far-field range of observable effects of the stressor. Patchy islands or periods of unsuitable conditions, within a generally intact system, give way to patchy islands or periods of suitable conditions, within a substantially degraded system.

Attribute X: Ecosystem connectance.
Access or linkage (in space/time) to materials, locations, and conditions required for maintenance of interacting populations of aquatic life; the opposite of fragmentation; necessary for metapopulation maintenance and natural flows of energy and nutrients across ecosystem boundaries.

***Scale-dependent factors* (ATTRIBUTES IX AND X).**
These attributes relate to interactions between the physical environment in all its aspects (spatial, temporal, structural, chemical, etc.), and the biota. Attributes IX and X are interpreted at different spatial and even temporal scales than the rest of the attributes, i.e., the reach, or sampled community perspective has been expanded to consider alterations occurring within entire catchments, basins, and regions, or within seasonal and annual cycles. These attributes were included in the BCG because the extent of ecosystem alteration has important environmental implications in terms of an individual waterbody's vulnerability to further effects from stressors as well as potential for mitigation. For example, ecosystem connectivity is fundamental to the successful recruitment and maintenance of organisms into any environment. A single impacted stream reach in an otherwise intact watershed has far more

restoration potential than a similar site in a basin that has undergone extensive land-scape alteration (Table 2-2). Tiers 1 and 2 depict a naturally connected or isolated system in which a natural disturbance regime, e.g. natural variability, is maintained. Detrimental effects in Tiers 3 and 4 are limited to the reach or seasonal scale. The two lowest tiers depict a system with detrimental effects extending to the catchment scale and affecting multiple seasons. A few "islands" of adequate physical/chemical conditions may serve as refugia in Tier 5, but extensive loss of connectance and refugia occur in Tier 6.

2.1.2 The BCG Tiers

Although the BCG is continuous in concept, it has been divided into six tiers to provide as much discrimination of different levels of condition as workgroup members deemed discernable, given current assessment methods and robust monitoring information (Figure 2-1). Defining the tiers between 3 and 5 was a challenge to the workgroup and entailed considerable discussion. The workgroup ultimately agreed some States and Tribes may only be capable of discriminating 3-4 tiers, while others might be capable of discerning 6 tiers based on characteristics of their database and monitoring program. However the workgroup agreed that the important role of the BCG model is to be a starting point for a State or Tribe to think about how to use information to better define their designated aquatic life uses and to communicate more clearly about biological condition. There is no expectation that States and Tribes establish six tiers of use classes. The ultimate number of the tiers is a State or Tribal determination.

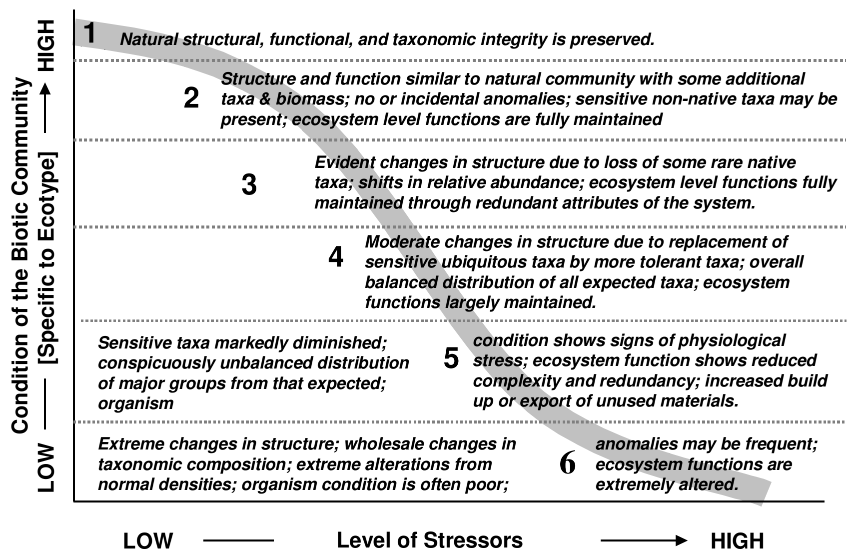


FIGURE 2-1. Conceptual model of the Biological Condition Gradient.

Tier 1: Natural or native condition.
Native structural, functional, and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability.

Tier 1 represents biological conditions as they existed (or still exist) in the absence of measurable effects of stressors. The Tier 1 biological assemblages that occur in a given biogeophysical setting are the result of adaptive evolutionary processes and biogeography that selects in favor of survival of the observed species. For this reason, the expected Tier 1 assemblage of a stream from the arid southwest will be very different from that of a stream in the northern temperate forest. The maintenance of native species populations and the expected natural diversity of species are essential for Tiers 1 and 2. Non-native taxa (Attribute VI) may be present in Tier 1 if they cause no displacement of native taxa, although the practical uncertainties of this provision are acknowledged (discussed in Section 2.2).

Attributes I and II (e.g., historically documented and sensitive taxa) can be used to help assess the status of native taxa and could be a surrogate measure to identify threatened or endangered species when classifying a site or assessing its condition.

Tier 2: Minimal changes in structure of the biotic community and minimal changes in ecosystem function.

Virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability.

Tier 2 represents the earliest changes in densities, species composition, and biomass that occur as a result of slight elevation in stressors (such as increased temperature regime or nutrient enrichment). There may be some reduction of a small fraction of highly sensitive or specialized taxa (Attribute II) or loss of some endemic or rare taxa as a result. Tier 2 can be characterized as the first change in condition from natural and it is most often manifested in nutrient enriched waters as slightly *increased* richness and density of sensitive ubiquitous taxa and taxa of intermediate tolerance (Attributes III and IV). These early response signals have been observed in many State programs as illustrated in Figure 2-2, showing slight to moderate increases in conductivity in Maine streams.

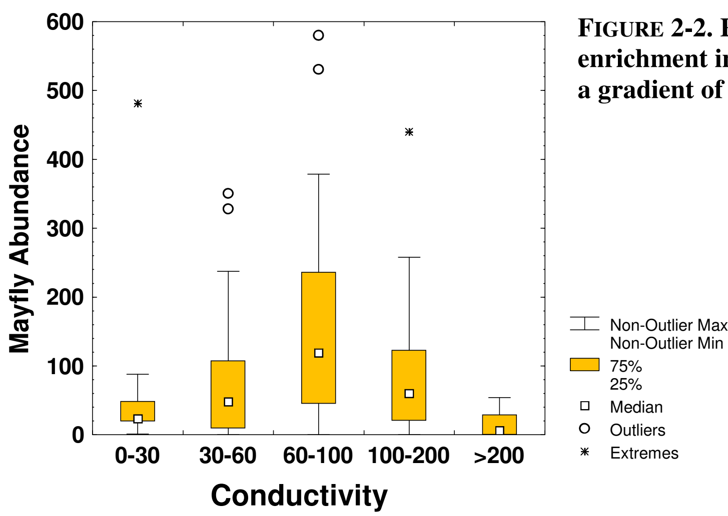


FIGURE 2-2. Response of mayfly density to enrichment in Maine streams as indicated by a gradient of increasing conductivity.

Tier 3: Evident changes in structure of the biotic community and minimal changes in ecosystem function.

Evident changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa but sensitive-ubiquitous taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system.

Tier 3 represents readily observable changes that, for example, can occur in response to organic enrichment or increased temperature. The "evident" change in structure for Tier 3 is interpreted to be perceptible and detectable decreases in sensitive-rare or highly sensitive taxa (Attribute II) and increases in sensitive-ubiquitous taxa or opportunist organisms (Attributes III and IV). Attribute IV taxa (intermediate tolerants) may increase in abundance as an opportunistic response to nutrient inputs.

Tier 4: Moderate changes in structure of the biotic community with minimal changes in ecosystem function.
Moderate changes in structure due to replacement of some sensitive-ubiquitous taxa by more tolerant taxa, but reproducing populations of some sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes.

> Moderate changes of structure occur as stressor effects increase in Tier 4. A substantial reduction of the two sensitive attribute groups (II and III) and replacement by more tolerant taxa (Attributes IV and V) may be observed. A key consideration is that some Attribute III sensitive taxa are maintained at a reduced level but are still an important functional part of the system (function maintained).

Tier 5: Major changes in structure of the biotic community and moderate changes in ecosystem function.
Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from those expected; organism condition shows signs of physiological stress; ecosystem function shows reduced complexity and redundancy; increased build-up or export of unused materials.

> Changes in ecosystem function (as indicated by marked changes in food-web structure and guilds) are critical in distinguishing between Tiers 4 and 5. This could include the loss of functionally important sensitive taxa and keystone taxa (Attribute I, II and III taxa) such that they are no longer important players in the system, though a few individuals may be present. Keystone taxa control species composition and trophic interactions, and are often, but not always, top predators. As an example, removal of keystone taxa by overfishing has greatly altered the structure and function of many coastal ocean ecosystems (Jackson et al. 2001). Additionally, tolerant non-native taxa (Attribute VI) may dominate some assemblages and changes in organism condition (Attribute VII) may include significantly increased mortality, depressed fecundity, and/or increased frequency of lesions, tumors and deformities.

Tier 6: Severe changes in structure of the biotic community and major loss of ecosystem function.
Extreme changes in structure; wholesale changes in taxonomic composition; extreme alterations from normal densities and distributions; organism condition is often poor; ecosystem functions are severely altered.

> Tier 6 systems are taxonomically depauperate (low diversity and/or reduced number of organisms) compared to the other tiers. For example, extremely high or low densities of organisms caused by excessive organic enrichment or severe toxicity may characterize Tier 6 systems.

2.2 How the BCG was developed, tested, and evaluated

In developing the BCG, the workgroup believed it was important that the model be grounded in sound theory, easy to apply, and meet the needs of practitioners around the country.

The BCG model was developed and tested by the TALU Workgroup. Based on recommendations from the full workgroup, a steering committee created a matrix that summarized biologists' experience and knowledge about how biological attributes change in response to stress in aquatic ecosystems (Table 2-1). In developing the BCG, the workgroup believed it was important that the model be grounded in sound theory as well as actual empirical observations, easy to apply, and meet the needs of users around the country. In building the model, the workgroup followed an iterative, inductive approach, similar to means-end analysis (Martinez 1998). The model was tested by determining how consistently workgroup members assigned samples of macroinvertebrates or fish to the six tiers, the results of which support the contention that the BCG represents aspects of biological condition common to all existing assessment methods.

The workgroup began by testing whether biologists from different parts of the country would draw similar conclusions regarding the condition of a waterbody using simple lists of organisms and their counts. This approach was based on Maine's experience, in which expert biologists independently assigned samples of macroinvertebrates to *a priori* defined classes of biological condition defined by differences in assemblage attributes (Davies et al. 1995). Decision instructions were provided to biologists in the form of a matrix, which outlined expected trajectories of quantifiable aspects of invertebrates *(See Case Example 3-3 in the next chapter)*. These corresponded with biological expectations for four water quality classes (A, B, C and Non-Attainment; *See Appendix A, Tables A-1 and A-2*). The high level of majority and unanimous agreement (98% and 64% respectively) among experts in placing samples into the different classes allowed Maine to develop a predictive statistical model that is now used to assess the biological condition of new sites (Courtemanch 1995) (*See Case Example 3-3)*.

To provide a functional framework for practitioners, the TALU Workgroup described how each of the ten attributes varies across six tiers of biological alteration (Table 2-1). The general model was then described in terms of the biota of a specific region (Maine). Based on 20 years of biomonitoring data, the Maine example describes how the relative densities of specific taxa with varying sensitivities to stressors change across the BCG tiers (Table 2-3, located at the end of this chapter).

To test the general applicability of the BCG to sampling data taken from real ecosystems, the workgroup evaluated how consistently individual biologists classified samples of aquatic biota based on the attributes incorporated into the BCG. Governmental and research biologists from 23 States and one Tribe participated in the data exercise. The full workgroup was divided into breakout groups according to regional (Northeast, South-Central, Northwest, Arid Southwest/Great Plains) or assemblage (fish, invertebrates) expertise. Samples were selected from invertebrate and fish data sets to span as many of the BCG tiers as possible. The invertebrate samples and fish samples used in the tests were collected from six different regions within the U.S. (Northeast, Mid-Atlantic, Southeast, Northwest, Southwest, Central) and included only basic descriptors of stream physical characteristics (substrate, velocity, width, depth, etc.), taxonomic names, densities, and in some cases, metric values. These data represent the basic core elements common to nearly all biological monitoring programs. Participants were asked to place each sample into one of the six condition tiers, though they were cautioned not to apply a simple relative quality ranking since all six tiers did not necessarily occur within the data sets. Biologists relied primarily on differences in relative abundances and sensitivities of taxa (i.e., Attributes I-VI) to make tier assignments because information needed to evaluate the status of the other Attributes was not available. Percent concurrence among the individuals was calculated to assess the level of agreement among biologists when applying the BCG to raw data. Perfect concurrence was set to equal the product of the number of raters by the number of streams. *Case Examples 3-2, 3-3, and 3-7, at the end of Chapter 3, outline how Maine and New Jersey biologists described tiers and assigned sites.*

In the first stage of the data exercise, between-biologist differences were evaluated by asking workgroup participants to rate a single data set of 6-8 samples. The breakout groups were then asked to classify samples from larger and more variable datasets. The groups were also instructed to summarize their interpretations and to identify biological responses to changes in conditions not captured by the BCG. Finally, the groups identified which tiers corresponded to how they currently assess biological integrity and the CWA interim goal for protection and propagation of aquatic life.

Workgroup members placed 82% of the benthic macroinvertebrate samples and 74% of the fish samples into the same BCG tiers. The range of variation among individuals was within one tier's distance in either direction. Tiers were revised following full workgroup discussion so that transitions were more distinct. Each of the breakout groups independently reported that the ecological characteristics approximately described by Tiers 4 and above were compatible with how they currently assess the CWA's interim goal for protection and propagation of aquatic life. These groups also identified the characteristics described by Tiers 1 and 2 as indicative of biological integrity.

Workgroup members reported that key concepts were important with respect to classifying samples into tiers and identifying the boundaries in between. For Tiers 1 and 2, biologists identified the maintenance of native species populations as essential to their understanding of biological integrity. Although many participants noted that criteria for distinguishing differences between tiers in Attribute VIII (ecosystem function) were poorly defined, most nevertheless identified ecosystem function changes (as indicated by marked changes in food-web structure and guilds) as critical in distinguishing between Tiers 4 and 5.

Discussion following the BCG exercise revealed that participants readily agreed on some of the condition attributes, but not others. For example, participants indicated they mostly used Attributes I-V (taxonomic composition and tolerance), Attribute VI (non-native taxa, for Tiers 2-6 only) and Attribute VII (organism condition) to evaluate biological conditions. In contrast, because Attributes VIII - X (ecosystem function and scale-dependent features) are rarely directly assessed by biologists, the evaluation of these attributes was accompanied by relatively high uncertainty. Even so, workgroup members strongly advocated retaining these attributes in the BCG because of the importance of this information in making restoration decisions.

The presence of non-native taxa in Tier 1 was also the subject of considerable discussion. Knowledge of the extensive occurrence of some non-native taxa in otherwise near-pristine systems conflicted with the desire by many to maintain a conceptually pure and natural tier. Further discussion resulted in agreement that the presence of non-native taxa in Tier 1 is permissible only if they cause no displacement of native taxa, although the practical uncertainties of this provision were acknowledged. The resulting tier descriptions, which allow for non-native species in the highest tiers as long as there is no detrimental effect on the native populations, has practical management implications. For example, introduced European brown trout (*Salmo trutta*) have replaced native brook trout (*Salvelinus fontinalis*) in many eastern U.S. streams. In some catchments, brook trout only persist in stream reaches above waterfalls that are barriers to brown trout. The downstream reaches are nearly pristine except for the presence of brown trout (D. Lenat, North Carolina Department of Natural Resources, personal communication). In these places, if society decided to remove the introduced brown trout and if stream habitat is preserved throughout the catchment, brook trout can potentially repopulate downstream reaches. In the use designation process, recognizing that the entire catchment has the *potential* to attain Tier 1 conditions will inform the public that a very high quality resource exists.

Critical gaps in knowledge were uncovered during the development of the BCG. For example, the workgroup identified the need for regional evaluations of species tolerance to stressors associated with pressure. Tolerance information presented in the current version of the BCG tends to be based on generalized taxa responses to a non-specific stressor gradient. At this time, tolerance information is not available for most taxa and for many common stressors (temperature, nutrients, sediments). In some cases, tolerance values are based on data collected in other geographic regions or for other purposes (e.g., van Dam's European diatom tolerances are used for North American taxa) (van Dam et al. 1994). Improved tolerance value information is needed to refine the BCG and improve its precision.

Additionally, taxa that are considered tolerant to stressors in one region of the country may not be similarly classified in another region. For example, long-lived taxa have generally been characterized as sensitive to increasing pressure and tend to be replaced by short-lived taxa in stressed systems. As such, the presence of long-lived taxa in a waterbody has been used to indicate high quality conditions, whereas the predominance of short-lived taxa indicates degradation. However, in small streams in the arid western U.S., extreme changes in hydrology define the natural regime for some systems and an opposite trend has been observed: short-lived taxa can dominate the biological community in natural settings. In these systems, a shift to long-lived taxa may be an indicator of altered, less variable flow regimes.

2.3 The relationship between the BCG and designated uses

The BCG is a model that provides a rational and consistent way to identify and communicate waterbody condition. It can thus be used to establish appropriate ALUs in State water quality standards and to assess attainment. The ecological condition to support an ALU for a specific waterbody can be described in terms of the BCG tiers and can be related to specific use categories such as fishery-based uses. For example, the ecological condition needed to support salmon spawning is an exceptional, high-quality natural stream and will likely be either a Tier 1 or 2 on the BCG. The ecological attributes that characterize the BCG tiers can be measured with methods used by each State, and these condition assessments can be directly linked to a State's ALUs.

Maine and Ohio are examples of States that have adopted uses based on a biological condition gradient into water quality standards (Courtemanch et al. 1989, Yoder and Rankin 1995a). Both of these States have incorporated multiple tiers of resource quality in their water quality standards (State of Maine 1985, 2003; Davies et al. 1995; State of Ohio 2003). As discussed above, the tiers in these States' TALUs describe aquatic-life management goals and attainment criteria for different waterbody types. For example, in Maine a waterbody is assigned to one of four management tiers by considering both its existing biological condition and its highest attainable condition as determined by a public and legislative process. These four tiers of biological quality in Maine's water quality standards are based on Odum's subsidy stress gradient (Odum et al. 1979, Odum 1985) (S*ee Appendix A, Figure A-2a and Table A-1*). Attainment of standards is assessed by determining to which tier a sample of macroinvertebrates is most similar (Courtemanch et al. 1989). Site-specific taxonomic composition data and other metrics are used in a discriminant model to identify the class of a particular waterbody (*See Case Examples 3-3 and 3-6 in Chapter 3*). Maine has found multiple tiers to be useful in 5 ways:

1) identifying and preserving the highest quality resources,
2) depicting existing conditions more accurately,
3) setting realistic and attainable management goals,
4) preserving incremental improvements, and
5) determining appropriate management action when conditions decline.

Over the past thirty years, States have independently developed technical approaches to assess condition and set ALUs specific to the biology of the State and its regulatory and political settings (U.S. EPA 2002a). Although these different approaches have fostered innovative technical approaches, they have also complicated the development of a nationally consistent approach to interpreting the condition of aquatic resources. Assessment results are often difficult to compare when quantitative outcomes (i.e., index or indicator values) represent different qualitative conditions. Additionally, without a common interpretative framework, use of different methods can hinder collaboration among natural resource agencies that have complementary missions. A consistent approach to interpreting biological condition will allow scientists and the public to more effectively evaluate the current and potential conditions of specific waters and watersheds and use that information to set appropriate ALUs.

The BCG can help promote consistent interpretation of scientific data by applying a common framework to diverse conditions and different assessment methods at national, regional, state, or watershed levels. By providing a means for managers and the public to identify outstanding resources, recognize incremental improvements, more appropriately allocate resources and prioritize management actions, aquatic and natural resource agencies will be able to coordinate and target resources more effectively.

2.4 Key points from Chapter 2

1. **The biological condition gradient is a descriptive model predicting biological response to increasing levels of stressors**. The biological gradient can be thought of a field-based dose-response curve where dose (x-axis) is level of stressors and response (y-axis) is biological condition.

2. **The purpose of the Biological Condition Gradient is to provide an ecologically-based model about biological condition and to promote clearer understanding of current conditions relative to natural conditions**. This should result in more meaningful engagement of the public in the designation of aquatic life uses in State and Tribal water quality standards programs.

3. **The model must be validated with data**. The BCG model does not reduce the necessity of developing robust methods for the quantitative and statistical validation of biological conditions. The list of attributes is intended to organize how we interpret biological information concerning a given aquatic community response to increasing levels of stressors. The approach should be thought of as seeking to identify a "best fit" tier, which consists of weighing the importance and signal-strength of the different attributes as they pertain to a specific waterbody or as used to describe a designated use class.

4. **The conceptual framework is not defined by any one method**. As presented in Chapter 3, the attributes have a quantifiable aspect that can potentially be assessed and validated in many different ways. The BCG has been designed to be independent of different assessment methodologies (i.e. Rapid Biological Assessment, Index of Biological Integrity; RIVPACS, multivariate analyses, etc.). The intent is for the ecological premises that support the model to reflect the same basis that underlies all successful methods used to quantify biological response to increasing levels of stressors.

5. **The number of useful tiers is flexible**. The purpose of the number of tiers is to provide a highly resolved biological condition gradient. There is no expectation that State or Tribal programs adopt six tiers, or categories, of designated uses. While step-wise progress toward refinement of designated aquatic life uses in State and Tribal water quality standards programs is desired over the long term, the ultimate number and type of tiers of uses is a State or Tribal determination.

6. **The BCG was designed to facilitate communication of the current biological condition of a waterbody compared to natural conditions**. For example, the BCG is grounded in natural conditions, which can help users and the public understand that current conditions do not necessarily represent natural conditions. In areas where natural or near-natural conditions exist, people are generally familiar with what is natural and what is altered. But in extensively altered regions practitioners and the public alike tend to accept the "best of what is left" as the potential for a system. In such places, it is difficult to visualize the natural conditions that were once present and designated uses may end up based on a diminished perspective. Natural conditions may not be achievable in many places, but an improved understanding of the changes that have occurred will result in a more scientifically defensible evaluation of current conditions and what can potentially be restored.

The next chapter provides information on how to adapt the national BCG model to reflect the specific ecology and stressor gradient characteristics of a particular state or region, and introduces some ways to quantify a biological condition gradient with monitoring data.

TABLE 2-3. Biological Condition Gradient: Maine example scenario for a cold-water stream catchment. [1]

Resource Condition "Tiers"	**Biological Condition Characteristics (Effects)**
1 **Natural or native condition** *Native structural, functional and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability*	**I** ***Historically documented, sensitive, long-lived, or regionally endemic taxa*** ➔ Long-lived native species of fish-host specialist or long-term brooder mussels such as Brook floater- *Alasmodonta varicosa;* Triangle floater- *Alasmodonta undulata;* Yellow lampmussel- *Lampsilis cariosa* are present in naturally occurring densities ➔ **Fishes**: Brook stickleback, Swamp darter **II** ***Sensitive- rare taxa*** ➔ The proportion of total richness represented by rare, specialist and vulnerable taxa is high, for example, without limitation, the following taxa are representative: **Plecoptera**: Capniidae, *Taeniopteryx, Isoperla, Perlesta, Pteronarcys, Leuctra*; **Ephemeroptera**: *Cinygmula, Rhithrogena, Epeorus, Serratella, Leucrocuta*; **Trichoptera**: *Glossosoma; Psilotreta; Brachycentrus*; **Diptera**: *Stempellina, Hexatoma, Probezzia*; **Coleoptera**: *Promoresia;* **Fishes***:* Slimy sculpin, Longnose sucker; Longnose dace **III** ***Sensitive- ubiquitous taxa*** ➔ Densities of Sensitive-ubiquitous taxa are as naturally occur. The following taxa are representative of this group for Maine: **Plecoptera**: *Acroneuria*; **Ephemeroptera**: *Stenonema, Baetis, Ephemerella, Pseudocloeon*; **Fishes**: Brook trout, Burbot, Lake chub **IV** ***Taxa of intermediate tolerance*** ➔ Densities of *intermediate tolerance taxa* are as naturally occur. The following taxa are representative of this category: **Trichoptera**: Hydropsychidae, Chimarra, *Neureclipsis*, *Polycentropus;* **Diptera**: *Tvetenia, Microtendipes, Rheocricotopus, Simulium;* **Fishes**: Common shiner, Fallfish **V** ***Tolerant taxa*** ➔ Occurrence and densities of Tolerant taxa are as naturally occur. The following taxa are representative of this category: **Diptera**: *Dicrotendipes, Tribelos, Chironomus, Parachironomus*; **Non-Insects**: Caecidotea, Isopoda, *Physa*, *Helobdella;* **Fishes***:* White sucker, Blacknose dace, Creek chub **VI** ***Non native or intentionally introduced taxa*** ➔ Non native taxa such as Brown trout, Rainbow trout, Yellow perch, are absent or, if they occur, their presence does not displace native biota or alter native structure and function **VII** ***Physiological condition of long-lived organisms*** ➔ Anomalies are absent or rare; any that occur are consistent with naturally occurring incidence and characteristics **VIII** ***Ecosystem Function*** ➔ Rates and characteristics of *life history (e.g., reproduction, immigration, mortality, etc.)*, and materials exchange processes *(e.g., production, respiration, nutrient exchange, decomposition, etc.)* are comparable to that of "natural" systems ➔ The system is predominantly heterotrophic, sustained by leaf litter inputs from intact riparian areas, with low algal biomass; P/R<1 (Photosynthesis: Respiration ratio) **IX** ***Spatial and temporal extent of detrimental effects*** ➔ Not applicable- disturbance is limited to natural events such as storms, droughts, fire, earth-flows. A natural flow regime is maintained. **X** ***Ecosystem connectance*** ➔ Reach is highly connected with groundwater, its floodplain, and riparian zone, and other reaches in the basin, at least annually. Allows for access to habitats and maintenance of seasonal cycles that are necessary for life history requirements, colonization sources and *refugia* for extreme events.

[1] This scenario presents Maine biologists' summary of the ecological characteristics of the six tiers in the Biological Condition Gradient model as observed in Maine (see Appendix A, Sections II and III). It is based on analysis of genus/species level benthic macroinvertebrate data (400 samples from rivers and streams spanning conditions from near-natural to severely altered) (Davies et al. 1999).

2

Minimal changes in structure of the biotic community and minimal changes in ecosystem function

Virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability

I *Historically documented, sensitive, long-lived, regionally endemic taxa*

- ➔ Some regionally endemic, long-lived species (e.g., some mussel species such as the Dwarf wedgemussel- *Alasmidonta heterodon*, and/or fish species, such as the Brook stickleback are absent due extirpation from Maine prior to the enactment of the CWA; some mussel species of Special Concern in Maine are present (e.g., Brook floater- *Alasmodonta varicosa;* Triangle floater- *Alasmodonta undulata;* Yellow lampmussel- *Lampsilis cariosa*)

II *Sensitive- rare taxa*

- ➔ Richness of rare and/or specialist invertebrate taxa is high though densities may be low (e.g., for Maine- **Plecoptera**: Capniidae, *Taeniopteryx, Isoperla, Agnetina, Perlesta, Pteronarcys, Leuctra*; **Ephemeroptera**: *Cinygmula, Rhithrogena, Epeorus, Serratella, Leucrocuta*; **Trichoptera**: *Glossosoma, Psilotreta, Brachycentrus*; **Diptera**: *Stempellina, Rheopelopia, Hexatoma, Probezzia*; **Coleoptera**: *Promoresia*). Densities of scrapers such as Glossosoma are increased
- ➔ Fish assemblage is predominantly native including such sensitive fish as Slimy sculpin, Longnose sucker, Longnose dace.

III *Sensitive- ubiquitous taxa*

- ➔ Superficial scraper-grazers and collector-gathers are favored due to slightly increased periphyton biomass on hard substrates, which results in higher relative abundance of these groups (e.g., **Ephemeroptera**: *Stenonema, Stenacron, Baetis, Ephemerella, Pseudocloeon*). Predatory stoneflies are common (e.g., *Acroneuria, Agnetina*). Populations of such native fish taxa as Brook trout, Lake chub, Burbot are common.

***IV* Taxa of intermediate tolerance**

- ➔ Increased biomass of diatom species that respond positively to increased nutrients and temperatures, but sensitive diatom species are maintained. Diatom richness is increased; filamentous forms are rare or as naturally occur
- ➔ May be slight increases in densities of macroinvertebrate taxa such as **Trichoptera**: Hydropsychidae, Philopotamidae, *Neureclipsis;* **Diptera**: *Rheotanytarsus, Microtendipes, Rheocricotopus, Simulium*
- ➔ Common shiner and Fallfish are in good condition

V *Tolerant taxa*

- ➔ May be slight increases in occurrence of tolerant taxa such as **Diptera**: *Polypedilum, Tvetenia,* **Non-Insects**: *Isopoda, Physa*; **Fishes**: White sucker; Creek chub, Blacknose dace

VI *Non-native or intentionally introduced* taxa

- ➔ Any intentionally introduced fish species (e.g., Brown trout- *Salmo trutta*, Rainbow trout- *Oncorhynchus mykiss)* occupy non-detrimental niche space

VII *Physiological condition of long-lived organisms*

- ➔ Any anomalies on fish are consistent with naturally occurring incidences and characteristics such as rare occurrence of gill or anchor parasites, blackspot, etc.
- ➔ Spawning areas of native fishes are evident during spawning season

VIII *Ecosystem Function*

- ➔ Rates and characteristics of *life history* (e.g., reproduction; immigration; mortality etc.), and materials exchange processes (e.g., production; respiration; nutrient exchange; decomposition e*tc.)* are unimpaired and not significantly different from the range of natural variability.
- ➔ The system is predominantly heterotrophic, sustained by leaf litter inputs from intact riparian areas; P/R/ is<1

IX *Spatial and temporal extent of detrimental effects*

- ➔ Extent is limited to small pockets or brief periods

X *Ecosystem connectance*

- ➔ Unimpaired access to habitats and maintenance of seasonal cycles that are necessary to fulfill *life history requirements*, and to provide colonization sources and *refugia* for extreme events.

3 **Evident changes in structure of the biotic community and minimal changes in ecosystem function** *Some changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa but <u>sensitive-ubiquitous</u> taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system*	**I *Historically documented, sensitive, long-lived, or regionally endemic* taxa** ➔ Brook floater- *Alasmodonta varicosa;* Triangle floater- *Alasmodonta undulata;* Yellow lampmussel- *Lampsilis cariosa;* are uncommon; Dwarf wedgemussel- *Alasmidonta heterodon* (and/ or a fish species) absent due to extirpation from Maine prior to CWA **II *Sensitive- rare taxa*** ➔ Some replacement of taxa having narrow or specialized environmental requirements, with functionally equivalent *sensitive-ubiquitous* taxa; coldwater obligate taxa are disadvantaged. Taxa such as **Plecoptera**: Capniidae, *Taeniopteryx, Isoperla, Perlesta, Pteronarcys, Leuctra*, *Agnetina*; **Ephemeroptera**: *Cinygmula, Rhithrogena, Epeorus, Serratella, Leucrocuta*; **Trichoptera**: *Glossosoma, Psilotreta, Brachycentrus*; **Diptera**: *Stempellina, Rheopelopia; Hexatoma, Probezzia*; **Coleoptera**: *Promoresia;* **Fishes**: Brook stickleback, Longnose sucker, Longnose dace are uncommonly encountered or absent **III *Sensitive- ubiquitous or generalist* taxa** ➔ Sensitive- ubiquitous or generalist taxa are common and abundant; taxa with broader temperature-tolerance range are favored (e.g., **Plecoptera**: *Acroneuria*; **Ephemeroptera**: *Stenonema, Baetis, Ephemerella, Pseudocloeon*) ➔ Overall mayfly taxonomic richness is reduced relative to the Tier 2 condition, with the preponderance of richness represented by sensitive- ubiquitous taxa; densities of remaining taxa are high and are sufficient to indicate healthy, reproducing populations ➔ Native Brook trout are significantly reduced due to the introduction of non-native Brown trout and the increased temperature regime **IV *Opportunist or facultative taxa of intermediate tolerance*** ➔ Filter-feeding blackflies (*Simulium*) and net-spinning caddisflies (e.g., *Hydropsyche, Cheumatopsyche, Polycentropus, Neureclipsis*) show increased densities in response to nutrient enrichment, but relative abundance of all expected major groups is well-distributed ➔ Increased temperature and increased available nutrients result in increased algal productivity causing an increase in the thickness of the diatom mat. This results in a "slimy" covering on hard substrates. ➔ Fish assemblage exhibits increased occurrence of Common shiner and Fallfish **V *Tolerant taxa*** ➔ Richness of **Diptera**: Chironomidae is increased; relative abundance of Diptera and Non-insects is somewhat increased but overall relative abundance is well-distributed among taxa from Groups III, IV and V, with the majority of taxa represented from Groups III and IV. Blacknose dace and white sucker are more common. **VI *Non-native or intentionally introduced* taxa** ➔ Brown trout have largely replaced native brook trout **VII** *Physiological condition of long-lived organisms* ➔ Incidence of *anomalies* such as gill parasites, anchor parasites, blackspot, etc., is low; serious anomalies such as tumors or deformities are essentially absent ➔ Environmental quality is sufficient to fully support reproduction of most long-lived species **VIII *Ecosystem Function*** ➔ Increased temperature and algal metabolism causes small diurnal sags in dissolved oxygen, compensated by adequate aeration from turbulence over riffle areas ➔ Algal biomass somewhat exceeds what can be utilized by resident grazers, resulting in evidence of die-back and slight downstream export of sloughed material. ➔ Patchy loss of high food quality riparian vegetation (e.g., oak; maple, beech) and elevated temperature, results in decreased growth and survival of some specialized shredder taxa (Pteronarcidae; Taeniopterygidae) with replacement by shredders capable of utilizing lower quality organic matter (Lepidostomatidae; Limnephilidae; Tipulidae). **IX *Spatial and temporal extent of detrimental effects*** ➔ Filamentous green algae occur in small patches within reaches; low dissolved oxygen levels occur only during the high temperature and low flow summer periods. ➔ Interstitial spaces, within the substrate of pools, are filled with fine sediment resulting in localized losses of interstitial habitats but riffle areas continue to provide adequate water flow and oxygen through interstitial habitats. **X *Ecosystem connectance*** ➔ Some downcutting has resulted in a patchy decrease in *connectance* of the stream from its floodplain except at unusually high flows. ➔ Thinning and patchy loss of riparian vegetation has altered the microclimate of the surrounding landscape causing a decrease in survival and reproductive success of adult mayflies and stoneflies.

4 **Moderate changes in structure of the biotic community and minimal changes in ecosystem function** *Moderate changes in structure due to replacement of some Sensitive-ubiquitous taxa by more tolerant taxa, but reproducing populations of some Sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes*	**I Historically documented, *sensitive, long-lived, regionally endemic* taxa** ➔ Healthy, reproducing populations of generalist mussel species are present (such as Eastern elliptio- *Eliptio complanata*; or Eastern lampmussel- *Lampsilis radiata radiata or* Eastern floater- *Pyganodon cataracta*) but Brook floater- *Alasmodonta varicosa;* Triangle floater- *Alasmodonta undulata;* Yellow lampmussel- *Lampsilis cariosa* are absent. **II *Sensitive- rare, specialist, vulnerable taxa with narrow environmental requirements*** ➔ Richness of specialist and vulnerable taxa is notably reduced; if present, densities are low (e.g., **Plecoptera**: Capniidae, *Taeniopteryx, Isoperla, Perlesta, Pteronarcys, Leuctra*; *Agnetina*; **Ephemeroptera**: *Cinygmula, Rhithrogena, Epeorus, Serratella, Leucrocuta*; **Trichoptera**: *Glossosoma; Psilotreta; Brachycentrus*; **Diptera**: *Stempellina, Rheopelopia; Hexatoma, Probezzia*; **Coleoptera**: *Promoresia,* **Fishes***:* Occurrence of Slimy sculpin, Longnose sucker and Longnose dace is reduced **III *Sensitive- ubiquitous or generalist taxa*** ➔ Densities of sensitive- ubiquitous scraper and gatherer insects (e.g., *Stenonema, Heptagenia, Baetis, Ephemerella, Pseudocloeon*) are sufficient to indicate that reproducing populations are present but relative abundance is reduced due to increased densities of opportunist invertebrate taxa (Group IV); ➔ Predatory stoneflies are reduced (e.g., *Acroneuria*) **IV *Opportunist or facultative taxa of intermediate tolerance*** ➔ Many substrate surfaces are covered by bryophytes and macro-algae responding to increased nutrients, resulting in displacement of lithophytic (stone-dwelling) micro-algae in favor of epiphytic (plant-dwelling) and filamentous forms (e.g., *Cladophora*). ➔ Increased loads of suspended particles favor collector-filterer invertebrates resulting in notably increased densities and relative abundance of filter-feeding caddisflies and chironomids (e.g., **Trichoptera**: Hydropsychidae, *Chimarra*, *Neureclipsis*, *Polycentropus;* **Diptera***: Tvetenia, Microtendipes, Rheocricotopus, Simulium;* **Fishes***:* Common shiner and Fallfish are common and abundant **V *Tolerant taxa*** ➔ There is an increase in the relative abundance of tolerant generalists (for example, *Polypedilum, Eukeifferiella, Cricoptopus*) and/or in numbers of non-insect scrapers and gatherers (e.g., *Physa, Sphaerium, Asellus, Hyalella)* but they do not exhibit significant dominance ➔ Overall relative abundance is well distributed among taxa from Groups III, IV and V, with the majority of the total abundance represented from Group IV. ➔ **Native fish such as White sucker, Blacknose dace, Creek chub are common.** **VI *Non-native or intentionally introduced* taxa** ➔ **Brook trout are absent or transient but such taxa as Smallmouth bass, Golden shiner and Yellow perch are common.** **VII** Physiological condition of long-lived organisms ➔ Incidence of anomalies such as blackspot and gill and anchor parasites is slightly higher than expected ➔ Occurrence of tumors, lesions and deformities is rare **VIII Ecosystem Function** ➔ Increased available nutrients increase algal productivity causing increased diatom, macro-algae and macrophyte biomass, and consequently lowering evening dissolved oxygen levels and increasing daytime oxygen levels. Invertebrate biomass is high but production has shifted to result in greater biomass of intermediate tolerance organisms than sensitive organisms. For example, filter-feeders utilizing suspended material shift from mayflies and sensitive mussels and caddisflies (e.g., *Isonychia, Elliptio, Brachycentrus*) to facultative types (e.g., Hydropsychidae, *Rheotanytarsus*, Sphaeriidae, *Musculium, Pisidium*); grazers of diatoms shift from sensitive mayflies and caddisflies (e.g., *Heptagenia, Leucrocuta*, Glossosomatidae) to facultative scrapers and collector gatherer organisms (e.g., *Baetis*, *Callibaetis*, Physidae, Leptoceridae). The suspended organic matter load somewhat exceeds what can be utilized by resident filterers resulting in increased levels of exported material. Sloughing of excess macro-algae and macrophyte biomass results in increased downstream export of course particulate organic matter. ➔ The system is becoming more autotrophic due to algal photosynthesis. The P/R ratio shows a slight increase. **IX *Spatial and temporal extent of detrimental effects*** ➔ Increased macrophyte and algal biomass extends downstream beyond the confluence with the next tributary; filamentous algae first appears in the stream as temperatures warm in late spring; pools and depositional areas are silt-filled; the interstitial spaces in the substrate of runs is becoming obstructed by sand and silt ➔ Early morning low dissolved oxygen levels occur occasionally during late spring and fall as well as during the mid summer **X *Ecosystem connectance*** ➔ Filling of interstitial spaces obstructs access to hyporheic zone for early instar stonefly nymphs, eliminating nursery areas and *refugia* for storm-events and low flows. Adult stoneflies from upstream reaches continue to oviposit but reproductive success is limited; stonefly nymphs continue to colonize by drift, with limited success. ➔ Poorly managed culverts on some tributaries impede fish passage and access to some spawning areas.

5	
Major changes in *structure* of the biotic community and moderate changes in *ecosystem function* *Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from that expected; organism condition shows signs of physiological stress; system function shows reduced complexity and redundancy; increased build-up or export of unused materials*	**I *Historically documented, sensitive, long-lived, or regionally endemic taxa*** ➔ Mussel fauna, including commonly occurring, generalist taxa (e.g., Eastern lampmussel- *Lampsilis radiata radiata;* Eastern floater- *Pyganodon cataracta;* Eastern elliptio- *Elliptio complanata*) is markedly diminished due to poor water quality **II *Sensitive- rare taxa*** ➔ Only the rare occurrence of individual representatives of specialist and vulnerable taxa with no evidence of successful reproduction **III *Sensitive- ubiquitous taxa*** ➔ Either absent or present in very low numbers, indicating impaired recruitment and/or reproduction **IV *Opportunist or facultative taxa of intermediate tolerance*** ➔ Filter-feeding invertebrates such as Hydropsychid caddisflies (e.g., *Cheumatopsyche*) and filter-feeding midges (e.g., *Rheotanytarsus, Microtendipes*) occur in very high numbers **V *Tolerant taxa*** ➔ Frequent occurrence of tolerant collector-gatherers (e.g., Orthocladiini, *Micropsectra, Pseudochironomus, Dicrotendipes,* Isopoda- *Caecidotea;* Amphipoda- *Hyalella, Gammarus*); ➔ Relative abundance of non-insects often equal to or higher than relative abundance of insects ➔ Deposit-feeders such as Oligochaeta are increased ➔ Numbers of tolerant predators are increased (Hirudinea, *Thienemannimyia, Cryptochironomus*) ➔ Native fish species are essentially absent with the exception of tolerant taxa like White sucker, Blacknose dace and Creek chub **VI *Non-native or intentionally introduced taxa*** ➔ Golden shiner, Smallmouth bass, and Yellow perch are common **VII** Physiological condition of long-lived organisms ➔ Biomass of young of year age classes is low; overall fish biomass is reduced; ➔ Sex ratio of remaining fish does not equal 1 ➔ Occurrence of parasitic infestations and disease is common ➔ Incidence of serious anomalies such as tumors and anatomical deformities is higher than expected **VIII *Ecosystem Function*** ➔ High algal photosynthetic activity results in daytime dissolved oxygen supersaturation accompanied by nighttime dissolved oxygen levels less than 4 ppm. Extremely high algal biomass significantly alters the habitat structure of the substrate; ➔ The P/R ratio is significantly > 1; the system is predominantly autotrophic ➔ Loss of coarse particulate shredders and alteration of bacterial decomposer community contributes to build-up and/or export of unused organic matter; ➔ Mechanisms for nutrient spiraling are significantly simplified and less efficient resulting in increased export of nutrients from the system **IX Spatial and temporal extent of detrimental effects** ➔ Substrate has become armored by increased sediment loading, altered flow regime and altered channel morphology resulting in compaction of interstitial space habitat, leaving only patches of well-scoured gravel substrate in high-gradient riffle areas; ➔ Armoring is resistant to spring scouring events, preventing annual spring sediment flushing and re-sorting of substrate; ➔ Near complete canopy removal results in all day insolation of stream and surrounding land surface causing abnormally elevated temperature regime in early spring and late fall. This causes unnaturally elevated seasonal temperature cues and results in failures of *life history requirements.* **X *Ecosystem connectance*** ➔ Lateral connectance to floodplain areas is eliminated except at peak flows, due to altered channel morphology caused by human intervention (bank riprapping, dikes) and altered flow regime. ➔ All appropriate high quality spawning gravel in upstream areas is destroyed by silt deposition, preventing spawning of white suckers, leaving only mature adults. Culverting is common, contributing to impairment of fish passage ➔ Lack of riparian vegetation eliminates habitat for adult flying aquatic insects, reducing survival and reproduction of resident organisms and reducing successful recruitment of immigrating organisms (i.e., flight dispersal of ovipositing females).

6

Severe changes in structure of the biotic community and major loss of ecosystem function

Extreme changes in structure; wholesale changes in taxonomic composition; extreme alterations from normal densities and distributions; organism condition is often poor; ecosystem functions are severely altered

I *Historically documented, sensitive, long-lived, regionally endemic taxa*
- ➔ Poor water quality, compaction of substrate, elevated temperature regime and absence of fish hosts for reproductive functions preclude the survival of any mussel fauna

II *Sensitive- rare taxa*
- ➔ These taxa are absent due to poor water quality, elevated temperature regime, alteration of habitat, loss of riparian zone, etc.

III *Sensitive- ubiquitous taxa*
- ➔ Absent due to above listed factors, though an occasional transient individual, usually in poor condition, may be collected.

IV *Taxa of intermediate tolerance*
- ➔ Filter-feeding insects and other macroinvertebrate representatives of this group are severely reduced in density and richness, or are absent.

V *Tolerant taxa*
- ➔ Low dissolved oxygen conditions preclude survival of most insect taxa except those with special adaptations to deficient oxygen conditions (e.g., *Chironomu*s)
- ➔ The macroinvertebrate assemblage is dominated by tolerant non-insects (Planariidae, Oligochaeta, Hirudinea, Sphaeriidae, etc.)

VI *Non-native or intentionally introduced* taxa
- ➔ Native species are essentially absent
- ➔ Only very tolerant invasive alien fish taxa are collected (Golden shiner, Yellow perch);
- ➔ Number of individuals collected is abnormally low

VII *Physiological condition of long-lived organisms*
- ➔ Fish biomass is very low; individuals that are collected appear to be transients and are in poor condition
- ➔ Incidence of parasitic infestations and disease is high; anatomical deformities and/or tumors are common
- ➔ Minimal evidence of recruitment or reproduction except some extremely tolerant groups may have high production; young of year age classes are absent

VIII *Ecosystem Function*
- ➔ Water quality has degraded to such an extent that algal photosynthesis is negligible
- ➔ Decomposition of organic matter creates P/R markedly <1; the system is predominantly heterotrophic as a result of high bacterial respiration and minimal photosynthesis
- ➔ Reproductive success is very low
- ➔ Recruitment of emigrating organisms into upstream and downstream habitats is impaired due to low fecundity and high mortality rates of resident biota.

IX *Spatial and temporal extent of detrimental effects*
- ➔ The *reach* and all tributaries are affected by widespread alteration of within stream conditions as a result of severely altered land-use and poor water quality.

X *Ecosystem connectance*
- ➔ *Watershed*-wide land use changes and alteration of stream morphology has affected all tributaries eliminating sources of recruitment and destroying spawning habitat;
- ➔ Physical and chemical requirements to fulfill *life history functions* (e.g., seasonal temperature cues for mating behavior and egg development; intact nursery habitats; optimal levels of dissolved gases, etc.) are severely disrupted resulting in very low reproductive success and high mortality rates.

CHAPTER 3. HOW DO YOU DEVELOP AND CALIBRATE A BIOLOGICAL CONDITION GRADIENT?

Figure 3-1 shows the overall approach for calibrating the Biological Condition Gradient, BCG, for a specific region. This chapter discusses the technical elements and steps for calibrating a regional BCG. The calibration process includes:

- Identification of defensible biological goals (also see Chapters 1 and 5)
- Development of the conceptual foundation of the regional BCG (Section 3.1)
- Assessment and modification, if necessary, of the State's biological monitoring program to support quantitative calibration of a regional BCG (Section 3.2)
- Calibration of a regional quantitative BCG model for operational assessment (Section 3.3)

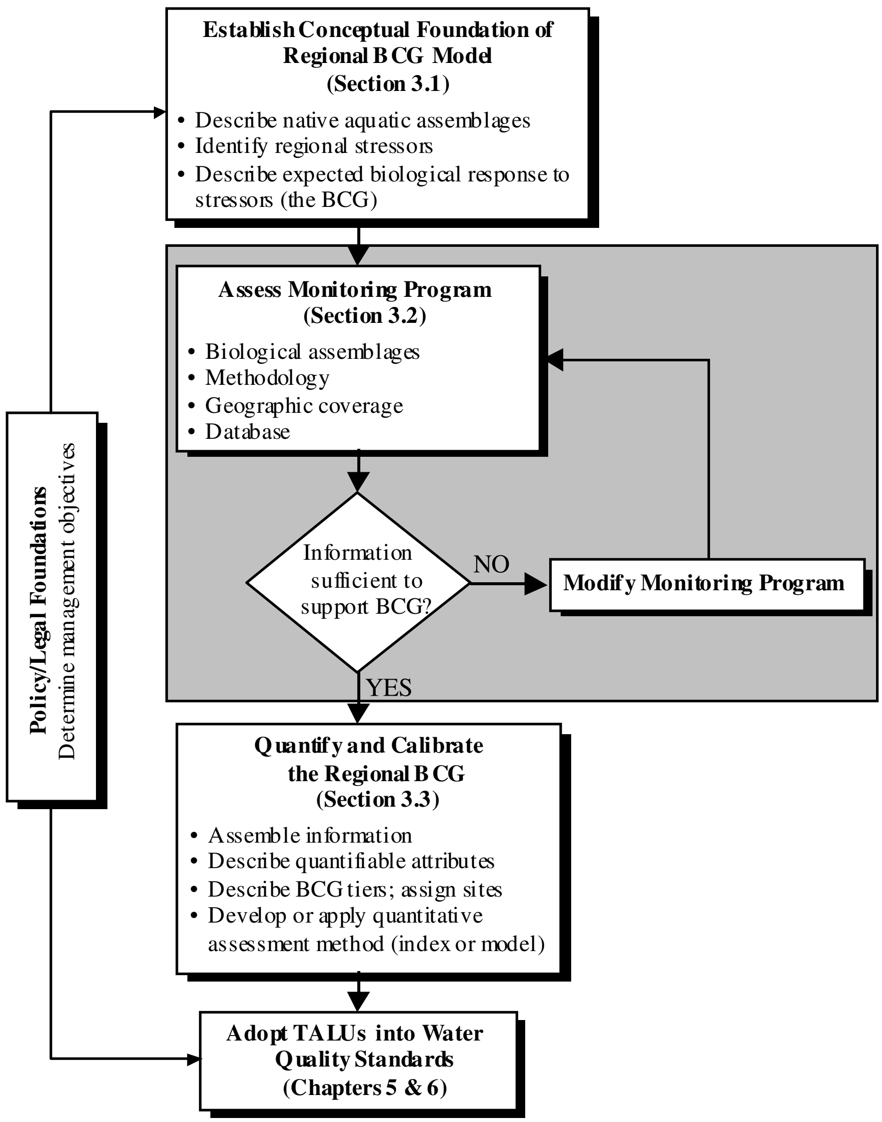


FIGURE 3-1. Technical components of the Biological Condition Gradient.

A State's water management program can support development of tiered aquatic life uses if it is flexible with respect to improvements in scientific knowledge and acknowledges that scientific advances may support adjustment of biological goals. State and Tribal designated uses form the aquatic life goals and water quality criteria (biological, chemical and physical) to protect the uses provide the basis for measuring attainment of the goals.

3.1 Conceptual foundation of a regional BCG model

The first technical component of calibrating a regional BCG is to adapt the national BCG model to regional conditions. Model development includes three components that, together, provide a complete ecological description of biological response to stressors that is consistent with ecological theory and empirical observation:

- Describe the native aquatic assemblages under natural, undisturbed conditions
- Identify the predominate regional stressors
- Describe the BCG, including the theoretical and empirically observed foundation of assemblage response to stressors

Similar to the national BCG model development process, regional BCG calibration can take place through technical panels and workshops that bring together aquatic biologists and ecologists knowledgeable about the waterbodies and assemblages in their regions. The technical experts describe native aquatic assemblages, regional stressors, and patterns of biological alteration based on both empirical observations and theoretical foundation to develop a regional biological condition gradient. The technical experts can include scientists from State and federal water quality agencies and natural resource departments, interstate river commissions, universities, and the private sector.

Expert participants in the regional model and calibration exercise should be knowledgeable about the assemblages sampled in the applicable monitoring programs (invertebrate biologists, ichthyologists, algologists, endangered species experts, etc.). The group should also include scientists involved in monitoring programs who are familiar with the sites and the organisms, plus other State, federal, university, and private sector biologists with relevant expertise. In some cases, BCGs have been initially drafted by a single experienced and knowledgeable individual, followed by a consensus process to confirm and modify the model.

3.1.1 Describe native aquatic assemblages

The BCG is grounded in natural biological assemblages that are present in ecosystems with no or minimal disturbance. Developing the BCG entails specific descriptions of the natural aquatic assemblages. The description of natural conditions requires biological knowledge of the region, classification of the natural assemblages, and, if available, historical descriptions of the habitats and assemblages.

Existing information – Information on biota in undisturbed or minimally disturbed habitats is required to develop a regional BCG model. If the State has an extensive monitoring program with undisturbed reference sites, its existing monitoring data will play an important role in developing the descriptions of reference biota. In addition to monitoring data, participants should also consult general references on biota of the region, especially references showing the historical and present-day geographic distribution of flora and fauna. These references often exist for fish and vascular plants, or may be unpublished reports and lists for threatened invertebrates such as mussels, snails, and dragonflies. However, such references are often unavailable for benthic macroinvertebrates or algae.

Classification – Developing a description of the BCG requires that biologists take into account the natural variability in assemblage structure and composition among sites and explain that variability where possible. This requires a classification system or model to predict the natural variation among sites (e.g., Wright et al. 1984, Barbour et al. 1999, Bailey et al. 2004). In this document, the term "classification" refers to identifying consistent differences between biological assemblages from undisturbed or minimally disturbed aquatic systems, if information available, and explaining those differences in terms of natural environmental gradients. Such natural gradients are encompassed within the regional descriptions of the undisturbed or minimally disturbed condition of the stressor gradient (Chapter 4).

Distributions of the organisms that make up aquatic communities are controlled by the effects of temperature, water velocity, light, oxygen, water quantity, dissolved substances (e.g., DOC, alkalinity, pH), food resources, cover, reproductive habitat, variability of physical and chemical factors, competitors, and predators. These physical and chemical factors vary geographically enabling biologists to characterize several community types by geographic location, such as cold water/warm water fish communities and low gradient/high gradient invertebrate communities. Scientists have also recognized geographic boundaries characterized by geology or vegetation (ecoregions: Omernik 1987; fish communities: Hughes and Larsen 1988; macroinvertebrate communities: Gerritsen et al. 2000). Some variables, notably measures of stream size (e.g., order, catchment area, length, total flow), have a more continuous effect on biological variables (e.g., increase of fish species richness with stream size; Karr et al. 1986).

Reference condition – Closely connected with classification of undisturbed or minimally disturbed systems and communities is the definition and measurement of reference condition. Methods for establishing reference condition need to be consistent for differing waterbody conditions to be compared (Hughes 1985, 1994; Hughes et al. 1986; Moss et al. 1987; Bailey et al. 2004; Stoddard et al. in press). Undisturbed or minimally disturbed conditions are comparable to "natural conditions," e.g. BCG tiers 1 and 2. Therefore, defining "natural" reference conditions is the starting point for development of a regional BCG. Ideally, empirical data assembled from reference sites with no or minimal levels of stressors characterize Tiers 1 and 2 of the BCG. This is because Tier 1 biological condition is, by definition, an assemblage structure, function, and taxonomic composition that is "naturally derived" from a physical environment not effected by stressors (Angermeier and Karr 1994).

Minimally disturbed sites (as defined by physical, chemical, and landscape measures) can be slightly altered from undisturbed condition, but should retain most characteristics of the resident biota in undisturbed sites. In many regions of the country where Tier 1 and Tier 2 sites may no longer exist, the reference sites used by agencies are considered "least disturbed." These sites have also been termed as the "best available," or "best existing," in the region but may be substantially altered from pristine, natural conditions. In extensively altered regions where undisturbed or minimally disturbed sites are absent, the best means to accurately characterize Tiers 1 or 2 may be through historical records of the taxonomic distributions of different assemblages and descriptions of the physical setting of undisturbed conditions (see below).

Historical descriptions – Historical descriptions help reconstruct undisturbed aquatic habitats and may help identify present-day sites that approximate historical conditions. This information is especially critical in areas where the best existing sites are significantly altered. Sources of historical information include early photographs and taxonomic collections, pre-dam and pre-irrigation physical data (USGS flow data, BLM data), and the descriptions of pioneers, naturalists, and scientists. Recent compilations and summaries of historical information have been developed where local or conservation interest is strong (e.g., Kuzelka et al. 1993, Johnson 1994). *See Case Example 3-1 on considering historical stream characteristics to estimate minimally disturbed conditions and support reference stream selections in Kansas*.

If no undisturbed or minimally disturbed reference sites exist in a region, the stressor gradient provides a means for determining the best regional candidates to act as benchmarks for comparison, i.e., “least disturbed” or “best available conditions.” Chapter 4 discusses the stressor gradient and a framework to organize stressor information derived from measures of the physical, chemical, and landscape variables of a sampled site. Applying monitoring information that is organized into the stressor gradient framework will help managers evaluate the status of their waters relative to change, or departure, from reference condition.

3.1.2 Identify regional stressors

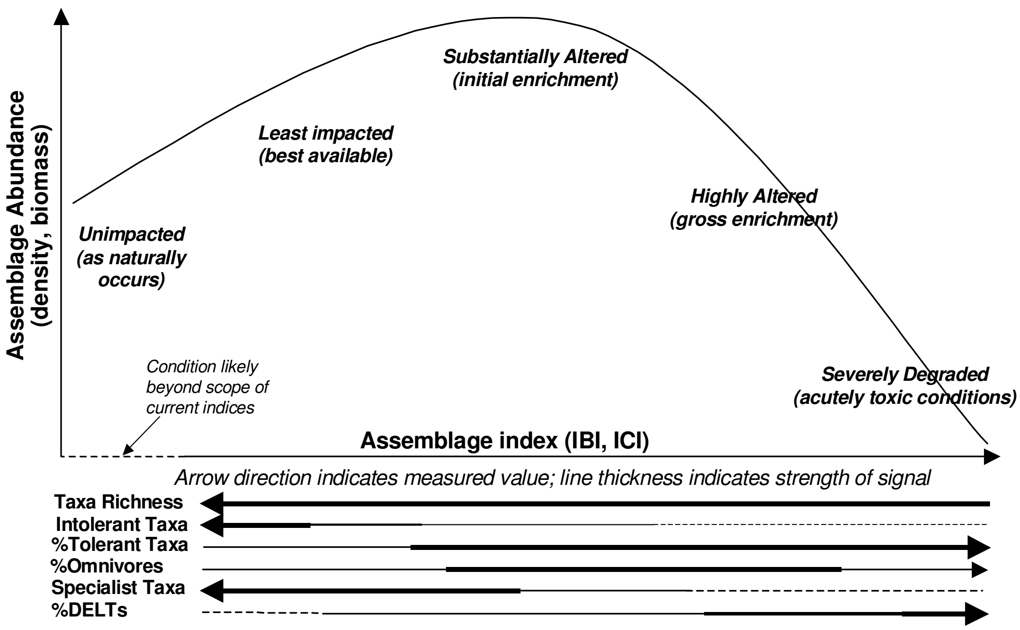
A description of regionally dominant stressors will help define expectations for biological responses that are likely to occur. This step considers sources of physical and chemical stressors and causes of landscape or habitat disturbance (the stressor gradient; Chapter 4). For example, if an ecoregion is primarily mountainous, then stressors from extensive row-crop agriculture will be relatively less frequent than stressors from other sources. Other examples of regionally important stressors include hydrologic alteration from urbanization; effluent-created permanent streams in the arid west; and acid mine drainage and related metals contamination in coal mining regions of the Appalachians and metal mining regions of the Rocky Mountains.

Identification of stressors and their sources is the first step in characterizing the stressor gradient (Chapter 4). The stressor gradient is the combination of causal factors that induce an adverse response in the aquatic biota. A conceptual model of fish and macroinvertebrate assemblage response to a regional stressor gradient ranging from undisturbed or minimally disturbed conditions to severely altered conditions was developed based on empirical observations of assemblage responses to multiple sources in Ohio (Figure 3-2). The graphic represents measured assemblage abundance (y-axis) against an assemblage index (fish IBI, macroinvertebrate ICI; x-axis) with the generalized response of selected metrics. Biological descriptions correspond to the six tiers of the BCG model and include descriptions of assemblage characteristics, chemical water quality conditions, physical habitat and flow regime, and sources of stress that are typically associated with each. This was modified from an original conceptual model by Ohio EPA (1987) and Yoder and Rankin (1995b). It demonstrates that understanding the relationship between assemblage responses and stressors is a fundamental aspect of bioassessments.

3.1.3 Describe the Biological Condition Gradient

In testing the national BCG model, regional experts calibrated it to specific regional sites and assemblages. Biologists familiar with the regions’ natural aquatic communities and their responses to stress worked collaboratively to calibrate the BCG model to conditions in the following regions: Maine, Kentucky, the Central Great Plains, and selected areas in the arid west (Arizona and eastern Washington). Table 2-3 shows the resulting model for Maine.

The equivalent step in developing a regional BCG model is to develop a local counterpart to the national BCG model. The objective is to ground the BCG in local conditions. The regionally calibrated BCG describes the undisturbed or minimally disturbed aquatic ecosystems of the region, and the responses of the biota to the predominate regional stressor gradient. To the extent possible, the regional model should describe undisturbed or minimally disturbed conditions.



Biological and Stressor Gradient Descriptors

"As Naturally Occurs" *(Pristine)*	"Least Impacted" *(Exceptional)*	"Initial Enrichment" *(Good)*	"Moderate Enrichment" *(Fair)*	"Gross Enrichment" *(Poor)*	"Severely Altered" *(Very Poor)*
Assemblage Characteristics					
Native assemblages; no symptoms of stress	"Best of what's left" assemblages; high richness; intolerants, specialists predominate	"Typical" assemblages; good richness; emerging symptoms of stress in selected metrics	"Impaired" assemblages; tolerants & generalists predominate numbers/bio-mass; loss of intolerants	"Degraded"; highly tolerant taxa pre-dominate; reduced abundance; anomalies increasing	"Severely degraded"; very low numbers; few taxa; very high % anomalies; toxic signatures
Chemical Water Quality Conditions					
As natural; no human-made compounds present	"Best reference" quality; toxics < detection; high D.O., low nutrients	"Background reference" quality; toxics < chronic; adequate D.O., nutrients = reference	"Enriched" quality; toxics < chronic; marginal D.O. regime, nutrients > reference	"Degraded" quality; toxics > chronic; low D.O., nutrients >> reference	"Extremely poor" quality; toxics ≥ acute; very low D.O., nutrients >> reference; contaminated sediments
Physical Habitat & Flow Regime					
Natural habitat and flow regime; no human-made modifications	Excellent quality habitat & flow regime; recovered from human-made modifications	Good quality habitat & flow regime; *de minimis* human modifications	Fair quality habitat & flow regime; active human modifi-cations; incomplete recovery	Poor quality habitat & flow regime; active human modifi-cations; no recovery	Severe modifi-cations; ephemeral flows; active human modifications; no recovery potential
Examples of Sources and Activities					
No effects of human activity are evident	Point sources present, do not dominate flows; NPS impacts buffered by extensive riparian system	Point sources may dominate flows; NPS impacts buffered by good riparian zones	PS/NPS enrich-ment impacts; NPS unbufferd; channel modifi-cations; im-poundments	Gross PS/NPS enrichment impacts inc. CSOs; NPS unbufferd; chan-nel modifications; urbanization	Severe PS/NPS toxic impacts; extreme channel modifications; urbanization; acid mine drainage, severe thermal

FIGURE 3-2. Conceptual model of the response of fish and macroinvertebrate assemblages to a gradient of impacts in warmwater rivers and streams throughout Ohio (modified from Ohio EPA 1987 and Yoder and Rankin 1995b).

The BCG model may require some example data from sites to empirically ground-truth conclusions. An example regional BCG was described in Chapter 2, the Maine scenario for cold-water, high gradient streams (Table 2-3). Ohio also developed a conceptual model of the BCG, shown in Figure 3-2, as part of its tiered aquatic life use development. In addition to the description of undisturbed, natural assemblages and the predominate stressor gradient in a region, the regional model also requires a narrative description of the tiers and their biological attributes.

***A narrative description of the tiers of the BCG for the region* –** The regional model includes description of individual tiers along the gradient of biological response to stressors, including organisms present and organisms absent. The descriptions of changes in the attributes corresponding to the different tiers are derived from the consensus among technical experts as well as agreement on the number of tiers that can be discriminated across the entire gradient. The regional narrative descriptions refine the national model's descriptions of changes across the stressor gradient to reflect local conditions. (e.g., see Maine example, Table 2-3 and Ohio example, Figure 3-2). The description of the Ohio BCG is in the row titled "Assemblage Characteristics" (Figure 3-2). In Ohio, enrichment occurs at intermediate disturbance levels for the metrics (numbers or biomass).

The descriptions should account for the natural classification that applies to the region. As noted in Section 3.1.1, "classification" is defined as the process of stratifying according to natural gradients. It may be necessary to develop separate narrative descriptions for major classes of natural gradients if the biological expectations differ widely among classes. For example, the biota of low-gradient streams with fine, sandy substrates may be dominated by invertebrates adapted to those conditions, such as midges and worms. These same organisms are often indicators of degraded conditions in fast-flowing streams with coarse substrate, but may be expected to occur under the best conditions in naturally silty streams.

***A narrative description of the ecological attributes that are used to determine the tiers* –** Ecological attributes are measurable characteristics of the system (described in Chapter 2). For bioassessment programs that sample biota of target assemblages, the critical attributes are those most closely related to taxonomic information contained in the sampled assemblages. Many species can be assigned to an attribute group, and the change in the attributes is described in the conceptual model. In the Ohio example (Figure 3-2), attributes include intolerants, generalists, specialists, etc. listed in the descriptions in the first row (Assemblage Characteristics).

3.2 Data needs: Assess and modify technical program

Consistent, quality assured and controlled (qa/qc) monitoring information is key to developing a quantitative assessment system within a BCG framework. Key elements of a biological monitoring programs are listed below, correspond to design and data collection elements outlined in *Technical Guidelines: Technical Elements of a Bioassessment Program* (see Appendix C) (Barbour and Yoder unpublished manuscript). Elements of a monitoring program for quantitative calibration of the BCG are discussed below.

3.2.1 Biological assemblages

Development of a quantitative BCG can include one or more biological assemblages (e.g., benthic macroinvertebrates, fish, periphyton, phytoplankton). Choice of each of these assemblages, and field sampling methods, are discussed in *Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish* (EPA/841-B-99-002; Barbour et al. 1999).

3.2.2 Consistent methodology

Consistent and demonstrated methodology is important for calibration of a regional BCG. Methodological consistency includes sampling methods that obtain representative samples of relevant biota in the assessment unit, choice and use of sampling equipment, index period, definition of sampling site (e.g., stream reach), and allocation of sampling and subsampling effort to obtain representative estimates of composition and structure. Field sampling considerations are discussed in *Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish* (EPA/841-B-99-002; Barbour et al. 1999), and statistical considerations are discussed in *Statistical Guidance for Developing Indicators for Rivers and Streams* (Appendix E).

3.2.3 Geographic coverage

The monitoring program should have sufficient spatial and temporal coverage to provide adequate quantitative information to describe biological community expected undisturbed/minimally disturbed conditions (Section 3.1.1). This would include major geographic regions, waterbody types, and environmental gradients of pressure and stressors.

Natural Classifications –There should be sufficient reference site data in the State's database to classify natural conditions and account for natural spatial variability among sites. Classification was discussed in Section 3.1.1.

Stressor gradient – To describe the BCG, examples are used for each of the tiers that occur in the state or region. Hence, data must span the entire condition gradient from the least disturbed to the most disturbed sites in a particular region, along the entire stressor gradient.

Geographic information – In addition to routine monitoring data, geographic information helps to develop natural classification of waterbodies to refine the expected condition. As noted above, one of the requirements for developing a description of the BCG is to have a natural classification of the resource, which provides a framework for organizing and interpreting natural variability among sites. Useful geographic information includes:

- Watershed delineations – catchments of the specific sampling sites
- Physical characteristics of sampling site catchments (catchment area, distance to source, mean slope, etc.)

In addition to natural characteristics, geographic information should include information for characterizing the stressor gradient, the x-axis of the BCG and evaluating whether there are undisturbed or least disturbed reference sites. This would include information on discharges, non-point sources of pollutants, and watershed and landscape characteristics.

Reference condition – The no or low stressor end of the stressor gradient, whether undisturbed or least disturbed condition, should be well represented as reference sites and reference condition in the database. Considerations for establishing reference condition were discussed in Section 3.1.1.

3.2.4 Database

A comprehensive and complete database is critical to BCG calibration. The database should include all information collected in the monitoring program, as well as stressor and pressure information that may be collected on a geographic basis. The data must be organized and made accessible so that expert participants can easily view and interpret the data.

3.2.5 Modify monitoring program

If the specific data and information from a State monitoring program are not sufficient to support a quantitative BCG calibration, then the State may need to strengthen its technical program. Monitoring and sampling program design are not covered here. See *Technical Guidelines: Technical Elements of a Bioassessment Program* and *Statistical Guidance for Developing Indicators for Rivers and Streams* (Appendixes C and E).

3.3 Calibrate a regional BCG model

The final step in developing an assessment method using the BCG framework is to quantify and calibrate a model or system for routine assessment of waterbodies. In this step, the conceptual model that was adjusted for regional conditions is further refined and validated with data and, where possible, with quantitative relationships. The same expert panel that developed the regional conceptual model is best suited to calibrate the BCG model with quantitative information.

Regional BCG models have been calibrated for routine use in bioassessment and biocriteria programs. These calibrations can be used independently as stand-alone assessment methods, or in conjunction with existing biotic indexes. The earliest operational development took place in Maine and Ohio (Ohio EPA 1987, Courtemanch et al. 1989, Davies et al. 1995, Yoder and Rankin 1995a, Davies et al. 1999) and was the basis for the development of the national conceptual model. Regional calibration extends beyond application of the conceptual model and requires consistent operational rules so that sites can be assigned to tiers in a consistent fashion.

The following sections outline the process of regionally calibrating and developing a BCG model.

3.3.1 Assemble information

The information required to complete these tasks includes the database of consistently collected biological monitoring data from a subset of sites throughout the region and geographic and historical information where available (Section 3.2). If the State or agency has a very large data set from a long-standing monitoring program, then it is not practical to make all of the data available to the regional BCG workshop participants. Instead, select a subset of sites that represent the entire stressor gradient, from the minimally or least disturbed to the most stressed sites in the state. The objective of the rating exercise is to select a variety of representative sites across the gradient so that all tiers occurring in the region are represented in the calibration sample of sites. Some reference sites should be included in this set as well as intermediate and severely stressed sites. The data must be organized and made accessible so that expert participants can easily view and interpret the data. The following information should be available:

- A comprehensive species list for each assemblage that is monitored (e.g., macroinvertebrates, periphyton, fish), which can be sorted by higher taxonomic categories (order and family). To the extent known, tolerance values (to various stressors), trophic status (functional feeding group), habit, breeding guild, etc. should be included in each taxa list.
- Counts of abundance, by taxon, for each sample. If necessary, the database program can adjust for unequal effort among samples.
- Complete habitat data
- Field notes
- Complete field physical and chemistry data (e.g., streamflow, pH, conductivity, temperature, velocity, etc.)
- Complete laboratory chemistry results
- Landscape and hydrologic alteration of the catchments of the sampling sites, if available; otherwise land use of the smallest hydrologic accounting unit that contains the sampled catchments
- Site identification (name, ID, location)

Sites from a comprehensive monitoring program should span the range of water and habitat quality found in the state, from the best to the worst. At this point, the data will have passed QA checks and will meet the requirements for developing a BCG, outlined briefly in Section 3.2 and in Appendix C, and in greater detail in *Technical Guidelines: Technical Elements of a Bioassessment Program* (Barbour and Yoder unpublished manuscript).

Rather than expecting the expert group to work with stacks of printed data, it is useful to develop a spreadsheet that can be manipulated by participants or projected onto a screen for use during group discussions. The spreadsheet displays data from a single site at a time and calculates taxa and abundances of attribute groups. One person should be assigned responsibility for assembling all relevant data for the workshop exercise. If the State data are not well organized (i.e., not housed in a single comprehensive database), then assembling the data may require substantial time and effort.

Classification – In this stage, it may be necessary to develop, refine, or empirically test classification schemes proposed in conceptual model development (Section 3.1) if the State does not have a fully tested classification scheme for aquatic assemblages in natural waterbodies. The purpose of classification for this document was also explained in Section 3.1. Classification is influenced by the components of a monitoring design: methods, measured variables, sample size (number of sites), etc. There are several quantitative approaches to developing a classification system, including categorical models, continuous models, *a priori* methods (use of existing models), and *a posteriori* methods (empirical models using data in hand). Many references are available to help analysts develop biological classifications of waterbodies (bioassessment case studies and methods: Barbour et al. 1999, Wright 2000, Gerritsen et al. 2000, Hawkins et al. 2000, Hawkins and Vinson 2000, Smith et al. 2001, Bailey et al. 2004; textbooks: Jongman et al. 1987, Ludwig and Reynolds 1988, Legendre and Legendre 1998, Davies et al. unpublished manuscript).

3.3.2 Describe attributes

Ecological attributes are measurable characteristics of the system described in Chapter 2. These are the measures used to determine a waterbody's position along the BCG. As described in Chapter 2, attributes that are derived from taxonomic composition or organism condition (Attributes I to VII) are routinely measured and interpreted in State and Tribal water programs. As a practical matter, these are the key attributes that need to be quantitatively characterized for routine assessment.

The technical expert panel should work through the list of taxa collected in the monitoring program and assign the taxa to Attributes I through VI. In this process, the specific definitions of the attributes may be adjusted to reflect local knowledge. For example, New Jersey biologists redefined Attribute II from "sensitive-rare" taxa to "highly sensitive" taxa because rarity was not considered to be related to sensitivity to pollution, and sampling methods do not capture rare taxa with any predictable reliability. *See Case Example 3-2 for further discussion of New Jersey's tier descriptions for high and low gradient streams.*

- Attribute I consists of rare and endemic taxa, which are not often encountered by routine biological sampling methods. Their presence may be known from larger-scale surveys designed to assess rare species.
- Attributes II through V are taxonomic groupings organized according to tolerance to pollution, where Attribute II taxa are the most sensitive and Attribute V taxa are the most tolerant. These four attributes are the quantitative workhorses for assessment on the BCG and must be thoroughly characterized to calibrate a regional BCG. The tolerances of these attributes can be initially assigned based on existing tolerance estimates, but the panel should consider whether the existing tolerance estimates are accurate based on their experience and observations of the organisms.
- Attribute VI consists of introduced taxa.

Due to incomplete information, rarity in the database, or lack of knowledge, not all taxa will be assigned to an attribute.

3.3.3 Describe tiers and assign sites to tiers

Similar to the national BCG model development process, regional development can occur in workshops that bring together aquatic biologists and water quality standards experts familiar with streams in their regions. Workshop participants are asked to develop both the ecological attributes and the rules for assigning sites to tiers along the gradient. Workshops proceed as follows:

1. Participants consider the conceptual model of the BCG to identify specific biological changes that can be observed along the stressor gradient in their region. Specific metrics or attributes that can be measured within the BCG framework are identified.
2. The groups consider data from selected monitoring sites and assign the sites to tiers in the BCG based on the biological monitoring information from each site. Initially there may be disagreement among the group members, but as they become familiar with the process, sites are rated more consistently.
3. From the discussions and decisions, a set of rules is developed for assigning sites to individual tiers in the BCG.

Using the regionally adapted conceptual model (Section 3.1.3), participants examine data from selected sites throughout the region. Sites are selected from the preliminary stressor gradient (See Chapter 4) to represent the gradient as it occurs in the region. The group should consider the biological condition, species present and absent, and come to consensus on the tier to which each site should be assigned. Experience has shown that assemblages are best kept separate at this stage. The group should describe the tiers and assign sites to tiers separately for macroinvertebrates, fish, periphyton, and other assemblages.

Groups typically start with several candidate reference sites in the region in an effort to establish a reference baseline. Depending on the completeness of the database, the best sites in that database may not reflect undisturbed or minimally disturbed conditions. Additionally, if the ecoregion spans more than one state, the best sites might be in a different state or tribal land—and may not be part of the database. Ideally, calibration of the BCG in physiographic or ecological regions that cross state boundaries should be multi-state and tribal efforts. The important point here is that the best sites are *not* automatically assigned to Tiers 1 or 2. The assemblages from the candidate reference sites should be compared to the descriptions of Tier 1 and Tier 2 sites developed in the initial theoretical exercise. The following questions should be addressed:

- Do the candidate reference sites meet the theoretical expectations of Tier 1 or Tier 2? Then, if the answer is no, first validate the model's Tier 1 and 2 expectations by addressing the following questions:
    - Are these candidate reference sites minimally disturbed, that is, are there no or negligible effects from stressors?
    - Can the level of stressors be documented?
    - Is historical information available that would suggest that they are minimally disturbed?
- If these three questions are answered "yes" then the theoretical expectations and descriptions of Tiers 1 and 2 may need to be reassessed and altered. If the candidate reference sites apparently have more then than minimal or negligible levels of stressors, then they do represent examples of Tiers 1 and 2, undisturbed or minimally disturbed conditions. In many areas, sites identified as reference, especially those that are the "least disturbed," may be rated Tier 3 or even Tier 4 in the BCG.

Following development of the tier descriptions, participants continue to assign sites to tiers using the descriptions they have developed. Both the tier descriptions and the original taxa assignments may be revisited and revised in order to resolve any anomalies or issues that arise throughout the assignment process. Sites are frequently deemed intermediate (between adjacent tiers), and assigning sites to tiers does not require group unanimity. *See Case Example 3-3 on Maine's assignment of stream sites to waterbody classes (tiers) using benthic macroinvertebrate metrics*.

Tier assignments can also be tested against stressor gradients from the database. Stressor gradients (e.g., toxic metal concentrations, habitat conditions, nutrient concentrations, etc.) can be considered partial components of the stressor gradient (Chapter 4). Figure 3-3 shows an example from Ohio, showing copper concentration in the BCG tiers. In general, lower tier sites have a greater likelihood of elevated copper above the criterion level, although all tiers except the poorest (NA; very poor) included at least some sites with copper not exceeding the criterion.

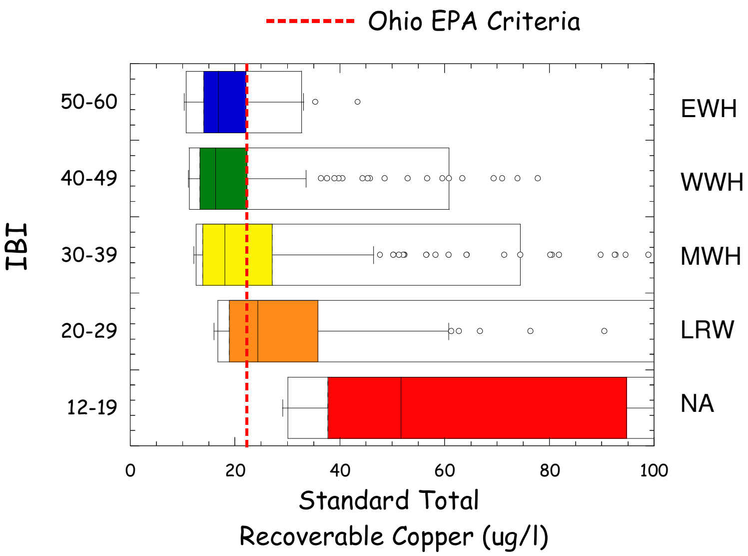


FIGURE 3-3. Ohio BCG tiers and copper concentration. Each horizontal bar approximates the tier shown on the right: EWH - exceptional warmwater habitat; WWH - warmwater habitat; MWH - modified warmwater habitat; LRW - limited resource waters; NA - non-attaining (very poor). Shaded areas are interquartile ranges of copper concentration in each BCG tier. Note that all sites in the very poor tier had copper concentration above the Ohio copper criterion (dashed line).

Setting expectations in significantly altered landscapes
In some regions, the historical conditions describing Tier 1 and 2 sites no longer exist. Many native species have been extirpated or greatly reduced, and the physical and chemical habitat of streams is completely different from the pristine, or undisturbed, condition. For example, the breaking up of native prairie sod and ongoing agricultural practices has resulted in high sediment and nutrient loads in midwestern prairie streams (e.g., Kuzelka et al. 1993). Removal of forest cover in eastern agricultural areas (e.g., Corn Belt Plains, Interior Plateau, Southeastern Plains, Riverine Lowlands) has had similar effects, although large tracts of forest cover remain or have regrown. In the western Great Plains, damming of snowmelt-fed streams and rivers has eliminated annual scouring flows and reduced sediment loads of rivers such as the Missouri, Platte, Arkansas, Rio Grande (e.g., Johnson 1994). Biological conditions comparable to Tiers 1 and 2 may no longer exist in some ecological regions of the continent. Mitigation of the resource to pristine conditions may not be currently possible *(See Case Example 3-1)*.

3.3.4 Develop quantitative assessment methods

To developing a regional BCG water quality agencies should consider ecological information critically in making assessments. Biological condition tiers are narrative statements on presence, absence, abundance, and relative abundance of several groups of taxa, as well as statements on system connectivity and ecosystem attributes (e.g., production, material cycling). The statements are consensus best professional judgments based on the years of experience of many biologists in a region, and reflect accumulated biological knowledge.

Consistent application of the BCG to routine assessment and ultimately to better define designated aquatic life uses in water quality standards, will require an operational system that does not depend on reconvening the same group of experts to rate all sites. Assessments should minimize individual variability or bias, as might occur if individual assessors then interpret the rules developed by expert consensus.

Accordingly, there are a variety of ways to automate the decision tool, ranging from application of existing biotic indexes (multimetric IBI type indexes, RIVPACS indexes, BEAST applications) to development of new expert systems that specifically replicate the decision-making of the expert group that defined the BCG for the region (Appendix A; Davies et al. unpublished manuscript). Below are discussions of three methods for developing an operational assessment system, two of which use existing indexes, and the third of which develops and calibrates a system specifically for identifying tiers of the BCG. Other methods are also possible (e.g., expert systems), but the three explained below are currently used for operational bioassessment into tiers of the BCG.

Any quantitative model or procedure that is developed to assign sites to tiers should be tested with independent data that were not used to calibrate the model. This applies to all three quantitative model approaches discussed here. In general, the models are calibrated using tier assignments developed by the expert panel (Section 3.3.3). A second data set of tier assignments (also assigned by the expert panel) is then required to test the model.

Calibrating biotic indexes to the BCG

Biotic indexes such as IBIs (multimetric approaches under a variety of acronyms; Barbour et al. 1999), predictive model indexes (RIVPACS approaches; Wright 2000), and true multivariate indexes (BEAST models; Bailey et al. 2004) are all attempts to describe a biological condition gradient. As such, index approaches may be suited to identifying tiers in the gradient and for assessment in the context of the BCG.

Simple division of an index scoring range is not recommended because most indexes were not explicitly developed on a BCG framework. For example, metrics in an IBI-type index may have been selected because of strong responsiveness to stressors, rather than reflecting the conditions expressed in the BCG (see Table 2-1). If a State is to develop tiered aquatic life uses based on the national BCG model, it therefore may be necessary to recalibrate existing index models to the BCG or develop new biological models and can be used to assign sites to tiers. For example, Vermont has designated aquatic life uses as differentiated by biological threshold criteria *(See Case Example 3-4).*

Through an iterative process, scoring criteria may be developed for existing indexes that correspond with biologists' consensus on narrative descriptions of the tiers in the biological gradient. If tiers are established based on other designated uses (e.g., hydrologically modified canals), then each tier or use class can be calibrated to an index score reflecting the best potential condition for that use. Ohio used this approach to set biological criteria for four use classes (see Chapter 5).

An existing index may be calibrated to the BCG model at the level of index scores, or by deriving a new index that better reflects the BCG. Both approaches require a set of sites that have been assigned to the tiers of the BCG that were determined by the expert panel to be appropriate for the specific aquatic ecosystem (Section 3.3.3).

Calibrating index scores – The set of sites that have been assigned to tiers of the BCG are used to calibrate index scores. Index scores for the sites are examined (Figure 3-4). If separation of the index scores among tiers is good, then index thresholds can be selected to maximize the ability to discriminate among the tiers. Figure 3-4 shows a hypothetical example with five tiers (BCG Tiers 2 - 6). Separation of scores among tiers is generally good, and the solid lines indicate scoring thresholds between adjacent tiers. The exception here is that the index does not discriminate as reliably between Tiers 2 and 3 as it does between other pairs of tiers.

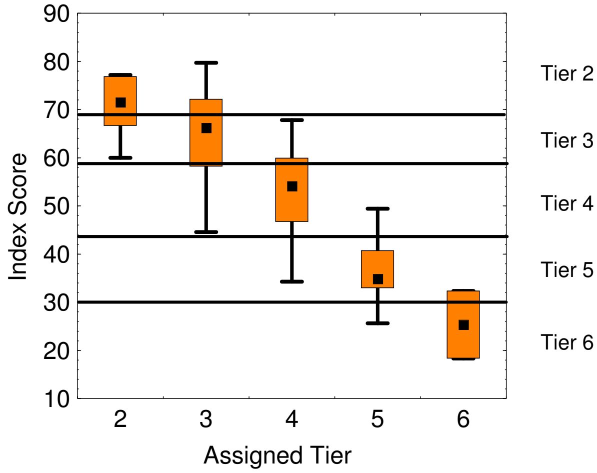


FIGURE 3-4. Hypothetical example of biotic index scores of sites assigned to BCG tiers, where the index is able to discriminate tiers most of the time. Boxes represent interquartile range of each tier; points are medians, and whiskers represent range of score outside the quartiles. Horizontal lines represent thresholds that could be applied to discriminate tiers using the index scores. In this example, there are no undisturbed reference conditions in the region.

The British Environment Agency recalibrated two RIVPACS indexes in a similar way. Initially, index scores were divided into four equal tier categories based on the statistical distribution of reference site scores (90% interval; Helmsley-Flint 2000). However, regional field biologists observed that four equal categories based on a 90% interval were insufficient to discriminate exceptional from good sites, and poor from very poor sites (Helmsley-Flint 2000). Accordingly, the indexes were recalibrated so that categories matched those determined by the regional experts. The resultant six categories are similar to the six tiers of the BCG (Table 3-4). *See Case Example 3-5 for a description of this process.*

Calibrating metrics – However, index scores may show a great deal of variation within BCG tiers, such that assigning tiers based on index scores is an inaccurate process (Figure 3-5). In the hypothetical example shown in Figure 3-5, the index is unable to discriminate among Tiers 2 through 4. In this instance, it would be necessary to revise the index to reflect tiers of the regionally calibrated BCG. Revision and recalibration of an IBI, or of other indexes, can be part of a State's routine recalibration process that occurs periodically when substantial new data have been collected.

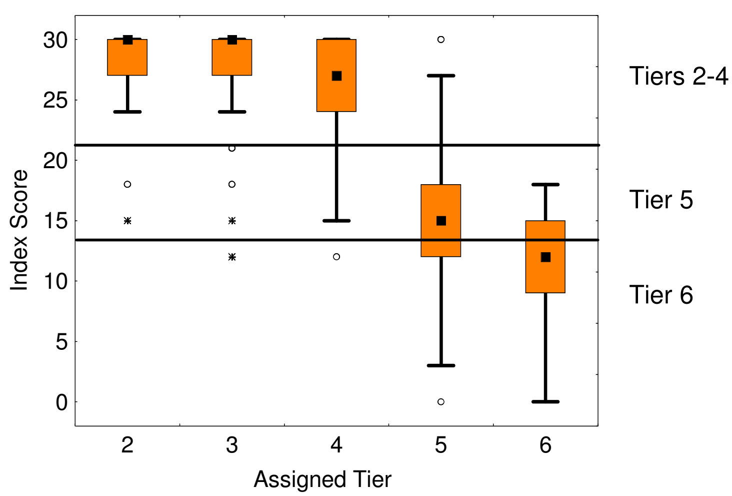


FIGURE 3-5. Hypothetical example of biotic index scores of sites assigned to BCG tiers, where the index is not able to discriminate tiers.

Model development to support BCG tiers: Discriminant model

Simple recalibration of index scores to BCG tiers may not yield distinct break-points (or benchmarks) between adjacent tiers. This is the case when sites in different tiers (as determined by the expert panel) have the same or similar index scores, showing that the index cannot discriminate among tiers of the BCG. Development of an operational tiered assessment system may require a separate index or model calibrated to the tiers.

Discriminant analysis may be used to develop a model that will divide, or discriminate, observations among two or more classes. A discriminant function model is a linear function combining the input variables. It obtains the maximum separation (discrimination) among the classes. The model is developed from a "learning" dataset where the classes have been identified. The model is then used to determine class membership of new observations where the class is unknown. Thus, a discriminant function model can be developed from a biological data set where sites have been assigned to BCG tiers. The analysis identifies variables that will discriminate among the tiers. The resultant model is then used to identify the tier to which a site should be assigned. Maine uses this method to determine whether streams are meeting biological criteria for multiple tiered uses. *See Case Example 3-6 on Maine's development of linear discriminant functions to assess tiers*.

Although it requires considerable statistical expertise to develop, the advantage of discriminant analysis is that it uses established and well-documented statistical methodology. However, it requires a relatively large set of assigned sites to calibrate the model, approximately 20 per tier. Accuracy of the model to the expert-assigned calibration and test sites can be as high as 89 - 97% (based on jack-knife tests; Davies et al. unpublished manuscript).

Using a discriminant model to develop biocriteria requires both a set of training data to develop the model and confirmation data to test the model. The training and confirmation data may be from the same biosurvey, randomly divided into two, or they may be two or more years of survey data. All sites in each data set are assigned to BCG tiers by the expert workgroup (Section 3.3.3).

One or more discriminant function models are developed from the training set to predict tier membership from biological data. Once developed, the model is applied to the confirmation data set to determine how well it can assign sites to classes using independent data not used to develop the model *(See Case Example 3-6)*. More information on discriminant analysis can be found in any textbook on multivariate statistics (e.g., Jongman et al. 1987, Ludwig and Reynolds 1988, Legendre and Legendre 1998).

Quantitative rules for tier assignments

Tier descriptions in the conceptual model tend to be rather general (e.g., "reduced richness"). To allow for consistent assignments of sites to tiers, it is necessary to operationalize, or codify, the general tier descriptions into a set of rules that anyone can follow and obtain the same tier assignments as the group of experts.

Operational rules are used to define the tier descriptions ("as naturally occur," "reduced," "greatly reduced," etc.) to quantitative or semi-quantitative rules for each attribute ("Attribute II taxa > 50% of any other attribute, ± 10%"). These rules preserve the collective professional judgment of the expert group and set the stage for the development of models that reliably assign sites to tiers without having to reconvene the same group. In essence, the rules and models capture the group's collective decision criteria.

Rule development can take place during the expert panel workshop to describe the detailed BCG and assign sites to tiers (Section 3.3.3). It requires discussion and documentation of tier assignment decisions and the reasoning behind the decisions. During this discussion, facilitators should elicit and record:

- each participant's tier decision ("vote") for the site;
- the critical or most important information for the decision – for example, the number of taxa of a certain attribute, the abundance of an attribute, the presence of indicator taxa, etc.; and
- any confounding or conflicting information and how this was resolved for the eventual decision.

See Case Example 3-7 for an example of decision rules developed during New Jersey's calibration exercise (Table 3-6).

Testing
Rule development should be iterative. Following the initial development phase, the draft rules should be tested by a group of experts to ensure that new data and new sites are assessed in the same way. This usually requires a second workshop, during which a set of test sites not used in the initial rule development and also spanning the range of stress should be assessed. Any remaining ambiguities and inconsistencies from the first iteration can also be resolved. Rules can be used directly for assessments, for calibrating one of the previous assessment methods (IBI, discriminant model), or as the basis of an expert system.

Thresholds and uncertainty
For each of the quantitative models described above, it is possible to estimate predictive uncertainties. Index variability is estimated from repeated measures at sites over one or more years, and accuracy of the quantitative model to expert consensus is estimated from the number of "correct" calls by the model. Several methods exist to estimate overall predictive uncertainty. For uncertainty of the models discussed here, see Helmsley-Flint (2000) and Davies et al. (unpublished manuscript).

Not all uncertainty is statistical, and not all issues of uncertainty can be reduced to a statistical probability. Experience with the BCG workgroups suggests that there will always be sites that fall on the border between tiers. It is important to recognize that some sites are borderline or intermediate, not that we are uncertain about where they are. This is a consequence of forcing a more-or-less continuous gradient into discrete management categories.

While thresholds between tiers do not need to reflect true discontinuities in nature, the tiers should represent detectable and consistent differences in assemblages, their taxonomic composition, and ecological function. To the extent they are consistent and detectable, they serve to inform management on how well we are protecting against degradation and making progress towards restoration goals.

Disagreement among assemblages
Once a BCG has been regionally calibrated, a possible scenario in assessment is that two assemblages collected at the same site indicate different tiers of the BCG. For example, to what tier should a site be assigned if the fish indicate Tier 2 but the macroinvertebrates indicate Tier 4? Options include:

- averaging the two assemblages (Tier 3 in this example),
- selecting the lowest assessment among the assemblages (Tier 4), or
- selecting the highest assessment among the assemblages (Tier 2).

In making this decision, it is important to consider the level of rigor in the tier assessments among the assemblages, particularly if an assessment is based on an absence, rather than presence, of information (absence of evidence is not evidence of absence). This requires considering the strength of evidence for each assemblage. Automatic calculation of an average or use of the highest assessment is neither conservative nor protective of the resource. Both Ohio EPA and the British Environment Agency have chosen to select the lowest assessment among indexes and assemblages for final tier assignments (Yoder and Rankin 1995b, Helmsley-Flint 2000).

3.4 Key points from Chapter 3

1. **The conceptual Biological Condition Gradient can be quantified and calibrated to local conditions for use in assessment and water quality criteria.** The tiers of condition described in the BCG conceptual model can be applied to local or regional conditions by regional biological experts with a sufficient monitoring database.

2. **A quantified BCG is not defined by any one monitored assemblage or methodology.** BCGs have been developed from different assemblages and methodologies (fish, benthic macroinvertebrates, artificial substrates, etc.) and by calibrating different assessment indicators to the BCG (IBI, RIVPACS, and multivariate analysis).

3. **Quantification and development of a BCG is data driven.** A regional monitoring database should be used to calibrate a BCG that meets performance requirements and QA requirements. The monitoring agency should have access to biological expertise, and should be committed to provide sustained support.

Chapter 3 has discussed transforming the conceptual scientific model of the BCG into a quantified and calibrated model for biological assessment. Chapter 4 discusses the Stressor Gradient model, the x-axis of the BCG. Chapter 5 discusses key concepts and milestones for developing tiered aquatic life uses in water quality standards that two states, Maine and Ohio, have learned based on their experience in adopting tiered uses, and is supported by their individual case histories of TALU development (Appendixes A and B). Chapter 6 presents examples of how Maine and Ohio have applied tiered uses in their water quality management program.

Chapter 3 Case Examples

CASE EXAMPLE 3-1. USING HISTORICAL INFORMATION TO IDENTIFY REFERENCE STREAMS IN KANSAS

Historical information can be used to reconstruct the pre-settlement biological baseline and estimate undisturbed or minimally disturbed conditions. Potential sources of historical data include museum fish and shellfish collections, historical notes and writings, journal entries, indigenous knowledge, published archeological studies, photographs and maps, and early biological surveys or studies.

Some knowledge of pre-settlement baseline conditions is needed when planning long-term restoration efforts in areas where undisturbed or minimally disturbed reference waterbodies no longer exist. For example, in Kansas, few streams have completely escaped the effects of large-scale agricultural and livestock practices implemented over the past 150 years. Therefore, biologists within the Kansas Department of Health and Environment (KDHE) consider available information on historical stream characteristics to estimate minimally disturbed conditions and support contemporary reference stream selections.

KDHE recognizes six general categories of aquatic biological responses to increasing levels of disturbance (Table 3-1). Class A represents natural or pre-settlement stream conditions, equivalent to Tier 1 in the BCG, in which "native structural, functional and taxonomic integrity is preserved; ecosystem function is presented within the range of natural variability." Some indication of the native character of streams in the Great Plains can be found in the narrative accounts of early nineteenth century explorers, including Lewis and Clark, Zebulon Pike, and George Sibley, among others. Railroad surveys and other investigations yielded additional information on the aquatic flora and fauna and generated maps and the earliest known photographs of many streams.

TABLE 3-1. Kansas stream biological integrity categories.

Class A:	Historical (natural) reference condition
Class B:	Contemporary (quasi-natural) reference condition
Class C:	Fully supportive of designated aquatic life use
Class D:	Partially supportive of designated aquatic life use
Class E:	Non-supportive of designated aquatic life use
Class F:	Grossly non-supportive of designated aquatic life use

Source: Kansas Dept. of Health and Environment

Although many of the biological surveys from the mid-1800s were performed after the start of intensive agriculture, they still provide valuable documentation of the occurrence of several freshwater species that soon disappeared from specific watersheds or the region as a whole. Museum collections and other historical records indicate that many creeks and smaller rivers in the Great Plains supported a variety of predominately eastern fish and shellfish species, most requiring clear water and relatively stable stream bottoms. In fact, this region was once home to more than 50 unionid mussel species. Today, several mollusca species are no longer found in most of their original habitats (Figure 3-6). Over the past 150 years, at least 11 aquatic molluscan taxa have become extinct in Kansas, and an additional 23 species are currently designated as endangered, threatened, or vulnerable.

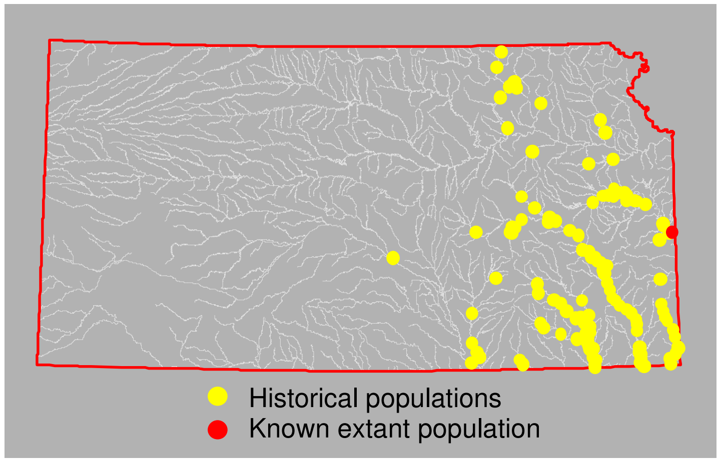


FIGURE 3-6. Decline in geographical distribution of black sandshell mussel in Kansas.

Because typical biological indexes (e.g., IBI) are usually developed from ambient "least disturbed" reference sites, they may lack sensitivity to discriminate among tiers or levels in the BCG. Surviving populations of historically occurring key species and indicator taxa can be used to further verify the minimally disturbed condition. KDHE considers historical fish, mussel, and prosobranch snail communities, and has created a "mussel loss" indicator metric that compares the taxa richness of the contemporary and historical unionid mussel assemblage for use in 305(b) and 303(d) list development (Figure 3-7). Sites retaining 90-100 percent of their pre-settlement species are deemed fully supportive of the aquatic life use, sites with 75-89 percent are considered partially supportive, and sites retaining 0-74 percent are assigned to the non-supportive category. In establishing long-term restoration goals, KDHE intends to continue drawing upon historical information sources to help ensure that the projected changes in aquatic plant and animal assemblages trend toward the pre-settlement biological condition.

There are some challenges and drawbacks when using historical data to reconstruct natural stream conditions. It takes a great deal of time and commitment to piece together numerous bits of information, especially considering the limitations and inconsistencies inherent in historical data. Much of the information is not directly comparable to modern assessment data, largely because results from previous studies and observations are often based on different sampling methodologies. Sometimes the data are not applicable because they were obtained after settlers significantly impacted the land, but often such physical habitat data are missing or incomplete. Finally, some regions settled early in the history of the nation may simply lack definitive data on the baseline biological condition.

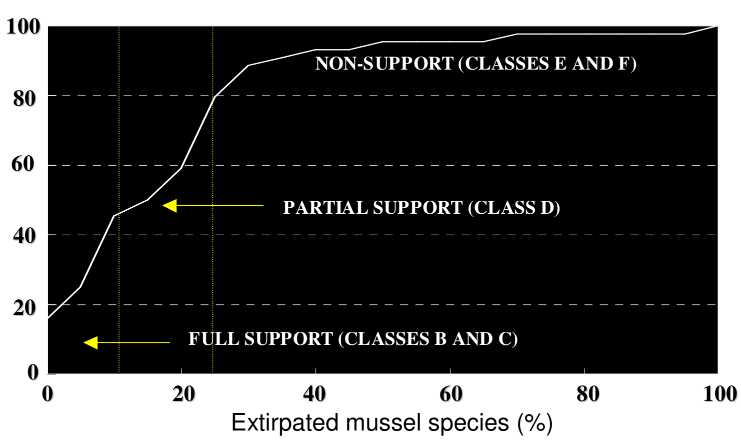


FIGURE 3-7. Cumulative frequency distribution for Kansas streams with minimum three-year period-of-record and five or more species historically.

CASE EXAMPLE 3-2. NEW JERSEY TIER DESCRIPTION

Aquatic biologists in New Jersey described tiers of the BCG for benthic macroinvertebrate assemblages of both high and low gradient streams of the state. The expert panel first assigned invertebrate taxa to Attributes I to VI. The panel redefined Attribute II from "sensitive-rare" taxa to "highly sensitive" taxa because rarity was not considered to be related to sensitivity to pollution, and sampling methods do not capture rare taxa with any predictable reliability. In addition, the panel determined that five tiers are applicable to New Jersey high gradient streams, and that four tiers describe the State's low gradient streams. For both high and low gradient streams, the panel thought that Tier 1 sites may not exist.

Table 3-2 shows the attribute matrix for high gradient streams. Attributes VII to X are not measured for the invertebrate assemblage at this time, and are not included in the matrix. The group was able to distinguish five separate tiers (Tiers 2-6) for high-gradient streams of New Jersey. The first tier described in the Maine model (Davies and Jackson in press) was not initially useful because it was not clear to the group whether Tier 1 (pristine) sites occur in New Jersey based upon benthic macroinvertebrate data alone. Other data sets (i.e. finfish communities and/or rare and endangered species) may be more useful in determining whether a site is in Tier 1. The group also determined that several indicator taxa are useful in discriminating tiers, in particular the tolerant hydropsychid caddisflies as indicators of moderate organic enrichment for Tiers 3 and 4; abundance of tubificid worms as an indicator of extreme enrichment and hypoxia for Tier 6; and complete absence of mayflies as an indicator of toxicity, also for Tier 6.

In contrast to high gradient streams, participants could only distinguish four separate tiers for low gradient streams (Tier 2, Tiers 3-4 combined, Tier 5, and Tier 6) (matrix not shown). The best-known sites in the Coastal Plain contain moderate numbers of tolerant taxa, which is a consequence of low water velocity and absence of cobble habitat rather than poor water quality. As a result, the group concluded that it was not feasible to distinguish Tier 3 from Tier 4, and combined them into a single tier.

In general, participants were able to achieve consensus on tier assignments for the sites reviewed. In some cases, there was discussion and some disagreement on which of two adjacent tiers a site should be assigned to. These intermediate sites, with characteristics of both adjacent tiers, are to be expected since ecological response to stressors is relatively continuous.

TABLE 3-2. Summary attribute matrix for New Jersey high gradient streams.

Ecological Attributes	1 Natural Condition	2 Minimal Loss	3 Some Replacement; Function Maintained	4 Notable Replacement Function Largely Maintained	5 Tolerants Dominant, Loss of Function	6 Severe Alter Structure and Function
I *Historically documented, sensitive, long-lived or regionally endemic taxa*	As predicted for natural occurrence except for global extinctions	As predicted for natural occurrence except for global extinctions	Some may be absent due to global extinction or local extirpation	Some may be absent due to global, regional or local extirpation	Usually absent	Absent

TABLE 3-2. Summary attribute matrix for New Jersey high gradient streams.

Ecological Attributes	1 Natural Condition	2 Minimal Loss	3 Some Replacement; Function Maintained	4 Notable Replacement Function Largely Maintained	5 Tolerants Dominant, Loss of Function	6 Severe Alter Structure and Function
II Highly sensitive taxa	As predicted for natural occurrence, with at most minor changes from natural densities	Virtually all are maintained and well represented (both taxa and abundance)	May be markedly diminished (in either taxa or abundance), with replacement by functionally equivalent *Sensitive and common* taxa	Significantly diminished (taxa and abundance)	Usually absent	Absent
III *Sensitive & common taxa*	As predicted for natural occurrence, with at most minor changes from natural densities	Present and may be increasingly abundant.	Common and abundant; relative abundance greater than *Highly Sensitive* taxa. Similar to good taxa (sensitive & common taxa).	Present with reproducing populations maintained; some replacement by functionally equivalent *taxa of intermediate tolerance.*	Frequently absent or significantly diminished (if present incidental)	Absent
IV *Taxa of intermediate tolerance*	As predicted for natural occurrence, with at most minor changes from natural densities	As naturally present at low abundances	Often evident increases in abundance	Common and often abundant; relative abundance greater than *Sensitive and common* taxa	Often exhibit excessive dominance	Richness of all taxa is low
V *Tolerant taxa*	As naturally occur, with at most minor changes from natural densities. If present, at very low abundance.	As naturally present at low abundances. May have several taxa at low abundances.	May be increases in abundance of functionally diverse tolerant taxa	May be common but do not exhibit significant dominance	Often occur in high densities and may be dominant	Usually comprise the majority of the assemblage; often either very low or very high densities.
VI *Non-native or intentionally introduced taxa*	Non-native taxa, if present, do not displace native taxa or alter native structural or functional integrity	Non-native taxa may be present, but occurrence has a non-detrimental effect on native taxa	Sensitive or intentionally introduced non-native taxa may dominate some assemblages (e.g. fish or macrophytes)	Some replacement of sensitive non-native taxa with functionally diverse assemblage of non-native taxa of intermediate tolerance	Some assemblages (e.g., fish or macrophytes) are dominated by tolerant non-native taxa	Often dominant; may be the only representative of some assemblages (e.g., plants, fish, bivalves)
XI *Potential Supplemental Attributes; Indicator taxa*	No apparent response of indicator taxa	No apparent response of indicator taxa	Initial response of indicator taxa, (e.g., increase of suspension feeders with enrichment)	Some response of indicator taxa, (e.g. increase of Caenids with silt, etc.)	Response of indicator taxa (e.g., loss of mayflies with toxic stress)	

CASE EXAMPLE 3-3. MAINE BIOLOGISTS' ASSIGNMENT OF SITES TO CLASSES (TIERS)

Maine DEP assembled a panel of three biologists to assign sites to each of Maine's three stream classes (A, B, C), and a fourth class representing non-attainment (NA). Each biologist independently reviewed biological information for each sampling event, including identities and abundances of taxa occurring in the biological sample and computed index values for the biological data (e.g. diversity, richness, EPT, etc). Physical habitat information was also reviewed including water depth, velocity, substrate composition, canopy cover, etc., in order to evaluate the effects of various habitat conditions on the structure of the macroinvertebrate community. Sample information was reviewed for the values of the given measures, relative to values for other samples in the data set. The actual classification assignment was determined by how closely the biological information conformed to the aquatic life classification standards, correcting for habitat effects. Numerical ranges, per se, were not established, *a priori*, for each measure. Instead, the information was reviewed for its compatibility with the mosaic of findings expected for each Class, listed in Table 3-3. The biologists did not have any knowledge of the actual location of the sampled sites, nor did they have knowledge of any pollution influences. Following the independent assignment of classes the biologists established a consensus classification, following an open exchange of justifications for each biologist's assignment.

Each biologist reviewed the sample data for the values of a list of measures of community structure and function. Criteria used by biologists to evaluate each measure are listed in Table 3-3.

In 64% of the cases there was unanimous agreement among the independent raters, and in an additional 34% of the samples two of the raters were in agreement and one had assigned a different classification. In three of the rated samples there was disagreement among all three raters (2%).

TABLE 3-3. Relative findings chart.

Measure of Community Structure	Relative Findings			
	A	B	C	NA
Total Abundance of Individuals	often low	often high	variable	variable: often very low or high
Abundance of Ephemeroptera	high	high	low	low to absent
Abundance of Plecoptera	highest	some present	Low to absent	Absent
Proportion of Ephemeroptera	highest	variable depending on dominance by other groups	low	zero
Proportion of Plecoptera	highest	variable depending on dominance by other groups	low	zero
Proportion of Hydropsychidae	intermediate	highest	variable	low to high
Proportion of Ephemeroptera & Plecoptera	highest	variable	Low	absent
Proportion of *Glossosoma*	highest	low to intermediate	very low to absent	absent
Proportion of *Brachycentrus*	highest	low to intermediate	very low to absent	absent
Proportion of Oligochaetes	low	low	low to moderate	highest
Proportion of Hirudinea	low	variable	variable	variable to highest
Proportion of Gastropoda	low	low	variable	variable to highest
Proportion of Chironomidae	lowest	variable depending on the dominance of other groups	highest	variable
Proportion of *Conchapelopia* & *Thienemannimyia*	lowest	low to variable	variable	variable to highest

TABLE 3-3. Relative findings chart.

Measure of Community Structure	Relative Findings			
	A	B	C	NA
Proportion of *Tribelos*	low to absent	low to absent	low to variable	variable to highest
Proportion of *Chironomus*	low to absent	low to absent	low to variable	variable to highest
Generic Richness	variable	highest	variable	lowest
Ephemeroptera Richness	highest	high	low	very low to absent
Plecoptera Richness	highest	variable	low to absent	absent
EPT Richness	high	highest	variable	low
Proportion Ephemeroptera Richness	highest	high	low	low to zero
Proportion Plecoptera Richness	highest	variable	low	zero
Proportion Diptera Richness	low to variable	variable	highest	variable to high
Proportion Ephemeroptera & Plecoptera Richness	highest	high	low to variable	low to absent
EPT Richness divided by Diptera Richness	high	highest	low to variable	lowest to zero
Proportion Non-EPT or Chironomid Richness	high	high	low	lowest
Percent Predators	low	low	high to variable	highest
Percent Collector, Filterers & Gatherers divided by Percent Predators & Shredders	high	highest	low	lowest
Number of Functional Feeding Groups Represented	variable	highest	variable	lowest
Shannon-Weiner Generic Diversity	low to intermediate	Highest	Variable to intermediate	lowest
Hilsenhoff Biotic Index	lowest	low	intermediate	highest

CASE EXAMPLE 3-4. VERMONT'S USE OF EXISTING BIOLOGICAL INFORMATION FOR THE BCG

Vermont used reference condition as the anchor point for assessing biological condition, and tiers of biological condition were established and described in terms of deviation from the reference condition. Biological narratives were developed, which provided guidance for evaluating degrees of deviation from the reference condition. The proposed language was intended to formalize Best Professional Judgment (BPJ) assessments by technical experts while remaining close to historical implementation. It was also critical that the new classification system maintain consistent assessment results, particularly for non-attainment findings.

Vermont tapped into more than 20 years worth of biological data collected from wadeable streams to develop biocriteria. Existing macroinvertebrate and fish assemblage monitoring data were evaluated for "reference" and "non-reference" condition in order to classify wadeable stream ecotypes and define biological reference conditions for each. Reference, or minimally disturbed, sites were determined based on BPJ. Various macroinvertebrate and fish community metrics were evaluated in order to describe their usefulness in detecting responses to disturbance.

Macroinvertebrate analysis identified four distinct wadeable stream ecotypes exhibiting unique biological characteristics: small high-gradient mountain streams; medium-sized high gradient streams and rivers; warmwater moderate gradient rivers and streams; low gradient soft bottom rivers and streams. A suite of eight macroinvertebrate community metrics was selected for the purpose of setting threshold criteria based on responsiveness to disturbance and impact. The eight metrics represent a range of structural and functional characteristics and were evaluated to minimize information redundancy. The range of reference condition was described for each metric and ecotype. Threshold criteria, based on deviation from the reference condition, were established for each ecotype consistent with the language contained in the water quality standards for each classification (Figure 3-8). Uncertainties associated with each threshold are recognized through the establishment of threshold ranges. The eight metrics are not combined into a single index number, but are evaluated separately in a BPJ analysis of use support status.

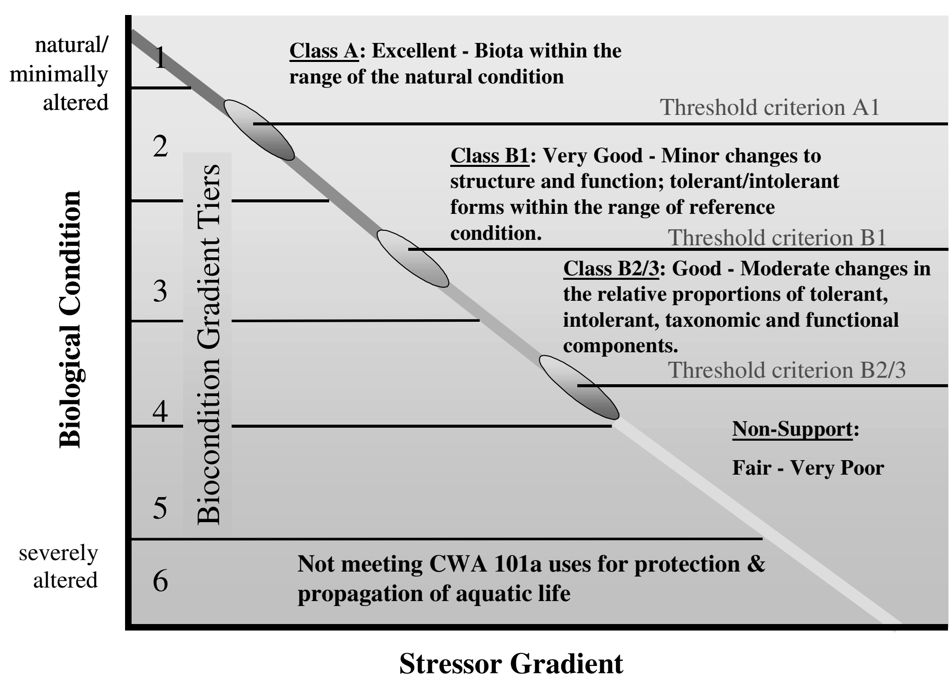


FIGURE 3-8. Vermont's designated aquatic life uses as differentiated by biological threshold criteria.

Two fish community indices of biotic integrity differentiating between strictly coldwater and mixed water assemblages were developed and calibrated to the Vermont Water Quality Standard narrative thresholds based on deviations from the reference condition. The indices combine multiple metrics representing a range of structural and functional characteristics into a single index number.

Since the BCG is continuous, it can be subdivided into any number of categories. The fish and macroinvertebrate criteria thresholds used by the Department were able to differentiate four categories of "support" status – Class A (near natural condition), high quality Class B1, general Class B2/3, and non-support (Figure 3-8). Common narrative descriptors – excellent, very good, good and fair-very poor were used to describe the thresholds. A determination of less than good was indicative of aquatic life use non-support. Categories of non-support (fair, poor, very poor) were not described.

When Vermont's new standards became effective in July 2000, all waters previously designated Class B were categorized as general Class B2/3 by default. The idea was to use the watershed planning process to propose and implement designated use reclassifications, particularly to the high quality Class B1. VtDEC is assembling candidate lists of waterbodies exhibiting high quality biological condition consistent with the Class B1 designated use. Final consideration of candidates is made via public process in order to ensure compatibility with local watershed plans and interests. Although no reclassifications have been made to date, the BCG has provided a clear visualization of the concepts of disturbance and impact, and this has been a useful tool in explaining the WQS to the public.

CASE EXAMPLE 3-5. DEVELOPING BIOLOGICAL CONDITION TIERS IN GREAT BRITAIN

In the 1980s, the Environment Agency of the United Kingdom sponsored the development of a nationwide monitoring and assessment program based on benthic macroinvertebrates. A four-year initiative, aimed at determining whether the macroinvertebrate community at a site could be predicted using physical and chemical features, led to the development of RIVPACS (River Invertebrate Prediction and Classification System). Other countries, and some states in the U.S. such as Oregon and Illinois, have subsequently integrated RIVPACS models into their biological assessment programs.

Predictive models like RIVPACS base assessments on the compositional similarity between observed and expected biota. To create a RIVPACS model for a particular region, standard protocols are followed to sample the region's biota and habitat at a network of reference sites that span the range of that region's environmental conditions. Sites are then classified based on biological similarity. Next, a multivariate model relates environmental setting (elevation, watershed area, geology) to the biological classification – this is used to estimate, or predict, the probabilities of sites belonging to biologically-defined groups and the probabilities of capturing each taxon. The current RIVPACS model, RIVPACS IIIa (Wright 2000), estimates two indexes for assessment – one based on the total number of expected taxa and a second based on expected average tolerance of the taxa. For both indexes, the model generates a list of taxa expected to occur under unstressed conditions, at greater than 50% probability for a particular assessment site. This list is then used to estimate the site's expected average tolerance value, and the probabilities are summed to generate the expected number of species. Both the number of predicted taxa that were actually observed and the tolerance value actually observed are divided by the expected values to obtain the final indexes. These indexes are compared against the model predictions to determine if the values are significantly different from the reference condition. Index values close to 1.0 indicate the site is similar to reference, and values less than 1.0 indicate deviation from reference.

Initially, the Environment Agency created four categories for the indexes – the scoring range below the 5th percentile of the index distribution of reference sites was divided into three equal categories, and the range above the 5th percentile made up the fourth. These categories, or grades, correspond to tiers of a BCG (Wright et al. 1994, Helmsley-Flint 2000). Review and application of the grades by regional biologists revealed that they did not discriminate between "good" and "very good" sites, or between "poor" and "very poor" sites (Helmsley-Flint 2000). Through cycles of data analysis and discussions with regional biologists, the Environment Agency was able to establish index thresholds for six grades, ranging from "very good" to "bad" (Table 3-4). The grades do not represent equal intervals of the index scores (Helmsley-Flint 2000). Although the British grades are determined solely by benthic macroinvertebrates, there is a distinct similarity between the narrative descriptions of the grades and the tiers of the BCG.

Assignment of a site to a grade is based on both the tolerance and total taxa indexes (Table 3-4). The indexes are independently applicable, and the lower of the two index scores determines the site grade. For example, if the total taxa index indicates "Good" but the tolerance index indicates "Fair", the site will be rated "Fair." To achieve the status of "Very Good", a site must have at least 85% of the expected taxa of an equivalent reference site and must have a tolerance index value (average score per taxon) as high as the expected value from a reference site.

Through an iterative process, the British Environment Agency was able to develop scoring criteria for existing indexes (RIVPACS N-Taxa and RIVPACS ASPT) that corresponded to regional biologists' consensus on tiers of a biological condition gradient.

TABLE 3-4. Definitions of six biological grades, developed by regional biologists of the Environment Agency in England and Wales (Helmsley-Flint 2000).

Grade	Definition	1 RIVPACS Index Scores	
		Tolerance Index (EQI ASPT)	**Taxa Index (EQI N-taxa)**
Grade a VERY GOOD	The biology is similar to (or better than) that expected for an average and unpolluted river of this size, type and location. There is a high diversity of Families, usually with several species in each. It is rare to find a dominance of any one Family.	≥ 1.0	≥ 0.85
Grade b GOOD	The biology shows minor differences from **Grade a** and falls a little short of that expected for an unpolluted river of this size, type and location. There may be a small reduction in the number of Families that are sensitive to pollution, and a moderate increase in the number of individual creatures in the Families that tolerate pollution (like worms and midges). This may indicate the first signs of organic pollution.	≥ 0.90	≥ 0.70
Grade c FAIRLY GOOD	The biology is worse than that expected for an unpolluted river of this size, type and location. Many of the sensitive Families are absent or the n umber of individual creatures is reduced, and in many cases there is a marked rise in the numbers of individual creatures in the Families that tolerate pollution.	≥ 0.77	≥ 0.55
Grade d FAIR	The biology shows big differences from that expected for an unpolluted river of this size, type and location. Sensitive Families are scarce and contain only small numbers of individual creatures. There may be a range of those Families that tolerate pollution and some of these may have high numbers of individual animals.	≥ 0.65	≥ 0.45
Grade e POOR	The biology is restricted to animals that tolerate pollution, with some Families dominant in terms of the numbers of individual creatures. Sensitive Families will be rare or absent.	≥ 0.50	≥ 0.30
Grade f BAD	The biology is limited to a small number of very tolerant families, often only worms, midge larvae, leeches, and the water hoglouse. These may be present in very high numbers. Even these may be missing if the pollution is toxic. In the very worst case there may be no life present in the river.	< 0.50	< 0.30

CASE EXAMPLE 3-6. MAINE'S USE OF LINEAR DISCRIMINANT MODELS TO ASSESS AQUATIC LIFE USE TIERS

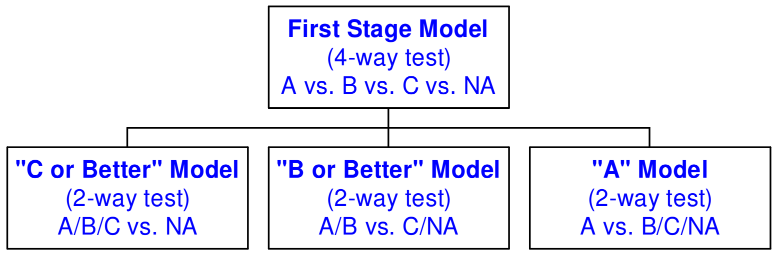
Maine identifies three aquatic life use classes for its streams – AA/A, B, and C – and also has a 4th category of non-attainment (NA) for streams that do not meet minimum water quality criteria (Table 3-5). The Maine Department of Environmental Protection (DEP) has developed a procedure using linear discriminant models (LDMs) to classify samples. LDMs are multivariate predictive models that use biological variables to determine whether a stream meets the biological criteria for classes A, B, or C, or if it falls into the category of non-attainment (Davies et al. 1995).

TABLE 3-5. Maine water quality classification system for rivers and streams, with associated biological standards (Davies et al. 1995).

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 minimally impaired. 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 Clean Water 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

To calibrate the LDMs, stream biologists from Maine DEP assigned an initial set of streams to the four aquatic life categories: A, B, C, and NA. Assignment of samples was based on presence-absence of taxa, abundance of taxa, richness, community structure, and ecological theory. Four linear discriminant models were calibrated from the initial data set. The four models function as a two-step process to evaluate individual sites:

> Step 1: First stage model – Estimates the probability of a site's membership into each of the four classes (4-way test)
> Step 2: Second stage models – Develop more accurate membership probabilities. Each is a two-way discriminant function, which perform better than multi-way models. There are three second stage models that estimate the probabilities of membership in a given class(es) versus any lower classes (Figure 3-9).



* Aquatic life use attainment decisions are based on the three 2-way tests.

FIGURE 3-9. Series of four linear discriminant models.

The models use 31 quantitative measures of community structure, including the Hilsenhoff Biotic Index, Generic Species Richness, EPT, and EP values to classify sites. In operational assessment, monitored test sites are run through the two-step hierarchical models and assigned to one of the four categories based on the probability results. Uncertainty is expressed for intermediate sites that fall between two categories. The assessment becomes the basis for management action if a site is rated as NA, or if its assessed category (B, C, or NA; the result of the LDM) is less than the site's assigned life use class (A, B, or C). Thus, if a site was assigned life use class A, but assessment shows that it only meets life use class B or C (model assessment was B or C), then management action may be required. If a site has improved, it requires further evaluation as a candidate for reclassification to a higher class.

Maine's numeric biocriteria provide an expert system for determining attainment of aquatic life uses. The LDMs provide an empirical model for expert judgment, which in turn is ultimately derived from years of empirical observations, ecological theory, data analysis, and clearly stated aquatic life management goals. They establish a direct relationship between the model's outcomes and management objectives (the aquatic life use classes). Therefore, broad resource goals and objectives can be directly translated to scientifically defensible, quantitative thresholds (Table 3-5). 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 would be necessary. Details of Maine's approach and statistical analysis procedures are in Shelton and Blocksom (2004) and Davies et al. (unpublished manuscript).

CASE EXAMPLE 3-7. NEW JERSEY QUANTITATIVE RULE DEVELOPMENT

After describing the BCG for high gradient streams of New Jersey (Table 3-2), the New Jersey DEP workgroup developed decision rules for assigning sites to the tiers (Table 3-6). Biologists in the New Jersey workgroup generally preferred to use taxa richness as the first and most important criterion for determining site tier assignments. Thus, the number of highly sensitive taxa was most often used to distinguish between Tier 2 and Tier 3 sites. Tier 2 should have several highly sensitive taxa (Attribute 2), but their richness may be reduced in Tier 3. For example, a preliminary rule for Tier 2 was that highly sensitive taxa richness (Attribute 2 taxa richness) should be at least 50% of the richness of any other attribute group (3 through 5). Similarly, the difference between Tiers 3 and 4 was viewed primarily as changes in the sum of richness of highly sensitive and sensitive-common taxa, such that in Tier 3 sites, the sum of taxa richness of the two sensitive groups should be at least 50% of the sum of richness of the two more tolerant groups.

Although taxa richness was generally the first criterion for the higher tiers (Tiers 2 and 3), relative abundance could override richness in extreme cases: Tier 3 was required to have more than 25% relative abundance of the two sensitive groups combined, and severely reduced abundance (< 50 organisms in the total sample, after QA determined that the sample was properly collected and processed) can downgrade a site to Tier 6 in combination with signals of potential toxicity.

Tier 5 was discriminated from Tier 4 by a significant reduction of sensitive taxa (Attributes 2 and 3) to the point where they are merely incidental if present and are not a functional part of the community. Approximately 10% relative abundance was deemed a functional part of the community. Tier 6 was discriminated from Tier 5 by increasing loss of all taxa and dominance by tolerant taxa (Attribute 5). Tier 6 could also be indicated by extreme low numbers combined with signals of toxicity (complete absence of mayflies, presence of Cricotopus), without other Attribute 5 taxa.

The rules are applied as a downward cascade: for a site to be rated as Tier 2 (the highest defined tier for New Jersey), all attributes must meet the Tier 2 condition (Table 3-6). A Tier 3 rating requires one or more failures of Tier 2 rules, but the site must meet all remaining Tier 3 rules. These rules cascade to Tier 5. Tier 6 has special rules of exceedingly low taxa richness, or abundance, or complete dominance of tolerant taxa (Attribute 5). Tier 5 consists of sites that fail Tier 4 conditions, yet also fail Tier 6.

TABLE 3-6. Proposed decision rules for New Jersey high gradient streams.

Attributes	**Tiers**					
	1	**2**	**3**	**4**	**5**	**6**
All Taxa						Low richness (<10 taxa) Low abundance (<50 individuals)
I Sensitive, regionally endemic taxa		(No rules determined for Attribute 1)				
II Highly sensitive taxa		Taxa >= 50% of any other Attribute (± 10)%	*Taxa >=2* (± 2 taxa)	May be absent	May be absent	May be absent
III Sensitive & common taxa		Taxa (2 + 3) >= Taxa (4 + 5) (± 2 taxa)	Taxa (2 + 3) > 50% of Taxa (4 + 5) (± 10%)	Taxa (2 + 3) ≥ 3 (± 2 taxa) Abund (2+3) >10%	May be absent; abund (2 + 3) <10% (or less than 3 taxa) (± 5%)	May be absent

TABLE 3-6. Proposed decision rules for New Jersey high gradient streams.

Attributes	Tiers					
	1	2	3	4	5	6
IV Taxa of intermediate tolerance					Taxa (4) ~= Taxa (5) Abund (4) >= Abund (5)	
V Tolerant taxa		<20% of total abundance (± 5%) (if tiers 2 and 3 ambiguous)	<50% of total abundance (± 10%)		High density, abundance of Attributes 4, 5	Taxa (5) > Taxa (4) (± 2 taxa) Abund (5) > Abund (4)
V.a. Indicator taxa		Tolerant Hydropsych. <= 10% abundance (± 5%)	Tolerant Hydropsych. <= 50% abundance (± 10)%		Hydropsych. may dominate Tubificidae not dominant	Mayflies absent Tubificidae dominate Attrib 5
Combinatorial rules (RxC format)		(II,2) and (III,2) and (IV,2) and (V.a,2)	(not (II,2) or not (III,2)) and (II,3) and (III,3) and ((V,3) or (V.a,3))	(Not (II,3) or Not (III,3)) and (III,4)	Not (III,4) and Not (All,6) and Not (V,6)	(All,6) or (V,6)

CHAPTER 4. THE X-AXIS: A GENERALIZED STRESSOR GRADIENT

The x-axis of the Biological Condition Gradient Model (BCG) illustrates how increasing levels of stressors in aquatic ecosystems change biological condition. This chapter presents a conceptual model that helps characterize stressor gradients by focusing on the progression from sources (changes in key environmental processes) to stressors and ultimately to their effects on biotic condition (Figure 4-1). The model also looks at the mechanisms through which these biotic components are affected. The stressor gradient model can be used to organize data and information on watershed characteristics, hydrologic modifications and stressors to thoroughly evaluate these relationships. This information will provide a foundation for States and Tribes to use the BCG to address both current conditions and ecological potential of their waterbodies, develop realistic restoration options for impaired waters, and communicate this information to the public.

4.1 The scientific foundation for the stressor gradient

Stressors affect biological assemblages and ecosystem processes both directly and indirectly, including altering metabolic pathways, energy availability and behavior of the organisms (Karr et al. 1986, Adams 1990, Poff et al. 1997). Historically, point source pollution and in-stream hydrological modifications were the dominant alterations (see 4.2.1) to fresh waters. While these issues continue today, water quality management now faces a wider variety of changes stemming from mining, forest harvest, agriculture, urbanization, industry, and even recreation (Richter et al. 1997, Bryce et al. 1999). In addition, non-contaminant related changes to aquatic ecosystem factors (see text box below) commonly impact biological conditions (Figure 1-3) and can also influence other stressors (Karr and Dudley 1981, Karr et al. 1986, Poff et al. 1997, Slivitzky 2001). Consideration of these factors and their interactions in water quality management can lead to greater improvements to biological condition than a focus on contaminants alone (Karr et al 1986).

1. **Chemical factors** (e.g., hardness, nutrients, toxic compounds)
2. **Flow regime** (including the timing and amount of water in the channel; diversions)
3. **Biotic factors** (competition, predation, disease, invading species, etc.)
4. **Energy source** (photosynthesis, inputs from land, etc.)
5. **Habitat structure** (channel shape and features, siltation, etc.) *(from Karr et al. 1986, see Figure 1-3)*

The influence of each factor on biological condition in specific waterbodies can be difficult to evaluate and quantify because each of these factors reflect both indirect and direct forces. Flow regime, for example, affects biological condition and the other in-stream factors (e.g., habitat structure, water quality) (Poff et al. 1997). Altered stream flows are associated with poor channel habitats, erosion, bank instability, and lower base flows (Poff et al. 1997). Species distributions, abundances, and competitive interactions all rely on natural flow regimes (Poff and Allan 1995, Greenburg et al. 1996, Reeves et al. 1996, Poff et al. 1997). Stream ecosystem structure and function (Vannote et al. 1980) and the riverscape concept (Ward 1998, Fausch et al. 2002) integrate the influences of all stressors. These individual and collective influences, represented by the BCG model's x-axis – the Generalized Stressor Gradient (GSG) – drive the biological condition of streams and reveal the need for a more holistic approach to stream monitoring and management. Because of the dynamic nature of aquatic ecosystems, however, all of these factors are in a state of constant flux. The natural range of conditions that native biota are adapted to may be narrow, wide, or seasonally variable, depending on the climate, topography and ecoregion in which the system occurs. A simplified model, therefore, is needed to help organize environmental factors and their relationships to stressors and biological responses.

4.2 The conceptual model for a Generalized Stressor Gradient

Building upon the Karr conceptual model, the Generalized Stressor Axis model characterizes the environmental processes and mechanisms that generate stressors which lead to biological responses within waterbodies (Figure 4-1). An event or activity that alters the aquatic system is called a **disturbance**. Ecosystems normally have some level of disturbances that characteristically occur within a range of natural variability. Disturbances beyond this range, however, can exert **pressure**[1] upon an aquatic system by altering fundamental environmental processes and ultimately generating stressors. Stressors are physical, chemical or biological factors that cause an adverse response from aquatic biota (U.S. EPA 2000b). The term "pressure" conceptually and mechanistically links larger scale landscape and hydrological disturbances with the ecological processes that are ultimately changed, leading to pressure(s) being "felt" by the aquatic biota. Stressors are what link pressures to effects on biota, via exposure mechanisms. A stressor, therefore, can be traced back to its source or tracked forward to the biological response, via a causal pathway (Figure 4-1). For example, destabilized stream banks due to removal of riparian plants could be the source of excess fine sediment to a stream. Erosion by high flows is the mechanism by which the excess fine sediments are generated, and the resulting in-stream siltation is the stressor. Smothering of bottom substrate habitat and organism gills by these fine sediments are two mechanisms by which biota are exposed and adversely affected. Invertebrate mortality and fish emigration could be some of the environmental outcomes or changes in biotic condition.

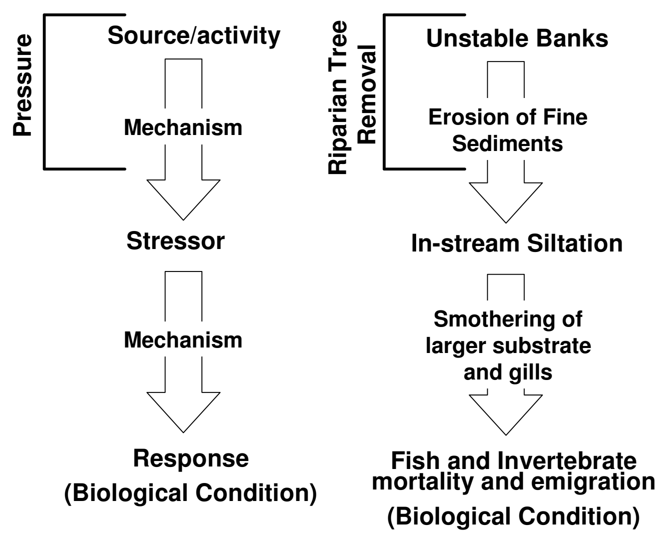


FIGURE 4-1. Conceptual model illustrating the linkages between pressure and biological condition. The specific stressor(s) and their intensity (the BCG x-axis) are created via pressure(s) acting through specific mechanisms. An example for each step of the model is also shown.

The effects of stressors on biota, however, depend on the magnitude, frequency, and duration of exposure to the stressors. Developing a BCG for a given system characterizes the general relationship between its combined stressors (the model's x-axis) and its overall biological condition (the y-axis). Multiple stressors are usually present, and thus the stressor x-axis of the BCG seeks to represent their cumulative

[1] The use of the word pressure in this context has a well-established history in the European environmental literature. Pressure is a term originally used by the European Union in its Water Framework Directive (OECD 1993). SOLEC (State of the Lakes Ecosystem Conference) also used the term pressure and defined it to be the outcomes of human activities that have the potential to cause environmental effects (Shear et al. 2005).

influence as a **Generalized Stressor Gradient (GSG)**, much as the y-axis generalizes biological condition.

4.2.1 How the model supports development of a GSG

The conceptual model provides a theoretical basis for relating single or multiple stressors to biotic responses and condition. This concept is taken further in developing a generalized stressor gradient, which, as the BCG's x-axis, is used in relating cumulative stressors to cumulative biotic effects. The factors that drive biological condition (Figure 1-3) and how condition is affected by a range of stressor intensities are used in defining the gradient. Two example GSGs are provided below.

Tables 4-1A and 4-1B outline example scenarios for humid-temperate (Table 4-1A) and arid (Table 4-1B) regions of the U.S. under differing levels of stressors. The high, medium and no/low stressor levels are used only to describe relative differences in magnitude and are not formal categories for classifying stressors. The five factors from Figure 1-3 were modified to six factors by separating toxics (e.g., copper, cadmium, mercury) from conventional chemical pollutants (e.g., nitrates, phosphorous).

When stressors are absent or low, natural or near-natural conditions of the aquatic ecosystem prevail. However, as stressors increase, one or more of the six factors can deviate from natural conditions. In humid temperate regions, for example, the loss of a watershed's forested landscape generally increases in-stream stressors by affecting flow, soil erosion, water quality and aquatic habitat structure. In arid regions, loss of riparian vegetation and cryptogamic crusts (a tightly bound mesh of lichen, algae and lower plants that prevent erosion and provide a hospitable environment for germinating plants) has the same kind of effects.

TABLE 4-1. Example scenarios for humid-temperate (A) and arid (B) regions of the US under three levels of stressors. The stressor levels are used only to describe relative differences in magnitude and are not formal categories. Karr's five factors (Figure 1-3) were modified to six factors by separating toxics from conventional chemical pollutants. These scenarios were written primarily from the reach-scale perspective, both local and watershed scale factors, however, are important for determining the condition of streams.

(A) Humid-temperate Scenario

Stressor Level	*Flow Regime*	*Habitat Structure*	*Water Quality*	*Toxics & Bioengineered Chemicals*	*Energy Source*	*Biotic Interactions*
No/Low	As naturally occurs, includes floods & low flows at natural rates and extent; High connectivity with ground water maintained	As naturally occurs, varies with size & slope, typically large wood abundant, coarse substrate, overhanging vegetation, and undercut banks are present	As naturally occurs or only minimal increase in nutrients & sediments; no point sources, includes flood turbidity & summer warming; usually cool or cold & dissolved oxygen (DO) saturated; sediments & nutrients low & pulsed seasonally	As naturally occur, typically rare and no toxics in amounts toxic to aquatic biota	As naturally occurs, varies with channel width, typically dominated by riparian woody vegetation, unless naturally autochthonous	As naturally occur, anadromy and potamodromy common or only slightly reduced by distant dams or fishing; beavers common; aliens non-detrimental; DELT anomalies absent; no or insignificant historical range changes
Medium	Flashier, increased drought frequency; some water withdrawals; low to moderate wetland drainage; dams may reduce annual floods and droughts	Reduced LWD in channel; fines slightly to moderately more abundant than expected from stream power; pool substrate moderately embedded; reduced undercut banks, overhanging vegetation, and habitat complexity; some loss of pool volume and pool/riffle proportions may be altered.	Enriched, turbidity may increase, moderate diel warming, small DO sags may occur but these rarely violate criteria; point sources minor or if they exist are treated; fish kills rare	Toxics rarely in amounts toxic to aquatic biota, but Hg may be of chronic concern to top piscivores due to bioaccumulation; sediment contamination may be detectable but not causing effects in biota.	Autochthonous production higher than expected in lower order streams; filamentous algae may be present	Altered fish age structure from fishing; beavers diminished; anomalies infrequent; sensitive aliens may dominate, tolerant aliens may be present; minor to moderate historical range reductions; cosmopolitan species may extend distributions further upstream. DELT anomalies rare; stocking may be influencing native populations
High	Flashy; highly altered drought/flood regime; mostly or entirely human controlled in urban areas; water withdrawals & impoundments if present, fundamentally alter the nature of the ecosystem	Simplified or manmade; wood, undercut banks & overhanging vegetation absent or non-functioning; rubble & trash common, substrates highly armored or embedded. Dam impoundments often present.	Highly enriched, turbid, warm; large diel DO & temperature changes; chemical and point sources inadequately treated or overwhelmed by untreated diffuse toxic pollution. Dams when present produce altered thermal regime and nutrient dynamics	Fish kills may be common in low summer flows; toxics may be present in chronic or acutely toxic amounts; bioengineered chemicals can affect growth & reproduction; high to extreme sediment contamination; fish consumption advisories serious	Mostly autochthonous or imported fine particulate organic matter or dissolved organic matter; may be too turbid for filamentous algae	Dominated by transient fishes or tolerant aliens; many historically common species extirpated; anomalies when associated with toxic impacts are abundant & serious; beavers transient or absent.

TABLE 4-1. (B) Arid Scenario

Stressor Level	***Flow Regime***	***Habitat Structure***	***Water Quality***	***Toxics & Bioengineered Chemicals***	***Energy Source***	***Biotic Interactions***
No/Low	As naturally occurs or only slightly altered, includes floods & low flows at 10-20 & 20-50 yr intervals, respectively; floods flashy; annual scouring flows; high connectivity with ground water.	As naturally occurs, varies with geology, substrate, flow, size, slope, soil, latitude, elevation & orography; relatively stable riparian vegetation, LWD in flats.	As naturally occurs with only minimal increase in nutrients & sediments, includes flood turbidity & summer warming; depending on soils, may be naturally saline or alkaline; enriched where beaver present; ash from 5-20 year fire cycles. No point sources	As naturally occur, typically rare, but may be natural sources of arsenic & selenium. No toxics in amounts toxic to aquatic biota	As naturally occurs, varies with channel width, typically dominated by riparian woody vegetation in small unconstrained channels; heterotrophic & autochthonous in wider systems	***A***s naturally occur, potamodromy (long distance river migrants) common and only slightly reduced by distant dams; beavers common; aliens absent or non-detrimental
Medium	Altered, increasingly flashy; increased drought frequency; some water withdrawals and wetland drainage; flow alterations mitigated to some extent by environmental flow releases.	Minor amounts of incision, widening or shallowing; reduced LWD in channel; fines greater than expected from stream power; bed coarsening from upstream dams; pool substrate increasingly embedded; reduced aquatic macrophytes, undercut banks, & overhanging vegetation.	Enriched, warmer & saltier, turbid at low flows, small DO sags; point sources if present with treatment. No fish kills	No acute toxicity is observed, but chronic toxicity is possible due to bioaccumulation. Fish consumption advisories likely for sensitive populations	Mostly allochthonous, but increasingly autochthonous in narrow streams; wide streams heterotrophic or autochthonous with increasing amounts of filamentous algae.	Altered fish age structure from fishing; aliens more common and beginning to reduce competitors & prey; potamodromy reduced; beavers markedly diminished.
High	Human controlled; large inter-basin transfers; ground water overdrawn; effluent dominated streams below cities. Highly altered drought/flood regime; droughts yield more dry channels; withdrawals & dams severely alter nature of the ecosystem.	Largely manmade; little or no LWD, undercut banks & overhanging vegetation; highly sedimented; construction rubble & trash common.	Highly enriched, turbid, warm; large diel DO changes; most point sources inadequately treated or overwhelmed by untreated diffuse toxic pollution; dams produce altered thermal regime.	Fish kills in low summer flows or after rains; toxics seasonally or always present in acutely toxic amounts; mine spills; bioengineered chemicals affect growth & reproduction.	Mostly autochthonous or imported fine particulate or dissolved organic matter; filamentous algae common if turbidity allows it.	Dominated by transient fishes or tolerant aliens; fish consumption advisories serious; once-common species now threatened, endangered or extirpated from large portions of their historical ranges; potamodromy rare and erratic; beavers transient or absent.

4.3 How the BCG model and management actions are linked

Pressure, as used in this document, applies to the environmental processes that can be altered by certain activities and the mechanisms from those activities that generate stressors. Many landscape altering activities can be quantified with such measures as population density, proportion of land devoted to agriculture, total miles of roadway, or quantities of water used /released. These activities, however, may or may not generate stressors. Actions can be taken that insulate stream processes from the environmental pressure of certain activities, helping to maintain or restore the ecological potential of an aquatic system.

Controls and Best Management Practices (BMPs) are management actions designed to mitigate or reduce the levels and effects of stressors that adversely alter stream ecosystem function. BMPs can function in a number of ways: they may reduce the stressors being generated by sources, reduce the exposure of biota to stressors, or increase the resistance of an aquatic ecosystem to adverse changes. For example, urbanization without controlling for the effects of added impervious surface is a pressure that often results in reduced biological condition. The typical alteration of water flow (such as more frequent flooding due to increased runoff) causes stressors. The mechanism for flow alteration is the creation of large expanses of impervious surfaces, characteristic of most cities. Impervious surface speeds up the flow of water over the land during rain events often resulting in more frequent and more intense floods. Constructing retention ponds to store run-off water is a control measure that doesn't alter the pressure of urbanization, but may reduce the stressors acting on the stream system. Mechanistic processes operate between pressures and stressors, and between stressors and biological response (Figure 4-1). Understanding these mechanisms, and how they operate, is the key to identifying the likely effect of a particular management action and its likelihood to produce the desired response in biological condition. In the retention ponds example above, the pressure (urbanization) and mechanism for stressor generation (excessive surface run-off) still exist, but their influence on in-stream stressors has been neutralized by a management action, and therefore the exposure mechanism influencing the biological community was reduced or eliminated.

The basis of the BCG model is that increased pressures can generate increased stressors, and in turn, increased stressors are associated with decreasing biological condition (Figure 4-2A through D). Systems that are minimally affected by stressors exhibit natural condition (Tables 4-1A and 4-1B). Human activities may exert pressure and generate stressors on aquatic systems, resulting in changes from the natural state. Typically, the stressors on aquatic systems increase as pressures increase (Figure 4-2A dashed line). Effective management practices, however, can alter the effects of pressures and reduce stressors. The solid, curved line in Figure 4-2B represents this theoretical relationship graphically. With effective controls and/or BMPs, a given amount of pressure (vertical fine dashed arrow rising from the pressure axis) results in a lower stressor level (where the dashed arrow intersects the stressor axis). Figure 4-2B illustrates the influence of effective management in changing the pressure/stressor relationship in ways that will subsequently improve the biological condition.

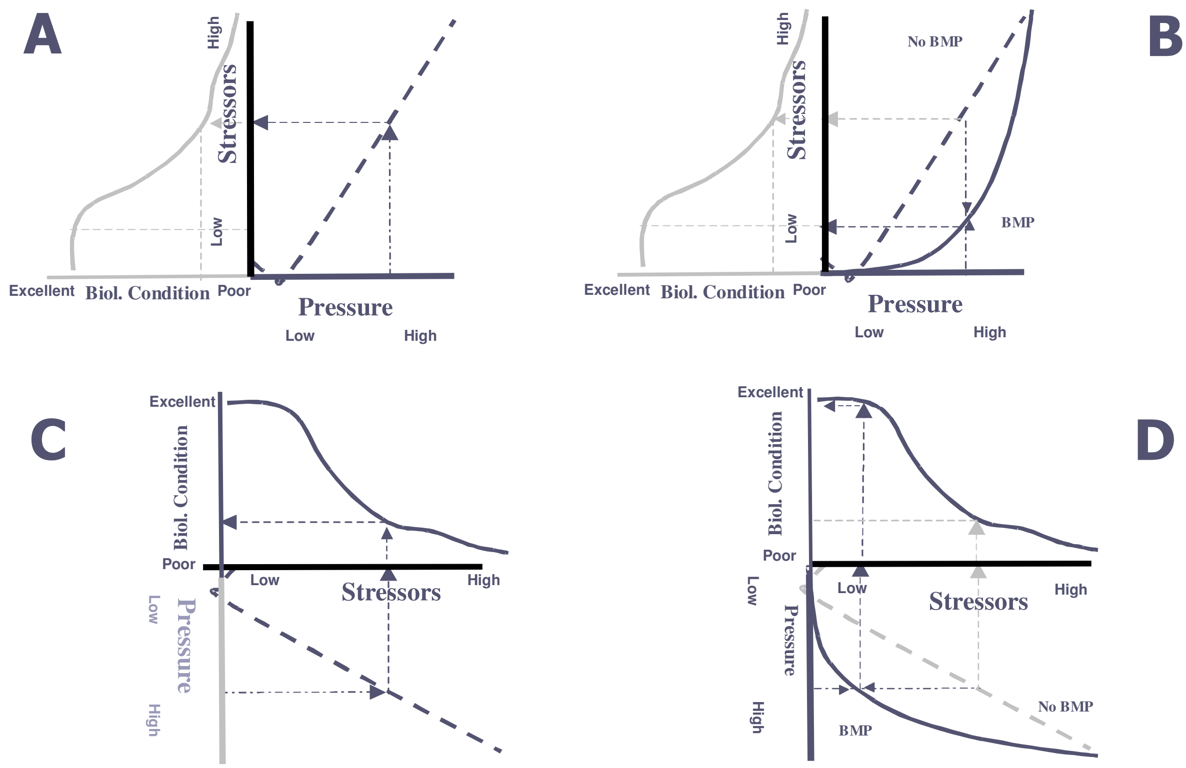


FIGURE 4-2. Relationship between pressure, stressors, and biological response.

Figure 4-2C is a 90 degrees clockwise rotation of Figure 4-2A. Stressors (which are shown to increase in response to increasing pressure in Figure 4-2A) are now on the x-axis. Biological condition is shown as the response variable on the y-axis. This represents the biological condition-stressor relationship developed in Chapter 1. In this example, the moderate-high effect of the stressors (dashed arrow rising from the stressor axis) results in poor biological condition (the point where the dashed arrow intersects the biological condition axis).

Figure 4-2D shows Figure 4-2B rotated 90 degrees clockwise. As in Figure 4-2C, stressors are on the x-axis, and biological condition is shown as the response variable on the y-axis. The effect of low levels of stressors (dashed arrow rising from the stressors axis in Figure 4-2D) results in near excellent biological condition (where the dashed arrow intersects the biological condition axis). The pressure-stressor relationship has been shaded out. But it reminds us how, together, pressure and management actions (i.e., permit limits, BMPs, channel restructuring) can determine stressor levels, and ultimately, the condition of the biota. The specific effects of stressors on biological responses will depend on the type, magnitude, duration, and frequency with which the stressor occurs. These stressor attributes are, in turn, a result of the cumulative pressures exerted on the ecosystem and relevant management decisions to mitigate these pressures.

Different types of disturbances can exert pressure on an ecosystem through altering fundamental processes such as water flow, transport of materials, watershed/riparian structural dynamics, channel structural dynamics and biological activities. For example, dams and impoundments alter flow, natural biological activities and material transport by creating lake conditions in a stream environment, and creating barriers to fish movements and migration. Sediment, nutrient and organic matter transport are all reduced downstream of impoundments and water quality attributes such as natural temperature fluctuations and dissolved oxygen are often altered by dams. When severe enough, these alterations act as stressors to the downstream community.

Tools can be developed that characterize the relationships among pressures, altered processes, the stressors they generate, and the resulting biological responses. Information from pressure and stressor indicators provides insight on how changes in these fundamental processes may be affecting the biological condition of water resources (Table 4-2). Understanding how specific stressors are generated and the influence of specific stressors on biological condition, provides the underpinnings for the BCG's stressor axis. Further, it reveals potential opportunities for management actions to reduce stressors and counteract the alteration of fundamental processes.

TABLE 4-2. Fundamental environmental processes typically altered by disturbances that ultimately generate stressors. For each process, example mechanisms that link pressures (pressure indicators) to stressors and typical stressor indicators for the major environmental factors impacted are listed. Management Actions (indicated by the grey bar place holder), when effective, will alter the effect of the pressures, thereby alleviating some or all of the mechanism for increasing stressors.

Process	**Pressure Indicators** (potential for stress)	**Mechanism for Stressor Production**		**Stressor Indicators for each Major Factor**	**Comments**
Flow alteration	-% impervious area -road density -% urban -population-density -storm sewer miles -# diversions -# of dams -point source dischargers	-acceleration of water flow -reduced groundwater infiltration -increased peak flow -more frequent elevated events -reduced base flow -suspended fines during floods increased subsequent deposition -increased incision and changes to channel structure due to increased power during flood	Management Actions (BMPs /Controls)	<u>Flow regime</u> -changes in flood frequency -changes in base flow -changes in drought frequency -changes in stream power <u>Habitat Structure</u> -increased fines -increased armouring of substrate <u>Water Quality</u> no direct indicators <u>Toxics</u> no direct indicators <u>Energy Source</u> no direct indicators <u>Biotic Interactions</u> -lose certain fish life histories	Alteration of flow includes changes to the rate, volume and timing of discharge Alteration of water flow also changes materials transported and channel structure; the consequences also affect water quality, toxics, and energy sources (See Table 4-1) Alterations of habitat lead to changes in fish life histories adapted to natural flow regimes leading to migrators, rheophils and nonguarding lithophils & lithopelagophils being replaced by residents, generalists, and polyphils

TABLE 4-2. Fundamental environmental processes typically altered by disturbances that ultimately generate stressors.

Process	**Pressure Indicators** (potential for stress)	**Mechanism for Stressor Production**		**Stressor Indicators for each Major Factor**	**Comments**
Alteration of materials transported	-point source dischargers and discharge constituent levels -km of riparian buffers -% impervious surfaces -road density -row crops -population-density -atmospheric deposition -CAFOS -km^2 of tile drains -non-point sources -logging -# mines -# septic systems -fertilizer use -irrigation -# quarries	-erosion of surface solutes, sediments and warmer water -increased discharge of chemicals from point sources -increased algal biomass results in greater daytime photosynthesis and night-time respiration -increased material input adds carbon and nutrients that increase biological activity	Management Actions (BMPs /Controls)	<u>Flow regime</u> See above <u>Habitat Structure</u> See above <u>Water Quality</u> -altered nutrient concentrations -altered TSS -altered TDS -altered temperature -altered turbidity -increased coliform <u>Toxics</u> -increased contaminants in fish -increased pesticides -increased bioaccumulative organics -increased metals in water and sediments <u>Energy Source</u> -suspended algae could increase or decrease <u>Biotic Interactions</u> -changes in invertebrate functional groups to more filterers or grazers -sensitive species diminished; % tolerant species increased -algal community changes	Sediment deposition alters habitat complexity and structure; during floods, remobilization of buried materials can reintroduce nutrients and bioaccumulative toxics to the food web Algal growth may increase due to increased nutrients or decrease due to high turbidity. These state changes will affect invertebrate functional group composition (e.g. more filterers due to high levels of suspended organic matter or more grazers if alterations in material transport are reduced).

TABLE 4-2. Fundamental environmental processes typically altered by disturbances that ultimately generate stressors.

Process	**Pressure Indicators** (potential for stress)	**Mechanism for Stressor Production**		**Stressor Indicators for each Major Factor**	**Comments**
Changes to channel structure	-km channelized -# dams -# culverts -density of road crossings -valley fills -diversions -levees -bank-stabilization -riprap/ Concrete -floodplain losses -snagging of LWD	-flow alteration -solute sediment transport -direct engineering activities	Management Actions (BMPs /Controls)	<u>Flow regime</u> -changes in mean velocity -changes in discharge	Alteration of normal channel shape and depth alters the dissipation and flow of energy during hydrological transport both laterally and longitudinally which ultimately reduces habitat complexity and therefore reduces community diversity.
				<u>Habitat Structure</u> -reductions in number of habitat types -reduced pool depth -reduced substrate heterogeneity -increased embeddedness -changes in width: depth ratio -reduced number of snags -reduced woody debris -reduced off channel habitat (including braiding index) -reduced flood plain connectivity, bank angle/stability	
				<u>Water Quality</u> no direct indicators	
				<u>Toxics</u> no direct indicators	
				<u>Energy Source</u> -autochthonous allochthonous shift	
				<u>Biotic Interactions</u> -reduced pool dwelling organisms -loss of specialized insectivores -blocked migrations -reduced spawning habitat	

TABLE 4-2. Fundamental environmental processes typically altered by disturbances that ultimately generate stressors.

Process	**Pressure Indicators** (potential for stress)	**Mechanism for Stressor Production**		**Stressor Indicators for each Major Factor**	**Comments**
Changes to watershed/ riparian structure	-% of area developed -# trapping permits -levees -tile number-drains/ditches -km of streamside roads -surface area of off stream ponds or wetlands -valley bottom grazing -riparian width -riparian continuity -aggregate mining -devegetation -fragmentation	-landscape alterations -clearing of vegetation -paving surfaces -building structures -agricultural practices -draining or filling wetlands -riparian disturbance or removal -increased algal biomass results in greater daytime photosynthesis and night-time respiration -increased material input adds carbon and nutrients that increase biological activity	Management Actions (BMPs /Controls)	<u>Flow regime</u> -changes in flood severity and response time <u>Habitat Structure</u> -reductions in amount of woody debris -loss of riparian trees -reduced riparian structural complexity -reduced off channel habitat (including braiding index) -reduced flood plain connectivity -increased riparian fragmentation <u>Water Quality</u> -increased fine -increased sediments -increased nutrients -increased ions <u>Toxics</u> no direct indicators <u>Energy Source</u> -P/R changes -increased algal contributions; -shift from allochthonous toward autochthonous <u>Biotic Interactions</u> -loss of sensitive species -addition of intermediate species and total species -changes in invertebrate functional groups to more filterers or grazers -algal community changes	Watershed/riparian bottom and vegetation provides hydrological assimilative capacity during flooding and contributes water and nutrients during interflood periods; these are also important nursery habitats and refugia that maintain biodiversity

TABLE 4-2. Fundamental environmental processes typically altered by disturbances that ultimately generate stressors.

Process	Pressure Indicators (potential for stress)	Mechanism for Stressor Production		Stressor Indicators for each Major Factor	Comments
Changes in biological activity	-# non-indigenous invasive species (NIS) -# fish stocked (species individuals) -baitfish sales -fishing licenses -creel census results -# Guide licenses	-stocking programs -foraging by invasive species (grazing, predation) -habitat modification by invasive plants and fish	Management Actions (BMPs /Controls)	<u>Flow regime</u> -macrophytes (decreases-> accelerate flow; increases->reduce flow) ***Habitat Structure*** -macrophytes/algalmats alter substrate <u>Water Quality</u> -increased BOD <u>Toxics</u> no direct indicators <u>Energy Source</u> no direct indicators <u>Biotic Interactions</u> -NIS (native gamefish decline, hatchery fish increase) -native fish/benthos and riparian vegetation and birds increasingly replaced by aliens -sensitive specialists replaced by tolerant generalists (birds, fish, invertebrates, plants) -increased fish diseases and anomalies -riparian vegetation may be eliminated	Loss of species from invasives and stocking practices

4.3.1 Additional considerations for the stressor axis

The concepts of spatial and temporal scale are critical issues in adequately defining a stressor axis. Stressors may be introduced through diffuse or point sources delivered from upstream in the channel or watershed, or laterally from riparian, floodplain or upland sources. Pollutants can also be delivered through atmospheric sources from above, or below from groundwater sources. Activities in the watershed or along the waterbody corridor will influence the connectivity and integrity of the water resource. Stressors are expressed over temporal and spatial scales ranging from a one-time, localized event to chronic exposures occurring continuously over vast landscapes. Pressures, stressors, and responses operate at different spatial and temporal scales (Figure 4-3). These are not independent of one another in either space or time; therefore, consideration of multiple pressures is essential. An additional consideration is that any given pressure creates multiple stressors, which in turn affect biological condition. The steady accumulation of small pressures in watersheds results in "cumulative impacts," which present added challenges for characterizing, evaluating, and managing stressors.

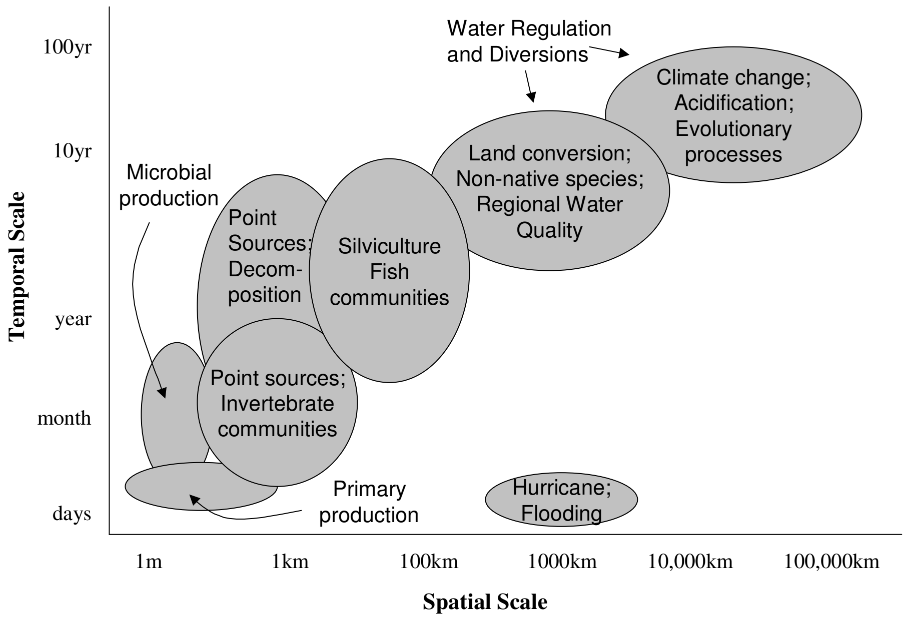


FIGURE 4-3. Perspective of scale for pressure-stressor-response variables (modified from Richards, C. and L.B. Johnson. 1998. Landscape perspectives on ecological risk assessment. In *Risk Assessment: Logic and Measurement*, M.C. Newman and C. Strojan (eds.). Ann Arbor Press.).

The complexity of the relationships between biological condition and stressors at various spatial and temporal scales, underscores the importance of using sound information to identify and link these stressors back to the pressures that cause them. To a large degree, this is the critical step in gaining stakeholder support for restoration and protection actions as well as for changes in activities or behaviors. As discussed earlier, fine sediment is commonly identified as a stressor across the United States because of the smothering of important habitat. Identifying the relative contributions of various sources of these sediments is more challenging (e.g., bank erosion, upland erosion, spatial sources), but also critical to remediation efforts.

4.4 How a GSG can be developed and calibrated

Developing and calibrating a stressor gradient must be based on appropriately classifying aquatic resources and establishing reference conditions or other scientifically defensible approaches. Classification (e.g., biogeographic regions, basins, biological considerations) is a critical first step so that the temporal and spatial scales of the dominant stressor categories and sources can be addressed (Herlihy et al. in press, VanSickle and Hughes 2000, McCormick et al. 2000, Waite et al. 2000). Of equal importance is establishing the appropriate reference condition for a particular area (Hughes 1985, 1994; Hughes et al. 1986; Moss et al. 1987; Stoddard et al. in press), because that is the benchmark against which areas to be evaluated will be compared (as discussed in BCG Section 3.1.1).

Like the biological condition axis, the stressor axis is anchored in the natural, or undisturbed or minimally disturbed, condition (i.e., Tier 1 BCG). However, reference may represent minimally-disturbed (i.e., nearly natural) or least-disturbed (i.e., best available) conditions depending on the level of disturbance that exists across the geographic area of interest (Stoddard et al. in press, Hughes 1994). Linking regional factors, pressures, and stressors with biological condition into a BCG will assist States and Tribes in identifying levels of disturbance and the primary drivers of biological condition in their watersheds. If no undisturbed or minimally disturbed reference sites exist in a region, a stressor axis provides a means for determining the best condition or regional candidates to act as benchmarks for comparison, i.e., “least disturbed” or “best available conditions.” The stressor axis concept will enable managers to place the status of their stream ecosystems into a regional context and prioritize actions. The reference condition approach, which describes the potential biological condition of the region’s waters, provides a framework to set appropriate restoration endpoints for that resource and region.

The next step involves quantification of in-stream stressors, riparian condition, landscape characteristics and riverscape alterations, as well as point source discharges and other localized pressures. Calibrating stressors along natural gradients (waterbody size, catchment area, stream power, elevation, latitude, and geology) can improve ability to detect pressure effects by removing the confounding effects of stressor gradients with natural gradients (Fausch et al. 1984, Hughes et al. 2004, Kaufmann and Hughes in press). There have been many efforts to characterize pressures and incorporate quantitative information into environmental assessment programs (Table 4-3). Riparian condition has been widely recognized as affecting the physical habitat and biological condition of streams (Naiman and Decamps 1990, Fitzpatrick et al. 2001, Lammert and Allan 1999, and Lattin et al. 2004). In some circumstances, watershed condition was more important (Roth et al. 1996, Snyder et al. 2003). Wilhelm et al. (unpublished manuscript) used both catchment (i.e., watershed) and riparian disturbance for the development of their non-wadeable habitat index for streams in Michigan. Wang and others (in press) found that fish assemblages were most influenced by local environmental factors in largely undisturbed catchments. However, as the level of catchment disturbance increased, the importance of catchment-scale factors increased and that of local-scale factors decreased. These studies indicate how important regional and local factors are for determining the relationship among sources, stressors, and biological condition and the most appropriate scale for addressing these relationships.

TABLE 4-3. Percent variance in biological response (R^2) explained by catchment and riparian land use, and percent land use producing poor IBI scores (modified from Hughes et al. unpublished manuscript).

Authors	Response Variable	R^2 Catchment	R^2 Riparian	N	Location & % Land Use for "Poor" rating
Bryce & Hughes (2002)	Fish IBI	0.40	--------	13	OR/ 50% urban
	Fish IBI	0.35	--------	16	Appalachia/ 15% urban
	Diatom IBI	0.29-0.36	--------	16	App./ declines w/ ag.
	Benthos IBI	0.48-0.67	--------	16	App./ 50% ag., 20% mined
Fitzpatrick et al. (2001)	Fish IBI	0.31	0.58	25	WI/ 70% ag.
	Diatom IBI	0.16	ns	25	WI/ag.
	Benthos IBI	ns	ns	25	WI/ag.
Hughes et al. (unpublished)	Fish IBI	0.42	0.38	104	OR/ rd. density >1.9 km/km2
Karr & Chu (2000)	Benthos IBI	0.25	---------	66	WA/ 40% impervious
Klauda et al. (1998)	Fish IBI	0.68	---------	61	MD/ 60% urban
Lammert & Allan (1999)	Fish IBI	0.01	0.22-0.28	18	MI/ declines w/ riparian ag.
	Benthos IBI	ns	ns	18	MI/ag.
Lattin et al. (2004)	Fish IBI	ns	0.20-0.46	25	OR/20% network riparian ag.
Leonard & Orth (1986)	Fish IBI	0.60	---------	44	WV/ rd. density >1.7 km/km2
McCormick et al. (2001)	Fish IBI	0.05-.08	---------	313	App./ declines as deforested
Mebane et al. (2003)	Fish IBI	0.45-0.56	---------	41	OR/ 25% deforested
	Fish IBI	0.56	---------	30	ID/ 15% irrigated ag.
Morley & Karr (2002)	Benthos IBI	0.53	0.00-0.82	34	WA/ 45% impervious
Roth et al. (1996)	Fish IBI	0.50	0.02-0.38	21	MI/ 80% ag.
Snyder et al. (in press)	Fish IBI	0.16-0.64	0.02-0.17	20	WV/ 15% urban
Steedman (1988)	Fish IBI	0.64	0.67	10	ONT/ 95% ag., 60% urban
Wang et al. (1997)	Fish IBI	0.48	---------	134	WI/ declines w/ deforesting
Wang et al. (2000)	Fish IBI	0.34	---------	47	WI/ 5% impervious
Wang et al. (2001)	Fish IBI	0.04-0.31	0.26-0.34	47	WI/ 5% impervious
Yoder et al. (2000)	Fish IBI	0.41	---------	101	OH/ 30% urban

Once the suite of stressors and pressures are measured or quantified for a given group of waterbodies, the next step is to determine if more than one stressor gradient exists and how they are related (i.e., are there several gradients based on different pressures, activities or landscapes?). Dealing with these multiple stressors and pressures can be complicated. A direct multiple correlation approach was taken by EMAP in the mid-Appalachian Highlands where poor quality streams were most often associated with alien fish, channel sedimentation, and riparian habitat alteration out of several hundred possible stressors (U.S. EPA 2000a). Kaufmann and Hughes (in press) used correlation and multiple linear regression analyses to determine that low stream IBI values were associated with excess streambed fines, bed instability, higher water temperature, higher dissolved nutrient concentrations, and lack of deep pools and cover complexity. These stressors were most strongly associated with riparian disturbance and road density. Effects were more pronounced in streams draining erodible sedimentary bedrock than in those draining more resistant volcanic terrain. States and Tribes could use similar multivariate approaches for identifying the stressor(s) most associated with measures of biological condition in their regions

A method employed in the Great Lakes Environmental Indicators (GLEI) project to characterize disturbance to the U.S. Great Lakes coastal region, used principle components analysis to reduce over 200 GIS variables into a single gradient (Danz et al. 2005). The GLEI approach individually considered six different kinds of disturbance: agriculture, atmospheric deposition, land cover, human population, point sources, and shoreline alteration. A watershed-based approach was used to reflect the premise that the environmental effects of these activities in coastal watersheds can influence environmental conditions in (downstream) coastal ecosystems. The first principle component from their analysis explained 73% of the variance in the agriculture variables and was interpreted as an overall gradient in stressors across the basin (Figure 4-4). Environmental responses such as water quality, fish assemblage metrics, and bird abundances were strongly correlated with this stressor gradient.

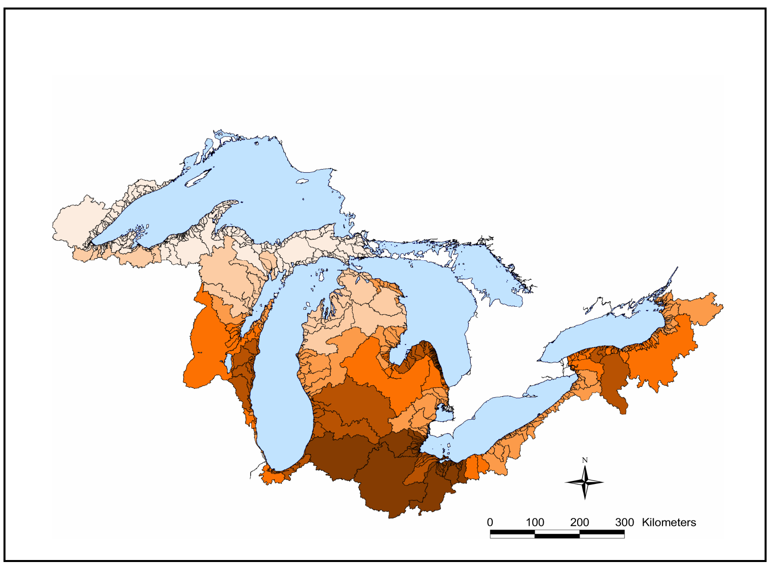


FIGURE 4-4. The first principal component of the agricultural variables for the U.S. Great Lakes basin. Darker shading indicates greater amounts of agriculture.

When multiple sources and stressors interacted to form the stressor gradient for a given watershed, GLEI found it desirable to develop a visual display of PCA axis 1 that subsumes the multiple stressors by portraying a single disturbance gradient. While the pressure-stressor model could eventually be developed and visualized as a single gradient from low to high levels of stressors (Figure 4-4), different individual and combinations of stressors are expected to dominate in different regions. Furthermore, the depiction of individual categories of stress provides important information about potential mechanisms affecting the state of the system. The GLEI researchers created a flow diagram (Figure 4-5) that details their steps for quantifying a stressor gradient (modified from Danz et al. 2005).

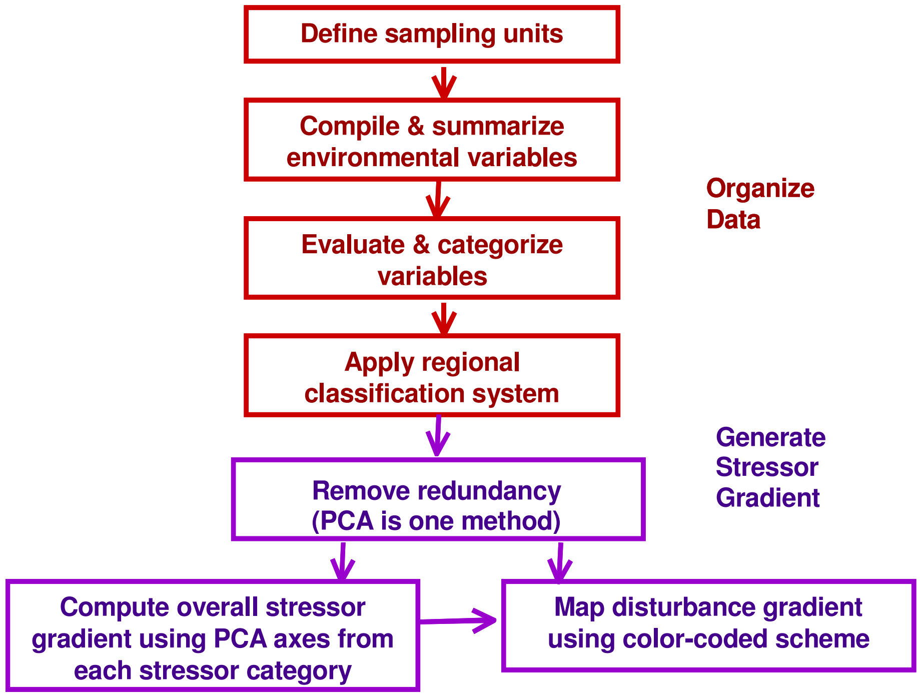


FIGURE 4-5. Flow diagram detailing the steps used by GLEI researchers in quantifying their stressor gradient (modified from Danz et al. 2005).

Whether using a single or multiple stressor gradient, all this information needs to be assembled to develop a model that integrates the components of pressures and establishes a baseline for using stressors to interpret biological responses. Relationship models that describe the associations among stressors, the processes that generate them, and biological conditions (responses) need to be developed. If possible, the extent of management actions (e.g., controls/BMPs) needs to be identified and ways to characterize these actions need to be considered (although this is an area of active research). The degree of deviation from natural conditions and the types of stressors present will affect restoration potential and therefore BMP effectiveness. Examples of tools that are currently available for characterizing a suite of pressures are: Analytical Tools Interface for Landscape Assessments (ATtiLA), National Land Cover Database (NLCD), and air photos.

Calibrating a stressor axis depends on the scale of the question to be addressed. The stressor axis should be developed independently of the biological information to avoid circularity when developing the BCG. In the development of their non-wadeable habitat index (NWHI), Wilhelm et al. (unpublished manuscript) used catchment and riparian disturbance gradients (CDG and RDG respectively) to select and weight habitat metrics at both watershed and reach scales. While the final NWHI was strongly correlated to disturbance measures and included habitat metrics that supported this relationship, a true test of the relationship between their stream response measure and disturbance measures would require a new, independent data set. The GLEI researchers used a wide range of publicly available data sets to quantify five different classes of disturbances. Their stressor axis is currently being calibrated. Stressor development and calibration involves using sufficient information to characterize relative positions along the axis and, in particular, being able to anchor the upper end (i.e., low or no stressors) and the lower end (i.e., high level of stressors) (Whittier et al. in press). This can be accomplished via a combination of public consensus, best professional judgment, and empirical approaches (e.g., Areas Of Concern (AOC), Great Lakes Environmental Indicators (GLEI) approach, and index development) (Whittier et al. in press, Danz et al. 2005, U.S. EPA 2000b).

4.5 Key points from Chapter 4

1. The stressor gradient provides a framework for organizing and interpreting information about watershed characteristics and using those characteristics to predict aquatic ecosystem biological responses. It helps us understand the observed biological conditions and the stressors related to those conditions. It can help identify the predominant stressors affecting the aquatic biota and develop effective management actions to mitigate their effects.

2. Understanding how specific stressors are generated and how they affect biotic condition provides the underpinnings for the BCG's stressor axis and ultimately the basis for interpreting the influence of stressors on biological condition.

Incorporating Tiered Aquatic Life Uses Into State and Tribal WQS: Case Examples

Tiered aquatic life uses are descriptive narratives of designated uses that are supported with numeric biocriteria and chemical and physical criteria.

As a key component of State and Tribal water quality standards, designated uses define the goals for a waterbody, determine the criteria to protect it, guide management outputs, and, ultimately, environmental outcomes. Aquatic life tiers couple descriptive narratives (tiered uses) with supporting numeric criteria. The specificity of designated uses greatly influences the level of precision at which a water quality management program operates. Incorporating tiered aquatic life uses into water quality standards can have a positive effect on water quality management outcomes. States that have made this transition have demonstrated that tiered aquatic life uses promote both the development of more appropriate aquatic life use goals and biological criteria to measure attainment of those goals. The data and experience developed from tiered uses supported by comprehensive monitoring have multiple uses in the water quality based approach to pollution control (Figure 5-1).

The preceding chapters of this document describe ways of better characterizing and defining the biological and physical condition of waterbodies and their aquatic life uses. These next two chapters discuss the underlying principles and processes involved in developing tiered aquatic life uses and applying them in water quality management based on "lessons learned" from State experiences. Maine and Ohio are two States that have adopted tiered aquatic life uses in their WQS and have implemented them through systematic monitoring and assessment. The experiences of Maine and Ohio provide a sequence of steps, or milestones, that can serve as a template for other States to follow. These milestones are:

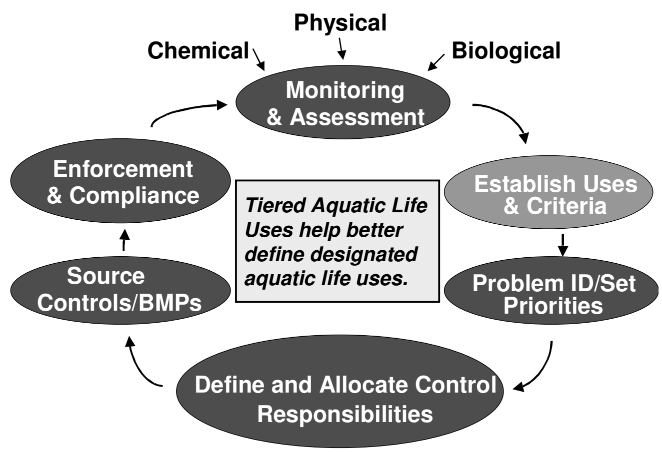


FIGURE 5-1. U.S. EPA Water Quality Based Approach to Pollution Control based on Chapter 7, Water Quality Standards Handbook.

1. **Establish conceptual foundation**
2. **Merge scientific and policy foundations**
3. **Establish monitoring program**
4. **Develop and validate quantitative thresholds**
5. **Apply tiered uses in water quality management**

Both States developed tiered aquatic life uses for similar reasons: 1) to incorporate ecologically relevant outcomes in goal setting; 2) to guide cost-effective, defensible management decisions; 3) to measure incremental progress in meeting management goals; and 4) to merge the design and practice of monitoring and assessment with the development and implementation of WQS. Chapter 5 captures the "lessons learned" by Maine and Ohio in their development of tiered uses (Milestones 1 – 4) and Chapter 6 presents case examples about how each State has benefited from this approach (Milestone 5).

CHAPTER 5. KEY CONCEPTS AND MILESTONES IN THE DEVELOPMENT OF TIERED AQUATIC LIFE USES

Tiered aquatic life uses should be derived based on knowledge of the aquatic biota (specifically the assemblages used in biological assessments) and the factors that determine their distribution, abundance, and composition. Tiered narrative statements and numeric biological criteria can represent measurable benchmarks along a regionally calibrated biological condition gradient (BCG). Maine and Ohio have determined that these benchmarks represent attainable conditions in their States for the protection or restoration of surface waters through implementation of WQS. Since waterbodies are assigned to tiered uses based on a comprehensive ecological database (biological, chemical, physical assessments), Maine and Ohio are more confident that both the uses themselves and any changes to a waterbody's condition are ecologically relevant.

5.1 Key concepts for developing tiered aquatic life uses

Maine and Ohio's tiered aquatic life uses represent the goals for individual waterbodies. Their tiered uses share the following common characteristics:

- uses are ecologically-based
- uses include the structural and functional properties of the specific aquatic communities that inhabit an aquatic ecosystem
- attainment is based on measurable biological criteria, which are indexed to a regionally relevant reference condition
- implementation integrates monitoring, assessment, and WQS

These characteristics are discussed more fully in the Maine and Ohio case histories (Appendixes A and B). But, two key concepts that Maine and Ohio have learned are:

1. Tiered Aquatic Life Uses Should Be Ecologically Based
Tiered uses should be built on a strong ecological foundation that provides a credible basis for the protection and restoration of aquatic resources. Tiered uses should reflect the collective attributes of the BCG and encompass the structural and functional attributes and processes of an aquatic ecosystem. As discussed previously, Figure 1-4 illustrates how pollution leads to exposures and responses, both ecological and human health, which can affect the status of waterbody-specific designated uses. Because the designated use is initially stated in narrative and qualitative terms, the challenge is to logically and appropriately relate the chemical, physical, and biological criteria to the designated use. The more precise the statement of the designated use, the more accurate the associated criteria can be as an indicator of that use.

Linking tiered uses to a regionally calibrated BCG provides the scientific framework for determining the biological condition and potential of individual waterbodies, which is the basis for assigning the appropriate tier. Tiered use narratives should include explicit references to the protection of aquatic life and specify the structural and functional properties that are to be protected. The derivation and calibration of numeric biocriteria should assure ecological relevance consistent with the properties of the regional aquatic fauna.

2. Linkage of Tiered Uses to the BCG via Biocriteria
The BCG attributes are incorporated directly into tiered aquatic life uses through biological assessments and with biological criteria. The development of biocriteria is an important part of the process in accomplishing this task and should adhere to the technical components of the overall TALU process

(Appendixes C, D, and E). Karr et al. (1986) recommended six key elements in the development of bioassessment tools and biocriteria:

1) measure(s) must be biological
2) measure(s) should be interpretable at different trophic levels and provide a connection to other organisms and assemblages not included in the biological assessment process
3) measure(s) must be sensitive to the environmental conditions being assessed
4) response range must be suitable for the intended application, i.e., encompassing the full range of the BCG
5) measure(s) must be reproducible and precise within acceptable limits for data collected over space and through time
6) variability of the measure(s) must be low enough to detect changes along the entirety of the BCG

Representative indicator assemblages are used to measure attainment of the biocriteria as part of the derivation process. As such, biocriteria represent the measurable ecological properties of a tiered aquatic life use.

5.2 Key milestones for developing tiered aquatic life uses

The Maine and Ohio case histories (Appendixes A and B) reveal conceptually consistent, but technically different ways of developing tiered uses including numeric biological criteria and a comprehensive monitoring and assessment program. However, the process followed by each demonstrates common tasks and milestones that States and Tribes can use as a template for developing tiered uses. These milestones and tasks are illustrated in Table 5-1 and consist of five major steps:

Milestone 1. Establish Conceptual Foundation *(Maine and Ohio Case Histories, part I)*

- Establish an interdisciplinary, collaborative approach to the development of tiered uses (ecological, technical, and legal)
- Identify and acquire appropriate staff and management expertise

Milestone 2. Merge Scientific & Policy Foundations *(Maine and Ohio Case Histories, part II)*

- Link management objectives with technical program
- Evaluate for consistency with existing water quality standards framework
- Draft or refine narrative aquatic life use descriptions

Milestone 3. Establish Monitoring Program *(Maine and Ohio Case Histories, part III)*

- Develop methods and monitoring design, establish reference conditions, build baseline database and database management system
- Logistics: staffing, facilities, and equipment

Milestone 4. Develop/Validate Quantitative Thresholds *(Maine and Ohio Case Histories, part IV)*

- Program implementation: develop biocriteria and water quality program support (initiating the process of using TALUs and biological assessments to support water quality management tasks)
- Validate the accuracy of ecological expectations with empirical data
- Program maintenance: refine biocriteria and maintain water quality program support (maintaining the process of using TALUs and biological assessments including the continuous evaluation of tools, criteria, and processes based on what is being learned via a systematic approach to monitoring and assessment; includes expansion to other aquatic ecotypes)

Milestone 5. Application in Water Quality Management *(Chapter 6; Maine Case History, part IV)*

- Apply biocriteria to support WQS
- Integrate tiered biocriteria with other types of chemical and physical criteria

Milestones 1 - 4 describe the key tasks in the development of tiered aquatic life uses. Milestone 5 addresses the application of tiered uses in water quality management. Ideally, the milestones can be accomplished sequentially, each laying the appropriate scientific or policy foundation for the next step. However, many States will have already accomplished some or even a majority of the tasks, particularly under Milestone 3 (Establish Monitoring Program). Some may also use biological assessments for support functions beyond status assessments, but perhaps lack the formal tiered use framework in their WQS or have remaining technical development issues. Maine and Ohio found that capacity for conducting biological assessments is an equally important issue and generally included 5-10% of State water quality management program resources. They found that this level of funding should make available sufficient resources to carry out the development, maintenance, and assessment tasks on a statewide basis.

Table 5-1 and Figure 5-2 include many of the major tasks in the development of a program and they can serve as a “road map” to determine where a particular State program stands regarding the goal of developing and applying tiered uses in its water management programs. Figure 5-2 can also be used as a guide for identifying, prioritizing, and organizing outstanding and remaining tasks. Furthermore, there is a transition under Milestone 4 from an emphasis on development of a tiered aquatic life use approach to program maintenance. Program maintenance includes ongoing evaluation and “fine tuning” of the bioassessment tools and criteria as the program matures. It also includes the further development and refinement of assessment and management tools and criteria as data, experience, and knowledge are gained via systematic monitoring and assessment. Maine and Ohio initially developed tiered uses and biocriteria for streams and wadeable rivers and currently either have developed or are evaluating tiered uses and biocriteria for other waterbody types (e.g. nonwadeable rivers, wetlands, lakes and estuaries). Program maintenance can also include the development of tiered uses for these other types of waterbodies. Evaluating whether there is a need to change existing use designations for specific waterbodies is another important task. This is accomplished during the triennial review process with decisions based directly on outcomes from systematic watershed monitoring and assessment and historic data.

Milestones 1 - 4 and Figure 5-2 reflect a sequence of strategic steps in the development of tiered aquatic life uses. A functional and effective program will emerge if essential theoretical, technical, and legal elements are addressed and fully integrated throughout the development process. Table 5-1 shows typical tasks associated with each founding element and the type of professional expertise required to accomplish them. One of the key “lessons learned” in Maine and Ohio is that problems arise when technical and management activities are done in isolation from each other. A collaborative and interdisciplinary approach that blends technical and management activities yields better decisions at all levels.

The triennial review process is readily adaptable to developing and then refining uses on a watershed basis, and to making needed adjustments to bioassessment tools and criteria. As the program develops and matures over time, and as resources become available, application of a tiered use framework can advance from condition assessment to formal incorporation into water quality standards.

TABLE 5-1. Expertise and tasks for key TALU milestones.

Conceptual Foundations	**Technical Foundations**	**Policy/Legal Foundations**
Professional Expertise Required		
✓ Senior professional biologists ✓ Regional ecological experts	✓ Professional biologists ✓ Taxonomists ✓ Field support staff ✓ Statistician ✓ Database managers	***Initial concept formulation:*** ✓ Senior professional biologists ✓ WQS managers ***Later stages***: All of the above plus… ✓ Senior management ✓ State legal counsel ✓ Legislature or WQS board ✓ Stakeholders
Milestones 1, 2 and 4	**Milestones 3 and 4**	**Milestones 1, 2 and 4**
Essential Elements		
▪ Literature review of stress ecology studies for locale ▪ Develop regional BCG model ▪ Determine expected biological assemblage response to typical stressor scenarios; ▪ Identify ecological attributes necessary to maintain a functioning ecosystem (to help establish goals for protection or restoration)	▪ Clarify classification issues (confounding natural gradients of locale); ▪ Define reference conditions ▪ Determine monitoring approach and strategy ▪ Exploratory data analyses to validate/refine BCG model ▪ Best available, best tested metrics to assess status of ecological attributes of interest ▪ Set thresholds that correspond to BCG tiers, that protect essential ecological attributes	▪ Determine management objectives; ▪ Identify priority aquatic resources ▪ Cross-walk BCG to WQS context- (how good a fit is provisional BCG/TALU conceptual model to existing use classes and WQ criteria) ▪ Seek early review of the legal standing of any proposed changes to WQS- strengthen and clarify language ▪ Account for public values and economic constraints/realities

Based on the commonalities between Maine and Ohio's experiences, several important "lessons learned" were identified for States and Tribes that are considering developing tiered aquatic life uses.

- **Interdisciplinary approach to development:** Development of tiered aquatic life uses is most successful when active cooperation and close working relationships exist among the individuals charged with technical/scientific development and oversight of water quality standards.
- **Plan enough to be certain of success… and use adaptive management approach:** Clear knowledge of scientific and legal principles should guide every step of planning and development. An adaptive management approach is beneficial throughout the development process because new technical information and management understanding are gained as part of the process. An adaptive management approach incorporates needed flexibility into a program by building on the new knowledge and insights.
- **"Proper" sequencing versus logical decisions:** The exact sequence of developmental events is not as critical as the necessity of following a plan that is logical for a particular State or Tribe, builds on current program strengths and reflects rigorous adherence to scientifically and legally sound foundations.
- **Graduated application to support water quality management decisions:** Some level of condition assessment and regulatory decision-making (application in water management) can happen as soon as a credible monitoring program is established and linked to narrative TALU goal statements.

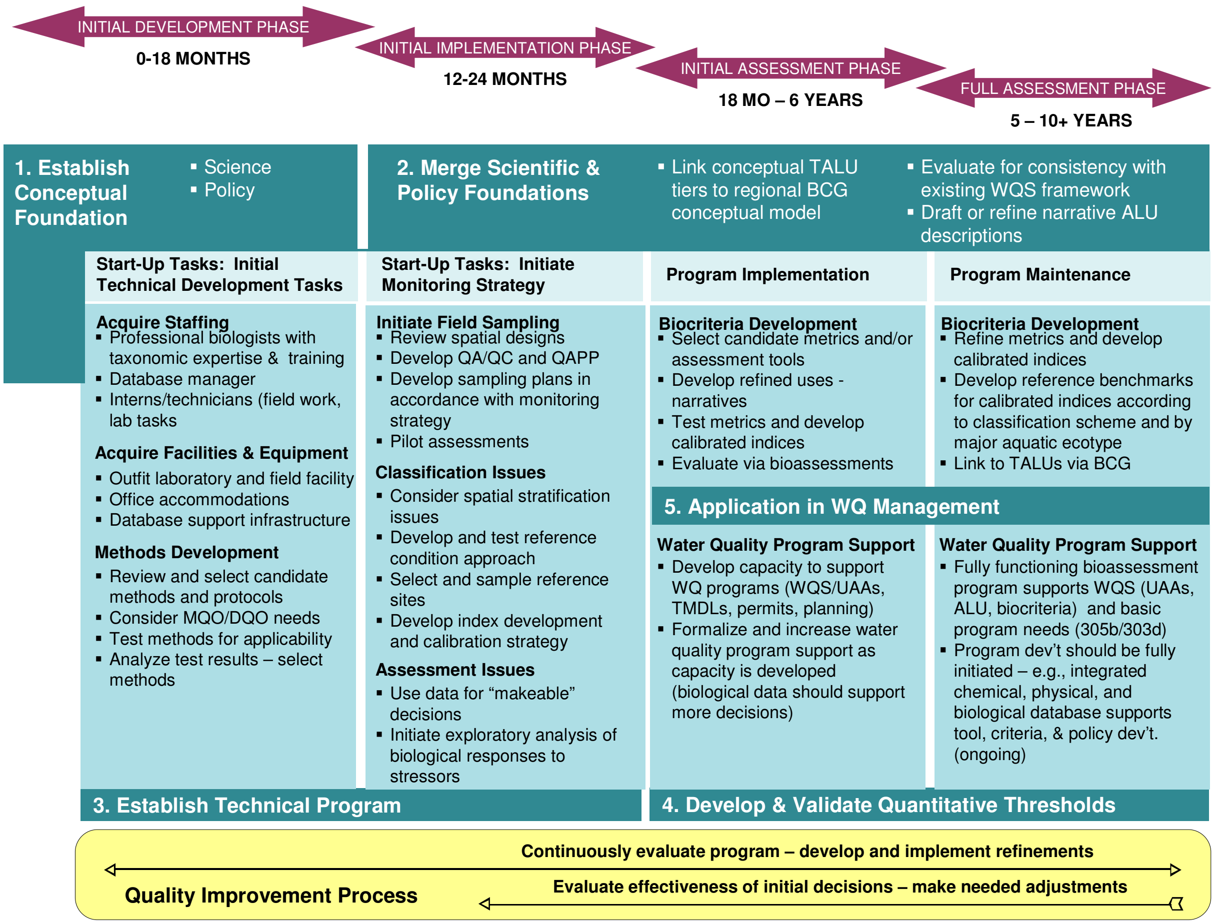


FIGURE 5-2. TALU and biocriteria program development tasks: Timeline and key milestones. A process of sequential tasks and milestones that States can follow in the development and implementation of tiered aquatic life uses and attendant biological criteria.

5.3 Using TALUs to support water quality management

The adoption of tiered uses should positively influence water quality management outputs and outcomes. Tiered uses in State and Tribal water quality standards, coupled with a systematic and comprehensive monitoring and assessment program, can provide an essential link among a wide variety of water quality management programs. In Maine and Ohio, the end result have supported baseline CWA management programs such as NPDES permitting, construction grants, and, more recently, the revolving loan program, basin planning (including TMDLs, listings of impaired waters, development of restoration plans), and nonpoint source assessment. The comprehensive support of water quality management that emerges from systematic monitoring and tiered aquatic life uses in Maine and Ohio is made possible by following the milestones shown in Table 5-1 and Figure 5-2 to establish and develop a program. Monitoring supports day-to-day water quality management needs and can take place at multiple scales including a statewide, regional, watershed, or site-specific basis.

A sustained monitoring and assessment program naturally incorporates strategic functions and results in improved criteria, tools, policies, awareness, and legislation. The aggregated database comprises the experience gained by conducting systematic assessments and includes the regular resampling of reference sites and long-term monitoring of reference condition. The database allows comprehensive analysis and interpretation of spatial and temporal trends and tracking the effectiveness of different water quality management programs. The overall program thereby fosters continuous improvement through adaptive management because the relevant information and the interpretation of that information is made available to managers.

As an example, full documentation of the results and benefits of improvements in wastewater treatment on multiple waterbodies in both Ohio and Maine would not have been possible without a comprehensive biological monitoring network and tiered uses to put the results into a communication and management context *(See Case Example 6-4. Long-term Monitoring and Use Re-establishment in Maine).* Furthermore, tiered uses allowed the two States to secure and retain the gains made by upgrading some of the affected rivers to higher tiers, a development that had not been anticipated before the wastewater treatment was improved. These examples also validated the process of setting TALU-based WQS and using them to develop regulatory requirements. The outcomes allayed many of the original uncertainties about the cost-effectiveness of water quality based permitting and gave regulatory programs the confidence to implement new requirements. This was critical in Ohio where the virtues of municipal wastewater treatment more stringent than secondary treatment were widely debated and doubted in the early 1980s. Advanced treatment (also known as best available demonstrated control technology or BADCT) is now widely supported because not only did it work as a treatment technology, but it delivered the end outcome of improved biological condition.

The comprehensive, long-term programs in Ohio and Maine have demonstrated their value by improving prioritization of management actions and enabling more effective targeting of resources. Chapter 6 summarizes several case examples of how biological monitoring and tiered uses contribute to many different aspects of the water quality management cycle (Figure 5-1).

5.4 Key points from Chapter 5

States that have successfully implemented a TALU approach have found that:

1. The specificity of designated uses greatly influences the level of precision at which a water quality management program operates. Incorporating more refined, or tiered, aquatic life uses into water quality standards can have a positive effect on water quality management outcomes. States that have made this transition have demonstrated that tiered aquatic life uses promote both the development of more appropriate aquatic life use goals and biological criteria to measure attainment of those goals.

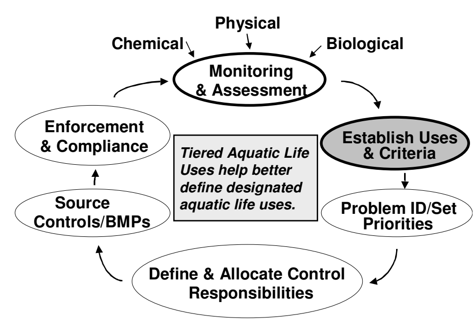
2. Tiered uses in State and Tribal water quality standards, coupled with a systematic and comprehensive monitoring and assessment program, can provide comprehensive support to water quality management programs. In Maine and Ohio, the end result supports baseline CWA management programs such as NPDES permitting, construction grants, and, more recently, the revolving loan program, basin planning (including TMDLs, listings of impaired waters, development of restoration plans), and nonpoint source assessment.

3. Though based on different technical approaches, their development of tiered aquatic life uses followed common tasks and milestones. Development of tiered uses has been most successful when there was early and consistent collaboration among their monitoring, criteria, and standards programs.

CHAPTER 6. HOW HAVE STATES AND TRIBES USED TALUS IN WATER QUALITY STANDARDS AND MANAGEMENT?

Tiered aquatic life uses supported by systematic assessments can provide the information needed for water quality management at watershed, regional, and statewide scales. A comprehensive monitoring and assessment program is a critical aspect of implementation of tiered aquatic life uses. The same data and information that provide baseline status assessments also address watershed-specific management needs such as the appropriate designation of individual waterbodies, TMDL development, and NPDES permits. This chapter presents several case examples in Maine and Ohio of how tiered uses and monitoring contribute to all aspects of the water quality based approach to pollution control (Figure 5-1). These include setting criteria and standards; problem identification and establishing priorities (stressor identification); defining and allocating control responsibilities (source identification); determining source controls or BMPs (TMDLs, UAAs, WLAs); and enforcement and compliance (NPDES permits and other compliance agreements). The following are case examples of how TALUs, coupled with systematic monitoring and assessment, have and can be used to support key water quality management programs and functions. These examples further exemplify what can be accomplished by following the developmental process described in Chapter 5. Accompanying each case example is a diagram of U.S. EPA's Water Quality Management Cycle (Figure 5-1) with the key component for that particular example shaded. Most of the following examples were accomplished during the Program Maintenance phase of the TALU development milestones (Figure 5-2) and demonstrate what can be produced as the bioassessment program matures; however, some of the initial assessments can be accomplished during the Program Implementation phase.

CASE EXAMPLE 6-1. REFINING WATER QUALITY CRITERIA IN OHIO



Ohio EPA developed empirical associations between aquatic life and ambient stressor levels for parameters such as ***dissolved oxygen*** from its monitoring program data beginning in the late 1970s. The known prevalence of organic enrichment from point sources and intensive watershed surveys identified dissolved oxygen (D.O.) as a major stressor limiting aquatic life throughout the 1980s (Ohio EPA 1988, 2000).

When the Exceptional Warmwater Habitat (EWH) aquatic life use was established in 1978, Ohio also established tiered dissolved oxygen criteria to protect "highly sensitive aquatic organisms; growth and reproduction of recreationally and commercially important species; [and] maintenance of populations of imperiled species" (Ohio EPA 1996). This was in contrast to the goal for the Warmwater Habitat (WWH) use, which was the "maintenance of typically representative warmwater aquatic organisms and recreationally important species" (Ohio EPA 1996). The original single criteria for EWH streams of 6 mg/l was largely based on pertinent literature of the time, best professional judgment using the knowledge that these streams supported populations of very sensitive aquatic species, and that the D.O. criteria should be more stringent than the WWH criterion (5 mg/l daily average, 4 mg/l minimum).

Since the original adoption of the EWH use and associated tiered D.O. criteria, analyses of ambient biological and chemical data suggested that the 6 mg/l minimum criterion was over-protective for these waters. Both statewide and reach specific data were used to document streams with dissolved oxygen concentrations below 6 mg/l (but typically above 5 mg/l) that fully attained the EWH aquatic life use as measured by the numeric biocriteria. These results were used to justify a two-number criterion of 6 mg/l average, 5 mg/l minimum for the EWH use (Ohio EPA 1996). Two examples of these data include the stressor-response relationship between grab sample D.O. data (Figure 6-1) and continuous D.O. data (Figure 6-2) and the IBI in the E. Corn Belt Plains (ECBP) and Huron/Erie Lake Plain (HELP) ecoregions of Ohio. Both graphs show an expected gradient of response between D.O. and IBI scores and show that minimum dissolved oxygen values between 5 and 6 mg/l were commonly associated with IBI scores in the EWH range.

Figure 6-1 illustrates a relationship that is commonly observed between stressors and biological measures where multiple stressors are prevalent. On Figure 6-1, to the left of the dashed line at 5.0 mg/l (grab samples), numerous D.O. values are found associated with low IBI scores, but very few at IBI scores above 50 (EWH). If D.O. is >5.0 mg/l, IBI scores are much more likely to attain WWH (>40) and EWH (>50). Figure 6-2 shows continuous D.O. data vs. IBI ranges that correspond to quality tiers ranging from exceptional to very poor. This also supports a similar conclusion as Figure 6-1, but captures the full range of D.O. values that occur over a 24-hour period, especially the early morning hours when the diel cycle yields the lowest values.

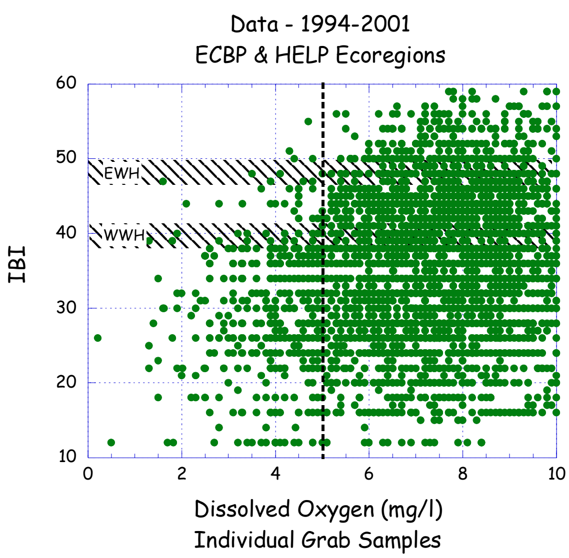


FIGURE 6-1. Dissolved oxygen concentrations (individual grab samples) vs. Index of Biotic Integrity (IBI) values in the Huron/Erie Lake Plain (HELP) and E. Corn Belt Plains (ECBP) ecoregions of Ohio. Hatched areas represent Exceptional Warmwater Habitat (EWH) and Warmwater Habitat (WWH) biocriteria for the ECBP ecoregion.

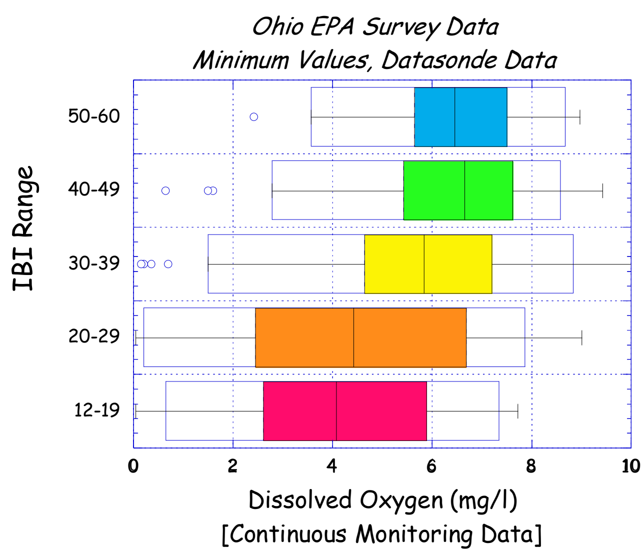
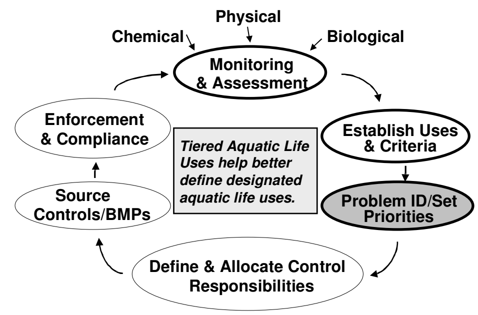


FIGURE 6-2. Box plots of minimum dissolved oxygen concentrations by IBI ranges for continuous monitoring data at all locations monitored in 1988 and 1994. IBI ranges are: 50-60 (exceptional, EWH); 40-49 (good, WWH), 30-39 (fair); 20-29 (poor); 12-19 (very poor).

The key message of this case example is that water quality criteria can be refined to reflect aquatic life use tiers if sufficient ambient data exists over sufficient spatial and temporal scales. It also provides more confidence in applying the water quality criterion as a design target for permitting and TMDL purposes. The previous EWH D.O. criterion (6 mg/l minimum) became a disincentive to redesignate rivers and streams that were fully attaining the EWH biocriteria because of the difficulty in meeting the permit limits. The criterion revision, based in part on the analyses presented here, resolved that situation in the majority of cases and allowed for the redesignation of such rivers and streams to EWH.

CASE EXAMPLE 6-2. DEVELOPMENT OF MORE PRECISE TARGETS FOR RESTORATION IN OHIO



Nutrients have been identified as a major stressor to aquatic life across the U.S. (U.S. EPA 2002b). Nutrients are not directly toxic under most conditions, but rather exert their influence on higher organism groups via interactions within energy pathways and by influencing D.O. dynamics within streams and rivers. Ohio EPA described biological gradients of response to nutrient concentrations in streams and rivers (Ohio EPA 1999a). This was accomplished by linking the primary nutrients (nitrate, total phosphorus) and other parameters to the biocriteria (IBI, ICI, etc.) on a statewide, ecoregion, and stream/river size basis. Thus ranges of these parameters consistent with attainment of the tiered aquatic life uses were accomplished (Ohio EPA 1999a; Table 6-1). While the values in Table 6-1 are not explicit water quality criteria, they are used as TMDL targets given the direct linkage they have with aquatic life use attainment. In addition to ambient fish and invertebrate data, ambient chemical data, and stream habitat data, Ohio is currently collecting information on chlorophyll and algal assemblages to improve understanding of the mechanisms of nutrient impact on aquatic life (Bob Miltner, Ohio EPA, personnel communication). This work should result in refined targets that can be used to determine which restoration activities should be most effective at restoring aquatic life. The identification of nutrient targets for each aquatic life use tier provides an appropriate and achievable level of protection for specific waterbodies. This application provides restoration targets for TMDLs that, if achieved, should result in full attainment of aquatic life uses.

TABLE 6-1. Statewide total phosphorus targets (mg/L) for Ohio rivers and streams.

Watershed Size	**Aquatic Life Use**		
	EWH	WWH	MWH
Headwaters (drainage area <20 mi^2)	0.05	0.08	0.34
Wadeable rivers (20 mi 2 <drainage area <200 mi	0.05	0.10	0.28
Small rivers (200 mi 2 <drainage area <1,000 mi	0.10	0.17	0.25
Large rivers (drainage area >1,000 mi	0.15	0.30	0.32
EWH =Exceptional Warmwater Habitat; WWH =Warmwater Habitat; MWH =Modified Warmwater Habitat			

As for nutrients, Ohio does not have explicit ***habitat and sediment*** criteria in the WQS. However, targets for habitat and sedimentation outcomes were developed by demonstrating a relationship between specific good quality and poor quality attributes and their ratios. Unlike water quality parameters, single numeric criteria for habitat and sedimentation do not exist and are inappropriate because 1) there are complexities in identifying expected values or ranges of values for specific attributes, 2) the resultant effects on the aquatic biota are explained by aggregations of good (warmwater) and poor (modified; see HIMA in Table 6-2) habitat attributes, and 3) the spatial scale over which these stressors exert their effects on aquatic life includes multiple dimensions (Rankin 1995). Rather than generating tiered criteria for habitat and sediment attributes, Ohio has developed quantitative habitat and sediment targets for TMDLs based on regional stream types (e.g., low vs. high gradient) and stream-size dependent "dose-response" relationships with the numeric biocriteria associated with the tiered aquatic life uses (Rankin 1995). The Stillwater River TMDL (Ohio EPA 2004) in the E. Corn Belt Plains (ECBP) ecoregion is an example of how nutrient, sediment, and habitat targets ("criteria") were developed and used along with more traditional chemical criteria to direct TMDL development in the watershed (Table 6-2).

TABLE 6-2. Numeric targets for biological, habitat, and water quality parameters for the Stillwater River in western Ohio. From Ohio EPA (2004) TMDL report for the Stillwater River watershed. The targets and criteria vary in accordance with the tiered uses, which are resolved prior to impaired water delineations and TMDL development.

	Biological Criteria		Habitat Targets		Water Quality Criteria				Nutrient Targets		
Aq. Life Use	Min. ICI	Min. IBI	QHEI	HIMA[a]	Ammonia-N* Max	Ammonia-N* Mean	Dissolved Oxygen* Min	Dissolved Oxygen* Mean	TKN[b]	Nitrate[b]	TP[b]
MWH	22	24	45	≤3	7.3	1.2	3.0	4.0	4.0	3.0	0.30
WWH	32	36	60	≤1	7.3	0.8	4.0	5.0	1.0	1.0	0.08
EWH	42	46	75	0	4.5	0.8	5.0	6.0	1.0	0.5	0.05

[a]HIMA - High Influence Modified Habitat Attributes
[b]Target values are adopted from Ohio EPA (1999)
*Specific numeric water quality exist in OAC 3745- 1-07, Tables 7-3 through 7-8; target values are guidelines based on the 75^{th} percentile values of temperature (24°C) and field pH (8.1) from all samples collected during the 1999 Stillwater survey.
MWH = Modified Warmwater Habitat; WWH = Warmwater Habitat; EWH = Exceptional Warmwater Habitat

All of the targets in Table 6-2 were either wholly or partially generated based on responses between the parameters, biological assemblage data, and the tiered aquatic uses to which they are related. This is important because most of these parameters, habitat in particular, are not amenable to the traditional laboratory based derivation. When these parameters are altered from "naturally occurring" conditions, they can induce an adverse response for the biota, thus behaving as stressors. Targets for TMDLs or other restoration strategies would either be difficult to generate, or lead to potentially incomplete solutions without being ground-truthed in ambient data relationships and a tiered aquatic life use framework, the latter of which is typically associated with a stressor gradient based on habitat or landscape characteristics. Since many of the targets in Table 6-2 were generated directly from ambient stressor and response relationships, their interpretations are likely less ambiguous than a rote application of lab derived criteria, although causative associations may be weaker. This approach is consistent with a recommendation in the NRC TMDL report (NRC 2001) that criteria or targets be positioned as closely as possible to the designated use and that indicators representing the full causal chain of events from stress to exposure to response be used.

Understanding the role of habitat as an influence on the biological restoration potential for a waterbody may be one of the greatest values of tiered aquatic life uses coupled with a systematic assessment process. Habitat and landscape changes compose a common stressor gradient along which States and Tribes may derive tiered uses. Tiered uses provide a useful framework for evaluating restoration potential, prioritizing management actions, and allocating abatement resources.

CASE EXAMPLE 6-3. DETERMINING APPROPRIATE LEVELS OF PROTECTION IN OHIO

Hurford Run is a small stream located in an ***urban/industrial*** area (steel finishing, petroleum refineries) of Canton, Ohio that drains an area of 8.5 square miles (Figures 6-3, 6-4). The entire stream has been subjected to direct channel modifications from the 1900s up to the time of the study. During the biological surveys in the mid 1980s, the stream was severely impaired by chemical pollutants, so much so that some sites had no fish. Because of the severity of the impairment, the use attainability analysis (UAA) relied on the assessment of habitat quality by the Qualitative Habitat Evaluation Index (QHEI; Rankin 1995).

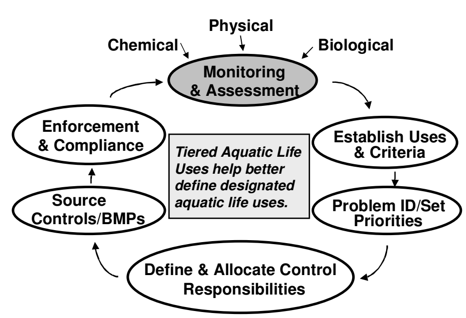


FIGURE 6-3. 1986 photograph of Hurford Run near Canton, Ohio looking upstream at the reach that is classified as a Limited Resource Water. Disturbed soil was caused by efforts to remove soils contaminated by nearby industrial operations.

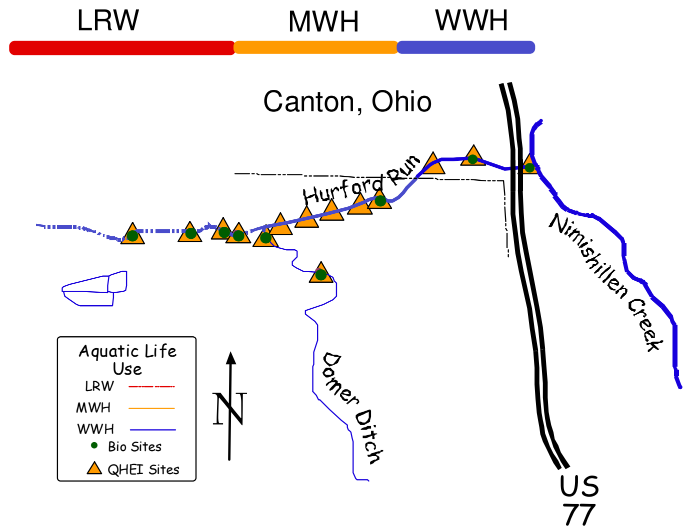


FIGURE 6-4. Map of Hurford Run near Canton, Ohio showing Ohio EPA IBI (solid circles) and habitat (QHEI, triangles) sampling stations. Spatial extent of stream aquatic life use designations is denoted along the top.

Established relationships between attributes of habitat as measured by the QHEI and levels of biological performance consistent with the tiered aquatic life uses provide an important tool to evaluate use attainability and assign appropriate uses to specific streams and rivers (Rankin 1989, 1995; Ohio EPA 1990). For example, Ohio has identified which habitat features may limit aquatic communities and which are predictive of streams with warmwater (WWH) and exceptional warmwater (EWH) biological communities. Figure 6-5 summarizes the IBI (left) and QHEI scores (right) for Hurford Run from 1985 to 1998. Very poor habitat quality from recent and historical channelization in the upper reach (RM 1.8 - 2.5) of Hurford Run and the associated hydrological characteristics (e.g., ephemeral flows) resulted in a Limited Resource Waters (LRW) designation for this upper reach. The middle reach beginning at the confluence of Domer Ditch (RM 1.7-1.0) was subject to extensive, maintained channel modifications and resulted in degraded habitat features (Figure 6-5, right), but water was always present. Channel maintenance practices resulting in poor quality substrates, undeveloped pools and riffles, and a lack of instream cover preclude biological recovery to assemblages consistent with the WWH use. Following a use attainability analysis (UAA), the middle reach was designated as Modified Warmwater Habitat (MWH), reflecting the biological restoration potential for a channel-modified stream.

The lower one mile of Hurford Run, although previously relocated and channelized, naturally recovered sufficient warmwater (good) habitat attributes such as coarse substrates and better developed riffle and pool features to achieve QHEI scores (>60-70) that are typical of the WWH use for this ecoregion, hence this segment was left at WWH. The tiered aquatic life uses that were assigned represent the highest attainable potentials given the existing level of sanctioned channel maintenance in this urban stream.

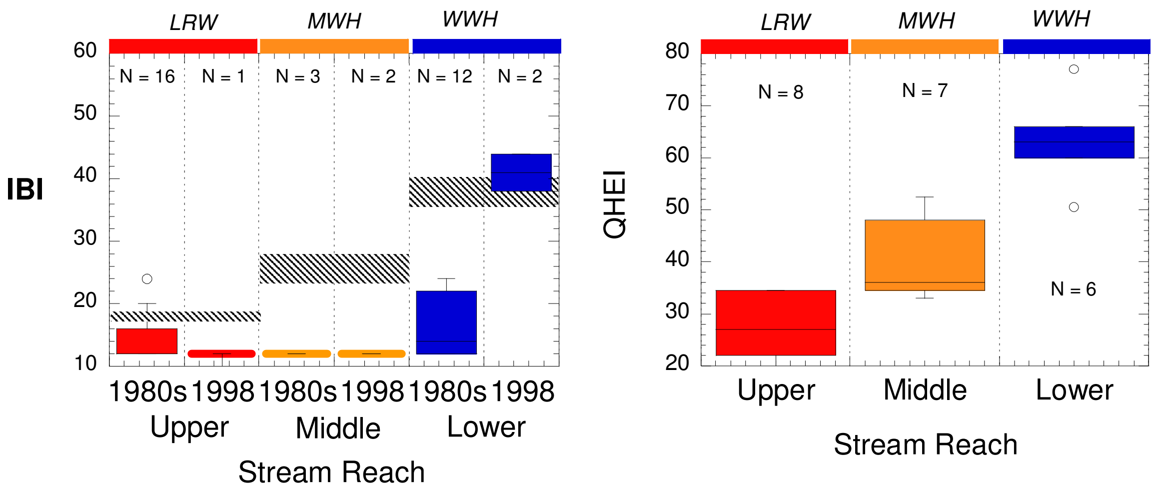
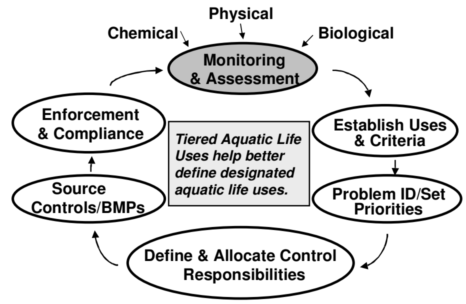


FIGURE 6-5. Box and whisker plots of IBI (left) and QHEI (right) by stream segment in Hurford Run near Canton, Ohio. Aquatic life use designations for segments are denoted along the top of each plot. 1998 data is separated from the 1980s data for the IBI, but data are combined for the QHEI. Data collected between 1985 and 1998. Lines are sites with no variability in scores (IBIs = 12). The hatched bars denote Ohio biocriteria for each tiered use.

All of the designated uses required additional abatement of the major point sources discharging to Hurford Run. Following the initial abatement of point source discharges in the late 1980s, data collected in 1998 demonstrated recovery of the IBI score near the mouth of the stream to the WWH biocriterion as predicted by the QHEI (Figure 6-5, left). Because this reach was designated WWH, it is protected from any further alteration below this quality. The MWH designated middle reach and LRW designated upper reach of Hurford Run have been subjected to ongoing channel maintenance activities (e.g., dredging, bank mowing), which has limited the amount of biological restoration that can be expected. However, even these less-than-CWA goal uses are impaired due to unresolved toxic impacts (reflected in very poor IBI scores; Figure 6-5, left) presumably from the point sources and/or legacy impacts associated with the industrial sites bordering the stream.

Urban/industrial streams such as Hurford Run present challenges in terms of setting and attaining restoration goals. Visually, the lower reach of Hurford Run may not exemplify the classic depiction of a natural stream because of its urban/industrial setting and location adjacent to major highways. The instream habitat, however, indicated a WWH potential, which was eventually verified as the effects of chemical stressors were reduced. The feedback provided by bioassessments based on the systematic collection of biological and habitat data, which is essential to using tiered aquatic life uses, is an important impetus for achieving water quality goals.

CASE EXAMPLE 6-4. LONG-TERM MONITORING AND USE RE-ESTABLISHMENT IN MAINE



Between 1974 and 1981, an estimated 33 million dollars was spent by industry, State, and federal sources to implement primary and secondary wastewater treatment technology on facilities discharging into a 100 km section of the Penobscot River between Millinocket and Costigan, Maine. These expenditures resulted in an 80% reduction in loadings of biochemical oxygen demand and total suspended solids discharged from the kraft and sulfite pulp and paper mills in the study area. In 1974, the benthic macroinvertebrate community was determined to be highly degraded at three stations in closest proximity to pulp and paper effluents (Stas. 129, 131, 133). An additional two sites, somewhat downstream of pollution outfalls (Stas. 125, 126), were determined to be degraded (Rabeni 1977). The benthic community of the study area has been re-evaluated several times following major water quality changes in the 1970s, with the conclusion that the investments have resulted in dramatic improvements in the river's ability to support aquatic life.

Station 129 is located 4 km downstream of the Lincoln Pulp and Paper Company outfall. Figure 6-6 provides a graphical summary of changes in two metrics of aquatic community structure for the period of record at Station 129. Maine DEP uses the metrics shown in a linear discriminant model to assign aquatic life classification attainment. In 1974, Station 129 was designated as "highly polluted." The substrate at Station 129 was covered with sewage bacteria (*Sphaerotilus*) and the invertebrate community was restricted to worms, leeches, and pollution tolerant midge larvae. Numbers of individuals were very high, indicating a "bloom" of tolerant, opportunist organisms. Diversity and richness values were very low (Figure 6-6), and there was a complete absence of pollution-sensitive mayflies and stoneflies. In terms of aquatic life classification, this station did not meet minimum State or federal standards.

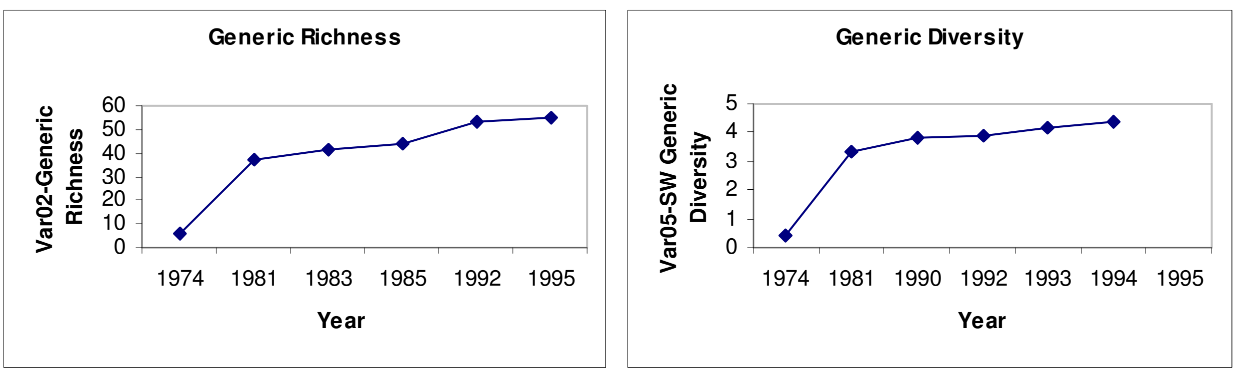


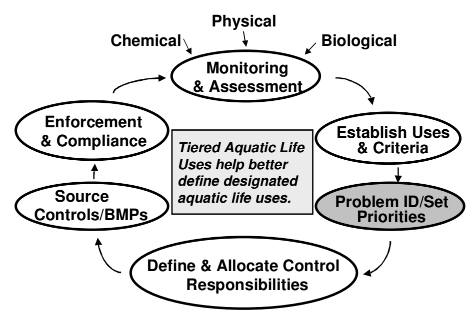
FIGURE 6-6. Scatter plots showing values for two biological community variables, generic richness (left) and generic diversity (right), from Sta. 129, the Penobscot River below Lincoln Pulp and Paper, between 1974 and 1996.

Dramatic improvements in the benthic macroinvertebrate community were evident by 1981 (Davies 1987). Total abundance was down, richness and diversity were greatly improved (Figure 6-6), and the proportion of tolerant midge larvae was lower. Low numbers of stoneflies and mayflies were also present. Overall, attainment had improved to Class C standards. The station has been sampled four times since 1981, each time meeting Class B standards and showing continued improvement in community structure, including high diversity and richness and healthy stonefly and mayfly populations. This long-term dataset provides a valuable example of the responsiveness of biota to water quality improvements. It also highlights the unique usefulness of biological monitoring to document and summarize the real world benefits of responsible stewardship of aquatic resources.

As a result of investment in wastewater control, the Penobscot River improved dramatically, from not attaining Class C standards in 1974 to attaining Class B standards throughout most of the river today. As a result, Maine upgraded the river from Class C to Class B in two steps. As of 1999, the entire mainstem, with the exception of an impounded section, is now Class B and must attain Class B standards. Without TALUs, the upgrade could not have taken place and the river would be maintained today as the equivalent of Class C. With Maine's TALUs, the river is now protected as Class B, which has been demonstrated to be attainable throughout. Documentation of the improvement and subsequent protection of the improved conditions is not possible without TALUs.

In addition to the Penobscot, many other streams in Maine have been upgraded in class as a result of effective wastewater treatment or dam removal, which has led to dramatic improvements in biological condition and class attainment.

CASE EXAMPLE 6-5. DEVELOPMENT OF LIMITS FOR NPDES PERMITS IN MAINE



Decoster Egg Farm, located in Turner, Maine, is the largest producer of brown eggs in New England. The Farm has a long history of environmental concerns including levels of ammonia and nitrates in violation of drinking water standards. This case example presents a unique example of the ***detection of biological impacts*** in a stream attaining surface water quality standards but affected by polluted groundwater recharge. Permitting staff had recorded nutrient levels in leachate draining poorly managed manure and chicken carcass waste piles. Stream violations were not sufficiently high to trigger enforcement action based on surface water quality violations but the high levels resulted in contaminated leachate entering groundwater on the Decoster property. In 1989, the Department brought enforcement action against Decoster Egg Farm to prohibit any further spreading of manure on the property and to enforce proper management of other animal waste products.

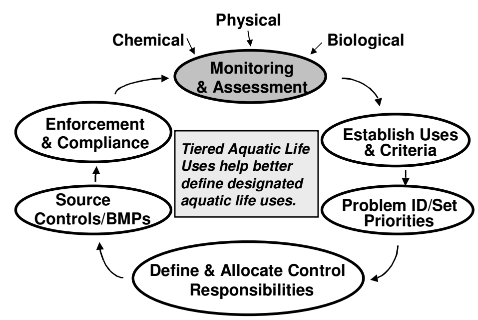
In 1991, the company was required to evaluate the condition of the aquatic life in streams affected by leachate or groundwater upwelling. Two of the streams, Lively Brook and House Brook, were designated by the State to maintain Class B water quality conditions. The use designation process had deemed this to be an appropriate management goal for these streams based on the tiered use designations of other streams of comparable habitat and watershed condition. Field investigations included probes of the hyporheic zone (the water flowing through the stream substrate) to measure the conductivity of the upwelling groundwater. Conductivity is a measure of the ionic strength of water and is a very good means of detecting certain types of pollutants. The streambed investigation uncovered several areas of contaminated groundwater recharge to the stream. Aquatic life sampling, completed in 1992, confirmed impacts to the benthos at three stations affected by groundwater upwelling on Lively Brook and one station on House Brook. Station 188, on House Brook, is located downstream of a failing treatment system that receives waste from the egg washing operation. The waste stream is severely contaminated by nitrates. This station failed to attain minimum Class C aquatic life standards in 1992. Repeat sampling in 1997 demonstrated attainment of Class C standards but the stream still failed to attain its assigned Class B status, indicating the need for additional management intervention. Biomonitoring information was used to issue a consent order requiring termination of manure spreading practices and improved treatment of the products of the egg washing facilities. The egg washing facility was removed.

The Lively and House Brooks case study illustrates the full water program cycle (Figure 5-1). Monitoring and characterization of the habitat and watersheds of the two streams revealed that, with best management practices in place, they should be able to attain Class B status, but in fact were not attaining minimum Class C status. Problem identification showed that contaminated groundwater due to poor management practices was causing the impairment. A set of source controls were applied, the facility complied with the controls, and monitoring of the streams' condition continued. The monitoring showed that although the streams had improved to Class C, they were still not attaining their designated Class B status. Maine DEP applied further source controls on the facility to achieve Class B status.

Ongoing monitoring, iterative management intervention, and tiered use goals confirmed that the streams had the potential to attain Class B status. Without tiered uses, source controls would have stopped when a minimal condition was reached (consistent with a Class C condition) and the two streams would never have recovered to Class B. Tiered aquatic life uses create attainable goals and best uses for waterbodies, resulting in better quality waters than are possible with a single use. If a general aquatic life use system had been in force, it likely would have resulted in a biological quality comparable to Maine's Class C, with no impetus for improvement to the actual potential (Class B).

CASE EXAMPLE 6-6. NPDES PERMITTING AND USE ATTAINABILITY ANALYSIS IN OHIO

Ecologically-based TALUs, a systematic approach to monitoring and assessment, and a sound UAA process can provide substantial benefits for NPDES permitting related to both the derivation of permits and assessing the effectiveness of a permit in restoring an aquatic life use. A system for identification of the attainable potential for the aquatic life of a waterbody using a systematic approach can set credible restoration goals and support measured responses to environmental risks. This case example illustrates the use of TALUs, systematic monitoring and assessment, and a consistent process for conducting UAAs in support of NPDES permitting issues.



The Ottawa River in northwest Ohio has been heavily polluted for more than a century. The river is impacted by the city of Lima, rural communities, and agricultural activities (row crops). Heavy industry in Lima was identified as a major source of water pollution since the 1880s (Leeson 1885 c.f. Ohio EPA 1992) being especially severe in the 1960s " . . . when more than 37 miles were devoid of fish, including the Auglaize River downstream from the Ottawa River" (Ohio EPA 1992). Point sources include one major municipal and two major industrial discharges, industrial contributors to the Lima sewer system, combined sewer overflows (CSOs), and partial or untreated sewage discharges from semi-rural areas in the watershed. The effluent flow from the three major point sources enter the Ottawa River within a 0.8 mile reach and comprise the majority of the river flow during dry weather months. Improvements consistent with CWA technology standards have been made at the major wastewater treatment facilities since the late 1970s. The major causes of impairment include organic enrichment and low D.O., general toxicity, habitat alterations (impoundments), nutrients, ammonia, heavy metals, oil and grease, and chlorine in both the water column and bottom sediments (Ohio EPA 1998).

This case example focuses on a 25-mile segment of the Ottawa River that is directly impacted by major point sources (Figure 6-7) and includes zones of immediate and acute impacts and various phases of recovery downstream. Physical habitat in the mainstem downstream from the major point sources is good to excellent, and the mainstem is designated WWH as the result of a use attainability analysis and upgrade conducted in the late 1980s. Prior to this analysis, most of the river was assigned the Limited Warmwater Habitat (LWH) aquatic life use, which was assigned to rivers thought to be so polluted that restoration was considered unfeasible. The LWH use was developed and applied prior to the development and adoption of TALUs by Ohio EPA and is no longer used.

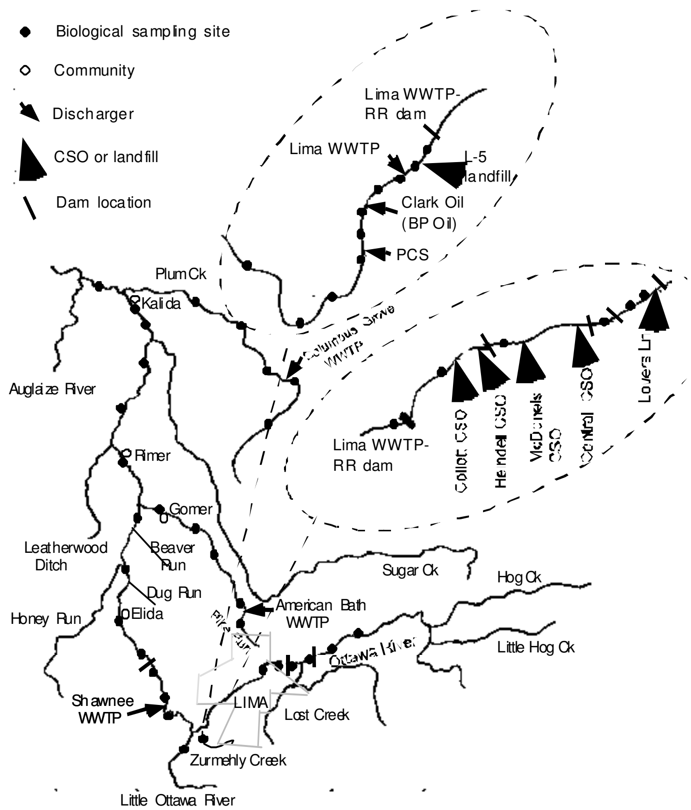


FIGURE 6-7. Map of the Ottawa River with magnification of two reaches in the Lima, Ohio area (after Ohio EPA 1998).

Toxic stressors, exposures, and responses reached a maximum in the segment directly impacted by the three major point sources (Ohio EPA 1998; Yoder and DeShon 2003). Evidence of multiple toxic exposures occurred in the water column chemistry, sediment chemistry, whole effluent toxicity, frequency of DELT anomalies, fish tissue contaminants, and biochemical markers (Table 6-3). These indicators pointed strongly to impacts of a toxic character and the biological response signatures provided the corroborating feedback. Low D.O. can occur in the Ottawa River (Ohio EPA 1998), but the more serious toxic effects that are evident in the biological response signatures presently mask its less serious effects.

TABLE 6-3. A matrix of stressor, exposure, and response indicators for the Ottawa River mainstem based on data collected in 1996 (after Ohio EPA 1998). The darkness of shading indicates the degree of severity of effect or exceedance expressed by an indicator.

SEGMENT	DES. USE	RESPONSE INDICATORS				EXPOSURE INDICATORS						STRESSORS				
	Attain-ment Status	QHEI	IBI	MIwb	ICI	Water Chem.	Sedi-ment Chem.	Tox-icity	% DELT	Fish Tiss.	Bio-marker	# Dams/ Pools	Urban-Indust. Landuse	Cumulative Loads	Spills	CSO SSOs
Ottawa River mainstem - 1996																
Thayer Rd to Sugar St.	FULL-PART.	68	Fair-Good	Fair-Good	Good	Nitrates	Low	NA	Mod-High	Mer-cury	Low	Mod-e	Low	Low	Low	Low
Sugar St. to Lima WWTP	**NON**	47	**Poor** to Fair	**Poor** to Fair	**Poor** to M.G.	CBOD TSS D.O.	As,Cr Cd,Cu Ni,Zn	Mod-erate	**High**	Pesti-cides	BUN Naph B(a)p	**High**	**High**	Mod-erate	Mod-e	**High**
Lima WWTP Allentown dam	**NON**	72	**Poor**	**Poor** to Fair	Fair to Good	Amm. CBOD TSS D.O. Nitrates Phos Chrom. PAH Pesticid	As,Cr Cd,Cu Ni,Zn PAH	Mod-erate	**Very High**	Selen-ium Pest-icides	EROD Naph B(a)p BUN	Mod-e	**High**	**High**	**High**	**High**
Allentown dam to Kalida	PAR-TIAL	69	**Poor** -Fair	Fair-Good	Good -Exc.	TSS	Low	NA	**High**	Pesti-cides	Low	Low	Low	**High**	Low	Low
Kalida to mouth	FULL	69	Good	Good	Exc.	TSS	Low	NA	**Very High**	Pesti-cides	Low	Low	Low	**High**	Low	Low

QHEI scores for the Ottawa indicated more than adequate habitat to support the WWH use designation (Rankin 1989, 1995). In a growing recovery zone immediately below the impacted reach, the biota eventually exhibited recovery to WWH status in the lower reaches of the river. In the impaired sections, the biological response signatures strongly indicate general toxicity, which is a fundamentally different response than what would occur in response to habitat or low D.O. alone (Figure 6-8; Yoder and Rankin 1995b; Yoder and DeShon 2003). Results from a similar time period for the Scioto River are shown for comparison. This river is impacted by non-toxic causes and sources including organic enrichment and oxygen demanding wastes from sources that dominate the low flow of the river and emanate from a similar municipal infrastructure and watershed setting. Taken together, these considerations led Ohio EPA to redesignate (upgrade) the Ottawa River from LWH to WWH in 1989. The redesignation was controversial and resulted in legal actions challenging the WWH use. Plaintiffs contended that the habitat could not support a WWH assemblage and further argued that D.O. concentrations consistent with WWH criteria were unattainable due to upstream impoundments and the flow regime. The WWH designation was upheld because Ohio had a substantial record demonstrating the relationship between habitat condition (as QHEI) and attainable biological condition described in the tiered uses. The response signatures indicated that the cause of non-attainment in the Ottawa River was primarily toxicity.

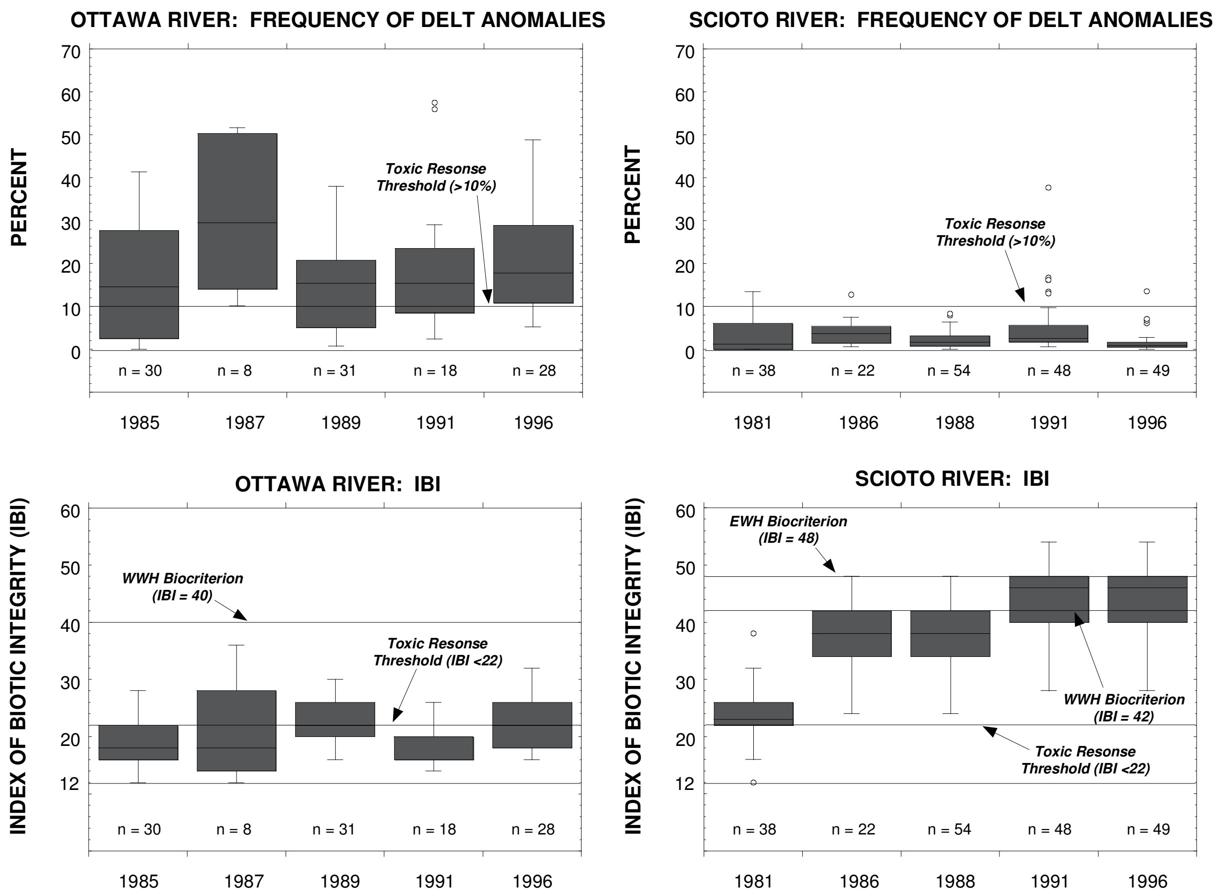


FIGURE 6-8. Results for two key fish assemblage measures (%DELT anomalies, upper left panel and IBI, lower left panel) showing the thresholds for toxic responses in the Ottawa River study area between 1985 and 1996. The results are shown with those from the Scioto River between 1981 and 1996 to illustrate the different responses shown in a river impacted by non-toxic stressors.

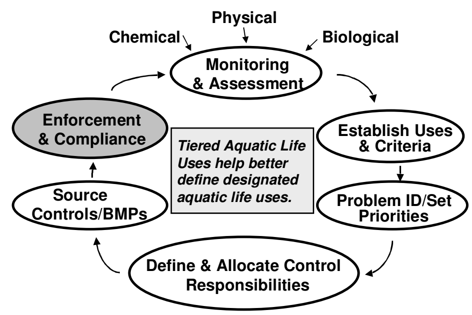
The WWH redesignation and the subsequent permitting of the three major point sources could have taken a significantly different path in the absence of the TALU approach employed by Ohio EPA. Instead of keeping the focus on the most limiting problem of complex toxicity, the outcome could have been diverted by the initial claims of habitat limitations and D.O. issues. Ohio's systematic approach to monitoring directly tied to its TALUs was upheld in a court case on the redesignation to WWH, which has averted subsequent legal actions in other similar permitting cases. This is related to the soundness and consistency of the UAA approach and the perception that the TALUs are reasonably attainable and protective.

One tool the NPDES program uses to identify potential problems from dischargers is non-compliance with permit terms and conditions. In this case, none of the individual point sources involved were considered in non-compliance of their NPDES permits at the time of the assessments. However, their cumulative effect on biological condition resulted in severe biological impairment of the river. As a result, Ohio EPA imposed controls to significantly improve water quality, including chronic WET limits, close scrutiny of intermittent releases and spills, and internal audits conducted by two of the industrial facilities involved. In addition, an unregulated landfill leachate was discovered and subsequently required remediation.

Under a tiered system, the biocriteria endpoints vary with the specific use and thus can affect the NPDES permit. For example, a WWH designation requires better biological condition (higher IBI, MCI and MIwB scores) than the LRW use. Accordingly, LRW waters can tolerate higher nutrients and lower D.O. than WWH waters (See Figure 6-2, Table 6-2, and Appendix B), which would affect permit limits. A decision that the stream was either habitat limited or dissolved oxygen limited alone would have diverted attention away from the severe toxic impacts that were in reality limiting the aquatic life in this river. The magnitude of these influences would have been underestimated on the sole basis of administrative measures, without the stressor analysis that identified the causes of impairment in the Ottawa River.

CASE EXAMPLE 6-7. SUPPORT FOR DREDGE AND FILL PERMITTING IN OHIO

The losses of habitat diversity or habitat-mediated stressors such as increased siltation are now the most prevalent causes of aquatic impairment in Ohio (Figure 6-9, Ohio EPA 2000). This is also true across much of the U.S. (U.S. EPA 2002b). Environmental effects of extensive landscape changes and in stream habitat alterations are a primary stressor gradient along which the tiered aquatic life uses were developed. Some habitat alterations are readily restorable while others are essentially permanent either because they are continuously maintained for flood control or drainage purposes or they exceed the natural capacity for recovery.



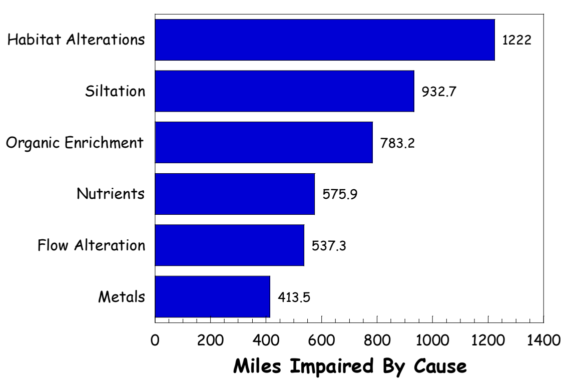


FIGURE 6-9. Six leading causes of aquatic life impairment in Ohio up to the year 2000 (from Ohio EPA 2000).

States can use Sections 401 and 404 of the CWA to manage direct alterations to aquatic habitats. Tiered aquatic life uses have proved useful in 404 permitting and 401 certification of those permits. Those wanting to modify a stream that will result in the discharge of dredge or fill material into waters of the U.S. must obtain a Section 404 permit from the U.S. Army Corps of Engineers (ACOE) and a Section 401 water quality certification from the State. The State must certify that proposed activities will comply with, not violate, WQS. The existence of biocriteria in the Ohio WQS makes this linkage a valid tool for evaluating the impacts of habitat alterations that are covered under the CWA. Ohio EPA used a 20+ year database to develop habitat stressor gradients along several aspects of habitat quality at both site and watershed scales, including overall habitat quality as measured by the QHEI and for specific attributes such as substrate and channel condition. Examples of these stressor gradients from the E. Corn Belt Plains (ECBP) and Huron/Erie Lake Plain (HELP) ecoregions are illustrated in Figure 6-10.

Tiered aquatic life uses have enabled a range of management responses to dredge and fill projects related to the quality and sensitivity of the waterbody in question. Tiered uses are an important consideration in the implementation of nationwide permits. Nationwide permits are designed to minimize site-specific oversight where ecological risks are assumed to be low. Frequently, however, the criteria for which places are eligible can overlook high quality waters and lead to their alteration. The Ohio EWH use designation requires high habitat quality and stable hydrological regimes (especially in headwater and wadeable streams). Because these essential attributes can be altered by direct modifications to the stream channel and other habitat features, Ohio requires individual reviews of projects that occur in such high quality streams. Under a general use system, these would be lumped with all other streams under the nationwide permit system.

The same information embodied in the tiered aquatic life uses allows Ohio to expend less oversight on streams that cannot attain the WWH use designation. Such streams are generally ephemeral or continuously maintained as drainage conveyances. This does not mean that physically degraded streams are ignored. The attention gained by habitat impacts has prompted the development of mitigation standards that will take the tiered aquatic life uses into account and require enhancement or restoration wherever feasible. The stressor-response relationships (Figure 6-10) that have been developed between biological assemblages and key habitat attributes have been applied to the 401 program in Ohio. For nationwide 404 permits a series of general and specific exclusions and conditions have been derived that vary with tiered aquatic life uses (ACOE 2002). These include a general exclusion (of nationwide permits) for streams that are EWH and for certain antidegradation tiers (State Resource Waters and Outstanding State Resource Waters), the delineation of which was based primarily on the same biological assemblage attributes that are in common with Ohio's tiered aquatic life uses.

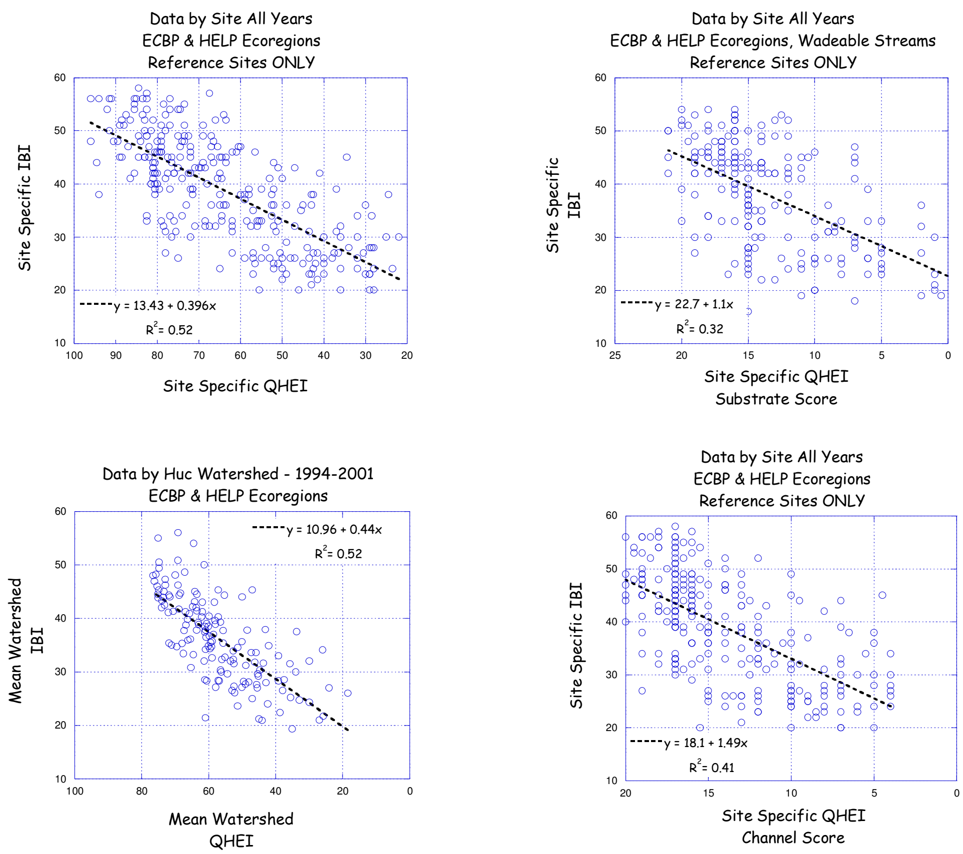


FIGURE 6-10. Examples of habitat stressor gradients vs. IBI for Ohio wadeable streams in the ECBP and HELP ecoregions.

Aside from the general considerations discussed above, tiered uses have also proved useful for specific nationwide permits. For example, Nationwide Permit 21 is for surface coal mining activities. Higher quality uses such as WWH or EWH and Coldwater Habitat (CWH) require individual 404 permits in all cases. Only MWH or LRW uses can be exempted from site-specific review under a nationwide permit for mining (and for these there are stream length limitations). Again this is a significant benefit of having tiered uses and the knowledge of the relationships between activities (e.g., habitat alterations) and the biological responses in the indexes that compose the tiered biocriteria. The 404/401 program in Ohio is still evolving. One goal is to move away from a case-by-case review of every permit by developing mitigation standards tied directly to the tiered aquatic life uses that will be protective, relatively rapid, accurate, and efficient in terms of resource expenditures. Making similar decisions within a single use system would be more difficult and require either more case-by-case oversight to account for habitat gradients, or risk being over-protective in some cases and under-protective in others.

References & Additional Resources

Aarts, B.G.W., F.W.B. van den Brink and P.H. Nienhuis. 2004. Habitat loss as the main cause of the stagnating recovery of the fish faunas of regulated large rivers in Europe: the transversal floodplain gradient. *River Research and Applications*.

Adams, S.M. 1990. Status and use of biological indicators for evaluating the effect of stress on fish. *American Fisheries Society Symposium* 8(1).

Adams, S.M., K.D. Ham, and R.F. LeHew. 1998. A framework for evaluating organism responses to multiple stressors: mechanisms of effect and importance of modifying ecological factors. In *Multiple Stresses in Ecosystems*, J.J. Cech, B.W. Wilson, and D.G. Crosby (eds.), pp.13-22. Lewis Publishers, Boca Raton, FL.

Allan, J.D. 1995. *Stream Ecology: Structure and Function of Running Waters*. Chapman & Hall, New York.

Allan, J.D., D.L. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple scales. *Freshwater Biology* 37:149-161.

Allan, J.D. 2004. Landscapes and riverscapes: The influence of land use on stream ecosystems. *Annual Review of Ecology, Evolution, and Systematics* 35:257-84.

Allen, A.P. and R.J. O'Connor. 2000. Interactive effects of land use and other factors on regional bird distributions. *Journal of Biogeography* 27:889-900.

Angermeier, P.L. and J.R. Karr. 1994. Biological integrity versus biological diversity as policy directives. *BioScience* 44:690-697.

Appelberg, M., B.I. Henrikson, L. Henrikson, and M. Svedang. 1993. Biotic interactions within the littoral community of Swedish forest lakes during acidification. *Ambio* 22:290-297.

Armitage, P.D., D. Moss, J.F. Wright, and M.T. Furse. 1983. The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running-water sites. *Water Research* 17:333-347.

Army Corps of Engineers (ACOE). 2002. Memo: Grant of Section 401 Certification Authorization of discharges of dredged or fill material to various waters of the state [Ohio] for Nationwide Permits as detailed in the January 15, 2002, Federal Register (Volume 67, Number 10). CECW-OR Washington, D.C. 20314-1000.

Bailey, R.C., R.H. Norris, and T.B. Reynoldson. 2004. *Bioassessment of Freshwater Ecosystems Using the Reference Condition Approach*. Kluwer, Dordrecht, NL.

Baird, J.D., and G.A. Burton, Jr. (eds.). 2001. *Ecological Variability: Separating Natural from Anthropogenic Causes of Ecosystem Impairment*. SETAC Press, Pensacola, FL. 336 pp.

Bakus, G.J., W.G. Stillwell, S.M. Lather and M.C. Wallerstein. 1982. Decision-making: with application for environmental management. *Environmental Management* 6:493-504.

Ballentine, L.K. and L.J. Guarraia (eds.). 1977. *Integrity of Water*. EPA 055-001-010-01068-1. U.S. EPA, Office of Water and Hazardous Materials, Washington, DC.

Barbour, M.T., J.M. Diamond, and C.O. Yoder. 1996a. Biological Assessment Strategies: Applications and Limitations. In *Whole effluent toxicity testing: An evaluation of methods and prediction of receiving system impacts*, D.R. Grothe, K.L. Dickson, and D.K. Reed-Judkins (eds.), pp. 245-270. SETAC Press, Pensacola, FL.

Barbour, M.T., J. Gerritsen, G.E. Griffith, R. Frydenborg, E. McCarron, J.S. White, and M.L. Bastian. 1996b. A framework for biological criteria for Florida streams using benthic macroinvertebrates. *Journal of the North American Benthological Society* 15:185-211.

Barbour, M.T. 1997. The re-invention of biological assessment in the U.S. *Human and Ecological Risk Assessment* 3(6):933-940.

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. *Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish. Second Edition.* EPA 841-B-99-002. U.S. EPA, Office of Water, Washington, DC.

Barbour, M.T. and C.O. Yoder. Technical Guidelines: Technical Elements of a Bioassessment Program. Unpublished manuscript.

Bendellyoung, L.I., K.E. Bennett, A. Crowe, C.J. Kennedy, A.R. Kermode, M.M. Moore, A.L. Plant, and A. Wood. 2000. Ecological characteristics of wetlands receiving an industrial effluent. *Ecological Applications* 10:310-322.

Benke, A.C. and C.E. Cushing (eds.). 2005. *Rivers of North America*. Academic Press, New York.

Booth, D.B., J.R. Karr, S. Schauman, K.P. Kinrad, S.A. Morley, M.G. Larson, and S.J. Burges. 2004. Reviving urban streams: land use, hydrology, biology, and human behavior. *Journal of the American Water Resources Association* 40:1351-1364.

Brinely, F.J. 1942. Biological studies, Ohio River pollution survey. I. Biological zones in a polluted stream. *Sewage Works Journal* 14(1):147-152.

Brinkhurst, R. 1993. Future directions in freshwater biomonitoring. In *Freshwater Biomonitoring and Benthic Macroinvertebrates*, D.H. Rosenberg and V. H. Resh (eds.), pp. 442-460. Chapman and Hall, New York.

Brinson, M.M. and A.I. Malvarez. 2002. Temperate freshwater wetlands: types, status, and threats. *Environmental Conservation* 29:115-133.

Brönmark, C. and L-A. Hansson. 2002. Environmental issues in lakes and ponds: current state and perspectives. *Environmental Conservation* 29:290-306.

Bryce, S.A., D.P. Larsen, R.M. Hughes, and P.R. Kaufmann. 1999. Assessing relative risks to aquatic ecosystems: a mid-Appalachian case study. *Journal of the American Water Resources Association* 35:23-36.

Bryce, S.A., R.M. Hughes, and P.R. Kaufman. 2002. Development of a bird integrity index: using bird assemblages as indicators of riparian condition. *Environmental Management* 30:294-310.

Bunn S.E. and Arthington A.A. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management* 30:492-507.

Cairns, J. Jr. 1977. Quantification of biological integrity. In *The Integrity of Water*, R.K. Ballentine and L.J. Guarraia (eds.), pp. 171-187. Proceedings of a Symposium, March 10-12, 1975. U.S. Environmental Protection Agency, Washington, DC.

Cairns, J. Jr. 1981. Biological monitoring part VI-future needs. *Water Research* 15:941-952.

Cairns, J. Jr. and J. R. Pratt. 1993. A history of biological monitoring using benthic macroinvertebrates. In *Freshwater Biomonitoring and Benthic Macroinvertebrates*, D.M. Rosenberg and V.H. Resh (eds.), pp. 10-27. Chapman & Hall, New York.

Cairns, J. Jr., P. V. McCormick, and R. R. Niederlehner. 1993. A proposed framework for developing indicators of ecosystem health. *Hydrobiologia* 263:1-44.

Contreras-Balderas, S., R. J. Edwards, M. L. Lozano-Vilano and M. E. García-Ramírez. 2002. Fish biodiversity changes in the Lower Rio Grande/Rio Bravo, 1953-1996. *Reviews in Fish Biology and Fisheries* 12(2):219-240.

Campbell, M.S.A. 1939. Biological indicators of intensity of stream pollution. *Sewage Works Journal* 11(1):123-127.

Canada and the United States. 2001. State of the Great Lakes 2001. Toronto and Chicago. 82 p.

Carlson, R.E. 1992. Expanding the trophic state concept to identify non-nutrient limited lakes and reservoirs. In *Proceedings, National Conference on Enhancing the States' Lake Management Programs*, Chicago, IL, 1991, pp. 59-71. North American Lake Management Society.

Carpenter, S., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley, and V.H. Smith. 1998. Nonpoint pollution of surface waters with phosphorous and nitrogen. *Issues in Ecology*, No. 3.

Cech Jr., J.J., B.W. Wilson, and D.G. Crosby. 1998. *Multiple stressors in ecosystems*. Lewis Publishers, 202p.

Chaloud, D. J. and M. S. Nash. 2001. Using cononical correlation to detect association of landscape metrics with water biological and chemical properties in Savannah River Basin. Presented at Above & Beyond 2001, An EPA Remote Sensing Conference, Las Vegas, NV, March 20-21, 2001.

Courtemanch, D.L. 1984. A closing artificial substrate device for sampling benthic macroinvertebrates in deep rivers. *Freshwater Invertebrate Biology* 3(3):143-146.

Courtemanch, D.L., S.P. Davies, and E.B. Laverty. 1989. Incorporation of biological information in water quality planning. *Environmental Management* 13(1):35-41.

Courtemanch, D.L. 1995. Merging the science of biological monitoring with water resource management policy: Criteria development. In *Biological assessment and criteria: Tools for water resource planning and decision making*, W.S. Davis and T.P. Simon (eds.), pp. 315-325. Lewis Publishers, Boca Raton, FL.

Courtney, L. A. and W. H. Clements. 2000. Sensitivity to acidic pH in benthic invertebrate assemblages with different histories of exposure to metals. *Journal of the North American Benthological Society* 19:112-127.

Croonquist, M. and R. P. Brooks. 1993. Effects of habitat disturbance on bird communities in riparian corridors. *Journal of Soil and Water Conservation* 48:65-70.

Danz, N.P., R.R. Regal, G.J. Niemi, V.J. Brady, T. Hollenhorst, L.B. Johnson, G.E. Host, J.M. Hanowski, C.A. Johnston, T. Brown, J. Kingston, and J.R. Kelly. 2005. Environmentally stratified sampling design for the development of Great Lakes environmental indicators. *Environmental Monitoring and Assessment.*

Davies, P.E. 2000. Development of a national river bioassessment system (AusRivAS) in Australia. In *Assessing the Biological Quality of Fresh Waters*, J.F. Wright, D.W. Sutcliffe and M.T. Furse (eds.), pp. 113-124. Freshwater Biological Association, Ambleside.

Davies, S.P. 1987. Benthic macroinvertebrate response to pollution abatement in the Penobscot River, Maine. MS thesis, University of Maine, Orono, ME. 83 pp.

Davies, S.P., L. Tsomides, D.L. Courtemanch and F. Drummond. 1995. *Maine biological monitoring and biocriteria development program*. DEP-LW108, Maine Department of Environmental Protection, Augusta, ME. P.61.

Davies, S.P., L. Tsomides, J.L. DiFranco, and D.L. Courtemanch. 1999. *Biomonitoring Retrospective: Fifteen Year Summary for Maine Rivers and Streams*. MDEP (DEP LW1999-26).

Davies, S.P. and L. Tsomides. 2002. *Methods for Biological Sampling and Analysis of Maine's Rivers and Streams*. MDEP (DEP LW0387-B2002).

Davies, S.P., F. Drummond, D.L. Courtemanch, L. Tsomides. Probabilistic Models Based on Expert Judgment Protocols to Assess Attainment of Tiered Aquatic Life Uses in Maine Rivers and Streams. Unpublished manuscript.

Davies, S.P. and S.K. Jackson. In press. The Biological Condition Gradient: A descriptive model for interpreting change in aquatic ecosystems. *Ecological Applications*.

Davis, W.S. 1995. Biological assessment and criteria: Building on the past. In *Biological assessment and criteria: Tools for water resource planning and decision making*, W.S. Davis and T.P. Simon (eds.), pp. 15-29. Lewis Publishers, Boca Raton, FL.

Day, J.W., C.A.S. Hall, W.M. Kemp, and A. Yanez-Arenciba. 1989. *Estuarine Ecology*. John Wiley & Sons, New York.

DeShon, J.D. 1995. Development and application of the invertebrate community index (ICI). In *Biological assessment and criteria: Tools for water resource planning and decision making*, W.S. Davis and T.P. Simon (eds.), pp. 217-243. Lewis Publishers, Boca Raton, FL.

Diamond, J.M. and V.B. Serveiss. 2001. Identifying sources of stress to native aquatic fauna using a watershed ecological risk assessment framework. *Environmental Science and Technology* 35:4711-4718.

Dixit, S. S., J. P. Smol, D. F. Charles, R. M. Hughes, S. G. Paulsen, and G. B. Collins. 1999. Assessing water quality changes in the lakes of the northeastern United States using sediment diatoms. *Canadian Journal of Fisheries and Aquatic Sciences* 56:131-152.

Doudoroff, P. and C.E. Warren. 1957. Biological indices of water pollution with special reference to fish populations. Biological Problems in Water Pollution, U.S. Public Health Service, Robert A. Taft Sanitary Engineering Center, Cincinnati, OH. 144-163.

Dynesius, M. and C. Nilsson. 1994. Fragmentation and flow regulation of river systems in the northern third of the world. *Science* 266:753-762.

Ellis, M.M. 1937. Detection and measurement of stream pollution. *Bulletin of the Bureau of Fisheries* 48:365-437.

Emery, E. B., T. P. Simon, F. H. McCormick, P. A. Angermeier, J. E. DeShon, C. O. Yoder, R. E. Sanders, W. D. Pearson, G. D. Hickman, R. J. Reash, and J. A. Thomas. 2003. Development of a Multimetric Index for Assessing the Biological Condition of the Ohio River. *Transactions of the American Fisheries Society* 132:791-808.

Fausch, K.D., J.R. Karr, and P.R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. *Transactions of the American Fisheries Society* 113:39-55.

Fausch, K.D., J. Lyons, P. L. Angermeier, and J. R. Karr. 1990. Fish communities as indicators of environmental degradation. *American Fisheries Society Symposium* 8:123-144.

Fausch, K.D., C.E. Torgersen, C.V. Baxter, and H.W. Li. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. *BioScience* 52:483-498.

Federal Water Pollution Control Act, as amended by the Clean Water Act of 1977: [Commonly referred to as Clean Water Act] [Enacted by Public Law 92-500, October 18, 1972, 86 Stat. 816; 33 U.S.C. 1251 et seq.; Amended by PL 93-207, December 28, 1973, and PL 93-243, January 2, 1974; PL 93-592, January 2, 1975; PL 94-238, March 23, 1976; PL 94-273, April 21, 1976; PL 94-558, October 19, 1976; PL 95-217, December 28, 1977; PL 95-576, November 2, 1978; PL 96-148, December 16, 1979; PL 96-478, PL 96-483, October 21, 1980; PL 96-510, December 11, 1980; PL 96-561, December 22, 1980; PL 97-35, August 13, 1981; PL 97-117, December 29, 1981; PL 97-164, April 2, 1982; PL 97-440, January 8, 1983; Amended by PL 100-4, February 4, 1987].

Fitzgerald, D. G., R. P. Lanno, and D. G. Dixon. 1999. A comparison of a sentinel species evaluation using creek chub (*Semotilus atromaculatus mitchill*) to a fish community evaluation for the initial identification of environmental stressors in small streams. *Ecotoxicology* 8:33-48.

Fitzhugh, T.W. and B.D. Richter. 2004. Quenching urban thirst: growing cities and their impacts on freshwater ecosystems. *BioScience* 54: 741-754.

Fitzpatrick, F. A., B. C. Scudder, B. N. Lenz, and D. J. Sullivan. 2001. Effects of multi-scale environmental characteristics on agricultural stream biota in eastern Wisconsin. *Journal of the American Water Resources Association* 37:1489-1507.

Freeman, M.C., E.R. Irwin, N.M. Burkhead, B.J. Freeman, and H. L. Bart, Jr. 2005. Status and Conservation of the Fish Fauna of the Alabama River System. In *Historical Changes in Large River Fish Assemblages of the Americas*, J.N. Rinne, R. M. Hughes, and B. Calamusso (eds.). American Fisheries Society Symposium 45, Bethesda, Maryland.

Frey, D.G. 1977. Biological Integrity, a Historical Approach. In *The Integrity of Water*, R.K. Ballentine and L.J. Guarraia (eds.), pp. 127-140. Proceedings of a Symposium, March 10-12, l975. U.S. EPA, Office of Water and Hazardous Materials, Washington, DC.

Frissell, C.A. 1993. Topology of extinction and endangerment of native fishes in the Pacific Northwest and California (USA). *Conservation Biology* 7:342-354.

Furse, M.T., D. Moss, J.F. Wright, and P.D. Armitage. 1984. The influence of seasonal and taxonomic factors on the ordination and classification of running-water sites in Great Britain and on the prediction of their macro-invertebrate communities. *Freshwater Biology* 14:257-80.

Gakstatter, J., J.R. Gammon, R.M. Hughes, L. Ischinger, M. Johnson, J.R. Karr, T. Murphy, T.M. Murray, and T. Stewart. 1981. *A recommended approach for determining biological integrity in flowing waters*. U.S. EPA, Corvallis, OR. 26 pp.

Gammon, J.R. 1976. *The fish population of the middle 340km of the Wabash River*. Purdue University Water Resources Research Center, LaFayette, IN. Technical Report 86.

Gammon, J.R., A. Spacie, J.L Hamelink, and R.L. Kaesler. 1981. Role of electrofishing in assessing environmental quality of the Wabash River. In *Ecological Assessments of Effluent Impacts on Communities of Indigenous Aquatic Organisms*, J. Bates and C.I. Weber (eds.), pp. 307-324. ASTM STP 730. Philadelphia, PA.

Ganasan, V. and R. M. Hughes. 1998. Application of an index of biological integrity (IBI) to fish assemblages of the rivers Khan and Kshipra (Madhya Pradesh), India. *Freshwater Biology* 40:367-383.

Gaufin, A.R. and C.M. Tarzwell. 1953. Discussion of R. Patrick's paper, "Aquatic organisms as an aid in solving waste disposal problems." *Sewage and Industrial Wastes* 25(2):214-217.

Gerritsen, J., M.T. Barbour, and K. King. 2000. Apples, oranges and ecoregions: On determining pattern in aquatic assemblages. *Journal of North American Benthological Society* 19:487-496.

Gerritsen, J. and E.W. Leppo. 2004. *Tiered aquatic life use development for New Jersey*. Prepared by Tetra Tech, Inc. for U.S. EPA Office of Water, EPA Region 2, and New Jersey DEP.

Gessner, M.O. and E. Chauvet. 2002. A case for using litter breakdown to assess functional stream integrity. *Ecological Applications* 12: 498-510.

Gibson, G.R., M.T. Barbour, J.B. Stribling, J. Gerritsen, and J.R. Karr. 1996. *Biological criteria: Technical guidance for streams and small rivers (revised edition)*. EPA/822/B/96/001. U.S. EPA, Office of Water, Washington, DC.

Graf, W. L. 2001. Damage control: restoring the physical integrity of America's rivers. *Annals of the Association of American Geographers* 91:1-27.

Greenberg L, P. Svendsen, and A. Harby. 1996. Availability of microhabitats and their use by brown trout (*Salmo trutta*) and grayling (*Thymallus thymallus*) in the River Vojman, Sweden. *Regulated Rivers: Research & Management* 12: 287–303.

Griffith, J.A., E.A. Martinko, J.L. Whistler, and K.P. Price. 2002. Interrelationships among landscapes, NDVI, and stream water quality in the U.S. Central Plains. *Ecological Applications* 12(6): 1702-1718.

Grove, R.H. 1995. *Green imperialism: colonial expansion, tropical island edens, and the origins of environmentalism, 1600-1860*. Cambridge University Press, Cambridge, United Kingdom.

Halliwell, D. B., R. W. Langdon, R. A. Daniels, J. P. Kurtenbach, and R. A. Jacobson. 1998. Classification of freshwater fish species of the Northeastern United States for use in the development of indices of biological integrity, with regional applications. In *Assessing the sustainability and biological integrity of water resources using fish communities*, T. P. Simon (ed.), pp. 301-337. CRC Press, Boca Raton, FL.

Harding, J.S., E.F. Benfield, P.V. Bolstad, G.S. Helfman, and E.B.D. Jones III. 1998. Stream biodiversity: the ghost of land use past. *Proceedings of the National Academy of Science* USA 95: 14843-14847.

Hawkins, C.P. and M.R. Vinson. 2000. Weak correspondence between landscape classifications and stream invertebrate assemblages: implications for bioassessment. *Journal of North American Benthological Society* 19: 501-517.

Hawkins, C.P., R. Norris, J. Gerritsen, R.M. Hughes, S.K. Jackson, R.K. Johnson, and R.J. Stevenson. 2000. Evaluation of the use of landscape classifications for the prediction of freshwater biota: synthesis and recommendation *Journal of North American Benthological Society* 19:541-556.

Helmsley-Flint, B. 2000. Classification of the biological quality of rivers in England and Wales. In *Assessing the Biological Quality of Fresh Waters*, J.F. Wright, D.W. Sutcliffe and M.T. Furse (eds.), pp. 55-70. Freshwater Biological Association, Ambleside, UK.

Herlihy, A.T., R.M. Hughes, and J.C. Sifneos. In Press. National clusters of fish species assemblages in the conterminous United States and their relationship to existing landscape classification schemes. In. *Influences of landscapes on stream habitats and biological assemblages*, R.M. Hughes, L. Wang, and P.W. Seelbach (eds.), Pages xx-xx. American Fisheries Society Symposium, Bethesda, Maryland.

Herricks, E.E. and D.J. Schaeffer. 1985. Can we optimize biomonitoring? *Environmental Management* 9:487-492.

Hilsenhoff, W.L. 1987. An improved biotic index of organic stream pollution. *Great Lakes Entomologist* 20: 31-39.

Holden, P. B. and C. B. Stalnaker. 1975. Distribution and abundance of mainstem fishes of the middle and upper Colorado River Basins, 1967-1973. *Transactions of the American Fisheries Society* 104:217-231.

Hughes, R.M. 1985. Use of watershed characteristics to select control streams for estimating effects of metal mining wastes on extensively disturbed streams. *Environmental Management* 9:253-262.

Hughes, R.M., D.P. Larsen, and J.M. Omernik. 1986. Regional reference sites: a method for assessing stream potential. *Environmental Management* 10:629-635.

Hughes, R. M. and J. R. Gammon. 1987. Longitudinal changes in fish assemblages and water quality in the Willamette River, Oregon. *Transactions of the American Fisheries Society* 116:196-209.

Hughes, R.M. and D.P. Larsen. 1988. Ecoregions: an approach to surface water protection. *Journal of the Water Pollution Control Federation* 60: 486-493.

Hughes, R. M. 1994. Defining acceptable biological status by comparing with reference conditions. In *Biological assessment and criteria: tools for water resource planning and decision making*, W. S. Davis and T. P. Simon (eds.), pp. 31-47. CRC Press, Boca Raton, FL.

Hughes, R. M., P. R. Kaufmann, A. T. Herlihy, T. M. Kincaid, L. Reynolds, and D. P. Larsen. 1998. A process for developing and evaluating indices of fish assemblage integrity. *Canadian Journal of Fisheries and Aquatic Sciences* 55:1618-1631.

Hughes, R. M., and T. Oberdorff. 1998. Applications of IBI concepts and metrics to waters outside the United States and Canada. In *Assessing the sustainability and biological integrity of water resources using fish communities*, T. P. Simon (ed.), pp. 79-93. CRC Press, Boca Raton, FL.

Hughes, R.M., S. Howlin, and P.R. Kaufmann. 2004. A biointegrity index (IBI) for coldwater streams of western Oregon and Washington. *Transactions of the American Fisheries Society* 133:1497-1515.

Hughes, R. M., S. A. Bryce, and D. Drake. Use of a Generalized Stressor Gradient for Comparing Reference Conditions of USA Surface Waters. Unpublished manuscript.

Intergovernmental Task Force on Monitoring Water Quality (ITFM). 1992. *Ambient Water-Quality Monitoring in the United States. First Year Review, Evaluation, and Recommendations*. Interagency Advisory Committee on Water Data, U.S. Geological Survey, Reston, Virginia.

Intergovernmental Task Force on Monitoring Water Quality (ITFM). 1995. *The strategy for improving water-quality monitoring in the United States: Final report of the Intergovernmental Task Force on Monitoring Water Quality*. U.S. Geological Survey, Reston, VA.

Jackson, J.B.C., M.X. Kirby, W.H. Berger, K.A. Bjorndahl, L.W. Botsford, B.J. Bourque, R.H. Bradbury, R. Cooke, J. Erlandson, J.A. Estes, T.P. Hughes, S. Kidwell, C.B. Lange, H.S. Lenihan, J.M. Pandolfi, C.H. Peterson, R.S. Steneck, M.J. Tegner, and R.R. Werner. 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293: 629-638.

Jenks, G.F. and F.C. Caspall. 1971. Error on choroplethic maps - definition, measurement, reduction. *Annals of the Association of American Geographers* 61(2):217-244.

Johnson, R.K., T. Wiederholm, and D.M. Rosenberg. 1993. Freshwater biomonitoring using individual organisms, populations, and species assemblages of benthic macroinvertebrates. In *Freshwater Biomonitoring and Benthic Macroinvertebrates*, D.M. Rosenberg and V.H. Resh (eds.), pp. 40-125. Chapman & Hall, New York.

Johnson, W.C. 1994. Woodland Expansion in the Platte River, Nebraska: Patterns and Causes. *Ecological Monographs* 64:45-84.

Jongman, R.H.G., C.J.F. ter Braak and O.F.R. Tongeren (eds.). 1987. *Data Analysis in Community and Landscape Ecology*. Pudoc, Wageningen, NL.

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6(6):21 27.

Karr, J.R. and D.R. Dudley. 1981. Ecological perspective on water quality goals. *Environmental Management* 5:55-68.

Karr, J.R., K.D. Fausch, P.L. Angermeier, P.R. Yant, and I.J. Schlosser. 1986. Assessing biological integrity in running waters: a method and its rationale. *Illinois Natural History Survey Special Publication* 5.

Karr, J.R. 1987. Biological monitoring and environmental assessment: A conceptual framework. *Environmental Management* 11:249-256.

Karr, J.R., P.R. Yant, K.D. Fausch, and I.J. Schlosser. 1987. Spatial and temporal variability of the Index of Biotic Integrity in three midwestern streams. *Transactions of the American Fisheries Society* 116:1-11.

Karr, J.R. 1990. Biological integrity and the goal of environmental legislation: lessons from conservation biology. *Conservation Biology* 4:244-250.

Karr, J.R. 1998. Rivers as sentinels: Using the biology of rivers to guide landscape management. In. *River Ecology and Management: Lessons from the Pacific Coastal Ecosystems*, R. J. Naiman and R. E. Bilby (eds.), pp. 502-528. Springer, NY.

Karr, J.R. and E. W. Chu. 1999. *Restoring Life In Running Waters: Better Biological Monitoring*. Island Press, Washington, DC.

Karr, J.R. 2000. Health, integrity, and biological assessment: The importance of whole things. In *Ecological Integrity: Integrating Environment, Conservation, and Health*, D. Pimentel, L. Westra, and R. F. Noss (eds.), pp. 209-226. Island Press, Washington, DC.

Karr, J.R. and E.W. Chu. 2000. Sustaining living rivers. *Hydrobiologia* 422:1-14.

Karr, J.R. and E. M. Rossano. 2001. Applying public health lessons to protect river health. *Ecology and Civil Engineering* 4: 3-18.

Karr, J.R. and C. O. Yoder. 2004. Biological assessment and criteria improve total maximum daily load decision making. *Journal of Environmental Engineering* 130: 594-604.

Kaufmann, P.R., and R.M. Hughes. In Press. Geomorphic and anthropogenic influences on fish and amphibians in Pacific Northwest coastal streams. In *Influences of landscapes on stream habitats and biological assemblages*, R. M. Hughes, L. Wang, and P. Seelbach (eds.), pages xx-xx. American Fisheries Society Symposium xx. Bethesda, Maryland.

Kerans, B. L., and J. R. Karr. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. *Ecological Applications* 4:768-785.

Klemm, D.J., P.A. Lewis, F. Fulk, and J.M. Lazorchak. 1990. *Macroinvertebrate field and laboratory methods for evaluating the biological integrity of surface water*. EPA/a600/4-90/030. U.S. Environmental Protection Agency, Cincinntti, OH, 256 pp.

Klauda, R., P. Kazyak, S. Stranko, M. Southerland, N. Roth, and J. Chaillou. 1998. Maryland Biological Stream Survey: A state agency program to assess the impact of anthropogenic stresses on stream habitat quality and biota. *Environmental Monitoring and Assessment* 51: 299-316.

Krumholz, L.A. and W.L. Minckley. 1964. Changes in the fish population in the upper Ohio River following temporary pollution abatement. *Transactions of the American Fisheries Society* 93(1)1-5.

Kuzelka, R.D., C.A. Flowerday, R.N. Manley, B.C. Rundquist, S.J. Herrin, and C.A. Flowerday (eds.). 1993. *Flat water: A history of Nebraska and its water.* Resource Report No. 12, Conservation and Survey Division, Institute of Agriculture and Natural Resources, University of Nebraska – Lincoln.

Ladson, A. R., L. White, J. A. Doolan, B. L. Finlayson, B. T. Hart, S. Lake, and J. W. Tilleard. 1999. Development and testing of an Index of Stream Condition for waterway management in Australia. *Freshwater Biology* 41(2):453-468.

Lammert, M., and J. D. Allan. 1999. Assessing biotic integrity of streams: effects of scale in measuring the influence of land use/cover and habitat structure on fish and macroinvertebrates. *Environmental Management* 23:257-270.

Lattin, P. D., P. J. Wigington Jr., T. J. Moser, B. E. Peniston, D. R. Lindeman, and D. R. Oetter. 2004. Influence of remote sensing imagery source on quantification of riparian land cover-land use. *Journal of the American Water Resources Association* 40:215-227.

Leeson, M. A. 1885. *History of Montana 1739-1885*. Warner, Beers & Co., Chicago.

Legendre, P. and L. Legendre. 1998. *Numerical Ecology*. Second English Edition. Elsevier, Amsterdam.

Leonard, P. M., and D. J. Orth. 1986. Application and testing of an index of biotic integrity in small coolwater streams. *Transactions of the American Fisheries Society* 115:401-414.

Lorentz, C. M., G. M. Van Dijk, A. G. M. Van Hattum, and W. P. Cofino. 1997. Concepts in river ecology: implications for indicator development. *Regulated Rivers: Research and Management* 13:501-516.

Lubinski, K. and C. Theiling. 1999. Assessments and forecasts of the ecological health of the Upper Mississippi River System Floodplain Reaches. Ch. 2. in *Ecological Status and Trends of the Upper Mississippi River System 1998*, Lubinski and Theiling, eds. United States Geological Survey, Report of the Long Term Resource Monitoring Program, LTRMP 99-T001, LaCrosse, WI.

Ludwig, J.A. and J.F. Reynolds. 1988. *Statistical Ecology*. John Wiley & Sons, New York.

Malmqvist, B. and S.D. Rundle. 2002. Threats to the running water ecosystems of the world. *Environmental Conservation* 29: 134-153.

Margalef, R. 1963. On certain unifying principles in ecology. *American Naturalist* 97:357-374.

Margalef, R. 1981. Stress in ecosystems: a future approach. In *Stress effects on natural ecosystems*, G.W. Barrett and R. Rosenberg (eds.), pp. 281-289. Wiley, London, UK.

Martinez, M. E. 1998. What is problem-solving? *Phi Delta Kappan* 79:605-610.

McCormick, F.H., D.V. Peck, and D.P. Larsen. 2000. A comparison of ecological classification hierarchies for Mid-Atlantic stream fish assemblages. *Journal of North American Benthological Society* 19:385-404.

McCormick, F.H., R. M. Hughes, P. R. Kaufmann, D.V. Peck, J. L. Stoddard, and A.T. Herlihy. 2001. Development of an index of biotic integrity for the Mid-Atlantic Highlands. *Transactions of the American Fisheries Society* 130: 857-877.

Mebane, C. A., T. R. Maret, and R. M. Hughes. 2003. An index of biological integrity (IBI) for Pacific Northwest rivers. *Transactions of the American Fisheries Society* 132:239-261.

Meyer W.B., Turner B.L., II (eds.). 1994. *Changes in Land-Use and Land-Cover: A global perspective*. Cambridge University Press, Cambridge, UK.

Miller, R.R., J.D. Williams, and J.E. Williams. 1989. Extinctions of North American fishes during the past century. *Fisheries* 14:22-38.

Mills, H.B., W.C. Starrett, and F.C. Bellrose. 1966. Man's effect on the fish and wildlife of the Illinois River. *Illinois Natural History Survey Biological Notes* 57.

Miltner, R.J., D. White, and C.O. Yoder. 2003. The biotic integrity of streams in urban and suburbanizing landscapes. *Landscape and Urban Planning* 69(2004): 87-100.

Morley, S. A. and J. R. Karr. 2002. Assessing and restoring the health of urban streams in the Puget Sound Basin. *Conservation Biology* 16:1498-1509.

Moss, D., M.T. Furse, J.F. Wright, and P.D. Armitage. 1987. The prediction of the macroinvertebrate fauna of unpolluted running-water sites in Great Britain using environmental data. *Freshwater Biology* 17:41-52.

Moyle, P. B. 1986. Fish introductions into North America: patterns and ecological impact. In *Ecology of biological invasions of North America and Hawaii*, H. A. Mooney and J. A. Drake (eds.), pp. 27-43. Springer, New York.

Myers, R.A. and B. Worm. 2003. Rapid worldwide depletion of predatory fish communities. *Nature* 423:280-283.

Naiman and Decamps (eds.). 1990. *The ecology and management of aquatic-terrestrial ecotones*. Parthenon Publishing, Carnforth, UK.

Nash, M. S. and D.J. Chaloud. 2002. *Multivariate Analyses (Canonical Correlation Analysis and Partial Least Square, PLS) to Model and Assess the Association of Landscape Metrics to Surface Water Chemical and Biological Properties using Savannah River Basin Data*. EPA/600/R-02/091. United States Environmental Protection Agency, Washington, DC.

National Research Council. 1992. *Restoration of Aquatic Ecosystems*. National Academy Press, Washington DC.

National Research Council. 2001. *Assessing the TMDL Approach to Water Quality Management*. Water Science and Technology Board, Division on Earth and Life Studies. National Academy Press, Washington, DC.

Nehlsen, W., J. E. Williams, and J. A. Lichatowich. 1991. Pacific salmon at the crossroads: stocks at risk from California, Oregon, Idaho, and Washington. *Fisheries* 16(2):4-21.

Nerbonne B.A. and B. Vondracek. 2001. Effects of local land use on physical habitat, benthic macroinvertebrates, and fish in the Whitewater River, Minnesota, USA. *Environmental Management* 28:87-99.

Nilsson C., A. Ekblad, M. Gardfjell, and B. Carlberg. 1991. Long-term effects of river regulation on river margin vegetation. *Journal of Applied Ecology* 28:968-987.

Odum, E.P., J.T. Finn, and E.H. Franz. 1979. Perturbation theory and the subsidy-stress gradient. *BioScience* 29:349-352.

Odum, E.P. 1985. Trends expected in stressed ecosystems. *BioScience* 35:419-422.

Ohio Administrative Code 3745-1-07 c. 1978.

Ohio Department of Natural Resources (Ohio DNR). 1960. *Gazetteer of Ohio streams*. Ohio Water Plan Inventory Report Number 12. Division of Water, Columbus, Ohio. 179 pp.

Ohio Environmental Protection Agency (Ohio EPA). 1980. *Manual of surveillance methods and quality assurance practices*. Office of Wastewater Pollution Control, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1981. *5-year surface water monitoring strategy, 1982-1986*. Office of Wastewater Pollution Control, Division of Surveillance and Standards, Columbus, Ohio. 52 pp. + appendices.

Ohio Environmental Protection Agency (Ohio EPA). 1987. *Biological criteria for the protection of aquatic life: volumes I-III*. Ohio Environmental Protection Agency, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1988. *Ohio water quality inventory – 1988 305(b) report, volume I and executive summary*. E.T. Rankin, C.O. Yoder and D.A. Mishne (eds.). Ohio EPA, Division of Water Quality Monitoring and Assessment, Columbus, OH.

Ohio Environmental Protection Agency (Ohio EPA). 1989a. *Biological criteria for the protection of aquatic life: Volume III. Standardized biological field sampling and laboratory methods for assessing fish and macroinvertebrate communities.* Division of Water Quality Monitoring and Assessment, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1989b. *Addendum to biological criteria for the protection of aquatic life: Volume II. Users manual for biological field assessment of Ohio surface waters*. Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1990. *The use of biocriteria in the Ohio EPA surface water monitoring and assessment program.* Ohio Environmental Protection Agency, Division of Water Quality Planning and Assessment, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1992. *Ohio water resource inventory. Volume I: Summary, status, and trends*. E. Rankin, C. Yoder, and D. Mishne (eds.). Ohio Environmental Protection Agency, Division of Water Quality Monitoring and Assessment, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1996. *Justification and Rationale for Revisions to the Dissolved Oxygen Criteria in the Ohio Water Quality Standards*, OEPA Technical Bulletin MAS/1995-12-5, State of Ohio Environmental Protection Agency, Division of Surface Water, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1998. *Ohio Water Resource Inventory, Volume I: Summary, Status and Trends*. E. T. Rankin, C. O. Yoder, and D.Mishne, (eds.). Division of Surface Water, Ecological Assessment Section. Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1999a. *Associations between nutrients, habitat, and the aquatic biota of Ohio's rivers and streams*. Technical Bulletin MAS/1999-1-1, Division of Surface Water, Monitoring and Assessment Section, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1999b. *Total maximum daily load TMDL team report*. Division of Surface Water, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 2000. *Ohio Water Resource Inventory, Volume I: Summary, Status and Trends*. E. T. Rankin, C. O. Yoder, and D.Mishne, (eds.). Division of Surface Water, Ecological Assessment Section. Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 2003. *Ohio EPA's primary headwater habitat project: key findings*. Fact sheet. Division of Surface Water, Columbus, Ohio. 2 pp.

Ohio Environmental Protection Agency (Ohio EPA). 2004. *Total maximum daily loads for the Stillwater River Basin*. Division of Surface Water, Columbus, Ohio.

Omernik, J.M. 1987. Ecoregions of the conterminous United States. *Annals of the Association of American Geographers* 77:118-125.

Omernik, J.M. 1995. Ecoregions: A spatial framework for environmental management. In *Biological assessment and criteria: Tools for water resource planning and decision making*, W.S. Davis and T.P. Simon (eds.), pp. 49-62. Lewis Publishers, Boca Raton, FL.

Omernik, J.M. 2003. The misuse of hydrologic unit maps for extrapolation, reporting and ecosystem management. *Journal of the American Water Resources Association* 39(3):563-573.

Organization for Economic Cooperation and Development (OECD). 1993. Core set of indicators for environmental performance reviews. *Environmental Monographs* 83.

Patrick, R.M. 1950. Biological measure of stream conditions. *Sewage and Industrial Wastes* 22:926-938.

Patrick, R.M. 1953. Aquatic organisms as an aid in solving waste disposal problems. *Sewage and Industrial Wastes* 25(2):210-214.

Paul, M.J. and J.L. Meyer. 2001. The ecology of urban streams. *Annual Review of Ecology and Systematics* 32:333-365

Pianka, E. R. 1970. On r- and k-selection. *American Naturalist* 104:592-597.

Plafkin, J.L., M.T. Barbour, K.D. Porter, S.K. Gross, and R.M. Hughes. 1989. *Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish*. EPA 440-4-89-001. Office of Water Regulations and Standards, U.S. Environmental Protection Agency, Washington, DC.

Poff, N.L. and J.D. Allan. 1995. Functional organization in stream fish assemblages in relation to hydrologic variability. *Ecology* 76:606-627.

Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, and J.C. Stromberg. 1997. The natural flow regime: a paradigm for river conservation and restoration. *BioScience* 47:769-784.

Poff, N.L. and A.D. Huryn. 1998. Multi-scale determinants of secondary production in Atlantic salmon (*Salmo salar*) streams. *Canadian Journal of Fisheries and Aquatic Sciences* 55(Suppl. 1):201-217.

Potter, K.M., F.W. Cubbage, and R.H. Schaberg. 2005. Multiple-scale landscape predictors of benthic macroinvertebrate community structure in North Carolina. *Landscape and Urban Planning* 71:77-90.

Pretty, J.L., S.S.C. Harrison, D.J. Shepherd, C. Smith, A.G. Hildrew, and R.D. Hey. 2003. River rehabilitation and fish populations: assessing the benefit of instream structures. *Journal of Applied Ecology* 40:251-265.

Pulliam, H. R. 1988. Sources, sinks, and population regulation. *American Naturalist* 132: 652-661.

Quinn, J.M. 2000. Effects of pastoral development. In *New Zealand Stream Invertebrates: Ecology and Implications for Management*, K.J. Collier and M.J. Winterbourn (eds.), pp. 208-229. Caxton, Christchurch, NZ.

Rabeni, C.F. 1977. Benthic macroinvertebrate communities of the Penobscot River, Maine, with special reference to their role as water quality indicators. Ph.D thesis, University of Maine, Orono. 169 pp.

Rabeni, C. F., S. P. Davies, and K. E. Gibbs. 1988. Benthic macroinvertebrate response to pollution abatement: structural changes and functional implications. *Water Resources Bulletin* 21:489-497

Rankin, E.T. 1989. *The qualitative habitat evaluation index (QHEI), rationale, methods, and application*. Ohio Environmental Protection Agency, Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, OH.

Rankin, E.T. 1995. The use of habitat indices in water resource quality assessments. In *Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making*, W.S. Davis and T.P. Simon (eds.), pp. 181-208. Lewis Publishers, Boca Raton, FL.

Rapport, D. J., H. A. Regier, and T. C. Hutchinson. 1985. Ecosystem behavior under stress. *American Naturalist* 125:617-640.

Reeves, G. H., L. E. Benda, K. M. Burnett, P. A. Bisson, and J. R. Sedell. 1995. A disturbance-based ecosystem approach to maintaining and restoring freshwater habitats of evolutionarily significant units of anadromous salmonids in the Pacific Northwest. In *Evolution and the aquatic ecosystem: defining unique units in population conservation*, J. L. Nielsen (ed.), pp. 334-349. American Fisheries Society Symposium 17, Bethesda, Maryland, USA.

Regier, H.A., R.L. Welcomme, R.J. Steedman, and H.F. Henderson. 1989. Rehabilitation of degraded river ecosystems. In *Proceedings of the international large river symposium (LARS)*, D.P. Dodge (ed.), Canadian Special Publication of Fisheries and Aquatic Sciences 106, pp. 86-97. Department of Fisheries and Oceans, Ottawa, Ontario.

Reisner, M. 1986. *Cadillac desert: the American west and its disappearing water*. Penguin Books, New York.

Riebesell, J.F. 1974. Paradox of enrichment in competitive systems. *Ecology* 55: 183-187.

Richards, C., L.B. Johnson, and G.E. Host. 1996. Landscape-scale influences on stream habitats and biota. *Canadian Journal of Fisheries and Aquatic Sciences* 53:295-311.

Richards, C., L.B. Johnson. 1998. Landscape perspectives on ecological risk assessment. In *Risk Assessment: Logic and Measurement*, M.C. Newman and C. Strojan (eds). Ann Arbor Press.

Richardson, J. S., T. J. Lissimore, M. C. Healey, and T. G. Northcote. 2000. Fish communities of the lower Fraser River (Canada) and a 21-year contrast. *Environmental Biology of Fishes* 59:125-140.

Richter, B.D., D.P. Braun, M.A. Mendelson, and L.L. Master. 1997. Threats to imperiled freshwater fauna. *Conservation Biology* 11:1081-1093.

Rinne, J.N., R.M. Hughes, and B. Calamusso. 2005. *Historical changes in large river fish assemblages of the Americas*. American Fisheries Society, Symposium 45, Bethesda, Maryland.

Roni, P., T.J. Beechie, R.E. Bilby, F.E. Leonetti, M.M. Pollock and G.R. Pess. 2002. A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific Northwest watersheds. *North American Journal of Fisheries Management* 22:1-20.

Roth, N.E., J.D. Allan, D.L Erickson. 1996. Landscape influences on stream biotic integrity assessed at multiple spatial scales. *Landscape Ecology* 11:141-156.

Roy, A.H., A.D. Rosemond, M.J. Paul, D.S. Leigh, and J.B. Wallace. 2003. Stream macroinvertebrate response to catchment urbanisation (Georgia, USA). *Freshwater Biology* 48:329-346.

Schindler, D.W. 1987. Detecting ecosystem responses to anthropogenic stress. *Canadian Journal of Fisheries and Aquatic Sciences* 44:6-25.

Shear, H., N. Stadler-Salt, P. Bertram, and P. Horvatin. 2003. The development and implementation of indicators of ecosystem health in the Great Lakes basin. *Environmental Monitoring and Assessment* 88: 119-152.

Shelton, A.D. and K.A. Blocksom. 2004. *A Review of Biological Assessment Tools and Biocriteria in New England States*. EPA/600/R-04/168. U.S. Environmental Protection Agency, Cincinnati, OH.

Simon, Thomas P. Ed. 2003. *Biological Response Signatures: Indicator patterns using aquatic communities*. CRC Press, Boca Raton, FL.

Simpson J.C and Norris R.H. 2000. Biological assessment of river quality: development of AusRivAS models and outputs. In *Assessing the Biological Quality of Fresh Waters: RIVPACS and Other Techniques*, J.F. Wright, D.W. Sutcliffe and M.T. Furse (eds.), pp. 125-142. Freshwater Biological Association, Ambleside, UK.

Slivitsky, M. 2001. *A Literature Review on Cumulative Ecological Impacts Of Water Use and Changes in Levels and Flows*. The Great Lakes Commission, October 15, 2001. 63 pp.

Smith, P.W. 1971. Illinois streams: a classification based on their fishes and an analysis of factors responsible for the disappearance of native species. *Illinois Natural History Survey Biological Notes* 76.

Smith, P.W. 1979. *The fishes of Illinois*. Univ. Illinois Press, Urbana, IL. 314 pp.

Smith, R.A., G.E. Schwartz and R.B. Alexander. 1997. Regional interpretation of water quality monitoring data. *Water Resources Research* 33: 2781-2798.

Smith, R.W., M. Bergen, S.B. Weisberg, D. Cadien, A. Dalkey, D. Montagne, J.K. Stull, and R.G. Velarde. 2001. Benthic response index for assessing infaunal communities on the southern California mainland shelf. *Ecological Applications* 11: 1073-1087.

Smith, S.H. 1972. The future of salmonid communities in the Laurentian Great Lakes. *Journal of the Fisheries Research Board of Canada* 29:951-957.

Snyder, C.D., J.A. Young, R. Villella, and D.P. Lemarie. 2003. Influences of upland and riparian land use patterns on stream biotic integrity. *Landscape Ecology* 18(7):647-664.

Solimini, A. G., G. A. Tarallo, and G. Carchini. 1997. Life history and species composition of the damselfly assemblage along the urban tract of a river in central Italy. *Hydrobiologia* 382:63-86.

Southerland, M.T. and J.B. Stribling. 1995. Status of biological criteria development and implementation. In *Biological assessment and criteria: tools for water resource planning and decision making*, W.S. Davis and T.P. Simon (eds), pp. 81-96. Lewis Publishers, Boca Raton, Florida.

Sparks, R.E. and W.C. Starrett. 1975. An electrofishing survey of the Illinois River, 1959-1974. *Illinois Natural History Survey Bulletin* 295-316.

State of Maine. 1985. Maine Laws Ch. 698 15 (in part). An Act to Amend the Classification System for Maine Waters and Change the Classifications of Certain Waters. Augusta, Maine.

State of Maine. 2003. Code of Maine Rules 06-096. Chapter 579: Classification Attainment Evaluation Using Biological Criteria for Rivers and Streams. Augusta, Maine.

State of Ohio. 2003. Ohio Administrative Code at OAC Chapter 3745-1. State of Ohio Water Quality Standards. Columbus, Ohio.

Steedman, R. J. 1988. Modification and assessment of an index of biotic integrity to quantify stream quality in southern Ontario. *Canadian Journal of Fisheries and Aquatic Sciences* 45:492-501.

Stevenson, R. J., Y. Pan, and P. Vaithiyanathan. 2002. Ecological assessment and indicator development in wetlands: the case of algae in the Everglades, USA. *Verhandlungen Internationale Vereinigung für Theoretische und Andgewandte Limnologie* 28:1248-1252.

Stoddard, J.L., D.P. Larsen, C.P. Hawkins, R.K. Johnson, and R.H. Norris. In Press. Setting expectations for the ecological condition of running waters: the concept of reference condition. *Ecological Applications*.

ten Brink, B.J.E., S.H. Hosper, and F. Colijn. 1991. A quantitative method for description and assessment of ecosystems: the AMOEBA approach. *Marine Pollution Bulletin* 23:265-270.

ten Brink, B.J.E. 1991. The AMOEBA approach as a useful tool for establishing sustainable development. In *In search of indicators of sustainable development*, O. Kuik and H. Vebruggen (eds.), pp. 71-87. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Thoma, R.F. 1999. Biological monitoring and an index of biotic integrity for Lake Erie's nearshore waters. In *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities*, T.P. Simon (ed.), pp. 417-462. CRC Press, Boca Raton, FL.

Tockner, K. and J. A. Stanford. 2002. Riverine flood plains: present state and future trends. *Environmental Conservation* 29:308–330.

Trautman, M.B. 1957. *The Fishes of Ohio*. The Ohio State Univ. Press, Columbus, OH. 683 pp.

Trautman, M.B. 1981. *The Fishes of Ohio. Second edition*. Ohio State University Press, Columbus, OH.

Troelstrup, N. H. Jr., and G. L. Hergenrader. 1990. Effect of hydropower peaking flow fluctuations on community structure and feeding guilds of invertebrates colonizing artificial substrates in a large impounded river. *Hydrobiologia* 199:217-228.

Tsai, C. 1968. Effects of chlorinated sewage effluents on fishes in upper Patuxent River, Maryland. *Chesapeake Science* 9(2):83-93.

Tsai, C. 1973. Water quality and fish life below sewage outfalls. *Transactions of the American Fisheries Society* 102(2):281-292.

United States and Canada. 1987. Great Lakes Water Quality Agreement, as amended by protocol signed November 18, 1987. Windsor, ON.

U.S. Environmental Protection Agency (U.S. EPA). 1985. *Questions and Answers on Antidegradation*. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. Environmental Protection Agency (U.S. EPA). 2000a. *Mid-Atlantic Highlands streams assessment*. EPA/903/R-00/015. U.S. Environmental Protection Agency, Philadelphia, PA.

U.S. Environmental Protection Agency (U.S. EPA). 2000b. *Stressor Identification Guidance Document*. EAP/822/B-00/025. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. Environmental Protection Agency (U.S. EPA). 2002a. *Summary of Biological Assessment Programs and Biocriteria Development for States, Tribes, Territories, and Interstate Commissions: Streams and Wadeable Rivers*. EPA-822-R-02-048. U.S. Environmental Protection Agency, Office of Environmental Information and Office of Water, Washington, DC.

U.S. Environmental Protection Agency (U.S. EPA). 2002b. *National Water Quality Inventory 2000*. EPA-841-R-02-001. U.S. Environmental Protection Agency Office of Water, Washington DC 20460, August 2002.

U.S. Environmental Protection Agency (U.S. EPA). 2003. *Mid-Atlantic Integrated Assessment (MAIA): State of the flowing waters report*. U.S. Environmental Protection Agency, Corvallis, Oregon.

van Dam, H., A. Mertenes, and J. Sinkeldam. 1994. A coded checklist and ecological indicator values of freshwater diatoms from the Netherlands. *Netherlands Journal of Aquatic Ecology* 28:117-33.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* 37: 130-137.

Vansickle, J., and R.M. Hughes. 2000. Classification strengths of ecoregion, catchments and geographic clusters for aquatic vertebrates in Oregon. *Journal of the North American Benthological Society* 19:370-384.

Vitousek P.M., J.D. Aber, R.W. Howarth, G.E. Likens, P.A. Matson, D.W. Schindler, W.H. Schlesinger, and D.G. Tilman. 1997. Human alteration of the global nitrogen cycle: sources and consequences. *Ecological Applications* 7:737-750.

Vollenweider, R.A. 1968. *The scientific basis of lake and stream eutrophication with particular reference to phosphorus and nitrogen as eutrophication factors*. Technical report DAS/DSI/68.27. Organization for Economic Cooperation and Development, Paris.

Waite, I.R., and K.D. Carpenter. 2000. Associations among fish assemblage structure and environmental variables in Willamette basin streams, Oregon. *Transactions of the American Fisheries Society* 129:754-770.

Wallace, J.B., J.R. Webster, and W.R. Woodall. 1977. The role of filter feeders in flowing waters. *Archiv fur Hydrobiologie* 79:506-532.

Wallin, M., T. Wiederholm, and R.K. Johnson. 2003. *Guidance on establishing reference conditions and ecological status class boundaries for inland surface waters. European Union Common Implementation Strategy (CIS) for the Water Framework Directive.* Swedish University of Agricultural Sciences, Uppsala, Sweden.

Walsh, C.J., A.K. Sharp, P.F. Breen, and J.A. Sonneman. 2001. Effects of urbanisation on streams of the Melbourne region Victoria, Australia, I. Benthic macroinvertebrate communities. *Freshwater Biology* 46:553-565.

Wang, L., J. Lyons, P. Kanehl, and R. Gatti. 1997. Influences of Watershed Land Use on Habitat Quality and Biotic Integrity in Wisconsin Streams. *Fisheries* 22(6):6-12.

Wang, L., J. Lyons, P. Kanehl, R. Bannerman, and E. Emmons. 2000. Watershed Urbanization and Changes in Fish Communities in Southeastern Wisconsin Streams. *Journal of American Water Resources Association* 36(5):1173-1189.

Wang, L., J. Lyons, P. Kanehl, and R. Bannerman. 2001. Impacts of urbanization on stream habitat and fish across multiple spatial scales. *Environmental Management* 28:255-266.

Wang, L., J. Lyons, and P. Kanehl. 2002. Effects of watershed best management practices on habitat and fish in Wisconsin streams. *Journal of American Water Resources Association* 38:663-80.

Wang, L., P. Seelbach, and J. Lyons. In Press. Effects of levels of human disturbance on the influence of catchment, riparian, and reach scale factors on fish assemblages. In *Influences of landscapes on stream habitats and biological assemblages*, R. M. Hughes, L. Wang, and P. Seelbach (eds.), Pages xx-xx. American Fisheries Society Symposium xx. Bethesda, Maryland.

Ward, J.V. 1998. Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. *Biological Conservation* 83:269-278.

Welsh, H. H. Jr. and L. M. Ollivier. 1998. Stream amphibians as indicators of ecosystem stress: a case study from California's redwoods. *Ecological Applications* 8: 1118-1132.

Whittier, T. R., A. T. Herlihy, and S. M. Pierson. 1995. Regional susceptibility of Northeast lakes to zebra mussel invasion. *Fisheries* 20(6):20-27.

Whittier, T. R. and T. M. Kincaid. 1999. Introduced fish in Northeastern USA lakes: Regional extent, dominance and effect on native species richness. *Transactions of the American Fisheries Society* 128:769-783.

Whittier, T. R., J. L. Stoddard, R. M. Hughes, and G.A. Lomnicky. In Press. Associations among catchment- and site-scale disturbance indicators and biological assemblages at least- and most-disturbed stream and river sites in the western USA. In *Influences of landscapes on stream habitats and biological assemblages*, R. M. Hughes, L. Wang, and P. Seelbach (eds.), Pages xx-xx. American Fisheries Society Symposium xx. Bethesda, Maryland.

Wiens, J. A. 2002. Riverine landscapes: taking landscape ecology into the water. *Freshwater Biology* 47:501–515.

Wickham, J.D., K.B. Jones, K.H. Riitters, R.V. O'Neill, R.D. Tankersley, E.R. Smith, A.C. Neale and D.J. Chaloud. 1999. An integrated environmental assessment of the US Mid-Atlantic Region. *Environmental Management* 24:553–560.

Wilhelm, J.G.O., J.D. Allan, K.J. Wessell, R.W. Merritt and K.W. Cummins. Habitat evaluation of non-wadeable rivers. Unpublished manuscript.

Williams, J.E., C.A. Wood, and M.P. Dombeck (eds.). 1997. *Watershed restoration: principles and practices*. American Fisheries Society. Bethesda, Maryland.

Wissmar, R.C. and R.L. Beschta. 1998. Restoration and management of aquatic ecosystems: a catchment perspective. *Freshwater Biology* 40: 571-585.

Witte, F., T. Goldschmidt, J. Wanink, M. Van Oijen, K. Goudswaard, E. Witte-Maas, N. Bouton. 1992. The destruction of an endemic species flock: quantitative data on the decline of the haplochromine cichlids of Lake Victoria. *Environmental Biology of Fishes* 34:1-28.

Wright, J.F., D. Moss, P.D. Armitage, and M.T. Furse. 1984. A preliminary classification of running-water sites in Great Britain based on macro-invertebrate species and the prediction of community type using environmental data. *Freshwater Biology* 14: 221-256.

Wright, J.F., M.T. Furse, and P.D. Armitage. 1994. Use of macroinvertebrates communities to detect environmental stress in running waters. In *Water quality and stress indicators in marine and freshwater systems: linking levels of organisation*, D. W. Sutcliffe (ed.), pp. 15-34. Freshwater Biological Association, Ambleside.

Wright, J.F. 1995. Development of a system for predicting the macroinvertebrate fauna of flowing waters. *Australian Journal of Ecology* 20:181-197.

Wright, J.F. 2000. An introduction to RIVPACS. In *Assessing the Biological Quality of Fresh Waters*, J.F. Wright, D.W. Sutcliffe and M.T. Furse (eds.), pp. 1-24. Freshwater Biological Association, Ambleside, UK.

Wurtz, C.B. 1955. Stream biota and stream pollution. *Sewage & Industrial Wastes* 27(11):1270-1278.

Yang, J. R., F. R. Pick, and P. B. Hamilton. 1996. Changes in the planktonic diatom flora of a large mountain lake in response to fertilization. *Journal of Phycology* 32:232-243.

Yoder, C.O. 1978. *A proposal for the evaluation of water quality conditions in Ohio's rivers and streams*. Division of Industrial Wastewater, Columbus, OH. 43 pp.

Yoder, C.O. 1995. Policy issues and management applications for biological criteria. In *Biological assessment and criteria: tools for water resource planning and decision making*, W.S. Davis and T.P. Simon (eds.), pp. 327-343. Lewis Publishers, Boca Raton, FL.

Yoder, C.O. and E.T. Rankin. 1995a. Biological response signatures and the area of degradation value: New tools for interpreting multimetric data. In *Biological assessment and criteria: tools for water resource planning and decision making,* Davis, W.S. and Simon, T.P. (eds.), pp. 263-286. Lewis Publishers, Boca Raton, FL.

Yoder, C.O. and E.T. Rankin. 1995b. Biological criteria program development and implementation in Ohio. In *Biological assessment and criteria: tools for water resource planning and decision making*, Davis, W.S. and Simon, T.P. (eds.), pp. 109-144. Lewis Publishers, Boca Raton, FL.

Yoder, C.O. and E.T. Rankin. 1998. The role of biological indicators in a state water quality management process. *Environmental Monitoring and Assessment* 51:61-68.

Yoder, C.O. and E.T. Rankin. 1999. Biological criteria for water resource management. In *Measures of Environmental Performance and Ecosystem Condition*, P.C. Schulze and R.A. Frosch (eds.), pp. 227-259. National Academy of Engineering, National Academy Press, Washington, DC.

Yoder, C.O. and M. A Smith. 1999. Using fish assemblages in a state biological assessment and criteria program: essential concepts and considerations. In *Assessing the sustainability and biological integrity of water resources using fish communities*, T.P. Simon (ed.), pp. 17-56. CRC Press, Boca Raton, FL.

Yoder, C.O., R.J. Miltner, and D. White. 2000. Using biological criteria to assess and classify urban streams and develop improved landscape indicators. In *National Conference on Tools for Urban Water Resource Management and Protection*, S. Minamyer, J. Dye, and S. Wilson (eds.), pp. 32-44. EPA/625/R-00/001. U.S. EPA, Cincinnati, OH.

Yoder, C.O. and J.E. DeShon. 2003. Using biological response signatures within a framework of multiple indicators to assess and diagnose causes and sources of impairments to aquatic assemblages in selected Ohio rivers and streams. In *Biological response signatures: indicator patterns using aquatic communities*, T. P. Simon (ed.), pp. 23-81. CRC Press, Boca Raton, FL.

Yuan, L. L. and S. B. Norton. 2003. Comparing the responses of macroinvertebrate metrics to increasing stress. *Journal of the North American Benthological Society* 22:308-322.

Zaroban, D. W., M. P. Mulvey, T. R. Maret, R. M. Hughes, and G. D. Merritt. 1999. Classification of species attributes for Pacific Northwest freshwater fishes. *Northwest Science* 73:81-93.

Zeeb, B. A., C. E Christie, J. P. Smol, D. L. Findlay, H. J. Kling, and H. J. B. Birks. 1974. Responses of diatom and chrysophyte assemblages in lake-227 sediments to experimental eutrophication. *Canadian Journal of Fisheries and Aquatic Sciences* 51:2300-2311.

Glossary

Ambient Monitoring	sampling and evaluation of receiving waters not necessarily associated with episodic perturbations
Allochthonous	organic matter that was produced outside the system (e.g., wood, leaves, berries, insects etc.)
Anadromy	fish that live most of life in oceans or lakes and migrate to streams to spawn
Antidegradation Statement	statement that protects existing uses, prevents degradation of high quality waterbodies unless certain determinations are made, and which protects the quality of outstanding national resource waters
Aquatic Assemblage	an association of interacting populations of organisms in a given waterbody, for example, fish assemblage or a benthic macroinvertebrate assemblage
Aquatic Community	an association of interacting assemblages in a given waterbody, the biotic component of an ecosystem
Aquatic Life Use	a beneficial use designation in which the waterbody provides suitable habitat for survival and reproduction of desirable fish, shellfish, and other aquatic organisms; classifications specified in State water quality standards relating to the level of protection afforded to the resident biological community by the State agency
Attribute	measurable part or process of a biological system
Autochthonous	organic matter produced within the system (e.g., algae, macrophytes)
BEAST	used in parts of Canada, the BEAST (BEnthic Assessment of SedimenT) multivariate technique uses a probability model based on taxa ordination space and the "best fit" of the test site(s) to the probability ellipses constructed around the reference site classes
Beneficial Uses	desirable uses that water quality should support. Examples are drinking water supply, primary contact recreation (such as swimming), and aquatic life support.
Benthic Macroinvertebrates or Benthos	animals without backbones, living in or on the sediments, 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). Also referred to as benthos, infauna, or macrobenthos
Best Management Practice	an engineered structure or management activity, or combination of these, that eliminates or reduces an adverse environmental effect of a pollutant
Biological Assessment or Bioassessment	an evaluation of the biological condition of a waterbody using surveys of the structure and function of a community of resident biota.
Biological Criteria or Biocriteria	**Scientific meaning**: quantified values representing the biological condition of a waterbody as measured by structure and function of the aquatic communities typically at reference condition. **Regulatory meaning**: narrative descriptions or numerical values of the structure and function of aquatic communities in a waterbody necessary to protect the designated aquatic life use, implemented in, or through water quality standards.

Biological Diversity or Biodiversity	refers to the variety and variability among living organisms and the ecological complexes in which they occur. Diversity can be defined as the number of different items and their relative frequencies. For biological diversity, these items are organized at many levels, ranging from complete ecosystems to the biochemical structures that are the molecular basis of heredity. Thus, the term encompasses different ecosystems, species, and genes.
Biological Indicator or Bioindicator	an organism, species, assemblage, or community characteristic of a particular habitat, or indicative of a particular set of environmental conditions
Biological Integrity	the ability of an aquatic ecosystem to support and maintain a balanced, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats within a region
Biological Monitoring or Biomonitoring	use of a biological entity as a detector and its response as a measure to determine environmental conditions. Ambient biological surveys and toxicity tests are common biological monitoring methods.
Biological Survey or Biosurvey	collecting, processing, and analyzing a representative portion of the resident aquatic community to determine its structural and/or functional characteristics
Bioregion	any geographical region characterized by a distinctive flora and/or fauna
Clean Water Act	an act passed by the U.S. Congress to control water pollution (formally referred to as the Federal Water Pollution Control Act of 1972). Public Law 92-500, as amended. 33 U.S.C. 1251 et seq.
Clean Water Act 303(d)	This section of the Act requires States, territories, and authorized Tribes to develop lists of impaired waters for which applicable water quality standards are not being met, even after point sources of pollution have installed the minimum required levels of pollution control technology. The law requires that these jurisdictions establish priority rankings for waters on the lists and develop TMDLs for these waters. States, territories, and authorized Tribes are to submit their list of waters on April 1 in every even-numbered year.
Clean Water Act 305(b)	biennial reporting requires description of the quality of the Nation's surface waters, evaluation of progress made in maintaining and restoring water quality, and description of the extent of remaining problems
Cosmopolitan Species	species with worldwide distribution or influence where there is suitable habitat
Criteria	limits on a particular pollutant or condition of a waterbody presumed to support or protect the designated use or uses of a waterbody. Criteria may be narrative or numeric.
DELT Anomalies	percentage of Deformities, Erosions (e.g., fins, barbels), Lesions and Tumors on fish assemblages
Designated Uses	those uses specified in water quality standards for each waterbody or segment whether or not they are being attained
Disturbance	human activity that alters the natural state and can occur at or across many spatial and temporal scales
Ecological Integrity	the condition of an unimpaired ecosystem as measured by combined chemical, physical (including physical habitat), and biological attributes. Ecosystems have integrity when they have their native components (plants, animals and other organisms) and processes (such as growth and reproduction) intact.
Ecoregion	a relatively homogeneous ecological area defined by similarity of climate, landform, soil, potential natural vegetation, hydrology, or other ecologically relevant variables

Ecosystem-level functions	processes performed by ecosystems, including, among other things, primary and secondary production; respiration; nutrient cycling; decomposition. See discussion concerning how this function is considered in the draft biological condition gradient in transmittal memorandum under "outstanding issues" and in the file: attribute explanation.
Existing Uses	those uses actually attained in a waterbody on or after November 28, 1975, whether or not they are included in the water quality standards (November 28, 1975 is the date on which U.S. EPA promulgated its first water quality standards regulation). Because an existing use has been attained, it cannot be removed unless uses are added that require more stringent criteria.
Function	processes required for normal performance of a biological system (may be applied to any level of biological organization)
Heterotrophic	obtaining organic matter from other organisms rather than synthesizing it from inorganic substrates
Hyporheic Zone	area below the streambed where water percolates through spaces between the rocks and cobbles. Also known as the interface between surface water and groundwater.
Historical Data	data sets from previous studies, which can range from handwritten field notes to published journal articles
Historically documented taxa	taxa known to have been supported in a waterbody or region prior to enactment of the Clean Water Act, according to historical records compiled by state or federal agencies or published scientific literature
Index of Biological/Biotic Integrity	an integrative expression of site condition across multiple metrics. An index of biological integrity is often composed of at least seven metrics
Invasive species	a species whose presence in the environment causes economic or environmental harm or harm to human health. Native species or non-native species may show invasive traits, although this is rare for native species and relatively common for non-native species. (Please note - this term is not currently included in the biological condition gradient)
Life-history requirements	environmental conditions necessary for completing life cycles (including, among other things, reproduction, growth, maturation, migration, dispersal)
Lithophils	organisms that thrive on rocks or stones
Lithopelagophils	organisms that spawn in open gravelly areas and have no guarding behavior
Maintenance of populations	sustained population persistence; associated with locally successful reproduction and growth
Metric	a calculated term or enumeration representing some aspect of biological assemblage, function, or other measurable aspect and is a characteristic of the biota that changes in some predictable way with increased human influence
Multimetric Index	an index that combines indicators, or metrics, into a single index value. Each metric is tested and calibrated to a scale and transformed into a unitless score prior to being aggregated into a multimetric index. Both the index and metrics are useful in assessing and diagnosing ecological condition. See Index of Biotic Integrity.
Multivariate Analysis	statistical methods (e.g. ordination or discriminant analysis) for analyzing physical and biological community data using multiple variables
Narrative Biocriteria	written statements describing the structure and function of aquatic communities in a waterbody necessary to protect a designated aquatic life use

Native	an original or indigenous inhabitant of a region; naturally present
Non-detrimental effect	does not displace native taxa
Non-native or intentionally introduced species	with respect to a particular ecosystem, any species that is not found in that ecosystem. Species introduced or spread from one region of the U.S. to another outside their normal range are non-native or non-indigenous, as are species introduced from other continents.
Numeric Biocriteria	specific quantitative measures of the structure and function of aquatic communities in a waterbody necessary to protect a designated aquatic life use
Periphyton	a broad organismal assemblage composed of attached algae, bacteria, their secretions, associated detritus, and various species of microinvertebrates
Piscivore	predatory fish that eats mainly other fish
Polyphils	organism with no specialized spawning requirements, behavior, or preferred habitat
P/R	ratio of photosynthesis to respiration in a system
Presently Attained Uses	those uses actually being attained in a waterbody at the present moment
Rapid Bioassessment Protocols	cost-effective techniques used to survey and evaluate the aquatic community to detect aquatic life impairments and their relative severity
Reference Condition (Biological Integrity)	the condition that approximates natural, un-impacted conditions (biological, chemical, physical, etc.) for a waterbody. Reference condition (Biological Integrity) is best determined by collecting measurements at a number of sites in a similar waterbody class or region under undisturbed or minimally disturbed conditions (by human activity), if they exist. Since undisturbed or minimally disturbed conditions may be difficult or impossible to find, least disturbed conditions, combined with historical information, models or other methods may be used to approximate reference condition as long as the departure from natural or ideal is understood. Reference condition is used as a benchmark to determine how much other water bodies depart from this condition due to human disturbance.
Reference Condition (Biological Integrity), cont.	**Least Disturbed Condition:** the best available existing conditions with regard to physical, chemical, and biological characteristics or attributes of a waterbody within a class or region. These waters have the least amount of human disturbance in comparison to others within the waterbody class, region or basin. Least disturbed conditions can be readily found, but may depart significantly from natural, undisturbed conditions or minimally disturbed conditions. Least disturbed condition may change significantly over time as human disturbances change. **Minimally Disturbed Condition:** the physical, chemical, and biological conditions of a waterbody with very limited, or minimal, human disturbance in comparison to others within the waterbody class or region. Minimally disturbed conditions can change over time in response to natural processes. **Best Attainable Condition:** a condition that is equivalent to the ecological condition of (hypothetical) least disturbed sites where the best possible management practices are in use. This condition can be determined using techniques such as historical reconstruction, best ecological judgment and modeling, restoration experiments, or inference from data distributions

Reference Site	a site selected for comparison with sites being assessed. The type of sites selected and the type of comparative measures used will vary with the purpose of the comparisons. For the purposes of assessing the ecological condition of sites, a reference site is a specific locality on a waterbody that is undisturbed or minimally disturbed and is representative of the expected ecological integrity of other localities on the same waterbody or nearby waterbodies
Refugia	accessible microhabitats or regions within a stream reach or watershed where adequate conditions for organism survival are maintained during circumstances that threaten survival, e.g., drought, flood, temperature extremes, increased chemical stressors, habitat disturbance, etc.
Regional Reference Condition	a description of the chemical, physical, or biological condition based on an aggregation of data from reference sites that are representative of a waterbody type in an ecoregion, subecoregion, watershed, or political unit
Rheophils	organisms that flourish in free-flowing water
Restoration	the re-establishment of pre-disturbance aquatic functions and related physical, chemical, and biological characteristics
River Invertebrate Prediction and Classification System (RIVPACS)	a predictive method developed for use in the United Kingdom to assess water quality using a comparison of observed biological species distributions to those expected to occur based on a model derived from reference data
Sensitive taxa	intolerant to a given anthropogenic stress; first species affected by the specific stressor to which they are "sensitive" and the last to recover following restoration
Sensitive or regionally endemic taxa	taxa with restricted, geographically isolated distribution patterns (occurring only in a locale as opposed to a region), often due to unique life history requirements. May be long-lived, late maturing, low fecundity, limited mobility, or require mutualist relation with other species. May be among listed E/T or special concern species. Predictability of occurrence often low, therefore, requires documented observation. Recorded occurrence may be highly dependent on sample methods, site selection and level of effort.
Sensitive - rare taxa	naturally occur in low numbers relative to total population density but may make up large relative proportion of richness. May be ubiquitous in occurrence or may be restricted to certain micro-habitats, but because of low density, recorded occurrence is dependent on sample effort. Often stenothermic (having a narrow range of thermal tolerance) or cold-water obligates; commonly k-strategists (populations maintained at a fairly constant level; slower development; longer life-span). May have specialized food resource needs or feeding strategies. Generally intolerant to significant alteration of the physical or chemical environment; are often the first taxa observed to be lost from a community.
Sensitive - ubiquitous taxa	ordinarily common and abundant in natural communities when conventional sample methods are used. Often having a broader range of thermal tolerance than Sensitive- Rare taxa. These are taxa that comprise a substantial portion of natural communities, and that often exhibit negative response (loss of population, richness) at mild pollution loads or habitat alteration.
Spatial and temporal ecosystem connectance	access or linkage (in space/time) to materials, locations, and conditions required for maintenance of interacting populations of aquatic life; the opposite of fragmentation; necessary for metapopulation maintenance and natural flows of energy and nutrients across ecosystem boundaries
Stressors	physical, chemical, and biological factors that adversely affect aquatic organisms

Structure	taxonomic and quantitative attributes of an assemblage or community, including species richness and relative abundance structurally & functionally redundant attributes of the system = characteristics, qualities, or processes that are represented or performed by more than one entity in a biological system
Subcategorized Uses	States and Tribes may adopt subcategories of a use and set the appropriate criteria to reflect varying needs of such subcategories of uses, for instance, to differentiate between cold water and warm water fisheries
Taxa	a grouping of organisms given a formal taxonomic name such as species, genus, family, etc.
Taxa of intermediate tolerance	comprise a substantial portion of natural communities; may be r-strategists (early colonizers with rapid turn-over times; "boom/bust population characteristics). May be eurythermal (having a broad thermal tolerance range). May have generalist or facultative feeding strategies enabling utilization of relatively more diversified food types. Readily collected with conventional sample methods. May increase in number in waters with moderately increased organic resources and reduced competition but are intolerant of excessive pollution loads or habitat alteration.
Tolerant taxa	comprise a low proportion of natural communities. Taxa often are tolerant of a broader range of environmental conditions and are thus resistant to a variety of pollution or habitat induced stress. They may increase in number (sometimes greatly) in the absence of competition. Commonly r-strategists (early colonizers with rapid turn-over times; "boom/bust" population characteristics), able to capitalize when stress conditions occur. Last survivors.
Total Maximum Daily Load	the sum of the allowable loads of a single pollutant from all contributing point and nonpoint sources; calculation of the maximum amount of a pollutant a waterbody can receive and still meet water quality standards and an allocation of that amount to the pollutant's source
Use Attainability Analysis	structured scientific assessment of the physical, chemical, biological or economic factors affecting attainment of the uses of waterbodies
Water Quality Standards	a law or regulation that consists of the designated use or uses of a waterbody, the narrative or numerical water quality criteria (including biocriteria) that are necessary to protect the use or uses of that particular waterbody, and an antidegradation policy
Water Resource Management (Non-Regulatory)	decisions on management activities relevant to a water resource such as problem identification, need for and placement of best management practices, pollution abatement actions, and effectiveness of program activity

Acronyms

ACOE	Army Corps of Engineers
ALU	Aquatic Life Use
BCG	Biological Condition Gradient
BEAST	BEnthic Assessment of SedimenT
BOD	Biological Oxygen Demand
BPJ	Best Professional Judgment
BMP	Best Management Practice
CDG	Catchment Disturbance Gradient
CAFO	Confined Animal Feeding Operation
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CSO	Combined Sewer Overflows
CWA	Clean Water Act
EPT	Ephemeroptera, Plecoptera, Trichoptera
FWPCA	Federal Water Pollution Control Act
GIS	Geographic Information Systems
GLEI	Great Lakes Environmental Indicators
HDG	Human Disturbance Gradient
HDI	Human Disturbance Index
IBI	Index of Biological/Biotic Integrity
ICI	Invertebrate Community Index
ITFM	Intergovernmental Task Force on Monitoring Water Quality
LDM	Linear Discriminant Model
LWD	Large Woody Debris
NPDES	National Pollutant Discharge Elimination System
NWHI	Non-Wadeable Habitat Index
ONRW	Outstanding Natural Resource Waters
QHEI	Qualitative Habitat Evaluation Index
RBP	Rapid Bioassessment Protocols
RCRA	Resource Conservation and Recovery Act
RDG	Riparian Disturbance Gradient

RIVPACS	River Invertebrate Prediction and Classification System
TALU	Tiered Aquatic Life Use
TMDL	Total Maximum Daily Load
SCI	Stream Condition Index
STP	Sewage Treatment Plants
UAA	Use Attainability Analyses
WLAs	Waste Load Allocations
WQS	Water Quality Standards
WWTP	Wastewater Treatment Plant

Appendix A

MAINE TALU IMPLEMENTATION CASE HISTORY[1]

I. Establish conceptual foundation

Since the early 1970s, prior to adoption of the CWA, Maine water quality law has had a tiered structure, based on a gradient of water quality conditions. An early articulation of the conceptual basis for a tiered approach to establishing aquatic life uses was made by John Cairns and others in a U.S. EPA-sponsored symposium on the biological integrity objective of the Clean Water Act (Ballentine and Guarraia 1977), with further elaboration in Cairns et al. (1993) and Karr and Chu (2000). The underlying basis depicts biological condition declining across a gradient of stressors.

Maine's goal-based management classes range from Class AA, the highest water quality standard and greatest restrictions on human activity, to Class C (and formerly Class D, discontinued), the lowest quality standard with more flexible allowances for human activities (MDEP 2004 305b report). Maine's current water quality classification law for rivers and streams establishes four tiers of aquatic life use (ALU) that represent the upper end of a gradient of biological condition that occurs in the State (State of Maine 1985, Courtemanch et al. 1989, Courtemanch 1995). Conditions worse than this upper end (i.e., worse than Class AA/A, B, or C) are deemed unacceptable. Numeric biocriteria are based on assessment of benthic macroinvertebrates (State of Maine 2003, Davies et al. unpublished manuscript). Assessment of algal assemblages also occurs in most waterbodies but numeric criteria have not yet been developed. Maine relies on the response of benthic macroinvertebrates to human influences for several reasons:

- Diverse life history strategies and a wide range of pollution tolerance;
- Relatively long-lived (+/- 1 year) compared to algae and bacteria;
- Limited mobility diminishes stressor avoidance behavior and emigration;
- The indigenous fish assemblage in Maine is not very diverse and information is limited to just a few species.

Biologists in Maine and elsewhere have long observed clear-cut differences in community structure and composition of benthic macroinvertebrate samples that are collected from waters across a continuum of increasing stressors. The conceptual foundation of the Maine Department of Environmental Protection (MDEP) Biological Monitoring Program (and resulting biocriteria) was framed by three factors: 1) the first-hand observations of such biological response patterns, 2) published empirical and theoretical work in aquatic stress ecology, and 3) Maine's pre-existing water management context. The first two factors are discussed in sequence in this section. The water management context is discussed in the next section, II. *Merge Scientific & Policy Foundations*.

Empirical Observations of Maine Biologists

Differences in resident biological assemblages are evident even to the untrained eye when there are substantial differences in water quality (Figure A-1). This can be illustrated with a very simple example based on a gradient of increasing enrichment. In the initial years of biological assessment in Maine, biologists observed that minimally disturbed sampling locations tended to support many invertebrate taxa (high diversity), but at low to moderate density. In contrast, streams receiving well-treated or well-diluted domestic effluents exhibited higher organism densities, though the types of organisms were similar.

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[1] Appendix A was written by Susan Davies, Maine Department of Environmental Protection.

Streams receiving heavy loadings of sewage or nutrient-laden industrial effluents showed obvious differences in taxa and numbers from that expected in minimally disturbed streams. Streams receiving toxic amounts of chlorine or industrial waste showed much lower densities and many more hardy types of organisms than would be expected in undisturbed areas.

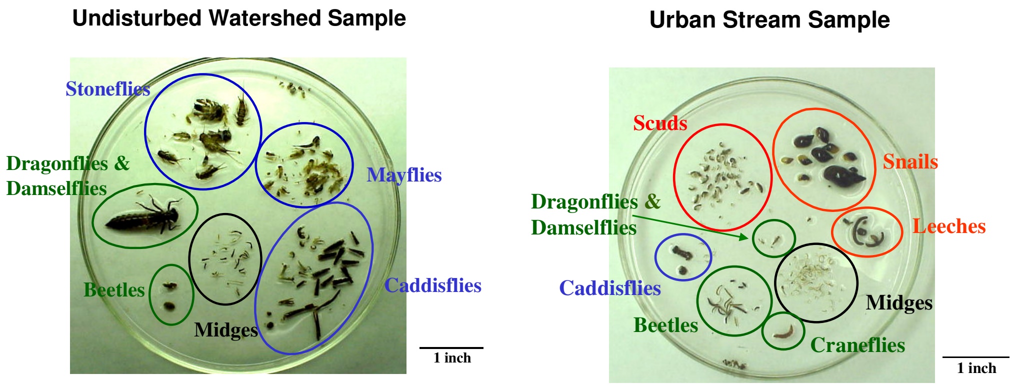


FIGURE A-1. Differences in numbers and types of organisms that are associated with different levels of disturbance can be evident even to the untrained eye.

Published Empirical and Theoretical Work in Aquatic Stress Ecology

The very obvious differences in biological responses for Maine streams, described above, are consistent with published conceptual models and empirical findings of stress ecology. The subsidy-stress gradient model of Reibesell (1974), and further developed by Odum et al. (1979) and Odum (1985), provided Maine DEP biologists with a theoretical model of expected patterns of biological change that was consistent with their own empirical observations (Figure A-2a and A-2b). Development of numeric biocriteria proceeded from this underlying ecological paradigm with the goal to statistically characterize the observed biological condition groups to determine aquatic life use class attainment.

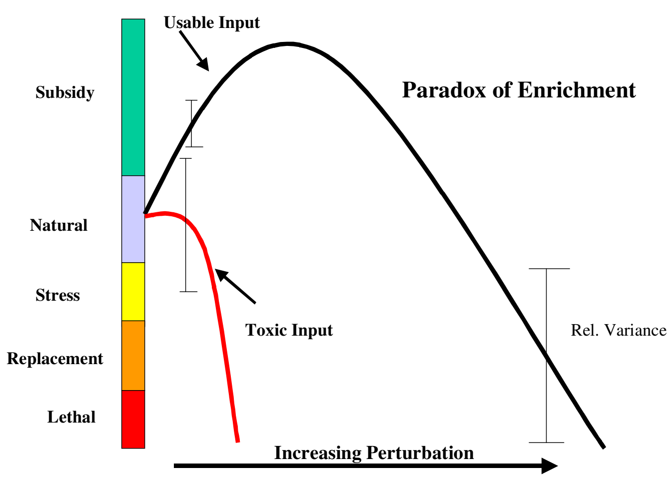


FIGURE A-2a. Subsidy-stress gradient: The ecological theory basis for Maine's aquatic life use descriptions (Odum et al. 1979). Some disturbances have an enriching or subsidizing effect on biological assemblages because they provide more than normal usable resources (nutrients, organic matter, etc.). Inputs in excess of what can be processed by the resident community have a detrimental effect (increased biochemical oxygen demand, accumulation of unusable resources, etc.) and lead to negative community response. Toxic or poisonous inputs have an immediate detrimental effect.

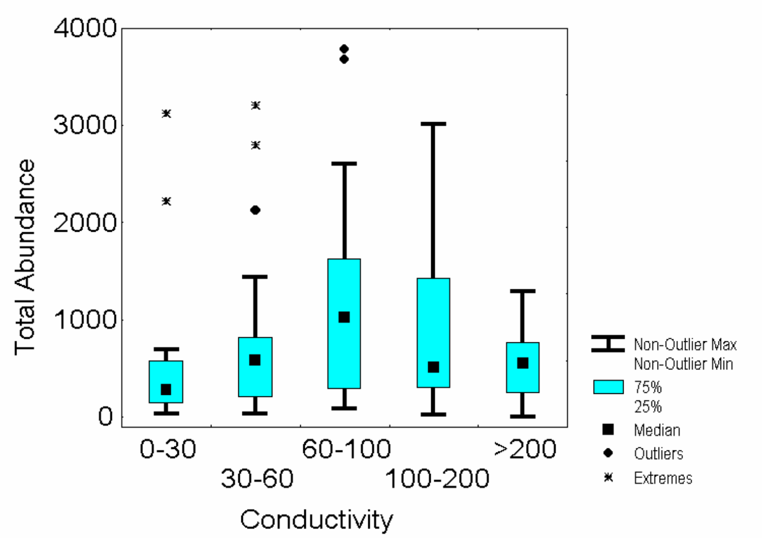


FIGURE A-2b. Empirically observed subsidy-stress gradient in Maine streams, documented by changes in benthic macroinvertebrate density. Low levels of conductivity are an indicator of slight enrichment while high levels are often associated with toxic contamination.

Stress ecology recognizes biological changes in response to increasing levels of stressors (i.e., gradients of environmental quality) as distinct from those that occur in responses to natural gradients, such as elevation, climate, alkalinity, stream size, and geographic location. While natural and ecoregional gradients can and do influence biological expectations in important ways, biological responses from the high to the low end of generalized stressor gradients in Maine streams tend to be far more obvious (Davies et al. 1999, Davies et al. unpublished manuscript). Odum's model supported our observation that structurally distinct biological groups exist across a gradient of water quality. Identifying predictable, characteristic differences among those biological condition groups could serve as the underlying conceptual basis for development of tiered aquatic life uses. Four biological condition groups would also fit well with the State's four-tiered standards for dissolved oxygen, bacteria, and habitat described in the existing water quality classification law.

II. Merge scientific and policy foundations

The narrative aquatic life use statements in Maine's TALUs describe conditions ranging from "as naturally occurs" (Class AA and Class A- the highest ALU designations) to "maintenance of structure and function" (Class C- the lowest ALU designation allowed in Maine) (Table A-1). The subsidy-stress gradient model helped guide the development of the ecologically-based definitions in the law. These specific definitions establish the biological characteristics that are required for attainment of each ALU classification (Table A-2).

TABLE A-1. Maine's narrative aquatic life and habitat standards for rivers and streams (M.R.S.A Title 38 Article 4-A § 464-465).

CLASS	MANAGEMENT	BIOLOGICAL STANDARD
AA*	High quality water for recreation and ecological interests. No discharges or impoundments permitted.	Habitat shall be characterized as natural and free flowing. Aquatic life shall be as naturally occurs.
A	High quality water with limited human interference. Discharges limited to non-contact process water or highly treated wastewater of quality equal to or better than the receiving water. Impoundments allowed.	Habitat shall be characterized as natural. Aquatic life shall be as naturally occurs
B	Good quality water. Discharge of well-treated effluent with ample dilution permitted. Impoundments allowed.	Habitat shall be characterized as unimpaired. Discharges shall not cause adverse impacts to aquatic life. Receiving water shall be of sufficient quality to support all aquatic species indigenous to the receiving water without detrimental changes in the resident biological community.
C	Acceptable water quality. Maintains the interim goals of the Federal Water Quality Act (fishable/swimmable). Discharge of well-treated effluent permitted. Impoundments allowed.	Habitat for fish and other aquatic life. Discharges may cause some changes to aquatic life, provided that the receiving waters shall be of sufficient quality to support all species of fish indigenous to the receiving water and maintain the structure and function of the resident biological community.
Impound-ments	Riverine impoundments not classified as Great Ponds and managed for hydropower generation	Support all species of fish indigenous to those waters and maintain the structure and function of the resident biological community.

*The narrative aquatic life standard is the same for Class AA and Class A.

TABLE A-2. Definitions of terms used in Maine's water classification law.

1. **Aquatic life** any plants or animals that live at least part of their life cycle in fresh water.
2. **As naturally occurs** conditions with essentially the same physical, chemical and biological characteristics as found in situations with similar habitats, free of measurable effects of human activity.
3. **Community function** mechanisms of uptake storage and transfer of life-sustaining materials available to a biological community, which determine the efficiency of use and the amount of export of the materials from the community.
4. **Community structure** the organization of a biological community based on numbers of individuals within different taxonomic groups and the proportion each taxonomic group represents of the total community.
5. **Indigenous** supported in a reach of water or known to have been supported according to historical records compiled by State and Federal agencies or published in scientific literature.
6. **Natural** living in or as if in, a state of nature not measurably affected by human activity.
7. **Resident biological community** aquatic life expected to exist in a habitat, which is free from the influence of the discharge of any pollutant. This shall be established by accepted biomonitoring techniques.
8. **Unimpaired** without a diminished capacity to support aquatic life.
9. **Without detrimental changes in the resident biological community** no significant loss of species or excessive dominance by any species or group of species attributable to human activity.

Consistency with other applicable WQ criteria

As shown in Figure A-3, MDEP designed the narrative ALUs to be parallel to the tiered dissolved oxygen and bacteria standards. This was done because Department biologists recognized that differences in allowed human activities and water quality criteria of the different classes (AA, A, B, C) would inevitably yield different expectations for aquatic community response. For example, it is unreasonable to expect the same biological assemblages to thrive in both Class AA waters (dissolved oxygen: "as naturally occurs"- >7 ppm for Maine; dams and discharges prohibited) and Class C waters (minimum dissolved oxygen 5 ppm; dams, industrial and municipal discharges allowed).

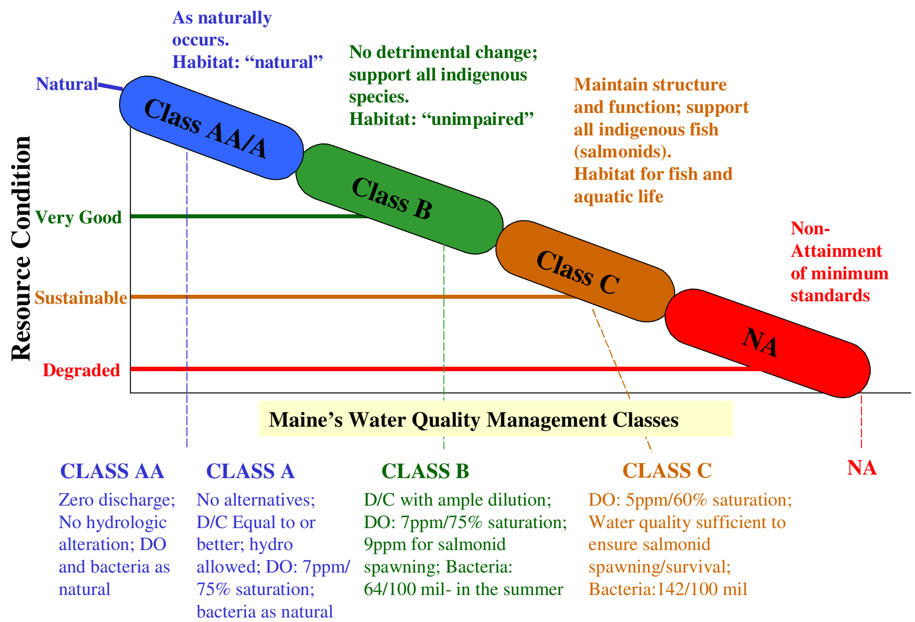


FIGURE A-3. Relation between Maine TALUs and other water quality standards and criteria.

The final language of the narrative aquatic life uses was the result of extensive negotiations between MDEP biologists and stakeholder biologists, under the purview of a legislative subcommittee. Lawyers on both sides weighed in regularly to ensure the fairness and legality of the statute. MDEP biologists drafted the narrative standards and definitions with careful attention to retaining a sound foundation in ecological theory. Furthermore, careful attention was given to how each biological attribute could be quantified (and thus assessed for attainment), with credible and widely accepted biological metrics (Table A-3).

TABLE A-3. Maine tiered uses based on measurable ecological values.

Narrative Standard		Ecological Value		Quantifiable Measures
CLASS A *natural*	➡	Taxonomic and Numeric Equality; Presence of Indicator Taxa	➡	Similarity, Richness, Abundance, Diversity; EPT, Indicator Taxa, Biotic Index
CLASS B *unimpaired, maintain indigenous taxa*	➡	Retention of taxa and numbers; Absence of hyperdominance; Presence of sensitive taxa	➡	Community loss; Richness; Abundance; Diversity; Equitability; Evenness; EPT; Indicator Taxa, Biotic Index
CLASS C *maintain structure*	➡	Resistance, Redundancy; Resilience; Balanced Distribution	➡	Richness; Diversity; Equitability; Evenness
and function	➡	Energy exchange; Resource assimilation; Reproduction	➡	Trophic groups; Richness; Abundance; Community loss; Fecundity; Colonization rate

How do Maine's tiered aquatic life uses relate to the Biological Condition Gradient?
Maine's aquatic life standards specify different levels (tiers) of water quality necessary to maintain designated aquatic life uses. These standards correspond to the tiers of the Biological Condition Gradient in Figure A-4.

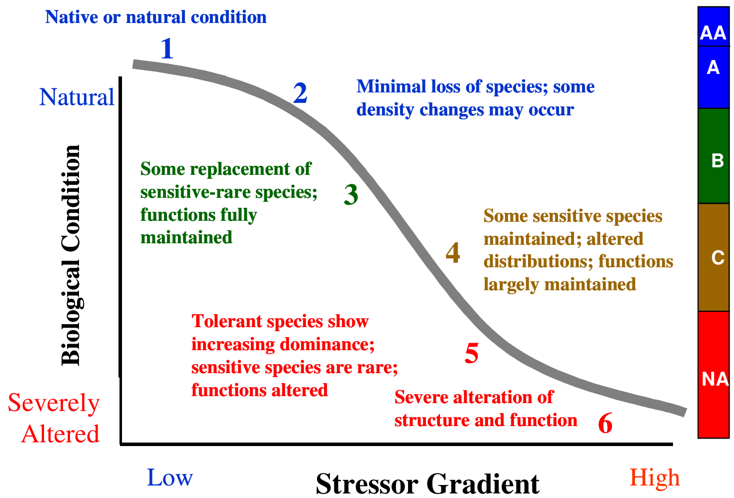


FIGURE A-4. Maine TALUs in relation to the BCG tiers.

<u>Class AA and Class A</u> have the same narrative aquatic life uses requiring that aquatic life be "as naturally occurs." This phrase is defined in the statute as "conditions with essentially the same physical, chemical, and biological characteristics as found in situations with similar habitats, free of measurable effects of human activity." The stated goal condition for Class AA/A thus conforms to Tier 1 or high Tier 2 conditions on the BCG.

Samples attaining MDEP Class A numeric criteria cover a range of conditions, some of which are fully consistent with BCG Tier 1 but some of which would have to be interpreted as BCG Tier 2. Examples of the latter are mildly enriched locations showing higher abundance of organisms (than "natural" for Maine) and increased algal biomass, and Class A locations that are influenced by dams.

<u>Class B</u> aquatic life standards require that there be "no adverse impacts" and that water quality be "sufficient to support all indigenous aquatic species without detrimental changes in the resident biological community." This phrase is defined as "no significant loss of species or excessive dominance by any species or group of species attributable to human activity." This wording was carefully chosen to allow for commonly observed increases in measures of biomass, density, and richness that occur in response to mild enrichment (as depicted by Odum's "subsidy hump" in Figure A-2a and A-2b) but to prohibit negative biological changes, such as notable loss of indigenous taxa. Thus the expectation for Class B is that sensitive taxa should be well represented with community structure comparable to Class A.

Samples attaining MDEP Class B numeric criteria cover a range of conditions, some of which are fully consistent with BCG Tier 2 but some of which would have to be interpreted as BCG Tier 3 because of the degree of structural change or the failure to collect Sensitive-Rare taxa. Dams, well-managed landscape changes, and well-treated point sources are allowed in Class B waters. These changes may result in detectable signals such as absence of migratory taxa, increased algal biomass, higher total abundance of organisms, and increased abundance of sensitive-ubiquitous taxa (i.e., higher relative abundance of some mayflies and some filter feeders; higher abundance of Perlid stoneflies) resulting in a community structure more consistent with Tier 3.

Class C aquatic life standards require that structure and function of the resident biological community be maintained. Numeric biocriteria in Maine document that waterbody segments meeting Class C dissolved oxygen and bacteria standards, but not attaining Class B standards, show obvious differences in biological assemblages. In terms of benthic macroinvertebrates, differences can be generally described as lower numbers and richness of cold-water obligate taxa and those taxa that have high dissolved oxygen requirements (e.g., gill-breathing mayflies and stoneflies), higher densities of filter-feeding organisms, and increased densities of some types of chironomid midges and other facultative or tolerant groups.

Samples attaining MDEP Class C numeric criteria cover a range of biological conditions, most of which are fully consistent with BCG Tier 3 and/or Tier 4. About 10% of samples that attain MDEP Class C numeric criteria would have to be interpreted as BCG Tier 5 because of the degree of structural change or very low numbers of Sensitive taxa (e.g., the mean abundance of Ephemeroptera in sites attaining Class C numeric criteria is 86 individuals per sampler but about 10% have less than 10 mayflies). Attainment of Class C numeric criteria usually indicates that other community structure attributes are present (e.g., evenness of distributions, richness and/or diversity of the assemblage of taxa of intermediate tolerance). Hyper-dominance of filter-feeders, complete absence of expected sensitive insect taxa (especially stoneflies and mayflies), and high proportions of tolerant taxa signal assemblages that fail to meet Class C water quality standards. These conditions represent BCG Tiers 5 and 6.

III. Establish technical program

How does Maine DEP collect biological data?

The MDEP's Biological Monitoring Program began standardized sampling of river and stream macroinvertebrates in 1983 (less rigorously standardized biological assessments had begun at least 10 years before). Experience gained on the Penobscot River (Davies 1987, Rabeni et al. 1988) had demonstrated the practical usefulness and reliability of rock-filled basket artificial substrates (Klemm et al. 1990). Maine has adapted the basic design of these devices to enable sampling of waterbody depths ranging from as little as 5 cm (using rock-filled mesh bags; Davies et al. 1999) to about 10 meters in large riverine impoundments (using boat-retrievable cones; Courtemanch 1984, Davies and Tsomides 2002, http://www.state.me.us/dep/blwq/docmonitoring/biological/biorep2000.htm). The success of these devices has enabled the MDEP to apply comparable field and analytical methods to nearly all rivers and streams of significant regulatory interest (Davies and Tsomides 2002), greatly simplifying the development and application of river and stream biocriteria. Further, the physiography of Maine is quite homogeneous with roughly 85% of the State falling within just two relatively similar ecoregions (Omernik 1987). For this reason stratification by ecoregion was not the critical concern that it is for States in some other regions of the country (Davies et al. unpublished manuscript).[2]

In 1999, Maine began an algal monitoring program to strengthen the interpretation of ecological condition by providing information from a second biological assemblage. Maine's fish assemblage is naturally depauperate, limiting its suitability as a candidate for bioassessment. The algal monitoring program will assist the Department in the development of river and stream nutrient criteria. The Department also has a companion biomonitoring program to assess wetland biological condition.

Database development

By the late summer of 2004, the Department had established about 800 monitoring stations in all major watersheds throughout the State (Figure A-5). Data from macroinvertebrate samples are stored in an Oracle© database and all stations are geo-referenced in the Department's geographic information system

[2]Maine's southern ecoregion is very small but recent data suggest that some improvement in accuracy of class prediction could result from better accounting for ecoregional differences there.

(ArcInfo©). Data collected in accordance with Maine's biocriteria protocol are analyzed using statistical models that estimate to which of the four water quality classes a sample belongs. Findings of the Biological Monitoring Program are used to document existing conditions, identify problems, set water management goals, assess the progress of water resource management measures, and trigger needed remedial actions.

Sampling methods
Samples of benthic macroinvertebrates are collected from flowing streams in rock bags (or baskets or cones). At least three substrate samplers are exposed in the waterbody for 28 days during the late summer, low flow period (July 1 to September 30). The MDEP usually conducts sampling, but others may also perform monitoring to determine attainment of classification if done according to a quality assurance plan.

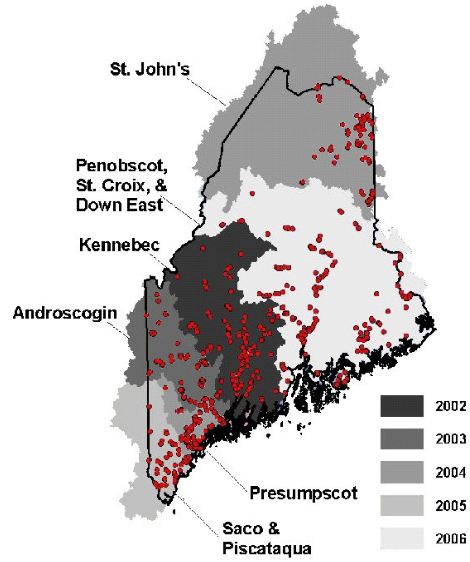


FIGURE A-5. Macroinvertebrate sampling stations in Maine.

Laboratory methods
Samples are retrieved, sorted, and stored for identification by a professional freshwater macroinvertebrate taxonomist. Organisms are identified to species whenever possible or otherwise to the lowest taxonomic level possible.

Analytical methods
If a sample satisfies the minimum data requirements (total mean abundance of at least 50 individuals, generic richness of at least 15 taxa for 3 replicate samplers), data are entered into the MDEP's computer software for further analysis through the numeric criteria statistical model. The model is able to take large amounts of information generated from a biological sample, describe which variables appear to be most significant in the classification decisions, and provide a mathematical summary that integrates the information. The model produces probability scores from 0 to 1 that indicate the likelihood that a sample attains each water quality class.

IV. Develop and validate quantitative thresholds

How does Maine quantify the tiered aquatic life uses so that attainment can be assessed?
In the late 1980's, the MDEP quantified the narrative aquatic life goals for each water quality class by developing a probability-based statistical model to serve as numeric biocriteria (Courtemanch et al. 1989, Courtemanch 1995, Davies et al. unpublished manuscript). The model uses 31 biological variables, many of which were specifically chosen because of their utility in measuring some important ecological attribute in the narrative standard. The model quantifies and standardizes the expert judgment of biologists and it now serves as an expert system for decision-making (*See Case Examples 3-3 and 3-6*).

To develop the model, biologists used agreed-upon decision rules and a Delphi technique (Bakus et al. 1982) to assign an aquatic life attainment classification (A, B, C, or non-attainment) to 144 samples of benthic macroinvertebrate data, based on conformity of the sampled community to one of the 3 narrative aquatic life standards in Maine's statute, or to a fourth category representing non-attainment of minimum State standards (Shelton and Blocksom 2004, Davies et al. unpublished manuscript). The samples evaluated represented 300 distinct taxonomic units and 70,000 organisms collected from rivers, streams, and riverine impoundments. Those data and their classification assignments were used as the baseline for construction of the expert system, in the form of a linear discriminant model, to evaluate future macroinvertebrate samples for water quality classification attainment. The original model was used from 1992 through 1999 when the model was recalibrated with an additional 229 (for a total of 373) sampling events. The recalibration resulted in relatively minor changes to the structure of the original model, involving simplification of the structure of two of the sub-models, the elimination of two poorly performing variables, and changes in model coefficients to account for the new data.

How has Maine established reference conditions?
Maine has taken a conceptually different approach to establishing baseline reference conditions from which to develop numeric biological criteria. Because we determined that detection of four distinct biological condition groups, characterized by differences in specified ecological attributes, was our management goal, it was also our goal for statistical analysis. We desired to develop numeric criteria that would enable us to assign sites to one of those four condition groups (A, B, C, non-attainment). Therefore, our task for characterizing reference conditions was to conceptually and then statistically define those four groups. Thus in a sense, initially by expert judgment and then by multivariate analysis, we created a Class A reference condition (deemed to be close to natural), a Class B reference condition, a Class C reference condition, and non-attainment reference conditions. Use of biological information to establish a minimally disturbed reference has been criticized due to the dangers of a too circular process. We have tested our biology-based a priori assignment of sites to Class A using more traditionally identified reference locations (i.e., based on high percent natural landcover) and found good correspondence with the biologically-defined Class A sites.

Adoption of the Numeric Biocriteria Rule
On April 17, 2003 the Maine Board of Environmental Protection adopted numeric freshwater biocriteria in rule. The biocriteria rule describes the process that the MDEP uses to make decisions about attainment of aquatic life uses in rivers and streams. The rule describes protocols for biological sampling of benthic macroinvertebrates, laboratory analyses, modeling analysis of laboratory data, and selective use of expert judgment. Adoption of this rule quantitatively interprets Maine's existing narrative 'aquatic life' standards for each riverine water quality classification.

V. Application in water quality management

How does the MDEP decide which waterbodies and locations to monitor?
For purposes of biological monitoring, the MDEP divided the State into five major river basins, which are sampled on a 5-year rotational schedule (Figure A-6): Androscoggin, Kennebec and Mid-Coast, Penobscot, St. Croix and North Coastal Rivers, Piscataqua, Saco and Southern Coast, St. John and Presumpscot. The decision to monitor specific locations on a waterbody can be based on a variety of factors such as:

- prior knowledge of human activities that could have a detrimental effect on a waterbody: sampling seeks to detect actual impacts on biological communities;
- knowledge of future potential threats to a waterbody: sampling can be done to collect baseline data before, for example, development occurs or a discharge is licensed; follow-up sampling can determine the effect, if any, on the biological community by said development or discharge;
- requirement/desire to monitor the effects of remediation activities or water quality management changes;
- desire to expand coverage of the monitoring program and to more fully document natural variability.

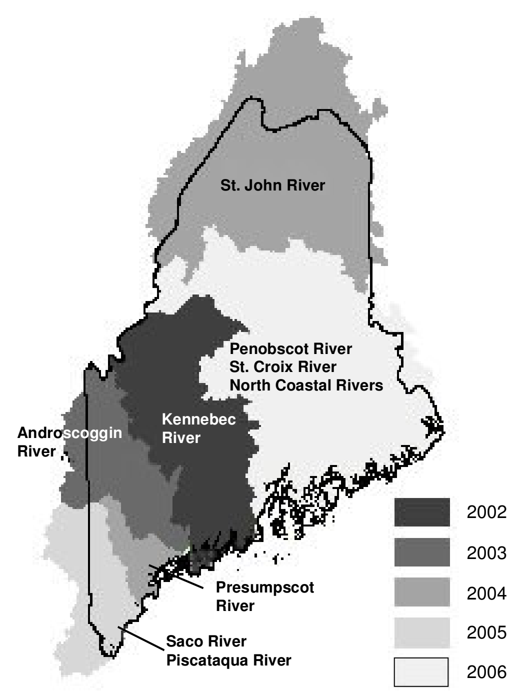


FIGURE A-6. Maine five-year rotating basin sampling schedule.

How are tiered aquatic life uses designated in Maine?
The quality of Maine's waters is described in terms of physical, chemical and biological characteristics associated with the State's water classification program. As established in Maine statute (38 MRSA Sections 464-470), the classification program consists of designated uses (e.g. drinking water supply, recreation in and on the water, habitat for fish and other aquatic life), criteria (e.g. bacteria, dissolved oxygen and aquatic life), and characteristics (e.g. natural, free flowing) that specify levels of water quality necessary to maintain the designated uses. All State waters have a classification assignment (Rivers and streams: AA, A, B, C; Lakes: GPA; Marine and estuarine: SA, SB, SC). Tiered narrative aquatic life uses specific to wetlands are currently under consideration by MDEP and a supporting wetland biomonitoring program is in place.

The classification system in Maine is goal-based in that assignment of a given waterbody to a use class (AA, A, B or C) may not necessarily reflect its current conditions. Rather, it establishes the level of quality the State has deemed the waterbody must achieve. Maine's classification system is also more risk based than quality based. Water quality differences among the various classes are not large, however, the different levels of restrictions put on human activities associated with each class establishes the level of risks that water quality could be degraded resulting in increased threats to designated use attainment. Rivers and streams are assigned to a tiered aquatic life use goal (Table A-1: AA and A -"*as naturally occurs,*" B- "*no detrimental change,*" C- "*maintain structure and function and water quality sufficient to support salmonids*") that represents the best fit after considering:

- The current condition in terms of dissolved oxygen, bacteria, and aquatic life (Figure A-3) and
- The highest attainable goal condition (taking into account ecological and socioeconomic factors).

The State water quality assessment provided in Maine's 305b report gives the status of attainment of the water resource goals established in the classification program. Thus, some waters may be listed as impaired even though they have relatively good water quality (Table A-4), e.g., a Class A river may be listed because it does not fully attain the standards of that class but may be of sufficiently good quality to attain Class B or C, and the Clean Water Act interim goal. The classification program is reviewed every three years (Triennial Review) by the Department and the Board of Environmental Protection (Board). The Board may, after opportunity for public review and hearing, make recommendations to the Legislature for changes in water quality standards or reclassification of selected waters. The most recent revisions to the classification program were completed in 2002-2003 when the Legislature authorized classification upgrades to 75 river, stream and coastal segments totaling over 800 miles of waters (Figure A-7).

TABLE A-4. Examples of how numeric biocriteria results determine whether or not a waterbody attains designated aquatic life uses in Maine.

Legislative Class	Monitoring Result	Attains Class?	Next Step
A	A	Yes	--
C	B	Yes	--
A	B	No	TMDL
B	NA	No	TMDL

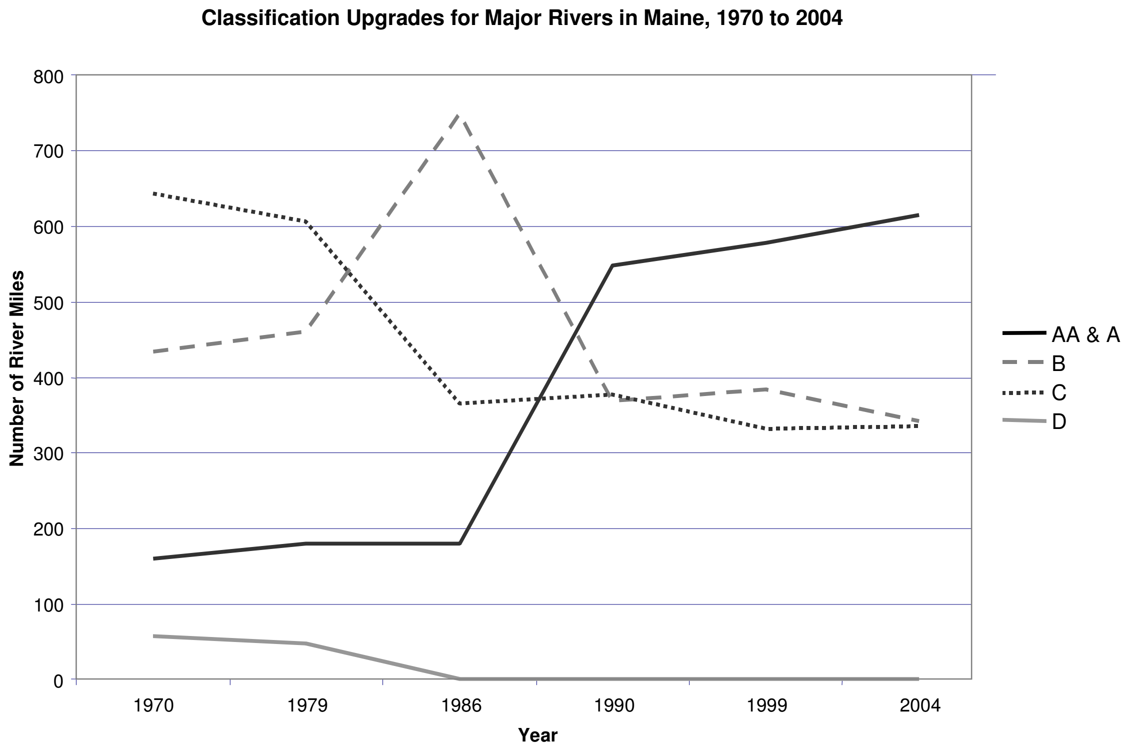


FIGURE A-7. Increased designation of Class AA and Class A uses on major Maine rivers (as shown by river miles) between 1970 and 2004, as a result of water quality improvements and public support for the Class AA/A goal in the Triennial Review Process.

What is the management perspective for TALU designations in Maine?
Class AA waterbodies, as compared to Class A, have significantly greater restrictions on allowed activities. For example, no discharge of wastewater and no dams are allowed in Class AA waterbodies. Class A waters carry a higher risk of degradation because discharges are allowed, though the risk is small because they must be of "equal to or better" water quality than the receiving water. Dams are also allowed. Obstructions to flow, whether man-made or natural can alter assemblage structure from free-flowing conditions (Poff et al. 1997, Davies et al. 1999). The definition in water quality standards for the term "natural" sought to limit the effects of altered flows to no greater than what could be expected from a "natural" obstruction to flow (e.g., a natural hydrological control or a beaver dam). Thus to accommodate dams in Class A, "*natural*" is defined as "occurring in, *or as if in*, a state of nature not measurably affected by human activities." Assemblages that are characteristic of the waters above and below beaver dams or low-head, run-of-river, man-made dams are deemed to pass this standard. Most dams in Class A provide for passage of anadromous fish.

Class B was originally applied as the default ALU for unmonitored waters though current use designations are nearly equal in stream miles for Class A and Class B, both of which far exceed Class C miles when all rivers and streams in the State are considered (Figure A-8). From the management perspective, a Class B designation often applies to waterbody segments exposed to well-treated or well-diluted domestic discharges or to areas subjected to landscape alterations that result in moderate increases in the nutrient and organic matter load.

Class C narrative aquatic life standards prohibit any activities that result in the loss of structure and function of the resident biological community. "Community structure" is defined as "the organization of a biological community based on numbers of individuals within different taxonomic groups and the proportion each taxonomic group represents of the total community," while community function is defined as "mechanisms of uptake storage and transfer of life-sustaining materials available to a biological community which determine the efficiency of use and the amount of export of the materials from the community." This management class is applied to waterbodies that may be impounded, altered by landscape changes, or that receive industrial wastewater.

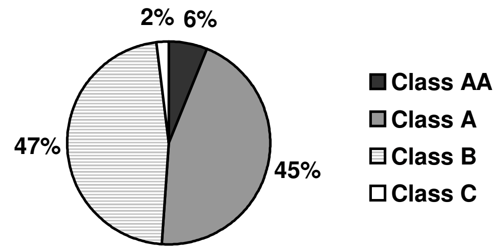


FIGURE A-8. Percent of linear miles of all rivers and streams in each of Maine's designated use classes (year 2000).

What process was used to bring the Maine TALU biocriteria rule through adoption?

The MDEP Biological Monitoring Program completed provisional numeric biocriteria in 1990. Those numeric thresholds were the basis for extensive regulatory and non-regulatory Department decisions between 1990 and 2003, e.g., issuance or denial of 401 water quality certificates and recommendations for flow management changes, 303d and 305b listings, prioritization of at-risk waterbodies, and problem identification. In April 2003, the State formally adopted tiered numeric biocriteria rules that were the result of the analysis of 15 years of biological data and the experience gained through 20 years of regulatory decision-making based on numeric biocriteria (Table A-5). Remarkably, the biocriteria rule was one of the most complicated and important, but least contested water quality rules that the Maine Department of Environmental Protection has adopted in the last 15 years. Stakeholders from all sides had become convinced of the merits of the approach.

TABLE A-5. Chronology of Maine's biocriteria development.

1983	The MDEP Biological Monitoring Program began a standardized program of sampling stream invertebrate communities.
1986	The revised Water Classification Program, which defined tiered narrative standards for aquatic life, became law.
1989	MDEP staff and University of Maine statistical ecologist, Dr. Frank Drummond embarked on the development of numeric criteria to support the narrative standards of the law.
1990	A technical advisory committee of stakeholder scientists was convened to provide peer review and oversight of the biocriteria development process. Over the course of approximately 2 ½ years, MDEP staff, Dr. Drummond, and the committee developed a statistical model based on expert judgment and linear discriminant analysis to address the scientific goals, as well as the policy and regulatory goals of the new biocriteria program.
1991-1993	Public informational workshops on the process were held in March 1991, September 1993, and December 1993.
1999	The original statistical model was recalibrated to take advantage of the expanded dataset available at that time.
2002	During a formal stakeholder review process, meetings were held in March and April and comments were solicited from representatives of the hydropower and paper industry, environmental advocacy groups, other State agency biologists (e.g., fish and wildlife), university scientists, and private consultants.
2002	A workshop on the rule and its background was held in early October for the Maine Board of Environmental Protection.
2003	The Board of Environmental Protection adopted the rule on April 3 and it was subsequently adopted by the Maine State Legislature.

Appendix B

OHIO TALU IMPLEMENTATION CASE HISTORY[1]

In 1990, Ohio EPA adopted numeric biological criteria in the Ohio Water Quality Standards (Ohio WQS; Ohio Administrative Code 3745-1). These criteria have been used to guide and enhance water quality management programs and assess their environmental outcomes. The numeric biocriteria are an outgrowth of an existing framework of tiered aquatic life uses and narrative biological assessment criteria that has been in place since 1980. This case history is intended to summarize the evolutionary development of the components of the WQS and monitoring and assessment programs that took place in the late 1970s and throughout the 1980s and 1990s.

I. Establish conceptual foundation

Initially developed and adopted by Ohio EPA in 1978, tiered aquatic life uses represented a major revision to the existing general use framework that was adopted in 1974. This level of tiered uses recognized the different types of warmwater aquatic assemblages that corresponded to the mosaic of natural features of the landscape and nearly two centuries of human-induced changes. The eventual development of more refined tiered uses and the attendant numeric biocriteria that are in place today was the result of a decade long development process. The important concepts that spurred and guided these developments in the Ohio EPA program are described as follows:

Natural History and Zoogeography
The empirical evidence used to develop the initial concepts for tiered uses can be found in comprehensive works on the natural history and zoogeography of the Midwest such as Fishes of Ohio (Trautman 1957, 1981) and Fishes of Illinois (Smith 1979). These texts documented the natural and human-induced variations in the distribution, composition, and abundance of biological assemblages over space and through time. Trautman (1957) not only provides a lesson in Ohio's natural history, but also describes the biological evidence that was used to formulate the initial concepts about biological integrity that emerged in the late 1970s and early 1980s. Such works also described the key features of the landscape that influence and determine the potential aquatic fauna of waterbodies and were the forerunners of the regionalization tools that appeared soon after. As an alternative to a "one-size-fits-all" approach, these provided an important foundation for the development of Ohio's tiered uses.

Landmark Stream and River Pollution Studies
The earliest studies of the effects of pollution on biological assemblages were the precursors of the approach eventually developed and used by Ohio EPA. Campbell (1939), Brinely (1942), and Wurtz (1955) described the classical zones of pollution in flowing waterbodies. Ellis (1937) conducted one of the first comprehensive studies of water pollution in the U.S. including an emphasis on the chronic impacts of wastewater discharges. Patrick (1950, 1953) employed the concept of species (or taxa) diversity as an indicator of the "health and well-being" of aquatic assemblages and described a "biodynamic cycle." Gaufin and Tarzwell (1953) also described pollutional zones using aquatic assemblages and were the first to advocate cost-effective assessments of one or two representative assemblages (e.g., fish and macroinvertebrates). Subsequent studies of that time included landmark pollution investigations of rivers and streams (Krumholz and Minckley 1964; Mills et al. 1966; Tsai 1968, 1973; Sparks and Starrett 1975; Gammon 1976), some of which introduced standardized approaches to

[1] Appendix B was written by Chris Yoder, Midwest Biodiversity Institute, Columbus, Ohio.

biological data collection and analysis. These were the key citations in the original proposal for the present-day Ohio EPA biological assessment program (Yoder 1978). Such works also provided the impetus for articulating the linkage between ecological symptoms of aquatic health and human-induced changes in aquatic ecosystem quality that came later.

Concepts of Biological Integrity

The articulation of a practical definition of biological integrity by Karr and Dudley (1981) provided a theoretical framework for the development of Ohio's numeric biological criteria. Key components of this framework are: 1) using biological assemblages as a direct measure of aquatic life use attainment status (Herricks and Schaeffer 1985, Karr et al. 1986), 2) the development and use of multimetric assessment tools (Karr 1981, Karr et al. 1986), 3) derivation of regional reference condition to determine appropriate aquatic life use goals and assessment endpoints (Hughes et al. 1986), and 4) systematic monitoring and assessment of the State's waters. This represented a major advancement over previous attempts to define and develop a workable framework to address the concept of integrity (Ballentine and Guarraia 1977). Embedded in this framework is the recognition that water quality management must be approached from an ecological perspective that is grounded in sound ecological theory *and* validated by empirical observation. This means developing monitoring and assessment and WQS to encompass the five factors that determine the integrity of a water resource (Figure 1-3; Karr et al. 1986).

Experiences in Applying Systematic Biological Assessments

A major aspect of the development of the Ohio biological assessment program and tiered uses is the experience gained through the initial and sustained development of systematic bioassessments beginning in the late 1970s and through the 1980s. This is where the previously described methods, concepts, and theories were applied, tested, and developed, resulting in a tractable system for measuring biological quality at multiple spatial scales and through time. An evolutionary process occurred in which qualitative, narrative biocriteria were initially used to assess rivers and streams via systematic watershed monitoring and assessments. The data and experiences gained in this process provided the raw materials for incorporating the concepts of biological integrity that emerged simultaneously. This resulted in further refinements to the biological assessment tools and criteria and the tiered uses including how they are assigned and assessed. Key to the success of this approach was the initial decisions about indicator assemblages and methods. These have remained stable throughout the entire development and implementation process, with no major modifications that would have resulted in major disconnections of the database. The specific methods, tools, and criteria are described in Section II.

When numeric biocriteria and refined uses were adopted in 1990, the development process continued with adaptations of that system to different waterbody types. A systematic process for classifying and assessing wetlands was developed in the early 1990s and narrative biocriteria were adopted in the Ohio WQS. Biological assessment methods and indexes were also developed for the Lake Erie near shore and lacustuary habitats (Thoma 1999). Routine application of the numeric biocriteria in support of dredge and fill permitting and 401 certifications exposed the need to develop new assessment tools for primary headwater streams, i.e., those draining less than one square mile. Dealing with these waters required a change in indicator groups emphasizing aquatic amphibians and invertebrates and a modified classification scheme (Ohio EPA 2003). Finally, the Ohio River Valley Water Sanitation Commission (ORSANCO) developed a systematic approach for assessing fish (Emery et al. 2003) and macroinvertebrate assemblages of the Ohio River mainstem as a precursor to the adoption of numeric biocriteria. Other innovations are expected to follow and include recalibration of the stream and river biocriteria following the resampling of reference sites that took place during 1990-1999, urban stream classification issues (Yoder et al. 2000, Miltner et al. 2003), and adaptation to level IV ecoregions and other geomorphic classification schemes. These are examples of a continuous improvement process that naturally follows the adherence to the fundamentals of integrating WQS with systematic monitoring and assessment.

II. Merge scientific and policy foundations

From the outset, biological and water quality assessments were intended to play a pivotal role in the application of tiered uses. Since designated uses were formulated and described in ecological terms, it followed that they should be applied and measured on an ecological basis. At that time, the readily available criteria were chemical-specific and the development of practical and systematic biological assessments was in pilot testing and development stages. The operational execution of tiered uses (WQS) was dependent on developing a more comprehensive and systematic approach to monitoring and assessment that supported the watershed and waterbody specific application of tiered uses. However, time was required to develop standardized data, tools and criteria, spatial design, and spatial coverage, which were part of the monitoring and assessment program that delivered full support for tiered uses (and all other water quality management programs). Figure B-1 illustrates the evolutionary and incremental process of the development of tiered uses, allied tools and criteria, and the monitoring and assessment approach that were necessary to achieve full implementation of TALU in Ohio.

• General ALU • Few Specific Chemical Criteria • Narrative "Free Froms" • "Pilot" biological monitoring program • Fixed station M&A design (chemical only; 100+ sites)	• Narrative TALUs • More Specific Chemical Criteria • Initial designation of specific waters (BPJ based; system-atic M&A envisioned) • Fixed station M&A design (chemical only; 100+ sites) • "Pilot" biological monitoring program (10-15 sites/yr.)	• Narrative TALUs • More Specific Chemical Criteria • Narrative Biocriteria (initial WQ program support) • Designation of specific waters based on M&A (UAA process) • Intensive river & watershed surveys initiated (integration of biol, chem, and physical indicators; 100 -200 sites/yr.)	• Refined TALUs • Specific & Complex Chemical Criteria • Numerical Biocriteria (BCG implicit) • Physical Habitat Assessment • WET Testing • Geometric watershed design (integration of biol, chem, and physical indicators; 400-600 sites/yr.) • Integrated Bioassessments (systematic WQ program support)
(1974 - 1978)	(1978 - 1980)	(1980 - 1987)	(1987 - present)

FIGURE B-1. Evolutionary development of TALU and allied tools, criteria and assessments from the baseline of the 1974 WQS based on general uses and few specific water quality criteria to refined TALUs and specific chemical, physical, and biological criteria implemented via an integrated monitoring and assessment framework. The three time periods beginning with 1978-1980 approximate the first three phases of biocriteria development and implementation in Figure 5-2.

Pre-development Phase: 1974-1978

The first WQS adopted in 1974 were consistent with the technology available at that time consisted of general uses, "free from" statements, and few numeric criteria of any kind (chemical, physical, or biological). The monitoring and assessment program adhered to contemporary U.S. EPA guidance, consisting of a fixed station network (approximately 100 sites, monthly and quarterly chemical sampling) and a "pilot" biological program. The baseline water quality management programs (i.e., NPDES permitting, funding, planning) were also in their initial stages of development and implementation. A comprehensive water quality based approach to pollution abatement and management had not yet been developed or envisioned – abatement efforts focused on technology based limitations for major point sources. The linkage between WQS and monitoring and assessment had not yet been made, the latter being viewed as a less important, optional activity.

Initial TALU Development Phase: 1978-1980

In 1978, tiered aquatic life and other uses (e.g., recreation, water supply) were described and adopted along with the development of numeric chemical criteria for parameters such as dissolved oxygen (D.O.), temperature, ammonia, and common heavy metals (e.g., copper, cadmium, lead, zinc, iron, chromium, and nickel). The tiered uses emanated from recognition of the broader ecological concepts described in section I, as well as the belief that a "one-size-fits-all" approach to water quality management (i.e., the result of applying general uses) was neither realistic, cost-effective, nor saleable to stakeholders and the public. While tiered uses promised more customized and cost-effective management outcomes, the integration of WQS and monitoring and assessment, which is necessary before these stated objectives could be realized, had not yet taken place.

Ohio's First Tiered Use Designations

Tiered aquatic life uses are articulated as narrative statements describing the ecological attributes that should be supported by each tier. The criteria associated with each tier consisted of pollutant-specific, single value criteria for a limited set of water quality parameters (i.e., D.O., temperature, ammonia, common heavy metals). There were no biological criteria at that time, although the vision was to eventually develop a biologically-based assessment process. The tiers included variations on a theme of warmwater aquatic assemblages as written in the narrative for the warmwater habitat (WWH) use designation:

> "These are waters capable of supporting reproducing populations of fish, normally referred to as warmwater species, and associated vertebrate and invertebrate organisms and plants on an annual basis. These standards apply outside of the mixing zone." (Ohio Administrative Code 3745-1-07 c. 1978)

The intent of the exceptional warmwater habitat (EWH) use designation is illustrated by the phrase "These are waters capable of supporting ***exceptional and unusual*** populations of fish . . . " In essence, the EWH designation required evidence of an exceptional or unusual assemblage of fish or associated aquatic organisms and plants on an annual basis. Initially, EWH designations were made based on the known locations of self-sustaining populations of fish and other aquatic species that were considered of exceptional value, most of which had exhibited historical declines in distribution throughout Ohio and the Midwest in response to human-induced changes. These locations also corresponded to a congruence of natural landscape features associated with Ohio's glacial geology that "insulated" these assemblages from the cascade of effects from alteration in the landscape that adversely impacted the same species in other more vulnerable waterbodies. The result was waters with more intact habitats, less altered hydrological characteristics, and water quality that was "much better than most." As such, a goal of EWH is to protect such aquatic habitats as a refuge for rare and sensitive species and is vital to the broader restoration goals of the 1972 Federal Water Pollution Control Act (FWPCA) amendments. A greater degree of protection was initially afforded to these waters via more stringent water quality criteria for key parameters such as D.O., ammonia, and temperature (Ohio Administrative Code 3745-1-07 c. 1978). WWH became the default designation for all other waters that lacked such "exceptional and unusual attributes", but which retained or had the potential to exhibit the minimum quality that met the baseline provisions of the FWPCA (Sec. 101[a][2]).

A coldwater habitat (CWH) designation was also developed, but primarily focused on fishery attributes (i.e., Salmonids), which are largely artificially propagated and maintained in Ohio. However, the possibility of incorporating broader ecological attributes into this use narrative was included in the designated use narrative as follows:

> "These are waters capable of supporting populations of fish, normally referred to as coldwater species and associated vertebrate and invertebrate organisms and plants on an annual basis. These waters are not necessarily capable of supporting successful reproduction of Salmonids and may be stocked periodically. These standards apply outside of the mixing zone." (OAC 3745-1-07)

The monitoring and assessment program was initially based on fixed stations and emphasized chemical assessments, but experimental approaches such as small-scale intensive surveys and biological assessments were being developed and tested. There were no empirically derived or narrative biological criteria to decide between EWH and WWH. Specific assignments of waters were made using expert consensus and best professional judgment based on the known ecological attributes inherent in each designation. Thus the assignments of individual water bodies were only as good as the information available for such waters, which was later found to be incomplete or inadequate. Other tiers in the Ohio aquatic life use designations included seasonal warmwater habitat (SWH) and limited warmwater habitat (LWH). Water quality criteria for common chemical parameters were tiered and/or varied for each use designation. Criteria were the most stringent for CWH and EWH and the least stringent for LWH, the latter use essentially functioning as a temporary variance to WWH.

Initial TALU Implementation and Development Phase: 1980-1987

While the tiering provided by EWH and WWH is conceptually consistent with the intent and attributes of the biological condition gradient (BCG; Chapters 2 and 3), the tools to quantify and implement the associated concepts were lacking in 1978. The inclusion of the concepts of biological integrity (Karr and Dudley 1981), operational measures of biological condition (Karr et al. 1986), and the concepts of regionalization and reference sites (Hughes et al. 1986, Omernik 1987) led to further refinements of the tiered uses in this phase. These refinements resulted in the present day hierarchy of the exceptional warmwater, warmwater, modified warmwater, and limited resource waters use designations. The narrative descriptions were modified to reflect the operational definition of biological integrity (Karr and Dudley 1981), further integrating the parallel development of numeric biological criteria.

The original tiered uses were devised with an eye toward the eventual development of a biological assessment based approach to their implementation. These initial developments took place in the early 1980s and included narrative (or qualitative) biological "criteria" (Tables B-1 and B-2) supported by biological assessments and the implementation of an intensive survey design executed on a mainstem river or watershed basis (Ohio EPA 1981). These early biocriteria were based on the experiences and best professional judgment of the agency biologists and reflected the analytical and assessment tools of that time. At the same time, t chemical criteria were being further developed and whole effluent toxicity (WET) testing was being explored.

The use of monitoring and assessment in support of water quality management programs emphasized WQS (assigning tiered uses), construction grants (advanced treatment justifications), and NPDES permits (water quality based effluent limits). At the same time, the statewide database that would support the eventual and more comprehensive development of biological, chemical, and physical assessment tools and criteria was being amassed via the systematic implementation of a an intensive survey and watershed assessment process. Comparatively complex chemical-specific criteria were adopted for 126 priority pollutants and included chronic, acute, and lethal endpoints for aquatic life; criteria were also adopted for human health exposures. Whole effluent toxicity testing was introduced and developed as a water quality based permitting tool (Figure B-1).

TABLE B-1. Biological criteria (fish) for determining aquatic life use designations and attainment of Clean Water Act goals (November, 1980; after Ohio EPA 1981).

Evaluation Class Category	"Exceptional" Class I (EWH)	"Good" Class II (WWH)	"Fair" Class III	"Poor" Class IV
1.	Exceptional or unusual assemblage of species	Usual association of expected species	Some expected species absent, or in very low abundance	Most expected species absent
2.	Sensitive species abundant	Sensitive species present	Sensitive species absent, or in very low abundance	Sensitive species absent
3.	Exceptionally high diversity	High diversity	Declining diversity	Low diversity
4.	Composite index >9.0 – 9.5	Composite index >7.0 – 7.5; <9.0 – 9.5	Composite index >4.5 – 5.0; <7.0 – 7.5	Composite index <4.0 – 4.5
5.	Outstanding recreational Fishery		Tolerant species increasing, beginning to dominate	Tolerant species dominate
6.	Rare, endangered, or threatened species present			

Conditions: Categories 1, 2, 3, and 4 (if data is available) must be met and 5 or 6 must also be met in order to designated in a particular class.

TABLE B-2. Biological criteria (macroinvertebrates) for determining aquatic life use designations and attainment of Clean Water Act goals (November, 1980; after Ohio EPA 1981).

Evaluation Class Category	"Exceptional" Class I (EWH)	"Good" Class II (WWH)	"Fair" Class III	"Poor" Class IV
1.	Pollution sensitive species abundant	Pollution sensitive species present in moderate numbers	Pollution sensitive species present in low numbers	Pollution sensitive species absent
2.	Intermediate species present in low numbers	Intermediate species present in moderate numbers	Intermediate species abundant	Intermediate species present in low numbers or absent
3.	Tolerant species present in low numbers	Tolerant species present in low numbers	Tolerant species present in moderate numbers	Tolerant species abundant (all types may be absent if extreme toxic conditions exist)
4.	Number of taxa >30[1]	Number of taxa 25-30	Number of taxa 20-25	Number of taxa <20
5.	Exceptional diversity Shannon index <3.5	High diversity Shannon index 2.9-3.5	Moderate diversity Shannon index 2.3-2.9	Low diversity Shannon index <2.3

[1]Number of quantitative taxa from artificial substrates

A key development that took place during this time period was the pilot testing of ecoregions (Omernik 1987) and the development of the regional reference condition concept (Hughes et al. 1986). Along with the emerging concepts of biological integrity (Gakstatter et al. 1981, Karr and Dudley 1981) and multimetric assessment tools (Karr 1981, Karr et al. 1986), these advances represent the foundational development of the tools and criteria that emerged out of this phase. During this phase, integrated biological, chemical, and physical assessments were emphasized in support of a wider array of management issues (including nonpoint sources) in addition to the mainstay priorities of construction grants and NPDES permitting. The results of these assessments were documented in Comprehensive Water Quality Reports, the production of which included the first true integration of the monitoring and assessment, WQS, water quality modeling, and permitting programs. Study teams were formed for each project and included staff membership from each program. The analyses and recommendations included in these reports provided the basis for WQS use revisions, water quality based NPDES permits (including water quality certifications), advanced treatment justifications, and other findings related to the observed impacts of nonpoint sources.

The WQS were modified in 1985 to include a listing of designations by individual waterbody, as opposed to default designations or tributary membership (Table B-3). The original listing of individual waterbodies in the WQS was based on the Gazetteer of Ohio Rivers and Streams (Ohio Dept. of Natural Resources 1960). Waterbodies listed in the Gazetteer that had not been assessed via the biological and water quality assessment process were assigned a “default” designation of WWH. Waterbodies that were originally designated in 1978, or subsequent to that version of the WQS, retained those uses and this was denoted for each waterbody in the rules (Table B-3). Unconfirmed non-WWH uses required validation by site-specific monitoring and assessment due to a public notice issued by Ohio EPA in 1981. In reality, many “default” WWH designations also required reassessment because the variations in watershed settings and stressor gradients had only begun to be recognized. The Gazetteer of Ohio Rivers and Streams did not include all jurisdictional streams in the State; thus “unlisted” streams were assigned use designations as they became known via the systematic assessment of Ohio watersheds and/or as site-specific management issues arose. This further emphasized the role of monitoring and assessment in the designation of individual waterbodies.

Ongoing TALU Implementation and Maintenance Phase: 1987- present

Prompted by the testing and developments that took place in the initial implementation and development phase, Ohio EPA proposed and adopted numerical biological criteria (Figure B-2) and further refinements to the tiered uses. The narratives of the tiered uses first developed in 1978 were revised and new uses were added, both of which were influenced by the developments and the monitoring and assessment experience that took place in the preceding time period. The aquatic life use narratives were revised to reflect the operational definition of biological integrity (Karr and Dudley 1981) and provided direct reference to how the numerical biological criteria were developed and derived. These definitions follow:

> “Warmwater” – these are waters capable of supporting and maintaining a balanced, integrated, adaptive community of warmwater aquatic organisms having a species composition, diversity, and functional organization comparable to the twenty-fifth percentile of the identified reference sites within each of the following ecoregions: the interior plateau ecoregion, the Erie/Ontario lake plains ecoregion, the western Allegheny plateau ecoregion and the eastern corn belt plains ecoregion. For the Huron/Erie lake plains ecoregion, the comparable species composition, diversity and functional organization are based on the ninetieth percentile of all sites within the ecoregion. For all ecoregions, the attributes of species composition, diversity, and functional organization will be measured using the index of biotic integrity, the modified index of well-being, and the invertebrate community index as defined in “Biological Criteria for the Protection of Aquatic Life: Volume II, Users Manual for Biological Field Assessment of Ohio

Surface Waters," as cited in paragraph (B) of rule 3745-1-03 of the Administrative Code. In addition to those water body segments designated in rules 3745-1-08 to 3745-1-32 of the Administrative Code, all upground storage reservoirs are designated warmwater habitats. Attainment of this use designation (except for upground storage reservoirs) is based on the criteria in Table 7-14 of this rule. A temporary variance to the criteria associated with this use designation may be granted as described in paragraph (F) of rule 3745-1-01 of the Administrative Code.

TABLE B-3. Example of individual stream and/or segment use designations in the Ohio water quality standards showing aquatic life, water supply, and recreational use designations. Designation with a "+" means the use has been confirmed by monitoring and assessment. Designation with an "*" indicates a "default" designation or unverified designation – these waters will eventually be assessed via the rotating basin approach [excerpted from Ohio Administrative Code 3745-1-09].

Waterbody Segment	Use Designations													Comments
		Aquatic Life Habitat						Water Supply			Recreation			
	SRW	WWH	EWH	MWH	SSH	CWH	LRW	PWS	AWS	IWS	BW	PCR	SCR	
↓ ↓ ↓ ↓ ↓ ↓ ↓ ↓														
Scioto River – Frank Rd. (RM 127.7) to downstream from Bridge St. in Chillicothe (RM 70.7)		+							+	+		+		
- Greenlawn Dam (RM 129.8) to Frank Rd. (RM 127.7)	+	+							+	+		+		
- Olentangy R. (RM 132.3) to Greenlawn Dam (RM 129.8)	+			+					+	+		+		ECBP ecoregion – impounded MWH
- Dublin Rd. WTP dam (RM 133.4) to Olentangy R. (RM 132.3)	+	+							+	+		+		
- O'Shaughnessy Dam (RM 148.8) To Dublin Rd. WTP dam (RM 133.4)	+	+						+	+	+		+		
- all other segments		*							*	*		*		
Scippo Cr.			+						+	+		+		
Congo Cr. (Scippo Cr. at RM 1.64)			+						+	+		+		
Unnamed trib. Scippo Cr. (RM 16.31)		+							+	+			+	
Unnamed trib. Scippo Cr. (RM 18.87)							+		+	+			+	Small drainageway maintenance
Yellowbud Cr. - Ebenhack Rd. (RM 3.0) to mouth			+						+	+		+		
- all other segments		+							+	+		+		
RCA Tributary (Scioto R. RM 96.5)							+		+	+			+	Small drainageway maintenance

SRW = State Resource Water; WWH = Warmwater Habitat; EWH = Exceptional Warmwater Habitat; MWH = Modified Warmwater Habitat; SSH = Seasonal Salmonid Habitat; CWH = Coldwater Habitat; LRW = Limited Resource Waters; PWS = Public Water Supply; AWS = Agricultural Water Supply; IWS = Industrial Water Supply; BW = Bathing Waters; PCR = Primary Contact Recreation; SCR = Secondary Contact Recreation

Huron Erie Lake Plain (HELP)

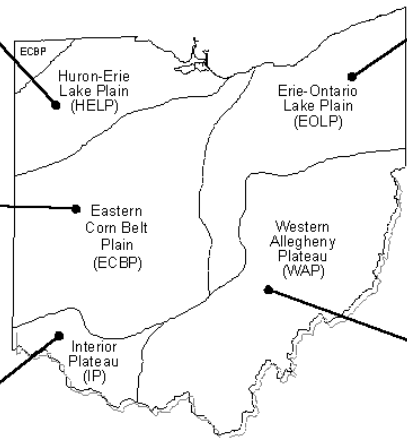
Use	Size	IBI	MIwb	ICI
WWH	H	28	NA	34
	W	32	7.3	34
	B	34	8.6	34
MWH-C	H	20	NA	22
	W	22	5.6	22
	B	20	5.7	22
MWH-I	B	30	5.7	NA

Erie Ontario Lake Plain (EOLP)

Use	Size	IBI	MIwb	ICI
WWH	H	40	NA	34
	W	38	7.9	34
	B	40	8.7	34
MWH-C	H	24	NA	22
	W	24	6.2	22
	B	24	5.8	22
MWH-I	B	30	6.6	NA

Eastern Corn Belt Plains (ECBP)

Use	Size	IBI	MIwb	ICI
WWH	H	40	NA	36
	W	40	8.3	36
	B	42	8.5	36
MWH-C	H	24	NA	22
	W	24	6.2	22
	B	24	5.8	22
MWH-I	B	30	6.6	NA



Western Allegheny Plateau (WAP)

Use	Size	IBI	MIwb	ICI
WWH	H	44	NA	34
	W	44	8.4	34
	B	40	8.6	34
MWH-C	H	24	NA	22
	W	24	6.2	22
	B	24	5.8	22
MWH-A	H	24	NA	30
	W	24	5.5	30
	B	24	5.5	30
MWH-I	B	30	6.6	NA

Interior Plateau (IP)

Use	Size	IBI	MIwb	ICI
WWH	H	40	NA	30
	W	40	8.1	30
	B	38	8.7	30
MWH-C	H	24	NA	22
	W	24	6.2	22
	B	24	5.8	22
MWH-I	B	30	6.6	NA

Statewide Exceptional Criteria

Use	.Size	IBI	MIwb	ICI
EWH	H	50	NA	46
	W	50	9.4	46
	B	48	9.6	46

FIGURE B-2. Numeric biological criteria adopted by Ohio EPA in 1990, showing stratification of biocriteria by biological assemblage, index, site type, ecoregion for the warmwater habitat (WWH) and exceptional warmwater habitat (EWH) use designations.

The narrative for the exceptional warmwater habitat (EWH) use designation retained the same application language with the following differences (in bold italics):

> ***"Exceptional warmwater"*** - these are waters capable of supporting and maintaining ***an exceptional or unusual*** community of warmwater aquatic organisms having a species composition, diversity, and functional organization comparable to the ***seventy-fifth percentile of the identified reference sites on a statewide basis*** . . . ***all lakes and reservoirs, except upground storage reservoirs, are designated exceptional warmwater habitats.*** Attainment of this use designation ***(except for lakes and reservoirs)*** is based on the criteria in Table 7-14 of this rule."

The narrative for coldwater habitat (CWH) was also revised and reflected a broader application of this use for reasons other than the existence of maintenance stocking of Salmonid fish species:

(i) "Coldwater habitat, inland trout streams" – these are waters which support trout stocking and management under the auspices of the Ohio department of natural resources, division of wildlife, excluding waters in lake run stocking programs, lake or reservoir stocking programs, experimental or trial stocking programs, and put and take programs on waters without, or without the potential restoration of, natural coldwater attributes of temperature and flow. The director shall designate these waters in consultation with the Director of the Ohio department of natural resources.

(ii) "Coldwater habitat, native fauna" – these are waters capable of supporting populations of native coldwater fish and associated vertebrate and invertebrate organisms and plants on an annual basis. The director shall designate these waters based upon the result of use attainability analyses.

The WWH, EWH, and CWH use designations are considered consistent with the minimum goals of the CWA (Section 101[a][2]) and the associated Federal Regulation (40CFR Part 130). However, the public notice issued in 1981 by Ohio EPA required that designated uses other than WWH be validated on a waterbody specific basis prior to basing permitting requirements on the attendant water quality criteria. Furthermore, a waterbody must reflect the capability to attain the EWH biological criteria at a sufficient number of sampling locations to be designated EWH (Ohio EPA 1987) and the CWH designation has its own set of requirements in the narrative. Such showings are not required for WWH, except that the potential to attain must be determined by biological and habitat assessments.

> "Coldwater" – these are waters that meet one or both of the characteristics described in paragraphs (B)(1)(f)(i) and (B)(1)(f)(ii) of this rule. A temporary variance to the criteria

Use designations that do not meet the minimum goals of the CWA, and thus require a use attainability analysis on a water body specific and/or segment-by-segment basis include:

> "Modified warmwater" – these are waters that have been the subject of a use attainability analysis and have been found to be incapable of supporting and maintaining a balanced, integrated, adaptive community of warmwater aquatic organisms due to irretrievable modifications of the physical habitat. Such modifications are of a long-lasting duration (i.e., twenty years and longer) and may include the following examples: extensive stream channel modification activities permitted under sections 401 and 404 of the act or Chapter 6131 of the Revised Code, extensive sedimentation resulting from abandoned mine land runoff, and extensive, permanent impoundment of free-flowing water bodies. The attributes of species composition, diversity and functional organization will be measured using the index of biotic integrity, the modified index of well-being, and the invertebrate community index as defined in "Biological Criteria for the Protection of Aquatic Life: Volume II, Users Manual for Biological Field Assessment of Ohio Surface Waters," as cited in paragraph (B) of rule 3745-1-03 of the Administrative Code. Attainment of this use designation is based on the criteria in Table 7-14 of this rule. The modified warmwater habitat designation can be applied only to those waters that do not attain the warmwater habitat biological criteria in Table 7-14 of this rule because of irretrievable modifications of the physical habitat. All water body segments designated modified warmwater habitat will be reviewed on a triennial basis (or sooner) to determine whether the use designation should be changed. A temporary variance to the criteria associated with this use designation may be granted as described in paragraph (F) of rule 3745-1-01 of the Administrative Code.

The Limited Resource Waters (LRW) use designation is defined as:

> "Limited resource water – these are waters that have been the subject of a use attainability analysis and have been found to lack the potential for any resemblance of any other aquatic life habitat as determined by the biological criteria in Table 7-14 of this rule. The use attainability analysis must demonstrate that the extant fauna is substantially degraded and that the potential for recovery of the fauna to the level characteristic of any other aquatic life habitat is realistically precluded due to natural background conditions or irretrievable human-induced conditions. All water body segments designated limited resource water will be reviewed on a triennial basis (or sooner) to determine whether the use designation should be changed. Limited resource waters are also termed nuisance

prevention for some water bodies designated in rules 3745-1-08 to 3745-1-30 of the Administrative Code. A temporary variance to the criteria associated with this use designation may be granted as described in paragraph (F) of rule 3745-1-01 of the Administrative Code. Waters designated limited resource water will be assigned one or more of the following causative factors. These causative factors will be listed as comments in rules 3745-1-08 to 3745-1-30 of the Administrative Code.

(i) "Acid mine drainage" – these are surface waters with sustained pH values below 4.1 s.u. or with intermittently acidic conditions combined with severe streambed siltation, and have a demonstrated biological performance below that of the modified warmwater habitat biological criteria.
(ii) "Small drainageway maintenance" – these are highly modified surface water drainageways (Usually less than three square miles in drainage area) that do not possess the stream morphology and habitat characteristics necessary to support any other aquatic life habitat use. The potential for habitat improvements must be precluded due to regular stream channel maintenance required for drainage purposes.
(iii) Other specified conditions.

The designation of specific waterbodies as MWH or LRW requires a use attainability analysis (UAA) based on a waterbody specific assessment. These do not meet the minimum conditions prescribed by the CWA (Section 101[a][2]). All of these were adopted in the Ohio WQS in 1990.

Relationship of Ohio's Tiered Uses to the Biological Condition Gradient
Ohio's current tiered uses represent refinements to the original tiered uses adopted in 1978 and reflect the developments that benefited from ten years of experience in applying a tiered use system. The practical impacts of these refined and tiered uses on water quality management are described in Table B-4 and include the designated use, the key attributes of that use, why a waterbody would be designated for that use, and some of the practical impacts to water quality management. All of the biological criteria and some of the chemical/physical criteria associated with each use are tiered in a logical relationship to the ecological attributes, which are ascribed by the designated use narrative and the translation of that narrative to specific criteria. This is consistent with the concepts of the BCG in that expectations and attainment of each use are measured by the biological criteria that are in turn designed to describe and measure increments in quality along the BCG (Figure B-3). Chemical-specific and physical parameters are cast in the role of stressor and exposure indicators and criteria (i.e., they are best used as design criteria in modeling and TMDLs). They directly support the development and implementation of abatement and management strategies via water quality management programs by providing the translation between associations of cause and effect via monitoring and assessment to enforceable controls via permitting and best management practices via TMDLs. The biological criteria are cast in the role of response indicators and as the primary criteria for determining use attainment status, measuring relative quality, and documenting the effectiveness of abatement and management strategies (Yoder and Rankin 1998, Karr and Yoder 2004). The logical relationship between exposure and response follows in that some of the key chemical criteria are more stringent for the uses that are representative of the higher tiers of the BCG (i.e., EWH) and least stringent for the lowest tiers (i.e., MWH, LRW). These are then translated accordingly to wastewater and other water quality management requirements. However, criteria that do not demonstrate an empirical relationship along the BCG are not tiered.

TABLE B-4. Key features associated with tiered aquatic life uses in the Ohio WQS (OAC 3745-1-07).

Aquatic Life Use	Key Attributes	Why a Waterbody Would Be Designated	Practical Impacts (compared to a baseline of WWH)
Warmwater Habitat (WWH)	Balanced assemblages of fish/invertebrates comparable to least impacted *regional* reference condition	Either supports biota consistent with numeric biocriteria for that ecoregion **or** exhibits the habitat potential to support recovery of the aquatic fauna	Baseline regulatory requirements consistent with the CWA "fishable" and "protection & propagation" goals; criteria consistent with U.S. EPA guidance with State/regional modifications as appropriate
Exceptional Warmwater Habitat (EWH)	Unique and/or diverse assemblages; comparable to upper quartile of *statewide* reference condition	Attainment of the EWH biocriteria demonstrated by both organism groups	More stringent criteria for D.O., temperature, ammonia, and nutrient targets; more stringent restrictions on dissolved metals translators; restrictions on nationwide dredge & fill permits; may result in more stringent wastewater treatment requirements
Coldwater Habitat (CWH)	Sustained presence of Salmonid or non-salmonid coldwater aquatic organisms; bonafide trout fishery	Bioassessment reveals coldwater species as defined by Ohio EPA (1987); put-and-take trout fishery managed by Ohio DNR	Same as above except that common metals criteria are more stringent; may result in more stringent wastewater treatment requirements
Modified Warmwater Habitat (MWH)	Warmwater assemblage dominated by species tolerant of low D.O., excessive nutrients, siltation, and/or habitat modifications	Impairment of the WWH biocriteria; existence and/or maintenance of hydrological modifications that cannot be reversed or abated to attain the WWH biocriteria; a use attainability analysis is required	Less stringent criteria for D.O., ammonia, and nutrient targets; less restrictive applications of dissolved metals translators; Nationwide permits apply without restrictions or exception; may result in less restrictive wastewater treatment requirements
Limited Resource Waters (LRW)	Highly degraded assemblages dominated exclusively by tolerant species; *should not* reflect acutely toxic conditions	Extensive physical and hydrological modifications that cannot be reversed and which preclude attainment of higher uses; a use attainability analysis is required	Chemical criteria are based on the prevention of acutely lethal conditions; may result in less restrictive wastewater treatment requirements

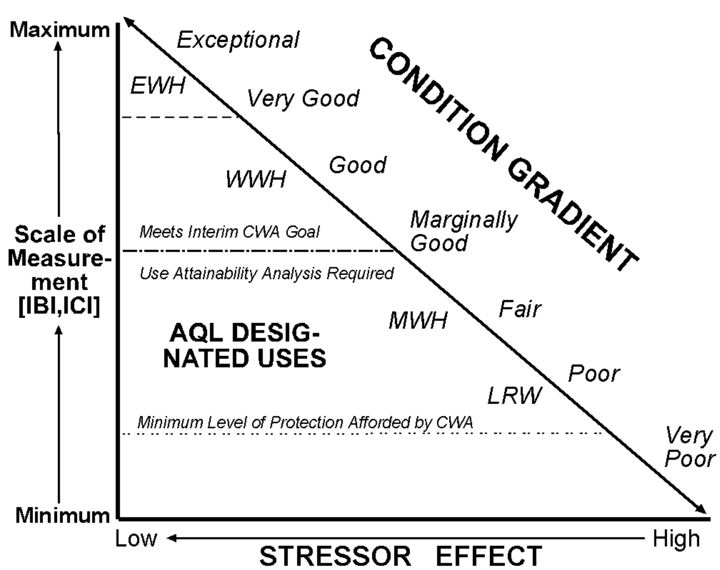


FIGURE B-3. The relationship of Ohio's tiered designated uses and numerical biological criteria to the Biological Condition Gradient.

Because of bioaccumulation concerns, many toxicant criteria are designed to protect all aquatic life uses even though they may demonstrate a graded response to the numeric biocriteria and tiered uses. For some of the heavy metals criteria where translators were developed between dissolved and total forms, concerns about the effects of potentially increased discharges of total metals resulted in a risk assessment that examined the relationships between the numeric biocriteria and total metals (Ohio EPA 1999a). This led to the derivation of "caps" on the amount of additional total metals that are permitted as a result of the dissolved metals translator process. These caps varied in accordance with the relationships demonstrated with the numeric biocriteria and tiered uses. Other parameters that do not demonstrate an empirical relationship along the BCG are not tiered. Future data exploration may well result in tiered chemical or physical criteria for stressors that are presently based on fixed, single value criteria. Such refined chemical criteria are expected to provide benefits to watershed-based management related to the prioritization of BMPs and in the application of emerging tools such as pollutant trading.

III. Establish technical program

From the outset, the implementation of tiered uses was intended to include a comprehensive and systematic monitoring and assessment program. The integration of the tiered uses with monitoring and assessment was an evolutionary development that followed the process outlined in Chapter 5 (Table 5-1; Figure 5-2) and Figure B-1.

How Does Ohio Collect Biological Data?
Ohio EPA employs a multiple chemical, physical, and biological indicators approach that utilizes each according to their most appropriate roles as indicators of stress, exposure, and response (Yoder and Rankin 1998). This approach leads to more effective regulation of pollution sources, improved assessment of diffuse and non-chemical impacts, and improves our ability to implement management strategies for successfully protecting and restoring the ecological integrity of watersheds. Key attributes that the biological indicators were developed to reflect include:

1) cost-effective collection of data
2) readily available science
3) be indicative of or extend to different trophic levels
4) integrate multiple effects and exposures
5) exhibit reasonable response and recovery times
6) be precise and reproducible
7) be responsive to a wide range of perturbations
8) be relevant to managerial and programmatic issues

Because it is impractical to monitor the entire organism assemblages present in an aquatic ecosystem, choices must be made. Ohio's choice of two organism groups (benthic macroinvertebrates and fish) is consistent with the ITFM (1992, 1995) recommendations and was done for a number of reasons. Each assemblage has been widely used in assessments and there is abundant information about their life histories, distributions, and environmental requirements. The benefit of having two different groups independently showing the same result is obvious and lends considerable strength to a bioassessment. However, differences in the responses by each group can lead to the definition of problems that might otherwise have gone undetected, underrated, or misunderstood in the absence of information from either organism group. For example, representatives of one assemblage may be able to tolerate and metabolize toxic substances that are highly detrimental to representatives of the other assemblage. The differences in recovery rates between each assemblage provide an added dimension to the understanding of how abatement processes work and document incremental changes through time. The value of such information in a risk management process should be obvious. Comparisons between the performance of fish and macroinvertebrates as arbiters of aquatic life use attainment showed non-agreement between assemblages at 33% in non-wadeable rivers, 21.2% in wadeable streams, and 28.2% in headwater streams (Yoder and Rankin 1995a). Assessments based on a single group would have overlooked proportions of the impairment that actually existed, let alone the loss of signal in diagnosing causal associations. Some of the concepts in Appendix C are based on this knowledge and experience.

Overview of the Technical Approach

The development and refinement of Ohio's biological assessment tools and criteria reflects an evolutionary process that is summarized in Figure B-1 and in Chapter 5 (Figure 5-2). The standardization of sampling and laboratory methods occurred first and illustrates the importance of the initial decisions about methods, taxonomic resolution, and professionalism early in the process (Ohio EPA 1987, DeShon 1995, Rankin 1995, Yoder and Smith 1999). From the outset of the systematic collection of biological data in Ohio, choices about sampling methods and laboratory procedures were the most important of the initial decisions that were made. These determine the attributes and characteristics of the resulting data and the usefulness and accuracy of the analytical tools and criteria that are developed. This, in turn, determines the quality of the entire approach including its ability to accurately determine biological impairments and discriminate relative quality along the BCG. Because of its primary role as a response indicator, it determines our perceptions of environmental quality and the effectiveness of our responses via water quality management programs and policies.

Sampling Methods

A number of decisions need to be made concerning the adoption of sampling methods. Decisions about sampling methods and gear, seasonal considerations, which organism groups to monitor, which parameters to measure and record, which level of taxonomy to use, etc. all were made early in the process. This was a critical juncture in the process since the decisions made here determined the effectiveness of the bioassessment effort.

The development of standardized sampling methods was the most important initial task in the implementation of Ohio's biological monitoring program. While many sampling methods and techniques existed for both macroinvertebrates and fish, many lacked adequate testing or standardization. The primary task was the testing, development, and validation of the chosen methods, which involved testing each for its ability to deliver good information at a reasonable cost. The goal was to use methods and protocols that would require 1-3 hours at a sampling site making it possible to sample several sites each day, tens of sites each week, and hundreds of sites each sampling season. A seasonal index period was also established during the summer-early fall (mid June to mid October).

For macroinvertebrates, artificial substrates were the method of choice and this was consistent with the U.S. EPA guidance of that time. The application of this method was further tested to refine the general approach in the early 1980s. A cluster of five artificial substrates bound to a concrete block are set in detectable current for a colonization period of six weeks. A dip net/hand pick sample of the surrounding natural substrates including all available habitats is collected at the time of substrate retrieval. This technique, known as qualitative sampling, employs a triangular frame dip net and can be used as a stand-alone sampling method. A site description data sheet is completed by a crew leader and includes information about the site habitat, environmental setting, and other pertinent information. Samples are retrieved, preserved in 10% formalin in the field, and transported to the laboratory for later processing. The specific methods are documented in written guidance manuals (Ohio EPA 1980, 1987, 1989b) that are codified by reference in the Ohio WQS.

Fish are collected using various wading and boat-mounted pulsed D.C. electrofishing gears, depending on the width and depth of the stream or river. These also had their origin in already available techniques, but the stratification of their use in different sizes of waterbodies was an issue that required prior testing and development. Sampling is standardized by lineal distance of stream or river and reach lengths were determined by sampling standard increments at methods test sites in the early 1980s. Fish samples are processed in the field and include identification to species, enumeration (counts and biomass) by age groups (adult, 1+, 0+), and delineation of external anomalies. A qualitative habitat assessment (QHEI; Rankin 1989, 1995) is completed over the entirety of the electrofishing reach. Fish sites are sampled once, twice, or three times within the seasonal index period, the frequency being determined by the complexity of the setting and the potential for episodic impacts. The specific methods are documented in written guidance manuals (Ohio EPA 1980, 1987, 1989b) that are codified by reference in the Ohio WQS.

Laboratory Methods
Each artificial substrate (quantitative) and natural substrate (qualitative) sample is processed in accordance with standardized procedures (Ohio EPA 1989b). This includes an initial pre-pick and visual scan for rare and large organisms, subsampling by major taxa group (mayflies, stoneflies, caddisflies, midges, others), and identification and enumeration to the lowest practicable taxonomic level. Ohio EPA staff perform both field sampling and laboratory processing.

Fish specimens that cannot be verified in the field are preserved in 10% formalin and transported to the laboratory for later processing. These are changed to 70% ethyl alcohol and identified to species. Verification of difficult specimens is performed by at least one qualified non-Ohio EPA taxonomist.

Analytical Methods
Ohio EPA analyzes biological data using routines available in the Ohio ECOS data storage, retrieval, and management system. Data is entered into Ohio ECOS following a data validation and QA/QC process to eliminate transcription and other errors. The principal indexes are based on multimetric techniques that were modified and calibrated for use in Ohio. For fish this includes the Index of Biotic Integrity (IBI; Karr 1981, Fausch et al. 1984, Karr et al. 1986) and the Index of Well-Being (IWB; Gammon 1976, Gammon et al. 1981). For macroinvertebrates it includes the Invertebrate Community Index (ICI; Ohio EPA 1987, DeShon 1995). In addition to the primary indexes, data analyses include the index metric values, relative abundance, and other aggregations of the data that exhibit ecologically meaningful patterns and information over space and time. This can include the use of multivariate analyses, parametric and non-parametric statistical techniques, and data mapping.

Staffing and Professionalism
Qualified and regionally experienced staff are employed to carry out the sampling and data analysis activities. Skilled and experienced staff direct, manage, and supervise all activities. This includes a high level of expertise in the field since many of the critical pieces of information are recorded and, to a degree, interpreted here. The same professional staff who collect the field data also interpret and apply the information derived from the data in a "cradle to grave" fashion. Thus the same staff who perform the field work also plan that work, process the data into information, interpret the results, and apply the results via assessment and reporting. Such staff, particularly those with sufficient experience, also contribute to policy and program development. The majority of data used by Ohio EPA is collected by agency staff. However, the methods and approach can be carried out by other entities and practitioners. Since 1999, Ohio EPA has operated a voluntary certification process and this will soon be mandated by the Ohio Credible Data Law.

How Does Ohio Decide What Waterbodies and Locations to Monitor?
In 1980, Ohio EPA initiated an intensive watershed survey design that included chemical/physical and biological assessments or surveys. A biological and water quality survey, or "biosurvey," is an interdisciplinary monitoring effort coordinated on a waterbody specific or watershed scale. The effort may involve a relatively simple setting focusing on one or two small streams, one or two principal stressors, and a handful of sampling sites or a much more complex effort including entire drainage basins, multiple and overlapping stressors, and tens of sites. Through the 1980s, Ohio EPA conducted biosurveys in 6-10 different study areas with an aggregate total of 250-300 sampling sites sampled/year.

• Rotating basin approach for determining annual monitoring activities.
• Correlated with NPDES permit schedule.
• Supports annual WQS use designation rule-making.
• Aligned with 15 year TMDL schedule in 1998.

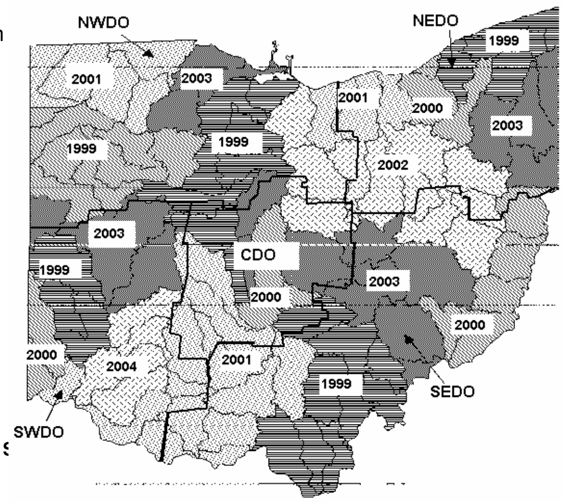


FIGURE B-4. Five-year basin approach for determining annual watershed monitoring and assessment activities and correspondence to support major water quality management programs.

While the purpose of these surveys was to support multiple program objectives, the schedule of water quality management program outputs was not always coordinated with the biosurvey schedule. In 1990, this process was formally coordinated beginning with a revision to the schedule for reissuance of major and significant NPDES permits. Ohio EPA formally adopted a five year basin approach in which biosurveys were scheduled two years in advance of the reissuance of NPDES permits (Figure B-4). The rotating basin approach proved its utility in two other instances. The first was in support of Ohio nonpoint source assessment in 1990, and the second when TMDLs became a major priority in 1998. The latter was seamlessly integrated into the rotating basin approach (Ohio EPA 1999b). In the 1990s, the demand for the watershed assessments increased further with up to 700 sites being sampled within 10-12 study areas in some years. The process of program integration was further institutionalized with a structured process for selecting watersheds, planning the monitoring, and analyzing and reporting the results (Table B-5).

TABLE B-5. Important timelines and milestones in the planning and execution of the rotating basin approach conducted annually and since 1990 by Ohio EPA.

Milestone	Timeline
December - February: (Months 1-3)	Initial screening of the major hydrologic areas takes place by soliciting input from the various program offices and other stakeholders.
February - March: (Months 3 thru 4)	Final prioritization of issues and definition of specific study areas. Resource allocation takes place and study team assignments are made.
March - May: (Months 4 thru 5)	Study planning takes place and consists of detailed map reconnaissance, review of historical monitoring efforts, and initial sampling site selection by the study team. Final study plans are reviewed and approved.
May - June: (Months 5 thru 6)	Final study plans are used to develop logistics for each field crew. Preparations are made for full-scale field sampling.
June - October: (Months 6 thru 10)	Field sampling takes place with field crews operating somewhat independently on a day-to-day basis, but coordinated by the study plan and the team leader. Study team communication takes place as necessary, especially to resolve unexpected situations.
October - February: (Months 10 thru 14)	Laboratory sample analysis takes place for chemical and biological parameters. Raw data is entered into databases for reduction and analysis. The study team meets to review the information base generated by the field sampling and to coordinate the data analysis and reporting effort.
November - May: (Months 11 thru 17)	Information about indicator levels 3-6 is retrieved, compiled, and used to produce analyses that will support the evaluation of status and trends and causal associations within the study area. Integration of the information (*i.e.,* assessment) is initiated.
May - December: (Months 17 thru 24)	The assessment process is completed by producing working copies of the assessment for review by the study team and a final edit for an internal peer review. Final assessment approved by management for use within and outside of Ohio EPA. It is used to support 305b /303d, NPDES permitting, water quality standards (*e.g.,* use designation revisions), and other programs where surface water quality is of concern.

Each biosurvey is designed and conducted to meet three major objectives:

1) determine the extent to which use designations assigned in the Ohio WQS are either attained or not attained;
2) determine if use designations assigned to a given waterbody are appropriate and attainable; and
3) determine if any changes in key ambient biological, chemical, or physical indicators have taken place over time, particularly before and after the implementation of point source pollution controls or best management practices.

The data gathered by a biosurvey is processed, evaluated, and synthesized in a biological and water quality report. Each biological and water quality study contains a summary of major findings and recommendations for revisions to WQS (e.g., Table B-6), future monitoring needs, or other actions which may be needed to resolve existing impairment(s) of designated uses. At the same time, the systematic execution of basin surveys builds a long-term database over space and time, creating and sustaining a resource of the development and improvement of tools, criteria, policies, and legislation (Figure B-5).

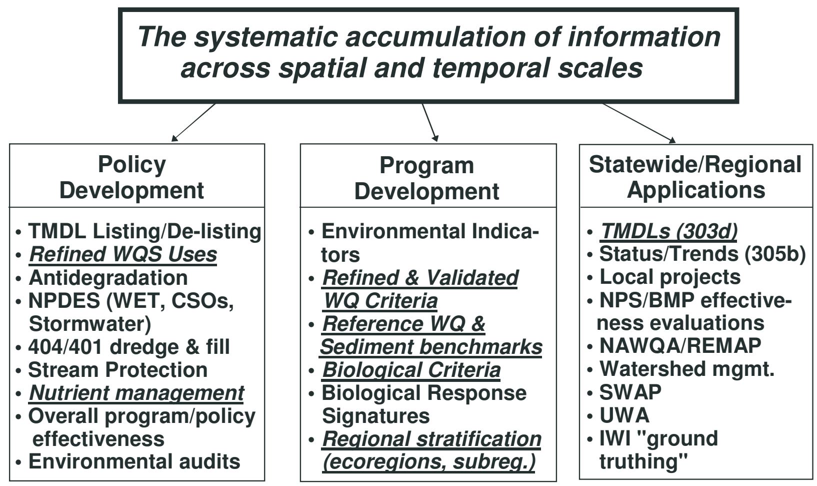


FIGURE B-5. Strategic support provided over time by systematic monitoring and assessment; functions related to the implementation of TALUs are italicized and underlined.

The recommendations for use designation revisions are a direct result of the biological and water quality assessment. Uses are designated on demonstrated potential to attain a particular use based on the following sequence (in order of importance):

1) attainment of the biocriteria (if attaining WWH or higher – attainment of EWH is required to be designated as EWH); and

2) if a WWH biocriterion is not met, the habitat potential determined by the Qualitative Habitat Evaluation Index (QHEI; Rankin 1989, 1995) and an associated assessment of warmwater: modified habitat attributes is used to determine the potential to attain WWH.

For uses less than WWH (i.e., MWH or LRW), a use attainability analysis is required and includes consideration of the factors that essentially preclude WWH attainment including the feasibility of restoring the waterbody. A use attainability analysis requires the following information:

1) the present attainment status of the waterbody based on a biological assessment performed in accordance with the requirements of the biocriteria, the Ohio WQS, and the Five-Year Monitoring Strategy (the latter pertains to adequacy of spatial design);

2) a habitat assessment to evaluate the potential to attain at least WWH; and

3) a reasonable relationship between the impaired status and the precluding human-induced activities based on an assessment of multiple indicators used in their appropriate indicator roles and a demonstration consistent with 40CFR Part 131.10 [g][1-6].

In the example from the Big Darby Creek watershed assessment conducted in 2000, all of the streams and segments listed in Table B-6 were sampled in accordance with Ohio EPA's geometric and intensive survey design. A number of the streams in Table B-6 were originally assigned aquatic life use designations in the 1978 and 1985 WQS based largely on best professional judgment or by tributary membership, while others were not yet designated. The current biological assessment methods and numerical biocriteria did not exist at that time. Most of the larger tributaries and the mainstem were previously designated based on biosurveys of specific segments and streams in 1979, 1981, 1988, and 1992. The use designations of most of the mainstem and some of the major tributaries were resolved by those efforts. However, many of the smaller streams in this watershed were evaluated for the first time using a standardized biological approach in 2000. Ultimately, the designations for each stream and river segment are based on direct sampling and assessments of each individual waterbody and the processes previously described. Extrapolation of sampling results for this and other purposes (e.g., status assessment) is minimal and occurs only within individual waterbodies. The application of the geometric watershed and intensive survey design included all tributaries and resulted in the addition of 26 previously unlisted and/or undesignated streams. Of these 26 streams, four were designated EWH, 18 as WWH, four as MWH, and two as LRW; an additional five stream segments were simultaneously designated CWH. Under the 1978 WQS, all 26 tributaries would have been designated as EWH by virtue of their tributary membership in the Big Darby watershed. This was extended to only the 19 named tributaries in the 1985 WQS, of which nine were later changed based on earlier biosurvey data. This example illustrates the comparative lack of accuracy in extrapolating uses by tributary membership within a watershed and the need to sample and assess individual streams for use designation purposes.

TABLE B-6. Summary of recommendations for use designations in the Big Darby Creek watershed based on a biological and water quality assessment completed in 2000. Symbols are listed for the existing designation/recommended designation (_ - undesignated; + - verified by biosurvey; * - unverified default designation from 1978 or 1985 WQS).

Water Body Segment	Use Designations												
		Aquatic Life Habitat						Water Supply			Recreation		
	SRW	WWH	EWH	MWH	SSH	CWH	LRW	PWS	AWS	IWS	BW	PCR	SCR
Big Darby Creek (02-200) [a] - Headwaters to RM 79.2		*	+			+			*+	*+		*+	
- RM 79.2 to mouth			*+						+	+		+	
Flat Branch (02-223) (RM 78.48)[b]			*	+					+	+		+	
Tributary to Flat Branch (02–365) (RM 1.5)				_+					_+	_+		_+	
Little Darby Creek (02-251) (RM 78.34) RM 3.5 to mouth			+			+			_+	_+		_+	
U.T. to B. Darby Cr. (02-361) (RM 74.91) RM 0.75 to mouth			_+						_+	_+			_+
Spain Creek (02-222) (RM 74.3) - Headwaters to RM 5.0		+	*			+			+	+		*+	
RM 5.0 to mouth			*+			+			+	+		*+	
Pleasant Run (02-221) (RM 72.01)			*+						+	+		*+	

Water Body Segment	Use Designations												
		Aquatic Life Habitat						Water Supply			Recreation		
	SRW	WWH	EWH	MWH	SSH	CWH	LRW	PWS	AWS	IWS	BW	PCR	SCR
U.T. to Big Darby Creek (02-360) (RM 69.4) RM 1.8 to mouth		_+							_+	_+			_+
Hay Run (02-220) (RM 67.6) RM 1.1 to mouth			_+						+	+		*+	
Prairie Run (02-219) (RM 63.84)							_+		_+				_+
Buck Run (02-209) (RM 63.74)		+	*						+	+		_+	
Robinson Run (02-207) (RM 53.69)		+	*						+	+		*+	
Sweeney Run (02-357) (RM 52.11) RM 1.7 to mouth		_+							_+	_+		_+	
Sugar Run (02-206) (RM 50.92) - Headwaters to RM 7.0			*	+					+	+		*+	
- RM 7.0 to mouth		+	*						+	+		*+	
U.T. to Sugar Run (02-358) (RM 7.39)				_+					_+	_+		_+	
Worthington Ditch (02-2356) (RM 50.62) RM 0.4 to mouth		_+							_+	_+		_+	
Ballenger-Jones Ditch (02-355) (RM 49.68) RM 3.72 to mouth		_+							_+	_+		_+	
Yutzy Ditch (02-364) (RM 47.1) RM 1.38 to the mouth		_+							_+	_+		_+	
Fitzgerald Ditch (02-272) (RM 44.96) RM 1.75 to mouth		+							_+	_+		_+	
Little Darby Cr.(02-210) (RM 34.1) Headwaters to RM 36.9			*+			+			+	+		+	
Little Darby Cr.(02-210) (RM 34.1) RM 36.9 to mouth			*+						+	+		+	
Clover Run (02-218) (RM 39.8)		_+							*+	*+			_+
Lake Run (02-216) (RM 36.9)		_+							*+	*+		*+	
Jumping Run (02-217) (RM 3.9)		_+							*+	*+		*+	
Treacle Creek (02-213) (RM 31.3)			*+						*+	*+		*+	
Howard Run (02-215) (RM 5.4)			*+						*+	*+		*+	
Proctor Run (02-214) (RM 3.69)			*+						*+	*+		*+	
Barron Creek (02-212) (RM 24.4)		_+							*+	*+			_+
Wamp Ditch (02-363) (RM 23.0)		_+							_+	_+			_+
Spring Fork (02-211) (RM 17.46)			*+						*+	*+		*+	
Bales Ditch (02-362)(RM 3.64) RM 1.72 to mouth		_+							_+	_+			_+
Smith Ditch (02-353) (RM 31.69)			_+						_+	_+		_+	
Tributary to Smith Ditch (02-354)(RM0.06)			_+						_+	_+		_+	
Gay Run (02-298) (RM 26.48)		_+							_+	_+		_+	
Hellbranch Run (02-204) (RM 26.1) Headwaters to RM 5.0		+	*						+	+		+	
Hellbranch Run (02-204) (RM 26.1) RM 5.0 to mouth			*+						+	+		*+	
Hamilton Ditch (02-259) (RM 11.19) -Hdwtrs to Feder Rd.				_+					*+	*+		_+	

Water Body Segment	Use Designations												
		Aquatic Life Habitat						Water Supply			Recreation		
	SRW	WWH	EWH	MWH	SSH	CWH	LRW	PWS	AWS	IWS	BW	PCR	SCR
Feder Rd. to mouth		_+							*+	*+		_+	
Clover Groff Ditch (02-245) (RM 11.19 - Hdwtrs to Feder Rd.				_+					*+	*+		_+	
Feder Rd. to mouth		_+							*+	*+		_+	
Springwater Run (02-203) (RM 24.0)		+	*						+	+		*+	
U.T. to Big Darby Creek (02-352) (RM 23.77)		_+							_+	_+			_+
U.T. to Big Darby Creek (02-270) (RM 20.2)		_+							_+	_+		_+	
U.T. to Big Darby Creek (02-366) (RM 18.41)		_+							_+	_+			_+
Greenbrier Creek (02-202) (RM 16.75)		+	*						+	+		*+	
Georges Creek (02-201) (RM 14.4)		+	*						+	+		*	_+
Lizard Run (02-273) (RM 12.93)							_+		+	+			_+

a - River code of the river or stream segment; b - River Mile of the confluence point with applicable receiving stream

While the principal focus of a biosurvey is on the status of aquatic life uses, the status of other uses such as recreation and water supply, as well as human health concerns are also addressed (Table B-6). The findings and conclusions of a biological and water quality study may factor into regulatory actions taken by Ohio EPA (e.g., NPDES permits, Director's Orders, the Ohio Water Quality Standards [OAC 3745-1]), and are eventually incorporated into Water Quality Permit Support Documents (WQPSDs), State Water Quality Management Plans, the Ohio Nonpoint Source Assessment, and the Integrated Report (combined 303[d] and 305[b] report). Periodic rulemakings are conducted to incorporate the use revision recommendations into the Ohio WQS, thus resolving the issue *prior to* the application of water quality management (see Figure 5-1, U.S. EPA's Water Quality Management Cycle). Figure B-6 summarizes the number of stream and river segments (mostly whole streams) where aquatic life uses have been revised as the result of a biological and water quality assessment in Ohio since 1978. This became a routine practice once the assessment criteria and decision-making process

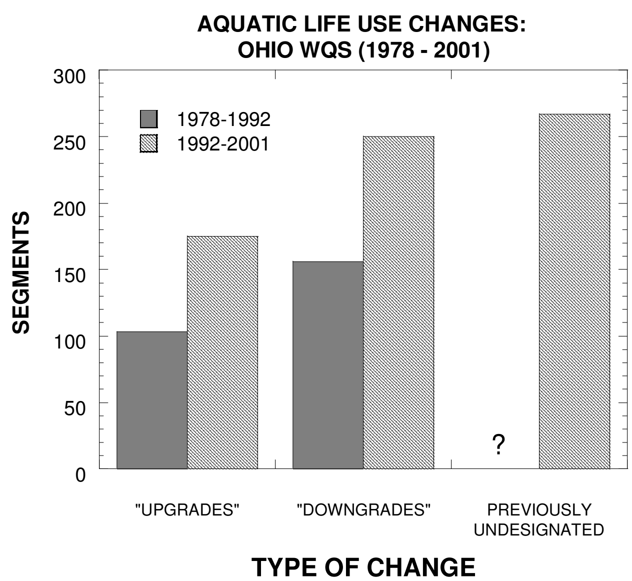


FIGURE B-6. The number of individual stream and river segments in which aquatic life use designations were revised during 1978-1992 and 1992-2001. Cases where the use was revised to a higher use are termed "upgrades" and cases where a lower use was assigned are termed "downgrades". Previously undesignated refers to streams that were not listed in the 1985 WQS, but which have been added via the Five-Year Basin Approach to monitoring and assessment.

for UAAs were established earlier in the assessment process. It required the development of reliable tools, particularly for determining status, assessing habitat, and determining causal associations, all of which are part of the developmental process described in Figure 5-2. The terms "upgrade" and "downgrade" are used figuratively here and in Figure B-6 as descriptors of the direction of change from the default use to that produced by a standardized assessment process. The majority of these changes are from the baseline of the original designations made in 1978 or 1985 without the benefit of systematic monitoring and assessment data, numerical biocriteria, and refinements in the process that occurred in the late 1980s. Thus, the original use designations are merely being "corrected" to the appropriate use based on a standardized process and more robust criteria and assessments.

Monitoring and assessment information, when based on a sufficiently comprehensive and rigorous system of environmental indicators, is integral to protecting human health, preserving and restoring ecosystem integrity, and sustaining a viable economy (ITFM 1992). Such a strategy is intended to achieve a better return on public and private investments in environmental protection and natural resources management. More and better monitoring and assessment information is needed to answer the fundamental questions about the condition of our water resources and to shape the strategies needed to address both existing and emerging problems within the context of watershed-based management. These principles have guided the development of surface water monitoring and assessment at Ohio EPA for the past 25 years and will continue to do so in the future.

IV. Develop and validate quantitative thresholds

The lack of adequate and reliable decision criteria for biological assessment has historically limited its usefulness, reliability, and wider acceptance in water quality management. In 1980, Ohio EPA developed an initial set of decision criteria for fish and macroinvertebrate assemblages that consisted of narrative quality ratings based in part on numerical biological index "guidelines" (Tables B-1 and B-2). These were intended to more directly reflect and assess the ecological goals espoused by the tiered aquatic life uses adopted in 1978. These early narrative biocriteria were comprised of contemporary measures such as taxa richness, indicator guilds, the Shannon diversity index, and the Index of Well-Being (Gammon 1976). Attainable expectations for a set of narrative community attributes were based on Ohio's experience with sampling approximately 150-200 sites statewide. This approach was used between 1980 and 1987 and was applied uniformly on a statewide basis. As the technology did not yet exist, no effort was made to account for background variability by using landscape partitioning frameworks such as ecoregions.

The narrative classification system consisted of assigning narrative quality ratings such as exceptional (consistent with the Exceptional Warmwater Habitat use), good (Warmwater Habitat use), fair, and poor. Exceptional and good met the goals of the Clean Water Act while fair and poor reflected a failure to attain those goals (Tables B-1 and B-2). The purpose of this narrative classification system was essentially two fold: 1) to provide an objective, systematic basis for assigning aquatic life uses to surface waters; and 2) to provide an objective, standardized approach for determining the magnitude and severity of aquatic life impairments for assessment purposes. Considerable judgment was used in applying these early narrative biological criteria on a site-specific basis and the system was characteristic of between a level 2 and 3 program *(See Appendix C)*. The aggregate impact of these assessments played a major role in setting and evaluating WQS use designations, designing water quality management plans, and developing advanced treatment justifications for municipal sewage treatment plants. These criteria also provided a basis for designating stream and river segments as attaining, partially attaining, or not attaining designated aquatic life uses in the 1982, 1984, and 1986 Ohio EPA 305b reports. They were, however, inherently prone to under-estimating impairment (DeShon 1995).

Regionally Referenced Numerical Biological Criteria

In 1986, a major effort was undertaken to develop regionally referenced and calibrated numeric biological criteria using a statewide set of regional reference sites. This was spurred by the Ohio Stream Regionalization Project in which the application of Omernik's (1987) ecoregions and the regional reference site concept (Hughes et al. 1986) was tested. For the fish assemblage, the Index of Well-Being was modified (Ohio EPA 1987) and the Index of Biotic Integrity (IBI; Karr 1981, Karr et al. 1986) was added. For macroinvertebrates, the Invertebrate Community Index (ICI; Ohio EPA 1987, DeShon 1995) replaced the narrative evaluations used previously. The IBI and ICI consist of metrics that include community production, function, tolerance, and reproduction in an aggregated index. This provides for a more rigorous, ecologically oriented approach to assessing aquatic community health and well-being. The process of deriving the numerical biological criteria is described more extensively in Ohio EPA (1987, 1989a,b) and Yoder and Rankin (1995a).

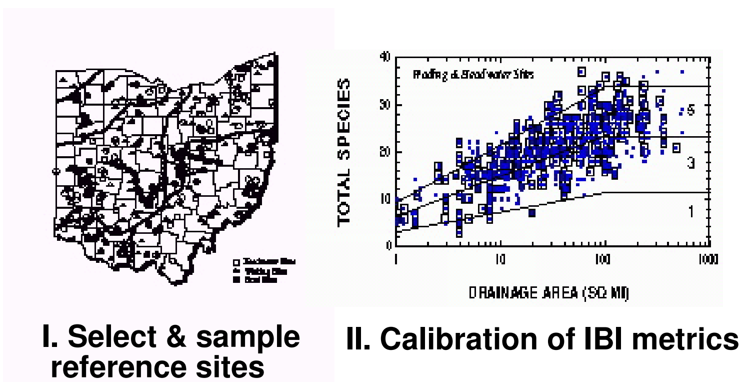
The derivation of the current numerical biological criteria is based on the biological "performance" that is demonstrated at least impacted, regional reference sites. This is consistent with the operational definition of biological integrity as defined by Karr and Dudley (1981), which provides the theoretical basis for this framework. The numerical biological criteria resulting from the application of this framework represent the assemblage performance that can reasonably be attained given contemporary background conditions. Although these do not emanate from an attempt to define "pristine," pre-Columbian conditions, the design framework includes a provision to "maintain" the biocriteria by continually resampling the reference sites – reference condition is monitored so that all reference sites are resampled once each decade. This promotes the periodic and orderly reassessment of reference condition and the database that drives the calibration of the biological indexes and the derivation of the numeric biocriteria. Furthermore, the knowledge base used in the development of the multimetric indexes includes an awareness of pre-settlement faunas and their characteristics. This is entirely consistent with the BCG and the description of attributes from "as naturally occurs" to an increasingly disturbed state. Thus, if pristine conditions do return this would be reflected by the periodic adjustments to the multimetric indexes, their calibration, and/or the numerical biological criteria.

Biological criteria in Ohio are based on two principal organism groups, fish and macroinvertebrates. Numerical biological criteria for rivers and streams were derived by utilizing the results of sampling conducted at more than 400 reference sites that represent the "least impacted" conditions within each ecoregion (Ohio EPA 1987, 1989a). This information was then merged within the existing framework of tiered aquatic life uses to establish attainable, baseline biological assemblage performance expectations on a regional basis. Biological criteria vary by ecoregion, aquatic life use designation, site type, and biological index (Figure B-2).

The framework within which biological criteria were established and used to evaluate Ohio rivers and streams includes the following major steps:

- selection of indicator organism groups;
- establish standardized field sampling, laboratory, and analytical methods;
- selection and sampling of least impacted reference sites;
- calibration of multimetric indexes (e.g., IBI, ICI);
- set numeric biocriteria based on attributes specified by each tiered aquatic life use designation;
- reference site re-sampling (10% of sites sampled each year beginning in 1990); and,
- making periodic (i.e., once per 10 years) adjustments to the multi-metric indexes, numeric biocriteria, or both as determined by reference site resampling results (Note: this latter step has yet to be undertaken by Ohio EPA).

The major steps in the biological criteria calibration, derivation, and application process are summarized in Figure B-7. The process integrates the technical process of index derivation and calibration with narrative statements about the desired biological assemblage condition and regionalization (e.g., ecoregions). This latter step is particularly important as it is needed to stratify regional landscape variability within a tractable framework. Figure B-7 portrays the calibration of the IBI for wading sites. A similar stepwise procedure was used to calibrate the Invertebrate Community Index for macroinvertebrates (Ohio EPA 1987, DeShon 1995) and the IBIs for the headwater and boatable site types. Once reference sites are selected and sampled (Step 1 in Figure B-7) the biological data is first used to calibrate the IBI (Step 2) and ICI. For fish three different IBIs were derived, one each for headwaters, wading (Step 3), and boat sites. The reference site IBIs are then used to establish numerical biological criteria (Steps 4 and 5). A notched box-and-whisker plot method was used to analyze the distribution of IBIs by ecoregion (Step 4). These plots contain sample size, medians, ranges with outliers, and 25th and 75th percentiles. Box plots have one important advantage over the use of means and standard deviations (or standard errors) because they do not assume a particular distribution of the data. Furthermore, outliers (i.e., data points that are two interquartile ranges beyond the 25th or 75th percentiles) do not exert an undue influence as they can on means and standard errors. In establishing biological criteria for a particular area or ecoregion we attempted to represent the "typical" biological community performance, not the extremes and outliers. These can be dealt with on a case-by-case or site-specific basis, if necessary. Once numerical biological criteria are determined, they are then used in making assessments of specific rivers and streams (Step 6).



Metric	5	3	1
Number of Species	Varies x Drainage Area		
No. of Darter Spp.	Varies x Drainage Area		
No. of Sunfish Spp.	>3	2-3	<2
No. of Sucker Spp.	Varies x Drainage Area		
Intolerant Species			
>100 sq. mi.	>5	3-5	<3
<100 sq. mi.	Varies x Drainage Area		
%Tolerant Species	Varies x Drainage Area		
%Omnivores	<19	19-34	>34
%Insectivores			
<30 sq. mi.	Varies x Drainage Area		
>30 sq. mi.	>55	26-55	<26
%Top Carnivores	>5	1-5	<1
%Simple Lithophils	Varies x Drainage Area		
%DELT Anomalies	>1.3	0.5-1.3	<0.5
Relative Abundance	>750	200-750	<200

III. Calibrated IBI modified for Ohio waters

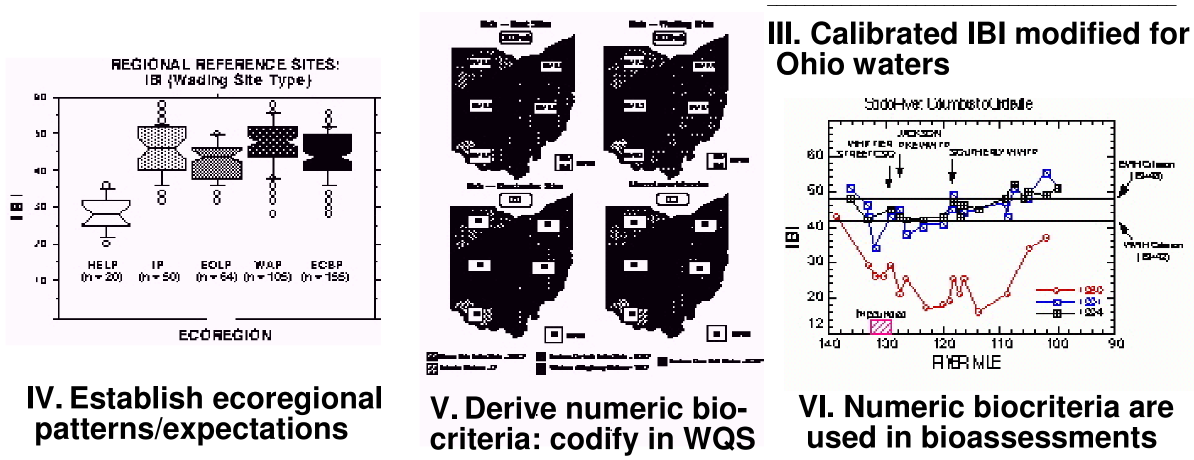


FIGURE B-7. The major steps of the Ohio EPA numeric biological criteria calibration and derivation process leading to their application in biological and water quality assessments; this example is for the Index of Biotic Integrity (IBI) for wading sites.

The outcome is a systematic process for measuring the essential products of aquatic structure and function that represent symptoms of ecosystem health. BCG derived and calibrated numeric biocriteria provide tangible measures of aquatic assemblages by which ecosystem health and well-being can be inferred. The tangible products of healthy watersheds are desirable biomass, water quality that is suitable for all uses, and an ability to assimilate background inputs that do not alter the key characteristics or processes associated with the aquatic assemblages detailed in the BCG (Table B-7). The key indicators of each are biological assemblage performance consistent with the designated use (measured by the biological indexes and compared to the numeric biocriteria) and chemical and physical quality comparable to least impacted regional reference conditions and other acceptable exposure thresholds.

TABLE B-7. The tangible products that are symptomatic of aquatic ecosystem health and the measurable biological, chemical, and physical indicators of healthy and degraded aquatic systems.

Tangible "Products"	Healthy	Degraded
Biomass	Desirable forms (quality biodiversity, game fish, birds, mammals, inverts., plants, algae, microbes)	Undesirable forms (low quality biodiversity, nuisance abundances, tolerant species dominate)
Water Quality	Comparable to regional reference	Poorer than regional reference
Assimilative Capacity	Processes background runoff and materials without adverse changes in biota	Inability to process background inputs due to reduced capacity to biologically and physically process excess materials
Measurable Indicators:		
Biological assemblages	Meet or exceed numeric biocriteria for TALU	Does not meet biocriteria for TALU; response varies by impact type and severity of impairment
Chemical indicators	Meets numeric criteria (some are TALU based) and is within reference thresholds	Exceeds numeric criteria and/or reference thresholds
Physical Indicators	Provides essential habitat attributes and hydrology	Degraded habitat and altered hydrology

What Process Was Used to Adopt Biocriteria in the Ohio WQS?

The adoption of numeric biocriteria and tiered uses in the Ohio WQS has been an evolutionary process over the preceding 25 years. There were many important events that determined the make-up and acceptance of the biocriteria and TALU in Ohio. These milestones are summarized in Table B-8. Some of the key events that resulted in a wider acceptance of the present day biocriteria and tiered uses were the legal proceedings on the use changes that occurred in the lower Cuyahoga River in 1988 and Ottawa River in 1989. Ohio EPA adopted a recommendation that the use designation of the Cuyahoga River mainstem be changed from a Limited Warmwater Habitat use designation to Warmwater Habitat based on biological and water quality surveys conducted between 1984 and 1987 and the ensuing UAA process. The former use was adopted in 1978 as a variance for specific point source derived pollutants. The biological assessments concluded that while the mainstem was severely impaired, the potential to attain WWH with achievable water quality based management of point sources was supported by the habitat assessment that showed a sufficiently intact habitat. This was eventually resolved via a legal process that included appeals of the initial decision up to the Ohio Supreme Court.

TABLE B-8. Key events and milestones that occurred in the evolutionary development, adoption, and implementation of biological assessments, numeric biocriteria, and tiered aquatic life uses in Ohio between 1974 and the present.

YEAR	MILESTONE	DESCRIPTION
1974	First Ohio WQS	General use, few numeric criteria, narrative “free froms”
1978	Initial TALUs	Tiered uses adopted, specific chemical criteria
1980	Narrative “biocriteria”	First organized approach to biological assessment; systematic monitoring & assessment
1983-4	Stream Regionalization Project	Testing and validation of Omernik’s ecoregions and reference site concepts in Ohio
1986-7	Derivation of numeric biocriteria	Statewide data collected to date was used to develop, derive, and calibrate numeric biocriteria based on multimetric indexes; biocriteria “User Manuals” published
1987	Biocriteria proposed in WQS	Initial proposal for numeric biocriteria
1987-89	Hearings on Cuyahoga River use change	Litigation of revision of a segment of the river form LWH to WWH; regulated entities Contested basis for the “upgrade”; the first test of the technical and policy aspects of the numeric biocriteria and TALU implementation; resolved at Ohio Supreme Court
1989	Hearings on Ottawa River use change	Litigation of revision of a segment of the river from LWH to WWH; regulated entities challenged; issue settled after Cuyahoga case ruling; led to more stringent regulation of point and nonpoint sources.
1990	Biocriteria adopted	Numeric biocriteria and refinements to TALUs were formally adopted in WQS
1990	Five-Year Basin Approach	A rotating basin approach that integrated key WQ management program outputs (e.g., NPDES permits) was initiated; use changes processed in annual rulemakings
1991	Internal training and orientation	All water program staff receives training in WQS, monitoring & assessment, modeling, and permit development and their integration.
1995	Lake Erie Bioassessment	Biological assessment methods and indexes developed for application to Lake Erie near shore and lacustuary habitats
1998	Wetlands bioassessment methods and biocriteria	Bioassessment methods and narrative criteria were developed for wetlands; includes various standardized assessment methods (beyond delineation) and a classification scheme.
1998	TMDL development process & schedule	TMDL development was integrated into the Five-Year Basin Approach ad schedule through 2015
1999	Re-sampling of regional reference sites	First re-sampling of regional reference sites was completed via the Five-Year Basin Approach
2003	Primary Headwater Habitat	Assessment and classification scheme for primary headwater streams that are not included in the existing numeric biocriteria are developed as a result of stream management applications.
2003	Ohio River	ORSANCO develops biological assessment tools and indexes as a precursor to numeric biocriteria for the Ohio R. mainstem.

Data collected via follow-up monitoring between 1984 and 2000 shows that attainment of the WWH biocriteria is increasing in the mainstem and proving the validity of both the WWH designation and the water quality based pollution abatement that the redesignation spurred (Figure B-8). A similar case involving the Ottawa River was resolved when this legal decision was made. No other appeals of the hundreds of use changes that have been made since that time have been filed. The systematic process of resolving use designation issues ahead of water quality management actions (permitting, listing, funding, planning) has proceeded as one of the most important outcomes of the Five-Year Basin Approach since that time. The next major milestone for the program will be the analysis of the first set of reference sites re-sampling that took place in the 1990s. In addition, level IV subregions have been delineated, which offers an additional level of potential stratification to the biocriteria derivation process.

The developments that occurred in the late 1990s including biological assessment and classification schemes for wetlands, Lake Erie near shore and lacustuary habitats, primary headwater stream habitat, and the Ohio River all happened as a result of the ground work laid in the 1980s for streams and rivers. It illustrates the natural growth process that can occur once the fundamentals of the approach are developed, tested, and adopted.

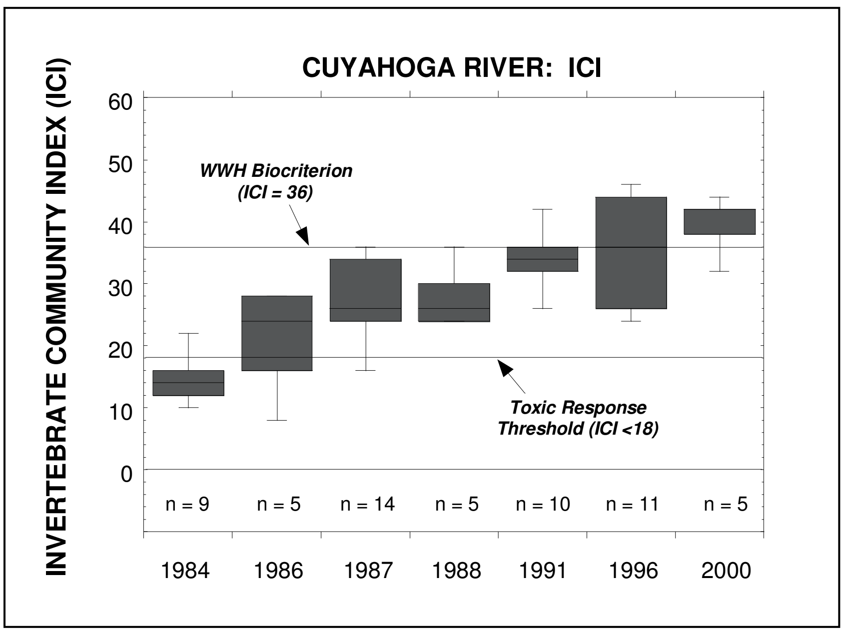


FIGURE B-8. Box-and-whisker plots of Invertebrate Community Index (ICI) results in the mainstem of the Cuyahoga River between Akron and Cleveland between 1984 and 2000.

Appendix C

Technical Guidelines: Technical Elements of a Bioassessment Program

(SUMMARY OF DRAFT DOCUMENT)

[This document has undergone review by State and U.S. EPA Regional biologists and managers. Data analyses are currently being conducted to refine certain technical elements (e.g., subsampling level, taxonomic resolution, spatial array of sites) that determine the level of rigor. A revised version will be prepared for a more comprehensive review by States and Tribes prior to finalization. A draft document will be available to the public in 2006.]

What are these technical guidelines and what is the purpose of the document?

This document is intended primarily for use by State and Tribal program managers and staff responsible for monitoring and assessment and WQS programs. States and Tribes can use this information to assess and communicate the precision of biological programs and, if deemed necessary, to refine and modify those programs. As States and Tribes increasingly use biological assessments and criteria to refine designated aquatic life uses, the need to recognize and communicate the level of precision of the biological program takes on greater importance. In addition, when the majority of States are in various stages of developing and improving their biological assessment programs, States and Tribes can use the type of detailed guidelines and milestones provided in this document to evaluate their progress.

Bioassessment is a major component of monitoring and assessment programs that include other chemical, physical, and environmental measures and indicators (ITFM 1992, 1995; Yoder and Rankin 1998). This document describes the critical, or key, technical attributes and processes of State and Tribal biological assessment programs. State and Tribal monitoring programs can also use the technical information presented in this document as a procedural template for evaluating the technical elements of their chemical and physical monitoring and assessment approaches. Ultimately, the integration of chemical and physical assessment with biological assessment will provide information to help States and Tribes better determine priorities and make more informed management decisions. State and Tribal programs can achieve appropriate levels of precision in their monitoring and assessment programs using currently available methods and technologies, and these approaches will produce a sufficiently accurate, comprehensive, and cost-effective program capable of supporting all water quality management programs.

What are the key technical elements of a bioassessment program?

There are 12 key technical elements that compose three basic methodological components: sampling design, methods, and data interpretation (see box at right). To better understand each technical element in a bioassessment program, it is important to

The 12 Key Technical Elements of a Bioassessment Program

<u>Sampling Design Component</u>

1. *Temporal Periodicity* of the sampling
2. *Spatial Coverage* of the sites within the area of interest
3. *Natural Classification* of the waterbodies as a framework for assessment
4. *Regional Reference Condition* development
5. *Reference Sites Selection Criteria*

<u>Methods Component</u>

6. Number and kinds of *Indicator Assemblages*
7. Methods for *Sample Collection*
8. Methods for *Sample Processing*

<u>Data Interpretation Component</u>

9. Attention to *Ecological Attributes* for indicators
10. Calibration of *Biological Endpoints*
11. *Diagnostic Capability* of the indicators
12. Use of *Professional Review* of documentation and methods

articulate the underlying rationale for each. The technical elements will be described in detail in the draft document that is being prepared for review.

How do environmental managers use these guidelines to evaluate the precision of their bioassessment program?

Included in the *Technical Guidelines* document is a checklist that enables managers and technical staff to evaluate their program's level of rigor for each of the 12 key technical elements. The checklist includes four levels of rigor, with Level 4 being the most rigorous. For an overall assessment of a water quality agency's bioassessment program, a checklist should be completed for each assemblage and waterbody ecotype, as bioassessment programs may have different levels of rigor for different waterbody ecotypes. It is important for the water quality agency to determine and reconcile these for management purposes since differing levels of rigor provide different levels of confidence in decision-making.

Evaluation of a program's level of rigor should be conducted collaboratively with State and Tribal technical staff and managers. Documentation will support completion of the checklist regarding aspects of the technical elements. Some variation between different elements will likely occur in terms of performance level (i.e., one element may receive a Level 4, while another is determined to be Level 2). Therefore, a scale that combines the rating of all elements will provide an overall indication of bioassessment program rigor. This cumulative evaluation provides a detailed analysis of the strengths and weaknesses of the comprehensive bioassessment and biocriteria program. In this rating system, we have considered all elements to be of equal weight. However, the data acquisition (sampling, processing) and treatment (analysis) phase is the linchpin of any program. One of the questions under discussion in preparation of this draft document is how to evaluate the influence of these particularly key elements.

What are the implications of having a bioassessment program with a high level of rigor?

The rigor and quality of biological assessments may vary among water resource agencies. The quality of the biological data is integral to effectively and accurately answering questions about condition, protection, restoration, or other management decisions regarding surface water resources. For example, bioassessment data obtained using a low level of rigor may provide a lesser degree of resolution needed to differentiate many stressor effects from natural variability.

The guidelines focus on four levels of rigor, where Level 4 is the most rigorous and provides the highest quality of data. The lower levels of rigor may detect and describe severely altered waters, and to a more limited extent, waterbodies in the best condition. As the level of rigor increases, the ability to discern more precisely different levels of biological condition increases. Figure C-1 illustrates the theoretical performance of the four levels of rigor of bioassessment techniques in assessing condition and the level of confidence in those assessments.

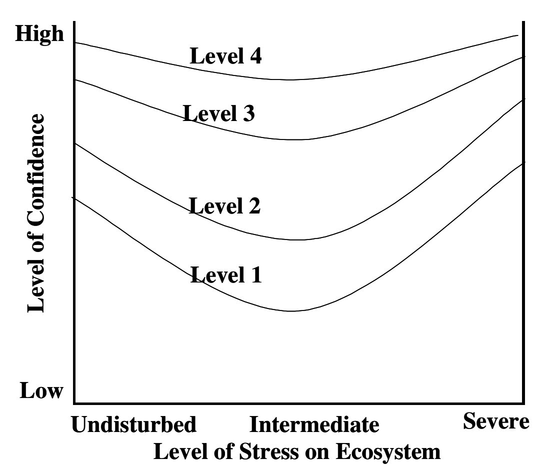


FIGURE C-1. Conceptual illustration of confidence in detecting different stress levels as a function of assessment rigor (Levels 1-4 with 4 being most rigorous).

Detecting and quantifying intermediately stressed sites, accurately describing associated causes and sources, and measuring along a stressor gradient will be done more accurately and with more confidence as the level of rigor increases. In Figure C-1, Level 4 provides the highest confidence in the biological assessment along the stressor gradient. Progressively less rigorous methods provide higher confidence in the assessment at the extremes of the stressor continuum, often only useful for status assessments. The difference in levels of rigor will be more apparent in applications requiring diagnostic capability. By first identifying the level of rigor attained by each of the key elements and the overall approach, States and Tribes can better use the data and information. For instance, States and Tribes may need a high level of confidence in an assessment, such as that associated with a Level 3 or 4 bioassessment, to determine level of stress along a gradient (Figure C-1). Less rigorous methods would not reliably detect disturbance.

Figure C-2 is a conceptual illustration depicting how increasingly comprehensive bioassessments better detect and discriminate differences along the BCG. As currently defined, Level 4 employs numeric biocriteria, based on calibrated and refined assessment tools (e.g., calibrated indexes or model output) that, in turn, are based on regional reference conditions at a sufficiently detailed level of geographic stratification and classification of aquatic ecotypes. This approach can discriminate different condition tiers (e.g., as in the Biological Condition Gradient) within a known margin of uncertainty. Level 3 usually employs a numeric and/or narrative assessment methodology that discriminates among fewer condition categories and reflects an ordinal scale of measurement (i.e., excellent, good, fair, poor). Assessment programs that rate as Level 2 may be unable to differentiate more than two broad categories or classes of condition. This level has a large degree of uncertainty about assessing stressors, and the pass/fail boundary may reflect an under- or over- protective threshold. Level 1 functions as a general screening tool and may identify best conditions from the worst in only a very coarse sense. The uncertainty with a Level 1 rated program precludes resolution to many management questions without further monitoring and assessments.

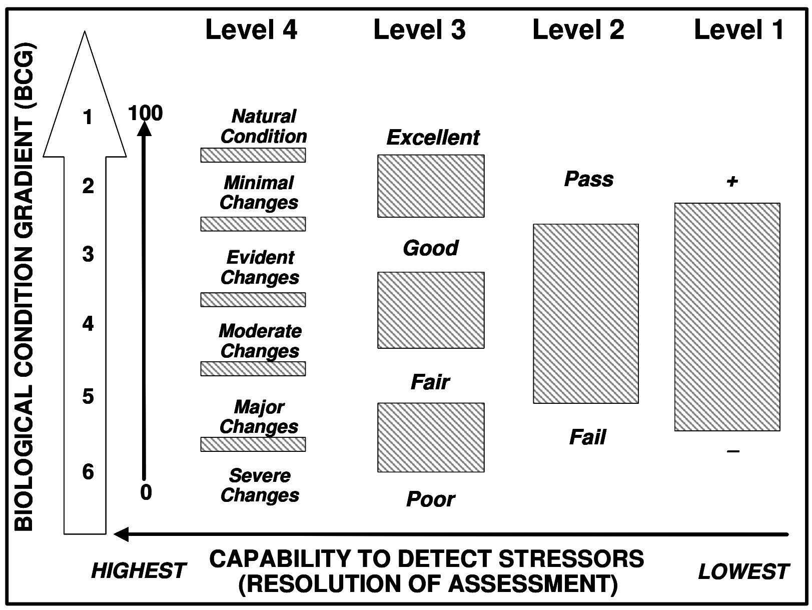


FIGURE C-2. Conceptual illustration of the capability of increasingly comprehensive bioassessments to detect and discriminate along the biological condition gradient. Shaded areas represent relative degree of uncertainty.

Appendix D

The Role of Reference Condition in Biological Assessment and Criteria

(INTRODUCTION TO DRAFT DOCUMENT ON DEVELOPMENT AND APPLICATION OF THE REFERENCE CONDITION CONCEPT)

The Clean Water Act's biological integrity objective and fishable swimmable goals pose significant challenges to States and Tribes charged with evaluating whether aquatic resources under their management achieve the objective and goals. One of the critical challenges is the development of a standard or benchmark by which to judge whether particular water bodies are in accord with the objective and goals. The concept of a "reference condition" and its implementation form the foundation on which to make such judgments.

This document provides States, Tribes, and other practitioners with guidelines on the reference condition concept and how to apply it in their water management programs, particularly for assessing the condition of aquatic resources. These guidelines are intended to broaden the implementation of biological monitoring and assessment, to increase the consistency among States and Tribes, and to improve the success of individual programs.

States, Tribes, and others have developed and implemented the concept of reference condition in a variety of ways to meet their individual needs, without comprehensive guidance from the U.S. EPA. This "bottom-up" approach has both advantages and disadvantages. Advantages include the exploration of a variety of interpretations of the concept and their implementation, yielding information on successes and difficulties. From these experiences comes an evaluation of what works and what does not. Disadvantages include the diversity of opinions about the concept and its role, leading to potential confusion and sometimes contradictory interpretation and implementation. The technical and policy challenges inherent in this effort have resulted in considerable variation in how individual States and Tribes define and use the concept. Establishing and using the reference condition concept appropriately is critical to implementing biological criteria and tiered aquatic life uses to protect and restore water resource quality. Part of the purpose of this document is to encourage consistency, both in the language that is used to express the concept, and in its everyday application.

This document will cover the following topics: a description of the concept of reference condition as well as related terms and concepts (including minimally disturbed, least disturbed, and best attainable conditions); methods for characterizing reference and related conditions; using water body classification to partition natural variability; setting thresholds to determine achievement of a target condition; and application of the concept in heavily modified regions (e.g., urban landscapes, agricultural regions) and waterbodies (reservoirs, regulated rivers). Both technical and implementation issues are addressed to increase the understanding of the concepts. A section on frequently asked questions and answers is included to address topics of particular concern to practitioners. Throughout the document, examples are drawn from existing State and Tribal programs to illustrate specific applications that are consistent with the guidelines.

In April 2003, U.S. EPA's Office of Water sponsored a National Biological Assessment and Criteria Workshop in Coeur d'Alene, Idaho. This workshop contained sessions on a variety of related topics including sessions on the reference condition concept, water quality standards, biocriteria, tiered aquatic life uses, and index development. A CD that contains many of the presentations at this workshop is included as an appendix to provide a snapshot of the state of the science at that time, and a means to flesh out some of the issues not addressed in detail in the body of this document. Material in this document supersedes any contradictory material presented on the CD because thinking has evolved since that time.

Technical and implementation issues

A principal technical challenge facing States and Tribes is accurately determining a reference or related condition from the range of historical and current ecological conditions. This may involve the analysis of data from existing reference sites and/or the modeling of historical information and expert opinion. Technical issues include understanding and taking into account natural variability through classification and/or modeling of natural gradients. Both classification and modeling need to be ecologically valid, yet practical for States and Tribes. A related issue is determining whether an existing condition is significantly (both ecologically and statistically) different from a specified condition (e.g., as specified in water quality standards). Scientific rigor is necessary, tempered by ease of understanding and implementation.

Implementation issues revolve around how States and Tribes can apply the reference condition concept to protect and improve an existing biological condition through application in water quality standards, including:

- Refinement of aquatic life uses through a) setting condition thresholds, b) interpreting/translating narrative aquatic life uses, and c) establishing subcategories of tiered aquatic life uses;
- Establishment of numeric biological criteria;
- Quantitative biological description of existing designated uses through bioassessments; and
- Determination of departure of existing condition from biological integrity.

U.S. EPA guidance on the implementation of the reference condition concept balances the need for scientific rigor and the need for practical application, that together result in the protection and improvement of water quality.

Appendix E

Statistical Guidance for Developing Indicators for Rivers and Streams: A Guide for Constructing Multimetric and Multivariate Predictive Bioassessment Models

(SUMMARY OF DRAFT DOCUMENT)

[This document has undergone various levels of review by a technical workgroup and U.S. EPA representatives. The current version is being prepared for a more comprehensive review prior to finalization. The final document is anticipated in 2005.]

States are faced with the challenges of not only developing tools that are both appropriate and cost-effective (Barbour 1997), but also the ability to translate scientific data for making sound management decisions regarding water resources. The approach to analysis of biological (and other ecological) data should be straightforward to facilitate a translation for management application. This is not meant to reduce the rigor of data analysis but to ensure its place in making crucial decisions regarding the protection, mitigation, and management of the nation's aquatic resources. In fact, biological monitoring should combine biological insight with statistical power (Karr 1987). Karr and Chu (1999) state that knowledge of regional biology and natural history (not a search for statistical relationships and significance) should drive both sampling design and analytical protocol.

A central premise of biological assessment is comparison of the biological resources of a waterbody to an expected reference condition. The condition of the waterbody is evaluated by its departure from the expected condition. Biological assessment of waterbodies depends on our ability to define, measure, and compare an assessment endpoint between similar systems. This guidance outlines analytical methodologies to perform two tasks:

- Characterize biological expectation.
- Determine whether a site deviates from that expectation.

The methods considered here use the same general approach: sites are assessed by comparing the assemblage of organisms found at a site to an expectation derived from observations of many relatively undisturbed reference sites. The expectations are modified by classifying the reference sites to account for natural variability. Biological variables are tested for response to stressors by comparison of undisturbed or minimally disturbed reference sites and disturbed sites. A set of "rules" is developed from this information, which are then used to determine if the biota of a site deviate from the expectation, indicating the degree to which the site is impacted.

Several analytical methods have been developed to assess the condition of water resources from biological data, beginning with the saprobien system in the early 20th century to present-day development of biological markers (Cairns and Pratt 1993). This document provides guidance for two methods for analyzing and assessing waterbody condition from assemblage and community-level biological information:

1. Multimetric assessment using an index that is the sum of several metrics. This is the basis of the Index of Biotic Integrity (IBI) (Karr et al. 1986), the Invertebrate Community Index (ICI) (Ohio EPA 1990); the Rapid Bioassessment Protocol (Plafkin et al. 1989); and State indexes developed from these (e.g., Southerland and Stribling 1995, Barbour et al. 1996a, Barbour et al. 1996b).

2. Assessment comparing actual species composition at a site to an idealized reference site predicted from a multivariate statistical model. This is the basis of the River InVertebrate Prediction And Classification System (RIVPACS; Wright et al. 1984, Furse et al. 1984, Moss et al. 1987, Wright 1995, Wright 2000) and the AUStralian RIVer Assessment System (AUSRIVAS; Davies 2000, Simpson and Norris 2000).

Many other methods are possible, as well as permutations of the two methods above, all of which are beyond the scope of this document. The two approaches were selected because:

- They use community and assemblage data.
- The methods are not restricted to any one assemblage. The examples all use freshwater benthic macroinvertebrates, but any other assemblage could also be used, such as fish phytoplankton, zooplankton or macrophytes.
- The methods are general, and have been used by several agencies in many areas. The examples used to illustrate the methods have also been carried out over wide geographic areas with many sites, demonstrating the generality of the methods.
- The methods have been fully documented and illustrated with case examples.
- These analysis methodologies are cost-effective and easy to communicate to managers and the public.

Once the framework for bioassessment is in place, conducting bioassessments becomes relatively straightforward. Either a targeted design that focuses on site-specific problems or a probability-based design, which has a component of randomness and is appropriate for 305(b), area-wide, and watershed monitoring, can be done efficiently. Routine monitoring of reference sites may be based on a probability design, which will allow cost efficiencies in sampling while monitoring the status of the reference condition of a State's streams. Potential reference sites of each stream class would be randomly selected for sampling, so that an unbiased estimate of reference condition can be developed. A randomized subset of reference sites can be resampled at some regular interval (e.g., a 4-year cycle) to provide information on trends in reference sites.

This document outlines the steps required to complete multimetric and multivariate predictive assessment models. It includes sections briefly covering the conceptual principles behind each step and then uses an example dataset that demonstrates the practical application of those principles step by step. It begins with a discussion of some concepts and approaches common to both techniques and then moves into multimetric and multivariate predictive models. At the end, it concludes with a discussion of how biocriteria can be developed from either of the approaches.

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Washington, DC 20460
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EPA-822-R-05-001
August 2005