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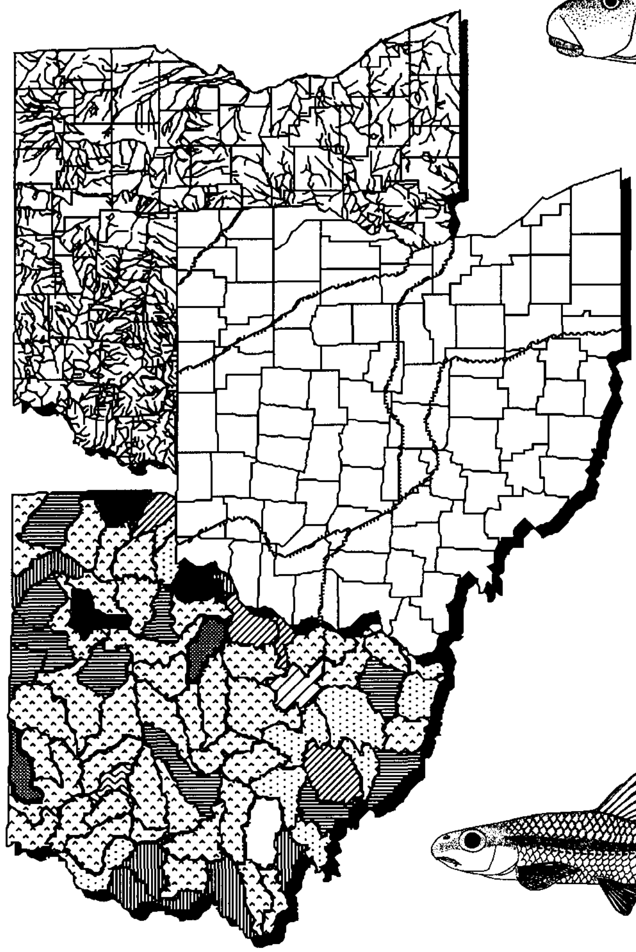
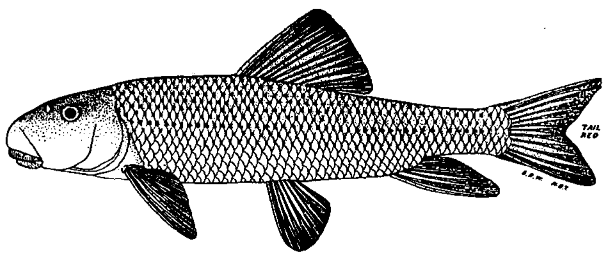
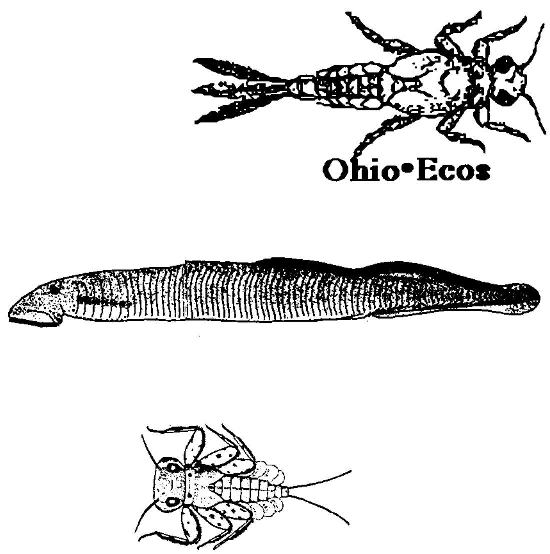
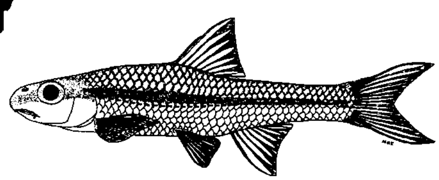
The Role of Biological Criteria in Water Quality Monitoring, Assessment, and Regulation









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Environmental Regulation in Ohio
Institute of Business Law
Cleveland, Ohio

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The Role of Biological Criteria in Water Quality Monitoring, Assessment, and Regulation

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Abstract

The measurement of progress towards meeting the goals of national clean water legislation have historically been predominated by measures of administrative activity and chemical water quality. These measures have increasingly come into question largely because neither communicates about nor measures real changes and conditions in the environment. In an effort to improve this situation more direct and comprehensive ecological indicators and associated indices, and bioassessment frameworks have recently been developed. The Index of Biotic Integrity (IBI) typifies what are referred to as multimetric indices which essentially are aggregations of complex aquatic community information. The underlying theory about how to practically define the Clean Water Act goal of biological integrity and regional partitioning frameworks such as ecoregions have prompted the concept of biological criteria as a direct measure of aquatic ecosystem condition. Within a regional reference site framework multimetric indices are calibrated and numerical biocriteria derived which are reflective of baseline, reference conditions. The resultant biological criteria provide an important ecosystem response indicator which heretofore has been lacking. Ohio EPA adopted numerical biological criteria into the Ohio Water Quality Standards (WQS) regulations as part of an existing water quality management process. These criteria compliment the existing stressor and exposure indicators which are comprised of chemical, toxicological, and physical parameters.

Comparisons of biological criteria and chemical criteria based assessments demonstrate that the risk inherent to relying solely on the latter carries a significant risk of making type II errors in resultant statements of waterbody condition. The implications to water resource management are significant in that potentially major problems will either be overlooked, underestimated, or improperly

characterized by conventional water quality management programs. The problems encountered in 20 years of chemical and toxicological based water quality management have resulted largely from the inappropriate uses of stressor and exposure indicators as response indicators. Many challenges remain to fully implement biological criteria within an integrated water resource management system. The U.S. EPA policy of independent application and concerns expressed by both the regulatory community and States presently serve as impediments to the wider use and adoption of biological criteria. Potential solutions including program and regulatory incentives are described herein.

Biological criteria in the Ohio WQS regulations has provided the Ohio EPA with some substantial advantages in surface water quality management and regulation. Some of these include: determining appropriate aquatic life use designations for surface water bodies, "reality checks" on prescriptive regulations, assessing cumulative impacts, extending anti-degradation concerns to nonpoint sources including habitat influences, defining high quality waters, and managing regulatory programs for real world environmental results. Adopting a biological criteria approach to monitoring and assessment fosters a more complete integration of important ecological concepts with water resource policy and management and strategic planning. Biological criteria and the attendant monitoring and assessment design provides a means to incorporate the broader concept of water resource integrity while preserving the appropriate roles of the traditional chemical/physical and toxicological approaches developed over the past three decades.

Introduction

The goal of this paper is to describe the framework used by Ohio EPA to develop an ecological response and exposure indicator of the condition of ecological resources and suggest what roles such information should have in water resource management and policy. A principal objective of the Clean Water Act (CWA) is to restore and maintain the physical, chemical, and biological integrity of the nation's surface waters (CWA Section 101[a][2]). Although this goal is fundamentally ecological, the specific methods by which regulatory agencies have attempted to reach this goal have been predominated by such non-ecological measures as chemical/physical water quality (Karr *et al.* 1986). The rationale for this process is well known - chemical criteria developed through toxicological laboratory studies of selected aquatic organisms serve as surrogates for determining attainment of the ecologically-based goals of the CWA.

Goals of State Water Quality Standards

How can effective and meaningful measures be devised to determine progress towards meeting the goals of the CWA? On this there is a breadth of comprehension and expectations. Although the language of the CWA regarding biological integrity is fairly concise, the expectations that legislators, regulators, the regulated community, environmental activists, and the public in general have are varied and at times widely divergent. In some instances it seems the goal is to merely achieve clear water (*i.e.*, maximum visibility similar to that exhibited by oligotrophic lakes common to the northern latitudes). During the last decade, regulatory programs concentrated almost exclusively on controlling toxic chemicals in the water column, tactically ignoring other factors that affect the biological integrity goal of the CWA. This approach is unquestioningly applied, particularly in the NPDES permitting and WQS programs, and many of the recently emerging programs. Thus, we assert here that there has been a significant failure to adequately understand or visualize the goals of the CWA. Simply stated, clear water is not a prerequisite for clean water as clean water frequently fails the clearness test in many areas of the U.S.

The underlying presumption that improvements in chemical water quality will be followed by a restoration of biological integrity has increasingly come into question. Although the traditional chemical/physical water quality approach may give an impression of empirical validity and legal defensibility it does not directly measure the ecological health and well-being of surface water resources. Nor does it comprehensively follow with the definition of pollution in Section 502 of the CWA as "....manmade or man-induced alteration of the chemical, physical, biological or radiological integrity of water...." which clearly is broader than a singular concern for chemical pollutants. The notion that controlling point source discharges of chemicals holds the key to attaining the biological integrity goal of the CWA has become so ingrained into the present regulatory culture that some interesting conceptions about water quality standards and what the goals of the CWA actually are about have arisen. This relatively narrow focus has lead to disparities in the understanding of environmental processes and widely varied expectations regarding the goals, objectives, and results of water quality management and water pollution control in general.

Water quality can easily become a confused and nebulous concept, especially when no demonstrable or tangible end-product can be easily identified. Regulatory advocates assert that the attainment of administrative goals (*e.g.*, permit compliance) will logically be followed by actual environmental improvements. However, providing support for environmental verification via ambient monitoring is frequently inadequate or lacking altogether. Do we simply continue to assume that environmental improvement occurs without making an effort to confirm this with environmentally-based and standardized monitoring? The presumptions of an administrative, surrogate indicators dominated approach follow: 1) *any* chemical water quality criteria exceedence is bad; 2) the observation of no exceedences is good; and, 3) an emphasis on the control of toxic chemicals will result in the attainment of CWA goals. In fact well intentioned, but simplistic quests for clear and/or chemically cleaner water have fostered management strategies which have actually resulted in *increased* net damage to the environment (Ohio EPA 1992a) because of an over-reliance on these sometimes flawed presumptions.

Some characteristics of a reliance on these presumptions, which are also symptomatic of an incomplete foundation in water resource policy and legislation, include:

- a reliance on prescriptive approaches to management and regulation;
- common usage of and reliance on anecdotal information;
- an emphasis on administrative activities;
- a skewed allocation of resources between different programs;
- emphasis on a single component to the near exclusion of others (*e.g.*, point vs. nonpoint sources, toxics vs. nutrients and/or habitat, etc.); and,
- inconsistent environmental statistics reported between States (*e.g.*, 305b report statistics, 304l toxic discharge list, etc.).

Unfortunately, these are representative of most State water quality management programs which, under Federal direction and mandate, are comprised of administrative predominated, point source biased, and toxics oriented approaches. However, U.S. EPA, the States, and others are beginning to address these foundational shortcomings through efforts such as the Strategic Goals Initiative, ecological indicators development, and the Intergovernmental Task Force on Monitoring Water Quality (ITFM 1992, 1993).

Major Factors Which Determine Water Resource Integrity

A growing body of information shows that other factors in addition to chemical water quality are responsible for the resultant condition of surface water resources (Karr and Dudley 1981; Karr *et al.* 1986; Rankin and Yoder 1990a; Rankin 1995). Because biological integrity is affected by multiple factors in addition to chemical water quality, controlling chemicals *alone* does not in itself assure the restoration of biological integrity (Karr *et al.* 1986). If real progress towards restoration and protection of aquatic ecosystems is to occur, a broader focus on the water resource as a whole is needed. The concepts inherent to the biological integrity goal statement implicitly includes such a focus. While whole effluent toxicity testing (acute and chronic bioassays) offers an improvement over a strictly chemical-specific approach, it also lacks an ability to broadly assess ecosystem effects, particularly those caused by physical, episodic, and non-toxic chemical impacts. Biological criteria and the attendant monitoring and assessment design provides a means to incorporate the broader concept of water resource integrity while preserving the appropriate roles of the traditional chemical/physical and toxicological approaches developed over the past three decades.

Aquatic Ecosystem Condition Indicators

The health and well-being of the aquatic biota in surface waters is an important barometer of how effectively we are achieving environmental goals, namely the Clean Water Act (CWA) goal of maintaining and restoring the biological integrity of the nation's surface waters. This concept underlies the basic intent of State WQS. Yet this tangible end-product of CWA regulatory and water quality planning and management efforts is frequently not linked to source control and other

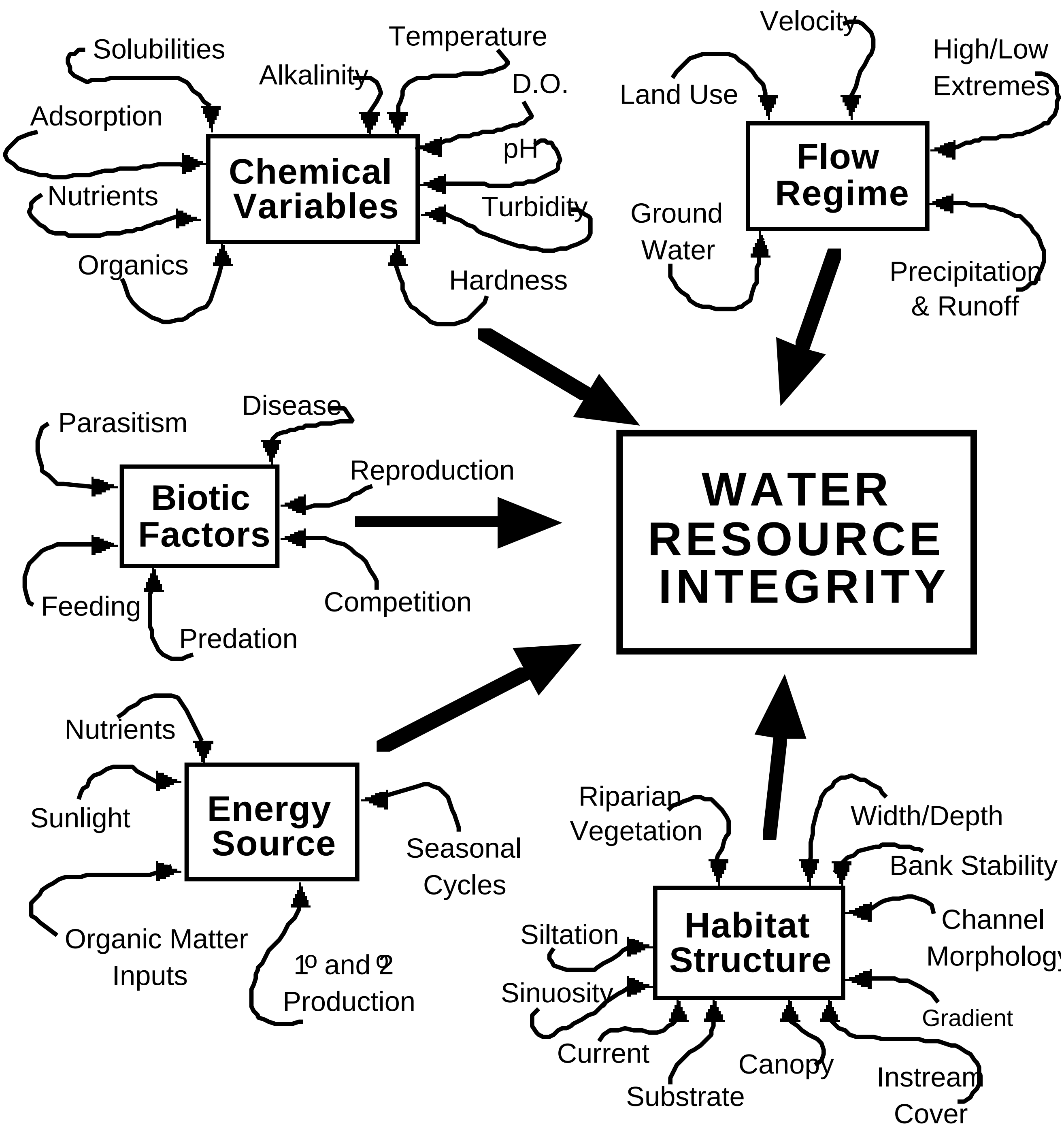


Figure 1. The five principal factors, with some of their important chemical, physical, and biological components that influence and determine the integrity of surface water resources (modified from Karr *et al.* 1986).

program performance measures. Simply stated biological integrity is the integrated result of chemical, physical, and biological processes in the aquatic environment (Figure 1). The interaction of these processes is readily apparent in the functioning of aquatic ecosystems. Thus management efforts which rely on comparatively simple, surrogate approaches to assessment and management carry a significant risk of failure in attempting to achieve a real restoration of ecological integrity (Karr 1991). Therefore, ecological concepts, biological criteria, and attendant monitoring and assessment tools need to be further incorporated into the management of surface water resources.

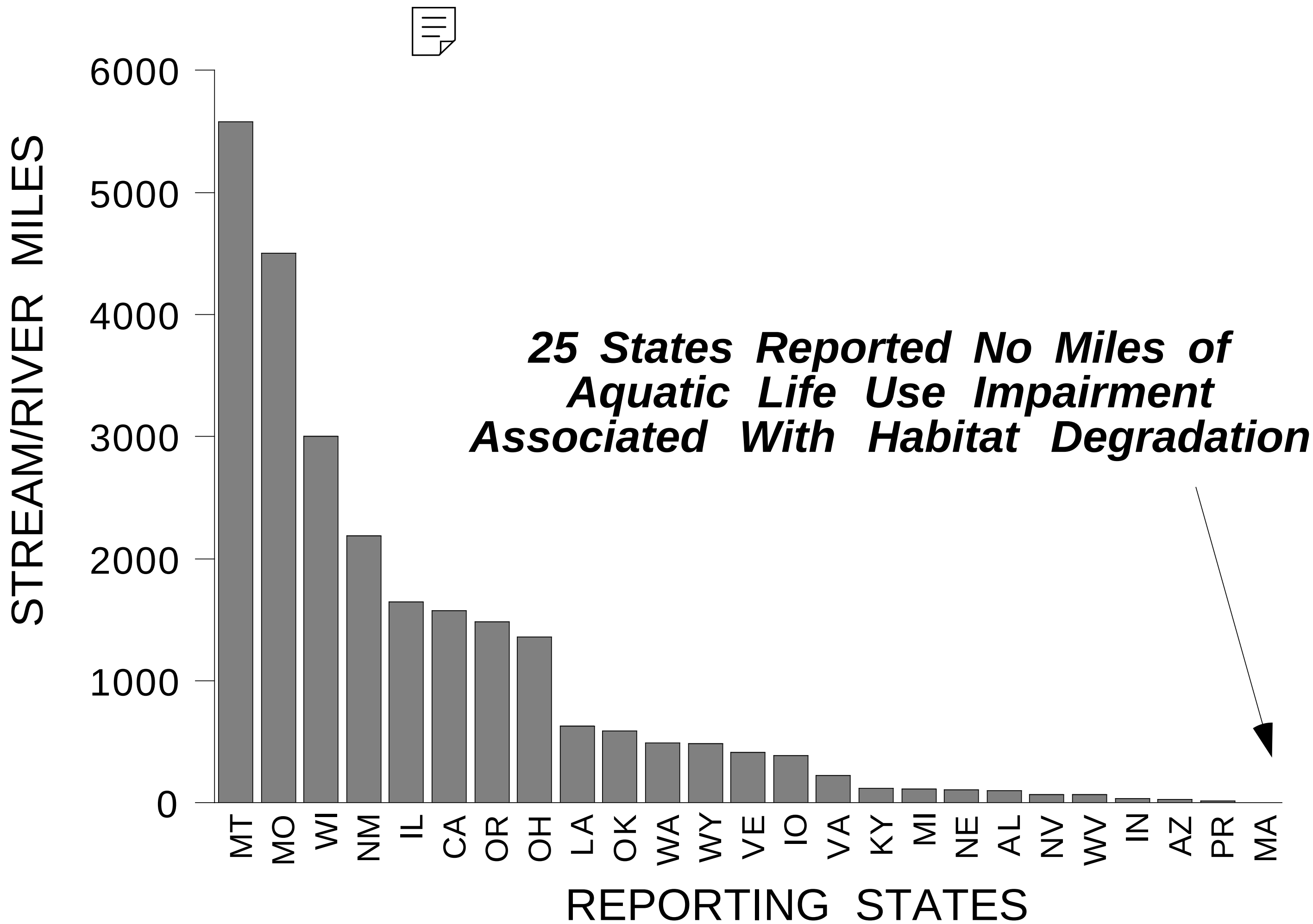
Disparities in the Use of Indicators: Case Examples

The use of and *appropriate* roles of different chemical, physical, and biological indicators to portray the biological integrity goal of the CWA is still being debated. While many States and organizations rely primarily on biological indicators to assess the condition of their water resources, others emphasize chemical/physical indicators. Thus the "playing field" is comparatively uneven and inconsistent on a national scale. The following case examples demonstrate the inherent risks of inaccurately portraying the condition of aquatic resources when chemical/physical indicators alone are used as programmatic end-points.

A comparison of the capabilities of chemical water quality criteria and biological criteria to detect aquatic life impairment based on ambient monitoring in Ohio exemplifies the need to use a suite of chemical, physical, and biological indicators. Out of 645 stream and river segments analyzed, impairment revealed by biological community measures was evident in 49.8% of the segments where no impairments of chemical water quality criteria were observed (Ohio EPA 1990; Yoder 1991a). A smaller proportion (2.8%) revealed significant exceedences of chemical criteria where the biological criteria were fully attained. While these discrepancies may seem remarkable, the causes for each are related to the fundamental differences between what chemical and biological indicators actually measure. Biological communities respond to and integrate a wide variety of chemical, physical, and biological factors in the environment whether of natural or anthropogenic origin. Simply stated, controlling chemicals alone does not assure the ecological integrity of water resources (Karr *et al.* 1986). The inclusion of numerical biological criteria into the assessment framework at Ohio EPA has had some important ramifications in how environmental quality is accounted for. This was most apparent in an "increase" in the proportion of stream and river miles classified as non-attaining from 9% in 1986 to 44% in 1988. This difference was due primarily to the superior ability of numerical biological criteria to detect and quantify impairments beyond those caused by exceedences of chemical criteria alone. This example should also lend understanding as to why significant differences can occur in the statistics reported to U.S. EPA by different States. Such differences exist not only because of a reliance on different indicators, but also because of different frameworks for deriving and calibrating indicator endpoints.

A good example of the net effect of different indicator frameworks is in the proportion of waters reported as impaired by habitat degradation by the States to U.S. EPA (Figure 2; U.S. EPA 1994). Out of 58 States and Territories which report such statistics on a biennial basis, 25 (nearly one-half) reported zero (0) miles of rivers and streams as impaired by habitat modification activities (includes channelization, impoundments, riparian encroachment, hydrological modifications, or the physical degradation of substrates). Of the remaining States, only 15 reported more than 100 miles (0.3% of Ohio's perennial streams) of streams and rivers impaired by degraded habitat. These statistics are difficult to accept at face value given the pervasive nature of habitat modifying activities such as flood control, agricultural practices (including silviculture), resource extraction, and urban development throughout the U.S. The discrepancies between State statistics for this impairment category is likely related to the non-use of habitat relevant indicators and programmatic biases towards the control of chemicals and point source discharges.

These examples demonstrate that the significant risk inherent to relying on surrogate information (*i.e.*, using chemical criteria *alone* to assess aquatic life uses) is overwhelmingly towards making a type II error in resultant statements of aquatic ecosystem condition. The implications of this to water resource management are significant in at least two ways; 1) potentially major problems will either be overlooked or underestimated by protection and restoration programs, and 2) an inappropriate amount of attention will be devoted to problems of comparative and diminishing consequence. While type I errors were detected in the Ohio EPA chemical/biological criteria comparison, these were much less frequent and were related more to the relative inability of chemical criteria to adequately stratify landscape variability, something which biological criteria



6000
5000
4000
3000
2000
1000
0
STREAM/RIVER MILES
25 States Reported No Miles of Aquatic Life Use Impairment Associated With Habitat Degradation
MT MO WI NM IL CA OR OH LA OK WA WY VE IO VA KY MI NE AL NV WV IN AZ PR MA
REPORTING STATES

based on multimetric indices and a regional reference site approach automatically incorporates.

Indicators Hierarchy

The U.S. EPA Environmental Monitoring and Assessment Program (EMAP; U.S. EPA 1991) categorizes environmental indicators as measuring stresses (stressor indicators), exposure (exposure indicators), and response (response indicators). Stressor indicators include activities and their byproducts which exert an impact to the aquatic biosphere and include such factors as land use changes and discharges of pollutants. Exposure refers to evidence that a component of the natural environment has been subjected to the effect of a substance or activity which has the potential to directly or indirectly change the environment. Response refers to the status of a particular resource component, usually biological, in relation to external stresses and exposure to those stresses. Individual indicators function most appropriately in one of the three indicator classes although some may double as a secondary evaluation of another indicator class. For example, chemicals generally function best as an indicator of exposure, but may also indirectly provide insights to response. Biological measures are inherently response oriented and, depending on the "robustness" of the bioassessment framework, may provide additional insights to exposure and stress.

The previous comparisons of chemical and biological indicator frameworks illustrate a national problem - *the inappropriate use of stressor and exposure indicators as response indicators*. While no single indicator measures everything, some indicators are inherently better at expressing the end-point condition of the aquatic ecosystems which we are striving to restore and maintain via legislation such as the CWA and State WQS. In attempting to resolve the obvious inconsistencies in measuring the condition of our aquatic resources a fundamental first step should be to recognize and establish appropriate roles for chemical, physical, and biological indicators. More accurately portraying the condition of the nation's surface waters also depends on using suites of indicators, each used in their most appropriate role as measuring stress, exposure, or response. While some U.S. EPA programs have accomplished this step, the regulatory programs are conspicuous in having yet to incorporate these concepts.

Biological Criteria: A Relatively Recent Concept for Measuring Condition

Because of the varied issues which States must deal with, indicator frameworks are needed which summarize the interaction of the fundamental driving factors (Figure 1). Biological criteria offer a way to measure the end-result of water resource management efforts and successfully accomplish a reasonable protection of aquatic ecosystems. Ohio EPA adopted biological criteria in its water quality standards regulations in May, 1990. Biological criteria are based on measurable characteristics of aquatic communities such as species richness, key taxonomic groupings, functional feeding guilds, environmental tolerance, and evidence of stress. These criteria are structured in the Ohio WQS regulations within a system of tiered aquatic life uses. Numerical biological criteria were derived using a regional reference site approach which has been extensively described and documented (Ohio EPA 1987a,b; Ohio EPA 1989a; Yoder 1989; Yoder and Rankin 1995a). The resulting numerical expressions essentially reflect the health and well-being of reference aquatic communities and are the end-product of an ecologically complex and structured derivation process.

Understanding Biological Integrity: A Prerequisite to Biological Criteria

Without a sufficient theoretical basis it would be very difficult, if not impossible, to develop biological criteria and meaningful measures of aquatic ecosystem condition. While it was the perception of biological degradation (*i.e.,* fish kills, spills, odors, colors, foams, gross pollution episodes, etc.) which stimulated the landmark environmental legislation of the past two decades, an ecological focus was diminished in the quest for easily measured surrogates (Karr 1991). The term biological integrity originates from the Federal Water Pollution Control Act (FWPCA) amendments of 1972 and has remained a part of the subsequent reauthorizations. Early attempts to define biological integrity in ways that it could be used to measure attainment of the legislative goals were inconclusive. One of the better known of these efforts failed to produce a consensus definition or framework for determining biological integrity, although several contributors urged that a holistic or systems approach be employed (Ballentine and Guarria 1975). Biological integrity was considered relative to: 1) conditions that existed prior to European settlement; 2) the protection and propagation of balanced, indigenous populations; and, 3) ecosystems that are unperturbed by human activities. These criteria (especially 1 and 3) could be construed as referring to a pristine condition which exists in only a few, if any, ecosystems in the conterminous United States. Subsequent to this initial effort a U.S. EPA sponsored work group concluded that biological integrity, when defined as some pristine condition, is difficult if not impractical to precisely define and assess (Gakstatter *et al.* 1981). The pristine vision of biological integrity was considered as a conceptual goal towards which pollution abatement efforts should continue to strive, although current, past, and future uses of surface waters may prevent its full realization in many parts of the U.S. A recent and comprehensive treatment of this issue was recently accomplished by Jackson and Davis (1995) and Polls (1995).

Contemporary efforts to construct a workable, practical definition of biological integrity have provided the supporting theory that necessarily precedes the development of standardized measurement techniques and criteria for determining compliance with that goal. Biological integrity is defined herein as "....the ability of an aquatic ecosystem to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitats of a region" (Karr and Dudley 1981). This is a workable definition which directly alludes to measurable characteristics of biological community structure and function found in the least impacted habitats of a region. It is this definition and the underlying ecological theory which provides the fundamental basis for the development of numerical biological criteria using a regional reference site approach. U.S. EPA adopted a facsimile of this definition in their national biological criteria program guidance (U.S. EPA 1990).

The recently emerging issue of biodiversity should not automatically be equated with biological integrity even though the two concepts share many common attributes and goals (Karr 1991). There seems to be a dichotomy between each in that efforts to protect and restore biodiversity are primarily focused on ecosystem "elements" (*i.e.,* genetic diversity, populations, bioreserves, etc.). While biological integrity certainly includes these elements it additionally encompasses ecosystem "processes" (*i.e.,* nutrient cycles, trophic interactions, speciation, etc.). The eventual attainment of the CWA goal of biological integrity is much more than merely achieving a high level of species diversity, numbers, and/or biomass. In fact there are situations when increases in any one of these parameters can be a sign of degradation.

The oft cited ecosystem approach to environmental management (*e.g.*, Great Lakes Water Quality Initiative) can even be more restricted to dealing with specific elements, some of which are not direct ecological parameters (*i.e.*, toxicological surrogates), as compared to a broader ecological approach which relies on direct measures of biological integrity. The current debates about biodiversity (*e.g.*, Endangered Species Act) and ecosystem approaches (*e.g.*, Great Lakes Initiative) urgently need to be expanded to include the concepts of biological integrity in order to improve the chances for the acceptance and overall success of each effort and to assure that environmental problems are addressed from an ecological perspective. Conservation policy should promote management practices which maintain integrity, prevent endangerment, and enhance the recovery of species *and* ecosystems (Angermier and Williams 1993). Achieving this state of aquatic ecosystem health will bring along the other goals as well since functionally healthy communities include the elements of biodiversity and rare species inherent to the more narrowly focused management efforts. We believe that biocriteria can play an important role in meeting these challenges.

New Generation Biological Community Evaluation Mechanisms

New methods and approaches to assessing aquatic community data have been developed over the past 10-15 years which have provided a significant advancement in being able to utilize biological community information for resource characterizations and as arbiters of environmental goal attainment/non-attainment. These include such innovations as the Index of Biotic Integrity (IBI) , as originally developed by Karr (1981), and as subsequently modified by many others (Leonard and Orth 1986; Steedman 1988; Ohio EPA 1987b; Miller *et al.* 1988; Simon 1991; Lyons 1992), the Index of Well-Being (Iwb) developed by Gammon (1976), the Invertebrate Community Index (ICI; Ohio EPA 1987b; DeShon 1995), the U.S. EPA Rapid Bioassessment Protocols (Plafkin *et al.* 1989), and most recently the Benthic IBI for macroinvertebrate assemblages (BIBI; Kerans and

Karr 1993). These represent what are termed here as new generation community evaluation mechanisms since they represent recent and substantial progress in the seemingly continuous effort to develop improved community indices. While the pursuit of a "better" index has been criticized and even trivialized, it is nevertheless an important process not only as an effort to make ambient aquatic community data more useable, but a more integral part of the overall water resource management process.

The Multimetric Approach

While numerical biological indices have been criticized for potentially oversimplifying complex ecological processes (Suter 1993; Polls 1995), the need to distill such information to commonly comprehended expressions is both practical and necessary. The advent of the new generation evaluation mechanisms have filled important practical and theoretical gaps not always fulfilled by previously available single dimension indices (*e.g.,* diversity indices). Multimetric evaluation mechanisms such as the IBI extract ecologically relevant information from complex biological community data while preserving the opportunity to analyze such data on a multivariate basis. The problem of biological data variability is also addressed within this system. Variability is controlled by specifying standardized methods and procedures (*e.g.,* Ohio EPA 1989b), compressed through the application of multimetric evaluation mechanisms (*e.g.,* IBI, ICI), and stratified by accounting for regional and physical variability and potential (*e.g.,* ecoregions, tiered aquatic life uses) The result are evaluation mechanisms such as the IBI and ICI that have acceptably low, replicate variability (Davis and Lubin 1989; Rankin and Yoder 1990; Stevens and Szczytko 1990; Yoder and Rankin 1995a).

Multimetric indices have been criticized as representing a loss of rich information in the reduction of the data to a single index value (*e.g.,* Suter 1993). However, this presumes that the supporting data is never viewed, analyzed, or examined beyond the calculation of an index value. The need to examine sub-components of the indices and even the raw data is implicit throughout the biocriteria process thus these criticisms are yet without foundation. The need for numerical indices is clearly evident throughout the environmental management process, provided that each emanates from a sound theoretical basis and robust information base. While the need for interpretation by qualified biologist will always be necessary, it is unrealistic to expect that expert judgement alone will be acceptable as a substitute for a more empirical process. Fortunately, biological judgement can be incorporated into structured frameworks such as that developed by the State of Maine using multivariate techniques (Davies *et al.* 1993), the U.S. EPA Rapid Bioassessment Protocols (Plafkin *et al.* 1989), and the regional reference site approach (Hughes *et al.* 1986; Ohio EPA 1987b, 1989a; Yoder and Rankin 1995a).

Some of the early attempts to satisfy the demand for numerical assessment tools produced indices of only limited or even questionable theoretical rigor. However, the theoretical underpinnings behind the IBI type indices are much more robust especially when used within a framework where regional considerations, reference condition, and background variability have been adequately incorporated. Steedman (1988) described the IBI as being based on simple, definable ecological relationships which is quantitative as an ordinal, if not linear measure, and which responds in an intuitively correct manner to known environmental gradients. Further, when the IBI is incorporated with mapping, monitoring, and modeling information it should be invaluable in determining management and restoration requirements for warmwater streams. As an aggregation of community information the IBI (and facsimiles thereof) provide a way to organize complex data and reduce it to a scale which is interpretable against communities of a known (reference) condition. There is no complex transformation of data accomplished, just an improved stratification and organization of complex ecological information. Simply stated, multi-metric indices can satisfy the demand for a straightforward numerical evaluation that expresses a relative value of aquatic community health and well-being which allows program managers (who are frequently non-biologists) to, in effect, “visualize” relative levels of biological integrity. These measures also provide a means to establish numerical biological criteria.

Karr’s Index of Biotic Integrity (IBI) Modified by Ohio EPA

The IBI originally proposed by Karr (1981) and later refined by Fausch *et al.* (1984) and Karr *et al.* (1986) incorporates 12 aggregations of community information termed “metrics”. These fall within three broad categorical groupings: species richness and composition, trophic composition, and fish abundance and condition. Some metrics respond positively (*i.e.,* their raw value increases) to increasing environmental quality and are termed positive metrics. Other metrics respond positively to increasing degradation (*i.e.,* their raw value decreases) and are termed negative metrics. Some metrics respond across the entire range of environmental quality whereas others respond more strongly to a portion of that range (Karr *et al.* 1986). While no single metric can consistently function across all types of impacts, the aggregation of metrics combined in the IBI provides sufficient redundancy to provide a consistent and sensitive measure of biological integrity (Angermier and Karr 1986). The IBI relies on multiple parameters, an essential attribute when the system being evaluated is complex (Karr *et al.* 1986). While the IBI incorporates elements of professional judgement, it also provides the basis for establishing quantitative criteria for determining what constitutes exceptional, good, fair, poor, and very poor condition.

An important example of using biological judgement in the biological criteria process is in the tailoring of the IBI to local/regional conditions. Several examples of this exist across the U.S. most of which are summarized by Miller *et al.* (1988). In this process Ohio EPA derived three different modifications of the original IBI (Table 1) which resulted from an extensive process of testing more than 60 candidate metrics. A wide variety of stream and river sizes occur throughout Ohio and these not only contain different fish assemblages, but obtaining a sample requires the use of different methods. Thus it was necessary to modify Karr's original IBI for application to these different stream sizes and make additional adjustments to account for bias induced by different sampling gear. All modifications were made in keeping with the guidance provided by Karr *et al.* (1986). Three different modified IBIs were developed for Ohio rivers and streams (Table 1); 1) a headwaters IBI for application to headwater streams (defined as stream locations with a drainage area <20 mi.^{2}; 2) a wading site IBI for application to stream locations >20 mi.^{2} sampled with wading methods; and, 3) a boat site IBI for locations sampled with boat methods. These divisions were made based on inherent differences in faunal associations (*e.g.,* headwaters vs. wading sites) and sampling gear bias considerations (*e.g.,* wading vs. boat sites). In all of these modifications the basic ecological structure and content of Karr's original IBI was preserved.

Table 1. Modification of Index of Biotic Integrity (IBI) metrics used by the Ohio EPA to evaluate wading sites, headwaters sites, and boat sites. The original IBI metrics of Karr (1981) are given first with substitute metrics following.

IBI Metric	Headwaters Sites[1,2]	Wading Sites[2]	Boat Sites[3]
1. Number of Native Species[4]	X	X	X
2. Number of Darter Species		X	
Number of Darter and Sculpin Species	X		
%Round-bodied Suckers[5]			X
3. Number of Sunfish Species[6]		X	X
Number of Headwater Species[7]	X		
4. Number of Sucker Species		X	X
Number of Minnow Species	X		
5. Number of Intolerant Species		X	X
Number of Sensitive Species[8]	X		
6. %Green Sunfish			
%Tolerant Species	X	X	X
7. %Omnivores	X	X	X
8. %Insectivorous Cyprinids			
%Insectivores	X	X	X
9. %Top Carnivores		X	X
%Pioneering Species [9]	X		
10. Number of Individuals			
Number of Individuals (less tolerants)[10]	X	X	X
11. %Hybrids			
%Simple Lithophils		X	X
Number of Simple Lithophils	X		
12. %Diseased Individuals			
%DELT Anomalies[10]	X	X	X

[1] applies to sites with drainage areas <20 mi^2; [2] sampled with wading electrofishing methods; [3] sampled with boat electrofishing methods; [4] excludes all exotic and introduced species; [5] includes all species of the genera *Moxostoma*, *Hypentelium*, *Minytrema*, and *Ericymba*, and excludes *Catostomus commersoni*; [6] includes only Lepomis species; [7] species designated as permanent residents of headwaters streams; [8] includes species designated as intolerant and moderately intolerant (Ohio EPA 1987b); [9] species designated as frequent and predominant inhabitants of temporal habitats in headwaters streams; [10] excludes all species designated as tolerant, hybrids, and non-native species; [10] includes only individuals with deformities, eroded fins or barbels, lesions, and tumors.

While the IBI has worked well in Ohio and many parts of the U.S. (Miller *et al.* 1988), Canada (Steedman 1988), and Europe (Oberdorff and Hughes 1992), there are situations in which problems have been encountered. Two of these include applications to western U.S. drainages (Bramblett and Fausch 1991) and cold water streams in the northern tier of States (Lyons 1992). In both cases the make-up of the fauna is somewhat counter to some of the initial presumptions of the IBI. Bramblett and Fausch (1991) encountered difficulties due to an inherent lack of intolerant species in the harsh and highly variable hydrological conditions of the Arkansas River in Colorado. Lyons (1992) found that degradation of cold water streams in Wisconsin results in an increased diversity of fish species (due to the invasion of warmwater species) which is counted as a positive occurrence under the basic presumptions of the original IBI. This is not to say that solutions to these problems do not exist, but rather points up the need to consider the inherent characteristics of the regional fauna to react to man-induced environmental changes in developing IBI type applications and to account for these characteristics in determining which beneficial uses (*e.g.*, a choice between cold water or warmwater?) will be managed for ahead of time. This latter consideration also implies that a tiered system of aquatic life use designation is needed.

Initial Decisions and Other Considerations

There are a number of fundamental decisions which need to be made prior to adopting a set of biological indicators and attendant monitoring methods. This is a critical juncture in the process since these initial decisions will determine program effectiveness well into the future. Decisions about which sampling methods and gear to use, seasonal considerations, which organism groups to monitor, which parameters to measure, which level of taxonomy to use, etc. all need to be made up front in the process. The following axiom should apply "....when in doubt choose to take more measurements than seem necessary at the time since information not collected is impossible to retrieve at a later date". However, this does not apply equally to all factors. For example, seasonality is a well developed concept thus it may not be necessary to sample in multiple seasons for the sake of data redundancy. However, parameters which require little or no extra effort to acquire should be included until enough evidence is amassed to prove or disprove its relative worth. One example in Ohio is with external anomalies on fish. We decided to record this information even though it was not apparent what its later usefulness would be. This parameter has proven over time to be one of our most valuable assessment tools. For macroinvertebrates the decision to identify midges to the genus/species level also proved far sighted given the value of this group in diagnosing impairments. Of course, samples could be archived for later processing, but the logistical burdens that this would entail later are equally undesirable.

Another important consideration is assuring that qualified and regionally experienced staff are available to implement the monitoring and assessment activities. Ecological assessment is not unlike other professions in which the most skilled and experienced personnel are sought to direct, manage, and supervise. However, biological field assessment requires an equivalent level of expertise in the field since many of the critical pieces of information are recorded and, to a degree, interpreted in the field. There is simply no substitute for the intangibles gained by direct experience in the field. This is not a job to be left to technicians. In addition, it is only prudent that 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 assessments (*e.g.*, reporting). Such staff, particularly those with more experience, also contribute to policy development.

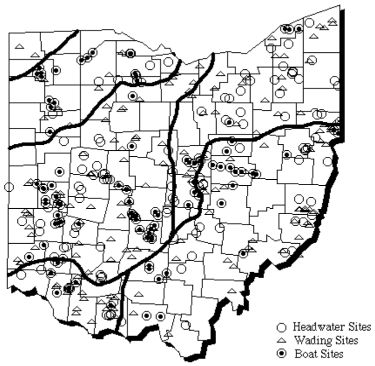
Derivation and Application of Numerical Biological Criteria

Describing a framework for the derivation of numerical biological criteria has been done thus far in only a very few places (Davies *et al.* 1993; U.S. EPA 1991b; Yoder and Rankin 1995a). However, this is the most critical part of the entire biological criteria process since the results are the actual numeric expressions of the reference aquatic community performance which will be used to arbitrate CWA aquatic life goal attainment/non-attainment. A standard framework from within which regional and site-specific biological criteria can be derived needs to be implemented on a national scale. Ideally, this framework should be adaptable to all aquatic ecosystem types provided that relevant ecological indicators and concepts are incorporated by experienced practitioners who are familiar with regional-specific and ecosystem-specific peculiarities. Nationally uniform biological criteria, unlike national water quality criteria for specific chemicals, are neither feasible nor desirable as it is widely recognized that ecological resources vary considerably across the nation. However, a nationally uniform or consistent framework is possible and it must include different tiers of criteria complexity. The key is to “capture” this inherent variability within a common national goal such as the attainment and restoration of biological integrity. Once this is understood and accepted, a nationally uniform framework is then possible.

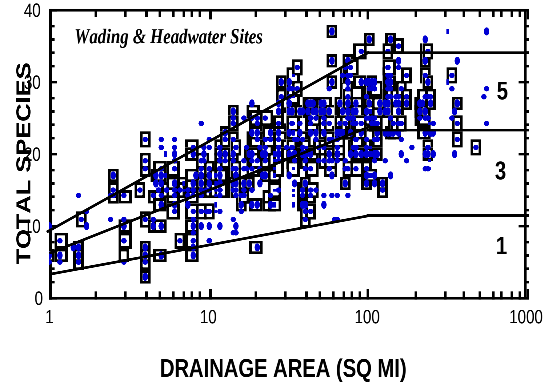
Framework for Deriving Numerical Biological Criteria: Ohio Case Example

The derivation of biological criteria for Ohio surface waters is essentially based on the biological community performance which occurs at 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 such a framework. The numerical biological criteria which result from the application of this framework represent the aquatic community performance that can reasonably be attained given present-day background conditions. Although these criteria are not an attempt to define “pristine”, pre-Columbian conditions, the design framework includes the flexibility to make future changes to the criteria which would take place when changes in background conditions are detected. Thus, if pristine conditions truly returned this would eventually be reflected by future, periodic adjustments to the multi-metric indices, their calibration, and 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 350 reference sites that typify the "least impacted" condition within each ecoregion (Figure 3; Ohio EPA 1987b; 1989a). This information was used within an existing framework of tiered aquatic life uses in the Ohio Water Quality Standards to establish attainable, baseline biological community performance expectations on a regional basis. Biological criteria vary by ecoregion, aquatic life use designation, site type, and biological index. The resulting criteria for two of the “fishable, swimmable” uses, Warmwater Habitat (WWH) and



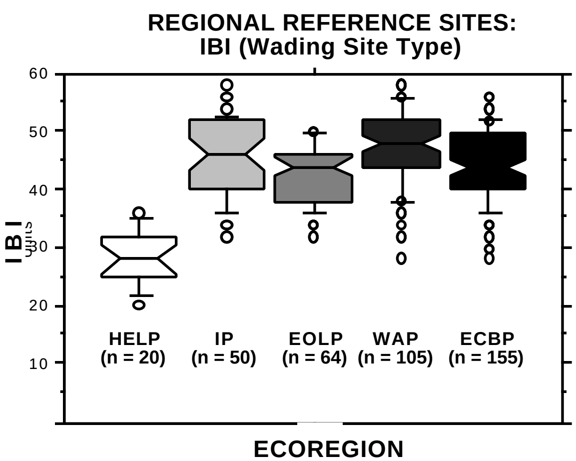
I. Select & sample reference sites



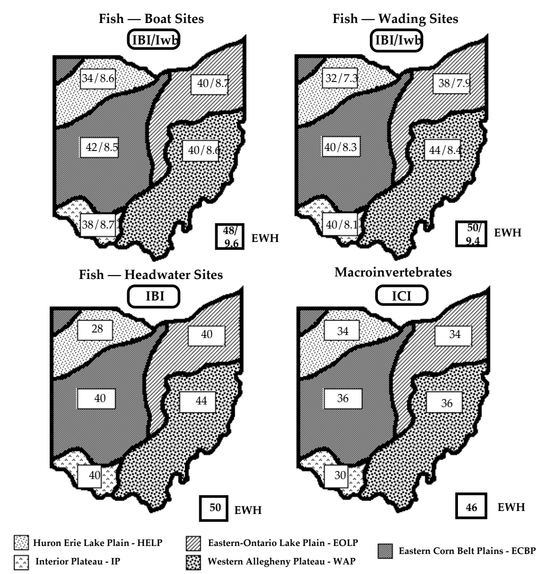
II. Calibration of IBI metrics

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

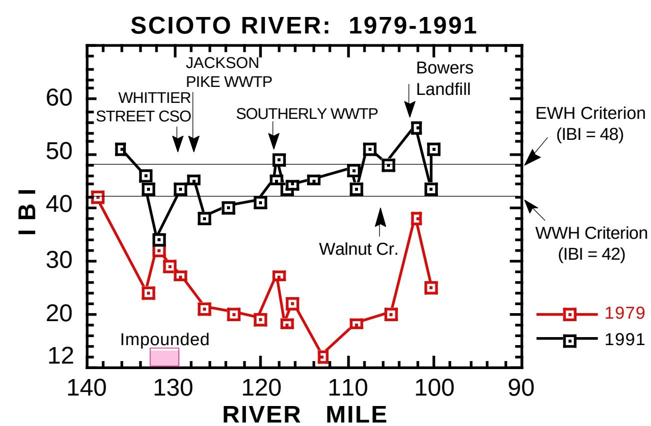
III. Calibrated IBI modified for Ohio waters



IV. Establish ecoregional patterns/expectations



V. Derive numeric bio-criteria/codify in WQS



VI. Numeric biocriteria used in assessments

Exceptional Warmwater Habitat (EWH) are shown in Figure 4.

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 multi-metric indices (*e.g.,* IBI, ICI);
- set numeric biocriteria based on attributes specified by tiered aquatic life use designations;
- reference site re-sampling (10% of sites sampled each year); and,
- make periodic (*i.e.,* once per 10 years) adjustments to the multi-metric indices, numeric biocriteria, or both as determined by reference site re-sampling results.

The major steps in the biological criteria calibration, derivation, and application process are summarized in Figure 3 This process also presumes that narrative statements about the desired biological community condition already exist in the State WQS and that some type of broad-scale regionalization(*e.g.,* ecoregions) has also been accomplished. The former is a critical part of the framework as this specifies the ecosystem goals which management programs are striving attain. The latter step is particularly important as it is necessary to stratify regional landscape variability within a manageable framework. This is best accomplished when a landscape partitioning framework such as ecoregions (Omernik 1987) and sub-components are used as an initial step in accounting for natural landscape variability. It is because of landscape variability that uniform and overly simplified approaches to water resource management will fail to produce desired results (Omernik and Griffith 1991). We used Omernik's (1987) ecoregions to accomplish this task, but other regionalization methods may work equally well in other areas. The following Ohio case example portrays the calibration of the IBI modification for wading sites. A similar stepwise procedure was used to calibrate the Invertebrate Community Index for macroinvertebrates (Ohio EPA 1987b; DeShon 1995) Once reference sites are selected and sampled (step 1 in Figure 3) the biological data is first used to calibrate the modified IBI (step 2) and ICI. For fish three different

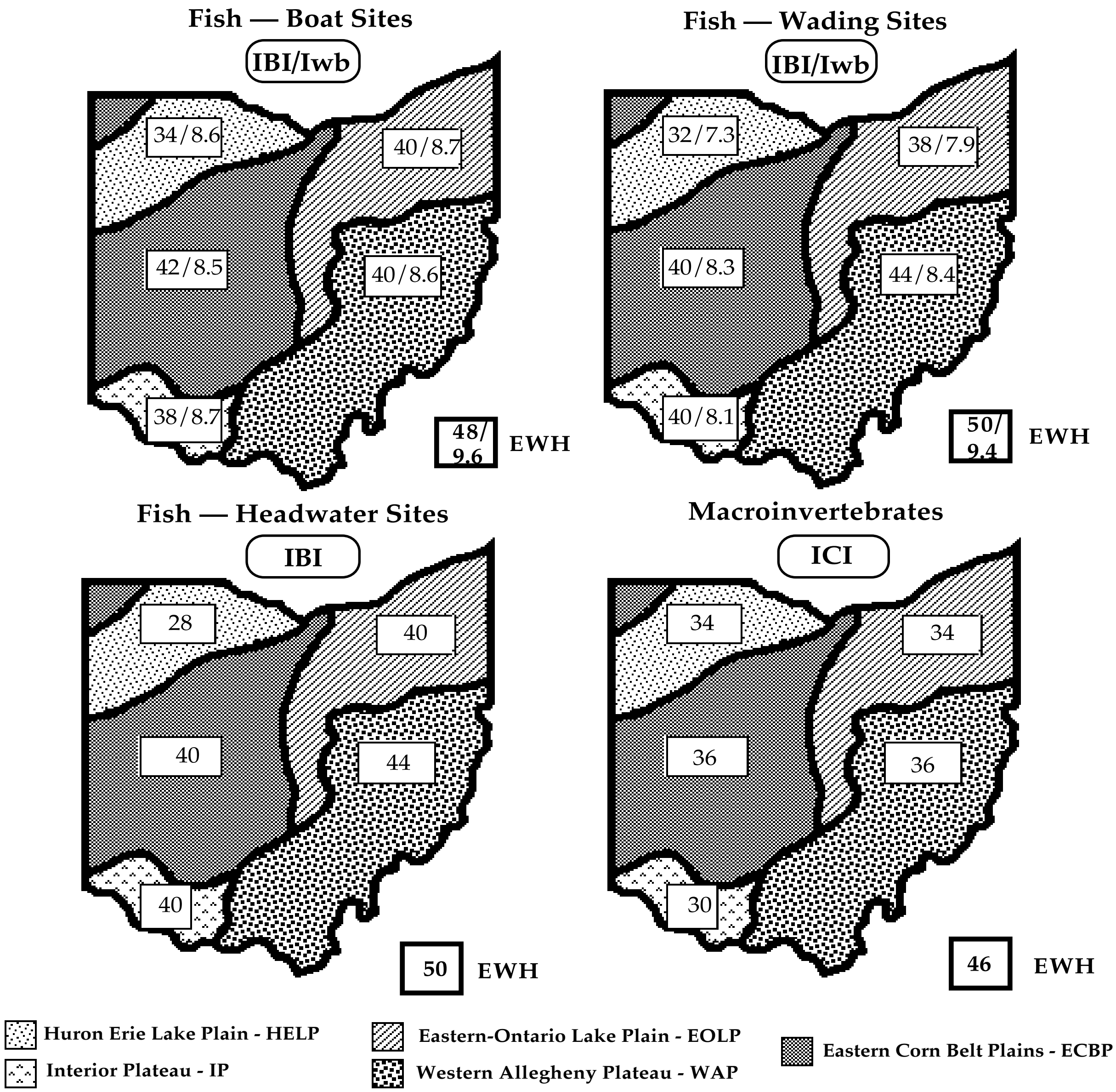


Figure 4. Biological criteria in the Ohio WQS for the Warmwater Habitat Use (WWH) and Exceptional Warmwater Habitat (EWH) use designations arranged by biological index, site type for fish, and ecoregion. The EWH criteria for each index and site type is located in the boxes located outside of each map.

IBIs are constructed, one each for headwaters, wading (step 3), and boat sites. The reference site results for the IBI are then used to establish numerical biological criteria (steps 4 and 5). A notched box-and-whisker plot method was used to portray the results for each biological index 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 are attempting 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).

Applications of Numerical Biological Criteria: Determining Use Attainment Status

Procedures for determining the use attainment status of Ohio's lotic surface waters were also developed (Ohio EPA 1987b; Yoder 1991b). Using the numerical biocriteria as defined by the Ohio WQS, use attainment status is determined as follows:

1) FULL - aquatic life use attainment is considered full if all of the applicable numeric indices exhibit attainment of the respective biological criteria; this means that the aquatic life goals of the Ohio Water Quality Standards are being attained.

2) PARTIAL - at least one organism group exhibits non-attainment of the numeric biocriteria, but no lower than a narrative rating of “fair”, and the other group exhibits attainment.

3) NON - neither organism group exhibits attainment of the ecoregional biocriteria, or one organism group reflects a narrative rating of “poor” or “very poor”, even if the other group exhibits attainment.

Following these rules a use attainment table is constructed on a longitudinal mainstem or watershed basis. An example of how a use attainment table is constructed is found in Tables 2 and 3.

To demonstrate the implementation of the preceding guidelines the attainment tables from the 1987 biological survey of the Kokosing River mainstem and the 1982 and 1990 surveys of the Hocking River, both located in central Ohio, are used here as examples (Tables 3 and 4). Each river represents different extremes in potential aquatic community performance and aquatic life use attainment status. A use attainment table includes the sampling locations for both fish and macroinvertebrates by river mile (upstream to downstream), the IBI, MIwb, and ICI scores, the Qualitative Habitat Evaluation Index (QHEI; Rankin 1989, 1995) score, the use attainment status (*i.e.,* full, partial, or non), and any pertinent comments with regard to the proximity of the site to pollution sources and other influences. This table is the official designation of use attainment status using the numerical biological criteria and forms the basis for all other uses including reporting (*e.g.,* basin reports, 305b report) and assessment (*e.g.*, Water Quality Based Effluent Limit [WQBEL] reports).

The Kokosing River example demonstrates longitudinal changes in use designation, different site types, and a change in ecoregion (Table 2). As a result of the 1987 survey, the original Warmwater Habitat (WWH) use designation was revised to Exceptional Warmwater Habitat (EWH) for a portion of the mainstem (designated as recommended segments in Table 2). In this situation it was the demonstrated attainment of the EWH biological criteria for both organism groups in the recommended segments which prompted the change. The original WWH designation was originally made in 1985 without the benefit of site-specific biological data. In addition to the longitudinal changes in use designation, there are also changes in site type, with the transition from wading sites in the upper mainstem to boat sites in the lower mainstem. The decision about where a change in ecoregion takes place involves more than merely following the boundaries on the ecoregion maps. Ecoregion boundaries are more transitional than discrete and defining where a change takes place is important for cross-boundary rivers such as the Kokosing. This involves examining the base maps of surficial geology (*i.e.*, glacial geology in Ohio), soils, climax vegetation potential, and land use. Since rivers tend to export the characteristics of the parent ecoregion into the next this phenomenon is taken into account as well. The areas defined as most typical and generally typical by Omernik and Gallant (1988) in the receiving ecoregion are

Table 2. Aquatic life use attainment status for the Warmwater Habitat (WWH) and recommended Exceptional Warmwater Habitat (EWH) use designations in the Kokosing River mainstem based on data collected during July - September 1987.

RIVER MILE Fish/Invert.	IBI	Modified Iwb	ICI	QHEI[a]	Attainment Status[b]	Comment
Kokosing River (1987)						
Erie/Ontario Lake Plain - WWH Use Designation (Existing)						
49.8[c]/49.8	56	N/A	38	85	FULL	
- /48.4	-	-	46	-	(FULL)	
46.4[d]/46.3	50	8.4	42	87.5	FULL	
45.3[d]/45.2	50	7.7[ns]	44	36.5	FULL	
40.6[d]/40.5	53	8.8	48	68.5	FULL	
36.6[d]/35.0	46	7.8[ns]	48	65	FULL	
30.7[d]/30.6	43	7.9	48	72	FULL	
Erie/Ontario Lake Plain - EWH Use Designation (Recommended)						
28.7[e]/28.6	50	9.6	48	79	FULL	Reg. Ref. Site; dst. N. Br.
25.5[e]/25.2	51	9.7	46	78	FULL	Reg. Reference Site
24.0[e]/24.2	50	9.6	46	91.5	FULL	Dst. Mt. Vernon WWTP
23.0[e]/22.9	48	10.1	48	90.5	FULL	
20.9[e]/18.0	53	9.8	46	76	FULL	Reg. Reference Site
16.1[e]/16.2	47[ns]	9.8	46	80	FULL	Dst. Gambier WWTP
11.7[e]/11.6	48	9.5[ns]	54	97.5	FULL	Reg. Reference Site
8.8[e]/8.7	51	9.9	38*	93.5	PARTIAL	
W. Allegheny Plateau - EWH Use Designation (Recommended)						
6.3[e]/6.2	54	10.0	52	92.5	FULL	
0.5[e]/1.5	46[ns]	10.1	48	87	FULL	Reg. Reference Site

* - significant departure from ecoregional biocriteria; poor and very poor results are underlined.
ns - nonsignificant departure from ecoregional biocriteria for WWH or EWH (4 IBI or ICI units; 0.5 Iwb units).
a - Qualitative Habitat Evaluation Index (QHEI) values based on the new version (Rankin 1989).
b - Attainment status based on one organism group is parenthetically expressed.
c - Headwaters site type; d - Wading site type; e - Boat site type.

Ecoregion Biocriteria: Erie/Ontario Lake Plain (EOLP)

INDEX - Site Type	WWH	EWH	MWH[f]
IBI - Headwaters/Wading	40	50	24
IBI - Boat	40	48	24
Mod. Iwb - Wading	7.9	9.4	5.8
Mod. Iwb - Boat	8.7	9.6	5.8
ICI	34	46	22

Ecoregion Biocriteria: W. Allegheny Plateau (WAP)

INDEX - Site Type	WWH	EWH	MWH[f]
IBI - Boat	40	48	24
Mod. Iwb - Boat	8.6	9.6	5.5
ICI	36	46	22

f - Modified Warmwater Habitat for non-acidic mine runoff impacted areas.

also used to assist in making this determination. In some cases the site-specific habitat attributes are used to help separate where the transition from one ecoregion to the next takes place.

The Hocking River (Table 3) presents a stark contrast in attainment status compared to the Kokosing (Table 2). The extensive non-attainment observed in 1982 was due to a combination of factors related to point source discharges, combined sewers, urbanization, and habitat impacts. Improvements made primarily in municipal wastewater treatment and industrial pretreatment were revealed in the increased partial and full attainment observed in 1990 (Table 3). This example additionally demonstrates the use of the biological criteria to serve as a feedback tool for determining the effectiveness of pollution control programs.

Interpreting Results on a Longitudinal Reach or Subbasin Basis

A longitudinal analysis of biological sampling results is also performed in an attempt to visually interpret and describe the magnitude and severity of departures from the numerical biological criteria. This is done by plotting the biological index results (IBI, MIwb, or ICI) by river mile for the subject survey area. Major sources of potential impact and the applicable numerical biological criteria are indicated on each graph. These graphs are also a standard reporting feature in basin specific reports. The results of fish and macroinvertebrate community sampling in the Scioto River during 1980 and 1991 are used as an example (Figure 5). One of the best proven uses for this type of analysis is demonstrating changes through time (*i.e.*, trends). Unlike chemical parameters, biological indices integrate chemical, biological, and physical impacts to aquatic systems and portray use attainment/non-attainment in aggregate and direct terms. Frequency and duration considerations, which are difficult to adequately account for with most chemical monitoring approaches, are integrated by the resident aquatic life in the receiving water body. Figure 5 represents the results of bioassessment in a 40 mile river segment which has been sampled repeatedly over multiple years. The obvious improvements exhibited by both the IBI and ICI illustrate the benefits of improved municipal wastewater treatment in the Columbus, Ohio area. While this is also communicated by a use attainment table (see Tables 2 and 3) the extent and magnitude of the incremental improvements along each index axis can only be demonstrated

Table 3. Aquatic life use attainment status for the Warmwater Habitat (WWH) use designation in the upper Hocking River mainstem based on data collected during July - October 1982.

RIVER MILE Fish/Invert.	IBI	Modified Iwb	ICI	QHEI[a]	WWH Attain-ment Status[b]	Comment
Hocking River (1982)						
	Erie/Ontario Lake Plain - WWH Use Designation (Existing)					
95.2/ -	27*	6.1*	-	46	(**NON**)	Old channelization
93.2/ -	23*	5.5*	-	-	(**NON**)	Channelization
92.0/92.0	17*	4.5*	44	48	**NON**	Urban development
- /91.2	-	-	8*	-	(**NON**)	Raw sewage evident
90.7/89.3	17*	4.0*	2*	40	**NON**	dst. Wheeling lift sta.
88.8/88.5	12*	0.6*	0*	48	**NON**	dst. Lancaster WWTP
- /87.3	-	-	0*		(**NON**)	
85.7/85.4	12*	1.8*	0*	62	**NON**	
	W. Allegheny Plateau - WWH Use Designation (Existing					
83.1/82.9	20*	4.0*	0*	67	**NON**	ust. Rush Cr./Sugar Gr.
- /81.8	-	-	0*		(**NON**)	ust. Rush Cr./Sugar Gr.
81.4/81.3	17*	2.4*	12*	84	**NON**	dst. Sugar Gr, Rush Cr.
77.2/77.3	29*	6.8*	22*	63	**NON**	dst. Clear Cr.
73.3/73.5	31*	7.3*	18*	66	**NON**	at Enterprise
Hocking River (1990)						
	Erie/Ontario Lake Plain - WWH Use Designation (Existing)					
95.2/95.1	35[ns]	8.2	50	66.5	FULL	Background
92.2/91.9	27*	6.5*	52	44	**NON**	Channelization
90.8/90.7	28*	6.9*	46	37	PARTIAL	dst. CSOs, urban
89.4/89.4	24*	5.9*	38	42	**NON**	dst. Wheeling Lift Sta.
89.1/89.1[c]	25	4.7	26	49.5	N/A	Lancaster WWTP Mix.
89.0/88.9	30*	6.5*	32[ns]	38	PARTIAL	dst. Lancaster WWTP
87.1/87.2	25*	4.9*	MG[d]	59	**NON**	
	W. Allegheny Plateau - WWH Use Designation (Existing					
82.0/82.9	33*	6.5*	46	62	PARTIAL	
81.2/81.3	39[ns]	8.1[ns]	44	57.5	FULL	dst. Rush Creek
77.2/77.1	34*	7.5*	G	63.5	PARTIAL	dst. Clear Cr.
73.2/73.4	43	8.1[ns]	G	63.5	FULL	at Enterprise

* - significant departure from ecoregional biocriteria; poor and very poor results are underlined.
ns - Nonsignificant departure from ecoregional biocriterion (4 IBI or ICI units; 0.5 Iwb units).
a - all Qualitative Habitat Evaluation Index (QHEI) values are based on the most recent version (Rankin 1989).
b - use attainment status based on one organism group is parenthetically expressed.
c - biocriteria do not apply in mixing zones.
d - narrative rating used in lieu of ICI (G = good; MG = marginally good).

Table 3. (continued)

Ecoregion Biocriteria: Erie/Ontario Lake Plain (EOLP; RM 84.1 - 93.2)

INDEX - Site Type	WWH	EWH	MWH[e]
IBI - Wading (RM96.2-93.2)	40	50	24
IBI - Boat (RM92.0-84.1)	40	48	24
Mod. Iwb - Wading	7.9	9.4	6.2
Mod. Iwb - Boat	8.6	9.6	5.5
ICI (RM92.0-85.4)	36	48	30

Ecoregion Biocriteria: Western Allegheny Plateau (WAP; RM 73.3 - 83.1)

INDEX - Site Type	WWH	EWH	MWH[e]
IBI - Boat (RM82.4-74.3)	40	48	24
Mod. Iwb - Boat	8.6	9.6	5.5
ICI (RM82.9-73.5)	36	48	30

e - Modified Warmwater Habitat for channelized habitats.

graphically. While the biological improvement is correlated with the overall reduction of loadings of conventional pollutants at both of the major WWTPs which serve the Columbus metropolitan area, non-attainment remains in some reaches, particularly those in close proximity to combined sewer overflows and habitat impacts.

Statewide Reporting and Assessment Applications

Biological data and biological criteria are the principal arbiter of aquatic life use attainment status for the biennial Ohio Water Resource Inventory (CWA section 305b report), the principal purpose of which is to report on the status and direction of the State's waters. Perhaps the most frequently asked question that we encounter follows: "Is water quality improving or worsening?" The General Accounting Office (U.S. GAO 1986) criticized U.S. EPA for not being able to quantify improvements in water quality for the billions of dollars spent to improve WWTP effluents through the construction grants program. The failure to provide support, both programmatically and monetarily, for adequate State ambient monitoring programs has resulted in a lack of consistent and useable information to determine national trends. Nichols (1992) indicates that bioassessment information which documents changes over time could be found for only three

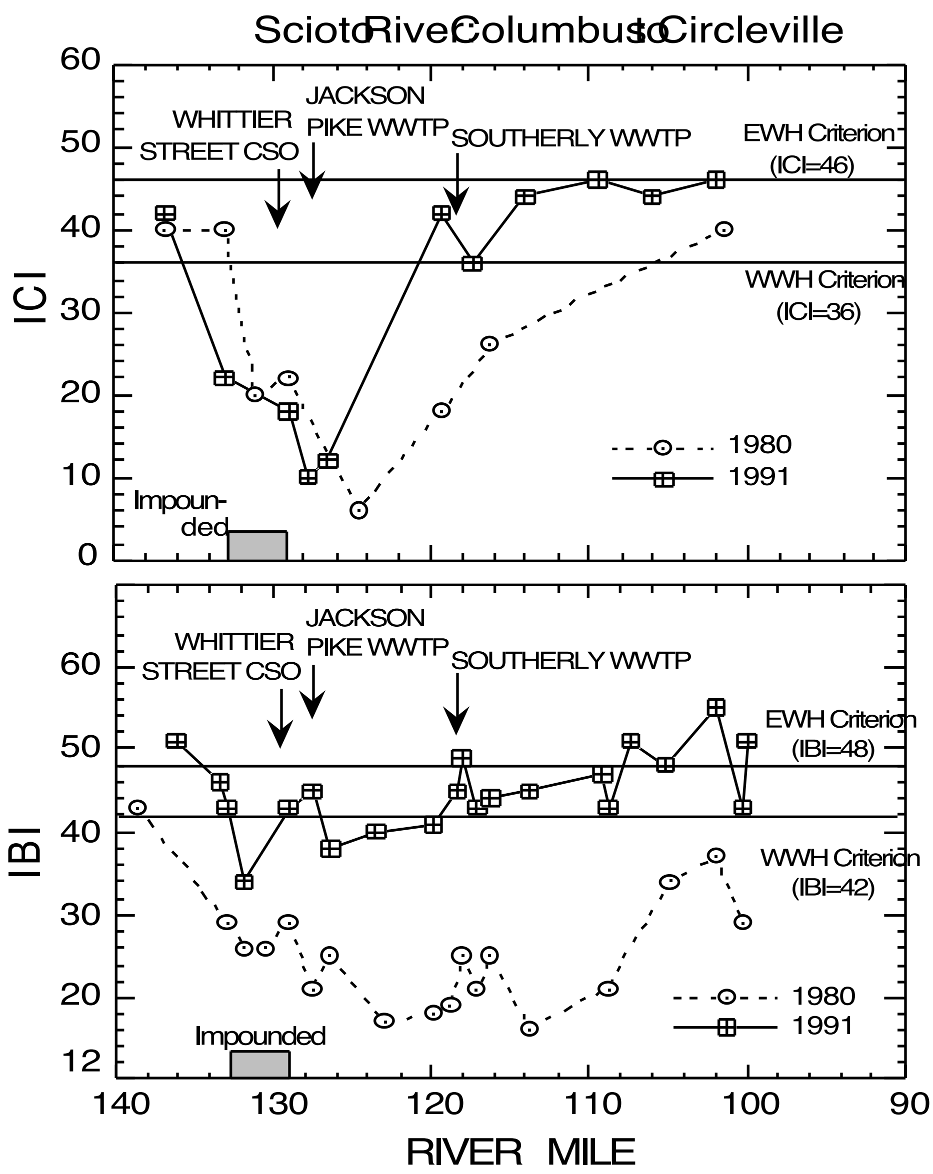


Figure 5. Longitudinal profile of the Index of Biotic Integrity (IBI; lower) and the Invertebrate Community Index (ICI; upper) for the Scioto River between Columbus and Circleville, Ohio based on electrofishing samples collected during July-October 1980 and 1991.

water systems in the U.S. While this is likely a very conservative estimate of the national data base, it nevertheless indicates the chronic problem of a lack of consistent data over wide areas.

In Ohio we have attempted to develop a long-term data base which will meet several of these needs. One analysis of the statewide database performed as part of the 1992 Ohio Water Resource Inventory analyzed changes at paired sites within the Ohio EPA statewide database (Figure 6; Table 4). The Ohio EPA database was not collected under a statistically random design for the location of sampling sites and while the aggregate design is spatially biased, the large number of sites sampled (>4500 locations) and thorough coverage of the streams and rivers with drainage areas greater than 100 $mi.^2$ (70% coverage statewide) makes valid statewide comparisons possible. In this analysis, for sites with more than two years of data, trends represent the difference between the earliest and latest results, most of which are approximately 10 years apart. Sample sizes are 443 sites, 404 sites, and 268 sites for the IBI, MIwb, and ICI, respectively. Table 4 summarizes pertinent percentile shifts in the biological indices between the earliest and latest time periods and the results of a paired t test (using a t statistic and Wilcoxon's Z test) between these periods as well. The comparison of the two different time periods showed that the increased index scores for the later period were highly significant ($p < 0.0001$). For each index there has been a significant positive change over time at most sites (Figure 6). Although substantial improvements in aquatic life have been observed, it is clear that a significant proportion of Ohio's rivers and streams remain chemically polluted and/or physically degraded to a detectable degree. The macroinvertebrate ICI showed both the largest magnitude of increase and shift in the frequency of sites entering into the good and exceptional performance ranges (*i.e.,* ICI scores equivalent to the WWH and EWH use designations), and sites exiting the poor and very poor ranges (ICI scores <14). In contrast the fish community IBI had the greatest number of sites exiting the poor and very poor ranges, but the fewest sites entering the good and exceptional ranges (*i.e.,* IBI scores equivalent to the WWH and EWH use designations, respectively). Although the improvements which have been documented in many Ohio rivers and streams are considerable, the applicable biological criteria have not yet been completely attained throughout the length of any single water body. The predominant pattern which we have observed is for the macroinvertebrate community (as measured by the ICI) to

Table 4. Summary of paired sites with at least two years of biological data from Ohio streams and rivers sampled before and after 1988. Data pairs represent earliest and latest index values for the ICI, MIwb, or IBI at each paired site.

	Biological Index Scores					
	IBI		**MIwb**		**ICI**	
Category	Earliest	Latest	Earliest	Latest	Earliest	Latest
10th %ile	14	19	3.1	4.3	6	10
25th %ile	21	24	5.1	6.1	16	20
Median	28	31	6.9	7.9	30	34
75th %ile	38	40	8.3	8.8	40	46
90th %ile	46.2	48	9.2	9.5	46	50
Mean	29.6	32.6	6.4	7.3	27.7	32.6
Paired t-test	442 df		403 df		267 df	
t value	-8.558		-10.678		-6.689	
Mean difference	2.8		0.8		4.8	
(L minus E)	p <0.0001		p <0.0001		p <0.0001	
Wilcoxon (Z)	-8.039		-9.794		-6.403	
	p <0.0001		p <0.0001		p <0.0001	

recover first, followed later on by fish abundance and biomass (MIwb), with structural and functional indicators (IBI) responding last. Although this sequence is not invariable it is the most often observed pattern in Ohio thus far.

Forecasting Future Conditions

One recently realized benefit of the type biological assessment and criteria framework used by Ohio EPA is the ability to forecast or estimate future conditions in terms of overall aquatic life use attainment/non-attainment. This was accomplished in the 1994 Ohio Water Resource Inventory (305b report; Ohio EPA 1994) by first estimating the observed rate of improvement (in terms of the proportion of miles of rivers and streams which are fully attaining applicable designated use criteria in each subsequent biennial 305b reporting cycle) and then projecting this into future years. Figure 7 is an example of this process and summarizes the change in the proportion of waters which fail to attain designated use criteria, namely the biological criteria, between 305b report cycle years. Results obtained since 1988 were used since this represents the approximate period when major municipal wastewater treatment plant (WWTP) discharges were mandated to attain compliance with water quality-based effluent limits. Most of the monitoring information collected since 1988 represents follow-up visits to streams and rivers which were monitored prior to these major treatment upgrades. Thus the aggregated information obtained during each subsequent 305b report cycle year constitutes the overall environmental improvement which has resulted mostly from these WWTP upgrades. A major challenge facing Ohio EPA's water quality management program is reaching 75% full attainment by the year 2000 (*i.e.*, the Ohio 2000 goal). Projecting beyond 1994 through the year 2000 by assuming the observed rate of improvement between 1988 and 1994 shows that we will fall nearly 20% short of the Ohio 2000 goal. The predominant limiting influences in terms of pollution sources (*i.e.*, point, nonpoint, or both) were also identified in each waterbody segment and aggregated on a statewide basis. The proportion of non-attainment predominantly influenced by point sources is projected to shrink from 40% in 1988 to less than 10% of the remaining impairment by the year 2000. Thus nonpoint sources will constitute an increasingly larger share of the impairment to aquatic life uses through the year 2000. One benefit of having this type of analysis is to support a re-examination of water program priorities, which have historically been driven by point source concerns. While point source management is important, increasing attention will need to be devoted to solving and abating nonpoint sources which, in Ohio, includes habitat degradation in addition to over-land runoff. Thus the regulatory "culture" of U.S. EPA and the States will need to mature beyond the present heavy emphasis on chemical/physical water quality and dilution-based methods to include more innovative and ecosystem-based approaches oriented to dealing with both point and nonpoint sources.

The Role of Biological Criteria in The Management of Aquatic Resources

We define the management of aquatic resources here as being broader than the traditional pervue of water quality management. In keeping with this, efforts to attain the goals espoused by the CWA and other initiatives (*e.g.*, maintenance and recovery of aquatic biodiversity) ought to recognize the

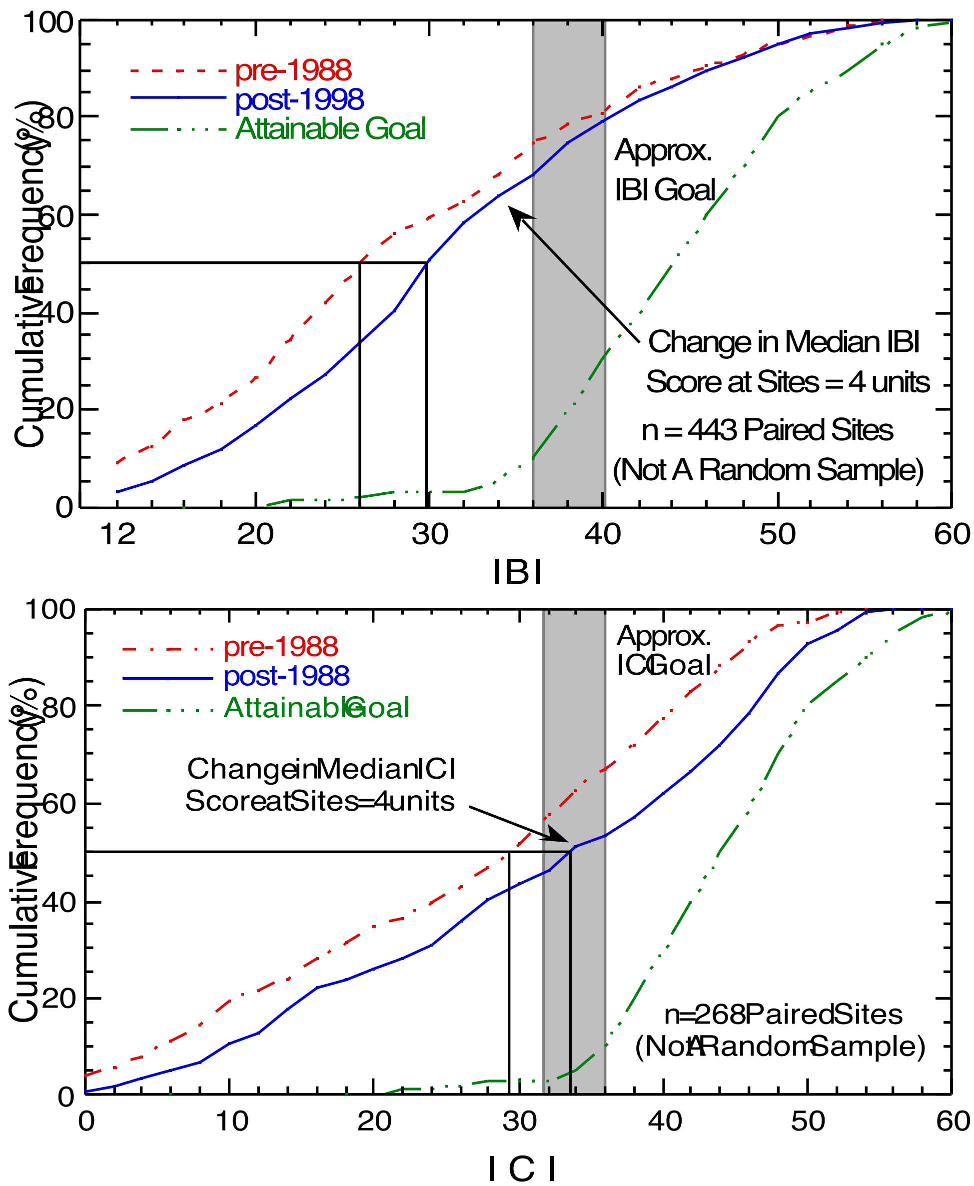


Figure 6. Cumulative frequency diagram (CFD) for the IBI and ICI (lower) in which the pre-1988 and post-1988 scores for 443 and 268 paired samples, respectively are compared on a statewide basis.

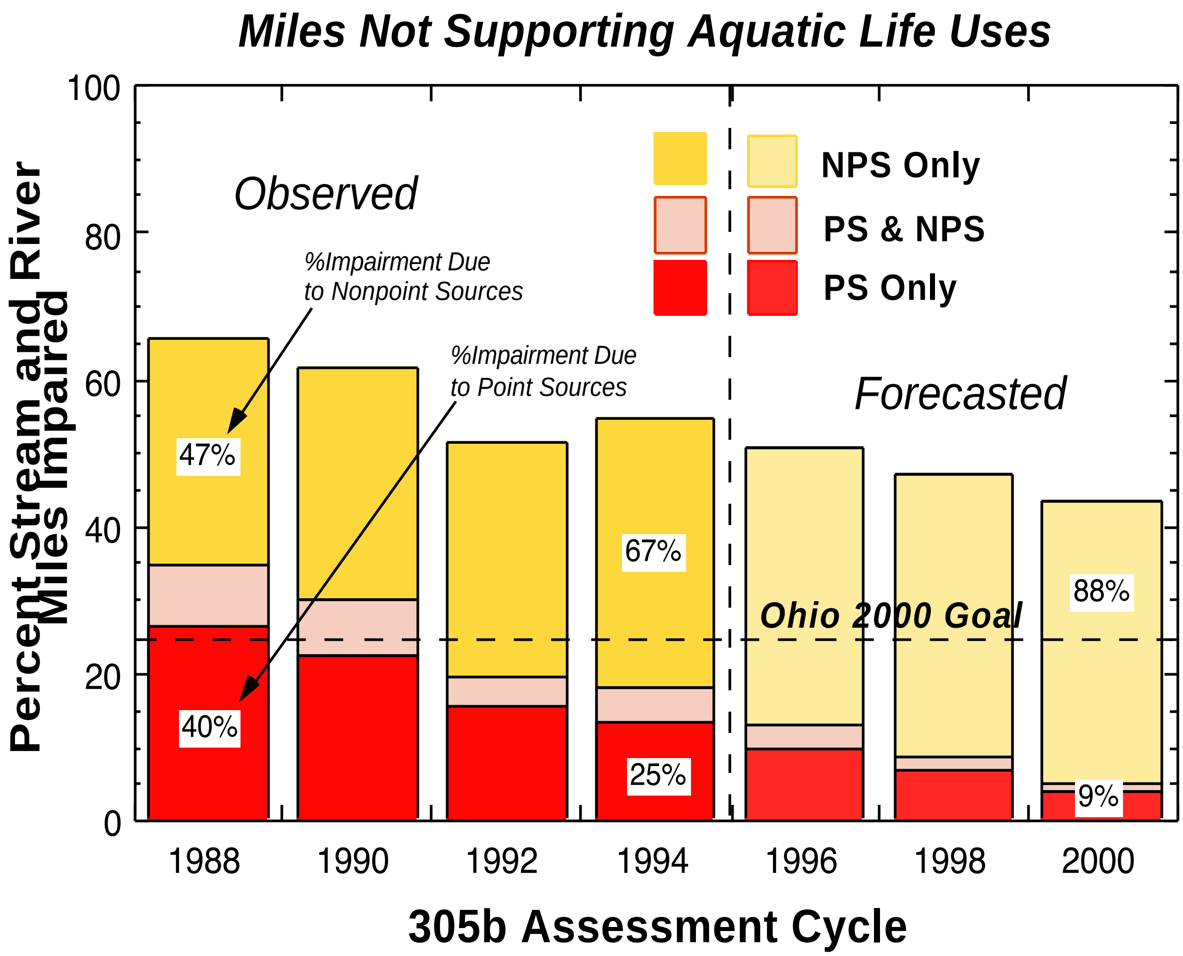


Figure 7. Observed reduction in the proportion of river and stream miles failing to attain criteria for designated aquatic life uses between 1988 and 1994 (left of dashed vertical line) and that projected through the year 2000 (right of dashed vertical line) based on the observed rate of restoration.

potentially broad role that biological criteria and assessment have in each area. It would be unfortunate to limit biological criteria to the traditional regulatory focus of water quality management programs (*i.e.*, NPDES permits, water chemistry, etc.) the ability to be useful in virtually any issue involving water resources, where a goal is to protect, enhance, or restore aquatic communities and ecosystems, has been amply demonstrated.

Uses of Biological Criteria in Ohio EPA Water Programs

The Ohio EPA water programs have relied extensively on ambient bioassessments since the late 1970s. The program areas within which biological criteria have found the most widespread uses are the biennial water resource inventory (305b report), water quality standards (aquatic life use classifications), NPDES permits (includes enforcement and litigation support), the construction grants program (now the State Revolving Loan Fund), the Ohio Nonpoint Source Assessment (CWA section 319), evaluation of wet weather flow impacts (stormwater, CSOs), the State certification of CWA section 404 permits (401 program) and petitioned ditches, ranking of CERCLA sites, and comparative risk (Figure 8). In addition the biological data has proved useful to other State agencies including the Ohio Department of Natural Resources (rare, threatened, and endangered species, scenic rivers, nonpoint source management, fisheries management) and the Ohio Department of Transportation (environmental impact statements). These applications are discussed in further detail by Yoder and Rankin (1995).

Policy Applications of Biological Criteria

This is perhaps the most controversial and certainly the least understood aspect of biological criteria, at both the national and state levels. When addressing the policy implications of biological criteria it is important to understand the applications of biocriteria and how this overlaps with the more traditional uses of chemical/physical and toxicological tools and criteria. Biological criteria are principally limited to ambient assessment applications whereas chemical and toxicological criteria can be used in ambient assessments and as design criteria. Understanding the basis behind these differences is important. For example, biocriteria are not intended to function the same way as a chemical criterion, from which effluent limitations for specific chemical substances are

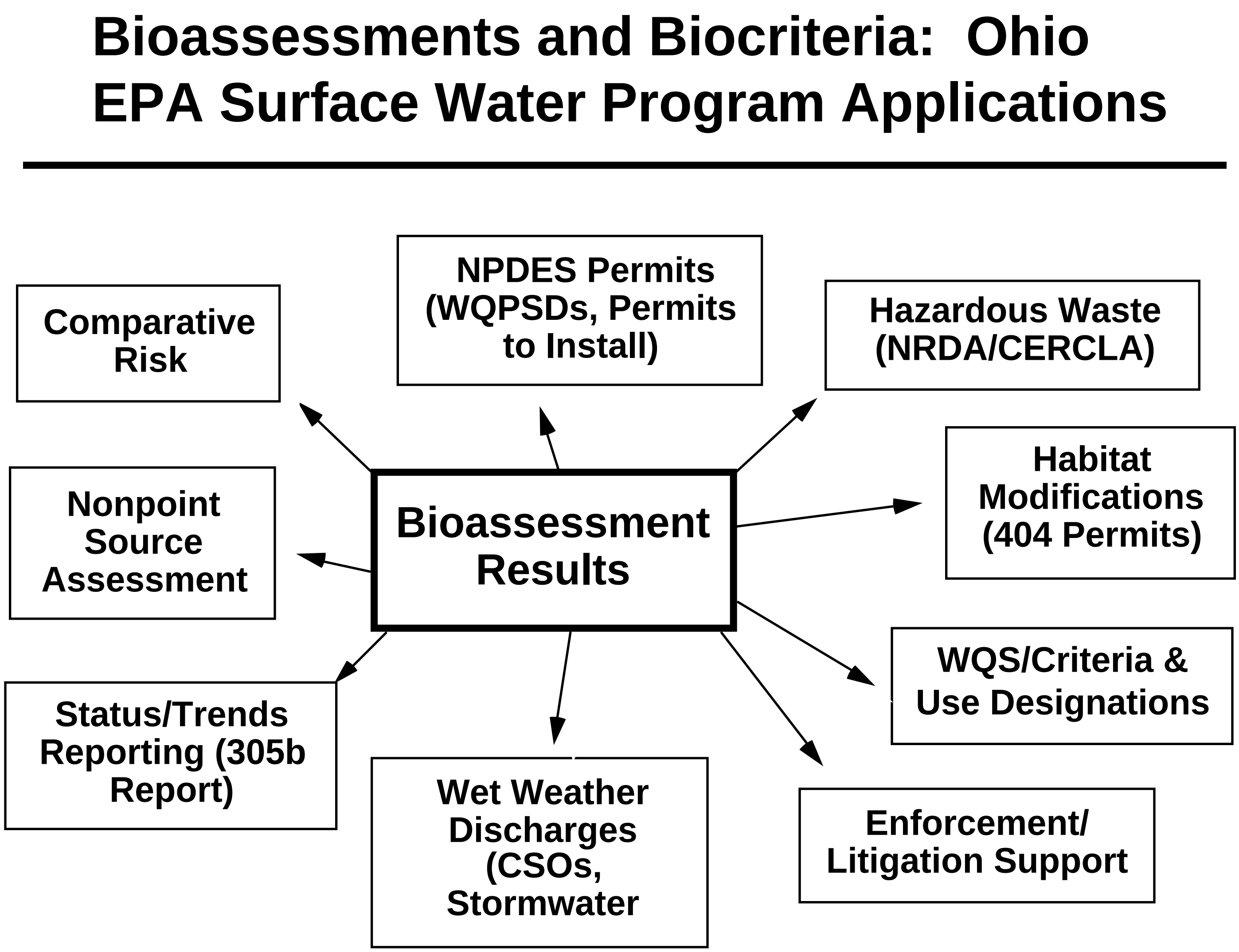


Figure 8. The various environmental management programs at Ohio EPA which are supported by information from biological surveys (biosurveys).

derived, even though both employ the common term “criterion”. Also, biocriteria are limited to aquatic life issues thus they play no more than an ancillary role in human health risk assessment. Despite these intuitively obvious limitations, biocriteria are frequently criticized for not being able to function for purposes which they were not originally intended to address.

There is broad agreement that bioassessments and biocriteria are one of the best ways to determine and characterize aquatic life use impairment. Beyond that, however, there are varied opinions about the policy and regulatory role of biocriteria. Presently these issues are still being discussed within U.S. EPA and between U.S. EPA, the States, environmental groups, and the regulated community. U.S. EPA and most environmental groups favor the policy termed “independent application”. Others, principally States and the regulated community, have proposed a “weight of evidence” approach. We attempt here to summarize some of the technical issues underlying this debate. Also we emphasize that the following deals entirely with aquatic life issues and does not transcend the importance of criteria for persistent toxicants as they pertain to human health, wildlife, and other non-aquatic life uses.

Policy of Independent Application

The U.S. EPA policy of independent application was first outlined in the national biocriteria program guidance (U.S. EPA 1990) and later reaffirmed in an “eco policy” statement (U.S. EPA 1991b) and the revised Technical Support Document for Water Quality-Based Toxics Control (U.S. EPA 1991c). A panel discussion at the Third National Water Quality Standards for the 21st Century conference dealt with this policy and included perspectives from U.S. EPA (Jackson 1992), the regulated community (Ruffier 1992), a State (Schregardus 1992), and an environmental group (Schmidt 1992).

Since biological criteria are applied as a direct measure of aquatic life use attainment/non-attainment an obvious overlap with chemical/physical and toxicological surrogate criteria occurs. This can happen in at least two different ways: 1) where concurrent biological, chemical and/or toxicological data are being used to assess aquatic life use attainment/non-attainment *on an ambient* *basis*, and 2) in determining appropriate effluent limitations for point sources based on the reasonable expectation that one or more criteria (including whole effluent toxicity) might be exceeded based on worst case assumptions about receiving water conditions and other watershed characteristics. In both cases conflicts may arise between the three major assessment tools (chemical-specific, whole effluent, biocriteria). U.S. EPA's definition of independent applicability means that the validity of the results of any one of the three approaches is independent of the need for any confirmation by the others. Each assessment is interpreted independently with none being viewed as superior or more powerful than another. U.S. EPA bases this policy on the . . . "unique attributes, limitations, and program applications of each of the three tools". Jackson (1992) asserts that each method independently provides sufficient evidence of aquatic life use impairment irrespective of what the other tools show or fail to show. Thus appropriate regulatory action should be taken when any one of the three tools determines that a standard is not attained. It is clear that the policy has a regulatory emphasis and prior familiarity with and dependence on the chemical-specific and whole effluent approaches no doubt contributes to the policy.

The alternative to independent application is to employ what has been termed a weight-of-evidence approach in which no one tool is assumed to be either equal or superior, but an informed examination of the results may lead to giving one of the tools more "weight" in the decision making process. In this process the respective power and site-specific applicability of each tool is considered and no *apriori* decision about the independence of one tool from the others is made. It should be noted here that U.S. EPA has not yet fully developed policies and rules pertaining to biocriteria and is presently examining these issues, thus the debate remains open.

The flaws inherent to the U.S. EPA policy of independent application lie in the *apriori* equating of the three major tools, chemical-specific, whole effluent toxicity, and biocriteria. In appearance this policy is only a more complex facsimile of the long abandoned U.S. EPA policy of presumptive applicability of the mid-1970s U.S. EPA WQS program. We believe that the site-specific circumstances relevant to the five factors which determine water resource integrity (see Figure 1) should determine how much weight should be given to each tool. This type of decision can be made only when comprehensive and adequate data from all three tools is available. Under independent application a decision is only as "good" as the least powerful assessment tool, whereas under weight-of-evidence the "strongest" data play a more appropriate role. In many situations the bioassessment and biocriteria will provide the most powerful information, but may not be entirely conclusive. Thus the integrated application of the chemical-specific, whole toxicity, and other tools (*e.g.,* habitat, biomarkers) will be required to successfully employ a weight-of-evidence approach. Yoder (1995) provides a more extensive discussion of the details surrounding the current policy debates involving biological criteria.

Other Policy Concerns

Not all of the concerns about biocriteria and bioassessment are being expressed by U.S EPA and environmental groups. States and the regulated community, while generally in favor of the overall biocriteria and bioassessment approach, have also expressed concerns. Cost and resource constraints are frequently raised by States which are facing an ever increasing burden of mandates without external funding increases. Thus the up-front investment required by biocriteria, while no more expensive than the other tools, represents an added cost. This is why it is important to provide incentives, additional or offset funding, or both for States to adopt biocriteria. States should also look to capabilities outside of their immediate purview such as sister State or Federal agencies which possess bioassessment capabilities. We do not intend to minimize the difficulties of actually accomplishing this type of inter-jurisdictional cooperation, but examples do exist and this will be strongly encouraged in the future.

The regulated community is concerned about the potential for more stringent permit limits and other restrictions that may be leveraged by biocriteria (Polls 1995). There is little doubt that biocriteria will enhance the ability to detect environmental degradation thus the concern that regulatory requirements will increase as a result. However, a finding of non-attainment downstream from a discharge does not automatically translate into significantly more stringent effluent limitations or additional requirements. Biocriteria are principally an assessment tool and current science does not include a direct translation into effluent limits. However, biocriteria can give important cues about what types or classes of impairment exist (Yoder and Rankin 1995b) from which further regulatory efforts may emanate. The Ohio WQS have recently been modified with conditional statements about how the agency will proceed in cases where findings of non-attainment seem to contradict the existing performance of a permitted discharge.

Pihfer (1991) leaves us with the notion that as waters improve biocriteria will become more stringent leaving the regulated community on a "never ending merry-go-round" of increasingly stringent requirements. We disagree with this position because it presumes that the biota will continue to improve as pollutant concentrations are reduced on an infinite and linear basis. As has been pointed out previously the biota are more oriented towards threshold responses thus there is a point beyond which additional pollutant removal will have little or no beneficial effect. In fact, if a weight-of-evidence approach is employed a regulated entity is more likely to know when to "get off" rather than continue to be subjected to the uncertainties of independent application. Another concern is that an entity may be in full compliance with an NPDES permit, yet degradation is detected downstream from the discharge. In this case it would seem, as we have found to be the case in Ohio, that either the permit limitations are not sufficient, the entity self-compliance monitoring is inadequate or unrepresentative, there are undetected or unreported violations, or there are other pollutant releases not covered by the permit. While an entity may be reluctant to have these facts revealed, the lack of prior knowledge should not be a license to continue with the status quo.

In addition to technical concerns discussed by Polls (1995), the regulated community is also concerned about taking on responsibility for conducting the ambient monitoring required to implement biocriteria. We strongly advocate that the States have primacy in this area since this is necessary to develop the appropriate expertise in maintaining the biocriteria and in conducting the ambient bioassessments. States are in a much better position to implement a comprehensive program which includes issues beyond point sources such as habitat effects and integrated watershed management. Regulated entities that have existing bioassessment capabilities can contribute significantly to this process, but they should not be expected to shoulder the burden for the entire program. Volunteer programs will not fill this gap either as these rarely, if ever, have the expertise and resources required to operate the level of bioassessment necessary for a credible biocriteria approach. As we have already acknowledged this will be a difficult area for some States primarily because of the start-up costs. This is an area where U.S. EPA must examine the trade-offs between not having an adequate bioassessment capability and having existing impairment remain undetected or underrated. The regulatory community should have an interest in seeing good programs develop and be maintained as the information base will, on balance, foster better decision making by the States. Based on the overall water program costs incurred by Ohio EPA, this would constitute a shift of approximately 5 to 15% of water program resources depending on what bioassessment capabilities already exist.

The allowance of policy flexibility, which should be contingent on a commitment to using a higher level of bioassessment, might serve as an incentive for States to invest the resources necessary to develop a reference site database, derive numerical biocriteria, and establish sufficient case histories. This would not only serve the needs of individual States, but would also provide U.S. EPA with a better national assessment, something which has been sorely lacking over the past twenty years (see Figure 2).

Aquatic Resources at Risk - The Consequences of Inaction

There is little question that aquatic resources have been and continue to be degraded by a myriad of land use and resource use activities. Benke (1990) summarized the status of the nation's high quality rivers and streams concluding that fewer than 2% remain in this category. Judy *et al.* (1984) indicate that the declining status of surface waters across the U.S. is largely the result of nonpoint source impacts. A continued reliance on technology based and even water quality-based solutions to these problems will simply be insufficient. Water resources in Ohio and elsewhere

have historically been and will continue to be impacted by human activities beyond those targeted by the NPDES permit process. These remaining problems are comparatively more complex and subtle, but are no less important or real. In fact, it is these more subtle and diffuse impacts which imperil aquatic resources to the point where additional species are declining in distribution and abundance, this in addition to those already declared as rare, threatened, or endangered. In Ohio this amounts to an additional 10% of the fish fauna against the 30% which already fall into the various categories of imperilment (Ohio EPA 1992b).

A monitoring approach, integrating biosurvey data that reflects the integrity of the water resource directly, including water chemistry, physical habitat, bioassay, and other monitoring and source information, is essential to accurately defining these varied and complex remaining problems. Such information must also be used in tracking the progress of efforts to protect and rehabilitate water resources. The arbiter of the success of water resource management programs must shift from a sole reliance on administrative activities (numbers of permits issued, dollars spent, or management practices installed) and a pre-occupation with chemical water quality *alone* to more integrated and holistic measurements with overall water resource integrity as the goal. Biocriteria is an essential component in making this program shift.

Emphasizing aquatic life use attainment is important because: 1) aquatic life criteria oftentimes result in the most stringent regulatory requirements compared to those for the other use categories, (*i.e.*, protection for the aquatic life use criteria will assure the protection of other uses); 2) aquatic life uses apply to virtually all waterbody types and the diverse criteria (*i.e.*, includes conventionals, nutrients, toxics, habitat, physical, and biological factors, etc.) apply to all water resource management issues; and, 3) aquatic life uses and the accompanying chemical, physical, and biological criteria provide a comprehensive and accurate ecosystem perspective towards water resource management which promotes the protection of ecological integrity. The need for an ecological perspective in water resource management is especially evident in the following:

- the assessment and control of wet weather flows (stormwater, combined sewers);
- nonpoint source assessment and watershed management;
- site-specific criteria modifications; and,
- regulation of activities which directly impact aquatic habitat.

Finally, biocriteria can aid greatly in the visualization and classification of aquatic resource values and attributes. This is a critical need if we are to change the prevailing view of watersheds and streams merely as catchments and conveyances for municipal and industrial wastes, excess surface and subsurface drainage, or as obstacles to further land developments. In an effort to stem the virtually unabated loss of riparian habitat and watershed integrity Ohio EPA has proposed a stream protection policy which sets forth guidelines under which various activities will need to be conducted in order to conserve biological integrity. Without biocriteria and the case examples developed over the past 15 years this would not have been possible and any opportunity to stem these degrading influences would have been lost. Conversely, biocriteria and the attendant habitat assessment tools provide an opportunity to determine the resource value through the assignment of designated uses. Streams and rivers are designated for levels of protection consistent with their realistic potentials. The guidelines in proposed stream protection policy takes these factors into account, something which prescriptive programs would fail to recognize. Thus truly high quality resources are afforded an appropriate level of protection and lesser quality resources are not unnecessarily over-protected.

Remaining Challenges

While we have been able to demonstrate how biological criteria can be developed and used within a State water resource management framework, there are some important challenges which remain. Recently, the cumulative costs associated with environmental mandates, many of which consist of prescriptive regulations, have come into question. Former EPA Administrator Reilly frequently cited the need for the increased use of “good science” in formulating regulatory requirements. Both the regulated community and the public desire evidence of “real world” results in return for the expenditures made necessary by Federal and State mandated requirements. Biological criteria seem particularly well suited to meet some of these needs in that the underlying science and theory is robust (Karr 1991) and biocriteria certainly qualifies as “real world”.

The historical tendency in water quality management has been to make biological measurements fit the perceptions and use of chemical criteria, rather than the reverse. This is a paradox because an aquatic community is the embodiment of the temporal and spatial chemical, physical, and biological dynamics (*i.e.*, the "pieces") of the aquatic environment, not the reverse. Perhaps the inability of biologists to agree on a set of empirical measurements of biological integrity, or at least a common framework, has resulted in this tendency (Karr *et al.* 1986). One solution to this deficiency is to employ biological criteria which can directly indicate the degree to which biological integrity is or is not being achieved. This does not mean that biological criteria are a substitute for chemical criteria or bioassay techniques as these will continue to play an important role in water quality management. Their value, however, is greatly enhanced when used in combination with biological criteria. Such an approach will undoubtedly lead to more effective regulation of pollution sources, improved assessment of diffuse and non-chemical impacts, and a broadened capability to implement management strategies for protecting and restoring watersheds.

While no single environmental indicator can “do it all”, particularly in the more complex situations (i.e. multiple discharges, habitat alterations, presence of toxic compounds, etc.), it is obvious that biological criteria have a major role to play. A lack of information from or an over-reliance on any one indicator can result in environmental regulation that is less accurate and potentially under- or overprotective of the water resource. Accounting for cost is not only a matter of dollars spent, but is also a question of environmental accuracy and technical validity. In short, a credible and genuinely cost-effective approach to water quality management should include an appropriate mix of chemical, physical, and biological measures, each in their appropriate roles as stressor, exposure, and response indicators.

Adopting an increased reliance on direct measurements of biological community performance to establish regulatory direction and priorities will require a modification of some current regulatory attitudes and approaches. This will involve linking treatment processes and performance, ambient water quality, habitat, toxicity units, best management practices, etc. with observed biological community response in a feedback loop arrangement. Steedman (1988) provided a good example of how empirical data was used in a similar fashion to establish land use/riparian zone criteria for attaining prescribed levels of performance for Lake Ontario tributary fish communities. With some types of sources and degradation, a certain amount of trial and error application may be required especially where chemical tools begin to approach and exceed uncertainty, key knowledge about the chemical substances involved is lacking, or the cause of degradation is primarily nontoxic.

What, then, can be done to improve the process? The most logical avenue is to modify State WQS. As part of the State WQS regulations the management and implementation tools associated with biological criteria become legitimized beyond their usual "optional" status. An understanding of how the fundamental goals of the Clean Water Act (CWA) influence State WQS is necessary to comprehending the broad role that biological criteria can play in the surface water quality management process.

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