247

Answering Some Concerns About Biological Criteria Based on Experiences in Ohio

Chris O. Yoder

Manager, Ecological Assessment
Ohio Environmental Protection Agency
Division of Water Quality Planning & Assessment
Columbus, Ohio

Introduction

Biological criteria have been receiving increased national attention among the States and from the U.S. Environmental Protection Agency (EPA). The Agency has published national program guidance for biological criteria (U.S. Environ. Prot. Agency, 1990) and will require States to develop narrative biological criteria by 1993, evidence that this is a priority in its water quality program.

In Ohio, biological assessments and corresponding evaluation criteria have been used extensively since 1980. Use and evaluation of ambient biological data underwent an evolutionary process, from narrative descriptions of community attributes in the early 1980s to the numerical biological criteria adopted into Ohio's water quality standards regulations in February 1990.

The way regulatory agencies have assessed and managed surface water resources has undergone significant changes in the past 10 years. What was primarily a system of simple chemical criteria that served as surrogates for the biological integrity goal of the Clean Water Act has matured into a multidisciplinary process that includes complex chemical criteria and standards for whole effluent toxicity and biological community performance. This integrated approach has allowed surface water management programs to focus beyond water quality and consider the surface water resource as a whole.

Simply stated, controlling chemical water quality alone does not assure the integrity of water resources (Karr et al. 1986; Ohio Environ. Prot. Agency, 1990a); this results from the combination of chemical, physical, and biological processes (Fig. 1). To be truly successful in meeting this goal, we need monitoring and assessment tools that measure both the interacting processes and integrated result of these processes. Biological criteria offer a way to measure the end result of water quality management efforts and successfully protect surface water resources.

In addition to accurately assessing water resource health, the challenge of accounting for the landscape's natural variability was addressed through the use of ecoregions (Omernik, 1987) and regional reference sites (Hughes et al. 1986, 1990). Ecoregions delineate variability in major landscape features at a level of resolution that is easy to apply in statewide water quality standards (Gallant et al. 1989). Ecoregions in Ohio are transitional; they range from the flat, extensively farmed northwest section to the highly dissected, unglaciated east and southeast part of the State (Omernik and Gallant, 1988). In Ohio, numerical biological criteria are organized by ecoregion, organism group, site type, and use designation (Yoder, 1989; Ohio Environ. Prot. Agency, 1990b).

24654

Biological Criteria: Questions and Concerns

Although biological assessments have been a part of some State monitoring efforts for many years, only recently has the need for and acceptance of ambient biological criteria been recognized. In many traditional water quality circles, the validity and efficacy of biological criteria are often questioned or misunderstood. This presents a paradox because biological criteria directly express what water quality criteria are designed to achieve.

In an effort to address some of these concerns, we have posed the following five questions about biological criteria and answered them with real world examples from our experiences in Ohio.

1. Are ambient biological measures too variable to use in assessing surface water resources?

A frequent criticism of ambient biological data is that it is subject to natural and anthropogenic variations and therefore too "noisy" to function as a reliable component of surface water resource management. Natural biological systems are variable and seemingly "noisy," but no more than the chemical and physical components that exist within them. Certain components of ambient biological data are quite variable, particularly those measures at the population or sub-population level.

Single dimension community measures can also be quite variable. However, the advent of new generation community evaluation mechanisms such as the Index of Biotic Integrity (IBI) (Karr, 1981; Karr et al. 1986) have provided sufficient redundancy as to compress and dampen some of this variability. Rankin and Yoder (1990) examined replicate variability of the IBI from nearly 1,000 sites in Ohio and found it to be quite low at least-impacted sites (Fig. 2). Coefficient of variation (CV) values were less than 10 percent at IBI ranges indicative of exceptional biological performance, which is lower than that reported for chemical laboratory analyses and interlaboratory bioassay variability (Mount, 1987). Variability as portrayed by CV values increased at the IBI ranges indicative of impaired

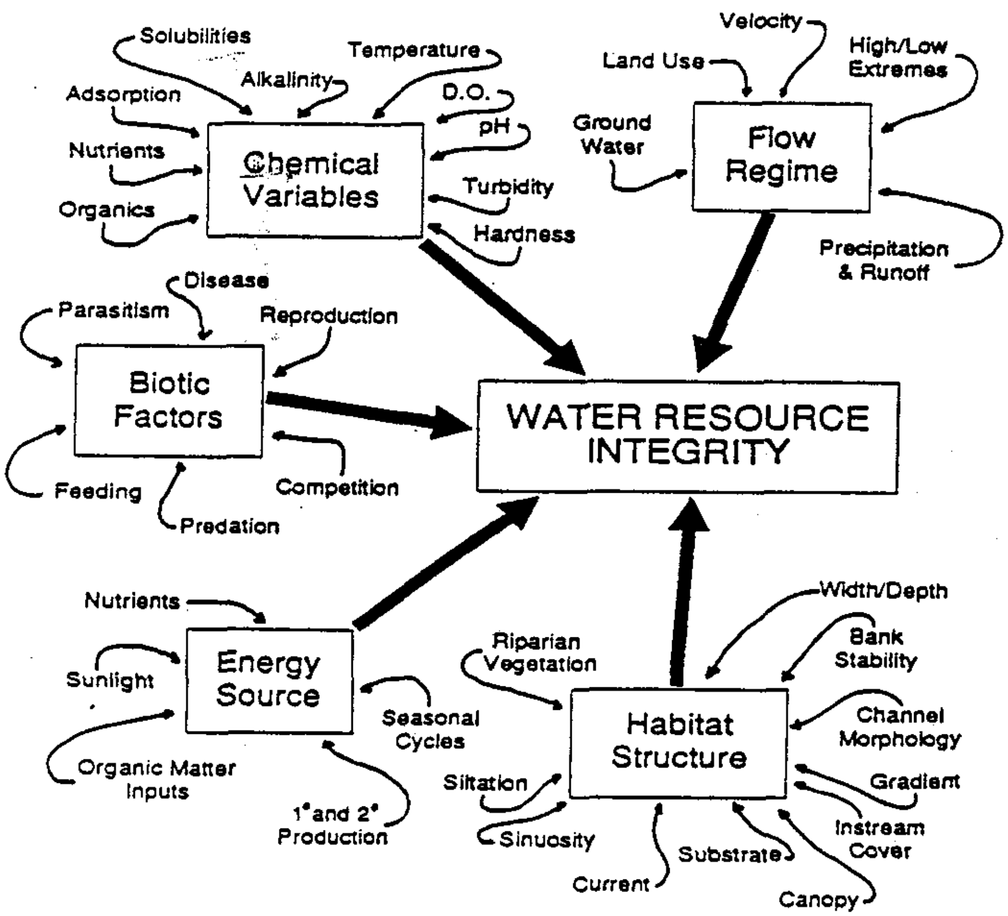


Figure 1.—The five principal factors, with some of their important chemical, physical, and biological components, that influence and determine the resultant integrity of surface water resources (modified from Karr et al. 1986).

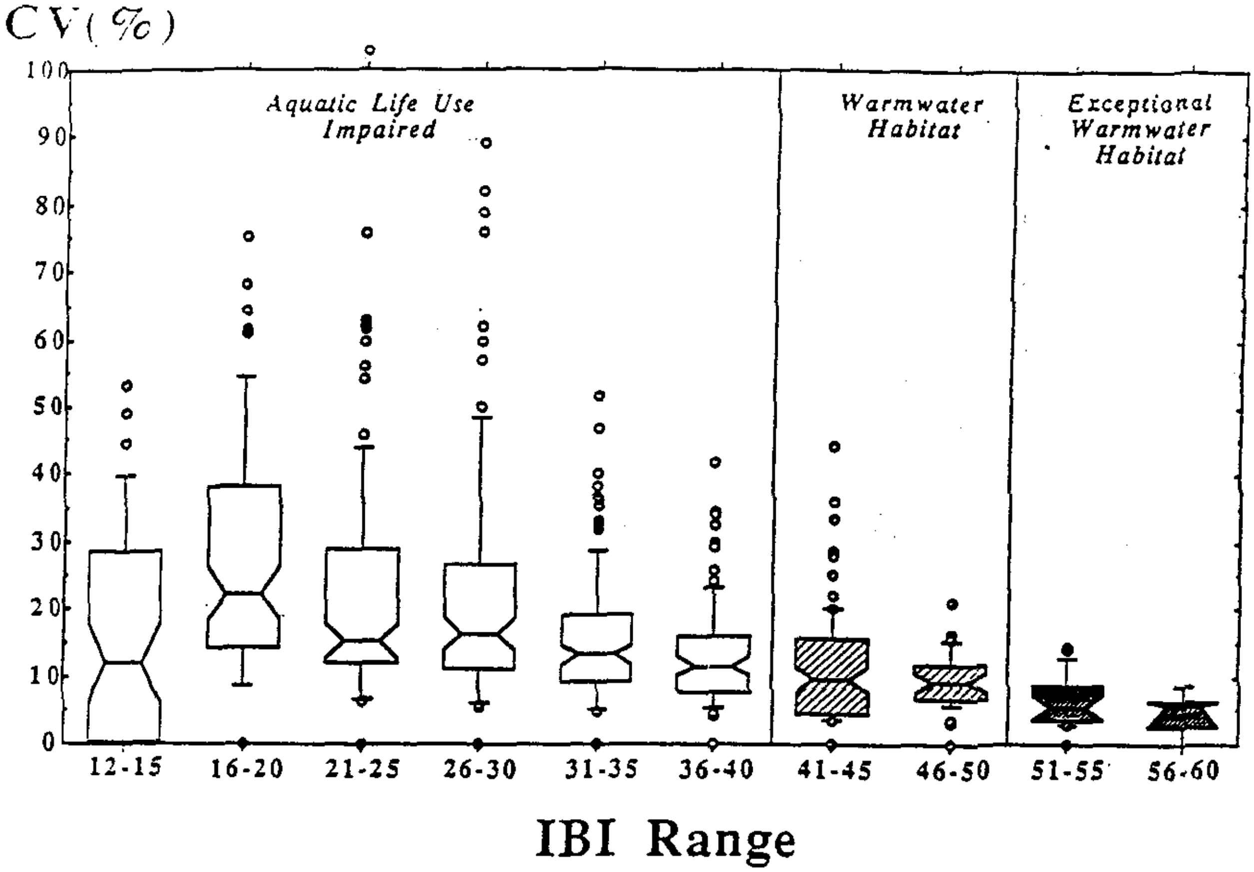


Figure 2.—Coefficient of variation (CV) for a range of IBI scores at sites with three sampling passes per year. Boxes show median, 25th and 75th percentiles and minimum, maximum, and outlier values.

biological performance. Low variability was found for Ohio's Invertebrate Community Index (ICI) with a CV of 10.8 percent for 19 replicate samples at a relatively unimpacted test site. Other researchers have reported similarly low variability with ambient biological evaluations (Davis and Lubin, 1989; Stevens and Szczytko, 1990).

Cairns (1986) suggested that differences in variability rather than differences in averages or means might be the best measure of stress in natural systems. Not only is the variability of the measures used to implement biological criteria low, the degree of variability encountered can be a useful assessment and interpretation tool.

Ohio EPA has addressed the variability inherent to biological measures in three general ways:

1. Variability is compressed through the use of multimetric evaluation mechanisms such as the IBI and ICI.
2. Variability is stratified through use of a tiered stream classification system, ecoregions, biological index calibration, and site type.
3. Variability is controlled through standard sampling procedures that address seasonality, effort, replication, gear selectivity, and spatial concerns.

Lenat (1990) also described similar approaches to controlling and thus reducing variability in ambient biological samples.

2. Are biological criteria sufficiently sensitive to serve as a measure of surface water resource integrity?

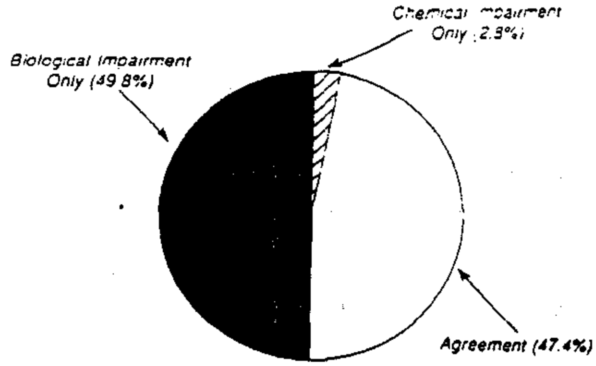
Conceptually, direct biological measures should be sufficient to measure water pollution control goals and end points that are fundamentally biological. However, this fact alone is an insufficient test of the efficacy of biological criteria and attendant assessment methodologies. Evaluation against currently accepted assessment methods is one way to test the comparative sensitivity of biological criteria. This was accomplished in the 1990 Ohio 305b report (Ohio Environ. Prot. Agency, 1990a), where comparisons were made of the relative abilities of biological and chemical water quality criteria and whole effluent toxicity tests to detect aquatic life use impairment.

In comparing biological with chemical water quality criteria, a database was used that consisted of 625 waterbody segments. Individual waterbody segments averaged 10.6 miles in length (range: 0.5-41.2 mi.) and had one or more chemical and biological sampling locations. Biological data consisted of fish and/or macroinvertebrate results. Water chemistry data consisted of grab samples at an average of 3.6 samples per site (range: 1 to 13 samples) and included parameters commonly measured by most ambient monitoring networks. (Ambient grab samples usually consist of dissolved oxygen, temperature, conductivity, pH, suspended solids, ammonia-N, nitrate-N, nitrite-N, total Kjehdahl nitrogen, phosphorus, and toxics such as cyanide, phenolics, copper, cadmium, chromium, lead, nickel, iron, and zinc on an as-needed basis.)

Ohio's recently adopted biological criteria were used to define biological impairment and the Ohio Water Quality Standards (WQS) were used to determine exceedances of chemical results. The comparison showed that biological impairment was evident in 49.8 percent of the segments where no ambient chemical water quality criteria exceedances were observed (Fig. 3). Both the biological and chemical assessments agreed about impairment (or lack thereof) in 47.4 percent of the waterbody segments. Chemical impairment was evident in the remaining 2.8 percent of the segments where no biological impairment was found. While much of the concern expressed about biological criteria has been with its potential use to "dismiss" chemical exceedances, such as the latter case, the most important finding of this analysis was with the ability of the biota to detect impairment in the absence of chemical criteria exceedances. An initial reaction to these results might be to view chemical criteria as not being sufficiently protective. However, further analysis of the reasons behind these results shows that the stringency of the chemical criteria is not an important issue. In the 49.8 percent of the segments with biological impairment alone, the predominant causes of impairment were organic enrichment/dissolved oxygen, habitat modification, and siltation (60.4 percent of the impaired segments). None of these, except very low dissolved oxygen, are measurable by direct exceedances of chemical water quality criteria.

Chemical causes of impairment were predominant in a minority of the cases (30.7 percent). In the absence of chemical criteria exceedances from the water column, this cause was deemed important because of information such as sediment contamination or effluent data that indicated periodic chemical problems not readily detectable by grab sampling. In this case, it was the failure of the chemical sampling effort to detect exceedances in the water column, primarily because of an insufficient sampling frequency, parameter coverage, or both. In many segments, both chemical and nonchemical causes occurred simultaneously, resulting in cumulative effects evident only in the biological results.

Case I: Relative performance of chemical water quality criteria vs. biological criteria



Case II: Ecoregional threshold concentrations for nutrients improves the performance of water chemistry

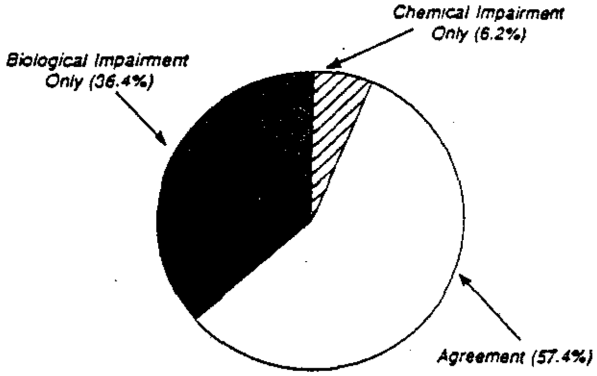


Figure 3.—Comparison of the abilities of biocriteria and chemical criteria to detect impairment of aquatic life uses in 625 waterbody segments throughout Ohio. Data were based on chemical water quality criteria currently in Ohio's water quality standards (upper) and supplemented with nutrient data using threshold values from ecoregional analysis (lower).

Another important factor to consider is that chemical criteria in this evaluation are used in an ambient application. Thus, factors such as sampling frequency, temporal variability, parameter coverage, and dilution dynamics can be of equal, if not overriding, importance as the stringency of the chemical criteria. One of the most important applications of chemical criteria is as design standards where factors such as design flows and safety factors tend to make up for their apparent inadequacies. This is not to say that chemical criteria can never be too stringent or lenient. Such situations are likely to arise on a site-specific basis, where unique regional or local conditions result in differences.

The performance of the chemical assessment relative to the biological was improved by including ecoregional threshold exceedances for nutrient parameters (nitrogen series, phosphorus), for which no aquatic life criteria exist (Fig. 3). By using the Ohio regional reference site database, threshold values for these parameters were established as 75th percentile concentrations. This reduced the frequency of segments with biological impairment alone to 36.4 percent. Again, the reasons are complex and were most often related to the coincidental occurrence of higher nutrient concentrations with predominant impacts such as organic enrichment, siltation, and habitat modification. Further work with ecoregional threshold values for additional chemical parameters may enhance the use of ambient water chemistry results for broad scale assessments such as the biennial 305b report and nonpoint source assessment.

An initial comparison was also made with bioassay results from 43 entities where receiving stream biosurvey data was available. The bioassay results represent 96-hour acute-definitive tests of the effluent and immediate mixing zone area. In-stream biological impairment was observed in nearly 60 percent of the comparisons where acute toxicity >20 percent was observed only in the effluent (Fig. 4).

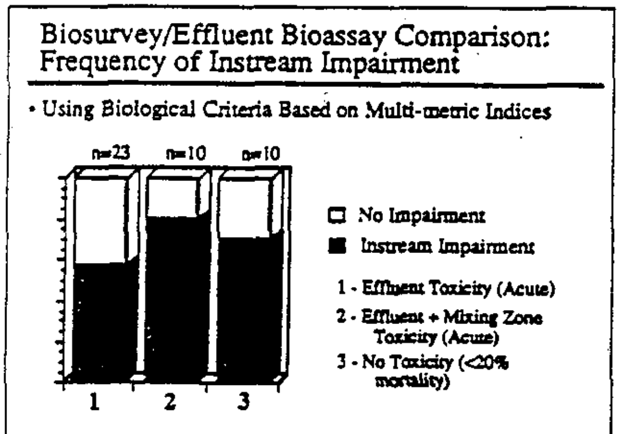


Figure 4.—Comparison of the abilities of biocriteria and acute bioassays to detect impairment of aquatic life uses at 43 locations throughout Ohio. Frequency of in-stream impairment is compared against: (1) effluent toxicity >20 percent only; (2) effluent and mixing zone toxicity >20 percent; and (3) no toxicity (≤20 percent).

For the cases where >20 percent mortality was observed in both the effluent and mixing zone, 8 of 10 comparisons showed in-stream biological impairment. In the remaining cases where no significant mortality (≤20 percent) of bioassay organisms was observed, biological impairment was observed in 7 of 10 comparisons. Again, the reasons for these discrepancies are complex but similar to the previously discussed comparison where biological impairment was observed in the absence of chemical criteria exceedances. Although more detailed analysis of these comparisons is needed, there was a general relationship between the severity of the bioassay toxicity and the existence of in-stream biological impairment (Ohio Environ. Prot. Agency, 1990a).

3. By using a regional reference site approach for establishing biological criteria, are aquatic life goals being set too low?

The debate about how attainable condition should be defined began in the 1970s with discussions on how to define and measure the Clean Water Act goal of biological integrity. Initial attempts failed to bring about a quantitative approach (Ballantine and Guarraia, 1975), but an acceptable definition was eventually forthcoming. This has been referred to as the operational definition of Karr and Dudley (1981), which essentially translates into the "biological performance and characteristics exhibited by the natural habitats of a region."

This provides the theoretical basis for designing a regional monitoring network of least impacted reference sites (Hughes et al. 1986) from which quantitative, numerical biological criteria can be derived. The specific approach used by Ohio is discussed elsewhere (Ohio Environ. Prot. Agency, 1987, 1989a; Yoder, 1989). The methods used to select and monitor reference sites, calibrate the biological evaluation mechanisms (IBI, ICI), and set the ecoregional biological criteria are inherently conservative and guard against biases that may result in underprotective biological criteria.

Reference-site selection guidelines are necessarily qualitative and are described in detail in Whittier et al. (1987) and Ohio EPA (1987, 1990b). In Ohio, which has had extensive landscape disturbance, the goal is to select least impacted watersheds to serve as a reflection of the current-day biological potential. Reference sites are selected according to stream size, habitat characteristics, and the absence of direct point source or obvious nonpoint source pollution impacts.

The "least impactedness" of reference sites in the extensively disturbed Huron/Erie Lake Plain (HELP) ecoregion of northwest Ohio is much different from that in the less-disturbed Western Allegheny Plateau (WAP) of southeastern Ohio and the other three ecoregions. Such background conditions can be unique to each region and, as such, define the present-day potential.

A criticism of this approach is that it relegates these areas to being no better that they are presently. However, an important element of regional reference sites is the re-monitoring effort designed to take place once every 10 years after which any changes in the background potential can be reflected in the calibration of the biological evaluation mechanisms, the biological criteria, or both. This maintenance effort will ensure that the biological criteria do not underrate the attainable biological performance within each region of the State.

The method of calibrating the biological evaluation mechanisms, such as the IBI and ICI also protects against underprotective criteria that might result from including possible suboptimal reference sites. The calibration methods for the IBI as specified by Fausch et al. (1984) include plotting reference site results for each IBI metric against drainage area (a reflection of stream size). The first step is to draw a maximum species richness line, beneath which 95 percent of the data points occur. This represents the line beneath which the area of the graph is trisected resulting in the 5, 3, and 1 scoring criteria common to each of the 12 IBI metrics (Fig. 5).

The Ohio EPA ICI for macroinvertebrates is calibrated in a similar manner, except that the area beneath the 95 percent line is quadrisected in conformance with the 6, 4, 2, 0 scoring configuration of the 10 ICI metrics (Fig. 5). Where the 95 percent line is drawn is controlled by the upper surface of points that represent the best results obtained statewide for that metric. Thus, the influence of any sub-optimal or marginal data (whether these are due to unknown impacts or poor sampling) in the calibration of the IBI or ICI is virtually nil. This technique induces an inherent element of conservatism into the eventual biological criteria.

When the biological index values for the IBI and ICI are calculated for each reference site sample, the biological criteria for each index can then be derived. This process is not entirely mechanical and involves making some value judgments about how biological criteria will be selected. Ohio's water quality standards specify a tiered system of aquatic life use designations, each with a narrative definition that specifies the biological attributes that waters attaining that use should exhibit. For the warmwater habitat (WWH) use designation, which is the most commonly applied aquatic life use in Ohio, the 25th percentile value of the reference site results was selected as the applicable biological criterion. Ohio EPA decided that most of the reference results should be encompassed by this base level use for Ohio's inland rivers and streams. Also, by excluding a fraction of the reference results, any unintentional bias induced by suboptimal or marginal results caused by factors that were not apparent in the initial selection process would be minimized or eliminated.

When the insignificant departure tolerances for each index are considered, less than 5 to 10 percent of the reference results fail to attain the biological criteria for the WWH use. For instance, insignificant departure from IBI and ICI values are 4 units each (Ohio Environ. Prot. Agency, 1987). If the ecoregion IBI criterion is 42, a value of 38 would be considered to attain the biological criterion but would be regarded as

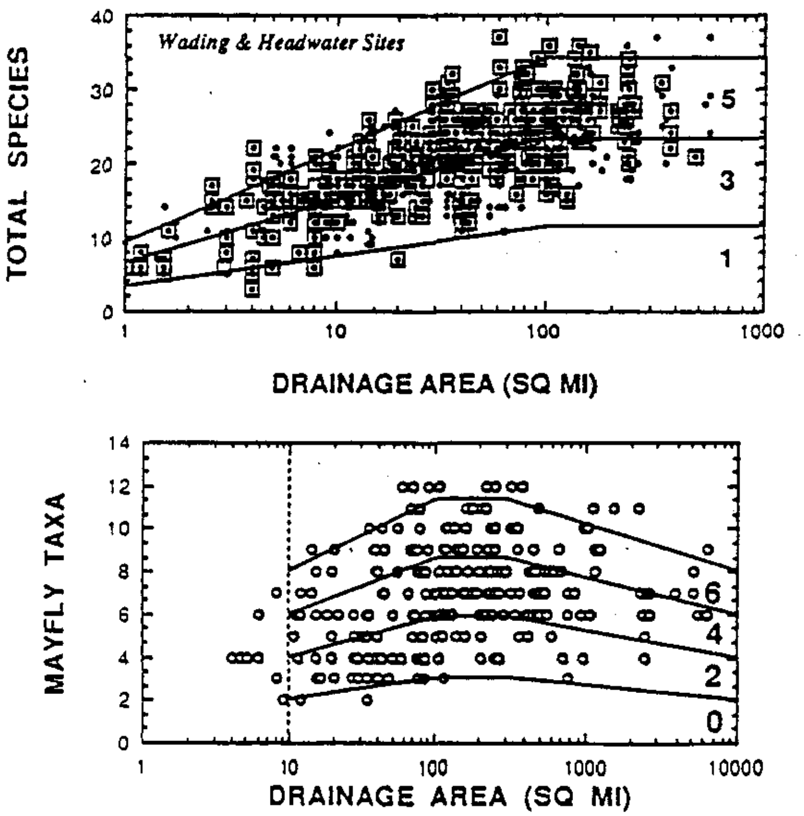


Figure 5.—Example of the technique used to calibrate the Index of Biotic Integrity (IBI) and the Invertebrate Community Index (ICI) for the metrics of each index. The number of fish species vs. drainage area for headwaters and wading site types (top panel) and number of mayfly taxa vs. drainage area (bottom panel) demonstrate the use of the 95 percent maximum line and the trisection and quadrisection methods used to establish the IBI and ICI metric scoring criteria.

an insignificant departure for risk management purposes.

This process is similar to the use of safety factors for toxicological applications and has previous precedents such as using the 75th percentile pH, temperature, and hardness to derive design un-ionized ammonia-nitrogen and heavy metals criteria, 20 percent mortality for bioassay results, or even using the 10^{-6} risk factor for carcinogens. In this sense, the 25th percentile acts as a safety factor in the derivation process. Because of unique problems with selecting reference sites in the highly modified HELP ecoregion, a different benchmark (upper 10 percent of all sites) was used to set the WWH biocriteria. The approach of setting attainable biological criteria is stratified by ecoregion (WWH use), site type for fish, and a tiered system of aquatic life use designations (Fig. 6). Rules for determining use attainment also provide safeguards: full attainment of a use requires achievement of the biological criteria for both fish and macroinvertebrates.

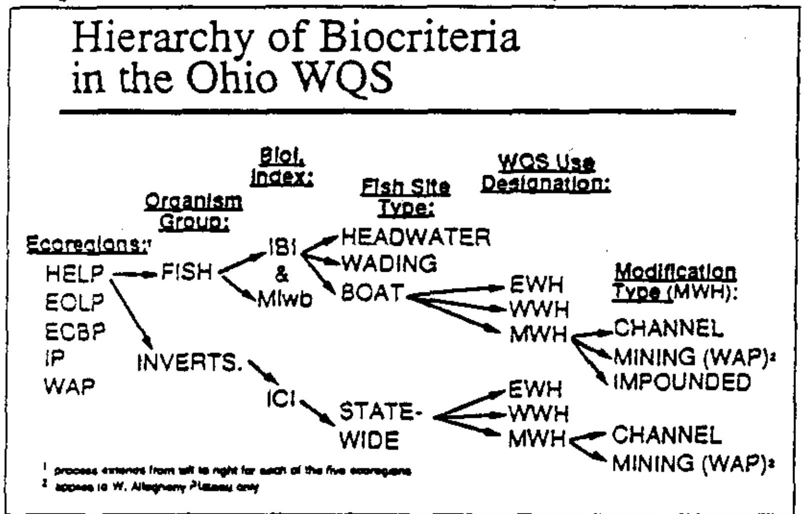


Figure 6.—Hierarchy of biological criteria in the Ohio water quality standards (WQS) showing organization by ecoregion, organism group, biological index, site type (fish), WQS-use designation, and modification type for the modified warmwater habitat use. The process above begins in the HELP ecoregion and extends from left to right through the fish and macroinvertebrate biocriteria. The ICI (statewide) and IBI (boat-site type) are portrayed and extend to the possible aquatic life use choices and the modification types possible for the WWH use. The possible pathways are the same for each of the other four ecoregions in Ohio.

4. Are the data collection costs associated with biosurveys and biological criteria unduly expensive?

Ambient biological assessments have had the unfortunate reputation of being time-consuming, intensive, and expensive. Oftentimes, this reputation has been a deterrent to using biosurveys in assessing surface water resources and in promoting surrogate methods of assessment (U.S. Environ. Prot. Agency, 1985).

The issue of cost has been addressed extensively in Ohio, where we have compared the relative resource requirements of ambient chemical assessment, bioassays, and biosurveys employing both fish and macroinvertebrates (Ohio Environ. Prot. Agency, 1990c). This comparison found that, for entity evaluation and stream surveys, biosurveys employing both fish and macroinvertebrates were cost-competitive with ambient chemistry and effluent bioassays (Table 1). While biosurveys may be comparable in terms of cost, it does not seem prudent to view these data in a competitive sense. Rather, the integrated use of all tools is necessary to ensure accuracy of evaluation and hence regulation. The well-worn metaphor of the three-legged stool is still appropriate.

A renewed focus on ambient biological assessment methods has resulted in the development of cost-effective strategies that also yield reliable and accurate information. Accuracy and reliability must accompany the cost effectiveness of the chosen approach. The importance of this concept is partially illustrated by an analysis of the different accuracies inherent to narrative and numerical biological as-

Table 1.—Comparison of the cost of ambient chemical, bioassay, and biosurvey assessment on an entity and stream survey evaluation basis, using cost data from Ohio EPA in FFY 1987 and 1988. This is based on an example that includes three point sources discharging to a medium-sized river in an urban and rural setting in Ohio.

CATEGORY	CHEMICAL	BIOSURVEY	BIOASSAY
Samples	90	12	9
Unit cost sample	$360	$1,850	$ 1,850 (acute)[1] $ 3,050 (7-day)[2]
Survey cost	**$32,400**	**$22,200**	**$16,650 (acute)[1]** **$27,450 (7-day)[2]**

Source: *The Cost of Biological Monitoring* (Ohio Environ. Prot. Agency, 1990c)
[1] 96-hour definitive test using *Ceriodaphnia* and fathead minnow
[2] 7-day acute chronic test using a 24-hour composite sample

24660

sessments. The evaluations yielded by Ohio's narrative macroinvertebrate criteria used from 1979 to 1986 and the ICI calibrated by using regional reference sites were compared across more than 400 sites sampled between 1981 and 1987.

The results indicated that the narrative approach overrated sites as being better than indicated by the calibrated ICI (Fig. 7). The narrative approach rated as "good" (attaining the WWH use) 36 percent of sites classified by the ICI as impaired, and as "fair," 21 percent of sites classified "poor" by the ICI. Only 1.3 percent of sites rated "poor" by the narrative method were classified "fair" by the ICI.

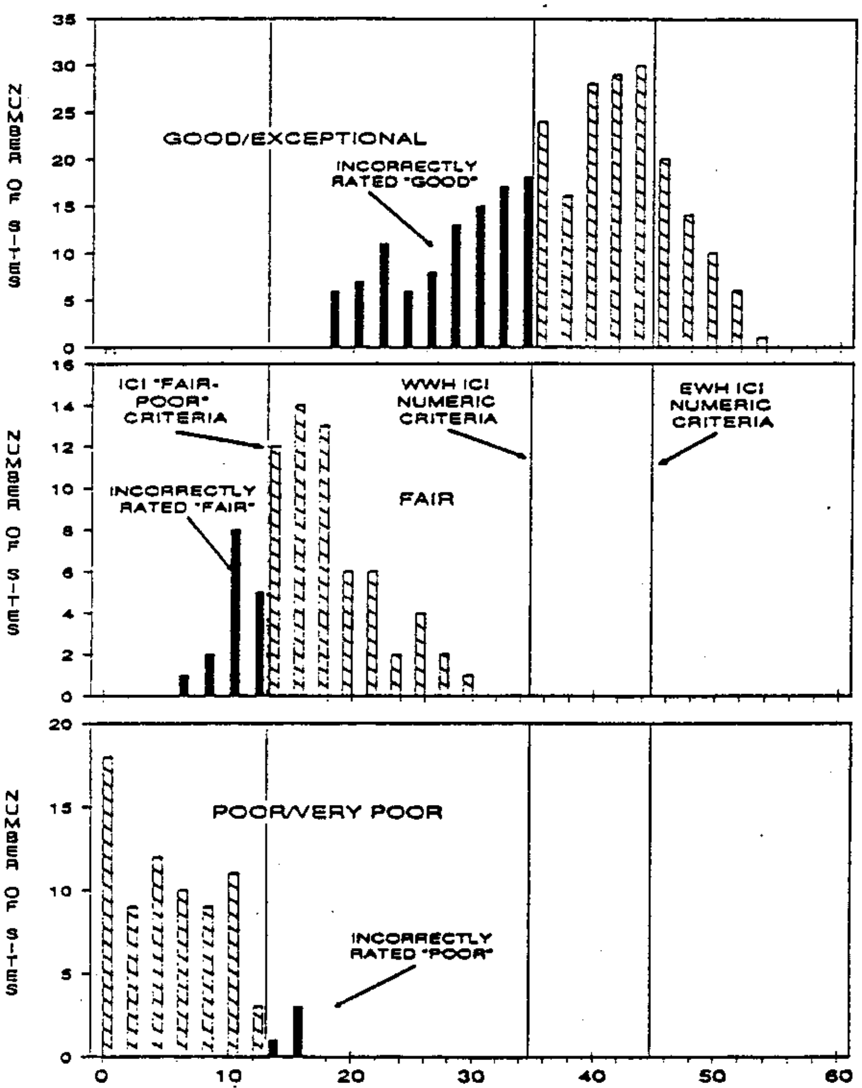


Figure 7.—Frequency distribution of ICI scores for more than 400 sites rated as Exceptional/Good, Fair, and Poor/Very Poor using the qualitative, narrative biocriteria developed in 1980 compared to the ICI biocriteria based on the regional, reference site approach. The solid bars are sites that were incorrectly rated by the narrative system vs. the ICI scoring derived from a numeric, regional, reference site system.

The predominant error orientation of the narrative approach was to rate sites as better than they were as determined by a calibrated evaluation mechanism. While it may seem premature to assume that the ICI is more accurate, the fact that it is a multimetric evaluation mechanism designed to produce the essence of the narrative system, but with greater precision, and that it extracts information directly from the regional reference sites argues in favor of the ICI.

The narrative evaluation system, on the other hand, relies on the best professional judgment of the biologist examining a completed sample sheet by eye aided by single dimension attributes such as number of taxa and a diversity index. An initial evaluation of Ohio EPA fish community narrative evaluations and Ohio Department of Natural Resources Scenic Rivers volunteer monitoring data revealed similar but more pronounced biases. Hilsenhoff (1990) recognized that such coarse assessments, although less expensive, result in less precise and discriminating results.

The impact of the type of biological evaluation used can be quite striking, particularly in broad-scale assessments such as the biennial 305b report. In the 1986 Ohio 305b report, judgments about use impairments were based largely on narrative biological assessments. Statewide results included:

- Nonattaining waters at 9 percent,
- Partial attainment at 30 percent, and
- Full attainment at 61 percent.

In 1988, Ohio used quantitative, numerical biological criteria employing multimetric evaluation mechanisms based on a regional reference site derivation process. The waterbodies assessed in the 1986 305b report were re-evaluated in addition to the new assessments completed

In 1987 and 1988, 44 percent of the waters were in nonattainment with only 34 percent fully attaining.

The marked increase in nonattaining waters between 1986 and 1988 was not wholly a result of poorer water quality but rather the different methods employed. Not only were the numerical criteria capable of more accurately assessing impairment, but the types of environmental problems that could be assessed were expanded to include more subtle nonchemical and nontoxic chemical impacts. In this example, the same data were analyzed in different ways. The aforementioned discrepancies would likely have been further compounded if methods of data collection had also changed.

This example not only illustrates the usefulness of the regional reference site approach, but also the importance of making the correct initial data collection decisions early in the monitoring process. A misplaced preoccupation with minimizing the cost of data collection could have some unfortunate consequences later in the process.

5. Does the collection and analysis of biosurvey data delay NPDES permits?

This question is more rhetorical than real since the lack of ambient environmental data seldom supersedes a regulatory agency's schedule for issuing National Pollutant Discharge Elimination System (NPDES) permits. However, if the proper organization of monitoring and NPDES issuance is achieved, neither need be a major concern.

Recently, Ohio implemented a rotating five-year basin approach to monitoring and NPDES permit reissuance. This approach allows enough lead time to ensure that biosurvey and other important information such as bioassays, chemical data, and Form 2C are available in time to support the drafting and issuance of NPDES permits. In Ohio, biosurvey data are deemed necessary for only a fraction of the NPDES permits issued. Prioritization and direction of resources are also important since resources are insufficient to monitor everywhere.

Within the five-year approach, some issues are evaluated every five years whereas other issues are evaluated on a 10-year or even 15-year rotation. Inevitably "fire drills" do occur and are responded to as needed. Ohio EPA can respond to specific requests—including both fish and macroinvertebrate field sampling, laboratory analysis, and data processing according to Ohio EPA protocols and procedures—on a one-week turnaround schedule (Ohio Environ. Prot. Agency, 1987, 1989b).

Conclusions

While the value and need for biological assessment have recently been recognized (U.S. Environ. Prot. Agency, 1990), many questions remain concerning the details of deriving and including biological criteria in State water quality standards regulations. Ohio EPA has attempted to answer five of the most commonly asked questions about the States' biological criteria. Some of the most important findings efforts have been:

- Biological criteria have a broad ability to assess and characterize a variety of chemical, physical, and biological impacts and detect cumulative impacts;
- Biological and integrated chemical-toxicity assessments can serve a broad range of environmental and regulatory programs, including water quality standards, NPDES permitting, nonpoint source management and assessment, natural resource damage assessment, habitat protection, and any other surface water efforts where aquatic life protection is a goal;
- Integrated approaches to surface water resource assessment yield more environmentally accurate results;
- Nontoxic and nonchemical causes of impairment predominate in Ohio; and
- Narrative and numerical-based biological assessment approaches differ widely in precision and accuracy.

The latter finding seems particularly important given the policy concerns about use of biosurvey data and biological criteria in the regulatory process. EPA favors an independent approach in the application of chemical-specific, bioassay and biosurvey results (U.S. Environ. Prot. Agency, 1990). Others have proposed a weight-of-evidence approach, where the weight given to any one assessment tool is considered site-specifically in a risk-based management process (Ohio Environ. Prot. Agency, 1989c). Based on the results of the narrative-numerical comparison, it would seem prudent to require independent application for narrative-based biological assessments, given the error tendencies of that approach. However, a discretionary use of the weight-of-evidence approach could be granted for States that have a fully developed numerical approach based on regional reference sites and multiple organism groups.

States are required to include at least narrative biological criteria in their water quality standards by 1993, but development of a numerical approach is not mandated. However, basing policy discretion on the strength of the biological assessment approach could serve as an incentive for States to develop a numerical system if they want to use the weight-of-evidence policy. This would not only result in a more powerful and environmentally accurate assessment tool for the individual States and EPA but would provide maximum flexibility within the entire water program. Thus, development of the more detailed numerical system would benefit both EPA's and individual State's environmental awareness and program flexibility.

References

Ballantine, R.K. and J.L. Guarraia, eds. 1975. The Integrity of Water: A Symposium. U.S. Environ. Prot. Agency, Washington, DC.

Cairns, J. 1986. Freshwater. *In* Proc. Workshop on Cumulative Environ. Effects: A Binatl. Perspective. Can. Environ. Assess. Res. Counc., Ottawa, ON. and Natl. Resourc. Counc., Washington, DC.

Davis, W.S. and A. Lubin. 1989. Statistical validation of Ohio EPA's invertebrate community index. EPA 905/9-89/007. Pages 23-32 *in* W.S. David and T.P. Simon, eds. Proc. 1989 Midwest Pollut. Biol. Meet., Chicago, IL.

Fausch, D.O., J.R. Karr, and P.R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. Trans. Am. Fish. Soc. 113:39-55.

Gallant, A.L. et al. 1989. Regionalization as a Tool for Managing Environmental Resources. EPA/600/3-89/060. Off. Water, U.S. Environ. Prot. Agency, Washington, DC.

Hilsenhoff, W.L. 1990. Data variability in arthropod samples used for the biotic index. EPA-905-9-90/005. Pages 47-52 *in* W.S. Davis, ed. Proc. 1990 Midwest Pollut. Biol. Meet., Chicago, IL.

Hughes, R.M., T.R. Whittier, C.M. Rohm, and D.P. Larsen. 1990. A regional framework for establishing recovery criteria. Environ. Manage. 14(5):673-83.

Hughes, R.M., D.P. Larsen, and J.M. Omernik. 1986. Regional reference sites: a method for assessing stream potentials. Environ. Manage. 10:629-35.

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6(6):21-7.

Karr, J.R. and D.R. Dudley. 1981. Ecological perspective on water quality goals. Environ. Manage. 5(1):55-68.

Karr, J.R. et al. 1986. Assessing biological integrity in running waters: a method and its rationale. Ill. Nat. Hist. Surv. Spec. Publ. 5. Urbana.

Lenat, D.R. 1990. Reducing variability in freshwater macroinvertebrate data. EPA-905-9-90/005. Pages 19-32. *in* W.S. Davis, ed. Proc. 1990 Midwest Pollut. Biol. Meet., Chicago, IL.

Mount, D.I. 1987. Comparison of test precision of effluent toxicity tests with chemical analyses. (Unpubl.) U.S. Environ. Prot. Agency, Environ. Res. Lab., Duluth, MN.

Ohio Environmental Protection Agency. 1987. Biological Criteria for the Protection of Aquatic Life: Vol. II. Users' Manual for Biological Field Assessment of Ohio Surface Waters. Div. Water Qual. Monitor./Assess., Surface Water Section, Columbus, OH.

———. 1989a. Addendum to Biological Criteria for the Protection of Aquatic Life: Users' Manual for Biological Field Assessment of Ohio Surface Waters. Div. Water Qual. Plann./Assess., Surface Water Section, Columbus, OH.

———. 1989b. Biological Criteria for the Protection of Aquatic Life: Vol. III. Standardized Biological Field Sampling and Laboratory Methods for Assessing Fish and Macroinvertebrate Communities. Div. Water Qual. Plann./Assess., Ecol. Assess. Section, Columbus, OH.

———. 1989c. Ohio EPA Policy for Implementing Chemical Specific Water Quality Based Effluent Limits and Whole Effluent Toxicity Controls in NPDES Permits. Div. Water Pollut. Control/Water Qual. Plann. Assess., Columbus, OH.

———. 1990a. Ohio Water Resource Inventory. Exec. Summ., Vol. I. E.T. Rankin, C.O. Yoder, D. Mishne, eds. Div. Water Qual. Plann./Assess., Ecol. Assess. Section, Columbus, OH.

———. 1990b. Uses of Biocriteria in the Ohio EPA Surface Water Monitoring and Assessment Program. Div. Water Qual. Plann./Assess., Ecol. Assess. Section, Columbus, OH.

———. 1990c. The Cost of Biological Monitoring. Div. Water Qual. Plann./Assess., Ecol. Assess. Section, Columbus, OH.

Omernik, J.M. 1987. Ecoregions of the conterminous United States. Ann. Am. Ass. Geogr. 77:118-25.

Omernik, J.M. and A.L. Gallant. 1988. Ecoregions of the Upper Midwest States. Map (scale 1:2,500,000). EPA/600/3-88/037. U.S. Environ. Prot. Agency Res. Lab, Corvallis, OR.

Rankin, E.T. and C.O. Yoder. 1990. The nature of sampling variability in the index of biotic integrity (IBI) in Ohio streams. EPA-905-9-90/005. Pages 9-18 *in* W.S. Davis, ed. Proc. 1990 Midwest Pollut. Biol. Meet., Chicago, IL.

Stevens, J.C. and S.W. Szczytko. 1990. The use and variability of the biotic index to monitor changes in an effluent stream following wastewater treatment plant upgrades. EPA-905-9-90/005. Pages 33-46 *in* W.S. Davis, ed. Proc. 1990 Midwest Pollut. Biol. Meet., Chicago, IL.

U.S. Environmental Protection Agency. 1985. Technical support document for water quality-based toxics control. Off. Water Enforc. Permits, Off. Water Reg. Stand., Washington, DC.

———. 1990. Biological Criteria: National Program Guidance for Surface Waters. EPA-440/5-90-004. Criteria/Stand. Div., Off. Water Reg./Stand., Washington, DC.

Whittier, T.R. 1987. The Ohio Stream Regionalization Project: A Compendium of Results. EPA/600/3-87/025. Environ. Res. Lab., U.S. Environ. Prot. Agency, Corvallis, OR.

Yoder, C.O. 1989. The Development and Use of Biological Criteria for Ohio Surface Waters. Pages 139-46 *in* Proc. Water Qual. Stand. 21st Century, U.S. Environ. Prot. Agency, Criteria/Stand. Div., Washington, DC.