Biological Assessment and Criteria

Tools for Water Resource Planning and Decision Making

Edited by
Wayne S. Davis
Thomas P. Simon

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CHAPTER 21

Policy Issues and Management Applications of Biological Criteria

Chris O. Yoder

1.0 INTRODUCTION

1.1 Overview: Should Biocriteria Be Part of State Water Programs?

Biological criteria as a part of ambient chemical, physical, and biological assessments (i.e., biosurveys) have a potentially broad role in many surface water resource management and policy issues as has been demonstrated throughout this volume (Karr, Chapter 2; Bode and Novak, Chapter 8; Courtemanch, Chapter 20; DeShon, Chapter 15; Yoder and Rankin, Chapter 9, 17; Rankin, Chapter 13). Despite these proven uses the role of biological criteria in surface water resource management and policy has yet to be fully implemented by most states and USEPA. This chapter is an overview of the management and policy uses of biological criteria in Ohio and an examination of the challenges to broader usage throughout the United States.

Recently, the cumulative costs associated with environmental mandates, many of which are the result of prescriptive regulations, have come into question. Former USEPA Administrator William 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 pollution control 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 administrative dominated direction of the traditional surface water regulation strategies partially emanates from the belief that it is neither practical nor feasible to directly measure compliance with the CWA goal of biological integrity (Jorling 1977). Since concise and practical frameworks for using direct biological measures were not forthcoming in the early days of the CWA, surrogate approaches were relied upon. Two reasons for this include the perception that direct biological information is simply not obtainable from a technical and resource/cost standpoint, and natural biological communities are simply too complex to measure and too poorly understood to use. The alternative was to use surrogate indicators of *potential* impairment as the basis for regulation and to provide feedback about current and future conditions. However, a continued reliance on this approach *alone* is questionable when considering the growing body of information that demonstrates the usefulness of the presently available bioassessment approaches.

The 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 that 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 and bioassay techniques as these will *always* 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 nonchemical impacts, and a broadened capability to implement management strategies for protecting and restoring watersheds.

In both the 1988 and 1990 Ohio Water Resource Inventories (305[b] reports) a comparison of the indications of aquatic life use attainment/nonattainment with the Ohio EPA numeric biocriteria was made with ambient chemical indications of the same. The ambient database for this analysis was generated from biosurveys dating to the late 1970s (Ohio EPA 1988, 1990b). Out of 645 waterbody segments analyzed, biological impairment was evident in 49.8% of the cases where no impairments of chemical water quality criteria were observed (Ohio EPA 1990b). While this discrepancy may at first seem remarkable, the reasons for it are many and complex. Biological communities respond to and integrate a wide variety of chemical, physical, and biological factors in the environment of both natural and anthropogenic origin. Simply stated, controlling chemical water quality *alone* does not assure the ecological integrity of water resources (Karr et al. 1986). The results of this analysis indicate not only the broad ability of the aquatic biota to reflect and integrate multiple chemical, physical, and biotic influences, but also the more important issues of accuracy and comprehensiveness within a state water quality management program.

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

1.2 Goals of State Water Quality Standards

A principal objective of the CWA is to restore and maintain the biological integrity of the nation's surface waters. Although this goal is fundamentally biological in nature, the specific methods by which regulatory agencies have attempted to reach this goal have been predominated by such nonbiological measures as chemical/physical water quality (Karr et al. 1986). The presumption is that improvements in chemical water quality will be followed by a restoration of biological integrity. This approach does not directly measure the ecological health and well-being of surface waters nor does it follow the definition of pollution in Section 502 of the CWA as "man-made 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 as the cornerstone of regulatory efforts towards attaining the biological integrity goal of the CWA has become so ingrained into the system that some interesting misconceptions about water quality standards and CWA goals have arisen.

Water quality can easily become a confused and nebulous concept, especially when no demonstrable or tangible endproduct can be identified. Regulators assert that the attainment of administrative goals will logically be followed by actual environmental improvements. But how can this be verified? Do we simply continue to assume that improvement occurs without making an effort to confirm this with environmentally based measures? 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 that have actually resulted in *increased* damage to the environment (Ohio EPA 1992a) because of a reliance on these sometimes-flawed presumptions.

1.3 The Role and Applicability of Biological Criteria

The existing status of the biota resident in any surface waterbody is the integrated result of complex and interrelated chemical, physical, and biological processes over time and is the summation, or result,

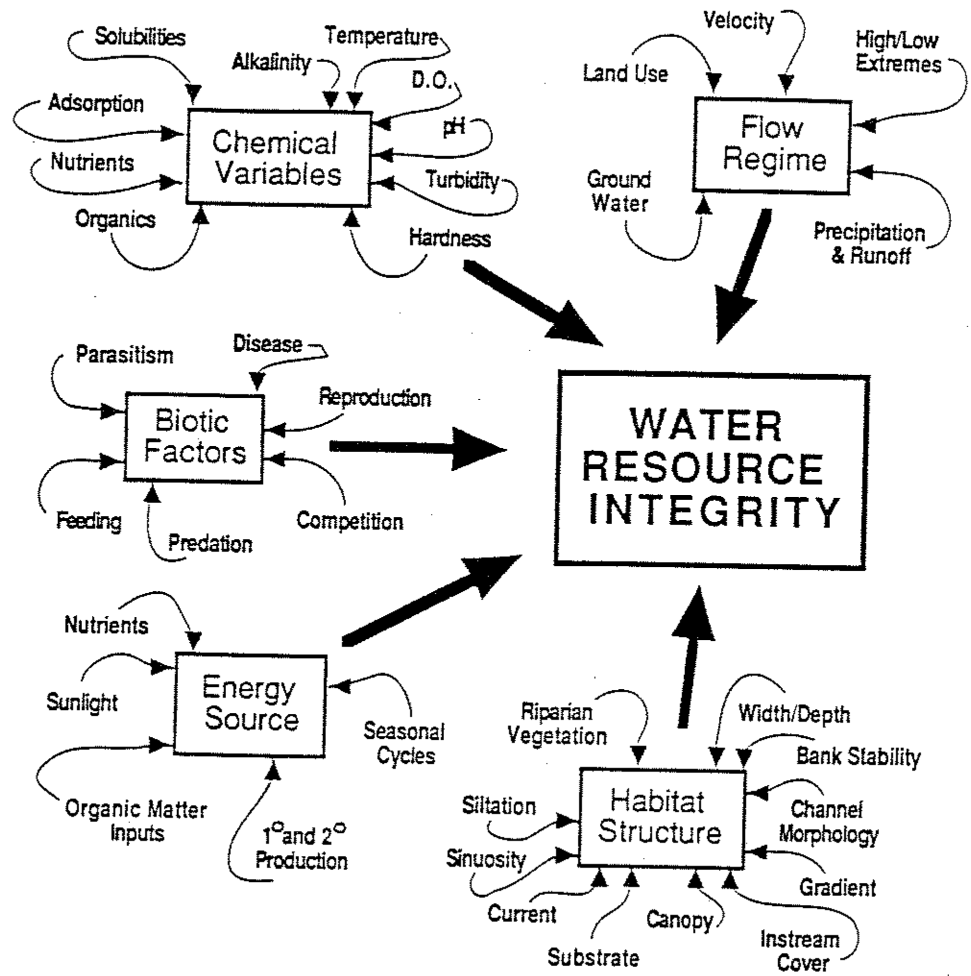

Figure 1. The five principal factors, with some of their important chemical, physical, and biological components that influence and determine the integrity of surface water resources. (Modified from Karr et al. 1986.)

of these processes in their dynamic sequences (Figure 1). Biological communities are precise indicators of actual conditions since they inhabit the receiving waters continuously and are subject to the array of chemical and physical influences that occur over time including both common and extreme events. This includes all of the chemical and physical variables that are commonly measured by most ambient monitoring programs plus additional important variables that are frequently not measured. If these chemical, physical, and biological variables are considered as "pieces" then the resultant biological condition is the integrated result of the assembly of these pieces in the proper dynamic sequence. In this sense biological criteria represent a top-down evaluation where the end product (biological community performance) is used to characterize the summed or integrated result of *all* the pieces (i.e., the chemical, physical, and biological processes that affect biological performance). By comparison the reductionist chemical/toxicity approach represents a bottom-up evaluation where *some* of the pieces are used in an attempt to simulate, predict, or explain complex processes using surrogate end points.

Adopting an increased reliance on direct measurements of biological community performance to establish regulatory direction and priorities may require a modification of some current regulatory attitudes and approaches. In addition to attempting to estimate a protection level for the end point of concern (i.e., biological integrity) via the chemical-specific and/or narrative (i.e., "free from") approaches, this process should ultimately involve the development of pollution abatement strategies to achieve or maintain biological community end points by having *prior* quantitative knowledge about that end point. 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.

USEPA regulations (40 CFR Parts 35 and 130) encourage the use of ambient biological data in water quality decision making. The USEPA technical guidance manual for performing wasteload allocations (USEPA 1984a) specifically states that it is preferable to coordinate the determination of Total Maximum Daily Loads (TMDL) with a biological survey because:

> As the numerical criteria of water quality standards are mostly derived from single species laboratory tests, an observation that a criterion is violated for a certain time period may provide no indication of how the integrity of the ecosystem is being affected. In addition to demonstrating the impairment of use, a biological survey, coordinated with a chemical survey, can help in identifying culprit pollutants and in substantiating the criteria values. The resulting database may also provide information transferable to other sites.

Any of these liabilities are further compounded when no violations of chemical water quality are detected especially considering our findings in comparing the relative abilities of chemical criteria and biocriteria to reveal impairments (Ohio EPA 1990; Yoder 1991a). As water quality management expands into watershed-level applications the need for biocriteria-based assessment becomes even more urgent. These shortcomings are serious enough for point sources, but are further compounded with nonpoint and intermittent sources of pollutants (e.g., combined sewer overflows, urban storm water discharges, spills/dumping, etc.). Nonpoint sources tend to be predominated by natural constituents (e.g., nutrients, sediments, etc.), overlap with physical impacts (e.g., habitat modification, riparian zone degradation), and are frequently more subtle than are point sources. Even with near-continuous chemical monitoring the need remains to interpret the biological meaning of the chemical results. Simply put, biological communities are broader indicators of environmental problems than is chemical sampling *alone* because they reflect the integrated dynamics of the chemical, physical, and biological processes that are constantly at work in aquatic ecosystems (Figure 1).

No single monitoring component can "do it all," particularly in the more complex situations (i.e., multiple discharges, habitat alterations, presence of toxic compounds, etc.). A lack of information from any one component or an overreliance on a single component can result in environmental regulation that is less accurate and potentially underprotective 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 all monitoring components. Prioritizing the use of these components must be based on experience, existing information, and best professional judgement.

2.0 POLICY AND PROGRAM APPLICATIONS

2.1 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 purview of water quality management and that efforts to attain the goals espoused by the CWA and other initiatives (e.g., maintenance and recovery of aquatic biodiversity) ought to recognize the 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) as it has the demonstrated ability to be useful in virtually any issue involving water resources where a goal is to protect, enhance, or restore aquatic communities and ecosystems. We believe that biological criteria and the attendant concepts of regionalization and reference sites have a broad applicability beyond the CWA.

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),

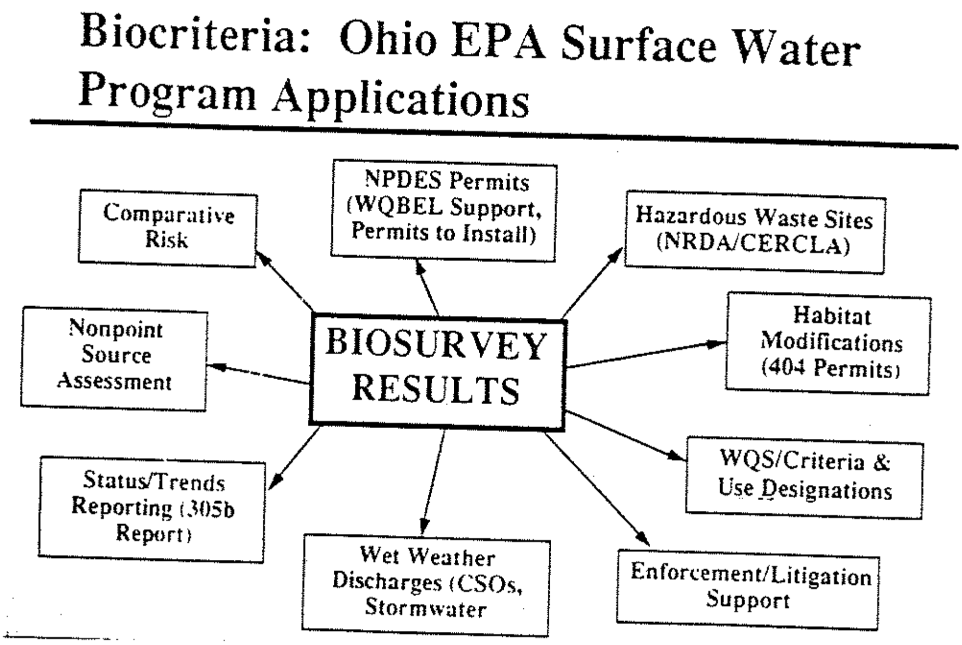

Figure 2. The various environmental management programs at Ohio EPA which are supported by information from biological surveys (biosurveys).

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 2). 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, and fisheries management) and the Ohio Department of Transportation (environmental impact statements). Some of these applications were discussed in detail by Yoder and Rankin (Chapter 9).

2.2 Technical Concerns With Biocriteria

Some of the technical concerns that have been expressed by USEPA (1991a; Jackson 1992) and others (Schmidt 1992) about the application of biocriteria include:

1. Bioassessments are more costly than chemical-specific assessments or bioassays and data interpretation is more difficult.
2. Biocriteria and bioassessment have not been sufficiently developed for uniform application and specified levels of protection are lacking.
3. The scientific basis of biocriteria remains untested over a sufficiently broad range of conditions.
4. Bioassessments and biocriteria are not applicable to other water use concerns such as human health, wildlife health, fish consumption, etc.
5. Biocriteria, chemical criteria, and whole effluent toxicity each measure different end points.
6. Bioassessments only provide a short-term evaluation and may not encompass critical conditions (i.e., the $Q_{7,10}$ flow used to portray critical conditions for wasteload allocation purposes).
7. The ability of biocriteria to indicate cause-and-effect relationships and instream response has not been sufficiently established.
8. Criteria are lacking for what constitutes a sufficiently comprehensive bioassessment.
9. Bioassessments are limited in their ability to define unimpaired waters, i.e., no definitive statement pertaining to unimpaired status can be made with bioassessments.
10. Bioassessment lacks a predictive capability as compared to chemical-specific and whole effluent tests.

Based on our experiences in Ohio and familiarity with other state efforts we take some issue, to varying degrees, with all of the above stated reservations.

2.2.1 Cost and Resource Issues

USEPA (1985, 1991a) and others previously have pointed to the "high" costs associated with biological surveys. Chemical surveys are viewed by USEPA (1991a) as being cheaper because of market availability. Independently, Yoder and Rankin (Chapter 9) examined the relative cost of chemical-specific, bioassay, and biosurveys and found that this was not the case. In fact, chemical assessment was the most expensive of the three tools examined based on costs incurred by Ohio EPA. Costs for chemical analysis will continue to increase not only because basic analytical costs are rising, but also because the number of analyses and parameters required is likewise increasing. As long as a cost-effective bioassessment approach is used, cost should not be a concern.

2.2.2 Biocriteria Development and Scientific Basis

Criticisms 2 and 3 pertain to the apparent lack of a framework and procedures for developing biocriteria, establishing specified levels of protection, and the untested scientific basis for biocriteria. These criticisms tend to ignore the significant progress made in a number of important areas, not the least of which has been the operational definition of biological integrity (Karr and Dudley 1981). Add to this the development of robust and information-rich multimetric indices, regionalization, and tiered aquatic life uses that are embodied in the peer-review literature (Karr et al. 1986; Hughes et al. 1986), USEPA guidance documents (e.g., Plafkin et al. 1989; Gibson 1994), and state frameworks (Davies et al. 1993; Ohio EPA 1987, 1989a, 1989b; Bode and Novak, Chapter 8; Courtemanch, Chapter 20).

Specified levels of protection defined by Yoder and Rankin (Chapter 9) employ threshold criteria that are not unlike those that have routinely been used in chemical-specific and whole effluent toxicity applications. In addition, biocriteria are not subject to the assumptions necessary in the chemical-specific criteria derivation and application process thus the need for artificial safety factors is greatly diminished. The fundamental nature of the calibration of multimetric indices and the biocriteria setting process each contain sufficient analogs to the safety factors specified as being necessary for water quality criteria by the Section 303 of the CWA.

With regard to the untested scientific basis of biocriteria we point first to the long history of water pollution biology. While the early studies may not "look" like what is being proposed today, the fundamental tenets of using the indigenous biota as an indicator of aquatic ecosystem quality is well developed. The usefulness of biocriteria as an ambient assessment tool to show impairment (Ohio EPA 1990a; Yoder 1991b) and portray changes over time (Ohio EPA 1992b) demonstrates the ability to fulfill basic and practical needs. In addition, we have shown extensively throughout Ohio the different and consistent patterns of response to different types of impacts (Yoder and Rankin, Chapter 17). We firmly believe that many of these patterns, and more importantly the basic concepts, are transferable to other states in the Midwest and possibly elsewhere. This is also something that can be further refined as the process evolves much like it has with the chemical-specific and whole effluent toxicity tools.

2.2.3 Applicability of Biocriteria

We agree that bioassessments are not directly applicable to other uses including human health, wildlife health, and other nonaquatic life uses (criticism 4). However, biocriteria should not be held accountable for failing to perform tasks other than that for which they were originally designed. We would be remiss in failing to point out that healthy aquatic communities, though, occur in waters where other uses such as public water supply, recreation, and wildlife flourish. With respect to the latter, the health of the ecosystem as portrayed by the instream biota is more often than not relevant to the habitat requirements of most aquatic and semi-aquatic avian, reptilian, amphibian, and mammalian species. For example, the riparian zones associated with our high-quality streams and rivers correlates well with critical habitat for a number of Ohio breeding birds (Ohio EPA 1992b).

2.2.4 Measurement End Points and Inferences About WQS Goal Attainment

In the strictest technical sense we agree with criticism 5 that each of the three tools measure different end points. However, this is not entirely germane to the debate over biocriteria applicability. With regard to the very practical and widespread purpose of each tool to indicate aquatic life use impairment, each is attempting to measure the *same* end point. Is each tool equally capable of determining whether a particular waterbody is attaining a designated aquatic life use? It is unrealistic to expect three different approaches to each have the same power of assessment, simply because they measure different end points. Our comparisons of ambient chemical assessment, acute bioassays, and instream bioassessment (Ohio EPA 1990a; Yoder 1991a) showed extensive disagreement between the three tools with regard to reflecting impairment. Furthermore, the inherent error tendencies of the ambient chemical and bioassay methods are to miss or underrate impairments. While none of the three tools are perfect measures of actual conditions, ambient bioassessment and biocriteria are the nearest to being a direct measurement. Instead of pitting the strengths and exploiting the weaknesses of each tool we would be better served in the long run by defining under what conditions each of the three tools are the most powerful. The current debate seems to result more in diminishing the strengths of each tool by simply equating their status in policy matters.

A frequent criticism of bioassessments and biocriteria has been the relevance of results obtained at stream and river flows other than the critical design flows (criticism 6) used in the TMDL process, i.e., the $Q_{7,10}$ flow. One problem with the logic behind this assertion is that biological communities do not respond along a linear continuum with flow and chemical concentrations, but rather are more threshold oriented. This assertion does not make ecological sense in that communities that are functionally intact can withstand extreme conditions; thus, the history of coping with such rare events should be reflected in the community performance measured at other flows. We would also point to the characteristics of intact communities outlined by Karr et al. (1986), particularly the capacity for self-repair. This is not to say that certain elements of an aquatic ecosystem cannot become unacceptably stressed over the short term during critical conditions. We frequently observed such stress in our results, particularly in the functional metrics of the biocriteria indices. However, the observation of an intact community following such events indicates that environmental health has been maintained to the point that no discernable effects took place or that the capacity for self-repair was not exceeded. We agree with Stephan et al. (1985) that a community should not be kept in a perpetual state of recovery. However, a community undergoing recovery would yield indications of this state at times other than critical low flows. It would be irresponsible to categorically deny the importance of employing critical flows as NPDES permit design criteria, but this is much different than maintaining that bioassessments are valid only under these critical low flows.

2.2.5 Cause-and-Effect Relationships

Another criticism (number 7) is with the perceived inability of the biota to determine cause-and-effect relationships, i.e., bioassessments can detect impairment, but provide no insights as to the sources or causes (Suter 1993). While this hypothesis seems plausible there are consistent patterns that emerge from the rich information contained in biological data, as we have demonstrated with the biological response signatures (Yoder and Rankin, Chapter 17). Others (e.g., Eagleson et al. 1990) have also observed discernable patterns in biological community data. Indeed, bioassessment has some very distinct limitations in pinpointing specific causative chemicals and, in complex areas, specific sources. However, bioassessments, when properly planned, designed, and implemented are not performed in a vacuum. Proximity to sources, source loadings, chemical results, toxicity tests, sediment chemistry, and habitat quality are all important factors and we routinely collect this type of information during biosurveys. Hence the integrated use of chemical, physical, and biological data. Even with this information at hand, the resolution of all cause-and-effect relationships is not completely accomplished in one or even multiple years of monitoring. However, this should not be construed as a failure of the process, but rather a statement of the complexity of some situations and the inherent limitations of all tools. The strength of bioassessment in these situations is the feedback provided in terms of an ambient, instream reality check on the application of the chemical-specific and whole effluent tools. For many situations this will only become evident through an iterative process.

2.2.6 Bioassessment Levels

The question about what constitutes a sufficiently comprehensive bioassessment (criticism 8) is another key contemporary issue facing the implementation of bioassessments and biocriteria. Yoder (1994) discussed the relative abilities and biases of different levels of bioassessment to discriminate varying levels of aquatic community performance. A hierarchy of bioassessment types was identified and is related to the number and complexity of data dimensions generated by each (Table 1). The need to recognize the existence of different levels of bioassessment is illustrated by the experience of Ohio EPA in their reporting of the status of stream and river miles attaining and not attaining designated uses (i.e., Clean Water Act goals) between the 1986 and 1988 Ohio Water Resource Inventories (305[b] report). Because of a change in the type of bioassessment used between the two 305(b) report years from level 5 to level 8 (Table 1), the miles of streams and rivers *failing* to attain their designated uses changed from 9% in 1986 to 44% in 1988, an *increase of nearly five times*. This remarkable change illustrates the important influence that the differing capabilities of the various bioassessment types listed in Table 1 can exert on the relative level of accuracy of an assessment and the need to categorize and classify each according to their respective abilities to detect and discriminate impairments both spatially and temporally.

2.2.7 Basic Hypothesis Testing and Bioassessments

USEPA (1990a, 1991f) employs the null hypothesis that there is an effect on water resource integrity, and the alternative hypothesis that there is no effect. Rejection of the null hypothesis is not interpreted to mean that the alternative is accepted since the initial rejection is not entirely decisive, but leaves the possibility that there is an effect too small to detect. In fact, others (Schmidt 1992) maintain that there is always an effect no matter how difficult it is to measure, with the implication that it is always significant. This approach can lead to the presumption that only negative findings, i.e., those that indicate an adverse effect, are the only valid findings. Thus, any findings of no effect are potentially invalid and, if taken to extremes, could mean that showing *attainment* of a goal or standard is a statistical impossibility.

There are some practical problems with this position: (1) it is impractical in a regulatory and administrative framework to become incapable of affirming that a goal or standard has been attained; (2) this approach assumes that each assessment tool is a unique and perfect indicator of adverse effects in the sense of CWA goal attainment (Ruffier 1992); and (3) this approach tends to dismiss any positive findings from one tool when it is contradicted by the negative findings of another. Miner and Borton (1991) argued that some effects are indeed insignificant in practical terms. There is precedence for this type of reasoning, e.g., in the declaration by USEPA under the general permitting regulations that certain discharges are classified as *de minimis*, hence they have little risk of a *significant* impact and are subjected to comparatively less scrutiny in the regulatory process.

In the application of the three tools the relative strengths (not weaknesses) of each need to be emphasized in determining the extent to which we should trust a finding of no adverse effect. Based on comparisons of the performance of each tool (Ohio EPA 1990a,b; Yoder 1991a) this should be a very different approach for the chemical-specific and whole effluent results than for bioassessment results. Because of the inherent error biases of each tool, a finding of no adverse effect based on the chemical-specific and whole effluent tools should be regarded as inconclusive without a concurrent finding of full attainment from the bioassessment. It is difficult for a bioassessment, assuming an adequate bioassessment level and biocriteria framework is employed, to give a false indication of attainment since the basic method is contingent upon finding a sufficient number and the right types of organisms. The error propensity inherent to bioassessment is to fail to find enough of the right organisms because of poor sampling and ignorance of the admonishments concerning certain prohibitive field conditions, and, hence, the false determination of an adverse effect. This error propensity is the *opposite* of the chemical-specific and whole effluent tools, which are prone to missing effects, particularly if they are episodic and/or sampling is conducted at an insufficient frequency. Thus, the use of bioassessments and biocriteria is viewed as a safeguard against being "fooled" by a showing of no effect by chemical and physical measures. Simply stated, the three tools are not equal in their respective abilities to detect adverse effects

Table 1. Hierarchy of Ambient Bioassessment Approaches That Use Information about Indigenous Aquatic Biological Communities

Bioassessment Type	Skill Required[1]	Organism Groups[2]	Technical Components[3]	Ecological Complexity[4]	Environmental Accuracy[5]	Discriminatory Power[6]	Policy Restrictions[7]
1. Stream walk (visual observations)	Nonbiologist	None	Handbook[8]	Simple	Low	Low	Many
2. Volunteer monitoring	Nonbiologist to technician	Invertebrates	Handbook[9], simple equipment	Low	Low to moderate	Low	Many
3. Professional opinion (e.g., RBP Protocol V)	Biologist w/experience	None or fish/inverts.	Historical records	Low to moderate	Low to moderate	Low	Many
4. RBP Protocols I and II	Biologist w/training	Invertebrates	Tech. manual,[10] simple equip.	Low to moderate	Low to moderate	Low to moderate	Many
5. Narrative evaluations	Aquatic biologist w/training and experience	Fish and/or inverts.	Std. methods, detailed taxonomy specialized equip.	Moderate	Moderate	Moderate	Moderate
6. Single dimension indices	(Same)	(Same)	(Same)	Moderate	Moderate	Moderate	Moderate
7. Biotic indices (HBI, BCI, etc.)	(Same)	Invertebrates	(Same)	Moderate to high	Moderate to high	Moderate	Moderate to few
8. RBP Protocols III and V	(Same)	Fish and inverts.	Tech. manual,[10] detailed taxonomy, specialized equip., *dual* organism groups	High	Moderate to high	Moderate to high	Few
9. Regional reference site approach	(Same)	Fish and inverts.	Same plus baseline calibration of multimetric indices and *dual* organism groups	High	High	High	Few
10. Comprehensive Bioassessment	(Same)	All organism groups	Same except *all* organism groups are sampled	Highest	High	High	Few

Note: This applies to aquatic life use attainment only — it does not apply to bioaccumulation concerns, wildlife uses, human health, or recreation uses.

[1] Level of training and experience needed to accurately implement and use the bioassessment type.
[2] Organism groups that are directly used and/or sampled; fish and macroinvertebrates are most commonly employed in the midwestern states.
[3] Handbooks, technical manuals, taxonomic keys, and data requirements for each bioassessment type.
[4] Refers to ecological dimensions inherent in the basic data that is routinely generated by the bioassessment type.
[5] Refers to the ability of the ecological endpoints or indicators to differentiate conditions along a gradient of environmental conditions.
[6] The relative power of the data and information derived to discriminate between different and increasingly subtle impacts.
[7] Refers to the relationship of biosurveys to chemical-specific, toxicological (i.e., bioassays), physical, and other assessments and criteria that serve as surrogate indicators of aquatic life use attainment/non-attainment.
[8] Water Quality Indicators Guide: Surface Waters (Terrell and Perfetti 1989)
[9] Ohio Scenic River Stream Quality Monitoring (Kopec and Lewis 1983).
[10] U.S. EPA Rapid Bioassessment Protocol (Plafkin et al. 1989).

and each has varying degrees of uncertainty; thus, the foundation for relying too much on the null hypothesis concept can be flawed. If nothing else, the position that a significant effect is always present despite the showing of no impairment by a bioassessment or other test, will become increasingly difficult to defend.

2.2.8 Predictive Abilities of Bioassessment

Bioassessments and biocriteria are widely perceived as not being predictive in the sense that a specified level of protection can be anticipated in the same manner as using chemical-specific criteria via the TMDL process or whole effluent toxicity test results through the use of toxic unit limitations (Suter 1993). The concerns expressed by some is that while biocriteria are a valid and useful measure of instream impairment, by the time it is detected the environmental damage is already done. While this may be true, we argue here that a significantly larger number of environmental problems will go undetected if bioassessments and biocriteria are not widely used by states.

There are some problems with the perception of a lack of predictive ability. Rankin (1989; Chapter 13) developed a relationship between the IBI and the Qualitative Habitat Evaluation Index (QHEI) sufficient to forecast nonattainment due to habitat degradation with a reasonable degree of certainty. This is most applicable to the review of proposed activities that would alter riparian and instream habitat. By knowing the present habitat condition using the QHEI, changes can be projected to the various QHEI attributes based on the details of the proposed activity. It can then be determined if there is a reasonable potential to degrade the habitat so that the applicable biocriteria will be violated.

Another example of a predictive capability is with the development of a model relationship between riparian corridor condition and percentage of urban land use (Steedman 1988). The model was "calibrated" using data from a cross section of watersheds and developing a linear relationship with the two landscape attributes as covariates. The usefulness of this to watershed planning is obvious since quantifiable estimates of land use and riparian corridor compatible with IBI values that attain a particular use or goal can be made with a known degree of certainty. The predictive abilities of biocriteria lie primarily in establishing precedents in the ambient environment. Thus, if we are to use and improve biocriteria as a predictive tool, robust databases will need to be amassed in order to develop model relationships.

2.3 Policy Applications of Biological Criteria

This is perhaps the most controversial and certainly the least understood aspect of biological criteria, at least at the national level. When addressing the policy implications of biological criteria it is important to understand the applications of biocriteria and how this overlaps with the uses of the more traditional chemical/physical and toxicological tools and criteria. Biological criteria are largely 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 as a chemical criterion from which effluent limitations for specific chemical substances are derived, even though both employ the common term "criterion." Also, biocriteria are limited to aquatic life use 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 that they were not originally designed to address. Hopefully the following discussions will provide a firmer definition of the appropriate role of biocriteria in a state water quality management program.

There is a consensus that application of bioassessments and biocriteria is 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. USEPA and most environmental groups favor the policy termed "independent application" when considering how to apply the results of bioassessments, ambient chemical data, and whole effluent toxicity. Others, principally states and the regulated community, favor a "weight of evidence" approach. We attempt here to examine the technical issues underlying this debate. Also, we emphasize that the following deals entirely with aquatic life use issues and does not transcend the importance of criteria for persistent toxicants as they pertain to human health, wildlife, and other nonaquatic life uses. These uses are truly independent of the aquatic life concerns in terms of how the criteria are applied.

2.4 Policy of Independent Application

The USEPA policy of independent application was first outlined in the national biocriteria program guidance (USEPA 1990a) and later reaffirmed in the "eco policy" statement (USEPA 1991c) and the revised *Technical Support Document for Water Quality-Based Toxics Control* (USEPA 1991f). A panel discussion at the Third National Water Quality Standards for the 21st Century conference dealt with this issue and included perspectives from USEPA (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/nonattainment, 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 ambient data are being used to assess aquatic life use attainment/nonattainment, and (2) in determining appropriate wasteload allocations for point sources or load allocations for nonpoint 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 and watershed characteristics. In both cases conflicts may arise between the three major assessment tools (chemical-specific, whole effluent, and biocriteria). USEPA's definition of independent applicability means that the validity of the results of any one of the three approaches is independent of any confirmation by the others. Each assessment operates independently with none being viewed as superior or more powerful than another regardless of the situation. USEPA 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 assessments determines that a standard is not attained.

USEPA (1991a) bases some of their equivalency of the biological, physical, and chemical components on a conceptual model of ecological integrity (Figure 3). However, we agree with Karr (1991) that this model is inadequate and not representative of the variable interaction of the three key components. We offer an alternative model, which shows that the overlapping influence of the three major components is not only disproportionate, but is dynamic (Figure 3). This depends on a given situation relative to the five major factors that influence water resource integrity (see Figure 1); thus, the influence of any one component will likely be disproportionate to the others in most situations. We also believe that for the purpose of aquatic life protection the biological component will dominate the other two because it is the product of the integrated interaction of the chemical and physical components.

2.5 The Case for Weight-of-Evidence

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 *a priori*, 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 prior decision about the independence of one tool from the others is made.

Based on the evidence we have examined and our own experiences in conducting bioassessments within a regulatory framework for more than 15 years, we believe there is a case to be made for employing a weight-of-evidence approach *with some important restrictions and limitations.* Unlike the policy of independent application, where all of the tools are considered to have an equal weight and ability, this approach acknowledges and accounts for the attributes of each tool. Weight-of-evidence takes advantage of the strengths of each, emphasizes the role of site-specific data, and promotes controlled flexibility in the process, an attribute that has the advantage of allowing new advances to be incorporated into the process (Ruffier 1992). This approach also seems to be consistent with the USEPA emphasis on ecological risk-based management and the well-known call for ensuring good science in the process. The policy of independent application is admittedly a regulatory approach (Jackson 1992) and has the appearance of being administered for the sake of regulatory expediency, allowing water pollution measures to keep pace with science without having to modify the permit program, and without the need to reconcile or justify discrepancies (Ruffier 1992). Independent application certainly simplifies the process and minimizes best professional judgement and site specificity, but at the potential loss of accuracy in the process. Weight-of-evidence does not make assumptions ahead of time, but, rather, relies

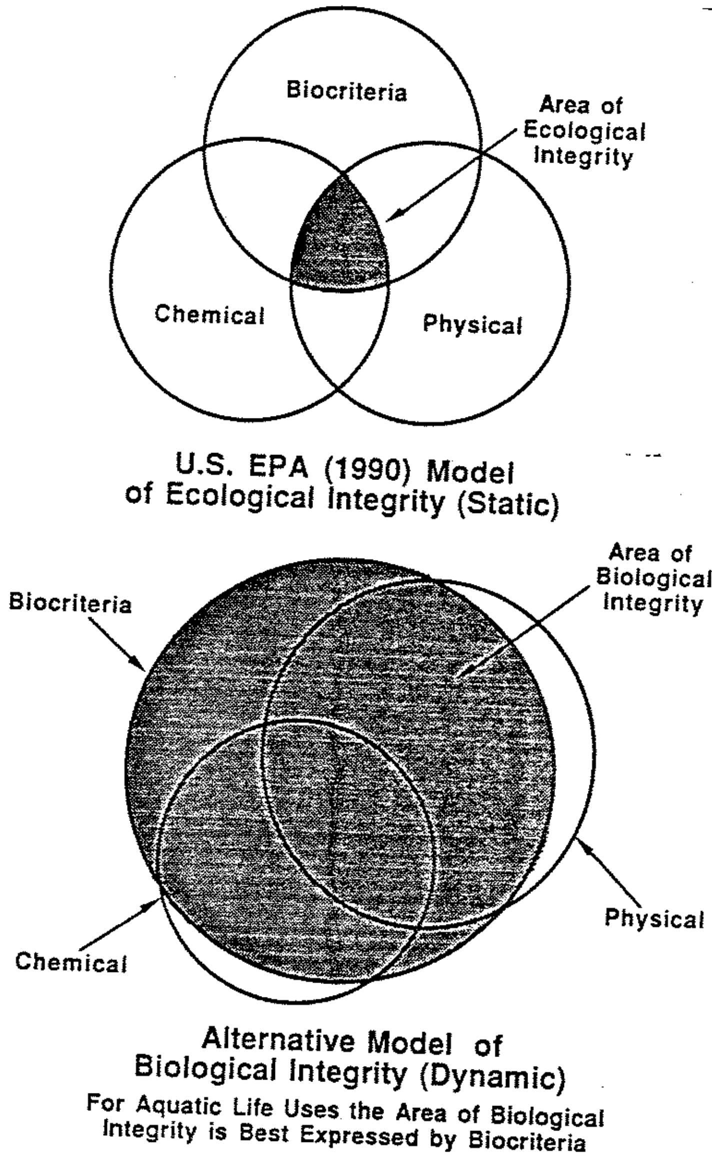

Figure 3. The elements of ecological integrity as envisioned by USEPA (1990a, 1991f; upper) and our view of the dynamic relationship between the biological, chemical, and physical elements of overall biological integrity as it pertains to aquatic life uses in Ohio (lower).

on the strengths of each assessment and application tool in a complimentary manner. One of the most serious concerns of independent application is that it limits the abilities of any one tool and reduces the process to possibly relying on the least powerful. Even though it may appear that the most stringent result is being applied, stringency does not necessarily imply accuracy or legality. Ruffier (1992) ranked each tool in a hierarchical manner as chemical-specific < whole effluent < bioassessment. Although we generally agree with this hierarchy, especially based on ours and others comparison tests, this is not invariable. To make it so in all cases would violate an important precept of weight-of-evidence.

Another concern with independent application is that it does not promote incentives for addressing sources or causes of water resource impairments other than those that are identifiable and controllable by the chemical-specific or whole effluent approaches. Ruffier (1992) presented the situation in which a relatively minor exceedence of a chemical-specific criterion (copper) was *predicted* alongside an *absence* of whole effluent toxicity and instream biological impairment. Independent application would require that the copper exceedence be the driving force behind any regulatory solution, thus essentially omitting the results of the other two tools. Any contribution of information from these tools is excluded from the development of permit limits. Furthermore, the uncertainties (i.e., weaknesses) of the chemical-specific criteria application are amplified because of the inherent liabilities imposed by the *predicted* chemical-specific violation. No investigation into why the discrepancy occurred would be pursued, which could effectively rule out other important information. Under the weight-of-evidence approach the reasons why the discrepancy existed would be investigated prior to initiating regulatory action. It is the *predicted* (as opposed to actual) exceedences of chemical-specific criteria, and subsequent indications of the *potential* for non-attainment, that pose the most frequent conflict. A predicted exceedence would certainly carry more weight if it was accompanied by a measured instream exceedence and more so by an instream biological impairment and/or effluent toxicity. Oftentimes it is the wasteload allocation assumptions, which are inherently conservative, that result in the prediction of an exceedence under specific conditions. If the other tools fail to show any risk of impairment, the reasonable potential to violate may not be so reasonable after all. USEPA (1991f) acknowledges this scenario and recommends that a more detailed and sophisticated monitoring and modeling approach be undertaken in an attempt to bring closer agreement. We certainly endorse this approach as being consistent with weight-of-evidence.

The concern is not solely with the potential for overregulation, but the far greater risk of "hidden" degradation. As was previously pointed out, relegating ambient bioassessment information to an equal place with the chemical-specific and whole effluent tools will lead state program managers to the conclusion that biocriteria are a nonproductive addition and are hence unnecessary. Thus, an opportunity to expose permit writers, plant operators, administrators, and others involved in decision making to more direct information about the aquatic resource will be lost. The NPDES process will essentially remain detached from the ambient environment. Not only will this result in a loss of valuable information, it will likely result in the continued degradation of the aquatic resource because the superior ability of bioassessment and biocriteria to detect and characterize problems will be missing.

Much of the concern about weight-of-evidence is with the potential for misuse and abuse by attempts to justify increasing loadings of pollutants. However, given the ability of the higher level of bioassessment types (Table 1) to detect degradation we doubt that significant increases in pollutant loadings will be justified with biocriteria on a widespread basis. What is more likely is that the magnitude of required pollutant load *reduction* may in a few cases be modified by the application of weight-of-evidence. In our experience the following are the situations where conflicts have arisen in Ohio:

1. Oxygen-demanding substances, ammonia-N, and copper are the most frequently involved parameters. Although the latter is a priority pollutant the chemistry is complex and is likely the root cause of many of the conflicts. We have yet to encounter a situation where a truly bioaccumulative toxicant has been involved; where such toxicants are elevated the bioassessments usually show severe degradation.
2. Conflicts can arise with the non-WWH uses, particularly for the MWH use. Presently the Ohio WQS have chemical-specific criteria for the MWH use different from WWH for dissolved oxygen (D.O.) and ammonia-N only — all other chemical parameters are equivalent to the WWH use designation.
3. The strict adherence to the results of chemical-specific criteria applications without the site and regional specific information fostered by a biocriteria approach can lead to abatement measures which actually result in increased environmental degradation. We observed this in a major suburban area where there was a "need" to eliminate small wastewater treatment plants due to noncompliance with their NPDES permits. This resulted in the destruction of stream habitat by the instream construction of interceptor sewers intended to deliver the sewage flow to newly constructed regional facilities (Ohio EPA 1992a). While this eliminated the "paper compliance" problem of the small plants, biosurveys showed that the zones of degradation below each were small in comparison to the miles of habitat destruction wrought by the sewer line construction and subsequent maintenance activities.
4. A "reverse conflict" can also occur with the application of the EWH use designation. Here the chemical-specific criteria for most substances are the same as the WWH use (D.O. is an exception), but the communities are more sensitive. This has led to efforts to oppose redesignation of stream segments to EWH from WWH based on the argument that the chemical-specific criteria cannot be attained, even though the more difficult-to-meet EWH biocriteria are attained, this being a prerequisite to assigning the EWH use.

With regard to the latter example, the projected inability to meet the EWH D.O. criterion has been presented as an argument for not adopting the EWH use. In order to redesignate a segment as EWH there must be a demonstration of the ability of the segment to attain the EWH biocriteria. Thus, we are faced here with the classic independent application scenario — the biocriteria are attaining, but the chemical-specific criteria are predicted to be exceeded under the critical, worst-case design conditions. However, the stakes in this situation are different than with the other use designations. Do we want to leave a stream segment that is performing at the EWH biocriteria levels vulnerable to future degradation by nonchemical impacts or impacts that are not regulated via the NPDES process? This is the risk if the redesignation to EWH cannot take place because of the *predicted* inability to meet a chemical-specific criterion. Thus, a strict adherence to independent application in this case can leave stream and river segments underprotected and vulnerable to future degradation.

The flaws inherent to the policy of independent application lie primarily in the equating of the three major tools, chemical-specific, whole effluent toxicity, and bioassessment/biocriteria. In many respects this policy is only a more complex facsimile of the long-abandoned USEPA policy of presumptive applicability of the 1976 water quality criteria. The site-specific circumstances relevant to the five factors that 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 are available. Under independent application a decision may only be 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 possibly other tools (e.g., habitat, biomarkers) will be required to successfully employ this policy.

2.6 Recommendations for Resolving the Independent Application/Weight-of-Evidence Conflict

Based on a review of the existing policy debate we believe the conflict can be resolved by establishing program criteria that states must follow in order to gain the desired policy flexibility. There are several advantages to codifying biocriteria in the WQS, not the least of which is the legal standing relative to other criteria. For aquatic life uses the basic narrative descriptions are generally written in biological terms and in the more sophisticated frameworks these qualify as narrative biocriteria. The addition of numeric biocriteria in this scheme then provides the quantitative benchmarks of aquatic life use attainment and nonattainment. For applications to habitat-modifying activities the biocriteria provide a powerful tool for minimizing degradation or in some cases preventing it altogether. This is of critical importance in states like Ohio where habitat and related sedimentation impacts are among the leading causes of impairment (Ohio EPA 1992b). Thus, one of our strongest recommendations is for numerical biocriteria to be part of the states' WQS.

Policy restrictions on the use of biocriteria in the overall water quality management program should be based on the level of bioassessment employed (Yoder 1994; Table 1). This is dependent on the relative sophistication of the bioassessment framework, which consists of methods, level of taxonomy, number of organism groups, number of data dimensions involved, and the biocriteria derivation framework. In short, it is not just the use of biological information, but the level of bioassessment and the framework within which biocriteria are developed that is most important. As the complexity and sophistication of the bioassessment and biocriteria framework increase, policy restrictions should decline accordingly.

This in no way implies an unrestricted use of bioassessments over chemical-specific and whole toxicity approaches. Rather, this approach advocates an informed and integrated use of each tool with the recognition that bioassessment is more likely to detect impairment than the other tools. We also recommend that special consideration be given to reserving policy flexibility for specific classes of pollutants such as conventional parameters (e.g., D.O. and suspended solids), nutrients and ammonia-N, and other "troublesome" parameters such as copper. Nearly all CWA Section 307(a) pollutants would be excluded from such a policy since most are persistent and bioaccumulative, most result in severe impairments, and ambient information about all except the most common heavy metals is generally lacking. While we do not advocate the unrestricted use of a weight-of-evidence approach, we believe a controlled application will be necessary to better deal with some of the emerging problems, particularly with nonpoint source constituents. Another safeguard could include making the use of a weight-of-evidence approach subject to a case-by-case justification similar to a use attainability analyses (USEPA 1984b), which is required prior to the designation of segments for less than a CWA goal use. Requiring a showing of biocriteria attainment for three consecutive years might be another safeguard to ensure that any showing of attainment was not spurious.

The allowance of policy flexibility, which is contingent on using a higher level of bioassessment (Table 1), might serve as an incentive for states to invest the resources necessary to develop a reference site database, numerical biocriteria, and sufficient case histories to fully implement bioassessments and biocriteria. Since the resultant accuracy of each bioassessment type is different, placing policy restrictions on the use of a particular level of bioassessment becomes an important issue. This is an unprecedented area of opportunity for USEPA to resolve some of the state and regulated entity objections to the policy of independent application in that policy flexibility would be influenced by the level of bioassessment. Under this framework policy restrictions decline as the level of bioassessment increases in power and accuracy. This would not only provide an incentive for states to develop adequate biocriteria frameworks, but would also benefit USEPA and the public in general in that the improved bioassessment capabilities would provide a more accurate and comprehensive assessment of the states' and nation's waters. We believe that a rigid adherence to the policy of independent application will discourage already reluctant states to develop adequate bioassessment programs since biocriteria will simply become another layer in the water quality management process (Pifher 1991). This would not only serve the needs of individual states, but would in turn provide a better statewide assessment of water resource conditions, which would lead to a much improved national assessment, something that has been sorely lacking over the past 20 years.

2.7 Other Concerns

Not all of the concerns about biocriteria and bioassessment are being expressed by USEPA and environmental groups. States and the regulated community, while generally in favor of the approach, have also expressed concerns. Cost and resource constraints are most frequently raised by states that are facing an ever increasing burden of mandates without external funding increases. 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, 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 that possess bioassessment capabilities. We do not minimize the difficulties of actually accomplishing this type of interjurisdictional 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. There is little doubt that biocriteria enhance the ability to detect degradation. However, this does not necessarily translate into significantly more stringent limitations. Pifher (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. As we have previously pointed out the biota are more threshold-response oriented and there is a point beyond which additional pollutant removal will have little or no beneficial effect on the biota. 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 subject 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 observed in Ohio, that either the permit limitations are not sufficient, the entity self-compliance monitoring is inadequate, 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.

The regulated community is also concerned about taking on responsibility for conducting the ambient monitoring required to implement biocriteria. We strongly advocate that states have primacy 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 that includes issue beyond point sources such as 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. Additionally, this is not an area for volunteer programs either as these rarely, if ever, have the expertise and resources required to operate the level of bioassessment necessary for a credible biocriteria approach (see Table 1). 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 USEPA must examine the trade-offs between not having an adequate bioassessment capability and having existing impairment remain undetected or underrated. 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.

3.0 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 United States 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 that 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 (Ohio EPA 1992b).

A monitoring approach, integrating biosurvey data that reflects the integrity of the water resource directly, with water chemistry, physical habitat, bioassay, and other monitoring and source information, must be central to accurately defining these varied and complex 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 heavy reliance on administrative activity accounting (numbers of permits issued, dollars spent, or management practices installed) and a preoccupation with chemical water quality *alone* to more integrated and holistic measurements with overall water resource integrity as a goal. Biocriteria seems an essential component in making this shift.

Emphasizing aquatic life use attainment is important because: (1) aquatic life criteria oftentimes result in the most stringent requirements compared to those for the other use categories, (i.e., protection for the aquatic life use criteria should 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 that 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 and combined sewers)
- Nonpoint source assessment and watershed management
- Site-specific criteria modifications
- Regulation of activities that directly impact aquatic habitat

Finally, biocriteria can greatly aid the visualization of aquatic resource values and attributes. This is a critical need if we are to change the prevailing view of watersheds and streams as merely 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 that 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 affect these degrading influences would have been lost.

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