State of Ohio
Environmental Protection Agency

Ecological Assessment Section
Division of Water Quality
Planning & Assessment

Biological Criteria for the Protection of Aquatic Life: Volume I: The Role of Biological Data in Water Quality Assessment

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NOTICE TO USERS

All methods and procedures for the use of biological criteria contained and/or referred to in these volumes supercede those described in any previous Ohio EPA manuals, reports, policies, and publications dealing with biological evaluation, designation of aquatic life uses, or the determination and evaluation of aquatic life use attainment. Users of these criteria and the supporting field methods, data analyses, and study design should conform to that presented or referenced in these volumes (and subsequent revisions) in order to be applicable under the Ohio Water Quality Standards (WQS; OAC 3745-1).

Three volumes comprise the supporting documentation for setting and using biological criteria in Ohio. All three volumes are needed to use the biological criteria, implement the field and laboratory procedures, and understand the principles behind their development, use, and application. These volumes are:

Ohio Environmental Protection Agency. 1987. Biological criteria for the protection of aquatic life: Volume I. The role of biological data in water quality assessment. Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, Ohio.

Ohio Environmental Protection Agency. 1987. Biological criteria for the protection of aquatic life: Volume II. Users manual for biological field assessment of Ohio surface waters. Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, Ohio.

Ohio Environmental Protection Agency. 1987. Biological criteria for the protection of aquatic life: Volume III. Standardized biological field sampling and laboratory methods for assessing fish and macroinvertebrate communities. Division of Water Quality Monitoring and Assessment, Columbus, Ohio.

In addition, one other publication from the Stream Regionalization Project is recommended to all users:

Whittier, T.R., D.P. Larsen, R.M. Hughes, C.M. Rohm, A.L. Gallant, and J.M. Omernik. 1987. The Ohio stream regionalization project: a compendium of results. U.S. EPA - Environmental Res. Lab, Corvallis, OR. EPA/600/3-87/025. 66 pp.

These documents can be obtained by writing:

Ohio Environmental Protection Agency
Division of Water Quality Monitoring and Assessment
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Other recommended and helpful literature is listed in the references of each volume.

FOREWARD

This volume represents a concept paper intended to support an initiative to adopt biological criteria into the Ohio Water Quality Standards (WQS) regulations (OAC 3745-1). Specific details of field methods and data analysis procedures are documerted in volumes II and III of this series (Ohio EPA 1987a; Ohio EPA 1987b).

Although the principle goal of the Water Quality Act is to restore and maintain chemical, physical, and biological integrity the methods by which regulatory agencies have been attempting to achieve it are primarily chemical and toxicological. Difficulties with defining an ecological approach to assessing biotic integrity have probably led to this reliance on surrogate measures. One purpose of this volume is to define biotic integrity as a practical and workable concept upon which objective biological criteria can be based. Thus compliance with a major directive of the Water Quality Act can be measured directly. This also responds to a mandate of the Water Quality Act of 1987 for the develcpment of biological monitoring and assessment methods as both a supplement and an alternative to the pollutant-by-pollutant criteria approach for toxic chemicals (Section 308).

This biocriteria approach can also be described as a systems approach in which the focus is on the resource (i.e. aquatic life) and its response to different environmental impacts. This approach permits a variety of different resource management options to be examined and used as a strategy to restore or protect the performance of the resource. In contrast, the current chemical specific/toxicity approach can be characterized as a regulatory approach in which the focus is on specific pollutants with specific rules for discharge being specified. This proposal advocates the complimentary use of both approaches, not one to the exclusion of the other.

The use of biological communities, particularly fish and macroinvertebrates, offers a holistic, systems approach to surface water quality assessment and management. Aquatic organisms not only integrate a variety of environmental influences (chemical, physical, and biological), but complete their life cycles in the water body and as such are continuous monitors of environmental quality. Focusing on major organism groups such as fish and macroinvertebrates represents biological evaluation at the sub-community level. This differs from past biological monitoring protocols which advocated the resource intensive monitoring of a variety of different organism groups (e.g. algae, macrophytes, zooplankton, diatoms, etc. in addition to fish or macroinvertebrates) at the same time. Another attractive feature of the biocriteria approach is that sampling need not be conducted under absolute worst case or critical conditions (i.e. Q7,10 flow) to determine attainment/non-attainment of aquatic life uses. This certainly presents a powerful assessment tool compared to the steady state approaches inherent to commonly applied chemical specific and toxicity methods. Including this type of biological field assessment along with traditional chemical and toxicity tools can significantly enhance decision making and regulatory resource allocation, particularly with complex issues.

The type of biological field assessments advocated by this document (i.e. sub-community level analysis) is cost competitive with chemical specific and toxicity testing methods. It is also equally cost effective when the power of the information derived from each is considered. The cost analysis presented in this document tends to refute the widely-held reputation of biological surveys as being prohibitively expensive.

Biological criteria were developed for Ohio rivers and streams using the biosurvey/ecoregion approach and the design of the Stream Regionalization Project in conjunction with the U.S EPA Environmental Research Laboratory - Corvallis. A set of least impacted reference sites from across the state and within each of the five ecoregions of Ohio were carefully selected and sampled for fish and macroinvertebrates. These sites represent watersheds with the least disturbance from human activity within each ecoregion. Based on these results criteria for three biological indices, the Index of Biotic Integrity (IBI, based on fish), the Modified Index of Well-Being (Iwb, fish), and the Invertebrate Community Index (ICI, macroinvertebrates) were derived. This design satisfies the definition of biological integrity as the biological performance achieved by the natural habitats within a region.

Practical uses of this approach include determining appropriate and attainable aquatic life uses for surface waters, extending anticegradation concerns to nonpoint and habitat impacts, enhanced problem discovery for toxics, prioritizing the use of regulatory resources (e.g. permits, grants, 3041 lists), and as a check on the attainment of Water Quality Act goals (e.g. 305b reporting).

Several examples from past Ohio EPA biological surveys are presented as a demonstration of how the biological criteria can be used and the complex combination of point source, nonpoint source, and habitat factors that are common to most study areas. The problem discovery capabilities of biological assessment are emphasized.

ACKNOWLEDGEMENTS

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Biological Criteria for the Protection of Aquatic Life: Volume I. The Role of Biological Data in Water Quality Assessment

I. INTRODUCTION

A principal objective of the Water Quality Act (WQA; previously referred to as the Clean Water Act) is to restore and maintain the physical, chemical, and biological integrity of surface waters. Although this goal is fundamentally "biological" in nature the specific methods by which regulatory agencies are attempting to reach this goal are predominated by such non-biological measures as chemical/physical water quality (Karr et al. 1985). The rationale for this process is well known - chemical criteria developed through toxicological studies of representative aquatic organisms serve as surrogates for measuring the attainment of the biological goals of the WQA. The presumption is that improvements in chemical water quality will be followed by a restoration of biological integrity. Although this type of approach may give the impression of empirical validity and legal defensibility it does not directly measure the ecological health and well-being of surface waters. In addition, recent information shows that other factors in addition to chemical water quality are responsible for the continuing decline of surface water resources in many cases (Judy et al. 1984). Because biological integrity is affected by these factors in addition to chemical water quality, controlling chemical discharges alone does not in itself assure the restoration of biological integrity (Karr et al. 1986). Whole effluent toxicity testing offers an improvement over a strictly chemical approach, but itself lacks the ability to broadly assess ecosystem effects, particularly those caused by physical and non-toxic chemical impacts.

This proposal advocates the adoption of biological criteria in the Ohio Water Quality Standards (WQS) regulations for the protection of aquatic life. While it is recognized from the outset that biological criteria and evaluations cannot perform every task necessary in a water quality management program they do offer some significant advantages over the traditonal chemical and/or bioassay approaches alone. Currently available techniques for future load projection (e.g. wasteload allocation), bioassay testing, compliance, and enforcement will continue to be important components in water quality management. However, the addition of biological criteria can be a valuable aid in supporting these activities if they are combined in a truly integrated program. It is important to recognize and exploit the links between the chemical, bioassay, and biosurvey/ecoregion approaches to water quality assessment and regulation. It is also important to recognize that the former are surrogates for biological community performance. Including direct, quantitative assessment of biological communities significantly broadens the base from which regulatory agencies can manage and protect surface water resources. This type of approach makes sense given the biological goals of the Water Quality Act and the important role that biological principles have in water resource management in general.

The need for this program also arises, in part, from a shift in emphasis on an "end of the pipe" approach to water quality management to a more integrated approach that considers site specific characteristics of the receiving water

body. Water quality management has been dominated by what can be characterized as a "regulatory approach" in which the focus of activity is on specific pollutants and specific rules for controlling discharges. By comparison the use of biological criteria can be characterized as a "systems approach" in which the focus of activity is on the resource (biological community performance) which should permit the consideration of more than one possible management strategy to restore and protect aquatic life resources. Both approaches should be factored into current water quality management programs. The challenge for regulatory agencies is to recognize situations where the elements of each approach should be used and to what degree they should influence decisions. The call for a greater emphasis on biological assessment and a more holistic, systems approach to water quality management is not unprecedented. Karr et al. (1986) provides a thorough discussion of the underlying theory behind this type of approach and compares it with other chemical and physical based measures. More recently, Section 308 of the Water Quality Act requires U.S. EPA to develop biological assessment methods and criteria in addressing toxics.

Ohio EPA has had more than ten years of experience in conducting biological evaluations of surface waters (Table 1). This base of experience has been invaluable in providing insights into the potential uses of biological criteria, their advantages, and their limitations. A wide array of different types and degrees of environmental perturbation (both chemical and non-chemical) have been observed and evaluated. Nearly 100 reports which include the results and analysis of biological assessments of different watersheds have been produced since 1979. This experience provides the basis for many of the concepts that are presented in all three volumes. The evaluation of surface waters routinely conducted by Ohio EPA includes chemical analyses (water column, effluent, sediment, and fish tissue), bioassay tests, and biological evaluations at the sub-community level (primarily fish and macroinvertebrates). This has provided the opportunity to observe and evaluate the similarities and differences between a direct measure of biotic integrity (i.e. biosurvey/ecoregion approach) with surrogate measures (e.g. chemical, bioassay, sediment) under a wide variety of conditions. Finally, the development and use of standardized field evaluation and data analysis techniques have been essential for developing biological criteria. This often underrated aspect of field evaluation has permitted the establishment of objective assessment criteria which can be used on a statewide and regional basis.

Why Biological Criteria?

Why does the use of biological criteria in water quality management present certain advantages over a chemical and/or bioassay approach alone? First, it must be understood what an assessment of the instream biota represents. The existing condition of the biota resident in any surface water body is the integrated result of many chemical, physical, and biological processes over time. Thus the existing biological condition is the "summation", or result, of these processes in their dynamic sequences. Biological communities themselves are precise indicators of actual conditions since they inhabit the receiving waters continuously and are subject to the variety of chemical and physical influences that occur over time. Chemical data, on the other hand,

Table 1. Measured progress of the Ohio EPA biological and water quality survey program, 1977-1987.

Report Year	Work Years	Streams Surveyed	River Mi.	Sampling stations Chem (% freq)	Fish	Macro	Point Source Issues: Major/sign.[a] Muni.	Ind.	CG/AT[b]	Minor Muni.	Ind.	Total Sources	WQS use issues other than WWH[c]
1977-1980	2-9	20	460	175 (7.6)	189	83	22	16	11	11	91	140	6 LWH, 1 MD
1981	14	26	420	180 (8.7)	220	149	18	6	27	27	42	93	10 LWH, 1 MD, 2 CWH, 1 EWH, 1 SSH
1982	18	35	830	252 (6.0)	338	248	24	6	42	44	57	131	7 LWH, 1 MD, 7 EWH, 2 SSH
1983	19	65	1150	324 (6.2)	404	360	34	21	51	50	20	125	16 LWH, 16 MD, 6 EWH, 5 CWH
1984	19	85	1180	312 (5.0)	313	266	37	20	12	18	29	104	12 LWH, 6 MD, 5 EWH, 1 CWH
1985	18	62	700	263 (3.9)	280	231	8	26	7	56	20	90	9 LWH, 2 MD, 7 EWH, 1 CWH
1986[d]	15	42	600	178 (3.8)	204	181	7	19	6	13	27	66	1 LWH, 2 EWH, 1 CWH
1987[d]	13	57	465	174 (3.0)	215	190	9	10	8	37	52	116	5 LWH, 4 EWH, 2 SSH, 3 MD
TOTAL	136	394	5805	1858 (6.0)	2193	1758	159	124	164	236	338	865	75 LWH, 30 MD, 30 EWH, 10 CWH, 5 SSH
1981-87	106	374	5345	1683	2004	1675	137	108	153	225	247	725	150 issues

[a] evaluations of municipal WWTPs also include an assessment of combined sewer overflow (CSO) effects, regulated and unregulated WWTP by-passes, and urban storm water runoff effects (these are not quantified in the above table).

[b] Construction Grants and Advanced Treatment issues.

[c] Water quality standards use designation issues:
WWH = Warmwater Habitat
LWH = Limited Warmwater Habitat
EWH = Exceptional Warmwater Habitat
CWH = Coldwater Habitat
SSH = Seasonal Salmonid Habitat
MD = Mine Drainage

[d] Includes two projects for the assessment of nonpoint source impact only.

are biased toward short-term conditions that exist at the time a sample is collected. This is also true of bioassay tests to a comparatively lesser degree. The condition of resident biological communities is the result of recent and past conditions including both usual and extreme events. This includes all of the chemical and physical variables that are used in the previously described chemical approach and additional important variables that may not be quantified or considered. If we consider these chemical, physical, and biological variables as "pieces" then the resultant biological condition is the integrated result of the assembly of these "pieces" in there proper dynamic sequence. In this sense the biosurvey/ecoregion approach represents a "top down" evaluation where the end product (biological community performance) is used to characterize the summed result of all the "pieces" (i.e. the chemical, physical, and biological processes that affect biological performance). By comparison the chemical/toxicity approach represents a "bottom up" evaluation where some of the "pieces' are used in an attempt to simulate, predict, or explain an assumed end result. Furthermore, commonly used techniques of the chemical approach are usually limited to static applications that consider the interactions of some of these variables at one point in time and under one set of conditions. Thus important dynamic phenomena and interactions that are reflected by the condition of the resident biota are not considered in commonly used chemical application techniques. In short, the biota exceeds our capacity to assemble the pieces via even the most sophisticated chemical application techniques. Thus, the observed performance of the biological system that we are attempting to protect should greatly influence how the commonly applied chemical and emerging bioassay techniques are used in a regualatory program. This would apply to both site-specific observations and broader, regional applications. For instance the biotic response observed at many locations over a broad geographic region may be used to influence how a chemical criterion is eventually derived. However, the response of the biota in a specific stream may influence how the chemical criterion is used to establish limits for a point source discharge.

A legitimate question that might be asked is if the biosurvey/ecoregion approach has so much potential why has it not been proposed prior to now? We believe that the answer to this question lies in the following:

- a practical, workable definition of biological integrity was not forthcoming, thus the underlying support for the biocriteria approach was lacking;
- biological field sampling methods were variable and essentially lacked standardization;
- a general failure to develop and use consistent benchmarks for measuring attainment of the biological integrity goals of the Clean Water Act; and,
- widely divergent opinion among biologists with regard to which organism groups should be monitored, which methods should be used, and a general lack of effective communication with the non-biologists involved in environmental management.

Although most of these shortcomings were certainly not intentional they were, nevertheless important in determining the place of biological assessment in surface water management in the past. An example of how these shortcomings were revealed was with the many 316(a) thermal discharge studies that were performed in the mid and late 1970s. Some of these problems remain today and will require further resolution in the near future. Fortunately there is a renewed interest in biological field evaluation. This can be credited to, in part, the advocation of standardized assessment techniques, recent advances in laboratory and field methods, a practical and implementable definition of biological integrity, and the ecoregion/reference site concept. Together these advances have provided a workable framework from which the use of biocriteria can be "institutionalized" in surface water quality management programs.

Concerns With the Current Approach

Chemical-numerical criteria and/or toxic units logically serve as the basis for the calculation of water quality based effluent limits. A commonly employed practice is to establish NPDES permit limits that will assure maintenance of the chemical criterion and/or compliance with toxicity units under a fixed set of discharge and receiving water quality and quantity design conditions intended to approximate critical circumstances (most frequently the Q7,10 stream flow and a discharge design flow). It is assumed in such cases that if any exceedence of a chemical criterion is predicted beyond a margin of safety then an impairment of the designated use will occur. Under the surrogate concept this is translated into an assumed impairment of biological community structure and function. A similar conclusion is made based on the results of bioassay tests if acute and/or chronic toxicity is observed. As was mentioned previously important dynamic relationships involving discharge and receiving water quantity and quality that can and do occur with reasonable regularity may either be inadequately addressed or rarely considered in such applications of WQS. To consider such dynamics in all applications would require considerably more resources and data than are presently or anticipated to be available.

Commonly used methods for deriving water quality-based controls requires the making of assumptions which are largely derived from best professional judgement. These assumptions should have a sound ecological and/or toxicological basis. However, many are based on an overly simple or incomplete understanding about the ecological and/or toxicological relationships that are actually involved. In addition, some assumptions are made out of practical necessity because measuring the actual environmental processes is time consuming, costly, and beyond the practical capabilities of most state agencies. One example of this type of assumption is the "instantaneous" mixing of effluent and receiving water flows. Even though mixing zones are three-dimensional and dynamic, a one-dimensional, static approach to accounting for this important phenomenon is commonly used. This may lead to regulatory strategies that require meeting stringent chemical limits at the end-of-the-pipe, but which provide for little real benefit in the receiving waters. It may also result in regulatory agencies encouraging discharge strategies such as high-rate diffusers which in certain situations can result in a greater impact than that from conventional outfalls. Other

important assumptions include pH, temperature (ammonia toxicity), and hardness (metals toxicity) duration values (e.g. which percentile of actual values are ecologically critical?). Other factors (e.g. sorbtion, chelation, additivity) are rarely considered, even though it is acknowledged that they occur and greatly influence the resultant effects on aquatic life (U.S. EPA 1985a). When this knowledge or information is lacking, uncertainty factors (in addition to any margin of safety already in the criteria) are employed. Since there are several points in the process where detailed knowledge is lacking, these uncertainty factors tend to "accumulate" through the criteria development, criteria implementation, and compliance evaluation processes. These shortcomings are not intentional, but remain an environmental reality.

Furthermore the chemical criteria, application, and data evaluation techniques themselves have some important limitations which have become increasingly evident:

- Organisms in the natural environment behave differently than laboratory organisms. Biological communities often respond to pollution stresses in a different manner from what laboratory criteria might predict or imply. This is due to the complex combination of prevalent ecological factors (e.g. habitat variations, refuge availability, biotic interactions, food sources, etc.) that cannot be adequately accounted for in the criteria development and application process.

- Exposure scenarios in the natural environment are dynamic, i.e. they are both spatially and temporally variable. In contrast most laboratory criteria derivations and commonly used criteria application techniques assume static or steady-state conditions on a pollutant-by-pollutant basis. Hence chemical criteria applications can be either over-protective or under-protective. This can result in continued harm to the environment or the imposition of treatment costs that are disproportionate to the environmental benefits gained.

- Laboratory criteria are not available for several species that comprise a significant component of biological communities in Ohio's surface waters (e.g. six species of the fish genus Moxostoma, 10 common darter species, several minnow species, etc.). This can result in under or over-protective criteria.

- Incremental variations in community structure and function, species richness, and population abundance that can be measured via biological field monitoring are not simulated or predicted with currently available chemical criteria derivation and application techniques. For example Gammon (1983) showed that even for a parameter with a complete data base (temperature) comparative predictive success was very limited. Only gross change was predictable; forecasting the degree of change requires detailed information that is beyond current science.

- Aquatic life criteria are generally not available for substances and phenomena that exert effects by means other than toxicity. This includes several substances that are natural constituents of watersheds such as nutrients and sediment.

- The current approach does not adequately account for non-chemical factors that may preclude full attainment of aquatic community potential. Examples of this include habitat modifications (e.g. channelization, impoundment, siltation) that prevent attainable biological performance via physical alteration and biological simplification. The resultant effects of these modifications are variable and are only generally transferable from site to site. This is also a limitation of effluent and ambient bioassay testing.

- Tiered aquatic life uses are not able to account for many of the incremental, biological variations that are observed between streams of the same use designation. Although different levels of biological performance can be measured between different streams of the same use designation one chemical criterion applies to all. Additionally, current criteria derivation techniques are insensitive to differences between two important use designations, Warmwater Habitat and Exceptional Warmwater Habitat, for all chemical parameters except dissolved oxygen.

- Satisfying some of the emerging water quality management needs (e.g. toxics control, nonpoint source evaluation, storm water management) may require chemical criteria and sophisticated application techniques for more substances than we can afford to consider on a parameter-by-parameter basis.

- When does a measured or predicted violation of a chemical WQS become significant in terms of actual harm to the aquatic community? Because this type of knowledge is lacking chemical data evaluations are limited to "rule of thumb" estimates of the percent exceedence that contributes to attainment, partial attainment, or non-attainment of legislative goals.

Several responses to meet some of these challenges have been proposed. These include site-specific criteria modification (U.S. EPA 1983a), advanced criteria application techniques (U.S. EPA 1985b), consideration of additivity in criteria application, sediment criteria, and whole effluent toxicity assessment (U.S. EPA 1985b). Although some of these techniques may get "closer" to the endpoint of concern, all have common limitations -- they employ a surrogate approach (which requires the use of uncertainty factors) and they can be resource intensive to implement. This also raises the question of how many pieces do we need to adequately simulate the whole, how long will it take, and how much will it cost to get there? For instance the recently published Technical Support Document for Water Quality-Based Toxics Control (U.S. EPA 1985b) describes some of the chemical and physical data requirements needed to implement such an approach. No fewer than 10-15 additional measurements of the water column and sediment (in addition to the current routine parameters) are needed at each site. Additionally, the logistics involved may be substantial (e.g. short sample holding times, analytical costs for organics). Answers for some of the other questions raised simply do not exist under current science. Eventually it is the performance of the natural system (which we are charged to protect) that is the endpoint measure that determines whether or not regulatory programs have been successful.

The past direction of traditional surface water regulation strategies likely emanates from the belief that it is not really feasible to measure the extant, quantitative condition of the instream biota. Therefore, it became that surrogate approaches to surface water regulation and assessment were relied upon. Reasons for this include the perception that biological information is simply not obtainable from a technical and resources/cost standpoint and that natural biological communites are simply too complex to measure and poorly understood to use. Therefore, the only alternative left was to use surrogate indicators of environmental impact to provide indirect insights into current and future conditions. However, continued reliance on this philosophy is questionable when the growing body of information that demonstrates the usefulness of the biosurvey/ecoregion approach is considered. Since it is possible to produce quantitative, standardized information about instream biotic response on a routine and cost-effective basis, how should the availability of this information influence the traditional chemical and toxicity approaches? Should a higher priority be placed on making sure such information is available? We believe that it should. U.S. EPA (1985b) states that there is a significant risk of incorrectly imposing a wasteload allocation generated by steady-state methods (refered to as Level 1 and 2 exposure assessments; see Table 2) if effluent variability and the probability basis for the allocation and permit limits are not considered. However, quantitative knowledge of aquatic community response can "make-up" for some missing information and deficiencies inherent to Level 1 and 2 assessments when used as part of a holistic, comprehensive risk assessment approach.

II. GENERAL CONCEPTS

Defining Biological Integrity

The term "biological integrity" originates from the Water Pollution Control Act ammendments of 1972 (PL 92-500) and has remained a part of subsequent revisions (PL 95-217; PL 100-4). Early attempts to define biological integrity in ways that it could be used to measure attainment of legislative goals were inconclusive. One of the most comprehensive of these efforts failed to produce a clear procedure or methodology for determining biological integrity, although several contributors urged that a holistic or systems approach be employed (Ballentine and Guarrie 1975). Biological integrity was considered relative to; 1) conditions that existed prior to human civilization, 2) the protection and propagation of balanced, indigenous populations, and 3) ecosystems that are unperturbed by human activities. These criteria (at least 1 and 3) refer to a pristine condition that probably exists in few, if any, ecosystems in the conterminous United States. One U.S. EPA sponsored work group (Gakstatter et al. 1981) concluded that biological integrity, when defined as some pristine condition, is difficult if not impractical to precisely define and assess. The pristine definition of biological integrity is considered a conceptual goal toward which pollution abatement efforts should strive, although current, past, and future uses of surface waters may prevent its full realization. More recently efforts to provide a workable, ecological definition of biological integrity have provided the supporting theory that necessarily precedes the development of standardized measurement techniques and criteria for determining compliance with that goal. Biological integrity is defined herein as the ability of an aquatic ecosystem to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitats within a region (Karr and Dudley 1981). This is a workable definition of biological integrity that is based on measurable characteristics of biological community structure and function in least impacted habitats. It also provides the fundamental underlying theory for the eventual development of biocriteria using the biosurvey/ecoregion approach.

Systems that possess or reflect biological integrity can withstand or rapidly recover from most perturbations imposed by natural environmental processes and some of those induced by humans (Cairns 1975; Karr et al. 1986). The reaction of an aquatic ecosystem to perturbation(s) depends largely on the frequency, magnitude, and duration of the effect and the inherent sensitivity of the system itself. Thus biological communities that are degraded and therefore lack integrity have had their capacity to withstand and rapidly recover from perturbation(s) exceeded. Some communities are likely to become even further degraded under incremental increases in stress. In contrast communities that reflect biological integrity do so because their capacity to withstand stress has not been exceeded to result in a temporally extended degradation of structural or functional organization. A biological system can be considered to have integrity when its inherent potential is realized, its condition is stable, its capacity for self-repair when perturbed is preserved, and minimal external support for management is needed (Karr et al. 1986). Biological

integrity is not necessarily equated with harvestable products of economic or recreational value, although those systems that reflect integrity often have the better commercial and recreational opportunities.

Factors That Affect Biological Integrity

Karr et al. (1986) grouped the environmental factors that most affect aquatic ecosystems into five major classes (Fig. 1). Alterations of the physical, chemical, or biological processes associated with these classes may adversely affect aquatic biota and therefore the biological integrity of the water body. Efforts to protect and restore water resources that focus on only one or two of these major classes, or only a few factors within a class, will fail if other factors are wholly or partially responsible for the observed degradation (Karr et al. 1986). Thus efforts to maintain and improve the quality of surface water resources in general and aquatic life in particular must be guided by methods and monitoring that identify perturbations associated with the factors in all five classes, not just one or two. Broad-based approaches to water resource management are not only more likely to provide solutions with real results, but they are more likely to prove cost-effective because the development of abatement measures is guided by a directly measured response instead of indirect surrogate measures that rely heavily on "rule of thumb" assumptions. Also, more reliance might be placed on the natural processes of the environment to protect and improve aquatic resources. For example, one solution to agricultural nonpoint runoff impacts on a stream system might be to protect or enhance the vegetative riparian buffer margin in addition to (or in place of) land treatment options. The presence of a shading tree canopy will deter the development of nuisance algal blooms which in turn affect biological integrity through changes in nutrient cycling and habitat alteration. Leaves from deciduous trees supply coarse particulate organic matter which is essential in the upper reaches of streams. Although this strategy directly concentrates on one of the five major classes (energy dynamics) it indirectly includes habitat structure, biotic interactions, flow regime, and chemical variables. In this case chemical water quality would not be the driving factor in the recommended solution. It is difficult to find a situation where chemical water quality exclusively controls or determines the biotic potential of a receiving water.

Applicability of Biological Criteria

This proposal advocates the adoption of biological criteria in the Ohio WQS regulations as an aid in performing surface water regulation in a more broad-based, technically sound, and cost-effective manner. The continued use of the more traditional chemical and emerging bioassay approaches is also advocated and is viewed as essential to having a truly integrated surface water quality management program. Compliance with and enforcement of NPDES requirements is primarily a chemical-numerical approach. However, the imposition of chemical (and toxic unit) limitations should bear some logical relationship to the observed biological community response in the receiving waters.

The broad applicability of field biological evaluation is demonstrated in Table 2. The comparative ability and "power" of some traditional water quality and bioassay assessment tools to measure or reflect key factors of the five major classes of environmental influences are compared with biological

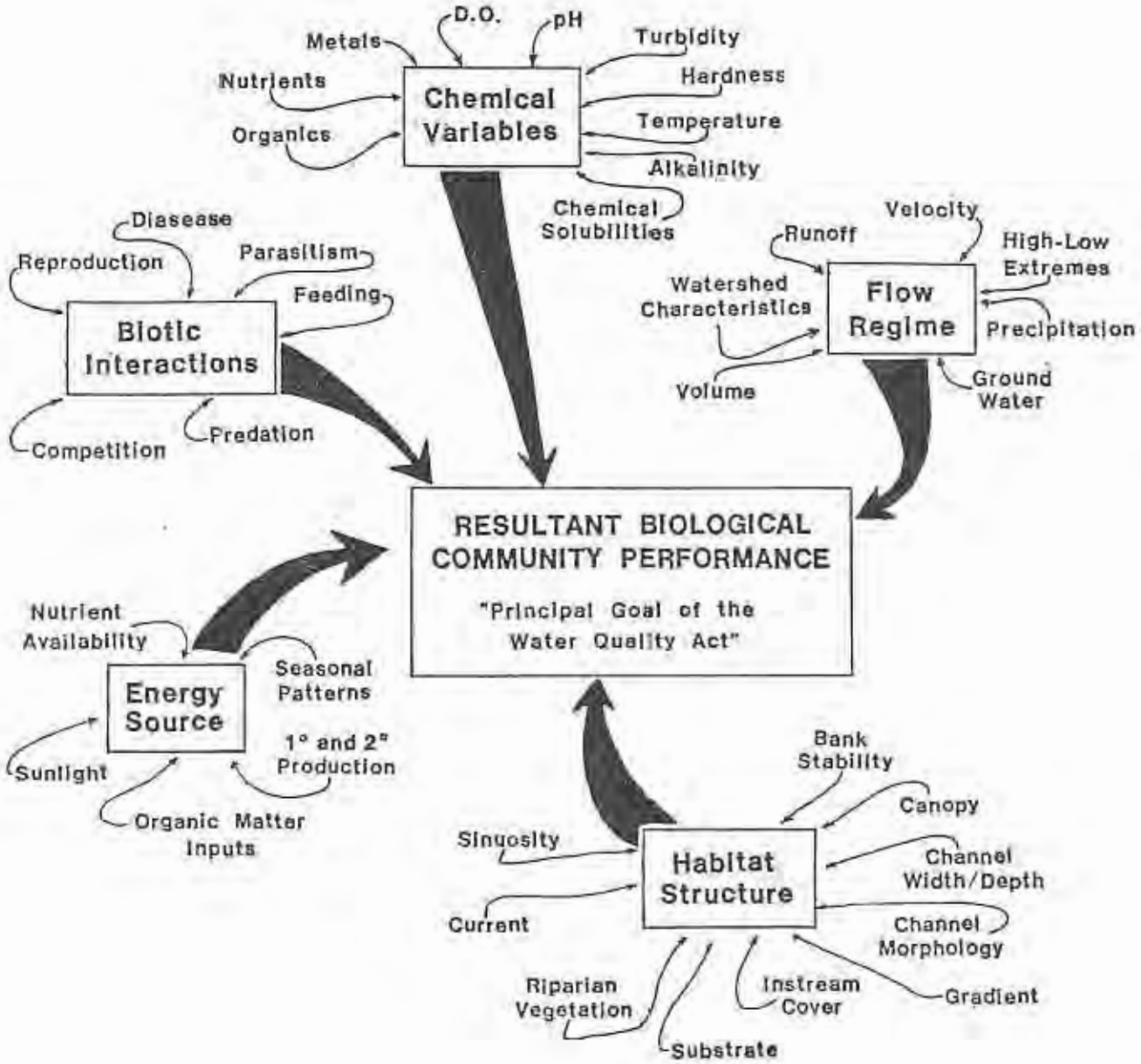

Figure 1. Some of the important chemical, physical, and biological factors which influence and determine the resultant biological community performance in surface waters showing the five principal factors and their various components (modified from Karr et al. 1986; 1° and 2° are primary and secondary, respectively).

Table 2. The comparative ability and "power" of various chemical, physical, and biological assessment techniques to measure or indicate key components of the five major factors that affect the biological integrity of surface waters (D - directly measures; I - indirectly measures; S - strongly reflects; C - casual relationship).

Factors/Components	Level I&2 Exposure Assess.[1]	Level 3 Exposure Assess.[2]	Toxicity (acute)	Toxicity (chronic)	Physical Assessment	Biological Field Eval.
I. CHEMICAL WATER QUALITY						
Conventional substances	D	D	I	I	–	S
Heavy metals	D	D	I	I	–	S
Toxic Organics	–	D	I	I	–	S
Static interactions	S	S	I	I	–	N/A
Dynamic interactions	–	I	–	I	–	S
II. ENERGY DYNAMICS						
1° and 2° dynamics	C	I	–	–	–	I
Nutrient cycling	C	I	–	–	–	I
Organic inputs	–	C	–	–	–	I
III. HABITAT QUALITY						
Substrate	–	–	–	–	D	S
Water velocity	–	–	–	–	D	S
Instream cover	–	–	–	–	D	S
Channel integrity	–	–	–	–	D	S
Riparian buffer	–	–	–	–	D	S
Habitat diversity	–	–	–	–	D	S
IV. FLOW REGIME						
Low extremes	I	I	–	–	–	S
High extremes	–	–	–	–	–	S
Temporal cycles	–	C	–	–	C	S
Volume	D	D	–	–	D	S
V. BIOTIC RESPONSES						
Acute effects	I	I	D	D	–	S
Chronic effects	I	I	I	S	–	S
Abundance, biomass	–	–	–	–	–	D
Structural	–	–	–	–	–	D
Functional	–	–	–	–	–	D
Diasease, etc.	–	–	C	C	–	D
Tolerances	–	–	–	–	–	D
Competition	–	–	–	–	–	S
Predation	–	–	–	–	–	S
Growth	–	C	–	S	–	D

1 consists primarily of models for oxygen demanding substances and simple mass-balance dilution calculations for other substances; steady-state conditions are assumed.

2 consists of applications ranging from probabalistic dilution to dynamic fate-assessment models.

field assessment. Each tool was "graded" as to whether it directly measures (D), indirectly measures (I), strongly reflects (S), or casually indicates (C) the given factor. A dash (-) indicates that no relationship is either demonstrated or possible. For example, Level 1 (steady state, mass balance dilution models) and Level 2 (probabalistic dilution models) exposure assessments directly measure conventional and heavy metal substances in the water column because they include chemical sampling and analysis. However, they only indirectly indicate acute and/or chronic biotic responses because of the extrapolations and assumptions that are necessarily made. Furthermore there is no direct relationship to the structural and functional factors in the biotic response class, although this is often assumed. Some acute toxicity assessment tools (e.g. bioassays) indirectly measure the effects of chemical substances, but may not measure or quantify chemical concentrations. Biological field evaluations strongly reflect these factors because community response patterns among different chemical effects (e.g. low D.O. vs. acute toxicity) are discernable. No credit was given to some tools even though they include an instantaneous measurement of listed factors in their methodology. For example, Level 1 or 2 exposure assessments may include water velocity measurements as part of that analysis, but this alone contributes little direct or indirect information about how the water velocity regime of the water body affects biological integrity. From Table 2 it is clear that biological field evaluation is the only tool that has some ability to measure, indicate, or reflect the factors in each of the five major classes. Thus it appears to be the "broadest" single tool for assessing the effect of a variety of environmental influences. When it is used in conjunction with the other assessment tools the ability to identify and quantify the component area influences is greatly enhanced. Such an approach will undoubtedly lead to more effective regulation of pollution sources.

Specific attributes of biological communities that make them particularly well suited to define environmental impacts include:

- Some organism groups, particularly fish and many macroinvertebrates, inhabit the receiving waters continuously and as such are a reflection of the chemical, physical, and biological history of the receiving waters.

- Resident biological communities are integrators of the prevailing and past chemical, physical, and biological history of the receiving waters, i.e. they reflect the dynamic spatial and temporal interactions of stream flow, pollutant loadings, toxicity, habitat, and chemical quality that are not comprehensively measured by chemical or short-term bioassay results alone.

- Many fish species and invertebrate taxa have life spans of several years (2-10 yrs. and longer), thus the condition of the biota is an indication of past and recent environmental conditions. Biological surveys need not be conducted under absolute "worst case" conditions to provide a comprehensive and meaningful evaluation of use attainment/non-attainment. A finding that biological integrity is being achieved not only reflects the current healthy condition, but also means that the community has withstood and recovered from any short-term stresses that may have occurred prior to field sampling.

- o Biological community condition portrays the results of water quality management efforts in direct terms, i.e. increases and decreases in community health (as reflected by structure and function, abundance of certain species, etc.) is a meaningful measure of regulatory program progress and attainment/non-attainment of legislative goals.

- o Minimal manipulation of data using adjustment or uncertainty factors is necessary (U.S. EPA 1985b).

- o Biological assessment techniques have progressed to the point that incremental degrees and types of degradation can be determined and presented as numerical evaluations (e.g. Index of Biotic Integrity, Invertebrate Community Index, etc.) that have relative meaning to non-biologists.

The common tendency in water resource management has been to make biological measurements fit the perceptions and use of chemical criteria, rather than the reverse. This is clearly illogical because the structure and function of the aquatic community is the embodiment of the temporal and spatial chemical, physical, and biological dynamics (i.e. the "pieces") of the aquatic environment. Perhaps the inability of biologists to agree on an empirical measurement of biological integrity has resulted in this situation (Karr et al. 1986). The solution to this problem is to have usable biological criteria which can quantitatively indicate the degree to which biological integrity is or is not being achieved. Chemical criteria and bioassay application techniques will always play an important role in water quality regulation. Their value, however, would be greatly enhanced when used in combination with holistic assessments of the resident biota.

Adopting an approach of increased reliance on direct measurements of biological performance to aid in setting permit limits and establishing 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 endpoint of concern (i.e. biological integrity) via the chemical and/or narrative (i.e. "free from") approaches, this process will involve the development of control requirements to achieve or maintain the biological endpoint by prior quantitative knowledge about that endpoint. This will involve linking treatment processes, entity performance, water quality, habitat, toxicity units, best management practices, etc. with observed biological community response in a "feedback loop" type of approach. With some types of degradation, a certain amount of trial and error application may have to be accepted especially where chemical tools begin to approach uncertainty, key knowledge about the chemical substances involved is lacking, or the degradation source is primarily non-toxic. One example of this approach would be in defering tertiary WWTP filters pending the outcome of post advanced treatment monitoring. Another example of a "feedback loop" type of approach is with Toxicity Reduction Evaluations which can be recommended when significant toxicity is found in an effluent.

U.S. EPA water quality planning and management regulations (40 CFR Parts 35 and 130) encourage the use of biological data in decision making. Total Maximum Daily Loads (TMDLs) may be calculated using either a pollutant by

pollutant approach based on mathematical modeling, or a biomonitoring approach using bioassays or biosurveys. In many cases U.S. EPA believes both approaches will be necessary (40 CFR Part 130.4, p. 1780). The U.S. EPA Technical Guidance Manual for Performing Wasteload Allocations (U.S. EPA 1984) specifically states that it is preferable to coordinate chemical sampling 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 data base may also provide information transferable to other sites."

The shortcomings of the chemical approach are serious enough for point sources, but they are further compounded with nonpoint and intermittent sources of pollutants (e.g. combined sewer overflows, storm water discharges, spills/dumping). Nonpoint sources are temporally more variable than point sources, tend to be predominated by "natural" constituents (nutrients, sediments, etc.), and are frequently more subtle in appearance than are point sources. In addition, depending on the time and conditions prevalent during sampling, chemical monitoring can overlook or underestimate nonpoint source impacts that are biologically evident. At best, near-continuous chemical monitoring is necessary to quantify the temporally dynamic nature of these impacts and then it is still left 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 chemical, physical, and biological processes.

Critical Flow Values and Biological Integrity

Water Quality Standards contain rules which define minimum stream flows at which chemical and narrative criteria must be met. This is most commonly the seven-day average flow that has a probability of recurrence once every ten years (i.e. Q7,10 flow). Other low-flow values are also used (95% duration flow, Q30,10 flow) and these approximate the Q7,10 relative to the annual hydrograph. Because the common implementation of chemical and narrative criteria is essentially a static, dilution oriented process, a defined "critical" flow is necessary. This has been a widely accepted and essentially unquestioned concept in surface water quality regulation for many years. However, a direct ecological basis for such flow values is lacking.

There have recently been efforts to define critical flow thresholds using a toxicological rationale (U.S. EPA 1986). This involved making judgements about the number of exceedences of acute and chronic chemical criteria that could occur without causing harm to the aquatic community. This effort also attempted to establish a minimum flow at which it is assumed that chemical and/or toxic unit limits can be set and not have the aquatic communities in a perpetual state of recovery. Establishing a single critical flow (i.e. Q7,10, Q30,10, etc.) on an ecological basis, however, is not only improbable under current science, but is probably inappropriate. There are simply too

many other variables that simultaneously affect the response and resultant condition of aquatic communities both spatially and temporally. Some can be estimated (e.g. duration of exposure, chemical fate dynamics), but many cannot because of the intersive data collection and analysis requirements (see Fig. 1). Other phenomena are simply not understood well enough to include in an analysis, yet their influence is integrated in the biological end result (e.g. complex biological, chemical, and physical interactions).

The ecological ramifications of low-flow conditions (particularly drought) in small streams have probably contributed to much of the attention given to critical low-flow. The results of low stream flow alone can be devastating in small watersheds, particularly during extended periods of severe drought (Larimore et al. 1957). This is largely due to a loss of habitat via dessication; organisms either leave or die during these periods. The sustaining flow provided by a point source effluent can mitigate the effects of dessication provided that chemical conditions are satisfactory for organism functioning and survival. The presence of water with even a marginal chemical quality can successfully mitigate what otherwise would be a total community loss. As was previously mentioned, this is dependent on the frequency, duration, and magnitude of any chemical stresses and local faunal tolerances. Small headwater streams (typically less than 10-20 sq. mi. drainage areas) in Ohio commonly experience near zero flows during extended dry weather periods, sometimes during several consecutive summers.

Chemical-numerical applications necessarily have their basis in dilution scenarios. However, these types of simplified analyses are no match for the insights provided into the chemical, physical, and biological dynamics that are reflected by the condition of the resident biota. The resolution of simplified chemical application techniques suffers somewhat when applied to extreme low-flow conditions. Site specific factors that outweigh the importance of flow alone are the availability of permanent pools and other refugia, gradient, organism acclimatization, and riparian characteristics such as canopy to name a few. Together these and other factors determine the ability of a biological community to function under "worst case" low-flow conditions.

What relevance do critical flow values have to an assessment of the resident biological communities? The most frequent misconception is that biological data collected during any time other than the Q7,10 critical flow does not represent the effect of "worst case" conditions. Sampling under such "worst case", low-flow conditions is simply not necessary when measuring the condition of communities that have relatively long life spans and carry out most or all of their life functions in the water body. It is inappropriate to expect biological community condition (which is the integrated result of physical, chemical, and biological factors) to be so controlled by a temporal extreme of one physical variable (see Fig. 1). The observed condition of the aquatic biota at any given time is the end result of the chemical, physical, and biological dynamics that have occured in a water body over time. This not only includes critical low-flow, but also the probabalistic relationship of low flows and effluent variability. Add to this the influence of high flow and the more usual in-between conditions. Unlike chemical water quality the aquatic biota does not respond instantaneously to normal short-term events. This strongly implies that one variable used in chemical criteria application

does not "make or break" the aquatic biota on its own. It would indeed be a poor survival strategy if aquatic organisms were so affected by such short-term and temporally extreme events. Harmful short-term episodes ranging from the onset of rapidly lethal conditions to a protracted chronic stress will be manifiested in the response of the resident biota. Thus the biota can reveal the real world effects of exceedences and consequent harm more precisely than can be predicted or measured on a chemical or toxicity basis alone. A finding that biological integrity is being achieved not only reflects a current healthy condition, but also means that the community has withstood and recovered from any short-term stresses as a result of critical low-flow (or other temporal events) that may have occurred prior to or during field sampling.

Therefore, because biological communities inhabit the receiving waters all of the time and will show the truly harmful effects of past stresses, it is not necessary to conduct sampling coincidental with critical low-flow conditions (or other temporal extremes) to gain a representative picture of community health and well-being. Indeed more important and significant stresses can and do take place as a result of events that occur at times other than critical low-flow. The condition of the aquatic biota is generally representative of environmental conditions even though maximum stresses might have occurred at times other than the sampling dates.

The Costs of Biological Field Monitoring

Biological field data collection has the reputation of being too costly and resource intensive to use on a routine basis. This reputation has been earned, in part, by the perception that a community level analysis is needed in most cases. However, Ohio EPA has used a sub-community level approach focusing on two major aquatic organism groups, fish and macroinvertebrates. Influences on the aquatic ecosystem will eventually appear in one or both groups since they are the end-product of aquatic system functional processes. This is not to imply that other organism groups are completely ignored. Quite the contrary, since a response elicited in the higher organism groups may have its beginnings in the lower trophic levels (e.g. phytoplankton).

Biological community data (particularly for fish and macroinvertebrates) are reasonably obtainable. Advances in sampling techniques and taxonomy over the past 10 years make routine biological field monitoring a workable concept for regulatory agencies. We recently compared the cost per sample between fish and macroinvertebrate sampling (according to Ohio EPA methods), and intensive survey chemical grab sampling and bioassay testing (Ohio EPA 1986). These costs are based on field sampling, data analysis, and personnel requirements (Table 2). Sampling both fish and macroinvertebrates was less costly than either chemical sampling or bioassay evaluation on an entity evaluation basis. When the comparative usefulness and "power" of each data set is compared, the "real" per unit cost of the fish and macroinvertebrate data is much less than the corresponding cost of the chemical and bioassay analyses. More comprehensive chemical monitoring and the inclusion of organic chemical scans and sediment analyses significantly boosts the costs for chemical data. This is a real concern because the list of chemical substances requiring analysis is growing. However, the chemical and bioassay tools are needed for adequate environmental evaluation and regulation and their value should not be

Table 2. Cost comparison of fish community and macroinvertebrate community evaluations with chemical/physical grab sampling and acute and acute/chronic bioassay tests (after Ohio EPA 1986).

Monitoring Component	Sample Collection	Analytical Cost	Cost per Test	Cost per Evaluation
Macroinverte-brate Community	N/A	N/A	N/A	$699
Fish Community				
2 passes/site	N/A	N/A	N/A	$673
3 passes/site				$897
Chemical/Physical Water Quality				
4 samples/site	$1,073[a]	$428[b]	$375	$1,501
6 samples/site	$1,610[a]	$642[b]	$375	$1,715
Bioassay				
Screening[c]	$ 209	N/A	$1,053[d]	$ 3,159
Definitive[e]	$ 209	N/A	$1,967	$ 5,901
Seven-day[f]	$ 209	N/A	$2,846	$ 8,538
Seven-day[g]	$1,578	N/A	$4,214	$12,642

a includes cost of sample collection and data analysis only; based on an average frequency of 3.9 samples/site in 1985.

b analytical costs based on each sample being analyzed for 5 heavy metals ($7.00 ea.), 4 nutrients ($10.00 ea.), COD or BOD ($15.00 ea.), and 2 additional parameters ($17.00 for both).

c 48 hour exposure to determine acute toxicity.

d represents the cost of both sample collection and laboratory components.

e 48 and 96 hour exposure to determine LC50 and EC50.

f seven-day exposure to determine acute and chronic effects using a single 24-hour sample; cost based on analysis of one pipe only; costs for chemical analyses in sole support of the bioassay test are not included.

g seven-day exposure using a composite sample collected daily (renewal); other factors in footnote f apply.

diminished. The purpose of this comparison is to show that the common perception of biological field data as being "prohibitively expensive" is unfounded. This conclusion tends to refute the perception that resource constraints will prohibit the use of instream biological community response on a routine basis (U.S. EPA 1985b).

III. BIOLOGICAL CRITERIA DEVELOPMENT

Three biological indices, the Index of Well-Being (Gammon 1976; Gammon et al. 1981), the Index of Biotic Integrity (Karr 1981; Fausch et al. 1984), both of which are based on fish, and the Invertebrate Community Index (DeShon et al., unpublished), which is based on macroinvertebrates, are used by Ohio EPA. Criteria for each index are defined by organism group, biological index, site type (fish), ecoregion, and aquatic life use designation. The concept has as its basis the design of the Stream Regionalization Project (SRP) and the ecoregion approach (Hughes et al. 1986; Whittier et al. 1987; Ohio EPA 1987a). Modified Iwb and IBI criteria were defined for each of the five ecoregions for three site types; headwaters (drainage area <20 sq. mi.), wading sites (streams sampled with wading methods, usually 20-300 sq. mi.), and boat sites (streams and rivers sampled with boat methods, usually 200-6000 sq. mi). ICI criteria are based primarily on an artificial substrate sampling method and consider the effect of stream and river size. A general rationale and basis for developing these criteria can be found in Volume II of this series (Ohio EPA 1987a) and Whittier et al. (1987).

In 1980 Ohio EPA developed narrative and numerical criteria based on biological evaluation at the sub-community level. Fish (Table 3) and macroinvertebrate (Table 4) criteria were based on the best professional judgement of Ohio EPA biologists and their experiences with a data base including about 150 sampling locations. Some revisions have occurred since 1980, but the approach remained essentially the same. The purpose of the classification system represented by Tables 3 and 4 was essentially two-fold: 1) to provide an objective, systematic basis for assigning aquatic life uses to surface waters, and 2) to provide an objective, standardized approach for determining the magnitude and severity of surface water impacts on the aquatic biota. These criteria also provided the basis for designating stream and river segments as attaining, partially attaining, or not attaining their designated aquatic life uses in the 1982, 1984, and 1986 Ohio 305b reports.

In 1983, Ohio EPA and the U.S. EPA Environmental Research Laboratory in Corvallis, Oregon, initiated a joint effort designated the Stream Regionalization Project (SRP). Conceptual methods for identifying regional patterns in aquatic ecosystems based on the commonality of a variety of physiographic characteristics in a region (i.e. ecoregion) were applied. It was expected that ecoregions could provide a useful framework for examining the potential aquatic life attainabilities for various sizes of flowing waters (Whittier et al. 1987). This approach has been successfully tested in Arkansas (Rohm et al. 1987) and Oregon (Hughes et al. 1987) and is currently being applied in a number of other states. The first phase of the SRP involved delineating the principal ecoregions of Ohio and identifying watersheds within each that are representative of "least disturbed" conditions. This planning effort provided the basis for choosing reference sites (Hughes et al. 1986) that were sampled for fish and macroinvertebrates to determine the biological expectations of the "least impacted" or "reference" sites within each ecoregion. This design is in keeping with the adopted definition of biological integrity. The process for the selection of regional refernce sites is described in Volume II (Ohio EPA 1987a).

Table 3. Ohio EPA biological criteria (fish) that were used for determining water quality use designations and attainment of Water Quality Act (WQA) goals during 1980-1987 (November 1980; revised January 1987).

Category	--- MEETS WQA GOALS ---		------ DOES NOT MEET WQA GOALS ------		
	"Exceptional" Class I (EWH)	"Good" Class II (WWH)	"Fair" Class III	"Poor" Class IV	"Very Poor" Class V
1.[a]	Exceptional, or unusual assemblage of species	Usual association of expected species	Some expected species absent, or in low abundance	Many expected species absent, or in low abundance	Most expected species absent
2.	Sensitive species abundant	Sensitive species present	Sensitive species absent, or in very low abundance	Sensitive species absent,	Only most tolerant species remain
3.	Exceptionally high species richness	High species richness	Declining species richness	Low species richness	Very low species richness
4.[b]	Composite index Greater than 9.0 - 9.5	Composite index Greater than 7.5 - 8.0, Less than 9.0 - 9.5	Composite index Greater than 6.0 - 6.5, Less than 7.5 - 8.0	Composite index Greater than 4.0 - 4.5, Less than 6.0 - 6.5	Composite index Less than 4.0 - 4.5
5.	Outstanding recreational fishery		Tolerant species increasing, beginning to predominate	Tolerant species predominate	Community organization lacking
6.	Species with an endangered, threatened, or special concern status are present				

a Conditions: Categories 1, 2, 3 and 4 (if data is available) must be met and 5 or 6 must also be met in order to be designated in that particular class.

b Based primarily on electrofishing samples, ranges may vary for other sampling methods; does not apply to the Coldwater Habitat (CWH) use designation.

Table 4. Biological criteria (macroinvertebrate) that were used for determining aquatic life use designations and attainment of Water Quality Act goals during 1981-1987 (March 1981).

	CWA GOALS MET		CWA GOALS NOT MET	
Evaluation Class Category	"Exceptional" Class I (EWH)	"Good" Class II (WWH)	"Fair" Class III	"Poor" Class IV
1.	Pollution sensitive species abundant	Pollution sensitive species present in moderate numbers	Pollution sensitive species present in low numbers	Pollution sensitive species absent
2.	Intermediate species present in low numbers	Intermediate species present in moderate numbers	Intermediate species abundant	Intermediate species present in low numbers or absent
3.	Tolerant species present in low numbers	Tolerant species present in low numbers	Tolerant species present in moderate numbers	Tolerant species abundant (all types may be absent if extreme toxic conditions exist)
4.	Number of taxa 30	Number of taxa 25-30	Number of taxa 20-25	Number of taxa 20
5.	Exceptional diversity Shannon index 3.5	High diversity Shannon index 2.9-3.5	Moderate diversity Shannon index 2.3-2.9	Low diversity Shannon index 2.3

The choice of using both fish and macroinvertebrates as the routine organism groups to monitor was made because both groups have been widely used in environmental assessment and there is an abundance of information about their life history, distribution, and environmental tolerances. The need to use both groups is apparent in the ecological differences between them, differences that tend to be complimentary in an environmental evaluation. The value of having both groups independently showing the same result cannot be overstated and lends considerable strength to the evaluations. Apparent differences in the responses of these two groups has usually led to the definition of problems that would have otherwise gone undetected in the absence of information from either organism group. The differing sensitivities of the two groups is not the same to all substances or in every situation. For example representatives of one group may be able to tolerate and metabolize some toxic substances that are highly detrimental to representatives of the other. This information can influence the decision to control certain substances or processes that may have been either overlooked or underrated in relying on an evaluation of only one group. The use of these two groups is somewhat analogous to the use of a fish species and an invertebrate species as standard bioassay test organisms. However, biological field evaluations by definition include a broad range of organism sensitivities and trophic levels from each of the two groups.

Other organism groups such as algae, diatoms, zooplankton, macrophytes, naiad mollusks, and others are not routinely included. However, this does not diminish the ability of Ohio EPA to evaluate and quantify surface water conditions. Fish and macroinvertebrates not only provide some of the most sensitive components of the aquatic biota, but their functioning and overall well-being are dependent on the primary and secondary productivity of the aquatic ecosystem. Thus problems that occur in the lower trophic groups that result in a loss of biological integrity will likely show up in these higher organism groups. It may be necessary on occasion to monitor and evaluate other organism groups for specific purposes such as refining our understanding of cause/effect relationships or verifying the presence of specific chemical contaminants. Our primary goal in using biosurveys is to measure attainment/non-attainment of aquatic life uses. Thus focusing on two of the better understood, higher organism groups will adequately satisfy that goal.

IV. USES AND APPLICATIONS

Having biological criteria in the WQS provides the Ohio EPA with some substantial advantages in surface water quality management and regulation. Some of the current, past, and anticipated future uses include:

- Biological criteria provide objective measurements to determine appropriate aquatic life use designations for surface water bodies.
- Biological criteria provide a means to determine the applicability and use of the antidegradation policy in the WQS (OAC 3745-1-05; e.g. extending antidegradation concerns to nonpoint source influences).
- Chemical and narrative WQS can be more appropriately applied and take into account the integrated dynamics of the receiving waters when relevant biological assessment information is available.
- Management decisions concerning point vs. nonpoint trade-offs can be legally implemented.
- Enforcement against entities discharging substances for which there are no existing water quality standards or effluent guidelines will have a legal basis. At least the impetus to provide a new chemical criterion or regulate the source with bioassay testing will be evident. This is consistent with the requirements of Section 308 of the Water Quality Act of 1987.
- Regulatory programs dealing with habitat manipulation and degradation will have a basis to regulate "non-chemical" environmental degradation (e.g. 401 program) to include consideration of antidegradation provisions.
- Attainment or non-attainment of biological criteria can be used as an initial screening tool for deciding where to invest biomonitoring (bioassay), chemical specific, or wasteload allocation resources. This represents a more comprehensive approach to problem discovery and problem solving than reliance on surrogate measures alone because the evaluation is inherently broad-based (see Fig. 1 and Table 2).
- Agency program results can be evaluated and based on a direct measurement of surface water impairment and condition rather than on surrogate or symptomatic measures alone. This gets at the objective of managing regulatory programs for environmental results rather than administrative goals only (i.e. number of permits issued, etc.).
- The use of biological community response as a supplemental factor in setting water quality-based effluent limitations can elevate a level 1 or 2 exposure assessment (U.S. EPA 1985b) to the equivalent of a level 3 exposure assessment because the biota themselves integrate and reflect factors that are not measured, used, or considered in the traditional level 1 or 2 assessments. The broad abilities of field data to indirectly measure or strongly reflect chemical, physical, and toxicological processes "makes-up" for the deficiencies inherent to the level 1 and 2 approaches.

- Biological community health can be an indirect assessment of potential human health concerns (e.g. tumors and deformities on fish may indicate the influence of toxic organics) and can prompt screening areas for potential cancer risk concerns.

- A knowledge of instream community impairment can more efficiently guide monitoring and regulatory activities that might otherwise rely on a "blanket" approach to information gathering (e.g. effluent characterization, major dischargers lists, etc.).

- In a practical sense, biological assessments allow the reporting of use attainment at the local, state, regional, and national levels.

The use and implementation of biological criteria is in some ways analogous to the use and implementation of chemical criteria. Implementation of this concept will also require the consideration of physical habitat since it plays an important role in determining the biological potential of any surface water.

The longitudinal study design that is fundamental to the Ohio EPA biological monitoring approach will continue to be an important factor in determining water body and entity specific regulatory options. Results of past Ohio EPA biological surveys are presented as examples of how the biological criteria can be used to define and quantify use impairment. Some of these represent a complex combination of point source, nonpoint source, and/or habitat degradation. The discussion of each example is focused primarily on the problem discovery and problem definition capabilities of biosurveys and is not intended to be an all inclusive discussion of each study area.

Mill Creek Example - Typical Point Source Impact

In this example the modified index of well-being (fish), the IBI (fish), and the invertebrate community index (ICI) all show the impact of point sources (Fig. 2) discharging to Mill Creek. The magnitude of the degradation combined with the steady improvement with distance downstream indicate that the effect is predominantly toxic (Ohio EPA 1987c). A comparison of the results from a 1978 survey (not shown) revealed an improvement with the number of miles degraded shrinking from 17 miles to five miles. This was due to the upgrading of the Marysville WWTP. The remaining degradation is attributed to toxicity from an electroplating process discharge which was thought to be in compliance with its NPDES permit. Like many urban areas the effect of general urban runoff and combined sewer overflows (CSO) was evidenced by only slightly depressed biological communities immediately upstream from the point sources. Upstream sampling sites in this study area meet the proposed WWH criteria. A significant source of toxicity was also discovered to be degrading a tributary during this survey.

Wills Creek Example - Diffuse Nonpoint Source Impact

In the second example (Fig. 3) the modified Iwb and IBI are impaired below the WWH criterion well upstream from the principal point sources. The ICI

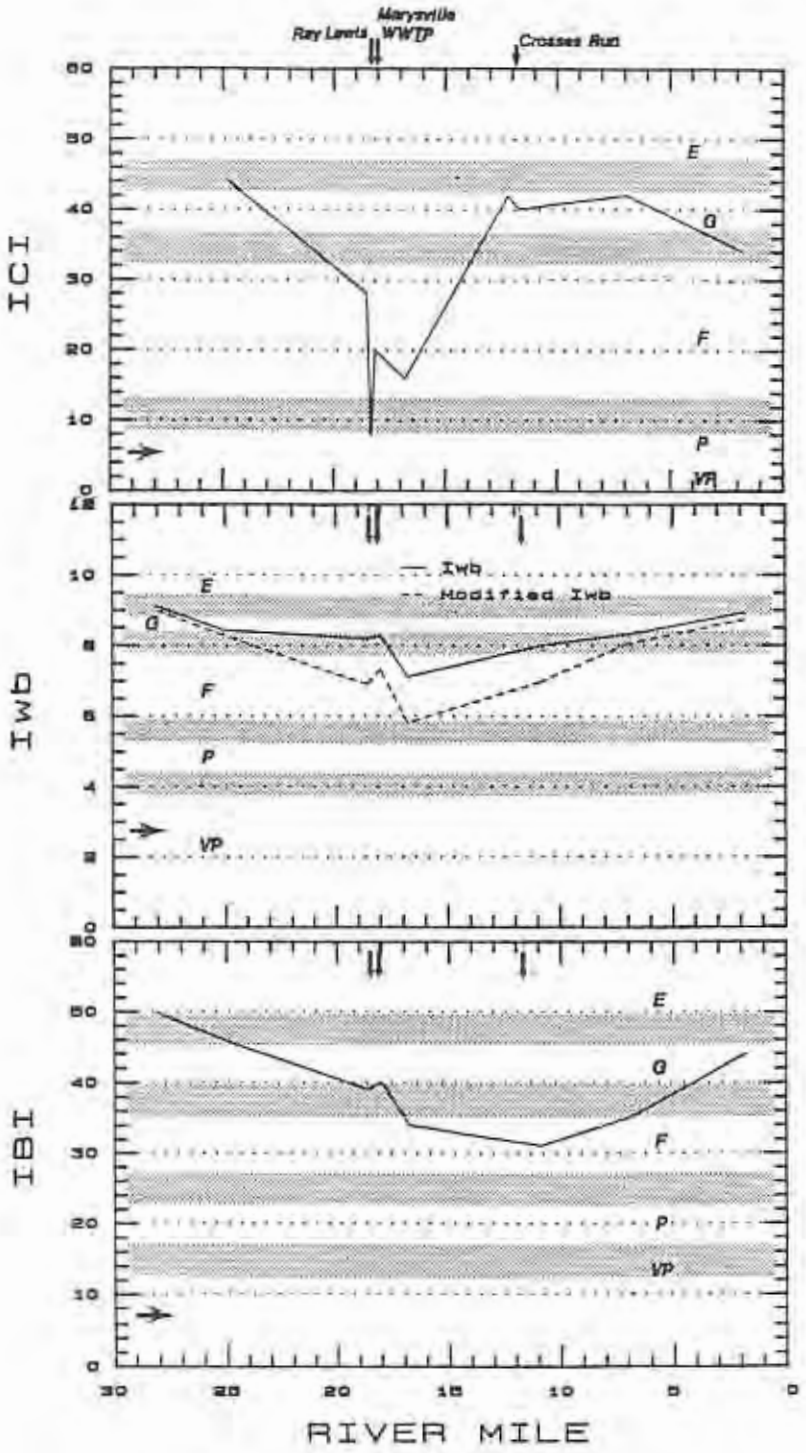

Figure 2. Longitudinal trend of the Invertebrate Community Index (ICI), Index of Biotic Integrity (IBI), and Modified Index of Well-Being (Iwb) in the Mill Creek study area based on samples collected during the period June-September, 1986. Shading indicates boundaries between exceptional (EWH), good (WWH), fair, poor, and very poor conditions and the variability of each index. Environmental influences are indicated on the ICI tier; horizontal arrows indicate direction of flow.

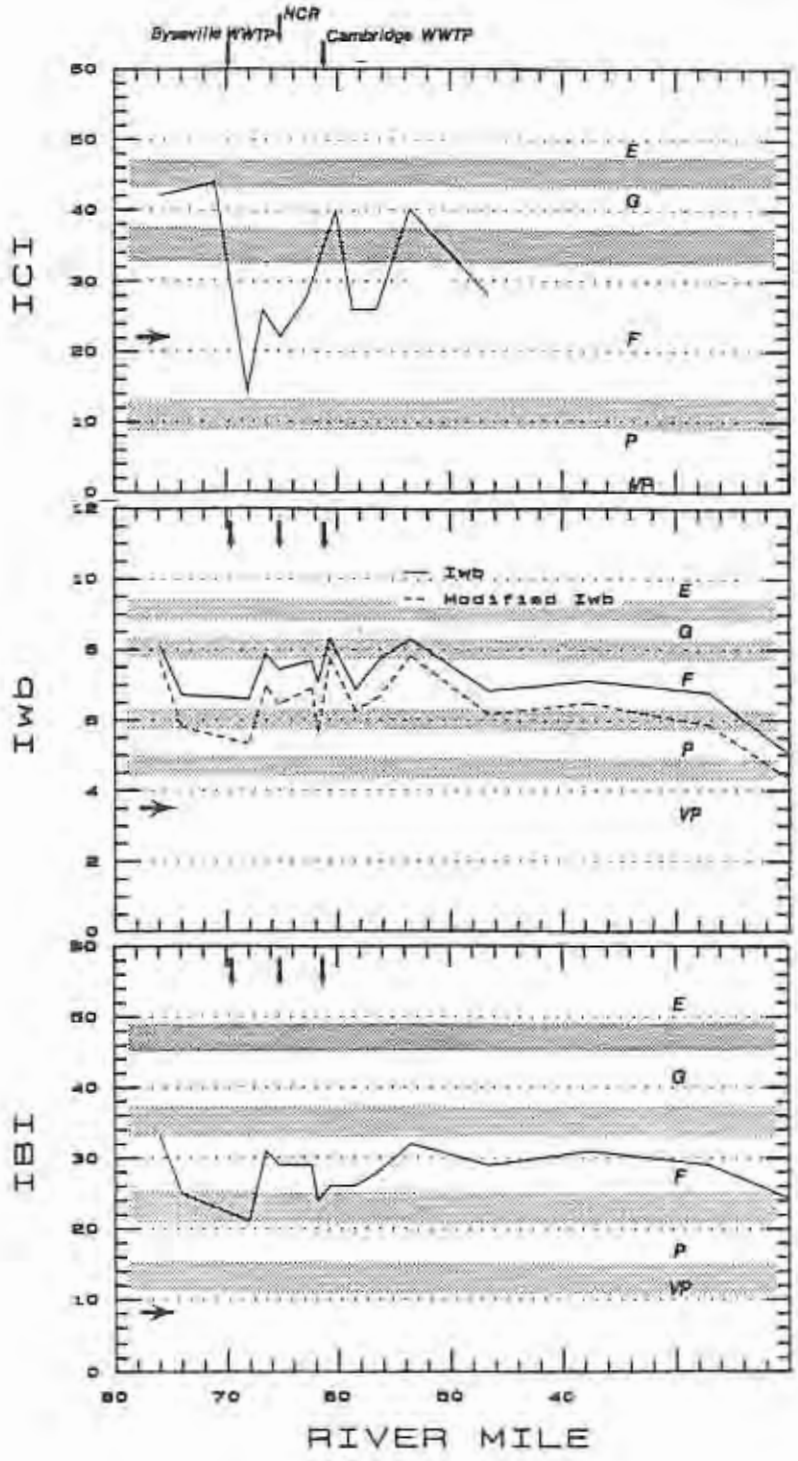

Figure 3. Longitudinal trend of the Invertebrate Community Index (ICI), Index of Biotic Integrity (IBI), and Modified Index of Well-Being (Iwb) in the Wills Creek study area based on samples collected during the period June-September, 1984. Shading indicates boundaries between exceptional (EWH), good (WWH), fair, poor, and very poor conditions and the variability of each index. Environmental influences are indicated on the ICI tier; horizontal arrows indicate direction of flow.

demonstrates this pattern to a lesser degree. In addition, the longitudinal pattern is erratic and largely unrelated to point source influences. The primary source of the degradation was identified as extensive siltation originating from surface mining operations in the upper watershed (Ohio EPA 1985a). The depression of the biota is mostly into the fair range which is typical of non-toxic sediment impacts. Although the point source loadings in the study area might impact a healthy biota, they do not in Wills Creek because the pervasive background siltation problem prevents full biological attainment of WWH. Abatement programs should focus on the nonpoint source problem first because little is to be gained by further controlling point sources.

Scioto River Example - Multiple Year Comparison

Biological data can also be used to document improvements due to pollution abatement efforts over a period of time. Figure 4 shows the summarized results of the modified Iwb and IBI based on six years of electrofishing in the Scioto River downstream from Columbus. The improvement in the biological indices downstream from both the Jackson Pike and Columbus Southerly WWTPs between 1979-81 and 1985-87 corresponds to reductions in discharged loadings of suspended solids, NH_3-N, and BOD. The effluent loading reductions were greatest at the Southerly WWTP, particularly that discharged via the raw sewage bypass. Although structural improvements in the fish community as portrayed by the Iwb have been substantial, evidence of subacute stresses still remain as indicated by the IBI and an elevated percentage of external deformities, eroded fins, lesions, and tumors. The ICI, IBI, and modified Iwb results for selected sampling locations downstream from the Jackson Pike (RM 126.4) and Columbus Southerly (RM 118.1, 113.5, and 102.0) WWTPs show improvement over a 14 year period (Fig. 5). Attainment of WWH criteria for each index occurred first at the sites farthest from the WWTP outfalls and is the most complete downstream from the Columbus Southerly WWTP. This type of analysis which compares instream results with effluent loading reductions over a period of years is what a recent U.S. General Accounting Office report (U.S. GAO 1986) pointed out was lacking in state and federal strategies to assess the direct benefits of improved wastewater treatment. Figures 4 and 5 certainly reveal the benefits of improved wastewater treatment.

Cuyahoga River Example - Complex Toxics Impact

Biological field data have also led to the discovery and improved understanding of significant environmental problems that would otherwise have gone unnoticed, or received less critical attention. An example of this is the toxic impact of the Akron WWTP on the fish and macroinvertebrate communities of the Cuyahoga River. The magnitude, severity, and pattern of the response indicated a severe toxic impact unlike the usual response observed downstream from most WWTPs where low D.O., elevated ammonia-N, and organic solids enrichment is the usual problem (Fig. 6). According to instream and effluent monitoring data, conventional parameters such as D.O., BOD_5, ammonia-N, and phosphorus have improved to levels that are generally compatible with healthy aquatic communities. Concurrent and follow-up bioassay testing found acute toxicity in the influent, bypass, and the receiving stream in 1984 and early 1985. Toxicity was reduced in late 1985 and substantially gone in 1986. The fish community showed some very modest structural improvements in 1985 and 1986 (e.g. higher numbers and biomass of

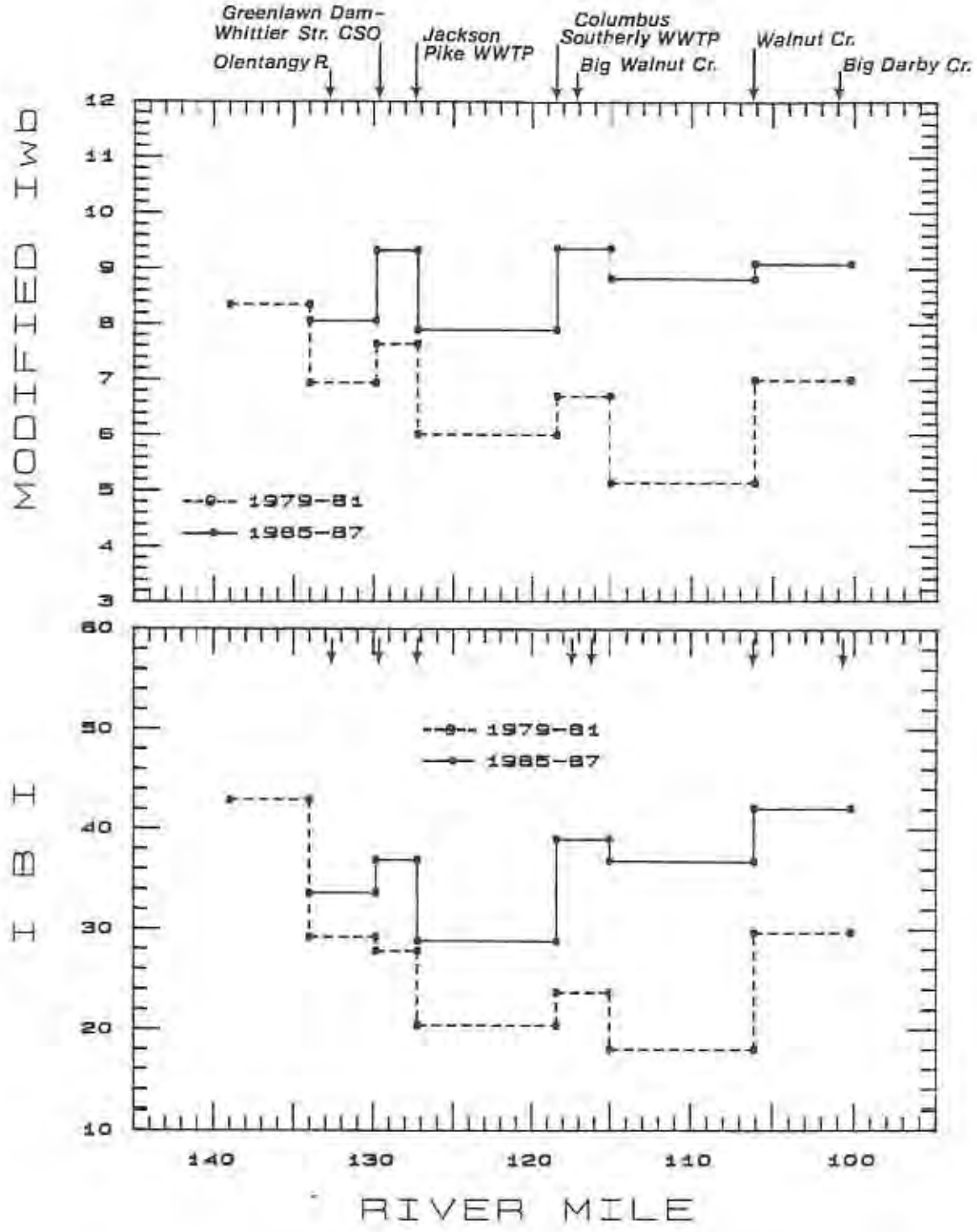

Figure 4. Summarized results of mean Modified Index of Well-Being (Iwb) and Index of Biotic Integrity (IBI) values in seven segments of the Scioto River between Columbus and Circleville for 1979-1981 and 1985-1987. Samples were collected during the period July-October of each year. Environmental influences are indicated; horizontal arrows indicate direction of flow.

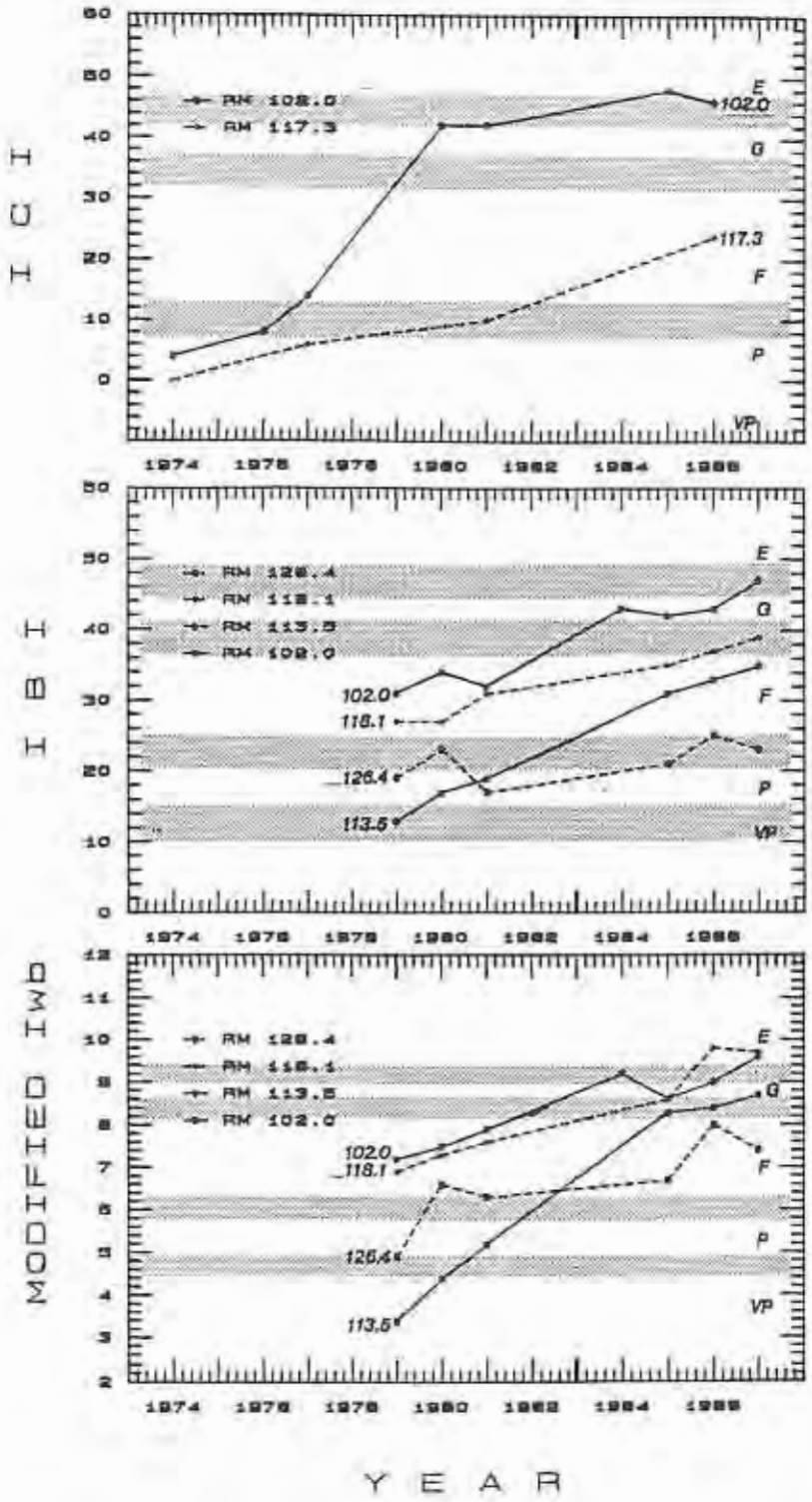

Figure 5. Results of the Invertebrate Community Index (ICI), Index of Biotic Integrity (IBI), and Modified Index of Well-Being (Iwb) at selected sampling locations in the middle Scioto River between 1974 and 1987. Shading indicates boundaries between exceptional (EWH), good (WWH), fair, poor, and very poor conditions and the variability of each index. Sampling was conducted during July-October of each year.

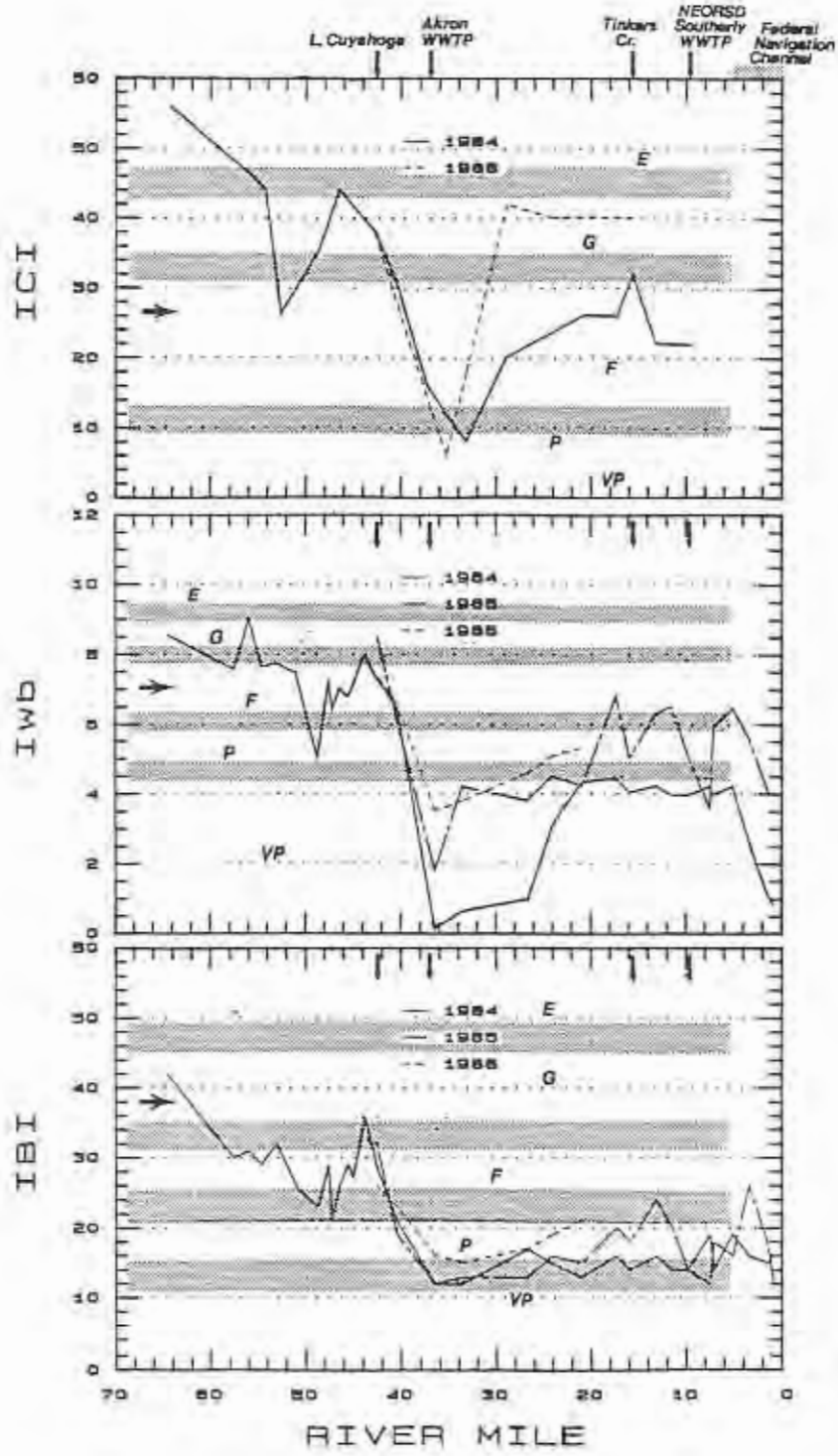

Figure 6. Longitudinal trend of the Invertebrate Community Index (1984, 1986) and Modified Index of Well-Being (fish; 1984, 1985, and 1986) in the Cuyahoga River study area principally from the Little Cuyahoga River to the diversion dam based on samples collected during the period June-September. Shading indicates boundaries between exceptional (EWH), good (WWH), fair, poor, and very poor conditions and the variability of each index. Environmental influences are indicated; horizontal arrows indicate direction of flow.

tolerant species), but was functionally degraded showing the remaining effects of a significant toxic stress. Of particular note was the very low IBI and remaining high incidence of skeletal deformities and other anomalies on fish which is interpreted as an indication of continued subacute stress. Macroinvertebrate results showed a similar magnitude of impact in 1986, but the length of the impact was lessened compared to the 1984 results. Both the fish and macroinvertebrate evaluations revealed that poor and very poor conditions remained for several miles downstream in 1986, although the number of miles at this condition was reduced from 1984. This is a classic example of how toxics problem discovery, quantification and identification can be accomplished using instream biological community response. In this situation (and possibly others) the use of only the chemical and bioassay testing techniques advocated by U.S. EPA (1985b) could have significantly underrated a serious environmental problem.

Little Cuyahoga River Example - Biological Problem Discovery

The results of a 1986 survey of the Little Cuyahoga River subbasin show the important use of biological data to reveal environmental degradation that would otherwise have gone unnoticed. The pattern of biological community response (Fig. 7) indicates severe toxic impacts in the upper and middle portions of the mainstem and impacts of a combined toxic and organic sewage nature in the lower segment. Of particular note was the lack of any violations of chemical WQS under low flow conditions. Visual observations revealed mixed, hard substrates (no extensive sludge deposits, etc.) and good water clarity, i.e nothing that seemed to show the magnitude of the problem that was biologically evident. One set of high flow samples following a period of rainfall indicated some WQS violations for heavy metals associated with combined sewer discharges in the lower portion of the mainstem. Still, the biological response was typical of what we have observed in other similarly affected areas. The biological response in the middle portion of the Little Cuyahoga River indicated the severe impact of toxic substances. Several point sources located in this segment are authorized to discharge non-contact cooling water and sanitary wastes. Thus if the scope of information covered by the NPDES permits was comprehensive there should be no such impact. Most of these entities have plastic and rubber manufacturing processes and therefore use and handle organic chemical products on their premises. The observed instream biological response indicates that some contamination of the receiving stream is occuring at least frequently enough to keep the resident biota suppressed. A follow-up inspection of the study area might focus on how chemicals could be reaching the stream, i.e. accidental spills, contamination of cooling water, contaminated area runoff, or unauthorized discharges. In this case current NPDES monitoring and discharge requirements may be inadequate.

The Cuyahoga and Little Cuyahoga River examples demonstrate the important use of biological field evaluation in supplementing toxics strategies for point sources. Toxics programs currently have a strong tendency to dwell on process and effluent characterizations, effluent toxicity, and chemical specific analyses of the process wastewaters. These are traditional "end-of-the-pipe" focuses. A significant concern with this approach is seemingly its inability to accurately assess and characterize impacts that occur via pathways to the receiving waters other than process discharges (e.g. runoff, "non-contact" cooling water, spills, dumping).

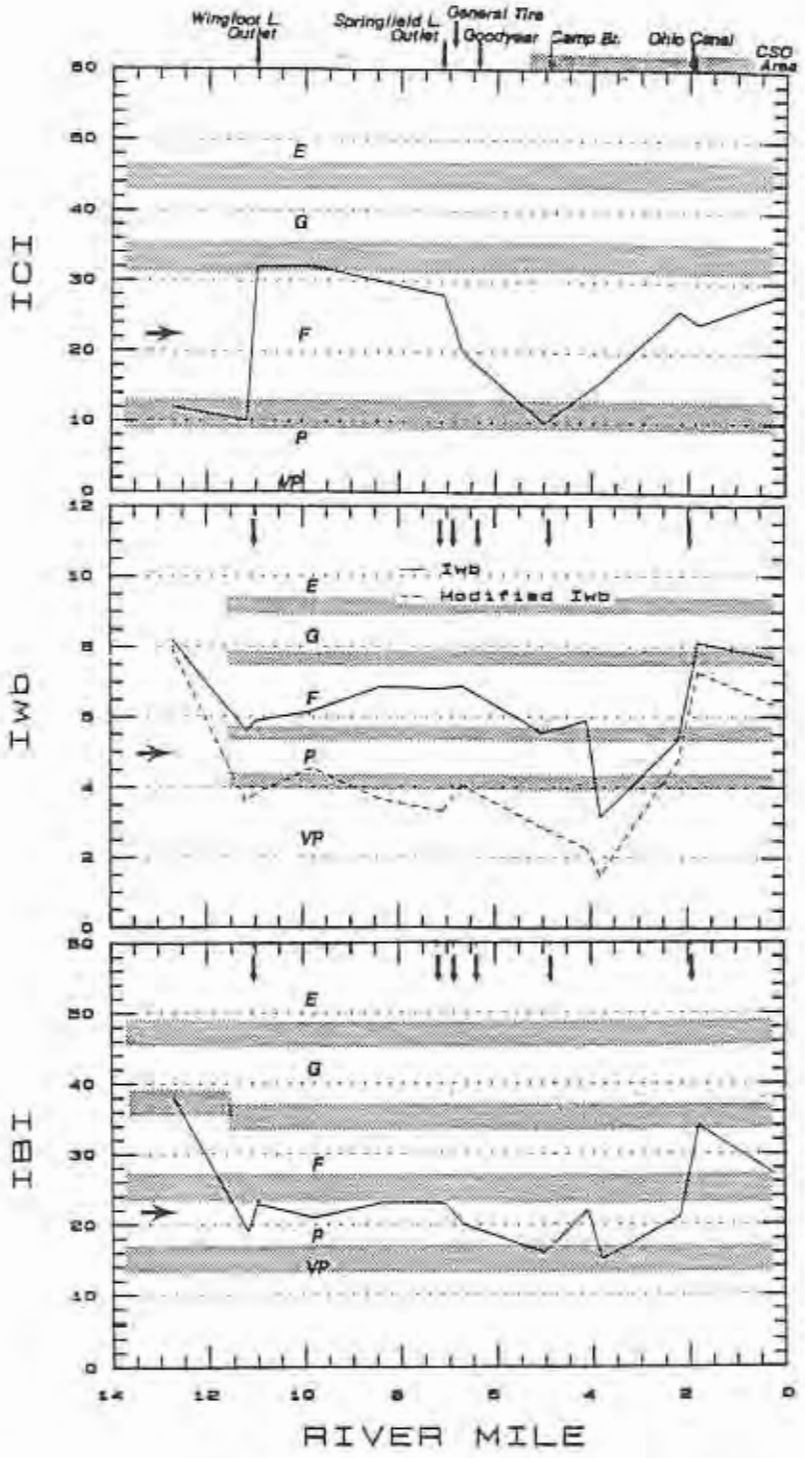

Figure 6. Longitudinal trend of the Invertebrate Community Index (ICI), Modified Index of Well-Being (Iwb), and the Index of Biotic Integrity (IBI) in the Little Cuyahoga River mainstem based on samples collected during the period July-September 1986. Shading indicates boundaries between exceptional (EWH), good (WWH), fair, poor, and very poor conditions and the variability of each index. Environmental influences are indicated on the ICI tier; horizontal arrows indicate direction of flow.

Nimishillen Creek Example - The Added Value of Biological Assessment

There are also impacts that result from discharges via permit regulated pipes, but which may go undetected because of unknown and/or unmonitored substances in the process or because of intermittent releases coupled with inadequate monitoring frequencies. An example of where emphasis on limited aspects of toxics evaluation and assessment can result in overlooking significant environmental problems is with the results of two different types of monitoring conducted in the Nimishillen Creek basin in northeastern Ohio. One effort was conducted by U.S. EPA and included chemical monitoring of selected surface waters, effluents, and sediments in the Nimishillen Creek basin. This included sampling efforts during July 29-30, 1980 (U.S. EPA 1982), August 2-3, 1982 (U.S. EPA 1983b), and August 4-5, 1982 (U.S. EPA 1983c). The stated objectives of these studies were to identify and document water quality problems in the study area, evaluate point source discharges with respect to NPDES permit limitations, and provide baseline data for evaluating future changes that are anticipated with improved wastewater treatment. Chemical concentrations of several substances in excess of applicable water quality standards (WQS) or U.S. EPA criteria were found in the receiving streams downstream from the principal point sources. Concentrations of several substances in the bottom sediment were at levels considered to be "polluted" and "heavily polluted". This included several EPA priority pollutants.

Ohio EPA conducted a biological and water quality survey of the Nimishillen Creek basin in 1985. This effort included chemical (water column, sediment, effluents) and biological (fish, macroinvertebrates) sampling at more than 50 locations. Follow-up work was conducted in selected locations in 1986. Significant impairment of the WWH use designation was biologically evident in all major tributaries and the mainstem of Nimishillen Creek. Approximately 25 cumulative miles of biological impairment was attributed to two point sources. Nonpoint sources (mainly urban encroachment and runoff) and combined sewer overflows caused biological non-attainment of WWH in other parts of the study area. Of the two point sources causing the 25 miles of degradation one is not permitted and the other is "under" permitted. The unpermitted discharge (Gregory Galvanizing) has been a chronic source of complaints for approximately 15 years. Previous attempts to locate a discharge pipe have apparently failed. Although discharge pipes were located in 1986 the source of the continuing impact appears to be general contamination and leaching from the plant site. Figure 8 shows the longitudinal trend of the modified Iwb, IBI, and ICI in the vicinity of this problem. Non-attainment of the biological criteria for this ecoregion is clearly indicated. Concurrent chemical sampling showed this facility to be the principal source of high zinc and very high iron concentrations in the Nimishillen Creek mainstem and downstream into Sandy Creek.

The "under-permitted" point source caused a significant biological impairment in the E. Branch of Nimishillen Creek (Fig. 9). The macroinvertebrate results (ICI) reflected a severe toxic impact from a steel finishing facility (J&L Specialty Steel) while the fish results (Iwb, IBI) additionally reflected the impacts of sewage enrichment from a WWTP. Concurrent and follow-up chemical monitoring revealed elevated concentrations of two substances (arsenic, nitrite-N) that were not found by the U.S. EPA surveys or previously known of in either the receiving stream or the effluents.

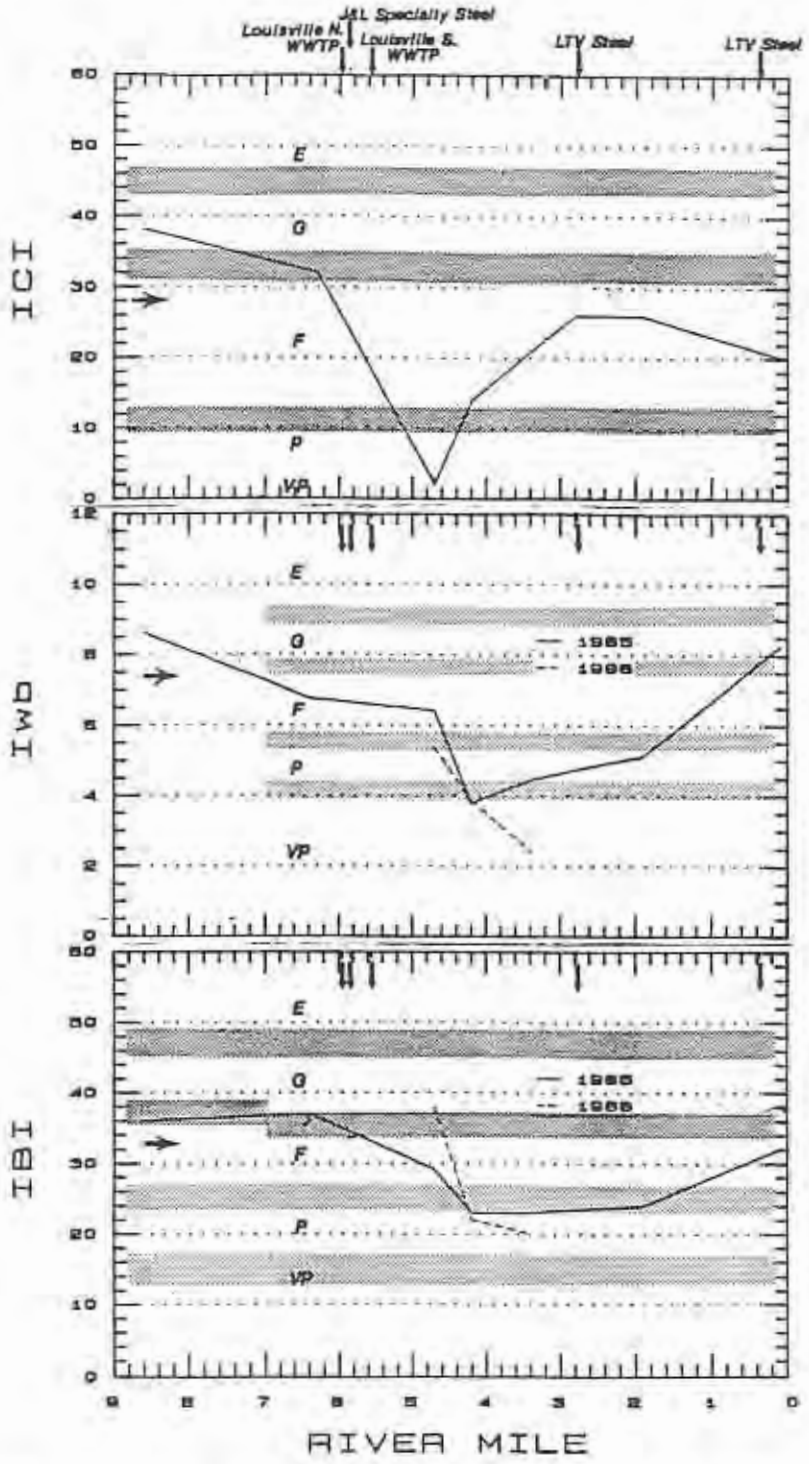

Figure 8. Longitudinal trend of the Invertebrate Community Index (ICI), Modified Index of Well-Being (Iwb), and the Index of Biotic Integrity (IBI) in the Nimishillen Creek mainstem and the W. Branch based on samples collected during the period July-September 1985 and 1986 (solid circles are mainstem values upstream from the W. Branch). Shading indicates boundaries between exceptional (EWH), good (WWH), fair, poor, and very poor conditions and the variability of each index. Environmental influences are indicated on the ICI tier; horizontal arrows indicate direction of flow.

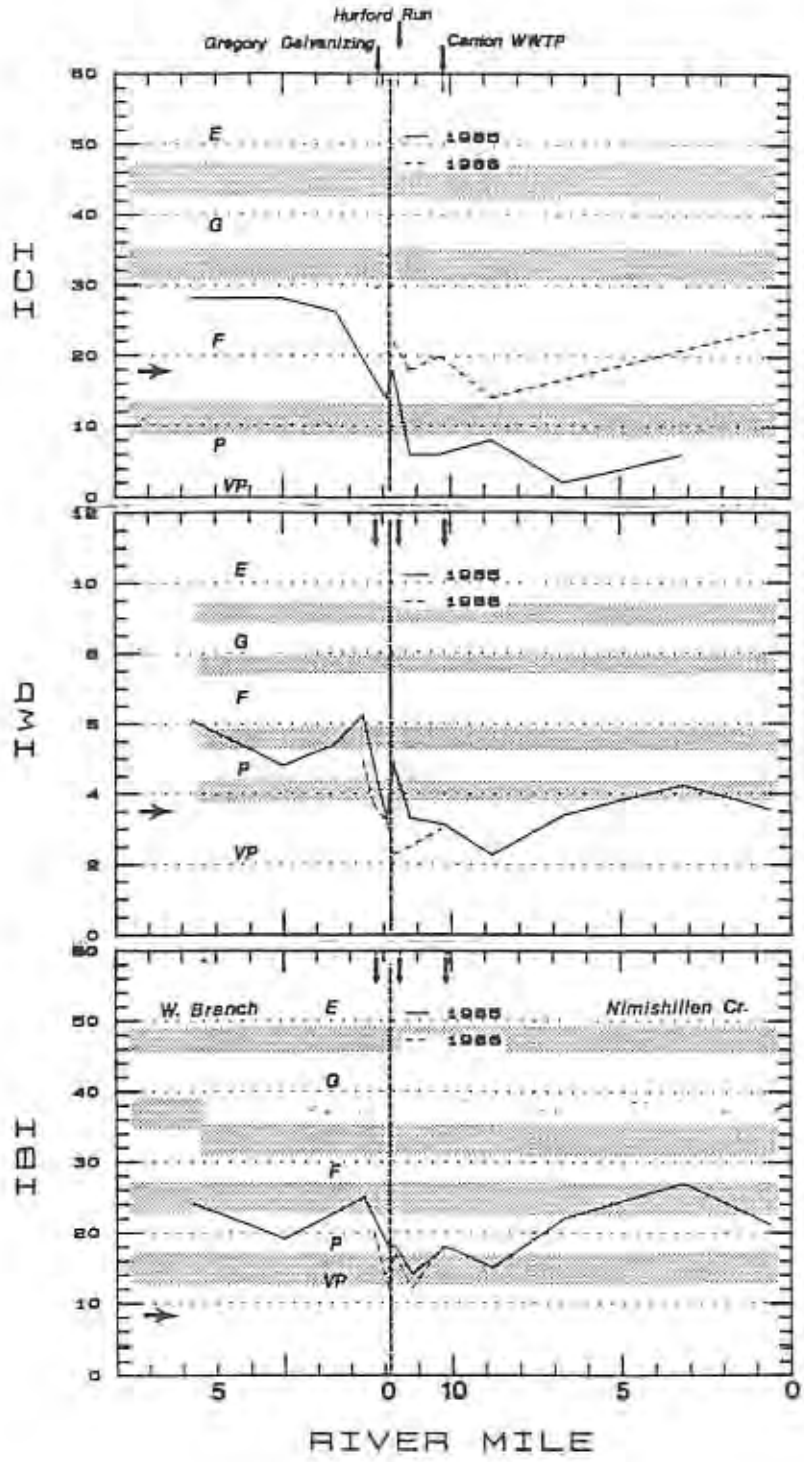

Figure 8. Longitudinal trend of the Invertebrate Community Index (ICI), Modified Index of Well-Being (Iwb), and the Index of Biotic Integrity (IBI) in the E. Branch Nimishillen Creek based on samples collected during the period July-September 1985 and 1986. Shading indicates boundaries between exceptional (EWH), good (WWH), fair, poor, and very poor conditions and the variability of each index. Environmental influences are indicated on the ICI tier; horizontal arrows indicate direction of flow.

While both the U.S. EPA and Ohio EPA monitoring efforts revealed water quality problems due to point sources the U.S EPA studies were limited to identifying problems with only the chemical substances that were analyzed. While it is important to know about the presence and concentrations of chemicals in surface water systems this information alone is inadequate for quantifying and fully understanding complex pollution problems. A much different understanding of the water quality problems in the E. Branch Nimishillen Creek was gained from the Ohio EPA studies. The inclusion of biological sampling for both fish and macroinvertebrates and the inclusion of additional chemical parameters led to the defintion of problems that were either missed or underrated by the U.S. EPA studies. The Ohio EPA results were used to holistically quantify (biologically and chemically) the magnitude and severity of aquatic life use impairment. The nature and magnitude of the biological impacts observed in the mainstem and E. Branch were used as a "cue" to chemically pinpoint and identify the problem sources. This example demonstrates the significant risks of partially and/or incorrectly characterizing environmental problems that is inherent to "chemical only" monitoring.

General Observations

These examples demonstrate that aquatic life impacts do appear in places other than the effluent and in modes other than acute toxicity to bioassay test organisms, or as numerical violations of chemical criteria. A broad-based monitor such as the biosurvey/ecoregion approach is needed to ensure that such problems are adequately detected and understood. Simply put there is no substitute for the receiving stream biota to indicate and quantify environmental impacts because they are present all of the time and are subject to all of the pollution episodes. In this way biological community response can also be used as a prioritizing mechanism for deciding where to conduct more detailed site-specific analyses such as bioassay, chemical specific characterizations, etc. This seems prudent from an environmental cost-effectiveness standpoint because limited regulatory resources can be focused on problems that are having the greatest impacts and hence the most potential for environmental benefit when they are corrected. This approach is currently being used to develop the annual list of Priority Water Quality Areas for construction grants and is the focus of the 1988 effort to develop lists of toxic and nonpoint impacted waters.

Our experiences show that biological field evaluation using fish and macroinvertebrates represents an accurate and sensitive measurement of environmental problems and progress in pollution control. For example D.O. (and associated chemical measures such as sediment oxygen demand, etc.) is a popular measurement for showing instream improvements following sewage plant upgrades. While this measure is important in understanding certain instream processes it remains an indirect and incomplete measure of biological integrity. Other influences exerted by sewage plant discharges that can influence the aquatic commmunities will be missed by dwelling on D.O. alone. These other influences are detected in the response of the aquatic communities. It is the broad-based ability of the biota to integrate environmental conditions that makes this important distinction.

Biological Criteria - Advantages and Limitations

Biological data alone cannot perform every task necessary in a water quality management program. Disciplines related to effluent characterization, chemical specific assessment, bioassay testing, and wasteload allocation are all essential. The need is for all of these disciplines to be truly integrated in an analytical (as opposed to rigidly mechanistic) framework in their application to water quality-based surface water regulation. Although biological field evaluations alone can be used to identify general degradation "types", a precise definition of cause usually requires the inclusion of other tools (i.e. chemical, bioassay). However, they can provide insights into site-specific applications of those tools and suggest possible cause and effect relationships. This is best accomplished when the results are interpreted by a trained biologist aided by examples of biological response under a wide range of perturbations.

The use of the biosurvey/ecoregion approach is an essential requirement for establishing biological criteria. However, like any other technical endeavor it is important to recognize and be aware of potential limitations. What might be perceived as some of the potential limitations of this approach include:

- o the accuracy of the ecoregions is limited - some flexibility in their use and application must be incorporated;
- o the approach cannot be mechanically applied - expert judgement is required;
- o natural or real world variability is not eliminated - it is an essential part of the process; and,
- o one set of ecoregions does not apply equally well to all situations - again judgement by regional experts is required.

In contrast the biosurvey/ecoregion approach has some important advantages. These include:

- o it provides a framework for setting ecologically meaningful uses and criteria;
- o it provides a framework for organizing monitoring programs and the reporting of results;
- o it provides a systematic, standardized basis for assessing progress by pollution control programs; and,
- o the approach requires a standardized, consistent application of methods and procedures.

The adoption of biological criteria does not mean that the current system of tiered aquatic life uses, accompanying chemical criteria, and application techniques should be radically altered. End-of-pipe wastewater regulation is

and will continue to be primarily a numerical system, therefore the use of chemical and bioassay techniques for establishing discharge limitations will continue to be an important part of the process. However, they will not be the only parts as has been the case.

A "systems approach" to surface water assessment and regulation is needed to compliment the traditional "regulatory approach". Although the latter employs specific methods that identify and quantify specific chemicals and levels of toxicity to aquatic organisms, extrapolating these results to an assumed level of impairment in a receiving water body still depends on making assumptions about exposure periods and exceedence frequencies. Determining "how much is too much" is too often made on an indirect, best professional judgement type basis. By comparison using biocriteria based on the functional and structural attributes of resident biological communities allows for a much more direct and holistic assessment. This information is used to determine the degree to which a water body is attaining or not attaining its designated use. This type of assessment is strengthened further when the overall approach has been "callibrated" by knowing what level of biological performance can be attained by the natural habitats within an ecoregion. Thus the need to make assumptions about exceedence frequencies for chemical and bioassay results can be eliminated by having a direct assessment of use attainment/non-attainment.

V. MANAGEMENT APPLICATIONS

There are two principal areas where biological criteria can improve and enhance overall water quality management programs. These are the demonstration of use attainment/non-attainment of aquatic life uses in surface waters and the discovery and quantification of aquatic environmental degradation. Within the context of these two major areas of application are a number of benefits to different classes of surface water regulation (i.e. NPDES permits, nonpoint source management, habitat protection, etc.).

Environmental Problem Discovery

The overall success of environmental regulation is determined in part by how successful we are at defining problems. The role of biological criteria in this process is to provide a holistic characterization of problems that eventually leads to their correction. The Nimishillen Creek and Cuyahoga River examples show that we cannot regulate that which we do not know about. Furthermore, we will not be aware of many significant problems without direct information from the receiving waters. The inherently broad ability of biological communities to detect problems makes them particularly well suited for problem discovery and quantification. One distinct advantage is that sampling need not be conducted during "critical" periods such as extreme low-flows or high flow runoff periods. It was pointed out earlier that the inherent characteristics of biological communities make them continuous indicators of past events and conditions. Thus surface water regulatory programs should derive significant benefits from having biological field monitoring and objective evaluation criteria.

Water Quality-Based Effluent Limits

The Ohio EPA Five-year Surface Water Monitoring Strategy (Ohio EPA 1985b) outlines general management procedures for using surface water monitoring data to conduct water quality-based effluent limit determination. In this application the role of the biological criteria will be to provide an objective assessment of whether or not aquatic life uses are attained in the surface water body affected by the point source(s). The general rationale of this approach is described in Chapter 3 of the Five-year Monitoring Strategy.

There are examples where biological impairment has not been evident under effluent loadings that were predicted to cause WQS violations by a Level 1 exposure assessment. Obviously in these situations the complexities and dynamics of the receiving water environment are such that these WQS applications are incomplete and the indications of use attainment should take precedence. However, when biological impairment is observed there are currently few practical alternatives to using a Level 1 or 2 exposure assessment to set water quality-based effluent limits. The role of the biological impact assessment data is to provide a site-specific perspective on the application of these predictive techniques.

A shift in emphasis on only numerical limits to one which includes the relationship between entity performance and the response of the biota in the receiving waters is needed. Too often the success of pollution abatement

programs are measured by the degree of compliance with numerical limits in the absence of knowledge about instream conditions. This has resulted in a tendency to dwell on what it will take to meet permit limits 100 percent of the time rather than directing attention to what the actual operation of a facility has achieved in the receiving waters. This approach is likely to be more cost-effective because it includes reliance on observed rather than predicted environmental results alone.

Finally, this approach further fills some important "gaps" in surface water resource management. Biosurveys and biocriteria provide a "reality check" on regulatory actions. Given the potential for the traditional chemical/toxicity approach to be under or over protective this is a significant benefit. A direct measure of use attainment is provided that portrays the resultant efforts of pollution control agencies in tangible terms. For example, it is much more meaningful to show compliance with calibrated and verified IBI, ICI, and/or Iwb than it is to only dwell on the number of actions (e.g. number of permits issued, etc.) taken, although the latter is an indispensible function of a regulatory agency. Quantifying conditions for the application of anti-degradation to the higher quality resources is another important function of biosurveys. The biocriteria and biosurvey/ecoregion approaches provide a "feedback loop" for pollution control efforts by quantifying the use attainment status of surface waters. These latter two functions will be particularly useful in the emerging nonpoint source management and abatement programs.

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