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

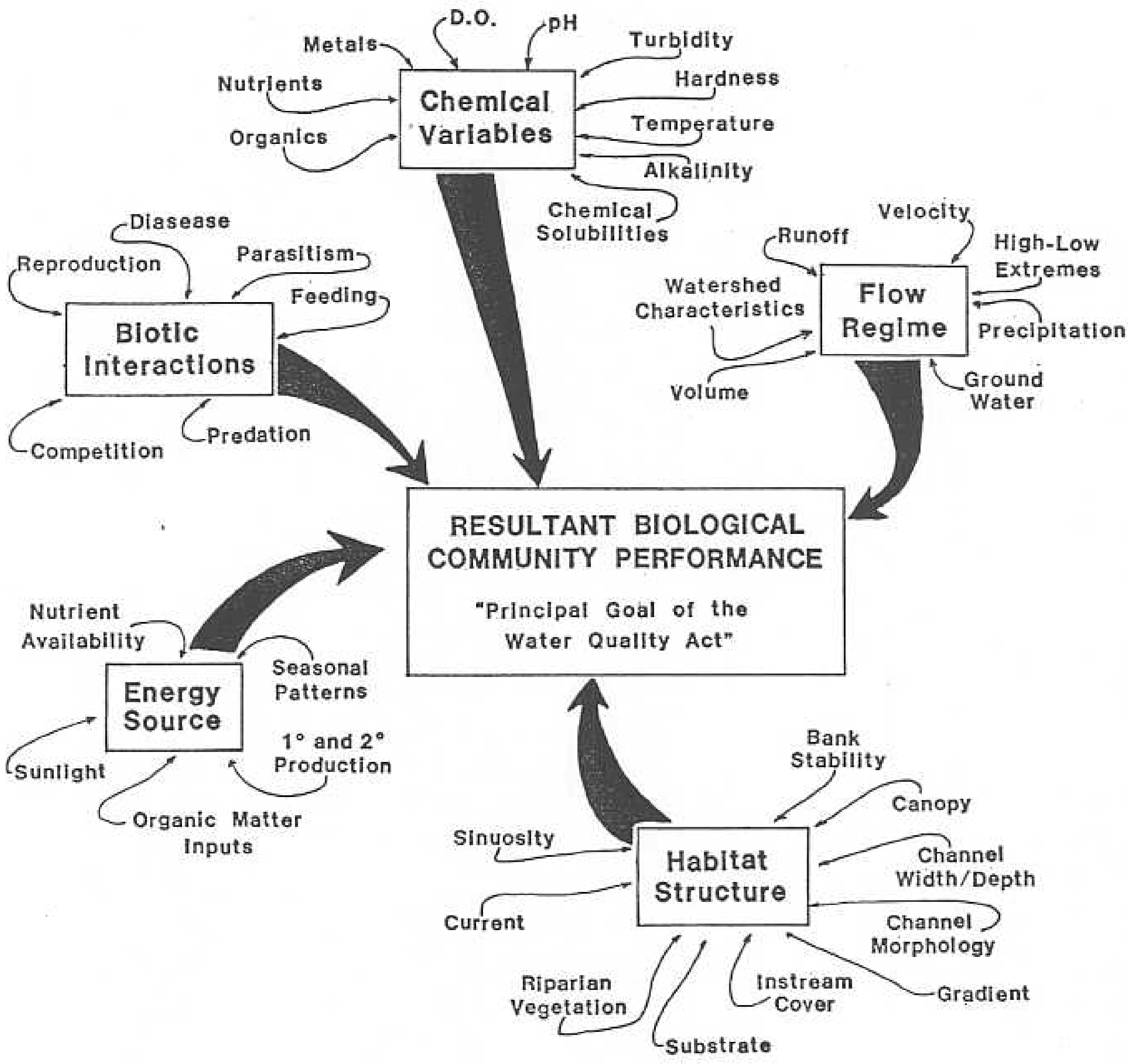

Figure 1. Some of the important chemical, physical, and biological factors which influence and determine the resultant biological community performance in surface waters showing the five principal factors and their various components (modified from Karr et al. 1986; 1° and 2° are primary and secondary,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

- 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.

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

- 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. $Q_{7,10}$ flow). Other low-flow values are also used (95% duration flow, $Q_{30,10}$ flow) and these approximate the $Q_{7,10}$ 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. $Q_{7,10}$, 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 intensive 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
Macroinvertebrate 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).