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BIOLOGICAL INTEGRITY: A LONG-NEGLECTED ASPECT OF WATER RESOURCE MANAGEMENT[1]

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Abstract. Water of sufficient quality and quantity is critical to all life. Increasing human population and growth of technology require human society to devote more and more attention to protection of adequate supplies of water. Although perception of biological degradation stimulated current state and federal legislation on the quality of water resources, that biological focus was lost in the search for easily measured physical and chemical surrogates. The "fishable and swimmable" goal of the Water Pollution Control Act of 1972 (PL 92-500) and its charge to "restore and maintain" biotic integrity illustrate that law's biological underpinning. Further, the need for operational definitions of terms like "biological integrity" and "unreasonable degradation" and for ecologically sound tools to measure divergence from societal goals have increased interest in biological monitoring. Assessment of water resource quality by sampling biological communities in the field (ambient biological monitoring) is a promising approach that requires expanded use of ecological expertise. One such approach, the Index of Biotic Integrity (IBI), provides a broadly based, multiparameter tool for the assessment of biotic integrity in running waters. IBI based on fish community attributes has now been applied widely in North America. The success of IBI has stimulated the development of similar approaches using other aquatic taxa. Expanded use of ecological expertise in ambient biological monitoring is essential to the protection of water resources. Ecologists have the expertise to contribute significantly to those programs.

Key words: biological integrity; biological monitoring; fish community; Index of Biotic Integrity (IBI); indexes of degradation; indicators; water pollution; water resources.

INTRODUCTION

Degradation of water resources has long been a concern of human society. Regions with dense human populations were the earliest areas at risk, but waters in isolated areas have also experienced degradation. The earliest anthropogenic threats to water resources were often associated with human health, especially disease-causing organisms and oxygen-demanding wastes (Meybeck and Helmer 1989). Early emphasis (e.g., the saprobic system: Kolkwitz and Marsson 1908, 1909) was on controlling these contaminants in urban areas where effluents exceeded the natural waste assimilation capabilities of waters. An industry developed to collect, treat, consolidate, and release household sewage through point-source outflows. The goal was to see that the streams' or lakes' ability to assimilate those wastes were not exceeded, using the philosophy that "dilution is the solution to pollution." As technology advanced, chemical and physical indicators became the primary regulatory tool to protect water resources.

However, continuing declines in the quality and quantity of water resources despite massive regulatory efforts call attention to the inadequacies of existing programs (EPA 1987, 1988*a, c,* 1989*c,* 1990, General Accounting Office 1987, National Resource Council 1987, Simon et al. 1988, Davis and Simon 1989, Day 1989). A Government Accounting Office study (General Accounting Office 1989) in United States Environmental Protection Agency (USEPA) Region 10 (Northwest United States) showed that 602 segments of rivers and streams are water-quality limited (i.e., limited by chemical contamination). Further, a nationwide United States Fish and Wildlife Service survey found reduced fishery potential because of chemical problems in 56% of the stream segments with water resource degradation (Judy et al. 1984). Of equal concern, the study found that 49% were impaired by degradation in physical habitat and 67% by flow alteration, neither of which are treated by existing USEPA programs. In 1986, USEPA acknowledged that nonpoint sources affect 65% of impaired stream miles, 76% of impaired lake acres, and 45% of impaired estuary square miles (General Accounting Office 1989; citing USEPA 1986 National Water Quality Inventory Report to Congress). From 1972 to 1982, four times more lake acreage deteriorated than improved in quality (Johnson 1989). Because most water resource programs concentrate on human health rather than a broader array of natural resource issues, many water resource problems persist (Huber 1989).

As human populations and their technology increase, impacts, such as the following, are too diverse

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for chemical control approaches to protect the resource (Karr and Dudley 1981, Karr et al. 1985*b*): (a) production of domestic effluents, (b) erosion following alteration of landscapes by agriculture, urbanization, and forestry, (c) alteration of stream channels and lake margins through dams, channelization, drainage and filling of wetlands, and dredging for navigation, (d) diversion or other flow alteration, (e) overharvest of biological resources, and (f) proliferation of toxic chemicals from point and nonpoint sources. Treatment of the impact of multiple stresses is in its infancy (Preston and Bedford 1988, Cairns and Niederlehner 1989). Recognition of these problems stimulated research to develop improved approaches for assessing the integrity or ecological health of water resource systems (Karr 1981, Karr et al. 1986, Ohio EPA 1988, Plafkin et al. 1989). Growing concern about the need to resolve the biodiversity crisis (OTA 1987) is a parallel but broader problem. For the first time in several decades, the opportunity to alter society's approach to the protection of water resources presents itself.

Several federal agencies and many states are calling for evaluation and implementation of programs of direct biological monitoring. New philosophies guide these efforts and signal major shifts that will be instrumental in "restoring and maintaining" biological integrity of the nation's waters, the explicit mandate of PL 92-500 (Water Quality Act Amendments of 1972) and amendments. USEPA has called for the following: (a) inclusion of biological criteria in its water-quality standards program (EPA 1988*a*), (b) restructuring of existing monitoring programs to document the impact of regulatory programs (EPA 1989*a*), (c) evaluation and control of nonpoint pollution (EPA 1989*b*), (d) coordination of chemical sampling with biological surveys (EPA 1984), (e) ecological risk assessment (EPA 1988*b*), (f) the incorporation of "good science" at all levels of water resource policy (EPA 1988*c*, 1989*b*), and (g) adoption of narrative biological criteria into state water-quality standards during the FY 1991-1993 triennium (EPA 1990). (Biological criteria [Biocriteria] is a code word for "indicator of ecological conditions" or "biological integrity compared to some least disturbed reference community.") EPA (1988*a*) and others (Van Putten 1989) call for development of biological criteria to protect terrestrial wildlife from the negative effects of human activities on water resources. United States Geological Survey (Hirsch et al. 1988) and Tennessee Valley Authority (Saylor and Scott 1987) are expanding their use of biological monitoring as well.

Most states are expanding their biological monitoring programs (Simon et al. 1988, Davis and Simon 1989, Kilkelly Environmental Associates 1989), although the approaches differ among states. Some have adopted legal biological criteria (Florida, Vermont), biological-based use designations (e.g., excellent warmwater habitat: Maine, Courtemanch et al. 1989; Arkansas, Rohm et al. 1987), and biological criteria in assessments and monitoring (Michael et al. 1989). Following a detailed, statewide program to evaluate ambient (field) biological monitoring, Ohio is incorporating biological monitoring into regulations for attainment of the goals of the Clean Water Act. An additional 16 states have active interest and 23 other states are expressing interest in biocriteria (Marcy 1989). Some (Colorado) retain a toxic and effluent focus while others (Ohio) incorporate a broader biological integrity goal. These advances came about because of recognition that water resource problems involve biological as well as physicochemical and socioeconomic issues.

Philosophical shifts within state and federal agencies suggest that the short-sighted and incomplete approach to water resources management ("making clean water will solve water resource problems") can be overcome. Replacement of this approach with sophisticated, quantitative assessments based on ecological principles is more likely to protect water resources from the wide range of human actions that degrade those resources.

Whereas the foundations for these advances have existed for perhaps two decades three factors have contributed to rapid advances in the last decade: (a) the development of integrative ecological indexes (Karr 1981, Karr et al. 1986, Ohio EPA 1988, Plafkin et al. 1989), (b) the development of the ecoregion approach (Hughes et al. 1986, 1987, Omernik 1987, Hughes and Larsen 1988), and (c) recognition of the importance of cumulative impact assessment at regional scales (Preston and Bedford 1988). The challenge for basic and applied ecologists in the next decade will be to ensure that ecological principles are used to improve the nation's programs to protect and manage water resources. Another benefit will be the opportunity to conduct additional ecological research.

Here, I describe impediments to an integrative ecological approach to protection of water resources, describe a recently developed approach for assessment of biological integrity of a water resource, and speculate on the future of biology in the protection of water resources.

Why Has It Taken So Long?

Although degradation in the ability of water resources to support biological activity was the first sign of a problem, society embraced approaches to improve water quality that were dominantly not biological. Reasons for limited use of integrative biological approaches to protect water resources are complex. I identified the following eight factors.

Dominance of reductionist viewpoints.—The dominant reductionist views of several disciplines involved in state and federal water management have been, and remain, a major impediment. First, few knowledgeable ecologists are willing to either help develop and implement criteria and standards or challenge the conceptual underpinnings of such approaches. Standards are the legally established rules consisting of two parts, designated uses and criteria. Designated uses are the purposes or benefits to be derived from a water body (e.g., aquatic life, irrigation water, and drinking water), and criteria are the conditions presumed to support or protect the designated uses. Dissolved oxygen (DO) may not fall below 5 mg/L, for example, if the designated use is a coldwater fishery. Second, engineers often fail to incorporate concern for biotic impairment. Third, politicians implement programs based on local interests and short time scales. Finally, planners attack problems as if ecosystem dysfunctions could be reversed without broad understanding of the whole system. Overall, a lack of interdisciplinary breadth, and especially a lack of grounding in biological theory, hampers development of sound water resource policy. A special need exists to account for actual behavior and variability of populations, communities, and ecosystems through adequate understanding of historical and current behavior (Schindler 1987).

Limited legal and regulatory programs.—Water law within the American legal system is a complex integration of federal and state constitutions (fundamental law), statutes and ordinances (acts at state or federal and local levels), administrative regulations (formulated and implemented by agencies), executive orders (orders by state and federal chief executives), and common law court decisions (Goldfarb 1988).

As a result, responsibility for regulation, protection, and development of water resources is vested in a patchwork of local, state, national, and international agencies. Although protection of water resources is the primary goal of water law, the law is not adequate to protect water resources. For example, current water law evolved before relationships between ground and surface water were understood. This issue highlights the fundamentally different approaches used in legal and scientific circumstances. Historically, courts could only impose effluent controls with proof, based on a preponderance of evidence, that an effluent was degrading a water resource. Science is more concerned with risk management that involves evaluation of probabilities of damage.

Another common but technically indefensible dichotomy that is firmly ingrained in water law is separate legal frameworks for water quality and quantity (McDonnell 1990). For example, water rights law dominates in the southwest where water supplies are limited (6% of United States supply but 31% of use), and water quality issues dominate in the east where water supplies generally exceed demand (37% of supply and 8% of use; Anonymous 1990). Toxicological (water quality) approaches dominate efforts to protect biological components of water resources in the east while methods to protect in-stream flows (water quantity) dominate in the west (Bain and Boltz 1989).

The evolution of federal water quality legislation (and the regulations that support it) illustrate additional constraints on the use of biology in protection of water resources. The first water-quality act was probably the Refuse Act of 1899 created to treat the growing problem of disease and oil pollution of navigable waters. Since the 1940s a series of laws and amendments were passed under the general rubric, Water Pollution Control Act (WPCA; also Water Quality Act or Clean Water Act). Amendments passed in 1948, 1956, 1961, 1965, 1966, and 1970 established several trends: (1) more money for construction and technology development, (2) expanded lists of pollutants for treatment, and (3) increases in enforcement efforts to control point sources of pollution. These efforts successfully controlled disease and, secondarily, reduced discharge of suspended solids (especially particulate organic carbon) that produced high biological oxygen demand (BOD) near wastewater treatment outflows. The growing array of chemicals from industrial plants, urban runoff, and agricultural sediments, nutrients, and pesticides were inadequately treated.

The burden of proof in documenting ecosystem degradation and establishing causal links to specific discharges fell on the government (Ward and Loftis 1989). However, establishing the cause of degradation was difficult and enforcement actions were rarely successful. First, no rigorously defined water-quality criteria were available. Second, few tools existed to accurately and effectively portray the results of regulatory programs.

By 1972, Congress recognized the need to revamp water resource programs. The WPCA Amendments of 1972 (PL 92-500), which came on the heels of the first "Earth Day" and heightened environmental awareness at the national level in the 1960s, contained far-reaching provisions, including stronger enforcement, increased federal involvement in water resource programs, and strict deadlines to end pollution by 1985. These were to be implemented primarily by achieving technology-based limits for point-source effluents (Levin and Kimball 1984, Ward and Loftis 1989). For the first time, water quality standards covered intrastate and interstate waters. Two visionary phrases in the act dealt with a "fishable and swimmable goal" and the charge to "restore and maintain the physical, chemical, and biological integrity of the Nation's waters." These phrases explicitly call attention to the need to permit ". . . all forms of natural aquatic life (the ultimate goal of water quality management)" (Meybeck and Helmer 1989).

Although the new emphasis on technology-based controls was heralded as revolutionary, implementation programs were usually limited to establishment of effluent limitations, an improved discharge permit system (National Pollutant Discharge Elimination System [NPDES]), performance standards for new plants and industries, and the call for sewer and waste treatment plants in all municipalities in the United States. Regulations continued to stress rules and standards for effluents rather than measuring the biological effects in the receiving water body, because regulators feared a return to the 1960s "burden of proof" days. (Recall the law vs. science dichotomy mentioned previously.) Thus, the focus on chemical parameters continued, or, when a biological perspective was used, the emphasis was on acute and later chronic effects of chemical pollutants on laboratory organisms. Many have expressed disappointment that the visionary law was used so inadequately (Anonymous 1981*a, b,* 1983).

Although the call for protection of biotic integrity was explicit in the 1972 amendments, point-source effluents remained the primary target of regulatory efforts for at least three reasons: (a) biological integrity was only one of several aspects explicitly protected, (b) politically and logistically point sources were easier to clean up, and (c) numerical pollution standards were thought to be both legally defensible and sufficient to protect water resources.

Success with the effluent-control approach on point sources of pollution made the effects of nonpoint-source (NPS) problems more obvious. However, programs to control NPS pollution were (and remain today) largely unsuccessful because of difficulties involved in applying point-source approaches to diffuse nonpoint-source problems (Karr 1990) and because of our unwillingness as a society to limit private land rights for the public good. Thompson (1989) provides a comprehensive, chemically oriented analysis of the continuing problems of NPS pollution.

The next major legislative action came in 1977 with passage of the Clean Water Act (CWA). As a result, emphasis shifted from conventional pollutants (e.g., fecal coliform and BOD) to the growing list of toxic chemicals released into the nation's waters. Although a wider perspective appeared in the 1972 and 1977 legislation (e.g., "fishable," "swimmable," and "biotic integrity"), the primary regulatory approach of both EPA and the states focused on technology-based controls to limit point-source pollutants discharged into bodies of water. All too frequently efforts to measure progress toward water-quality goals used administrative accounting (counts of permits issued or point sources regulated) rather than assessment of environmental results. An inability to associate water-quality based standards with biological integrity also limited the success of efforts to protect water resources, especially in view of the combined (synergistic, antagonistic, and additive: Risser 1988) effects of numerous pollutants and other human impacts (cumulative impacts: Preston and Bedford 1988). Although chemical and physical approaches are legally defensible (Mount 1985), they cannot measure complex attributes such as ecological health or "biotic integrity."

Because of widespread public support, the Congress passed the Water Quality Act of 1987, overriding a presidential veto. When combined with regulations developed in 1983 and 1985, this act changed the emphasis from technology-based controls with simple chemical water-quality standards to protection of specific water bodies (Plafkin 1989).

The 1980s have seen a major shift in philosophy with recognition of the inadequacy of earlier approaches as outlined in a report entitled "Surface Water Monitoring: A Framework for Change" (EPA 1987). At last, ambient monitoring of biological integrity is being recognized as a direct, comprehensive indicator of ecological conditions and, thus, the quality of a water resource. Although some argue that "the water quality criteria approach has served the science and the needs of society well" (Kimerle 1986), continuing degradation of water resources stimulated evolution in three areas of USEPA policy: (1) efforts to document environmental variability across landscapes and thus develop appropriate regional adjustments of standards (EPA 1983), (2) efforts to develop and implement approaches for the direct assessment of biotic integrity (Karr et al. 1986, Plafkin et al. 1989), and (3) recognition of the need to assess and mitigate cumulative impacts of human society (Karr and Dudley 1981, Karr 1987, Bedford and Preston 1988, Preston and Bedford 1988, EPA 1988*b*).

Definition of biological integrity.—USEPA convened a symposium (Ballentine and Guarraia 1975) on the integrity of water soon after passage of PL 92-500, but no clear definition of biotic integrity emerged. Many authors advocated use of a holistic perspective. Karr and Dudley (1981) argued that the "integrity" objective encompasses all factors affecting the ecosystem and developed a now widely quoted definition of *biological integrity* as the ability to support and maintain "a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region."

A more recent paper defined *ecological health* (an umbrella goal, the maintenance of which motivates virtually all environmental legislation) as follows: "a biological system . . . can be considered healthy 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).

While these definitions establish broad biological goals to replace the more narrowly defined chemical criteria, their use depends upon development of biological criteria based on ecological principles. Unfortunately, most theoretically oriented ecologists have been reluctant to participate in the application of their knowledge to applied problems (Schindler 1987), and a cynical attitude about the utility of ecologists dominates in some quarters (Wilk 1985, Kareiva 1990). Widespread use of single-species bioassays, complicated models, and impact-statement studies have been singularly unsuccessful at predicting the effects of anthropogenic stress on biological systems (Schindler

1987). Studies of population dynamics, food-web organization, and taxonomic structure of communities have been more successful (Schindler 1987).

Besides defining biological integrity, success at incorporating biotic integrity into water resource management depends on an appropriate, cost-effective procedure to measure biotic impairment. The index of biotic integrity (described in *How to measure biotic integrity,* below) provides one such approach.

Indexes to assess biological integrity.—Early efforts to develop biological indexes concentrated on detecting a narrow range of variation in biological integrity (Taub 1987, Ford 1989, Fausch et al. 1990), yielded indexes sensitive to only a few types of degradation (reduced DO, selected toxins, etc.), or provided only a binary (degraded/not degraded) evaluation. Some evaluated fecal contamination (Geldreich 1970) while others focused on effects of chemical stress on organisms at population or community levels (Ford 1989). Although valuable for measurement of selected anthropogenic effects, they were less useful for screening all types of degradation, including complex cumulative impacts.

The saprobin system developed in Europe (Sladacek 1973) focuses on biological oxygen demand (BOD). Use of selected intolerant species may reflect changes such as high levels of oxygen-demanding wastes or sedimentation resulting from soil erosion. Even more complex community-based indexes like the Hilsenhoff index for benthic macroinvertebrates (Hilsenhoff 1982, 1987) are primarily sensitive to domestic effluents. Coliform counts can identify inputs of untreated sewage, although contamination from wildlife and livestock may also affect bacterial counts (Dudley and Karr 1979). Finally, many existing biological indexes may only apply to a narrow geographical area (e.g., lake trout stocks in Laurentian Great Lakes). Such approaches are appropriate when specific narrow impacts are known to be present, but protection of water resources from a broad range of human impacts requires a more comprehensive approach.

One successful tactic of the past decade is to combine two or more biological metrics into a single index. Comparisons to assessment of human health (using metrics such as blood pressure, urine analysis, white blood cell count, and temperature), or several economic indicators are appropriate. Each metric provides information about the sampling site and even the region (Steedman 1988). When combined, these metrics characterize the biotic integrity in much the same way that a battery of medical tests are indicators of individual health. However, it is important to note that good health, human, economic, or ecosystem, is not a simple function of those metrics (Karr et al. 1986).

The ideal index would be sensitive to all stresses placed on biological systems by human society while also having limited sensitivity to natural variation in physical and biological environments. An array of indicators would be combined into one or more simple indexes and could be used to *detect degradation* and identify its cause and to determine if *improvement* results from management actions. In the best of all worlds these could be used in a regulatory context to *prevent* degradation and thus preserve high quality water resource systems. Indicators for general usage must be applicable in a wide range of water resource systems and be successful "in measuring attainment of the biological integrity goals of the Clean Water Act" (Ohio EPA 1988).

Region-based quantitative definitions of ecological health.—The idea of chemical-specific toxicological criteria and water-quality standards involves defining contaminant levels above which negative effects on water resources can be expected (Levin et al. 1989). But, standardized values for chemical criteria fail to recognize natural geographic variation in water chemistry. Natural heavy metal concentrations in western rivers are often well above EPA standards levels and dissolved oxygen levels often fall below standards established in water-quality regulations. Many recent efforts to develop biological criteria call for definition of regional (Fausch et al. 1984, Hughes et al. 1986, Gallant et al. 1989) and stream size expectations (Karr 1981, Ohio EPA 1988, Plafkin et al. 1989).

Many states have adopted Omernik's (1987) ecoregion concept as a framework for refining biological expectations (Gallant et al. 1989). Ecoregions are geographic areas within which stream communities are relatively homogeneous. However, their boundaries should not be flatly accepted because other boundaries, including river basins and physiographic provinces, may be important. Recognition of the existence of natural geographic variation in the ecological features of undisturbed aquatic systems is essential, a reality that has been overlooked for several decades in efforts to set rigid nationwide chemical and physical standards.

Standardization of field methods.—Standardization of methods (quality control/quality assurance [QC/QA]) is a fundamental prerequisite for any monitoring program. Without these, the utility of environmental monitoring data can and will be challenged (Plafkin et al. 1989). The first step is definition of standards for sampling ecosystems or lower levels of biological organization. Karr et al. (1986), Ohio EPA (1988), and Plafkin et al. (1989) provide examples of efforts to define acceptable methods for sampling biological communities in the field with minimal sampling effort. Finally, they also attempt to formalize analytic procedures.

Linking field measurements to enforceable management options.—As noted above, the 1965 Federal Water Quality Control Act (PL 89-234) provided a national framework for water-quality management. That framework considered water-quality management to be a task requiring policies and goals (standards) against which in-stream water-quality conditions would be evaluated (Ward and Loftis 1989). Problems developed when it was recognized that knowledge of the connections between in-stream conditions and the action of dischargers was inadequate. Operating under the assumption that point-source discharges were causing problems, the 1972 Act shifted from in-stream conditions to effluent conditions. Enforcement actions were based on discharge permits whose limits were exceeded. By the early 1980s, many recognized that ignoring in-stream conditions (Ward and Loftis 1989), especially their biological context (Karr and Dudley 1981, Karr et al. 1983), resulted in continued degradation of water resources. Although credit is appropriate for improved water quality in some areas due to discharge limitations through permits, money was sometimes used to provide wastewater treatment that did not improve in-stream conditions (Karr et al. 1985*a,* Meybeck and Helmer 1989). Simply put, the shift to effluent monitoring was a high-cost program that failed to protect the quality of many key water resources due to continuing impacts from unregulated sources (e.g., nonpoint sources).

Mount (1985) reflects one extreme view on this issue when he contends that the best defense for dependence on testing that is based on toxicity (effluent) is that it is decisive; that is, toxicity-based criteria provide clear standards for establishing water-quality impacts. However, decisiveness does not overcome their many deficiencies when one has ecological goals (endpoints) in mind. Most chemical standards have no meaning outside the legal/regulatory context, and they only protect environmental values explicitly included in the standard-setting process. In addition, toxicity-based criteria are not adequate as early warning devices for detection of degradation. Finally, poorly understood but important biological mechanisms and effects are not incorporated into the standard-setting process (Suter 1990). Single-species toxicity testing may in selected situations be well informed and decisive, but in many circumstances its decisiveness may be misleading and even dangerous for the resource.

In short, for nearly two decades a narrow perspective on standards was imposed that was presumed to be effective because it was decisive, legally defensible, and enforceable in a regulatory context. While I would not argue against this approach to control point-source discharges, I would also say that sole dependence on the approach cripples society's ability to detect, much less reverse, degradation due to nonpoint sources of pollution, habitat destruction, modification of flow, and changes in the energy base of the stream biota.

Need for cost-effective approaches to biological monitoring.—Water resource managers have long argued that the costs of ambient biological monitoring are too high (Loftis et al. 1983, EPA 1985). However, a recent compilation by Ohio EPA (Table 1) shows that biological monitoring may not be prohibitively expensive compared with more conventional approaches. Although costs may vary among agencies and circumstances, Table 1 demonstrates that biotic monitoring should not be discounted on cost criteria. Further, accounting must go beyond the cost of data collection and analysis to include costs of building and operating potentially unnecessary or poorly designed treatment plants and costs of bad management decisions. Karr et al. (1985*a*) provide an example of implementation of tertiary denitrification that probably yielded little benefit to the water resource. The mandate in the Clean Water Act of 1987 to reduce emphasis on construction suggests more widespread recognition of this issue.

While progress toward reducing the effects of these eight constraints varies, all have now been widely recognized and considerable energy is being expended to overcome them.

How To Measure Biotic Integrity

Human activities may alter the physical, chemical, or biological processes associated with water resources and thus modify the resident biological community. Biological criteria are valuable for assessing these alterations because they "directly measure the condition of the resource at risk, detect problems that other methods may miss or underestimate, and provide a systematic process for measuring progress resulting from the implementation of water quality programs" (EPA 1990). They do not replace chemical and toxicological methods, but they do increase the probability that an assessment program will detect degradation due to anthropogenic influences.

During the past decade five primary sets of variables have been identified that, when affected by human activities, result in ecosystem degradation (Fig. 1). Many individual studies demonstrate correlations (if not cause and effect) between degradation and some biological indicator (e.g., species richness, changing abundance of an indicator species, production/respiration ratio; see Taub 1987, Ford 1989 for recent reviews). However, few attempts have been made to integrate several of those indicators into a single index. I developed such an index, the index of biotic integrity (IBI) using a set of attributes that measure organization and structure of fish communities.

The index of biotic integrity (IBI)

The index of biotic integrity was conceived to provide a broadly based and ecologically sound tool to evaluate biological conditions in streams (Karr 1981). IBI incorporates many attributes of fish communities to evaluate human effects on a stream and its watershed. Those attributes cover the range of ecological levels from the individual through population, community, and ecosystem. Although initially developed for use with fish communities, the ecological foundation of IBI can be used to develop analogous indexes that apply to other taxa, or even to combine taxa into a more comprehensive assessment of biotic integrity.

Calculation of a fish IBI for a stream reach requires

TABLE 1. Comparative cost analysis for sample collection, processing, and analysis for evaluation of the quality of a water resource. Data from Ohio EPA, provided by C. O. Yoder (1989*b*).

	Per sample*	Per evaluation†
Chemical/physical water quality		
4 samples/site	$1436	$ 8616
6 samples/site	$2154	$12924
Bioassay		
Screening (acute—48-h exposure)	$1191	$ 3573
Definitive (LC50‡ and EC50§—48 and 96 h)	$1848	$ 5544
7-d (acute and chronic effects—7-d exposure single sample)	$3052	$ 9156
7-d (as above but with composite sample collected daily)	$6106	$18318
Macroinvertebrate community‖	$ 824	$ 4120
Fish community¶	$ 740	$ 3700
Fish and macroinvertebrates (combined)	$1564	$ 7820

* The cost to sample one location or one effluent; standard evaluation protocols specify multiple samples per location.
† The cost to evaluate the impact of an entity; this example assumes sampling five stream sites and one effluent discharge.
‡ Dose of toxicant that is lethal (fatal) to 50% of the organisms in the test conditions at a specified time.
§ Concentration at which a specified effect is observed in 50% of organisms tested; e.g., hemorrhaging, dilation of pupils, stop swimming.
‖ Using Invertebrate Community Index (ICI) (see text and Table 4).
¶ Using Index of Biotic Integrity (IBI) (see text and Table 2).

a single sample that represents fish species composition and relative abundances. (See Karr et al. 1986, Ohio EPA 1988, and Plafkin et al. 1989 for detailed sampling and data-handling protocols.) Application of IBI requires careful standardization of procedures as noted above (*Why has it taken so long? Standardization of field methods*).

An additional innovation of IBI is that the value for each metric is based on comparison to a regional reference site with little or no influence from human society (Fausch et al. 1984). Expected values are based on that reference site(s) with observed values compared to expected values. For each metric, an index score of 5 is assigned if the study site deviates only slightly from the reference site, 3 if it deviates moderately, and 1 if it deviates strongly from the undisturbed condition. These assessments require experienced biologists to set standards based on knowledge of the regional biota (e.g., higher species richness expected in Tennessee than in Nebraska) and the size of stream sampled (e.g., higher species richness in large than small streams). Thus, assessment of biotic integrity explicitly incorporates biogeographic variation into evaluation of biological systems.

Twelve attributes (Table 2) of a fish community are rated. The sum of those ratings (5, 3, or 1) provides an IBI value, an integrative and quantitative assessment of local biological integrity (Table 3). IBI uses three groups of metrics: species richness and composition, trophic composition, and fish abundance and condition.

Species richness and composition metrics.—The first group of six metrics evaluates the extent to which the sample area supports reduced species richness and altered species composition. Because richness varies as a function of region, stream size, elevation, and stream gradient, all sites must be evaluated against the expected richness from a similar undisturbed site (or, regionally, least disturbed site). For the Midwest, IBI includes five species richness metrics (Table 2); these include three taxon-specific (Catostomidae [suckers], Etheostomatinae of Percidae [darters], and Centrarchidae [sunfish]), one assessing the presence of species intolerant of human activities, and one assessing total species richness (excluding exotics). The three taxa are selected to represent groups that consume benthic invertebrates (suckers and darters) or drifting and terrestrial invertebrates (sunfish) as their primary food. Assessments using these taxa confirm by their presence (or not) whether requisite spawning habitat and food are available. Other taxa with similar ecological attributes should be substituted in regions where these families are not abundant (see *How to measure biotic integrity: Adaptability of IBI,* below and Table 2 for modification of IBI metrics in regions outside the Midwest). The intolerant-species metric uses the common observation that within a region a few species in most regions are especially sensitive (intolerant) to human disturbance. Intolerance to siltation is common, but other types of intolerance may also be present. The intolerant class should be restricted to the 5–10% of species that are most susceptible to degradation, and should not be taken as equivalent to rare and endangered (Karr et al. 1986). The fifth species richness/composition metric is the total species richness of the community. The sixth metric in this group relates to species composition. Green sunfish (*Lepomis cyanellus*) increase in abundance in degraded streams of the Midwest, reflecting the extent to which disturbance permits a species to dominate the community. Other species used in this metric in other regions included common carp (*Cyprinus carpio*), white sucker (*Catastomus commersonii*), and gardon (*Rutilus rutilus*) (Miller et al. 1988, Oberdorff and Hughes, *in press*).

ecological impact of human-induced alterations

1. food (energy) source
   - type, amount, and particle size of organic material entering a stream from the riparian zone versus primary production in the stream
   - seasonal pattern of available energy

- decreased coarse particulate organic matter
- increased fine particulate organic matter
- increased algal production

2. water quality
   - temperature
   - turbidity
   - dissolved oxygen
   - nutrients (primarily nitrogen and phosphorus)
   - organic and inorganic chemicals, natural and synthetic
   - heavy metals and toxic substances
   - pH

- expanded temperature extremes
- increased turbidity
- altered diurnal cycle of dissolved oxygen
- increased nutrients (especially soluble nitrogen and phosphorus)
- increased suspended solids
- increased toxics
- altered salinity

3. habitat structure
   - substrate type
   - water depth and current velocity
   - spawning, nursery, and hiding places
   - diversity (pools, riffles, woody debris)
   - basin size and shape

- decreased stability of substrate and banks due to erosion and sedimentation
- more uniform water depth
- reduced habitat heterogeneity
- decreased channel sinuosity
- reduced habitat areas due to shortened channel
- decreased instream cover and riparian vegetation

4. flow regime
   - water volume
   - temporal distribution of floods and low flows

- altered flow extremes (both magnitude and frequency of high and low flows)
- increased maximum flow velocity
- decreased minimum flow velocity
- reduced diversity of microhabitat velocities
- fewer protected sites

5. biotic interactions
   - competition
   - predation
   - disease
   - parasitism

- increased frequency of diseased fish
- altered primary and secondary production
- altered trophic structure
- altered decomposition rates and timing
- disruption of seasonal rhythms
- shifts in species composition and relative abundances
- shifts in invertebrate functional groups (increased scrapers and decreased shredders)
- shifts in trophic guilds (increased omnivores and decreased piscivores)
- increased frequency of fish hybridization
- increased frequency of exotic species

TABLE 2. Metrics used to assess biological integrity of fish communities based on Index of Biotic Integrity (IBI) (after Karr 1981, Karr et al. 1986). Ratings of 5, 3, and 1 are assigned to each metric according to whether its value approximates, deviates somewhat from, or deviates strongly from the value expected at a comparable site that is relatively undisturbed.

Metrics	Rating of metric* 5	3	1
Species richness and composition			
1. Total number of fish species* (native fish species)†			
2. Number and identity of darter species (benthic species)	Expectations for metrics 1–5 vary with stream size and region.		
3. Number and identity of sunfish species (water-column species)			
4. Number and identity of sucker species (long-lived species)			
5. Number and identity of intolerant species			
6. Percentage of individuals as green sunfish (tolerant species)	<5	5–20	>20
Trophic composition			
7. Percentage of individuals as omnivores	<20	20–45	>45
8. Percentage of individuals as insectivorous cyprinids (insectivores)	>45	45–20	<20
9. Percentage of individuals as piscivores (top carnivores)	>5	5–1	<1
Fish abundance and condition			
10. Number of individuals in sample	Expectations for metric 10 vary with stream size and other factors.		
11. Percentage of individuals as hybrids (exotics, or simple lithophils)	0	>0–1	>1
12. Percentage of individuals with disease, tumors, fin damage, and skeletal anomalies	0–2	>2–5	>5

* Original IBI metrics for midwest United States.
† Generalized IBI metrics (see Miller et al. 1988).

Because expectations for the species richness metrics vary with stream size, we formalized a method to establish those expectations. A "maximum species richness line" is determined with a plot of number of species on stream size (order, watershed area, or flow). A plot of data from a watershed (Fig. 2) or ecoregion yields a distinct right triangle, the hypotenuse of which approximates the upper limit of species richness (Fausch et al. 1984, Karr et al. 1986, Ohio EPA 1988).

Trophic composition metrics.—This group of three metrics evaluates the trophic composition of the fish community to assess the energy base and trophic dynamics of the resident biota. All organisms require reliable sources of energy and major efforts have been

TABLE 3. Total Index of Biological Integrity (IBI) scores, integrity classes, and the attributes of those classes (modified from Karr 1981).

Total IBI score (sum of the 12 metric ratings)*	Integrity class of site	Attributes
58–60	Excellent	Comparable to the best situations without human disturbance; all regionally expected species for the habitat and stream size, including the most intolerant forms, are present with a full array of age (size) classes; balanced trophic structure.
48–52	Good	Species richness somewhat below expectation, especially due to the loss of the most intolerant forms; some species are present with less than optimal abundances or size distributions; trophic structure shows some signs of stress.
40–44	Fair	Signs of additional deterioration include loss of intolerant forms, fewer species, highly skewed trophic structure (e.g., increasing frequency of omnivores and green sunfish or other tolerant species); older age classes of top predators may be rare.
28–34	Poor	Dominated by omnivores, tolerant forms, and habitat generalists; few top carnivores; growth rates and condition factors commonly depressed; hybrids and diseased fish often present.
12–22	Very poor	Few fish present, mostly introduced or tolerant forms; hybrids common; disease, parasites, fin damage, and other anomalies regular.
···†	No fish	Repeated sampling finds no fish.

* Sites with values between classes assigned to appropriate integrity class following careful consideration of individual criteria/metrics by informed biologists.
† No score can be calculated where no fish were found.

⟵

FIG. 1. Five major classes of environmental factors that affect aquatic biota (Karr et al. 1983). Arrows indicate the kinds of effects that can be expected from human activities (modified from Karr et al. 1986).

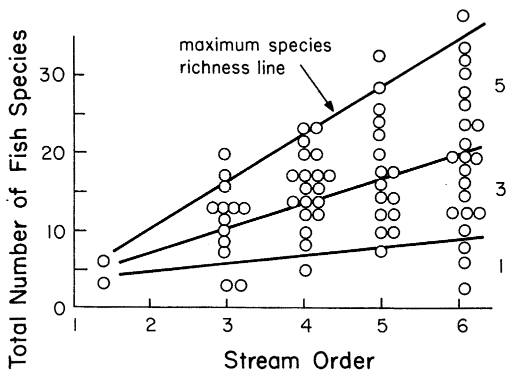

FIG. 2. Total number of fish species vs. stream order for 72 "least disturbed" sites along the Embarras River, Illinois. The area below the line of maximum species richness is trisected and used to rate IBI metric 1, total number of species. Ratings (5, 3, 1) indicate whether species richness at a given site on a stream of given order approximates, is somewhat less than, or is far less than the species richness expected from an "excellent" fish community in the region (modified from Fausch et al. 1984 by Karr et al. 1986). Stream order is a way of measuring stream size.

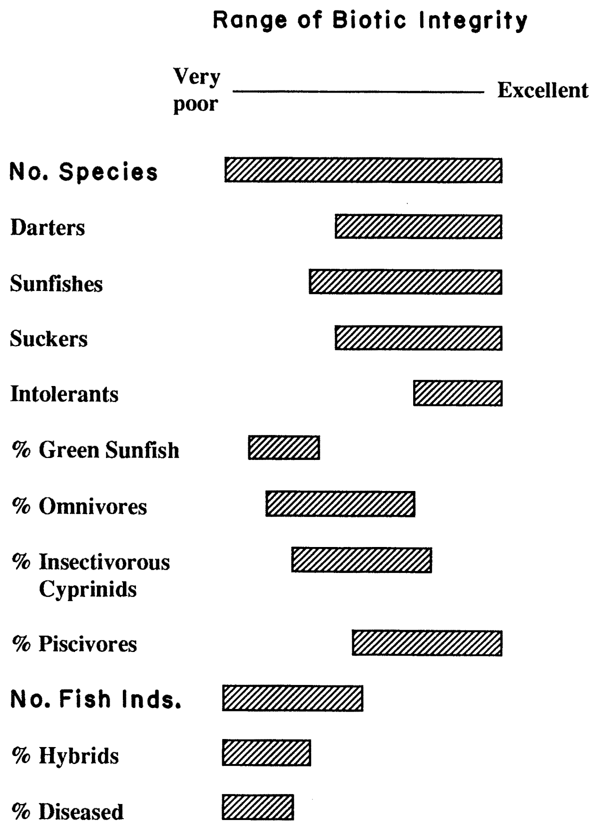

FIG. 3. Range of primary sensitivity (shaded area) for each metric (see Table 2) in the Index of Biotic Integrity (from Angermeier and Karr 1986).

made to measure the many dimensions of productivity. Direct measurement of productivity is costly and time consuming, especially if attempted at several trophic levels, and the interpretation of results may be ambiguous, or even misleading (Schindler 1987). Thus, several metrics that measure divergence from expectation were developed as a way to assess energy flow through the community (trophic structure). The proportion of omnivores increases and the proportion of insectivorous cyprinids and top carnivores decreases in degraded systems. In all three cases the proportion of individuals in the sample is used to rate stream reaches for each of these metrics. Scoring criteria for these functional metrics have been remarkably consistent throughout North America, suggesting a general pattern for stream fishes.

Fish abundance and condition metrics.—The last group of three metrics evaluates population density and fish condition. The total number of individuals in the sample is an important parameter for evaluation because disturbed areas often have reduced fish abundances. This metric must be based on catch per unit sampling effort, such as number per area sampled or per unit sampling time. The final two metrics evaluate the frequency of hybridization, apparently a function of habitat destruction and mixing of gametes in best available spawning areas (Hubbs 1961, Greenfield et al. 1973), and proportion of individuals with disease, tumors, fin damage, and major skeletal anomalies. The latter increases in degraded areas, especially in areas with major toxic contamination (Brown et al. 1973, Baumann et al. 1982). The anomaly metric uses only rapid survey procedures in search of anomalies that can be discovered by external examination without sacrificing fish.

Each metric reflects the quality of components of the fish community that respond to variation in different aspects of the aquatic system. The relative sensitivity of the metrics varies from region to region (Angermeier and Karr 1986, Karr et al. 1986, Steedman 1988), in part because metrics are differentially sensitive to various perturbations (siltation, flow alteration, toxins). In addition, natural variation in conditions among watersheds reduces the probability that indexes based on one or a few metrics will provide reliable assessments over a wide geographic area. The total number of fishes decreased, and trophic structure of the community shifted, in areas exposed to municipal effluent in one study (Karr et al. 1985*a*). Sedimentation and other habitat alteration reduced the number of fishes feeding on benthic invertebrates (e.g., darters). In the most degraded sites, many IBI metrics reflect the serious degradation and reinforce the strength of the inference. Finally, IBI metrics have differential sensitivity along the gradient from undisturbed to degraded (Fig. 3).

IBI scores can be used to (1) evaluate current conditions at a site, (2) determine trends over time at a site with repeated sampling, (3) compare sites from which data are collected more or less simultaneously, and (4) to some extent, identify the cause of local deg-

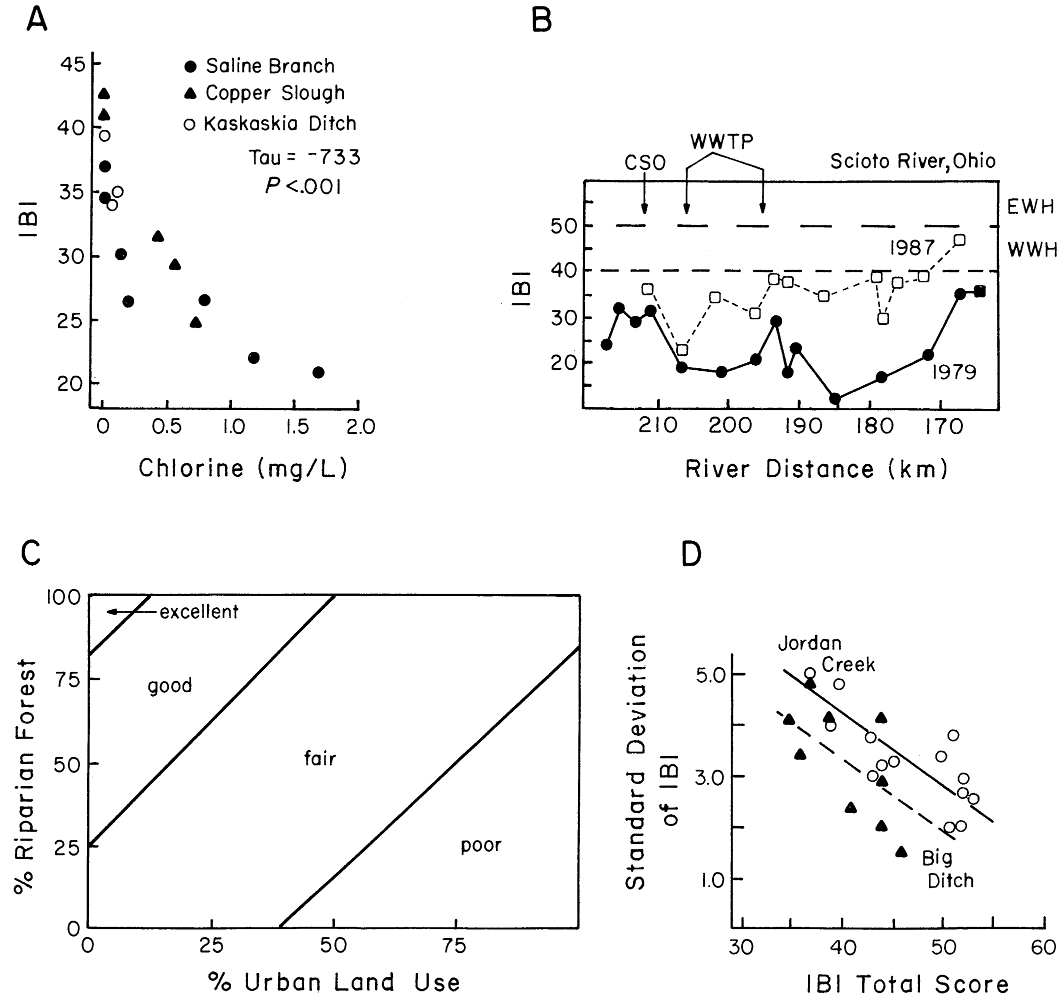

FIG. 4. (A) Index of Biotic Integrity (IBI) as a function of total residual chlorine content in three streams in east-central Illinois with wastewater inflow from standard secondary treatment with chlorination (from Karr et al. 1985*a*). (B) Longitudinal trend in IBI for the Scioto River, Ohio, in and downstream from Columbus, Ohio, 1979 and 1987. CSO = combined sewer overflow; WWTP = wastewater treatment plant inflow; WWH = warmwater habitat; EWH = excellent warmwater habitat. Stream flow is from left to right (from Yoder 1989*a*, originally published with River Mile as horizontal dimension). (C) Contour plot of qualitative IBI ratings as a function of urbanization and % retention of riparian forest (from Steedman 1988). (D) Standard deviation of IBI values as a function of IBI values in Jordan Creek (O) and Big Ditch (▲), Illinois (from Karr et al. 1987).

radation (Karr et al. 1986). Over 30 states and provinces and several federal agencies have used IBI (or modifications of IBI; see *How to measure biotic integrity: Adaptability of IBI,* below). At least four states and the Tennessee Valley Authority have incorporated IBI into their standards and monitoring programs (Miller et al. 1988), and many others are expanding use of IBI and conceptually similar approaches into their routine monitoring programs.

Many advantages of IBI have been cited (Karr 1981, Karr et al. 1986, Miller et al. 1988, Plafkin et al. 1989, Fausch et al. 1990) including: (1) it is quantitative; (2) it gauges a stream against an expectation based on minimal disturbance in the region; (3) it reflects distinct attributes of biological systems, including temporal and spatial dynamics; (4) there is no loss of information from constituent metrics when the total index is determined because each metric contributes to the total evaluation; and (5) professional judgment is incorporated in a systematic and ecologically sound manner.

IBI does not serve all needs of detailed biological monitoring (Karr et al. 1986, Fausch et al. 1990) and certainly cannot be advocated as a replacement for physical and chemical monitoring or toxicity testing. However, ecologically sophisticated biological monitoring provides direct information about conditions at a sample site compared with a site with little or no human influences or to the expectation under a designated biological use classification (e.g., high-quality warmwater fishery).

A sampling of successful uses of IBI

Successful use of IBI in a variety of contexts (effects of mine drainage, sewage effluent, habitat alteration, etc.) and in a diversity of geographic areas prove the utility of the IBI concept (Karr et al. 1986, Miller et al. 1988, Steedman 1988, Fausch et al. 1990, Oberdorff and Hughes, *in press*).

Studies have included evaluation of chemical factors, as illustrated by the inverse relationship between IBI and residual chlorine concentration in three watersheds in east-central Illinois (Karr et al. 1985*a*; see Fig. 4A). In another study, the effects of general watershed condition and wastewater treatment outflows were evaluated in the Scioto River, Ohio (Fig. 4B). IBI values were low throughout the river in 1979, especially declining below the inflow of wastewater from large sewage treatment plants. After 8 yr of efforts to control the plants' effluents, downstream biotic integrity improved appreciably by 1987, although regional nonpoint source and habitat degradation keeps IBI below optimal levels.

In southern Ontario, Steedman (1988) found that IBI was strongly associated with independently derived measures of watershed condition. He found that a threshold of degradation for Toronto area streams was reached under conditions ranging from 75% removal of riparian vegetation in areas with no urbanization to no (0%) removal of riparian vegetation at 55% urbanization. IBI ratings could be expressed as a function of proportion of basin in urban land use (URB) and proportion of order 1–3 channels with intact riparian forest (RIP) using the following equation: IBI = 30 − 19URB + 14RIP. IBI has also been used to show that degraded sites (low IBI) are more variable than less degraded sites (Fig. 4D). Further, between-stream differences in IBI variability were due to mobility of fish and nearby presence of habitat refuges (sources of colonists) in Jordan Creek (Karr et al. 1987), suggesting that cumulative regional impacts may also be important in determining local biotic integrity.

In short, IBI satisfies three conditions named by Schindler (1987) for useful monitoring programs: inexpensive, simple, and highly sensitive to changes in ecosystems.

Adaptability of IBI

No single index or set of metrics can be expected to detect all water resource problems. However, IBI is very successful as a broadly based approach to assess the quality of a water resource. IBI can be modified to incorporate other aspects of the fish community. Four such aspects are (1) species composition within major taxa, (2) population structure (e.g., size frequency distribution), (3) growth rates, and (4) relative health of individuals within populations of selected species. Karr (1981) mentioned all, but none were incorporated into the index because the necessary information was not easily obtained, especially from historical databases, the primary data available for initial development and testing of IBI. For example, a site with johnny darter (*Etheostoma nigrum*) and orange-throated darter (*E. spectabile*) is likely to be degraded compared with another site with banded (*E. zonale*) and slenderhead darters (*Percina phoxocephala*). One approach to scoring these situations (Hughes and Gammon 1987) is to give a plus (+) to sites with a preponderance of species that suggest high quality. When IBI scores are totaled, two or three species richness metrics with a plus appended would be scored by adding one unit to IBI. Such differences could be incorporated into future IBI applications when relative rankings of several species as indicators of degradation are known. As another example, one could incorporate information on health of individual fish through metrics such as condition factor (K) where L is total length (in millimetres) and M is mass (in grams) $K = (M/L^3) \times 10^5$. Some effort must be made to define a length class for determination of K (Lagler 1956). Alternatively, the age structure of the population might be used by examination of the masses and/or lengths of individuals of selected species or through reading of growth annuli on scales. Use of either of these may improve the resolution of IBI evaluations, especially when applied to specific groups of sportfish species, or when single species dominate assemblages (e.g., trout).

Adaptation of IBI to geographic regions outside the midwestern United States requires modification, deletion, or replacement of selected IBI metrics (Table 2). Miller et al. (1988) provide the most up-to-date review of changes needed to reflect regional differences in biological communities and fish distribution. The kind of flexibility illustrated by IBI results from an integrative framework with a strong ecological foundation. Areas as diverse as the streams of Colorado, New England, northern California, Oregon, southeast Canada, France (Seine River), and Appalachia and estuaries in Louisiana, Chesapeake Bay, and New England have been evaluated using the conceptual approach of IBI.

In California, the principal attributes that must be accommodated are reduced species richness, high endemism among watersheds, absence of midwestern taxa such as darters and sunfish, and high salmonid abundances. Modifications in IBI needed for use in estuarine areas of Louisiana included variation in salinity regimes and estuary size. New IBI metrics reflect aspects of fish residency, presence of nearshore marine fishes and large freshwater fishes, and a measure of seasonal variation in community structure. Other special considerations include the importance of stream gradient in Appalachia and geographic variation in tolerances of some species. For example, the creek chub (*Semotilus atromaculatus*) varies appreciably in its tolerance of stream degradation and food habits from Colorado to Illinois to the New River drainage of Virginia.

Modifications adopted by Ohio EPA include the replacement of several of the original IBI metrics with alternates for analysis of conditions in large rivers. They propose replacement of darters with round-bodied suckers in large rivers sampled with boat-mounted electrofishing gear, an excellent suggestion in a situation where darters are likely to be undersampled. They have, in addition, field tested and evaluated other aspects of IBI.

Recent use of IBI by the Tennessee Valley Authority has shown its value in assessing declining biotic integrity (Saylor and Scott 1987, Wade and Stalcup 1987). In one case, release of cold water limited fish communities in a reservoir tailwater stream and in another case low-flow periods left much of the channel dry. In both cases, IBI detected this degradation when general reviews of habitat conditions and water quality did not alert biologists to problems of water resource degradation.

Perhaps the most innovative use of IBI is the work of Steedman (1988) in southern Ontario. He sampled fishes at 209 stream sites in 10 watersheds near Toronto. All are in tributaries on the northwestern shore of Lake Ontario. His 10-metric IBI included several adaptations to accommodate both cold- and warm-water reaches, such as combining taxonomic groups in selected metrics: sculpins plus darters, salmonids plus centrarchids, and suckers plus catfishes. He found that within-year variation at sample sites on large rivers generally found IBI values within ±4 points (4 out of 50 = 8%), and most were within ±1 point. For between-year comparisons, >80% of sample sites had IBI values that varied among years by <10%.

Steedman's analysis of threshold effects in degradation of riparian habitat (Fig. 4C) raises a persistent but yet unanswered question about the threshold of riparian vegetation destruction within a watershed that results in major degradation of biotic integrity (Karr and Schlosser 1978). His approach using IBI may provide an indirect approach to answering that question. It deserves considerable study in many geographic areas.

Miller et al. (1988) encouraged modification of IBI to make it suitable for a wide range of geographical areas. Three cautions come to mind. First, avoid modifications unless they yield significant improvement in the utility of the index (Angermeier and Karr 1986). For example, Leonard and Orth (1986) modified many IBI metrics for study of streams in West Virginia. In a reanalysis of their data, the ability of the IBI approach to detect degradation was not improved by their modification of metrics (Fig. 5). Second, modifications of IBI should be undertaken only by experienced fish biologists familiar with the conceptual framework of IBI, local fish faunas, and watershed conditions.

Finally, efforts should be made to develop IBI-type indexes for use in other environments such as wetlands, lakes, and terrestrial ecosystems. Successful uses of IBI include large rivers in Oregon (Hughes and Gammon 1987), Ohio (Ohio EPA 1988), and France (Oberdorff and Hughes, *in press*). Similar advances have been made in evaluations of lakes in Minnesota (Heiskary et al. 1987) and wetlands (Brooks and Hughes 1988). I attempted with limited success to apply IBI concepts using data from a study of birds of forest islands (Blake 1983, Blake and Karr 1984, 1987). For example, the number of omnivorous birds increases as forest island size declines (Karr 1987). Apparently, the altered food base in remnant forest islands parallels changes in species richness and abundance of omnivores in disturbed headwater streams.

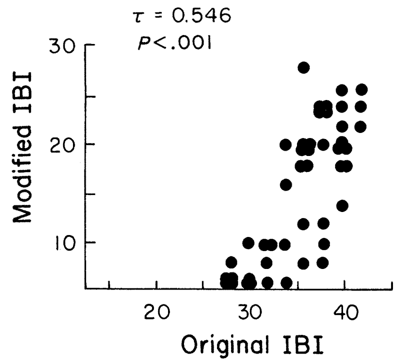

FIG. 5. Correlation between original IBI (Karr 1981) and modified IBI (Leonard and Orth 1986) at sites in West Virginia sampled by Leonard and Orth.

Assessment of biotic integrity with other taxa

The framework of the fish IBI has been adopted by invertebrate biologists in efforts to develop robust methods to measure degradation using benthic invertebrate (Ohio EPA 1988, Plafkin 1989) and protozoan communities (J. R. Pratt, *unpublished manuscript*). The most extensively tested, integrative effort is the Invertebrate Community Index (ICI) developed by Ohio EPA (1988). ICI is a 10-metric index (Table 4A) that emphasizes structural attributes of invertebrate communities. Ohio EPA used this approach because of "accepted historical use, simple derivation, and ease of interpretation." Metric 10 is scored based on a qualitative field sample, while metrics 1–9 are based on artificial-substrate sampling.

As part of its effort to establish biological metrics, USEPA has also supported development of a hierarchy of methods for biological monitoring. Rapid Bioassessment Protocol III (Plafkin et al. 1989) is similar to the ICI but has only eight metrics (Table 4B). Both structural and functional metrics are included. RBP III combines sampling invertebrates from a riffle/run habitat and from a grab sample of coarse particulate organic matter (e.g., leaf packs) at each sampling site.

Both efforts are promising developments designed to strengthen the role of biology in assessment of the

TABLE 4. Metrics used to assess biological integrity of benthic invertebrate communities.

A. Invertebrate Community Index (ICI) (after Ohio EPA 1988*). Ratings of 6, 4, 2, and 0 are assigned to each metric according to whether its value is comparable to exceptional, good, slightly deviates from a good, or strongly deviates from a good community.
   1. Total number of taxa
   2. Total number of mayfly taxa
   3. Total number of caddisfly taxa
   4. Total number of dipteran taxa
   5. Percent mayfly composition
   6. Percent caddisfly composition
   7. Percent Tribe Tanytarsini midge composition
   8. Percent other dipteran and noninsect composition
   9. Percent tolerant organisms
   10. Total number of qualitative EPT† taxa

B. Rapid Bioassessment Protocol III (after Plafkin et al. 1989‡). Ratings of 6, 3, and 0 are given based on values of each of the metrics with 6 being high quality and 0 being heavily degraded site.
   1. Taxon richness
   2. Family biotic index
   3. Ratio of scraper/filtering collector
   4. Ratio of EPT† and chironomid abundances
   5. Percent contribution of dominant family
   6. EPT† index
   7. Community loss index
   8. Ratio of shredders/total

* Metrics 1–9 based on artificial substrate sampler; metric 10 based on qualitative stream sampling.

† EPT—taxa in the Emphemeroptera, Plecoptera, and Trichoptera.

‡ Metrics 1–7 based on qualitative riffle/run sample; metric 8 based on leaf-pack (CPOM) sample.

quality of water resources. The ICI is a robust addition to the arsenal of assessment tools. The RBP III has been less extensively tested, and many validation studies remain to be done. For example, the use of a random sample of 100 invertebrates seems inadequate to me. Invertebrate communities often include 50 or more species, making it unlikely that a 100-individual subsample will be representative of taxa and ecological groups in such communities. Efforts are now being made to replace some of the "ratio" metrics of RBP III with other metrics (M. T. Barbour, *personal communication*). Other aspects of RBP III that require more intensive testing include its use over a wider range of geographic areas, the adequacy of genus vs. species level identifications, and the constraint that sampling concentrate on only a single local habitat (riffle/run or use of artificial substrates). These and other approaches that use invertebrates (Berkman et al. 1986, Lenat 1988, J. R. Pratt, *unpublished manuscript*) in assessment of biotic integrity are not as widely validated as is IBI, but they show considerable promise as additional water resource tools.

Fish and invertebrate IBI approaches are a major improvement over past programs in river and stream environments. Ecologists should strive to develop suites of metrics that integrate taxa (fish, invertebrates, protozoa, and diatoms) and levels of ecological organization (population, community, ecosystem, and landscape).

The Future of Biological Monitoring

Growing dissatisfaction with the adequacy of current water resource programs and recognition of the potential contribution of improved biological monitoring has stimulated nationwide interest in use of biological monitoring to attain the goals of the Clean Water Act. The need for more rigorous use of ecological principles provides an unprecedented opportunity for ecologists (biologists) to influence and even guide decisions about water resources. How might ecologists contribute?

Biology has always been a player (Ford 1989), but not a principal player, in the water resource arena. The saprobic system (Kolkwitz and Marsson 1908, 1909) was perhaps the first major effort to use biology in the assessment of water resource degradation. A classic paper (Patrick 1949) proved the importance of biological assessments to evaluation of the impacts of chemical pollutants. Cairns and his colleagues (Cairns 1974, 1988*a, b,* Matthews et al. 1982) have built substantially on that foundation. Virtually all efforts, however, continue the focus on the evaluation of degradation deriving from chemical contamination (e.g., Worf 1980, Cairns 1981). During the past decade the call for increased use of biological assessment has been motivated by the need to deal with chemical pollutants, both point and nonpoint sources, and the need to reverse other forms of water resource degradation (Karr and Dudley 1981, Karr 1987). The solution of water resource problems will not come from better regulation of chemicals or the development of better assessment tools to detect degradation caused by chemicals. The most critical need is to develop monitoring, assessment, regulatory, and restoration approaches that evaluate the complex dynamics of degradation at local levels and the cumulative regional impacts of anthropogenic disturbance (Karr and Dudley 1981, Bedford and Preston 1988). Rarely can environmental problems be traced to "simple causes, single disturbances" (Bedford and Preston 1988).

Many indicators (Table 5) of the health of biological systems have been tested in recent years (National Academy of Sciences 1986, Schindler 1987, Taub 1987, Ford 1989, Gray 1989, Levin et al. 1989, Pontasch et al. 1989, Karr 1990). Each has sensitivity at different levels of degradation and to different kinds of anthropogenic stress. In addition, measurement difficulty varies considerably among them. The common occurrence of several biological indicators among studies by so many biologists, however, suggest an unusual consensus. Yet, the complexity of biological systems and the diversity of factors responsible for degradation, makes it unlikely that any metric will have sufficient sensitivity to be useful under all circumstances. As a result, biologists are prone to reject many of the more specific approaches that show promise because they cannot be generalized. In fact, we should be integrating aspects of those promising indicators to create a more robust approach to biological monitoring.

The success of IBI in a wide range of stream sizes and geographic areas comes from its integrative use of the independent discoveries of many investigators. However, it is also not adequate. First, IBI was initially restricted to fish communities. Too much time and energy has been expended arguing about which taxon is most appropriate. I believe that just about any taxon could be selected and produce a reasonable level of insight about the water resource if appropriate wisdom is brought to bear on development of robust and general metrics. (Realistically, we must recognize that sampling, identification, or other problems might shift the balance among taxa.) Use of the term IBI with appropriate taxonomic modifiers could help to reduce the tension (e.g., fish IBI, macroinvertebrate IBI). Development of suites of metrics that effectively integrate taxa might also be a goal. However, any use of the IBI concept should reflect the use of a broadly based array of metrics that evaluate conditions from individual, population, community, and ecosystem levels.

Several suggestions are obvious. First, ecologists should support efforts to incorporate biology into the assessment process. Second, ecologists conducting environmental assessments must strive to overcome the tendency to amass unorganized data. In many respects, the inability or reluctance to distill the biological meaning from large quantities of data with rigorous, accurate, yet easily understood analyses has diminished the role of biology in water resource management.

Finally and most important, ecologists should make efforts to develop ways to apply advances in ecological theory to the solution of water resource problems. The need for biological input into evaluation of water resources is in many ways similar to the need that stimulated the synthesis called conservation biology (Schonewald-Cox et al. 1983, Soulé 1986, 1987). A core issue is how to use ecological knowledge to improve our ability to measure and interpret the effects of pollutant exposure or other human impacts on biological assemblages. We must be able to translate this knowledge into statements about the condition (ecological health) of these systems. Ecology as a discipline must contend with questions such as: (1) How can we optimize sampling design to detect patterns given existing spatial and temporal variation? (2) How do we identify and define impairment? (3) How can we improve our ability to detect initial impairment (sensitive indicators, early warning indicators) as opposed to detecting only massive degradation? (4) What should be done to apply integrative, ecological approaches to monitoring in a wide diversity of biological systems?

Recent advances in understanding the ecology of streams (see Matthews and Heins 1987, Stanford and Covich 1988, Yount and Niemi 1990) should also be used to improve the conceptual foundation and use of IBI. Three major world views are emerging with respect to structure and function in stream ecosystems: river continuum (Vannote et al. 1980), patches and boundaries (Schlosser 1982, Townsend 1989), and climatic variability (Schlosser 1987). Each draws attention to specific attributes of stream biotas and the pattern that they exhibit in space and time. The stream continuum hypothesis depicts the stream as an upstream–downstream gradient of gradually changing physical conditions and associated adjustments in energy processes and functional attributes of the biota. The food webs of headwater streams are assumed to be primarily allochthonous, while downstream areas are more autochthonous. The trophic structure of fish and invertebrate communities vary in concert with food resource availability. Some divergence from the stream continuum predictions have been documented (Winterbourn et al. 1981). For example, the shift from heterotrophy to autotrophy changes geographically (2nd–3rd order streams in the northwest vs. 4th or 5th order in the northeast: Minshall et al. 1983). Cummins et al. (1989) provide a perceptive discussion of the way the

TABLE 5. Biological indicators used to assess condition of a water resource with the goal of protecting human health, biotic integrity, or a specific resource. An integrative index approach should include representative metrics across a range of these levels.

Bioassay—procedure of exposing test organisms, in a laboratory setting, to various concentrations of suspected toxicants or dilutions of whole effluent.
Single species test
Multispecies (microcosm, mesocosm) tests
Biosurvey—process of collecting a representative portion of the organisms living in the water body of interest to determine the characteristics of the aquatic community.
Individual/species population (may involve selection of indicator species)
Tissue analysis for bioaccumulation
Biomarkers—genetics or physiology
Biomass/yield
Growth rates
Gross morphology (external or internal)
Behavior
Abundance/density
Variation in population size
Population age structure
Disease or parasitism frequency
Community/ecosystem (may involve indicator taxa or guilds)
Structure
Species richness/diversity
Relative abundances among species
Tolerants/intolerants
Abundance of opportunists
Dominant species
Community trophic structure
Extinction
Function
Production/respiration ratio
Production/biomass ratio
Biogeochemical cycles/nutrient leakage
Decomposition
Landscape
Habitat fragmentation/patch geometry
Linkages among patches
Cumulative effects across landscapes

stream continuum might be used to evaluate anthropogenic effects on stream biota.

According to the patches and boundaries idea, streams are complex landscapes (pools/riffles, small vs. large streams, channel/wetland/upland transition) with flows across the boundaries (Naiman et al. 1988) and the regional pattern of patches being of major consequence in determining the attributes of specific streams. Fuchs and Statzner (1990) provide a recent example of the role of patches and boundaries at the landscape scale in determining the time scale of stream restoration.

Finally, climatic variability affects the biota as it influences the seasonal patterns of flooding, or in a larger sense, the hydrology of the watershed. These influence the evolution of stream biotas (Horwitz 1978) and, following watershed modifications by humans, the persistence time of species (Toth et al. 1982) and communities in flowing waters.

In a larger sense, biological responses to stress have been explored at virtually all levels, including individual (Selye 1973), population (Emlen and Pikitch 1989, Underwood 1989), community (Gray 1989), ecosystem (Rapport et al. 1985, Schindler 1987, Rapport 1989) and landscape (Peterson et al. 1987, Bedford and Preston 1988, Preston and Bedford 1988, Hunsaker et al. 1989, Suter 1990). Community structure changes in three ways in response to stress (Gray 1989): reduction in diversity, retrogression to dominance by opportunist species, and reduction in mean size. Gray argues that the first stages of impact are shown by moderately common species, although most monitoring attention concentrates on common species. Emlen and Pikitch (1989) suggest that models to understand stress responses at the population level may involve purely mechanical descriptors of dynamics based on demographic parameters or models that describe the mediation of demographic parameters by environmental factors, such as those imposed on aquatic biota by a variety of human actions.

Ecological research has greatly expanded knowledge of stream ecology and the dynamics of water resource systems. Unfortunately those insights have not been effectively used in the protection of water resources. Biological inputs to water resource programs have often been limited to narrow toxicological considerations. The time is ripe for ecology and ecologists to use their science more effectively in the protection of water resources.

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