

The Development and Application of Multimetric Indices for the Assessment of Impacts to Fish Assemblages in Large Rivers: A Review of Current Science and Applications

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ABSTRACT

The biological assessment of lotic resources in much of the U.S. and Canada initially focused on wadeable rivers and streams. However, increased emphasis is being placed on larger, non-wadeable rivers. Many of these efforts include the development of multimetric indices represented by the Index of Biotic Integrity (IBI) following the original developmental work in the U.S. They include the pioneering work in the Wabash River of the Midwestern U.S., the inland rivers of Ohio and Wisconsin, the Ohio River mainstem, large western rivers and Quebec rivers, all of which focused on the fish assemblage. Monitoring fish assemblages in large rivers includes logistical and technical considerations that affect obtaining reliable estimates of relative abundance for all species that are amenable to efficient capture. A single gear approach is preferred for practical reasons and electrofishing is the sampling method of choice. Sampling effort is expressed in terms of distance sampled at a site and includes formulas based on fixed distances or multiples of river channel width. Relative abundance data are analyzed via multimetric indices (e.g., Index of Biotic Integrity), which are contingent on the development of a reference condition that supports a derivation and calibration process. Defining reference for large rivers represents a different challenge than with smaller, wadeable streams. For the latter, sufficient and suitable reference analogs usually exist, thus reference condition can be empirically derived. However, such analogs are either rare or do not adequately reflect the restorable potential for large rivers. Thus in developing the expectations that are necessary for metric calibration and IBI development, adequate historical knowledge of the assemblage is critical. Once developed, the metrics and indices provide meaningful measures of assemblage quality and response to chemical, physical, and biological influences and perturbations. This has been demonstrated for a wide variety of human impacts including water pollution, habitat and flow alterations, and land use changes. Successfully applying this protocol to large rivers involves taking the correct sequence of steps in the initial development of sampling

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and assessment methodologies. The IBI serves not only as an important benchmark of aquatic resource quality and condition, but also as a test of the significance of human impact on the aquatic environment. Developing and implementing a multimetric approach for large, coldwater rivers is feasible and would serve as a useful assessment and planning tool for determining the magnitude and severity of impacts from riverine flow and habitat modifications.

RÉSUMÉ

Les évaluations des ressources biologiques lotiques aux États-Unis et au Canada ont principalement porté sur des rivières dans lesquels on peut travailler à gué. Cependant, une importance de plus en plus grande est portée sur les cours d'eau plus profonds. Un effort particulier a été consacré au développement d'indices d'intégrité biologiques depuis la publication des premiers travaux dans ce domaine aux États-Unis. Ces efforts incluent les travaux originaux sur la rivière Wabash située dans le Midwest américain, ceux réalisés dans les états d'Ohio et du Wisconsin, la rivière Ohio et plusieurs grandes rivières de l'ouest américain et du Québec dont la plupart traitent de la communauté ichthyenne.

Les défis associés à l'échantillonnage des grands cours d'eau sont à la fois d'ordre technique et logistique et les contraintes peuvent affecter l'estimation des abondances relatives. La pêche électrique est la méthode de prédilection et certains protocoles exigent un échantillonnage nocturne. L'effort de pêche est alors exprimé en terme de distance pêchée et cette distance est souvent un multiple de la largeur du cours d'eau. L'objectif est de produire une estimation de l'abondance relative en incluant les espèces propices à une capture efficace. Les données ainsi récoltées, souvent très abondantes, sont analysées à l'aide d'indices multimétriques, qui sont calculées par rapport à une condition définie comme point de référence. La définition des conditions de références dans les cours d'eau plus profonds représente un défi plus grand que dans le cas des rivières où l'échantillonnage peut être fait à gué. Dans le cas des cours d'eau peu profonds, il existe déjà de nombreuses banques de données comparatives et une condition de référence peut être développée de manière empirique. Cependant, les points de comparaison sont rares dans le cas des cours d'eau plus profonds. La connaissance historique de la faune aquatique devient alors plus critique. Une fois les métriques développées et les indices calculés, ils donnent une indication adéquate de la qualité de l'assemblage communautaire de l'écosystème et de leur réponse aux perturbations chimiques, biologiques et physiques du milieu. Le succès de l'application de telles méthodes à des rivières plus froides et plus profondes est possible et une fois développés, ces IIB pourront servir pour l'évaluation de l'amplitude des impacts de la modification du régime hydraulique des cours d'eau et des modifications d'habitats.

INTRODUCTION

The challenges facing aquatic resource management involve complex interactions of biotic and abiotic factors that take place at multiple scales. Practical yet ecologically sound concepts, criteria, and attendant monitoring and assessment tools must be better used and institutionalized in this process. Without these tools, the methods and criteria used to assess and support aquatic resource management will continue to be dominated by chemical and physical surrogates. While this has been especially true of the U.S. *Clean Water Act* (CWA), it also exists within other water resource management policies and regulations. This is due frequently to the emphasis by management and regulatory agencies on these more easily measured and implemented surrogates (National Research Council, 2001), which is a partial result of the constant demand for quick results. It has been only during the past two decades that biological assessment methodologies and criteria that are practicable to implement yet ecologically robust, have become available. Key among these advances is the Index of Biotic Integrity (IBI) (Karr, 1981), which introduced the concept of multimetric assessment indices to the assessment of aquatic assemblages. While much of the ensuing use of the IBI in North America has supported water quality applications, it offers a largely unrealized potential to contribute to other water resource management issues using the same approach.

An important goal of our review is to show that developing and implementing a multimetric approach for large, coldwater rivers is not only feasible, but could serve as a useful assessment and planning tool for determining the magnitude and severity of impacts from riverine flow and habitat modifications. The best examples of developing and using IBI to assess a variety of impacts to riverine resources (i.e., water quality, land use, habitat, etc.) are presently from warmwater systems with additional examples only beginning to emerge for coldwater systems. The lessons learned in these situations provide ample guidance for more widely applying multimetric assessments to all types of rivers including coldwater rivers of the northern latitudes. In 2002, we completed field sampling as part of a fish assemblage assessment of the lower 130 km of the Kennebec River (Maine) mainstem. An important goal of this study is the eventual development of a fish IBI for the assessment of multiple stressors and multiple resource management objectives, some of which include flow, habitat and fish passage issues.

The IBI was developed originally in response to the desire to have an ecological assessment tool to meet the needs of government agencies, yet not sacrifice technical rigor and competence for cost-effectiveness. It was initially developed to monitor fish assemblages in wadeable streams of the Midwestern U.S. (Karr, 1981), but was soon adapted to other regions of the U.S. (Fausch *et al.*, 1984; Karr *et al.*, 1986; Miller *et al.*, 1988; Hughes *et al.*, 1998), in Canada (Steedman, 1988; LaViolette and Richard, 1996; Rioux and Gagnon, 2001; Kovacs *et al.*, 2002; LaViolette *et al.*, 2003), and in Europe (Oberdorff and Hughes, 1992). Modifications of the IBI and its metrics were later adapted to different assemblages such as aquatic macroinvertebrates (Ohio EPA, 1987; Plafkin *et al.*, 1989; Kerans and Karr, 1992; DeShon, 1995; Barbour *et al.*, 1996); wetland invertebrates (Euliss *et al.*, 2001),

vertebrates (Miccachion, 2002), and plant assemblages (Kantrud and Newton, 1996; Gernes and Helgen, 1999; Mack, 2002); coastal marine systems (Engle *et al.*, 1994; Deegan *et al.*, 1997; Weisberg *et al.*, 1997; Van Dolah *et al.*, 1999); and terrestrial invertebrate assemblages (Kimberling *et al.*, 2001). The IBI is used by many states to report on aquatic resource condition (Davis *et al.*, 1996); at least one state has formally included IBI as a numeric criterion in their water quality standards (Yoder and Rankin, 1995a) and others are near adoption. Like any tool, *IBI must be used appropriately and within the guidance provided by the originators.*

IBI was designed to monitor assemblage condition at a large number of sites to identify areas in need of remedial attention and for determining changes over time; it has proven adequate for these purposes in many places. Steedman (1988) described the IBI as “. . . based on simple, definable ecological relationships performing as an ordinal, if not linear measure, which responds in an intuitively correct manner to known environmental gradients (and) . . . when incorporated with mapping, monitoring, and modeling information it should be invaluable in determining management and restoration requirements.” One beneficial aspect has been the numeric expression of professional judgement about the condition of an aquatic assemblage. This satisfies the need for an assessment tool that expresses aquatic assemblage health and well-being in an easily grasped, intuitive manner, which allows program managers (who frequently are not biologists) to ‘visualize’ biological condition and response.

Two areas of IBI development present contemporary challenges: non-wadeable, large rivers and coldwater assemblages. Large rivers present the challenges of obtaining representative samples, determining reference conditions, understanding how assemblages respond to anthropogenic stressors and dealing with human impacts that occur across multiple spatial and temporal scales. Unlike wadeable streams, obtaining a representative sample from a larger, deeper, and frequently swifter flowing river involves what are perceived to be greater, even daunting logistical challenges. Because of the historical legacy, severity, and regional scope of human impacts, few if any analogs of least impacted reference condition exist in large rivers. Coldwater assemblages exhibit naturally low species diversity, which is widely perceived as a disadvantage for IBI development. These issues have direct relevance to the management challenges involving the flow and habitat alterations in many rivers of the U.S. and Canada.

METHODOLOGICAL ISSUES

This review focuses on the assessment of fish assemblages in large rivers. The definition of the term ‘large’ has been somewhat varied and other descriptors such as ‘great rivers’ have been advocated to describe the largest continental rivers such as the Ohio, Mississippi and St. Lawrence Rivers. Several have attempted to use watershed area as an indicator and while it is useful as a general descriptor, it does not serve as a decision criterion. Others have used terms such as wadeable and non-wadeable to describe the transition from smaller streams to large rivers. However, this can vary

by organism group – what is wadeable for sampling macroinvertebrates, periphyton or mussels may be non-wadeable for fish. For the purposes of this review, a large river is a lotic watercourse (at least in its natural, unimpeded flowing state) that requires the use of boat-mounted equipment to effectively and efficiently sample. In our experience in the Midwestern U.S. and Maine, and the experience of others elsewhere, this includes small rivers or even large streams that are either too deep or too wide to *effectively* sample using wading methods and gear. At the transition between wading and boatable sampling sites, there is considerable overlap in terms of watershed area and an independent development and calibration of each method is required (Ohio EPA, 1987; Yoder and Smith, 1999). Deciding which method to employ is an on-site decision by an experienced biologist. However, taking either method outside of the calibration domain is prohibited by the Ohio EPA (1987) methodology. This is the essential issue involved, using a consistent application of a sampling methodology within the bounds of its development and calibration.

As rivers become larger and more continuously navigable with sampling boats, the distinction focuses more on the difference between large and great rivers, since great rivers contain unique faunal assemblages distinctive from large rivers (Simon and Emery, 1995; Gammon and Simon, 2000). They also present the potential for different sampling methodologies such as night electrofishing (Emery *et al.*, 2003) and multiple sampling gears (Simon and Lyons, 1995; Penczac *et al.*, 1998; Casselman *et al.*, 1990; La Violette *et al.*, 2003). However, U.S. government agencies are usually interested in the single most effective gear type for conducting multimetric assemblage assessments over large geographic areas.

Biological assessments have been conducted in large freshwater rivers of the U.S. since the late 1940s. These focused on the most easily measured biota of the time (macroinvertebrates, periphyton, plankton); fish assemblage assessments were considered too difficult and expensive to accomplish. However, technological advances in electrofishing in the late 1960s and early 1970s were a major leap forward in sampling technology. These developments made the cost-effective and relatively rapid assessment of fish assemblages practicable (Yoder and Smith, 1999). Standardized sampling procedures are designed to produce a sufficiently representative sample of the assemblage with reasonable effort (1–3 hours/site, several sites sampled per field day, 10s of sites sampled per season, etc.). As such, this type of assessment is distinguished from resource intensive efforts that employ multiple collection gear, and those used to estimate populations, standing crop or an inventory of all species present.

Electrofishing has been determined to be the single most effective gear for obtaining fish assemblage data in Midwestern U.S. streams (Funk, 1958; Larimore, 1961; Boccardy and Cooper, 1963; Bayley *et al.*, 1989), in large rivers (Vincent, 1971; Novotny and Priegel, 1974; Gammon, 1973; 1976; Hendricks *et al.*, 1980; Ohio EPA, 1987), in the Ohio River (Sanders, 1992; Simon and Emery, 1995; Simon and Sanders, 1999), and the Lake Erie shoreline (Thoma, 1999). It is the gear of choice for local, State, and Federal agencies in many other parts of the U.S. (U.S. EPA, 1991; Meador *et al.*, 1993) and elsewhere (Kovacs *et al.*, 2002). Gammon's (1973, 1976) pioneering work in the Wabash River in Indiana serves as the conceptual model for the collection

of relative abundance data from large rivers. While we acknowledge that more information can be gained by adding gear types, the goal of assemblage sampling to develop IBI assessments is to produce estimates of overall assemblage condition as a reflection of overall environmental quality by assessing a portion of the total assemblage. To meet this goal, boat electrofishing has been consistently shown to produce the most information in return for effort.

Representativeness

Gammon (1973; 1976) assessed the representativeness of a standardized, pulsed D.C. large river electrofishing technique that has been followed by many subsequent large river practitioners. Gammon determined that sampling along the shoreline with the greatest depth, most diverse habitats, and abundant cover was the most effective *single* method for producing a representative assessment of a riverine fish assemblage. While this approach does not collect all of the species present in a river, it can account for 75–80% of the species that are present and also approximate their relative abundances. This meets the purposes and requirements for biological assessment in that sufficiently representative data are produced to provide reliable signal about assemblage condition without the need to accomplish a complete faunal inventory or population estimate. The large river IBI development studies that followed Gammon's pioneering work have also demonstrated the utility of the approach. Lyons *et al.* (2001) developed an approach for the large rivers of Wisconsin, but correctly observed that single gear assessments may not be completely applicable for rare or single species assessments or for detailed fisheries management tasks such as stock assessments of commercially or recreationally important resources. However, broad agreement between overall assemblage condition and the correspondence of suitable conditions for rare species and resource management goals has been amply demonstrated (Hughes and Gammon, 1987; Yoder and Rankin, 1995a). Protecting the assemblage generally results in the adequate protection of the fundamental attributes and processes that determine the individual resource attributes of a large river resource. While such an approach is not without criticism, it offers a relatively rapid screening tool that can be followed with more detailed investigation.

Precision and Accuracy

Ohio EPA (1987) extensively tested the reproducibility, accuracy and precision of the types of boat electrofishing methods used by most large river fish assemblage assessments. Based on a combination of data analyses from controlled methods testing studies and the aggregate statewide database, the reproducibility of an IBI score was determined to be four units on a 12 to 60 scoring scale. Rankin and Yoder (1990a) determined coefficient of variation (CV) values of 8–10% at least impacted and high quality sites. CVs increased above this range at sites with lower IBI scores (up to 50–60%), presumably due to spatial instability of the assemblage

under increased stress and exposure. Fore *et al.* (1993) performed extensive statistical analyses of the Ohio database and determined that IBI scores were reproducible to an error margin of 2–3 units. Their power analysis confirmed that the Ohio IBI was capable of distinguishing six discrete scoring ranges that correspond to delineations of the IBI scale into qualitative descriptions of exceptional, good, fair, poor, very poor and no fish present. Angermier and Karr (1986) analyzed other statistical properties of the IBI focusing on the degree of redundancy among metrics. The results of their analysis showed that careful construction and derivation of an IBI following the guidance of Karr *et al.* (1986) should produce a robust and non-redundant set of metrics. Ohio EPA (1987) and the recent work of Lyons *et al.* (2001) and Mebane *et al.* (2003) detail a similar approach for large rivers, the latter dealing with coldwater assemblages.

Accuracy is determined by the consistency and representativeness of assignments of environmental quality produced by the subject method. A key attribute of accuracy is the minimization of Type I and Type II assessment errors. Biological assessments are accepted as a direct measure of the aquatic life protection goals of the U.S. *Clean Water Act* and State water quality standards (as compared to surrogate assessments by chemical water quality criteria). This has given rise to the national biological criteria program at U.S. EPA (U.S. EPA, 1990) and includes methods guidance (Barbour *et al.*, 1997) and the development of formal implementation procedures (U.S. EPA Aquatic Life Uses Working Group). The issue at stake is the accuracy of the delineation of waters as impaired or unimpaired for CWA purposes (e.g. TMDLs). Historically, states and U.S. EPA based these decisions on chemical water quality data and comparison to state and national chemical water quality criteria. However, studies that compared the relative performance of chemical and biological data showed that biological indicators based on IBI detected more impairment and minimized Type II assessment errors (Rankin and Yoder, 1990b; Yoder and Rankin, 1998). The issue is not so much that biological assessments detect more impairment, but that they shed important light on the types and categories of impairment, which might be under-rated or overlooked altogether. The net effect of missing these things is an incomplete response to the management of aquatic resources. The inclusion of biological assessment alone is insufficient; it must include the right mix of methods and tools that are sufficiently developed and calibrated to ensure a high level of accuracy in the resulting assessment of biological quality and status.

ASSEMBLAGE ASSESSMENT PROCESS

Early fish assemblage assessments in large rivers of the U.S. were most frequently associated with studies of thermal effluents in response to CWA Section 316[a]. A common frustration with those studies was the lack of a systematic and conceptually robust framework for characterizing the assemblages, analyzing the data, and producing meaningful assessments. The later advent of the IBI provided a conceptual framework for improving the assessment of large river fish assemblages. Contemporary examples of comprehensive application to aquatic resource assessment

include the inland large rivers of Ohio (Ohio EPA, 1987; Yoder and Smith, 1999), the Ohio River mainstem (Simon and Emery, 1995; Emery *et al.*, 2003), and inland Wisconsin rivers and the Mississippi River mainstem (Lyons *et al.*, 2001). The Wisconsin and Ohio examples especially demonstrate the utility of the IBI as an assessment tool for assessing multiple impacts from multiple sources as shown in subsequent parts of this paper. Extensive work in Quebec includes the assessment of numerous rivers, including the St. Lawrence mainstem, using the IBI as a primary assemblage assessment tool (Richard, 1994; 1996; LaViolette and Richard, 1996; Martel and Richard, 1998; St. Jaques, 1998; LaViolette, 1999; St. Jaques and Richard, 2002; LaViolette *et al.*, 2003). This has improved our understanding of the ecological consequences of human impacts on large warmwater rivers and the sequence in which they occur.

While the aforementioned examples have dealt with the comparatively species-rich, warmwater and coolwater fish assemblages of the Midwestern U.S. and Quebec, examples of work with comparatively species-poor and cool to coldwater systems are emerging. Hughes and Gammon (1987) applied the large river and IBI assemblage assessment procedures to the Willamette River, Oregon. They not only found that a single gear approach (pulsed D.C. electrofishing) could work in a river that changes from cold to cool water, but that an IBI could be developed for the species-poor rivers of the Pacific northwest U.S. Sampling of other large rivers in the western U.S. was accomplished as part of U.S. EPA's Western Environmental Monitoring and Assessment Program (EMAP), the U.S. Geological Survey National Water Quality Assessment (NAWQA), and the state of Idaho (Mebane *et al.*, 2003; Robert Hughes, personal communication), which included large, coldwater rivers of the intermountain west and coastal northwest.

Index of Biotic Integrity (IBI)

Karr's (1981) original IBI is comprised of twelve measures or 'metrics' that represent species composition, trophic composition, abundance, and organism condition (Table 1; Karr *et al.*, 1986). Data obtained by sampling at a site are evaluated with respect to what is 'expected' in terms of a relatively unimpacted reference condition for a similar ecotype and region. Each metric is scored in accordance with the outcome of a calibration process as to how it compares to reference condition. Higher metric scores are awarded to results that most closely match the reference condition; the scoring is scaled to reflect the magnitude of departures and metric scores are summed into a final index score. The strength of this approach is the integration of information from the individual, population, community, zoogeographic and ecosystem levels into an ecologically-based index that reflects the quality of the water resource (Karr *et al.*, 1986). Like the multifaceted indices that are used to measure economic and human health (e.g., econometrics), IBI integrates multiple biological attributes to measure and communicate biological condition and thereby the ecological health of a river. The IBI is adaptable to diverse types of aquatic ecosystems, geographies, and environmental conditions and has proven to be a

Table 1. Modification of Index of Biotic Integrity (IBI) metrics used by the Ohio EPA to assess wading sites, headwater sites, and boat sites. The original IBI metrics of Karr (1981) are listed first (*italics*) with substitute metrics following (after Yoder and Smith, 1999).

IBI Metric	Headwater Sites ^{3,4}	Wading Sites ⁴	Boat Sites ⁵
<i>Number of Species</i> Number of Native Species ⁶	X	X	X
<i>Number of Darter Species</i> Number of Darter and Sculpin Species	X	X	
%Round-bodied Suckers ⁷			X
<i>Number of Sunfish Species</i> ⁸ Number of Headwater Species ⁹	X	X	X
<i>Number of Sucker Species</i> Number of Minnow Species	X	X	X
<i>Number of Intolerant Species</i> Number of Sensitive Species ¹⁰	X	X	X
% <i>Green Sunfish</i> %Tolerant Species	X	X	X
% <i>Omnivores</i>	X	X	X
% <i>Insectivorous Cyprinids</i> %Insectivores	X	X	X
% <i>Top Carnivores</i> %Pioneering Species ¹¹	X	X	X
<i>Number of Individuals</i> Number of Individuals (less tolerants) ¹²	X	X	X
% <i>Hybrids</i> %Simple Lithophils Number of Simple Lithophils	X	X	X
% <i>Diseased Individuals</i> %DELT Anomalies ¹³	X	X	X

³applies to sites with drainage areas <20 mi²; ⁴sampled with wading electrofishing methods; ⁵sampled with boat electrofishing methods; ⁶excludes all exotic and introduced species; ⁷includes all species of the genera *Moxostoma*, *Hypentelium*, *Minytrema*, *Erimyzon*, *Cycleptus*, and *Ericymba*, and excludes *Catostomus commersoni*; ⁸includes all *Lepomis* species except green sunfish; ⁹species designated as permanent residents of headwater streams; ¹⁰includes species designated as intolerant and moderately intolerant (Ohio EPA, 1987b); ¹¹species designated as frequent and predominant inhabitants of temporal habitats in headwater streams; ¹²excludes all species designated as tolerant, hybrids, and non-native species; ¹³includes individuals with deformities, eroded fins or barbels, lesions, and tumors.

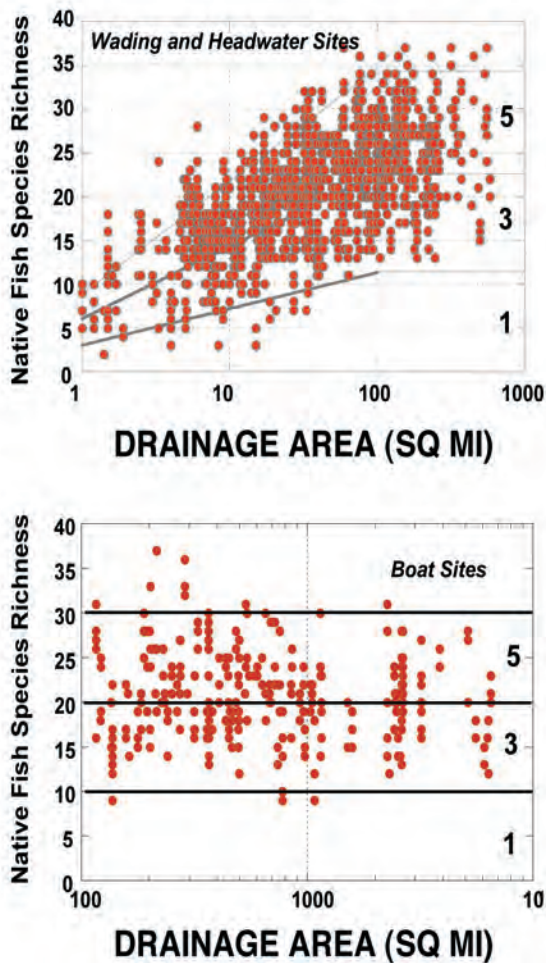


Figure 1. Ohio EPA Index of Biotic Integrity (IBI) Species Richness Metric Calibration Curves for Wadeable and Headwater Sites (Upper Panel) and Boatable (Non-Wadeable) Sites (Lower Panel). Data Points are From Least Impacted Regional Reference Sites Sampled Between 1980 and 1989 Using Standardized Methods (Ohio EPA, 1987). Scoring is Accomplished by Finding the Sample Result (Knowing the Watershed Area at the Site) within One of the Three Scoring Ranges. The Cumulative Scores for Multiple Metrics are then Summed to Produce the IBI Value.

more sensitive indicator of human impacts than conventional chemical and physical measures of those impacts (Karr and Chu, 2000).

The first IBIs employed a 5–3–1 scoring sequence with 5 being closest to reference and 1 deviating strongly from reference for each of 12 metrics making the possible scoring range from a maximum of 60 to a minimum of 12. An example from Wadeable streams and inland large rivers in Ohio illustrates the metric substitution (Table 1) and the scoring and calibration processes (Figure 1; Ohio EPA, 1987). Later improvements and innovations include modified scoring procedures for highly degraded sites (Rankin and Yoder, 1999), 0–100 point scaling (Lyons, 1992; Lyons *et al.*, 2001; Mebane *et al.*, 2003), and statistically–derived calibration techniques (Mebane *et al.*, 2003). Some misapplications have occurred and include the inappropriate accumulation of data from multiple sites and dates, including highly degraded sites in the calibration process, and the incorporation of inappropriately transformed data within metrics. The IBI is to be calculated for individual samples and dates, calibrated against minimally or least impacted reference conditions, and include relative abundance or taxa richness data as metrics.

Karr *et al.* (1986) recognized that the IBI would need to be modified to account for differences in assemblages across regional and ecological gradients. As such, the process of metric substitution (Table 1) was envisioned and guidance for accomplishing this task was developed. Hughes and Gammon (1987) applied this guidance in developing an IBI for the Willamette River, Oregon and found that an IBI could be developed for a naturally species–poor system. Frequently, we encounter the notion that depauperate faunas are less than suitable for IBI development, thus protocols involving fish are not widely attempted. The experience of Mebane *et al.* (2003) demonstrates that capturing the fundamental mosaic of structural and functional characteristics and attributes in the IBI is critical (Table 2). Key to their success was the inclusion of alternate metrics such as invasive alien species, which also function as undesirable generalists and tolerant species in these systems. They also incorporated the fundamentals of how coldwater systems respond to stress, both in the taxa richness and proportional metrics. Unlike warmwater faunas, which exhibit nearly linear declines in species richness with increasing stress, coldwater systems may show an overall increase, the result of the addition of alien and opportunistic warmwater species that invade as conditions become favourable (Lyons *et al.*, 1996; Mundahl and Simon, 1999). Other coldwater IBIs appear in Table 3.

USING IBI IN WATER BODY ASSESSMENTS

Karr *et al.* (1986) originally envisioned the strength of the IBI for supporting ecologically–based assessments of water resource condition. They provided some initial examples of regional and local scale uses, but the wider implementation of IBI in water resource programs had not yet occurred. Since that time, examples of the systematic application of IBI as a routine assessment tool have emerged (Karr and Chu, 1999).

Table 2. Assessment of the ability of candidate metrics of a large river, coldwater IBI to discriminate between minimally–disturbed reference sites and anthropogenically–disturbed sites. Probability values are from the Mann–Whitney test (after Mebane *et al.*, 2003).

No.	Candidate Metric	P	Included in Index?
1	# of coldwater native species	<0.01	Yes
2	% native coldwater individuals	<0.01	No. Redundant with no. 1 and 3
3	% coldwater individuals	<0.01	Yes
4	# of alien species	<0.01	Yes
5	% alien individuals	0.30	No. Nondiscriminatory
6	% sensitive native individuals	<0.01	Yes
7	# of sculpin age classes	<0.01	Yes
8	% sculpin individuals	<0.01	Alternate if data for no. 7 missing
9	# of salmonid age classes, excluding whitefish	<0.01	Yes
10	# of salmonid age classes, including whitefish	0.02	No. Redundant with no. 9 and less sensitive
11	% catchable trout	0.31	No. Nondiscriminatory
12	% tolerant individuals	<0.01	Yes
13	% carp	<0.01	Yes
14	% individuals with any anomalies	0.21	No. Nondiscriminatory
15	% individuals with DELT anomalies	<0.01	Yes
16	# coldwater individuals per minute electrofishing	<0.01	Yes.

Table 3. Metrics used for coldwater streams and rivers and metrics from the original IBI of Karr (1981); + or – indicates metric value increases or decrease, respectively, in response to degraded water quality.

Original IBI (Karr, 1981)	Coldwater Stream of the Upper Midwestern United States (Mundahl and Simon, 1999)	Northern and Central Idaho Erosional (e) or Depositional (d) Streams (Fisher, 1989)
# fish species (-) # of sucker species (-) # of darter species (-) # of sunfish species (-) # of intolerant species (-) % green sunfish (+) % omnivores (+) % insectivorous cyprinids (-) % top carnivores (-) % hybrids (+) (CPUE) (-) % with anomalies (+)	# fish species (+) # coldwater fish species (-) # minnow species (+) # of benthic species (+) # of tolerant species (+) % salmonids as brook trout (-) % intolerant individuals (-) % coldwater individuals (-) % white suckers (+) % top carnivores (-) # of coldwater individuals (-) # warmwater individuals (+)	# fish species (e -, d +) # salmonid species (e -) # non-salmonid species (e -, d +) # introduced species (e -) % salmonid individuals (e +) average salmonid length (e -) average salmonid weight (e -) salmonid density (e +, d -) # amphibian species (e +, d -) amphibian biomass (d +) # intolerant species (d -) % hybrids (d +) salmonid biomass (d -) fish biomass (d -) macroinvertebrate density (d +)
Idaho large rivers (Royer and Minshall, 1996)	Southern Idaho small streams (Robinson and Minshall, 1992)	Rivers and streams, Upper Snake River basin, Idaho and Wyoming (Maret, 1997)
% insectivores (-) # intolerant taxa (-) native cyprinid species (+) % carp biomass (+)	# salmonid species (-) # tolerant species (+) % tolerant individuals (+) % Salmonidae (-) salmonid biomass (-) salmonid condition factor (-)	# native fish species (-) # total species (+) % introduced species (-) (+)% cottids (-) # salmonid species (-) % coldwater species (-)

The Ohio EPA has since 1979 conducted biological and water quality surveys of large rivers involving intensive sampling of major river reaches of more than 100 km in length that are influenced by numerous and diverse stressors. Fish assemblage assessments are part of an integrated biological, chemical and physical monitoring effort that is designed to meet three principal objectives: 1) determine the extent to which water quality standards are either attained or not attained; 2) determine if each water body is appropriately classified; and 3) determine if changes in biological, chemical or physical indicators are reflected in efforts to implement pollution controls or best management practices.

The segment of the Scioto River that is impacted by the city of Columbus is an example of such efforts. A nearly 70 km segment of the mainstem is affected by effluent from two major municipal wastewater treatment plants that discharge nearly 200 million gallons of treated wastewater per day. During the summer–fall period this effluent comprises nearly 95% of the flow in the Scioto River downstream from Columbus. Land use in the basin is dominated by row crop agriculture and by urban land use within Columbus. Fish assemblage assessments have been conducted in all but three years since 1979, preceding and following major efforts to reduce pollution effects. Departures from the biological criteria for the IBI indicated poor and very poor quality in 1979, 1980, and 1981 (Figure 2). As major technological improvements were made at the municipal wastewater treatment plants, improvements in the IBI were measured in subsequent years. At least 50% and usually more of IBI values have been higher than the applicable biological criterion since 1991. A more quantitative tool, the Area of Degradation Value¹⁴ (ADV; Yoder and Rankin, 1995b) shows that impairment has been virtually eliminated since 1991, three years after a multimillion-dollar, mandatory upgrade to wastewater treatment was completed. Similar successes have been demonstrated in several other Ohio rivers and streams and these have contributed, in the aggregate, to nearly all of the improvements made in Ohio's aquatic resource condition since 1988. A significant benefit of taking the time and effort to document these changes is the confidence that it lends to wastewater treatment practices. Prior to full implementation, the type of treatment achieved by Columbus was regarded by many as technologically 'too

¹⁴The ADV is a quantitative expression that measures the extent (length of river) and severity of the departure from a biological criterion (how much the IBI is below the goal or criterion) using the changes in IBI over a length of a river segment that includes multiple IBI values. ADV is a mathematical expression of the polygon formed by sequentially connecting the IBI values along a length of river and the minimum biocriterion value – it can be calculated as a negative (below the criterion) or positive (above the criterion) value. It is used to determine the negative departure (impairment) or the degree to which a river exceeds (attainment) the minimum biological criterion and provides a better depiction of quality beyond pass/fail descriptors. It is used to compare results between years in the same river or between different rivers, provided the same calibrated index is used.

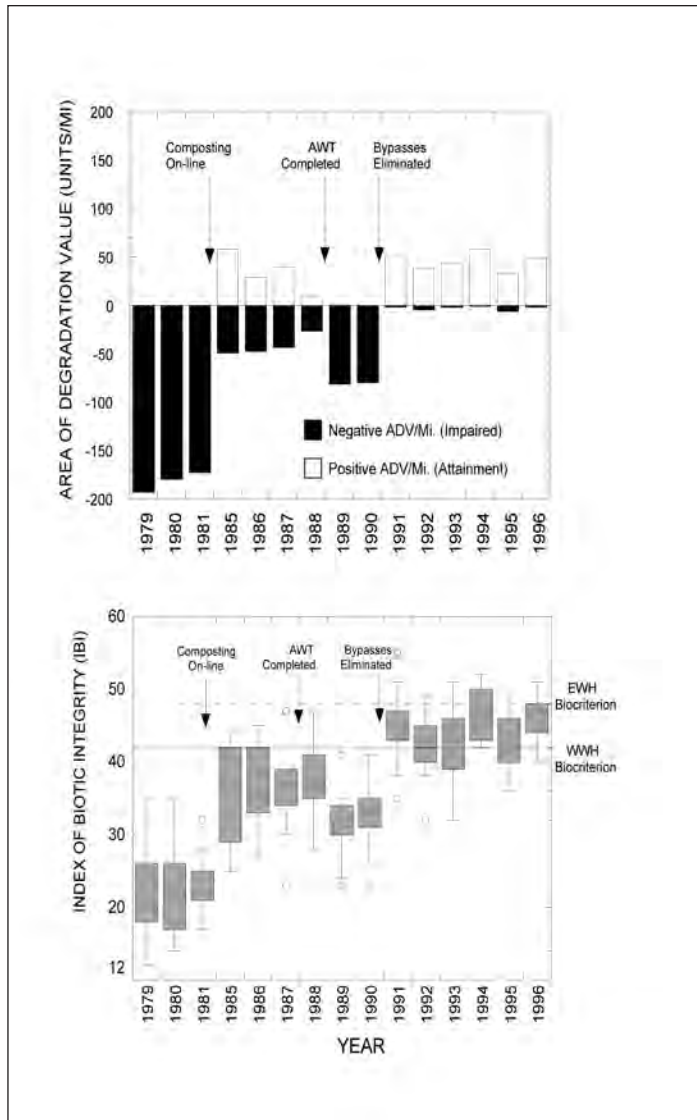


Figure 2. Annual Index of Biotic Integrity (IBI) Results from the Mainstem Scioto River Directly Impacted by Municipal Sewage Discharges and Urban Runoff from Columbus, Ohio between 1979 and 1996 (Lower Panel; WWH = Warmwater Habitat; EWH = Exceptional Warmwater Habitat; AWT = Advanced Wastewater Treatment) and Area of Degradation Values (ADV) Based on IBI Results from the Same Segment and Time Period (Upper Panel). Significant Changes in the Operation of the Sewage System are Noted on Each Panel. The ADV Values Represent the Longitudinal Extent and Severity (Departure from the IBI Biocriterion) of Impairments (Negative ADV) or the Extent to which the IBI is above the Biocriterion (Attainment) in 30 Miles of the Scioto River.

difficult and costly' and the environmental benefits as 'dubious'. The demonstration of the beneficial results in the aquatic resource in this and many other cases proves the benefits of stringent treatment regulations and allays the need to justify its application in each and every case.

Wisconsin DNR

Lyons *et al.* (2001) developed an IBI for the large rivers of Wisconsin to address a pressing need for a tool to assess the diverse impacts from pollution and other stressors including flow and habitat influences. Typical of many Midwestern U.S. rivers, they have been altered by physical modifications, primarily impoundments for hydroelectric power generation and flood control, and polluted by municipal, industrial and agricultural wastes. Wisconsin also has several relatively unmodified and unimpacted large rivers that offer direct data on reference condition. The IBI was developed after a large, statewide database of standardized fish assemblage data was produced and analyzed. Following a validation and testing phase, the IBI was used to assess common impacts to the large rivers of Wisconsin (Figure 3).

Sampling sites were classified into one of six classes, five of which reflected the predominant human impacts to which each was subjected. These included impoundments (upper reaches of impounded sections formed by dams), peaking sites (located within 30 km downstream from hydropower dams with daily flow fluctuations of at least a factor of two), nonpoint sources (watersheds with at least 50% as intensive agriculture or 20% as urban), point sources (affected by major pollution effluents), and multiple impact sites (combination of two or three of the preceding impacts). Reference sites comprised the sixth class and reflected least impacted conditions with minimal human disturbance. These sites not only had the highest IBI scores, but also exhibited the least variability (Figure 3). This is a desirable trait of an IBI and reflects both comprehensive index development and stable reference condition. In terms of mean IBI scores, multiple impacts resulted in the poorest quality assemblages and point sources the best compared to reference condition. The latter reflects the substantial progress made in the control of point sources via U.S. CWA regulation. Hydrologic modifications ranked behind point sources, but ahead of nonpoint sources and multiple impacts. The results within any single impact category were variable reflecting differing degrees in the severity and magnitude of an impact at a given site. The hydropower peaking sites ranged in quality from poor to excellent (Figure 3). The severity of the impact by flow fluctuations downstream from the peaking hydroelectric facilities was increased when the distance to the next dam downstream was shortest, presumably due to less refugia being available in these situations.

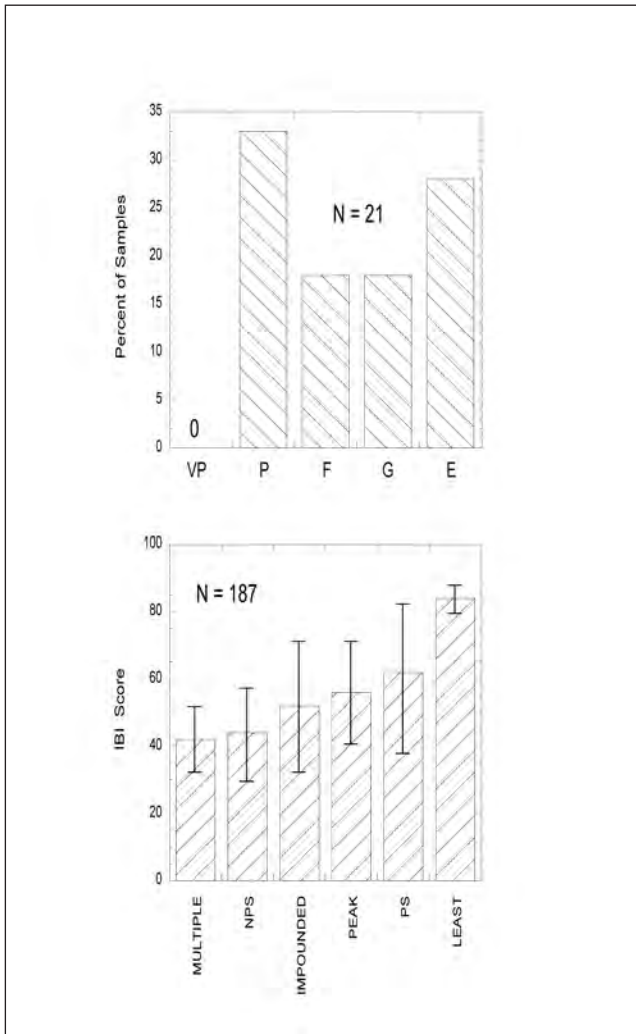


Figure 3. Mean IBI Scores and 95% Confidence Intervals (Lower Panel) and the Frequency Distribution (%) of IBI scores among Five Condition Ratings (Upper Panel; VP = Very Poor; P = Poor; F = Fair; G = Good; E = Excellent) for the Peaking Impact Type. Data are from 187 Large River Sampling Sites in Wisconsin (After Lyons *et al.*, 2001).

St. Francois River (Quebec)

Kovacs *et al.* (2002) used the IBI developed by Richard (1996) to assess the impacts from pulp mill effluents. The intent is to determine the adequacy of Canadian regulations that pertain to pulp mills and the sublethal effects of their effluents. Part of this process includes an assessment of macroinvertebrates and fish in the receiving waters once every three years. In addition, the regulations require an examination of morphological and life history characteristics of two sentinel fish species. The results of comparing multiple years of biological assessment before and after the upgrading of pulp mill effluent treatment processes showed the IBI was the most sensitive and responsive indicator of improved water quality. Various biochemical and fish health measurement by comparison either did not detect changes or were inconclusive as to the associated cause.

DISCUSSION

The strengths of the IBI approach to the assessment of large river fish assemblages are: 1) its cost-effectiveness as a relatively rapid assessment tool; 2) as an assessment of aquatic ecosystem quality at the local, river reach, and multi-jurisdictional regional scales; and 3) its ability to measure biological condition in an incremental manner. Many large river systems in the U.S. and elsewhere have never been systematically monitored with standardized assessment tools. This approach provides a way to characterize the comparative quality of rivers for a variety of water resource management efforts including the protection of high quality reaches and those that require immediate intervention and remediation. However, the scale of sampling will determine the eventual ability of a program to know about and manage specific stressors, and understand the sequence and patterns involved with delineating and managing multiple stressors. Therefore, the supporting monitoring and assessment must be comprehensive in terms of integrating chemical, physical and biological indicators, and conducted on a routine basis and at the same scales at which management is applied (Yoder, 1998).

Even though the IBI has been used to demonstrate incremental changes in assemblage condition over time and space at both local and regional scales, it does not express all demonstrable changes that take place within an aquatic assemblage or community. For example, a subtle change in a single species may or may not be reflected in the resultant index score, depending on how that species contributes to the index. Lyons *et al.* (2001) did not see this as a major disadvantage. In fact, they see the IBI as a test of the significance of stressor effects in that those detected by an IBI are the ones that merit the attention of aquatic resource management. However, this will be true only when a comprehensive and robust database and systematic monitoring and assessment process support the development, calibration and verification of an IBI. It also requires careful construction of the IBI and its component metrics, a process that can make or break the ability of the IBI to serve as a meaningful and sufficiently sensitive indicator of environmental change. The

development and calibration process is best achieved at a multi-jurisdictional regional scale (as opposed to reach or site-specific) and over multiple years of methods testing, data collection, index development, and index testing, necessary to capture the full breadth of variability and natural patterns that occur in regional faunas. It also contributes to a better understanding of how the potential of different rivers actually varies across the landscape.

Frequently, chemical and/or physical surrogates and other indirect means of predicting adverse effects drive the pursuit, selection and implementation of water resource management options. These practices fill important gaps that are not addressed by ambient assessment tools like the IBI. However, the role of the IBI is to serve as an indicator of response to stress and exposure that is comprised of a mosaic of chemical, physical and biological phenomena (Yoder and Rankin, 1998; Karr and Yoder, 2003). The linkages between stressor management processes, their effects on key attributes of aquatic ecosystems, and the condition of the impacted waterbody (Figure 4) transcends the comparatively bivariate and linear assumptions inherent to individual stressor-by-stressor management. The sequence of integrating management actions with the environmental reality provided by the proper use of chemical, physical and biological assessment tools and indicators (Yoder and Rankin, 1998), and as suggested in Figure 4, is frequently incomplete or missing altogether in aquatic resource decision-making. In addition, the non-linear interactions of multiple stressors are best captured by the biological response. Human-induced stresses influence water body condition through their effect on five classes of factors (Karr *et al.*, 1986; Karr, 1991; Karr and Chu, 2000) thereby determining biological and other characteristics of a water body (Figure 4). Biological assessment using IBI can provide confirming evidence of our imperfect simulations and estimates of stress-exposure relationships and processes (Figure 4; Karr and Yoder, 2003). Once these relationships are validated via adequate monitoring and assessment, they can be applied more prescriptively and with greater confidence to similar types of management issues.

A better awareness and understanding of these relationships can improve the basis for relevant water resource management policies and management responses. An improved and more comprehensive understanding of these relationships revealed by adequate monitoring and assessment should lead to the development of more effective and productive management responses. Without it we are likely to continue dealing with one problem at a time, which perpetuates the reliance on simplistic management responses and prescription and continues the risk of unintended consequences from our actions. Biological assessment based on the concepts of the IBI represent a profound shift in thinking, putting biological response and quality at the forefront of water resource management. It also emphasizes 'all of the biology' as opposed to approaches that singularly emphasize fragments of the biology such as endangered and commodity species (Karr and Chu, 2000). The latter can also result in an over-emphasis of aesthetic or recreational values as opposed to an emphasis on data and the assessment of what it means to the whole system (Karr *et al.*, 2000).

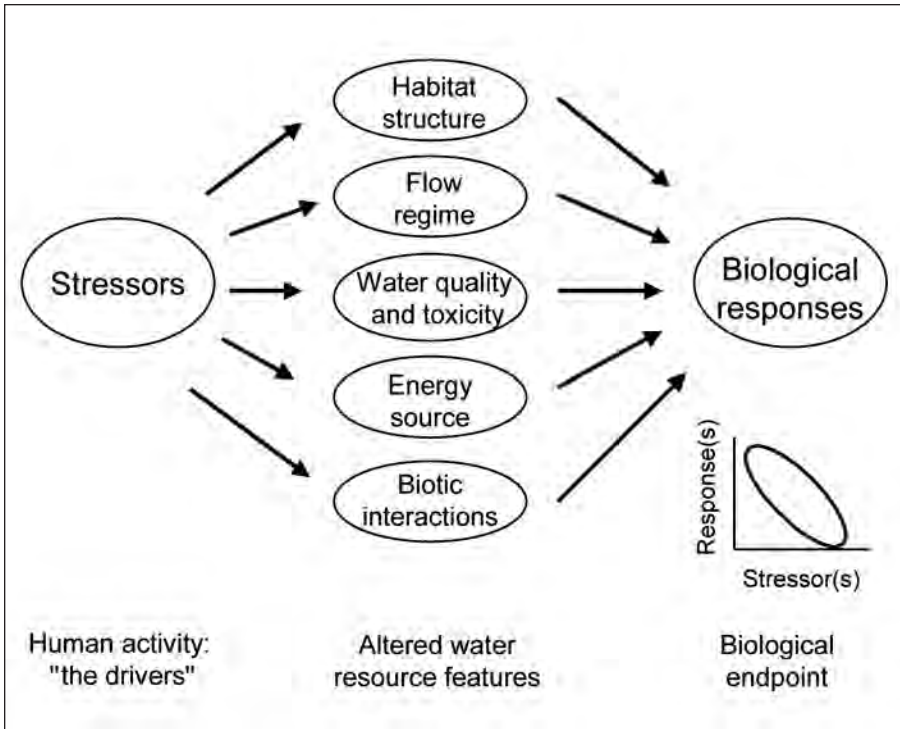


Figure 4. Linkages from Human Activity (the Stressors or Drivers of System Change) through the Five Major Features of Water Resources Altered by Human Activity to the Biological Responses that Produce Ambient Condition, the Biological Endpoints of Primary Interest in Biological Assessment Programs. This Model Illustrates the Multiple Causes of Water Resource Changes Associated with Human Activities. Insert Shows Graphical Relationship between Stessor Dose and the Gradient of Biological Responses Indicative of a Good Biological Metric (After Karr and Yoder, 2003).

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