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An Index of Biological Integrity for Northern Mid-Atlantic Slope Drainages

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Abstract.—An index of biological integrity (IBI) was developed for streams in the Hudson, Delaware, and Susquehanna River drainages in the northeastern United States based on fish assemblage data from the Mohawk River drainage of New York. The original IBI, developed for streams in the U.S. Midwest, was modified to reflect the assemblage composition and structure present in Mid-Atlantic Slope drainages. We replaced several of the Midwestern IBI metrics and criteria scores because fishes common to the Midwest are absent from or poorly represented in the Northeast and because stream fish assemblages in the Northeast are less rich than those in the Midwest. For all replacement metrics we followed the ecology-based rationale used in the development of each of the metrics of the Midwestern IBI so that the basic theoretical underpinnings of the IBI remained unchanged. The validity of this modified IBI is demonstrated by examining the quality of streams in the Hudson, Delaware, and lower Susquehanna River basins. The relationships between the IBI and other indicators of environmental quality are examined using data on assemblages of fish and benthic macroinvertebrates and on chemical and physical stream characteristics obtained during 1993–2000 by the U.S. Geological Survey's National Water Quality Assessment Program in these three river basins. A principal components analysis (PCA) of chemical and physical variables from 27 sites resulted in an environmental quality gradient as the primary PCA axis (eigenvalue, 0.41). Principal components analysis site scores were significantly correlated with such benthic macroinvertebrate metrics as the percentage of Ephemeroptera, Plecoptera, and Trichoptera taxa (Spearman $R = -0.66$, $P < 0.001$). Index of biological integrity scores for sites in these three river basins were significantly correlated with this environmental quality gradient (Spearman $R = -0.78$, $P = 0.0001$). The northern Mid-Atlantic Slope IBI appears to be sensitive to environmental degradation in all three of the river basins addressed in this study. Adjustment of metric scoring criteria may be warranted, depending on composition of fish species in streams in the study area and on the relative effort used in the collection of fish assemblage data.

The biological integrity of a site is a measure of its naturalness; high integrity is associated with populations of native species that interact under natural community processes and functions (Angermeier and Karr 1986). Thus, biological integrity is closely allied with environmental quality, and an assessment of integrity can serve as a surrogate measurement of health. Karr (1981) introduced the index of biotic integrity (IBI) as a bioassessment tool that integrates several attributes of stream fish assemblages and provides a rapid and relatively inexpensive way to assess the general health of streams and evaluate environmental change (Karr et al. 1985). Karr et al. (1986) argued that management of water resources will improve if biological indices that incorporate easily quan-

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tified and understood results from biological monitoring are used as a direct measure of water quality. This concept has been promoted with some success during the last decade (e.g., Simon 1999; Angermeier et al. 2000; McCormick et al. 2001). In the Northeast, IBIs have not been used widely except for statewide IBIs developed for New Jersey, Vermont, and Connecticut (Halliwell et al. 1999; Langdon 2001).

The IBI developed by Karr (1981) for Midwestern drainages consists of 12 fish community metrics that include information from the broad ecological categories of species richness and composition, trophic resource use, fish abundance, and the condition of individual fish in the sample. This concept, although initially developed for warmwater streams in Illinois, has been applied effectively in other regions during the past two decades (e.g., Fausch et al. 1984; Miller et al. 1988; Simon 1991; Klemm et al. 1993; Hughes et al. 1998; Angermeier et al. 2000; Roth et al. 2000; Schleiger 2000; Smogor and Angermeier 2001). The IBI approach has a firm foundation in ecological theory, is simple, is relatively consistent (e.g., Covert 2001), and provides a quantified basis for environmental decision making.

We propose a watershed-based IBI for three large Mid-Atlantic Slope basins—the Hudson, Delaware, and Susquehanna River basins, which together drain much of New York, New Jersey, and Pennsylvania. A distinct IBI is needed for this region because the structure of stream-fish assemblages in the Northeast differs from that in other parts of North America. Although we modified the individual metrics from the Midwestern IBI, we retained the original rationale for our substituted metrics, a need emphasized by Plafkin et al. (1989) and Barbour et al. (1999). The rationale behind and validity of the IBI, as applied to northern Mid-Atlantic Slope drainages, have not been previously documented, although Angermeier et al. (2000) and McCormick et al. (2001) developed an index for the Mid-Atlantic Highlands region, including upland streams in parts of the lower Susquehanna River system. We are not proposing a regional or statewide IBI because (1) streams in New England and most of the small, coastal mid-Atlantic drainages are relatively depauperate and require additional modifications beyond those suggested here (Miller et al. 1988; Halliwell et al. 1999), and (2) fish fauna in the western drainages in New York and Pennsylvania are more diverse than in these northern Mid-Atlantic Slope drainages. An IBI originally developed for the Northeast by Miller et al. (1988) was found generally to be inadequate for streams in the region (Langdon 1989; Jacobson 1994; Kurtenbach 1994; Keller 1995), which led to the development of IBIs designed for application to smaller geographic areas (see Halliwell et al. 1999). The need to extensively adjust the IBI by drainage indicates that a regional IBI for use in all northeastern states and provinces is impractical. Perhaps because of this, the original northeastern IBI (Miller et al. 1988) was never widely used by natural resource management agencies in this region.

Several approaches have been used in developing IBIs for different regions of the country (Simon 1999). The approach we took in developing the northern Mid-Atlantic Slope IBI was to combine our general knowledge of fish ecology with specific data from stream surveys conducted during the past seven decades. Survey results provide data suitable for determining fish assemblage composition and structure despite variation in sampling methods and environmental conditions over time. These data have allowed us to formulate appropriate metrics and develop suitable scoring criteria. The key to the success of any index is its ability to reflect accurately the system it is designed to examine. An index also needs to be robust, that is, usable in a variety of studies, regardless of differences in personnel and data collection methods. Our choice of an approach reflected our desire to develop an accurate and robust index applicable to data collected by a variety of sampling protocols, not to recommend a particular sampling protocol.

Our goal here is to present a useful IBI for assessing stream conditions in three northern Mid-Atlantic Slope drainages that comprise cold-, cool-, and warmwater fish assemblages. We explain the reasons for modifying the metrics of earlier indices that were deemed inappropriate for use with the fish assemblages of these drainages. To test the usefulness of this IBI, we have evaluated its effectiveness in assessing environmental conditions at 27 sites in the three drainages, and we compare the fish IBI results with assessments based on benthic macroinvertebrate and chemical and physical habitat data.

Methods

Historical database.—Development of the northern Mid-Atlantic Slope IBI follows the approach originated by Karr (1981). Establishment of scoring criteria for the metrics requires fish-assemblage data from the least degraded sites in a wa-

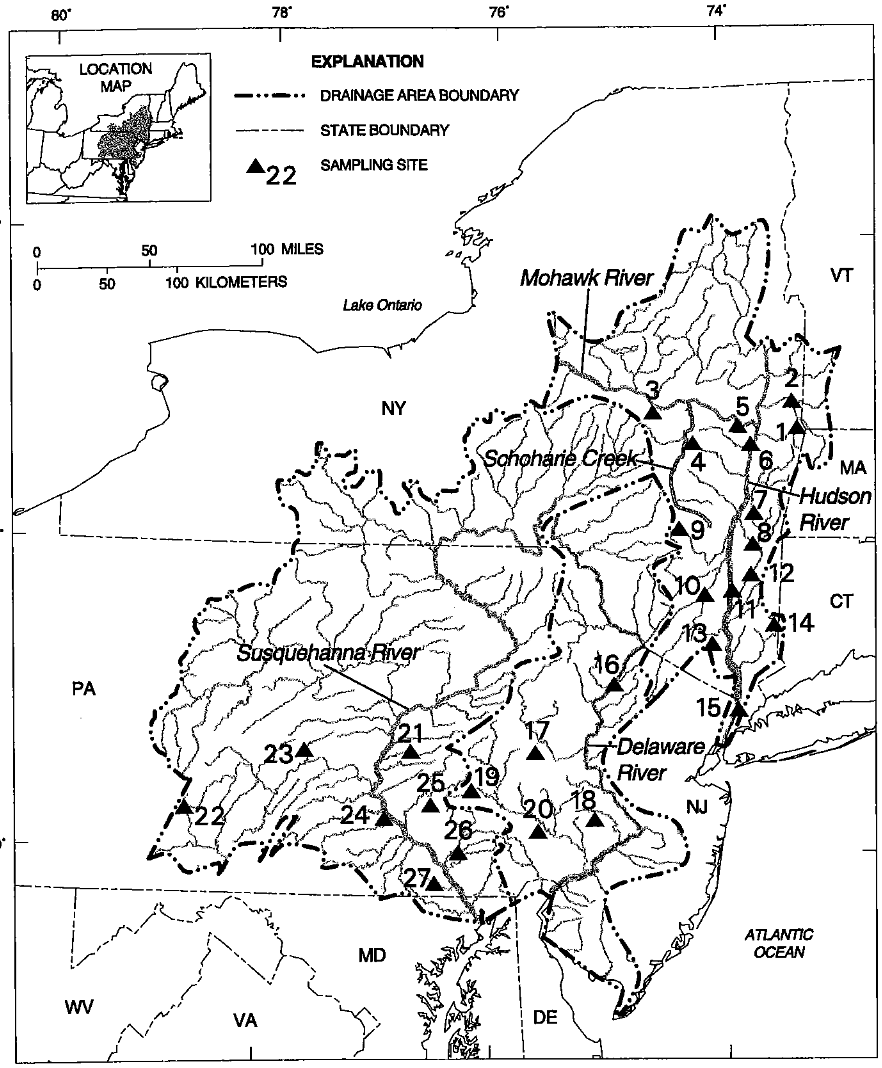

FIGURE 1.—Mid-Atlantic Slope drainages for which the index of biological integrity (IBI) was developed. Sites on the Mohawk River and its tributary, Schoharie Creek, were used to develop the metrics. Fish, macroinvertebrates, and information on chemical and physical characteristics were collected at the numbered sites within the Hudson, Delaware, and Susquehanna River drainages and were used to verify the IBI. Sampling site numbers correspond to those in Table 2.

tershed to provide a standard, or reference condition, against which other sites in the drainage can be scored (Fausch et al. 1984). Historical, quantitative data on fish assemblages in the northeast are uncommon. When such data are available, the identification of a reference condition is usually complicated by the long history of human settlement (Marston and Gordon 1938; Marston 1939).

A comprehensive fish survey of the Mohawk River drainage (Figure 1) in eastern upstate New York included information from 864 sites (Moore 1935; field catalogs housed at Cornell University, Ithaca, New York). The data from that survey are among the most thorough of any historical database on fish assemblages in the Northeast. Some of the data are limited, however, because often the numbers of fishes collected were not recorded. Instead, fish species abundance was estimated and classified in terms of rare, common, many, or abundant, a subjective approach that probably varied among the many workers. Specimens from most sites were preserved and are now stored at the New York State Museum (NYSM). Seines were used as the primary sampling device. The corresponding field notes include little information on sampling effort or area surveyed, but they do indicate that all available habitats within each study reach were sampled. Often, associated notes on flow, bottom type, water temperature, and other environmental attributes are included.

At the time of the 1934 Mohawk River survey, environmental change, including the presence of dams, channelization, pollution, and species introductions, had already affected the resident assemblages of native fish in the drainage. For example, slopes were deforested throughout the Northeast in the 19th century, which led to increases in water temperature and turbidity (Lynch et al. 1984). The tanning industry that developed on the banks of upper Schoharie Creek and West Canada Creek, major tributaries to the Mohawk River, was notorious for its release of pollutants into those streams (Faigenbaum 1935). The unpublished field notes from the 1934 survey of the Mohawk River drainage occasionally mention areas affected by milk pollution or pollution from other agricultural activities or large numbers of diseased fish. Exotic fishes such as brown trout *Salmo trutta*, common carp *Cyrpinus carpio*, and largemouth bass *Micropterus salmoides* had already become established in the Mohawk River drainage by the time of the 1934 survey (Greeley 1935). Despite these confounding issues, adequate samples were taken at enough sites and in sufficient detail to provide a comprehensive database broad enough to develop metrics 1 through 4, dealing with fish species richness and composition (Table 1).

The remaining metrics (5–12; Table 1) require quantitative data on fish species abundance. Therefore, these metrics and their scoring criteria were developed by using data collected more recently in a survey of the abundance and distribution of fishes in Schoharie Creek during 1982–1983 (R. A. Daniels, unpublished). In this study, fish were collected by using seines or a backpack electroshocking unit; all captured fish were identified, counted, and usually released. Voucher specimens were preserved and are housed at the NYSM. Mean stream width of the sample reach (m) was measured at the site. Substrate is represented by an index (Daniels 1987) calculated by summing the Wentworth scale rank multiplied by the proportion represented by that rank. Flow (m^3/s) was calculated from stream width, depth, and velocity measured at the site with an electromagnetic flowmeter. Elevation (m) and gradient (m/km) were estimated from U.S. Geological Survey (USGS) topographic maps. Drainage area upstream of a site (km^2) was measured from watershed maps with a compensating polar planimeter having an accuracy of 0.5 km^2. For this analysis, values were log-transformed.

Data from some sample reaches were not suitable for use in developing the IBI. Information from sites was rejected if the site was too close (1.6 km) to lakes or confluences of larger streams, canals, dams, or other obstructions, and if data were obviously incomplete (Fausch et al. 1984). Sites that included bridges were rejected, but sites entirely up- or downstream of bridges were used. Information from upland coldwater sites containing a maximum of three species and including a salmonid species was rejected (Langdon 2001), but lowland coldwater sites, which usually have more than three species including a salmonid, were used. Because variation in stream gradient could affect the scoring of species richness metrics (Leonard and Orth 1986), we examined elevation as a surrogate measure for gradient, but observed no relation between elevation and species richness for similar-sized stream sites. These screening criteria provided 79 sites from the 1934 Mohawk River survey and 61 sites from the 1982–1983 Schoharie Creek survey for use in developing metrics for this IBI.

Certain metrics require that species be classified by trophic group, habitat affiliation, or both. We follow Halliwell et al. (1999) when classifying inland freshwater and diadromous fishes found in the northeastern United States.

Scoring criteria.—Two types of data were used to develop IBI metric scoring criteria. The comprehensive stream surveys conducted in 1934 and 1982–1983 provided information on richness and composition of fish species. We used these data for the richness metrics (1–4). Stream surveys conducted during 1982–1983 were used to develop the IBI metric scoring criteria for the abundance or relative abundance metrics (5–12).

Fausch et al. (1984) presented the maximum species richness line (MSRL) concept. The MSRL is based on empirical data that suggest that an increase in species richness corresponds to an increase in stream size. We used watershed area as a measure of stream size (Miller et al. 1988) in this study because the use of stream order (Strahler 1957) may not be the best measure of relative stream size (Hughes and Omernik 1983). MSRLs compensate for variation in fish species richness

TABLE 1.—Metrics, justification for change from those of the Midwestern index of biological integrity (IBI), and scoring criteria developed for the IBI for northern mid-Atlantic streams. A criterion score of 5 is best. The acronym MSRL refers to the maximum species richness line, MDL to the maximum density line; both are described in the text.

Metric number	Metric for Midwestern IBI	Metric for mid-Atlantic streams	Environmental assessment	Justification for change
		Resident fish species richness and composition		
1	Total number of fish species	Total number of fish species	Richness decreases with degradation	
2	Number of darter species	Number of benthic-insectivorous species	Sensitive due to specific use of benthic (often riffle) habitats for reproduction and feeding	Only 6 darter species in mid-Atlantic drainages
3	Number of sunfish species	Number of water column species	Responsive to degradation of pool habitats, loss of riparian vegetation and stream cover	Only 13 sunfish (excluding black basses) species in mid-Atlantic drainages, usually no more than 3 at any site
4	Number of sucker species	Number of terete minnow species	Most are intolerant of habitat and chemical degradation and are long-lived	Only 6 sucker species in area. Terete minnow species are widely distributed; several show distinct habitat preferences and many are long-lived.
5	Number of intolerant species	Percentage of dominant species	Intolerant species respond rapidly to environmental change and disturbance	Intolerance of habitat degradation is difficult to identify. The replacement metric is a measure of evenness in the assemblage, which is also a measure of assemblage tolerance to degraded conditions.
6	Proportion of individuals that are green sunfish *Lepomis cyanellus*	Percentage of individuals that are white suckers *Catostomus commersoni*	Green sunfish increases in abundance in degraded streams	Green sunfish is not native to or widely distributed in mid-Atlantic drainages, as is white sucker.
		Trophic composition		
7	Proportion of individuals that are omnivores	Percentage of individuals that are generalists	Omnivores (Schlosser 1982) become dominant when certain components of the food base become less reliable	Generalists (Halliwell et al. 1999) become dominant under similar circumstances.
8	Proportion of individuals that are insectivorous cyprinids	Percentage of individuals that are insectivores	Relative abundance of insectivores decreases with degradation in response to availability of the insect supply, which reflects alterations of water quality and instream habitat	Including all insectivores increases the number of species counted in this metric.
9	Proportion of individuals that are top carnivores	Percentage of individuals that are top carnivores	Presence of top carnivores is indicative of a diverse and healthy community	
		Fish abundance and condition		
10	Fish per sample	Fish per sample	Usually, degraded sites yield fewer individuals than healthy sites	
11	Proportion of individuals that are hybrids	Percentage of species represented by two size-classes	Assesses the effect of habitat degradation on reproductive isolation	Hybrid fishes are rare. Counts of species represented by juveniles and adults assess recruitment, not reproductive isolation and constitute a valid substitute metric.
12	Proportion of individuals with disease, tumors, fin damage, or skeletal anomalies	Percentage of individuals with disease, tumors, fin damage, or other anomalies	Fish condition is related to habitat degradation	

TABLE 1.—Extended.

Metric number	Comments	Scoring 5	Scoring 3	Scoring 1
	Resident fish species richness and composition			
1	This metric should be modified to exclude from the count young-of-year, exotic, stocked, transient, and lentic species.		Use of MSRL	
2	This group includes darters *Etheostoma* and *Percina* spp., certain daces *Rhinichthys* spp. and other minnows, madtoms *Noturus* spp., and sculpins *Cottus* spp.		Use of MSRL	
3	These are deep-bodied forms such as sunfishes *Lepomis* and *Enneacanthus* spp., certain minnows *Notemigonus, Clinostomus, Luxilus,* and *Cyprinella* spp., suckers *Erimyzon* spp., and yellow perch *Perca flavescens*, along with surface-oriented fishes *Fundulus* and *Labidesthes* spp. Black basses excluded.		Use of MSRL	
4	Species include *Campostoma, Couesius, Exoglossum, Hybognathus, Margariscus, Nocomis, Notropis, Pimephales, Phoxinus,* and *Semotilus.*		Use of MSRL	
5	In the Northeast, native fishes are postglacial migrants that can tolerate a wide range of environmental conditions	<40%	40–55%	>55%
6	White suckers can dominate degraded sites, although their mere presence is not indicative of degradation.	<3%	3–15%	>15%
	Trophic composition			
7	Generalist feeders consume a variety of foods from a variety of habitats. Omnivores are species whose diets contain at least 25% plant and 25% animal material. All omnivores are generalists but not vice versa.	<20%	20–45%	>45%
8		>50%	25–50%	<25%
9	Metric counts include only individuals that function as top carnivores, excluding, for example, juvenile black bass.	>5%	1–5%	<1%
	Fish abundance and condition			
10			Use of MDL	
11	During the 1934 sampling of the Mohawk River drainage, 20,516 specimens in 2,086 lots from 483 sites were collected; of these, only 23 individuals in 18 lots from 17 sites were identified as hybrids. Presence of hybrids per se is not an indication of environmental degradation.	>40%	15–40%	<15%
12		0%	0–1%	>1%

related to region and stream size (Fausch et al. 1984). Because the first four metrics evaluate species richness, we developed MSRLs for total number of fish, number of benthic insectivores, number of water column species, and number of species that are terete minnows (Figure 2), based on data from Mohawk River drainage streams. Scoring criteria for the remaining two species richness and composition metrics were based on percentage composition because species richness versus watershed area relationship was not detected.

Occasionally, the computed index for sites may fall either on or very near the lines that separate criteria score regions in MSRL-type graphs. This makes the scoring of a particular species richness and composition metric for these sites problematic. One solution is to assign sites that fall on the lines a value of either 4 or 2, which compensates for the variability associated with drafting the graph. For example, the thickness of the line and the accuracy of point placement will affect the final score. We used this method for metrics in cases where scoring was dependent on the MSRL.

Karr (1981) and Fausch et al. (1984) did not suggest guidelines for establishing criteria for the number of individuals per sample (metric 10), other than to state that relative criteria were used after conversion to catch per unit effort. In our study, plotting the number of individuals per 100 m versus watershed area showed no obvious relationship; therefore, we replaced this metric with a density measure (number of individuals per 100 m^2; see Miller et al. 1988). Fish density has been shown to decrease as watershed area (stream size) increases (Thompson and Hunt 1930; Hallam 1958; Larimore and Smith 1963). This density decrease may be wholly or partly a result of decreasing fish-capture efficiencies in large, deep streams. The line that forms the upper boundary for about 95% of the sites (Figure 3) is the maximum density line (MDL; Miller et al. 1988). The MDL is drawn with slope fit by eye (Fausch et al. 1984), and its intersection with the abscissa provides the focal point for trisecting the ordinate into criteria-score regions similar to those of the MSRL. Originally, we calculated the upper 90% prediction band about each regression line (for Figure 3: $y = 1.95 - 0.42x$; $r = -0.57$, $P < 0.01$) as the MDL. Because of annual variability in fish population size, use of prediction bands about the sample are more appropriate than confidence bands about the mean, and 90% in this case approximated the upper limit of the data. We favor the simpler technique for determining the MDL, however, because it cor-

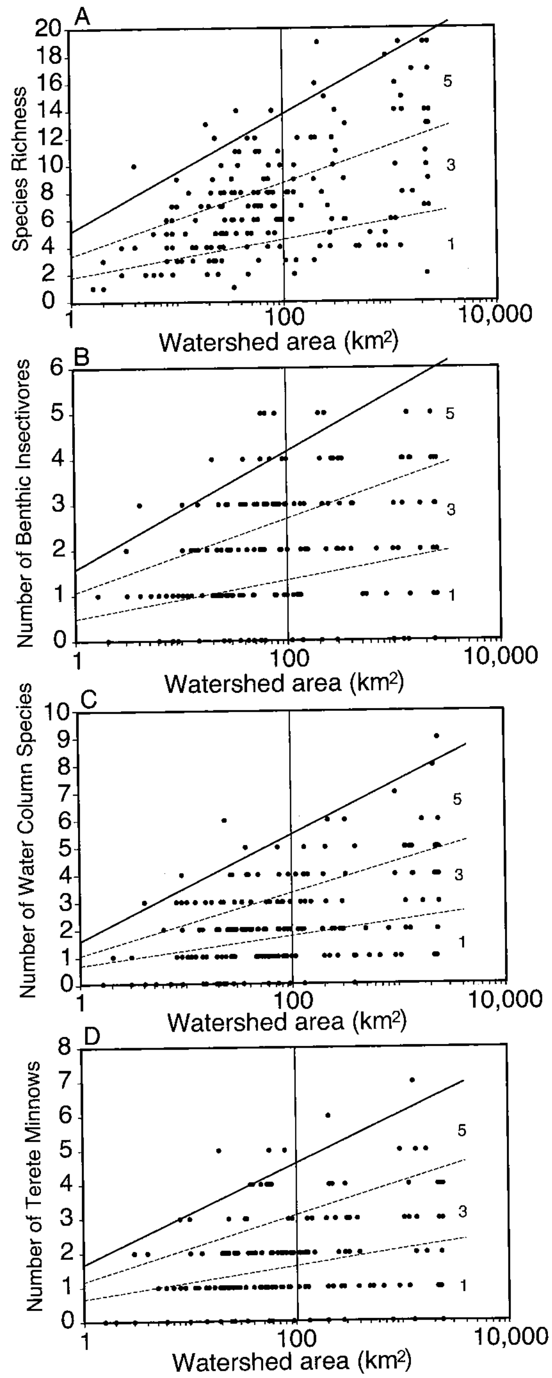

FIGURE 2.—(A) Species, (B) benthic insectivore, (C) water column species, and (D) terete minnow species richness from the Mohawk River drainage sites, showing maximum species richness lines and associated scoring criteria.

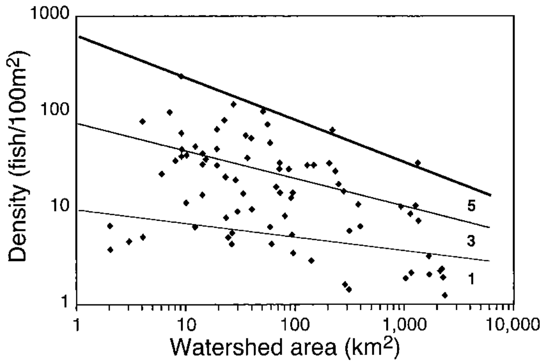

FIGURE 3.—Fish density from the Mohawk River drainage sites, showing maximum density line and associated scoring criteria.

responds to the way in which MSRLs are defined and differs little from the upper 90% prediction band. Specification of the MDL requires that sample-reach area be recorded in addition to fish abundance. The extra time required to obtain this measurement is compensated for by increased objectivity in scoring this metric.

The IBI is designed as a robust index that can be modified to suit individual projects in different regions. The relation between fish density and drainage area differs among regions and watersheds (Hynes 1970) and therefore needs to be evaluated for each watershed. Differences in fish-sampling effort also may warrant adjustments in the scoring criteria. In addition, regional differences in the number of fish species among and within river basins may require adjustments of MSRL scoring criteria from those that have been developed here on the basis of the Mohawk River Basin streams.

Validation of the northern Mid-Atlantic Slope drainage index of biological integrity.—To assess the validity of the northern Mid-Atlantic Slope IBI, we calculated the index scores for 27 sites in the Hudson River basin (HDSN), the lower Susquehanna River basin (LSUS), and the Delaware River basin (DELR; Figure 1; Table 2), and examined the relation between IBI scores and other biological, chemical, and physical indicators of environmental quality. In addition to information on the fish assemblage at these sites, we had access to information on the benthic macroinvertebrate assemblage, water chemistry, and physical indicators of water and habitat quality, as well as watershed characteristics such as land use, human population density, and watershed size. These data were collected during 1993–2000 as part of the USGS National Water Quality Assessment Program (NAWQA; Leahy et al. 1990; Gurtz 1994). If the fish-based IBI adequately represents biological integrity, we expect to see concordance with assessments based on other biological, chemical, and physical characteristics (e.g., Allan et al. 1999a, 1999b).

All fish sampling sites were on wadable streams draining watersheds of 19–1,903 km^2 and were sampled during summer low-flow conditions. Sites were selected to represent urban, agricultural, and undisturbed land uses within the study areas. We eliminated sites proximal to lakes, reservoirs, or confluences with larger streams. We excluded a headwater site that had only three fish species and two sites in the Pine Barrens ecoregion of New Jersey that were low-gradient coastal plain streams. Fish were sampled from representative reach lengths of 150–320 m and were collected in 1 or 2 passes by electrofishing with pulsed direct current in an upstream direction. Fishing gear used (e.g., backpack or tote barge unit; several backpack electrofishers) and crew size configuration were adjusted for flow and water depth conditions at each site. Fish specimens were identified and counted, and a subsample of 30 or more individuals per species was measured, weighed, and examined for anomalies in the field. Some individuals were preserved for laboratory identification and vouchers for all fish species were retained. Further details on protocols for sampling fish assemblages are presented in Meador et al. (1993a).

In computing the IBI, we did not include the catadromous American eel *Anguilla rostrata* in the fish-assemblage calculations because this species was ubiquitous and did not reflect water or habitat quality. Brown trout and rainbow trout *Oncorhynchus mykiss* were not included if they showed signs of hatchery origin (e.g., frayed, deformed, or clipped fins, lack of coloration, poor overall condition). We used data from only the first electrofishing pass because not every site had two passes.

Benthic macroinvertebrates were collected from riffle habitats according to Cuffney et al. (1993a). A modified Slack sampler equipped with 425-μm mesh netting was held in place while benthos were scrubbed from rocks within a 0.25-m^2 section directly in front of the net. After all rocks were cleaned, finer substrate was disturbed by stirring with a metal rod to a depth of 10 cm and then the area was kicked for 30 s. Each sample was a composite of five collections per reach. Samples were processed at the USGS National Water Quality Laboratory according to Cuffney et al. (1993b) and

TABLE 2.—List of sites from U.S. Geological Survey (USGS) National Water Quality Assessment Program studies in the Hudson, Delaware, and Susquehanna River basins, from which data were obtained to test the mid-Atlantic index of biological integrity, 1993–2000. Site locations are shown in Figure 1.

Site number	USGS station number	Locality	Basin area (km^2)	Population density (per km^2)	Elevation (m)	Forest (% basin area)	Dominant land use
		Hudson River basin					
1	01333500	Little Hoosic River at Petersburg, New York	141	13	179	92	Agricultural
2	01334500	Hoosic River near Eagle Bridge, New York	1,324	56	108	71	Mixed
3	01349150	Canajoharie Creek near Canajoharie, New York	155	18	104	38	Agricultural
4	01351200	Fox Creek near Schoharie, New York	256	19	195	55	Agricultural
5	01356190	Lisha Kill northwest of Niskayuna, New York	40	525	76	34	Urban
6	01359135	Patroon Creek at Albany, New York	37	965	12	17	Urban
7	01361200	Claverack Creek at Claverack, New York	142	37	43	71	Agricultural
8	0136216850	Roeliff Jansen Kill at Jackson Corners, New York	440	19	90	69	Agricultural
9	01362200	Esopus Creek at Allaben, New York	165	25	304	99	Forested
10	01371500	Wallkill River at Gardiner, New York	1,903	105	57	36	Mixed
11	01372051	Fall Kill at Poughkeepsie, New York	49	413	15	65	Urban
12	01372200	Wappinger Creek near Clinton Corners, New York	233	30	71	74	Agricultural
13	01373690	Woodbury Creek near Highland Mills, New York	29	168	180	78	Urban
14	01374494	Haviland Hollow Brook near Putnam Lake, New York	32	70	131	97	Urban
15	01376500	Saw Mill River at Yonkers, New York	62	1,055	37	38	Urban
		Delaware River basin					
16	01440000	Flat Brook near Flatbrookville, New Jersey	168	14	106	88	Forested
17	01451800	Jordan Creek near Schnecksville, Pennsylvania	136	63	122	33	Agricultural
18	01464907	Little Neshaminy Creek near Neshaminy, Pennsylvania	72	393	62	36	Urban
19	01470779	Tulpehocken Creek near Bernville, Pennsylvania	185	110	93	13	Agricultural
20	01472157	French Creek near Phoenixville, Pennsylvania	152	61	51	63	Agricultural
		Lower Susquehanna River basin					
21	01555400	East Mahantango Creek at Klingerstown, Pennsylvania	116	971	158	43	Agricultural
22	01559795	Bobs Creek near Pavia, Pennsylvania	43	11	183	91	Forested
23	01564997	Kishacoquillas Creek at Lumber City, Pennsylvania	149	111	189	30	Agricultural
24	01571490	Cedar Run at Eberlys Mill, Pennsylvania	33	1,213	107	37	Urban
25	01573095	Bachman Run at Annville, Pennsylvania	19	58	120	18	Agricultural
26	01576540	Mill Creek near Lyndon, Pennsylvania	140	289	76	11	Agricultural
27	01577300	Muddy Creek at Muddy Creek Forks, Pennsylvania	186	268	107	30	Agricultural

Moulton et al. (2000). These data were used to calculate total taxa richness; the number of Ephemeroptera, Plecoptera, and Trichoptera taxa; the percentage of taxa that were not insects; and the percentage represented by the three numerically dominant taxa.

Stream habitat data were collected according to Meador et al. (1993b) and Fitzpatrick et al. (1998) during summer low-flow conditions. Measurements and visual estimates were made at either 6 or 11 transects along the stream reach. Habitat variables included wetted channel width and depth, canopy-closure angle, bank-full dimensions, bank vegetative cover, and presence or absence of erosion. A habitat quality index was calculated on the basis of the following characterisics: (1) percentage of bank observations with erosion, (2) an index of bank stability (based on bank height, angle, substrate, and percent vegetative cover, as described in Fitzpatrick et al. 1998), (3) estimates of riparian tree density, and (4) degree of channelization or other habitat alteration. The index is described further in Gilliom et al. (1998). Stream slope was determined either on site with a hand-held or surveyor's level or from 1:24,000 scale USGS maps.

Water column samples were collected by a depth-integrated, equal-width method and were analyzed for major and minor ions, nutrients, pH, total alkalinity, and sediment at the USGS National Water Quality Laboratory (Shelton 1994). Field measurements of specific conductance and temperature were made at the time of chemical and (or) fish sampling. Chemical data from a single base flow sampling event were used for these analyses; most were August samples. Stream discharge was determined at or near base flow (usually in August) during the fish-sampling year from automated gauging stations or handheld flowmeters.

The dominant environmental gradients were identified by principal components analysis (PCA) of chemical and physical variables. Principal components analysis integrates most of the variation associated with many input variables and creates a smaller number of synthetic variables called eigenvectors or PCA axes. Multiple axes that explain a large proportion of the variation can be interpreted in terms of an underlying environmental gradient. The relation between IBI and environmental quality was then examined by Spearman rank correlation analysis of the PCA axes. Before performing PCA, variables were log-transformed, square-root-transformed, or arcsine-transformed to approximate a normal distribution. Variables for PCA were selected to minimize highly correlated pairs of variables, variables with skewed or otherwise nonnormal distributions even after transformation, variables with many values missing, and those likely to exhibit high diel variation. We used DELR fish, macroinvertebrate, chemical, and habitat data collected during 2000; LSUS data collected during 1993 and 1994; and HDSN data collected from 1993 and 1994 (seven sites), 1993 only (one site), or 1993 (chemistry and benthic macroinvertebrate data) and 1995 (fish data, seven sites). Mean values were used for sample sites with multiyear HDSN and LSUS data.

Certain adjustments may be warranted in scoring criteria for the NAWQA data set because of differences in fish faunal composition between HDSN (including the Mohawk River basin used to develop scoring criteria) and the other two river basins and differences between NAWQA sampling methods and those used to collect the data utilized in developing the scoring criteria. Thus, we scored the IBI both with and without any adjustment in metric scoring criteria and compared the outcomes. Adjustments were made in metric scoring criteria for (1) total species richness of resident fish, (2) terete minnow species richness, (3) percentage of taxa with both juveniles and adults found, and (4) percentage of individuals with anomalies. These adjustments were done to account for (1) a substantially richer fish fauna in LSUS and DELR than in HDSN, (2) larger potential terete minnow species richness in LSUS than in the other two drainages, and (3) a more intensive and extensive effort in the NAWQA sample and data-collection methods than in those used to collect the data used to develop the scoring criteria. Rescaling of the MSRLs was done by selecting the sites in the top 25th percentile of environmental quality, as determined by PCA results. Of these sites, we then selected the site with the greatest richness values for LSUS and DELR and used those values as the basis for rescaling the y-axis of the MSRLs. We also adjusted the anomalies and juvenile/adult metrics by setting the cutoff values to correspond to approximately the 25th and 75th percentile levels from the NAWQA data set.

Results

The IBI for northern Mid-Atlantic Slope drainages differs in several ways from the IBI developed for the Midwest (Karr 1981; Fausch et al. 1984) and for the Northeast and for New England (Miller et al. 1988; Halliwell et al. 1999). Eight metrics in the Hudson, Delaware, and Susquehanna IBI differ from those in the Midwestern IBI because the fish fauna in the former is less rich and the assemblage structure less complex and of more recent origin. Four metrics differ from those in the northeastern IBI developed by Miller et al. (1988). Although the Hudson, Delaware, and Susquehanna River drainages differ slightly in species richness and composition, similarity among the fish faunas is great (Daniels 1993). Thus, a single IBI can be used in these drainages despite slight drainage-specific differences in species composition and assemblage structure. Table 1 lists the metrics developed for the northern Mid-Atlantic Slope drainages. Included are explanations of what the metric measures, a justification for any change in the metric from the Midwestern IBI, and scoring criteria. Although most metrics are explained fully in Table 1, some require additional explanation.

The first metric, a measure of site-specific species richness, is the same as the metric used in the Midwestern IBI (Karr et al. 1986). We suggest that several types of species can be excluded from the count (Table 1; Scott and Helfman 2001). The rationale for excluding these groups is that they are temporary components to the assemblage and not ecologically significant in the assemblage structure. For example, fishes such as bluegills and yellow perch, which are typical of stocked upland ponds, may be washed into the stream or may use the stream temporarily for dispersal. In developing this IBI, we included all long-term, resident species, whether they were native or naturalized. We did not include transient or stocked species.

Metric 5, the percentage of dominant fish species, represents a major departure from its counterpart in the Midwestern IBI, the proportion of intolerant fish species. Intolerant species, those that are sensitive to habitat degradation (Karr et al. 1986), would appear to be an ideal group for assessing water quality. Intolerance of habitat degradation is difficult to identify, however, particularly in the Northeast, where native fishes, as post-

TABLE 3.—Correlation coefficients of the $\log_e$- or arcsine-transformed data used to develop the 12 metrics of the mid-Atlantic drainages. Coefficients in bold are significantly correlated at $P < 0.05$; $n = 61$.

Metric	Richness	Benthic insectivores	Water column species	Terete minnows	Dominant species	White suckers	Generalists
1. Total number of fish species		**0.55**	**0.72**	**0.76**	**−0.59**	**0.23**	−0.05
2. Number of benthic-insectivorous species			0.08	**0.54**	**−0.37**	**0.23**	0.16
3. Number of water column species				**0.25**	**−0.35**	−0.10	**−0.26**
4. Number of terete minnow species					**−0.41**	0.16	0.16
5. Percentage of dominant species						−0.07	0.16
6. Percentage of individuals that are white suckers							0.19
7. Percentage of individuals that are generalists							
8. Percentage of individuals that are insectivores							
9. Percentage of individuals that are top carnivores							
10. Fish per sample							
11. Percentage of species represented by 2 size-classes							
12. Percentage of individuals with anomalies							

glacial migrants, are present there partly because they can tolerate a wide range of environmental conditions. Roth et al. (2000) evaluated two forms of a species richness metric for intolerant fish—one ranked species on the basis of information gleaned from the literature, whereas the second ranked species based on their presence or absence at degraded or minimally disturbed sites. Roth et al. (2000) rejected the use of the literature-based ranking but found the data-based ranking acceptable. In the absence of better information on the tolerance ranges of fishes, particularly minnows (see Whittier and Hughes 1998), we recommend the use of percentage dominant species in the sample as a replacement metric. Roth et al. (2000) evaluated this metric as well and found it satisfactory. This replacement metric is a measure of evenness in the assemblage, rather than richness, and is a measure of tolerance of degraded conditions.

We also considered using the number of exotic or introduced species as a replacement metric; it could be a strong metric and clearly is appropriate as an index that measures the naturalness of a site (Angermeier and Karr 1986). The simple presence of exotic species, by definition, indicates that an assemblage is compromised. A problem with using this metric, however, is that exotic or introduced species, as relatively recent introductions to the fish assemblages, tend to display wide tolerance ranges for many habitat variables, even if the populations are naturalized (Halliwell et al. 1999). Nonetheless, the presence of numerous exotic species has been associated with habitat degradation in other areas (Moyle et al. 1982). The designation of exotic or introduced fish species, inclusive of transplanted or transient native species (Halliwell et al. 1999), between and within river drainage systems can be a difficult task. However, if the number of exotic fish species metric is to be used effectively, counts should reflect accurate knowledge of the status of all species in the individual drainages (Halliwell et al. 1999). Such classifications are primarily based on the availability of documented fish stocking/transfer records but ultimately will be based on best professional judgment. Problems may arise when there are differences of opinion in the scientific community. For example, Schmidt (1986) regarded emerald shiner *Notropis atherinoides* in the Hudson River system as native, whereas Mills et al. (1997) listed the species as introduced. In the final analysis, care should be taken to prevent the "mismeasure of fish assemblage integrity" (Scott and Helfman 2001) through inclusion of nonnative or nonresident fish species and artificial inflation of metrics of fish species richness (Halliwell et al. 1999).

Assessment of the Mohawk River Index of Biological Integrity

The IBI metrics derived from the Mohawk River surveys are robust, deal with the major aspects of the integrity of the fish assemblage, and follow the criteria that were used successfully by Karr et al. (1986) in developing the Midwestern IBI. Ideally, correlation among metrics should be low to minimize redundancy. For each variable, we used the log-transformed count or measurement or arcsine-transformed relative abundance measures from each site to produce a 12×12 correlation coefficient matrix (Table 3). The values used in the fish richness metrics (1–4) and sample density metric (10) tended to be significantly positively correlated with each other but negatively correlated with the

TABLE 3.—Extended.

Metric	Metric				
	Insectivores	Top carnivores	Density	Size-classes	Anomalies
1. Total number of fish species	−0.04	**−0.33**	**0.59**	0.14	0.16
2. Number of benthic-insectivorous species	**−0.33**	−0.16	**0.45**	0.07	**0.23**
3. Number of water column species	**0.45**	**−0.45**	0.18	0.09	−0.04
4. Number of terete minnow species	−0.02	0.18	−0.14	−0.01	0.05
5. Percentage of dominant species	−0.02	0.18	−0.14	−0.01	0.05
6. Percentage of individuals that are white suckers	−0.15	−0.08	**0.28**	−0.08	0.15
7. Percentage of individuals that are generalists	**−0.69**	**−0.39**	**0.25**	0.02	0.18
8. Percentage of individuals that are insectivores		−0.18	**−0.22**	−0.14	−0.14
9. Percentage of individuals that are top carnivores			**−0.36**	0.01	−0.05
10. Fish per sample				**0.23**	0.16
11. Percentage of species represented by 2 size-classes					0.08
12. Percentage of individuals with anomalies					

percentage of the dominant species (5). The percentages of species with juvenile and adult size classes (11) and of individuals with anomalies (12) were not correlated with the other measures used as metrics. Realistically, development and selection of metrics that are not correlated with each other is difficult, so some redundancy is inherent in IBI development. Despite the significant correlations between certain pairs, each metric individually contributes information to the IBI score not found in the other metrics (see Fore et al. 1994).

IBI scores were not significantly correlated with most environmental variables associated with natural variation among sites, indicating that scores do not reflect gross environmental characteristics. We examined correlations between scores and six environmental measurements from each site. Scores were positively correlated with drainage area ($r = 0.30$, $P < 0.005$) and stream width ($r = 0.28$, $P < 0.05$) and were negatively correlated with elevation ($r = -0.24$, $P < 0.10$) and gradient ($r = -0.27$, $P < 0.05$). Scores were not significantly correlated with flow or with substrate composition ($P > 0.10$). In general, these results indicate that higher scores tend to be found at downstream sites. However, weak correlations, and the lack of significance in the correlation between scores and some environmental variables, indicate that the scores are not merely a reflection of natural physical variation among sites.

Validation of the Index of Biological Integrity for Northern Mid-Atlantic Slope Drainages

Index of biological integrity scores ranged from 14 for an urban–industrial HDSN site to 52 for an agricultural HDSN site; the median IBI score was 42. Scores yielded IBI classifications (following Karr et al. 1986) of "very poor" to "good," with the median site classified as "fair." None of the NAWQA sites was classified as "excellent" during any year. This is not surprising, given the extent of human activity in the watersheds studied. There was no statistically significant difference in IBI score among study units or ecoregions (Tukey's studentized range test on ranked scores, $P > 0.05$). Fish assemblage data are reported in Firda et al. (1993, 1994, 1995), DeLuca et al. (1999, 2000), and Bilger and Brightbill (1998).

Environmental gradients based on chemical, physical, and macroinvertebrate variables.—Water quality data are reported in Firda et al. (1993, 1994, 1995), DeLuca et al. (1999, 2000), and Durlin and Schaffstall (1993, 1994, 1995). Use of PCA of selected chemical and physical variables resulted in three eigenvectors that, together, explained 61% of the variation among sites. Eigenvalues of axes I, II, and III were 0.41, 0.31, and 0.20, respectively (Table 4). The first eigenvector (PC1) can be interpreted as a water and habitat quality gradient, according to correlations (loadings) between original variables and eigenvector site scores (Table 4). Spearman rank correlations between site scores on PC1 and other biological, environmental, and watershed variables support this water- and habitat quality interpretation. Site IBI scores were positively correlated with percentage of urban land in the watershed (0.75, $P < 0.0001$), human population density (Spearman $R = 0.55$, $P < 0.01$), percentage of macroinvertebrate taxa richness as noninsect taxa (Spearman $R = 0.64$, $P < 0.001$), and percentage of macroinvertebrate abundance composed of the three most dominant taxa (Spearman $R = 0.67$, $P = 0.0001$).

TABLE 4.—Correlations (loadings) of environmental variables with site scores on principal components axes I–III from an analysis of environmental variables collected from 27 sites during summer low-flow conditions. Eigenvalues are given in parentheses. Correlations of 0.30 or more are shown in bold.

Variable	Transformation	Axis I (0.41)	Axis II (0.31)	Axis III (0.20)
Alkalinity	None	**0.36**	0.01	−0.04
Chloride	Log_e	**0.40**	0.15	−0.28
Discharge	Log_e	−0.03	**0.35**	**0.41**
Elevation	Square root	−0.29	−0.24	0.03
Erosion (%)	None	0.21	−0.28	**0.32**
Habitat quality index	None	**−0.39**	−0.01	−0.18
Nitrate + nitrite	Log_e	0.18	0.04	**0.35**
Open canopy angle	Log_e	−0.24	0.10	**0.31**
Organic carbon, dissolved	Log_e	0.04	**0.36**	−0.01
Organic carbon, suspended	Log_e	**0.30**	0.26	0.29
Percent reach as run	None	−0.13	**0.41**	−0.23
Percent reach as riffle	None	0.16	**−0.33**	0.20
Specific conductance	Log_e	**0.41**	0.11	−0.17
Water temperature	None	−0.09	**0.38**	−0.16
Wetted width	Log_e	−0.18	0.28	**0.41**

Site scores were negatively correlated with percentage of watershed with forest cover (Spearman $R = -0.42$, $P < 0.05$), benthic macroinvertebrate taxa richness (Spearman $R = -0.58$, $P < 0.01$), and percentage of taxa as Ephemeroptera, Plecoptera, and Trichoptera taxa (Spearman $R = -0.66$ $P < 0.001$). PC1 was negatively correlated with elevation (Spearman $R = -0.55$, $P < 0.01$) and drainage area (Spearman $R = -0.43$, $P < 0.05$) but was not significantly correlated with stream gradient. The second and third eigenvectors are associated with factors other than environmental quality. PC2 represents a gradient of natural features associated with elevation, stream channel morphology, and temperature (Table 4). The percentage of benthic macroinvertebrate taxa as noninsects was the only macroinvertebrate metric having a significant correlation with site scores on PC2 (Spearman $R = 0.46$, $P < 0.05$), suggesting the absence of a clear water quality gradient. The third eigenvector may be associated with degree of agricultural influences versus urban influences (Table 4). Site scores were weakly positively correlated with percentage of watershed occupied by agricultural land (Spearman $R = 0.41$, $P < 0.05$) and percentage of macroinvertebrate taxa made up Ephemeroptera, Plecoptera, and Trichoptera taxa (Spearman $R = 0.40$, $P < 0.05$) and negatively correlated with gradient (Spearman $R = -0.44$, $P < 0.05$). These PCA results indicate that the most important source of variation among sites is that of environmental quality associated with human influences. Thus, we determined that PC1 site scores could function as a reasonable surrogate for a gradient of water quality and habitat quality conditions with which to compare our IBI results.

Index of biological integrity results after rescaling metric scoring criteria.—Changes in mean IBI score, after rescaling two of the fish species richness MSRLs and two of the relative abundance metric scoring criteria, were relatively small. Median change was 2 points (minimum 0, maximum 4). Mean scores ranged from 14 to 52, with a median score of 41. Adjustment scoring criteria resulted in no change in IBI classification for 19 of the 27 sites, a half-class change (i.e., "poor" to "poor–fair") for seven sites, and a complete class change for a single site.

Correspondence between index of biological integrity and the environmental quality gradient.—Index of biological integrity scores were significantly negatively correlated (Spearman $R = -0.78$, $P < 0.0001$) with PC1 (Figure 4a), indicating that the index successfully captures the fish assemblage's response to water and habitat quality for the sites and range of conditions considered in this study. Index of biological integrity scores were not significantly correlated (at $P < 0.05$) with either PC2 or PC3, which were inferred to represent gradients not associated with water or habitat quality. The small change in IBI score for some sites after adjustment of metric scoring criteria did not markedly change the relation between IBI score and PC1 score (Figure 4b; Spearman $R = -0.75$, $P < 0.0001$).

Discussion

The IBI metrics developed and the scoring criteria selected were successful in assessing water and habitat quality for northern Mid-Atlantic Slope drainages on the basis of fish assemblage data. The three NAWQA data sets used to test the

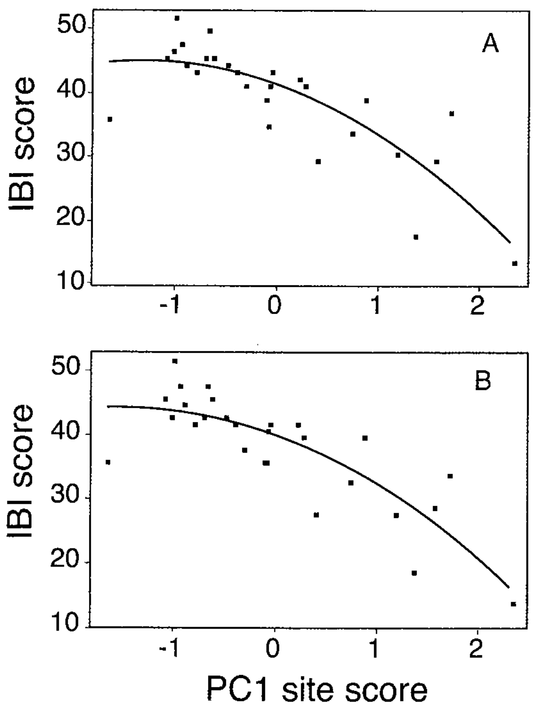

FIGURE 4.—IBI scores in relation to decreasing environmental quality, as represented by site scores on axis 1 (PC1) from a principal components analysis of chemical and physical variables. Panel (**A**) shows the IBI calculated without adjustment of metric scoring criteria, panel (**B**) the IBI calculated after adjustment of scoring criteria for selected metrics.

northern Mid-Atlantic Slope IBI were collected with the same overall study design and protocols. However, team leaders, crew composition, and conditions varied among the river basins and, in some cases, at sites within river basins. The northern Mid-Atlantic Slope IBI appears to be robust enough to withstand these potential sources of variation. An effective multi-metric biological/environmental assessment index should be robust, easy to use, and based on accurate (although not precise) data, and the results should be capable of duplication. This northern Mid-Atlantic Slope IBI appears to meet these criteria fully. However, we do not propose that this IBI be rigidly adopted and used without modification. Rather, we recommend that individual investigators consider modifying the metrics and scoring criteria as required by the natural features of the aquatic system under study, the objectives of the study, and the resources available to conduct the study.

Recent work (e.g., Lyons et al. 1996; Simon 1999; Angermeier et al. 2000; Roth et al. 2000; Langdon 2001; McCormick et al. 2001) has provided ample evidence that no one set of metrics or scoring criteria is appropriate for all situations. This is particularly true in the northeastern United States (see examples in Halliwell et al. 1999).

Scoring criteria can be altered to meet the needs of modified IBIs. For example, all individuals of fish species identified as top carnivores (metric 9) in the sample are counted, as recommended by Karr et al. (1986). In some cases, however, counting only the functional top carnivores and classifying juveniles and young of year of these species as functional insectivores may be more reasonable. Percentages delimiting IBI scores may also need careful assessment. Both the percentage of dominant species (metric 5) and the percentage of species represented by juvenile and adult size classes (metric 11) are heavily dependent on the amount of effort put into sampling. The scoring criteria should be adjusted as needed.

Investigators also may choose to broaden or narrow the range of characteristics used in scoring any given metric. Metric 12 in the proposed IBI represents the count of all fish with anomalies, but workers may choose to ignore certain conditions in certain studies. For example, we did not include blackspot (Trematoda), leeches (Hirudinea), and other external parasites such as anchorworm *Lernaea* spp. in the NAWQA calculations. Research protocols, metric definitions, and scoring criteria should be appropriately developed, on a case by case basis and take into account particular study goals and objectives.

The IBI for northern Mid-Atlantic Slope drainages uses 12 metrics similar to those of the Midwestern IBI (Karr 1981). Each metric represents a different range in primary sensitivity to changes in stream biological integrity. Angermeier and Karr (1986) showed that the amount of information conveyed by a particular metric varied among data sets from Illinois, Ohio, and West Virginia, all differing in spatial scale. Therefore, inclusion of all 12 metrics should increase the precision of an assessment of biotic integrity. This is a major advantage of the IBI over other single- or multiple-metric bioassessment indices.

Comparisons with benthic bioassessment, chemical and physical data, and watershed variables suggest that the northern Mid-Altantic Slope IBI proposed here is sensitive to measurable environmental degradation. We did not find large changes in IBI results or IBI classification after final adjustment of selected scoring criteria. However, even small changes in classification may be important, depending on how the information is to be used. Thus, development of IBI scoring criteria for particular geographic regions, research protocols, and study objectives may require the collection of fish assemblage data from a suitable number of "reference" or "minimally–impaired" sites.

The IBI developed here for the Hudson, Delaware, and Susquehanna River drainages appears to have broad applicability and could perhaps be directly used elsewhere in the Northeast, such as in the Saint Lawrence River system and the Connecticut River drainages. However, most smaller, interior New England river drainages and small coastal drainages in New Jersey and on Long Island would be better served by IBIs developed specifically for these areas (Halliwell et al. 1999; Chang et al. 2000; Langdon 2001). In any case, IBI metrics may need to be further modified, new metrics developed, or scoring criteria may need to be changed as appropriate for observed differences in regional northeastern U.S. fish faunal assemblages.

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