Index of Biotic Integrity Applied to a Flow-regulated River System

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Abstract: We developed a modified index of biotic integrity (IBI) for the Tallapoosa River system based on small-bodied fishes. The modified IBI comprised 9 metrics in 4 categories: (1) species richness and composition, (2) indicator species, (3) trophic function, and (4) abundance. We used distribution records and collection data from 1990–1995 to derive expected values for metrics. The IBI was most sensitive to changes in percentage of insectivorous cyprinids, percentage of intolerant species, fish abundance, and number of darter species, and least sensitive to total species richness. IBI scores generally were lower at sites experiencing more severe flow fluctuations as a result of hydropeaking dam operation. We recommend that the IBI be further tested, refined, and used as part of long-term monitoring programs in regulated southeastern river systems.

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The IBI offers resource managers an ecologically based method for assessing the health of aquatic ecosystems. The original IBI developed for midwestern streams (Karr 1981, Karr et al. 1986) consisted of 12 fish community parameters, or metrics, divided into categories of species richness, trophic structure, and fish abundance and condition. The 12 metrics were selected to evaluate different aspects of the health of stream ecosystems, and were therefore used to reflect changes in community structure

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or function that might not be assessed by measures of water chemistry or contaminant levels alone. The IBI provided a tool for quantifying changes in ecosystem health as a result of habitat degradation or flow alteration, in addition to chronically poor chemical water quality (Karr and Dudley 1981).

The IBI was first used to assess the biological quality of streams in the central United States (e.g., Berkman et al. 1986, Osborne et al. 1992). As the IBI gained popularity, new versions were developed for use in other regions throughout the United States (reviewed by Miller et al. 1988, Simon and Lyons 1995), France (Oberdorff and Hughes 1992), and Canada (Steedman 1988). Versions of the IBI developed for the central United States retained most of the metrics used in the original IBI with minor modifications. However, versions developed for the eastern and western United States and for lakes and estuaries used substantially different metrics that reflected faunal differences among systems (Simon and Lyons 1995). Simon and Lyons (1995) provided a review of changes in IBI metrics for use in different regions and noted the need for development of IBIs for the species-rich streams of the Southeast.

The Tallapoosa River, a major tributary to the Alabama River system, has been extensively altered for production of electricity, flood control, water supply, and recreation. Natural flow patterns exist in only about 25% (93 km) of the uppermost Piedmont portion of the river, in West-central Georgia and East Alabama. Harris Dam, located near the town of Wedowee, Alabama, is the most upstream impoundment on the Tallapoosa River. Peaking hydropower production at Harris Dam (with hourly discharges that vary from leakage flow, near 0, to >200 m^3/sec during generation near capacity) alters the flow regime in the 80-km reach of the river between the dam and the backwaters of Lake Martin, the next downstream impoundment. Thurlow Dam is the downstream-most impoundment on the Tallapoosa River and regulates flow in the lower 75-km Coastal Plain section of the river, to its confluence with the Coosa River near Montgomery, Alabama. Although a minimum flow requirement at Thurlow Dam of 34 m^3/sec (mandated by the Federal Energy Regulatory Commission in 1991) reduces the magnitude of hourly discharge fluctuations, daily peak flows are typically about 660% (225 m^3/sec) of the minimum flow.

The physical and biological diversity of the Tallapoosa River system presents an opportunity to evaluate the sensitivity of an IBI to faunal community changes corresponding to various degrees of streamflow alteration. At least 125 species of fishes historically occurred in the Tallapoosa River (Pierson et al. 1986, Swift et al. 1986), including at least 3 endemic species. The river continues to support a significant portion of the native fish fauna in the unregulated headwater section as well as in the flow-regulated Piedmont and Coastal Plain reaches (Travnichek and Maceina 1994, U.S. Geol. Surv., unpubl. data). Thus, although extensively managed, the Tallapoosa River system represents an important natural resource. If an IBI adapted for use in the Tallapoosa River system is sensitive to faunal changes associated with hydrologic alteration, it could be used to monitor effects of future development in this basin and, with appropriate modification, in other flow-regulated rivers in the southeastern United States.

Our objectives were to 1) develop an IBI for application to a flow-regulated, species-rich river system in Alabama, 2) quantify fish assemblage differences among sites variously affected by flow alteration, and 3) provide a baseline for evaluating the effects of future changes in the watershed.

Methods

Study Sites

Sampling was conducted at 7 study sites located in the Tallapoosa River system in Alabama (map provided in Travnichek and Maceina 1994). Five sites were located in the Piedmont subregion. Three of the Piedmont study sites (RP1, RP2, and RP3) were in the regulated portion of the Tallapoosa River downstream from Harris Dam. One of the 2 unregulated Piedmont sites (UP1) was located on the Little Tallapoosa River, a major tributary flowing into Harris Reservoir. The second unregulated Piedmont site (UP2) was located in the Tallapoosa River upstream from Harris Reservoir. Two sites (RCP1 and RCP2) were located below Thurlow Dam in the Coastal Plain subregion of Alabama. The 7 sites correspond to those sampled by Travnichek and Maceina (1994) during 1990–1992 and were chosen to reflect a variety of physical habitat conditions and to continue long-term data collection efforts.

Fish Sampling

Fish were collected June–September 1994 and July–August 1995 using 1.5- × 6.0-m prepositioned area electrofishers (PAE, Bain et al. 1985). The length of individual study sites ranged from 2.5 to 4.0 km with each site divided into sampling sections about 180 m long. At the start of each day, a section was randomly selected and 10–20 PAE samples were collected in the section depending on the availability of shallow-water habitat. Visibility limitations and safety concerns restricted sampling to areas ≤1.5 m deep. If a section was selected where there were no such areas, another section was randomly selected. Sampling continued until evening when low light limited visibility. Usually 2 or 3 sections were sampled in a day. We collected 100 PAE samples at each site in both 1994 and 1995 (total N = 1,400).

Selection and Analysis of Raw IBI Metrics

We reviewed metrics used in published versions of the IBI (Simon and Lyons 1995). Karr (1981) and subsequent authors grouped metrics into 5 categories: (1) species richness and composition; (2) indicator species; (3) trophic function; (4) reproductive function; and (5) abundance and condition. We used these categories as a framework for selecting metrics. Because of sampling limitations associated with PAEs, we based the metrics on small-bodied fishes (usually <150 mm TL) that were typically collected in shallow-water habitats. In order to examine redundancy in potential metrics, we calculated Pearson correlation coefficients for all raw metric scores based on our 1994–1995 collection data.

Development of Metric Expectations

Using distributional data (Lee et al. 1980; M. Pierson, Ala. Power Co., unpubl. data) and collection records from 1990–1995 (Ala. Coop. Fish and Wildl. Res. Unit, unpubl. data; Natl. Biol. Serv., unpubl. data; B. Freeman, Univ. Ga. Museum of Nat. History, unpubl. data), we estimated the number of species common enough to be collected in a thorough sampling effort at Piedmont and Coastal Plain study sites. This was done to establish expected values for species richness and composition metrics. We did not include historically rare species (those that occurred in the basin but were represented in few collections) or large-bodied fishes that were unlikely to be captured by PAEs in our estimates in order to minimize negative bias in IBI scores. Because of differences in stream size and faunal composition, we established different expectations for total species richness at UP1, UP2, and the remaining Piedmont sites. For the Coastal Plain sites, where fewer collection data were available and distributional data indicated as many as 70 common fish species, 70 was averaged with the highest number of species collected at either site in 1994 or 1995 to estimate expected total species richness. We used our 1994–1995 collection data to establish expectations for metrics that were based on relative abundances by selecting the best value observed in both years at Piedmont and Coastal Plain sites. The best value approach to establishing metric expectations has been widely used (e.g., Fausch et al. 1984, Karr et al. 1986, Lyons 1992, Osborne et al. 1992) and typically works well when unimpaired reference sites are difficult to identify (Simon and Lyons 1995). For metrics based on relative abundances, we chose to use the same expected values at each Piedmont site because there was not sufficient evidence in historical collection records to suggest appropriate values that would reflect possible longitudinal differences in relative abundances among sites.

Metric Standardization, Formulation, and Sensitivity Analysis

We followed the approach of Minns et al. (1994) for standardizing metric scores, formulating the IBI, and analyzing effects of individual metrics on the overall IBI score. Raw metrics were standardized to a scale of 0–10. A value of 10 would be assigned if the raw metric equalled or exceeded the expected value. Otherwise the standardized score was calculated as $B \times$ raw score, where B = 10/expected value. Standardized IBI metrics were summed and multiplied by $10/N_m$ (N_m, number of metrics) to obtain an IBI score that varied continuously from 0 to 100. A score of 0 would indicate that sampling produced no fish and a score of 100 would indicate raw scores $\geq$ the expected value for each of the metrics. IBI scores were divided into 5 categories: >0–20, >20–40, >40–60, >60–80, and >80.

In order to determine relative contributions of individual metrics to the IBI score, we calculated a reduced IBI for each metric: reduced IBI = $10(N_m \times \text{IBI}/10 - \text{test metric})/(N_m - 1)$ (Minns et al. 1994). We then computed the difference between reduced and complete IBIs for each site and year, for each metric. The total variances (i.e., across sites and years) of the differences were used to assess the sensitivity of the complete IBI to individual metrics. Because each metric did not contribute uniformly to the complete IBI score across the range of possible IBI scores (Angermeier and Karr 1986), we also calculated the variances of the differences for IBI scores grouped by score category (i.e., >40–60, >60–80). The ratio of within-category variance to total variance for a metric indicated range sensitivity. For example, a metric with a high ratio affected IBI scores in a particular category more than a metric with a low ratio (Minns et al. 1994).

Application of the Tallapoosa River IBI

Travnichek and Maceina (1994) sampled the 7 study sites during 1990–1992 and found reduced fish species richness, diversity, and density at the most strongly regulated sites compared to unregulated sites. To assess the ability of the IBI to detect faunal changes that were reflected in species richness, diversity, and density, we calculated IBI scores using Travnichek and Maceina's data. Because sampling methods were identical and total sample sizes were similar between their study and ours, we also included IBI scores for Travnichek and Maceina's data in our sensitivity analysis.

Results and Discussion

IBI Metrics and Expectations for the Tallapoosa River System

We collected a total of 20,686 individuals of 78 species from 1,400 PAE samples during 1994 and 1995. Sixty-two species were collected in both years, 8 species were collected in 1994 but not in 1995, and 8 different species were collected in 1995 but not in 1994. Cyprinids and percids (primarily *Etheostoma* and *Percina* spp.) dominated collections in terms of relative abundances in both years. We collected very few individual fish >200 mm TL. Sampling gear bias against larger individuals was evident in the relatively low proportion (0.005–0.060) of top predators collected at all sites in both years.

The 9 metrics selected for the IBI and expected values for each of the metrics are presented in Table 1. Metric categories found in published IBIs were used when possible and substitutions or omissions were made only when metrics did not discriminate among any of the sites or when metrics were highly intercorrelated. Species richness and composition metrics were similar to those in Karr's (1981) original IBI with the exception that we included *Micropterus* spp. in the number of sunfish species and we did not include a top predator trophic category. These changes were made based on the low proportion of top predators in collections.

Indicator species metrics included percentage of intolerant species but did not include percentage of green sunfish (*Lepomis cyanellus*). Based on published accounts of habitat requirements (Lee et al. 1980, Etnier and Starnes 1993), we identified 8 species as intolerant, including the endemic Tallapoosa shiner (*Cyprinella gibbsi*) and Tallapoosa darter (*Etheostoma tallapoosae*) (Table 2). We chose to use percentage of individuals as intolerant species rather than number of intolerant species to reduce redundancy with species richness and composition metrics. We based our decision on the assumption that sites with high biological integrity support larger populations of intolerant species compared to degraded sites. Because green sunfish accounted for <1% of the total number of individuals at all sites, the percentage of green sunfish metric was omitted. We substituted evenness, calculated as Shannon diversity (H') / ln (H_{max} or total number of species), for the green sunfish metric as proposed for streams in the Midwest (Simon and Lyons 1995).

Table 1. Metrics, expected values, and coefficients *B* for a modified IBI for the Tallapoosa River system. *B* = 10/expected value (Minns et al. 1994).

Category and metric	Expected values: Piedmont	Expected values: Coastal Plain	Metric coefficient *B*: Piedmont	Metric coefficient *B*: Coastal Plain
Species richness and composition				
Total *N* of fish species	49[a]	56	10/49	10/56
N of sucker species	4	5	10/4	2
N of darter species	6	10	10/6	1
N of sunfish species	10	10	1	1
Indicator species				
% of individuals as intolerant species	22	16	10/22	10/16
Evenness multiplied by 100	100	100	10/100	10/100
Trophic function				
% of individuals as insectivorous cyprinids	49	51	10/49	10/51
% of individuals as benthic fluvial specialists	85	65	10/85	10/65
Abundance				
Density (mean *N*/PAE sample)	24.8	20.8	10/24.8	10/20.8

[a]This value applies to sites RP1, RP2, and RP3 only; because of smaller stream size, 40 and 44 were used as expected values for species richness (based on historical collection data) at sites UP1 and UP2, respectively.

Trophic function metrics included percentage of individuals as insectivorous cyprinids and percentage as benthic fluvial specialists (Table 1). Benthic fluvial specialists were small species that forage on the stream bottom in areas with moderate to swift current. Fishes in the insectivorous cyprinids and benthic fluvial specialists groups were selected based on published accounts of habitat use and behavior (Lee et al. 1980, Etnier and Starnes 1993). We chose to omit the omnivore metric because it did not discriminate among sites and was highly correlated with the percentage of individuals as cyprinids ($R = 0.76$, $P = 0.002$), which resulted in reduced scores at species-rich Coastal Plain sites where cyprinids dominated the fish assemblage.

Reproductive function is often assessed by the percentage of hybrids (Simon and Lyons 1995). Because only 7 hybrids (all *Lepomis* spp.) were collected in both years, the percentage of hybrids metric was not useful for comparing sites and therefore not included in the IBI. However, if warranted by future collection data, the percentage of hybrids metric could be used in the IBI. We used density expressed as mean *N*/PAE sample as an abundance metric. We did not include a metric based on fish condition because we did not collect the required data.

Relationships among Metrics

Correlations of raw metric scores across sites and years ($N = 14$) suggested some overlap between metrics. Overall species richness was correlated with number of darter species ($R = 0.62$, $P = 0.016$) and number of sucker species ($R = 0.72$, $P = 0.004$),

Table 2. List of species classified as intolerant, insectivorous cyprinid, or benthic fluvial specialist for use in the Tallapoosa River IBI.

Common name	Scientific name	Intolerant species	Insectivorous cyprinid	Benthic fluvial specialist
Southern brook lamprey	*Icthyomyzon gagei*			+
Largescale stoneroller	*Campostoma oligolepis*			+
Alabama shiner	*Cyprinella callistia*		+	+
Tallapoosa shiner	*Cyprinella gibbsi*	+	+	
Blacktail shiner	*Cyprinella venusta*		+	
Lined chub	*Hybopsis lineapunctata*			+
Striped shiner	*Luxilus chrysocephalus*		+	
Bandfin shiner	*Luxilus zonistius*		+	+
Pretty shiner	*Lythrurus bellus*		+	
Speckled chub	*Macrhybopsis aestivalis*	+	+	+
Silver chub	*Macrhybopsis storeriana*		+	+
Bluehead chub	*Nocomis leptocephalus*			+
Golden shiner	*Notemigonus crysoleucas*		+	
Orangefin shiner	*Notropis ammophilus*		+	+
Emerald shiner	*Notropis atherinoides*		+	
Rough shiner	*Notropis baileyi*		+	
Silverjaw minnow	*Notropis buccatus*			+
Silverside shiner	*Notropis candidus*		+	
Fluvial shiner	*Notropis edwardraneyi*		+	
Silverstripe shiner	*Notropis stilbius*		+	
Weed shiner	*Notropis texanus*			+
Skygazer shiner	*Notropis uranoscopus*			+
Mimic shiner	*Notropis volucellus*		+	
Clear chub	*Notropis winchelli*		+	+
Riffle minnow	*Phenacobius catostomus*	+	+	+
Creek chub	*Semotilus atromaculatus*		+	
Highfin carpsucker	*Carpiodes velifer*			+
Alabama hogsucker	*Hypentelium etowanum*			+
Spotted sucker	*Minytrema melanops*			+
River redhorse	*Moxostoma carinatum*			+
Black redhorse	*Moxostoma duquesnei*			+
Golden redhorse	*Moxostoma erythrurum*			+
Blue catfish	*Ictalurus furcatus*			+
Black madtom	*Noturus funebris*			+
Speckled madtom	*Noturus leptacanthus*			+
Shadow bass	*Ambloplites ariommus*	+		
Crystal darter	*Crystallaria asprella*	+		+
Muscadine darter	*Percina (Alvordius)* sp.			+
Logperch	*Percina* sp. cf. *caprodes*			+
Blackbanded darter	*Percina nigrofasciata*			+
Bronze darter	*Percina palmaris*			+
River darter	*Percina shumardi*			+
Saddleback darter	*Percina vigil*	+		+
Naked sand darter	*Ammocrypta beani*	+		+
Lipstick darter	*Etheostoma chuckwachatte*			+
Greenbreast darter	*Etheostoma jordani*			+
Johnny darter	*Etheostoma nigrum*			+
Goldstripe darter	*Etheostoma parvipinne*			+
Speckled darter	*Etheostoma stigmaeum*			+
Tallapoosa darter	*Etheostoma tallapoosae*	+		+
Banded sculpin	*Cottus carolinae*			+
Tallapoosa sculpin	*Cottus* sp. cf. *carolinae*			+

highlighting the contribution of percids and catastomids to total species richness. Similarly, the number of darter species was positively correlated with number of sucker species ($R = 0.57$, $P = 0.033$). Percentage of benthic fluvial specialists was correlated with percentage of intolerant species ($R = 0.59$, $P = 0.027$). This relationship was expected because 6 of the 8 intolerant species were also in the benthic fluvial specialists category. Other correlations that were difficult to explain suggested possible site effects or spurious correlations: the percentage of insectivorous cyprinids was inversely correlated with number of sunfish species ($R = -0.70$, $P = 0.006$) and percentage of intolerant species ($R = -0.55$, $P = 0.043$). All other correlations were not significant ($P > 0.05$).

Sensitivity Analysis

The highest total variances of the differences between reduced and complete IBIs were associated with percentage of insectivorous cyprinids and percentage of intolerant species (Table 3). Metrics with high total variances of the differences contribute more to the complete IBI score than metrics with low variances. Other high variance values were associated with the number of darter species and density. The lowest variance (i.e., least contribution to complete IBI score) was associated with species richness. Because the 4 metrics with the highest variances were distributed among the 4 metric categories (Table 3), the IBI should be sensitive to changes in assemblage composition, trophic structure, or fish abundance.

IBI sensitivity to metrics was not the same in different score categories. The IBI was most sensitive to species richness and composition metrics, and percentage of benthic fluvial specialists when IBI score was in the >40–60 category. IBI values in

Table 3. Total variance of differences between reduced and complete IBI's, and ratios of within to total variance for IBI score categories, for individual metrics. Underlined values were greater than the median ratio and indicate the most sensitive metrics within each score range.

Category and metric	Total variance	IBI score category >40–60	IBI score category >60–80
Species richness and composition			
Total *N* fish species	1.23	<u>1.10</u>	0.77
N sucker species	3.69	<u>1.36</u>	0.53
N darter species	6.28	<u>1.48</u>	<u>0.87</u>
N sunfish species	4.86	0.78	0.66
Indicator species			
% individuals as intolerant species	11.41	0.14	<u>1.04</u>
Evenness multiplied by 100	1.40	0.37	0.43
Trophic function			
% individuals as insectivorous cyprinids	13.52	0.49	<u>0.92</u>
% individuals as benthic fluvial specialists	3.48	<u>0.92</u>	<u>0.83</u>
Abundance			
Density (mean *N*/PAE sample)	6.71	0.32	<u>1.04</u>

the >60–80 category were most sensitive to trophic function metrics, percentage of intolerant species, and density. Because only 2 IBI scores were in the >80 category, we were unable to assess IBI metric sensitivity for scores >80.

Application of the IBI

IBI scores for the 7 study sites based on Travnichek and Maceina's (1994) data were consistent with their original observations of reduced species richness, diversity, and density at regulated Piedmont sites in the Tallapoosa River compared to unregulated sites (Table 4). Similarly, IBI scores for the Coastal Plain sites accurately reflected differences observed by Travnichek and Maceina (1994). The IBI score for the downstream Coastal Plain site (RCP2) was considerably higher than the score for the upstream site (RCP1), located about 2 km downstream from Thurlow Dam. These results suggest the IBI is sensitive enough to accurately summarize differences in fish assemblages in the Tallapoosa River.

IBI scores from the 3 available data sets were variable among years but patterns were evident (Table 4). IBI scores for UP2 were consistently higher than scores for RP1 and RP3 and, in 2 of the 3 data sets (1990–1992 and 1995), than the score for RP2. This result was consistent with our subjective ranking of site UP2 as least impacted by human activity (i.e., flow is unregulated and local riparian disturbance is minimal at this site, whereas UP1 is largely bordered by pasture). The rank order of IBI scores for regulated Piedmont sites was inconsistent across years but the range of IBI scores among regulated sites was consistently low (4.16–5.83). At Coastal Plain sites the IBI score for RCP2 was higher than the score for RCP1 in 1990–1992 and 1995. The low IBI score for RCP2 in 1994 was largely attributable to a lower percentage of intolerant species.

IBI scores were higher in 1995 than in 1994 at 6 of the 7 sites. We speculate that the observed differences in IBI scores were related to differences in discharge between years. Mean hourly discharge for the Tallapoosa River measured near Montgomery, Alabama, was higher in 1994 than in 1995 during spring (136 m^3/sec vs. 75 m^3/sec) and summer (233 m^3/sec vs. 46 m^3/sec). High spring and summer discharges may have resulted in reduced reproductive success and survival during 1994 compared to 1995.

Conclusions

Simplification of biological data is the goal in the use of indices such as the IBI (Gerritsen 1995). Thus, at sites where low IBI scores suggest possible environmental degradation, additional sampling and analyses may be required to identify the nature and extent of changes in the ecosystem. The IBI represents a potentially useful tool for the resource manager because it summarizes information about fish communities into a single value that is more easily interpretable, especially by non-biologists, than more complex analyses. The index may also present a more accurate assessment of system function than individual measures such as species richness. For example, in 1994 we collected more species at RP1 than at any other site, although abundances

Table 4. IBI scores for 1994, 1995, and 1990–1992 and measures of species richness, species diversity (Shannon diversity, H'), and density (mean N fish/sample).

Metric	Site						
	UP1	UP2	RP1	RP2	RP3	RCP1	RCP2
				1994			
Species richness	7.25	6.82	7.14	6.73	5.71	4.82	5.89
N sucker species	7.50	7.50	7.50	5.00	7.50	6.00	8.00
N darter species	5.00	8.33	10.00	10.00	8.33	7.00	5.00
N sunfish species	9.00	6.00	8.00	9.00	6.00	6.00	5.00
% intolerant species	0.21	9.41	6.91	9.77	7.68	7.81	2.50
Evenness	7.21	7.59	7.60	6.09	7.89	7.44	6.56
% insectivorous cyprinids	7.59	4.08	4.59	3.24	8.45	5.06	9.98
% benthic fluvial specialists	4.91	9.92	7.27	10.00	7.11	9.97	7.22
Density	2.61	5.38	2.71	5.62	3.52	4.51	7.59
Total IBI score	56.98	72.26	68.58	72.74	69.11	65.12	64.16
				1995			
Species richness	6.50	7.95	5.92	6.73	6.53	6.07	7.32
N sucker species	5.00	10.00	7.50	7.50	7.50	8.00	10.00
N darter species	5.00	10.00	10.00	10.00	10.00	8.00	8.00
N sunfish species	5.00	7.00	5.00	9.00	6.00	8.00	4.00
% intolerant species	0.25	6.55	3.95	4.68	6.73	5.19	9.63
Evenness	6.52	7.34	6.97	6.89	8.16	7.29	7.89
% insectivorous cyprinids	9.63	6.76	9.90	2.61	6.06	7.20	9.53
% benthic fluvial specialists	8.86	9.47	8.08	6.67	7.11	9.69	9.98
Density	8.73	10.00	7.96	7.28	4.96	6.45	9.98
Total IBI score	61.66	83.40	72.54	68.19	70.05	73.20	84.81
				1990–1992			
Species richness	6.50	6.82	3.88	4.90	5.51	4.29	5.71
N sucker species	5.00	7.50	2.50	2.50	2.50	2.00	6.00
N darter species	6.67	10.00	8.33	10.00	10.00	5.00	6.00
N sunfish species	7.00	6.00	4.00	4.00	6.00	5.00	5.00
% intolerant species	0.05	1.14	0.45	1.27	1.23	1.63	8.06
Evenness	6.09	6.68	6.97	6.79	7.45	5.79	7.61
% insectivorous cyprinids	9.43	4.45	10.00	4.76	8.00	10.00	10.00
% benthic fluvial specialists	2.46	6.25	6.29	5.11	4.85	4.02	6.69
Density	4.68	5.27	1.49	2.70	1.73	3.07	9.88
Total IBI score	53.19	60.11	48.80	46.69	52.52	45.32	72.18
Individual Measures							
				1994			
Species richness	30	30	35	33	28	27	33
Species diversity	2.43	2.58	2.70	2.13	2.63	2.45	2.29
Density	6.48	13.35	6.71	13.94	8.74	9.38	15.79
				1995			
Species richness	26	35	29	33	32	34	41
Species diversity	2.13	2.61	2.35	2.41	2.83	2.57	2.93
Density	21.66	24.79	19.75	18.06	12.29	13.41	20.75
				1990–1992			
Species richness	26	30	19	24	27	24	32
Species diversity	1.98	2.27	2.05	2.16	2.46	1.84	2.64
Density	11.60	13.06	3.69	6.69	4.29	6.39	20.55

and overall IBI were low. This suggests species richness alone would be a poor indicator of biotic integrity at a site experiencing severe flow fluctuations and possibly influenced by variable colonization and recruitment processes.

Additional impacts on aquatic resources are inevitable given the rapid growth of the human population in the Southeast. For example, additional water development projects are being considered in the upper portion of the Tallapoosa River system to meet water supply demands for the metropolitan Atlanta area. Long-term sampling programs will be necessary to detect and respond to declining biological integrity in our rivers; an IBI similar to that formulated in this study, and refined by incorporation of additional data, could prove useful for monitoring biotic changes in southeastern river systems resulting from watershed and streamflow alteration.

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