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A Preliminary Index of Biotic Integrity for Monitoring the Condition of the Rio Paraiba do Sul, Southeast Brazil

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ABSTRACT / The biodiversity of many Brazilian rivers is seriously threatened by industrial and municipal pollution, and Rio Paraiba do Sul, located between two major industrial centers is one example of this situation. A survey of the fish assemblage was conducted from October 1998 to September 1999 and the data were used to develop an index of biotic integrity (IBI). We sampled three zones in bracketing a large urban–industrial complex to evaluate water quality changes and the usefulness of the IBI as a monitoring tool. Water quality was classified as poor upstream of the effluent discharges, very poor near the discharges, and poor–fair downstream of the discharges, with this latter situation revealing the current biological capacity of the river. Physical and chemical habitat characteristics were also measured at each site to construct an independent environmental index to validate the IBI. The habitat and IBI indices were highly correlated, suggesting this IBI would be applicable to other large rivers in southeast Brazil.

Public concern with the state of Brazilian rivers has quickly exposed deficiencies in the understanding of their ecology. One critical defect is the lack of proven procedures to assess the condition of rivers and to monitor their responses to remedial management. In Brazil and in most tropical countries, the few empirical studies of fishes that have occurred were not used to assess biological integrity and therefore were not understood by or useful for decision makers. Assessment procedures are needed that provide reliable answers to river management questions, are rapid and inexpensive, are repeatable, and are easily understood by environmental agencies, so that limited resources can be applied to numerous problems. Integration of traditional chemical and physical measures of water quality with biological measures is also necessary to provide more complete, sensitive, and easily comprehended assessment of river condition (Hellawell 1978, Loeb 1994). Sewage, toxic chemicals, and excessive phosphorus are the main water quality concerns, while altered flow regimes, lost habitat area and diversity, passage obstructions, and riparian degradation constitute the structural problems.

KEY WORDS: Fish assemblage; Fish community; IBI; Pollution; Tropical rivers; Biomonitoring

In Brazilian rivers, most fish assemblage research is restricted to species censuses and changes in assemblage structure, while no available information consistently links fish to water quality or structural conditions. Neither an "index of biotic integrity" nor any other quantitative measure of biotic integrity has been applied to Brazilian rivers. Additionally, none of the previously developed IBIs appeared to be appropriate for use in large tropical rivers, which are very common in South America, most of them in a highly deteriorated state. Our aim is to adapt the index of biotic integrity (Karr 1981) to southeast Brazilian conditions, investigate the contribution of each IBI metric to the index, validate the IBI as a measure of river health by comparing it with an independent measure, and evaluate the IBI's potential for assessing regional environmental quality.

Biological integrity was defined as "the ability to support and maintain a balanced, integrated, adaptive community of organisms having species composition, diversity, and functional organization comparable to that of natural habitat of the region" (Karr 1999). The IBI is commonly used and accepted worldwide as a reliable tool to assess water condition: it is an ecologically based index for quantitatively assessing the biological quality of surface waters. The IBI quantifies the impact of environmental deterioration, based on a series of compositional, structural, and functional metrics of fish assemblages. These metrics are closely aligned with the components that Miller and others (1988)

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Table 1. Underlying assumptions of index of biotic integrity (IBI) concerning how river fish assemblages change with environmental degradation[a]

1. Number of native species and those in specific habitat guilds decline
2. Number of intolerant species declines
3. Proportion of native individuals declines
4. Proportion of native fish species declines
5. Proportion of tolerant individuals increases
6. Proportion of trophic generalists, especially omnivores, increases
7. Proportion of trophic specialists such as invertivores or carnivores declines
8. Fish abundance generally declines
9. Proportion of individuals requiring silt-free coarse spawning substrate declines, and the incidence of hybrids increases
10. Incidence of externally evident disease and morphological abnormalities increase
11. Incidence of alien individuals increases

[a]From Fausch and others (1990).

concluded should be evaluated when monitoring the effects of environmental actions to ensure sensitivity to all forms of degradation.

The original IBI is a multimetric index that uses a combination of 12 attributes, called metrics, of local fish assemblages to assess biotic integrity directly (Karr and others 1986). Each metric is scored according to the range of values observed at the different types of sites, and these vary with stream size and region in 6 of the 12 metrics. Scores of 5.3, or 1 are assigned to each metric according to whether its value approximates, deviates somewhat from, or deviates strongly from the value expected at minimally disturbed sites. Metrics are based on ecological assumptions concerning fish assemblage responses to environmental changes (Table 1). Individual metric scores can be used to diagnose factors responsible for degradation at a site.

An IBI is based on the assumption that there are predictable relationships between fish assemblage structure and the physical, chemical, and biological conditions of waterbodies. Sampling IBI metrics in a river reach allows objective assessment of biotic integrity and ecological processes, and provides a basis for monitoring river trends. The IBI has potential for predictive modeling of aquatic environmental quality. To be truly predictive, such a model should be specific to particular zoogeographic regions and classes of river reaches. It should also be standardized so that consistent results are obtainable by different investigators, and results from different rivers and regions can be compared.

Ideally, undisturbed references sites would provide a standard against which all other sites of a region could be compared, instead minimally disturbed or least disturbed sites are often selected (Steedman 1988, Kerans and Karr 1994). Such sites enable metric evaluation based on abundance of individual animals, their species richness and condition, the level of intrusion of alien species, and the representation of relevant trophic and habitat guilds in the biotic assemblage. There are no undisturbed or minimally disturbed sites in any of southeastern Brazil's highly altered rivers. In many regions of Brazil it is difficult or impossible to find sites with excellent fish assemblages. Furthermore, no guide is available for ranking the ecological significance of the various kinds of disturbance, and hence for objectively choosing a set of least disturbed sites to be ecological standards. When there are no undisturbed or minimally disturbed sites, the least impacted regional sites can be used as standards (Hughes and others 1986). Additionally, historical data such as species records can be used to construct expected theoretical values for IBI metrics.

Methods

Study Area

The Rio Paraiba do Sul is an eighth-order river in southeast Brazil, draining one of the most important industrial regions in the country. Typical winter and summer flows are 109 m^3sec and 950 m^3/sec, respectively. It is 1080 km long, with a 57,000-km^2 watershed, in the states of São Paulo. Minas Gerais, and Rio de Janeiro. Mean annual flow in the study area is 318 m^3/sec at 370 m above sea level (Barroso 1989). The study reach is 70 km long, between the cities of Resende and Barra do Pirai and in a single ecoregion. This ecoregion is characterized by unconsolidated and semiconsolidated sand, gravel, silt, and clay with basalt outcroppings, 100–300 cm of annual precipitation, low mountains, low nutrient soils, semideciduous seasonal rain forest with some Atlantic forest fragments, and poor cropland areas.

The Rio Paraiba do Sul receives many organic and industrial effluents directly affecting the ichthyofauna (Coelho and Fonseca 1986). Over 160 m^3/sec is pumped from the study area for hydroelectric power and water supply purposes, and this further increases water quality concerns. The study area was divided in three zones: ZI, 40 km upriver of the major urban–industrial complex of Volta Redonda: Z2, just down river of the complex, and Z3, 30 km down river of the complex (Figure 1). The river is impounded ca. 30 km upstream of Z1 and agriculture is the main land use in this area. Fields are flood-irrigated, and woody riparian

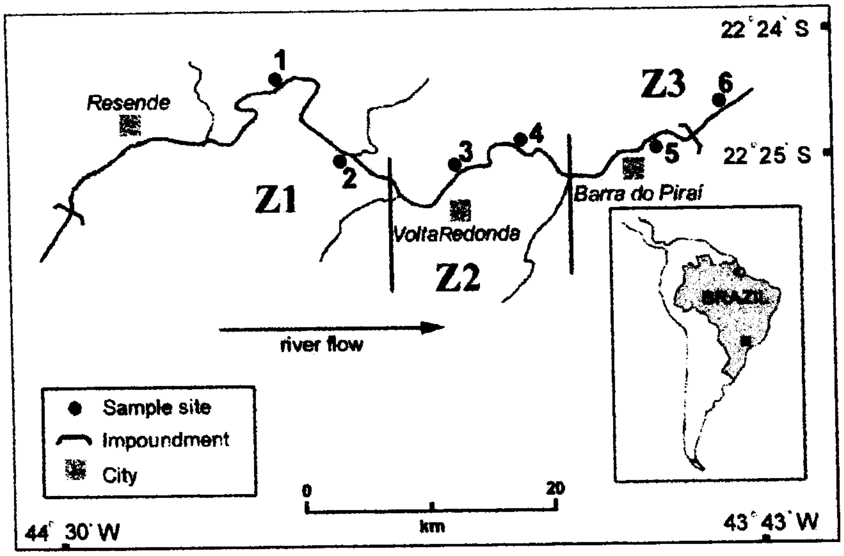

Figure 1. Rio Paraiba do Sul, indicating the three study zones and six sampling sites: Z1 (sites 1 and 2: upstream of the pollution focus). Z2 (sites 3 and 4 at Volta Redonda, near the largest urban-industrial discharges, and Z3 (sites 5 and 6 downstream of the pollution focus).

vegetation is largely absent, resulting in bank erosion, high silt loads, and little large woody debris. Z2 is the area that receives the most untreated sewage and industrial (mainly steel and siderurgic) effluents. Wastes that enter into the river at Z3 are comparatively lower and are from textile industries and agriculture fertilizers. This allowed us to assess conditions upriver and downriver of the highly impacted area and to estimate the capacity of Z2 to improve following waste treatment.

Sampling Methods

Six sites (two in each zone) were sampled monthly between October 1998 and September 1999. Sites were chosen on the basis of accessibility and similarity in physical habitat. During the survey, efforts were made to capture all species of fish at each sampling site, and all captured fish were identified and counted. Fishing effort was standardized and the sampling unit was defined as the pooled total number of fish collected in 15 cast net throws, 15 mesh trays lifts, and 1 seine haul in each site and covering different microhabitats, that is, riffles, pools, and the proximity of the river's margins. Each site was located near a relatively undisturbed tributary confluence, aiming to sample most fish diversity, and attempts were made to collect all fish encountered at the site. The seine was 10 × 2 m with 5-mm mesh. Hauls were performed near the tributary confluence and along the main channel margin, covering approximately 50 m; mesh trays were 80 cm in diameter with 1-mm mesh; and cast nets were 4 m in diameter with 2-cm mesh.

Temperature, turbidity, conductivity, pH, dissolved oxygen, margin alteration, vegetation cover, fish cover, and substrate diversity were measured monthly. This chemical and physical habitat conditions were noted during fish sampling.

IBI Development

Adaptation of Karr's (1981) original IBI requires deletion, adjustment, or inclusion of new metrics more appropriate for subtropical conditions, but the IBI conceptual model is suited to such modifications (Hughes and Oberdorff 1999).

We based our characterizations of the 52 fish species collected on Araujo (1996), Bizerril (1999), unpublished dietary data, plus our own observations (Table 2). Trophic groups included detritivores, omnivores, invertivores, herbivores, and carnivores. Criteria separating these groups are not sharply defined. We did not develop a detritivore metric because there is no evidence yet of a differential response to river degradation by members of this group. A modified 12-metric IBI was developed as follows (Table 3). Overall scoring criteria for compositional and structural metrics were established based on the collected data. For these metrics, a frequency distribution of values was generated, and the values of the 95th percentile (for metrics in which high values indicated high quality) or the 5th percentile (for metrics in which low values indicated high quality) were identified. Then 95% of the frequency distribution below or above this value was divided into thirds (i.e., the values for the 63rd and 32nd percentiles were determined). Values greater than or equal to the 63rd percentile value were given a score of 5 (good) for metrics in which high values indicated high quality and a score of 1 (poor) for metrics in which high values indicate low quality. Values between the 63rd and 32nd percentiles were given a score of 3 (fair), and values less than or equal to the value for the 32^{nd} percentile were given scores of either 1 or 5, depending on the relation between metric values and environmental quality. A similar scoring procedure was used by Lyons and others (1966). Scores for each metric were summed to give IBI scores, which were then compared among the three zones.

Number of native species. The number of native species for middle mainstream reaches of the Rio Paraiba do Sul was estimated at 57–69, with higher proportions of siluriforms (50%) than characiforms (35%) according to Fowler (1948, 1950, 1951, 1954), Britski (1994), Araujo (1996), and Bizerril (1999). These previous studies had indicated that this approach gave a representative sample of the fish community. We adjusted the expected number of native species and all other taxonomic groups to our sampling program, which focused on river margins and tributary mouths. Because the mid-channel was not sampled, species from this part of the river were excluded from our richness expectations. The number of species was plotted against the sites, which gives a more reliable measure of the species richness adjusted to employed fishing effort: three parallel ranges, similarly to MSRL (Fausch and others 1990) were fitted by eye to included about 95% of the sites, and it indicates the number of species expected to determine the ratings. We follow Hughes and Oberdorff (1999) rather than Karr (1981) in favoring only native species versus all species for this metric, because of the undesirable influence of alien species in our waters. The number of native species is a measure of biological diversity that typically decreases with increased degradation.

Number of intolerant species. Species that previously were very abundant but presently occur only occasionally because of environmental deterioration, were considered intolerant species. According to Lyons and others (1996), they are sensitive to many types of environmental stress and tend to be absent in the presence of environmental degradation. Intolerant species disappear early in the degradation sequence associated with agriculture and urban development (high suspended solids, increased temperatures and siltation, decreased dissolved oxygen), and are the last to reappear after restoration (Ganasan and Hughes 1998).

Percent of cyprinodontiform individuals. Cyprinodontiforms are very common in very polluted South American rivers, sometimes comprising the only group of fishes thriving under such harsh conditions. These fish are considered tolerant species because they are common in most rivers and tolerant to many pollutants. This metric is the complement to "number of intolerant species" and is comparable to Hughes and Oberdorff's (1999) percent tolerant individuals. Like Karr's (1981) original percent green sunfish metric, it distinguishes between low and moderate water quality.

Number of water column characiform species. The number of water column characiforms is an indicator of water column condition, especially for sight feeders. It is a substitute for Karr's (1981) number of centrarchid species (which are alien to Brazil) as well as Hughes and Oberdorff's (1999) number of water column species or number of cichlid species.

Number of siluriform species. The number of benthic siluriforms provides an indicator of benthic habitat degradation as a consequence of catchment erosion and sedimentation. Since suckers do not occur in Brazil, it is a substitute for Karr's (1981) number of sucker species metric. Hugueny and others (1996) used number of large siluriform species in their IBI for an African river. The proportions of siluriforms and characiforms are similar between mid-channel and near-shore areas, so expected proportions of these taxa were used to estimate the expected number of species.

Number of alien species. Like Ganasan and Hughes (1998), we used alien species to represent loss of reproductive isolation, instead of Karr's (1981) percent hybrid individuals. Alien species are mostly accidental releases from fish farms: these species can alter the structure of native fish assemblages by competition and predation, sometimes extirpating some native species.

Percent omnivorous individuals. Adults of these species eat large proportions of both plants and animals. Karr (1981) proposed this metric for assessing disruption of the food web by stressors, and it has been applied widely outside the United States and Canada (Hughes and Oberdorff 1999).

Percent carnivorous individuals. Carnivorous adults eat predominantly other vertebrates or large invertebrates. Karr (1981) proposed this metric for assessing loss of trophic diversity and keystone species. This metric is used widely outside the United States and Canada (Hughes and Oberdorff 1999).

Percent herbivorous individuals. Although not proposed by Karr (1981), this metric is occasionally used outside Canada and United States (Hughes and Oberdorff 1999), especially in the tropics where there are many herbivorous fish species. It is sensitive to disruptions in primary production.

Percent invertivorous individuals. Karr (1981) suggested this metric for assessing disruptions in secondary production, because in US streams invertebrates are responsible for much of the processing of organic matter. Disrupted invertebrate biomass and composition is presumably reflected in disruptions in invertivorous fishes. This metric is also widely used in IBIs outside the

Table 2. Trophic group, geographic origin, tolerance, occurrence, and habitat of fish captured from the Rio Paraiba do Sul. Brazil. October 1998–September 1999

Family/species	Trophic group	Origin	Tolerance	Occurrence	Microhabitat
Characiformes					
Anostomidae					Water column
Leporinus copelandii	Herbivore	Native			
Leporinus mormyrops	Herbivore	Native		Rare	Water column
Leporinus sp.	Herbivore	Native		Rare	Water column
Characidae					
Astyanax bimaculatus	Omnivore	Native			Water column
Astyanax fasciatus	Omnivore	Native			Water column
Astyanax giton	Omnivore	Native			Water column
Astyanax scabripinnis	Omnivore	Native		Rare	Water column
Astyanax taeniatus	Omnivore	Native		Rare	Water column
Astyanax sp 1.	Omnivore	Native			
Astyanax sp 2.	Omnivore	Native			
Deuterodon sp.	Invertivore	Native			Water column
Hyphessobrycon bifasciatus	Invertivore	Native			
Hyphessobrycon callistus.		Alien			
Hyphessobrycon reticulatus	Invertivore	Native		Rare	
Oligosarcus hepsetus	Carnivore	Native			
Probolodus heterostomus		Native			Water column
Curimatidae				Rare	
Cyphocharax gilberti	Detritivore	Native			
Erithrynidae					
Hoplias malabaricus	Carnivore	Native			Water column
Hoplerythrinus unitaeniatus	Carnivore	Native		Rare	Water column
Serrasalminae					
Metynnis maculatus		Alien			
Siluriformes				Rare	
Auchenipteridae					
Glanidium albescens		Native			Benthic
Trachelyopterus striatulus	Omnivore	Native		Rare	Benthic
Callichthyidae					
Callichthys callichthys		Native			Benthic
Corydoras nattereri		Native			Benthic
Loricariidae				Rare	
Hartia loricariformes	Detritivore	Native			Benthic
Hypostomus affinis	Detritivore	Native			
Hypostomus luetkeni	Detritivore	Native			
Hypostomus sp.	Detritivore	Native			Benthic
Loricariichthys spixii	Detritivore	Native		Rare	Benthic
Rineloricaria sp.	Detritivore	Native			Benthic
Pimelodidae					
Pimelodella sp.	Omnivore	Native		Rare	Benthic
Pimelodus fur	Omnivore	Native			Benthic
Pimelodus maculatus	Omnivore	Native			Benthic
Rhamdella sp.	Omnivore	Native		Rare	
Rhamdia parahybae	Carnivore	Native		Rare	
Rhamdia sp.	Carnivore	Native			
Gymnotiformes					
Gymnotidae					
Gymnotus cf. carapo	Invertivore	Native			
Gymnotus sp.	Invertivore	Native		Rare	
Sternopygidae				Rare	
Eigenmannia virescens	Invertivore	Native			
Cyprinodontiformes					
Poeciliidae					
Lebistes reticulatus	Omnivore	Native	Tolerant		
Phallocerus caudimaculatus	Omnivore	Native	Tolerant		
Perciformes					

Table 2. (Continued)

Family/species	Trophic group	Origin	Tolerance	Occurrence	Microhabitat
Cichlidae					
Cichla monoculus		Alien			
Cichlaurus fascetus		Native		Rare	
Crenicichla dorsocellata	Carnivore	Native		Rare	
Crenicichla lacustris		Native		Rare	
Geophagus brasiliensis	Omnivore	Native			
Oreochromis niloticus		Alien			
Tilapia rendalli		Alien			
Sciaenidae				Rare	
Pachyurus adspersus	Carnivore	Native			
Synbranchidae				Rare	
Synbranchus marmoratus	Carnivore	Alien			

Table 3. Fish assemblage metrics used to calculate the Index of Biotic Integrity (IBI) for middle-lower reaches of the Rio Paraiba do Sul. Brazil

Scores and criteria			
	1	3	5
Number of native species	< 4	4–7	> 7
Number of intolerant species	0	1	> 1
Percent of cyprinodontiform individuals	> 67	33–67	< 33
Number of water column characiform species	< 2	2–3	> 3
Number of siluriform species	< 2	2–3	> 3
Number of alien species	> 1	1	0
Percent of individuals as omnivores	> 45	20–45	< 20
Percent of individuals as carnivores	< 1	1–5	> 5
Percent of individuals as herbivores	< 1	1–5	> 5
Percent of individuals as invertivores	> 1	1–5	< 5
Number of individuals minus omnivores	< 5	5–10	> 10
Number of species comprising 90% of individuals	< 3	3–5	> 5

United States and Canada (Hughes and Oberdorff 1999).

Number of individuals minus omnivores. The total number of fish minus omnivores was used as an abundance metric. We excluded omnivores because they tolerate poor conditions. Similarly, Yoder and Smith (1999) subtracted tolerant species from their abundance metric. The expected values for the total number of fish minus omnivores were estimated from our samples.

Percent of species representing 90% of individuals. Fish with abnormalities were rare, so we replaced that metric with a dominance metric. Fore and others (1996) and Kerans and Karr (1994) used a similar metric (percent dominance by three and two taxa, respectively) for assessing macroinvertebrate assemblages.

Validating the IBI

To validate the IBI, we correlated it with an independent environmental index. The environmental index (EI) was based on nine variables, five associated with water quality and four with physical habitat structure, measured at each sampling occasion (Table 4).

The water quality variables (dissolved oxygen, conductivity, temperature, pH, turbidity) related mainly to municipal and industrial pollution. Based on our knowledge of the literature and field experience, we established ideal and stressed conditions for a tropical fish fauna (Loeb 1994). For instance, ideal oxygen conditions existed at > 60% saturation, while stress conditions were < 60% saturation. Expectations were developed in a similar manner for pH, conductivity, and temperature, and for each range we attributed a score 1.3, or 5 if it matched, deviated somewhat from, or deviated strongly from ideal conditions (Table 4).

The habitat structure variables (substrate diversity, margin condition, fish cover, and vegetation cover) were related to altered flows and modified channels. In our study area, numerous human disturbances and engineering works have led to widespread homogeni-

Table 4. Abiotic attributes and criteria for an environmental index (E1) in the Rio Paraiba do Sul. Brazil

Scores and criteria	1	3	5
Dissolved oxygen (% saturation)	< 60	–	> 60
pH	< 4 or > 9.1	4–6 or 8–9	6.1–7.9
Conductivity (μS/cm)	> 100 or < 20	–	20–100
Temperature (°C)	> 28	–	< 28
Turbidity (NTU)	> 25	5–25	< 5
Substrate diversity (rocks, sand, mud)	1 type	2 types	3 types
Vegetation cover (riparian and aquatic)	none	little	common
Fish cover (rocks, wood debris, roots, aquatic vegetation)	none	little	common
Margin alterations	common	little	none

zation of habitat structure. For each site, we qualitatively ranked the occurrence of all types of cover (rocks, wood debris, aquatic vegetation). For fishes, cover acts as a refuge from predators and provides food resources. Similarly, we ranked margin alterations, substrate diversity, and vegetation cover as none, little, or common.

By summing scores obtained for each variable, we obtained an EI varying from 9 to 40, which reflected the overall environmental condition of each site. This index is composed, in turn, by the sum of two indices, the physicochemical index (sum of the five water quality attributes) and the habitat index (sum of the four physical attributes). To better understand the effects of river degradation on fish assemblages and the IBI, we used a nonparametric Kendall concordance coefficient to examine links between EI, IBI, and each IBI and EI metric.

Results and Discussion

Index Response to Disturbance

While existing data provided some information for developing scoring criteria for IBI metrics, a stratified, randomized, and replicated survey of the Rio Paraiba do Sul fish fauna offered a consistent and comprehensive data set for a trial of the IBI concept. Differences between zones represent actual spatial degradation, because all data were collected from the same sites in the same manner. These data indicate that standardized netting techniques can be applied in the absence of boat electrofishing. Hugueny and others (1996) and Ganasan and Hughes (1998) also used standardized netting techniques in larger tropical rivers in developing their IBIs.

The IBI scores reflect our expectations for this highly altered reach of Rio Paraiba do Sul. The three river zones had average IBI scores between 21 and 31, out of a possible range of 12–60. IBI scores indicated poor conditions in Z1, upstream of the industrial plant, then deteriorated even more at the industrial discharge areas in Z2, where IBI indicated very poor conditions. IBI improved to fair in Z3, approximately 40 km downstream of the industrial plant (Figure 2). All three zones are highly modified as a result of their proximity to Volta Redonda (Z2). The capacity for river self-rehabilitation can be assessed at Z3, which attained the highest IBI scores.

Higher IBI scores are significantly correlated with higher habitat (physical) scores but not with physicochemical (water quality) scores (Table 5), and the habitat incorporates much of the variation in IBI scores (Figure 3). Hence, only physical attributes (habitat index) were considered in the analysis of relationship with IBI. Overall, plots between IBI scores and habitat values revealed that Z2 showed the lowest values for both indices, while Z3 showed the best ones, with some coincidences between Z3 and Z1. Refining IBI and E1 metrics (mainly water quality) and scoring could improve the relationship. A strong positive correlation (Pearson $r = 0.64$, $P = 0.00$: Kendall's tau = 0.47, $P =$ 00) was found between the value of IBI and the independent rating of habitat index, with this latter index associated with approximately 41% of the variability in IBI scores. This result further confirmed the ability of the proposed IBI as a tool for detecting environmental degradation. The IBI model developed in this paper should be viewed as preliminary and subject to further refinement. Its present geographic scope is limited to the Paraiba do Sul river and similar-size tropical rivers with a comparable faunal composition. It probably will also be useful in adjacent states, which have similar rivers, and perhaps in others regions of South America.

Measuring Human Influence

IBI responded more strongly to the physical habitat structure variables than to the water quality variables.

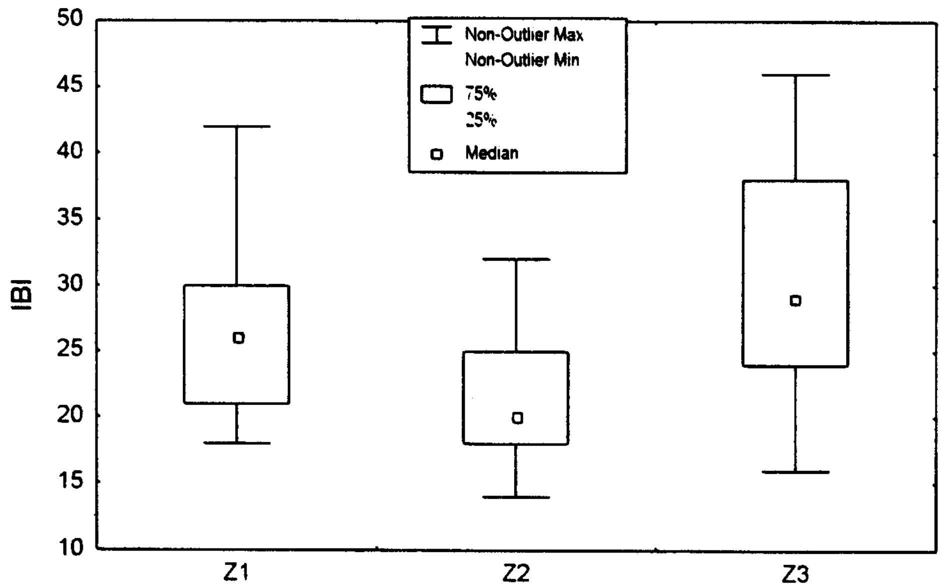

Figure 2. Box plots of index of biotic integrity in the three studied zones for Paraiba do Sul river: 1998–1999. Central marks in each box represent median values: boxes show the extent of the 25th–75th percentiles, and error bars are the 10th–90th percentiles.

Table 5. Coefficient of concordance (Kendall's tau) between IBI and each IBI and E1 metric

Metric	Kendall's tau[a]
Number of native species	0.71**
Number of intolerant species	0.22**
Percent of cyprinodontiform individuals	0.51**
Number of characiform species	0.66**
Number of siluriform species	0.52**
Number of alien species	0.07 NS
Percent of individuals as omnivores	0.05 NS
Percent of individuals as carnivores	0.52**
Percent of individuals as herbivores	0.16**
Percent of individuals as invertivores	0.41**
Number of individuals minus omnivores	0.53**
Number of species to comprise 90% of individuals	0.71**
Dissolved oxygen (% saturation)	−0.05 NS
pH	−0.08 NS
Conductivity (μS/cm)	0.11 NS
Temperature (°C)	0.06 NS
Turbidity (NTU)	−0.13*
Substrate diversity (rocks, sand, mud)	0.41**
Vegetation cover (riparian and aquatic)	0.28**
Fish cover (rocks, wood debris, roots, aquatic vegetation)	0.27**
Margin alteration	−0.42**

[a] *$P < 0.05$, **$P < 0.01$, NS = not significant.

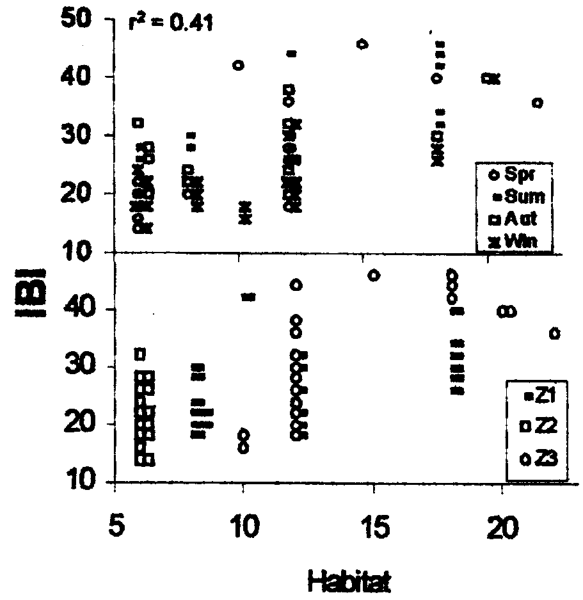

Figure 3. Relationship between IBI and habitat scores. Samples labeled by seasons (above) and by zones (below).

This response suggests that the fish assemblages in these zones of the Rio Paraiba do Sul are affected more by physical habitat alteration than water quality (Table 5). Impacts ranged from agricultural activities involving channelization and riparian destruction to altered hydrologic regime and channel modification related to urban development. Physical habitat diversity and quality, particularly related to riparian condition were important factors affecting fish assemblages: in general, poor riparian systems show increases in sediment that are associated with the degradation in the native fish assemblage. Belliard and others (1999) observed the same pattern. Another consideration is that flow and water quality conditions were consistently mediocre across all three zones, leaving physical habitat structure as the most variable. Finally, habitat was associated with only 41% of the variability in IBI scores, indicating that other factors, such as physicochemical conditions (water quality), sampling date, and hydrological conditions may be important. The measured physicochemical variables did not significantly correlated to IBI scores.

Compared with E1, which measures only conventional water quality and physical habitat variables, IBI is a more robust tool for assessing ecosystem quality because of its ability to integrate the biological, physical, and chemical effects of river perturbations. The three zones all had very poor water quality.

Measurement of such water quality parameters as dissolved oxygen, pH, turbidity, and conductivity poorly reflect toxic effluents or biological condition. Hughes and Gammon (1987) evaluated the effects of improved water quality on fish assemblages in a large river by using IBI, and found expected declines from the upper to the lower river, with small changes near large point sources of pollution. Hugueny and others (1996) and Ganasan and Hughes (1998) found longitudinal changes in IBI scores relative to pollution sources in large tropical rivers also. Similarly, Yoder and Smith (1999) reported on the longitudinal changes in IBI scores above and below a large city, as well as the temporal changes in IBI scores before and after improved sewage treatment.

As might be expected, seasonal variation IBI scores could occur due to either natural changes in fish assemblages or changes in discharge and concentration of pollutants in the river. Plots between IBI scores and habitat scores labeled by seasons did not reveal any clear temporal variation in these indices, although there is a trend for the highest values occur in summer and the lowest values in winter. As expected, highest rainfall in summer result in higher river flow and water level, with the river flooding marginal areas. This increases habitat diversity by enabling fish to reach riparian vegetation and logs that are unavailable during the dry season in winter. Additionally, the largest amount of allocotone material brought into the river increases food availability. An opposite situation occurs in the winter and was reflected in poorer IBI scores. These slightly seasonal changes in IBI confirm its robustness to evaluate and compare changes in environmental conditions at different temporal scales.

Biological Response at the Metric Level

Correlation analysis showed a significant ($P < 0.05$) concordance between IBI and all its metrics with two exceptions (Table 5). The responses of all but two of the IBI metrics were consistent with river degradation. Low correlations between IBI and percent of omnivores and number of alien species are due to different causes. Omnivores are abundant and widespread throughout the study area, while alien species are still rare and restricted to Z2. These conditions contribute to low correlations for these metrics and suggest that scoring criteria need revision once additional data are available.

The number of fish species increased from 24 at Z1 to 30 and 31 at Z2 and Z3, respectively. The lower species number in Z1 may partly result from a hydroelectric impoundment 40 km upstream together with heavily polluted conditions in Z2 acting as physical and chemical barriers, respectively, limiting fish migration. Pringle (1997) described how downstream chemical barriers alter upstream assemblages, and the effects of impoundments on fish assemblages are well established (Agostinho and others 2000, Pringle and others 2000, Schmutz and others 2000). Additionally, effluents from agriculture activities, absence of woody riparian vegetation, and bank erosion contributed to decreased number of fish species at Z1.

The number of individuals was highest at Z2 (2193), followed by Z3 (874), and Z1 (294). The increased number of fish at Z2 was mainly due to opportunists and tolerant omnivores (cyprinodontiforms *Lebistes reticulatus* and *Phallocerus candimaculatus* and, to a lesser extent, the cichlid *Geophagus brasiliensis*). These species thrive in poor water quality and high organic loads, which increase food availability; however, more sensitive species cannot tolerate such conditions. This is why such omnivores were omitted from our number of individuals metric.

There are several reasons for increased biological monitoring and assessment in large Brazilian rivers. The increased rate of degradation of the main lotic systems and the need to improve water quality to protect human healthy are the main issues that could be cited. Additionally, increased public concern about ecosystem quality and the desire to rehabilitate polluted rivers and to protect pristine systems could trigger public policies to address these issues.

Only the middle reaches of Rio Paraiba do Sul were considered in this study because the river's largest steel industry and water removal are located there. The principles and procedures of IBI development were sufficiently robust to cope with some shortcomings without undue loss of power or sensitivity to environmental change.

Preliminary evaluation of the required and available data for developing an IBI indicated that it was feasible to test the value of this index in Rio Paraiba do Sul (Araujo 1998). The underlying assumptions of the IBI concerning how river fish assemblages change with environmental degradation were generally applicable to

Rio Paraiba do Sul conditions. Challenges are expected in applying these assumptions to develop the IBI for other southeastern Brazil systems, but the robustness, flexibility, and sensitivity of the IBI model (Karr and others 1986) suggest that this index has potential value for assessing and monitoring the ecological health of Brazilian rivers. Before doing so, we must determine the degree to which our IBI metrics and scoring apply to other zoogeographic regions and rivers. These systems, although highly disturbed, are capable of considerable improvement, as was indicated for the very polluted study reach in the Paraiba do Sul river. Chemical monitoring alone, as it is presently practiced by Brazilian environmental agencies, is inadequate for assessing the ecological quality of such systems.

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