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Primary Research Paper

Effects of landscape and riparian condition on a fish index of biotic integrity in a large southeastern Brazil river

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Abstract

Environmental conditions of a large river in southeastern Brazil were assessed by evaluating fish assemblage structure (index of biotic integrity, IBI), landscape use (forest, pasture, urban area, and tributary water) and riparian condition. A survey of the 338 km-long middle reach of the Rio Paraiba do Sul, containing a large urban-industrial complex, was conducted in two seasons: summer/wet and winter/dry. Fish were sampled with a standardized level of effort twice at seven sites, between March 2001 and April 2002, by gill nets, cast nets, sieves and seines. Riparian condition was evaluated by direct observations, and land use maps were used to assess landscape condition of an 8 km^2 buffer surrounding each site. IBI scores ranged from 5 to 36 (out of a possible range of 4–40), with lowest values at an urban-industrial landscape, and highest scores upstream and downstream, indicating the river's recovery capacity. The most appropriate time to assess IBI was during the winter/dry period, when sampling was more effective and the IBI was more sensitive to changes in environmental quality. Landscape use and riparian condition were correlated, and IBI was positively correlated with % pasture, % tributary area, and riparian condition, but negatively correlated with % urban area. In some cases urban areas eliminated riparian woody vegetation, destabilizing site physical habitat structure.

Introduction

Indices of biotic integrity (IBIs) based on fish assemblage structure, which were first introduced by Karr (1981), are now used worldwide to assess fish assemblage condition (Hughes & Oberdorff, 1998). IBIs also have been successfully employed for benthic macroinvertebrates (Kerans & Karr, 1994; Klemm et al., 2003), algae (Hill et al., 2000; Fore, 2002), and riparian birds (Bryce et al., 2002), to indicate alteration in aquatic systems. These IBIs offer more comprehensive assessments than other biotic indicators based solely on species richness, diversity indices, indicator species, or multivariate analyses (Karr et al., 1986; Karr & Chu, 1999; Verdonschot, 2000). Although widely accepted as monitoring tools, IBIs are also useful for relating fish assemblage responses to degradation of riparian zones and landscapes.

Rivers are influenced by land use at regional scales (Richards et al., 1996) and basin land use and riparian zone condition can interact to affect the severity of water quality degradation (Meador & Goldstein, 2003). Riparian condition and landscape uses are micro/proximal and macro/distal indicators of environmental disturbance, respectively. Alterations in riparian cover and intensive land uses reduce IBI scores by degrading fish assemblage structure and dynamics (Steedman, 1988; Roth et al., 1996; Allan et al., 1997; Wang et al., 1997, 2000, 2001; Klauda et al., 1998; Lammert & Allan, 1999; Schleiger, 2000; Meador & Goldstein, 2003; Snyder et al., 2003; Hughes et al., 2004; Van Sickle et al., 2004). Riparian canopy cover is important for moderating water temperatures through shading, as well as for providing wildlife habitat, riverbank stability, and particulate organic material (Barling & Moore, 1991; Gregory et al., 1991; Osborne & Kovacic, 1993). Organic inputs from riparian vegetation are major food sources for river organisms (Cummins, 1974) and large woody debris provides structure to create and maintain complex channel habitats (Gregory et al., 2003). Riparian vegetation also affects aquatic macrophytes. Presence of complex and extensive riparian cover at large river margins often indicates favorable environmental quality, greater habitat diversity, increased food availability for aquatic biota, reduced bank erosion, and decreased nutrient and sediment loads into the main river channel (Karr & Schlosser, 1978; Gregory et al., 1991). Highly altered riparian zones impoverish riparian cover and river habitats and decrease the diversity and complexity of aquatic biota.

Landscape land use has a close relationship with riparian and river habitats and, consequently, with aquatic communities. Intensive land uses such as urbanization and row crop agriculture decrease riparian cover and increase physical habitat degradation, sedimentation, hydrographic alterations, temperature oscillations, and contaminant and nutrient concentrations (Bryce et al., 1999). Intensive urbanization and agriculture are indicators of poor environmental quality, while native forest is associated with good environmental condition (Steedman, 1988; Wang et al., 2001). However, agriculture and urbanization vary along natural gradients, confounding the understanding of biological processes related to land use (Allen et al., 1999; Snyder et al., 2003). Land use assessments are important for understanding environmental gradients and factors that structure fish assemblages, and such information is necessary for comprehensive ecological assessments of aquatic systems and their fish assemblages (Steedman, 1988; Roth et al., 1996; Lammert & Allan, 1999; Schleiger, 2000; Waite & Carpenter, 2000; Meador & Goldstein, 2003). The preceding studies focused mostly on sets of small streams. Our first objective was to apply the IBI along a single 338 km reach of the Rio Paraíba do Sul, and to relate the resulting IBI scores to landscape land use and riparian condition. A second objective was to determine the effect of sampling season on the ability of the IBI to detect major perturbations.

Study area

The middle reach of the Rio Paraíba do Sul flows 400–600 m above sea level and drains ancient, predominantly sedimentary, soil covered by tropical forest. This ecoregion is characterized by both un- and semi-consolidated sand, gravel, silt and clay, with basalt outcroppings, low mountains, low nutrient soils, fragmented semi-deciduous seasonal rain forest, and poor croplands. The climate is mesothermic with high relative humidity, hot and wet summers and dry winters. Annual rainfall ranges from 100 to 300 cm, with the average generally over 200 cm (DNAEE, 1983). Most precipitation occurs between November and January, and heavy rains occasionally cause large floods of the Paraíba do Sul River. June through August is the driest period of the year (Carvalho & Torres, 2002). Temperature ranges from minima of 20–22 °C in June through August and maxima of 32–34 °C in December through February, with an annual average of 26–28°.

River flow in this reach averages 318 $m^3 s^{-1}$, ranging from 109 $m^3 s^{-1}$ in the dry period to 950 $m^3 s^{-1}$ in the wet period (Hydroscience, 1977). The Rio Paraíba do Sul is 1080 km long, with a 57,000 km^2 watershed. The study reach was 338 km long, covering a drainage area of approximately 33,663 km^2 within a single ecoregion between the parallels 20°26′ and 23°38′ south and the meridians 41°00′ and 46°30′ west (Fig. 1). The Paraíba do Sul waters are widely used for human consumption, industrial use, irrigation, hydroelectric power plants and recreation. The total water volume removed for domestic uses is estimated at 60 $m^3 s^{-1}$; other uses like industry and agriculture lack official volume estimates (Carvalho & Torres, 2002). Human actions at the landscape scale disrupt the geomorphic processes that maintain the riverscape and its associated

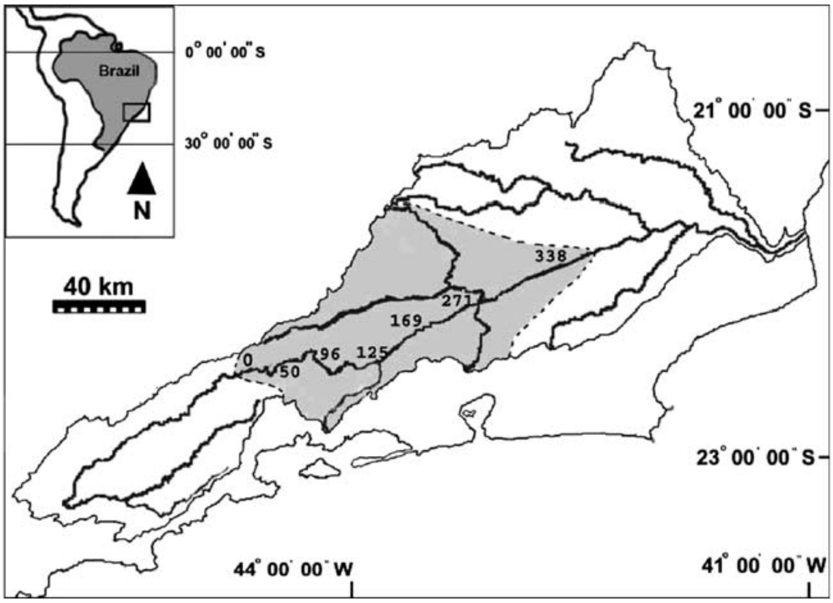

Figure 1. Rio Paraíba do Sul watershed, indicating the seven sampling sites from upstream to downstream.

biota and frequently result in habitat that is both degraded and less heterogeneous.

The study reach was chosen because it drains one of the most important industrial regions in Brazil, and includes the most polluted section of the river (Pfeiffer et al., 1986), with several textile, chemical and food industries, and a large industrial steel plant at Volta Redonda. Agriculture and sand mining also are common in the area. Diffuse pesticide and sediment pollution from agriculture is combined with point source organics and metals from untreated municipal and industrial effluents. Seven sampling sites were chosen, near the municipalities of Queluz, Resende, Barra Mansa, Volta Redonda, Barra do Piraí, Três Rios, and Além Paraíba, and each included a tributary to the mainstem.

The seven sites (Fig. 1) were each sampled in two seasons: high flow (summer) and low flow (winter), between March 2001 and April 2002. We examined these two seasons to evaluate their effects on IBI interpretation, because hydrologic period produces variance in habitat availability and also influences fish migrations and the efficiency of fishing techniques in large rivers. Distance between adjacent sites was 29 and 102 km. Sites were chosen on the basis of accessibility, similarity in habitat types, and to maximize the diversity of habitat types (pools, riffles, tributaries) at each site. Study sites were 70–100 m wide and

Table 1. IBI metric scoring criteria for the middle Rio Paraíba do Sul

Metrics	5	3	1	Best	Worst
Number of native species	> 23	19–23	< 19	27	15
Number of Characiform species	> 10	8–10	< 8	12	5
Number of Siluriform species	> 10	8–10	< 8	12	6
Number of sensitive species	> 5	3–5	< 3	8	0
% Cyprinodontiform individuals	< 31	32–63	> 64	0	95
Number of dominant species	> 10	7–10	< 7	16	1
% Omnivorous individuals	< 56	56–75	> 75	34	97
% Carnivorous individuals	> 13	8–13	< 8	20	0

10–300 cm deep depending on habitat type. The most upstream and downstream sites had predominately rock, cobble and sand substrates, abundant cover, and riparian grass and shrubs. The middle sites had sand and clay substrates, fair cover, and riparian grass or barren.

Methods

Fish assemblage sampling and analysis

Each site was sampled over a 24-h period to cover an area of 56,000 m^2, with reaches 560–800 m long and 70–100 m wide. Several fishing methods were used in a standardized manner to collect the maximum number of species and individuals in different sizes and microhabitats. Fishing equipment included gill nets, cast nets, seines and sieves. The sites were too deep for electrofishing by wading, and we lacked an electrofishing boat. At each site, a total of 18 gill nets (25×2.5 m, with 2.5–7.5 mm mesh) were deployed in deep water in the afternoon and retrieved the following morning, for a total of 16 h fished per net. Cast nets (3 m diameter and 2–3 cm mesh) were fished by two skilled persons for 2 h in water 2–3 m deep. A seine (10×3 m with 5 mm mesh) was employed by two persons in shallow areas for 2 h. A sieve (80 cm in diameter with 1 mm mesh) was used in macrophyte beds by one person for 1 h. Frequently, too few individuals were collected at a site visit with any single gear to be adequate alone for evaluating metric and IBI scores. Following Whittier et al. (1997), Ganasan & Hughes (1998), and Bozzetti & Schulz (2004), we pooled all fish caught by the different fishing equipment taken at each site visit into a single value, thus defining the unit effort. Fish were identified to the lowest taxonomic level possible (Appendix 1). Gut contents of some species were examined to confirm feeding habits. Voucher specimens were deposited at the Laboratory of Fish Ecology, Universidade Federal Rural do Rio de Janeiro.

An IBI was adapted for the Rio Paraíba do Sul by Araújo (1998) and Araujo et al. (2003). Eight metrics were chosen on the assumption that they represented key aspects of assemblage structure and function and changed with environmental deterioration (Karr, 1981). Native species richness was suggested by Karr (1981) to represent biological diversity, which typically declines with disturbance and is commonly used in IBIs (Hughes & Oberdorff, 1998). Number of characiform species is a substitute for Karr's (1981) number of sunfish (pool) species, which tend to decline with increased turbidity or reduced cover. Number of siluriform species is a substitute for Karr's (1981) numbers of darter and sucker (benthic) species, which are reduced by sedimentation and insufficient dissolved oxygen. Percent cyprinodontiforms substitutes for Karr's percent tolerant species, which increase or dominate in polluted environments (Ganasan & Hughes, 1998). The number of sensitive species metric behaves just the opposite, decreasing or disappearing altogether from disturbed environments (Karr, 1981). Percent omnivores increase as the food base is disrupted (Karr, 1981), but in subtropical rivers with highly fluctuating food sources, omnivores are usually abundant. Higher percent carnivores represent a more balanced food energy base (Karr, 1981) and species desired for consumption by humans. Number of dominant species is a measure of evenness, with lower numbers representing disrupted river environments (Karr & Chu, 1999; Klemm et al., 2003).

Metric scoring criteria for the IBI were based on the highest metric scores observed in the entire reach, because no minimally disturbed reference site data were available. Clearly, a high score does not indicate integrity, but it can reveal marked differences among sites. This approach was suggested by Karr et al. (1986) and employed by Hughes & Gammon (1987), Ganasan & Hughes (1998), and Bozzetti & Schulz (2004). Following the method used by Ganasan & Hughes (1998), metric scoring criteria were developed by trisecting the range of obtained values. For example, if the maximum number of species observed at any site was 27 and the least was 16 (for a range of 12), sites with 24–27 species were scored as 5, sites with 20–23 species were scored as 3, and those with <20 species received a score of 1 (Table 1). In addition, raw metric values at the low end of the range were given a minus, and those at the high end a plus. Sites were given an extra point for each two pluses, while a point was subtracted for each pair of minuses. Hughes & Gammon (1987) and

Ganasan & Hughes (1998) used this scoring to improve IBI sensibility and to help discriminate sites that tend to frequently score especially low or high within a 1, 3, or 5 scoring category. We used the same IBI metrics and scoring criteria in both seasons to allow us to compare the effect of season without confounding that comparison with a varying IBI. Bozzetti & Schulz (2004) used a similar approach to assess seasonal effects, and although species composition differed significantly among seasons, their IBI consistently discriminated poor site quality from fair quality.

Land use and riparian condition

Five classes of land use were assessed: secondary forest (since no original forest exists in the area), commercial forest (reforested with *Eucalyptus* sp. for timber), pasture, urban/industrial (industries, railroads, roads, urban centers, residential areas, playing fields, mines) and tributary water (surface area of tributary water in the buffer). Land use of each site was determined using a geographic information system (GIS). A local buffer area of 8 km^2 was assessed (4 km upriver and 1 km down river of the sampling site, and 0.8 km land ward from each river margin). This buffer area was chosen after examining alternative buffer areas, including the entire catchments. These buffer areas represent land uses immediately affecting the sites vs. land uses farther removed from the river. Immediate riparian zones were 25 m shoreward from both sides of the river and 580–800 m long, depending on the site length. Land uses were based on 1:50,000 scale maps from the Brazilian Institute of Geography and Statistics digitized by SAGA/UFRJ software (Xavier-da-Silva, 2001). Riparian condition (percent eroded, vegetation density, macrophyte abundance, human activities) was scored as 0–20 from field observations (Barbour et al., 1999; Table 2). This was a qualitative score prone to observer error, but with reasonable precision (Kaufmann et al., 1999). No data were collected for inchannel physical or chemical habitat because we were solely interested in relating landscape and riparian conditions to IBI scores, not in explaining mechanisms for those relationships.

Results

Index of biotic integrity

The IBI scores ranged from 5 to 36 (Table 3). In the winter/dry period IBI was lowest at site 125, and highest at site 338 (Fig. 1). During the summer/wet period, high IBI scores occurred at sites 0 and 169, and low scores occurred at sites 50, 271, and 125, with the lowest again at site 125 (Table 3). In the dry season, site 125 had the lowest raw values for six metrics, while in the wet season site 0 had the highest values for four metrics.

Land use and riparian condition

Pasture was the predominant land use in the area, comprising >50% of all site buffers, except for sites 125 and 50, where urban areas predominated (79 and 44%, respectively; Table 4). Secondary forest was greatest at site 271 (15%) commercial forest was greatest at sites 0 and 96 (5% each). Water surface area in tributaries was lowest at sites

Table 2. Scoring riparian and near shore condition (adapted from Barbour et al., 1999)

Acceptable (16–20)	Marginally acceptable (11–15)	Moderately degraded (6–10)	Degraded (0–5)
Width of vegetated zone >20 m. No human impact. Mixture of trees, shrubs and grass. No erosion. Abundant & diverse aquatic vegetation.	Width 15–20 m. Minimal human impact. Mixture of trees, shrubs and grass, with grass and herbaceous common. Little erosion. Low diversity of aquatic vegetation.	Width 10–15 m. Large areas of human impact. Margins largely grass and herbaceous. Erosion evident. Little aquatic vegetation.	Width of vegetated zone <10 m. Human impacts dominant. Little or no woody riparian vegetation. Margins 100% grass or eroding. No aquatic vegetation.

Table 3. IBI metric values and scores (in bold) for the middle Rio Paraíba do Sul[a]

Period	Site	No. native spp.	No. Characiforms spp.	No. Siluriform spp.	% Cyprinodontiforms	No. Sensitive spp.	% Omnivores	% Carnivores	No. Dominant spp.	IBI
Dry/Wet	0	19/26	5/9	8/12	6/0	4/8	65/34	6/7	11/15	
		3/5	**1-/3**	**3/5+**	**5/5+**	**3/5+**	**3/5+**	**1/1**	**3/5**	**22/36**
	50	18/21	7/9	8/7	3/9	0/4	46/81	20/9	13/12	
		1/3	**1/3**	**3/1**	**5/5**	**1−/3**	**5/1**	**5+/3**	**5/5**	**26/24**
	96	18/20	8/8	6/8	1/1	3/4	52/49	4/13	11/10	
		1/3	**3/3**	**1-/3**	**5/5**	**3/3**	**5/5**	**1/3**	**3/3**	**22/28**
	125	15/24	5/11	7/10	95/49	1/5	97/87	< 1/3	1/6	
		1-/5	**1-/5**	**1/3**	**1-/3**	**1/3**	**1-/1**	**1-/1**	**1-/1**	**5/22**
	169	20/27	9/12	8/11	8/10	2/6	85/72	< 1/6	9/14	
		3/5+	**3/5+**	**3/5**	**5/5**	**1/5**	**1/3**	**1-/1**	**3/5**	**20/35**
	271	19/24	7/10	9/11	8/16	2/1	81/86	14/4	8/9	
		3/5	**1/3**	**3/5**	**5/5**	**1/1**	**1/1**	**5/1**	**3/3**	**22/24**
	338	24/25	7/8	12/10	18/26	7/4	66/60	17/15	15/16	
		5/5	**1/3**	**5+/3**	**5/5**	**5/3**	**3/3**	**5/5**	**5/5+**	**34/32**

[a]A combination of 2 pluses or 2 minuses resulted in a 1 point increase or decrease, respectively.

50 and 125 and highest at sites 0, 96 and 338. The lowest buffer quality was at site 125, as indicated by increased urban area and decreased forest. Riparian condition scores ranged from 4 (degraded) at site 125 to 16 and 13 (acceptable) at sites 271 and 338, with little difference between seasons as expected (Table 4).

Relationships between IBI and land use

Correlation strengths between IBI and land use attributes varied with season (Fig. 2). Pasture had a significant positive correlation with IBI during the dry season, while tributary surface area was significant during the wet season ($R^2 > 0.43$ and

Table 4. Percentage of land use in the 8 km^2 buffer, riparian condition (see Table 2) and IBI (dry/wet) of each sampling site in the middle Rio Paraíba do Sul

% Land use							
Site	Secondary forest	Commercial forest	Urban area	Pasture	Tributary water	Riparian condition	IBI
0	0.14	5.27	28.49	58.4	0.99	10/11	22/36
50	0.43	0.00	43.49	47.6	0.59	7/7	26/24
96	6.10	5.15	7.77	69.16	0.90	11/12	22/28
125	2.76	1.39	78.98	5.58	0.37	4/4	5/22
169	3.34	0.08	11.92	72.99	0.88	9/10	20/35
271	14.83	0.00	3.07	68.37	0.71	16/16	22/24
338	1.82	0.00	23.87	56.72	0.90	14/14	34/32

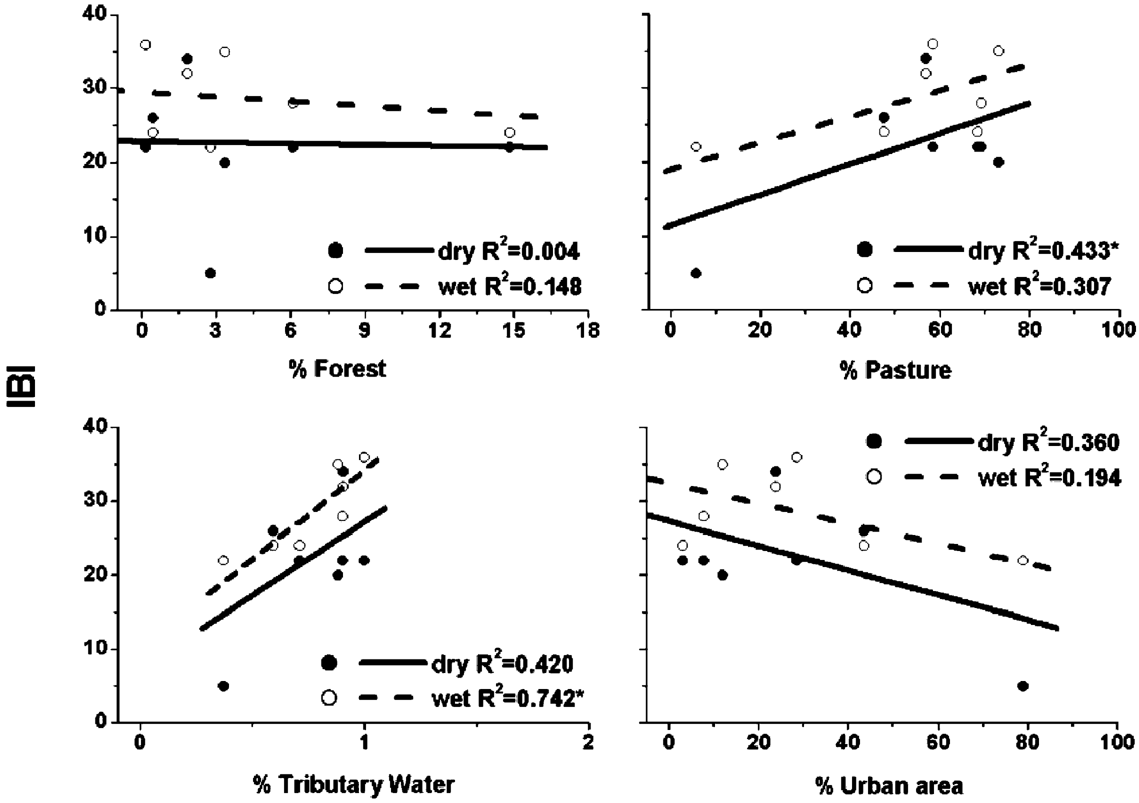

Figure 2. IBI and land use relationships in the middle Rio Paraíba do Sul during summer/wet and winter/dry periods; *indicates statistical significance.

0.74, respectively). Pasture was weakly correlated with IBI in the wet season and tributary surface area was weakly correlated with IBI during the dry season. Urban area had a weak negative correlation with IBI in both seasons ($R^2 < 0.36$), while percent forest had insignificant correlations in both seasons. The effect of the urban and steel manufacturing complex at site 125 was much more evident in the dry season than in the wet season. Riparian condition was positively correlated with IBI in both seasons, with significant values for the winter/dry season only ($R^2 = 0.44$; Fig. 3).

Percent pasture and tributary surface area had significant positive correlations with riparian condition ($R^2 > 0.51$), while urban area had highly significant negative correlations with riparian condition during the wet ($R^2 = 0.76$) and dry ($R^2 = 0.67$) seasons (Fig. 4). Correlations were insignificant between percent forest and riparian condition.

Discussion

The low IBI scores at site 125 confirmed expectations that the large urban and industrial complex at Volta Redonda is strongly associated with environmental alteration along the middle Rio Paraiba do Sul. Large untreated organic and industrial loads enter the river at this site. Low IBI scores associated with organic and industrial effluents have been reported for other large rivers (Hughes & Gammon, 1987; Oberdorff & Hughes 1992; Hugueny et al., 1996; Ganasan & Hughes, 1998; Yoder et al., 2005).

Physical barriers may have influenced the IBI along the reach. Although their fish assemblages should be similar, rheophilic and migratory species cannot pass Funil Reservoir and dam, between sites 0 and 50. Site 0 also receives effluents from upriver, but the reservoir serves as primary treatment by settling pollutants, possibly improving water quality in site 50. On the other hand, riparian condition is slightly lower at site 50 and urban area is higher. A markedly higher summer/wet IBI score (36) at site 0 compared with the score (24) at site 50 vs. comparable scores during the winter/dry season (22 and 26, respectively) indicate that spawning movements during the wet season are restricted. This is further supported by the substantially greater number of migratory siluriform species at site 0 during the wet season, as well as more native and sensitive species compared with site 50 (Table 3, Appendix 1). Conclusions on barrier effects are confounded by poorer riparian condition and increased urbanization (Table 4) at site 50. However, given our knowledge of how dams and reservoirs alter lotic fish assemblages (Bowen et al., 1996; Agostinho et al.,2000; Pringle et al., 2000; Schiemer, 2000; Schmutz et al. 2000; Dieterman & Galat 2004; Quist et al., 2004; Tiemann

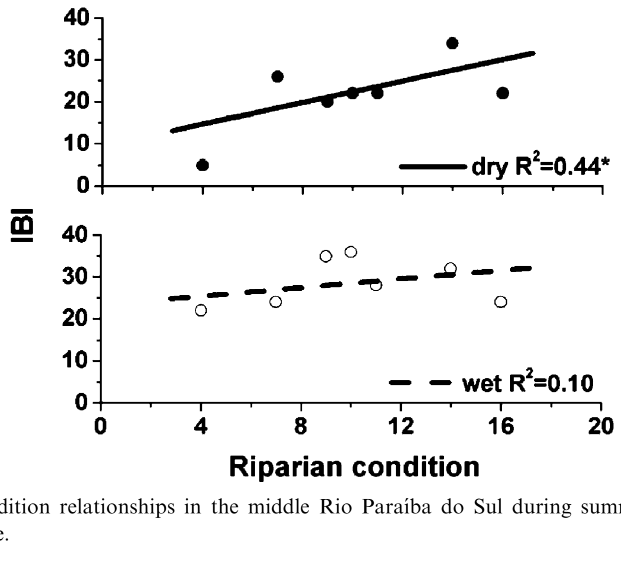

Figure 3. IBI and riparian condition relationships in the middle Rio Paraíba do Sul during summer/wet and winter/dry periods; *indicates statistical significance.

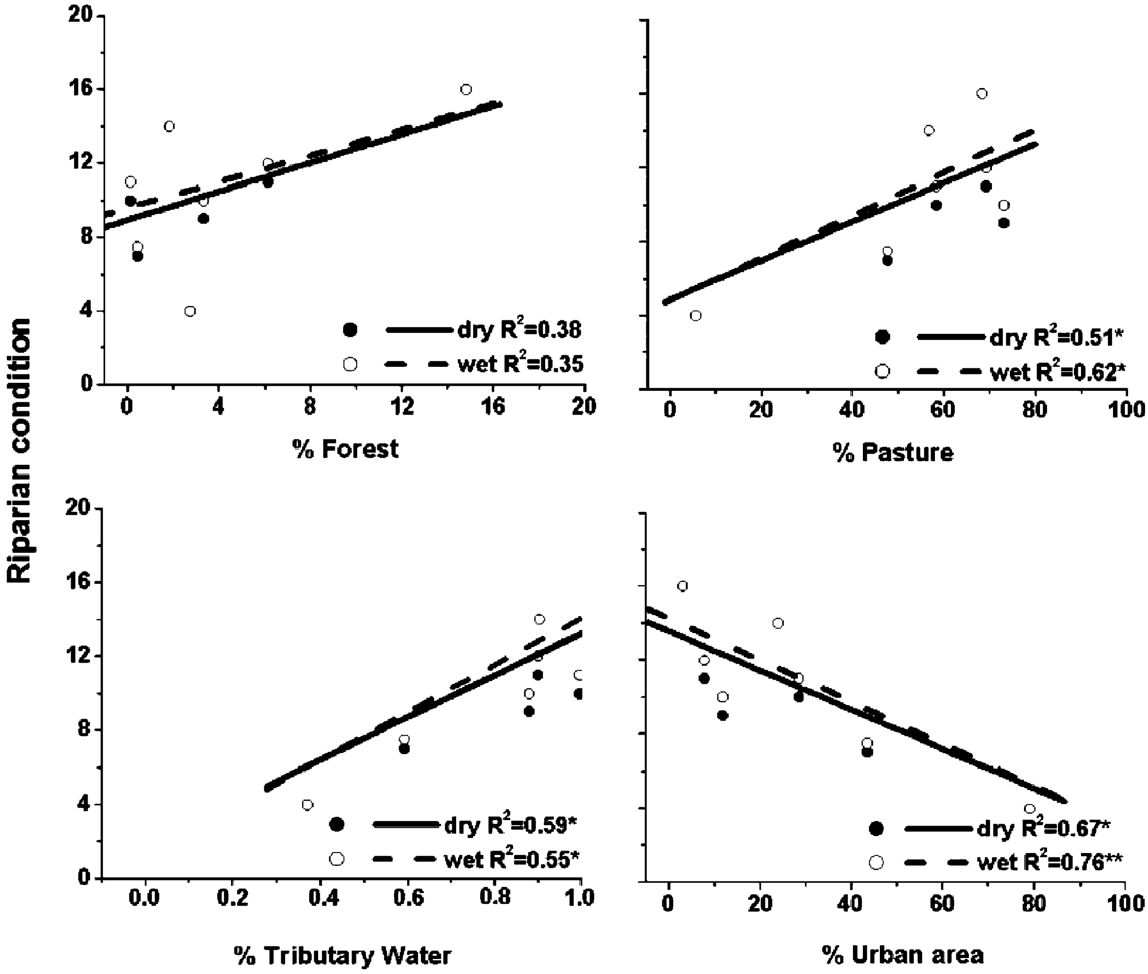

Figure 4. Relationships between riparian condition and land use in the middle Rio Paraíba do Sul, during winter/dry and summer/wet periods; *indicates statistical significance.

et al., 2004; Rinne et al., 2005), we believe further study of dam effects on the Paraiba do Sul is warranted.

Another impact in the studied section is the sand extraction by inchannel dredging which is very common along the river. Dredges were recorded at all sampling sites except km 0 and 271. Dredging causes increased suspended solids and substrate homogenization, margin destruction and erosion, and decreased fish reproduction.

Impact on IBI scores was clearer in the dry period than in the wet period, and IBI scores were lower in the dry season also (Table 3). Lower water levels reduced habitat volume and diversity, and provided less dilution of organic and industrial pollutants, which are associated with poorer environmental quality in the dry period. On the other hand, higher run-off in the summer/wet period increased habitat diversity and water area in the buffer, which was associated with higher IBI values. Grossman et al. (1998) and Bozzetti & Schulz (2004) found that flow variation due to hydrologic periods affected microhabitats and fish assemblage structure. Although our sampling effort per site was the same, fishing gears do not perform equally well in summer/wet and winter/dry periods. Cast nets and seines usually are more efficient at lower water levels because in wet periods shallows are too deep and part of the riparian vegetation is immersed, making access difficult and interfering with cast net and seine fishing. Fish taxa indicating poor water quality, such as

Cyprinodontiformes, *Geophagus brasiliensis* and *Oreochromis niloticus*, are less likely caught in the summer/wet period. Therefore, we agree with Bozzetti & Schulz (2004) that fish sampling to assess IBI in southeastern Brazil should be performed during the winter/dry season, when the ichthyofauna is more comprehensively sampled and the effects of pollutants and poor physical habitat quality are most distinct. Major USA biological assessment programs also recommend the summer low-flow season for sampling fish assemblages (Plafkin et al., 1989; Meador et al., 1993; Peck et al., 2004). However important life history information can be obtained by sampling fish assemblages in multiple seasons (Fausch et al., 2002; Araújo et al., 2003).

The relatively low correlations between IBI and land use attributes are partly due to low sample size ($n = 7$), but interesting patterns are nonetheless evident. Pasture was directly correlated with IBI; it did not negatively affect IBI, as might be expected, despite resulting from forest removal (Fig. 2). This likely occurred for three reasons. First there was a narrow range in forest (0–15%) and thus little variation to explain compared with the wide range in pasture (5–70%). Wang et al. (in press) also reported that landscape variables with narrow ranges correlated weakly with fish assemblages at sites. Second, the other predominant land use in the study was urban (5–80%), meaning the major alternative land use to pasture was urban. More pasture meant less urban land use, and consequently higher IBI scores than with high urban land use. In other words, higher IBI scores were associated with lower urban land use as reflected in greater pasture land use because the two major land uses were inversely related. Wang et al. (2000) observed similar IBI responses as agricultural land decreased relative to urban in the absence of forest. Third, the lowest IBI scores occurred at site 125, which had the least amount of pasture, meaning that high percent pasture would receive higher metric and IBI scores. As expected, increased urban land use correlated with lower IBI scores, as others have reported (Steedman, 1988; Wang et al., 1997, 2000, 2001; Klauda et al., 1998; Snyder et al., 2003).

At the scale of our study, correlation strengths with IBI were similar at riparian and buffer scales, possibly because our riparian zone size (25×800 m) did not differ greatly from our buffer size (1.6×5 km), as opposed to entire catchments of 3200–7000 km^2. The riparian condition score was highest at site 271, which also had the least urban land use, but the IBI score at site 271 was not the highest. Site 271 consistently supported few sensitive species and high percent omnivores, indicating a disturbed aquatic environment and food base (Table 3), regardless of riparian and buffer condition. On the other hand, site 125 had the most urban land use and lowest riparian condition by far, also had the lowest IBI score. Few correlations between IBI, land use, and riparian condition were significant. Those that were only accounted for 43–74% of the variance, leaving considerable variance to be explained by inchannel physical and chemical habitat conditions.

Urban land use and low riparian condition were associated with unbalanced fish assemblages, reflected in low IBI scores. The mosaic of habitat patches, ecotones, and successional stages – the riverscape in all its complexity – is largely responsible for the biodiversity of rivers (Ward, 1998; Fausch et al., 2002; Robinson et al., 2002). In this patch dynamics perspective (Allan, 2004), the interaction between species-specific habitat needs, life histories, and dispersal ability and the ever-shifting temporal and spatial mosaic of river habitats support greater diversity than would occur in unchanging habitats. Thus, both the variety and the variability of habitats are important in influencing the biological diversity of rivers. It is difficult to precisely determine the stressors most affecting fishes in such situations. Large urban areas incorporate industries and are prone to increases in contaminants, suspended solids, nutrients, water temperature, and flow and channel alterations, as well as decreased dissolved oxygen and riparian structure and function. Destruction of riparian vegetation leads to habitat simplification and consequently limits aquatic communities. Steedman (1988), Waite & Carpenter (2000), and Van Sickle et al. (2004) found a similar pattern for catchment agriculture. We suspect that pasture contributes to the preservation of riparian vegetation simply because it was associated with reduced urbanization in our study. A less positive perspective on the benefits of pasture would be expected if we had been able to use reference sites that were naturally forested, such as those employed by Hughes et al. (2004).

Conclusions

We conclude that large-scale assessments of fish assemblage condition in southeastern Brazil rivers are best conducted during the winter/dry season when gear is most efficient, fish most concentrated and least variable, and anthropogenic stressors greatest. Our IBI effectively indicated the effects of a large urban industrial complex, as well as the related differences in land use and riparian condition. A positive relationship between pasture and IBI occurred because increased pasture covaried with decreased urbanization and because pasture dominated at least disturbed sites in the absence of natural forest. A better understanding of how land use and riparian condition affect IBI scores requires quantitative evaluation of physical and chemical habitat at a large set of sites.

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Appendix

Appendix 1. Classification of fish species and numbers collected from the Rio Paraíba do Sul, 2001/2002, by site and season (winter/dry & summer/wet). Species listed according to Lauder & Liem (1983).

Order/Species	Classification				Sites (Dry/Wet)						
	Trophic group	Sensitivity	Micro habitat	Origin	0	50	96	125	169	271	338
Characiforms											
Astyanax bimaculatus	O		WC		51/36	2/17	7/7	3/5	13/14	3/137	40/18
Astyanax parahybae	O		WC		22/28	2/29	23/1	4/3	6/16	30/151	18/3
Astyanax giton	O		WC			0/4		0/1	37/16	1/13	0/19

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Appendix 1. (Continued)

Order/Species	Classification				Sites (Dry/Wet)						
	Trophic group	Sensitivity	Micro habitat	Origin	0	50	96	125	169	271	338
Astyanax scabripinnis	O	S	WC					0/1	0/1		
Astyanax sp 1	O		WC		0/9	1/0		0/1	0/5	3/8	11/3
Astyanax sp 2	O		WC		1/0	1/9	6/2		1/0	0/4	
Deuterodon sp	INV		WC						7/4		0/3
Hyphessobrycon bifasciatus	INV	S	S			0/3			0/1		
Hyphessobrycon luetkeni	INV		S							3/0	
Hyphessobrycon reticulatus	INV	S	S						0/1		
Hyphessobrycon callistus	INV		S	Alien	5/28	12/1	23/23	0/2	0/4	2/69	
Brycon sp	H		S					0/1			
Colossoma sp	IL		WC	Alien			0/1				
Probolodus heterostomus	O		WC		0/2	0/8	3/2	17/4	2/0		
Oligosarcus hepsetus	C		WC		5/1	10/14	2/27	23/14	1/13	39/35	18/18
Hoplias malabaricus	C		WC		4/4	3/0	4/10	1/1	0/2	0/7	4/2
Hoplerythrinus unitaeniatus	C	S	WC		1/2			0/2			
Prochilodus lineatus	IL		WC							0/1	6/3
Cyphocharax gilberti	IL		WC		0/2		1/0		1/1	0/2	10/30
Leporinus copelandii	H		WC		0/1	3/6	4/16	2/4	2/1	0/3	10/0
Leporinus sp.	H	S	WC		0/1			0/1			
Siluriforms											
Glanidium albescens	O	S	B				4/0				2/0
Trachelyopterus striatulus	INV	S	B						0/2	0/4	11/3
Pimelodus maculatus	O		B		43/16	4/52	19/39	17/4	1/0	5/0	0/1
Pimelodus fur	O		B		1/3	0/1		0/14	34/64	28/39	19/1
Pimelodella sp.	O	S	B								5/0
Rhamdia parahybae	C		B		0/1	1/0		0/2	0/1	5/0	
Rhamdia sp 1	C		B					15/0	0/2	1/6	4/0
Rhamdia sp 2	C		B			1/0		0/3		0/2	1/1
Callichthys callichthys	IL		B			0/3				0/3	
Corydoras nattereri	IL		B		4/51				8/22	1/0	3/4
Hoplosternum littorale	IL	S	B		57/37	2/1	0/6	3/13	0/7	6/11	3/4
Hypostomus affinis	IL		B		3/11	2/4	12/24	47/25	4/10	4/7	3/4
Hypostomus luetkeni	IL		B		12/27	1/1	42/40	11/5	2/16	0/5	23/0
Hypostomus sp	IL		B		0/1					0/1	
Harttia loricariformis	IL	S	B		10/25		0/2		2/0		
Rineloricaria sp	IL		B		0/14	4/8	11/7	111/30	14/3	1/3	0/6
Gymnotiforms											
Gymnotus cf. carapo	INV		WC		2/2	1/0	1/1	0/2	0/5	0/2	2/3
Eigenmannia virescens	INV	S	WC		0/6		0/3	1/3	2/4	2/0	3/2
Cyprinodontiforms											
Phalloceros caudimaculatus	O		S			1/7			5/34	21/170	68/1
Poecilia reticulata	O		S	Alien	19/0	1/16	3/4	7455/420	18/3	6/15	0/63
Xiphophorus heller	O		S	Alien				2/0			
Perciforms											

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Appendix 1. (Continued)

Order/Species	Classification				Sites (Dry/Wet)						
	Trophic group	Sensitivity	Micro habitat	Origin	0	50	96	125	169	271	338
Cichla monoculus	C		W	Alien	0/7						2/0
Crenicichla lacustris	INV	S			8/4	0/1	2/1		0/3	4/13	24
Cichlasoma facetus	O	S			1/0						0/2
Geophagus brasiliensis	O				16/16	21/50	30/71	44/102	22/99	78/38	8/5
Tilapia rendalli	O			Alien	48/6	1/21	16/30	16/142	109/2	113/238	19/17
Oreochromis niloticus	O			Alien	5/0		1/2	1/3	0/8	2/46	1/5
O. hornorum X O. niloticus	O			Alien	5/3	0/2	0/1	68/45	2/4	0/162	42/10
Pachyurus adspersus	C	S			0/1	0/8	1/3				10/4
Synbranchiforms											
Synbranchus marmoratus	C	S	WC		1/3	0/1					

Trophic group: omnivore (O), invertivore (INV), herbivore (H), iliovore (IL), carnivore (C). Sensitivity: Sensitive species (S). Microhabitat: water column (WC), surface (S), benthic (B).