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Influence of watershed land use and riparian characteristics on biological indicators of stream water quality in southeastern Brazil

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ABSTRACT

Land use changes and riparian corridor degradation strongly influence the water quality of low-order streams at different spatial scales. Stream water quality can be analyzed by chemical and biological indicators, but which are generally evaluated separately. Here, we evaluated if anthropogenic alterations at distinct spatial scales (estimated by land use, riparian zone composition, and riparian forest structure) influenced stream water quality according to chemical and biological (e.g., fish and macroinvertebrate) indicators in a tropical rural landscape (SE Brazil). We found higher total nitrogen concentrations and electric conductivity in streams with land use dominated by pasture and narrower riparian forests, higher diversity of macroinvertebrates and dissolved oxygen concentrations in streams with higher cover and width of riparian forests although land use was dominated by sugarcane, whereas fish diversity was related with natural vegetation cover at the watershed scale. Thus, stream water quality indicators responded to variation at both scales studied, with interactions between land use and riparian corridor characteristics. These results suggest that different degradation drivers may account for variation of different types of indicators (chemical vs. fishes vs. macroinvertebrates) at distinct spatial scales.

1. Introduction

Land use changes such as deforesting for conversion to agriculture and pasture areas are strong drivers that impact stream water quantity and quality in watersheds (Allan, 2004; Woodward et al., 2012). Therefore, the evaluation of water quality and ecological integrity of waterbodies is important for watershed monitoring and management (Chambers et al., 2012; Villeneuve et al., 2015). These evaluations, however, are complex, because the effects of anthropogenic changes can occur at different spatial scales, from the landscape scale to local disturbance effects (Stewart et al., 2001; Tran et al., 2010).

Changes on soil surfaces at the watershed scale can influence the flow of water and nutrients to the waterbodies, resulting in impacts on stream water quality (Quinn et al., 1997; Goldstein et al., 2007; Clapcott et al., 2012), due to increased sedimentation, nutrient and pollutant concentrations, and modified hydrological patterns (Harding et al., 1998; Allan, 2004; Neill et al., 2011). However, these effects can be modulated by the riparian corridor at local scales, so that the characteristics of the riparian zone influence surface runoff, streambank erosion, sedimentation, nutrient and pollutant transport, as well as environmental conditions for the aquatic biological communities (Sponseller et al., 2001; Naiman et al., 2005; Dosskey et al., 2010). Several studies found that both factors at the watershed scale and at the riparian corridor scale can influence structural (Stewart et al., 2001; Goforth and Bain, 2010; Tran et al., 2010) and functional variables (Sponseller and Benfield, 2001; Silva-Junior et al., 2014; Tanaka et al., 2015a) of lotic ecosystems.

At the riparian corridor scale, differences in cover of the dominant vegetation such as the proportion of forest cover relative to other vegetation types can influence stream nutrient concentrations, physical characteristics, and energy balance (Osborne and Kovacic, 1993; Tabacchi et al., 1998; Dosskey et al., 2010; Casatti et al., 2012; Jackson et al., 2014). Further, the local structure of riparian forests can influence the availability of coarse particulate organic matter (Paula et al., 2011), leaf breakdown rates (Tanaka et al., 2015a), and characteristics of stream water (Souza et al., 2013; Fernandes et al., 2014). For example, Stewart et al. (2001) found that the vegetation type, riparian forest fragmentation patterns, and forest cover at larger spatial scales were correlated with higher quality of the biological component of streams, such as higher diversities of aquatic communities, and higher proportion of fish and macroinvertebrate species intolerant to degradation.

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Nislow and Lowe (2006) found that streams with more recent riparian forests, with higher tree densities and smaller canopy cover had higher abundances of macroinvertebrates (mainly grazers) and fish (trout) than streams with older forests, where a higher proportion of shredders were found. Souza et al. (2013) and Fernandes et al. (2014) found effects both at the riparian zone scale, and at a finer spatial scale, of riparian forest structure on stream water quality in agricultural landscapes. Thus, both land use changes at the watershed scale and at the riparian corridor scale can influence stream water quality and quantity, although the relative contribution of factors operating at each scale is not well established yet (Wahl et al., 2013).

The evaluation of anthropogenic effects on stream water quality at different spatial scales also depend on the water quality indicators considered. In fact, acute effects on physical and chemical characteristics of the water may not persist long enough to detect their impact on biological organisms, so that biological indicators that can integrate effects at longer time scales are necessary to detect environmental impacts (Karr, 1981; Metcalfe-Smith, 1994; Marchant et al., 2006). For example, indicators related to fish (Paller et al., 2000; Bojsen and Barriga, 2002; Casatti et al., 2006) and macroinvertebrate communities (Bailey et al., 2004; Bonada et al., 2006; Death and Collier, 2010) have frequently been used to detect impacts on streams related to land use changes and deforestation of riparian forests. Recent studies comparing different groups of organisms used for biomonitoring studies found that there may be correlation among groups on their ability to detect environmental impacts, but this correlation can vary depending on river characteristics (Pinto et al., 2006). Also, distinct groups of organisms can differ in their relative tolerance to the environmental conditions found in degraded areas, and therefore in their ability to discriminate impacted areas from reference ones, as observed for different functional groups of fish, bryophytes, and diatoms (Paavola et al., 2003; Passy et al., 2004; Newall et al., 2006; Miserendino et al., 2011). Therefore, it is not always possible to extrapolate the results of a classification based on a single group of organisms to other groups. However, some studies found that combinations of different groups of organisms and biological indicators can result in higher precision in the evaluation of environmental impacts (Marchant et al., 2006; Mueller et al., 2014), since different types of organisms can respond to distinct stressors, or present different responses depending on the spatial scale considered (Dolph et al., 2011; Clapcott et al., 2012; Zuellig et al., 2012; Villeneuve et al., 2015). More precise methods of biomonitoring can then be used in risk predictive models, to support decisions on sustainable development strategies and management (Potter et al., 2004).

In this way, the present study aimed to evaluate if anthropogenic changes at different spatial scales (watershed, reach) influenced stream water quality in an agricultural landscape according to chemical and biological (fish, macroinvertebrates) indicators. We estimated variables related to land use, riparian zone composition, and riparian forest structure to evaluate at which scale and which factors influenced chemical and biological aspects of stream environmental quality in a tropical region.

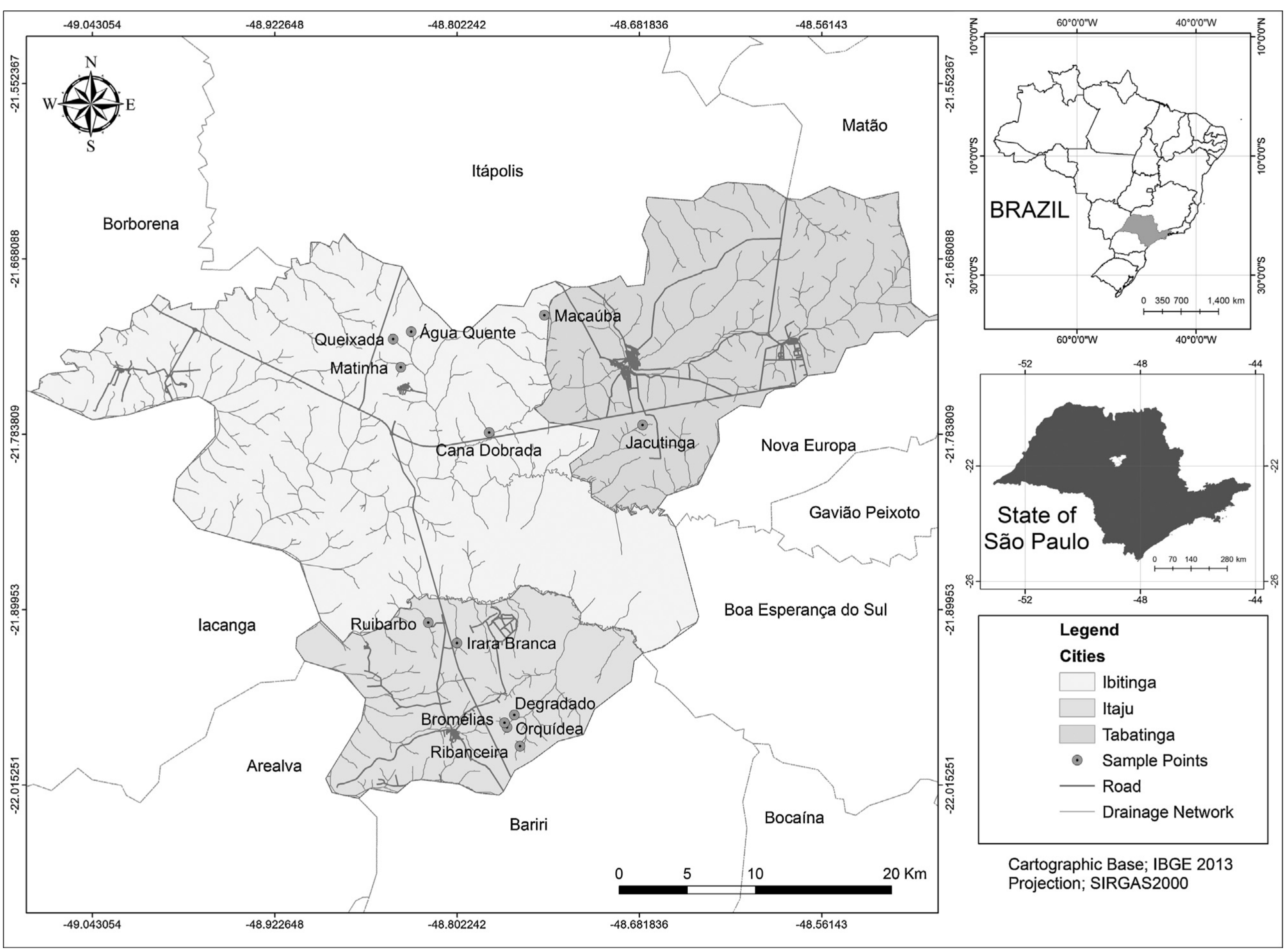

Fig. 1. Location of the studied streams in the Jacaré-Guaçu (upper streams) and Jacaré-Pepira (lower streams) watersheds in southeastern Brazil.

2. Methods

2.1. Study area

This study was carried out in low-order streams draining to the watersheds of Jacaré-Pepira and Jacaré-Guaçu rivers, both located in the central region of São Paulo State (Fig. 1). Both r ivers drain to the Tietê River, and their watersheds are in the Tietê-Jacaré Water Resources Management Unit (UGRHI-13) of São Paulo State. The Jacaré-Pepira River watershed is located between the coordinates 21°55′ and 22°30′S, 47°55′ and 48°55′W, extending throughout an area of 2612 km^2, and presenting a drainage density of 0.95 km/km^2 (Maier, 1983). The river flows from São Pedro city, at an altitude of 960 m amsl and, after 174 km reaches the Tietê River next to Ibitinga Reservoir, at an altitude of 400 m amsl. Part of the river's course is in the Cuestas Basálticas, which is characterized by steep terrain followed by large structural platforms of softened relief, whereas the final course of the river is in the Eastern Plateau (Maier, 1983). The Jacaré-Guaçu River watershed is located between the coordinates 21°37′ and 22°22′S, 47°43′ and 48°57′W, extending through an area of 4108 km^2 and drainage density of 0.88 km/km^2 (DNAEE/EESC, 1980). The headwaters are located in the cities of São Carlos and Itirapina, about 1040 m amsl, flowing 148 km until the Ibitinga Reservoir, in the Tietê River (IPT, 2000). Land use in the UGRHI-13 is predominantly for cultures of sugarcane, orange, *Pinus*, and *Eucaliptus*, with only 11.3% of remnant native vegetation; about 60% of mean water discharge is used for irrigation and other agricultural use (Tundisi et al., 2008).

The studied region comprised streams located in three municipalities (Ibitinga, Itaju, and Tabatinga), with hot and dry tropical climate (Aw) following Köppen's classification, with monthly mean temperatures between 19.3 and 25.4 °C, and mean annual precipitation of 1260 mm, concentrated between October and March (Miranda et al., 2012). Sampling was carried out in 2008 and 2009, with six streams in each watershed (Fig. 1): Água Quente, Cana Dobrada, Jacutinga, Macaúba, Matinha, Queixada (Jacaré-Guaçu), and Bromélia, Degradado, Irara-Branca, Orquídea, Ribanceira, Ruibarbo (Jacaré-Pepira).

2.2. Sampling

In each stream, we selected a 100 m long reach to sample the communities of fish and macroinvertebrates, and to characterize the riparian forest structure. Reach size was about 40 times the mean stream width (Freund and Petty, 2007) (Table 1).

Table 1
Range of stream environmental quality variables grouped by macroinvertebrates, fish, and chemical characteristics of the studied streams in southeastern Brazil.

Variable	Code	Range
Macroinvertebrates		
Shannon diversity index	H'_m	0.55–2.22
Biological monitoring working party index	BMWP	20–165
% Ephemeroptera, Plecoptera, Trichoptera	%EPT	0–0.53
Fish		
Shannon–Wiener diversity index	H'_f	0–1.88
Index of biological integrity	IBI	0–2.6
Chemical chacteristics		
Total nitrogen (mg l^{-1})	TN	0.776–1.905
Total phosphorus (mg l^{-1})	TP	0.015–0.094
Dissolved oxygen (mg l^{-1})	DO	3.60–9.12
Electrical conductivity (mS cm^{-1})	EC	0.013–0.103
pH	pH	6.01–7.28

Land use characterization was obtained by on-screen digitizing a LandSat TM-5 (221/75) image from 15 July 2008 (1:50,000), attributing a pixel to each land use, as proposed in the Brazilian Institute of Geography and Statistics (IBGE) technical manual on land use (IBGE, 2013), where vetorial training areas were created at a second layer with the software ArcGis 10.2.2. The drainage net and topography were described by digitizing topographic charts from IBGE (1:50,000). For each stream, we delimited the watershed from the downstream end of the studied reach and by the ridges of the drainage areas, to determine land use for each stream. The proportion of land occupied by the following uses was obtained: sugarcane plantation, orange plantation, forest, urban areas, and roads. Urban areas and roads were combined into a single variable (% urban) due to their low incidence in the studied watersheds. These variables were previously analysed for correlations by Tanaka et al. (2015b), and the only correlation found was between sugarcane and orange plantations (Pearson's correlation coeffiecient, $r = -0.725$, $P = 0.008$), so only sugarcane cover was used in the subsequent analyses due to its higher incidence in the watersheds studied.

The riparian forest structure and distribution was obtained by randomly establishing three 100 m^2 plots adjacent to the stream channel in each studied reach. In each plot, we estimated cover by vines and measured all trees with diameter at breast heigh (dbh) > 5.0 cm. Measurements included total tree height (estimated with a laser hypsometer) and dbh (estimated from the circumference measured with a measuring tape). From these measurements, the following variables were estimated for each reach: mean tree height, mean dbh, total basal area, tree density, and vertical canopy structure (a relative index estimated by the coefficient of variation of tree heights). For the analyses, we used mean values calculated for each stream. We also measured riparian forest width at each plot position, and used mean values for each stream in the analyses. These data were part of a larger study, presented in Souza et al. (2013).

To evaluate stream water quality, the following variables were determined with an YSI 556 multiparameter system: dissolved oxygen concentrations, electrical (specific) conductivity, and pH. We also collected two surface water samples at each location and sampling dates to determine total phosphorous and nitrogen concentrations, following standard methods (Koroleff, 1976; Mackereth et al., 1978); the mean values were used in the analyses.

Macroinvertebrates were sampled with a Surber sampler (area = 0.09 m^2, mesh size = 250 μm). Three samples were randomly obtained from each reach and transported to the laboratory, where macroinvertebrates were separated manually on a transilluminated white tray, and later fixed in ethanol 70%. Macroinvertebrates were identified at family level (except for Oligochaeta) following Froehlich (2007). Several indices of stream biological integrity based on macroinvertebrates were proposed (Metcalfe-Smith, 1994; Bonada et al., 2006; Baptista et al., 2007). Among these indices, Tanaka et al. (2015b) found high correlation for the studied region, so the following indicators were used: Shannon diversity index (related to community structure), proportion of EPT (Ephemeroptera, Trichoptera and Plecoptera) (related to community composition), and the adapted Biological Monitoring Working Party (BMWP) index, following Alba-Tercedor and Sánchez-Ortega (1988) and Junqueira et al. (2000), which weights the occurrence of macroinvertebrate families relative to their tolerance to pollution.

Fish were collected with a LR-24 electrofisher (Smith-Root, Inc.). Each sampled reach was delimited with nets with 5 mm mesh, and two surveys were carried out. Fish caught along the first survey were stocked in 5 gallons buckets filled with stream water and, after the second survey, all fish caught were fixed in formaldehyde 10%, and deposited in the fish collection of the

Ichtiology and Systematics Laboratory from Federal University of São Carlos under register numbers LISDEBE 3926-4045. Data from this study were part of a larger study on fish communities in the region (Nassin et al., unpublished data), and were previously analysed for correlations (Tanaka et al., 2015b). Variables related to the structure and diversity of species and functional groups were highly correlated, so we selected two variables as fish indicators: Shannon diversity index and an Index of Biological Integrity (IBI) recently proposed by Casatti et al. (2009) for streams in western São Paulo state.

2.3. Data analysis

Riparian patterns of forest structure were determined with a Principal Components Analyses following Souza et al. (2013), and included the variables tree density, basal area, tree height, diameter at breast height (dbh), vertical canopy structure, percent of vine cover, and riparian forest width. Data were previously standardized before analyses. The extracted axes represent patterns in riparian forest structure and were used in subsequent analyses.

There were two scales of independent variables, which included watershed land use and riparian forest variables, and three categories of stream water quality (dependent variables) that included chemical variables and community descriptors of macroinvertebrates and fish. To evaluate possible redundancies in information both in dependent and independent variables, we first calculated Pearson correlations between pairs of dependent variables, and then between pairs of independent variables. Variables with significant correlations ($P < 0.05$) were removed, so that the fewest dependent and independent variables were selected for the analyses.

We evaluated the influence of watershed land use and riparian forest structure variables on stream chemical and biological indicators with redundancy analysis (RDA), a linear method of direct gradient analysis, which is the canonical form of PCA (Jongman et al., 1995). RDA is the direct extension of multiple regression to model multivariate response data, constraining the ordination of dependent variables to linear combinations of the independent variables (Legendre and Legendre, 2012). Therefore, it generates a bidimensional space which allows the evaluation of the association of the higher values of the dependent variables (points in the graph) with the independent variables (vectors). All variables were checked for normality and transformed when necessary, using angular transformations for proportions and natural logarithms for concentrations and abundances. We used a stepwise procedure to reduce the number of independent variables, and the significance of the final model was verified using a Monte Carlo permutation test (999 permutations). Univariate analyses were carried out with Systat 13.1 software, whereas multivariate analyses were carried out in R (R Core Team, 2014), using the “vegan” package (Oksanen et al., 2013).

3. Results

Three streams of the twelve sampled presented riparian zones completely deforested, whereas the remaining represented a gradient in riparian forest cover. The first two axes of the PCA on riparian forest structure explained 80.8% of the variation. The first axis (hereafter referred to as FOR1, eigenvalue = 4.577) explained 65.4% of the variation (Fig. 2), representing a gradient from deforested reaches or with degraded riparian forest to reaches with larger riparian forest widths (0.838), higher trees (0.969), dbh (0.912), basal area (0.893), tree densities (0.599), and vertical canopy structure (0.930). The second axis (FOR2, eigenvalue = 1.077) explained 15.4% of the variation, ordering reaches with higher vine cover (−0.948) to reaches with higher basal area (0.335), representing, respectively, forests in early development and older, more mature forests (Fig. 2).

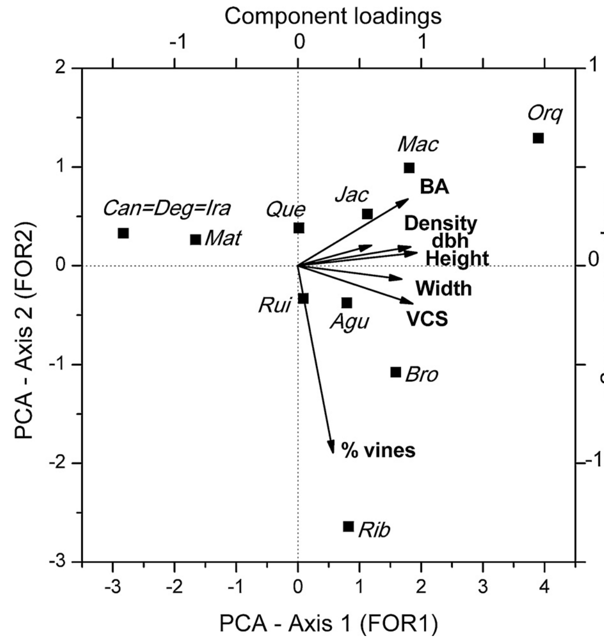

Fig. 2. Results of Principal Components Analysis on riparian forest structure variables relative to the studied streams. Agu = Água Quente, Bro = Bromélia, Can = Cana Dobrada, Deg = Degradado, Ira = Irara Branca, Jac = Jacutinga, Mac = Macaúba, Mat = Matinha, Orq = Orquídea, Que = Queixada, Rib = Ribanceira, Rui = Ruibarbo. Symbols are described in Table 2.

Drainage area in the studied watersheds varied between 0.86 and 5.49 km^2, and land use was predominantly for agriculture, mainly by cultures of sugarcane and orange. The area occupied by both cultures varied between 60.1 and 96.0% (Table 2), with the exception of Matinha (28.1%), where cover by urban structures dominated (58.1%). The cover by forests varied between 4.0% (Orquídea) and 26.8% (Ribanceira); in Orquídea, the remaining 96% of the watershed were covered by sugarcane culture and, nevertheless, the riparian forest studied presented the highest values for tree heights (mean = 11.2 m), dbh (29.4 cm), and vertical canopy structure (0.557). Watershed forest cover was negatively correlated with FOR2 ($r = -0.757$, $P = 0.004$), so we only used forest cover in subsequent analyses. Although cover by pastures was negatively correlated with FOR1 ($r = -0.630$, $P = 0.028$), the correlation was relatively low when considering criteria for

Table 2
Range of watershed and riparian forest variables of the studied streams in southeastern Brazil.

Variable	Code	Range
Riparian characteristics		
Tree height (m)	Height	3.1–11.2
Diameter at breast height (cm)	dbh	6.1–29.4
Basal area ($m^2\,ha^{-1}$)	BA	0.2–85.8
Density (ind ha^{-1})	Density	66.7–4233.0
Vertical canopy structure	VCS	0–0.56
Vine cover (%)	% vines	25–91.7
Riparian forest width (m)	Width	0–50
Watershed characteristics		
Drainage area (km^2)	Area	0.86–5.49
Forest (%)	% forest	4.0–26.8
Citrus (%)	% citrus	0–80.6
Sugarcane (%)	% sugarcane	6.1–96.0
Pasture (%)	% pasture	0–24.9
Urban (%)	% urban	0–58.1

redundancy in biomonitoring analyses ($r > 0.800$; Hering et al., 2006). Thus, both cover by pastures and FOR1 were used in subsequent analyses. The remaining correlations evaluated were not significant, so they were retained in the model.

Some water quality variables were correlated to each other. Concentrations of TP were positively correlated with TN ($r = 0.851$, $P = 0.002$) and EC ($r = 0.725$, $P = 0.018$), whereas pH was positively correlated with TP ($r = 0.655$, $P = 0.029$) and EC ($r = 0.651$, $P = 0.030$). Further, the Shannon diversity index estimated for macroinvertebrates (H'_m) was positively correlated with TP ($r = 0.667$, $P = 0.035$) and EC ($r = 0.657$, $P = 0.039$), but the values were relatively low according to Hering et al. (2006). The other variables did not present significant correlations between each other. Therefore, the following stream water quality and biological integrity variables were used in the RDA: DO, TN, EC, H'_m, BMWP, %EPT, H'_f and IBI, that presented large variation among streams (Table 1).

The RDA model explained 61.0% of the variation in stream chemical and biological indicators in relation to watershed and riparian forest structure, in a significant relationship between dependent and independent variables ($P = 0.035$). The first two axes of the RDA explained 75.5% of this variation. The first axis explained 43.6% of this variation (eigenvalue = 2.162) and separated streams with dominance by pasture and sugarcane culture from those with higher forest cover and urban structures (Fig. 3). Also, it separated the water quality variables and macroinvertebrate indicators (which occurred in watersheds with land use dominated by anthropogenic activities) from fish indicators (H'f and IBI). The orthogonal proximity of the points H'f and IBI to higher values of forest cover indicates their positive relationship with higher forest cover in the watersheds (Fig. 3). The second axis explained 31.8% of the variation (eigenvalue = 1.577), separating streams with land use dominated by sugarcane, some of them with better developed riparian forests, from streams dominated by pastures and urban structures. The indicators of higher stream water quality (DO, BMWP, %EPT, and H'_m) were associated to watersheds dominated by sugarcane culture and better developed riparian forests (FOR1), whereas indicators such as TN and EC were associated to watersheds dominated by pastures, where riparian forest width was smaller, with degraded or poor developed riparian forests (Fig. 3).

4. Discussion

Land use in the studied region is predominantly for agricultural acitivities, with dominance by pasture and cultures of sugarcane and orange. However, only pasture cover was negatively correlated with the variable FOR1 (representative of a forest gradient), suggesting a trend for watersheds with pasture in this region to have smaller riparian forest widths and cover, or even deforested riparian forests. In tropical regions, pasture areas can extend to stream margins, with negative results to water quality (Thomas et al., 2004; Wyman and Stein, 2010). In our study, streams in watersheds with higher pasture cover presented higher values of electric conductivity (EC) and TN concentrations, suggesting that the absence or reduction of riparian forests enabled higher nutrient loading to these streams. Neill et al. (2001) found higher concentrations of total suspended solids and organic N in deforested streams draining watersheds dominated by pasture in Amazon streams, but higher concentrations of dissolved inorganic N in forested streams, resulting in low difference in TN concentrations between both stream types. Sponseller et al. (2001), in a study carried out in North America, found higher variation in total inorganic N concentrations at the watershed than at the riparian corridor scale. In a previous study in the same region of our study, Souza et al. (2013) found no effects of riparian zone or riparian forest structure characteristics on TN concentrations, although they found higher EC values on deforested streams or with higher cover by grasses. Therefore, our results suggest that land use changes at the watershed scale can have more influence on TN concentrations than riparian zone characteristics (Fig. 3).

Other indicators of stream water quality evaluated in our study responded to distinct predictors, both at the watershed and at the reach spatial scales. Higher values of indicators related to macroinvertebrate communities and DO concentrations were associated with streams with wider riparian forests, with larger trees and values of vertical canopy structure, even though they were in watersheds dominated by sugarcane culture, as indicated by the effect of FOR1 in the analysis of the second RDA axis. These analyses suggest that despite high cover by sugarcane culture found in these watersheds, these indicators of stream water quality responded more to local effects of the riparian zone, which can

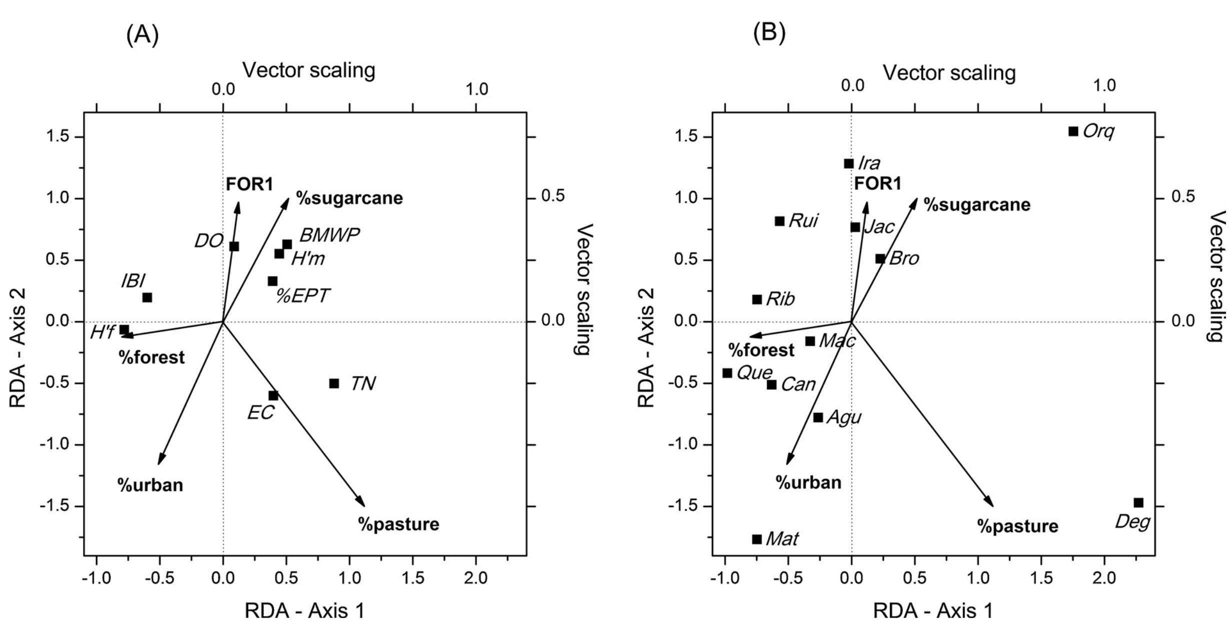

Fig. 3. Biplot of stream water quality variables (A) and sampling sites (B) in relation to the first ordination axes of RDA based on land cover percentages and riparian forest structure variables. Symbols are described in Tables 1 and 2.

alleviate the effects of land use by agricultural activities. Higher influence of the riparian zone in relation to watershed land use was also found by Sponseller et al. (2001), who found that indicators of macroinvertebrate assemblages responded only to land use changes at a 200 m scale of the riparian corridor, and by Tran et al. (2010), who found higher values of macroinvertebrate indicators in streams with higher DO concentrations, mainly wthin a buffering zone of 200 m. Also, Lammert and Allan (1999) found that land use adjacent to streams were better indicators than land use at a regional scale, but with different effects of local factors influencing fish (flow variability and adjacent land use) and macroinvertebrate (dominant substrate) communities.

On the other hand, Ligeiro et al. (2013) found local and watershed effects on EPT richness in a watershed with higher cover by agriculture, whereas in a watershed with land use dominated by pasture only effects at the watershed scale influenced this variable. Marzin et al. (2013) found similar contributions of the scales of reaches (10–11%), riparian corridor (10%) and watershed (11%) on the variation of macroinvertebrate community composition in streams studied in France, whereas for fish communities the contribution of variation at the reach scale was much higher (20–21%) than at the scales of the riparian corridor (6%) or the watershed (5%). Therefore, effects at distinct spatial scales can influence differently the biological communities in degraded regions.

The indicators of fish community quality, in our study, were more associated to watersheds with higher cover by forests. Tropical fish communities can respond to stressors both at the watershed scale and at the mesohabitat scale within streams (Casatti et al., 2009; Casatti and Teresa, 2012), with higher effects on taxonomic and functional diversities at the among-stream scale, and stronger effects on functional composition within streams (Teresa and Casatti, 2012). Differential input of resources (e.g., light, nutrients, organic carbon) is important both at the riparian and watershed scales (Bojsen and Barriga, 2002), but diferences in land use adjacent to the stream channel can strongly influence fish community composition (Lammert and Allan, 1999; Zeni and Casatti, 2014), by degradation of both physical and chemical conditions. On the other hand, Zuellig et al. (2012) found that indicators based on fish communities were strongly influenced by among-stream differences, both in undisturbed and developed watersheds, with weaker effects within streams.

5. Conclusions

These results suggest that, at larger spatial scales, differences between watersheds can influence the biological communities, but that combined effects of environmental changes at the riparian corridor scale can determine the final composition of these communities (e.g., Marzin et al., 2013). Recent studies indicate that stream water quality and ecosystem function variables can also vary at fine spatial scales (Dosskey et al., 2010; Souza et al., 2013), even within a riparian forest remnant (Fernandes et al., 2014; Tanaka et al., 2015a), and therefore differences in environmental conditions and resource availability at different spatial scales must influence the biological communities. Thus, the evaluation of different types of biological communities, together with chemical variables, can contribute to identify different types of stressors in watersheds, since each type of biological community can respond to distinct environmental impacts at different spatial scales. More studies are necessary to understand the mechanisms that influence these communities in streams with distinct land uses at different spatial scales, to predict stream water quality in these conditions.

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