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Contrasting land uses affect Chironomidae communities in two Brazilian rivers

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With 5 figures and 2 tables

Abstract: We investigated whether the conversion of riparian land use from indigenous forest to pasture influences Chironomidae (midge) community composition in two Brazilian rivers. Our study was conducted in the main rivers (São José dos Dourados 6th order, Aguapeí 7th order) of two large catchments in the State of São Paulo. Both are dominated by agricultural ecosystems but retain substantial patches of native riparian forest. Replicated artificial substrates (baskets filled with coarse clay gravel) were exposed at one forested reach and one pasture reach in each river. Each respective land use had to be present for at least 500 m along both river banks above and alongside the study reach, and the riparian forest at the forested reaches had to be at least 50 m wide. Colonisation baskets were sampled after 44 days and all midge larvae sorted and identified to the lowest level possible (a total of 5,286 individuals belonging to 27 taxa). Land use affected midge communities, but not as expected. Densities of three dominant midge genera were significantly higher in pasture reaches than in forested reaches, possibly in response to moderate nutrient enrichment at the pasture reaches. Total midge density and taxon richness were similar across land uses. These results imply that land-use related changes in habitat conditions were not severe enough to represent a stressor (with negative effects) for the midge communities in the investigated large rivers, but rather acted as a subsidy, with mainly positive effects.

Key words: Chironomidae, land use, feeding behaviour, community structure, rivers, Brazil.

Introduction

Human land use practices affect stream and river ecosystems worldwide, and the intensity of land use has increased dramatically in many regions of the world in recent decades (Allan 2004). In Brazil, changes in land use and a reduction in forest cover began soon after it's "discovery" by Europeans, and logging, mining, cattle farms, coffee and sugar cane plantations are the main activities causing this deforestation (Gonçalves 1997). Deforestation has also affected the riparian zones of many streams and rivers in Brazil (Azevedo 2000, Ditt 2000).

Because riparian forests represent an ecotone region linking terrestrial and aquatic ecosystems, their role as a buffer zone protecting the integrity of both systems and their respective biotic and abiotic components is broadly recognized (Dudgeon 1999, Allan 2004). Several studies have found a positive relationship between the presence of riparian forest and the water quality of both lentic and lotic ecosystems (e.g. Townsend et al. 2000, Iwata et al. 2003). Riparian forests act as an important food source (via organic matter inputs) for many aquatic organisms, moderate the temperature of rivers (via shading) and prevent excessive sediment input in stream via root retention (Aguiar et

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al. 2002, Herlihy et al. 2005). These positive effects of riparian forests of river ecosystems led to the adoption of laws to protect the riparian forests around the world. In Brazil, many riparian forests that had been severely degraded in earlier years are now protected (Rodrigues 1999), and programs of reforestation have begun in recent years (Simões et al. 2002, SMA 2005).

Nevertheless, ongoing land use practices still represent a potential threat to existing riparian forests. Livestock in cattle farms can degrade and even destroy riparian forests, increase the amount of fine sediment carried into rivers by surface runoff from farmland (Riley et al. 2003), and decrease the input of allochthonous organic matter (e.g. leaf litter and woody debris) into the rivers. Consequently, the chemical and physical properties of a river are altered by a change of forest cover and human land use in its catchment (Perry et al. 1999, Walsh et al. 2005, Burcher & Benfield 2006). To gain a better understanding of the mechanisms behind these changes, many studies have been carried out in temperate areas of the world. These studies focused on the influence of the land use changes on terrestrial (e.g. Ferreira & Horta 2001) or aquatic biota (Collier et al. 1997, Hall et al. 2003, Michailova et al. 2003) and in-stream physico-chemical characteristics (Davies-Colley 1997, Edgar 1999).

By contrast, similar research has only begun in recent years in subtropical and tropical parts of the world, including Brazil. The majority of these studies were conducted in small aquatic systems (Ferreira-Perruquetti & Fonseca-Gessner 2003, Gerhard 2005), as were most of the abovementioned studies in temperate climates, and there are few studies of the effects of human land use on the ecosystems of larger rivers, especially with a focus on the midge fauna (e.g. Sonoda 2005, Corbi 2006). The present study in two large Brazilian rivers had two specific objectives. The first was to determine whether the midge communities at forest and pasture reaches differed in terms of abundance, species richness, diversity, evenness and community composition. We expected midge communities to be less healthy and diverse at pasture reaches because of the positive effects that riparian forests have been shown to have on aquatic communities (see above). Our second objective was to compare the relative abundances of midge functional feeding groups between land uses. We expected the shredder guild within the Chironomidae to be more important and grazers to be less important at reaches with riparian forest than at pasture reaches, because forest reaches should be dominated by allochthonous inputs whereas pasture reaches should be dominated by autochthonous production.

Methods

Study sites

Our study was conducted in the main rivers of two catchments in the State of São Paulo, Brazil (Fig. 1). The two river basins were selected because they show a moderate degree of human impact compared to other catchments in the State, and we wanted to avoid rivers that receive highly polluted (and potentially toxic) water from predominantly urban or industrial catchments.

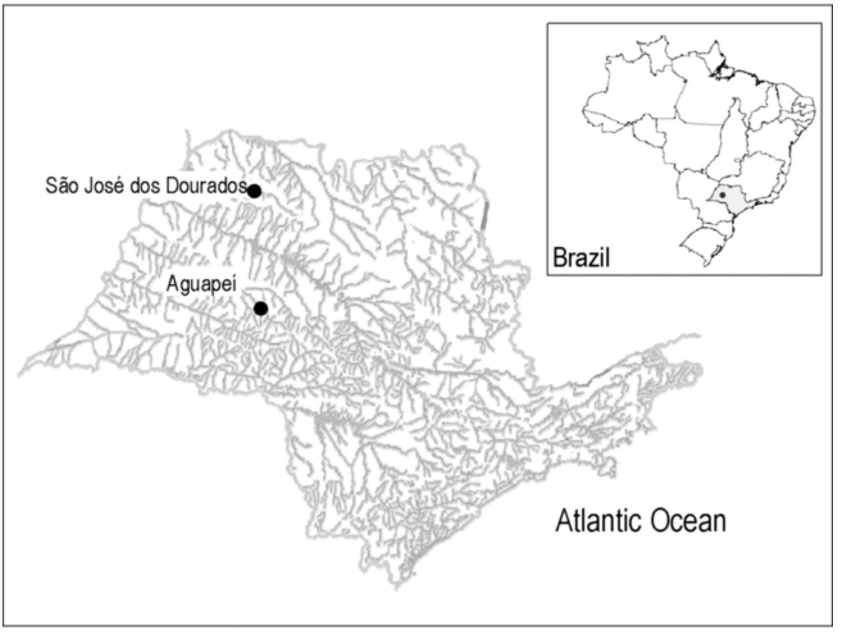


Fig. 1. Map of São Paulo State, with the study reaches in the Aguapeí and São José dos Dourados watersheds indicated by black dots. The smaller map shows the location of São Paulo State in Brazil.

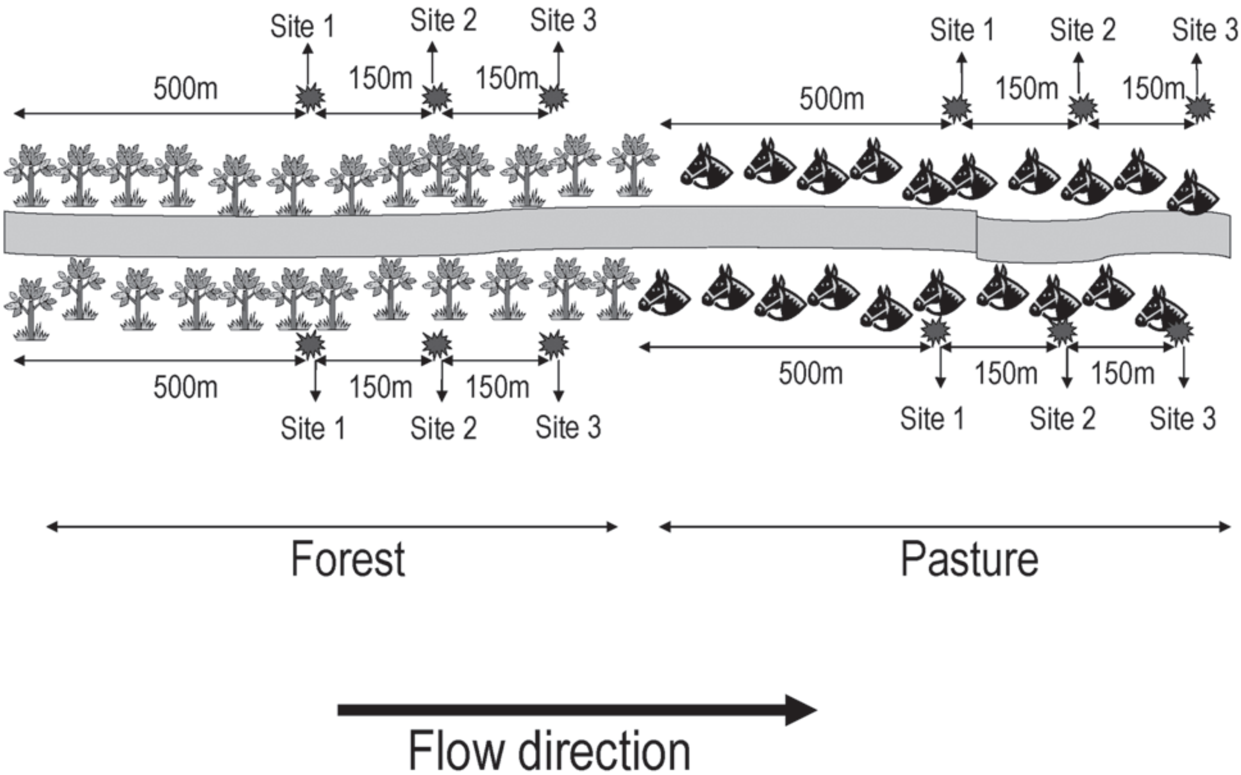


Fig. 2. Schematic drawing of the experimental design in each of the two rivers. Three colonisation baskets were exposed in both river margins at each of the three sampling sites within each study reach (distance between baskets 0.2–1.3 m). For more details see text.

The catchment of the Aguapeí River has an area of 12,200 km^2 and an annual rainfall of 1200 mm (average across 10 years; Groppo et al. 2001). The main land uses in the catchment are cattle farms (73.5 %) and perennial cropland (4.2 %); indigenous forest on conservation land comprises 12.6 %, and urban areas 0.7 %. Aguapeí River, the main river, is a 7th order river of 468 km length. At the study reaches, the river was about 48 m wide and 1.5 m deep in the main channel (during the dry season, n = 18 for both measures, see below), with a mean annual discharge of 83.6 m^3/sec (Groppo et al. 2001).

Because of the downstream drifting behaviour of larval midges (Armitage et al. 1995, Schreiber 1995, Merritt & Cummins 1996) and the downstream transport of dissolved nutrients and/or pollutants, the human land use practices in the riparian zones upstream of our study reaches should be most relevant for the midge communities at the reaches. Therefore, we determined the catchment area above our study reaches (3,611 km^2) and re-calculated the percentage of each land use in this sub-catchment. Once again, pasture was the dominant land use (70.9 % of the sub-catchment), followed by indigenous forest (16.1 %), perennial cropland (5.7 %) and urban areas (0.9 %).

São José dos Dourados River (henceforth called São José River) belongs to the homonymous catchment that is situated in the northwestern region of the State of São Paulo. This catchment has a total area of 5,159 km^2 and an annual average rainfall of 1,300 mm, is dominated by agrosystems (pasture, sugarcane and other crop plantations, 90.7 % of the total area), followed by indigenous forest (6.9 %) and urbanisation (0.6 %). At the study reaches, the river was about 22 m wide, had a maximum depth of 1.2 m in the main channel during the dry season and an annual discharge of 30.6 m^3/sec (average across 10 years, Groppo et al. 2001). São José River is a 6th order river of 301 km length. The sub-catchment of São José River upstream of our study reaches (area 1081.1 km^2) was dominated by cattle pasture (75.6 %), followed by indigenous forest (9.9 %), perennial cropland (5.6 %) and urban areas (0.7 %).

The investigated forest reaches in the Aguapeí River were located at 50º26 71"W and 21º70 47"S, and pasture reaches at 50º28 37"W and 21º70 88"S. The corresponding coordinates of forest and pasture reaches in the São José River were 20º32 14"S and 50º01 25"W, and 20º32 28"S and 50º01 46"W.

Width measurements of both rivers were taken once per month during 18 months, at the same locations where two state environmental agencies (Company of Technology on Environmental Sanitation [CETESB] and Department of Water and Electrical Energy [DAEE]) regularly obtain rainfall data and measure river discharge.

Plastic baskets (30 cm × 15 cm × 8 cm, mesh size 2.0 cm) filled with commercially produced clay rocks (Cinexpan Industry and Commerce of Clay Rocks Ltd, Várzea Paulista, Brazil) were used as artificial substrates for colonising Chironomidae. Clay rocks were classified as coarse gravel (particle size 16–32 mm) after Gordon et al. (1992). The clay rocks were slightly larger than the natural substrata in the beds of both rivers, which consisted of sand (particle size 0.06–2 mm), silt (0.004–0.06 mm) and clay (0.004–0.0002 mm) in the following percentages: Aguapeí River 74 % sand, 16 % silt and 10 % clay, São José River 91 % sand, 4 % silt and 5 % clay (averages of three replicate substratum samples per river collected with an Eckman dredge in January, March, May, July and September 2002).

Based on the recommendations of several methodological studies (Lammert & Allan 1999, Solimini et al. 2000, Cuffney et al. 2002), three criteria were used when selecting our study reaches. To prevent any interference of pasture practices on water chemistry, forest reaches had to be situated upstream of the pasture reaches. Both land use types had to be present for at least 500 m length along the river bank in both rivers margins above and alongside each study reach (Fig. 2). Moreover, the riparian forest strips at the forest reaches had to be at least 50 m wide. Upstream of the forest strips (which were less than 1 km long in both rivers), both forest reaches had pasture land use. All study reaches were 300 m long. Within each study reach, three sampling sites were selected in 150 m distance from each other (Fig. 2). At each site, three baskets filled with artificial substrates were placed randomly in both rivers margins (with 0.2–1.3 m distance between individual baskets), resulting in 18 baskets per land use and 36 baskets per river. At all sites, baskets were exposed in riffle habitats with similar water depth and current velocities. No baskets were exposed in the main channels of both rivers because this was logistically impossible. Further, riparian land use practices may be especially relevant for the aquatic communities inhabiting the margins of larger rivers because effects of riparian forests (e.g. shading, leaf litter input, reduced surface runoff during storms) should be greater in these margins than in mid-river.

In both rivers, substratum baskets were exposed during the last 44 days of the dry season (from 31 August to 13 October 2002). This period is generally considered long enough for the colonising insect assemblage to reach an equilibrium (Mihaljevic et al. 1998, Duxbury 2003). After this colonisation period, the artificial substrates in the baskets were removed from the rivers, placed into plastic containers, preserved with 80 % ethanol, and transported to the laboratory.

Out of the 18 baskets exposed in each land use at each river, twelve baskets could be retrieved at the forest and twelve from pasture reaches of Aguapeí River, whereas nine could be sampled at the forest reach and 15 at the pasture reach of São José River, resulting in a total of 48 samples. The remaining baskets were lost due to vandalism. Because these losses resulted in unequal numbers of baskets collected at each of the three sites within each study reach, we decided to pool the remaining samples from each entire reach in the statistical analysis (see below).

In the laboratory, each sample was washed under running water and all invertebrate and organic material was retained us-

Table 1. Relative abundances of midge taxa found in the two study rivers (per cent contributions to total midge density at each reach). Bold numbers represent taxa contributing > 10 % to total midge density at a given reach.

Taxon	Functional feeding group	Aguapeí River		São José River	
		Pasture	Forest	Pasture	Forest
Beardius spp.	Grazer (scraper)	0.1	0.0	0.1	0.0
Chironomus spp.	Filter feeder	0.0	0.0	1.2	2.3
Cryptochironomus spp.	Predator	1.1	4.2	0.0	0.0
Dicrotendipes spp.	Collector-gatherer	0.0	0.0	0.0	0.3
Endotribelos spp.	Shredder	0.3	0.0	0.0	0.0
Fissimentum spp.	Collector-gatherer	0.0	0.3	0.0	0.3
Goeldichironomus spp.	Filter feeder	0.0	0.0	0.0	4.2
Lauterborniella spp.	Collector-gatherer	0.0	0.0	1.2	1.7
Parachironomus spp.	Collector-gatherer	0.4	0.7	0.0	1.1
Pol. (*Polypedilum*) spp.	Collector-gatherer	0.8	1.9	0.0	0.3
Stenochironomus spp.	Shredder	0.0	0.0	0.3	0.0
Zavreliella spp.	Collector-gatherer	0.8	0.5	0.0	0.8
Pseudochironomus spp.	Collector-gatherer	0.0	2.7	0.0	0.0
Rheotanytarsus spp.	Filter feeder	**59.8**	**42.5**	**41.2**	**12.7**
Tanytarsus obiriciae Strixino & Sonoda, 2006	Collector-gatherer	0.0	0.0	7.3	9.6
Corynoneura spp.	Collector-gatherer	7.1	4.6	7.7	3.4
Lopescladius spp.	Collector-gatherer	8.3	**24.3**	0.3	1.1
Nanocladius spp.	Collector-gatherer	**14.7**	8.3	**21.2**	**20.4**
Ablabesmyia spp.	Predator	2.8	5.6	**12.4**	**37.1**
Coelotanypus spp.	Predator	0.0	0.0	0.2	0.3
Denopelopia spp.	Predator	0.0	0.3	0.2	0.0
Labrundinia spp.	Predator	1.2	0.8	5.1	2.8
Larsia spp.	Predator	0.5	1.5	1.1	1.1
Pentaneura spp.	Predator	1.1	1.7	0.3	0.3
Djalmabatista spp.	Predator	0.0	0.2	0.1	0.0
Morphotype M	Unknown	0.8	0.0	0.0	0.0
Morphotype Y	Unknown	0.1	0.0	0.0	0.0

ing a sieve with a mesh size of 0.2 mm (Trivinho-Strixino & Strixino 1998). The material retained in the sieve was placed inside clear plastic trays over a trans-illuminator (a wooden box covered with transparent plastic and lit up by two lamps underneath), and all Chironomidae in each sample were sorted and preserved in 70 % alcohol (Pinder 1989). The larvae were counted and identified to the genus level using a stereomicroscope and a microscope (both manufactured by Leica Microsystems AG, Wetzlar), and a key developed for Brazil's southeastern region (Trivinho-Strixino & Strixino 1995). Midge numbers in each sample were then extrapolated to densities per m^2 by multiplying with a factor based on the horizontal surface of the colonisation baskets (30 × 15 cm).

Data analysis

As biological response variables, we selected total density of midges, midge taxon richness, Rényi's diversity index (Tóthmérész 1995; calculated using 2 as the base for the logarithm) and Margalef's evenness index (Odum 1984), densities of the five most common midge genera (which also had to be present in more than 50 % of all 48 samples), and relative abundances of each of the five trophic guilds (collector-gatherers, filter feeders, predators, shredders and grazers, Merritt & Cummins 1996). All response variables were analysed using a nested 3-way Analysis of Variance (ANOVA) in SPSS® version 15.0. The factors included in the analysis were 'land use' as the fixed factor, 'river' as a random factor, and 'sample position' (1–15; the location of each sample determined from downstream to upstream) as a nested factor within 'land use'.

To correct for potential effects of midge abundance on community diversity (see McCabe & Gotelli 2000), we also determined abundance-corrected taxon richness by using total midge abundance as the covariate in a nested 3-way Analysis of Covariance (ANCOVA) on midge taxon richness. This procedure standardized taxon richness to the overall average number of

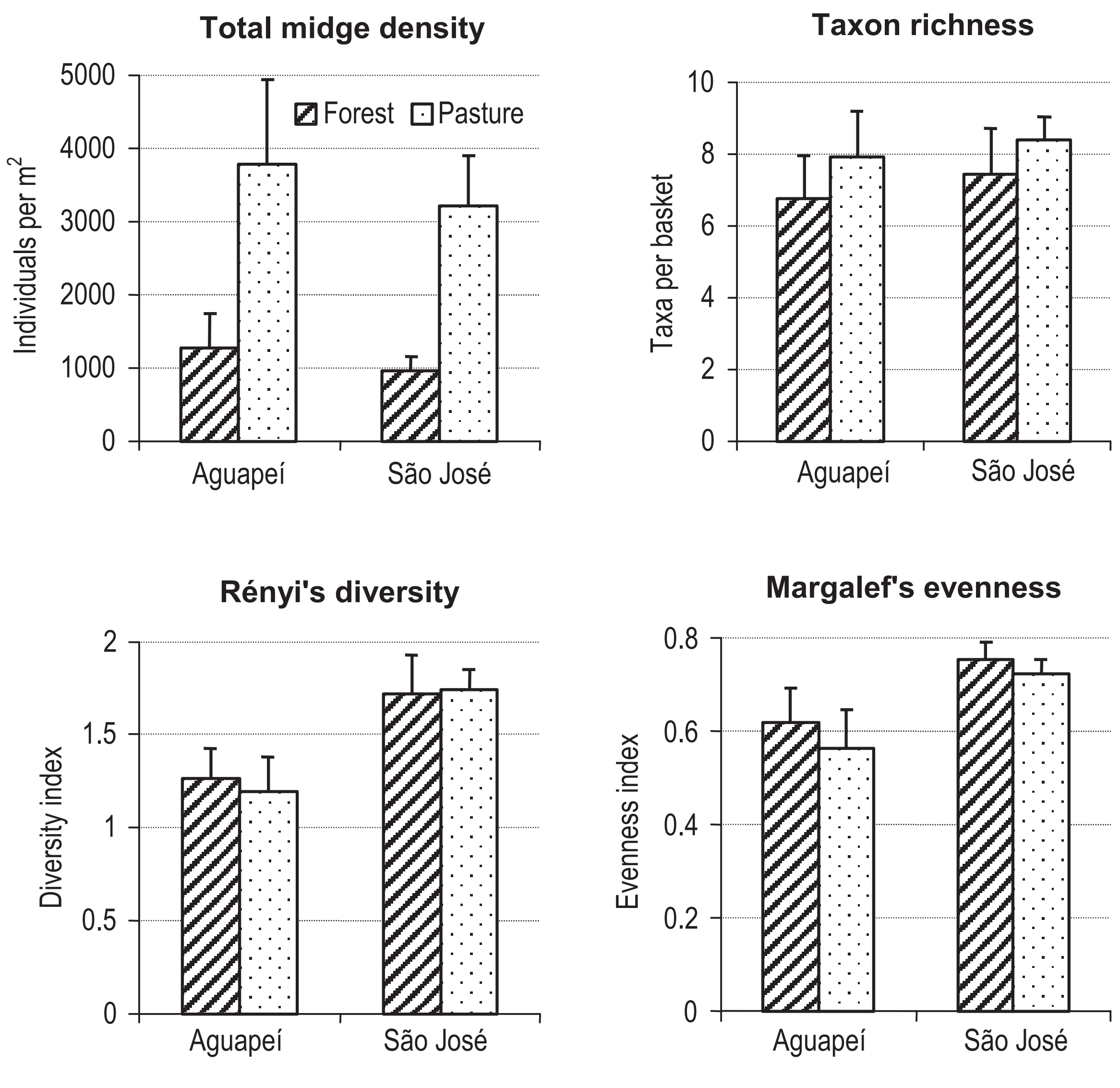


Fig. 3. Total density of Chironomidae per square metre, midge taxon richness (per sample), Rényi's diversity and Margalef's evenness at the two reach types within each river. For more details see text.

midges across all samples. We used ANCOVA here rather than the rarefaction technique used by McCabe & Gotelli (2000) because several of our samples contained very few (1–3) midge individuals and rarefaction based on the smallest sample would have resulted in uniformly low taxonomic richness across all samples.

Data were log-transformed where necessary to improve normality and homoscedasticity. Two samples (one from each land use category in the Aguapeí River) had to be omitted from the analyses of Rényi's diversity and Margalef's evenness because they contained only a single individual. To help avoid type II errors, we report exact *P*-values for all results and statistical power for all cases where $0.10 > P > 0.05$ (Toft & Shea 1983).

The results for the nested factor 'sample position' are presented (see Table 2 below) but not discussed any further because they merely indicate if the values of a given response variable differed between any of 9–15 sampling locations across the four reaches (regardless of land use type), and such differences are irrelevant to our research objectives.

Results

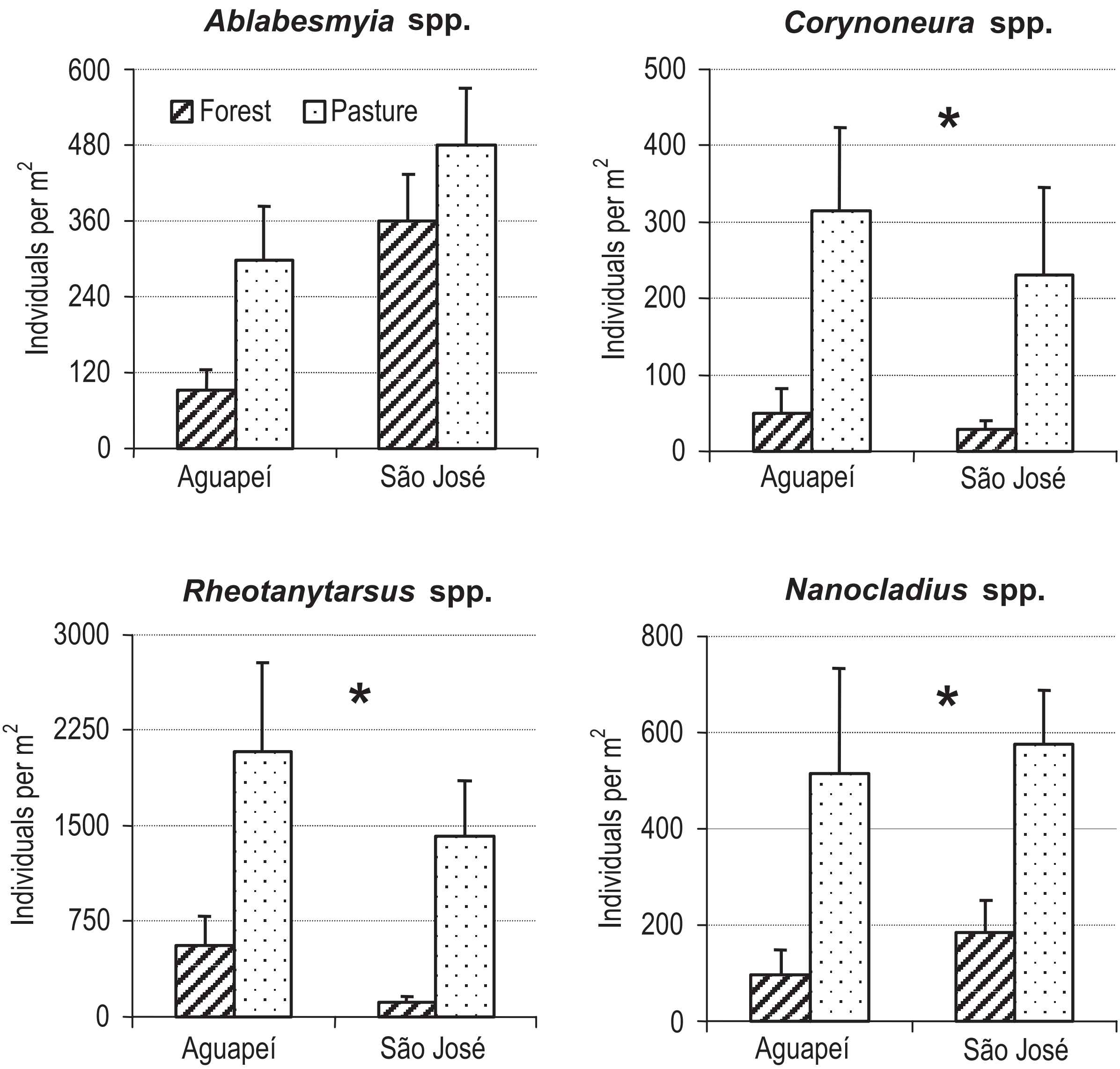
A total of 5,286 midge individuals were sorted and identified from both rivers, comprising 25 genera and two morphotypes (Table 1). The five most common genera (*Rheotanytarsus* spp., *Nanocladius* spp., *Corynoneura* spp., *Lopescladius* spp. and *Ablabesmyia* spp.) represented 88 % of all individuals in the 48 samples. Total midge density, "raw" taxon richness (uncorrected for midge abundance), abundance-corrected taxon richness and Margalef's evenness were all similar across land use categories and rivers (Fig. 3, Table 2). Rényi's diversity of the midge community was significantly higher in the São José River than in the Aguapeí River (Fig. 3, Table 2).

Three of the five most common midge genera (*Rheotanytarsus* spp., *Nanocladius* spp. and *Corynoneura* spp.) were more abundant at the pasture than at the forest reaches (Fig. 4, Table 2). *Nanocladius* was also more abundant in the São José River than in the Aguapeí River ($P = 0.06$, power = 0.46). Densities of the remaining two genera were not affected by land use but differed between rivers, with higher values of *Lopescladius* in the Aguapeí River and *Ablabesmyia* in the São José River.

The dominant midge genera also influenced the density distributions of the functional feeding groups, because filter feeders included *Rheotanytarsus*, predators *Ablabesmyia* and collector-gatherers *Nanocladius*, *Corynoneura* and *Lopescladius* (see Table 1). Therefore, we standardized the feeding group data by expressing them as relative abundances (per cent contributions to total midge density in each sample). Shredders and grazers were generally rare, each contributing less than 1 % of total midge abundance.

Table 2. Summary (*P*-values) of nested 3-way ANOVAs comparing total midge density, taxon richness, densities of the most common taxa and relative abundances (per cent contributions to total midge abundance per sample) of the five functional feeding groups between land uses and rivers. 'Land use' was the fixed factor in the analysis, 'river' a random factor, and the factor 'sample position' (1–15) was nested within 'land use'. * $P < 0.05$, ** $P < 0.01$. Statistical power is given for all results where $0.05 < P < 0.10$. A = Aguapeí River, SJ = São José River. The ANOVA model was intercept (degrees of freedom 1) + land use (1) + river (1) + sample (land use) (25) + error (20). N = 48.

	Land use	Ranking	River	Ranking	Sample (land use)
Total Chironomidae	0.11		0.24		0.89
Midge taxon richness	0.65		0.35		0.47
Abundance-corrected midge taxon richness (ANCOVA), $P < 0.0001$ for covariate log (total Chironomidae)	0.21		1.0		0.56
Rényi's diversity	0.69		0.02*	SJ > A	0.30
Margalef's evenness	0.42		0.14		0.36
Rheotanytarsus spp.	0.02*	Pasture > forest	0.44		0.87
Nanocladius spp.	0.007*	Pasture > forest	0.06/0.46	(SJ > A)	0.89
Corynoneura spp.	0.03*	Pasture > forest	0.53		0.95
Lopescladius spp.	0.31		0.0001**	A > SJ	0.24
Ablabesmyia spp.	0.35		0.001**	SJ > A	0.79
Collector-gatherers	0.50		0.09/0.40	(A > SJ)	0.45
Filter feeders	0.31		0.81		0.77
Predators	0.08/0.42	(Forest > pasture)	0.05/0.51	SJ > A	0.85
Shredders	0.07/0.44	(Pasture > forest)	0.56		0.15
Grazers (Scrapers)	0.31		0.14		0.33



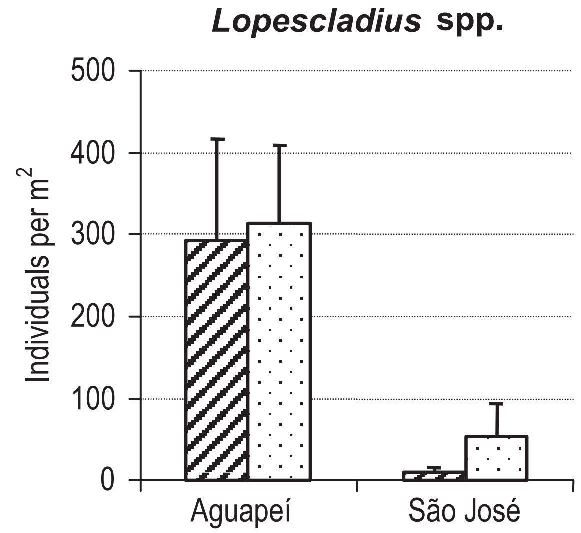


Fig. 4. Densities of the five most common midge taxa at the two reach types within each river. Asterisks indicate significant differences between land uses. For more details see text.

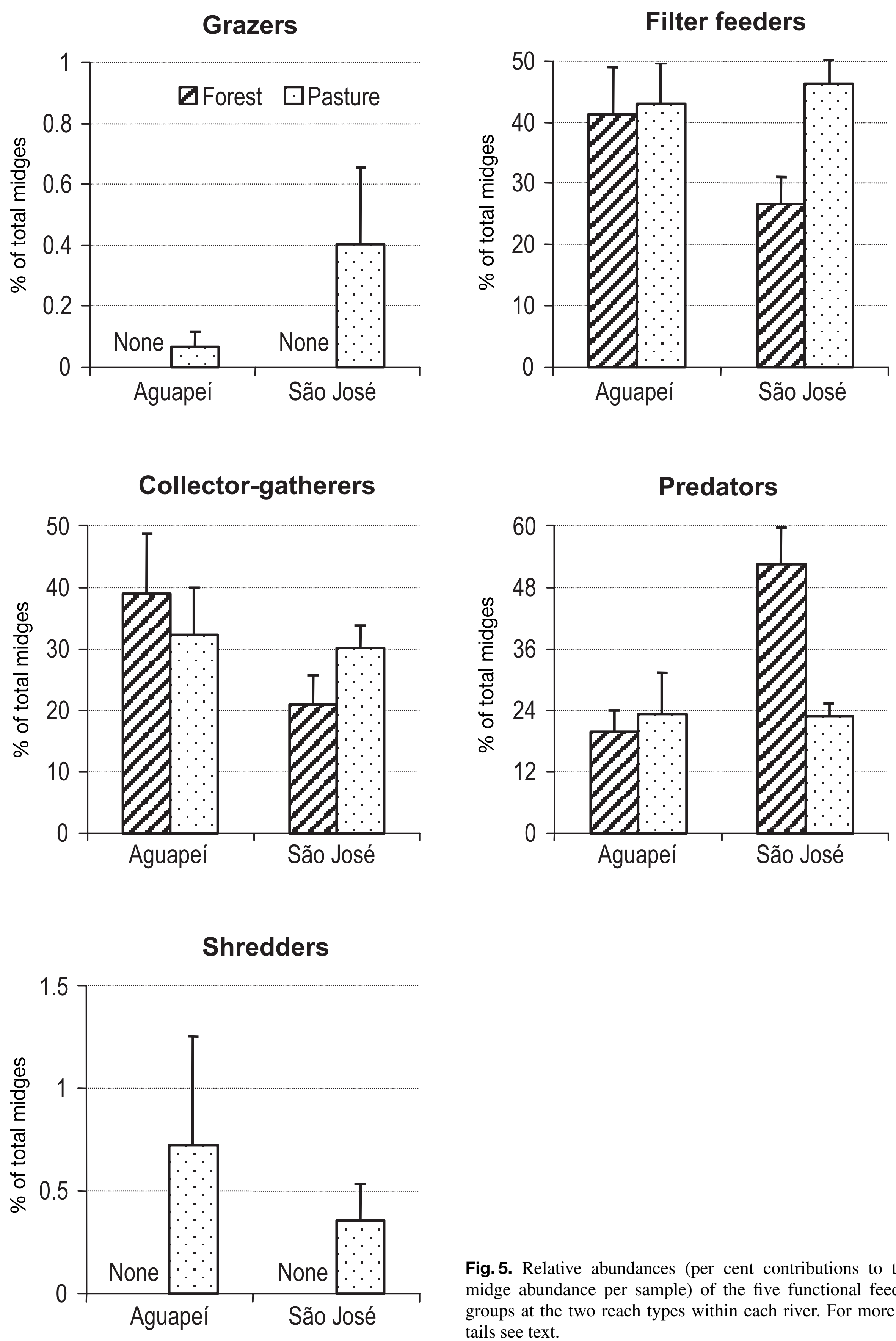


Fig. 5. Relative abundances (per cent contributions to total midge abundance per sample) of the five functional feeding groups at the two reach types within each river. For more details see text.

The relative abundances of collector-gatherers, filter feeders and grazers were all similar across land uses and rivers (Fig. 5, Table 2), except that collector-gatherers were relatively more common in the Aguapeí River ($P = 0.09$, power = 0.40). By contrast, predators were relatively more abundant at forest reaches than at pasture reaches ($P = 0.08$, power = 0.42), whereas the opposite pattern occurred for shredders, which were absent at forest reaches ($P = 0.07$, power = 0.44). Predators also made up a higher percentage of total midge density in the São José River ($P = 0.05$, power = 0.51).

Discussion

Midge communities in forest and pasture reaches

Our results add new knowledge about effects of human land use activities on Chironomidae assemblages in large Brazilian rivers. They also contrast with the majority of findings of earlier research on the contribution of riparian forests to the maintenance of healthy rivers (e.g. Friberg et al. 2005, Cetra & Petrere 2007, Momoli et al. 2007, Muotka & Syrjänen 2007). Because of these findings, we had expected the midge communities at our pasture reaches to be less healthy and diverse than at the forested reaches. However, none of the investigated five midge community parameters differed significantly between the two land uses, and three of the dominant midge genera were actually more common at the pasture sites. These results may indicate that land-use related changes in habitat conditions were not severe enough to represent a stressor (with negative effects) for the midge community, but rather acted as a subsidy, with mainly positive effects (see Niyogi et al. 2007a). Although we did not measure nutrient levels at our study reaches, many studies worldwide have shown that nutrient levels generally rise with increasing degree of catchment development (see e.g. Riley et al. 2003, Turner et al. 2003, Allan 2004, Niyogi et al. 2007a, 2007b), with moderate nutrient enrichment in pasture streams (Buck et al. 2004, Matthaei et al. 2006), and such moderate increases in nutrient levels may have been beneficial for the three midge taxa that increased in density at our pasture sites. For Brazilian fish communities, our results are paralleled by those of Gerhard (2005) who found that fish community richness was higher in the absence of riparian forest in a survey of 60 stream sites that differed in their riparian land use (forest, pasture and areas affected by erosion).

One might argue that our inability to find degraded midge communities at the pasture sites was partly caused by the relatively low level of replication (only two sites in each land use category), which was restricted by the availability of large river sites that fulfilled the selection criteria described in the Methods and the overall logistical effort involved. Nevertheless, we would like to emphasize that our design was powerful enough to detect several significant differences between land use categories (and/or study rivers; see Table 2), and none of these differences can be interpreted as a negative effect of the pasture land use. Consequently, we feel that our results are valid, in spite of this limitation of our study design.

In contrast to our findings, Marques et al. (1999) found a decrease in Chironomidae richness in Brazilian watersheds impacted by human land uses (iron and gold mining and *Eucalyptus* plantations), where midge communities were dominated by *Chironomus* spp., a genus known world-wide to be highly resistant to pollution. Similarly, Ometto et al. (2000, 2004) found that invertebrate communities in São Paulo State were generally most diverse in forested rivers, with decreasing diversity as human land-use activities increased. Corbi (2006) and Corbi & Trivinho-Strixino (2006) studied the midge fauna inhabiting streams surrounded by sugar cane crops in the same region of Brazil. They also observed that midge taxon richness was lower at sites that had lost their riparian forests. When compared to these five studies, the results from our present research may imply that pasture grazing by cattle is less harmful for lotic midge communities in Brazilian rivers than some other human land uses. Alternatively, the relatively large size and discharge of our two study rivers (6^{th} and 7^{th} order) may have moderated the effects of the land-use change on the aquatic community (due to efficient dilution of potentially harmful substances entering the rivers), because the streams investigated in the other five Brazilian studies were all considerably smaller (1^{st} to 3^{rd} order).

Evidence for the value of riparian forests for preserving biological water quality also exists from other regions of the world (see e.g. Bastian et al. 2007, Dallas 2007, Galbraith et al. 2008). For instance, in a study of 3^{rd} and 4^{th}-order streams in the USA, Burcher & Benfield (2006) emphasized the influence of watershed land-use practices on in-stream physical features and (indirectly) on river biota. In Japan, Nakamura & Yamada (2005) studied the importance of riparian forest along pasture farms and found the same pattern as in the five Brazilian studies cited above. Finally, the positive influence of riparian forest on faunal diversity has been documented not only for aquatic organisms. For example, Smith & Wachob (2006) investigated the importance of streamside forests for birds in the USA and found that forest clear-cutting practices were harmful for richness and diversity of the bird community.

Functional feeding groups

Our second research objective concerned the functional feeding groups within the Chironomidae. We expected a relative increase of grazers and a decrease of shredders at pasture sites. Grazers should increase due to a higher primary production of benthic algae caused by increased irradiation, resulting in more food for grazers. Shredders should decrease due to a lack of allochthonous input of organic material from riparian forests into the rivers. Our results for the relative abundances of the functional feeding groups (a few trends approaching significance but no significant differences between land uses) do not support these two hypotheses, possibly because both feeding groups made up only a small proportion of the midge communities in the two rivers. Instead, midge communities were dominated by filter feeders and collector-gatherers, and predators were also important.

So why were grazing and shredding midges so rare in our study rivers? Midge grazers may have been rare because the bed surface in both rivers was dominated by sand (see Methods) and other grazing invertebrates, such as mayflies and snails, generally prefer coarser substrata (e.g. Gonser 1997, Lysne & Koetsier 2006), presumably because the benthic algae they feed on tend to be more abundant on larger bed particles. For non-midge shredders, England & Rosemond (2004) found that a decline in the number of shredders in several North American streams was associated with low standings stocks of coarse particulate matter. However, regular field observations during our study period indicated that densities of fallen leaves were quite high at the forest reaches of both rivers: about 40-50 % of the bed surface in the river margins (where the colonisation baskets were exposed) was covered by leaves and branches (K. C. Sonoda, pers. observ.). Consequently, there was no lack of food for shredders in the locations where the baskets were exposed, in spite of the relatively large size of the investigated rivers (6^{th} and 7^{th} order).

It may be that shredders are generally quite rare in many South American streams and rivers, similar to certain other regions of the world, such as in New Zealand (see e.g. Winterbourn et al. 1981). For instance, Mathuriau & Chauvet (2002) investigated the breakdown of leaf litter in a Colombian stream and found that shredders played an insignificant role in this process. Callisto et al. (2001, 2004) found low densities of shredders at 2^{nd}-5^{th} order stream sites surrounded by riparian forest in a Brazilian national park during both the dry and the rainy seasons. Finally, Buss et al. (2002) also observed that shredders were rare in another Southeast Brazilian river.

Conclusions and outlook

Further studies are necessary to establish a clearer picture of the effects of land use changes on Chironomidae communities in Brazilian rivers. Nevertheless, our study adds to the emerging body of research investigating the importance of riparian forests for the maintenance of aquatic biodiversity in Brazil. In future research, midges should ideally be identified to the species level, and analysis of gut contents would also be desirable. Further, more replicates at the site/river level should be used where logistically feasible. Finally, investigating the entire aquatic community (i.e. including other invertebrates and fish communities) rather than focusing exclusively on the midges may increase overall power to detect differences between land use categories.

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