Abrupt change of a stream ecosystem function along a sugarcane-forest transition: Integrating riparian and in-stream characteristics

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ABSTRACT

Forest remnants can locally improve water quality of deforested streams in a reset effect, but few studies evaluated if leaf breakdown rates respond to forest remnants or, at a finer spatial scale, to riparian forest structure. We studied leaf breakdown rates along a deforested Neotropical stream as it flowed through a sugarcane/forest remnant transition; we adjusted a non-linear model to describe this relationship, and evaluated whether this model was further related to the effects of riparian forest structure, stream physical characteristics and shredder abundances. Modeled leaf breakdown rates rapidly increased as the stream entered the forest remnant, stabilizing in the forest interior after about 100 m. Observed leaf breakdown rates deviated from the model within the forest remnant. This unexplained residual variation was related to riparian forest structure, which was heterogeneous within the remnant. Leaf breakdown rates were not related to stream physical characteristics, but were significantly related to the abundance of shredders, especially with the dominant leaf-mining Chironomidae. Abundances of leaf-mining Chironomidae were strongly related to both distance along the forest remnant and riparian forest structure. Therefore, higher leaf breakdown rates as the stream flowed through the forest remnant were possibly due to increases in abundances of leaf-mining Chironomidae, which responded to variation at both spatial scales studied. These results suggest that forest remnants are important in rural landscapes not only by improving stream water quality but also by restoring ecosystem functions.

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1. Introduction

Leaf litter decomposition is an important process in ecosystems and, together with other functional variables, is commonly used to evaluate ecosystem health (Clapcott et al., 2010; Tank et al., 2010). Leaf breakdown rates can respond to gradients of human disturbance in both terrestrial and aquatic systems, and have been used as indicators of changes in ecosystem functions (Gessner and Chauvet, 2002; Young et al., 2008; Silva-Junior et al., 2014). In stream ecosystems, leaf breakdown rates are an important functional variable, since changes in these processes can indicate both changes in the structure of aquatic communities and stream water quality (Webster et al., 1999; Tank et al., 2010). These responses can be integrated at different spatial scales, within streams, among streams, and among watersheds (Tiegs et al., 2009). As the vegetation structure of the riparian zone can strongly

influence the aquatic communities by changing physical and chemical characteristics of the stream water and habitat (Storey and Cowley, 1997; Souza et al., 2013), land use changes both at the scales of the stream and watershed can have diverse effects on community structure and ecosystem functions (Kreutzweiser et al., 2008; McKie and Malmqvist, 2009; Clapcott et al., 2012).

Several studies evaluated the relationship between land use and leaf breakdown rates in stream ecosystems, with variable results. For example, leaf breakdown rates can be slower in agriculture streams when compared with forested streams due to the negative effects of pesticides and stream habitat simplification on aquatic organisms (Rasmussen et al., 2012), or because of lower shredder abundances (Piscart et al., 2009; Encalada et al., 2010; Lecerf and Richardson, 2010). On the other hand, leaf breakdown rates in clearcut, agriculture or urban streams can be higher due to nutrient enrichment, higher microbial activity, higher shredder biomass, and higher storm runoff (Paul et al., 2006; Mckie and Malmqvist, 2009). Some studies found similar leaf breakdown rates between reference and impacted streams. Fleituch (2013) found that in reference streams leaf breakdown rates were

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determined mainly by small shredders (e.g., stoneflies), whereas in impacted streams leaf breakdown rates were determined by larger gammarids or dipterans; thus, the reduction of small shredders in impacted streams was balanced by the increase of other shredder groups, enabling the maintenance of this function. Similar leaf breakdown rates between forested and impacted streams were also detected due to differences in the composition of decomposer communities (shredders vs microbes), with greater effects of shredders in forested streams, but higher effects of microbial breakdown in pasture (Hladyz et al., 2010) or agriculture streams (Hagen et al., 2006; Huryn et al., 2002). These results refer to comparisons among streams, but leaf breakdown rates can respond to land use within the riparian corridor, and not necessarily respond to land use at the scale of the watershed, both in temperate (e.g., Sponseller and Benfield, 2001) and tropical systems (Silva-Junior et al., 2014). Also, leaf breakdown rates can differ between streams with similar land use (as in forested watersheds) but distinct structure and composition of riparian forests, even with no changes to the instream fauna (Kominoski et al., 2011).

Within a stream, some studies suggest that reaches with higher proportion of riparian forests upstream (thus with differences in land use) may have higher leaf breakdown rates than downstream, pasture-dominated streams, due to higher abundances of decomposer species, higher dissolved oxygen (DO) concentrations, and lower sedimentation rates (Sponseller and Benfield, 2001; Encalada et al., 2010). However, the characteristics of streams in deforested areas can change as they flow through forest remnants, with changes in stream physicochemistry, and composition and structure of macroinvertebrate communities, sometimes with a local "reset effect" in stream water quality (Harding et al., 2006; Fernandes et al., 2014; Goss et al., 2014; Suga and Tanaka, 2013). These characteristics can also be locally influenced by the structure of the riparian forest, mainly in relation to tree density, mean trunk diameter, and total basal area (Souza et al., 2013). In fact, Fernandes et al. (2014) found that as a tropical deforested stream flows through a forest remnant, DO and nutrient concentrations increase and electric conductivity (EC) decreases but, except for DO concentrations, these variables were also influenced by riparian forest structure, which was not homogeneous along the forest remnant. On the other hand, Goss et al. (2014) found decreased nutrient concentrations as the agricultural streams flowed through forest remnants in Ohio, but no relationship was found between leaf breakdown rates and distance within the remnants. To our knowledge, there are no studies that simultaneously evaluated the effects of the presence of a forest remnant and riparian forest structure on leaf breakdown rates within the same stream.

In this study we sampled the same stream studied at the same time by Fernandes et al. (2014) and Suga and Tanaka (2013) to test the hypotheses that (1) leaf breakdown rates vary as a deforested stream flows through a forest remnant, reducing the effects of deforestation, and (2) that this variation would be related to shredder abundance. As the forest structure was not related to the distance along the forest remnant (Fernandes et al., 2014), we tested the hypothesis that (3) at a finer spatial scale, leaf breakdown rates would be locally influenced by the structure of the riparian forest, stream physical characteristics, and macro-invertebrate shredder communities.

2. Methods

2.1. Study site

This study was carried out in Vassununga State Park (21°20′–21°55′S, and 47°32′–47°40′W), which is located in the Mogi-Guaçu River watershed in São Paulo state, SE Brazil. The park is subdivided in six main units, which are inserted in a matrix predominantly used for perennial cultures, sugar cane, and pasture (Korman, 2003). We studied the Córrego da Gruta stream, whose source is located within sugar cane, hundreds of meters upstream a forest remnant that constitutes one of the six park units (Capetinga Oeste), with an area of 327.83 ha. Córrego da Gruta is a first-order stream that initially flows through active sugar cane plantation, than abandoned sugar cane plantation, before entering the Capetinga Oeste unit, where it flows through about 1 km along the forest remnant, before reaching the Mogi-Guaçu River. The climate of the region is Cwa according to Köppen classification, with mean annual temperatures between 17.6 °C (July) and 23.5 °C (February), and mean annual rainfall = 1478 mm concentrated in the austral summer months (Setzer, 1966).

2.2. Sampling

We sampled three reaches upstream the forest remnant at 150, 100, and 50 m from the remnant edge (denominated in this study −150, −100, −50, respectively), one reach just upstream the border of the forest remnant (point 0), and nine reaches downstream the border, within the forest remnant, at distances 50, 100, 150, 200, 250, 300, 400, 500 and 600 m. Each reach comprised a 50 m stretch of the stream from points −150 to 300, and a 100 m stretch from points 300 to 600 m; a detailed description of the area and sampling points can be found in Fernandes et al. (2014). Stream water characteristics were studied by Fernandes et al. (2014), and macroinvertebrate community structure by Suga and Tanaka (2013). Briefly, the effects of the forest remnant was to increase DO and phosphorus concentrations, decrease electric conductivity, increase macroinvertebrate total abundance but decrease taxon richness and diversity due to an increasing dominance by Chironomidae larvae (Table 1).

Leaf breakdown rates were estimated using 10 × 15 cm nylon bags, with 5.0 mm mesh size. We collected leaves directly from trees of *Cecropia pachystachya* Trécul (Urticaceae), a common tree species in riparian forests of the region; each litter bag was filled with 5.0 g of previously dried leaves. Five litter bags were fixed in the bottom of the stream bed, independently of the mesohabitat present, exactly at each sampling point; each litter bag was placed 5 m from each other. Litter bags were retrieved after 28 days by carefully enclosing each bag in a plastic bag underwater and taken to the laboratory where each sample was washed on a 500 μm sieve to separate the macroinvertebrates from the remaining leaf detritus. Two litter bags were lost, one from the sampling point 300 m and other from the point 500 m. The leaf detritus samples were oven-dried at 70 °C until leaf mass stabilized. Remaining leaf mass was determined with a digital scale (precision = 0.001 g). Leaf breakdown rates were estimated by the proportion of leaf mass loss (% LML) after 28 days as % LML = 1 – (remaining leaf mass/initial leaf mass), following Niu and Dudgeon (2011). The macroinvertebrates were classified to family and categorized in shredders following Cummins et al. (2005); leaf-mining Chironomidae (*Stenochironomus* spp.) were classified as shredders following Chará-Serna et al. (2012) and Henriques-Oliveira et al. (2003).

The structure of the riparian forest was estimated in each reach defined by the sampling points. Four 10 × 10 m plots adjacent to the stream were randomly marked, two in each margin. Within each plot, all trees with diameter at breast height (DBH) ≥ 3.0 cm were measured: circumference at breast height with a measuring tape, and canopy height with a laser hypsometer. The following variables were obtained from each plot: tree density, mean DBH, mean canopy height, total basal area, and vertical canopy structure (estimated by the coefficient of variation of canopy heights within each plot). Forest structure data were analyzed by Fernandes et al. (2014), who carried out a principal components analysis on the five variables to evaluate

Table 1
Range and response of selected variables measured at Córrego da Gruta (SE, Brazil) as it flows through the studied forest remnant.

Variable	Range	Response to the forest remnant	Reference
Stream characteristics			
Dissolved oxygen ($mg\,l^{-1}$)	3.9–8.2	Increase	Fernandes et al. (2014)
Total P ($\mu g\,l^{-1}$)	14.1–67.3	Increase	Fernandes et al. (2014)
Electric conductivity ($mS\,cm^{-1}$)	0.047–0.067	Decrease	Fernandes et al. (2014)
Mean depth (cm)	3.9–21.2	No response	This study
Mean width (cm)	16.0–130.0	No response	This study
Current velocity ($m\,s^{-1}$)	0.030–0.953	No response	This study
Macroinvertebrates			
Total abundance	2.2–931.7	Increase	Suga and Tanaka (2013)
Taxon richness	6.2–10.8	Decrease	Suga and Tanaka (2013)
Shannon–Wiener diversity index	0.83–1.65	Decrease	Suga and Tanaka (2013)
Shredders	0.0–36.9	Increase	Suga and Tanaka (2013)
Leaf-mining Chironomidae	0.0–29.1	Increase	This study
Leaf breakdown rates			
Proportion of leaf mass loss (%)	20.9–77.6	Increase	This study
k ($g\,day^{-1}$)	0.0084–0.0535	Increase	This study

the spatial distribution of forest structure. The first two axes explained 79.1% of the variation: the first axis (hereafter referred to as Forest Structure Axis 1) explained 45.0% of the variation and represented a gradient from more stratified forests with smaller trees to less stratified forests taller trees, whereas the second axis (hereafter referred to as Forest Structure Axis 2) explained 34.1% of the variation and represented a gradient of reaches with higher tree density but lower total basal area to reaches with lower tree density but higher total basal area (Fernandes et al., 2014). Neither forest structure axes were correlated with distance along the forest remnant, and therefore represented heterogeneity in riparian forest structure within the remnant.

Stream physical characteristics were estimated by flow velocities, width of flow, and flow depth. Stream flow velocities were determined with a flow meter, with measurement range between 0.1 and 4.5 $m\,s^{-1}$, by scanning all the transversal area of the water perimeter to obtain average velocities of the running flow. Average velocities are directly calculated by the flow meter, since it records and averages the velocity data obtained instantaneously along the scanning points. Readings of average velocities were obtained at three points within each sampling point, and the average value for each sampling point was used in the analyses. To describe the stream channel, we measured the width of flow and flow depth with tape measures or rulers (precision = 1.0 mm). Water depths were measured at least in six points along the channel width, and we used the average value at each sampling point.

2.3. Statistical analysis

To evaluate the effects of accumulated distance along the forest remnant on leaf breakdown rates, we used a non-linear model as suggested by Lim et al. (1998) and Harding et al. (2006) for analysing riparian zone buffering effects. The following first-order exponential model was fitted: $y = y_0 + A\exp^{(-x/t)}$, where A and t are constants, and x is the accumulated distance along the stream longitudinal gradient. The model was fitted by iteration with the Levenberg–Marquadt algorithm following Seber and Wild (2003), as implemented in the software Origin Pro 8.0. The dependent variables were the mean values of leaf breakdown rates (proportion of leaf mass loss after 28 days) and mean shredder abundances at each sampling point.

The effects of forest structure, stream physical characteristics, and composition of macroinvertebrate communities on leaf breakdown rates were evaluated using the residuals of the first-order exponential model fitted above as dependent variables, to evaluate if there

was remaining residual variation that could be explained by other independent variables. We used a stepwise multiple regression to select the variables that would be related to the residuals of (i) leaf breakdown rates, and (ii) shredder abundances. The set of candidate independent variables in (i) included Forest Structure Axis 1 and 2, shredder abundances, and stream flow velocities, whereas in (ii) they included only Forest Structure Axis 1 and 2, and stream flow velocities. Correlations between variables were calculated using Pearson's correlation coefficient, to evaluate multicollinearity among independent variables. We used stream flow velocities as a surrogate of stream physical characteristics because it was significantly correlated with width of flow and flow depth (see Section 3), and because it is an important factor influencing leaf breakdown rates in streams (Fonseca et al., 2013). The significance level used in the study was $P < 0.05$.

3. Results

Stream physical characteristics varied along the studied sampling points (Table 1). The width of flow ranged between 16 and 130 cm, averaging 49.8 cm ($\pm$38.89 SD), whereas mean depth varied between 3.9 and 21.2 cm, averaging 10.7 cm ($\pm$5.52) along the stream. Mean flow velocities in the studied reaches was 0.454 $m\,s^{-1}$ ($\pm$0.294), varying between 0.030 and 0.953 $m\,s^{-1}$. Stream flow velocities were correlated both with width of flow ($r = -0.770$; $P < 0.05$) and mean flow depth ($r = -0.767$, $P < 0.05$). These three variables were not significantly correlated ($P > 0.05$) to the distance along the forest remnant (stream flow velocities: $r = -0.582$; width of flow: $r = 0.630$; mean flow depth: $r = 0.559$).

Leaf breakdown rates increased exponentially as the stream flowed throughout the forest remnant [$y = 0.665 - 0.094\,e^{(x/94.5)}$, $R^2 = 0.79$, $P < 0.001$], with leaf mass losses varying between 20.9% and 77.6%. Leaf breakdown rates upstream the forest remnant were low, rapidly increasing as the stream flowed into the remnant, stabilizing after the first 50 m in the fragment interior, where values were 3× higher than upstream sampling points (Fig. 1a). Larger residual variation in leaf breakdown rates not explained by distance along the forest remnant was observed in sampling points within the forest remnant (Fig. 1a). The stepwise multiple regression model indicated that this residual variation was only negatively related to Forest Structure Axis 2 (Fig. 1b), and explained 56% of the variation in residuals ($y = 0.010 - 0.117x$, $R^2 = 0.56$, $P = 0.013$). Therefore, leaf breakdown rates were higher in reaches with higher tree density and lower total basal area; these results were also found when we directly adjusted a linear
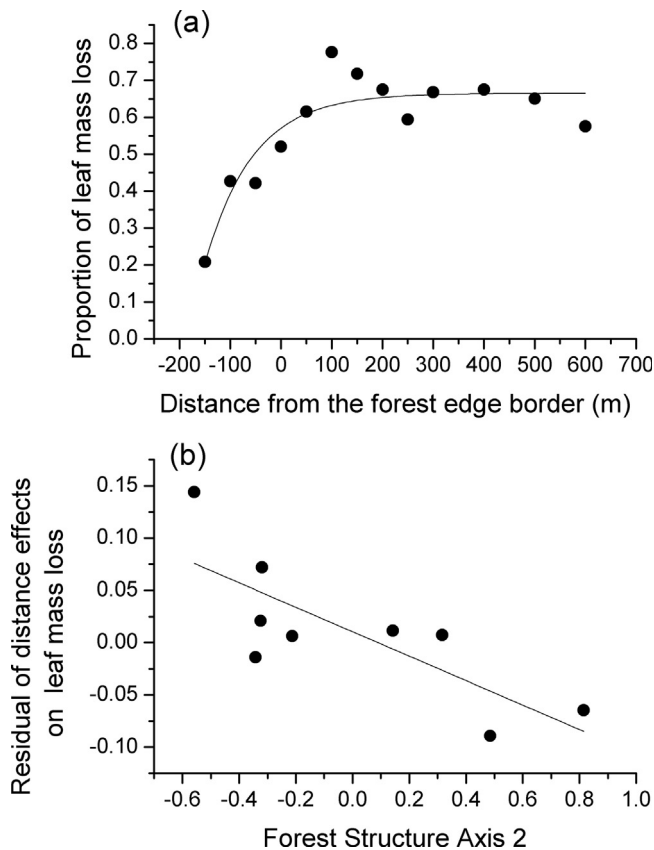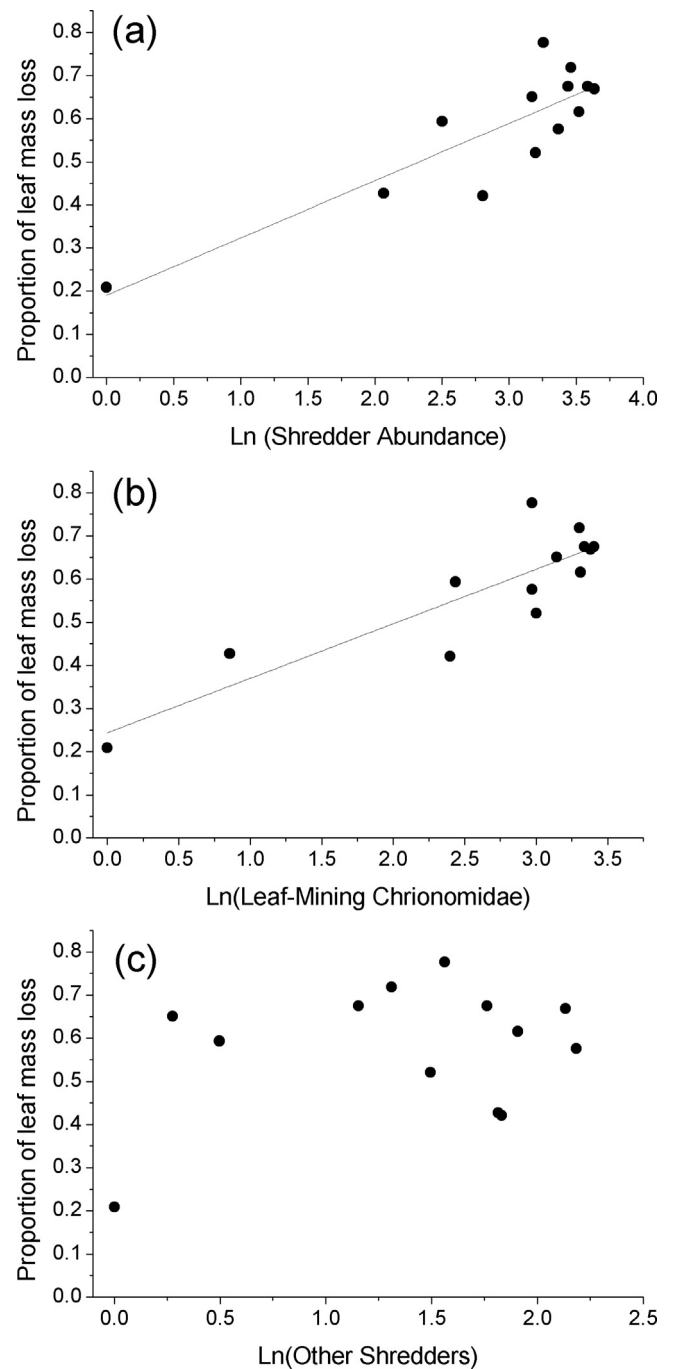
Fig. 1. Variation in leaf breakdown rates in relation to distance along the forest remnant (a), where the curve represents the fitted first-order exponential model ($R^2 = 0.79$), and relationship between the residuals of the fitted exponential model and the Forest Structure Axis 2 at each analysed reach (b), where the line represents the fitted linear regression model ($R^2 = 0.56$).

regression model of leaf breakdown rates against Forest Structure Axis 2 ($y = 0.661 - 0.097x$, $R^2 = 0.45$, $P = 0.028$).

The pattern of variation of leaf breakdown rates along the forest remnant was similar to the variation of shredder abundances in the same sampling points found by Suga and Tanaka (2013), suggesting a relationship between leaf breakdown rates and the abundance of this functional group. In fact, leaf breakdown rates increased linearly with shredder abundance (Fig. 2a), ($y = 0.190 + 0.133x$, $R^2 = 0.72$, $P < 0.001$). At the point -150 m, upstream the forest remnant, shredder abundance was zero and could be an influent point (Fig. 2a). However, when we repeated this analysis excluding this point, the relationship between leaf breakdown rates and shredder abundances was still significant ($y = 0.116 + 0.156x$, $R^2 = 0.41$, $P = 0.015$), indicating that it did not determine this relationship.

Shredders included the following taxa: Calamoceratidae, Hyallelidae, and leaf-mining Chironomidae. The latter constituted most of the shredders (61.2–98.0% of all shredders), except for the point -100 m (28.6%), which was dominated by Calamoceratidae. Leaf breakdown rates were strongly related to the abundance of leaf-mining Chironomidae ($y = 0.244 + 0126x$, $R^2 = 0.74$, $P < 0001$) (Fig. 2b), but were not related to the abundance of the other shredders ($P > 0.30$; Fig. 2c).

Therefore, the factors that determine the variation in leaf-mining Chironomidae could be responsible for the effect of the forest remnant on leaf breakdown rates that we found in this study (Fig. 1a). In fact, the abundances of leaf-mining Chironomidae increased exponentially with distance along the forest remnant [$y = 3.204 - 0.491e^{(x/77.5)}$, $R^2 = 0.88$, $P < 0.001$] (Fig. 3a). At this


Fig. 2. Variation of leaf breakdown rates in relation to abundances of shredders (a), leaf-mining Chironomidae (b), and other shredders (c). The lines are the fitted linear models.

spatial scale, most variation in abundance (88%) was due to the gradient formed by the presence of the forest remnant. At a finer spatial scale, the residual variation not explained by this model was related to the forest structure (Fig. 3b), in a significant quadratic relationship with Forest Structure Axis 2 ($y = 0.261 - 0.129x - 1.406x^2$, $R^2 = 0.86$, $P = 0.001$).

4. Discussion

In rural landscapes, where land use is dominated by agricultural activities, the presence of forest remnants can locally improve stream water quality (Storey and Cowley, 1997; Harding et al.,
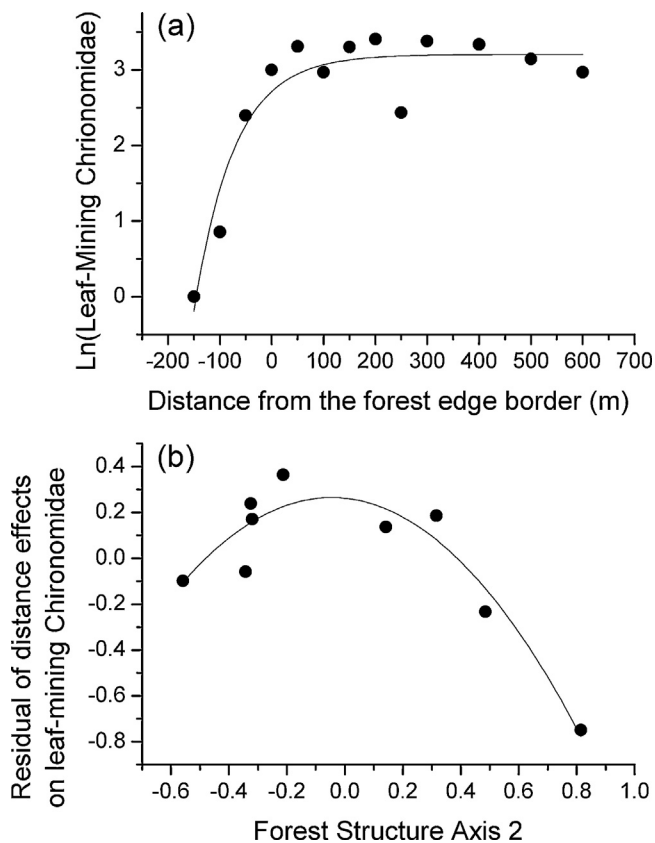
Fig. 3. Variation in abundances of leaf-mining Chironomidae in relation to distance along the forest remnant (a), where the curve represents the fitted first-order exponential model ($R^2 = 0.88$), and relationship between the residuals of the fitted exponential model and Forest Structure Axis 2 at each analysed reach (b), where the curve represents the fitted quadratic model ($R^2 = 0.86$).

2006; Arnaiz et al., 2011; Fernandes et al., 2014; Goss et al., 2014). Although an improvement in dissolved oxygen concentrations and electric conductivity can be detected, the responses of nutrient concentrations and macroinvertebrate community descriptors are more variable (Suga and Tanaka, 2013; Fernandes et al., 2014). Here we found that the presence of a forest remnant resulted in higher leaf breakdown rates within the same stream, a result commonly found in studies comparing forested and deforested streams (Sponseller and Benfield, 2001; Encalada et al., 2010). Further, higher shredder abundances were found in reaches within the forest remnant, a mechanism for the effect of the forest remnant on stream ecosystem functioning.

In studies comparing forested and deforested streams, several authors found higher leaf breakdown rates in forested streams due to higher abundances of shredders (Sponseller and Benfield, 2001), generally of a keystone taxon (Piscart et al., 2009; Encalada et al., 2010), although this is not a general result (e.g., Huryn et al., 2002; Mckie and Malmqvist, 2009). Within the same stream, Encalada et al. (2010) found higher abundances of shredders in upstream forested reaches when compared with pasture reaches downstream, whereas Goss et al. (2014) found increase in shredder abundances along forest remnants, but with no significant effects on leaf breakdown rates. Leaf-mining Chironomidae are classified as shredders in studies of Neotropical systems (Henriques-Oliveira et al., 2003; Chará-Serna et al., 2012 but see Rosemond et al., 1998), and, in our study, they contributed with most shredders. Their effects on leaf breakdown can be due to three mechanisms: (1) direct consumption of leaf material (Walker, 1988; Henriques-Oliveira et al., 2003; Chará-Serna et al., 2012), (2) physically facilitating the effects of water abrasion by mining the leaves (Rosemond et al., 1998), and (3) indirectly, as they

can be consumed by macroconsumers such as shrimps and fish, resulting in leaf breakdown (Henderson and Walker, 1986). In our study, mechanisms (1) and (2) likely predominate, because the samples were inserted in mesh bags, and no shrimp were observed in the area.

In this way, the increase in abundances of leaf-mining Chironomidae along the forest remnant is proposed as the mechanism responsible for the response of this ecosystem functioning variable to the presence of the forest remnant. The presence of the forest remnant improved stream water quality, as indicated by the non-linear increase in DO concentrations (Fernandes et al., 2014). The pattern of variation in leaf-mining Chironomidae abundances was similar to the variation in DO concentrations, suggesting that the abundance of leaf miners is related to the stream water quality. This is supported by the significant linear relationship between leaf-mining Chironomidae abundances and DO concentrations ($y = -1.976 + 0.665x$, $R^2 = 0.67$, $P < 0.001$). These results indicate that the presence of the forest remnant improved stream water quality, resulting in increased shredder abundances and associated increases in leaf breakdown rates.

An interesting result found in our study was the effect of the riparian forest structure, which explained additional variation unrelated to simple forest cover. Leaf breakdown rates were higher in reaches with higher tree density and lower total basal area. Differences in the composition of riparian forest communities can influence leaf breakdown rates within streams (Kominoski et al., 2011), macroinvertebrate community structure (Hernandez et al., 2005), large woody debris (Paula et al., 2011), and decomposer consumption rates (Kominoski et al., 2012). Further, Stone and Wallace (1998) found that shredder abundances increased along riparian forest succession after clearcut logging, stabilizing five years after the impact. These responses can be related to changes in the composition of the riparian forest along the succession, with differences in the quality of the available leaf detritus, and changes in the stream environment (Stone and Wallace, 1998). The effects of accumulated distance along the forest remnant were not correlated with the structure of the riparian forest in the current study (Fernandes et al., 2014), and the riparian forest presented small-scale variations possibly due to local disturbances. During our study, we verified that many trees in the forest showed marks of lightning strikes, and clearings could be found dominated by different plant species. Therefore, the riparian forest structure was not homogeneous in the forest remnant, with parts of the forest in different successional stages, causing heterogeneity in stream dynamics.

In this study, leaf breakdown rates were related to effects at two spatial scales (presence of the forest remnant and internal heterogeneity of the remnant), and leaf-mining Chironomidae similarly varied over these spatial scales suggesting that variation in Chironomids are the mechanism explaining the stream functional response to the forest remnant. The abundances of leaf-mining Chironomidae integrate the variation at both spatial scales, reflecting not only the effect of the forest remnant, but also the effect of heterogeneity resulting from differences in riparian forest structure. Therefore, changes in riparian forest structure at small spatial scales due to local disturbances can contribute to variation both in the structure of aquatic communities and stream ecosystem functioning (Souza et al., 2013; Fernandes et al., 2014). These spatial changes resulting both from the presence of forest remnants and the forest structure can contribute to stream ecosystem functioning.

5. Conclusion

The presence of riparian forest remnants can be important to improve stream water quality (Storey and Cowley, 1997; Fernandes

et al., 2014) and ecosystem function (this study) in rural, deforested landscapes. We found that leaf breakdown rates rapidly increased as the stream entered the remnant, stabilizing within the first 100 m, beyond which variation in leaf breakdown rates was explained by riparian forest structure. In our study, the forest remnant enabled the permanence of an important macroinvertebrate functional group (shredders), which possibly influenced leaf breakdown rates. This suggests that the conservation of even relatively small fragments of forest may be able to conserve important ecosystem functions that are otherwise absent in these landscapes. However, it is still not known for how long downstream these functions are maintained after the stream leaves the forest fragment. Our study was conducted on a single stream, and differences in land-use patterns, fragment sizes, and regional faunal composition could lead to distinct mechanisms in leaf breakdown and, therefore, in different patterns. The arrangement of forest remnants in landscapes and the heterogeneity among remnants resulting from differences in forest structure can be important for the management of water quality (Goforth and Bain, 2010; Miserendino et al., 2011) and biodiversity (Chazdon, 2003; Brooks et al., 2012). Hence, to evaluate these effects on stream ecosystem functions, it will be necessary to evaluate multiple forests under differing land-use contexts to assess the generality of these forest remnant effects on small streams.

Acknowledgements

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