Hydrobiologia (2021) 848:3503–3520
https://doi.org/10.1007/s10750-021-04604-y

PRIMARY RESEARCH PAPER

Riparian forests buffer the negative effects of cropland on macroinvertebrate diversity in lowland Amazonian streams

N. C. S. Marques · K. J. Jankowski · M. N. Macedo · L. Juen · A. Luiza-Andrade · L. A. Deegan

Received: 17 November 2020 / Revised: 13 April 2021 / Accepted: 21 April 2021 / Published online: 6 May 2021

Abstract Riparian forests regulate stream ecosystems and biodiversity. Therefore, changes to riparian structure may threaten stream ecosystem function by triggering taxonomic and functional changes to aquatic communities. Because macroinvertebrate assemblages are sensitive to environmental changes, they can be effective indicators of stream integrity in disturbed landscapes. To assess the role of riparian forests in maintaining tropical stream communities in areas experiencing large-scale watershed disturbance, we quantified the taxonomic and functional response of stream macroinvertebrate communities to forest clearing in the southeastern Amazon's agricultural frontier, a region experiencing widespread deforestation. Our results show that watershed deforestation can lead to significant changes in macroinvertebrate richness and community composition. We found a predominance of shredders in forested watersheds; scrapers in cropland watersheds with riparian forests; and collector-filterers in cropland watersheds without riparian forest buffers. Taxonomic composition was controlled by available organic material in forested watersheds and by periphyton in cropland sites regardless of whether they had a riparian buffer. Our results show that the clearing of riparian forests alters food sources supporting aquatic food webs, leading to ecosystem-level shifts through changes in light and temperature dynamics that affect aquatic communities in areas with intense land-use change such as the southeastern Amazon.

Handling editor: Verónica Ferreira.

Electronic supplementary material The online version of this article (https://doi.org/10.1007/s10750-021-04604-y) contains supplementary material, which is available to authorized users.

N. C. S. Marques (✉) · M. N. Macedo
Instituto de Pesquisa Ambiental da Amazônia, Canarana, Mato Grosso, Brazil
e-mail: marques.ncs@gmail.com

N. C. S. Marques
Laboratório de Ecossistemas, Universidade de Brasília, Brasília, Brazil

K. J. Jankowski
U.S. Geological Survey, Upper Midwest Environmental Sciences Center, La Crosse, USA

M. N. Macedo · L. A. Deegan
Woodwell Climate Research Center, Falmouth, USA

L. Juen · A. Luiza-Andrade
Programa de Pós-graduação em Ecologia, Instituto de Ciências Biológicas, Universidade Federal do Pará, Belém, Brazil

Keywords Amazon · Freshwater · Food webs · Functional diversity · Stream · Tropics

Introduction

Riparian forests provide essential protection for streams and represent an important link between terrestrial and aquatic ecosystems (Naiman et al., 2005). The benefits of riparian forests to stream structure and function include moderating stream water temperature (Osborne and Kovacic, 1993; Fernandes et al., 2014); increasing habitat diversity and food sources to stream organisms and sustaining habitat integrity and biodiversity (Naiman et al., 1993; Naiman and Décamps, 1997; Tolkkinen et al., 2020). Riparian forests may also serve as important landscape "filters" by reducing the input of pesticides and fertilizers to streams (Shortle et al., 2001) and contributing to bank stability, which reduces erosion and sediment inputs to streams (Naiman and Décamps, 1997.

Despite a large body of knowledge on the importance of conserving riparian forests, widespread deforestation of riparian zones continues to threaten stream and river ecosystems globally (Sweeney et al., 2004; Luke et al., 2019; Dala-Corte et al., 2020). The removal of riparian forests has implications for stream ecosystem function, including shifts in the energy base of stream food webs. It often results in a transition from allochthonous to autochthonous energy flows, because removal of stream-side trees tends to both reduce terrestrial subsidies and increase the light available to fuel primary production (Bormann et al., 1974; dos Reis Oliveira et al., 2020). Riparian forest degradation can also change the diversity and distribution of forest functional traits (e.g., phenology, chemical composition), with consequences for aquatic ecosystem function (Kominoski et al., 2013), including altering nutrient cycling (Pert et al., 2010), organic matter decomposition (e.g., Tank et al., 2010), and food web structure (Erdozain et al., 2019).

Given their important functions, the preservation of riparian forests has taken on increasing importance, particularly in tropical regions where agriculture is expanding and intensifying rapidly (Gibbs et al., 2010; Nunes et al., 2015). The Brazilian Forest Code places strict limits on deforestation of riparian and upland forests in agricultural regions of the Amazon, requiring landowners to conserve 35% to 80% of their property as intact forest and maintain riparian zones as "Areas of Permanent Protection" (Soares-Filho et al., 2014). However, it is not yet clear to what extent the Forest Code (Law N° 12.651; Soares-Filho et al. 2014) has succeeded in protecting riparian forest buffers or the streams and rivers they surround (Azevedo et al., 2017, Nunes et al., 2019). A recent meta-analysis found a sharp decline in invertebrate indicator biodiversity with under 10% loss of native vegetation in a 50 m buffer, suggesting that riparian buffers are critical to maintain stream function (Dala-Corte et al., 2020).

Benthic macroinvertebrates represent a critical link between terrestrial and aquatic systems and have long been used as indicators of stream integrity (e.g., Barbosa et al., 2001; Rizo-Patrón et al., 2013; He et al., 2019; Oliveira Júnior & Juen, 2019). For example, the occurrence and abundance of the EPT (Ephemeroptera, Plecoptera and Trichoptera) are often used as proxies to assess ecosystem integrity (e.g., Lake, 2000; Péru and Doledec, 2010). The degradation and loss of riparian forests tend to simplify macroinvertebrate communities (i.e., loss of species) and change the availability and variety of food resources, thus affecting the functional composition of aquatic insect communities (Firmiano et al., 2021). Changes to stream macroinvertebrate communities in response to land-use change have been well documented around the world (e.g., Sponseller et al., 2001; Tanaka et al., 2016; Ferreira et al., 2017), yet we know relatively little about the response of aquatic insect communities to the loss or degradation of riparian forests in the Amazon-Cerrado region (e.g., Juen et al., 2016; Leal et al., 2016; Paiva et al., 2017; Faria et al., 2017; Cardoso et al., 2018; Luiza-Andrade et al., 2020).

To understand if riparian forest degradation affects the diversity and function of stream communities in this rapidly developing region, we evaluated stream macroinvertebrate communities in headwater streams in forested watersheds ("Forest"), cropland watersheds with intact riparian forests ("Crop-Rip"), and cropland watersheds without riparian forest buffers ("Crop-NoRip) in the southeastern Amazon state of Mato Grosso, Brazil. Our study addressed two key questions: 1) Has the loss or removal of riparian forests altered the taxonomic or functional composition of aquatic insect communities (i.e., abundance, richness and diversity of species and functional feeding groups)? 2) Which environmental factors are the most important determinants of these observed changes and do they differ for functional and taxonomic diversity?

We hypothesized that forest streams would have more suitable habitat and support more abundant and diverse EPT assemblages, compared with streams in cropland watersheds. We expected a reduction in taxonomic and functional (measured as functional feeding groups; FFG) richness for aquatic insect communities in cropland streams without riparian forest, but anticipated that cropland streams with riparian forest would retain higher taxonomic and FFG richness (Question 1). Given that watershed deforestation has altered stream physical conditions such as discharge (Hayhoe et al., 2011; Riskin et al., 2017), temperature (Macedo et al., 2012), benthic organic matter standing stocks (K.J. Jankowski, U.S. Geological Survey, written comm., 2021), and riparian forests species composition (Maracahipes-Santos et al., 2020), we expected those factors to exert a strong influence on macroinvertebrates taxonomic and functional composition (Question 2). More specifically, in cropland streams with no riparian forest we expected to find a reduction in the abundance of shredders due to decreased litter input, but an increase in scrapers and collector-filterers due to a higher abundance of algal primary producers.

Methods

Study area

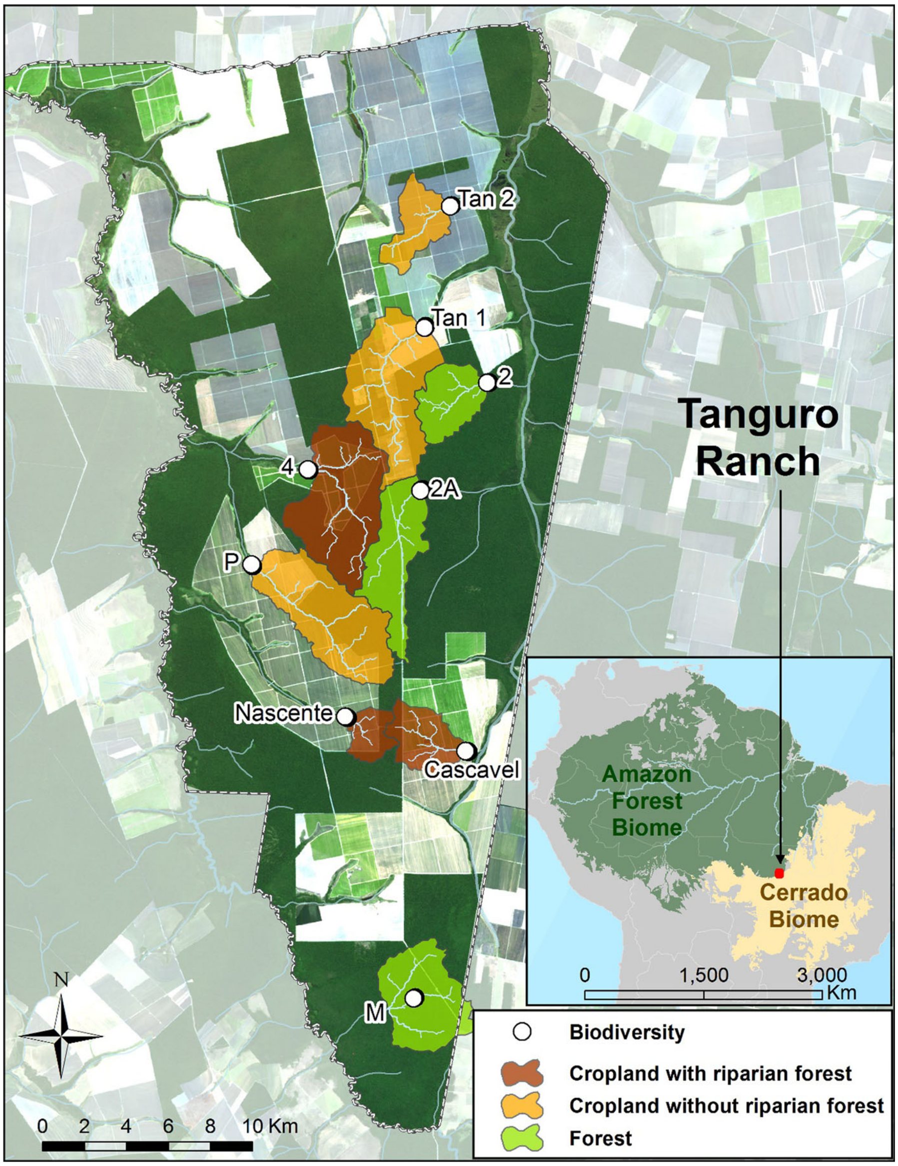
We conducted this study at Tanguro Ranch, an 800 km^2 farm located in the headwaters of the Xingu River in the state of Mato Grosso, Brazil (Fig. 1; Appendix A). Regional climate is characterized by high mean monthly temperatures (above 25°C). Annual precipitation varies from 1,700 mm to 2,200 mm (Balch et al. 2008), with a pronounced dry season from May to September (rare rainfall events > 10 mm).

Forests in the region are evergreen, semi-deciduous, seasonally dry transitional forests containing a mixture of tree species from ombrophilous rainforests (*i.e.* high annual rainfall) in the north to woody Cerrado (tropical savannas; lower annual rainfall) in the south. Tanguro Ranch is representative of the type and chronology of land-use changes occurring in the region. Deforestation for cattle pastures occurred in the mid-1980s, followed by conversion to soybean croplands occurring into the early 2000s (Macedo et al., 2012). Early forest clearing for pastures often included the removal of riparian forest to the stream edges, but the majority of cropped watersheds at Tanguro Ranch have wide riparian buffers that exceed Forest Code requirements (Macedo et al., 2012). Broad plateaus dominate the Tanguro Ranch landscape, with a gentle break sloping towards stream channels. Soils are deep, acidic and highly permeable (Neill et al., 2013, Jankowski et al., 2018). Several headwater streams are contained within the boundaries of Tanguro Ranch and drain either forested or cropland watersheds (Fig. 1). Further, cropland streams at Tanguro Ranch include a range of riparian forest protection—some watersheds have been completely deforested, while others have intact riparian forest buffers that are 30-200 m wide. Our study included nine streams, three in each of the following categories according to watershed land cover and riparian forest condition (Appendix A and B): completely forested watersheds (“Forest”); cropland watersheds with riparian forests (“Crop-Rip”); cropland watersheds without riparian forest buffers (“Crop-NoRip”).

Environmental variables

We measured standard environmental variables (following the protocol of Peck et al., 2006) at the time of macroinvertebrate sampling, including (i) habitat characteristics: substrate type, flow, presence of large wood; (ii) stream morphology: sinuosity, slope, width, depth; (iii) riparian structure: tree cover density, margin shading and; (iv) human disturbance in the channel and riparian forest: presence of pasture and crops. We selected variables that best described potential changes in EPT microhabitat and riparian forest characteristics in our analyses, based on previous work in similar Amazon streams (e.g., Luiza-Andrade et al., 2017; Paiva et al, 2021) (Appendix C). The variables included: (i) percentage of substrate cover (organic material, wood, macrophyte, periphyton); (ii) flow microhabitat (categorized as fast

Fig. 1 Tanguro Ranch. *Inset:* Red point indicates the location of Tanguro Ranch in the Amazon-Cerrado transition zone. Biodiversity: sampling points



[percentage covered by cascades, rapids, riffles and falls] and slow [percentage covered by glides and pools]) and; (iii) riparian forest structure (presence of large trees, woody understory, and herbaceous understory). Environmental variables were measured in 10 segments for each stream reach (Fig. 2; see *Macroinvertebrate Sampling* for details), and the mean and standard deviation calculated for the final analyses. In our study, each stream is defined as a sampling unit.

We measured pH at the top, middle and base of each 150 m stream reach using a multiparameter handheld meter (YSI Inc, Yellow Springs, Ohio, USA). Water temperature was measured every hour in each stream using a Hobo water level logger (Onset Computer, Bourne, Massachusetts, USA). We used the average temperature for August 2017 to characterize water temperature in each stream.

Macroinvertebrate sampling

We sampled all nine streams over nine days during the dry season of 2017 (August 07-15). In each stream, we sampled a 150 m reach that was subdivided into ten 15 m sections (Fig. 2). Each section was further subdivided into three segments of 5 meters (a total of 30 segments). We sampled macroinvertebrates in

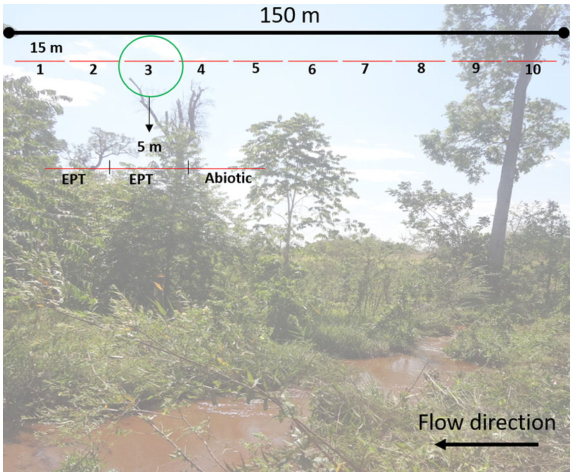


Fig. 2 Sampling design of EPT and environmental variables. In each stream, we sampled a 150 m reach that was subdivided into ten 15 m sections. Each section was further subdivided into three segments of 5 meters (a total of 30 segments)

the first two segments of each section and used the third segment to assess the environmental variables of the next upstream section, thus avoiding disturbance of the substrate before sampling (Juen et al. 2016; Faria et al. 2017; Shimano et al. 2018). We systematically assessed environmental variables and biodiversity by sections to ensure that the entire 150 m reach was equally represented. This approach increased the likelihood of capturing a representative sample of macroinvertebrate biodiversity. To estimate reach-scale values for abiotic and biotic variables, we used the subsamples to calculate the mean and standard deviations for environmental variables (10 samples, 1 per section), as well as total species abundance (20 samples, two per segment).

Macroinvertebrates were sampled with a hand net (known in Brazil as a “rapiché”; 18 cm diameter and mesh opening of 250 μm) following a systematic zigzag trajectory along each stretch (Shimano and Juen, 2016; Luiza-Andrade et al., 2017; Paiva et al., 2017). We identified only the individuals belonging to the orders Ephemeroptera, Plecoptera and Trichoptera. All functional groups were well represented among these three orders in our study streams. The specimens collected were fixed and stored with 85% ethanol and identified to the genus level using specialized literature specific to the Amazon-Cerrado region (Domínguez et al., 2006; Lecci and Froehlich, 2007; Hamada et al., 2014). Studies evaluating anthropogenic effects on aquatic insects at the taxonomic resolution of genus have presented consistent results (e.g., Ligeiro et al., 2013; Bertaso et al., 2015; Luiza-Andrade et al, 2017; Godoy et al., 2019; Paiva et al., 2021). Following processing, all specimens were deposited in the Zoological Collection of the Federal University of Pará (UFPA).

Functional feeding groups

Feeding traits are particularly useful functional traits since they indicate changes in the fundamental food sources supporting aquatic food webs (Poff et al.,2006). All individuals were classified into five functional feeding groups (*according to* Cummins and Klug, 1979; Appendix D): predators, scrapers, shredders, collector-gatherers and collector-filterers. The functional traits’ values were transformed using the Fuzzy coding approach (Chevenet et al.,1994). This method works as an affinity index, that considers differences registered (by different authors) for the same characteristic of the genus, taking into account the biological group’s trophic plasticity. The Fuzzy method score ranges from zero (no affinity) to three (high affinity). In the literature, we find records for specimens that contained two or more types of FFG categories. For example, the genus *Caenis* presents the values 0.4 in the Collector-Gatherer category and 0.6 for Scraper in the table (Supplementary Material - Appendix D). These values indicate that 40% of the consulted articles classified the genus in one FFG (in the Collector-Gatherer category), while the other 60% classified the same genus in another (such as Scraper). The fuzzy transformation converts the affinity into a percentage and separates the categories by blocks whose sum of the lines totals the value one. We used this approach because a particular genus can exhibit trophic plasticity, and we wanted to better reflect the range of potential feeding strategies represented by sampled taxa.

Data analysis

Abundance, richness, and diversity of EPT and FFG

To evaluate whether the loss or removal of riparian forest altered the abundance, richness, or diversity of

EPT at the genus level, we calculated the total number of individuals found in streams for each land use/management category (abundance) and used them as an estimate of taxonomic richness. Individuals from each stream were further classified into five functional feeding groups based on their genus, allowing us to calculate functional richness. Taxonomic and functional richness abundance were compared separately among streams of each riparian forest category using a univariate Analysis of Variance (ANOVA). Where significant differences were found across riparian forest categories ($p<0.05$), a post hoc Tukey test was used for pairwise comparison of differences among riparian forest categories.

To estimate the taxonomic diversity across riparian forest categories, we created rarefaction curves using individual-based accumulation (Gotelli and Colwell, 2011). The rarefaction curve method plots the number of individuals on the x-axis against the number of genera on the y-axis. We then calculated the mean taxonomic diversity across streams in each riparian forest category, as well as their confidence interval. We used the same analytical approach to evaluate FFG diversity.

To test if we sampled the communities sufficiently to capture their species richness we performed a sampling coverage test (proposed by Chao and Jost, 2012) using the rareNMtests (Cayuela and Gotelli, 2014) and Vegan packages in R (Oksanen et al., 2013).

Taxonomic and FFG community composition of EPT

To evaluate if community composition differed significantly among the three types of riparian forest categories, we used a Permutational Multivariate Analysis of Variance (PERMANOVA) using the Vegan package in R (Oksanen et al., 2013). PERMANOVA is a non-parametric analysis (analogous to MANOVA) that allows for the use of non-Euclidean distance measures among samples (Anderson, 2001). We ran the analysis using the Bray–Curtis distance metric and the *adonis* function. We also tested for the "homogeneity of group dispersion" which is a PERMANOVA assumption available in the PERMDISP (*betadisper*) function. This test evaluates variance within groups by using the average distance of group members (i.e., riparian forest categories) to the group centroid. It is a multivariate analog to Levene's test for homogeneity of variance (Anderson, 2001). To evaluate how many genera were shared and how many genera were unique among the three riparian forest stream categories (Forest, Crop-Rip, Crop-NoRip) we used a Venn Diagram (VennDiagramn package; Chen & Boutros, 2011). We used these same analytical approaches to evaluate if functional composition significantly changed among the three riparian forest categories.

Environmental drivers of taxonomic and FFG community composition

To examine which environmental variables most influenced taxonomic and functional composition we used a Redundancy Analysis (RDA; Rao, 1964). RDA is a constrained ordination technique that evaluates the effects of environmental variables on community composition (ter Braak and Verdonschot 1995) and is particularly useful when there are underlying linear abundance distributions. We applied a detrended correspondence analysis (DCA) to our data to determine the appropriate response model (linear [RDA] or unimodal [Canonical Correspondence Analysis]). The length of the first DCA axis was < 4 SD (units of standard deviation), indicating a homogenous dataset well suited to analysis with linear methods like RDA (Lepš and Šmilauer, 2003). We performed two separate RDAs, using either taxonomic (Appendix E) or FFG relative abundance (Appendix F) data as response variables and environmental factors as explanatory variables. To reduce the effect of multiple zeros in our abundance matrices, we did a Hellinger transformation (Dray & Dufour, 2007) to express each abundance value as a proportion of the total abundance. Environmental data were standardized to make them dimensionally homogenous, allowing direct comparison of regression coefficients (Legendre & Legendre, 2012). Before including the selected environmental variables (*see "environmental variables" section*) in the RDA, we tested for collinearity, removing variables that had a variance inflation factor higher than 10 (VIF $>$ 10). We then used forward selection of non-collinear environmental variables to select the variables that explained the most variance in the abundance data. The forward selection was done using the function "ordiR2step." The significance of the RDA was tested by an ANOVA-like permutation test. All analyses were done in the Vegan package in R (Oksanen et al., 2013).

Results

Abundance, richness, and diversity of EPT and FFG

We found a total of 465 individuals and 18 genera belonging to Ephemeroptera, Plecoptera and Trichoptera (EPT) orders across the nine streams (Appendix E). EPT insects were most abundant in the forest sites (225 individuals), followed by Crop-NoRip (138) and Crop-Rip (102 individuals) sites. The most abundant genera were *Macronema* (109 individuals), *Campylocia* (101 individuals) and *Leptonema* (95 individuals) (Appendix GA). The most abundant functional feeding guilds were collector-filterers (198 individuals) and predators (186 individuals) (Appendix GB). We found no differences in taxonomic abundance among watershed types (ANOVA; $P = 0.34$, $F = 1.262$, Appendix F). We did find differences in the abundance of functional feeding groups of collectors-filterers (ANOVA; $F = 0.142$, $P = 0.001$, Table 1) and shredders (ANOVA; $F = 37.22$, $P < 0.001$, Table 1), when comparing forested watersheds to both categories of cropland watersheds. The largest differences occurred between forested watersheds and Crop-NoRip watersheds (Table 2).

EPT taxonomic richness was similar in Forest sites (number of taxa: 11) and Crop-Rip (number of taxa: 11), but lower in Crop-NoRip (number of taxa: 9). We found no significant difference in the average EPT taxonomic diversity among riparian forest categories (Fig. 3A). The average EPT functional diversity in Crop-NoRip streams was significantly lower than that of Forest or Crop-Rip streams, which were similar (Fig. 3B). The rarefaction analysis showed that species accumulation curves were still increasing in all three riparian forest categories and did not reach a horizontal asymptote, indicating that more or larger samples are needed to fully represent absolute species richness (Fig. 4).

The sampling coverage test, on the other hand, showed that our samples had coverage values higher than 95% (Forest = 99%; Crop-Rip = 98%; Crop-NoRip = 98%), indicating that our sampling effort adequately characterized the EPT richness of these streams.

Taxonomic and FFG community composition of EPT

EPT taxonomic composition differed significantly among Forest, Crop-Rip, and Crop-NoRip sites (PERMANOVA, $F = 4.93$; $R^2 = 0.6$; $P = 0.02$), but within-group variance was similar (betadisper, $F = 0.5$; $P = 0.62$). Taxa that differed among riparian forest categories varied among riparian forest category pairs (Fig. 5A). Crop-Rip and Crop-NoRip streams shared a total of four genera absent in Forest streams: *Anacroneuria, Caenis, Cernotina* and *Nectopsyche*. Three genera were shared among all three riparian forest categories*: Leptonema, Macronema* and *Oecetis* (Fig. 5A).

The centroids of the functional community composition did not differ (PERMANOVA, $F = 1.46$; $R^2 = 0.32$; $P = 0.25$) and were similar in the amount of functional variation (betadisper, $F = 0.56$; $P = 0.59$) among the three riparian forest categories. Shredders were the only functional feeding guild that was not present in all riparian forest categories. Shredders were abundant in Forest streams, rare in Crop-Rip streams (i.e., only one individual found; Fig. 5B) and absent in Crop-NoRip streams.

Environmental drivers of taxonomic and FFG community composition

Taxonomic composition- Seven environmental variables had a VIF < 10 and four were selected (by forward selection) for inclusion in the RDA analysis (Appendix H). Environmental variables explained a total of 70.6% of variation (51.1% by first axis, 19.5% by second axis) in the taxonomic composition of streams. The EPT composition of Forest sites was distinct in multivariate space, compared with cropland sites, whether they had a riparian buffer or not (Fig. 6A). This separation was driven by four environmental variables that are likely related to land use: organic material, low flow microhabitat, periphyton, and temperature. EPT forest communities were associated with high levels of organic material and presence of low flow microhabitat. Communities in Crop-Rip and Crop-NoRip streams corresponded with higher temperature and periphyton. The ANOVA-like

Table 1 Results of ANOVA for abundance of functional feeding groups among the land-cover-riparian forest categories (Forested, Crop-Rip and Crop-NoRip)

	Df	Sum	Mean	*F*	Pr(> *F*)
Collector gatherers					
Riparian forest categories	2	142.9	71.4	0.14	0.87
Residuals	6	3026	504.3		
Collector-filterers					
Riparian forest categories	2	1286	643	21.6	**<0.001**
Residuals	6	178	29.7		
Scrapers					
Riparian forest categories	2	106.9	53.4	0.71	0.52
Residuals	6	449.3	74.8		
Predators					
Riparian forest categories	2	54	27	0.05	0.94
Residuals	6	3066	511		
Shredders					
Riparian forest categories	2	264.6	132.3	37.2	**<0.001**
Residuals	6	21.3	3.5		

Bold numbers indicate statistical significance ($P < 0.05$)

Df degrees of freedom, *Sum* sum of squares, *Mean* mean of squares

Table 2 Results of Tukey post hoc pairwise comparison among the riparian forest categories (Forest, Crop-Rip, Crop-NoRip)

	diff	lwr	upr	*P*
Collector filters				
Forest:Crop-NoRip	18	4.3	31.6	**0.016**
Crop-Rip:Crop-NoRip	− 11	− 24.6	2.6	0.106
Crop-Rip:Forest	− 29	− 42.6	− 15.3	**0.002**
Shredders				
Forest:Crop-NoRip	11.6	6.9	16.3	**0.001**
Crop-Rip:Crop-NoRip	0.3	− 4.3	5.0	0.974
Crop-Rip:Forest	− 11.3	− 16	− 6.6	**0.001**

Bold numbers indicate statistical significance

diff mean difference between groups, *lwr* lower boundary, *upr* upper boundary

permutation test showed that the RDA results were significant ($F = 3.57$; $P = 0.003$; DF = 4).

Functional composition- Seven environmental variables had a VIF < 10 and six were selected (by forward selection) for inclusion in the RDA analysis (Appendix H). Environmental variables explained 82.6% of the variation (52.9% by first axis; 29.7% by second axis) in functional composition. Interestingly, although the PERMANOVA found no significant compositional differences, the RDA showed that functional composition of Forest sites was distinct from sites with Crop-Rip and Crop-NoRip (Fig. 6B), similar to the taxonomic analysis. Functional differences were driven by six environmental variables: temperature, large wood, organic material, periphyton, low flow microhabitat and macrophytes. Composition in Forest streams was best explained by high organic material, low flow microhabitat, and the presence of wood, whereas communities in the Crop-Rip and Crop-NoRip sites corresponded with higher temperatures, periphyton, and macrophytes. The ANOVA-like permutation test showed that the RDA was significant ($F = 3.45$; $P = 0.04$; DF = 6).

Discussion

Our results showed that watershed deforestation resulted in changes in EPT macroinvertebrate community composition in small tropical streams. Interestingly, although we found significant differences in the taxon composition among riparian forest categories, functional composition remained similar, indicating that streams have retained some EPT functional resiliency to watershed-scale change. We detected subtle differences in EPT functional groups among Forest, Cropland with riparian forest and cropland without riparian forest indicating that functional capacity may shift over time as the landscape is altered (see Appendix A). Individual EPT functional groups showed different associations with sites and

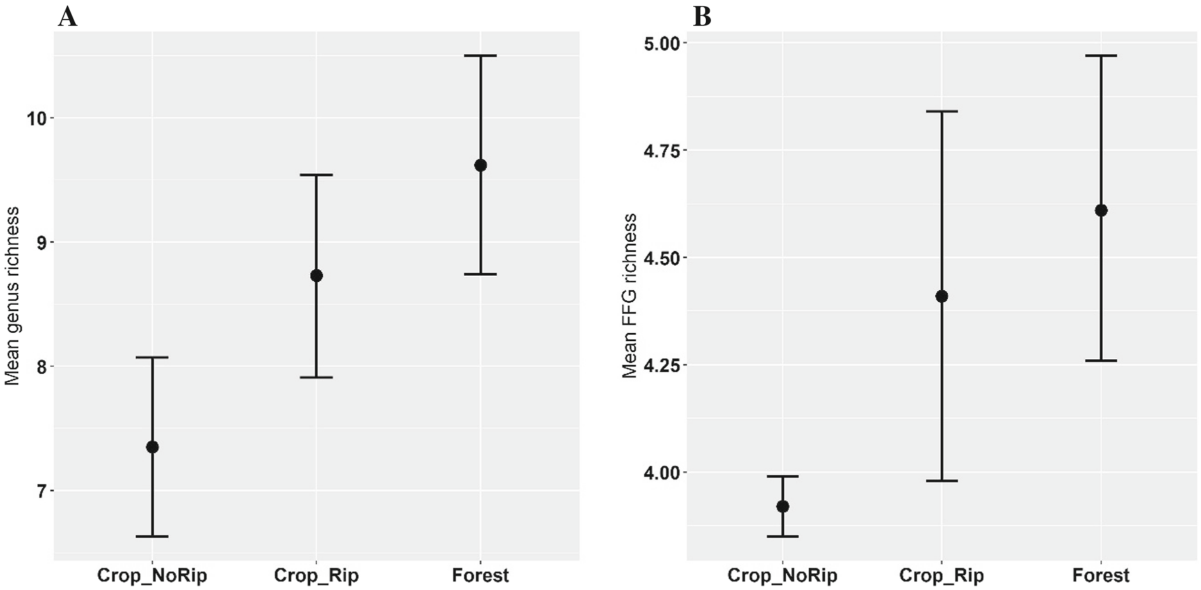


Fig. 3 Mean diversity and the confidence interval (95%) for the three riparian forest categories. EPT taxonomic diversity (**A**); FFG diversity (**B**)

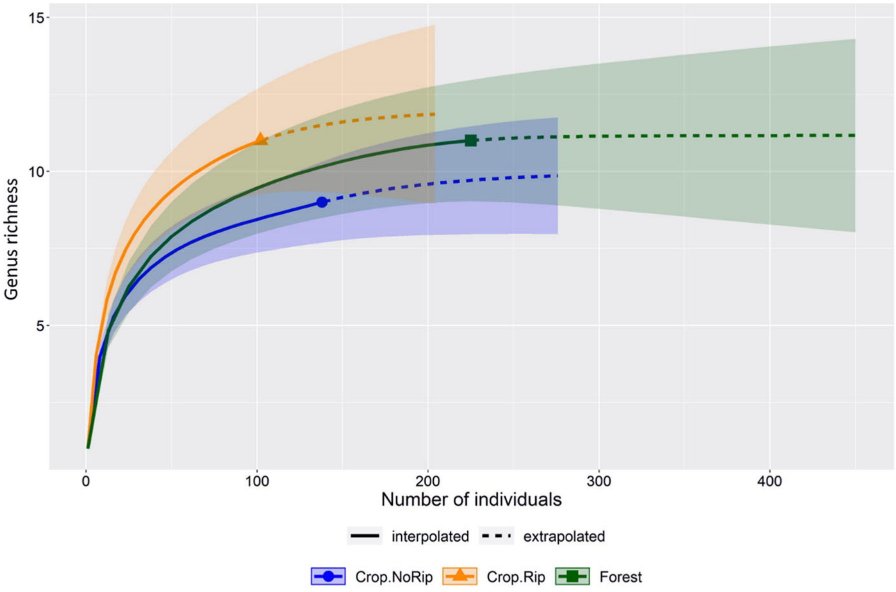


Fig. 4 Rarefaction curves for EPT genera collected in three riparian forest categories. The curve is based on the number of individuals recorded

environmental variables, however. Shredders showed the strongest differences in abundance among stream types. They were abundant in Forest streams, rare in cropland streams with riparian buffers, and absent in cropland streams with no riparian forest. Shredders and collector-gatherers were more abundant in Forest streams, which had higher organic material (K.J. Jankowski, U.S. Geological Survey, written comm., 2021) and more low flow microhabitat. Cropland streams at Tanguro Ranch have 2-4 times greater discharge on average than forested streams, as a result of reduced watershed evapotranspiration and greater runoff (Hayhoe et al., 2011, Riskin et al., 2017), which reduces the area of low flow habitat preferred by some macroinvertebrate species. Scrapers were more abundant in Crop-Rip sites, which had higher values of

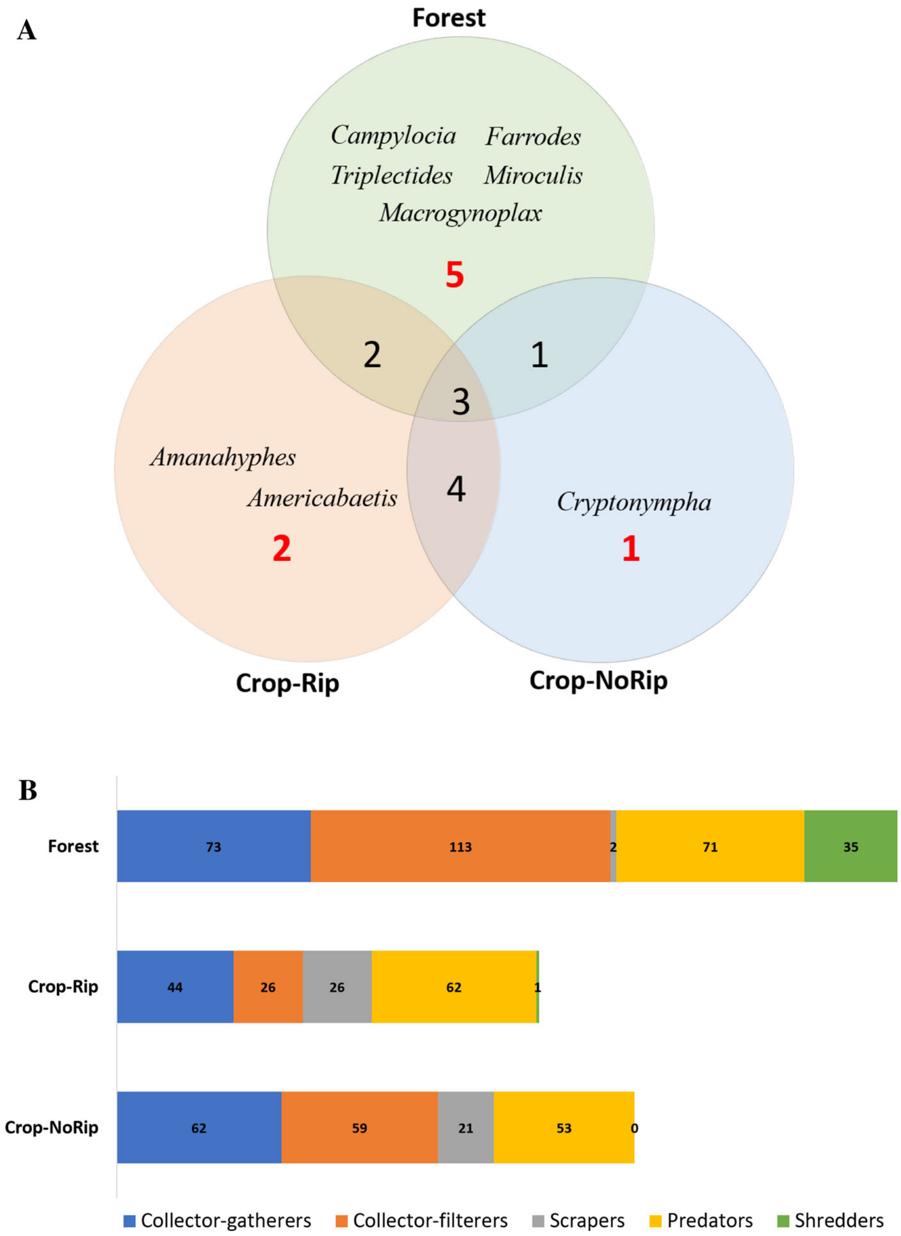


Fig. 5 Venn diagram showing the number of genera shared among the three riparian forest categories, as well as the number of the genera (in red) that are unique to each category (**A**). Bar chart showing how many individuals of each FFG were found in each riparian forest category (**B**)

periphyton and higher water temperature. Collector-filterers, the most generalist feeding type, were more common in Crop-NoRip sites associated with the presence of wood. These results indicate that streams on either end of the riparian forest categories (i.e., Forest and Crop-NoRip) had the most distinct functional communities. However, Crop-Rip and Crop-NoRip sites seemed to converge functionally and taxonomically due to subtle shifts in environmental conditions from watershed deforestation.

Taxonomic abundance, richness, diversity and composition

Our results support prior findings that degradation of riparian forests exerts a strong influence on macroinvertebrate communities (Siegloch et al., 2016; de Castro et al., 2018; Dala-Corte et al., 2020) and indicate that intact riparian forests provided protection against aquatic species loss in cropland watersheds (Carvalho et al., 2018). We observed a reduction in

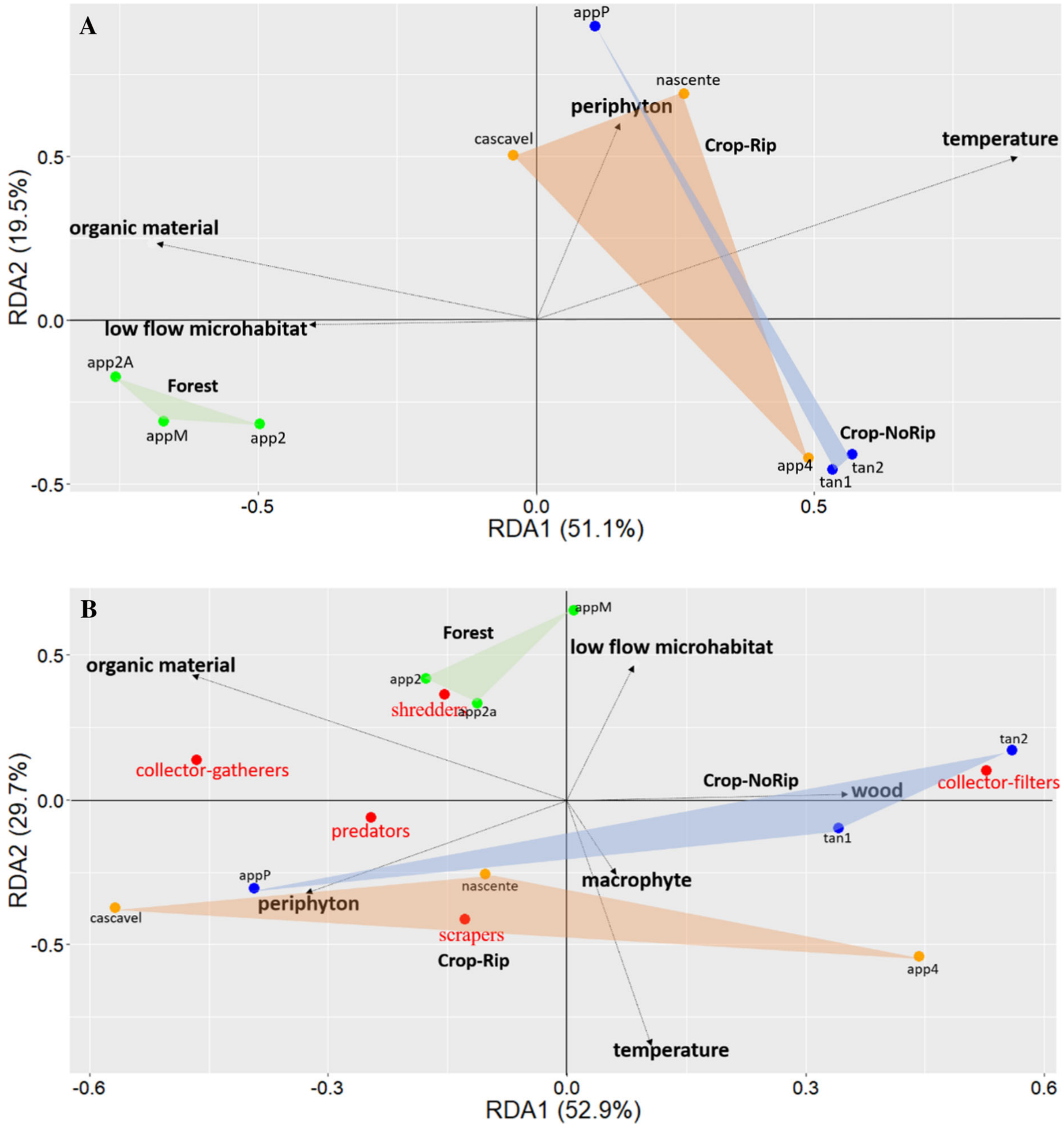


Fig. 6 Redundancy analysis (RDA) showing which environmental variables most contribute to EPT community composition in the nine streams across three riparian forest categories. Macroinvertebrate communities were classified according to taxonomic composition (A) and functional feeding group composition (B)

EPT abundance, richness, and diversity in cropland streams with no riparian forest, and more diverse macroinvertebrate communities in cropland streams with riparian forest. Forest streams had almost twice as many individuals as cropland streams, regardless of riparian buffer status, and five genera that were unique to streams with intact buffers. Riparian forest buffers in tropical systems reduce the effects of deforestation on aquatic biodiversity (Siegloch et al., 2016; de Castro et al., 2018; Dala-Corte et al., 2020) and water quality (Souza et al., 2013). Our results support those findings and are consistent with studies in other tropical systems, which have shown decreases in aquatic insect richness associated with the removal of riparian forests (Nessimian et al., 2008; Siegloch et al., 2014; Farias et al., 2018). Riparian forest protection helps mitigate some of the negative effects of watershed clearing on macroinvertebrate communities (*i.e.* by maintaining community composition), but other watershed-scale environmental factors are also important. In addition, these results show that the removal of riparian forests resulted in species loss rather than community change, at least whitin the time since deforestation at Tanguro Ranch. This underscores the importance of retaining forested watersheds in agricultural landscapes that can provide source populations to degraded or restored streams.

Although species richness in our sampled streams is low compared with other streams in the Amazon and Cerrado (e.g., Luiza-Andrade et al., 2017; de Castro et al., 2018), our sampling was sufficient, and our samples represented the full range of FFG variability (see "sampling coverage test"). Future work could help resolve whether the lower number of species is attributable to the study site's location in the Cerrado-Amazon transition or simply a reflection of interannual variability. In addition, the number of species does not fully characterize important aspects of biodiversity. Fundamental ecosystem processes, such as degradation of organic matter and productivity, can be threatened by the loss of functional groups and not necessarily by the number of genera (Mouillot et al., 2013).

Functional community composition

Understanding changes in functional community composition in response to disturbance can often provide more insight into potential shifts in ecosystem function and future trajectories than metrics of species composition alone (Petchey and Gaston, 2002, Poff, 1997). As a result, the analysis of functional traits has been widely used to understand the response of macroinvertebrate communities to environmental change (Poff et al., 2006). We hypothesized that both taxonomic and functional composition would shift in response to watershed deforestation and the removal of riparian forests but only found significant shifts in taxonomic composition and the shredder functional group. Although differences in total functional composition were not significant, our RDA analysis showed that forest communities were distinct from all cropland stream communities (i.e., with and without riparian forest cover; Fig. 6B). This may have resulted from the relatively small sample size of our study or from the exclusion of non-EPT species in our analyses. Additional sampling effort, including more streams and sampling throughout the year, may clarify this apparent difference. In addition, FFG are not necessarily static traits and can vary spatially or seasonally in some taxa (Tierno de Figueroa et al., 2019). If species are omnivorous or more plastic in their feeding responses, for example, we might not capture subtle differences in their response to environmental conditions. On the other hand, this result may indicate that these communities have either retained some functional resilience to these large-scale changes or that macroinvertebrate communities in cropland streams are still adjusting (in terms of their functional response) to land-use change effects on food resources.

We found differences in which functional groups were associated with watershed types and environmental variables. The degradation or absence of riparian forests often affects the availability of food resources for macroinvertebrates (Naiman et al, 2005) and can alter aquatic trophic structure (e.g., increasing the abundance of periphyton consumers; Kiffney et al., 2003). Our results show a predominance of collector-filterers in cropland streams with no riparian forest buffer; scrapers in cropland sites with riparian forest buffers; and shredders in forested sites. Shredders play an important role in litter breakdown in temperate streams (Cuffney et al., 1990; Hieber and Gessner, 2002), but in tropical streams their role and importance vary from region to region (e.g., Chara-Serna et al., 2012; Ferreira et al., 2015). Shredders are generally abundant in streams with high riparian vegetation density (Oliveira et al., 1999; Cummins et al., 2005). Because shredders feed on leaves, they can reduce their size and make them available for other taxa (e.g., collector-filterers) in the form of fine particulate organic matter (FPOM; Boyero et al., 2011) At our study site, leaves represent about 90% of litterfall biomass in sites with riparian forests. Although annual litterfall inputs are similar between forest and cropland streams, the biomass of benthic organic material is lower in cropland streams (K.J. Jankowski, U.S. Geological Survey, written comm., 2021). The interaction between shredders and organic material in forest streams highlights the important role that this group likely plays in the dynamics of organic material in tropical streams, and merits further study.

Scrapers typically feed on periphyton and are relatively tolerant to pollution and disturbance (Cummins and Klug, 1979). Periphyton growth can increase when light and temperature are higher, as often occurs in the absence of riparian vegetation (Bleich et al., 2015). Riparian forests in cropland watersheds at Tanguro Ranch have lower leaf area index (LAI) than forested sites (Nagy et al., 2015, Maracahipes-Santos et al., 2020), indicating that more light may reach streams and increase periphyton growth. Because higher primary productivity favors species with generalist feeding habitats, it is likely that the trophic relationships and energy flows in this altered ecosystem may be changing. Deforestation in the Upper Xingu River Basin has already changed fish assemblages, with fish abundance nearly 2.1 times higher in deforested streams (Ilha et al., 2019) than reference streams in primary forests, primarily due to the increase in fish with small adult body size. Thus, the higher abundance of scrapers in cropland areas likely reflects ecosystem-level shifts related to changes in riparian forests such as light and temperature dynamics, and perhaps the aquatic trophic structure.

Other functional traits can respond to changes in catchment land use, local physical habitat, and water quality. For example, macroinvertebrates in fully (upland and riparian) forested streams may have larger body sizes and < 1 reproductive cycle per year (de Castro et al., 2018). In this study we did not consider other functional traits such as those related to behavior, morphology, and physiology which may have displayed differing trends in response to riparian forest cover.

In addition, the functional feeding groups considered here are relatively coarse, and the life history and feeding strategies of diverse aquatic insect communities are not well understood in the Amazon-Cerrado region, compared with temperate regions. It is possible that we missed subtler functional shifts by using these general categories. Future studies could investigate a broader array of traits that may respond to different aspects of habitat change, including shifts in the thermal environment (Macedo et al., 2013) that has been found to influence fish body size in these streams (Ilha et al., 2018).

Environmental drivers

Small streams in cropland watersheds have higher and more variable water temperature (Macedo et al., 2013) and discharge rates (Hayhoe et al., 2011) compared with forested watersheds. On average, cropland streams are ~ 1.5°C warmer and have higher diel variability, with maximum daily temperatures ~ 3°C higher than in forested streams (Macedo et al., 2013). Likewise, discharge is 2 - 4 times higher than forested streams (Riskin et al., 2017). Both stream flow and temperature were linked to changes in taxonomic and functional composition observed in this study (Fig. 5). Water temperature has a significant effect on macroinvertebrate growth, fecundity, and survival (Sweeney, 1993) and some taxa may be especially sensitive to high water temperature (Lorion and Kennedy, 2009). While thermal traits of macroinvertebrates in this region are not well known, tropical organisms are generally adapted to low thermal variability (Tewksbury et al., 2008, Dillon et al., 2010), which could partially explain why richness was lower in the more environmentally variable cropland streams with no riparian forest. In addition, organic matter breakdown is typically more temperature sensitive than primary production (Allen et al., 2005), which could alter the relative availability of allochthonous and autochthonous organic matter (Richardson and Béraud, 2014), favoring taxa that preferentially feed on periphyton (i.e., scrapers). Water velocity can also act as a selective pressure for the distribution of stream organisms (Bispo et al., 2006). For example, high flows can favor the colonization of periphyton on rocks (Cattaneo et al., 1997) and other hard substrates such as wood, which attracts scrapers organisms and their predators (Feltmate et al., 1986). In contrast, it may limit successful colonization by either periphyton or macroinvertebrates if substrates are unstable (e.g., sand).

Taxonomic composition reflected the dominant environmental conditions across sites. Composition was driven by organic material in Forest streams and by periphyton and temperature in cropland streams, both with and without riparian forest buffers. The decrease in organic matter inputs with riparian forest removal reduces the diversity of available food sources. It can also increase sedimentation, and alter the availability of suitable habitat by changing light, temperature, or nutrient regimes (Vannote et al., 1980;

Davies et al., 2005). A separate study conducted in the same streams demonstrated a reduction in the biomass of terrestrial organic matter in cropland relative to forested streams (KK.J. Jankowski, U.S. Geological Survey, written comm., 2021), which supports our finding that high organic material was associated with high insect abundance in forested sites.

Conclusions

Riparian forest removal resulted in changes in EPT taxonomic community, but had a less pronounced effect on functional composition among streams. Taxonomic and functional biodiversity are complementary approaches that can reveal different responses to disturbance from species to ecosystem processes. Taxonomic richness in cropland streams with riparian forest buffers was intermediate between forest and cropland stream without riparian buffers, emphasizing the importance of the riparian forest buffers in mitigating aquatic species losses and, consequently, preserving the potentially diverse ecosystem functions provided by these species. Taxonomic community composition shifted in streams surrounded by crops, but broad groups in functional feeding composition did not change, indicating some functional redundancy in feeding habits in these communities with only subtle shifts in the dominance of certain functional feeding groups. Future work could investigate whether other functional traits (e.g., body size and body form) follow the same pattern in response to watershed deforestation and riparian forest preservation.

This study is among the few to evaluate stream taxonomic and functional macroinvertebrate community response to watershed deforestation and the effects of riparian conservation in the Amazon-Cerrado transition region of rapid land-use change. This region harbors high freshwater biodiversity (Abell et al., 2008), and the protection of riparian forests has been the subject of high-profile scientific and political debates (Azevedo et al., 2017). The conservation of riparian forests in areas with intense land-use change such as the southeastern Amazon helps to ensure the maintenance of aquatic communities, functional diversity, and important ecosystem processes. Using taxonomic and functional diversity in tandem may provide a better understanding of the relationship between land-use change and its effects on stream biodiversity in this rapidly changing tropical region than either metric alone.

Acknowledgments We thank IPAM and Grupo AMaggi for logistical support and use of the Tanguro field site. The expedition field team composed of Dr. Paulo Ilha, Dr. Murilo Sversut Dias, Dr. Tiago Barros Miguel, MSc. Pâmela Virgolino, MSc. Lorrane Gabrielle Cantanhede, Nathane Queiroz, Darlisson Nunes, and Raimundo Mota provided excellent field support and comradery. We thank Dra. Ludmila Rattis and MSc. Paul Lefebvre for producing the map used in Fig. 1. We also thank the two anonymous reviewers. Funding for this research came from the Conselho Nacional de Desenvolvimento Científico e Tecnológico CNPq (PELD/TANG #441703/2016-0); CAPES (Post-doctoral Fellowship 88887.136296/2017-00 to NM); and the National Science Foundation (DEB 1457602, INFEWS 1739724 to LAD and MNM). We thank CNPq for the productivity scholarship granted to LJ (304710/2019-9) and A.L.A. for the Doctoral Scholarship (141991/2016-0). KJJ was supported by the US Army Corps of Engineers' Upper Mississippi River Restoration Program. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the US Government.

Authors' contributions NCSM conceived the study, did the field collection, analyzed data, and wrote the paper. KJJ conceived the study, did the field collection, and wrote the paper. MNM conceived the study and wrote the paper. LJ conceived the study and wrote the paper; AL-A conceived the study, did the field collection, and wrote the paper. LD conceived the study and wrote the paper.

Funding Conselho Nacional de Desenvolvimento Científico e Tecnológico CNPq (PELD/TANG #441703/2016-0); CAPES (Post-doctoral Fellowship 88887.136296/2017-00 to NM); National Science Foundation (DEB 1457602, INFEWS 1739724 to LAD and MNM); Productivity scholarship granted to LJ (304710/2019-9); and Doctoral Scholarship to A.L.A. (141991/2016-0)

Data availability All data generated or analyzed during this study are included in this published article [and its supplementary information files].

Declarations

Conflict of interest Not applicable.

References

Abell, R., M. L. Thieme, C. Revenga, M. Bryer, M. Kottelat, N. Bogutskaya, B. Coad, N. Mandrak, S. Balderas, W. Bussing, M. Stiassny, P. Skelton, G. Allen, P. Unmack, A. Naseka, R. Ng, N. Sindorf, J. Robertson, E. Armijo, J.

Higgins, T. Heibel, E. Wikramanayake, D. Olson, H. López, R. Reis, J. Lundberg, M. Pérez & P. Pétry, 2008. Freshwater ecoregions of the world: a new map of biogeographic units for freshwater biodiversity conservation. BioScience 585: 403–414.

Allen, A. P., J. F. Gillooly & J. H. Brown, 2005. Linking the global carbon cycle to individual metabolism. Functional Ecology 192: 202–213.

Anderson, M. J., 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecology 26: 32–46.

Azevedo, A., A. R. Rajão, M. A. Costa, M. C. Stabile, M. N. Macedo, T. N. Dos Reis, A. Alencar, B. Soares-Filho & R. Pacheco, 2017. Limits of Brazil's Forest Code as a means to end illegal deforestation. Proceedings of the National Academy of Sciences 114: 7653–7658.

Balch, J. K., D. C. Nepstad, P. M. Brando, L. M. Curran, O. Carvalho Jr. & P. Lefebvre, 2008. Negative fire feedback in a transitional forest of southeastern Amazonia. Global Change Biology 1410: 2276–2287.

Barbosa, F. A. R., M. Callisto & N. Galdean, 2001. The diversity of benthic macroinvertebrates as an indicator of water quality and ecosystem health: a case study for Brazil. Aquatic Ecosystem Health & Management 4: 51–59.

Bertaso, T. R., M. R. Spies, C. B. Kotzian & M. L. Flores, 2015. Effects of forest conversion on the assemblages' structure of aquatic insects in subtropical regions. Revista Brasileira de Entomologia 59: 43–49.

Bispo, P. D. C., L. G. Oliveira, L. M. Bini & K. G. D. Sousa, 2006. Ephemeroptera, Plecoptera and Trichoptera assemblages from riffles in mountain streams of Central Brazil: environmental factors influencing the distribution and abundance of immatures. Brazilian Journal of Biology 662B: 611–622.

Bleich, M. E., M. T. F. Piedade, A. F. Mortati & T. André, 2015. Autochthonous primary production in southern Amazon headwater streams: novel indicators of altered environmental integrity. Ecological Indicators 53: 154–161.

Boyero L., R.G. Pearson., D. Dudgeon, M.A. Graça, M.O. Gessner, R.J. Albarino, V. Ferreira, C.M. Yule, A.J. Boulton, M. Arunachalam, M. Callisto, E. Chauvet, A. Ramırez, J. Chará, M.S. Moretti, J.F.Jr. Gonçalves, J.E. Helson, A.M. Chara-Serna, A.C. Encalada, J.N. Davies, S. Lamothe, A. Li, A.O.Y. Cornejo, L.M. Buria, V.D. Villanueva, M.C. Zuniga & C.M. Pringle, 2011 Global distribution of a keytrophic guild contrasts with common latitudinal diversity patterns. Ecology 92: 1839–1848

Cardoso, M. N., L. B. Calvão, L. F. de Assis Montag, B. S. Godoy & L. Juen, 2018. Reducing the deleterious effects of logging on Ephemeroptera communities through reduced impact management. Hydrobiologia 8231: 191–203.

Carvalho, F. G., F. Oliveira Roque, L. Barbosa, L. F. de Assis Montag & L. Juen, 2018. Oil palm plantation is not a suitable environment for most forest specialist species of Odonata in Amazonia. Animal Conservation 21: 526–533.

Cattaneo, A., T. Kerimian, M. Roberge & J. Marty, 1997. Periphyton distribution and abundance on substrata of different size along a gradient of stream trophy de Montréal. Hydrobiologia 3541: 101–110.

Cayuela, L. & N.J. Gotelli, 2014. rareNMtests: ecological and biogeographical null model tests for comparing rarefaction curves. R package version 1.0.

Chao, A. & L. Jost, 2012. Coverage-based rarefaction and extrapolation: standardizing samples by completeness rather than size. Ecology 93: 2533–2547.

Chara-Serna, A. M., J. D. Chara, M. D. Zuniga, R. G. Pearson & L. Boyero, 2012. Diets of leaf litter associated invertebrates in three tropical streams. Annales de Limnologie-International Journal of Limnology 48: 139–144.

Chen, H. & P. C. Boutros, 2011. VennDiagram: a package for the generation of highly-customizable Venn and Euler diagrams in R. BMC bioinformatics 12: 1–7.

Chevene, F., S. Doléadec & D. Chessel, 1994. A fuzzy coding approach for the analysis of long-term ecological data. Freshwater Biology 31: 295–309.

Cuffney, T. F., J. B. Wallace & G. J. Lugthart, 1990. Experimental evidence quantifying the role of benthic invertebrates in organic matter dynamics of headwater streams. Freshwater Biology 23: 281–299.

Cummins, K. W. & M. J. Klug, 1979. Feeding ecology of stream invertebrates. Annual Review of Ecology and Systematics 10: 147–172.

Cummins, K. W., R. W. Merritt & P. C. N. Andrade, 2005. The use of invertebrate functional groups to characterize ecosystem attributes in selected streams and rivers in South Brazil. Studies on Neotropical Fauna and Environment 40: 69–89.

Dala-Corte, R. B., A. S. Melo, T. Siqueira, L. M. Bini, R. T. Martins, A. M. Cunico, A. Pes, A. Magalhães, B. Godoy, C. Leal, C. Monteiro-Junior, C. Sternert, D. Castro, D. Macedo, D. Lima-Junior, E. Gubiani, F. Massariol, F. Teresa, F. Becker, F. Souza, F. Valente-Neto, F. Souza, F. Salles, G. Brejão, J. Brito, J. Vitule, J. Simião-Ferreira, K. Dias-Silva, L. Albuquerque, L. Juen, L. Maltchik, L. Casatti, L. Montag, M. Rodrigues, M. Callisto, M. Nogueira, M. Santos, N. Hamada, P. Pamplin, P. Pompeu, R. Leitão, R. Ruaro, R. Mariano, S. Couceiro, V. Abilhoa, V. Oliveira, Y. Shimano, Y. Moretto, Y. Súarez & F. Roque, 2020. Thresholds of freshwater biodiversity in response to riparian vegetation loss in the Neotropical region. Journal of Applied Ecology 57: 1391–1402.

Davies, P. E., L. S. J. Cook, P. D. McIntosh & S. A. Munks, 2005. Changes in stream biota along a gradient of logging disturbance, 15 years after logging at Ben Nevis, Tasmania. Forest Ecology and Management 219: 132–148.

de Castro, D. M. P., S. Dolédec & M. Callisto, 2018. Land cover disturbance homogenizes aquatic insect functional structure in neotropical savanna streams. Ecological Indicators 84: 573–582.

Dillon, M. E., G. Wang & R. B. Huey, 2010. Global metabolic impacts of recent climate warming. Nature 467: 704–706.

Domínguez E, C. Molineri, M. Pescador, M.D. Hubbard & C. Nieto, 2006. Ephemeroptera of South America. Aquatic Biodiversity in Latin America ABLAVol 2. Sofia: Pensoft.

dos Reis Oliveira, P. C., M. H. Kraak, M. Pena-Ortiz, H. G. van der Geest & P. F. Verdonschot, 2020. Responses of macroinvertebrate communities to land use specific sediment food and habitat characteristics in lowland streams. Science of the Total Environment 703:

Dray, S. & A. B. Dufour, 2007. The ade4 package: implementing the duality diagram for ecologists. Journal of Statistical Software 22: 1–20.

Erdozain, M., K. Kidd, D. Kreutzweiser & P. Sibley, 2019. Increased reliance of stream macroinvertebrates on terrestrial food sources linked to forest management intensity. Ecological Applications 29(4):

Faria, A. P. J., R. Ligeiro, M. Callisto & L. Juen, 2017. Response of aquatic insect assemblages to the activities of traditional populations in eastern Amazonia. Hydrobiologia 8021: 39–51.

Feltmate, B. W., R. L. Baker & P. J. Pointing, 1986. Distribution of the stonefly nymph Paragnetina media Plecoptera: Perlidae: influence of prey, predators, current speed, and substrate composition. Canadian Journal of Fisheries and Aquatic Sciences 438: 1582–1587.

Fernandes, J. D. F., A. L. de Souza & M. O. Tanaka, 2014. Can the structure of a riparian forest remnant influence stream water quality? A tropical case study. Hydrobiologia 724: 175–185.

Ferreira, W. R., R. Ligeiro, D. R. Macedo, R. M. Hughes, P. R. Kaufmann, L. G. Oliveira & M. Callisto, 2015. Is the diet of a typical shredder related to the physical habitat of headwater streams in the Brazilian Cerrado? Annales de Limnologie-International Journal of Limnology 51: 115–127.

Ferreira, W. R., L. U. Hepp, R. Ligeiro, D. R. Macedo, R. M. Hughes, P. R. Kaufmann & M. Callisto, 2017. Partitioning taxonomic diversity of aquatic insect assemblages and functional feeding groups in neotropical savanna headwater streams. Ecological indicators 72: 365–373.

Firmiano, K. R., D. M. Castro, M. S. Linares & M. Callisto, 2021. Functional responses of aquatic invertebrates to anthropogenic stressors in riparian zones of Neotropical savanna streams. Science of the Total Environment 753:

Gibbs, H. K., A. S. Ruesch, F. Achard, M. K. Clayton, P. Holmgren, N. Ramankutty & J. A. Foley, 2010. Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. Proceedings of the National Academy of Sciences 107: 16732–16737.

Godoy, B. S., A. P. J. Faria, L. Juen, L. Sara & L. G. Oliveira, 2019. Taxonomic sufficiency and effects of environmental and spatial drivers on aquatic insect community. Ecological Indicators 107:

Gotelli, N. J. & R. K. Colwell, 2011. Estimating species richness. Biological Diversity: Frontiers in Measurement and Assessment 12: 39–54.

Hamada, N., J. L. Nessimian & R. B. Querino, 2014. Insetos aquáticos na Amazônia Brasileira: taxonomia, biologia e ecologia. Manaus, INPA: 724p.

Hayhoe, S. J., C. Neill, S. Porder, R. McHorney, P. Lefebvre, M. T. Coe & A. V. Krusche, 2011. Conversion to soy on the amazonian agricultural frontier increases streamflow without affecting stormflow dynamics. Global Change Biology 17: 1821–1833.

He, X. Sun, X. Dong, Q. Cai & S.C. Jahnig, 2019. Benthic macroinvertebrates as indicators for river health in the Changjiang Basin. In Chinese Water Systems (pp. 207-217).

Hieber, M. & M. O. Gessner, 2002. Contribution of stream detrivores, fungi, and bacteria to leaf breakdown based on biomass estimates. Ecology 83: 1026–1038.

Ilha, P., L. Schiesari, F. I. Yanagawa, K. Jankowski & C. A. Navas, 2018. Deforestation and stream warming affect body size of Amazonian fishes. PloS one 135:

Ilha, P., S. Rosso & L. Schiesari, 2019. Effects of deforestation on headwater stream fish assemblages in the Upper Xingu River Basin. Southeastern Amazonia, Neotropical Ichthyology: 171.

Jankowski, K., C. Neill, E. A. Davidson, M. N. Macedo, C. Costa, G. L. Galford, L. Maracahipes-Santos, P. Lefevbre, D. Nunes, C. Cerri, R. McHorney, C. O'Connell & M. T. Coe, 2018. Deep soils modify environmental consequences of increased nitrogen fertilizer use in intensifying Amazon agriculture. Scientific reports 8: 1–11.

Juen, L., E. J. Cunha, F. G. Carvalho, M. C. Ferreira, T. O. Begot, A. L. Andrade, Y. Shimano, H. Leão, P. Pompeu & L. F. A. Montag, 2016. Effects of oil palm plantations on the habitat structure and biota of streams in eastern Amazon. River Research and Applications 3210: 2081–2094.

Kiffney, P. M., J. S. Richardson & J. P. Bull, 2003. Responses of periphyton and insects to experimental manipulation of riparian buffer width along forest streams. Journal of Applied Ecology 40: 1060–1076.

Kominoski, J. S., J. J. F. Shah, C. Canhoto, D. G. Fischer, D. P. Giling, E. González & Y. R. McElarney, 2013. Forecasting functional implications of global changes in riparian plant communities. Frontiers in Ecology and the Environment 11: 423–432.

Lake, P. S., 2000. Disturbance, patchiness, and diversity in streams. Journal of the North American Benthological Society 19: 573–592.

Leal, C. G., P. S. Pompeu, T. A. Gardner, R. P. Leitão, R. M. Hughes, P. R. Kaufmann, J. Zuanon, F. de Paula, S. Ferraz, J. Thomson, R. Mac Nally, J. Ferreira & J. Barlow, 2016. Multi-scale assessment of human-induced changes to Amazonian instream habitats. Landscape Ecology 318: 1725–1745.

Lecci L.S., T.V.D. Simões & A.R. Calor, 2014. Plecoptera do Semiárido: conhecimento atual e desafios. Artrópodes do Semiárido: biodiversidade e conservação. Printmídia, Feira de Santana, 91-98.

Legendre, P. & L. F. Legendre, 2012. Numerical ecology. Elsevier, Amsterdam.

Lepš, J. & P. Šmilauer, 2003. Multivariate analysis of ecological data using CANOCO. Cambridge University Press, Cambridge.

Li, A. O. & D. Dudgeon, 2008. Food resources of shredders and other benthic macroinvertebrates in relation to shading conditions in tropical Hong Kong streams. Freshwater Biology 53: 2011–2025.

Ligeiro, R., R. M. Hughes, P. R. Kaufmann, D. R. Macedo, K. R. Firmiano, W. R. Ferreira, D. Oliveira, A. Melo & M. Callisto, 2013. Defining quantitative stream disturbance gradients and the additive role of habitat variation to explain macroinvertebrate taxa richness. Ecological indicators 25: 45–57.

Lorion, C. M. & B. P. Kennedy, 2009. Relationships between deforestation, riparian forest buffers and benthic macroinvertebrates in neotropical headwater streams. Freshwater Biology 54: 165–180.

Luiza-Andrade, A., L. S. Brasil, N. L. Benone, Y. Shimano, A. P. J. Farias, L. F. Montag & L. Juen, 2017. Influence of oil palm monoculture on the taxonomic and functional composition of aquatic insect communities in eastern Brazilian Amazonia. Ecological Indicators 82: 478–483.

Luiza-Andrade, A., L. S. Brasil, N. R. Torres, J. Brito, R. R. Silva, L. Maioli, M. F. Barbirato & L. Juen, 2020. Effects of local environmental and landscape variables on the taxonomic and trophic composition of aquatic insects in a rare forest formation of the Brazilian Amazon. Neotropical Entomology 49: 821–831.

Luke, S. H., E. M. Slade, C. L. Gray, K. V. Annammala, J. Drewer, J. Williamson, A. Agma, M. Ationg, S. Mitchell, C. Vairappan & M. J. Struebig, 2019. Riparian buffers in tropical agriculture: scientific support, effectiveness and directions for policy. Journal of Applied Ecology 561: 85–92.

Macedo, M. N., R. S. DeFries, D. C. Morton, C. M. Stickler, G. L. Galford & Y. E. Shimabukuro, 2012. Decoupling of deforestation and soy production in the southern Amazon during the late 2000s. Proceedings of the National Academy of Sciences 109: 1341–1346.

Macedo, M. N., M. T. Coe, R. DeFries, M. Uriarte, P. M. Brando, C. Neill & W. S. Walker, 2013. Land-use-driven stream warming in southeastern Amazonia. Philosophical Transactions of the Royal Society B: Biological Sciences 368: 20120153.

Maracahipes-Santos, L., D.V. Silvério, M.N. Macedo, L. Maracahipes, K.J. Jankowski, L.N. Paolucci, C. Neill & P.M. Brando, 2020. Agricultural land-use change alters the structure and diversity of Amazon riparian forests. Biological Conservation 252:

Mouillot, D., D. R. Bellwood, C. Baraloto, J. Chave, R. Galzin, M. Harmelin-Vivien, M. Kulbicki, S. Lavergne, S. Lavorel, N. Mouquet, C. T. Paine, J. Renauld & W. Thuiller, 2013. Rare species support vulnerable functions in high-diversity ecosystems. PLoS biol 11(5):

Nagy, R. C., S. Porder, C. Neill, P. Brando, R. M. Quintino & S. A. D. Nascimento, 2015. Structure and composition of altered riparian forests in an agricultural Amazonian landscape. Ecological Applications 25: 1725–1738.

Naiman, R. J. & H. Decamps, 1997. The ecology of interfaces: riparian zones. Annual review of Ecology and Systematics 28: 621–658.

Naiman, R. J., H. Decamps & M. Pollock, 1993. The role of riparian corridors in maintaining regional biodiversity. Ecological applications 3: 209–212.

Naiman, R. J., H. Decamps & M. E. Mcclain, 2005. Riparia—Ecology. Elsevier Academic Press, London, Conservation and Management of Streamside Communities.

Neill, C., M. T. Coe, S. H. Riskin, A. V. Krusche, H. Elsenbeer, M. N. Macedo, R. McHorney, P. Lefevbre, E. Davidson, R. Scheffler, A. Michela, S. Porder & L. Deegan, 2013. Watershed responses to Amazon soya bean cropland expansion and intensification. Philosophical Transactions of the Royal Society B: Biological Sciences 368: 201–204.

Nessimian J.L., E.M. Venticinque, J. Zuanon, P.J.R. De Marco, M. Gordo, L. Fidelis, J.D. Batista & L. Juen, 2008. Land use, habitat integrity, and aquatic insect assemblages in Central Amazonian streams.Hydrobiologia 614:117–131.

Nunes, S. S., J. Barlow, T. A. Gardner, J. V. Siqueira, M. R. Sales & C. M. Souza, 2015. A 22 year assessment of deforestation and restoration in riparian forests in the eastern Brazilian Amazon. Environmental conservation 42: 193–203.

Nunes, S., J. Barlow, T. Gardner, M. Sales, D. Monteiro & Jr C. Souza Jr, 2019. Uncertainties in assessing the extent and legal compliance status of riparian forests in the eastern Brazilian Amazon. Land Use Policy 82: 37–47.

Oksanen J., F.G. Blanchet, R. Kindt, P. Legendre, P.R. Minchin, R.B. O'hara & M.J Oksanen, 2013. Package 'vegan'. Community ecology package, version, 29.

Oliveira, J. M. B. & L. Juen, 2019. Structuring of dragonfly communities (Insecta: Odonata) in Eastern Amazon: Effects of environmental and spatial factors in preserved and altered streams. Insects 10: 1–18.

Oliveira, L., P. Bispo, V. Crisci & K. Sousa, 1999. Distribuições de categorias funcionais alimentares de larvas de Trichoptera Insecta em uma região serrana do Brasil Central. Acta Limnologica Brasil 11: 173–183.

Osborne, L. L. & D. A. Kovacic, 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. Freshwater Biology 29: 243–258.

Paiva, C. K. S., A. P. J. de Faria, L. B. Calvão & L. Juen, 2017. Effect of oil palm on the Plecoptera and Trichoptera Insecta assemblages in streams of eastern Amazon. Environmental monitoring and assessment 189: 1–9.

Paiva, C. K. S., A. P. J. Faria, L. B. Calvao & L. Juen, 2021. The anthropic gradient determines the taxonomic diversity of aquatic insects in Amazonian streams. Hydrobiologia 848: 1073–1085.

Peck D.V., J.M. Lazorchak & D.J. Klemm, Eds, 2006. Environmental Monitoring and Assessment Program–surface Waters: Western Pilot Study Field Operations Manual for Wadeable Streams. National Health and Environmental Effects Research Laboratory [and] National Exposure Research Laboratory, Office of Research and Development, US Environmental Protection Agency.

Pert, P. L., J. R. A. Butler, J. E. Brodie, C. Bruce, M. Honzak, F. J. Kroon, D. Metcalfe, D. Mitchell & G. Wong, 2010. A catchment-based approach to mapping hydrological ecosystem services using riparian habitat: a case study from the Wet Tropics, Australia. Ecological Complexity 73: 378–388.

Péru, N. & S. Dolédec, 2010. From compositional to functional biodiversity metrics in bioassessment: a case study using stream macroinvertebrate communities. Ecological Indicators 10: 1025–1036.

Petchey, O. L. & K. J. Gaston, 2002. Functional diversity FD, species richness and community composition. Ecology letters 53: 402–411.

Poff, N. L., 1997. Landscape filters and species traits: towards mechanistic understanding and prediction in stream ecology. Journal of the North American Benthological Society 162: 391–409.

Poff, N. L., J. D. Olden, N. K. Vieira, D. S. Finn, M. P. Simmons & B. C. Kondratieff, 2006. Functional trait niches of North American lotic insects: traits-based ecological applications in light of phylogenetic relationships. Journal of the North American Benthological Society 254: 730–755.

Rao, C.R., 1964. The use and interpretation of principal component analysis in applied research. Sankhyā: The Indian Journal of Statistics Series A: 329-358.

Richardson, J. S. & S. Beraud, 2014. Effects of riparian forest harvest on streams: a meta-analysis. Journal of Applied Ecology 51: 1712–1721.

Riskin, S. H., C. Neill, K. Jankowski, A. V. Krusche, R. McHorney, H. Elsenbeer, M. Macedo, D. Nunes & S. Porder, 2017. Solute and sediment export from Amazon forest and soybean headwater streams. Ecological applications 271: 193–207.

Rizo-Patrón, F., A. Kumar, M. B. M. Colton, M. Springer & F. A. Trama, 2013. Macroinvertebrate communities as bioindicators of water quality in conventional and organic irrigated rice fields in Guanacaste, Costa Rica. Ecological Indicators 29: 68–78.

Shimano, Y. & L. Juen, 2016. How oil palm cultivation is affecting mayfly assemblages in Amazon streams. Annales de Limnologie-International Journal of Limnology 52: 35–45.

Shimano, Y., M. Cardoso & L. Juen, 2018. Ecological studies of mayflies Insecta, Ephemeroptera: can sampling effort be reduced without losing essential taxonomic and ecological information? Acta Amazonica 48: 137–145.

Shortle J.S., D.G. Able & M. Ribaudo, 2001. Agriculture and water quality: the issues. In Environmental Policies for Agricultural Pollution Control. CABI Publishing, London: 1–18.

Siegloch, A. E., M. Suriano, M. Spies & A. Fonseca-Gessner, 2014. Effect of land use on mayfly assemblages structure in Neotropical headwater streams. Anais da Academia Brasileira de Ciencias 86: 1735–1747.

Siegloch, A. E., R. Schmitt, M. Spies, M. Petrucio & M. I. M. Hernández, 2016. Effects of small changes in riparian forest complexity on aquatic insect bioindicators in Brazilian subtropical streams. Marine and Freshwater Research 68: 519–527.

Soares-Filho, B., R. Rajão, M. Macedo, A. Carneiro, W. Costa, M. Coe, H. Rodrigues & A. Alencar, 2014. Cracking Brazil's forest code. Science 344: 363–364.

Souza, A. L. T., D. G. Fonseca, R. Liborio & M. O. Tanaka, 2013. Influence of riparian vegetation and forest structure on the water quality of rural low-order streams in SE Brazil. Forest Ecology and Management 298: 12–18.

Sponseller, R. A., E. F. Benfield & H. M. Valett, 2001. Relationships between land use, spatial scale and stream macroinvertebrate communities. Freshwater Biology 4610: 1409–1424.

Sweeney, B. W., 1993. Effects of stream side vegetation on macroinvertebrate communities of White Clay Creekin eastern North America. Proceedings of the Academy of Natural Sciences of Philadelphia 144: 291–340.

Sweeney, B. W., T. L. Bott, J. K. Jackson, L. A. Kaplan, J. D. Newbold, L. J. Standley, C. Hession & R. J. Horwitz, 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. Proceedings of the National Academy of Sciences 10139: 14132–14137.

Tanaka, M. O., A. L. T. de Souza, L. E. Moschini & A. K. de Oliveira, 2016. Influence of watershed land use and riparian characteristics on biological indicators of stream water quality in southeastern Brazil. Agriculture, Ecosystems & Environment 216: 333–339.

Tank, J. L., E. J. Rosi-Marshall, N. A. Griffiths, S. A. Entrekin & M. L. Stephen, 2010. A review of allochthonous organic matter dynamics and metabolism in streams. Journal of the North American Benthological Society 291: 118–146.

Ter Braak, C. J. & P. F. Verdonschot, 1995. Canonical correspondence analysis and related multivariate methods in aquatic ecology. Aquatic Sciences 57: 255–289.

Tewksbury, J. J., R. B. Huey & C. A. Deutsch, 2008. Putting the heat on tropical animals. Science 320: 1296–1297.

Tierno de Figueroa, J. M., M. J. López-Rodríguez & M. Villar-Argaiz, 2019. Spatial and seasonal variability in the trophic role of aquatic insects: an assessment of functional feeding group applicability. Freshwater Biology 64: 954–966.

Tolkkinen, M. J., J. Heino, S. H. Ahonen, K. Lehosmaa & H. Mykrä, 2020. Streams and riparian forests depend on each other: a review with a special focus on microbes. Forest Ecology and Management 462:

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell & C. C. Cushing, 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37: 130–137.