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Avaliação da condição ambiental e diversidade de insetos aquáticos de riachos amazônicos submetidos à mútiplos impactos antrópicos

BELÉM, PARÁ
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Tese apresentada ao Programa de Pós-Graduação em Ecologia do convênio da Universidade Federal do Pará e Embrapa Amazônia Oriental, como requisito parcial para obtenção do título de Doutora em Ecologia.
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Orientador: Dr. Leandro Juen
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e científica.

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"The most general goal of science is to generate understanding".

Pickett et al. (1994)

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RESUMO

Os ecossistemas hídricos de água doce são estruturados hierarquicamente na paisagem e podem ser influenciados em diferentes escalas por atividades antrópicas. Na Amazônia, as atividades antrópicas que ameaçam os ecossistemas naturais e a diversidade de espécies estão relacionadas ao uso do solo para a pecuária, plantação de palma e extração de madeira. Esses usos do solo modificam a paisagem e os ecossistemas aquáticos, principalmente com a perda de hábitat físico dos riachos, o que diminui a heterogeneidade ambiental afetando a diversidade de espécies. Nesse contexto, o objetivo geral da tese foi avaliar como atividades antrópicas modificam a estrutura ambiental dos riachos e como essas modificações afetam a distribuição da biodiversidade de insetos aquáticos. A tese foi composta por três sessões. Na primeira, avaliamos a característica e heterogeneidade ambiental, a diversidade alfa e beta de Ephemeroptera, Plecoptera e Trichoptera (EPT) em categorias de uso de solo que são dominantes na paisagem amazônica (áreas de exploração de madeira, plantação de oléo de palma e pecuária). Nossos resultados mostraram que a estrutura ambiental local foi alterada pelas categorias de uso do solo e a heterogeneidade ambiental foi maior entre os riachos influenciados por pecuária e riachos controles. A diversidade alfa de EPT foi menor nos riachos influenciados pelas categorias de usos solo e a diversidade beta foi maior entre os riachos influenciados por pecuária e plantação de óleo de palma. Na segunda sessão, avaliamos as influências de preditores ambientais e espaciais na estruturação de metacomunidade de EPT e exploramos a resposta individual dos gêneros aos processos de estruturação da metacomunidade em múltiplas escalas espaciais. Observamos que o descritor espacial explicou 12% da variação total da comunidade, mas 6% foram explicados pelo ambiente espacialmente estruturado. Relações de codependência entre descritores espaciais, variáveis ambientais e gêneros de EPT foram encontradas apenas para *Amanahyphes*, *Anacroneuria*, *Askola*, *Americabaetis* e *Callibaetis*. Na terceira sessão, o nosso objetivo foi determinar os limiares de Odonata, Gerromorpha e EPT em gradientes de distúrbio antrópico na escala local e bacia de drenagem. A comunidade de Odonata apresentou uma mudança mais gradual no distúrbio local, mas os distúrbios na drenagem provocaram um pico de mudança acentuada e sincrônica na comunidade. Além disso, a maioria das espécies foi associada ao distúrbio na bacia de drenagem do que ao distúrbio local e os indivíduos foram representados em sua maioria por espécies tolerantes. No limiar comunitário de Gerromorpha e EPT, as espécies sensíveis apresentaram uma mudança acentuada no distúrbio local e na drenagem, ao passo que as espécies tolerantes apresentam uma faixa de mudança nesses gradientes. Gerromorpha teve mais espécies associadas aos gradiente de distúrbio local e na drenagem, as quais foram representadas tanto por espécies sensíveis como tolerantes. Por outro lado, EPT apresentou mais espécies sensíveis aos gradientes de distúrbio local e de drenagem, mas duas espécies aumentaram de frequência no distúrbio local e apenas uma no distúrbio de drenagem. Nossos resultados suportam o cenário de que o uso do solo em diferentes escalas espaciais é determinante na estruturação da comunidade de insetos aquáticos.

Palavras-chave: Insetos aquáticos; Diversidade alfa; Diversidade beta; Usos do solo; Gradientes de distúrbios; Escala espacial; Riachos; Codependência espacial; Paisagem; Metacomunidade; Limiar

Assessment of environmental condition and aquatic insect diversity in Amazonian streams submitted to multiple anthropic impacts

ABSTRACT

Freshwater ecosystems are structured hierarchically in the landscape and are influenced at different scales by anthropic activities. In the Amazon, anthropic activities that threaten ecosystems' integrity and species diversity are mainly related to livestock, oil palm plantations and unlawful logging. These land uses modify the landscape and freshwater ecosystems through the loss of stream physical habitat and decrease on environmental heterogeneity affecting the species diversity. In this context, the objective of the thesis was to assess how anthropic activities at different spatial scales modify the environmental structure of the streams and how these modifications affect the distribution of aquatic insect. The thesis consisted of three sections. In the first, we assessed environmental characteristic, environmental heterogeneity, alpha and beta diversity of Ephemeroptera, Plecoptera and Trichoptera (EPT) in human land use categories that are dominant in the Amazonian landscape (reduced-impact logging areas, conventional logging, oil palm plantation and livestock farming). Our results showed that the local environmental structure was altered by these land uses and environmental heterogeneity was higher among the streams influenced by livestock farming and control streams. The EPT alpha diversity was lower in streams influenced by land uses and beta diversity was higher among the streams influenced by livestock farming and oil palm plantation. In the second section, we assessed the environment and spatial influences on EPT metacommunity structuring and the individual response of EPT taxa to the metacommunity structuring processes on multiple spatial scales. We found that the spatial descriptor explained 12% of total community variation, but 6% was explained by the spatially structured environment. Codependence relationships between spatial predictors, environmental variables and genera of EPT were found only for *Amanahyphes*, *Anacroneuria*, *Askola*, *Americabaetis* e *Callibaetis*. In the third section, our objective was to determine the community and individual thresholds of Odonata, Gerromorpha and EPT in gradients of anthropic disturbances at local and catchment scales. The Odonata community presented gradual variation considering local disturbances, but catchment disturbances caused a sharp and synchronous change in the community. Furthermore, most species associated with disturbance in catchment than local disturbance and the individuals were represented mostly by tolerant species. At the community threshold of Gerromorpha and EPT, sensitive species showed a change sharp at catchment and local disturbance, while tolerant species presented a gradual change in both disturbance gradients. Gerromorpha had more species associated with the catchment and local disturbance gradients, which were represented bysensitive and tolerant species to these disturbances. On the other hand, EPT presented more species sensitive to local and catchment disturbances, with only two species increasing their abundance and frequency at the local disturbance gradient and only one at the catchment disturbance gradient. Our results support the scenario that the land-use at different spatial scales is important to structure aquatic insect community.

Keywords: Aquatic insect; Alpha diversity; Beta diversity; Land-uses; Disturbances gradients; Spatial scale; Streams; Spatial codependency; Landscape; Metacommunity; Threshold

SUMÁRIO

1. INTRODUÇÃO GERAL

A influência da escala espacial na estrutura de comunidades tem sido evidenciada nas últimas décadas (e.g. Ricklefs, 1987), mostrando que tanto fatores que ocorrem em escala local como regional são importantes estruturadores das comunidades (e.g. Sponseller et al., 2001; Marzin et al., 2013; Ochoa-Quintero et al., 2015; Jyrkänkallio et al., 2017). Em geral, nos ambientes aquáticos, os fatores locais têm sido representados por características dos habitats fluviais, como tipo de vegetação ripária e tamanho de susbstrato (Tonkin, 2014), enquanto a paisagem é mais relacionada com o tipo de uso e ocupação do solo na paisagem (Johnson & Goedkoop, 2002).

Pela teoria hierárquica de organização de ecossistemas lóticos, os processos que ocorrem em maior escala podem restringir os padrões de meso e microescala (Frissel et al., 1986), enfatizando a conexão entre os ecossistemas aquáticos e suas bacias hidrográficas (Pajunen, et al., 2017). Por exemplo, Pajunen et al. (2017) encontraram que maiores valores de condutividade em riachos podem ser relacionados a impactos antrópicos na bacia hidrográfica. Assim, os efeitos antrópicos nas comunidades de insetos aquáticos podem ser oriundos dos processos locais (Sponseller et al., 2001; Shimano & Juen, 2016) ou associados à drenagem (Feld & Henring, 2007), ou ainda pela interação de ambos (Macedo et al., 2014; Tonkin et al., 2016; Firmiano et al., 2017).

A maior extensão espacial também pode implicar maior heterogeneidade ambiental (Viana & Chase, 2018), um fenômeno natural que representa a variação ambiental entre os locais de uma região. Os ecossistemas aquáticos naturais geralmente apresentam alta heterogeneidade em suas características de habitat (Stein et al., 2014; Heino et al., 2015b), o que contribui para a alta diversidade biológica devido a quantidade de espécies com diferentes afinidades ambientais (Southwood, 1977; Muotka et al., 2002; Brown, 2003; Stein et al., 2014). No entanto, atividades antrópicas modificam o uso do solo ao redor dos ecossistemas lóticos, o que pode alterar a condição e heterogeneidade ambiental dos riachos, afetando as comunidades biológicas em toda a paisagem (Giam et al., 2015; Cunha & Juen, 2017).

O conceito de metacomunidade se encaixa nesse contexto, porque reconhece a importância do contexto multi-escala para explicar as comunidades locais e a variação entre elas (Leibold et al., 2004; Thompson e Townsend, 2006; Brown et al., 2011; Leibold & Chase, 2018) e tenta combinar perspectivas de nicho e neutras na compreensão dos padrões das comunidades (Leibold et al., 2004; Leibold & Chase, 2018). O modelo de *species sorting* prediz forte relação entre as espécies e as condições ambientais, de modo que os mecanismos relacionados ao nicho das espécies atuam mais fortemente na estruturação da comunidade Leibold et al., 2004; Vellend, 2010; Heino & Mykra, 2008; Leibold & Chase, 2018).

Complementarmente, a metacomunidade pode ser estruturada por processos espaciais, que podem estar relacionados à equivalência ecológica e à limitação de dispersão (Hubbell, 2001).

Assim, tanto processos *species sorting* quanto estocásticos podem atuar na estruturação da metacomunidade. Testes desses processos foram conduzidos em uma ampla variedade de ecossistemas e com diferentes organismos (por exemplo, Thompson & Towsend, 2006; Hájek et al., 2011; Provete et al., 2014; Jamoneau et al., 2018), e os resultados encontrados ressaltam a influência de processos ambientais (Heino & Mykra 2008; Grönroos et al., 2013) ou processos ambientais e espaciais conjuntamente estruturando as metacomunidades (Livingston et al., 2013; Provete et al., 2014; Jamoneau et al., 2018; Viana & Chase, 2018).

Medidas de diversidade de espécies também estão relacionadas com a escala espacial. Por exemplo, a diversidade alfa (e.g. riqueza de espécie) corresponde à diversidade em um habitat local, enquanto a diversidade beta mede a variação na composição de espécies entre locais (Whittaker, 1972). A ocorrência das espécies em um determinado local depende de processos de filtragem ordenados hierarquicamente que operam em diferentes escalas espaciais como, por exemplo, fatores climáticos, geologia, barreiras físicas (escala global), eventos históricos, processos de dispersão (escala regional), e características físicas e químicas do habitat local e interações bióticas (escala local) (Cox et al., 2016; Poff, 1977). Além desses complexos padrões naturais, a presença de atividades humanas pode causar mudanças significativas nos processos ecológicos, principalmente os que são relacionados a fatores locais (Benone et al., 2017; Cunha & Juen, 2017). Por exemplo, a diminuição na riqueza de espécie tem sido associada às características ambientais que foram modificadas por atividades humanas (Ligeiro et al., 2013; Murphy & Romanuk, 2014; Paiva et al., 2017) como pH, temperatura, turbidez da água (Astorga et al., 2011; Cunha et al, 2015) e proximidade com estradas (Paiva et al., 2017).

A heterogeneidade ambiental entre os riachos tem sido descrita como um importante direcionador da β-diversidade (Astorga et al., 2014). No entanto, apesar das evidências individuais, não há consenso sobre os efeitos das atividades humanas na heterogeneidade ambiental e na diversidade β (Heino et al., 2009; Gutiérrez-Cánovas et al., 2013; Bini et al., 2014; Johnson & Angeler, 2014; Larsen & Ormerod, 2014; Hawkins et al., 2015; Fugère et al., 2016). Alguns estudos realizados em diferentes regiões do mundo mostraram uma relação positiva entre heterogeneidade ambiental e diversidade β de macroinvertebrados aquáticos (Astorga et al., 2014) e diatomáceas (Zorzal-Almeida et al., 2017), mas a ausência de relação também foi documentada (Heino et al., 2013, Grman et al., 2015). Além das condições ambientais, a diversidade beta também pode ser afetada por processos relacionados à dispersão e estocasticidades (Johnson & Angeler, 2014; Heino et al., 2015a; McCreadie et al., 2017; Leibold & Chase 2018).

Todos os processos supracitados utilizam de princípios lineares entre as comunidades e as condições ambientais ou espaciais. No entanto, algumas dessas relações são não-lineares. Essa relação não-linear pode ser avaliada através da detecção de limiares ecológicos das comunidades e das espécies em relação a gradientes de distúrbio antrópicos. Os limiares ecológicos podem ser definidos como a mudança abrupta de uma variável resposta (e.g. riqueza ou composição de espécies) devido à mudança contínua em uma variável preditora (e.g. gradiente ambiental) (Muradian, 2001; Luck, 2005; Groffman et al., 2006). O gradiente ambiental pode variar de pouca ou nenhuma influência de atividades humanas até ambientes severamente alterados (Davies & Jackson, 2006) e podem ser representado quantitativamente por diferentes evidências de perturbação, como distúrbio antrópico na paisagem (Cardoso et al., 2013), variáveis físicas estruturais e químicas da água (Shimano e Juen, 2016; Firmiano et al., 2017; Sultana et al., 2019), fragmentação da floresta (Ochoa-Quinteiro et al., 2015; Roque et al., 2018) e perda na vegetação ciliar dos riachos (Rodrigues et al., 2016).

As espécies de uma comunidade podem apresentar respostas distintas a esses gradientes de distúrbios (Suriano & Fonseca-Gessner, 2013; Fu et al., 2016) dependendo da sua amplitude de exigências ecofisiológicas ou dos limites de tolerância a degradação ambiental (Southwood, 1977; Poff & Ward, 1990; Baker & King, 2010; Estavillo et al., 2013; Sundermann et al., 2015). A comunidade pode aumentar ou diminuir drasticamente sua frequência e abundância em algum ponto ao longo de um gradiente de distúrbio (Clements et al., 2010; Gido et al., 2010), podendo causar mudanças significativas na composição quando os limites de tolerâncias são atingidos (Roque et al., 2018). Uma vez atingido o limiar das espécies, a recuperação das comunidades passa ser um processo lento e difícil (Lyytimäki & Hilden, 2007).

Na Amazônia, as atividades antrópicas que ameaçam os ecossistemas naturais e a diversidade de espécies estão relacionadas ao uso do solo para a grilagem de terra e extração de madeira, plantação de palma e pecuária (Fearnside, 2005; IPAM, 2006; Peres et al., 2010; Cunha et al., 2015). Nas últimas décadas, essas atividades desencadearam uma intensa degradação na região amazônica conhecida como "Arco do desflorestamento", que abrange uma faixa contínua do estado do Acre até o Maranhão (Aldrich et al., 2014). Essa extensão de área alterada modificou a paisagem e os ecossistemas aquáticos, principalmente com a perda de hábitat físico e diminuição na heterogeneidade ambiental afetando a diversidade de organismos (Gardner et al.2013; Oliveira-Júnior et al., 2015; Leal et al., 2016; Calvão et al., 2016; Chen et al., 2017; Cunha & Juen, 2017). Portanto, estudos que agregam informações sobre os efeitos dessas atividades antrópicas no ambiente são importantes, uma vez que pode direcionar medidas durante e após a degradação.

Nesse contexto, o objetivo geral desta tese foi avaliar como atividades antrópicas em diferentes escalas espaciais modificam a estrutura ambiental dos riachos e como essas modificações afetam a distribuição da biodiversidade de insetos aquáticos. Para isso a tese foi dividida em três sessões apresentadas a seguir.

1º sessão

Na primeira sessão, nós avaliamos a característica ambiental, heterogeneidade ambiental, a diversidade alfa (α) e beta (β) de Ephemeroptera, Plecoptera e Trichoptera (EPT) em várias categorias de uso do solo que são dominantes na paisagem amazônica (áreas de exploração de madeira de impacto reduzido, exploração convencional de madeira, plantação de óleo de palma e pecuária). Nós testamos quatro hipóteses: H1. As características ambientais dos riachos são alteradas pelos diferentes usos do solo; H2. Riachos em área influenciada por usos do solo são ambientalmente menos heterogêneos do que os riachos em áreas relativamente não perturbadas (controle); H3. A diversidade α de EPT é menor nos riachos que envolvem os usos do solo; H4. A diversidade β de EPT é menor entre os riachos influenciada por usos do solo do que entre os riachos controle.

2º sessão

Na segunda sessão, o nosso objetivo foi avaliar a influência do ambiente e do espaço na estruturação da metacomunidade de Ephemeroptera, Plecoptera e Trichoptera (EPT). Nós também avaliamos a resposta individual de gêneros de EPT aos processos de estruturação da metacomunidade em diferentes escalas espaciais. Nós estudamos as relações entre características ambientais e distribuição de EPT utilizando o riacho como nosso grão, inserido em uma expressiva extensão espacial (duas bacias de drenagem).

3º sessão

Na terceira sessão, o nosso objetivo foi determinar os limiares de resposta das comunidades de Odonata, Gerromorpha e EPT em gradientes de distúrbio antrópico na escala local e de bacia de drenagem. Como a resposta da comunidade emerge das respostas observadas das populações individualmente (Haddad et al., 2015), nós também buscamos identificar os limiares das espécies, cujas frequências podem estar negativamente ou positivamente associadas aos gradientes de distúrbios.

2. Sessão I

Land-use determines environmental heterogeneity and beta diversity of aquatic insects in Amazonian ecosystems

A primeira sessão desta tese foi elaborada e formatada conforme as normas da publicação científica *Freshwater Biology*, disponível em: https://onlinelibrary.wiley.com/page/journal/13652427/homepage/forauthors.htm

Land-use determines environmental heterogeneity and beta diversity of aquatic insects in Amazonian ecosystems

2.1 Abstract

Understanding how different components of biological diversity are affected by changes in environmental characteristics is crucial for the implementation of effective conservation and restoration measures of the Amazonian streams. Based on this perspective, we tested whether environmental characteristics and environmental heterogeneity are altered by land use and how these alterations affect the α-diversity and β-diversity of Ephemeroptera, Plecoptera and Trichoptera assemblages (EPT). We sampled 83 streams distributed in relatively unimpacted (21 control) sites and sites impacted by conventional logging, reduced-impact logging, oil palm plantation and, livestock farming (62 land use). We used analysis of variance to examine whether land use affected environmental characteristics and α-diversity of EPT. We then performed multivariate dispersion analysis to assess if environmental heterogeneity and β-diversity of EPT decreased in areas under human land use. We found that environmental characteristics were altered by human land use; however, environmental heterogeneity was surprisingly higher in livestock farming sites. With regard to the components of diversity, EPT α-diversity was higher in control sites while β-diversity was higher in sites impacted by human land use. Our results advance the understanding of human land use impact on the environmental condition and biological diversity of tropical streams, demonstrating that human land use may have opposite effects on α- and β-diversity and that impacts differ across the different types of land use disturbance.

Keywords: environmental variation; macroinvertebrates; anthropic activities; diversity

2.2 Introduction

Environmental heterogeneity is a natural phenomenon that represents the spatial variation in environmental conditions within a region. Stream ecosystems are highly heterogeneous in their habitat features (Stein, Gerstner, & Kreft, 2014; Heino, Melo, & Bini, 2015), which contribute to high biology diversity owing to the accumulation of species with different environmental affinities (Southwood, 1977; Muotka, Paavola, Haapala, Novikmec, & Laasonen, 2002; Brown, 2003; Stein, Gerstner, & Kreft, 2014). However, anthropic activities modify the land use around streams, which may alter instream environmental condition, its spatial heterogeneity, and consequently, the biological communities across the landscape (Giam et al., 2015; Cunha & Juen, 2017).

The main anthropic activities responsible for changing the environmental characteristics of the Amazonian streams are livestock farming, agriculture (e.g. soybean and oil palm), and logging (Imazon, 2017). Their impacts include the loss of riparian vegetation, which results in an increased fine sediment loading in the streambed, a higher instability of the stream banks, and increased concentrations of nutrients and pollutants in the water (Belsky, Matzke, & Uselman, 1999; Allan, 2004; García-García, Vázquez, Novelo-Gutiérrez, & Favila, 2017). Furthermore, these activities can decrease thalweg depth, and dissolved oxygen concentration (Dias, Magnusson, & Zuanon, 2010; Nogueira, Calvão, Montag, Juen, & De Marco, 2016, Prudente, Pompeu, Juen, & Montag, 2017).

One of the most widely used methods to evaluate the impact of human land use on stream biota is measuring local species richness (α-diversity) and the variation in the species composition among sites (β-diversity) (e.g. Johnson & Angeler, 2014; McCreadie et al., 2017). Usually the α-diversity decreases in areas where human land use changes the environmental conditions (Ligeiro et al., 2013; Murphy & Romanuk, 2014; Paiva, Faria, Calvão, & Juen, 2017). For instance, some studies have showed that changes in pH, temperature and turbidity of the water due to land use (Astorga, Heino, Luoto, & Muotka, 2011; Cunha, Montag, & Juen, 2015) along with proximity to roads (Paiva, Faria, Calvão, & Juen, 2017) affected the richness of macroinvertebrate species in streams.

Environmental heterogeneity has been described as an important driver of β-diversity (Astorga et al., 2014). Despite a number of studies examining the impact of anthropic activities on environmental heterogeneity and β-diversity, there is little consensus among these studies (Heino et al., 2009; Gutiérrez-Cánovas, Millán, Velasco, Vaughan, & Ormerod, 2013; Bini, Landeiro, Padial, Siqueira, & Heino, 2014; Johnson & Angeler, 2014; Larsen & Ormerod, 2014; Hawkins, Mykrä, Oksanen, & VanderLaan, 2015; Fugère, Kasangaki, & Chapman, 2016). Some studies held in different regions of the world have showed a positive relationship between environmental heterogeneity and β-diversity of aquatic macroinvertebrates (Astorga et al., 2014) and diatoms (Zorzal-Almeida, Bini, & Bicudo, 2017), but other studies documented the lack of a relationship (Heino et al., 2013, Grman, Orrock, Habeck, Ledvina, & Brudvig, 2015).

Understanding how the components of diversity (α and β) are affected by changes in environmental characteristics is crucial to the implementation of effective environmental conservation and restoration measures (e.g. Arponen, Moilanen, & Ferrier, 2008; Ferrier & Drielsma, 2010; Ritter et al., 2017), mainly considering the current scenario of transformation of the Amazonian landscape by the intense land use (Fonseca et al., 2018; INPE, 2018). In this study, we assessed environmental heterogeneity and α and β-diversity of Ephemeroptera, Plecoptera and Trichoptera (EPT) in various human land-uses categories that are dominant in the Amazonian landscape (reduced-impact logging areas, conventional logging, oil palm plantation and livestock farming). We used EPT as model group in this study because they are notably sensitive to changes in water quality and environmental characteristics of streams (Ferro & Sites, 2007; Righi-Cavallaro, Roche, Froehlich, & Carvalho, 2010; Boyero et al., 2011; Ligeiro et al., 2013). We tested four hypotheses: H1. Environmental characteristics of streams are altered by the different land uses; H2. Streams surrounded by human land-uses are environmentally less heterogeneous than streams in relatively undisturbed areas (control streams); H3. EPT α-diversity is lower in streams surrounding human land uses; H4. EPT β-diversity is lower among streams surrounded by human land use than among control streams.

2.3 Materials and methods

2.3.1 Sampling area description

We sampled 83 streams of 1st to 3rd Strahler (1957) order (mean width of 2.7 m and mean depth of 0.28 m). The streams were distributed in the Northeastern and Southeastern mesoregion of the State of Pará, in an area known as the ‘Arc of Deforestation’ that extends across four Brazilian states (Pará, Mato Grosso, Rondônia, and Acre). The Arc of Deforestation is a region where the agricultural frontier and deforestation advance more intensively towards the forest (INPE, 1997; IBGE, 2011). These landscape changes began in the 1970s due to government financial incentives for farmers to establish plantations in the region (Mahar 1979, Smith, Serrão, Alvim, & Falesi, 1995, Aldrich, Walker, Simmons, Caldas, & Perz, 2012). The sampled streams were located in two hydrological units (Capim and Acará basins) in the Atlantic Coast-Northeast hydrological region (ANA, 2010). This hydrological region was defined according to the homogeneity of geophysiological characteristics (e.g., geomorphology, geology, hydrography, soils and hydroclimatic factor) (Resolution nº 07/2008). Therefore, we can state that all sites belonged to the same ecoregion called Atlantic Coast - Northeast (*sensu* Omernik & Griffith, 2014), what makes all the sites comparable in their biotic and abiotic components. We sampled streams in control sites (n = 21), conventional logging (n = 9), reduced-impact logging (n = 21), oil palm plantation (*Elaeis guineensis* Jacq.) (N = 22), and livestock farming sites (n = 10) (Figure 1).

Streams in control sites were located in two spatial clusters. We therefore assigned the two clusters of sites to separate groups: CON1 (n = 13) and CON2 (n = 8) in order for inter-stream distances within each control group to be comparable to the other human land uses, thus avoiding potential biases due to differences spatial dispersion across groups. Control streams are in native forest areas or inside large forest fragments. In reduced-impact logging (RIL) sites, logging cycles occur after 30 years of forest regeneration and riparian vegetation along streams are preserved. By contrast, in conventional logging (CL), timber is removed without regard for riparian protection, which results in riparian vegetation impairment and accumulation of woody debris on the forest surface (more details see Calvão, Nogueira, Montag, Lopes, & Juen, 2016).

Oil palm streams were divided into two groups, plantation of primary oil palm (PPP, n = 17) and plantation of secondary oil palm (PSP, n = 5), based on differences in the historical context of the type of land use at the time of planting the crop, which may affect aquatic invertebrates (Harding, Benfield, Bolstad, Helfman, & Jones, 1998; Al-Shami et al., 2013). In PPP sites, oil palm plantations were directly converted from forest. In these sites, riparian vegetation is preserved according to Brazilian legislation (Lei 12.651 of May 25, 2012), but the streams are used as irrigation source for young plantations (Agropalma, 2015). In the PSP area, the plantations were converted from non-forest land such as lands planted with other crops, pastures for livestock farming or deforested areas (Biopalma, 2015), following the federal proposal of the Ecological-Economic Zoning (EEZ) (Ramalho-Filho, Motta, Freitas, & Teixeira, 2010). In these sites, riparian vegetation is sparsely present in some stream reaches. While chemical fertilizers and pesticides are used in both PPP and PSP sites, organic fertilizers of industrial co-products (empty fruit bunch and palm fibers) are also used in PSP sites. There were fewer streams sampled in PSP than PPP and other human land uses because PSP is not as commonly found as the other land-use types since the EEZ (which stipulated for oil palm to be planted on non-forest lands) was only proposed in 2010. Lastly, in livestock farming sites, the cattle are allowed to consume all the grass in a particular area before moving to the next area on the farm (rotational grazing system)Then, the latter area is left to rest until it is restored (Silva & Barreto, 2014). In addition, livestock farmers use tanks, artificial dams or streams to provide drinking water to cattle.

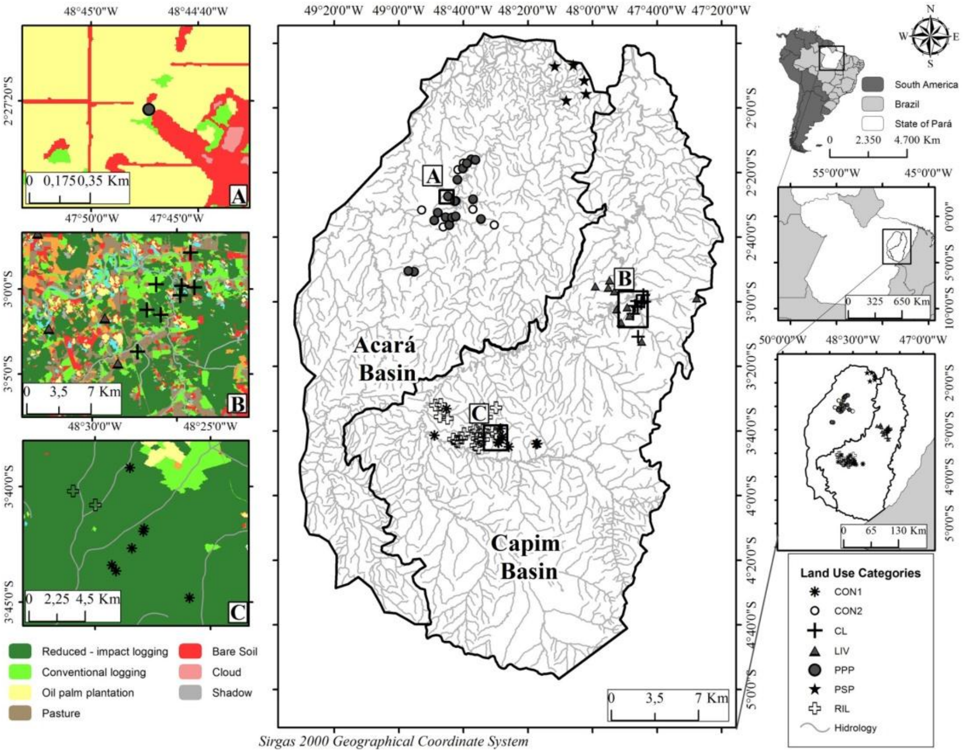

Figure 1. Geographic location of the 83 stream sites sampled in the Northeastern and Southeastern mesoregion of the state of Pará, Brazil. The streams were distributed in control areas (CON1 and CON2), conventional logging (CL), reduced-impact logging (RIL), plantation of primary oil palm (PPP), plantation of secondary oil palm (PSP) and livestock farming (LIV). The geographical coordinates of each stream are available in the supporting information on table SI1.

2.3.2 Physical and chemical factors

We measured the environmental characteristics along each 150 m stream reach used for aquatic macroinvertebrates samping. Each stream reach was subdivided into 10 longitudinal sections of 15 m and 11 transversal sections, following the protocol of the United States Environmental Protection Agency (US-EPA, Peck et al., 2006) adapted to tropical areas (Macedo et al., 2014). We used the following the environmental variables to characterize stream physical characteristics: mean thalweg depth (cm), % of riffle habitats, % of substrate organic detritus and mean canopy density at stream bank. These variables are considered important for assessing habitat conditions and channel disturbance (Kaufmann, Levine, Robison, Seeliger, & Peck, 1999).

Thalweg depth was measured in the deepest part of the longitudinal section of the streams and % of riffle habitats were measured along the thalweg. The % of organic substrate includes partially decomposed soil detritus and plant material between 0.053 mm and 2mm in size. Lastly, the canopy density at bank was measured at the wetted channel margin at both sides (left and right) with a spherical densiometer (convex type) positioned 0.30 cm above the surface of the water. Calculation of all these physical characteristics was made following Kaufmann et al. (1999).

The chemical characteristics of the water included the dissolved oxygen (mg/L) and pH, each one measured at three points along the longitudinal section in each stream (downstream, middle and upstream). Then, a mean value of each variable was calculated. The chemical water variables were measured before macroinvertebrate sampling and physical habitat measurements to avoid disturbing the water. The six environmental characteristics used as predictors are summarized in table 1.

Table 1. Environmental characteristics (mean ± standard deviation) of the 83 streams sampled in the control streams (CON1 and CON2), conventional logging (CL), reduced-impact logging (RIL), plantation of primary oil palm (PPP), plantation of secondary oil palm (PSP) and livestock farming (LIV).

Land uses	**Environmental characteristics**					
	Thalweg depth (cm)	**% of riffle habitats**	**% of organic substrate**	**Canopy density at bank**	**pH**	**Dissolved oxygen (mg / L)**
CON1	23.85 ± 10.79	18.56 ± 17.60	11.63 ± 10.39	94.65 ± 3.65	5.09 ± 0.93	6.06 ± 1.30
CON2	30.98 ± 15.20	32.94 ± 30.33	31.42 ± 16.27	96.79 ± 4.18	4.72 ± 0.24	7.03 ± 2.11
CL	20.96 ± 5.18	4.59 ± 5.10	0.97 ± 1.04	84.11 ± 3.30	4.74 ± 0.37	4.43 ± 1.31
RIL	25.24 ± 10.09	21.32 ± 20.92	8.89 ± 9.78	96.8 ± 1.72	4.56 ± 0.26	6.71 ± 1.11
PPP	35.38 ± 10.97	26.51 ± 15.75	24.01 ± 13.26	92.67 ± 7.30	4.79 ± 0.44	6.34 ± 1.14
PSP	33.64 ± 6.62	12.93 ± 9.69	42.50 ± 21.07	74.60 ± 5.03	4.68 ± 0.19	3.74 ± 0.72
LIV	38.62 ± 14.20	40.13 ± 31.79	4.00 ± 3.27	64.09 ± 34.77	5.28 ± 0.31	11.13 ± 1.51

2.3.3 EPT sampling

Each stream was sampled once, always in the period of least precipitation in the region (August - December) to avoid seasonal effects on environmental characteristics of the stream reach and on the EPT communities. For EPT sampling, a 150 m reach was defined in each stream, which was subdivided into 10 longitudinal sections of 15 m. Each longitudinal section was again subdivided into three segments of 5 m. Two substrate samples (forming one composite sample) were collected using an 18 cm diameter dipnet with 250 μm mesh in the first two segments of each section, totalizing twenty segments collected per stream (more details see Juen et al., 2016). The sampling covered all different habitats present at the stream, including inorganic substrate, macrophytes, leaf litter and parts of terrestrial vegetation immersed in the water. This methodology has been effective in studies that assess environmental impacts on aquatic ecosystems in the Amazon and the in the Cerrado Brazilian biomes (Shimano, Juen, Salles, Nogueira, & Cabette, 2013; Brasil, Juen, Giehl, & Cabette, 2016; Juen et al., 2016; Nogueira, Calvão, Montag, & Juen, 2016; Faria, Ligeiro, Callisto, & Juen, 2017; Paiva, Faria, Calvão, & Juen, 2017).

The EPT specimens collected were fixed in 85% ethanol and identified at the genus level using the specialized keys of Pes et al. (2014), Hamada & Silva (2014), Domínguez et al. (2006), Salles & Domínguez (2012) and Salles et al. (2014). Studies evaluating anthropogenic impacts on aquatic insects using the taxonomic resolution of genus have presented consistent results (e.g. Ligeiro et al., 2013; Bertaso, Spies, Kotzian, & Flores, 2015; Solar et al., 2016; Chen et al., 2017; Luiza-Andrade et al., 2017; Paiva, Faria, Calvão, & Juen, 2017). The specimens sampled were deposited in the collection of Aquatic Insects of the Laboratory of Ecology and Conservation at the Federal University of Pará, Brazil.

2.3.4 Statistical analyses

Each stream was considered an independent sample unit in the analyses. To test our first hypothesis (H1. Environmental characteristics of streams are altered by the different land uses), we applied a Permutational Multivariate Analysis of Variance (PERMANOVA; Anderson, 2001) to examine whether multivariate environmental characteristics were different among the different land uses categories, followed by pairwise comparisons between each pair of land use categories. In the PERMANOVA analysis, we used a Euclidean distance matrix calculated from the six standardized (mean = 0 and standard deviation = 1) stream environmental characteristics and we assessed statistical significance using a Monte Carlo method with 10,000 random permutations. P-values $\leq$ 0.05 were regarded as statistically significant. We used Principal Coordinates Analysis (PCoA) to visualize differences among land uses categories and correlations were made between environmental characteristics and PCoA axis to identify which characteristics contributed to ordination among sites (Legendre & Legendre, 2012).

We evaluated the differences of spatial dispersion between category of land use to verify the possibility of spatial bias in the analysis of environmental heterogeneity and β-diversity (following Heino et al., 2013; Tonkin, Heino, Sundermann, Haase, & Jähnig, 2016), since more clumped sites would be more prone to have lower habitat heterogeneity and beta diversity. For evaluating the spatial dispersion, the linear distances between all sites were calculated using the Euclidean distance on the geographic coordinates (lat-long) of the streams (see Supporting Information Table SI1). We did not observe a significant difference in the spatial dispersions between categories ($F_{(6,76)} = 1.970$; $P = 0.07$; Figure IS1). Therefore, any differences found in environmental heterogeneity and β-diversity was not due to the spatial arrangement of the streams.

To test our second hypothesis (H2. Streams surrounded by human land-uses are environmentally less heterogeneous than streams in relatively undisturbed areas) and fourth hypothesis (H4. EPT β-diversity is lower among streams surrounded by human land use than among control streams.), we used Multivariate Dispersion Analysis (PERMDISP, Anderson, Ellingsen, & Mcardle, 2006). The PERMDISP analyses calculated, for each response matrix (environmental and biological), the distance of each site to its group centroid (where each group is a land use category). Therefore, the mean distance to the centroid at each category of land use was used as a measurement of environmental heterogeneity and EPT β-diversity. The higher the mean distance to the centroid, the greater the dispersion in the response in a given land use category (Anderson, Ellingsen, & Mcardle, 2006). The differences observed in PERMIDISP were tested with 10,000 permutations (P-values ≤ 0.05 were taken as statistically significant). The environmental characteristics were used with the measure of environmental heterogeneity in each category of land use. For the analysis of environmental heterogeneity, the six environmental characteristics were standardized (mean = 0, standard deviation = 1) and the dissimilarity matrix was constructed using the Euclidean distance. The β-diversity was calculated using the Jaccard dissimilarity index, which only considers taxa occurrences (presence/absence data), since we were especially concerned about the changes in the taxonomic composition between assemblages.

To test our third hypothesis (H3. EPT α-diversity is lower in streams surrounding human land uses), we used the genera richness of EPT in each stream as a measure of α-diversity, and we tested if there were differences between the different land use categories with a one-way analysis of variance (ANOVA, Zar, 2010), followed by *post hoc* comparisons using the Tukey's test, when significant differences were observed. In addition, we tested the effects of the environmental characteristics on α-diversity using Generalised Additive Modelling (GAM) generated using the quasi-Poisson error distribution (Zuur, Ieno, Walker, Saveliev, & Smith, 2009).

We performed all analyses in R v. 3.3.2 (R Core Team 2016), using the "MASS" (Venable & Ripley, 2002), "mgcv" (Wood, 2006), “stats” (R Core Team 2016) and "vegan" packages (Oksanen et al., 2016) for multivariate analyses.

2.4 Results

2.4.1 Environmental characteristics of streams

There was a difference in environmental characteristics between control and human land uses (pseudo-$F_{(6,76)} = 6.377$; $p < 0.01$). The environmental difference was greatest between streams in livestock farming (LIV) and conventional logging (CL) sites (Table 2). LIV had greater thalweg depth, % of riffle habitats, pH and dissolved oxygen, but lower % of organic substrate and canopy density compared to other. This category of land use (LIV) also presented a greater variation in the values of canopy density between streams (Table 1). In the other side, CL has lower thalweg depth, % of rifle habitats, % of organic substrate and canopy density at stream bank (Figure 2, Table IS2).

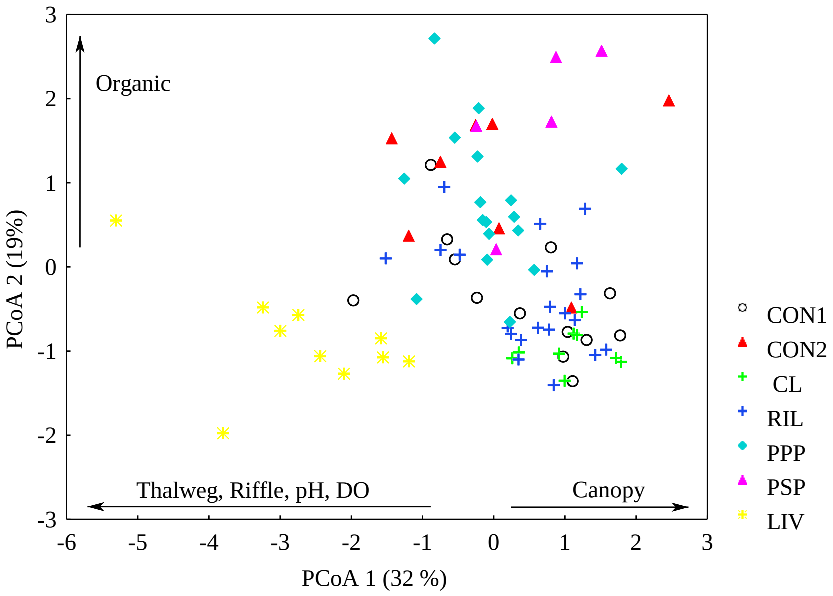

Figure 2. Principal Coordinate Analysis (PCoA) plot of the environmental characteristics among land uses categories. The arrows indicating the environmental characteristic relationship with the axis. The results of this relationship were shown in Table IS2. The categories control (CON1 and CON2), conventional logging (CL), reduced-impact logging (RIL), plantation of primary oil palm (PPP), plantation of secondary oil palm (PSP) and livestock farming (LIV).

Table 2. Pairwise comparisons from PERMANOVA analysis calculated from standardized stream environmental characteristics between categories control (CON1 and CON2), conventional logging (CL), reduced-impact logging (RIL), plantation of primary oil palm (PPP), plantation of secondary oil palm (PSP) and livestock farming (LIV). P-values ≤ 0.05 were regarded as statistically significant are marked in bold.

	CON1	CON2	CL	RIL	PPP	PSP	LIV
CON1							
CON2	**0.028**						

CL	**0.006**	**<0.001**				
RIL	0.782	**0.012**	**0.003**			
PPP	**0.011**	0.450	**<0.001**	**0.003**		
PSP	**<0.001**	0.070	**<0.001**	**<0.001**	**0.002**	
LIV	**0.003**	**0.015**	**0.005**	**<0.001**	**0.002**	**0.019**

2.4.2 Environmental heterogeneity

The environmental heterogeneity differed significantly among controls and land use types ($F_{(6,76)}$ = 4.539, p = 0.001). Streams in LIV and relatively unimpacted (control; CON1 and CON2) sites were most environmentally heterogenous. Streams in oil palm plantation (PPP and PSP) sites did not differ from control streams (CON1 and CON2), whereas streams in reduced-impact logging (RIL) and CL sites had lower environmental heterogeneity than controls (Figure 3, Table 3). However, among the land uses, LIV had greater environmental heterogeneity than CL, RIL and PPP, but LIV did not differ from PSP.

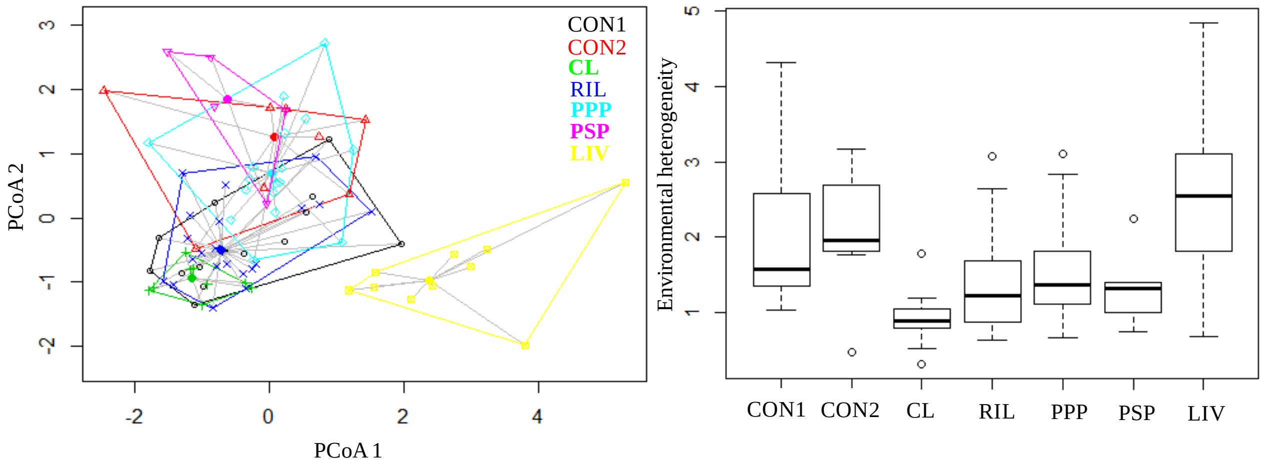

Figure 3. Principal Coordinate Analysis (PCoA) and Box plots of values of distances to group centroids calculated by PERMIDISP representing the environmental heterogeneity. Points in PCoA are colour-coded according to categories: control streams (CON1 and CON2), conventional logging (CL), reduced-impact logging (RIL), plantation of primary oil palm (PPP), plantation of secondary oil palm (PSP) and livestock farming (LIV).

Table 3. Pairwise comparisons of environmental heterogeneity between categories control (CON1 and CON2), conventional logging (CL), reduced-impact logging (RIL), plantation of primary oil palm (PPP), plantation of secondary oil palm (PSP) and livestock farming (LIV). *The mean distance to the group centroid was calculated for each category using PERMDISP.). P-values ≤ 0.05 were regarded as statistically significant are marked in bold.

	Environmental heterogeneity							
	CON1	CON2	CL	RIL	PPP	PSP	LIV	Mean distance to centroid*
CON1								2.080
CON2	0.992							2.075
CL	**0.006**	**0.003**						0.932

RIL	**0.030**	**0.036**	0.073					1.397
PPP	0.137	0.147	**0.021**	0.430				1.582
PSP	0.164	0.120	0.142	0.877	0.511			1.345
LIV	0.303	0.336	**0.001**	**0.002**	**0.013**	0.054		2.583

2.4.3 EPT diversity

We collected a total of 16.038 EPT specimens, representing 51 genera distributed across 18 families (Table SI3). The mean (± standard deviation) abundance of EPT was significantly greater in the streams in PSP (472 ± 219) and LIV (287 ± 256) sites compared to the other land-use categories.

The α-diversity of EPT was significantly higher in control streams (CON1 and CON2) and RIL (ANOVA, $F_{(6,76)} = 5.657$; $p < 0.001$) compared to streams in other land-use categories. Streams in CL, PPP and LIV sites had significantly lower α-diversity than control streams (CON1), but there was no significant difference between LIV and PSP streams (Figure 4, Table 4). The % of riffle habitat was positively associated with EPT α-diversity ($F = 4.75$, $p = 0.03$). Detailed results on the environmental correlates of EPT α-diversity are available in the supporting information in Table SI5.

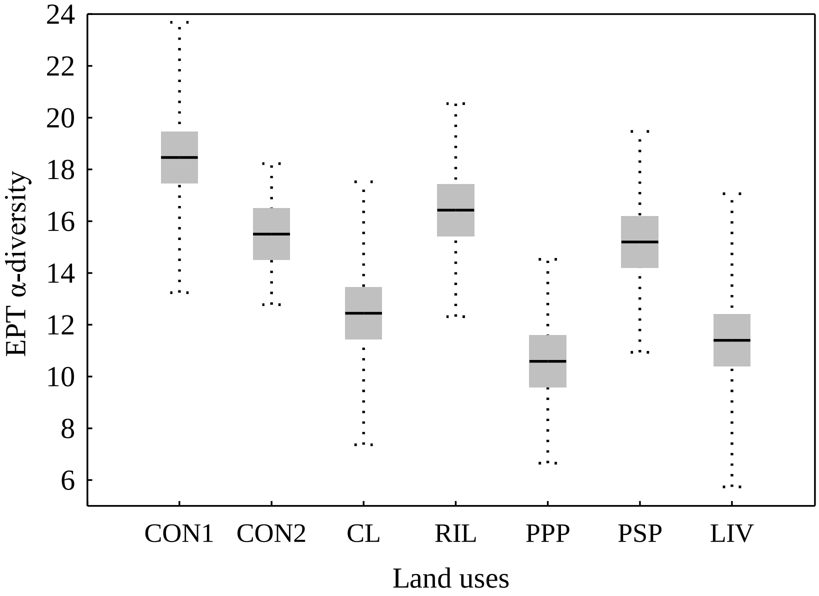

Figure 4. Box plot (mean and standard deviation) of the EPT α-diversity in the control streams (CON1 and CON2), conventional logging (CL), reduced-impact logging (RIL), plantation of primary oil palm (PPP), plantation of secondary oil palm (PSP) and livestock farming (LIV).

Table 4. *Post hoc* comparisons of one-way analysis of variance of EPT α-diversity between categories control (CON1 and CON2), conventional logging (CL), reduced-impact logging (RIL), plantation of primary oil palm (PPP), plantation of secondary oil palm (PSP) and livestock farming (LIV). P-values ≤ 0.05 were regarded as statistically significant are marked in bold.

EPT α-diversity

	CON1	CON2	CL	RIL	PPP	PSP	LIV	Mean genera richness
CON1								18
CON2	0.763							16
CL	**0.043**	0.800						12
RIL	0.858	0.999	0.294					16
PPP	**<0.001**	0.157	0.952	**0.003**				11
PSP	0.811	1.000	0.926	0.998	0.412			15
LIV	**0.006**	0.471	0.999	0.067	0.999	0.717		11

EPT β-diversity differed across category of land use (PERMDISP, $F_{(6,76)} = 2.964$, $p = 0.016$). The β-diversity was highest in LIV and PPP streams and the lowest in PSP streams. PPP streams had significantly higher EPT β-diversity than CON1, CON2, RIL, and PSP sites. Streams in LIV sites had significantly higher β-diversity than CON2 and RIL sites. (Figure 5, Table 5).

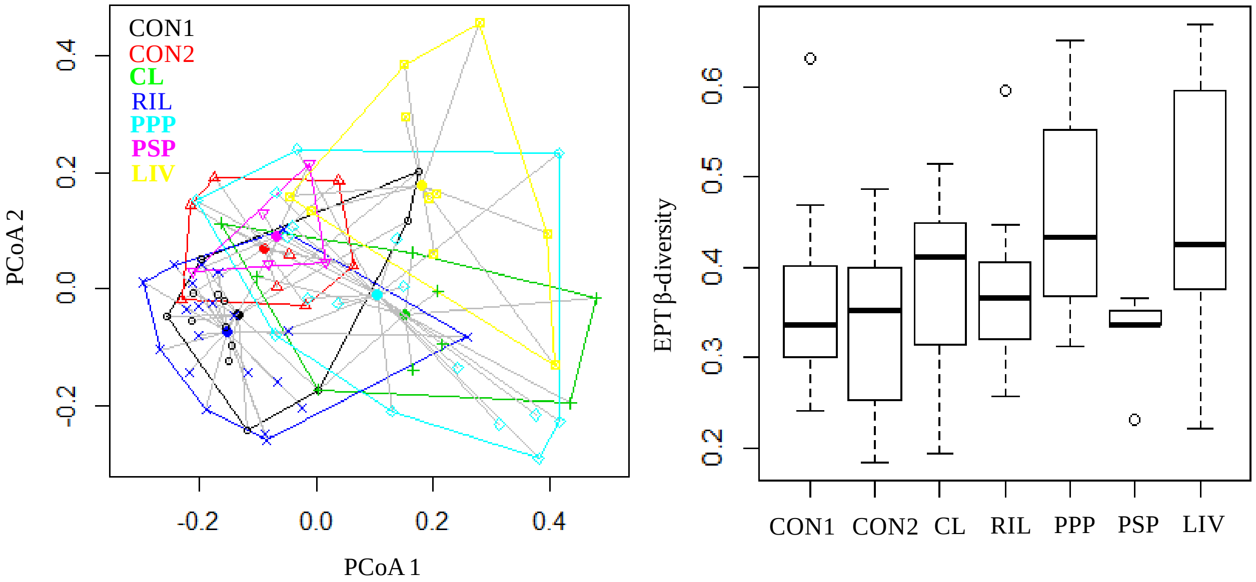

Figure 5. Principal Coordinate Analysis (PCoA) and Box plots of values of distances to group centroids calculated by PERMIDISP representing the environmental heterogeneity. Points in PCoA are colour-coded according to categories: control streams (CON1 and CON2), conventional logging (CL), reduced-impact logging (RIL), plantation of primary oil palm (PPP), plantation of secondary oil palm (PSP) and livestock farming (LIV).

Table 5. Pairwise comparisons of EPT β-diversity between categories control (CON1 and CON2), conventional logging (CL), reduced-impact logging (RIL), plantation of primary oil palm (PPP), plantation of secondary oil palm (PSP) and livestock farming (LIV). *The mean distance to the group centroid was calculated for each category using PERMDISP.). P-values ≤ 0.05 were regarded as statistically significant are marked in bold.

				EPT β-diversity				
	CON1	CON2	CL	RIL	PPP	PSP	LIV	Mean distance to centroid*
CON1								0.367
CON2	0.497							0.335

CL	0.782	0.391						0.380
RIL	0.881	0.286	0.807					0.371
PPP	**0.042**	**0.018**	0.125	**0.011**				0.451
PSP	0.405	0.841	0.307	0.189	**0.021**			0.324
LIV	0.063	**0.040**	0.149	**0.017**	0.736	0.048		0.467

2.5 Discussion

In this paper, we examined the effects of different human land uses on the environmental characteristics, environmental heterogeneity of these environmental characteristics, and the macroinvertebrate diversity patterns of Amazonian streams. Streams that drain areas used for livestock farming (LIV) and conventional logging (CL) had greater variation in terms of environment characteristics related to thalweg depth, riffle habitats, dissolved oxygen, organic substrate, and canopy density at bank compared to other land use categories.

The large variation and low canopy density in LIV streams can be attributed to the extremely patchy canopy cover as farmers often convert a large part of the riparian habitat into pastures for cattle grazing. Further, there is a large amount of variation in cattle densities among farms resulting in different grazing intensities (Toutain et al., 2010) and different levels of riparian vegetation disturbance (Scrimgeour & Kendall, 2003; Matthaei, Weller, Kelly, & Townsend, 2006). The high dissolved oxygen concentration in LIV streams may be associated with greater proportion of riffle habitats. In addition, as organic detritus can be removed due to water current (Lepori & Malmqvist, 2007) asymmetric sandbanks may arise along of channel resulting in a channel with greater thalweg depth. We would like to emphasize the greater heterogeneity found in LIV does not mean that streams have higher environmental quality since it measures the variation in environmental characteristics among streams but not the quality of the environment for stream biota. Environmental heterogeneity among LIV streams were likely a consequence of the highly varied management methods and techniques used across different farms and sampling sites.

The environmental characteristics evaluated in this study did not change substantially between RIL and control stream (CON1), but substrate organic detritus was lower in RIL. Thus, this variable may have been responsible for this category of land use to present lower heterogeneity, because the environmental heterogeneity of aquatic ecosystems is generally measured using variables related the physical structure of stream, such as riparian vegetation and channel morphology (Palmer, Menninger, & Bernhardt, 2010). The lower heterogeneity among CL streams is likely a consequence of the intense disturbance associated with conventional logging (Houghton et al., 2000; Saatchi, Houghton, Dos Santos Alvalá, Sorares, & Yu, 2007), where trees of all sizes were indiscriminately felled and dragged through the forest floor and across streams without any protections for these habitats. This results in changes to the ecological processes associated with streams as well as their physical properties, such as stream flow velocity (Troendle & Olsen, 1994), deposition of large organic debris on channel that cover up small organic debris used by the organisms for food or shelter (Campbell & Doeg, 1989; Boyero et al., 2011; Hamid & Rawi 2011), decrease of riparian vegetation and dissolved oxygen concentration (Cardoso, Calvão, Montag, Godoy, & Juen, 2018). We found similar results in our study because the streams in the CL area had lower riffle habitats, organic substrate, dissolved oxygen and canopy density at bank.

The environmental heterogeneity of the oil palm area (PPP and PSP) did not differ from the control (CON1 and CON2) streams likely because oil palm streams had riparian canopy cover that is generally high like the control streams. Although the environmental characteristics of these streams were altered, the similarity in riparian vegetation cover may mask the effects of anthropogenic stressors (Tonkin, 2014; Carvalho, Roque, Barbosa, Montag, & Juen, 2018). When we compare the heterogeneity among land uses, LIV had greater heterogeneity than CL, RIL and PPP. We believe it is associated with extremes of presence and absence of riparian vegetation that caused greater variation in the environmental characteristics in this category of land use since each stream was distinctly affected. In the other hand, the similarity in environmental heterogeneity between PSP and LIV may be associated the fact that PSP streams may still reflect the effects of past land use(s) prior to the establishment of current oil palm plantations, because land use legacies can be persistent and the resilience of the physical structure of environment advances slowly (Foster et al., 2003).

EPT α-diversity and β-diversity showed distinct responses to the effect of land use. As expected, the α-diversity was lower in streams affected by human land use. Human land use altered the stream environment, thus modifying the physical habitat experienced by the EPT communities. The local habitat is a determining factor driving the community structure of aquatic biota (Southwood, 1977), and habitat degradation usually results in a decrease of taxonomic richness (Rawi, Al-Shami, Madrus, & Ahmad, 2013). In this study, the riffle habitat variable was positively related with α-diversity. The relationship of richness with riffle was found in other studies (Roy, Rosemond, Leigh, Paul, & Wallace, 2003; Amaral, Silveira, Rosa, Oliveira, & Alves, 2015). The coarse substrate associated with riffle mesohabitats may be particularly important in providing food and shelter for a large number of species (Principe, Raffaini, Gualdoni, Oberto, & Corigliano, 2007). Interestingly, despite LIV having the greatest % of riffle habitat, its mean α-diversity remains lower than that of CON1 and CON2 sites, suggesting the impairment of other geomorphologic and hydrologic factors important for a rich EPT community (Principe Raffaini, Gualdoni, Oberto, & Corigliano, 2007, Heino et al., 2017).

The high β-diversity in LIV and PPP streams may be associated with the occurrence of genera sampled only in these streams. This suggests that taxa occupying the same habitat in different streams can be more similar than taxons occupying different habitats in the same stream (Silva, Ligeiro, Hughes, & Callisto, 2014). We believe that this factor may have been responsible for generating higher β-diversity in these two land-use categories. Thus, the environmental factors of the streams may determine the local diversity (Grönroos & Heino, 2012) and an increase or decrease in the environmental structure can enable the occurrence of determined genus because a habitat-specific (e.g. macrophyte and sandbanks) can increase in the changed environment. For example, *Harpagobaetis*, *Leptohyphes*, *Paracloeodes* e *Traverhyphes* (all Ephemeroptera) occurred only in streams influenced by livestock (LIV), and the habitat preferences of these taxa are related to presence of submerged vegetation, channel with sandy bottoms and higher proportion of riffle (Molineri, 2003; Domínguez, Molineri, Pescador, Hubbard, & Nieto, 2006; Schwiebert, 2007). These environmental characteristics were predominant in LIV streams with absence of riparian vegetation. Although these genera occurred only in this category, LIV had lower richness, but the decrease in richness may not necessarily mean change in community composition and simply a reduction in taxa (Larsen & Ormerod, 2010; Feld, de Bello, & Dolédec, 2013).

The Ephemeroptera genera *Cryptonympha* and *Macunahyphes* were only found in PPP streams. A study carried out in Amazonian streams also found these genera to be restricted to oil palm plantations (Shimano & Juen, 2016). In general, *Cryptonympha* is found in semi-lentic and lotic altered environments (Shimano, Cabette, Salles, & Juen, 2010; Shimano, Salles, & Cabette, 2011) as well as along sandbanks (Derka & Nieto, 2018). *Macunahyphes* is a newly described species (Dias, Salles, & Molineri, 2005); therefore, there is a paucity of information about its habitat affinity. We found this genus in a stream with a mean thalweg depth of 19cm, predominance of riffle habitat and high canopy density at bank. Compared to PPP, the lower β-diversity in PSP may be related to the environmental context in this area because community structuring may be dependent on the context between catchments (Heino, Grönroos, Soininen, Virtanen, & Muotka, 2012) and land use legacies, since it can affect communities for decades (Harding, Benfield, Bolstad, Helfman, & Jones, 1998; Foster et al., 2003).

2.6 Conclusion

Our study provides new information on effects of human land use on the environmental condition and biological diversity of streams and how their heterogeneity across space. Our results show that human land use affects α- and β-diversity of EPT taxa differently and that certain land-uses have larger impacts than others. While human land uses altered the stream environment, environmental heterogeneity was highest among streams located in livestock farming sites and control streams. The former result likely reflects the large variation in farming methods and techniques driving large differences in stream environmental conditions. In addition, the absence of systematic land use techniques makes it difficult to propose conservation measures because there is no pattern in use of the resource and the streams are altered distinctly. All human land-uses resulted in a decline in EPT α-diversity relative to control sites. However, EPT β-diversity was generally higher in streams influenced by human land use and highest in livestock farming sites. We hence suggest that the studies evaluating anthropic activities must include the evaluation of local environmental heterogeneity and that it is important to consider the historical context of land use change when evaluating the biodiversity impacts of human land use.

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2.9 Supporting Information

Table SI1. The geographic coordinate data set of the streams the control streams (CON1 and CON2), conventional logging (CL), reduced-impact logging (RIL), plantation of primary oil palm (PPP), plantation of secondary oil palm (PSP) and livestock farming (LIV).

Land Use	Latitude (Y)	Longitude (X)
CON1	-3.73127	-48.28472
CON1	-3.73216	-48.28948
CON1	-3.72335	-48.48768

CON1	-3.65307	-48.47472
CON1	-3.69891	-48.46499
CON1	-3.69696	-48.46444
CON1	-3.59157	-48.60641
CON1	-3.68783	-48.81531
CON1	-3.55106	-48.75340
CON1	-3.54578	-48.55860
CON1	-3.71111	-48.47323
CON1	-3.74678	-48.43165
CON1	-3.72741	-48.48464
CON2	-2.61359	-48.77034
CON2	-2.31874	-48.69424
CON2	-2.28428	-48.66568
CON2	-2.47869	-48.70494
CON2	-2.38092	-48.80201
CON2	-2.60319	-48.50678
CON2	-2.52248	-48.61725
CON2	-2.52660	-48.88210
CL	-2.96445	-47.73679
CL	-3.02065	-47.77956
CL	-3.02562	-47.76585
CL	-3.06172	-47.78866
CL	-2.99559	-47.77047
CL	-3.17981	-47.76374
CL	-2.99702	-47.74525
CL	-2.99861	-47.73275
CL	-3.00639	-47.74682
RIL	-3.75483	-48.58912
RIL	-3.67997	-48.49970
RIL	-3.66999	-48.51567
RIL	-3.59552	-48.54519
RIL	-3.68061	-48.66203
RIL	-3.70351	-48.67416
RIL	-3.71856	-48.70264
RIL	-3.72305	-48.69458
RIL	-3.70842	-48.71799
RIL	-3.60148	-48.74778
RIL	-3.59246	-48.78708
RIL	-3.54574	-48.49817
RIL	-3.53329	-48.81104
RIL	-3.53072	-48.78286
RIL	-3.65393	-48.61003
RIL	-3.66524	-48.59988
RIL	-3.71211	-48.57903
RIL	-3.70929	-48.56111
RIL	-3.66351	-48.56396

RIL	-3.70526	-48.59576
RIL	-3.74078	-48.60257
PPP	-2.54016	-48.79921
PPP	-2.60391	-48.73913
PPP	-2.56344	-48.75813
PPP	-2.57921	-48.81686
PPP	-2.57266	-48.57633
PPP	-2.56473	-48.72593
PPP	-2.47051	-48.61745
PPP	-2.55985	-48.70700
PPP	-2.45600	-48.74700
PPP	-2.31218	-48.66882
PPP	-2.26534	-48.62647
PPP	-2.36977	-48.69745
PPP	-2.28580	-48.64930
PPP	-2.26800	-48.60500
PPP	-2.84419	-48.92125
PPP	-2.84091	-48.95047
PPP	-2.48066	-48.71038
PSP	-1.77936	-48.09880
PSP	-1.78433	-48.19460
PSP	-1.92767	-48.03330
PSP	-1.85902	-48.04210
PSP	-1.96150	-48.13530
LIV	-3.02860	-47.82074
LIV	-3.07300	-47.80801
LIV	-2.88937	-47.91115
LIV	-2.98232	-47.46142
LIV	-2.92045	-47.98565
LIV	-2.92752	-47.91863
LIV	-2.94551	-47.88680
LIV	-3.10710	-47.85231
LIV	-3.03900	-47.87586
LIV	-3.20568	-47.74639

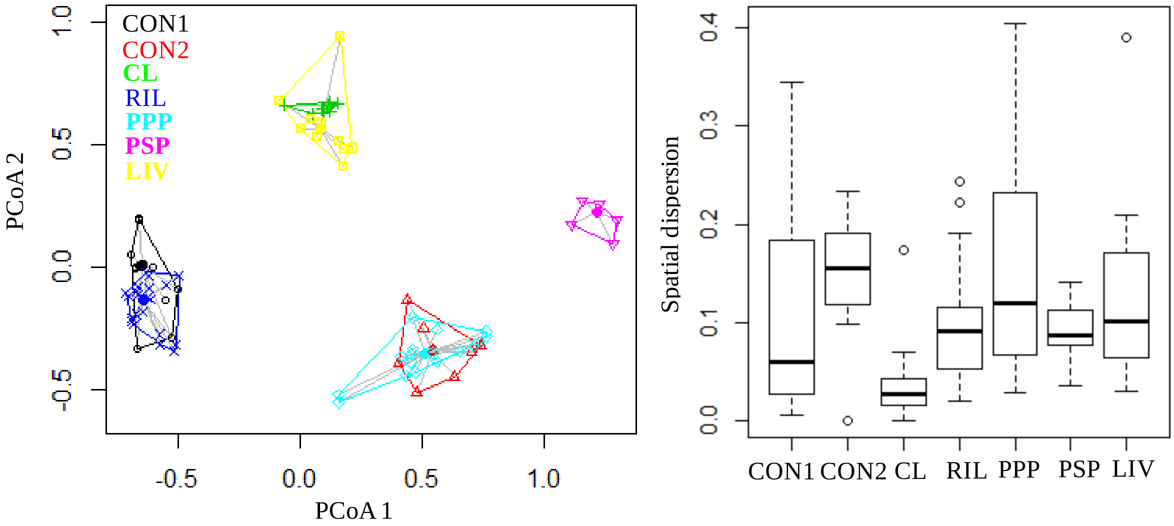

Figure SI1. Principal Coordinate Analysis (PCoA) and Box plots of values of distances to group centroids calculated by PERMIDISP representing the spatial dispersion. Points in PCoA are colour-coded according to categories: control streams (CON1 and CON2), conventional logging (CL), reduced-impact logging (RIL), plantation of primary oil palm (PPP), plantation of secondary oil palm (PSP) and livestock farming (LIV).

Table SI2. Result of correlation between environmental characteristics and PCoA axis. Bold values show a strong correlation with the axis.

Environmental characteristics	**PCoA 1**	**PCoA 2**
Thalweg depth (cm)	**-0.65**	0.37
% of riffle habitats	**-0.65**	0.26
% of organic substrate	0.13	**0.91**
Canopy density at bank	**0.55**	0.10
pH	**-0.52**	-0.12
Dissolved oxygen (mg / L)	**-0.69**	-0.25

Table SI3. The abundance of Ephemeroptera, Plecoptera and Trichoptera (EPT) genera sampled in 83 streams distributed in the control area (CON1 and CON2), conventional logging (CL), reduced-impact logging (RIL), plantation of primary oil palm (PPP), plantation of secondary oil palm (PSP) and livestock farming (LIV).

Taxonomic composition	**Land use**							
	CON1	**CON2**	**CL**	**RIL**	**PPP**	**PSP**	**LIV**	**Mean total abundance**
Ephemeroptera								
Baetidae								
Americabaetis Kluge, 1992		1			2		28	10
Aturbina Lugo-Ortiz & McCafferty, 1996	4	3				2	2	3
Brasilocaenis Puthz, 1975					1		2	2
Callibaetis Eaton, 1881	7		3	4		1	26	8
Callibaetoides Cruz, Salles & Hamada, 2013	6		1					4
Cloeodes Traver, 1938	13	6	3	5	3	2	1	5
Cryptonympha Lugo-Ortiz & McCafferty, 1998					1			1
Harpagobaetis Mol, 1986							2	2
Paracloeodes Day, 1955							3	3
Waltzoyphius McCafferty & Lugo-Ortiz, 1995	2	1	1	7	4	1	1	2
Zelusia Lugo-Ortiz & McCafferty, 1998	38	4	3	18	4	17	23	15
Caenidae								
Brasilocaenis Puthz, 1975	2	1	20	1	3	1	23	7
Caenis Stephens, 1835				4				4
Coryphoridae								
Coryphorus Peters, 1981	2	1		2	2			2
Euthyplociidae								
Campylocia Needham & Murphy, 1924	287	128	319	424	346	41	188	248
Leptohyphidae								
Amanahyphes Salles & Molineri, 2006	5		9	12	3	4	1	6
Hagenulopsis Ulmer, 1920						37	3	20
Hydrosmilodon Flowers & Domínguez, 1992							20	20

Leptohyphes Eaton, 1882							1	1
Macunahyphes Dias, Salles & Molineri, 2005					2			2
Traverhyphes Molineri, 2001		1					3	2
Tricorythopsis Traver, 1958	3	1		4				3
Leptophlebiidae								
Askola Peters, 1969					1		1	1
Farrodes Peters, 1971	64	1	181	68	22	65	3	58
Hagenulopsis Ulmer, 1920	8	1		8	3			5
Hydrosmilodon Flowers & Domínguez, 1992	7			7				7
Microphlebia Savage & Peters, 1983		1			6			3.5
Miroculis Edmunds, 1963	683	322	552	540	557	907	405	567
Simothraulopsis Demoulin, 1966	5		6	3	7	3		5
Ulmeritoides Traver, 1959	24	3	70	61	111	11	188	67
Polymitarcyidae								
Asthenopus Eaton, 1871							1	1
Campsurus Eaton, 1868	25	13	9	17	11		1	13
Plecoptera								
Perlidae								
Anacroneuria Klapálek, 1909	29	32	2	132	23	74		49
Enderleina Jewett, 1960	9	2		8				6
Macrogynoplax Enderlein, 1909	164	114	20	319	88	298	39	149
Trichoptera								
Calamoceratidae								
Phylloicus Muller, 1880	242	94	31	187	102	203	28	127
Ecnomidae								
Austrotinodes Schmid, 1955	3		1	2				2
Glossosomatidae Wallengren, 1891	**105**			90				98
Mortoniella Ulmer, 1906		9			1			5
Helicopsychidae								
Helicopsyche Siebold, 1856	284	31	3	385	17	8		121

Hydropsychidae								
Leptonema Guérin, 1843	41	87	15	102	91	108	55	71
Macronema Pictet, 1836	185	87	112	364	135	71	109	152
Macrostemum Kolenati, 1859	30	51	47	46	90	253	1231	250
Smicridea McLachlan, 1871	23	40	9	27	114	108	144	66
Hydroptilidae Stephens, 1836	**2**							2
Leptoceridae								
Amazonatolica Holzenthal & Pes, 2004		1						1
Gênero A	9			8				9
Nectopsyche Muller, 1879	4	12		10	4		6	7
Oecetis McLachlan, 1877	89	17	10	74	13	3	18	32
Triplectides Kolenati, 1859	105	83	22	186	100	12	13	74
Odontoceridae								
Marilia Müller, 1880	10	18		12	1	1		8
Philopotamidae								
Chimarra Stephens, 1829	40	50	19	49	212	24	34	61
Polycentropodidae								
Cernotina Ross, 1838	35	13	47	25	24	17	247	58
Cyrnellus Banks, 1913	7		21	2		1	10	8
Polycentropus Curtis, 1835	2							2
Polyplectropus Ulmer, 1905	40	2	40	45		88	5	37
Total	**2643**	**1231**	**1576**	**3258**	**2104**	**2361**	**2865**	**2291**

Table SI5. Results of GAM for α-diversity modeled by six environmental variables measured in 83 streams n the Northeastern and Southeastern mesoregion of the state of Pará, Brazil.

	Estimate	**Std. Error**	**F**	**p-value**
Thalweg depth (cm)	-0.02	0.05	0.24	0.63
% of organic substrate	-0.01	0.04	0.07	0.79
Canopy density at bank	0.06	0.05	1.88	0.17
pH	-0.06	0.04	1.83	0.18
Dissolved oxygen (mg / L)	-0.04	0.05	0.70	0.41

3. Sessão II

Multi-scale environmental variables predict the structuring of aquatic insects in Amazonian streams

A segunda sessão desta tese foi elaborada e formatada conforme as normas da publicação científica *Hydrobiologia*, disponível em: https://www.springer.com/life+sciences/ecology/journal/10750?detailsPage=pltci_911058

Multi-scale environmental variables predict the structuring of aquatic insects in Amazonian streams

3.1 Abstract

Understanding diversity patterns of aquatic insects at different spatial scales is important for inferring which process that shapes metacommunity structure. Our goal was to assess the environment and spatial influences of Ephemeroptera, Plecoptera and Trichoptera (EPT) metacommunity structuring and the individual response of EPT taxa to the metacommunity structuring processes on multiple spatial scales. We sampled 97 streams encompassing an area of 22.362km^2 and the measurements of all variables and the community sampling were made in a 150m reach in each stream. We analysed the contribution of environmental, spatial and shared descriptors using variance partitioning procedure (based on partial RDA) and individual response of taxa to the spatial structuring of environment on multiple spatial scales with multivariate multiscale codependence analysis. We found that spatial descriptors explained a higher proportion of the total variation of the community composition and only five genera had codependence relationships between spatial and environment predictors. We suggest that future studies that aim to assess different spatial scales some method that captures the individual response of the taxa.

Keywords: Ephemeroptera, Plecoptera and Trichoptera; spatial codependence; metacommunity

3.2 Introduction

Communities can be structured by environmental and spatial processes or by the interaction between them (Cottenie et al., 2003; Provete et al., 2014; Viana & Chase, 2018). However, the relative importance of these processes depends on the dispersal capacity of organisms (Heino, 2013). The metacommunity concept fits this framework because it recognizes the importance of the multi-scale context to explain local communities and the variation between them (Leibold et al., 2004; Thompson & Townsend, 2006; Brown et al., 2011; Leibold & Chase, 2018) and tries to combine niche and neutral elements into a synthetic view to understanding of community structure over space and time (Leibold et al., 2004; Leibold & Chase, 2018; Vellend, 2010).

The environmental process usually is assessed using measures of the physical and chemical characteristics of the aquatic systems that can influence the distribution of organisms through species sorting mechanisms (Heino & Mykra, 2008; Grönroos et al., 2013; Heino, 2013). In species sorting there is a strong relationship between species and environmental conditions (Leibold et al., 2004; Leibold & Chase, 2018), therefore, selective processes are predominant in local communities (Vellend, 2010). An alternative view suggests that the metacommunity structure differs due to the strong spatial process, which can be related to ecological equivalence and dispersal limitation (Hubbell, 2001). The more intuitive argument is that these two processes occur concomitantly contributing to the structure of metacommunities (e.g. Valente-Neto et al., 2018). Tests of these processes have been conducted in a wide variety of ecosystems and with different organisms (e.g. Thompson & Towsend, 2006; Hájek et al., 2011; Provete et al., 2014; Jamoneau et al., 2018). Most of these studies found that metacommunity was structured by environmental processes (Heino & Mykra 2008; Grönroos et al., 2013), and intermediate results between environmental and spatial process were also recorded (Livingston et al., 2013; Provete et al., 2014; Jamoneau et al., 2018; Viana & Chase, 2018).

The definition of spatial scale of observation is relevant in several studies of metacommunity (Chase, 2014; Leibold & Chase, 2018; Viana & Chase, 2018). The scale is related to the grain or spatial extension of observation (Heino et al., 2015), in which the grain is the individual size of the observation, while the extension is the total area of this observation (Wiens, 1989). Greater spatial extension implicitly involves greater habitat heterogeneity (Viana & Chase, 2018) and niche elements drive the metacommunity (e.g. Chase, 2014). However, if species are not able to access suitable habitats in that extension there can be an increased limitation of dispersion and stochasticity (e.g. Ng et al., 2009). On the other hand, when the grain is larger, the number of individuals sampled increases, while the environmental variables are restricted to this scale of observation (Viana & Chase, 2018). Therefore, if the grain of observation is smaller than the grain of the environmental variation, the importance of the niche decreases because at the fine scale the metacommunity can be structured by stochastic process (e.g., Chase, 2014). We cannot make inferences based only on the spatial scale of the study because different patterns emerge at different scales of investigation (Wiens, 1989) and organisms identify and respond at different scales (McGarigal et al., 2016).

The variation in species response at different spatial scales can be due to differences in the morphological or biomass characteristics of the organisms, such dispersal mode, body size that is expected to be related to the dispersal ability (De Bie et al., 2012). For instance, active dispersers with larger body size are less prone to dispersal limitation, whereas the passive dispersers with larger body size usually present higher dispersal limitation than the ones with smaller bodies (De Bie et al., 2012). Therefore, the dispersal mode of species has a key role in the spatial distribution of species and how they interact with their environment (Cottenie, 2015). The greater dispersal rate of individuals in the metacommunity is more affected by environmental process (Heino, 2013; Grönroos et al., 2013; Heino et al., 2015), but if species are weaker dispersers they can be more affected by the spatial process (Thompson & Towsend, 2006; Hájek et al., 2011).

Biodiversity patterns of freshwater invertebrates may be affected by processes that range sequentially from regional to local (Heino et al., 2003; Heino et al., 2007; Mykrä et al., 2007). For instance, Mykrä et al. (2007) found that spatially structured variables as substratum particle size and water depth were important in structuring the communities at local and regional scale, while conductivity and pH had spatial variation within drainage systems. On the other hand, Provete et al. (2014) found that canopy cover was related to broad scale. Freshwater invertebrates have been considered good models to assess metacommunity processes (Malmqvist, 2002; Bohonak & Jenkins, 2003) because they respond to both local scale and spatial drivers (Johnson et al., 2007; Heino & Mykra 2008; Shimano et al., 2013).

The goal of our study was to assess the environment and spatial influences on Ephemeroptera, Plecoptera and Trichoptera (EPT) metacommunity structuring. We also assessed the individual response of EPT taxa to the metacommunity structuring processes on multiple spatial scales. We studied the relationships between environmental characteristics and EPT distribution using the stream reach as our grain size, inserted in our spatial extension (sampled at two drainage basins in Amazon).

3.3 Methods

3.3.1 Study area

We sampled 97 streams in the Capim and Acará basins located in northeastern and southeastern mesoregion of the State of Pará, Brazil (Figure 1), encompassing an area of 22.362km^2. The coordinates of each stream were measuredusing a GPS and are available in Supporting Information (Table SI1). The database comprised a gradient of ecological conditions from nearnatural sites to those highly degraded by logging, agriculture and livestock farming. The climate of the area is characterized as "Af" according to Köppen's system (Peel et al., 2007), with few variations of temperature (maximum of 25.5°C and a minimum of 20°C) and high annual precipitation and relative humidity (2000-3000 mm and 85%, respectively) (SEMA, 2016). The remaining natural vegetation is classified as Dense Ombrophilous forest (IBGE, 2014) and reliefs have altimetry ranging from 50 - 350 m and declivity between 2 - 5% (Furtado & Ponte, 2014).

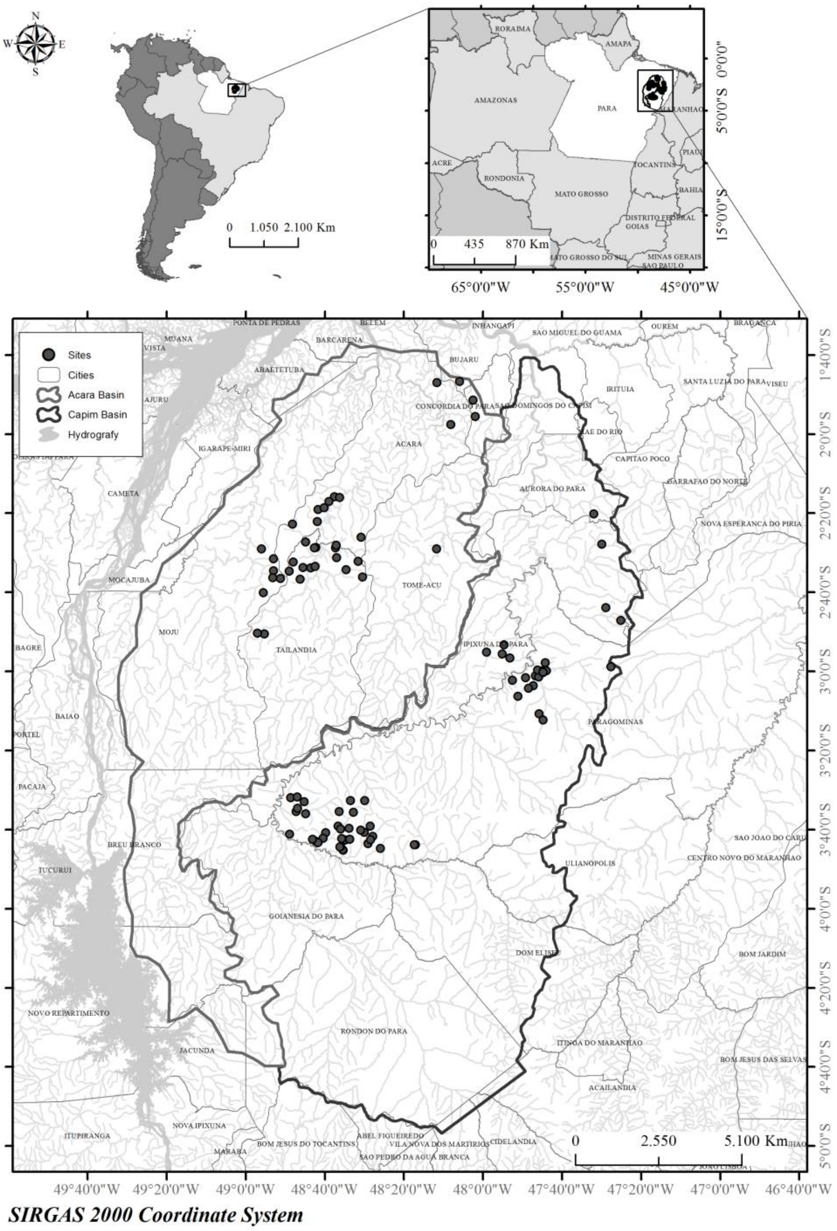

Figure 1. Location of the 97 streams that were used to analysis the spatial and environmental variation.

3.3.2 Environmental descriptors

We sampled nine environmental variables: canopy density at bank, riparian herbaceous cover, cover for aquatic organisms, number of woody large on channel, proportion of glide, proportion of substrate fine, proportion of substrate fine gravel, proportion of substrate of matter organic and dissolved oxygen. We select that set of the variable because previous empirical evidence shows an existing relationship between the stream biota and habitat structure, particularly those related to in-stream habitat heterogeneity (see Heino et al., 2003; Allan & Castillo, 2007 , Mykrä et al., 2007; Padial et al., 2014; Provete et al., 2014; Rocha et al., 2018 for more details). The environmental variables were measured along 150m reach in each stream, following the protocol guidelines of the United States Environmental Protection Agency (US-EPA, Peck et al., 2006) adapted to tropical streams (Macedo et al., 2014).

Canopy density at bank was measured at the wetted channel margin at both sides (left and right) with a spherical densiometer (convex type) positioned 0.30cm above the surface of the water. The mean riparian herbaceous cover is the riparian mid-layer (0.5 to 5m high) herbaceous cover measured through visual estimates both sides (left and right) of the stream. The cover for aquatic organisms corresponds to sum of woody debris (small - <0.3m and big - >0.3m), live trees or roots, leaf litter, overhanging vegetation (<1m above the water surface), undercut margin, boulder and artificial structures. The number of large woody debris on channel is the quantity of wood of length between 5m to 15m and diameter of 0.3m to > 0.8m observed in the channel of the streams. The proportion of glide habitat was measured along the thalweg of the streams. Three substrate types were measured: (i) the proportion of fine substrate (<0.6mm, silt or clay e.g.); (ii) proportion of gravel substrate included the % of cobble (64 to 250mm), % sand (0.06 to 2mm) and % silt; (iii) proportion of organic substrate included the % of roots, % fine litter, % leaf litter, % filamentous algae and % woody substrate. Calculation of all these variables was made following Kaufmann et al. (1999). Finally, the dissolved oxygen (mg / L) was measured into three points along the 150m reach in each stream (downstream, middle and upstream) to calculate a value mean for each stream. The environmental variables were not collinear according to the limit of $r \geq 0.7$ (Figure IS1).

3.3.3 EPT sampling

EPT larvae were sampled only during lower precipitation period in the region (August - December) to avoid seasonal effects on the structure of the community (Bispo et al., 2006). We sampled the EPT community at twenty segments in each stream reach using an 18cm diameter dipnet with 250μm mesh, collecting two substrate samples in each segment. The individuals from the twenty segments of each stream were combined into a single composite sample representing the stream (our sample unit). This methodology has already been used successfully in studies of Amazonian streams (Faria et al., 2017; Paiva et al 2017). The EPT specimens were identified at the genus level using the specialized keys of Pes et al. (2014), Hamada & Silva (2014), Domínguez (2006), Salles & Dominguez (2012) and Salles et al. (2014). The specimens were deposited in the collection of Aquatic Insects of the Laboratory of Ecology and Conservation at the Federal University of Pará, Brazil.

3.3.4 Statistical analyses

We used a principal coordinate of neighbour matrices (PCNM; Bocard & Legendre, 2002) to generate spatial variables as surrogates of the routes used by the communities for dispersal among the sampled streams. In addition, these eigenvectors capture the spatial arrangement of the streams under study at multiple scales (Diniz-Filho & Bini, 2005). We selected an adequate subset of explanatory spatial and standardized environmental variables using a forward selection procedure (Blanchet et al., 2008). The spatial component was represented by seven eigenvectors (PCNM) and the environmental component by five variables (Table SI2). Partial Redundancy Analysis (pRDA, Peres-Neto et al., 2006) with log-transformed abundance and environmental descriptor was applied to assess the role of spatial structure that is independent of the environmental variables (attributed to dispersal) and the environment that is independent of the spatial structures (attributed to environmental component).

We used the Multivariate Multiscale Codependence Analysis (mMCA) to assess the correlation between the variables (community and environmental) in multiple spatial scales and determine the individual response of taxa to the spatial structuring of environment (Guénard et al., 2010; Guénard & Legendre, 2018). The mMCA calculate the codependence coefficient vector that reflects the correlations of the response variable (**x**, EPT composition) and predictors (**y**, environmental variables) in different scales (**w**, PCNM eigenvectors).We evaluated the proportion of the effect of each predictor ($\mathbf{y}_i$) at the codependence on the response variable ($\mathbf{x}_i$) based on coregression coefficients (**β**). Positive coefficients occur when the predictive variable ($\mathbf{y}_i$) and response ($\mathbf{x}_i$) follow a common trend described by $\mathbf{w}_i$, whereas negative coefficients result from opposite trends. The genera with frequency of occurrence less than 5% of the total were excluded from the mMCA analysis, and we use the log transformation to the abundance [ln(x+1)] for all analysis. Thus, a combination of statistical methods allowed us to determine the individual responses of species to environmental characteristic at different spatial scales. Spatial eigenvectors and mMCA were calculated using R package "codep" (Guénard et al., 2018), forward selection procedure with the package "adespatial" (Dray et al., 2018) and "vegan" package (Oksanen et al., 2016) for pRDA.

3.4 Results

3.4.1. Environmental variation

The riparian herbaceous cover had greater variation among streams. Also, there was a high predominance of glide habitat in the sampled streams, with the exception of one stream. The number of large woody and fine substrate had greater variation than gravel substrate and organic substrate. In general, the streams had pretty much variation a high at the dissolved oxygen concentration of water and canopy density at bank among streams (Table 1).

Table 1. Environmental descriptors used to model the partitioning of components that influence the EPT community of Amazonian streams, Pará, Brazil.

Variables	**Minimum**	**Maximun**	**Mean**	**Standard deviation**	**Coefficiente Variation**
Canopy density at bank	5.35	99.7	89.2	15.7	0.18
Riparian herbaceous cover	1.14	60.7	17.1	11.9	0.70
Cover for aquatic organisms	23.0	282	115	43.6	0.38
Number of large woody on channel	0.00	11.3	1.31	1.93	1.47
% of glide	0.00	100	70.2	21.2	0.30
% of fine substrate	0.00	31.4	10.3	9.36	0.91
% of gravel substrate	0.00	73.3	34.5	19.3	0.56
% of organic substrate	0.06	100	51.8	24.1	0.46
Dissolved oxygen	2.87	14.5	6.65	2.32	0.35

3.4.2 Metacommunity structure

We sampled 18792 specimens, with an average of 194 (± 157 standard) specimens at streams. They comprised 54 genera of EPT (Table SI3). Only Ephemeroptera represented 47% of all specimens collected, following by Trichoptera 44% and Plecoptera 8%. Of the 54 genera, *Miroculis* occurred in 91% of the streams, while *Macronema* and *Campylocia* occurred in 84% (Table SI3). Pure spatial descriptors explained a higher proportion of the total variation in the structure of EPT metacommunity. On the other hand, the environment descriptors and spatially structured environmental variables remained lower (Table 2).

Table 2. Result of variance partitioning procedure (based on patial RDA) showing the processes that structure the EPT community in 97 streams distributed in Capim and Acará basin, Pará, Brazil.

	Adjusted R^2	pseudo-F	p
Environment descriptors	0.05	2.07	0.001
Spatially structured environmental	0.06	3.38	0.001
Spatial descriptors	0.12	3.01	0.001
Unexplained variation	0.77	-	-

Codependence relationships between spatial predictors, environment variables and genera of EPT were found for five genera. The relationship of those genera was high at fine scale (up to 93 km) (Table 4). The scale was defined based on the eigenvector retained by MCA, where the first eigenvector show broad-scale variation and subsequent eigenvector show smaller scale variation.

The environmental variables that best explained the abundance of genera considering the spatial structures were large woody channel, organic substrate, dissolved oxygen, fine substrate and canopy density at bank. Canopy density at bank, dissolved oxygen and fine substrate were related to fine scale (up to 93 km) while large woody on channel and organic substrate were related to medium and broad-scales (162 to 325 Km) (Table 3, Figure SI2). The relationship to genera could be positive as well as negative depending on the variable and scale. For example, in fine scale, canopy density at bank positively influenced the abundance of *Amanahyphes* and *Anacroneuria*, but negatively influenced *Askola* and *Americabaetis*. On the other hand, dissolved oxygen positively influenced *Americabaetis* and *Amanahyphes* and negatively influenced *Americabaetis* and *Askola*. The organic substrate negatively influenced *Callibaetis*, *Askola* and *Americabaetis* and positively *Anacroneuria* and *Amanahyphes* in medium and broad scale (Table 4).

Table 3. Result of multivariate multiscale codependence analysis (mMCA) showing the correlation among spatial predictors (eigenvectors PCNM), environmental variables and EPT genera. Results are in accordance with $\alpha \leq 0.05$ criterion.

Variable	Spatial scale (Km)	ϕ_{v_1,v_2}	v_1	v_2	Taxa
Number of large woody on channel	325	65	40	93	Ameri, Aman
% of organic substrate	217	145	40	95	Call, Aman, Ask, Anac
% of organic substrate	162	108	40	94	Ameri, Aman, Ask
Dissolved oxygen	93	28	40	90	Ameri
% of fine substrate	59	25	40	89	Ameri, Aman
Dissolved oxygen	46	71	40	92	Ameri, Aman, Ask
Canopy density at bank	28	52	40	91	Ameri, Aman, Anac
Canopy density at bank	10	37	40	88	Aman, Ask

ϕ_{v_1,v_2} is the Fisher-Snedecor *F* statistic to multivariate codependence coefficient; v_1 is the degree of freedom corresponding to the number of columns (genera) linearly independent; v_2 is the degree of freedom of residual associated with sample units (streams). Taxa codes: Ameri - *Americabaetis*, Aman - *Amanahyphes*, Call - *Callibaetis*, Ask - *Askola*, Anac - *Anacroneuria*.

Table 4. Coregression among spatial predictors (eigenvectors PCNM), environmental variables and EPT genera. ^{1}Coregression coefficients; ^{2}Standardised coregression coefficients.

Taxa	Spatial scale (Km)	Variable	β^1	$\beta_{stan}{}^2$
Amanahyphes	325	Number of large woody on channel	-0.10	-0.50
	217	% of organic substrate	<0.00	-0.01
	162	% of organic substrate	0.01	0.80
	59	% of fine substrate	0.01	0.16
	46	Dissolved oxygen	0.06	0.86
	28	Canopy density at bank	<0.01	0.41
	10	Canopy density at bank	0.01	1.28
Americabaetis	325	Number of large woody on channel	0.06	0.33
	162	% of organic substrate	-0.01	-1.09
	93	Dissolved oxygen	-0.04	-0.69
	59	% of fine substrate	-0.01	-0.23
	46	Dissolved oxygen	0.12	2.11
	28	Canopy density at bank	-0.01	-2.12
Anacroneuria	217	% of organic substrate	0.01	0.60
	28	Canopy density at bank	0.01	0.67
Askola	217	% of organic substrate	<0.00	-0.37
	162	% of organic substrate	0.00	0.27

	46	Dissolved oxygen	-0.01	-0.29
	10	Canopy density at bank	<0.00	-0.39
Callibaetis	217	% of organic substrate	-0.01	-1.50

3.5 Discussion

We found that the spatial descriptor explained 12 % proportion of total community variation, but 6% was explained by the spatially structured environmental. Communities structured by spatial processes had been reported in the literature (Magalhães et al., 2002; Tuomisto et al., 2003; Padial et al., 2014; Rocha et al., 2018) and the possible mechanisms behind these results could be dispersal limitation, biological interactions, neutral processes and historical dynamics (Legendre & Legendre, 2012; Cottenie, 2005; Astorga et al., 2011; Jamoneau et al., 2018).

The studies that relate the spatial processes acting in the community with dispersal limitation of the species (Tuomisto et al., 2003; Hájek et al., 2011; Soininen, 2016; Rocha et al., 2018) suggest that if species are strong dispersers the metacommunity is more affected by the environmental process (Heino, 2013; Grönroos et al., 2013; Heino et al., 2015), but if species are weak dispersers they can be more affected by the spatial process (Thompson & Towsend, 2006). On the other hand, stochastic processes of loss and gain of species can lead the variation at the metacommunity (Hubbell, 2001). However, before assuming that the stochastic process is the structuring factor of the metacommunity, some authors suggest that spatial autocorrelation may be important (Margalhães et al., 2002; Mykrä et al., 2007). The tendency of neighbouring sites to harbour similar communities (Magalhães et al., 2002; Cottenie et al., 2003; Mykrä et al., 2007) is possibly due to the emigration-extinction-recolonization dynamics through dispersion (Pulliam, 2000; Magalhães et al., 2002).

The historical dynamics may reflect natural events, biogeographical factors (Magalhães et al., 2002) or land-use legacy effects (Harding et al., 1998; Munteanu et al., 2015; Martin et al., 2017). Considering the streams sampled in this study are distributed in a single biogeographic unit without big rivers as barriers, we believe that biogeographical factors are the unlikely process in the metacommunity structure (e.g. Haffer, 1969; Morrone, 1994). In addition, large-scale natural events that may have occurred in the extension of the study are unknown. The most likely historical scenario suggests that the specimens were widely distributed in the extent of our study, but were excluded due to disturbances of anthropogenic causes that occurred over the time of colonization of the area (e.g. Mahar, 1979; Smith et al., 1995; Aldrich et al., 2012). Whatever the historical reason, these observations stay as predictive models to be include when knowledge about the distribution of organisms to be enough to allow the inclusion of these models. Therefore, evaluating these mechanisms still depends of information about life history, dispersal and interaction of the organisms, which are incipient in literature.

We believe that the historical processes of land-use in the extension this study may have generated the spatially structured environmental, because the environmental variation may be the result of a land-use legacy that has not been recorded (e.g. Harding et al., 1998; Astorga et al., 2011; Munteanu et al., 2015). Our study area are located at "Arc of Deforestation" in the Amazon where the historical process of land-use (Mahar, 1979; Smith et al., 1995; Aldrich et al., 2012) has left a landscape represented by vegetation fragments due to multiple anthropi impacts (Gardner et al., 2013; Gardner, 2014). Therefore, the landscape was modified to give space for cultures and human occupation (Aldrich et al., 2012) and the environment did not have sufficient time of regeneration before a new exploration, leaving an ecological memory and spatial pattern of species (Peterson, 2002; Fletcher & Fortin, 2018).

Environmental conditions may show variation related to scale (Wiens & Donoghue, 2004; Mykrä et al., 2007). In our study, the same variables at fine scale were important in structuring taxon at a broad scale, but the strongest relationships were observed between medium (~ 217km) and fine (up to 93 km) scales. For example, the variation of organic substrate in medium to broad scale (162 to 325 km) may reflect differences in regional characteristics, probably due to variation in vegetation cover type, soil lithology and pattern of land use and land cover, because variables are context dependent and their importance may change between regions (Mykrä et al., 2007). Besides the strong relationship of dissolved oxygen and canopy density at bank with fine scale suggests that these variables are locally affected probably due to the presence of anthropic access that modifies the marginal vegetation zone of the streams. Even minor changes in this vegetation zone destabilize aquatic habitats (Allan et al., 1997; Faria et al., 2017) and decreases important allochthonous inputs into the ecosystem, as well as change the physical and chemical properties of the ecosystem (Bleich et al., 2014).

The distribution of the organisms may be associated with a specific environment at different scale (Mykrä et al., 2007; Gilbert & Lechowicz, 2004) and can be related to the same environmental variables, even that the response to this variable is opposite (Gauch & Whittaker, 1972a, 1972b) because the effects of environmental variation can be positive at one scale but negative at another (Tamme et al., 2010; Giladi et al., 2011). Our results cope with this premise since the community that captured the fraction of the spatially structured environment was related positive or negative with the organic substrate, dissolved oxygen and canopy density at bank in different scales. For example, *Amanahyphes* and *Anacroneuria* were positively related to canopy density at bank and organic substrate whereas *Americabaetis*, *Askola* and *Callibaetis* were negatively related to this variable.

The organisms' responses to variables can be related to the environmental requirements of each organism. They can be found in roots near marginal vegetation (Salles & Molineri, 2006), from lotic to lentic habitats (Salles et al., 2003; Domínguez et al., 2006), from streams with good quality of canopy cover (Mesa et al., 2014) and even in bank undercuts in streams that have no marginal vegetation (Siegloch et al., 2014). Therefore, the distribution of these organisms may be the result of the interaction between habitat, physical conditions that characterize the habitat and food resource (Wallace & Anderson, 1984) because the organisms have physiological and biological characteristics that reflect how they to environmental conditions (Southwood, 1977; Southwood, 1988; Poff & Ward, 1990; Poff, 1997). Thus, organisms may not be structured spatially in a similar way because of the individual autoecology of each organism (Soininen, 2016), but the scale studied is very important to the diversity patterns of EPT species.

3.6 Conclusion

We found that the spatial descriptor explained the most proportion of EPT variation, but the spatially structured environmental also explained the metacommunity structuring. The *Amanahyphes, Anacroneuria, Americabaetis, Askola* e *Callibaetis* genera responded to the spatially structured environment on multiple spatial scales. Same variables at fine scale were important in structuring genera at a broad scale, but the strongest relationships were observed between medium (~ 217km) and fine (up to 93 km) scales. *Amanahyphes* and *Anacroneuria* were positively related to canopy density at bank and organic substrate whereas *Americabaetis, Askola* and *Callibaetis* were negatively related to this variable. We suggest that future studies that aim to assess different spatial scales regard including a method that captures the individual response of the taxa.

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2.9 Supporting Information

Table SI1. The geographic coordinate of the sampled site collection. Each sample unit (SU) is represented by a stream. Geographic coordinates in decimal degrees.

Sample Unit (SU*)	Coordinates**	
	Latitude (Y)	Longitude (X)
SU1	-3.7313	-48.2847
SU2	-3.7274	-48.4846
SU3	-3.7322	-48.2895
SU4	-3.6531	-48.4747
SU5	-3.6970	-48.4644
SU6	-3.7111	-48.4732
SU7	-3.7468	-48.4316
SU8	-2.6136	-48.7703
SU9	-2.3187	-48.6942
SU10	-2.5777	-48.8826
SU11	-2.5266	-48.8821
SU12	-2.4856	-48.9330
SU13	-2.4787	-48.7049
SU14	-2.3809	-48.8020
SU15	-2.6032	-48.5068
SU16	-2.5225	-48.6173
SU17	-3.5916	-48.6064
SU18	-3.6878	-48.8153
SU19	-3.5511	-48.7534
SU20	-3.5458	-48.5586
SU21	-3.7548	-48.5891
SU22	-3.7527	-48.5911
SU23	-3.6800	-48.4997
SU24	-3.6700	-48.5157
SU25	-3.5955	-48.5452
SU26	-3.6806	-48.6620
SU27	-3.7035	-48.6742
SU28	-3.7186	-48.7026
SU29	-3.7230	-48.6946
SU30	-3.7084	-48.7180
SU31	-3.6015	-48.7478
SU32	-3.5925	-48.7871
SU33	-3.5786	-48.7809
SU34	-3.5457	-48.4982
SU35	-3.5333	-48.8110
SU36	-3.5307	-48.7829
SU37	-3.6539	-48.6100
SU38	-3.6652	-48.5999
SU39	-3.7121	-48.5790

SU40	-3.7094	-48.5615
SU41	-3.7093	-48.5611
SU42	-3.6635	-48.5640
SU43	-3.7053	-48.5958
SU44	-3.7397	-48.5996
SU45	-3.7408	-48.6026
SU46	-2.9645	-47.7368
SU47	-3.0207	-47.7796
SU48	-3.0256	-47.7659
SU49	-3.0617	-47.7887
SU50	-2.9956	-47.7705
SU51	-3.1798	-47.7637
SU52	-2.9970	-47.7453
SU53	-2.9986	-47.7328
SU54	-3.0064	-47.7468
SU55	-3.0047	-47.7460
SU56	-2.3122	-48.6688
SU57	-2.5402	-48.7992
SU58	-2.6072	-48.8862
SU59	-2.5634	-48.7581
SU60	-2.2653	-48.6265
SU61	-2.5792	-48.8169
SU62	-2.5727	-48.5763
SU63	-2.4800	-48.6218
SU64	-2.5647	-48.7259
SU65	-2.6702	-48.9253
SU66	-2.4705	-48.6175
SU67	-2.6100	-48.8526
SU68	-2.5599	-48.7070
SU69	-2.4560	-48.7470
SU70	-2.2680	-48.6050
SU71	-2.2858	-48.6493
SU72	-2.3698	-48.6975
SU73	-2.8442	-48.9213
SU74	-2.8409	-48.9505
SU75	-2.4807	-48.7104
SU76	-1.7794	-48.0988
SU77	-1.7843	-48.1946
SU78	-1.9277	-48.0333
SU79	-1.8590	-48.0421
SU80	-1.9615	-48.1353
SU81	-2.4368	-48.5135
SU82	-2.5374	-48.5261
SU83	-2.4854	-48.1951
SU84	-3.0286	-47.8207
SU85	-3.0730	-47.8080

SU86	-2.8894	-47.9112
SU87	-2.9823	-47.4614
SU88	-2.9205	-47.9857
SU89	-2.9275	-47.9186
SU90	-2.9455	-47.8868
SU91	-3.1071	-47.8523
SU92	-2.3381	-47.5326
SU93	-2.4652	-47.4993
SU94	-3.0390	-47.8759
SU95	-2.7336	-47.4823
SU96	-3.2057	-47.7464
SU97	-2.7867	-47.4185

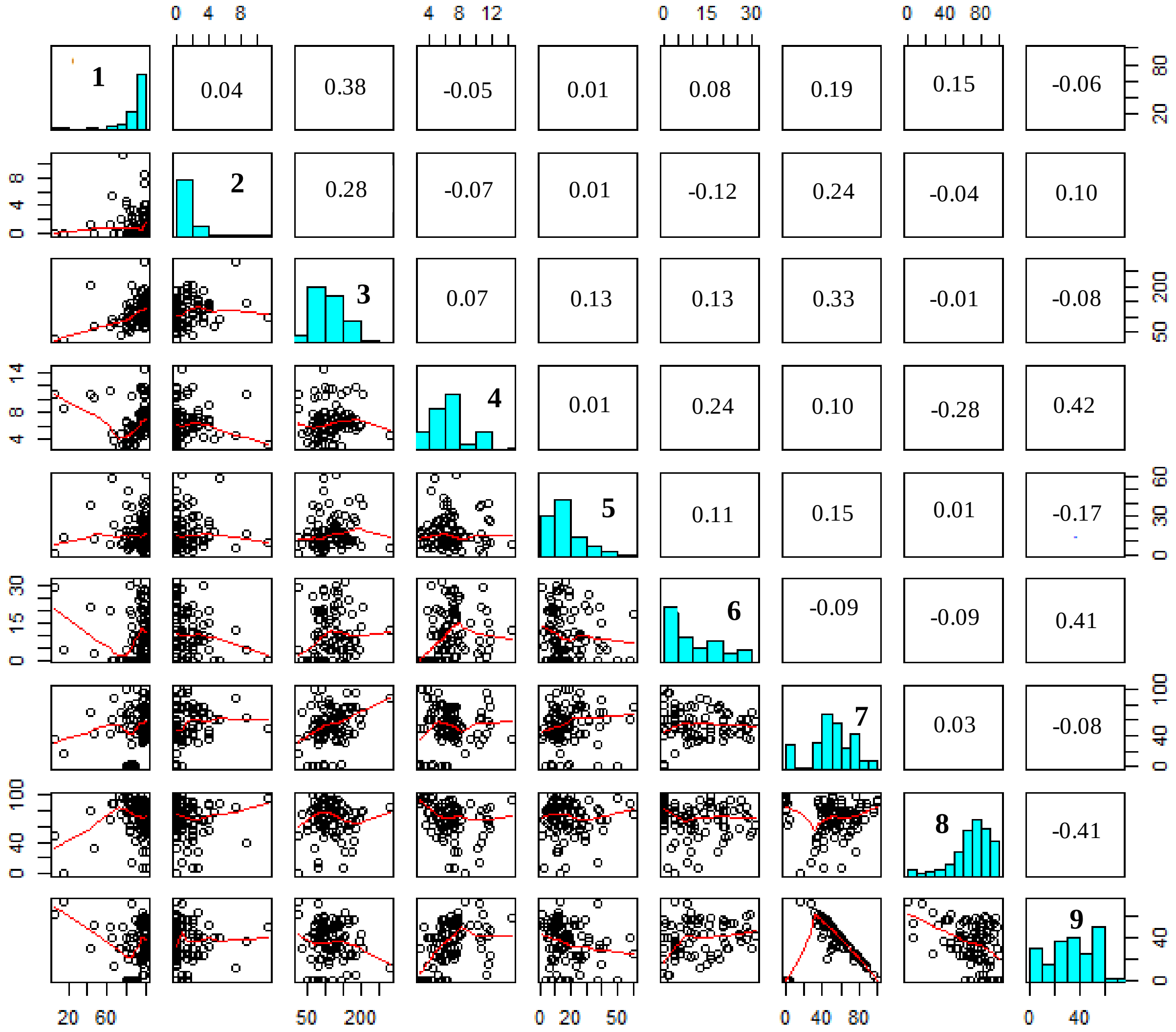


Figure IS1. Correlation among streams sampled. The histogram of each variable on the diagonal makes it possible to visualize the distribution of the variable and scatterplot the form of the relations between variables. The numbers inside histogram correspond to the variables (1) Canopy density at bank, (2) Number of large woody on channel, (3) Cover for aquatic organisms (4) Dissolved oxygen, (5) Riparian herbaceous cover, (6) % of fine substrate, (7) % of organic substrate, (8) % of glide and (9) % of gravel substrate.

Table SI2. Predictors used to describe environmental variables and spatial structures in the partial Redundancy Analysis (pRDA) obtained through multiple linear regression models with forward selection.

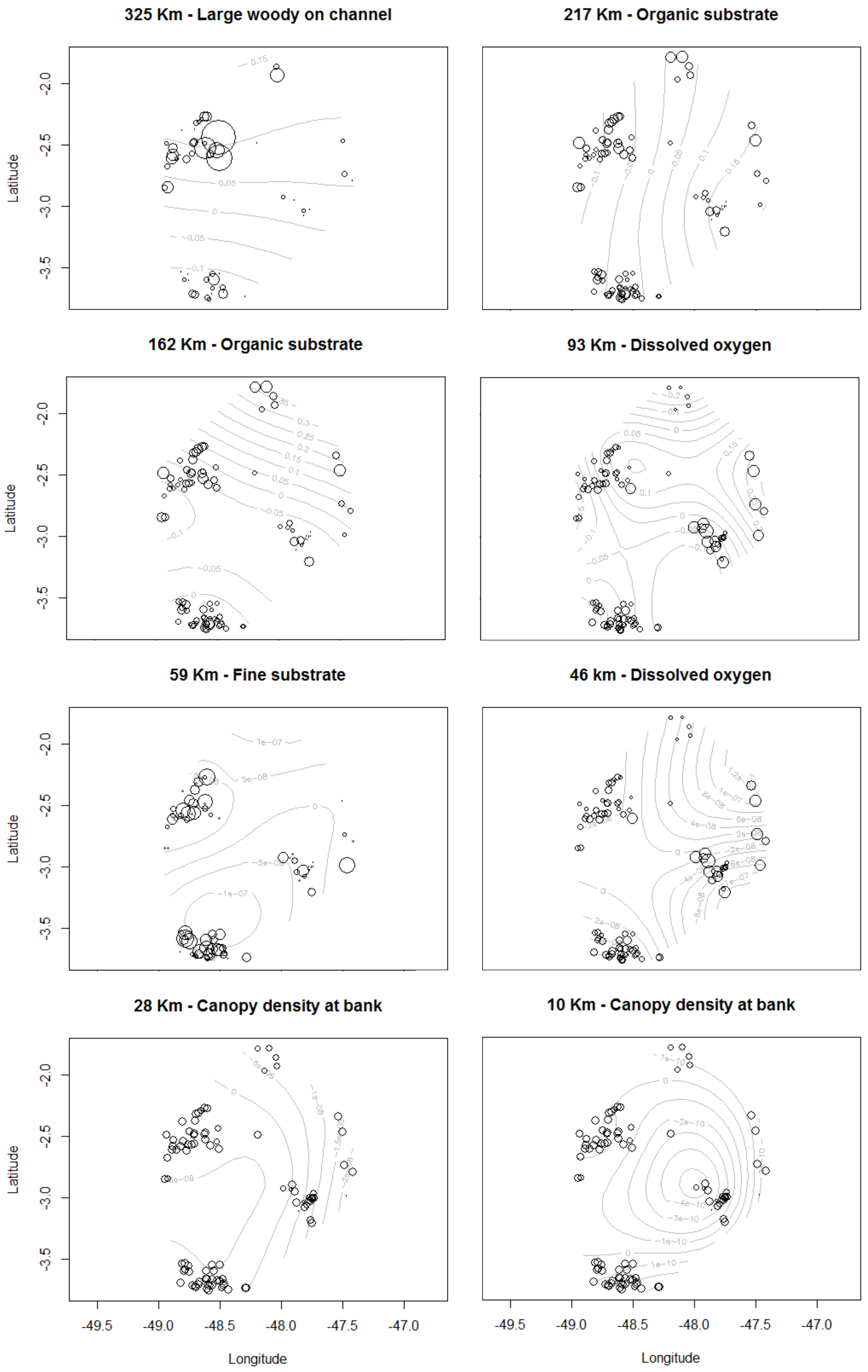
Environmental component				
	Order	Adjusted R2	F	p
Canopy density at bank	1	0.03	3.67	<0.001
Dissolved oxygen	4	0.05	3.36	<0.001
% of gravel substrate	9	0.07	3.06	<0.001
% of organic substrate	7	0.09	3.09	<0.001
Cover for aquatic organisms	3	0.11	2.95	0.01
Spatial component				
PCNM1	1	0.06	7.60	<0.001
PCNM2	2	0.11	5.60	<0.001
PCNM3	3	0.15	5.51	<0.001
PCNM9	9	0.16	1.98	0.04
PCNM5	5	0.17	1.99	0.03
PCNM18	18	0.17	1.93	0.04
PCNM11	11	0.18	1.90	0.04

Table SI3. Abundance of genera of Ephemeroptera, Plecoptera and Trichoptera (EPT) sampled in 97 streams distributed in the Capim and Acará basin located in northeastern and southeastern mesoregion of the State of Pará, Brazil.

Taxa	**Abundance**
Ephemeroptera	
Baetidae	
Americabaetis Kluge, 1992	30
Aturbina Lugo-Ortiz & McCafferty, 1996	12
Brasilocaenis Puthz, 1975	6
Callibaetis Eaton, 1881	41
Callibaetoides Cruz, Salles & Hamada, 2013	8
Cloeodes Traver, 1938	32
Harpagobaetis Mol, 1986	2
Paracloeodes Day, 1955	3
Waltzoyphius McCafferty & Lugo-Ortiz, 1995	18
Zelusia Lugo-Ortiz & McCafferty, 1998	130
Caenidae	
Brasilocaenis Puthz, 1975	56
Caenis Stephens, 1835	4
Coryphoridae	
Coryphorus Peters, 1981	6
Euthyplociidae	
Campylocia Needham & Murphy, 1924	2049
Leptohyphidae	
Amanahyphes Salles & Molineri, 2006	40
Hagenulopsis Ulmer, 1920	44
Hydrosmilodon Flowers & Domínguez, 1992	20

Leptohyphes Eaton, 1882	1
Macunahyphes Dias, Salles & Molineri, 2005	2
Traverhyphes Molineri, 2001	5
Tricorythodes Ulmer, 1920	2
Tricorythopsis Traver, 1958	8
Leptophlebiidae	
Hapsiphlebia Peters & Edmunds, 1972	1
Askola Peters, 1969	6
Farrodes Peters, 1971	428
Hagenulopsis Ulmer, 1920	19
Hydrosmilodon Flowers & Domínguez, 1992	14
Microphlebia Savage & Peters, 1983	8
Miroculis Edmunds, 1963	4959
Simothraulopsis Demoulin, 1966	24
Ulmeritoides Traver, 1959	803
Polymitarcyidae	
Asthenopus Eaton, 1871	1
Campsurus Eaton, 1868	142
Plecoptera	
Perlidae	
Anacroneuria Klapálek, 1909	343
Enderleina Jewett, 1960	28
Macrogynoplax Enderlein, 1909	1159
Trichoptera	
Calamoceratidae	
Phylloicus Muller, 1880	963
Ecnomidae	
Austrotinodes Schmid, 1955	6
Glossosomatidae Wallengren, 1891	130
Mortoniella Ulmer, 1906	11
Helicopsychidae	
Helicopsyche Siebold, 1856	723
Hydropsychidae	
Leptonema Guérin, 1843	565
Macronema Pictet, 1836	1352
Macrostemum Kolenati, 1859	1790
Smicridea McLachlan, 1871	514
Leptoceridae	
Amazonatolica Holzenthal & Pes, 2004	1
Gênero A	20
Nectopsyche Muller, 1879	37
Oecetis McLachlan, 1877	247
Triplectides Kolenati, 1859	554
Odontoceridae	
Marilia Müller, 1880	46
Philopotamidae	

Chimarra Stephens, 1829	556
Polycentropodidae	
Cernotina Ross, 1838	584
Cyrnellus Banks, 1913	49
Polycentropus Curtis, 1835	2
Polyplectropus Ulmer, 1905	228
Total	**18832**



325 Km - Large woody on channel

217 Km - Organic substrate

162 Km - Organic substrate

93 Km - Dissolved oxygen

59 Km - Fine substrate

46 km - Dissolved oxygen

28 Km - Canopy density at bank

10 Km - Canopy density at bank

Figure SI2. Ordination diagrams for the significant PCNM eigenvectors. The size of the symbols is associated with observed values of the environmental variable measured in each stream sampled. Splines show the values for each PCNM eigenvector with contours according to the scale of effect.

4. Sessão III

Limiares críticos de múltiplas comunidades aquáticas em função de gradientes de distúrbio antrópico local e de bacia de drenagem

A terceira sessão desta tese foi elaborada e formatada conforme as normas da publicação científica *Ecological Indicators* disponível em: https://www.elsevier.com/journals/ecological-indicators/1470-160X/guide-for-autho

Limiares críticos de múltiplas comunidades aquáticas em função de gradientes de distúrbio antrópico local e de bacia de drenagem

4.1 Resumo

O avanço no desmatamento na Amazônia devido atividades antrópicas tem sido uma preocupação crescente entre os ecólogos, porque mudanças na paisagem podem levar a perdas significativas da biodiversidade. Diante desse cenário, o nosso objetivo foi determinar o limiar de resposta das comunidades de Odonata, Gerromorpha e Ephemeroptera, Plecoptera e Trichoptera (EPT) em relação a um gradiente de distúrbio antrópico em escala local e em escala de bacia de drenagem. Como a resposta da comunidade emerge das respostas observadas das populações, nós também buscamos identificar os limiares das espécies, cujas abundâncias podem estar negativamente ou positivamente correlacionadas com os distúrbios. Amostramos 66 riachos distribuídos na extensão do arco do desmatamento no nordeste do estado do Pará. Os índices de distúrbio antrópico foram representados por medidas de atividades antrópicas no entorno dos riachos e pela porcentagem de uso de solo na drenagem a montante de cada riacho amostrado. Os limiares das comunidades e das espécies em relação aos gradientes de distúrbio local e de drenagem foram avaliadoss usando a Análise de Limiar de Táxons Indicadores (TITAN). Nossos resultados mostraram que os limiares das comunidades de Odonata, Gerromorpha e EPT tiveram picos de mudanças acentuados e graduais ao longo dos gradientes de distúrbio local e de drenagem. Em todas as comunidades houve espécies sensíveis e tolerantes aos gradientes de distúrbios. Dessa forma, nossos resultados apoiam o cenário de que o distúrbio antrópico, principalmente relacionado aos distintos usos do solo, exerce fortes efeitos sobre as comunidades de insetos aquáticos em riachos de pequeno porte.

Palavras-chaves: riachos; atividades humanas; escalas espaciais; ponto de mudança

4.2 Introdução

Os sistemas hídricos de água doce são estruturados hierarquicamente na paisagem (Frissell et al., 1986) e podem ser influenciados em diferentes escalas por atividades antrópicas, seja nas áreas imediatamente ao redor do riacho (escala local) ou nas áreas a montante correspondentes à drenagem dos mesmos (escala de drenagem) (Allan et al., 1977; Wang et al., 2006). Geralmente os estudos utilizam as características estruturais dos habitats para representar os fatores locais, como o tipo de vegetação ripária, tipos de substratos e morfologia do canal (Johnson e Goedkoop, 2002; Wang et al., 2006; Macedo et al., 2014). Já os fatores associados à drenagem são representados por característica relacionada à forma do canal, declividade, matéria alóctone e uso e ocupação do solo (Johnson e Goedkoop, 2002; Wang et al., 2006; Macedo et al., 2014; Astudillo et al., 2016).

Pela teoria hierárquica de organização de ecossistemas lóticos, os processos que ocorrem em maior escala podem restringir os padrões de meso e microescala (Frissell et al., 1986), enfatizando a conexão entre os ecossistemas aquáticos e suas bacias hidrográficas (Pajunen et al., 2017). Assim, as condições locais do habitat podem ser influenciadas por fatores que operam em escalas maiores. Por exemplo, Pajunen et al. (2017) encontraram que maiores valores de condutividade em riachos podem ser relacionados a impactos antrópicos na bacia hidrográfica. Os efeitos das atividades antrópicas nas comunidades de insetos aquáticos podem ser oriundos dos processos locais (Sponseller et al., 2001; Shimano e Juen, 2016), associados à drenagem (Feld e Henring, 2007), ou ainda pela interação de ambos (Macedo et al., 2014; Tonkin et al., 2016; Firmiano et al., 2017). Por exemplo, alguns estudos mostraram que os insetos aquáticos são afetados quando as características ambientais estruturais do ecossistema são modificadas, tais como, a vegetação ripária, a quantidade de madeira no canal, os tipos de substratos e as variáveis físico-químicas da água (Bertaso et al., 2015; Rodrigues et al., 2016; Vieira et al., 2015; Valente-Neto et al., 2016; Oliveira-Junior et al., 2017).

Uma forma de avaliar os efeitos dos processos descritos acima na distribuição das espécies é por meio do uso de gradientes antrópicos que quantificam as atividades antrópicas em diferentes escalas (Ligeiro et al., 2013; Terra et al., 2013; Carvalho et al., 2017; Chen et al., 2017). Esses gradientes podem variar de pouca ou nenhuma influência antrópica até ambientes severamente alterados (Davies e Jackson, 2006). Os gradientes podem ser representados quantitativamente por diferentes evidências de perturbação, como distúrbio antrópico na paisagem (Cardoso et al., 2013), variáveis físicas estruturais e químicas da água (Shimano e Juen, 2016; Firmiano et al., 2017; Sultana et al., 2019), fragmentação da floresta (Ochoa-Quinteiro et al., 2015; Roque et al., 2018) e perda na vegetação ripária dos riachos (Rodrigues et al., 2016).

As espécies de uma comunidade podem apresentar respostas distintas a esses gradientes de distúrbios (Suriano e Fonseca-Gessner, 2013; Fu et al., 2016), dependendo da sua amplitude de exigências ecofisiológicas ou dos limites de tolerância a degradação ambiental (Southwood, 1977; Poff e Ward, 1990; Baker e King, 2010; Estavillo et al., 2013; Sundermann et al., 2015). A comunidade pode aumentar ou diminuir drasticamente em algum ponto ao longo de um gradiente de distúrbio (Baker e King, 2010), podendo causar mudanças significativas na composição quando os limites de tolerâncias são atingidos (Roque et al., 2018). Uma vez atingido os limiares das espécies, a recuperação das comunidades passa a ser um processo lento e difícil (Lyytimäki e Hildén, 2007).

A especificidade por habitat das comunidades pode ser preponderante para predizer qual escala espacial a comunidade pode ser mais afetada. Por exemplo, Odonata adultos são particularmente sensíveis às condições na faixa de vegetação próxima dos riachos (Oliveira-Junior et al., 2015; Rodrigues et al., 2016), porque se dispersam pela vegetação adjacente aos sistemas aquáticos à procura de presas, parceiros ou poleiros (Corbet, 1999). Já as espécies de Gerromorpha são mais sensíveis às condições da qualidade da água e cobertura de dossel (Dias-Silva et al. 2010; Cunha et al., 2015), pois são exclusivamente aquáticos e vivem sobre a superfície da água ou nas margens dos riachos (Andersen, 1982; Yang et al., 2004). Enquanto a comunidade de Ephemeroptera, Plecoptera e Trichoptera (EPT) é mais sensível às condições dos substratos onde eles vivem (Wiggins, 1996; Domínguez et al., 2009; Froehlich, 2009). Portanto, essas comunidades são adequadas para avaliar os efeitos de distúrbio antrópico na paisagem e no riacho, porque elas apresentam especificidades ambientais diferentes, em que algumas espécies requerem habitats mais íntegros e outras conseguem persistir em ecossistemas mais alterados (Abhijna et al., 2013; Cunha et al., 2015; Miguel et al., 2017).

Nesse contexto, nosso objetivo foi determinar os limiares de resposta das comunidades de Odonata, Gerromorpha e EPT em gradientes de distúrbio antrópico na escala local e de bacia de drenagem. Como a resposta da comunidade emerge das respostas observadas das populações individualmente (Haddad et al., 2015), nós também buscamos identificar os limiares das espécies, cujas frequências podem estar negativamente ou positivamente correlacionadas com os distúrbios.

4.3 Métodos

4.3.1 Área de estudo

O estudo foi realizado em 66 riachos (nossas unidades amostrais) de primeira a terceira ordem (2,69 m de largura média e 28 cm de profundidade média) abrangendo uma região de 22.362 km^2 (Figura 1). Os riachos estão localizados dentro da extensão do “arco do desmatamento”, que corresponde a uma faixa contínua de pressão antrópica que avança sobre os remanescentes florestais na Amazônia. Esses têm sido convertidos em áreas de pastagens e diversos cultivos (Rodrigues et al., 2003; Gardner et al., 2013), resultando em uma paisagem fragmentada com drenagens sujeitas a múltiplos impactos de atividades antrópicas. O clima predominante na área de estudo é do subtipo *"Af"* (tropical úmido) (Peel et al., 2007) com precipitação média anual de 2.000 - 3.000 mm, sendo a estação mais chuvosa entre janeiro e maio. A umidade relativa se manteve ao longo do ano em torno de 85% (Sema, 2016). A vegetação característica da área é Floresta Ombrófila Densa representada pelo grupo de formação das terras baixas e submontana (IBGE, 2014).

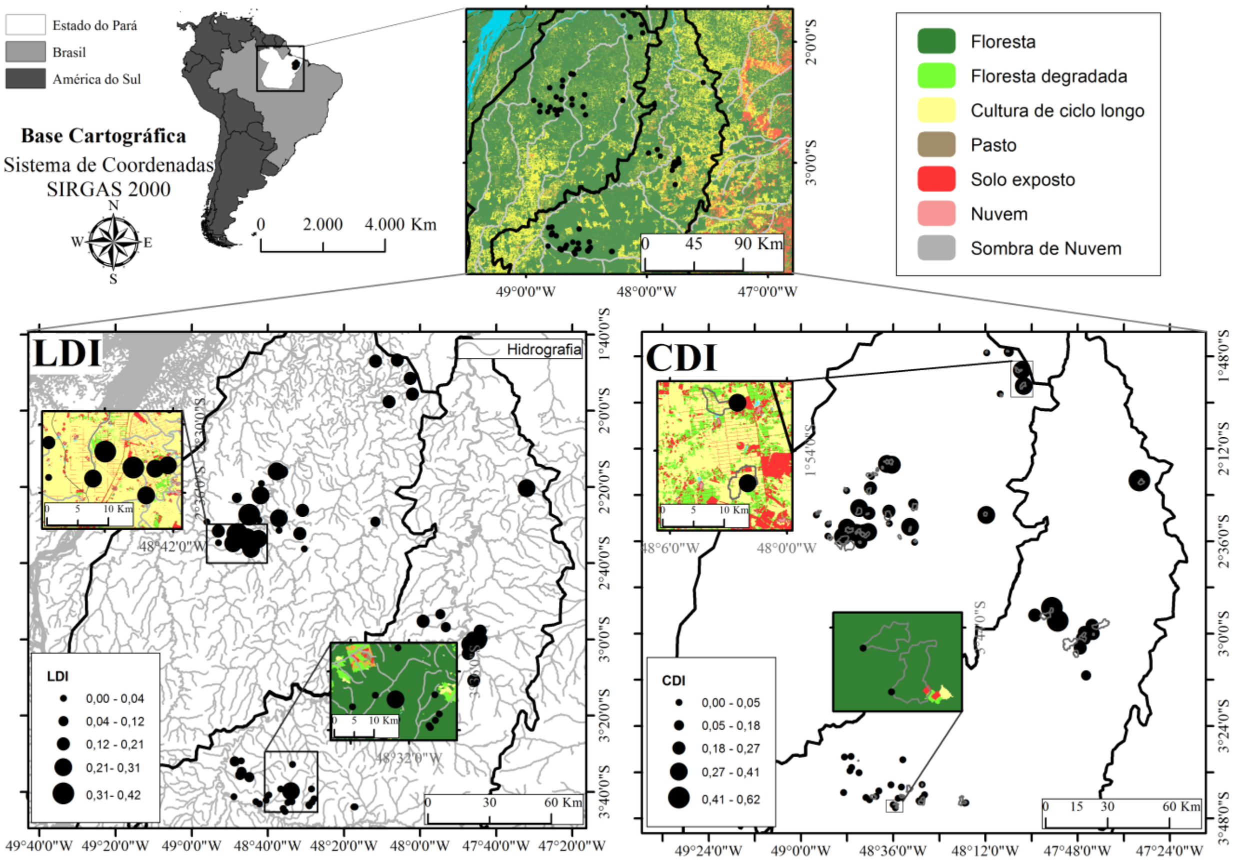


Figura 1. Localização geográfica dos 66 riachos onde foram realizadas as amostragens biológicas e caracterização dos gradientes de distúrbio antrópico (LDI e CDI). As coordenadas geográficas estão disponíveis na Tabela IS1 em Informação suplementar.

4.3.2 Gradiente de distúrbio antrópico

O índice de distúrbio local (*Local disturbance index* - LDI) e o índice de distúrbio na área de drenagem (*Catchment disturbance index* - CDI) foram usados para representar os gradientes de distúrbio antrópico nesse estudo. Esses índices foram calculados de acordo com método proposto por Ligeiro et al. (2013) que descrevem, respectivamente, a intensidade de distúrbio dentro do riacho (local) e na área de drenagem a montante de cada riacho.

O LDI foi representado pelo índice de distúrbio humano (*W1*_Hall), descrito por Kaufmann et al. (1999). Esse índice combina a observação visual de 11 tipos de distúrbios antrópicos (presença de construções, revestimento no canal, pavimentação, estradas, efluentes, lixo, gramados, agricultura, pastagem, extração de madeira e mineração) em três categorias de distância em relação ao canal: B - alterações dentro do canal do riacho ou nas margens; C - alterações em até 10m da margem do riacho, e P - alterações além de 10m da margem. Os valores atribuídos para cada distúrbio foram calculados de acordo com a proximidade do impacto, ou seja, peso de 1.5 quando o distúrbio estava presente no canal (B), 1.0 quando estava dentro da faixa ripária de 10m (C) e 0.667 quando a distúrbio estava fora da faixa de 10m (P). Assim, o *W1*_Hall pondera as observações de cada distúrbio com base na sua proximidade em relação ao canal do riacho (Kaufmann et al., 1999).

O CDI foi caracterizado pelo percentual do uso do solo quantificado na área de drenagem a montante de cada riacho amostrado. A área de drenagem foi delineada usando modelo de elevação digital (*Shuttle Radar Topograph Mission* - SRTM) de 30m de resolução (USGS, 2018). A definição do tipo de uso do solo foi feita com base no mapeamento do conjunto de satélites REIS (*RapidEye Earth Imaging System,* MMA, 2017), obtida gratuitamente para fins de pesquisa, através de um sistema de informação geográfica (SIG). Cada cena RapidEye cobre uma área de 77 km de largura e 1.500 km de comprimento, com resolução espacial de 5m e com cinco faixas no visível (RapidEye, 2015). Nós calculamos os percentuais de uso do solo a partir das cenas do REIS, considerando as classes de cobertura relevantes nessa paisagem (Tabela IS2). Foram classificados quatro tipos de usos do solo: solo exposto; pastagem; cultura de ciclo longo; e vegetação degradada. Como cada área de drenagem frequentemente era influenciada por múltiplos usos, nós atribuímos pesos para cada uso de acordo com o potencial de impacto, modificado de Ligeiro et al. (2013) e Rawer-Jost et al. (2004). Os pesos atribuídos foram classificados de acordo com a intensidade de alteração de cada uso: CDI = 4 x % solo exposto + 2 x % pastagem + 1 x % cultura de ciclo longo e floresta degradada. Para os índices ficarem na mesma escala numérica e permitir comparações visuais dos resultados foi feita a padronização do LDI e CDI. A padronização de cada índice foi feita dividindo o valor observado do índice (LDI e CDI) pelo valor equivalente a 75% (LDI = 5 e CDI= 300) do valor máximo que cada índice teoricamente poderia atingir (LDI=7 e CDI= 400) (Ligeiro et al., 2013) (Tabela IS3). Os gradientes (LDI e CDI) não foram colineares de acordo com o limite de $r \geq 0.7$ (Figura IS1a).

3.3.3 Amostragem Biológica

As amostragens foram realizadas no período de estiagem (agosto - dezembro) da região para evitar efeitos sazonais na estrutura das comunidades de insetos aquáticos (Bispo et al., 2006). As comunidades foram coletadas em 20 segmentos distribuídos sistematicamente ao longo de um trecho longitudinal de 150m em cada riacho (maiores detalhes em Juen et al., 2016). Os adultos de Odonata foram coletados nas margens dos riachos durante uma hora, entre 11h e 14h, sempre sob temperaturas acima de 19°C, usando uma rede entomológica de 40 cm de diâmetro e 65 cm de comprimento (e.g. Calvão et al., 2016). O tempo de coleta definido teve o objetivo de padronizar o esforço amostral, e a definição da condição climática deveu-se à dependência desses organismos à incidência solar (Corbet, 1999). Os espécimes da infraordem Gerromorpha foram coletados na superfície da água utilizando uma rede de mão (18 cm de diâmetro, 1 mm de malha) com tempo de varredura de três minutos em cada segmento do riacho (Cunha e Juen, 2017). Já os imaturos de

EPTs foram amostrados a partir de duas réplicas de substrato (e.g. raiz, folhas e detritos de madeira) coletado em cada segmento usando um coletor do tipo *rapiché* (peneira circular de 18 cm e malha de 0,05 mm). As identificações dos espécimes de cada comunidade foram feitas de acordo com chaves taxonômicas especializadas (Lencioni, 2005; Domínguez et al., 2006; Lencioni, 2006; Garrison et al., 2006; Garrison et al., 2010; Moreira et al., 2011; Salles e Domínguez 2012; von-Ellenrieder, 2013a; von-Ellenrieder, 2013b; Pessacq, 2014; Rodrigues et al., 2014; Hamada e Silva 2014; Pes et al., 2014; Salles et al., 2014; Floriano e Moreira 2015). O material biológico testemunho está depositado na Coleção de Insetos Aquáticos do Laboratório de Ecologia e Conservação, na Universidade Federal do Pará (UFPA), Campus Universitário de Belém, Pará, Brasil.

4.3.4 Identificando os limiares das espécies

Os limiares das espécies e de cada comunidade (Odonata, Gerromorpha e EPT) ao longo dos gradientes de distúrbio antrópico (LDI e CDI) foram detectados através da Análise de Limiar de Táxons Indicadores (*Threshold Indicator Taxa Analysis*- TITAN, Baker e King, 2010), usando os pacotes 'TITAN2' (Baker et al., 2015) e 'vegan' (Oksanen et al., 2016) na versão 3.3.2 do programa R (R Core Team, 2016). Essa análise utiliza as pontuações do valor de Espécies Indicadoras (IndVal) para identificar os potenciais pontos de mudança das espécies, comparando as diferenças na abundância relativa nos dois grupos do gradiente contínuo de distúrbio (e.g. menor intensidade de impacto *vs* maior intensidade de impacto). A magnitude relativa dos escores do IndVal em cada lado da divisão reflete os *z scores* positivos e negativos (z^+ e z^-). Respostas positivas (z^+) indicam que as espécies aumentam em frequência de ocorrência e abundância a partir do ponto de mudança (ou seja, são espécies tolerantes), enquanto que respostas negativas (z^-) indicam que as espécies diminuem em frequência de ocorrência e abundância no ponto de mudança (ou seja, são espécies sensíveis). Iterações realizadas pelo método de *bootstrap* (n=500) foram aplicadas para identificar as espécies com limiares confiáveis (confiabilidade) e a proporção de réplicas de *bootstrap* com o mesmo sinal de resposta do ponto de mudança (pureza). Assim, os valores de pureza indicam a proporção média da direção da resposta do táxon (z^+ ou z^-) e o valor de confiança representa os intervalos de confiança dos pontos de mudança do espécime. Portanto, uma espécie pode ser considerada significativamente associada a um gradiente de distúrbio antrópico se o IndVal for $p < 0,05$, pureza > 0,95 e confiabilidade > 0,95.

4.3.5 Identificando os limiares das comunidades

O TITAN também identifica o limiar da comunidade em relação ao gradiente de distúrbio. O limiar da comunidade foi obtido somando-se a mudança normalizada de cada táxon z^+ ou z^-

separadamente. O cálculo de um limiar para a comunidade é baseada na resposta sincrônica de todas as espécies indicadoras em que as pontuações (soma z^+ ou soma z^-) mostram um pico (Baker e King 2010; King e Baker, 2014). As espécies com baixa ocorrência (< 5) nas unidades amostrais e baixa abundância (< 5) foram removidas da análise, pois espécies com baixa ocorrência não estimam os valores de *z* de forma confiável (Baker e King 2010).

4.4 Resultados

4.4.1 Condição ambiental dos riachos

O índice de distúrbio local (LDI) variou de 0 a 0.42 e o índice de bacia de drenagem (CDI) variou de 0 a 0.62 (valores padronizados). Os valores próximos de zero representam ambientes que não são influenciados por atividades antrópicas na escala avaliada, ao passo que o aumento no valor do índice significa maior intensidade de atividades antrópicas.

4.4.2 Limiar das comunidades e das espécies

Nós coletamos 23.832 espécimes de Odonata, Gerromorpha e EPT nos 66 riachos amostrados. Odonata foi representado por 1.683 espécimes distribuídos em seis famílias, 16 gêneros e 23 espécies. Gerromorpha por 8.224 espécimes, quatro famílias, 12 gêneros e 28 espécies. EPT por 13.925 espécimes distribuídos em 15 famílias e 32 gêneros (Tabela IS4).

A avaliação dos limiares ao nível da comunidade para Odonata, Gerromorpha e EPT no gradiente de distúrbio local (LDI) e de bacia drenagem (CDI) é mostrada na Tabela IS5 e Figura 2. A resposta z+ da comunidade de Odonata em relação ao LDI é gradual, de forma que a soma (z) oscila no gradiente até 0.21 e depois diminui acentuadamente. A soma positiva (z+) para o CDI apresenta pico acentuado em 0.09. Já para a soma negativa (z-) houve um pico acentuado no gradiente CDI (0.08) e um pico gradual em LDI (0.13). A comunidade de Gerromorpha tem pico gradual na resposta z+ em ambos os gradientes (LDI com primeiro pico em 0.12 e CDI em 0.06). Já a resposta negativa (z-) é acentuada nesses gradientes (pico em 0.09 no LDI e 0.06 no CDI). A resposta negativa (z-) da comunidade de EPT para os gradientes de distúrbio (LDI e CDI) foi bastante acentuada, com pico de mudança em 0.08. Por outro lado, a resposta positiva é menos acentuada, com pico em 0.13 para LDI e 0.09 para CDI.

Em todas as comunidades, após o primeiro pico de mudança as frequências diminuem, mas ocorrem picos secundários ao longo do gradiente, o que representa outros grupos de espécies que estão mudando de forma sincrônica. Os picos acentuados e graduais das comunidades sugerem como as espécies sensíveis e tolerantes estão mudando no gradiente. Ou seja, a comunidade de Odonata teve uma faixa de mudança no gradiente local, mas os distúrbios na drenagem provocaram uma mudança acentuada e sincrônica na comunidade. Para Gerromorpha e EPT, as espécies sensíveis têm pontos de mudanças em ambos os gradientes de distúrbios, ao passo que as espécies tolerantes apresentam uma faixa de mudança nos gradientes.

Das 83 espécies avaliadas, foram identificadas 32 espécies com pontos de mudança no gradiente de distúrbio local (LDI) e gradiente na drenagem (CDI). Destas, 18 foram associadas aos dois gradientes (LDI e CDI), enquanto cinco foram associados exclusivamente ao gradiente local (LDI) e nove exclusivamente ao gradiente na drenagem (CDI). As espécies que foram associadas aos dois gradientes apresentaram as indicações dos *z-scores* (z+ ou -) similares, porém, o ponto de mudança ao longo dos gradientes LDI e CDI foram diferentes (Figura 3, Tabela IS6).

Para Odonata foram identificadas cinco espécies com limiares de LDI entre 0 a 0.27, das quais duas espécies tiveram indicadores negativos (z-) e três tiveram indicadores positivos (z+). Oito espécies apresentaram limiares entre 0 a 0.50 no CDI, das quais três com indicadores negativos e cinco positivos. *Chalcopteryx radians*, *Oligoclada walkeri* e *Oxystigma petiolatum* foram as espécies que apresentaram limiar somente no CDI. Para Gerromorpha foram 10 espécies com limiares de LDI entre 0.08 a 0.36, das quais cinco com indicadores negativos e cinco positivos. Em relação ao CDI, 12 espécies tiveram limiar entre 0 a 0.50, das quais seis com indicadores negativos e seis positivos. *Euvelia discala* e *Stridulivelia alia* foram as espécies com limiar somente em LDI e *Brachymetra albinervis albinervis*, *Microvelia mimula*, *Tachygerris celocis* e *Hydrometra guianana* apenas no CDI. Para EPT foram oito gêneros com limiares de LDI entre 0 a 0.24, dos quais seis com indicadores negativos e dois com indicadores positivos. Sete gêneros apresentaram limiares entre 0 a 0.50 no CDI, dos quais seis com indicadores negativos e um positivo. *Zelusia*, *Chimarra* e *Brasilocaenis* foram os gêneros que apresentaram limiares somente no LDI e *Phylloicus* e *Polyplectropus* apenas no CDI (Figura 3).

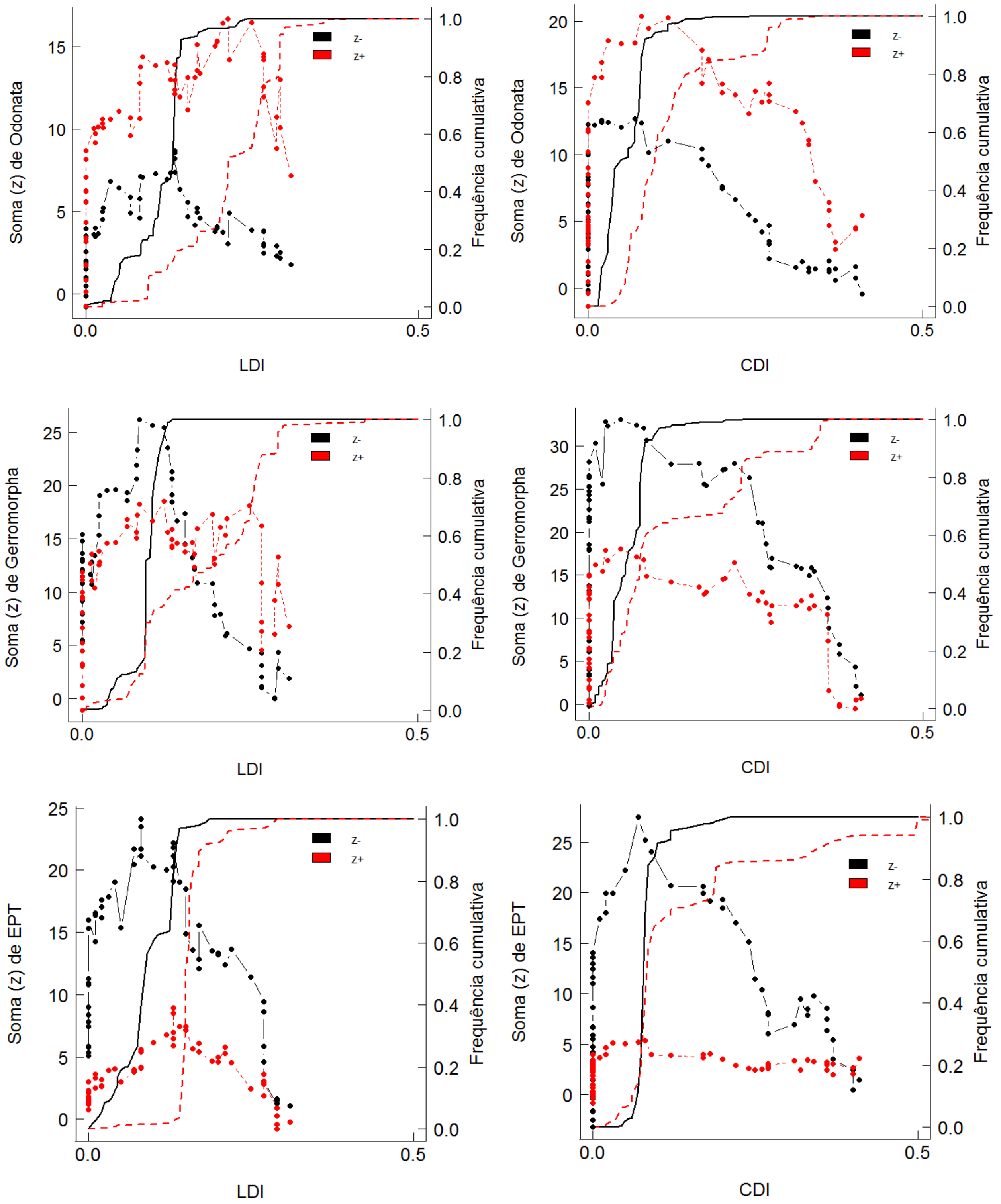
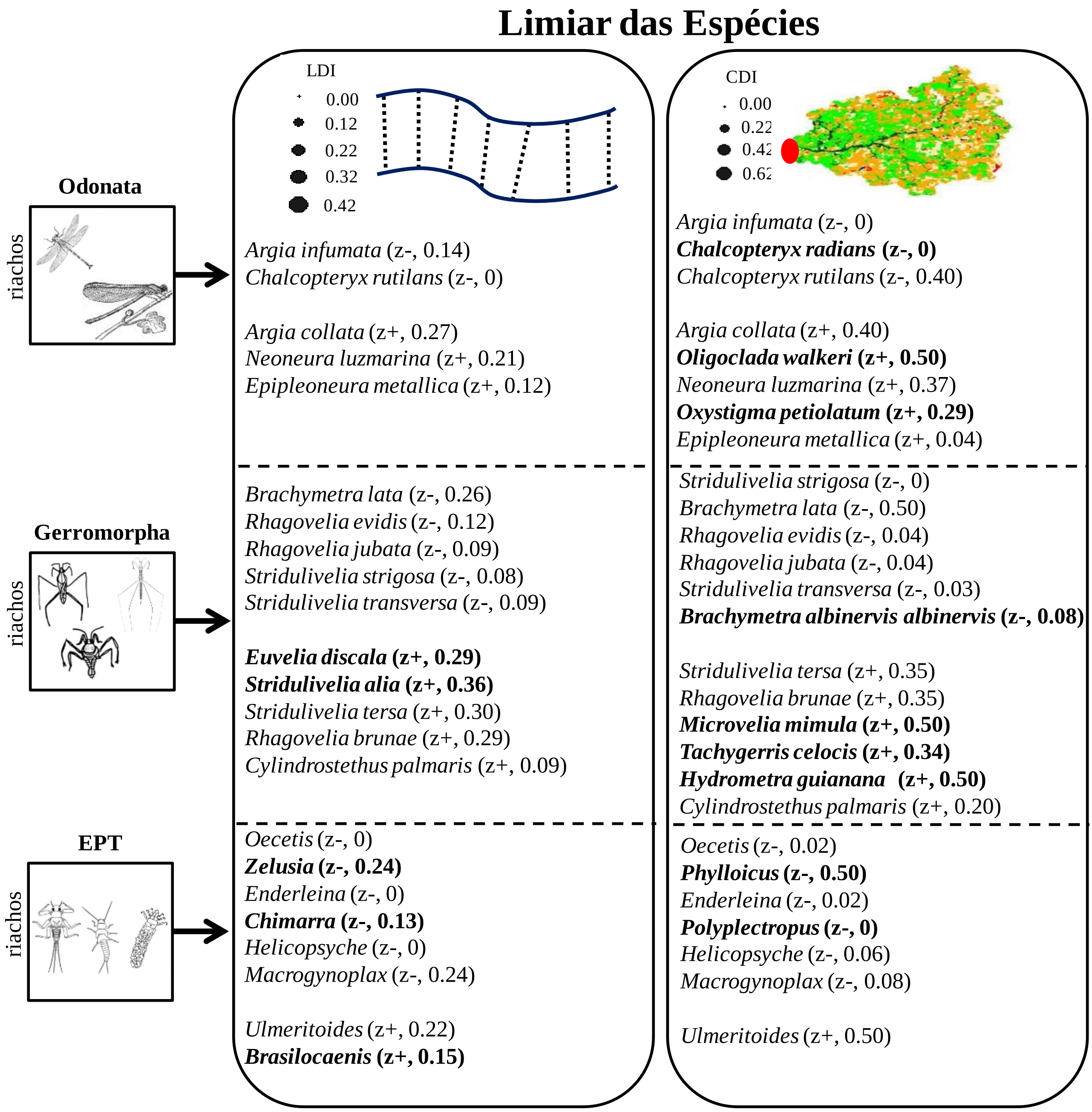


Figura 2. Valores de soma negativa (z-, pontos pretos) e positiva (z+, pontos vermelhos) extraído da análise TITAN para os pontos de mudança da comunidade de Odonata, Gerromorpha e EPT em resposta ao gradiente de distúrbio local (LDI) e de bacia de drenagem (CDI). Picos em soma z correspondem à localização no gradiente onde ocorrem declínios sincrônicos de espécies. As linhas contínuas e tracejadas representam a distribuição de frequência cumulativa de pontos de mudança entre 500 iterações de *bootstrap* para declínio (z-) e aumento (z+), respectivamente.



	LDI	CDI
Odonata	*Argia infumata* (z-, 0.14) *Chalcopteryx rutilans* (z-, 0) *Argia collata* (z+, 0.27) *Neoneura luzmarina* (z+, 0.21) *Epipleoneura metallica* (z+, 0.12)	*Argia infumata* (z-, 0) ***Chalcopteryx radians* (z-, 0)** *Chalcopteryx rutilans* (z-, 0.40) *Argia collata* (z+, 0.40) ***Oligoclada walkeri* (z+, 0.50)** *Neoneura luzmarina* (z+, 0.37) ***Oxystigma petiolatum* (z+, 0.29)** *Epipleoneura metallica* (z+, 0.04)
Gerromorpha	*Brachymetra lata* (z-, 0.26) *Rhagovelia evidis* (z-, 0.12) *Rhagovelia jubata* (z-, 0.09) *Stridulivelia strigosa* (z-, 0.08) *Stridulivelia transversa* (z-, 0.09) ***Euvelia discala* (z+, 0.29)** ***Stridulivelia alia* (z+, 0.36)** *Stridulivelia tersa* (z+, 0.30) *Rhagovelia brunae* (z+, 0.29) *Cylindrostethus palmaris* (z+, 0.09)	*Stridulivelia strigosa* (z-, 0) *Brachymetra lata* (z-, 0.50) *Rhagovelia evidis* (z-, 0.04) *Rhagovelia jubata* (z-, 0.04) *Stridulivelia transversa* (z-, 0.03) ***Brachymetra albinervis albinervis* (z-, 0.08)** *Stridulivelia tersa* (z+, 0.35) *Rhagovelia brunae* (z+, 0.35) ***Microvelia mimula* (z+, 0.50)** ***Tachygerris celocis* (z+, 0.34)** ***Hydrometra guianana* (z+, 0.50)** *Cylindrostethus palmaris* (z+, 0.20)
EPT	*Oecetis* (z-, 0) ***Zelusia* (z-, 0.24)** *Enderleina* (z-, 0) ***Chimarra* (z-, 0.13)** *Helicopsyche* (z-, 0) *Macrogynoplax* (z-, 0.24) *Ulmeritoides* (z+, 0.22) ***Brasilocaenis* (z+, 0.15)**	*Oecetis* (z-, 0.02) ***Phylloicus* (z-, 0.50)** *Enderleina* (z-, 0.02) ***Polyplectropus* (z-, 0)** *Helicopsyche* (z-, 0.06) *Macrogynoplax* (z-, 0.08) *Ulmeritoides* (z+, 0.50)

Figura 3. Quadro mostrando os limiares das espécies de Odonata, Gerromorpha e EPT em resposta aos gradientes de distúrbio local (*Local disturbance index* - LDI) e de bacia de drenagem (*Catchment disturbance index* - CDI). O limiar positivo (z+) ou negativo (z-) representa a direção de resposta da espécie ao gradiente, enquanto o valor apresentado entre parênteses corresponde ao ponto de mudança observado de cada espécie para cada gradiente analisado. As espécies em negrito são as que apresentam limiares apenas em um dos gradientes. Todas as espécies mostradas nesse quadro preencheram os critérios de p (≤ 0,05), pureza (≥ 0,95) e confiabilidade (≥ 0,95) (Informação suplementar Tabela IS6).

4.5 Discussão

Nossos resultados mostraram que os limiares das comunidades tiveram picos de mudanças acentuados e graduais ao longo dos gradientes de distúrbio local (LDI) e de drenagem (CDI). A comunidade de Odonata apresentou variação gradual no distúrbio local, mas os distúrbios na drenagem provocaram um pico de mudança acentuada e sincrônica na comunidade. No limiar comunitário de Gerromorpha e EPT, as espécies sensíveis apresentaram ponto de mudança nos dois gradientes de distúrbios, ao passo que as espécies tolerantes apresentam mudança gradual no gradiente. Nas comunidades avaliadas houve espécies sensíveis e tolerantes aos gradientes de distúrbios. Ao avaliar a resposta das espécies de cada comunidade, Odonata teve mais espécies associadas ao distúrbio em escala de drenagem do que ao distúrbio local, as quais foram representadas em sua maioria por espécies tolerantes. Já a comunidade de Gerromorpha teve mais espécies associadas aos gradientes de distúrbio local e de drenagem, representados tanto por espécies sensíveis como tolerantes. Por outro lado, EPT apresentou mais gêneros sensíveis aos gradientes de distúrbio local e de drenagem, mas dois gêneros aumentaram de frequência no distúrbio local e apenas um no distúrbio de drenagem.

A mudança gradual no limiar da comunidade de Odonata observado no distúrbio local pode ser devido à zona de vegetação ribeirinha disponibilizar habitat para as espécies (Sponseller et al., 2001). Esses habitats são caracterizados principalmente pela presença de cobertura de dossel, que contribui para reduzir a incidência solar e da temperatura do ambiente, e disponibilizam sítios de oviposição, como vegetação pendurada e presença de madeira no canal (De Marco et al., 2015; Calvão et al., 2016). Além disso, a presença da vegetação ribeirinha, ainda que reduzida em riachos alterados (e.g. vegetação degradada em virtude de exploração de madeira) pode ter sido o mecanismo chave pelo qual o distúrbio local apresentou menor efeito nas espécies de Odonata.

Por outro lado, a mudança acentuada e sincrônica de Odonata devido ao distúrbio na escala de bacia de drenagem pode estar relacionada à configuração da paisagem que intercala áreas de vegetação com usos do solo de distintas intensidades, como áreas de floresta degradada, cultura de ciclo longo e pastagens. Dessa forma, é esperado que a diminuição na qualidade ambiental causada por intensos usos do solo conduza à perda de habitat de Odonata adultos o que consequentemente diminui sua frequência e abundância. Os distúrbios antrópicos na bacia podem levar a mudanças acentuadas nas comunidades e nos habitats (Kovalenko et al., 2014; Luke et al., 2017) e os elementos na paisagem, como manchas de floresta, podem atuar como barreiras para o fluxo gênico de populações de Odonata (Keller et al., 2012). Assim, a configuração da paisagem na drenagem é um fator limitante para os organismos sensíveis, como *Argia infumata*, *Chalcopteryx radians* e *Chalcopteryx rutilans*, pois essas espécies têm preferências por locais que mantêm a vegetação ripária em contato com a floresta adjacente (Calvão et al., 2016). Além disso, as espécies tolerantes associadas ao distúrbio na drenagem em nosso estudo, foram encontradas em estudos anteriores ocorrendo em áreas de vegetação fragmentada (Calvão et al., 2016; Alves-Martins et al., 2018) e em plantações de óleo de palma, que ainda mantêm a faixa da vegetação ripária (Carvalho et al., 2018).

O limiar da comunidade de Gerromorpha e EPT associado à baixa perturbação (soma z^- no ínicio do gradiente) sugere que os espécimes sensíveis dessa comunidade são determinados principalmente pela presença de vegetação ripária nos riachos e nas suas bacias de drenagem. Por outro lado, as espécies tolerantes (z^+) dessas comunidades apresentaram uma mudança gradual ao gradiente de distúrbio. Isso significa que essas espécies podem ser capazes de lidar com uma amplitude maior de modificação na paisagem, que pode ser representada por riachos distribuídos em áreas de florestas degradadas e de agriculturas de ciclo longo. Espécies tolerantes podem aumentar suas abundâncias com o aumento na intensidade de uso do solo (Burdon et al., 2016), sugerindo que essas espécies podem persistir em ambientes alterados (Heino, 2013). Isso provavelmente ocorre porque as características dos riachos podem interagir com o componente da paisagem e gerar um habitat (*sensu* Southwood, 1977; Poff e Ward, 1990) em que as espécies podem ser beneficiadas. Contudo, é importante ressaltar que o padrão observado nos gêneros tolerantes (z+, figura 2, tabela IS6) para a comunidade de EPT no gradiente de distúrbio em escala de drenagem, pode ser devido às incertezas em relação a esse limiar. Essa incerteza pode estar relacionada ao amplo valor do intervalo de quartil encontrado, que possivelmente ocorreu devido à baixa quantidade de gêneros tolerantes (n=1) para o distúrbio na drenagem.

As associações das espécies de Gerromorpha com os gradientes de distúrbios podem ser relacionadas à capacidade de dispersão dos organismos. Nós encontramos um padrão onde distúrbios locais estão relacionados à maior abundância de *Euvelia discal*a e *Stridulivelia alia*, espécies da família Veliidae com menor capacidade de dispersão (Andersen, 1982). Por outro lado, na escala de bacia de drenagem, as espécies de maior tamanho corporal e com maior capacidade de dispersão terrestre, como *Tachygerris celocis* e *Hydrometra guianana*, estão relacionadas ao aumento da abundância em paisagens mais degradadas (Andersen e Weir, 2004; Cunha e Juen, 2017). Dessa maneira, acreditamos que a dispersão terrestre seja o mecanismo de maior associação dessas espécies a locais adequados, uma vez que espécies de menor capacidade de dispersão podem estar relacionadas a escalas locais devido à limitação da dispersão, enquanto que espécies com maior capacidade de dispersão conseguem alcançar locais adequados ou com maiores chances de sucesso quanto ao habitat (Karaouzas e Gritzalis, 2006; Cunha e Juen, no prelo).

A maior sensibilidade de EPT ao distúrbio no local e na drenagem sugere que os gêneros dessa comunidade podem diminuir devido à dependência em relação à estrutura do ambiente para alimentação, habitat e reprodução. As modificações na estrutura ambiental dos riachos podem ocorrer em virtude dos efeitos diretos e indiretos das alterações na paisagem (e.g. Castello e Macedo, 2016; Leitão et al., 2018). As alterações da paisagem podem diminuir a quantidade de detritos orgânicos e bancos de folhas nos canais dos riachos (Paula et al., 2011), remover estruturas que fornecem habitat para a biota aquática, como macrófitas e detritos lenhosos e aumentar a temperatura da água, podendo dessa forma afetar diretamente a fisiologia e o comportamento das espécies, causando modificações nas comunidades (Bojsen e Barriga, 2002; Sweeney et al., 2004; Macedo et al., 2013; Leal et al., 2016). Assim, a alteração no habitat pode gerar uma paisagem composta por um mosaico de distintos usos do solo que podem ser desfavoráveis para a ocorrência das espécies (Ricketts, 2001; Fischer e Lindenmayer, 2007), que são sensíveis as alterações mesmo que sutis no ambiente (Faria et al., 2017).

O aumento na frequência de *Brasilocaenis* e *Ulmeritoides* em relação aos distúrbios em escala local e na bacia de drenagem sugere que modificações nas características ambientais dos riachos decorrentes dos distúrbios beneficiam esses gêneros, provavelmente em virtude do aumento de sedimentos no canal e flutuação nos parâmetros físicos químicos da água. Esses gêneros foram relacionadas em estudos anteriores a habitats semilóticos com baixas concentrações de oxigênio (Da-Silva et al., 2010) e riachos com alta temperatura da água, pH, condutividade e sedimentos finos no canal (Shimano e Juen 2016; Faria et al., 2017).

4.6 Conclusões

Nossos resultados suportam o cenário de que o distúrbio antrópico, principalmente relacionados aos distintos usos do solo (e.g. culturas de ciclo longo e pastagem) e exploração dos recursos naturais exerce fortes efeitos sobre as comunidades de insetos aquáticos em riachos de pequeno porte. Os riachos estudados estão sujeitos a níveis de distúrbios acima da condição ideal para as espécies, uma vez que os limiares foram identificados para as comunidades de Odonata, Gerromorpha e EPT. Além disso, as espécies de insetos aquáticos estão diminuindo em frequência e abundância com o aumento tanto do distúrbio no local como do distúrbio na área de drenagem dos riachos. Esses resultados podem ser aplicados na gestão dos recursos hídricos, principalmente porque evidencia que existe variação na resposta entre e dentro das comunidades a gradientes de distúrbio em diferentes escalas. Isso reforça a necessidade de observar tanto as respostas individuais como da comunidade na tomada de decisão.

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3.9 Informação Suplementar

Tabela IS1. Coordenada geográfica em graus decimais do conjunto de riachos (unidade amostral - ua) amostrados.

Riachos	**Longitude**	**Latitude**	**Zona**	**Bacia**
ua1	-48.284722	-3.7313	22S	Capim

ua2	-48.484641	-3.7274	22S	Capim
ua3	-48.289478	-3.7322	22S	Capim
ua4	-48.487675	-3.7233	22S	Capim
ua5	-48.47472	-3.6531	22S	Capim
ua6	-48.464439	-3.697	22S	Capim
ua7	-48.473225	-3.7111	22S	Capim
ua8	-48.81531	-3.6878	22S	Capim
ua9	-48.558601	-3.5458	22S	Capim
ua10	-48.69424	-2.3187	22S	Acará
ua11	-48.88259	-2.5777	22S	Acará
ua12	-48.8821	-2.5266	22S	Acará
ua13	-48.933	-2.4856	22S	Acará
ua14	-48.70494	-2.4787	22S	Acará
ua15	-48.80201	-2.3809	22S	Acará
ua16	-48.506778	-2.6032	22S	Acará
ua17	-48.61725	-2.5225	22S	Acará
ua18	-48.591073	-3.7527	22S	Capim
ua19	-48.662025	-3.6806	22S	Capim
ua20	-48.674158	-3.7035	22S	Capim
ua21	-48.702643	-3.7186	22S	Capim
ua22	-48.717993	-3.7084	22S	Capim
ua23	-48.747775	-3.6015	22S	Capim
ua24	-48.78708	-3.5925	22S	Capim
ua25	-48.780851	-3.5786	22S	Capim
ua26	-48.811037	-3.5333	22S	Capim
ua27	-48.782855	-3.5307	22S	Capim
ua28	-48.610027	-3.6539	22S	Capim
ua29	-48.579028	-3.7121	22S	Capim
ua30	-48.561113	-3.7093	22S	Capim
ua31	-48.563955	-3.6635	22S	Capim
ua32	-48.602569	-3.7408	22S	Capim
ua33	-47.73679	-2.9645	23S	Capim
ua34	-47.77956	-3.0207	23S	Capim
ua35	-47.76585	-3.0256	23S	Capim
ua36	-47.78866	-3.0617	23S	Capim
ua37	-47.77047	-2.9956	23S	Capim
ua38	-47.76374	-3.1798	23S	Capim
ua39	-47.74525	-2.997	23S	Capim
ua40	-47.73275	-2.9986	23S	Capim
ua41	-47.74682	-3.0064	23S	Capim
ua42	-47.74602	-3.0047	23S	Capim
ua43	-48.79921	-2.5402	22S	Acará
ua44	-48.73913	-2.6039	22S	Acará
ua45	-48.75813	-2.5634	22S	Acará
ua46	-48.62647	-2.2653	22S	Acará
ua47	-48.81686	-2.5792	22S	Acará

ua48	-48.72593	-2.5647	22S	Acará
ua49	-48.61745	-2.4705	22S	Acará
ua50	-48.707	-2.5599	22S	Acará
ua51	-48.747	-2.456	22S	Acará
ua52	-48.605	-2.268	22S	Acará
ua53	-48.605	-2.268	22S	Acará
ua54	-48.69745	-2.3698	22S	Acará
ua55	-48.0988	-1.7794	22S	Acará
ua56	-48.1946	-1.7843	22S	Acará
ua57	-48.0333	-1.9277	22S	Acará
ua58	-48.0421	-1.859	22S	Acará
ua59	-48.1353	-1.9615	22S	Acará
ua60	-48.5135	-2.4368	22S	Acará
ua61	-48.52605	-2.5374	22S	Acará
ua62	-48.1951	-2.4854	22S	Acará
ua63	-47.91115	-2.8894	23S	Capim
ua64	-47.98565	-2.9205	23S	Capim
ua65	-47.8868	-2.9455	23S	Capim
ua66	-47.53262	-2.3381	23S	Capim

Tabela IS2. Porcentagem das classes de uso e cobertura do solo identificada através de cena RapidEye na área de drenagem a montante de cada riacho amostrado.

Riachos	**Solo exposto**	**Pastagem**	**Cultura de ciclo longo**	**Vegetação degradada**	**Vegetação densa**	**Água**	**Nuvem**	**Sombra de nuvem**
ua1	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua2	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua3	0.00	0.00	0.00	2.78	97.22	0.00	0.00	0.00
ua4	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua5	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua6	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua7	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua8	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua9	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua10	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua11	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua12	2.18	0.00	0.00	5.48	92.34	0.00	0.00	0.00
ua13	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua14	12.99	0.00	23.65	6.22	57.14	0.00	0.00	0.00
ua15	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua16	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua17	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua18	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua19	1.35	0.00	0.00	1.97	91.48	0.84	4.37	0.00
ua20	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua21	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00

ua22	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua23	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua24	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua25	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua26	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua27	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua28	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua29	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua30	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua31	0.00	0.00	0.00	8.54	91.46	0.00	0.00	0.00
ua32	0.00	0.00	0.00	0.06	99.94	0.00	0.00	0.00
ua33	6.93	19.93	0.00	10.16	61.24	0.24	0.46	1.04
ua34	2.04	15.77	0.34	20.08	61.76	0.00	0.00	0.00
ua35	3.72	13.68	0.00	10.55	72.05	0.00	0.00	0.00
ua36	0.23	28.89	2.19	21.14	47.16	0.39	0.00	0.00
ua37	2.73	5.07	0.00	40.01	47.38	0.00	1.83	2.97
ua38	8.91	0.28	0.00	13.38	68.67	1.90	6.56	0.29
ua39	8.49	7.68	0.00	2.57	64.56	14.31	2.40	0.00
ua40	6.13	20.40	0.00	0.00	69.59	0.00	3.88	0.00
ua41	1.25	9.86	0.00	0.00	88.81	0.09	0.00	0.00
ua42	3.63	5.66	0.00	0.00	85.45	5.26	0.00	0.00
ua43	8.43	0.00	86.13	0.20	5.24	0.00	0.00	0.00
ua44	7.15	0.00	51.00	0.00	41.85	0.00	0.00	0.00
ua45	10.50	0.00	65.30	0.00	24.20	0.00	0.00	0.00
ua46	0.00	0.00	92.96	7.04	0.00	0.00	0.00	0.00
ua47	18.90	0.00	36.86	0.00	44.23	0.00	0.00	0.00
ua48	8.73	0.00	84.19	0.53	6.55	0.00	0.00	0.00
ua49	4.25	0.00	55.85	3.15	36.51	0.25	0.00	0.00
ua50	7.80	0.00	78.28	2.88	11.05	0.00	0.00	0.00
ua51	13.63	0.00	60.18	7.64	18.55	0.00	0.00	0.00
ua52	2.52	0.00	94.46	3.02	0.00	0.00	0.00	0.00
ua53	3.95	0.00	75.90	1.53	18.62	0.00	0.00	0.00
ua54	4.05	0.00	50.65	5.28	40.01	0.00	0.00	0.00
ua55	1.27	0.00	16.22	0.00	82.51	0.00	0.00	0.00
ua56	0.00	0.00	0.00	6.01	81.07	0.00	4.77	8.15
ua57	10.88	1.96	45.79	2.02	39.36	0.00	0.00	0.00
ua58	7.66	0.00	60.83	9.68	20.82	0.00	0.00	1.00
ua59	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
ua60	2.59	0.00	26.38	0.00	71.03	0.00	0.00	0.00
ua61	9.13	7.33	50.47	5.47	27.60	0.00	0.00	0.00
ua62	9.82	0.00	59.66	0.00	30.52	0.00	0.00	0.00
ua63	17.87	37.00	0.00	31.23	12.92	0.00	0.94	0.00
ua64	14.11	11.70	0.00	2.39	64.35	0.00	2.76	4.69
ua65	26.42	35.50	0.00	1.55	11.26	6.16	16.47	2.74
ua66	6.47	75.80	0.00	7.56	10.21	0.00	0.00	0.00

Tabela IS3. Porcentagem do uso do solo após a atribuição dos pesos. Os pesos atribuídos foram classificados de acordo com a intensidade de alteração de cada uso: 4 x % solo exposto, 2 x % pastagem, 1 x % cultura de ciclo longo e floresta degradada. A padronização de cada índice foi feita dividindo o valor observado do índice (LDI_s* e CDI_s*) pelo valor extraído de 75% (LDI_p** = 5 e CDI_p** = 300) do valor máximo que o índice teoricamente poderia atingir (LDI_s =7 e CDI_s =400).

Riachos	**Solo exposto**	**Pastagem**	**Cultura de ciclo longo**	**Vegetação degradada**	**CDI_s***	**CDI_p****	**LDI_s***	**LDI_p****
ua1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ua2	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ua3	0.00	0.00	0.00	2.78	2.78	0.01	0.00	0.00
ua4	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ua5	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ua6	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ua7	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ua8	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ua9	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ua10	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.03
ua11	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.01
ua12	8.72	0.00	0.00	5.48	14.2	0.05	0.67	0.13
ua13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ua14	51.9	0.00	23.6	6.22	81.8	0.27	0.00	0.00
ua15	0.00	0.00	0.00	0.00	0.00	0.00	0.52	0.10
ua16	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ua17	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.01
ua18	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ua19	5.41	0.00	0.00	1.97	7.38	0.02	0.12	0.02
ua20	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.02
ua21	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.01
ua22	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.04
ua23	0.00	0.00	0.00	0.00	0.00	0.00	0.42	0.08
ua24	0.00	0.00	0.00	0.00	0.00	0.00	0.41	0.08
ua25	0.00	0.00	0.00	0.00	0.00	0.00	0.40	0.08
ua26	0.00	0.00	0.00	0.00	0.00	0.00	0.40	0.08
ua27	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ua28	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.02
ua29	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ua30	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
ua31	0.00	0.00	0.00	8.54	8.54	0.03	1.46	0.29
ua32	0.00	0.00	0.00	0.06	0.06	0.00	0.00	0.00
ua33	27.7	39.9	0.00	10.2	77.8	0.26	0.82	0.16
ua34	8.16	31.5	0.34	20.1	60.1	0.20	1.43	0.29
ua35	14.9	27.4	0.00	10.6	52.8	0.18	1.03	0.21
ua36	0.92	57.8	2.19	21.1	82.0	0.27	0.86	0.17
ua37	10.9	10.14	0.00	40.0	61.1	0.20	0.70	0.14

ua38	35.6	0.56	0.00	13.4	49.6	0.17	0.99	0.20
ua39	33.9	15.36	0.00	2.6	51.9	0.17	0.77	0.15
ua40	24.5	40.81	0.00	0.00	65.3	0.22	0.97	0.19
ua41	4.99	19.72	0.00	0.00	24.7	0.08	0.77	0.15
ua42	14.5	11.32	0.00	0.00	25.8	0.09	1.24	0.25
ua43	33.7	0.00	86.1	0.20	120	0.40	2.09	0.42
ua44	28.6	0.00	51.0	0.00	79.6	0.27	1.54	0.31
ua45	42.0	0.00	65.3	0.00	107	0.36	2.12	0.42
ua46	0.0	0.00	93.0	7.04	100	0.33	1.33	0.27
ua47	75.6	0.00	36.9	0.00	112	0.37	1.46	0.29
ua48	34.9	0.00	84.2	0.53	120	0.40	1.33	0.27
ua49	17.0	0.00	55.9	3.15	76.0	0.25	1.33	0.27
ua50	31.2	0.00	78.3	2.88	112	0.37	1.33	0.27
ua51	54.5	0.00	60.2	7.64	122	0.41	2.10	0.42
ua52	10.1	0.00	94.5	3.02	108	0.36	0.99	0.20
ua53	15.8	0.00	75.9	1.53	93.2	0.31	0.33	0.07
ua54	16.2	0.00	50.7	5.28	72.2	0.24	1.33	0.27
ua55	5.09	0.00	16.2	0.00	21.3	0.07	0.64	0.13
ua56	0.00	0.00	0.00	6.01	6.01	0.02	0.83	0.17
ua57	43.5	3.93	45.8	2.02	95.2	0.32	0.67	0.13
ua58	30.7	0.00	60.8	9.68	101	0.34	1.08	0.22
ua59	0.00	0.00	0.00	0.00	0.00	0.00	1.07	0.21
ua60	10.4	0.00	26.4	0.00	36.7	0.12	0.67	0.13
ua61	36.5	14.7	50.5	5.47	107	0.36	0.83	0.17
ua62	39.3	0.00	59.7	0.00	98.9	0.33	0.61	0.12
ua63	71.5	74.0	0.00	31.2	177	0.59	0.25	0.05
ua64	56.4	23.5	0.00	2.39	82.3	0.27	0.67	0.13
ua65	106	70.9	0.00	1.55	178	0.59	0.33	0.07
ua66	25.9	152	0.00	7.55	185	0.62	1.43	0.29

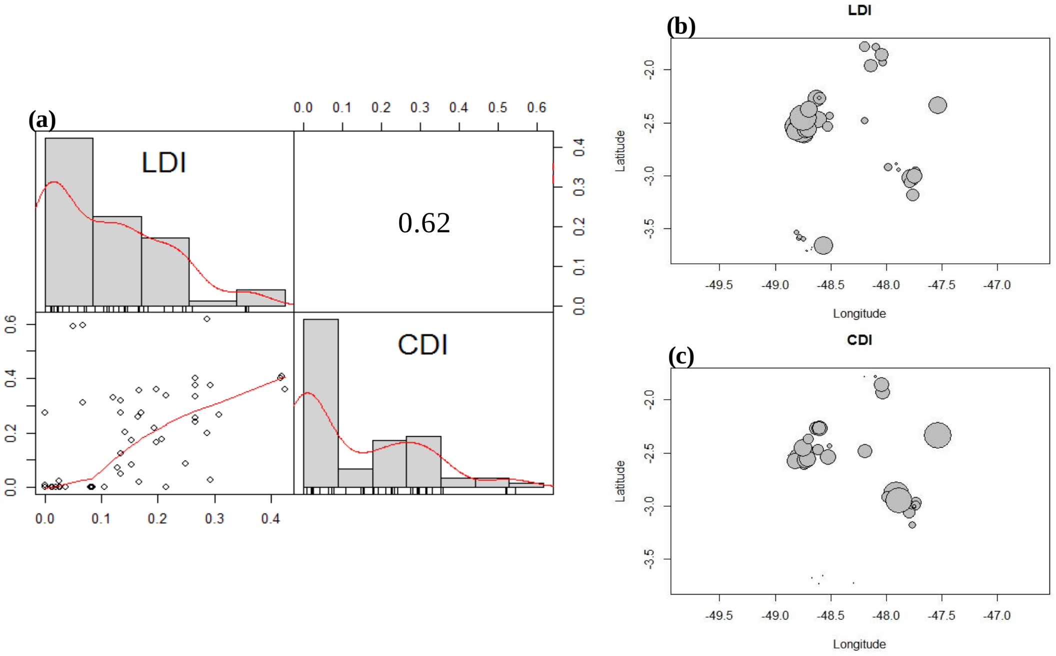

Figura IS1. Correlação entre os gradientes de distúrbios. O histograma de cada gradiente na diagonal possibilita visualizar a distribuição dos dados e o gráfico de dispersão permite verificar a forma das relações entre os gradientes. O coeficiente de correlação de Pearson é exibido no quadrante direito superior (a). Na representação gráfica dos gradientes de distúrbio LDI (b) e CDI (c), o tamanho dos círculos é proporcional ao valor observado do gradiente.

Tabela IS4. Composição das comunidades coletadas nos 66 riachos amostrados.

Composição	**Abundância**
Odonata	
Calopterygidae	
Chalcopteryx radians Ris, 1914	37
Chalcopteryx rutilans (Rambur, 1842)	179
Hetaerina rosea Selys, 1853	14
Mnesarete aenea (Selys, 1853)	261
Coenagrionidae	
Argia collata Selys, 1865	58
Argia indicatrix Calvert, 1902	81
Argia infumata Selys, 1865	192
Argia tinctipennis Selys, 1865	156
Dicteriadidae	
Dicterias atrosanguinea Selys, 1853	25
Libellulidae	
Argyrothemis argentea Ris, 1911	23
Fylgia amazonica Kirby , 1889	60
Oligoclada walkeri Geijskes, 1931	22
Perithemis thais Kirby, 1889	14
Megapodagrionidae	

Heteragrion silvarum Sjöstedt, 1918	282
Oxystigma petiolatum (Selys, 1862)	26
Protoneuridae	
Epipleoneura capilliformis (Selys, 1886)	10
Epipleoneura haroldoi Santos, 1964	10
Epipleoneura metallica Rácenis, 1955	61
Epipleoneura spatulata Rácenis, 1960	10
Neoneura luzmarina De Marmels, 1989	20
Phasmoneura exigua (Selys, 1886)	12
Protoneura tenuis Selys, 1860	72
Psaironeura tenuissima (Selys, 1886)	58
Gerromorpha	
Gerridae	
Brachymetra albinervis albinervis Nieser, 1970	36
Brachymetra lata Shaw, 1933	1845
Brachymetra shawi Hungerford & Matsuda 1938	21
Cylindrostethus palmaris Drake & Harris, 1934	226
Limnogonus recurvus Drake & Harris, 1930	14
Neogerris lotus (White, 1879)	112
Neogerris magnus (Kuitert, 1942)	116
Tachygerris adamsoni (Drake, 1942)	52
Tachygerris celocis Drake & Harris, 1931	317
Tachygerris opacus (Champion, 1901)	32
Hydrometridae	
Hydrometra comata Torre-Bueno, 1926	37
Hydrometra guianana Hungerford & Evans, 1934	24
Mesoveliidae	
Mesovelia amoena Uhler, 1894	14
Veliidae	
Euvelia discala Euvelia Drake, 1957	48
Euvelia lata Polhemus & Polhemus, 1984	31
Microvelia hinei Drake, 1920	371
Microvelia mimula White, 1879	102
Paravelia bullialata Polhemus & Polhemus, 1984	82
Paravelia conata (Hungerford, 1929)	22
Rhagovelia brunae Magalhães & Moreira, 2016	407
Rhagovelia elegans Uhler, 1894	912
Rhagovelia evidis Bacon, 1948	1742
Rhagovelia humboldti Polhemus, 1997	39
Rhagovelia jubata Bacon, 1948	1012
Stridulivelia alia (Drake, 1957)	74
Stridulivelia strigosa (Hungerford, 1929)	120
Stridulivelia tersa (Drake & Harris, 1941)	76
Stridulivelia transversa (Hungerford, 1929)	340
Ephemeroptera, Plecoptera e Trichoptera	
Baetidae	

Aturbina Lugo-Ortiz & McCafferty, 1996	9
Callibaetis Eaton, 1881	29
Cloeodes Traver, 1938	24
Waltzoyphius McCafferty & Lugo-Ortiz, 1995	12
Zelusia Lugo-Ortiz & McCafferty, 1998	103
Caenidae	
Brasilocaenis Puthz, 1975	29
Calamoceridae	
Phylloicus Müller, 1880	814
Euthyplociidae	
Campylocia Needham & Murphy, 1924	1489
Glossosomatidae	
Mortoniella Ulmer, 1906	11
Helicopsychidae	
Helicopsyche Siebold, 1856	471
Hydropsychidae	
Leptonema Guérin, 1843	396
Macronema Pictet, 1836	1042
Macrostemum Kolenati, 1859	818
Smicridea McLachlan, 1871	344
Leptoceridae	
Nectopsyche Müller, 1879	28
Oecetis McLachlan, 1877	177
Triplectides Kolenati, 1859	468
Leptohyphidae	
Amanahyphes Salles & Molineri, 2006	36
Leptophlebiidae	
Farrodes Peters, 1971	369
Hagenulopsis Ulmer, 1920	48
Miroculis Edmunds, 1963	4365
Simothraulopsis Demoulin, 1966	21
Ulmeritoides Traver, 1959	396
Odontoceridae	
Marilia Müller, 1880	38
Perlidae	
Anacroneuria Klapálek, 1909	306
Enderleina Jewett, 1960	11
Macrogynoplax Enderlein, 1909	938
Philopotamidae	
Chimarra Stephens, 1829	500
Polycentropodidae	
Cernotina Ross, 1838	320
Cyrnellus Banks, 1913	39
Polyplectropus Ulmer, 1905	199
Polymitarcyidae	
Campsurus Eaton, 1868	75

Total	**23832**

Tabela IS5. Resultado do limiar (TITAN) da comunidade de Odonata, Gerromorpha e EPT associadas ao gradiente de distúrbio local (*Local disturbance index* - LDI) e de drenagem (*Catchment disturbance index* - CDI). "Ponto" de mudança observado é mostrado para cada comunidade. A soma (z) é correspondente a filtragem obtida calculando apenas as pontuaçõe das espécies que foram determinadas como indicadoras puras e confiáveis. Essa soma representa os escores z negativos (z-) ou positivos (z+) em um ponto de mudança da comunidade.

		LDI					
		Ponto	5%	10%	50%	90%	95%
Odonata	soma (z-)	0.13	0.04	0.05	0.13	0.14	0.17
	soma (z+)	0.21	0.09	0.09	0.21	0.29	0.30
Gerromorpha	soma (z-)	0.09	0.04	0.05	0.09	0.12	0.12
	soma (z+)	0.12	0.07	0.08	0.20	0.29	0.29
EPT	soma (z-)	0.08	0.02	0.03	0.09	0.14	0.14
	soma (z+)	0.13	0.13	0.13	0.14	0.18	0.22
		CDI					
Odonata	soma (z-)	0.08	0.00	0.00	0.04	0.08	0.11
	soma (z+)	0.09	0.04	0.05	0.10	0.27	0.27
Gerromorpha	soma (z-)	0.06	0.01	0.03	0.06	0.08	0.10
	soma (z+)	0.06	0.02	0.03	0.08	0.33	0.35
EPT	soma (z-)	0.08	0.05	0.06	0.08	0.10	0.12
	soma (z+)	0.09	0.03	0.05	0.08	0.35	0.50

Tabela IS6. Resultados do limiar das espécies (TITAN) de Odonata (A), Gerromorpha (B) e Ephemeroptera, Plecoptera e Trichoptera (C) em resposta a gradiente de distúrbio local (*Local disturbance index* - LDI) e na drenagem (*Catchment disturbance index* - CDI). "Ponto" de mudança observado é mostrado para cada táxon e "+ / -" representa o limiar positivo (z+) ou negativo (z-). Todos os táxons analisados foram incluídos nessa tabela, mas apenas os taxa que preencheram os critérios de P (≤ 0.05), pureza (≥ 0.95) e confiabilidade (≥ 0.95) estão em negrito e foram apresentados nos resultados na figura3.

(A) Odonata

	LDI (Local)												
	Ponto	Frequência	"+ / -	IndVal	P	Z	5%	10%	50%	90%	95%	Pureza	Confiabilidade
Argia collata	**0.27**	**23.00**	**z+**	**44.86**	**0.00**	**4.64**	**0.01**	**0.01**	**0.13**	**0.26**	**0.27**	**1.00**	**1.00**
Argia indicatrix	0.26	30.00	-	38.68	0.13	0.97	0.00	0.00	0.08	0.26	0.29	0.60	0.55
Argia infumata	**0.14**	**41.00**	**z-**	**63.76**	**0.01**	**4.80**	**0.08**	**0.11**	**0.14**	**0.21**	**0.23**	**1.00**	**1.00**
Argia tinctipennis	0.00	44.00	-	53.98	0.01	3.18	0.04	0.05	0.13	0.28	0.29	0.92	0.84
Argyrothemis argentea	0.36	17.00	-	31.18	0.01	3.22	0.08	0.08	0.17	0.36	0.36	0.95	0.88
Chalcopteryx radians	0.09	7.00	-	21.88	0.01	3.72	0.00	0.00	0.09	0.12	0.13	1.00	0.90
Chalcopteryx rutilans	**0.00**	**35.00**	**z-**	**56.73**	**0.00**	**4.62**	**0.00**	**0.01**	**0.07**	**0.12**	**0.12**	**1.00**	**1.00**
Dicterias atrosanguinea	0.30	10.00	-	63.12	0.00	6.40	0.00	0.25	0.29	0.30	0.36	0.91	0.96
Epipleoneura capilliformis	0.04	5.00	-	19.23	0.00	4.12	0.00	0.00	0.04	0.07	0.07	0.99	0.85
Epipleoneura haroldoi	0.00	7.00	-	38.04	0.01	5.81	0.00	0.00	0.00	0.07	0.08	0.98	0.87
Epipleoneura metallica	**0.12**	**20.00**	**z+**	**53.64**	**0.00**	**6.72**	**0.04**	**0.04**	**0.12**	**0.15**	**0.20**	**1.00**	**1.00**
Epipleoneura spatulata	0.30	5.00	-	43.46	0.02	4.32	0.04	0.12	0.27	0.42	0.42	0.94	0.80
Fylgia amazonica	0.14	15.00	-	27.08	0.05	2.02	0.00	0.03	0.14	0.21	0.24	0.79	0.67
Hetaerina rosea	0.29	6.00	-	78.92	0.00	14.88	0.27	0.28	0.29	0.30	0.36	0.99	0.94
Heteragrion silvarum	0.00	49.00	-	54.65	0.10	1.49	0.02	0.02	0.22	0.29	0.37	0.79	0.69
Mnesarete aenea	0.00	41.00	-	52.39	0.02	3.18	0.00	0.00	0.04	0.29	0.29	0.83	0.93
Neoneura luzmarina	**0.21**	**8.00**	**z+**	**38.73**	**0.00**	**6.15**	**0.15**	**0.15**	**0.21**	**0.27**	**0.29**	**1.00**	**0.99**
Oligoclada walkeri	0.15	13.00	-	26.04	0.01	2.82	0.02	0.02	0.14	0.17	0.21	0.93	0.85
Oxystigma petiolatum	0.26	12.00	-	20.82	0.08	1.87	0.02	0.06	0.17	0.31	0.42	0.90	0.74
Perithemis thais	0.16	6.00	-	18.15	0.02	3.40	0.07	0.07	0.16	0.18	0.18	0.98	0.77
Phasmoneura exigua	0.17	6.00	-	12.77	0.20	1.20	0.00	0.00	0.07	0.17	0.17	0.61	0.39

Protoneura tenuis	0.00	23.00	-	35.61	0.03	2.31	0.00	0.00	0.12	0.21	0.21	0.88	0.85
Psaironeura tenuissima	0.29	24.00	-	37.88	0.10	1.55	0.00	0.00	0.21	0.27	0.29	0.79	0.69
CDI (Drenagem)													
Argia collata	**0.40**	**23.00**	**z+**	**43.16**	**0.00**	**4.70**	**0.04**	**0.05**	**0.15**	**0.37**	**0.40**	**0.99**	**0.97**
Argia indicatrix	0.26	30.00	-	42.12	0.04	2.22	0.00	0.00	0.23	0.27	0.40	0.83	0.81
Argia infumata	**0.00**	**41.00**	**z-**	**67.60**	**0.00**	**6.14**	**0.02**	**0.03**	**0.08**	**0.21**	**0.21**	**1.00**	**1.00**
Argia tinctipennis	0.06	44.00	-	54.55	0.01	3.01	0.00	0.00	0.06	0.29	0.34	0.89	0.84
Argyrothemis argentea	0.40	17.00	-	22.62	0.17	1.03	0.00	0.00	0.27	0.37	0.39	0.63	0.50
Chalcopteryx radians	**0.00**	**7.00**	**z-**	**36.84**	**0.00**	**7.07**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**1.00**	**0.99**
Chalcopteryx rutilans	**0.40**	**35.00**	**z-**	**55.23**	**0.00**	**4.70**	**0.00**	**0.00**	**0.02**	**0.05**	**0.08**	**0.98**	**0.99**
Dicterias atrosanguinea	0.40	10.00	-	14.91	0.24	0.63	0.00	0.00	0.16	0.36	0.36	0.72	0.51
Epipleoneura capilliformis	0.03	5.00	-	16.13	0.03	3.04	0.00	0.00	0.02	0.05	0.06	0.99	0.77
Epipleoneura haroldoi	0.00	7.00	-	39.74	0.05	3.42	0.00	0.00	0.02	0.25	0.29	0.90	0.66
Epipleoneura metallica	**0.04**	**20.00**	**z+**	**58.82**	**0.00**	**8.49**	**0.02**	**0.03**	**0.10**	**0.26**	**0.33**	**1.00**	**1.00**
Epipleoneura spatulata	0.50	5.00	-	16.42	0.03	2.81	0.00	0.19	0.23	0.50	0.50	0.94	0.70
Fylgia amazonica	0.50	15.00	-	70.00	0.01	3.97	0.02	0.06	0.36	0.50	0.50	0.92	0.87
Hetaerina rosea	0.36	6.00	-	28.61	0.00	4.43	0.00	0.18	0.35	0.37	0.39	0.89	0.81
Heteragrion silvarum	0.50	49.00	-	63.41	0.03	2.63	0.00	0.00	0.34	0.40	0.50	0.88	0.82
Mnesarete aenea	0.00	41.00	-	48.18	0.04	2.23	0.00	0.00	0.06	0.50	0.50	0.77	0.90
Neoneura luzmarina	**0.37**	**8.00**	**z+**	**24.07**	**0.00**	**5.01**	**0.01**	**0.02**	**0.23**	**0.37**	**0.37**	**1.00**	**0.96**
Oligoclada walkeri	**0.50**	**13.00**	**z+**	**32.60**	**0.00**	**4.53**	**0.06**	**0.06**	**0.08**	**0.50**	**0.50**	**0.98**	**0.95**
Oxystigma petiolatum	**0.29**	**12.00**	**z+**	**50.00**	**0.00**	**8.48**	**0.25**	**0.26**	**0.27**	**0.30**	**0.30**	**1.00**	**1.00**
Perithemis thais	0.08	6.00	-	14.05	0.13	1.52	0.00	0.00	0.08	0.26	0.26	0.67	0.51
Phasmoneura exigua	0.00	6.00	-	13.04	0.24	1.38	0.00	0.00	0.01	0.26	0.27	0.75	0.28
Protoneura tenuis	0.00	23.00	-	74.96	0.00	5.54	0.00	0.00	0.20	0.24	0.40	0.93	0.93
Psaironeura tenuissima	0.23	24.00	-	44.32	0.00	4.58	0.00	0.06	0.23	0.25	0.26	0.92	0.95

(B) Gerromopha (Hemíptera)

LDI (Local)													
	Ponto	Frequência	"+ / -	IndVal	P	Z	5%	10%	50%	90%	95%	Pureza	Confiabilidade

Brachymetra albinervis albinervis	0.00	10.00	-	24.91	0.00	3.50	0.00	0.00	0.09	0.13	0.13	0.99	0.91
Brachymetra lata	**0.26**	**60.00**	**z-**	**60.04**	**0.02**	**2.87**	**0.03**	**0.07**	**0.17**	**0.27**	**0.28**	**0.99**	**0.96**
Brachymetra shawi	0.00	7.00	-	13.14	0.23	0.94	0.00	0.03	0.13	0.22	0.27	0.78	0.36
Cylindrostethus palmaris	**0.09**	**36.00**	**z+**	**61.37**	**0.00**	**5.76**	**0.03**	**0.06**	**0.10**	**0.19**	**0.21**	**1.00**	**1.00**
Euvelia discala	**0.29**	**7.00**	**z+**	**39.63**	**0.00**	**6.74**	**0.14**	**0.21**	**0.27**	**0.29**	**0.29**	**1.00**	**0.97**
Euvelia lata	0.30	7.00	-	44.03	0.03	3.97	0.11	0.13	0.27	0.36	0.42	1.00	0.92
Hydrometra comata	0.00	13.00	-	41.74	0.01	3.40	0.01	0.08	0.13	0.14	0.15	0.99	0.94
Hydrometra guianana	0.07	7.00	-	17.95	0.02	2.36	0.04	0.05	0.13	0.20	0.28	0.96	0.75
Limnogonus recurvus	0.13	6.00	-	15.10	0.08	2.08	0.02	0.11	0.13	0.27	0.27	0.90	0.58
Mesovelia amoena	0.30	10.00	-	16.67	0.59	0.18	0.00	0.00	0.15	0.30	0.30	0.54	0.43
Microvelia hinei	0.14	20.00	-	38.88	0.02	2.65	0.03	0.04	0.14	0.20	0.22	0.93	0.84
Microvelia mimula	0.14	19.00	-	40.00	0.00	3.79	0.02	0.04	0.14	0.15	0.20	0.97	0.95
Neogerris lotus	0.07	16.00	-	34.52	0.00	3.27	0.03	0.05	0.07	0.20	0.26	0.87	0.90
Neogerris magnus	0.00	21.00	-	35.00	0.15	0.94	0.00	0.00	0.14	0.26	0.27	0.41	0.58
Paravelia bullialata	0.00	25.00	-	50.31	0.10	1.53	0.00	0.02	0.14	0.29	0.29	0.57	0.64
Paravelia conata	0.00	14.00	-	22.45	0.06	1.64	0.00	0.00	0.13	0.21	0.28	0.84	0.70
Rhagovelia brunae	**0.29**	**13.00**	**z+**	**61.04**	**0.00**	**8.29**	**0.17**	**0.20**	**0.26**	**0.29**	**0.29**	**1.00**	**1.00**
Rhagovelia elegans	0.00	58.00	-	55.26	0.26	0.56	0.00	0.00	0.01	0.27	0.28	0.50	0.65
Rhagovelia evidis	**0.12**	**47.00**	**z-**	**76.64**	**0.00**	**7.68**	**0.04**	**0.06**	**0.12**	**0.13**	**0.14**	**1.00**	**1.00**
Rhagovelia humboldti	0.13	8.00	-	20.51	0.02	3.29	0.04	0.05	0.13	0.14	0.14	0.96	0.86
Rhagovelia jubata	**0.09**	**29.00**	**z-**	**56.27**	**0.00**	**5.32**	**0.03**	**0.04**	**0.09**	**0.11**	**0.12**	**1.00**	**0.99**
Stridulivelia alia	**0.36**	**22.00**	**z+**	**41.23**	**0.02**	**3.20**	**0.00**	**0.01**	**0.04**	**0.36**	**0.36**	**0.98**	**0.97**
Stridulivelia strigosa	**0.08**	**23.00**	**z-**	**55.24**	**0.00**	**7.11**	**0.04**	**0.04**	**0.08**	**0.12**	**0.12**	**1.00**	**1.00**
Stridulivelia tersa	**0.30**	**15.00**	**z+**	**38.61**	**0.00**	**5.03**	**0.09**	**0.11**	**0.15**	**0.30**	**0.30**	**1.00**	**1.00**
Stridulivelia transversa	**0.09**	**27.00**	**z-**	**67.68**	**0.00**	**8.26**	**0.03**	**0.08**	**0.09**	**0.12**	**0.13**	**1.00**	**1.00**
Tachygerris adamsoni	0.00	17.00	-	31.68	0.02	2.42	0.01	0.01	0.09	0.17	0.21	0.92	0.92
Tachygerris celocis	0.30	41.00	-	57.57	0.02	2.91	0.08	0.13	0.26	0.29	0.29	0.97	0.92
Tachygerris opacus	0.17	12.00	-	22.86	0.09	1.78	0.00	0.00	0.17	0.18	0.20	0.77	0.62
CDI (Drenagem)													
Brachymetra albinervis albinervis	**0.08**	**10.00**	**z-**	**24.17**	**0.01**	**3.59**	**0.00**	**0.00**	**0.08**	**0.19**	**0.20**	**1.00**	**0.96**

Brachymetra lata	**0.50**	**60.00**	**z-**	**67.02**	**0.00**	**5.20**	**0.08**	**0.08**	**0.21**	**0.35**	**0.50**	**1.00**	**0.99**
Brachymetra shawi	0.50	7.00	-	13.71	0.23	1.06	0.00	0.00	0.19	0.50	0.50	0.70	0.42
Cylindrostethus palmaris	**0.20**	**36.00**	**z+**	**65.86**	**0.00**	**6.66**	**0.02**	**0.02**	**0.12**	**0.26**	**0.26**	**1.00**	**1.00**
Euvelia discala	0.23	7.00	-	23.57	0.00	4.09	0.00	0.17	0.23	0.35	0.35	0.94	0.86
Euvelia lata	0.25	7.00	-	21.16	0.01	3.22	0.04	0.08	0.24	0.35	0.35	0.96	0.80
Hydrometra comata	0.50	13.00	-	25.02	0.04	2.35	0.00	0.00	0.03	0.51	0.59	0.73	0.95
Hydrometra guianana	**0.50**	**7.00**	**z+**	**63.11**	**0.01**	**5.35**	**0.06**	**0.08**	**0.31**	**0.59**	**0.59**	**1.00**	**0.95**
Limnogonus recurvus	0.00	6.00	-	19.10	0.02	3.01	0.00	0.17	0.20	0.59	0.59	0.94	0.75
Mesovelia amoena	0.00	10.00	-	28.38	0.07	0.86	0.00	0.00	0.17	0.50	0.50	0.46	0.51
Microvelia hinei	0.50	20.00	-	39.59	0.00	3.40	0.04	0.05	0.08	0.50	0.50	0.78	0.96
Microvelia mimula	**0.50**	**19.00**	**z+**	**41.32**	**0.00**	**3.99**	**0.01**	**0.02**	**0.07**	**0.41**	**0.50**	**0.96**	**0.99**
Neogerris lotus	0.17	16.00	-	26.52	0.06	1.95	0.00	0.00	0.17	0.37	0.50	0.73	0.73
Neogerris magnus	0.50	21.00	-	74.24	0.00	4.99	0.00	0.08	0.49	0.50	0.59	0.64	0.89
Paravelia bullialata	0.00	25.00	-	40.63	0.02	2.98	0.00	0.00	0.23	0.50	0.59	0.75	0.93
Paravelia conata	0.50	14.00	-	86.07	0.00	6.87	0.04	0.06	0.40	0.50	0.50	0.96	0.90
Rhagovelia brunae	**0.35**	**13.00**	**z+**	**42.90**	**0.00**	**7.11**	**0.10**	**0.20**	**0.34**	**0.36**	**0.36**	**1.00**	**1.00**
Rhagovelia elegans	0.50	58.00	-	56.90	0.24	0.70	0.00	0.00	0.00	0.36	0.36	0.62	0.59
Rhagovelia evidis	**0.04**	**47.00**	**z-**	**71.47**	**0.00**	**6.46**	**0.00**	**0.00**	**0.02**	**0.06**	**0.07**	**1.00**	**1.00**
Rhagovelia humboldti	0.50	8.00	-	24.24	0.00	4.66	0.03	0.03	0.07	0.40	0.50	0.98	0.93
Rhagovelia jubata	**0.04**	**29.00**	**z-**	**46.81**	**0.00**	**3.90**	**0.00**	**0.00**	**0.04**	**0.30**	**0.31**	**0.99**	**0.97**
Stridulivelia alia	0.00	22.00	-	55.37	0.06	1.93	0.00	0.00	0.06	0.28	0.39	0.39	0.72
Stridulivelia strigosa	**0.00**	**23.00**	**z-**	**68.21**	**0.00**	**8.82**	**0.00**	**0.00**	**0.04**	**0.08**	**0.09**	**1.00**	**1.00**
Stridulivelia tersa	**0.35**	**15.00**	**z+**	**45.45**	**0.00**	**4.76**	**0.04**	**0.05**	**0.17**	**0.35**	**0.35**	**1.00**	**0.99**
Stridulivelia transversa	**0.03**	**27.00**	**z-**	**75.60**	**0.00**	**9.05**	**0.00**	**0.02**	**0.04**	**0.08**	**0.10**	**1.00**	**1.00**
Tachygerris adamsoni	0.50	17.00	-	37.50	0.17	0.95	0.00	0.00	0.06	0.50	0.50	0.69	0.61
Tachygerris celocis	**0.34**	**41.00**	**z+**	**61.70**	**0.00**	**4.66**	**0.12**	**0.22**	**0.24**	**0.34**	**0.34**	**1.00**	**0.99**
Tachygerris opacus	0.00	12.00	-	18.03	0.12	1.06	0.00	0.00	0.15	0.59	0.61	0.65	0.63

(C) Ephemeroptera, Plecoptera e Trichoptera (EPT)

LDI (Local)

	Ponto	Frequência	"+ / -	IndVal	P	Z	5%	10%	50%	90%	95%	Pureza	Confiabilidade
Amanahyphes	0.00	20.00	-	37.14	0.04	2.49	0.00	0.00	0.13	0.24	0.25	0.90	0.82
Anacroneuria	0.00	31.00	-	46.05	0.20	0.72	0.00	0.00	0.20	0.29	0.29	0.35	0.54
Aturbina	0.00	6.00	-	27.78	0.07	3.21	0.00	0.00	0.01	0.16	0.22	0.90	0.67
Brasilocaenis	**0.15**	**11.00**	**z+**	**34.44**	**0.01**	**4.79**	**0.13**	**0.13**	**0.15**	**0.18**	**0.21**	**1.00**	**0.99**
Callibaetis	0.00	8.00	-	23.60	0.24	0.52	0.00	0.00	0.12	0.27	0.29	0.46	0.32
Campsurus	0.00	18.00	-	57.35	0.02	2.58	0.00	0.00	0.01	0.30	0.37	0.87	0.86
Campylocia	0.37	63.00	-	76.65	0.06	1.66	0.00	0.02	0.15	0.37	0.37	0.58	0.74
Cernotina	0.29	41.00	-	63.16	0.06	1.66	0.00	0.05	0.26	0.29	0.29	0.35	0.51
Chimarra	**0.13**	**31.00**	**z-**	**54.90**	**0.00**	**3.17**	**0.00**	**0.00**	**0.08**	**0.21**	**0.21**	**0.99**	**0.96**
Cloeodes	0.00	11.00	-	36.40	0.01	3.62	0.00	0.00	0.01	0.28	0.30	0.86	0.85
Cyrnellus	0.29	11.00	-	16.43	0.21	0.70	0.00	0.04	0.14	0.22	0.27	0.45	0.44
Enderleina	**0.00**	**8.00**	**z-**	**92.31**	**0.00**	**7.87**	**0.00**	**0.00**	**0.04**	**0.09**	**0.11**	**1.00**	**0.98**
Farrodes	0.37	36.00	-	48.18	0.03	2.35	0.00	0.12	0.14	0.24	0.27	0.66	0.83
Hagenulopsis	0.22	11.00	-	25.19	0.05	1.71	0.00	0.00	0.21	0.27	0.29	0.78	0.63
Helicopsyche	**0.00**	**39.00**	**z-**	**71.72**	**0.00**	**7.15**	**0.01**	**0.03**	**0.09**	**0.13**	**0.13**	**1.00**	**1.00**
Leptonema	0.00	48.00	-	55.61	0.13	1.23	0.00	0.04	0.21	0.27	0.29	0.73	0.55
Macrogynoplax	**0.24**	**45.00**	**z-**	**65.13**	**0.00**	**4.38**	**0.09**	**0.13**	**0.14**	**0.24**	**0.25**	**0.99**	**0.97**
Macronema	0.37	60.00	-	67.12	0.09	1.71	0.02	0.03	0.28	0.29	0.37	0.54	0.83
Macrostemum	0.00	39.00	-	60.86	0.02	2.38	0.00	0.02	0.13	0.20	0.22	0.94	0.83
Marilia	0.00	12.00	-	35.36	0.06	2.34	0.00	0.00	0.05	0.17	0.28	0.82	0.63
Miroculis	0.24	63.00	-	63.83	0.17	1.10	0.00	0.01	0.18	0.29	0.36	0.51	0.70
Mortoniella	0.04	5.00	-	15.77	0.02	2.52	0.00	0.00	0.04	0.28	0.29	0.84	0.66
Nectopsyche	0.00	12.00	-	24.34	0.03	2.82	0.00	0.00	0.08	0.29	0.29	0.81	0.88
Oecetis	**0.00**	**35.00**	**z-**	**64.63**	**0.00**	**7.19**	**0.02**	**0.02**	**0.08**	**0.11**	**0.13**	**1.00**	**1.00**
Phylloicus	0.24	57.00	-	67.52	0.01	3.12	0.00	0.01	0.23	0.25	0.27	0.97	0.92
Polyplectropus	0.00	26.00	-	48.22	0.02	2.67	0.00	0.00	0.08	0.28	0.29	0.86	0.73
Simothraulopsis	0.00	10.00	-	19.23	0.06	1.39	0.00	0.00	0.03	0.24	0.25	0.74	0.63
Smicridea	0.00	39.00	-	46.18	0.07	1.63	0.04	0.08	0.18	0.28	0.29	0.85	0.70
Triplectides	0.37	50.00	-	56.42	0.05	2.14	0.00	0.00	0.12	0.27	0.29	0.84	0.77

Ulmeritoides	**0.22**	**37.00**	**z+**	**66.34**	**0.00**	**4.02**	**0.01**	**0.05**	**0.15**	**0.27**	**0.27**	**1.00**	**0.98**
Waltzoyphius	0.30	9.00	-	37.80	0.12	2.36	0.00	0.01	0.29	0.31	0.42	0.81	0.58
Zelusia	**0.24**	**28.00**	**z-**	**50.72**	**0.01**	**4.00**	**0.00**	**0.01**	**0.21**	**0.25**	**0.26**	**1.00**	**0.99**
CDI (Drenagem)													
Amanahyphes	0.39	20.00	-	33.33	0.22	0.93	0.00	0.00	0.03	0.36	0.37	0.81	0.57
Anacroneuria	0.00	31.00	-	80.02	0.02	3.25	0.00	0.00	0.35	0.37	0.37	0.89	0.81
Aturbina	0.35	6.00	-	10.44	0.28	0.80	0.00	0.00	0.00	0.33	0.33	0.70	0.59
Brasilocaenis	0.50	11.00	-	26.29	0.01	3.63	0.03	0.05	0.08	0.59	0.61	0.93	0.91
Callibaetis	0.50	8.00	-	64.50	0.01	5.49	0.00	0.09	0.41	0.59	0.59	0.91	0.78
Campsurus	0.00	18.00	-	31.63	0.02	2.80	0.00	0.00	0.00	0.30	0.37	0.91	0.79
Campylocia	0.00	63.00	-	58.52	0.16	1.05	0.00	0.00	0.32	0.50	0.59	0.65	0.54
Cernotina	0.50	41.00	-	55.70	0.06	1.54	0.00	0.02	0.36	0.59	0.59	0.82	0.67
Chimarra	0.00	31.00	-	58.32	0.06	1.81	0.00	0.00	0.18	0.32	0.33	0.81	0.63
Cloeodes	0.00	11.00	-	22.00	0.07	1.60	0.00	0.00	0.06	0.30	0.31	0.92	0.65
Cyrnellus	0.50	11.00	-	16.40	0.20	0.76	0.00	0.05	0.22	0.50	0.59	0.51	0.70
Enderleina	**0.02**	**8.00**	**z-**	**27.59**	**0.00**	**5.42**	**0.00**	**0.00**	**0.02**	**0.03**	**0.04**	**1.00**	**0.98**
Farrodes	0.37	36.00	-	52.35	0.06	1.83	0.00	0.02	0.27	0.37	0.37	0.67	0.70
Hagenulopsis	0.27	11.00	-	27.11	0.01	2.76	0.00	0.00	0.28	0.34	0.37	0.80	0.63
Helicopsyche	**0.06**	**39.00**	**z-**	**79.44**	**0.00**	**7.75**	**0.02**	**0.04**	**0.05**	**0.09**	**0.15**	**1.00**	**1.00**
Leptonema	0.50	48.00	-	49.70	0.27	0.58	0.00	0.06	0.27	0.39	0.50	0.53	0.51
Macrogynoplax	**0.08**	**45.00**	**z-**	**70.79**	**0.00**	**5.63**	**0.06**	**0.07**	**0.08**	**0.19**	**0.35**	**0.99**	**1.00**
Macronema	0.36	60.00	-	60.86	0.17	0.92	0.06	0.13	0.36	0.59	0.59	0.53	0.70
Macrostemum	0.00	39.00	-	62.69	0.01	2.38	0.00	0.00	0.04	0.33	0.34	0.89	0.78
Marilia	0.00	12.00	-	42.57	0.00	5.59	0.00	0.00	0.00	0.05	0.07	0.96	0.88
Miroculis	0.41	63.00	-	61.71	0.06	1.82	0.00	0.00	0.22	0.35	0.39	0.90	0.75
Mortoniella	0.00	5.00	-	13.86	0.02	1.88	0.00	0.00	0.00	0.36	0.37	0.87	0.65
Nectopsyche	0.01	12.00	-	26.52	0.01	3.44	0.00	0.00	0.00	0.05	0.07	0.99	0.94
Oecetis	**0.02**	**35.00**	**z-**	**65.52**	**0.00**	**6.22**	**0.00**	**0.00**	**0.02**	**0.07**	**0.08**	**0.98**	**1.00**
Phylloicus	**0.50**	**57.00**	**z-**	**65.40**	**0.00**	**3.82**	**0.02**	**0.06**	**0.08**	**0.36**	**0.37**	**0.97**	**0.98**
Polyplectropus	**0.00**	**26.00**	**z-**	**50.46**	**0.00**	**3.37**	**0.02**	**0.07**	**0.20**	**0.23**	**0.25**	**0.98**	**0.96**

Simothraulopsis	0.36	10.00	-	15.92	0.13	0.99	0.00	0.00	0.07	0.26	0.34	0.74	0.48
Smicridea	0.00	39.00	-	46.59	0.12	1.31	0.00	0.00	0.08	0.36	0.40	0.58	0.48
Triplectides	0.00	50.00	-	61.91	0.03	2.25	0.00	0.00	0.00	0.40	0.41	0.84	0.79
Ulmeritoides	**0.50**	**37.00**	**z+**	**71.52**	**0.00**	**6.01**	**0.03**	**0.03**	**0.08**	**0.34**	**0.39**	**1.00**	**1.00**
Waltzoyphius	0.00	9.00	-	16.07	0.15	0.47	0.00	0.00	0.02	0.40	0.41	0.53	0.39
Zelusia	0.00	28.00	-	85.79	0.00	4.29	0.00	0.00	0.03	0.19	0.37	0.94	0.88

5. CONCLUSÃO GERAL

Na primeira sessão, nós exploramos os efeitos do uso do solo na condição ambiental, heterogeneidade ambiental e diversidade biológica de EPT. Nós encontramos que o uso do solo influencia de forma distinta a diversidade alfa e beta dos gêneros de EPT e que determinado uso do solo têm maior impacto do que outros. Enquanto os usos do solo alteraram as características ambientais do riacho, a heterogeneidade ambiental foi maior entre os riachos localizados em área de pecuária. Todos os usos do solo resultaram em um declínio na diversidade alfa de EPT em relação aos locais de controle, mas a diversidade beta foi geralmente mais alta entre os riachos influenciados por pecuária.

Na segunda sessão, avaliamos a contribuição do componente ambiental e espacial na estruturação da metacomunidade de EPT. O componente espacial explicou a maior parte da variação de EPT, mas o ambiente espacialmente estruturado também foi importante na estruturação dessa metacomunidade. Os gêneros *Amanahyphes*, *Anacroneuria*, *Americabaetis*, *Askola* e *Callibaetis* responderam ao ambiente espacialmente estruturado em múltiplas escalas espaciais. As variáveis ambientais relacionadas à madeira no canal, dossel, substrato, oxigêncio dissolvido e proporção de silte foram importantes na estruturação desses gêneros em fina e ampla escala, mas as relações mais fortes foram observadas entre escala média (~ 217 km) e fina (até 93 km).

Na terceira sessão, buscamos identificar o ponto de mudança da comunidade e das espécies de insetos aquáticos em gradiente de distúrbio local de na drenagem. Nossos resultados suportam o cenário de que o distúrbio antrópico, principalmente relacionados aos distintos usos do solo (e.g. culturas de ciclo longo e pastagem) e exploração dos recursos naturais exerce fortes efeitos sobre as comunidades de insetos aquáticos em riachos de pequeno porte. Os riachos estudados estão sujeitos a níveis de distúrbios acima da condição ideal para as espécies, uma vez que os limiares foram identificados para as comunidades de Odonata, Gerromorpha e EPT. Além disso, as espécies de insetos aquáticos estão diminuindo em frequência e abundância com o aumento do distúrbio antrópicos na escala local e na área de drenagem dos riachos, mas tem espécies que ainda mantém a frequência e abundância nesses gradientes.

Baseado nos resultados encontrados, nós sugerimos que: (i) os estudos que avaliam as atividades antrópicas devem incluir as características ambientais locais; (ii) é importante considerar o contexto histórico da mudança do uso da terra ao avaliar os impactos do uso do solo na biodiversidade de insetos aquáticos; (iii) estudos futuros que visem avaliar diferentes escalas espaciais considerem a inclusão de um método que capture a resposta individual dos táxons; (iv) tanto as resposta da comunidade como as das espécies deve ser consideradas no planejamento de gestão dos recursos hídricos.

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