



The anthropic gradient determines the taxonomic diversity of aquatic insects in Amazonian streams

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Abstract Anthropic activities are one of the main drivers of change in the environmental characteristics of streams and the diversity of aquatic macroinvertebrates. We evaluated the influence of an anthropic gradient (varying degrees of impact) on the general level alpha and beta diversity of the Ephemeroptera, Plecoptera, and Trichoptera (EPT) in 48 eastern Amazonian streams. These insects were sampled using a dipnet, and the anthropic gradient was represented by the activities observed in the channel

and the catchment of each stream. We found that increasing anthropic impact reduced the alpha diversity of the EPT. The Total Beta Diversity (BD_{Total}) showed a moderate degree of variation in streams. The streams with the greatest Local Contribution to Beta Diversity (LCBD) had the lowest alpha diversity of EPT, while the genera with the greatest Species Contribution to Beta Diversity (SCBD) were the most abundant and widely distributed among the streams. Thus, the increase in anthropic impacts reduced the alpha diversity of the EPT and indirectly influenced the uniqueness, emphasizing the importance of using different components of the diversity to understand the effects of anthropic impacts on Amazonian streams.

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Introduction

The transformation of natural environments into an anthropic landscape also affects aquatic systems, leading to changes in environmental conditions and decreasing species diversity (Masese et al., 2009; Jun et al., 2011; Törnblom et al., 2011; Siqueira et al., 2015; Fierro et al., 2017). This is especially true for worldwide freshwater systems, which are considered the most threatened ecosystems (Malmqvist & Rundle, 2002; Strayer & Dudgeon, 2010). Land use in the drainage basin (catchment) of low-order streams alters the landscape (Allan, 2004), increasing the input of sediments (Törnblom et al., 2011; Taniwaki et al., 2019), reducing microhabitats and organic matter availability (Miserendino & Masi, 2010; Jun et al., 2011), changing dissolved oxygen levels and water temperatures (Brand & Miserendino, 2015; Taniwaki et al., 2019).

The effects of anthropic activities can be investigated by measuring different aspects of the biodiversity of the aquatic system (Ligeiro et al., 2013; Siqueira et al., 2015; Cunha & Juen, 2017; Fierro et al., 2017). Biological diversity can be measured at different levels of organization, that is, alpha (α)-diversity, which represents the diversity (species richness) at a given site, beta (β)-diversity, which corresponds to the variation among sites, and gamma (γ)-diversity, which measures the total diversity found in a region (Whittaker 1960, 1972). Measures of α -diversity have been commonly used as ecological indicators of local environmental conditions (Heino, 2009), and are widely-used in decision-making for the establishment of conservation targets (Myers et al., 2000). Given this, a large number of studies have focused on the anthropic impact on the α -diversity of benthic macroinvertebrates (Genito et al., 2002; Jun et al., 2011; Ligeiro et al., 2013; Ramezani et al., 2016; Alemneh et al., 2017). These studies have shown a negative relationship between the α -diversity of benthic macroinvertebrates and strong anthropic impacts (Genito et al., 2002; Jun et al., 2011; Ramezani et al., 2016).

The β -diversity reflects the way species respond to the environmental variation along an ecological gradient (Heino, 2009) and spatial distribution (Rocha et al., 2018). Previous studies have assessed β -diversity in a number of different ways using, for example, additive (Lande, 1996; Veech et al., 2002) and multiplicative partitioning (Jost, 2007) or the decomposition of dissimilarities into their nested and turnover components (Baselga, 2010). Legendre & De Cáceres (2013) recently proposed an approach that decomposes the total variation found in a community (Total Beta Diversity— BD_{Total}) into the Local Contribution to Beta Diversity (LCBD) and the Species Contribution to Beta Diversity (SCBD). The LCBD represents the degree of ecological uniqueness of a given site in comparison with all the other sites sampled in a given region, while the SCBD represents the relative contribution of each species to the observed patterns of β -diversity (Legendre & De Cáceres, 2013; Heino & Grönroos, 2017). Despite the growing number of studies that have evaluated the LCBD at different scales of time and space (Mimouni et al., 2015; Heino & Grönroos, 2017; Kong et al., 2017; Teittinen et al., 2017; Vilmi et al., 2017; Sieglöch et al., 2018; Sor et al., 2018; Li et al., 2020a), only a few studies have evaluated the influence of anthropic activities on the LCBD (Simões et al., 2015; Bourassa et al., 2017; Heino et al., 2017; Leão et al., 2020; Li et al., 2020b).

Empirical studies have found either a negative or a positive relationship between the LCBD and anthropic activities (Simões et al., 2015; Bourassa et al., 2017; Heino et al., 2017; Borges et al., 2020; Leão et al., 2020; Li et al., 2020b). For example, the LCBD is lower when the quality of the environmental conditions decreases with increasing anthropic effects (Bourassa et al., 2017; Borges et al., 2020; Li et al., 2020b), which means that the community is composed primarily of species that are relatively less specialized and more tolerant to environmental change (Bourassa et al., 2017; Borges et al., 2020). However, it is important to note that high LCBD values reflect the presence of unique species in the community due to specific ecological conditions in either preserved or altered systems (Legendre & De Cáceres, 2013). The LCBD may also have either a positive or negative relationship with α -diversity (Legendre & De Cáceres, 2013). A positive relationship between the LCBD and α -diversity (Kong et al., 2017; Pajunen et al., 2017)

may occur where streams with low levels of anthropic activity have a relative abundance of microhabitats and resources (Miserendino & Masi, 2010; Jun et al., 2011), which support a larger number of species with distinct ecological niches (Beisel et al., 2000), which can harbor unusual species that contribute to increasing LCBD in these sites (Legendre & De Cáceres, 2013). However, a negative relationship between the LCBD and α -diversity, would suggest that, despite having lower α -diversity altered sites may have distinct (rare) and tolerant species that make a greater contribution to β -diversity (Heino et al., 2017; Pajunen et al., 2017). The SCBD may be associated with certain characteristics of the species, such as their abundance and occupancy (Heino & Grönroos, 2017; Vilmi et al., 2017; Da Silva et al., 2018). If, for example, species abundance varies considerably among sites, this will increase β -diversity (Heino & Grönroos, 2017). On the other hand, species that are well distributed among sites, will tend to increase β -diversity (Heino & Grönroos, 2017). This variation in abundance and distribution has a major effect on the β -diversity values, which reinforces the importance of the systematic evaluation of the mechanisms and parameters that determine this variation.

To understand how aquatic biodiversity responds to anthropic disturbance, we used a dataset on genera level Ephemeroptera, Plecoptera, and Trichoptera (known collectively as the EPT) considered to be reliable environmental indicators (Ligeiro et al., 2013; Ramezani et al., 2016; De Faria et al., 2017; Luiza-Andrade et al., 2017). The EPT species also occupy distinct substrates (e.g., Fidelis et al., 2008) and feed on different resources (e.g., Cummins & Klug, 1979; Ceneviva-Bastos et al., 2017). The present study evaluated the influence of varying degrees of anthropic activity (the anthropic gradient) on the structure of the EPT community. Three hypotheses were tested: (i) streams influenced by more intense anthropic activity have a lower α -diversity of the EPT and a lower Local Contribution to β -diversity (LCBD); (ii) a positive relationship exists between the LCBD and the α -diversity of the EPT, and (iii) a positive relationship exists between the Species Contribution to β -diversity (SCBD) and the total abundance and occupancy of the EPT.

Materials and Methods

Study area

The present study focused on 48 streams located in the basin of the Capim River in the northeastern portion of the state of Pará, Brazil (Fig. 1). Most of the streams ($n = 35$) were distributed along a known gradient of anthropogenic impact, while the others ($n = 13$) were located in forested areas. The human colonization of the Capim basin became more intense in the 1960s, due to government incentives for the occupation of land in the Amazon region (e.g., Lui & Molina, 2009; Pinto et al., 2009). This occupation resulted in a mosaic landscape dominated by forest fragments, agriculture, silviculture (e.g., *Eucalyptus* spp.), logging, pasture, and bauxite and kaolin mining (Pinto et al., 2009; Dias et al., 2018). The region's climate is humid tropical (Peel et al., 2007) with a mean annual precipitation of 1,800 mm (Embrapa, 1986) and relative humidity of 81% (Pinto et al., 2009). Precipitation peaks between December and May, while the dry season lasts from June to November (Embrapa, 1986).

Anthropic gradient

The anthropic gradient was described using an Integrated Disturbance Index (IDI) which is defined by combining the anthropic activities in the riparian vegetation (Local Disturbance Index—LDI) and the proportion of Land Use and Land Cover (called LULC from here) in the catchment area (Catchment Disturbance Index—CDI) of each stream (Ligeiro et al., 2013). The LDI summarizes the presence and proximity of 11 categories of anthropic activity (crops, pasture, dams and dykes, buildings, pavement, roads, pipes, landfill/trash, parks/gardens, logging, and mining) within the channel and riparian vegetation within 11 cross-sections established at each stream (Kaufmann et al., 1999; Peck et al., 2006). Further details of the survey design are provided in the Characterization of the environment section, below. The values attributed to each anthropic activity were weighted according to their proximity to the channel, receiving a weight of 1.5 when present in the channel, 1.0 when present within the riparian vegetation (10 m \times 10 m), and 0.667 when outside the riparian vegetation (Kaufmann et al., 1999).

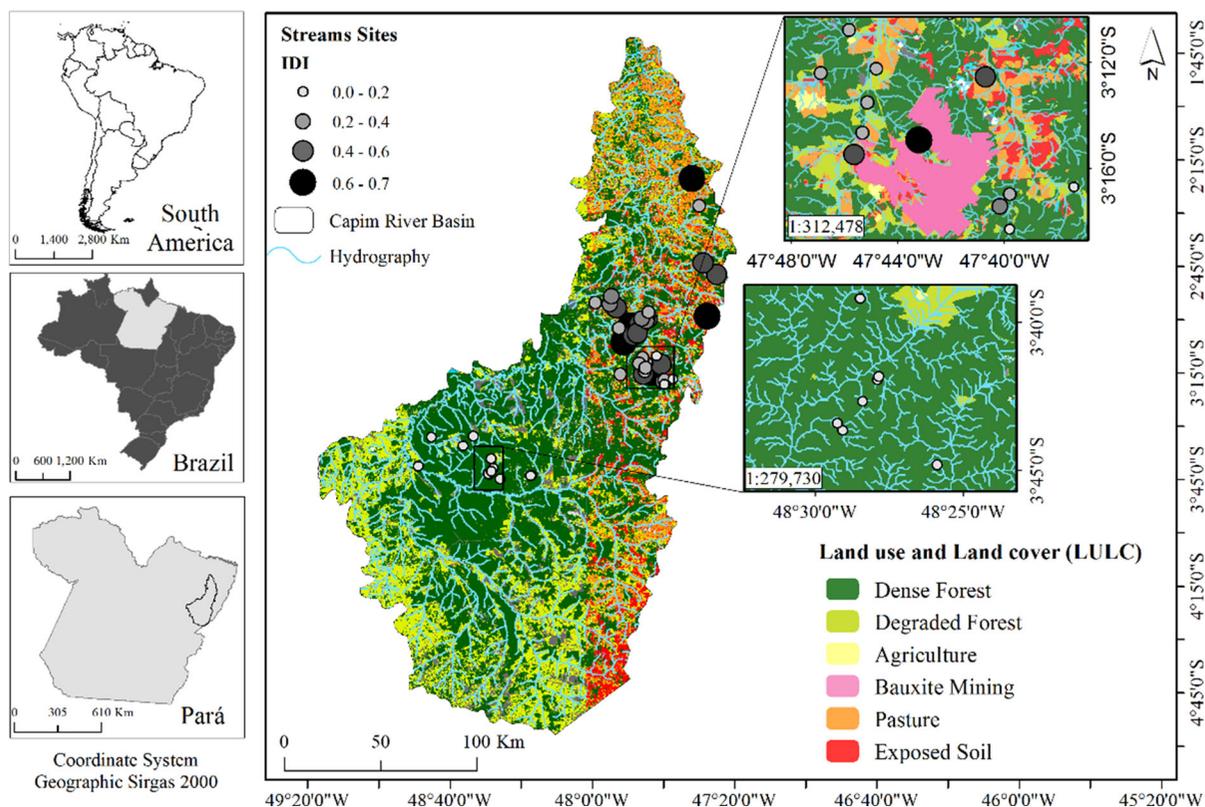


Fig. 1 Land Use and Land Cover (LULC) of 48 streams distributed within the Capim River basin in the northeastern state of Pará, Brazil. The circle size represents the anthropic

gradient on stream (IDI—Integrated Disturbance Index). Streams in forest areas (no impact) have IDI = 0, while streams with higher values have stronger anthropic activities

The CDI was calculated based on the LULC recorded for the catchment area, calculated from the digital elevation model with a spatial resolution of 30 m (Shuttle Radar Topography Mission—SRTM) using the QGIS geoprocessing software (version 2.14). The LULC was mapped through the interpretation of multispectral images of the REIS (RapidEye Earth Imaging System, <http://geocatalogo.mma.gov.br/>), which were acquired for the months in which the samples were collected during the study. The REIS images are 77 km wide and 1,500 km in length, with a spatial resolution of 5 m and spectral resolution of five bands. This fine spatial resolution delimits clearly the shape and texture of the elements, while the spectral resolution ensures a high mapping accuracy for the targets.

The images were submitted to three adjustments before being classified: (i) atmospheric correction, (ii) the creation of a mosaic of the images, and (iii) mosaic segmentation. The supervised classification was based

on the selection of samples (segments) from the image according to their texture, form, tonality, and color (De Almeida et al., 2016). The samples were grouped into five LULC classes: (i) dense forest (natural vegetation), (ii) degraded forest (forest fragments formed by predatory logging), (iii) pasture (extensive and/or intensive production of livestock), (iv) exposed soil, and (v) bauxite mining (see De Almeida et al., 2016; Montag et al., 2019; Leão et al., 2020). The LULC classification was validated using images from the TerraClass Project (2014), which were provided by the Brazilian National Institute of Space Research (INPE, <http://www.inpe.br/>). The percentage of each LULC class was measured within each catchment area and the weighted percentages of land use in the catchment were used to calculate the CDI (Online Resource, Table S1). The different types of land use were weighted according to the degree of anthropic change in environmental conditions (e.g., López et al., 1998; Miserendino & Masi, 2010; Eludoyin et al.,

2017). That is, areas of bauxite mining and exposed soil were weighted more than pasture, which was, in turn, weighted more than degraded forest ($CDI = 4^x \% \text{ bauxite mining and exposed soil} + 2^x \% \text{ pasture} + 1^x \% \text{ degraded forest}$).

The anthropic gradient (IDI) across the streams was defined by combining the LDI and CDI values (Ligeiro et al., 2013). For this, as the different indices were measured on different numerical scales, each index was first standardized to 75% of its maximum value (Ligeiro et al., 2013). In this case, the LDI values were standardized by dividing them by 5, given the empirical maximum score of 7, while the CDI values were divided by 300, given the empirical maximum of 400 (i.e., if the forest of the entire catchment had been cleared). The IDI was then calculated as $[(LDI/5)^2 + (CDI/300)^2]^{1/2}$, where streams with higher values have greater anthropic activity (Ligeiro et al., 2013) (Online Resource, Table S1).

Characterization of the environment

The study sites ranged from first to third order streams, based on Strahler's (1957) classification, with a mean width of 3 m (± 1 m, standard deviation) and a mean depth of 0.22 m (± 12 m). Environmental variables were measured along a 150 m reach, which was subdivided into 10 continuous sections spaced at intervals of 15 m, with 11 cross-sections along the entire reach (following Peck et al., 2006). At each of the 11 cross-sections, the total organic matter (percentage of roots, fine litter, coarse litter, woody debris, and filamentous algae) was measured visually, and the total immersion (percentage to which substrate is covered into the silt, sand or mud on the stream bottom) was determined by five equidistant measurements in each transect. The margin canopy cover of the cross-sections was also characterized at two points (left and right margins) of the stream channel using a spherical densiometer positioned at a height of 0.30 cm above the surface of the water. The wetted width corresponds to the cross-section of the stream, and the limits of the water at its margins, which was measured in each transect using a 1.5-m-long rigid ruler (a graded piece of plastic pipe).

The mean water temperature ($^{\circ}\text{C}$), dissolved oxygen concentration (mg/l), and pH were measured using a multiparameter Horiba device in three equidistant sections of the stream (downstream, middle and

upstream). The amount of glide habitat (slow-moving water) was also quantified visually in each of the 10 sections. Eight environmental parameters were calculated based on these measurements, following Kaufmann et al. (1999): wetted width (m), total immersion (%), total organic matter (%), margin canopy cover (%), glide (%), water temperature ($^{\circ}\text{C}$), dissolved oxygen (mg/l), and pH. A Pearson correlation analysis was then conducted, and three of the variables that were correlated significantly were omitted prior to the subsequent analyses, leaving only: wetted width, total organic matter, margin canopy cover, water temperature, and dissolved oxygen.

Sampling of the EPT

The EPT were sampled during the Amazonian dry season to guarantee the greatest possible diversity of aquatic insects (Bispo et al., 2001; Silva et al., 2018). As the water flow is reduced, the aquatic environment becomes more stable during the dry season, and the drift of aquatic organisms is also reduced (Bispo et al., 2001). For the EPT sampling, each stream section was subdivided into three longitudinal 5-m segments. These segments were evaluated visually to guarantee the collection of all the different types of substrates (e.g., root, sand, fine organic debris, leaf banks, woody debris, and macrophytes). The specimens were collected using a dipnet (18 cm in diameter with a 0.05 mm mesh) in two portions of the substrate in the first two segments of each section, with a total of 20 segments being sampled per stream (Shimano et al., 2018). The third segment of each longitudinal section of the stream was used to access the substrate of the next section, to avoid trampling this section prior to sampling. The specimens collected in each segment were grouped into a single sample representing the stream.

The specimens were separated from the substrate in the field and then fixed in 85% ethanol. The EPT were identified to the genus level using the taxonomic keys of Domínguez et al. (2006), Salles & Domínguez (2012), Hamada & Silva (2014), Pes et al. (2014), Salles & Lima (2014), and Salles et al. (2014). Previous studies have shown that a genus level taxonomic resolution is adequate to determine the response of aquatic insects to environmental changes caused by anthropic activities in streams of the Brazilian Amazon and Cerrado biomes (De Faria et al., 2017; Luiza-

Andrade et al., 2017; Godoy et al., 2019). Following identification, the specimens were deposited in the Aquatic Insects collection of the de Ecology and Conservation Laboratory (LABECO) at the Federal University of Pará (UFPA) in Belém, Brazil.

Data analysis

We use Principal Coordinates Analysis (PCoA, Gower, 1966) to visualize the ordination of streams based on environmental variables (wetted width, total organic matter, margin canopy cover, water temperature, and dissolved oxygen) in unconstrained ordination space. The euclidean distance matrix was built from standardized environmental variables. Then, the *envfit* function (permutations = 999) was used to relate the environmental variables to the PCoA axes. Only the significant environmental variables ($P < 0.05$) were adjusted in the PCoA ordination.

The α -diversity of the EPT was calculated as the genera richness per site. The total abundance of each genus was the total number of individuals present in all the streams, while occupancy was assessed by calculating the number of streams occupied by each genus (Heino & Grönroos, 2017; Da Silva et al., 2018). The LCBD and SCBD components were calculated from the total β -diversity (BD_{Total}) of the study area (Legendre & De Cáceres, 2013). For this, the composition matrix of EPT genera was first adjusted by Hellinger's transformation. The BD_{Total} was partitioned into the Local Contribution to β -diversity (LCBD) and the Species Contribution to β -diversity (SCBD), following Legendre & De Cáceres (2013). The streams and genera with above average LCBD and SCBD values, respectively, were considered to be those that most contributed to the BD_{Total} (Mimouni et al., 2015; Sor et al., 2018; Leão et al., 2020).

A Generalized Linear Model (GLM) with a negative binomial distribution, which is appropriate for overdispersed data (Zuur et al., 2009), was used to assess whether streams influenced by more intense anthropic activity have a lower α -diversity of EPT (hypothesis i). The model was validated visually by the simulated envelope approach (Moral et al., 2017). A beta regression (Cribari-Neto & Zeileis, 2010) with logit link function was used to assess whether streams influenced by more intense anthropic activities have a lower Local Contribution to β -diversity (LCBD) (hypothesis i), whether the LCBD has a positive

relationship with the α -diversity of the EPT (hypothesis ii), and whether the Species Contribution to β -diversity (SCBD) has a positive relationship with total EPT abundance and occupancy (hypothesis iii). The beta regression is the most appropriate for response data that vary between 0 and 1, as in the case of both the LCBD and SCBD, because this model is naturally heteroskedastic (for further details, see Cribari-Neto & Zeileis, 2010).

Moran's eigenvector maps based on distance, dbMEM (Dray et al., 2006), were used to evaluate and control for the possible influence of spatial structuring on the environmental variables (wetted width, total organic matter, margin canopy cover, water temperature, and dissolved oxygen) and EPT community (α -diversity and LCBD). The eigenvectors in this analysis were calculated from the Euclidean distance matrix (latitude and longitude) (Borcard & Legendre, 2002; Dray et al., 2006) and only positive eigenvectors were selected as spatial proxies (Borcard & Legendre, 2002). The relationships between spatial factors and environmental variables, α -diversity, and the LCBD of the EPT, were evaluated using the forward selection method (Blanchet et al., 2008), GLM, and beta regression, respectively.

Each stream sampled during the present study was considered to be a sample unit. The analyses were run in the R software (R Core Team, 2018), using the *betareg* (Cribari-Neto & Zeileis, 2010), *MASS* (Venables & Ripley, 2002), *vegan* (Oksanen et al., 2019), *hnp* (Moral et al., 2017), and *adespatial* packages (Dray et al., 2018).

Results

Environmental structure

In the PCoA, the first two axes explained 58% of environmental variation, with 37% on the first axis and 21% on the second (Fig. 2). The ordering of the PCoA axes was correlated with wetted width ($r^2 = 0.61$, $P = 0.001$), total organic matter ($r^2 = 0.42$, $P = 0.001$), margin canopy cover ($r^2 = 0.65$, $P = 0.001$), water temperature ($r^2 = 0.72$, $P = 0.001$) and dissolved oxygen ($r^2 = 0.47$, $P = 0.001$). When observing the streams' ordination, the most impacted have a higher water temperature and dissolved oxygen, greater wetted width and lower proportion of

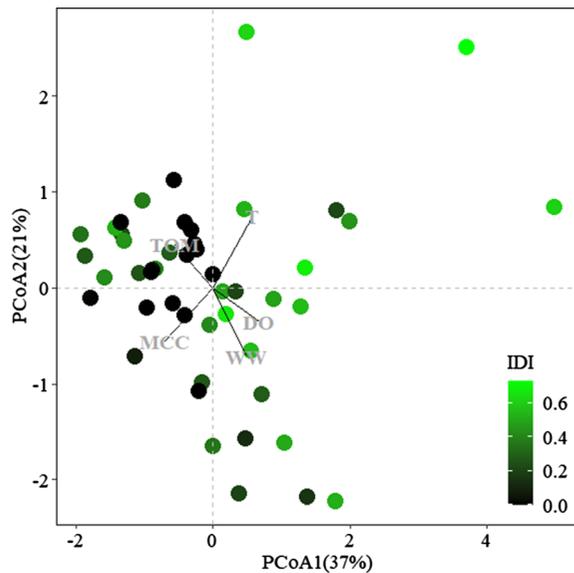


Fig. 2 Principal Coordinates Analysis (PCoA) ordination based on environmental variables each stream related to the anthropic gradient (IDI). The points correspond to the 48 streams sampled within the Capim River basin in the northeastern state of Pará, Brazil. Black arrows indicate the environmental variables significantly correlated with the ordination. Abbreviation of environmental variables: TOM (Total Organic Matter), MCC (Margin Canopy Cover), WW (Wetted Width), T (Temperature) and DO (Dissolved Oxygen)

margin canopy cover and total organic matter (Fig. 2). The relationship between spatial factors and environmental variables indicated that only the margin canopy cover was influenced significantly by broad-scale spatial factors ($r^2 = 0.119$, $P = 0.014$).

The EPT community

A total of 8,609 EPT specimens were collected, representing 18 families and 47 genera (Online Resource, Table S2). The order Ephemeroptera was represented by 4,287 individuals (mean \pm standard deviation, 89.313 ± 84.221), followed by the Trichoptera, with 4,042 individuals (84.208 ± 134.403), and the Plecoptera, with 280 individuals (5.833 ± 9.926). The α -diversity of the EPT was negatively related to the anthropic gradient (IDI) (Estimate = -0.723 , $z = -2.006$, $P = 0.045$) (Fig. 3) and was significantly influenced by broad-scale spatial factor (Estimate = 0.289 , $z = 4.056$, $P < 0.001$).

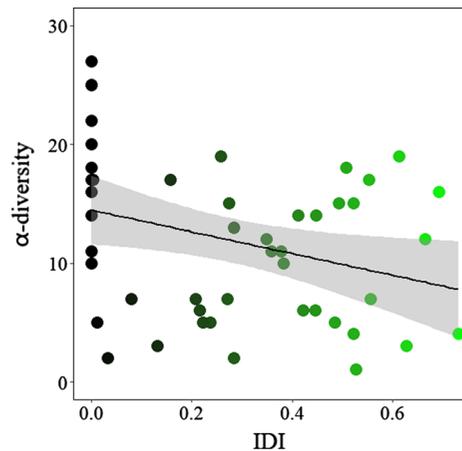


Fig. 3 Relationship between EPT α -diversity and anthropic gradient (IDI). Color gray represent the confidence interval of 0.95 for the linear model

The BD_{Total} was 0.57 and the SS_{Total} was 26.59. The LCB values ranged from 0.010 to 0.048 (Online Resource, Table S1). Nineteen streams had above average (> 0.021) LCB values (Fig. 4A) of which, 16 were influenced by varying degrees of anthropic activity, and only three were not impacted by anthropic activity (Online Resource, Table S1). The LCB was not related to the anthropic gradient (IDI) ($P = 0.594$), although it was negatively related to the α -diversity of EPT (Estimate = -0.039 ; $z = -4.889$; $P < 0.001$) (Fig. 5, Online Resource Table S3), although no relationship was found between the LCB and spatial factors (Online Resource, Table S4). The SCBD values ranged from 0.0001 to 0.1383 and 14 genera had an above average (> 0.0204) contribution to β -diversity (Fig. 4B). The SCBD was related positively to the total abundance (Estimate = 0.000 ; $z = 2.639$, $P = 0.008$) and occupancy (Estimate = 0.089 ; $z = 8.940$; $P < 0.001$) of the EPT (Fig. 6, Online Resource Table S3).

Discussion

The results of the present study indicated that the α -diversity of the EPT decreased with increasing anthropic impact. This response is similar to that found in other studies that have evaluated the influence of anthropic pressures on the α -diversity of aquatic insects (Genito et al., 2002; Jun et al., 2011; Ligeiro et al., 2013; Ramezani et al., 2016; Alemneh et al.,

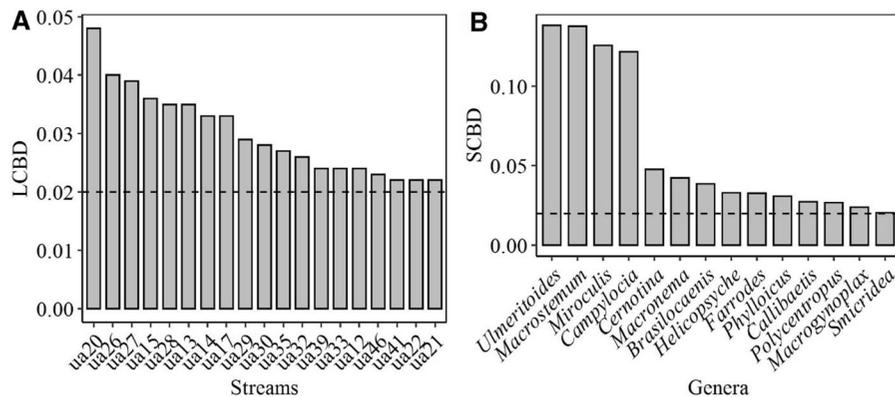


Fig. 4 Local Contribution to β -diversity (LCBD) (**A**) and (**B**) Species Contribution to β -diversity (SCBD) of only of streams and genera with above average values. The dashed line represents the average value of LCBD (0.021) and SCBD (0.0204)

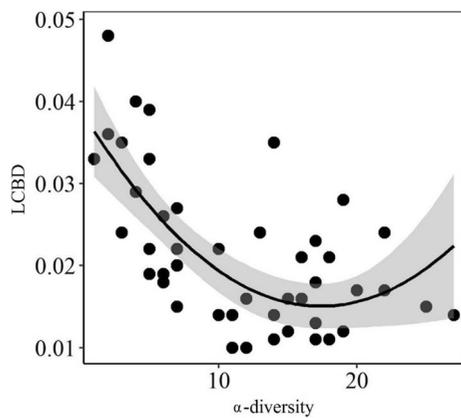


Fig. 5 Relationship between Local Contribution to β -diversity (LCBD) and EPT α -diversity. Color gray represent the confidence interval of 0.95 for the polynomial model

2017). This negative relationship was probably also due to the fact that more impacted streams had a series of characteristics, such as reduced margin canopy cover and total organic matter, and increased water temperatures, which are less suited to the ecological adaptations of many genera. Aquatic insects have morphological, physiological, and behavioral adaptations that reflect the physical features and conditions found in their habitats (Poof & Ward, 1990; Poff, 1997). However, species tend to respond differently to the anthropogenic changes that alter the characteristics of the environment (e.g., temperature, vegetation, and organic matter), with their response depending on their tolerance or ability to adapt (Poof & Ward, 1990; Poff, 1997). Thus, the loss of more favorable conditions, as

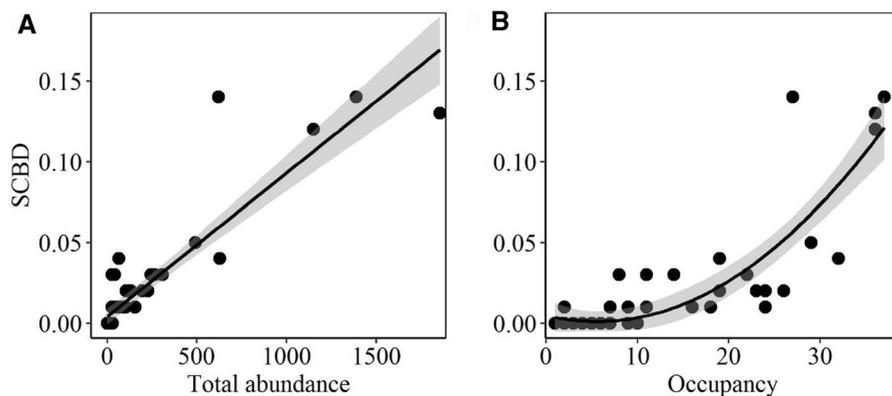


Fig. 6 Relationship between Species Contribution to β -diversity (SCBD) and total abundance (**A**) and (**B**) occupancy of EPT. Color gray represent the confidence interval of 0.95 for the linear and polynomial model, respectively

well as food resources and shelters, can provoke a reduction in the abundance of the less tolerant genera, contributing to a decrease in the α -diversity of the streams. On the other hand, the α -diversity of the EPT in the present study area was structured by broad-scale spatial factor, which is probably because the margin canopy cover has been structured by these factors, resulting in greater variation in this variable with increasing distance between streams. Li et al. (2019) obtained similar results, showing that the diversity of aquatic insects in streams is related to broad-scale spatial factors.

Contradicting the prior expectations of the present study, the LCBD of the EPT was not related to the anthropic gradient. The absence of any clear relationship between these variables may reflect the fact that all the streams were located within a single basin (Capim River), which may have resulted in the sampling of a single spectrum of anthropic impact, not strong enough to influence the uniqueness of the EPT group. Although environmental variables and land use have been identified as important factors for the LCBD of aquatic insects (Tonkin et al., 2016; Bourassa et al., 2017; Heino et al., 2017; Sor et al., 2018; Li et al., 2020a). The latter is still difficult to predict (Tonkin et al., 2016), especially in impacted streams (Leão et al., 2020). The difficulty of identifying the predictors that are most important for the LCBD may also result from the differential responses of biological groups to different predictors (Landeiro et al., 2018).

Different than expected, the LCBD was negatively related to the α -diversity of the EPT. This pattern has been reported frequently in studies of aquatic systems, which have focused on insects (Heino et al., 2017; Heino & Grönroos, 2017; Siegloch et al., 2018), fish (Legendre & De Cáceres, 2013; Kong et al., 2017; Borges et al., 2020; Leão et al., 2020), and diatoms (Pajunen et al., 2017; Teittinen et al., 2017; Vilmi et al., 2017). The data indicate that the streams with more unique species tend to have fewer species overall (Legendre & De Cáceres, 2013). In the present study, most of the streams with higher LCBD values were impacted and had a low α -diversity of EPT (1–7 genera), with at least one genus that had either an ample (e.g., *Ulmeritoides*, *Campylocia*, *Miroculis*, and *Macrostemum*) or a more restricted distribution (e.g., *Caenis*, *Callibaetis*, *Americabaetis* and *Ulmeritus*) among the streams. Overall, these findings indicate

that sites with higher LCBD values may have either lower or higher α -diversity, a pattern also reported by Kong et al. (2017) and Sor et al. (2018). This may be related to the proportion of common and rare species present in the communities, which will influence whether the relationship between the LCBD and α -diversity is positive, negative or neutral (Da Silva et al., 2018). One other factor that may account for the higher LCBD values recorded at sites with lower α -diversity is the fact that impacted streams may be occupied by species that are more tolerant of relatively harsh conditions (Pajunen et al., 2017). This is partially true in the present study, given that genera such as *Ulmeritoides*, *Campylocia*, *Miroculis* (Ephemeroptera), and *Macrostemum* (Trichoptera), which are considered to be tolerant of anthropic impacts (see Nogueira et al., 2011; De Faria et al., 2017; Cardoso et al., 2018), occurred frequently in streams with higher LCBD values, but also occurred at other sites. Regardless of whether the relationship between the α -diversity and the LCBD is negative or positive, sites with higher LCBD values should be prioritized in conservation programs, as this characteristic may indicate an urgent need for environmental management or the ecological restoration of the site (Legendre & De Cáceres, 2013; Da Silva et al., 2018).

The Species Contribution to β -Diversity (SCBD) was positively related to the total abundance and occupancy of EPT in the streams, in coincidence with previous studies (Heino & Grönroos, 2017; Vilmi et al., 2017; Szabó et al., 2019). This indicates that the genera that contributed most to β -diversity in the present study are more abundant overall and occupy streams with varying levels of environmental integrity. Given this, common species may have a considerable influence on β -diversity when their abundance varies widely between sites (Vilmi et al., 2017). The genera that contributed most to the β -diversity (SCBD) of the EPT in the present study were *Ulmeritoides*, *Miroculis*, *Campylocia* (Ephemeroptera), and *Macrostemum* (Trichoptera) which were all relatively abundant (620–1,856 individuals) and occurred in over half (56–77%) of the streams, including both impacted and undisturbed sites. The ample distribution or greater abundance of these genera in streams has been recorded in a number of previous studies (Nogueira et al., 2011; De Faria et al., 2017; Cardoso et al., 2018), and may be related with the great trophic plasticity, variability in substrates

(Fidelis et al., 2008; De Oliveira & Nessimian, 2010; Ceneviva-Bastos et al., 2017; Mboye et al., 2018), and possibly also to the varying levels of tolerance found in the species of each genus (see Poff and Ward, 1990; Poff, 1997).

Conclusion

The multiple anthropic activities that influence stream environmental structure in the basin of the Capim River have resulted in a complex scenario with some responsible for the establishment of a gradient of anthropic impact. In fact, the results of the study indicate that more intense anthropic activity reduced the α -diversity of the EPT and was indirectly related with the β -diversity (LCBD). When the structure of the EPT communities found in the study area is analyzed in terms of the Total Beta Diversity (BD_{Total}), the LCBD was negatively related to the number of genera in the streams, while the SCBD was positively related to the total abundance and genus occupancy. On the other hand, although the less abundant (rare) genera with more restricted distributions contributed less to β -diversity, they were important components of this metric, and should not be disregarded. It is also important to note that the present study was limited to the evaluation of the influence of spatial factors on α -diversity, and did not consider, for example, the dispersal capacity of the aquatic insects themselves. The findings of the present study support the combined use of α and β -diversity (that is, both the LCBD and the SCBD) for a more robust assessment of the impact of anthropic activities on the EPT community, given that each of these metrics assesses a distinct facet of community structure. Another important point is that streams with a more unique composition of genera do not always have higher α -diversity, which is important to consider when applying conservation or restoration efforts to streams with higher LCBD values.

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