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Local effects of deforestation on stream fish assemblages in the Amazon–Savannah transitional area

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The expansion of agriculture in the southeast Amazon–Savanah transitional area has greatly decreased forest cover and influenced stream structure and functioning. We assessed the effects of forest cover loss on the integrity of streams by assessing stream physical conditions and the taxonomic and functional diversity of fish assemblages in this transitional area. We hypothesize that low forest cover, especially adjacent to streams, results in poor physical conditions (*e.g.*, warmer temperatures, less physical structure, etc.), which in turn will decrease the taxonomic and functional diversity of fish assemblages. We detected that loss of forest cover negatively affects natural stream conditions and reduces the functional diversity of fish assemblages, but we did not find a strong effect on taxonomic diversity. Ambush and stalking predators, diurnal surface pickers (groups that exhibit opportunistic life history strategies), grazers, pickers, and browsers (groups that exhibit equilibrium life history strategies) were the functional groups with the strongest relation to altered environments. These groups can explore different niches, both with natural characteristics or altered by human activities. Our results suggest that the preservation of riparian zones can minimize the loss of specialized fish species in assemblages of Amazon–Savannah stream systems.

Keywords: Agricultural expansion, Environmental change, Habitat, Ichthyofauna, Land use.

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A expansão da agricultura na área de transição Amazônia-Savana diminuiu muito a cobertura florestal, influenciando a estrutura e o funcionamento dos sistemas de riachos. Avaliamos o efeito da perda de cobertura florestal na integridade dos riachos avaliando as condições físicas do ambiente e a diversidade taxonômica e funcional das assembleias de peixes nesta área de transição. Nossa hipótese é que a baixa cobertura florestal, especialmente adjacente ao riacho, resulta em más condições físicas (por exemplo, temperaturas mais quentes, e menos estrutura física) o que por sua vez diminuirá os valores da diversidade taxonômica e funcional das assembleias de peixes. Detectamos que a perda de cobertura florestal afeta negativamente as condições naturais e reduz a diversidade funcional das assembleias de peixes, mas não encontramos um efeito para a diversidade taxonômica. Predadores de emboscada e espreita, catadores de superfície diurnos, pastores, catadores e navegadores foram os grupos funcionais que tiveram a relação mais forte com ambientes alterados. Esses grupos podem explorar diferentes nichos, tanto com características naturais quanto alterados pela ação antrópica. Nossos resultados sugerem a preservação da zona ripária a fim de evitar a perda de espécies especializadas das assembleias de peixes de riachos.

Palavras-chave: Expansão agrícola, Habitat, Ictiofauna, Mudança ambiental, Uso do solo.

INTRODUCTION

The Amazon ecoregion is the largest tropical forest in the world and plays an important role in maintaining species diversity by providing food resources, shelter, different microclimates, and ecological processes, which also helps to maintain the structural integrity of habitats (Junk *et al*., 2007). South of this ecoregion is the transitional area with the Brazilian Savannah (Cerrado) ecoregion, an area that has been highly altered by the expansion of the agricultural frontier (Arc of Deforestation), which has deeply changed the region's natural characteristics (Brando *et al*., 2013; Nepstad *et al*., 2014) and contributed to the loss of physical and biotic integrity in aquatic ecosystems (Cunha *et al*., 2015; Ilha *et al*., 2019). The removal of forest cover, especially of the riparian vegetation, facilitates erosion processes and the carrying of sediment, nutrients, and pollutants to water systems (Allan, 2004; Nessimian *et al*., 2008). Forest loss also changes the flux of organic matter, channel morphology, substrate characteristics, and water physico-chemical characteristics, reducing the integrity and quality of habitats and modifying their distributions (Cunha *et al*., 2015; Leal *et al*., 2016; Juen *et al*., 2016). The changes in landscape and habitat quality also lead to biodiversity loss, both in taxonomic diversity (Montag *et al*., 2019) and functional diversity (Benone *et al*., 2020; Leão *et al*., 2020).

Quantifying the response of stream organisms to environmental change using classical measures of species diversity and abundance (taxonomic diversity) together with a functional trait approach provides the most comprehensive understanding of changes in stream function and integrity (Petchey *et al*, 2007; Batalha *et al*., 2010; Luiza-

Andrade *et al*., 2017). The functional approach refers to the morphological, behavioral, and physiological traits related to a species ability to exploit environmental resources, which can be used to understand how local communities are structured (MacArthur, Levins, 1967; Petchey *et al*., 2007; Leitão *et al*., 2018). Functional diversity becomes a link between species diversity and environmental change (Cadotte *et al*., 2011), and can be measured using indexes with various facets (*e.g.,* Villéger *et al*., 2008). Those indexes describe the distribution of species and their abundances inside the community functional space, which is defined by the functional traits used. Each index covers a facet of functional diversity (richness, evenness, or functional divergence), making them complementary (Mason *et al*., 2005).

Several taxonomic groups have been used in studies considering functional diversity in response to environmental change in terrestrial ecosystems (*e.g*., ants, beetles, and birds) and in aquatic ecosystems (*e.g*., macroinvertebrates, zooplankton, and fish) (Batalha *et al*., 2010; Barragán *et al*., 2011; Moreira *et al*., 2016; Luiza-Andrade *et al*., 2017). Due to the sensitivity of some fish species to environmental impacts, the taxonomic and functional composition of their communities can respond in different ways, depending on the type, size, time of impact, and the type of ecosystem in which they live (Burress, 2015; Prudente *et al*., 2017; Teresa, Casatti, 2017; Leitão *et al*., 2018).

In Amazonian streams, deforestation affects fish assemblages and alters functional identity (Leal *et al*., 2017; Brejão *et al*., 2018). Fish species are distributed among various trophic levels and require a wide range of resources for their success in different environments, from rivers to temporary ponds (Brejão *et al*., 2013). Environments with less riparian forest cover tend to have more homogeneous environments, favoring generalist fish species with feeding strategies that allow them to occur in habitats with anthropogenic changes (Vieira *et al*., 2015; Zeni *et al*., 2017).

Although several studies have examined these relationships in the Amazon Forest biome (Leal *et al*., 2017; Brejão *et al*., 2018), we understand little about these relationships in the transitional area between the Amazon and the Savannah ecoregion with accelerated deforestation (Trigueiro *et al*., 2020). We assessed the taxonomic and functional structure of fish assemblages from streams in transitional areas between the Amazon and the Savannah to assess three main questions: (a) How does physical habitat integrity in areas with less riparian vegetation affect species richness, abundance, and composition of fish assemblages?; (b) How does environmental change in areas with less riparian vegetation and low physical habitat integrity affect the functional diversity of fish assemblages?; and (c) How does suppression of riparian vegetation affect stream habitats, and what functional characteristics are related to the physical habitat integrity of streams? Our operational hypothesis is that sites with riparian vegetation suppression will be characterized with low physical habitat integrity, resulting in altered fish fauna composition and functional diversity. Conversion of riparian vegetation should also have a negative impact on species with specific functional traits, especially those groups that depend on the riparian vegetation (Casatti *et al*., 2012). These changes will decrease species richness, abundance, functional divergence, functional richness, and functional evenness of fish assemblages, since reduced habitat integrity reduces important resources for the species, mainly those used as food (Leitão *et al*., 2018).

MATERIAL AND METHODS

Study area. Nine streams (Fig. 1) were sampled in August 2017 on the Tanguro farm (Municipality of Querência, state of Mato Grosso, Brazil). Streams were selected to cover the variability of riparian and stream characteristics of the study area and included three forested watershed (*i.e.*, reference sites) and six agricultural watersheds (*i.e.*, altered sites). All streams with agricultural activities had a low degree of riparian forest, while the forested sites had extensive and well-preserved riparian vegetation. Therefore, to represent areas with less forest cover, we selected streams with less riparian forest and that were distant from each other in order to avoid spatial autocorrelation. If the impacts are strong, on a local scale, a small number of streams is capable of capturing the effects of vegetation loss on the fish fauna, as observed in other studies (Teresa *et al*., 2015; Virgilio *et al*., 2018, Ilha *et al*., 2018, 2019).

The Tanguro farm is located in the Savannah-Amazon transitional area, in the upper Xingu River basin, which is part of the larger Amazon River basin. The native vegetation consists of perennial seasonal canopy forests transitioning from rainforests in the north to woody savannahs in the south of the Xingu Basin (Velasquez *et al*., 2010). According to the 1927 Köppen, Geiger climate classification (Peel *et al*., 2007), the region's climate is the Aw type, with average annual precipitation of 1800 mm and an annual average temperature above 25°C (Rocha *et al*., 2014).

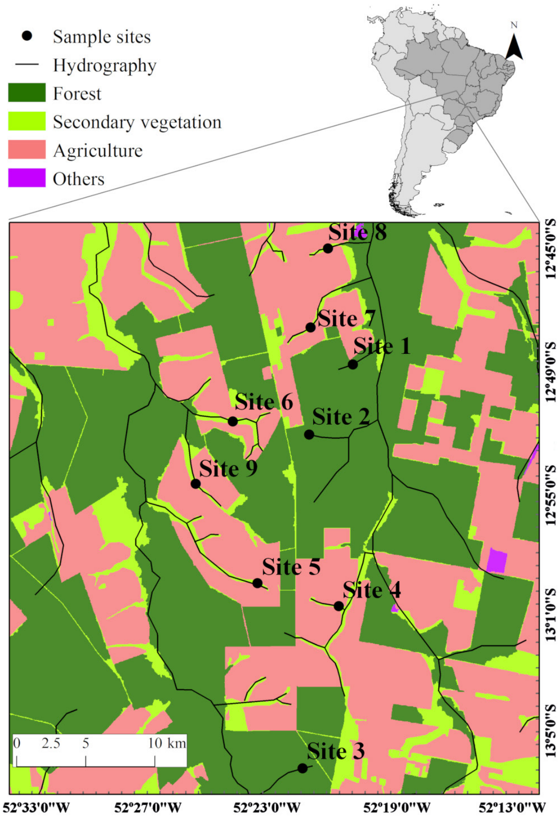

FIGURE 1 | Location of the nine streams sampled in the dry season of 2017, Tanguro Farm, state of Mato Grosso, Brazil.

Sampling

Forest cover. Using the ArcHydro tool, from ArcGis 10.1 software (Esri, 2012), with digital SRTM models of 30-meter resolution acquired from the NASA (National Aeronautics and Space Administration) homepage (https://earthexplorer.usgs.gov/), we delimited the catchments, their respective drainage systems, and the buffer zone. We evaluated land use within the riparian forest in a buffer zone measuring 60 m (30 m of land on each side of the drainage system) in the total catchment of each stream. We used this measure following the Brazilian Forest Code (Federal Law nº 12651/12), which designates riparian forests as Permanent Protection Areas (APP) in fluvial systems up to 10 m wide (Brasil, 2012). Land-use classes were identified using Digital Image Processing (PDI) of a set of Landsat 8 satellite images (USGS, 2018) using the PCI Geomatica and Ecognition programs.

The images were from the same year as the biological and physical habitat sampling (2017). Images were acquired in a geographic coordinate system projected in the geodesic DATUM SIRGAS UTM Zone 21. Next, we calculated the Kappa index according to Landis, Koch (1977), which reflects the quality of the supervised classification, ranging from 0 (low quality) to 1 (excellent quality). This index uses the classification made by the observer. In this image, the index places coordinate points where each land-use was considered. For example, where forest was considered, it seeks points that represent forest, and so forth for other land use classes. These points are compared with classified points to test the veracity of the information by counting how many forest points match the classified forest point.

The identified land use classes in riparian zone were: (a) forest, comprising areas occupied by rainforest; (b) secondary forest, comprising vegetation resulting from natural successional processes after total or partial suppression of the primary vegetation by anthropogenic activities or natural causes; and (c) agriculture areas, comprising mechanized or family farms and the presence of herbaceous plants, in addition to any indicators of homogenization seen in the satellite images (see Fig. **S1**). Even though it is in a forest-savannah transition area, the riparian vegetation was classified as forest due to its density of cover, a riparian vegetation characteristic.

Habitat. Environmental characterization of the sampled streams was carried out in the dry season, starting in the morning and extending into the afternoon, which is appropriate for our study due to the adequacy of the collection method (Frissell *et al*., 1986; Jaramillo-Villa, Caramaschi, 2008). In each stream, we delimited a 150 m stretch, which was then divided into 11 transects labeled from "A" (downstream) to "K" (upstream), obtaining 10 equidistant cross-sections of 15 m.

Environmental characterization (Kaufmann *et al*., 1999; Peck *et al*., 2006) consisted of 238 metrics including channel morphology, substrate, discharge and substrate, channel unit (water flow types), slope, sinuosity, riparian vegetation cover, woody debris tally, instream fish cover, and human impact (Tab. **S2**). We also measured water physical-chemical parameters including temperature, pH, conductivity, and dissolved oxygen (OD) at three sections (A, F and K) using a multiparameter probe. Habitat measurements were made within all cross-channel transects and longitudinal sections in each of the nine streams, and values are reported as averages and standard deviations for each stream.

Fish fauna. Fish sampling started in the morning and continued throughout the afternoon. Each stream was sampled only once. The sampling effort was six hours in each stream, with three being the number of collectors in each cross-section. Fish specimens were collected using circular 55 cm-diameter hand nets (sieves) and 40 cm-diameter dip nets ("puçá" in Portuguese), both with a 1 mm metallic mesh. We applied both methods simultaneously, always using two hand nets and dip nets in each section. The collected specimens were fixed in a 10% formalin solution and then transferred to 70% ethanol after 48 hours. Fish identification was carried out with the help of specialists and identification keys (Géry, 1977; Kullander, 1986). Species with taxonomic uncertainties were typed as morphospecies, taking care to standardize the nomenclature. The collected material was deposited in the Museu Paraense Emílio Goeldi (MPEG), Belém, state of Pará, Brazil. The list of catalog numbers for the species are listed in Tab. **S6**.

Functional traits. Fish specimens were classified into 11 trophic groups (Brejão *et al*., 2013) according to the feeding strategies of each species combined with their spatial distribution (stream margin, main channel, and water column). We also classified species into two life-history groups: 1) equilibrium strategy, with high juvenile survival rate, parental care, and large eggs, and 2) opportunistic strategy, characterized by small adult size, short generation period, high reproductive effort, and extended breeding seasons (Zeug, Winemiller, 2007). Feeding and reproductive strategies of individuals allow us to understand which factors determine the presence of a given species in a site, favoring not only their survival, but their perpetuation. Environmental changes modify these natural factors, causing species to leave the site looking for new resources and driving other species to occupy available resources the new trophic niche available (Zeug, Winemiller, 2007; Uieda, Pinto, 2011; Montag *et al*., 2019). Data for species were taken from the literature, and in the absence of information for a species, we used data for the genus (see **S3**).

Data Analyses. We first evaluated if there was spatial autocorrelation among the sampling units by calculating the fluvial distances among sampling sites. Using the ArcHydro tool, from ArcGis 10.1 software (Esri, 2012), with digital SRTM models of 30 meters resolution acquired from the NASA homepage (https://earthexplorer.usgs.gov/), we delimited the catchments and their respective drainage systems in order to verify if there was pseudo-replication among the data sampled in the streams. Next, a Principal Coordinates of Neighbor Matrices (PCNM) analysis was performed to transform the fluvial distances among the sites into a Euclidean distance matrix (Dray *et al*., 2006). The species abundance matrix and the spatial matrix generated by the PCNM were used to perform a Redundancy Analysis (RDA). The RDA result showed there was no correlation among the sampling sites (df = 2; F = 0.94; p = 0.552).

Given that land use variables are often autocorrelated, we performed a Principal Component Analysis (PCA), where the percentage of each land-use variable was transformed in a Euclidean distance matrix. Then, using the Broken-stick criterion, we selected the axes with the highest explanation power to represent forest cover (Legendre, Legendre, 2012). PCA was performed using the 'princomp' function of the Vegan package (Oksanen *et al*., 2017), in the R software (R Development Core Team, 2013).

Development of the Physical Integrity Index (PII). To calculate the Physical Integrity Index (PII) (Barbour *et al*., 1996), we used metrics from the physical habitat assessment in the three forested reference sites, and six agricultural sites. First, variables with more than 80% of values equal to 0 were removed. Second, we selected variables with reduced or no overlap between forested and agricultural watersheds based on Box-and-Whisker plots as indicators of the degree of habitat disturbance. Then, we used Student's t-test ($p < 0.05$), with the assumptions of normality and homoscedasticity, to verify whether the distributions were statistically different between both test and reference treatments (Zar, 2010). Metrics that did not show a statistically significant distribution were excluded. Metrics considered sensitive to environmental disturbances were submitted to a Spearman correlation analysis between pairs of metrics ($rS \geq 0.70$ and $p < 0.05$). Thus, when two metrics were correlated, only the one with the greatest relevance related to habitat characteristics was selected to determine the structure of aquatic communities (Baptista *et al*., 2007).

The criteria for scoring the index were based on Barbour *et al*. (1996), using the three reference forest sites as a baseline. Metrics that decrease with impact intensification obtained a positive score, with the 25th percentile as the divisor of the category with the highest score, called the "upper limit". For metrics that increase with impact intensification, with a negative score, the 75th percentile was used as the divisor of the category with the highest score.

The upper score represents 75% or more of the conditions found in reference sites and the lower score corresponds to the values that did not enter the distribution of these areas. The first case is given a score of five; in the second case, a score of one; intermediate conditions, values corresponding to 25% of the conditions found in reference sites, received a score of three. The index was expressed numerically by measuring average score values for each stream. We tested the hypothesis that forest cover reduction decreases the level of physical condition using linear regressions (Zar, 2010). For the analyses, the assumptions of homogeneity and normality of the residuals were analyzed. As predictor variables, we used the PCA axis selected by the Broken-Stick criterion, which represents forest cover, and as response variables, we used the Physical Integrity Index (PII) values for each stream.

Taxonomic diversity. To test the hypothesis that lower forest cover levels and lower physical conditions will decrease fish species richness and abundance, we performed simple linear regressions, where the assumptions of normality and homogeneity were reached (Zar, 2010). For these analyses, the predictive variables were the stream physical condition (represented by the PII) and forest cover (represented by the first PCA axis in the landcover analysis). We chose the latter because it represents the continuous gradient of forest cover, without stream categorization. Species richness and total abundance were the response variables.

To evaluate how fish species composition is distributed along the gradient of forest cover, we performed a Principal Coordinate Analysis (PCoA), using a Bray-Curtis distance matrix, where the abundance data was Hellinger-transformed. The PCoA was performed using the Vegan package (Oksanen *et al*., 2017), in the R software (R Development Core Team, 2013).

Functional diversity. We calculated three descriptive indexes of functional diversity (Villéger *et al*., 2008) (Tab. **S4**) using a categorical matrix with functional traits of life history and trophic groups together with the species abundance matrix:

i) Functional Richness (FRic) represents the volume occupied by the community in the functional space composed by community traits, allowing us to quantify the variation of each functional trait in each community (Mouchet *et al*., 2010). Decreased FRic values after disturbance indicate that disturbance acts as an environmental filter, selecting species with similar functional traits and that occupy a more restricted portion of the functional volume (Pakeman *et al*, 2011; Mouillot *et al*., 2013); ii) Functional Evenness (FEve) estimates the consistency with which species abundances are distributed in the functional space and can represent the efficiency with which available resources are used (Mouchet *et al*., 2010). With this index, we can infer if species are nested or dispersed evenly in the functional space. Under the influence of an environmental filter, coexisting species tend to be functionally similar, showing higher species grouping and consequently low heterogeneity in filling the functional space, resulting in low FEve values (Mouillot *et al*., 2013), and the iii) Functional Divergence (FDiv) assesses how much the taxa differ in the functional space. Higher FDiv values indicate a high level of niche differentiation between species, and thus low interspecific competition for resources. This index ranges from zero (when species are highly abundant and closer to the assemblage centroid in the occupied volume) to one (when highly abundant species are distant from the assemblage centroid) (Mouillot *et al*., 2013). All indexes were calculated using the FD package, using the dbFD function, which uses Principal Coordinate Analysis (PCoA) to scale samples on principal axes according to the Gower distance. The resulting axes are used as traits to calculate the indices. The indexes were calculated using R programming software (Laliberté, Legendre 2010; R Development Core Team, 2011; Laliberté *et al*., 2014).

We performed linear regressions (Zar, 2010), following the assumptions of normality and homogeneity of the residuals, for each descriptive index of functional diversity. We tested the hypothesis that lower values of physical condition and percentage of forest cover will lead to low values of richness, divergence and functional evenness of fish assemblages. We used the values obtained from the PII and the first PCA axis for land cover selected by the Broken-stick criterion as predictive variables. Additionally, we created ordination graphs, with the PCA axis representing the x axis and the abundance of trophic groups representing the y axis, to verify the organization of trophic groups related to forest cover loss. All analyses were conducted in the R programing environment (R Development Core Team, 2011).

RESULTS

Forest cover. The PCA analysis of the supervised classification of land use in the riparian buffer organized the sites according to the amount of forest cover on the first axis (84% of the variation; Fig. 2; Tab. 1). In relation to the Kappa index, which evaluates the quality of the classification of land-uses, we verified that the agricultural environments had excellent quality of classifications (1.0), followed by secondary forest (0.6) and forest (0.3). Even the value of 0.3 is classified by the literature as a good quality of interpretation (Niculescu *et al*., 2018); this value demonstrates the differences between high and low canopy that makes interpretation difficult, even in high resolution images.

Habitat. We used five metrics to construct the stream Physical Integrity Index (PII): 1) mean depth; 2) width and depth ratio in the section; 3) volume of woody debris in the stream/m2 – size class 2 (all woody debris larger than 0.3 m diameter); 4) mean herbaceous ground cover; and 5) cultivation proximity index (Tab. 2; Fig. 3). The

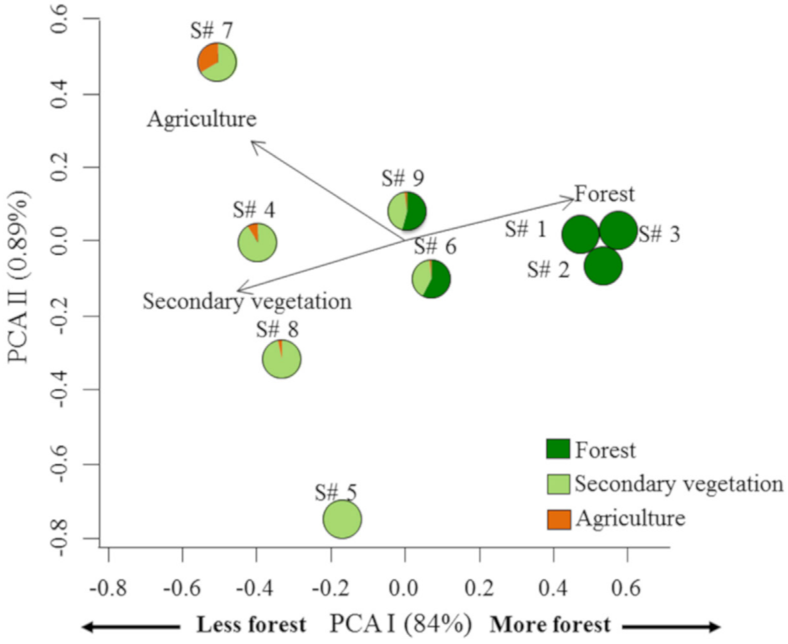

FIGURE 2 | Principal component analysis performed with the percentages of land use in 60 m riparian buffers in the catchment of nine streams located in the Tanguro Farm, Municipality of Querência, state of Mato GrossoT. Sites 1, 2, and 3 overlapped, and were rearranged for better visualization. Sgrid: Site.

TABLE 1 | Land-use classes identified in riparian buffers in the catchment of nine streams in the Tanguro Farm, Municipality of Querência, state of Mato Grosso, and their correlations to Axis I of the PCA.

Variable	Loadings	
	Axis I	Axis II
Forest	0.91	0.23
Secondary forest	-0.91	-0.26
Agriculture	-0.83	0.54
Eigenvalue	2.37	0.42
% of explanation	84%	0.89%
Broken-Stick	1.8	0.8

hypothesis that loss of riparian vegetation cover negatively affects the physical condition of streams was corroborated ($r^2 = 0.70$, $p = 0.004$) (Fig. 4). In general, the more forested streams in this study had lower mean depth ($t = 2.42$, $p = 0.04$), higher width and depth ratio in the section ($t = 2.94$, $p = 0.02$), lower cultivation proximity index ($t = 2.95$, $p= 0.02$), higher volume of wood debris in the stream ($t = 2.64$, $p = 0.03$), and lower mean herbaceous ground cover ($t = 2.41$, $p = 0.04$) compared to streams with less forest cover (Fig. 3).

Fish fauna. We collected 2,943 specimens, comprised of 29 species, 27 genera, and six orders (Tab. 3). Characiformes was the most representative order in terms of abundance, with 56% of individuals. The most abundant species were *Moenkhausia phaeonota* Fink, 1979 (27%), followed by *Hemigrammus* sp1. (24%), and *Melanorivulus megaroni* (Costa, 2010) (15%). The most representative (in terms of abundance) trophic functional groups were diurnal channel drift feeders (29%), diurnal backwater drift feeders (24%), and diurnal surface pickers (23%). For the life-history groups, 78% of individuals were opportunistic strategists.

Taxonomic Diversity. The hypotheses that lower forest cover in the riparian zone correlates with lower stream fish abundance ($r^2 = 0.175$, $p = 0.262$) and species richness ($r^2 = 0.041$, $p = 0.602$) were not supported. Additionally, we found no correlation between stream physical condition (PII) and abundance ($r^2 = 0.149$, $p = 0.304$) or richness ($r^2 =$

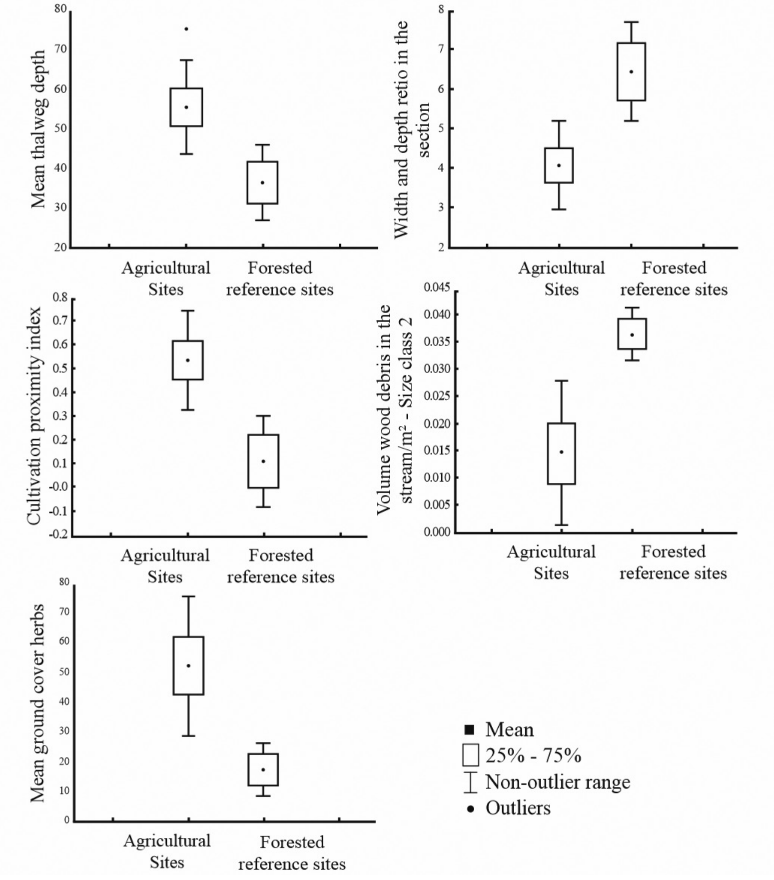

FIGURE 3 | Box-and-whiskers plots of the five metrics comprising the Physical Integrity Index.

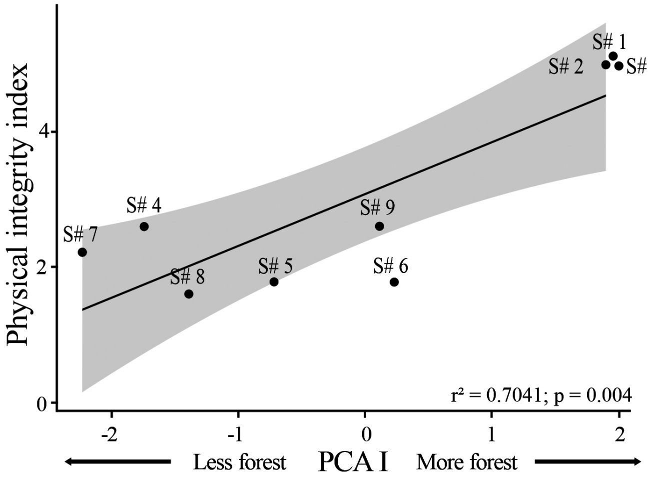

FIGURE 4 | Relationship between forest cover and the physical integrity index of streams located in the Tanguro Farm, Municipality of Querência, state of Mato Grosso. Forest cover is represented by the Axis I of the PCA performed with land use variables obtained from a 60 m buffer. Sites 1, 2 and 3 overlapped, and were rearranged for better visualization. Sgrid: Site.

TABLE 2 | Values of the metrics selected to calculate the scores of the Physical Integrity Index in the Amazon, state of Mato Grosso. PII: Physical Integrity Index; MEAN_TD: mean thalweg depth (cm); WDR_SEC: width and depth ratio in the section; VOL_WDS: volume of woody debris in the stream /m² –size class 2; MEAN_HERBS: mean ground cover herbs; CULTIV_PI: cultivation proximity index.

Sample sites	Coordinates		PII	MEAN_TD	WDR_SEC	VOL_WDS	MEAN_HERBS	CULTIV_PI
	Latitude	Longitude						
Site 1	-12.836	-52.333	5	45.00	4.41	2.70	16.59	0.33
Site 2	-12.881	-52.361	5	42.70	7.04	7.38	16.25	0.33
Site 3	-13.099	-52.366	5	56.86	4.57	2.77	37.04	0.21
Site 4	-12.993	-52.342	1.8	25.70	7.27	3.4	26.70	0.00
Site 5	-12.978	-52.395	2.2	43.82	2.66	0.00	72.04	0.66
Site 6	-12.873	-52.411	1.8	51.11	3.64	0.00	43.75	0.66
Site 7	-12.811	-52.360	2.2	75.27	3.41	6.28	75.22	0.66
Site 8	-12.760	-52.349	2.6	41.06	5.02	6.04	9.20	0.00
Site 9	-12.913	-52.435	1.8	61.11	5.80	3.76	68.29	0.66

0.003; p = 0.891) of fish assemblages (Fig. **S5**). The first two axes of the PCoA explained 62% of the variation in the data (Fig. 5; Tab. **S6**). The results from the PCoA show that species were grouped according to different levels of forest cover as represented by the color gradient, with lighter colors representing sample points with less forest cover.

Functional diversity. As hypothesized, we found a positive correlation between forest cover in the riparian zone and functional richness (r^2 = 0.479, p = 0.03) (Fig. 6). However, contrary to our hypothesis we did not find a positive correlation with functional evenness (r^2 = 0.111, p = 0.379) or divergence (r^2 = 0.211, p = 0.214).

TABLE 3 | Species list classified according to their trophic functional groups and life history. Species found in nine streams sampled in 2017, in the Tanguro Farm, Municipality of Querência, state of Mato Grosso. NIB: Nibblers; DBDF: Diurnal backwater drift feeders; DCDF: Diurnal channel drift feeders; ASP: Ambush and stalking predators; DSP: Diurnal surface pickers; NIP: Nocturnal invertebrate pickers; PB: Pickers and browsers; CNBP: Crepuscular to nocturnal bottom predators; GRU: Grubbers; GRA: Grazers; CNDF: Crepuscular to nocturnal drift feeders; OS: Opportunistic strategy; ES: Equilibrium strategy.

Taxon	Species code	Total N	Trophic group	Life-history
Characiformes				
Anostomidae				
Leporinus britskii Feitosa, Santos & Birindelli, 2011	sp1	2	NIB	OS
Characidae				
Hyphessobrycon sp.	sp2	1	DBDF	OS
Hemigrammus sp1.	sp3	1	DBDF	OS
Moenkhausia oligolepis (Günther, 1864)	sp4	11	DCDF	OS
Moenkhausia phaeonota (Fink, 1979)	sp5	789	DCDF	OS
Hemigrammus sp2.	sp6	712	DBDF	OS
Crenuchidae				
Characidium zebra (Eigenmann, 1909)	sp7	61	ASP	OS
Melanocharacidium sp.	sp8	63	DCDF	OS
Lebiasianidae				
Copella arnoldi (Regan, 1912)	sp9	159	DSP	OS
Cyprinodontiformes				
Rivulidae				
Melanorivulus megaroni (Costa, 2010)	sp10	453	DSP	OS
Poecilidae				
Pamphorichthys sp.	sp11	46	DSP	OS
Gymnotiformes				
Gymnotidae				
Gymnotus carapo Linnaeus, 1758	sp12	79	NIP	ES
Gymnotus coropinae Hoedeman, 1962	sp13	3	NIP	ES
Hypopomidae				
Hypopomus brevirostris (Steindachner, 1868)	sp14	47	NIP	ES
Rhamphichthyidae				
Gymnorhamphichthys rondoni (Miranda Ribeiro, 1920)	sp15	32	NIP	ES
Hypopygus lepturus Hoedeman, 1962	sp16	74	NIP	ES
Sternopygidae				
Eigenmannia trilineata (Lopes & Castello, 1966)	sp17	54	NIP	ES
Sternopygus macrurus (Bloch & Schneider, 1801)	sp18	4	NIP	ES
Cichliformes				
Cichlidae				
Aequidens tetramerus (Heckel, 1840)	sp19	178	PB	ES
Crenicichla inpa (Ploeg, 1991)	sp20	7	ASP	ES
Siluriformes				
Auchenipteridae				
Tatia aff. *aulopygia* (Kner, 1858)	sp21	8	CNBP	ES
Callichthydae				
Megalechis sp.	sp22	2	GRU	ES
Cetopsidae				
Helogenes marmoratus Günther, 1863	sp23	52	CNDF	ES
Heptapteridae				
Brachyglanis sp.	sp24	29	CNBP	ES
Cetopsorhamdia sp.	sp25	3	CNBP	ES
Rhamdia aff. *quelen* (Quoy & Gaimard, 1824)	sp26	1	CNBP	ES
Rhamdella sp.	sp27	15	CNBP	ES
Loricariidae				
Curculionichthys sp.	sp28	50	GRA	ES
Synbranchiformes				
Synbranchidae				
Synbranchus marmoratus Bloch, 1795	sp29	7	CNBP	ES

Additionally, our hypothesis that reduced habitat integrity would correlate with lower values of functional richness ($r^2 = 0.037$, $p = 0.618$), functional divergence ($r^2 = 0.238$, $p= 0.183$), and functional evenness ($r^2 = 0.262$, $p = 0.676$) of fish assemblages was not supported.

The graph based on the distribution of functional groups in response to forest cover loss showed that groups such as ambush and stalking predators, diurnal surface pickers (opportunistic life history strategists), and pickers and browsers (equilibrium life history strategists) are associated more with environments with low forest cover values, while groups comprised of crepuscular to nocturnal drift feeders are more associated with environments containing high forest cover (Fig. 7).

DISCUSSION

Our results indicate that forest cover loss in the 60 m riparian buffer resulted in lower physical habitat quality (indicated by the Physical Integrity Index – PII) and functional

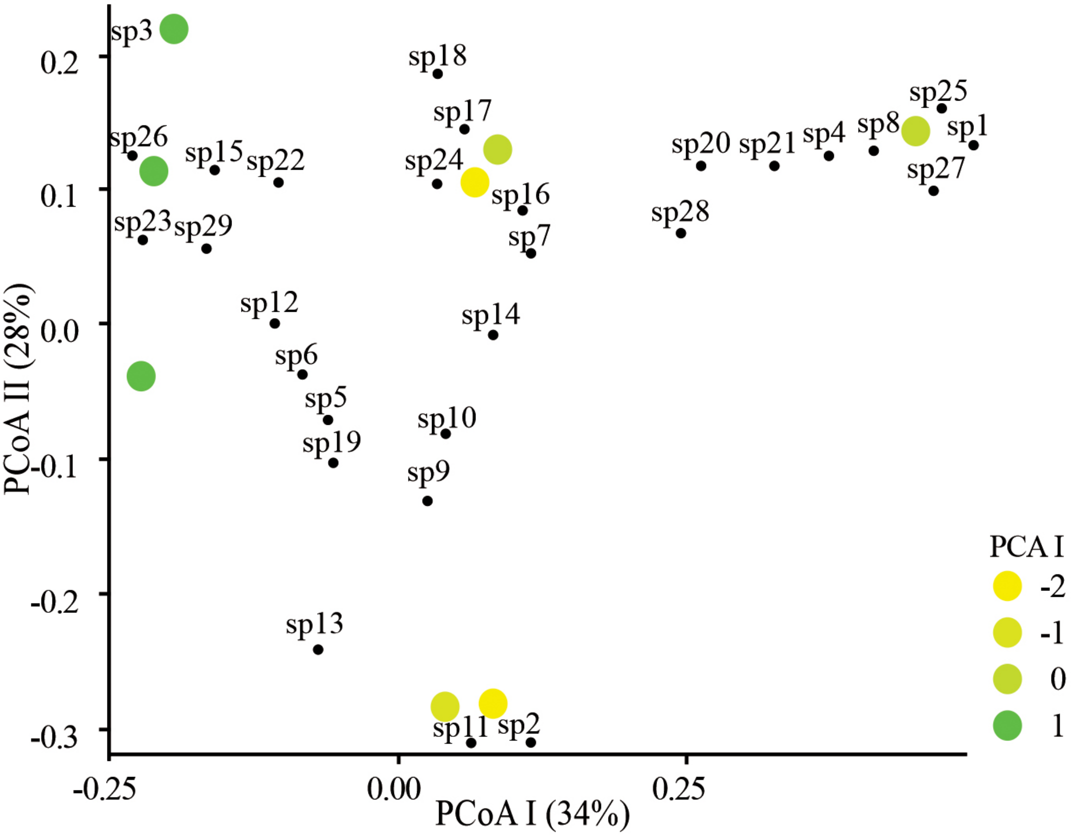

FIGURE 5 | Relationship between species composition and the first PCA axis, ordinated by a Principal Coordinate Analysis. Lighter colors represent sample points with less forest cover. The species corresponding to the codes are in Tab. 2.

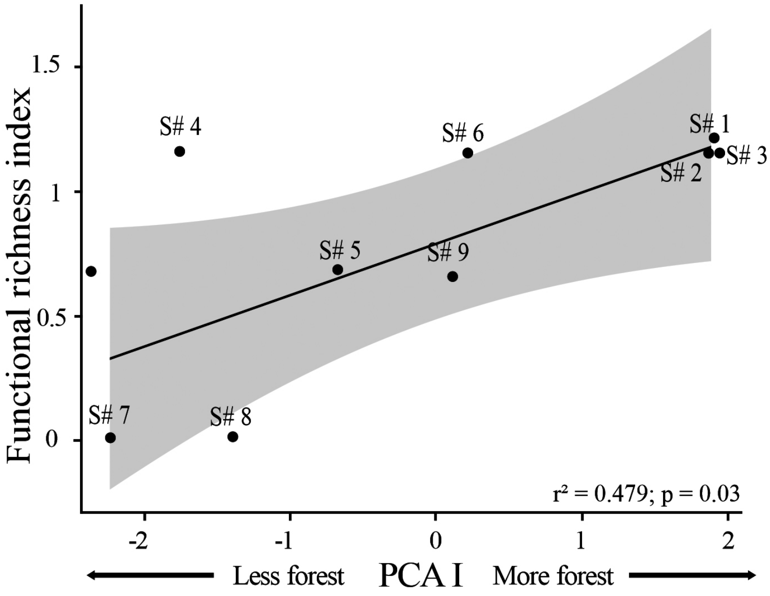

FIGURE 6 | Relationship between forest cover and the functional richness index observed in streams located in the Tanguro Farm, municipality of Querência, state of Mato Grosso. Forest cover is represented by the Axis I of the PCA performed with land use variables obtained from a 60 m buffer. Sites 1, 2 and 3 overlapped and were rearranged for better visualization.

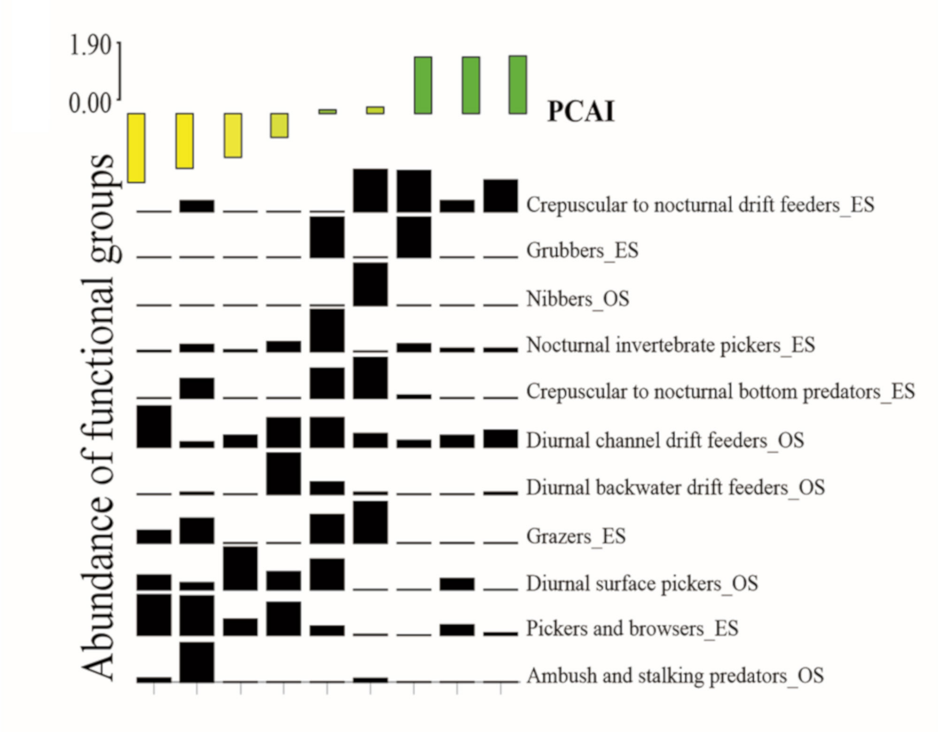

FIGURE 7 | Distribution of functional groups related to forest cover loss in Amazonian streams, state of Mato Grosso. OS: Opportunistic strategy; ES: Equilibrium strategy. Forest cover values vary from green (more forest cover) to yellow (less forest cover).

richness. However, at the scale of our study, forest cover loss did not affect taxonomic richness, abundance, divergence, and functional evenness of fish species. The physical condition of the streams also did not influence the abundance, richness, functional richness, functional divergence, and functional evenness of the fish fauna.

Several authors have studied the effects of deforestation on fish fauna diversity in different environmental contexts (Ferreira *et al*., 2018; Ilha *et al*., 2018, 2019; Leitão *et al*., 2018; Montag *et al*., 2019). The results found in these studies tend to vary due to different factors such as the degree of impact, type of ecosystem, local characteristics, fauna diversity, and others. Our results diverged somewhat from previous findings, perhaps because the area we studied has a high percentage of preserved vegetation along the river network and a lower percentage of agricultural areas, which may help maintain certain natural characteristics of the ecosystem and mitigate the effects deforestation in the area (Paula *et al*., 2021). This abundance of preserved vegetation may also be helping the community to recover after disturbances from agricultural practices by retaining resource availability, especially habitat and nutrient resources that favor specialist species (Karr, 1981; Teresa, Casatti, 2010).

We observed that even with the presence of the forest extension in the sampled sites, it is still possible to detect changes in their physical condition. The mean herbaceous ground cover was higher and the volume of woody debris was lower at the streams in agricultural watersheds (*i.e*., altered sites). Deforestation likely makes these sites susceptible to the proliferation of herbaceous plants due to increased solar incidence and lowered competition for resources; furthermore, it reduces the amount of allochthonous material in streams, as well as the supply of woody debris that are essential elements for communities of small water bodies that use them as shelter (Roth *et al*., 1996). The variable width and depth ratio also differed between reference and altered sites, becoming narrower and deeper in the altered sites, due to the removal of natural structures in the stream channel and resulting in slow water flow (Neill *et al*., 2006; Deegan *et al*., 2011).

Changes in the prevalence of certain functional groups between sites occurred possibly due to changes in the physical characteristics of streams. We observed that opportunistic life history strategist species were highly represented altered sites, as they can explore various environments. Groups such as crepuscular to nocturnal drift feeders, grubbers, and nibblers had a significant reduction across the forest gradient. Thus, the ecosystem functional volume is reduced since the available niches are underutilized, or contribute less efficiently toward ecosystem functions, such as the exchange of energy and matter or substrate bioturbation (Mason *et al*., 2005; Leduc *et al*., 2015).

The most representative functional group (diurnal channel drift feeders) was relatively homogeneous along the forest cover gradient. This distributional pattern may have occurred because this group was mostly composed of species from the Characidae family, which occupy a variety of habitats and have various feeding strategies (Brejão *et al*., 2013). Nocturnal invertebrate pickers showed little variation along the gradient, possibly because they belong to the order Gymnotiformes, which possess diverse feeding strategies and can make use of different habitat types (Albert, Crampton, 2006). Trophic plasticity suggests that each species can feed on many different prey types, which is not necessarily true for gymnotiforms. *Gymnotus* Linnaeus, 1758 can be piscivorous, but *Gymnorhamphichthys* Ellis, 1912 mostly feed on larval invertebrates.

Similar to other studies, grazers, represented by the Loricariidae family, were associated with environments that suffered environmental change (Bojsen, Barriga, 2002). We observed an increase of species from this family in areas with low forest cover. It is possible that additional light, resulting from reduced forest cover, increased the availability of periphyton on aquatic plants, pluricellular algae (epiphyton), and sediment (epilithon) (Sládeèková, 1962), which are important food resources for this group.

The presence of an ambush and stalking predator *Characidium zebra* Eigenmann, 1909, in sites with lower forest cover may be associated with prey availability, such as aquatic insect larvae (Cetra *et al*., 2011), which are more available in these types of environments (Juen *et al*., 2014; Marques *et al*., 2021). Diurnal surface pickers were also more abundant in environments with less forest cover. This group is represented by individuals of the Rivulidae family, which possess opportunistic feeding habits and are commonly found in shallow waters and deforested streams (Volcan *et al*., 2011; Ilha *et al*., 2019). Pickers and browsers showed a clear increase in abundance as the percentage of forest cover decreased. In this study, this group was mostly represented by the Cichlidae family, which, is comprised of some tolerant and generalist species that can present higher species richness in deforested environments and have been associated with sites that have suffered environmental degradation (Bojsen, Barriga, 2002; Burress, 2015; Ferreira *et al*., 2018).

Crepuscular to nocturnal drift feeders preferred forested streams; in this group, there was a predominance of *Helogenes marmoratus* Günther, 1863, a species that seeks shelter in the microhabitats between roots and wood in the riparian vegetation as well as in leaf banks, where they hide during the day (Zuanon *et al*., 2015). Crepuscular to nocturnal bottom predators had a high number of individuals in sites with reduced habitat integrity. The most representative individuals of this group belong to the Heptapteridae family, which has been recorded inhabiting various substrate types, *e.g*., leaf banks, sand, and groundwater (Zuanon *et al*., 2015), indicating their capacity to inhabit different environments.

Species with opportunistic reproductive strategies preferred environments with low forest cover values, suggesting that they can colonize disturbed or recently available habitats quickly. However, species with equilibrium strategies had high abundance in environments with intermediate forest levels and were inferred to be adaptive in habitats where resources are limited (Zeug, Winemiller, 2007).

The Amazon-Savannah transitional area is a region rich in biodiversity; however, due to the accelerated advance of deforestation, fish diversity and stream ecosystem functions are at risk. Our study was able to detect the effects of deforestation on the fish fauna, and, even with a low number of samples, it highlights the importance of complying with the Brazilian Forest Code (Brasil, 2012), which requires the creation of Permanent Protection Areas (APP) along the shoreline of any natural watercourse. Despite the observed decrease in functional richness, our results suggest that the presence of forest in the riparian zone, even if it is secondary forest, can buffer the effects of adjacent deforested areas on the taxonomic and functional diversity of the fish fauna, avoiding even greater losses to species diversity. However, we believe that additional studies in the area, with a larger number of samples, can increase the level of confidence in the results presented here.

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Pâmela V. Freitas: Data curation, Formal analysis, Investigation, Methodology, Software, Supervision, Validation, Visualization, Writing-original draf, Writing-review and editing.

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Karina Dias Silva: Conceptualization, Formal analysis, Investigation, Methodology, Software, Supervision, Validation, Visualization, Writing-original draf, Writing-review and editing.

ETHICAL STATEMENT

The sampling and transportation of the biological material were authorized by the Biodiversity Authorization and Information System (SISBIO), under the Permanent License number 8878–1 to collect zoological material. The biological material is filed under the Animal Use Ethics Commission – CEUA/ UFPA nº 8293020418 (ID 000954), following the rules issued by the National Council for the Control of Animal Experimentation (CONCEA).

COMPETING INTERESTS

The authors declare no competing interests.

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