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Does riparian vegetation affect fish assemblage? A longitudinal gradient analysis in three Amazonian streams

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ABSTRACT. The degradation of riparian vegetation in low-order streams can cause irreversible environmental damage, biodiversity loss and alter the structure of communities. In this context, the present study evaluated how the level of degradation of riparian vegetation influences the fish assemblage in three low-order streams in the Brazilian Amazon. In each stream, three different stretches were delimited, distributed in the environments of forest, edge and pasture, and the samplings were conducted in three hydrological regimes using dip nets. Influences of the stretches and streams on richness and abundance were tested with GLMM, on the composition with PERMANOVA and on indicator species with IndVal. Species richness was higher in forest environments and lower in pasture; while abundance was higher in pasture and lower in forest. Species composition differed between stretches, streams and in the interaction between stretches and streams, with a strong difference between stretches of forest and pasture environments. Six species dependent of the riparian vegetation were classified as indicators of forest environments; while a generalist species was indicator of pasture. These results reinforce the negative influence of riparian vegetation removal on biological communities and provide subsidies that can help in the conservation and management of these systems.

Keywords: Low-order streams; environmental alteration; diversity loss; fish diversity; indicator species; pasture.

A vegetação ripária afeta as assembleias de peixes? Uma análise de gradiente longitudinal em três riachos amazônicos

RESUMO. A degradação da vegetação ripária em riachos de baixa ordem pode causar danos ambientais irreversíveis, perda de biodiversidade e alterar a estrutura das comunidades. Neste contexto, o presente estudo avaliou como o nível de degradação da vegetação ripária influencia a assembleia de peixes em três riachos de baixa ordem na Amazônia brasileira. Em cada riacho, foram delimitados três diferentes trechos, distribuídos nos ambientes de floresta, borda e pastagem, onde as amostragens foram conduzidas em três regimes hidrológicos, usando redes de imersão. As influências dos trechos e dos riachos na riqueza e na abundância foram testadas com GLMM, na composição com PERMANOVA e nas espécies indicadoras com IndVal. A riqueza de espécies foi maior em ambientes de floresta e menor em pastagem, enquanto a abundância foi maior em pastagem e menor em floresta. A composição de espécies diferiu entre trechos, riachos e na interação entre trechos e riachos, com forte diferença entre trechos de ambientes florestais e de pastagem. Seis espécies dependentes da vegetação ripária foram classificadas como indicadoras de ambientes florestais, enquanto uma espécie generalista foi indicadora de pastagem. Estes resultados reforçam a influência negativa da remoção da vegetação ripária sobre as comunidades biológicas e fornecem subsídios que podem auxiliar na conservação e no manejo desses sistemas.

Palavras-chave: Riacho de baixa ordem; alteração do ambiente; perda de diversidade; diversidade de peixes; espécies indicadoras; pastagem.

Introduction

The Amazon basin, within its diversity of aquatic habitats, is formed by a network of small watercourses, called streams (Junk, 1983). Low-order streams are relatively fragile systems with low

autochthonous production, which makes them dependent on riparian vegetation to maintain their biotic integrity (Barrella, Petrere Jr., Smith, & Montag, 2000; Henry, 2003; Castello et al., 2013). Although relatively small, riparian areas influence and are influenced by the stream channel through

exchange of water, nutrients and organic matter (Williams, Fisher, & Melack, 1997). Thus, they play a key role in the maintenance of water quality and water flow in both flood and drought periods (Naiman, Décamps, & McClain, 2005). They also provide necessary conditions for the survival and reproduction of diverse life forms, as well as a considerable richness of environmental goods and services (Junk & Piedade, 2005; Castello et al., 2013).

The increase of environmental degradation in low-order streams, caused mainly by the agriculture progress (Rivero, Almeida, Ávila, & Oliveira, 2009; Ferreira & Coelho, 2015), has caused irreversible environmental damage and biodiversity loss. The removal of riparian vegetation reduces the supply of woody material to streams, affecting mainly species that seek food and shelter through these materials (Angermeier & Karr, 1984; Casatti et al., 2015). Unprotected soils favor erosion that modify stream channels and leave the environments more exposed to solar radiation and winds, causing more pronounced fluctuations in water temperature (Ferreira & Casatti, 2006). An increase in autochthonous production may also occur due to higher incidence of light and increase in oxygen consumption, mainly due to organic matter decomposition (Pusey & Arthington, 2003).

Among the high fish species richness known for the Amazon basin, a considerable part is composed of headwaters fishes, including countless endemism (Lévêque, Oberdorff, Paugy, Stiassny, & Tadesco, 2008; Zuanon et al., 2015; Froese & Pauly, 2018). Therefore, relating spatial distribution patterns of fish species with habitat characteristics can be an important tool to predict the impacts of forest removal on riparian environments of the Amazon Basin (Bojsen & Barriga, 2002; Mendonça, Magnusson, & Zuanon, 2005; Kemenes & Forsberg, 2014; Bordignon, Casatti, Pérez-Mayorga, Teresa, & Brejão, 2015).

Naturally, stream species composition varies in space (Winemiller, Agostinho, & Caramaschi, 2008). A historical set of studies shows that the species diversity of streams increases from the source to the mouth (Vannote, Minshall, Cummins, Sedell, & Cushing, 1980; Poff & Allan, 1995; Matthews, 1998), because as the stream increases in size, it offers a variety of resources, shelters and physical conditions that favor the addition of species (Tos, Gomes, & Rodrigues, 2014). However, deforestation can lead to several changes in the structure and energy sources of streams (Allan, Erickson, & Fay, 1997; De Paula et al., 2013; Leal et al., 2016), directly or indirectly influencing the distribution of

organisms along the longitudinal gradient (Perkin & Gido, 2012). In the case of fish, studies have shown that deforestation of riparian environments can alter the community structure, causing changes in species composition (Bojsen & Barriga, 2002), loss of sensitive species, dominance of generalist species (Ferreira & Casatti, 2006) and functional homogenization of communities (Bordignon et al., 2015).

In this context, the aim of the present study was to evaluate how riparian vegetation influences the fish assemblage, along the longitudinal gradient under different levels of degradation of riparian vegetation, in three low-order streams in the Brazilian Amazon. More specifically, we tested if different conditions have influence on (i) fish richness, abundance and composition between the streams; (ii) fish richness, abundance and composition between stream stretches (environments along a longitudinal gradient); and (iii) indicator species composition both between streams and between environments along a longitudinal gradient.

Material and methods

Study area

The study was conducted in the southwestern Brazilian Amazon in the Reserva Experimental Catuaba (REC; 10°4'36"S, 67°37'0"W), municipality of Senador Guimard, State of Acre, Brazil (Figure 1). The REC has 2111 ha of tropical rainforest cover, composed mainly of open rainforest with bamboo and palms (López-Rojas, Souza, & Morato, 2015). In this region, part of the forest has been removed due to livestock production that has transformed most of the once vast areas of rainforest into a mosaic of grasslands with sparse forest fragments (Acre, 2000).

Three low-order streams were selected and used as model ecosystems for this study purpose: *Bambú* Stream and *Pogo* Stream, both first-order; and *Floresta* Stream, a second-order stream formed by the two previous streams. Three stretches of 100 m length were selected in each stream, distributed as follows: a stretch inside the forest (Forest) inserted 200 m from the edge; a stretch in the ecotone between forest and grassland inserted in the forest fragment edge (Edge); and a stretch bordered by grassland, inserted 200 m outside the forest fragment (Pasture). In total, fish were sampled in nine stretches (Figure 1).

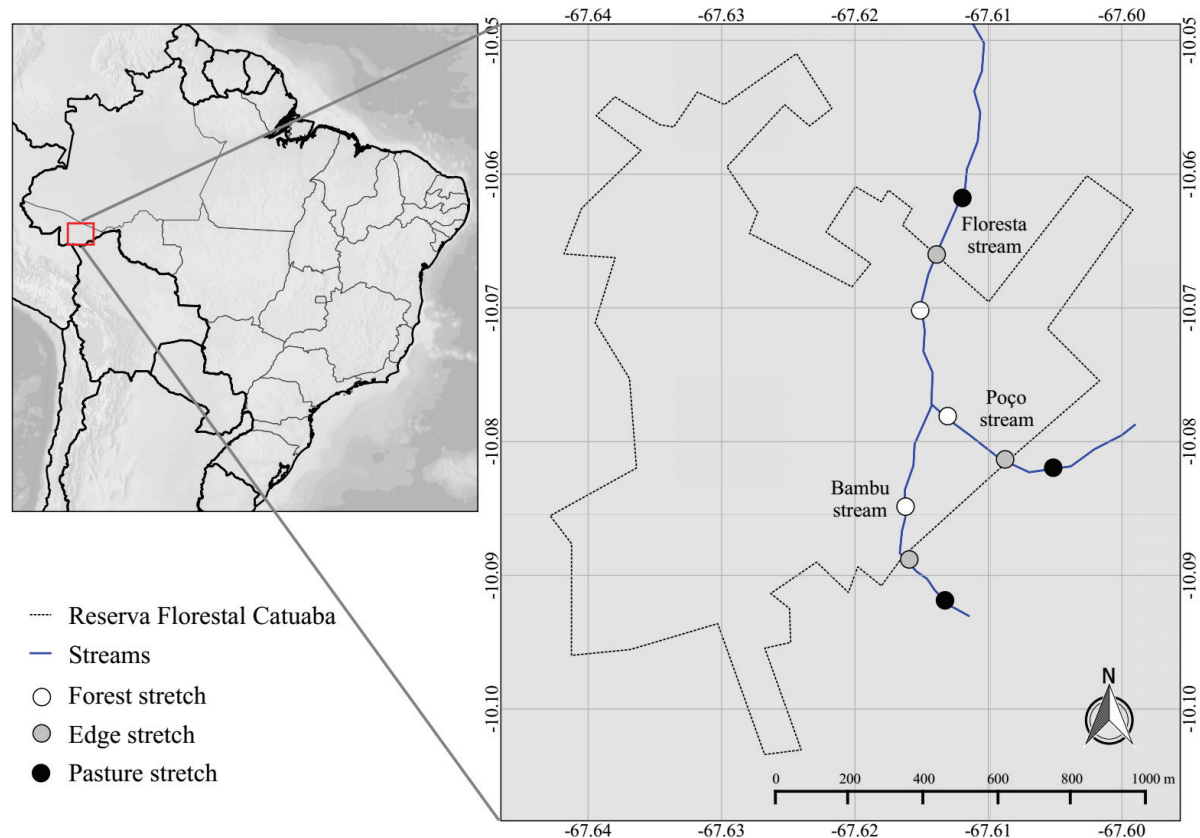


Figure 1. Study area in the Reserva Experimental Catuaba, State of Acre, Brazil.

Structure of streams

The stretches selected in the *Floresta* and *Poço* streams, which are bordered by both forest and forest fragments, are inserted in predominantly dense forest habitats, whereas *Bambu* Stream features vegetation composed of open forest with bamboo. Different vegetation creates substantial changes in physical and chemical characteristics of the distinct stream stretches. In nearly all the stream stretches bordered by forest, a high preservation degree was observed, as the surrounding vegetation provides stretches of dense forest, an abundance of arboreal stratification, shrubs and herbaceous layers, as well as epiphytes and closed canopy areas, which contribute to low luminosity to streambeds. In the stretches located at the edge of the forest fragment, a medium preservation degree was attributed, as they have relatively small proportion of preserved tree layer, with an abundance of shrubs, herbs, vines and other forest edge species. In the stretches inserted in pasture areas, a low preservation degree was assigned, related to the lack of trees, direct exposure of the aquatic environment and siltation process at an early stage.

Fish sampling

The region climate includes a rainy season from October to April and dry season in the remaining

months. The transition period between seasons may varies (Fonseca, 2006). Due to this variation, fish sampling was conducted every three months, considering the different hydrological regimes: flood, receding and drought (January to October 2008). Samplings in each stretch were carried out in the morning (8:00 am) and afternoon (5:00 pm). Two dip nets (80 x 60 cm) handled by two researchers were used simultaneously during a period of two hours along a 100 m stretch for collecting fishes. Sampling was conducted upstream, reaching different biotopes, such as backwaters, main channel, riffles and marginal pools. To minimize fish escape, 1.5 mm nylon nets were used to isolate the stretches. All collected fish individuals were fixed to obtain an abundance estimation. The collected fishes were anesthetized using clove oil, fixed in 10% formalin and transported to the Fish Collection at the *Universidade Federal do Acre*, Rio Branco, State of Acre, Brazil (MUFAC-IC).

Data analysis

Spatial autocorrelation in the overall data set was tested using a Mantel Multivariate Correlogram (Legendre & Legendre, 2012). Significant spatial autocorrelation was detected in the first two distance classes (smaller scales, $p < 0.01$), indicating that the nested arrangement of the sampling units (stretches

within the streams) did not constitute spatially independent observations. Due to spatial autocorrelation, Generalized Linear Mixed Models via Penalized Quasi-Likelihood (glmmPQL; Zuur, Ieno, Walker, Saveliev, & Smith, 2009) with Gaussian distribution were used to check for variations in richness and abundance as a function of the stretch (environments along a longitudinal gradient: forest, edge and pasture), stream and interaction between stretch and stream (fixed effects). Abundance and richness were log-transformed to reduce data heterogeneity and meet the assumptions of the test. The seasonal period was included as a random effect to incorporate temporal variability. To consider the spatial autocorrelation, an error term was included in the models that assumes an exponential correlation structure using the geographic coordinates of the sampled stretches. This technique is considered one of the best to deal with non-normal spatially autocorrelated data (Dormann et al., 2007). Model validation was performed graphically following the protocol of Zuur, Ieno, and Elphick (2010).

To access variations in fish species composition, an abundance matrix was converted into a dissimilarity matrix using the Bray-Curtis distance. For this, the samplings in each stretch were considered as sampling unit, totaling 27 samples. The distance matrix was summarized into two axes and plotted using the non-metric multidimensional scaling (nMDS). A PERMANOVA (Permutational Multivariate Analysis

of Variance) was used with the Bray-Curtis index to test for any differences in species composition, using the streams, stretches (environments along a longitudinal gradient: forest, edge and pasture) and the interaction between stretch and streams as factors. The significance was obtained using a Monte Carlo permutation method with 999 permutations.

To identify species highly associated with different environments and streams, we used the Indicator Species analysis (IndVal; Dufrêne, & Legendre, 1997). The indicator value ranges from 0 to 100 and is generated from the product of two values: A, a measure of specificity based on species abundance in each stretch or stream; and B, a measure of fidelity based on the occurrence of the species in each stretch or stream. The significance of the indicator value for each species was obtained by a procedure of 999 permutations, in which species with $p \leq 0.05$ were considered indicators of the respective stretches or streams (Dufrêne & Legendre, 1997).

All analyses were run in the R software (R Core Team, 2018). Spatial autocorrelation analyses were performed using the *mantel.correlog* function of *vegan* package (Oksanen et al., 2017); glmmPQL using the *glmmPQL* function of *MASS* package (Venables & Ripley, 2002); nMDS and PERMANOVA using the *metaMDS* and *adonis* functions, respectively, of *vegan* package (Oksanen et al., 2017); and IndVal using the *indval* function of *labdsv* package (Roberts, 2016).

Table 1. Composition and abundance of fishes in different environments stretches of three low-order Amazonian streams, State of Acre, Brazil.

Taxon	Bambú stream			Floresta stream			Poço stream			Total abundance
	Forest	Edge	Pasture	Forest	Edge	Pasture	Forest	Edge	Pasture	
Characiformes										
Acestrorhynchidae										
<i>Acestrorhynchus falcatus</i> (Bloch 1794)							7			7
Characidae										
<i>Ctenobrycon hauxwellianus</i> (Cope 1870)				5			19			25
<i>Hemigrammus</i> sp.	10			10						20
<i>Hyphessobrycon bentosi</i> Durbin 1908		4		102	159	17	2			284
<i>Moenkhausia comma</i> Eigenmann 1908	2	15		4		1	5	13	2	42
<i>Moenkhausia oligolepis</i> (Günther 1864)		1		1	1			55	8	66
<i>Moenkhausia</i> sp.	32	133	299	136	126	12	1	9	1	749
Erythrinidae										
<i>Erythrinus erythrinus</i> (Bloch & Schneider, 1801)	5	2		6	2		1			16
<i>Hoplias malabaricus</i> (Bloch 1794)	11	3	15	4	21	24	1	1	9	89
Lebiasinidae										
<i>Pyrhulina brevis</i> Steindachner 1876	113	165	222	95	10		38	169	436	1248
Cichliformes										
Cichlidae										
<i>Aequidens tetramerus</i> (Heckel 1840)	12	114	521	11	93	323	4	18	153	1249
<i>Apistogramma</i> sp.	60	186	399	118	360	144	64	61	966	2358
<i>Crenicichla</i> sp.	1	1			6	8		6	43	65
Cyprinodontiformes										
Cynolebiidae										
<i>Anablepsoides micropus</i> (Steindachner 1863)	24	90	11	7	2		100	104	43	381
Gymnotiformes										
Gymnotidae										
<i>Gymnotus coropinae</i> Hoedeman 1962		1			1	3	2			7
<i>Gymnotus</i> sp.		2					2			4
Hypopomidae										
<i>Brachyhypopomus pinnicaudatus</i> (Hopkins, Comfort, Bastian & Bass 1990)	12	37	22	8	10	35	7	27	88	246
Rhamphichthyidae										

Taxon	Bambú stream		Floresta stream			Poço stream			Total abundance	
	Forest	Edge	Pasture	Forest	Edge	Pasture	Forest	Edge		Pasture
<i>Gymnorhamphichthys rondoni</i> (Miranda Ribeiro 1920)	18	2		276			26	1		323
Sternopygidae										
<i>Eigenmannia virescens</i> (Valenciennes 1836)				1	37	58		8		104
<i>Sternopygus macrurus</i> (Bloch & Schneider 1801)	6			2		1	4			13
Siluriformes										
Auchenipteridae										
<i>Tatia</i> sp.								2		2
Callichthyidae										
<i>Callichthys callichthys</i> (Linnaeus 1758)	2	1		10	18	1		2		34
<i>Corydoras ourastigma</i> Nijssen 1972							5			5
<i>Megalechis thoracata</i> (Valenciennes 1840)				1						1
Cetopsidae										
<i>Cetopsis parma</i> Oliveira, Vari & Ferraris 2001	8			2		2	4			16
<i>Helogenes</i> cf. <i>gouldingi</i>	7	16		9			20	8		60
Loricariidae										
<i>Ancistrus</i> sp. 1	14			15			52	3		84
<i>Ancistrus</i> sp. 2	4			2			7			13
<i>Farlowella smithi</i> Fowler 1913	9			13		1	15			38
<i>Rineloricaria lanceolata</i> (Günther 1868)	6			6			1			13
<i>Rineloricaria</i> sp.	4			9	1	3				17
Synbranchiformes										
Synbranchidae										
<i>Synbranchius marmoratus</i> Bloch 1795	1		2		2	1	1		2	9
Abundance	361	773	1491	853	856	653	362	488	1751	7588
Richness	22	17	8	25	17	17	22	17	11	32

Results

A total of 7588 individuals were collected, corresponding to 32 species, 15 families and six orders. Siluriformes was the most species-rich order (11 species), followed by Characiformes (10), Gymnotiformes (six) and Cichliformes (three). Cyprinodontiformes and Synbranchiformes were represented by one species each. The most abundant species were *Apistogramma* sp. (2358 individuals), *Aequidens tetramerus* (Heckel 1840) (1249) and *Pyrhulina brevis* Steindachner 1876 (1248), which comprised 64% of all collected individuals. Conversely, *Gymnotus* sp., *Tatia* sp. and *Megalechis thoracata* (Valenciennes 1840) were considered rare, with less than five individuals found for each species (Table 1).

Species richness showed a longitudinal variation between the stream stretches (GLMM; F-value = 22.03, $p < 0.01$, $df = 16$), being greater in forest environments (30 species; mean = 15.88 ± 2.57) than in edge (24 species; mean = 11.55 ± 2.45 ; $t = -2.81$, $p = 0.01$) and pasture environments (20 species; mean = 9.11 ± 2.14 ; $t = -5.68$, $p < 0.01$). Among streams, the highest species richness was observed in the *Floresta* Stream (29 species; mean = 12.33 ± 3.53), followed by *Pogo* (28 species; mean = 12.33 ± 3.57) and *Bambú* streams (26 species, mean of 11.88 ± 4.28). However, this difference was not significant (GLMM; F-value = 0.21, $p < 0.82$, $df = 16$) (Figure 2). A significant interaction between stretch and stream (GLMM; F-value = 3.35, $p < 0.04$, $df = 16$) occurred due to richness increasing in

pasture environment in the *Floresta* Stream in comparison to the same environment in the other streams ($t = 2.58$, $p = 0.02$, $df = 16$) (Figure 2A).

Abundance also showed a longitudinal variation between the stream stretches (GLMM; F-value = 14.20, $p < 0.01$, $df = 16$), being lower in forest environments (1576 individuals; mean = 262.44 ± 132.32) than in edge (2117 individuals; mean = 235.33 ± 100.72 ; $t = 2.63$, $p = 0.02$) and pasture environments (3895 individuals; mean = 432.77 ± 226.06 ; $t = 4.91$, $p < 0.01$) (Figure 2B). Among streams, the highest abundance was observed in the *Bambú* Stream (2625 individuals; mean = 291.66 ± 206.66), followed by *Pogo* (2601 individuals; mean = 289.00 ± 228.52) and *Floresta* streams (2362 individuals; mean = 262.44 ± 132.32), but this difference was not significant (GLMM; F-value = 0.07, $p < 0.93$, $df = 16$; Fig. 3). A significant interaction between stretch and stream was due to abundance decrease in pasture environment in the *Floresta* Stream in comparison to the same environment in the other streams ($t = -5.52$, $p < 0.01$, $df = 16$) (Figure 2B).

The fish species composition differed longitudinally between the stream stretches (F model = 6.83, $p < 0.01$, $df = 2$), between streams (F model = 2.78, $p = 0.02$, $df = 2$) and in the interaction between stretches and streams (F model = 2.78, $p < 0.01$, $df = 4$). The clusters formed by ordination analysis (nMDS) reflected a strong difference in fish composition between forest and pasture stretches, the latter being mainly composed of species present in edge environments. The ordination also summarized the differences in fish composition between streams, highlighted by the grouping of the same symbols (Figure 3).

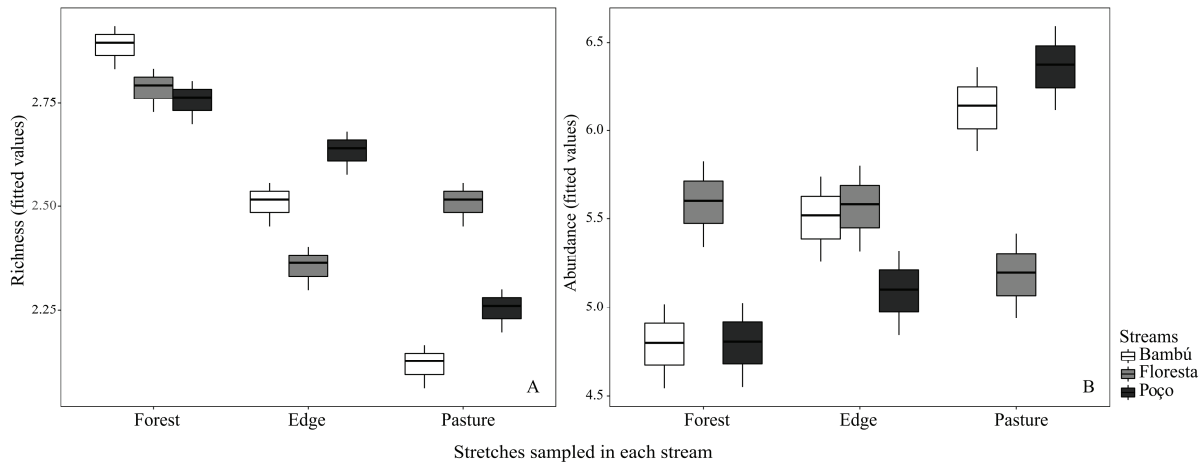


Figure 2. A - Species richness and B - Abundance between stretches in different environments of three low-order Amazonian streams, State of Acre, Brazil.

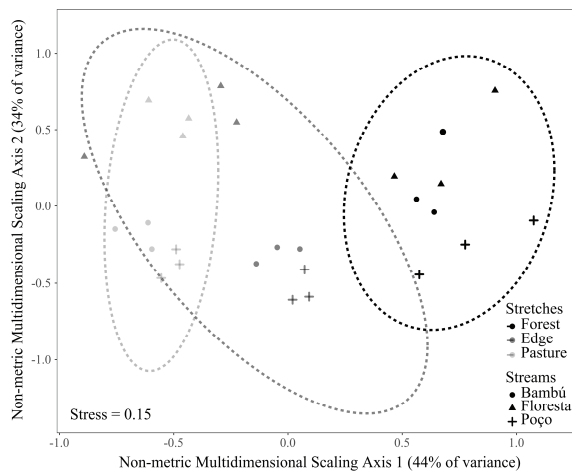


Figure 3. Ordination produced by non-metric multidimensional scaling to fish composition in forest, edge and pasture environments of three low-order Amazonian streams, State of Acre, Brazil.

Seven species were classified as indicators of environments in the longitudinal gradient (stretches) due to high specific and fidelity: *Ancistrus* sp. 1 (IndVal = 0.96; $p = 0.04$), *Ancistrus* sp. 2 (IndVal = 1.00; $p = 0.04$), *Farlowella smithi* Fowler 1913 (IndVal = 0.97; $p = 0.03$), *Gymnorhamphichthys rondoni* (Miranda Ribeiro 1920) (IndVal = 0.99; $p = 0.05$), *Rineloricaria lanceolata* (Günther 1868) (IndVal = 1.00; $p = 0.04$) and *Sternopygus macrurus* (Bloch & Schneider 1801) (IndVal = 0.92; $p = 0.04$) associated with forest environments; and *A. tetramerus* (IndVal = 0.80; $p = 0.05$) with pasture. None species was indicator of edge environments. Between streams, only *Hyphessobrycon bentosi* Durbin 1908 (IndVal = 0.98; $p = 0.03$) was indicator of the Floresta Stream.

Discussion

The analysis of longitudinal gradient in the three sampled streams allowed the proposed objectives to be achieved. In a clear way, there were variations in fish species richness, abundance and composition along the spatial scale as a consequence of changes in the riparian vegetation. The fish communities evaluated were composed mainly of small Characiformes and Siluriformes, a typical configuration of the fish fauna of the Amazon Basin (Lowe-McConnell, 1987; Barros et al., 2011; Ramalho et al., 2014; Zuanon et al., 2015) and other Neotropical areas in Brazil (Shibatta, Gealh & Bennemann, 2007; Casatti, Silva, Langeani, & Castro, 2006; Casatti, Romero, Teresa, Sabino, & Langeani, 2010; Ávila et al., 2018).

Community richness and abundance differed along the longitudinal gradient between stretches with riparian vegetation of forest, edge and pasture environments. The stretches in pasture environment presented few species and high abundance (low diversity), while forest environments presented a larger number of species and lower abundance (high diversity). This reinforces the predictions for the effects of habitat physical degradation on the structure of fish assemblages in pasture environments (Casatti et al., 2006; 2010) and the changes in fish abundance induced by environmental impacts in streams (Dias, Magnusson, & Zuanon, 2010). The diversity loss occurs because riparian vegetation is one of the main determinants of biodiversity maintenance and functioning of aquatic environments (Lorion & Kennedy, 2009; Wantzen & Mol, 2013).

Anthropogenic pressure in these regions can affect the entire ecosystem (Tudesque, Grenouillet, Gevrey, Khazraie, & Brosse, 2012) and therefore may also influence the structure of aquatic organisms such as fish fauna (Bordignon et al., 2015; Allard, Popec, Vigouroux, & Brosse, 2016; Taniwaki, Piggott, Ferraz, & Matthaei, 2016).

Fish composition varied along the longitudinal gradient as a function of riparian vegetation, with a conspicuous differentiation of forest environment communities in relation to the altered pasture environment. Longitudinal variations in the composition of riparian communities are naturally expected (Mendonça et al., 2005; Montag et al., 2018), but changes in riparian vegetation were more pertinent to determine the fish composition in the low-order streams sampled. This fact is directly associated with degradation of the marginal physical habitat, which causes homogenization of riparian vegetation, substrate microhabitats, water column reduction due to excessive sedimentation and silting (Casatti et al., 2006; 2010; Ramalho et al., 2014; Montag et al., 2018). This homogenization of the habitat structure can also cause homogenization of fish communities (Cetra & Petrere Jr., 2007). Here, the results show a change in composition and homogenization of fish communities in the forest-pasture direction, indicated by the richness decrease and total separation of the ordination analysis groups.

Differences in species composition are also reflected in the distinct indicator species. Among streams, the segregation in composition reflect a spatial segregation in communities, expected due to intrinsic characteristics of each stream (Kemenes & Forsberg, 2014), but strong associations were found only for *H. bentosi*, the single indicator species of the Floresta Stream. However, the composition of indicator species shows that the removal of riparian vegetation seems to favor opportunistic species that substitute the most sensitive ones, being more tolerant and adaptable to such variations (Devictor, Julliard & Jiguet, 2008; Ceneviva-Bastos & Casatti, 2007; Casatti et al., 2010; Teresa, Casatti, & Cianciaruso, 2015). The cichlid classified as indicator of pasture environments (*A. tetramerus*), as well as other species of the family Cichlidae, are widely distributed and are commonly associated with environmental conditions because they are tolerant and generalist in diet (Beatty, Morgan, Keleher, Allen & Sarre, 2013; Zuanon et al., 2015). In addition, because they occur in all environments studied, as well as *Apistogramma* sp. (not classified as indicator), these species seem to be favored by changes in the aquatic habitat caused by the removal of riparian vegetation.

Indicator species of forest environments are dependent on conditions provided by riparian vegetation, mainly because they feed on allochthonous material and aquatic insects (e.g. *G. rondoni* and *S. macrurus*; Zuanon, Bockmann, & Sazima, 2006; Zuanon et al., 2015) and periphyton and organic matter on the substrate (e.g. *Ancistrus* spp., *F. smithi* and *R. lanceolata*; Ibañez et al., 2007; Zuanon et al., 2015). In the case of *G. rondoni* for example, the sediment dependence, sand-dwelling habit and morphology makes the species highly specialized (Schwassmann, 1976; Zuanon et al., 2006; 2015; Soares, Rosa, Silva, Albrecht, & Caramaschi, 2017). Thus, the maintenance of riparian forest environments is extremely important for the survival of these and other species because they provide a wide supply of resources and maintain these resources (Castello et al., 2013), mainly in terms of woody debris (Montag et al., 2018). The vulnerability of these species (*Ancistrus* spp., *F. smithi*, *G. rondoni*, *R. lanceolata* and *S. macrurus*) due to alterations in riparian vegetation is reflected in the rarity or total absence in the stretches modified due to pasture insertion.

Conclusion

This study showed that changes in riparian vegetation in low-order Amazonian streams affect fish communities in terms of richness, abundance and composition. Thus, there is a reduction in diversity in the forest-border-pasture direction, attributed to richness decreasing and increasing of generalist species abundance. The environment alteration also favored the change in species composition and the homogenization of communities in pasture environments. This change in composition was reflected in the selection of a generalist species as indicator of pasture environments and specialist species dependent on complex environments as indicators of forest environments. Finally, the results presented reinforce the negative influence of riparian vegetation removal on biological communities, especially in small fish communities in low-order streams, and thus, provide subsidies that can help in the conservation and management of these systems.

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