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Fish community composition differs in rural and urban Neotropical streams

Composição da comunidade de peixes difere em riachos neotropicais rurais e urbanos

Eduarda Samantha Ribeiro[1], Tatiane Mantovano[1]*, Dyego Leonardo Ferraz Caetano[1], Luana Gabriela Marques da Silva[1], Leandro da Silva[1], Thiago Rodrigues Barbosa[1] and Fernando Emmanuel Gonçalves Vieira[1]

[1]Grupo de Estudos e Pesquisa em Recursos Hídricos e Ecologia Aplicada – GEPRHEA, Universidade do Norte do Paraná – UENP, Av. Manoel Ribas, 215, Centro, CEP 86400-000, Jacarezinho, PR, Brasil

*e-mail: mantovano.t@outlook.com

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Abstract: Aquatic ecosystems have a great diversity of habitats, including streams that are of extreme ecological and economic importance but have undergone impacts, such as the input of domestic and industrial waste and deforestation of riparian forest. However, few studies have been carried out on the diversity, specifically beta diversity, of fish assemblages in urban and rural streams. **Aim:** In this context, the objective of this work was to verify if there are differences in fish assemblage structure between urban and rural streams, with focus on beta diversity. **Methods:** We selected 12 streams, 6 urban and 6 rural, located in the municipalities of Ourinhos (SP) and Jacarezinho (PR) in the Paranapanema river basin. Fishes were collected using a sieve. Concomitantly water quality variables were also measured, such as temperature, pH, conductivity and total dissolved solids. To assess whether fish beta diversity differed between streams, a dispersion homogeneity test (Permutational Analysis of Multivariate Dispersions) was calculated. **Results:** Significant differences were observed in conductivity and total dissolved solids, both with higher values for urban streams. In urban streams, 12 species of fish were identified, five of which were exclusive, while in rural streams, 18 species were recorded, ten of which were exclusive. Furthermore, both composition and beta diversity were significantly different between rural and urban streams, with the highest values recorded in rural streams. **Conclusions:** This study provides evidence of how urbanization impacts the composition and beta diversity of fish in streams and highlights the importance of sustainable management such as effluent control, restoration of riparian vegetation, and the determination of protection areas.

Keywords: diversity; aquatic ecosystems; urbanization; Teleostei.

Resumo: Os ecossistemas aquáticos têm uma grande diversidade de habitats, incluindo riachos de extrema importância ecológica e econômica, mas que sofreram impactos, como a entrada de resíduos domésticos e industriais e o desmatamento da mata ciliar. No entanto, poucos estudos foram realizados sobre a diversidade, especificamente a diversidade beta, de grupos de peixes em córregos urbanos e rurais. **Objetivo:** Nesse contexto, o objetivo deste trabalho foi verificar se há diferenças significativas nos padrões de diversidade beta em assembleias de peixes de córregos urbanos e rurais. **Métodos:** Foram selecionados 12 riachos, 6 urbanos e 6 rurais, localizados nos municípios de Ourinhos (SP) e Jacarezinho (PR), na bacia do rio Paranapanema. Os peixes foram coletados por meio de uma peneira. Concomitantemente, também foram medidas as variáveis de qualidade da água, como temperatura, pH, condutividade e sólidos totais dissolvidos. Para avaliar se há variação na composição entre os riachos, foi calculada a Permidisp. **Resultados:** Foram observadas diferenças significativas na condutividade e no total de sólidos dissolvidos, ambos com valores mais altos para os córregos urbanos. Foram observadas diferenças significativas em relação à composição de peixes entre trechos de córregos urbanos e rurais, bem como para a diversidade beta, com os valores mais altos sendo encontrados em córregos rurais. **Conclusões:** A composição de peixes difere entre os riachos rurais e urbanos. A diversidade beta dos riachos rurais é significativamente diferente e maior do que a dos urbanos, o que nos permite inferir que eles estão melhor preservados. Esses resultados podem funcionar como uma ferramenta eficaz para medidas de gestão, como controle de efluentes e restauração da vegetação ribeirinha e determinação de áreas de proteção para córregos urbanos.

Palavras-chave: diversidade; ecossistemas aquáticos; urbanização; Teleostei.

1. Introduction

Human activities have caused profound and lasting impacts on the environment, mainly through changes in land use (Leal et al., 2016; Brejão et al., 2018). These changes lead to habitat destruction, degradation and fragmentation, being one of the main causes of biodiversity loss on a global scale (Souza et al., 2015). In addition, human impacts on landscapes often reduce the capacity of ecosystems to provide essential services, such as clean air and water, as well as natural resources (Smith et al., 2013).

Streams are of vast economic importance as they are generally used for water supply, irrigation and other human activities (Andrade et al., 2012) but are subject to numerous negative effects caused by surrounding land use (Bonato et al., 2012; Carvalho et al., 2017; Castro et al., 2018). Particularly, streams located close to urban areas are among the most impacted (Pusey & Arthington, 2003) due to soil impermeabilization as well as the constant input of high loads of pollutants of domestic or industrial origin (Pusey & Arthington, 2003; Cunico et al., 2012; Garcia et al., 2021). The expansion of urbanization often leads to the degradation of riparian vegetation, intensified by interventions such as the channelization and the homogenization of streams. These modifications reduce the structural complexity of aquatic habitats, causing negative impacts on the fish fauna. In addition to providing physical and food resources for aquatic organisms, the riparian vegetation plays a crucial role in regulating the physical and chemical parameters of water (Casatti, 2010; Daga et al., 2012; Collier et al., 2019).

The replacement of native vegetation with impermeable surfaces in urban streams increases the incidence of sunlight and sedimentation in water bodies, resulting in increased temperature and total dissolved solids levels, with direct impacts on local fish communities (Casatti, 2010; Larentis et al., 2022). Another process related to the total or partial removal of marginal vegetation is the increase in water turbidity, due to the large amount of particulate solids that flow into the channel. Soil leaching in areas adjacent to water bodies can reduce the heterogeneity of the water column, causing a decrease in fish species richness (Allan, 2004). Thus, deforestation affect the availability of habitats, food resources and physicochemical conditions of streams, reducing the occurrence of specialized species and benefiting those with greater tolerance and generalist characteristics (Teresa et al., 2015). Urbanization has adverse impacts on the alpha and beta diversity of fish assemblages (Borges et al., 2020), as it can eliminate more sensitive species from local assemblages and contribute to biotic homogenization on a regional scale (Hewitt et al., 2010; Ortega et al., 2021).

However, streams located outside urban areas can also suffer from human development, mainly due to agricultural and livestock activities, although they present more subtle effects (Garcia et al., 2021). Activities such as livestock farming and monocultures have generated significant environmental impacts, including greater sediment input into rivers, changes in the physical and chemical properties of water, reduction of natural shelters through the homogenization of environments and, consequently, a decline in species diversity (Casatti et al., 2009; Teresa & Casatti, 2012; Cunha & Juen, 2017).

One approach to measure environmental changes in urban and rural streams is through beta diversity, defined as the variation in species composition between two samples, or habitats, within a geographic area (Whittaker, 1960). Beta diversity has been used to describe various phenomena, such as compositional heterogeneity or differentiation between different locations (Al-Shami et al., 2013). Numerous environmental factors influence the distribution of species in space and, consequently, the structure of the community. The response is related to the organisms' niche requirements (Grinnell, 1917; Sales et al., 2021) and/or to the species' ability to disperse (Hubbell, 2001; Hill et al., 2017). More preserved environments, such as rural streams, generally have a greater availability of habitats and resources and, thus, a greater number of species can coexist in different locations, increasing beta diversity (Astorga et al., 2014; Bini et al., 2014; Ortega et al., 2018; Rodríguez et al., 2019).

The expansion of both agricultural and urban activities is causing significant changes in the biodiversity of aquatic ecosystems (Miiller et al., 2021). Therefore, this research aims to verify if there are differences in the structure of local fish assemblages in urban and rural streams located in a same drainage, the Middle Paranapanema River. To this end, it was predicted that beta diversity patterns between these environments would be significantly different, and the highest values would be recorded for rural streams, due to the lower degree of anthropogenic impacts.

2. Material and Methods

2.1. Study area

The Paraná River basin, with approximately 900,000 km^2, is the second largest hydrographic basin in South America and the fourth largest in the world (Langeani et al., 2007). The Paranapanema River (22°46'38"S, 50°02'55"W), with a length of 930 km, crosses 247 municipalities, 115 in the state of São Paulo and 132 in the state of Paraná. Having its headwaters in Capão Bonito-SP, on the western slope of the Serra de Paranapiacaba and flowing into the left margin of the Paraná River, it is one of the main tributaries of the upper Paraná River basin (Ziesler & Ardizzone, 1979; Jarduli et al., 2020).

The mesoregion comprising the Paranapanema river basin has an area of about 109,600 km^2 (Sampaio, 1944). It has 11 hydroelectric power plants distributed along its channel. The soil is mainly used for agricultural activities. Located in the Middle Paranapanema, the selected study area comprises 12 streams, six urban and six rural, in the municipalities of Ourinhos (SP) and Jacarezinho (PR), spanning a stretch of 328 km from the mouth of the Apiaí-Guaçu river to the Salto Grande dam (Sampaio, 1944) (Figure 1).

2.2. Sampling design

The sites were chosen in order to obtain sections with the same local coverage, with similar climatic and geomophological characteristics, with replicas at two different levels of human impacts, six rural and six urban sites. This was possible by considering two municipalities. The urban sites did not have consolidated riparian vegetation, with few trees,

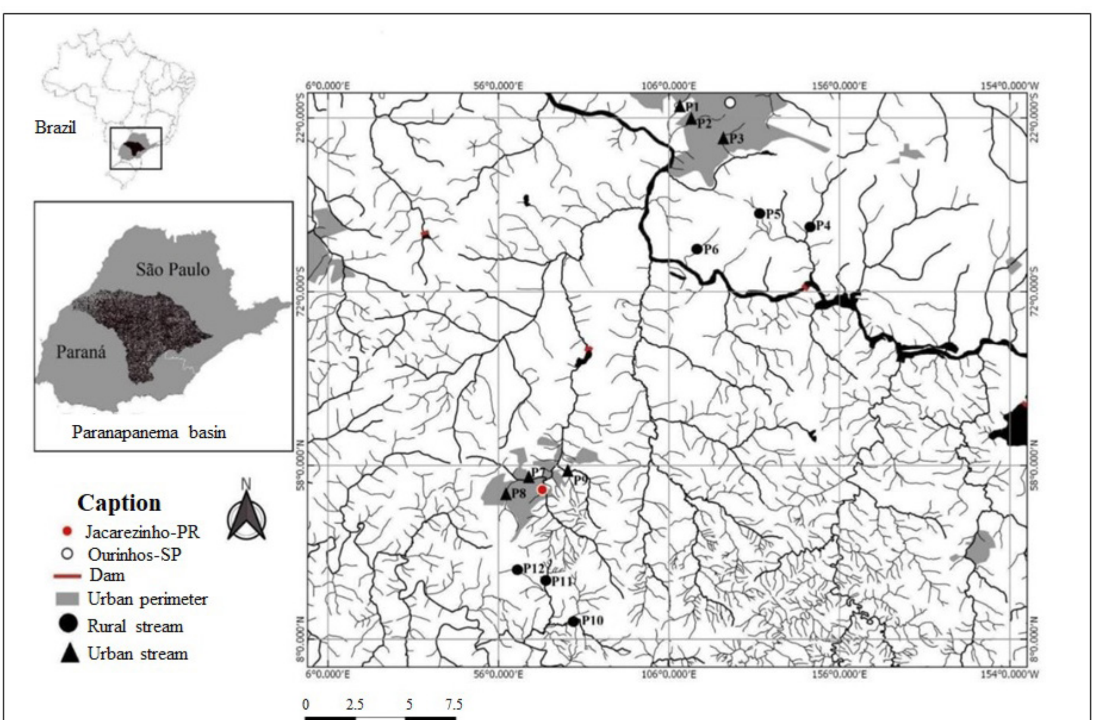

Figure 1. Location of urban and rural streams in the municipalities of Ourinhos (SP) and Jacarezinho (PR).

and the soil was more exposed, with some silted areas. The substrate of the urban sites showed little diversity, with a predominance of sand and pebbles, as well as clay, rocks and some stretches with fragments of concrete. The rural sites had, on average, a width ranging between 3 and 10 meters of riparian vegetation, with some trees, shrubs and some stretches with a predominance of grasses. The predominant substrates were clay and sand, but there were also logs, leaves, rocks and gravel.

Fish collection was carried out with the aid of three capture instruments, a round sieve (0.5 cm mesh × 0.78 m diameter), rectangular sieve (0.5 cm mesh × 1.02 m length × 0.81 m wide) and a seine (0.5 cm mesh × 3.0 m long × 1.0 m height), in the dry season of 2019 (autumn). The collections were carried out during the day, in the upstream to downstream direction in sections of 50 meters blocked by nets of 3.0 mm mesh for 50 minutes in each section. After capture, the specimens were anesthetized in a clove oil solution, euthanized by immersion in 10% formalin, subsequently preserved in 70% alcohol, and identified according to Ota et al. (2018).

Concomitantly with the collection of fish species, the water temperature (°C), pH, electric conductivity (µS/cm) and total dissolved solids (ppm) were obtained. These water quality variables were obtained with measurements at the edges and center of four random transverse transects using a Hanna HI-991300 multiparameter probe, placed 20 cm below the surface of the water.

2.3. Data analysis

To verify significant differences regarding the abiotic parameters between urban and rural streams, an Analysis of Variance [ANOVA with significance of $p < 0.05$ (Zar, 1999)] was performed, based on the data obtained from temperature, pH, conductivity and total solids. In case of significant differences, we used a Tukey's test at 5% probability. Assumptions of ANOVA were verified and Bonferroni correction was used to adjust significance values. The abiotic variables available for urban and rural streams were also summarized using principal component analysis (PCA; Gauch Junior, 1986), and the axes retained for interpretation were those that presented eigenvalues greater than those obtained by the random "broken stick" model (Jackson, 1993). To avoid collinearity, electrical conductivity variable was excluded.

We calculated the constancy index, which represents the frequency of fish species by taking into account the number of times they were found in the samples. We used the following formula: $C = p.100/P$, where C represents the constancy index, p is the number of samples in which the species was found, and P is the total number of samples. According to this index, if $C \geq 50\%$ the species is considered "constant", if $25\% \leq C \leq 50\%$ the species is "accessory" and if $C \leq 25\%$ the species is "accidental" (Dajoz, 1973).

To verify the efficiency of fish sampling in the streams studied, the species accumulation curve was calculated. The expected richness was estimated by the Chao 2 index (95% confidence interval), which has been used in short-term inventories (Chazdon et al., 1998).

To verify if the composition of fish species in the environments is different, a multivariate permutational analysis of variance (PERMANOVA; "adonis2" function from the vegan R package- Anderson, 2001) was performed using a dissimilarity matrix calculated by the Sørensen index with 999 permutations. This test is quite robust for identifying changes in community structure (Anderson & Walsh, 2013). Furthermore, to verify whether beta diversity is higher in rural than urban streams, the dispersion homogeneity test was used (PERMDISP; Anderson, 2006; "betadisper" function from the vegan R package). The calculation of pseudo-F and p values was based on 999 permutations of the residuals under a reduced model. The boxplot was used to visualize the dispersion of beta diversity based on the species presence and absence data. All analyses were performed using R software (R Core Team, 2018) with the vegan package (Oksanen et al., 2016).

3. Results

Considering the environmental variables analyzed for rural and urban streams, significant differences ($p<0.05$) were observed in total dissolved solids (ppm) and conductivity (µS), which showed higher values in urban streams (Figure 2). It was also possible to observe through the PCA a clear distinction between rural and urban systems based on the environmental variables evaluated. Rural samples were positively related to temperature and pH. On the other hand, urban samples were positively related to the total solids variable. The first two main axes (PC1 and PC2) together explain 66.89% of the total variation, consistently reflecting the distinct environmental patterns between the areas analyzed (Figure 3).

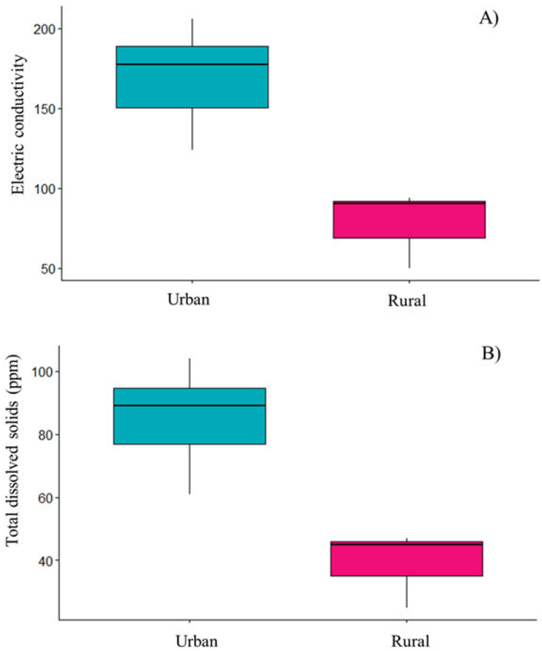

Figure 2. Electrical conductivity (A) and total solids (B) in urban (blue) and rural (red) streams.

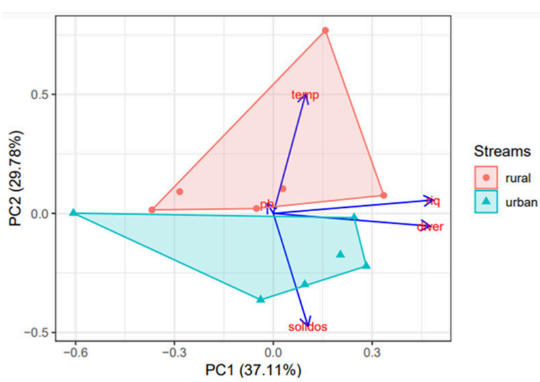

Figure 3. Principal Coordinate Analysis (PCoA) for abiotic data in in urban (green) and rural (purple) streams. Temp = temperature; solidos = total solids; diver = diversity of substrates; ph = pH; riq = richness of substrates.

Throughout the study, 23 fish species belonging to eight families were identified. In urban streams, 12 fish species were identified, of which five were exclusive, while in rural streams 18 species were recorded, of which ten were exclusive. Considering the families, one family was exclusive to urban streams (Erythrinidae), one to rural streams (Crenuchidae) and six occurred in both environments (Characidae, Callichthyidae, Loricariidae, Heptapteridae, Cichlidae and Poeciliidae) (Table 1). Among these families, the richest was Characidae, with nine species. Erythrinidae had only *Hoplias*

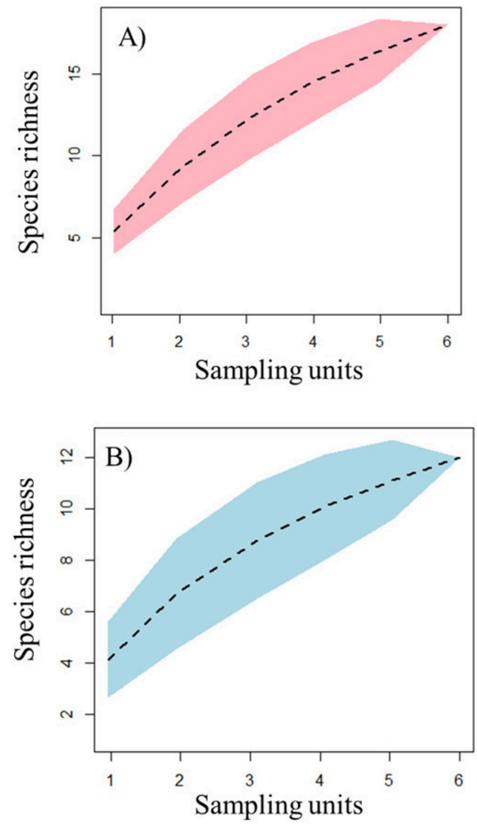

Figure 4. Species accumulation curves for rural (A) and urban (B) streams.

malabaricus. In urban streams, mean richness was 5.16 (coefficient of variation=25%), while in rural streams it was 5.83 (coefficient of variation=59%).

The results obtained by accumulation curves indicated that both rural (Figure 4A) and urban (Figure 4B) streams did not reach an asymptote.

The beta diversity differed significantly (PERMDISP, DF=1; F = 6.51; p =0.028), with higher values of distance from centroid and consequently beta diversity recorded for rural streams (average distance from centroid of 0.44) compared to urban (average distance from centroid of 0.23). Considering the composition of fish species, a significant difference (Pseudo-F = 3.24; p=0.023) was observed (Figure 5). Across urban streams, 4 species were constant, 2 accessory and 6 accidental, while in rural streams 2 were constant, 9 accessory, and 7 accidental (Table 1). Ten species were exclusive to rural streams, such as *Characidium zebra* and *Apareiodon piracicabae*, while five were exclusive to urban streams, such as *Hoplias malabaricus* (Table 1).

4. Discussion

The study observed significant differences in the structure of local fish assemblages between urban

Table 1. List of fish species (organized into families) found in urban and rural streams of the Paranapanema River Basin.

Taxon		CI	Cla		CI	CI
Characidae						
Apareiodon piracicabae (Eigenmann, 1907)	Urban	0		Rural	16.6	Ac
Astyanax bockmanni Vari & Castro, 2007	Urban	50	A	Rural	66.6	Ct
Astyanax lacustris (Lütken, 1875)	Urban	16.6	Ac	Rural	16.6	A
Brycon americus iheringii (Boulenger, 1887)	Urban	16.6	Ac	Rural	66.6	Ct
Brycon americus exodon (Eigenmann, 1907)	Urban	0		Rural	16.6	Ac
Brycon americus sp.	Urban	0		Rural	16.6	Ac
Piabarchus stramineus (Eigenmann, 1908)	Urban	0		Rural	16.6	Ac
Serrapinnus notomelas (Eigenmann, 1915)	Urban	0		Rural	33.3	A
Oligosarcussp.	Urban	0		Rural	16.6	Ac
Crenuchidae						
Characidium zebra Eigenmann, 1909	Urban	0		Rural	33.3	A
Erythrinidae						
Hoplias malabaricus (Bloch, 1794)	Urban	33.3	A	Rural	0	
Callichthyidae						
Corydora saeneus (Gill, 1858)	Urban	83.3	Ct	Rural	50	A
Hoplosternum littorale (Hancock, 1828)	Urban	16.6	Ac	Rural	0	
Loricariidae						
Hypostomus ancistroides (Ihering, 1911)	Urban	100	Ct	Rural	50	A
Hypostomus nigromaculatus (Schubart, 1964)	Urban	16.6	Ac	Rural	0	
Heptapteridae						
Rhamdia quelen (Quoy & Gaimard, 1824)	Urban	0		Rural	16.6	A
Imparfinis sp.	Urban	33.3	A	Rural	0	
Cichlidae						
Geophagus brasiliensis (Quoy & Gaimard, 1824)	Urban	16.6	Ac	Rural	33.3	A
Crenicichla britzkii Kullander, 1982	Urban	0		Rural	16.6	Ac
Coptodon rendalli (Boulenger, 1897)	Urban	16.6	Ac	Rural	0	
Oreochromis niloticus (Linnaeus, 1758)	Urban	0		Rural	16.66667	Ac
Poeciliidae						
Poecilia reticulata Peters, 1859	Urban	100	Ct	Rural	50	A
Phalloceros sp.	Urban	83.3[	Ct	Rural	0	
Phalloceros harpagos Lucinda, 2008	Urban	0		Rural	50	A

Constance index (CI) is indicated as: Constant (Ct) - (> 50% of the samples), accessory (A) - (between 25% and 50%) and accidental (Ac) - (< 25%) (Dajoz, 1973) - Cla - classification.

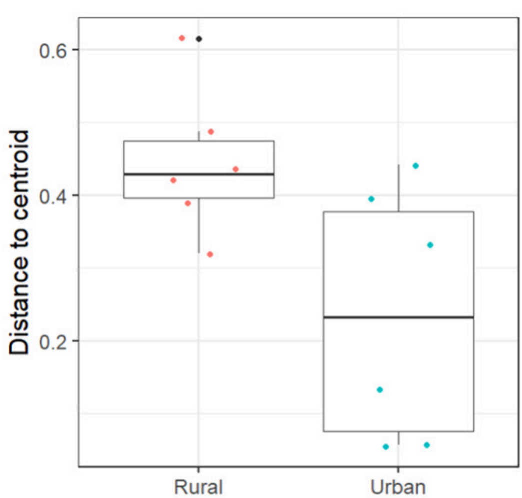

Figure 5. Boxplot of species composition in urban (green) and rural (purple) streams.

and rural streams. Furthermore, higher beta diversity values were recorded for rural streams, supporting the prediction. Despite being modified by the use of the surrounding land (e.g., contamination by agricultural effluents), as it is located in a rural area, these streams are less affected by domestic and industrial organic waste.

Environmental characterization through chemical and physical parameters of water is essential for environmental diagnostics. Significant differences were observed only in conductivity and total dissolved solids, both higher in urban streams due to the higher concentration of ions (Baggio et al., 2016). These variations may be related to land use, dumping of domestic and industrial waste, or siltation (Molina et al., 2017; Bhateria & Jain, 2016). The correlation between dissolved solids and electrical conductivity occurs due to the presence of ions (Dos-Santos et al., 2018). Increased total solids can impact aquatic ecosystems, retaining bacteria and organic waste and impairing fish spawning (Quinelato et al., 2020). Previous studies have also associated high concentrations of dissolved solids with poor water quality in urban streams (Quinelato et al., 2020; Pinto et al., 2009; Araújo & Oliveira, 2013).

In rural streams, the water temperature variable was positively related to sampling. Removing riparian vegetation in rural areas reduces shade over streams, allowing more sunlight and increasing water temperatures, which is detrimental to these aquatic ecosystems (Lóis et al., 2011). In contrast, urban streams may be partially shaded by residential infrastructure and bridges. Studies indicate that the absence of riparian vegetation can result in significant increases in water temperature. For example, after riparian vegetation was removed in a river basin, maximum water temperatures increased by up to 7°C after cutting (Lóis et al., 2011).

Rural streams had a higher number of fish families compared to urban streams, reflecting the impacts of urbanization, such as pollution, habitat loss, and land use changes, which reduce water quality and biodiversity (Miiller et al., 2021). Urban runoff intensifies water flow, increases flooding, and raises the concentration of nutrients, pesticides, and heavy metals, resulting in the loss of aquatic species (Marques & Cunico, 2021).

Environmental degradation favors opportunistic species, such as *Poecilia reticulata*, an exotic species abundant on the banks of degraded streams due to its tolerance to hypoxia, small size, and generalist habits (Kramer & Mehegan, 1981; Ferreira & Casatti, 2006; Langeani et al., 2007; Teresa & Casatti, 2012). Its introduction can alter food chain dynamics, increase primary productivity, and modify nitrogen fluxes (Collins et al., 2016), in addition to reducing the density of native species (Holitzki et al., 2013). Another exotic species found was *Oreochromis niloticus*, restricted to rural streams, possibly due to escape from fish farms (Caetano et al., 2021). Considered one of the most dangerous invasive species in tropical and subtropical regions, *O. niloticus* impacts trophic cascades, water quality, and ecosystem function (Zengeya et al., 2013; Gu et al., 2015; Stauffer Junior et al., 2022). In urban streams, *Hoplias malabaricus* (Erythrinidae) was the only species recorded. Opportunistic predator widely distributed in South America, it stands out for its resistance to disturbed environments, prolonged food deprivation and temperature variations (Petry et al., 2007, 2010; Daga et al., 2012).

The Erythrinidae family has sedentary habits, preferring calm water streams (Blanco et al., 2010). The Characidae family presents a great diversity of species due to the variety of body shapes, facilitating the occupation of different ecological niches and the obtaining of food (Dias & Fialho, 2009). Previous studies also identified Characidae as the most abundant family (Sant'Anna et al., 2006). The only species of the Crenuchidae family in the study, *C. zebra*, was found only in rural streams, associated with greater water flow, which may be linked to its fusiform body (Casatti & Castro, 2006). In addition, *C. zebra* is related to environments with greater species diversity, justifying its absence in urban areas (Caetano et al., 2016). Another species restricted to rural streams was *A. piracicabae* (Characidae), probably due to its predominantly insectivorous diet (Lampert et al., 2022), since dietary diversity is lower in urban streams.

Beta diversity of fish assemblages differed significantly between rural and urban streams, being higher in rural streams due to greater variation in species composition (Chase & Leibold, 2002; Leibold et al., 2004). This suggests that rural streams contribute more to regional diversity, influenced by environmental heterogeneity and the spatial configuration which increase local diversity (Teshima et al., 2017; Dala-Corte et al., 2019; Martins et al., 2021). Another factor that can justify the results are local drivers. One of the possible explanations is related to the ecological niche. Faced with anthropized environments, species considered sensitive to human alterations generally do not occur (Ferreira & Casatti, 2006). These processes consequently lead to differences in species composition, with an increase in the dominance of generalist and opportunistic species in more urbanized areas (Jones & Leather, 2012). It is well established in the literature that differences in environmental conditions, within a region, can generate a variation in species composition and an increase in beta diversity (Chase & Leibold, 2002; Leibold et al., 2004).

5. Conclusion

This study observed significant differences in fish beta diversity in stretches of rural and urban streams, with the highest values recorded in less disturbed (rural) environments. In this context, it is expected that the results found in this study support management measures, such as effluent control, the restoration of riparian vegetation and the determination of protection areas for urban streams.

Data availability

The datasets used and/or analysed during the current study are available from the corresponding author on request.

References

Allan, J.D., 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annu. Rev. Ecol. Evol. Syst. 35(1), 257-284. http://doi.org/10.1146/annurev.ecolsys.35.120202.110122.

Al-Shami, S.A., Heino, J., Che-Salmah, M.R., Abu-Hassan, A., Suhaila, A.H., & Madrus, A.M., 2013. Drivers of beta diversity of macroinvertebrate communities in tropical forest streams. Freshw. Biol. 58(6), 1126-1137. http://doi.org/10.1111/fwb.12113.

Anderson, M.J. 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecol. 26, 32-46. http://doi.org/10.1111/j.1442-9993.2001.01070.pp.x.

Anderson, M.J., 2006. Distance-based tests for homogeneity of multivariate dispersions. Biometrics 62(1), 245-253. PMid:16542252. http://doi.org/10.1111/j.1541-0420.2005.00440.x.

Anderson, M.J., & Walsh, D.C., 2013. PERMANOVA, ANOSIM, and the Mantel test in the face of heterogeneous dispersions: what null hypothesis are you testing? Ecol. Monogr. 83(4), 557-574. http://doi.org/10.1890/12-2010.1.

Andrade, C.F., Niencheski, L.F., Attisano, K.K., Milani, M.R., Santos, I.R., & Milani, I.C., 2012. Fluxos de nutrientes associados às descargas de água subterrânea para a Lagoa Mangueira (Rio Grande do Sul, Brasil). Quim. Nova 35(1), 5-10. http://doi.org/10.1590/S0100-40422012000100002.

Araújo, M.C., & Oliveira, M.B.M., 2013. Monitoramento da qualidade das águas de um riacho da Universidade Federal de Pernambuco, Brasil. Rev. Amb. Água 8, 247-257. http://doi.org/10.4136/ambi-agua.1192.

Astorga, A., Death, R., Death, F., Paavola, R., Chakraborty, M., & Muotka, T., 2014. Habitat heterogeneity drives the geographical distribution of beta diversity: the case of New Zealand stream invertebrates. Ecol. Evol. 4(13), 2693-2702. PMid:25077020. http://doi.org/10.1002/ece3.1124.

Baggio, H., Freitas, M.D.O., & Araújo, A.D., 2016. Análise dos parâmetros físico-químicos, oxigênio dissolvido, condutividade elétrica, potencial hidrogeniônico e temperatura, no baixo curso do Rio das Velhas-MG. Caminhos Geogr. 17(60), 105-117. http://doi.org/10.14393/RCG176008.

Bhateria, R., & Jain, D., 2016. Avaliação da qualidade da água do lago: uma revisão. Gest. Sustent. Recur. Hidricos. 2, 161-173.

Bini, L.M., Landeiro, V.L., Padial, A.A., Siqueira, T., & Heino, J., 2014. Nutrient enrichment is related to two facets of beta diversity for stream invertebrates across the United States. Ecology 95(6), 1569-1578. PMid:25039221. http://doi.org/10.1890/13-0656.1.

Blanco, D.R., Lui, R.L., Bertollo, L.A.C., Margarido, V.P., & Moreira Filho, O., 2010. Karyotypic diversity between allopatric populations of the group *Hoplias malabaricus* (Characiformes: Erythrinidae): evolutionary and biogeographic considerations. Neotrop. Ichthyol. 8(2), 361-368. http://doi.org/10.1590/S1679-62252010000200015.

Bonato, K.O., Delariva, R.L., & Silva, J.C., 2012. Diet and trophic guilds of fish assemblages in two streams with different anthropic impacts in the northwest of Paraná, Brazil. Zoologia 29(1), 27-38. http://doi.org/10.1590/S1984-46702012000100004.

Borges, P.P., Dias, M.S., Carvalho, F.R., Casatti, L., Pompeu, P.S., Cetra, M., Tejerina-Garro, F.L., Súarez, Y.R., Nabout, J.C., & Teresa, F.B., 2020. Stream fish metacommunity organisation across a Neotropical ecoregion: the role of environment, anthropogenic impact and dispersal-based processes. PLoS One 15(5), e0233733. PMid:32453798. http://doi.org/10.1371/journal.pone.0233733.

Brejão, G.L., Hoeinghaus, D.J., Pérez-Mayorga, M.A., Ferraz, S.F., & Casatti, L., 2018. Threshold responses of Amazonian stream fishes to timing and extent of deforestation. Conserv. Biol. 32(4), 860-871. PMid:29210104. http://doi.org/10.1111/cobi.13061.

Caetano, D.L.F., Oliveira, E.F., & Zawadzki, C.H., 2016. Fish species indicators of environmental quality of neotropical streams in southern Brazil, upper Paraná River basin. Acta Ichthyol. Piscat. 46(2), 87-96. http://doi.org/10.3750/AIP2016.46.2.04.

Caetano, D.L.F., Oliveira, E.F., & Zawadzki, C.H., 2021. Ichthyofauna of tributary streams of the Cinzas river Basin, Paranapanema river, Brazil. Oecol. Aust. 25(1), 142-153. http://doi.org/10.4257/oeco.2021.2501.13.

Carvalho, D.R., Castro, D.M.P., Callisto, M., Moreira, M.Z., & Pompeu, P.S., 2017. The trophic structure of fish communities from streams in the Brazilian Cerrado under different land uses: an approach using stable isotopes. Hydrobiologia 795(1), 199-217. http://doi.org/10.1007/s10750-017-3130-6.

Casatti, L., 2010. Alterações no Código Florestal Brasileiro: impactos potenciais sobre a ictiofauna. Biota Neotrop. 10(4), 31-34. http://doi.org/10.1590/S1676-06032010000400002.

Casatti, L., & Castro, R.M.C., 2006. Testing the ecomorphological hypothesis in a headwater riffles fish assemblage of the rio São Francisco, southeastern Brazil. Neotrop. Ichthyol. 4(2), 203-214. http://doi.org/10.1590/S1679-62252006000200006.

Casatti, L., Ferreira, C.P., & Carvalho, F.R., 2009. Grass-dominated stream sites exhibit low fish species diversity and dominance by guppies: an assessment of two tropical pasture river basins. Hydrobiologia 632(1), 273-283. http://doi.org/10.1007/s10750-009-9849-y.

Castro, D.M.P., Dolédec, S., & Callisto, M., 2018. Land cover disturbance homogenizes aquatic insect functional structure in neotropical savanna streams. Ecol. Indic. 84, 573-582. http://doi.org/10.1016/j.ecolind.2017.09.030.

Chase, J.M., & Leibold, M.A., 2002. Spatial scale dictates the productivity-biodiversity relationship. Nature 416(6879), 427-430. PMid:11919631. http://doi.org/10.1038/416427a.

Chazdon, R.L., Colwell, R.K., Denslow, J.S., & Guariguata, M.R., 1998. Statistical methods for estimating species richness of woody regeneration in primary and secondary rain forests of Northeastern Costa Rica. In: Dallmeier, F., & Comiskey, J.A., eds. Forest biodiversity research, monitoring and modeling: conceptual background and old world case studies. Paris: UNESCO, 285-309.

Collier, C.A., Almeida Neto, M.S., Almeida, G.M.A., Severi, W., Rosa Filho, J.S., & El-Deir, A.C.A., 2019. Effects of anthropic actions and forest areas on a neotropical aquatic ecosystem. Sci. Total Environ. 691(1), 367-377. PMid:31323582. http://doi.org/10.1016/j.scitotenv.2019.07.122.

Collins, S.M., Thomas, S.A., Heatherly II, T., MacNeill, K.L., Leduc, A.O.H.C., López-Sepulcre, A., Lamphere, B.A., El-Sabaawi, R.W., Reznick, D.N., Pringle, C.M., & Flecker, A.S.,., 2016. Fish introductions and light modulate food web fluxes in tropical streams: a whole-ecosystem experimental approach. Ecology 97(11), 3154-3166. PMid:27870030. http://doi.org/10.1002/ecy.1530.

Cunha, E.J., & Juen, L., 2017. Impacts of oil palm plantations on changes in environmental heterogeneity and Heteroptera (Gerromorpha and Nepomorpha) diversity. J. Insect Conserv. 21(1), 111-119. http://doi.org/10.1007/s10841-017-9959-1.

Cunico, A.M., Ferreira, E.A., Agostinho, A.A., Beaumord, A.C., & Fernandes, R., 2012. The effects of local and regional environmental factors on the structure of fish assemblages in the Pirapó Basin, Southern. Landsc. Urban Plan. 105(3), 336-344. http://doi.org/10.1016/j.landurbplan.2012.01.002.

Daga, V.S., Gubiani, É.A., Cunico, A.M., & Baumgartner, G., 2012. Effects of abiotic variables on the distribution of fish assemblages in streams with different anthropogenic activities in southern Brazil. Neotrop. Ichthyol. 10(3), 643-652. http://doi.org/10.1590/S1679-62252012000300018.

Dajoz, R., 1973. Ecologia geral. Petrópolis: Vozes.

Dala-Corte, R.B., Sgarbi, L.F., Becker, F.G., & Melo, A.S., 2019. Beta diversity of stream fish communities along anthropogenic environmental gradients at multiple spatial scales. Environ. Monit. Assess. 191(5), 288. PMid:31001723. http://doi.org/10.1007/s10661-019-7448-6.

Dias, T.S., & Fialho, C.B., 2009. Biologia alimentar de quatro espécies simpátricas de Cheirodontinae (Characiformes, Characidae) do rio Ceará Mirim, Rio Grande do Norte. Iheringia Ser. Zool. 99(3), 242-248. http://doi.org/10.1590/S0073-47212009000300003.

Dos-Santos, L.R.A., Araújo, S.M.S., De-Souza, M.Z.S.A., & Medeiros, L.E.L., 2018. Degradação ambiental no Açude de Bodocongó na cidade de Campina Grande, Paraíba. Rev. Rev. Verde Agroecol. Desenvolv. Sustent. 13(1), 74-78. http://doi.org/10.18378/rvads.v13i1.5377.

Ferreira, C.D.P., & Casatti, L., 2006. Integridade biótica de um córrego na bacia do Alto Rio Paraná avaliada por meio da comunidade de peixes. Biota Neotrop. 6(3), bn00306032006. http://doi.org/10.1590/S1676-06032006000300002.

Garcia, T.D., Strictar, L., Muniz, C.M., & Goulart, E., 2021. Our everyday pollution: are rural streams really more conserved than urban streams? Aquat. Sci. 83(3), 1-12. http://doi.org/10.1007/s00027-021-00798-4.

Gauch Junior, H.G., 1986. Multivariate analysis in community ecology. Cambridge: Cambridge University Press.

Grinnell, J., 1917. Field tests of theories concerning distributional control. Am. Nat. 51(602), 115-128. http://doi.org/10.1086/279591.

Gu, D.E., Ma, G.M., Zhu, Y.J., Xu, M., Luo, D., Li, Y.Y., Wei, H., Mu, X.D., Luo, J.R., & Hu, Y.C., 2015. The impacts of invasive Nile tilapia (*Oreochromis niloticus*) on the fisheries in the main rivers of Guangdong Province, China. Biochem. Syst. Ecol. 59, 1-7. http://doi.org/10.1016/j.bse.2015.01.004.

Hewitt, J., Thrush, S., Lohrer, A., & Townsend, M., 2010. A latent threat to biodiversity: consequences of small-scale heterogeneity loss. Biodivers. Conserv. 19(5), 1315-1323. http://doi.org/10.1007/s10531-009-9763-7.

Hill, M.J., Heino, J., Thornhill, I., Ryves, D.B., & Wood, P.J., 2017. Effects of dispersal mode on the environmental and spatial correlates of nestedness and species turnover in pond communities. Oikos 126(11), 1575-1585. http://doi.org/10.1111/oik.04266.

Holitzki, T.M., MacKenzie, R.A., Wiegner, T.N., & McDermid, K.J., 2013. Differences in ecological structure, function, and native species abundance between native and invaded Hawaiian streams. Ecol.

Appl. 23(6), 1367-1383. PMid:24147409. http://doi.org/10.1890/12-0529.1.

Hubbell, S.P., 2001. The unified neutral theory of biodiversity and biogeography (MPB-32) Princeton: Princeton University Press.

Jackson, D.A., 1993. Stopping rules in principal component analyses: a comparison of heuristical and statistical approaches. Ecol. Wash. DC. 74, 2204-2214.

Jarduli, L.R., Garcia, D.A.Z., Vidotto-Magnoni, A.P., Casimiro, A.C.R., Vianna, N.C., Almeida, F.S., Jerep, F.C., & Orsi, M.L., 2020. Fish fauna from the Paranapanema River basin, Brazil. Biota Neotrop. 20(1), e20180707. http://doi.org/10.1590/1676-0611-bn-2018-0707.

Jones, E.L., & Leather, S.R., 2012. Invertebrates in urban areas: a review. Eur. J. Entomol. 109(4), 463-478. http://doi.org/10.14411/eje.2012.060.

Kramer, D.L., & Mehegan, J.P., 1981. Aquatic surface respiration, an adaptive response to hypoxia in the guppy, *Poecilia reticulata* (Pisces, Poeciliidae). Environ. Biol. Fishes 6(3-4), 299-313. http://doi.org/10.1007/BF00005759.

Lampert, V.R., Dias, T.S., Tondato-Carvalho, K.K., & Fialho, C.B., 2022. The effects of season and ontogeny in the diet of *Piabarchus stramineus* (Eigenmann 1908) (Characidae: Stevardiinae) from southern Brazil. Acta Limnol. Bras. 34, e31. http://doi.org/10.1590/s2179-975x5621.

Langeani, F., Castro, R.M.C., Oyakawa, O.T., Shibatta, O.A., Pavanelli, C.S., & Casatti, L., 2007. Diversidade da ictiofauna do Alto Rio Paraná: composição atual e perspectivas futuras. Biota Neotrop. 7(3), 181-197. http://doi.org/10.1590/S1676-06032007000300020.

Larentis, C., Kliemann, B.C.K., Neves, M.P., & Delariva, R.L., 2022. Effects of human disturbance on habitat and fish diversity in Neotropical streams. PLoS One 17(9), e0274191. PMid:36084014. http://doi.org/10.1371/journal.pone.0274191.

Leal, C.G., Pompeu, P.S., Gardner, T.A., Leitão, R.P., Hughes, R.M., Kaufmann, P.R., Zuanon, J., de Paula, F.R., Ferraz, S.F.B., Thomson, J.R., Mac Nally, R., Ferreira, J., & Barlow, J., 2016. Multi-scale assessment of human-induced changes to Amazonian instream habitats. Landsc. Ecol. 31(8), 1725-1745. http://doi.org/10.1007/s10980-016-0358-x.

Leibold, M.A., Holyoak, M., Mouquet, N., Amarasekare, P., Chase, J.M., Hoopes, M.F., Holt, R.D., Shurin, J.B., Law, R., Tilman, D., Loreau, M., & Gonzalez, A., 2004. The metacommunity concept: a framework for multi-scale community ecology. Ecol. Lett. 7(7), 601-613. http://doi.org/10.1111/j.1461-0248.2004.00608.x.

Lóis, E., Santos, R.F., & Labaki, L.C., 2011. Efeitos de diferentes estruturas de vegetação ciliar sobre as variáveis de microclima e a sensação de conforto térmico. Ver Inst Flor 23(1), 117-136. http://doi.org/10.24278/2178-5031.2011231289.

Marques, P.S., & Cunico, A.M., 2021. Ecologia de peixes em riachos urbanos. Oecol. Aust. 25(2), 604-660. http://doi.org/10.4257/oeco.2021.2502.22.

Martins, I., Macedo, D.R., Hughes, R., & Callisto, M., 2021. Major risks to aquatic biotic condition in a Neotropical Savanna River basin. River Res. Appl. 37(6), 858-868. http://doi.org/10.1002/rra.3801.

Miiller, N.O.R., Cunico, A.M., Gubiani, É.A., & Piana, P.A., 2021. Functional responses of stream fish communities to rural and urban land uses. Neotrop. Ichthyol. 19(3), e200134. http://doi.org/10.1590/1982-0224-2020-0134.

Molina, M.C., Roa-Fuentes, C.A., Zeni, J.O., & Casatti, L., 2017. The effects of land use at different spatial scales on instream features in agricultural streams. Limnologica 65, 14-21. http://doi.org/10.1016/j.limno.2017.06.001.

Oksanen, J., Blanchet, F.G., Kindt, R., Legendre, P., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.H., & Wagner, H., 2016. Vegan: Community Ecology Package. R package version, 2.6. Vienna: R Foundation for Statistical Computing.

Ortega, J.C., Thomaz, S.M., & Bini, L.M., 2018. Experiments reveal that environmental heterogeneity increases species richness, but they are rarely designed to detect the underlying mechanisms. Oecologia 188(1), 11-22. PMid:29736864. http://doi.org/10.1007/s00442-018-4150-2.

Ortega, J.C.G., Bacani, I., Dorado-Rodrigues, T.F., Strüssmann, C., Fernandes, I.M., Morales, J., Mateus, L., Silva, H.P., & Penha, J., 2021. Effects of urbanization and environmental heterogeneity on fish assemblages in small streams. Neotrop. Ichthyol. 19(3), e210050. http://doi.org/10.1590/1982-0224-2021-0050.

Ota, R.R., Deprá, G.C., Graça, W.J., & Pavanelli, C.S., 2018. Peixes da planície de inundação do alto rio Paraná e áreas adjacentes: revised, annotated and updated. Neotrop. Ichthyol. 16(2), e170094. http://doi.org/10.1590/1982-0224-20170094.

Petry, A.C., Agostinho, A.A., Piana, P.A., & Gomes, L.C., 2007. Effects of temperature on prey consumption and growth in mass of juvenile trahira *Hoplias aff. malabaricus* (Bloch, 1794). J. Fish Biol. 70(6), 1855-1864. http://doi.org/10.1111/j.1095-8649.2007.01461.x.

Petry, A.C., Gomes, L.C., Piana, P.A., & Agostinho, A.A., 2010. The role of the predatory trahira (Pisces: Erythrinidae) in structuring fish assemblages in lakes of a Neotropical floodplain. Hydrobiologia 65(1), 115-126. http://doi.org/10.1007/s10750-010-0281-0.

Pinto, D.B.F., Silva, A.M.D., Mello, C.R.D., & Coelho, G., 2009. Qualidade da água do ribeirão Lavrinha na região Alto Rio Grande-MG, Brasil. Cienc. Agrotec. 33(4), 1145-1152. http://doi.org/10.1590/S1413-70542009000400028.

Pusey, B.J., & Arthington, A.H., 2003. Importance of the riparian zone to the conservation and management of freshwater fish: a review. Mar. Freshw. Res. 54(1), 1-16. http://doi.org/10.1071/MF02041.

Quinelato, R.V., Farias, E.S., Brito, J.M.S., Virgens, W.A., & Pires, L.C., 2020. Análise espaço temporal da qualidade da água dos rios Peruípe, Itanhém e Jucuruçu, Bahia. Sci. Plena. 16(7), http://doi.org/10.14808/sci.plena.2020.071701.

R Core Team, 2018. R: a language and environment for statistical computing. Vienna: R Foundation for Statistical Computing.

Rodríguez, P., Ochoa-Ochoa, L.M., Munguıa, M., Sánchez-Cordero, V., Navarro-Sigüenza, A.G., Flores-Villela, O.A., & Nakamura, M., 2019. Environmental heterogeneity explains coarse–scale β–diversity of terrestrial vertebrates in Mexico. PLoS One 14(1), e0210890. PMid:30682061. http://doi.org/10.1371/journal.pone.0210890.

Sales, L.P., Hayward, M.W., & Loyola, R., 2021. What do you mean by "niche"? Modern ecological theories are not coherent on rhetoric about the niche concept. Acta Oecol. 110, e103701. http://doi.org/10.1016/j.actao.2020.103701.

Sampaio, T., 1944. Relatório sobre os estudos efetuados nos rios Itapetininga e Paranapanema. Rev Inst Geog Ecol. 2(3), 30-81.

Sant'Anna, J.F.M., Almeida, M.C.D., Vicari, M.R., Shibatta, O.A., & Artoni, R.F., 2006. Levantamento rápido de peixes em uma lagoa marginal do rio Imbituva na bacia do alto rio Tibagi, Paraná, Brasil. Publ. UEPG Ciênc. Biol. Saúde 12(1), 39-46.

Smith, P., Ashmore, M.R., Black, H.I., Burgess, P.J., Evans, C.D., Quine, T.A., Thomson, A.M., Hicks, K., & Orr, H.G., 2013. The role of ecosystems and their management in regulating climate, and soil, water and air quality. J. Appl. Ecol. 50(4), 812-829. http://doi.org/10.1111/1365-2664.12016.

Souza, D.M., Teixeira, R.F., & Ostermann, O.P., 2015. Assessing biodiversity loss due to land use with Life Cycle Assessment: are we there yet? Glob. Change Biol. 21(1), 32-47. PMid:25143302. http://doi.org/10.1111/gcb.12709.

Stauffer Junior, J.R., Chirwa, E.R., Jere, W., Konings, F.A., Tweddle, D., & Weyl, O., 2022. Nile Tilapia, *Oreochromis niloticus* (Teleostei: Cichlidae): a threat to native fishes of Lake Malawi? Biol. Invasions 24(6), 1585-1597. http://doi.org/10.1007/s10530-022-02756-z.

Teresa, F.B., & Casatti, L., 2012. Influence of forest cover and mesohabitat types on functional and taxonomic diversity of fish communities in Neotropical lowland streams. Ecol. Freshwat. Fish 21(3), 433-442. http://doi.org/10.1111/j.1600-0633.2012.00562.x.

Teresa, F.B., Casatti, L., & Cianciaruso, M.V., 2015. Functional differentiation between fish assemblages from forested and deforested streams. Neotrop. Ichthyol. 13(02), 361-370. http://doi.org/10.1590/1982-0224-20130229.

Teshima, F.A., Mello, B.J.G., Ferreira, F.C., & Cetra, M., 2017. High β-diversity maintains regional diversity in Brazilian tropical coastal stream fish assemblages. Fish. Manag. Ecol. 23(6), 531-553. http://doi.org/10.1111/fme.12194.

Whittaker, R.H., 1960. Vegetation of the Siskiyou Mountains, Oregon and California. Ecol. Monogr. 30(3), 279-338. http://doi.org/10.2307/1943563.

Zar, J.H., 1999. Biostatistics analysis. New Jersey: Princeton-Hall.

Zengeya, T.A., Robertson, M.P., Booth, A.J., & Chimimba, C.T., 2013. Ecological niche modeling of the invasive potential of Nile tilapia *Oreochromis niloticus* in African river systems: concerns and implications for the conservation of indigenous congenerics. Biol. Invasions 15(7), 1507-1521. http://doi.org/10.1007/s10530-012-0386-7.

Ziesler, R., & Ardizzone, G.D., 1979. The inland waters of Latin America. Rome: Food and Agriculture Organization of the United Nations. Copescal Technical Paper, no. 1.

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