Check List the journal of biodiversity data
NOTES ON GEOGRAPHIC DISTRIBUTION

Xiphophorus hellerii Heckel, 1848 (Cyprinodontiformes, Poeciliidae): new record, updated distribution, and the role of urbanization in its invasion in Brazil

Flavia Begnini[1,2], Renan B. dos Reis[1], Érika Spanhol Lemunie[1], Luciano Lazzarini Wolff[2], Franco Teixeira-de-Mello[3], Rosilene Luciana Delariva[1,2]

1 Universidade Estadual do Oeste do Paraná, Centro de Ciências Biológicas e da Saúde, Programa de Pós-Graduação em Conservação e Manejo de Recursos Naturais, Rua Universitária 2069, 85819-110, Cascavel, Paraná, Brazil
2 Universidade Estadual do Oeste do Paraná, Centro de Ciências Biológicas e da Saúde, Laboratório de Ictiologia, Ecologia e Biomonitoramento, Rua Universitária 2069, 85819-110, Cascavel, Paraná, Brazil
3 Departamento de Ecología y Gestión Ambiental, Centro Universitario Regional del Este, Universidad de la República, 20000, Maldonado, Maldonado Department, Uruguay

Corresponding author: Rosilene Luciana Delariva (rosilene.delariva@unioeste.br)

Abstract. Biological invasions threaten freshwater biodiversity, particularly in anthropogenically impacted ecosystems. We report the first record of the fish *Xiphophorus hellerii* Heckel, 1848 in the Piquiri river basin (Upper Paraná), which was found in recent surveys, and investigate its distribution in Brazilian streams, using secondary data. We recorded 19 individuals in the Piquiri river basin. The results indicate predominance of occurrences in metropolitan areas, totalling 76 records in Brazil, likely linked to aquarium releases. We hypothesize that channelized urban streams may facilitate establishment by increasing environmental stressors, and we highlight the risks to the native fish fauna by the invasive *X. hellerii*.

Keywords. Exotic fishes, invasive species, management, urban impacts

Begnini F, dos Reis RBR, Lemunie ES, Wolff LL, Teixeira-de-Mello F, Delariva RL (2026) *Xiphophorus hellerii* Heckel, 1848 (Cyprinodontiformes, Poeciliidae): new record, updated distribution, and the role of urbanization in its invasion in Brazil. Check List 22 (4): 635–648. https://doi.org/10.15560/22.4.635

Introduction

Biological invasions, defined as the establishment of species beyond their natural distribution ranges, are mostly mediated by human activities and represent a major threat to native biodiversity, leading to losses in taxonomic, functional, and genetic diversity (Ricciardi 2007, 2013; Iacarella et al. 2015; Daga et al. 2016; Larentis et al. 2019, 2022a; dos Reis et al. 2025). When associated with environmental disturbances such as urbanization, deforestation, and pollution, the impacts of invasive species are further intensified, contributing to the degradation of natural ecosystems (McKinney 2009; Jacobson 2011; Iacarella et al. 2015; Roy et al. 2016; Ganassin et al. 2020). In the Neotropical region, freshwater-fish communities have been increasingly affected by the introduction and spread of non-native species (Vitule 2009; Vitule et al. 2012; Abilhoa et al. 2013; Ota et al. 2018; Garcia et al. 2021; Dagosta et al. 2024). Despite being the most species-rich region in the world for freshwater fishes (Birindelli and Sidlauskas 2018), several areas within the Neotropics exhibit high levels of biological invasions. A notable example is the Upper Paraná ecoregion (*sensu* Abell et al. 2008), which has undergone substantial ecological alteration following the flooding of the Sete Quedas waterfalls (Júlio Júnior et al. 2009; Ota et al. 2018). This event has facilitated the establishment of invasive species, and current estimates indicate that approximately 128 non-native fish species have been documented in this region (Dagosta et al. 2024) with effects at the ecosystem level (Moi et al. 2021).

Transformations in the natural landscape have also been driven by the increasing expansion of urban centres (McGrane 2016; Ortega et al. 2021; Pelicice et al. 2021), by the pressures imposed on headwaters located in urban areas (Cruz and Pompeu 2020), by the daily input of pollutants (domestic and/or industrial) and also by physical and hydrological changes related to the channelization

OPEN ACCESS

Academic editor: Felipe Araújo
Received: 24 February 2026
Accepted: 16 June 2026
Published: 14 July 2026

Copyright © The authors. This is an open-access article distributed under terms of the Creative Commons Attribution License (Attribution 4.0 International – CC BY 4.0)

process (Lau et al. 2006; Lennox and Rasmussen 2016; Stowe et al. 2023). The process of channelization involves the construction of drainage infrastructures, including widening and sealing the natural channel with artificial materials, such as concrete and rocks, and containing the banks with gabion structures (Walsh et al. 2005; McGrane 2016). Such artificialized environments are often used by non-native species, which exploit these structures as refuges against abiotic and biotic stressors, and are considered key points of introduction, due to their proximity to introduction vectors, including ports, roads, and urban parks (Boon et al. 2023).

These changes in the environment can accelerate the process of loss of sensitive species and the establishment of opportunistic and tolerant species, since the loss of environmental heterogeneity, the absence of marginal vegetation, and disturbed water quality are consequences of channelization that directly contribute to the composition and functionality of human settlements (Antoniazzi et al. 2023; Begnini et al. 2025). This scenario becomes more serious when we consider the introduction of exotic species that compete for analogous ecological niches, including food resources, shelter, and habitat (Daga et al. 2012; Ganassin et al. 2020). In addition, exotic species can be responsible for the transmission of diseases, parasites, and hybridization with native species (Júlio Júnior et al. 2009; Orsi and Britton 2014; Pelicice and Agostinho 2009; Vitule et al. 2012; Garcia et al. 2018).

The order Cyprinodontiformes comprises species distributed throughout nearly all regions of the world, occurring in both temperate and tropical climates, as well as in freshwater and saline environments (Rosen and Bailey 1963). Within this order, poecilids (family Poecilidae) are typically small-bodied fishes, reaching a maximum standard length (SL) of approximately 200 mm, although most species are considerably smaller, often less than half this size (Rosen and Bailey 1963; Lucinda and Garavello 2000; Lucinda 2005, 2008; Souto-Santos et al. 2023, 2025). Due to their aquatic, insectivorous feeding habits, some species have been used as a biological control strategy against mosquito populations in rivers and streams in Brazil and worldwide (Cavalcanti et al. 2007; Pamplona et al. 2009; Azevedo-Santos et al. 2017). In addition, the exuberant colouration of certain species has made them highly popular in the ornamental fish trade (Esmaeili et al. 2010; Larentis et al. 2019; Islamy et al. 2025). However, the excessive proliferation of some poecilid species kept in artificial aquariums has led to their introduction into natural environments (Ota et al. 2018; Larentis et al. 2019; Islamy et al. 2025). A notable example is *Xiphophorus hellerii* Heckel, 1848, a small exotic poecilid which has been consistently recorded in urban streams with high levels of anthropogenic disturbance in urban streams of southern Brazil (Magalhães and Jacobi 2017).

Xiphophorus hellerii, commonly known as the Green Swordtail, has a natural distribution from southern North America (Mexico) to parts of Central America (Guatemala, Honduras, and Belize) (Fricke et al. 2025). It exhibits internal fertilization, with females giving birth to fully formed juveniles that resemble miniature adults (Rosen 1960; Rosen and Bailey 1963). The species exhibits clear sexual dimorphism, with males having an elongated extension on the ventral lobe of the caudal fin rays, a characteristic that gives rise to the common name of the species (Rosen 1960). Other diagnostic features include highly polymorphic colour patterns, a small, prognathous mouth, a large dorsal fin, and cycloid scales covering the body from head to tail (Rosen 1960).

One of the main left-bank tributaries of the Upper Paraná river basin is the Piquiri River, which harbours 152 fish species, of which 30 are non-native species (Cavalli et al. 2018). Along its length, this river encompasses large urban centres (e.g. Cascavel, Paraná state, Brazil), with a significant portion of its headwater streams located within urban territory, where they are altered by channelization (Begnini et al. 2025). These streams are home to native poecilid species, such as *Phalloceros circummontanus* Souto-Santos, Mejia, Arcila & Buckup, 2025, previously identified as *P. harpagos* Lucinda, 2008 in the region (Cavalli et al. 2018; dos Reis et al. 2020; Souto-Santos et al. 2025). Based on public datasets and recent field surveys, we aim to: (i) document the distribution and geographical range of *X. hellerii* in Brazil by comparing its occurrence in streams classified as urban, rural, and forested; (ii) report, for the first time, the presence of this species in the Piquiri river basin, located in the Upper Paraná ecoregion; and (iii) discuss the potential implications of channelization as a refuge for the occurrence and establishment of this non-native species.

Methods

The city of Cascavel, located in western Paraná, is the fifth most populous city in the state, with over 364,000 inhabitants according to the IBGE (2024). It is located in the interfluve of three basins: Paraná III (lower Paraná ecoregion following dos Reis et al. (2020)), Piquiri (Upper Paraná Ecoregion *sensu*

Abell et al. (2008)), and Iguaçu river basins (Iguassu ecoregion *sensu* Abell et al. (2008)). Considered the third-largest drainage area in Paraná (Parolin et al. 2010), the Piquiri river basin has an area of 24,156 km^2, with altitudes ranging from 188 m to 1,180 m above sea level. This hydrological configuration consists primarily of headwaters and small bodies of water scattered throughout the city's urban areas.

The stream Amambay is formed by the confluence of three headwaters, all located in the Morumbi neighbourhood, part of the urban perimeter of Cascavel (Figure 1). It exhibits a high degree of anthropic influence, characterized by a narrow strip of mostly exotic riparian vegetation interspersed with a small strip of native vegetation. As a strategy for managing stormwater and implementing new recreational areas for the population, linear parks were built along watercourses, with the goal to include representative green spaces in urban landscapes along straightened or channelized watercourses.

The Amambay was channelized over approximately 890 m in 2018; this section of stream has been named S3. Recently, the linear park began to have its area expanded, and the stream was subjected to additional channelization along a 110 m stretch, named S2. Originally, S2 and S3 had heterogeneous environments, with rapids, watercourses, pools, mesohabitats, greater depth, and a substrate predominantly composed of pebbles, gravel, and sand/clay. After being channelized, the water bodies had a channelized substrate, with an increase in average width, and consequently a reduction in depth and an increase in flow. S1 and S4 are located, respectively, upstream and downstream, and their beds were not changed during the implementation of the park (Figures 1, 2), presenting areas of rapids and pools.

Fish samplings were conducted using electrofishing during three phases: one before channelization (May 2023), one during channelization (January 2024), and the last six months after channelization (October 2024) (Figure 2). We collected the fish under authorization from the Chico Mendes Institute for Biodiversity Conservation (ICMBio; license number 25039). After capture, the fish were euthanized according to the ethics protocol of the Animal Use Committee of the State University of Western Paraná (no. 14-22 and 08-24/CEUA) and fixed in 10% formalin. In the laboratory of ichthyology, ecology and biomonitoring (Laboratório de Ictiologia, Ecologia e Biomonitoramento, LIEB-UNIOESTE), counts and measurements were taken according to Miller (1948). Measurements were taken with digital callipers having a precision of 0.1 mm. The data acquired were compared with data from the specialized literature on *Xiphophorus* taxonomy (e.g. Rosen 1960; Rosen and Bailey 1963) and fish taxonomy in Brazil. The vouchers are deposited in the Nupelia Ichthyological Collection (Coleção Ictiológica do Nupelia, NUP-UEM, Maringá, Paraná, Brasil)

To compare the date of introduction and the categorization of streams with *X. hellerii* occurrence, we selected records of specimens deposited in museums or scientific collections, accessed on GBIF.org

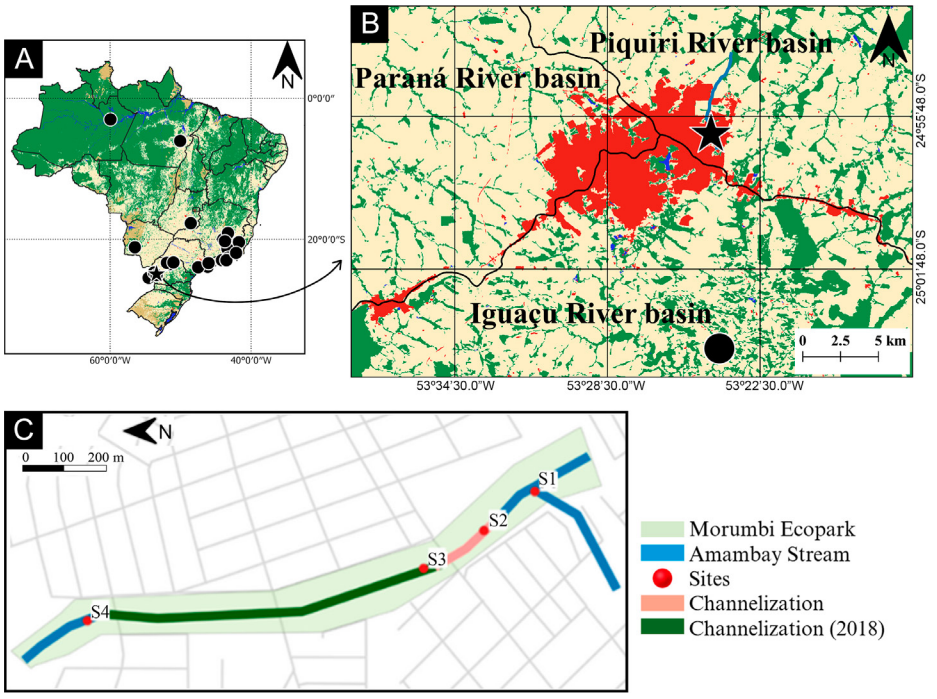

Figure 1. A. Map of Brazil, showing previous and new collection sites of *X. hellerii*. **B.** Map of Paraná state and the region close to Cascavel, highlighting sample sites of *X. hellerii* (black circles) and the new record (black star) from Amambay stream. Colours represent land cover and use from MapBiomas: red = urban areas; green = forests; yellow = farming areas; beige = non-forest natural formations; areas in blue = water. **C.** Sample sites in Amambay stream, municipality of Cascavel, Paraná, Brazil.

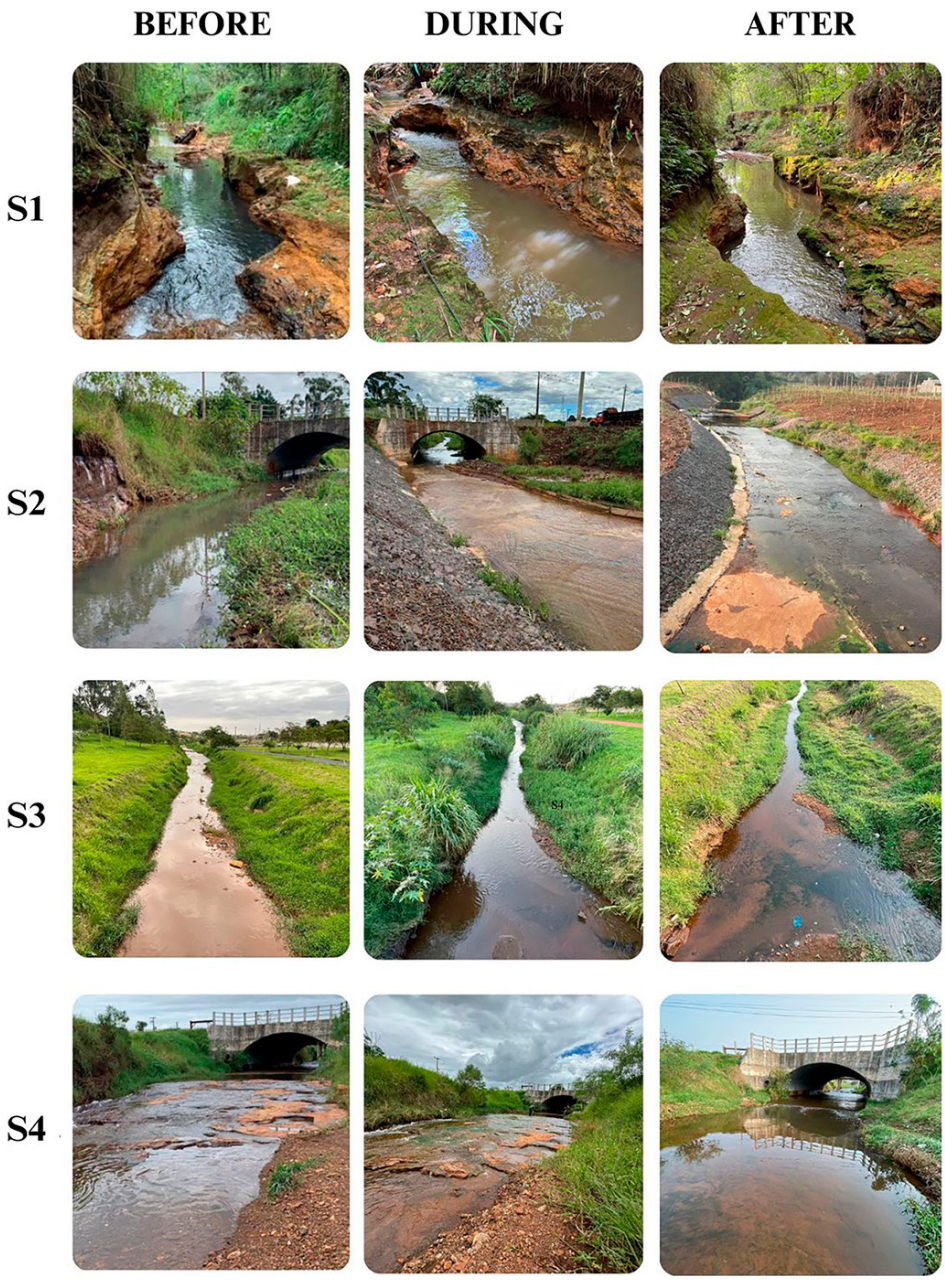

Figure 2. Sampling sites in Amambay stream that underwent the channelization process and their respective phases.

(2025; last updated list of identified material of *X. hellerii* in GBIF: https://doi.org/10.15468/dl.3r8byf). Afterwards, the geographical coordinates of each record were plotted in QGIS v. 3.16.7; a buffer of 500 m was built and overlapped with land-use-cover data from MapBiomas (MapBiomas Collection Official v. 10, downloaded at https://brasil.mapbiomas.org/colecoes-mapbiomas/). Based on the land-use cover within the polygon of the 500 m buffer, we categorized the streams (with sample points) by the presence of *X. hellerii* in Brazil, using the predominance of land-use cover (>50%) as follows: urban (in non-forested areas); rural (in farming area or non-forest natural formations); or forest (in forests).

Results

***Xiphophorus hellerii* Heckel, 1848**

Figure 3, Table 1

New records. BRAZIL — Paraná state • Upper Paraná basin, Rio Piquiri sub-basin, Amambay stream; 24°56′30″S, 053°24′26″W; 686 m alt.; 15.I.2025; R. Delariva leg.; 2 spec., 30.7–40.6 mm SL, NUP 26060.

Identification. The specimens analysed here fit the diagnosis of the genus *Xiphopohurs* in having a distinct gonopodium in males, including a large decurved hook and calcified blade arising from the

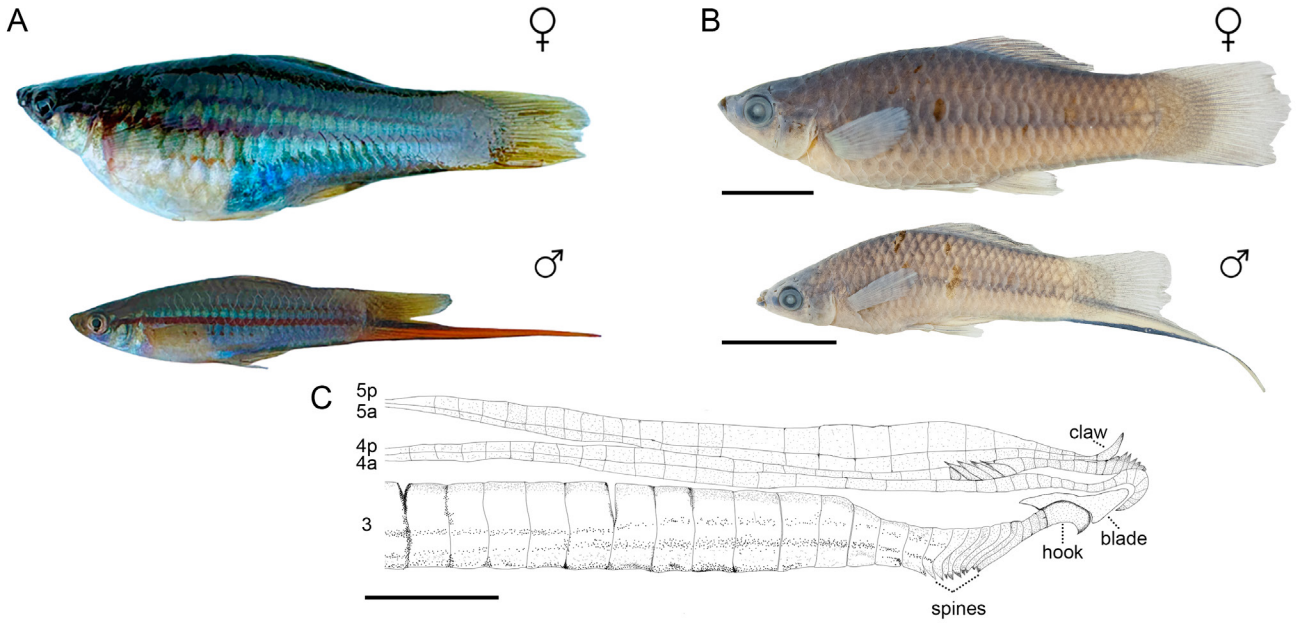

Figure 3. Lateral view of *Xiphophorus hellerii*, uncatalogued. **A.** Specimens photographed before fixation. **B.** Specimens fixed in alcohol (NUP 26060). **C.** Schematic draw of the structures of male gonopodium. Scale bars, 10 mm in specimens, 1 mm in gonopodium.

Table 1. Morphometric data for *Xiphophorus hellerii* based on seven specimens collected in the Piquiri river basin. Mininum and maximum values, as well as mean and standard deviation (SD) are provided.

Measurementss	Females (*n* = 4)				Males (*n* = 3)			
	Min.	Max.	Mean	SD	Min.	Max.	Mean	SD
Standard length (mm)	31.05	41.75	37.92	—	33.4	54.3	43.7	—
Percents of Standard Length								
Predorsal length	50.0	51.9	50.9	0.94	51.8	53.6	52.7	0.76
Prepelvic length	44.2	48.4	46.4	2.15	45.6	51.6	49.1	2.51
Body greatest depth	25.8	30.8	28.0	2.58	31.2	38.6	35.1	3.28
Dorsal-fin basal length	20.8	24.7	22.8	1.92	21.4	25.0	23.7	1.62
Anal-fin basal length	6.3	9.8	7.7	1.86	6.3	10.4	8.7	1.75
Dorsal-fin length	30.7	44.5	39.9	7.98	33.4	38.3	35.6	2.19
Anal-fin length	22.9	23.7	23.3	0.43	16.0	21.6	19.7	2.53
Middle caudal-ray length	18.9	26.9	24.0	4.44	24.2	28.9	27.0	2.01
Pectoral length of longest ray	21.9	24.4	23.1	1.24	19.3	22.6	21.3	1.48
Pelvic length of longer fin	20.4	25.5	23.5	2.70	15.9	17.8	16.7	0.86
Caudal-peduncle length	36.6	42.9	39.7	3.13	32.8	39.0	35.4	2.71
Head length	23.6	25.7	24.3	1.23	22.2	24.9	23.4	1.15
Percent of head length								
Head width	66.0	69.5	67.6	1.76	72.1	72.7	72.3	0.28
Interorbital	24.9	45.4	37.5	11.06	36.3	49.7	44.7	7.33
Snout length	20.8	34.3	29.8	7.78	34.0	39.0	36.3	2.57
Orbit length	31.5	32.5	32.0	0.54	28.5	32.8	30.5	2.18
Mouth width	25.5	35.8	32.0	5.65	36.2	39.1	37.6	1.49
Upper jaw length	19.7	22.9	20.9	1.71	17.4	20.5	19.1	1.57
Mandible length	18.8	21.7	20.2	1.45	18.2	20.5	19.1	1.20

dorsal margin of the hook on ray 3; ray 4a with the distal region curving downward and conforming to the shape of the blade and two series of serrae on ray 4p, separated by simple segments; and a ventrally directed ray 5a (Figure 3C). Furthermore, our specimens fit the diagnosis of *X. hellerii* and can be distinguished from *X. maculatus* introduced to Brazil in having two dark-brown (in fixed specimens) or dark-red (in live specimens) longitudinal stripes extending from the opercle to the caudal-fin base (in *X. maculatus*, distinct dark-brown or dark-red longitudinal stripes are absent); no spots or blotches on the caudal peduncle or on the caudal-fin rays (in *X. maculatus*, spots or blotches are present on the caudal peduncle and the caudal-fin rays); and the lobe of the ventral caudal-fin rays extending as a filament, forming a sword-like shape (in *X. maculatus*, this lobe does not extending as a filament). Seven specimens were measured and presented in Table 1.

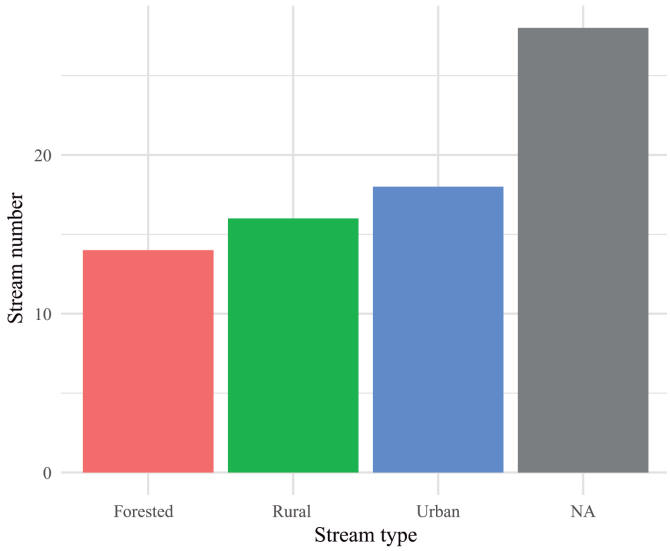

Figure 4. Abundance of *X. hellerii* by sample points and channelization phase.

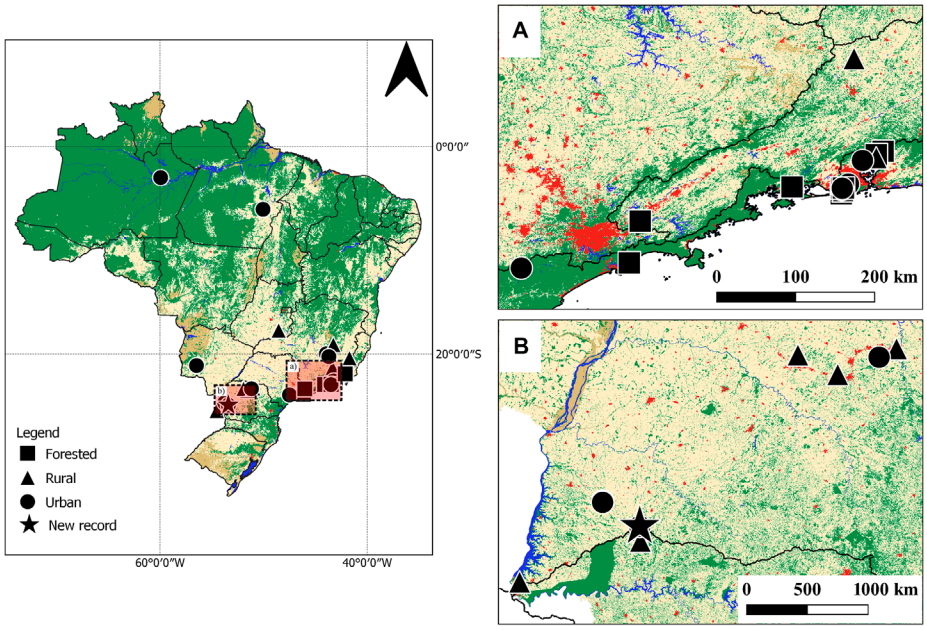

Figure 5. Map of Brazil showing stream classification based on the land cover and use of the MapBiomas collection 10. The letters highlight regions with many samples and overlapping locations on the map: (**A**) streams classification in southeastern Brazil; (**B**) streams classification in southern Brazil.

Meristic data. Scales in lateral line 27 (7 spec.); scale series between lateral line and dorsal-fin origin 3.5 (1 spec.) or 4.5 (6 spec.); scale series between anal-fin origin 3.5 (4) or 4.5 (3). Pectoral-fin rays i,10 (7 spec.); pelvic-fin rays i,5 (7 spec.); dorsal-fin rays i,10 (7 spec.).

In Amambay stream where the species was recorded, 19 specimens were collected across three sampling sites in the Amambay stream (Figure 4), located in the city of Cascavel, Paraná state, Brazil. No specimen was collected in S1. Of the 19 collected specimens, two were deposited in the NUP collection (sample point S3), while the remaining 17 encompassing all sample points are housed at the LIEB-UNIOESTE for future ecological studies.

A total of 76 records, including those of this study, of *X. hellerii* were found in natural or anthropized environments across Brazil, including one specimen fixed directly from an aquarium by Paulo Miranda-Ribeiro in 1944, and are not utilized in the categorized stream analysis. Of the 76 records, 47 had geographical coordinates that allowed for the categorization of the stream environment: 14 records were from streams with native forest, 16 from rural environments, and 17 from urban streams (Figures 5, 6; Appendix Table A1; see also Supplementary Material 1).

Forty-nine records included collection dates. The species was first included in scientific collections in 1999, which represents its earliest known occurrence in natural or anthorpized environments in Brazil. This first record was collected in Goiás by the NUP team. Following this initial record, the number of specimens deposited increased, with additional lots collected between 2003 and 2009, and a subsequent rise in records from 2015 to 2025 (Figure 7).

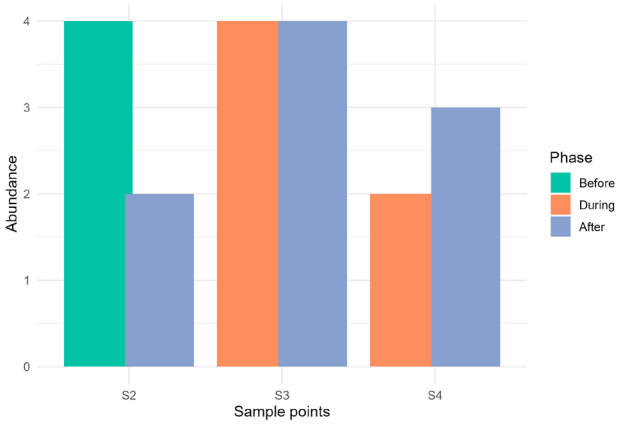

Figure 6. Proportion of collections sites and streams classifications of *Xiphophorus hellerii* in Brazil. NA = streams have no coordinates available.

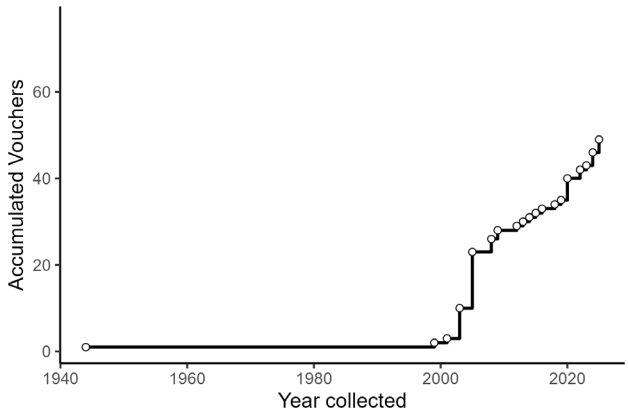

Figure 7. Accumulation curve of collections by year of *Xiphophorus hellerii* in Brazil.

Discussion

Our recent field surveys have yielded a new record of *Xiphophorus hellerii* from the state of Paraná and specifically from the Piquiri river basin, Upper Paraná ecoregion (*sensu* Abell et al. 2008). Nineteen adult individuals were electrofished from an urban stream (Amambay stream), which includes both channelized and unchannelized sites. Furthermore, based on our investigation of stream types (urban, rural, and forest streams) where *X. hellerii* has been found across Brazil, we found that this species occurs in all these stream types. Most records are from urban streams (Appendix Table A1), with many from metropolitan regions of Brazil, particularly in the southeast and south.

Globally, *X. hellerii* has been widely introduced into freshwater systems and is currently reported in at least 31 countries, including Brazil (Maddern 2009). The early records of the species in natural and anthropized Brazilian habitats date back to 1999 in the states of Goiás and Minas Gerais; later, the species was recorded from the state of Rio de Janeiro between 2003 and 2005, Paraná in 2008 and 2009, the state of São Paulo in 2012 and 2013, Mato Grosso do Sul in 2016, Pará in 2020, and Santa Catarina in 2021 (Supplementary Material Table S1). The records indicate a clear colonization of different river basins, particularly in regions with high-human population densities (Figures 1, 5). This species' occurrence is likely linked to mosquito-control programs and accidental or intentional releases by aquarium hobbyists; *X. hellerii* is prized by aquarists due to its small size, bright colouration, and ease of handling (Azevedo-Santos et al. 2017; Larentis et al. 2019). Furthermore, the absence of records in ichthyological collections and museums does not suggest that this species is not already in other areas, especially considering that approximately 10% of GBIF records (Supplementary Material Table S1) are via iNaturalist.

In Paraná, *X. hellerii* was first recorded in 2008 and 2009 in the municipalities of Maringá (Paranapanema river basin) and Toledo (Paraná III river basin), respectively. In 2012 and 2014, it was found in the municipalities of Apucarana and Ibiporã (Paranapanema river basin) (dos Reis et al. 2020), and later, in 2018, in the Iguaçu river basin (Larentis et al. 2019), and in the Refúgio da Boa Vista, in a branch of the Itaipu reservoir (Supplementary Material Table S1). Our new records represent a geographic expansion into a fourth major basin in Paraná state.

Although some authors collected in the surrounding locations of the new records (see Larentis et al. 2025), no specimens of *Xiphophorus hellerii* were captured, which suggests a hypothesis of facilitation

of the invasion process by channelized and/or urbanized areas, since our records demonstrate greater abundance of specimens during and after channelization (Figure 4). These results emphasize the potential for continued spread in the absence of effective control measures, despite the existence of Brazilian laws prohibiting the release of invasive species into natural environments (Faria et al. 2023). This introduction of exotic species into streams already impacted by urbanization and physical alterations, such as channelization, is a cause for concern, as these degraded environments become more vulnerable to biological invasion (Cruz and Pompeu 2020; Takács et al. 2021). In the case of the Amambay stream, the presence of a linear park along the channel-straightening process facilitates public access to the waterway, thereby favouring the intentional disposal of organisms by aquarists, a practice that contributes to the introduction and establishment of exotic species. The low abundance of *X. hellerii* and the absence of juveniles suggest a recent introduction. Furthermore, its absence in adjacent river basins, as reported in previous studies (e.g. Larentis et al. 2022b, 2025), further supports this hypothesis. However, its detection alongside another non-native species, as well as native species, at the same locations increases the risk of cumulative negative impacts on the native fauna.

The presence of both non-native species *X. hellerii* and *P. reticulata* Peters 1859 in the same habitats (Begnini et al. 2025) can lead to competitive exclusion of native poeciliids, such as *P. circummontanus*, especially considering the reduced ecological plasticity of native species (Magalhães and Jacobi 2017). Exotic poeciliids tend to exploit a broader range of resources and are often favoured in altered environments. Additionally, hybridization between exotic poeciliid species is well documented (e.g. *Xiphophorus* and *Poecilia*), posing further risks to native populations through genetic introgression (Meyer et al. 2005; Lampert and Schartl 2008).

Xiphophorus hellerii exhibits characteristics that increase its invasive capacity, including viviparity, lack of reproductive seasonality, and high fecundity (Esmaeili et al. 2010; Rosen 1960). These characteristics make population control difficult once individuals have established themselves. Despite the recent canalization of approximately 100 m of the Amambay stream (S2), individuals of the species have been recorded in both channelized and unchannelized areas (Figure 6). It is important to note that, in channelized environments, water temperatures tend to be higher due to the direct incidence of solar radiation on the concrete structures. Even so, these stretches have allowed the species to survive, even in the face of habitat modifications. No individuals have been recorded at site S1.

The physically altered sections of the Amambay stream are currently undergoing recolonization. However, features such as reduced depth, increased flow, and the installation of artificial steps act as environmental filters that select for certain life-history traits, particularly favouring small-bodied, highly fecund, and trophically flexible species (Bohus et al. 2023; Zhang et al. 2024; Begnini et al. 2025; Pessoa et al. 2025). These conditions may function as refuges for certain species by buffering them from abiotic and biotic stressors (Boon et al. 2023), thereby facilitating the colonization and establishment of non-native species in environments that are already degraded.

These conditions align with the ecological profiles of poeciliids found in the study area, supporting a shift toward communities dominated by tolerant, generalist species (Lamothe et al. 2018; Cruz and Pompeu 2020; Larentis et al. 2022b). This process contributes to biotic homogenization, characterized by a reduced richness of sensitive species and an increased dominance of tolerant and non-native species (Chen and Olden 2020; Barrios and Teixeira de Mello 2022; Miranda et al. 2023; Teixeira de Mello et al. 2024; Pessoa et al. 2025). Additional evidence suggests that the presence of *X. hellerii* may contribute to the decline in biodiversity of other aquatic organism groups, such as zooplankton and odonate populations, in addition to acting as a predator of amphibian eggs and larvae (Stockwell and Henkanaththegedara 2011).

Given the well-documented negative consequences of biological invasions on native fauna and the vulnerability of urbanized systems to such events, consistent monitoring of urban streams is essential. In the context of our study, channelization emerges as a key facilitator of species introductions. By increasing public access to urban watercourses and creating artificial structures that serve as refuges for non-native taxa, reducing both abiotic and biotic stress, channelized environments become more susceptible to the establishment of invasive species. In this way, understanding how invasive populations establish and interact with native biota, particularly in highly altered habitats, will be key to developing targeted management strategies and preventing further ecological degradation. Thus, the results presented reinforce the importance of preventive and integrated environmental management priority actions, such as strengthening environmental education initiatives, promoting responsible trade and possession of non-native species, restoring riparian vegetation, and protecting urban stream corridors (Magalhães and Jacobi 2010; Azevedo-Santos et al. 2023; Azevedo-Santos 2026; Garcia and Ribeiro 2026). In addition, systematic monitoring, as conducted here, in urban streams may

allow the early detection of invasive populations before they become widespread. Because urbanized watersheds frequently act as facilitation of invasion, management efforts focused on these environments may provide disproportionate benefits for the conservation of native fish assemblages and the ecological integrity of Neotropical freshwater ecosystems.

Acknowledgements

We gratefully acknowledge the support of the State University of Western Paraná (UNIOESTE), as well as the contributions of the LIEB-UNIOESTE team (Laboratório de Ictiologia, Ecologia e Biomonitoramento) for assistance with fieldwork and laboratory analyses. The Municipality of Cascavel provided access permissions support. We also thank the editors and reviewers for their valuable time, constructive comments, and careful evaluation of this manuscript.

Additional information

Conflict of interest

The authors declare that no competing interests exist.

Ethical statement

The care and use of animals complied with the ethics protocol of the Federal Council of Veterinary Medicine, Brazil (No. 1000/2012), animal welfare laws, guidelines and policies as approved by the Animal Use Committee of the State University of Western Paraná (No. 14-22 and 08-24/CEUA).

Use of artificial intelligence (AI)

The authors accept full responsibility for the content of the manuscript, including the disclosure of any use of AI. No AI tools were used in the preparation of this manuscript.

Funding

This study was financially supported by the Municipality of Cascavel. Funding acknowledgements include a master's scholarship for FB awarded by the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (CAPES) and a postdoctoral scholarship for RBR provided by Fundação de Apoio ao Ensino, Extensão, Pesquisa e Pós-graduação (FUNDEP/Unioeste). FTM received funding from the Universidad de la República through Espacio Interdisciplinario, Comisión Sectorial de Investigación Científica (CSIC), PEDECIBA, and SNI.

Author contributions

Conceptualization: FB, RLD, and RBR. Funding acquisition: RLD. Project administration: RLD. Resources: RLD. Supervision: RLD. Formal analysis: FB, RBR, LLW, ESL, FTM, RLD. Data curation: FB, RBR, LLW, ESL, FTM, RLD. Investigation: FB, RBR, LLW, ESL, FTM, RLD. Methodology: FB, RBR, LLW, ESL, FTM, RLD. Visualization: FB, RBR, LLW, ESL, FTM, RLD. Validation: FB, RBR, LLW, ESL, FTM, RLD. Writing – review and editing: FB, RBR, LLW, ESL, FTM, RLD.

Author ORCID iDs

Flavia Begnini https://orcid.org/0009-0003-7519-0013
Renan Borges dos Reis https://orcid.org/0000-0002-3704-0749
Érika Spanhol Lemunie https://orcid.org/0009-0002-8383-6630
Luciano Lazzarini Wolf https://orcid.org/0000-0003-1770-7697
Franco Teixeira de Mello https://orcid.org/0000-0003-4904-6985
Rosilene Luciane Delariva https://orcid.org/0000-0002-6489-2437

Data availability

All data that support the findings of this study are available in the main text, Appendix, and Supplementary Materials.

Supplemental material 1. Vouchers and collection locations for *Xiphophorus hellerii* acquired through GBIF data (XLSX format).

References

Abell R, Thieme ML, Revenga C, Bryer M, Kottelat M, Bogutskaya N, Coad B, Mandrak N, Balderas SC, Bussing W, Stiassny MLJ, Skelton P, Allen GR, Unmack P, Naseka A, Ng R, Sindorf N, Robertson J, Armijo E, Higgins JV, Heibel TJ, Wikramanayake E, Olson D, López HL, Reis RE, Lundberg JG, Sabaj MH, Petry P (2008)

Freshwater ecoregions of the world: a new map of biogeographic units for freshwater biodiversity conservation. BioScience 58 (5): 403–414. https://doi.org/10.1641/b580507

Abilhoa V, Bornatowski H, Vitule JR (2013) Occurrence of the alien invasive loach *Misgurnus anguillicaudatus* in the Iguaçu river basin in southern Brazil: a note of concern. Journal of Applied Ichthyology 29: 257–259. https://doi.org/10.1111/jai.12007

Antoniazzi R, Montaña CG, Peterson D, Schalk CM (2023) Changes in taxonomic and functional diversity of an urban stream fish assemblage: a 30-year perspective. Frontiers in Environmental Science 10: 965291. https://doi.org/10.3389/fenvs.2022.965291

Azevedo-Santos VM, Vitule JRS, Pelicice FM, García-Berthou E, Simberloff D (2017) Nonnative fish to control *Aedes* Mosquitoes: a controversial, harmful tool. BioScience 67 (1): 84–90. https://doi.org/10.1093/biosci/biw156

Azevedo-Santos VM, Daga VS, Tonella LH, et al (2023) Brazil's urban ecosystems threatened by law. Land Use Policy 131: 106721. https://doi.org/10.1016/j.landusepol.2023.106721

Azevedo-Santos VM (2026) Plastic pollution: restoring urban riparian vegetation. BioScience biag006. https://doi.org/10.1093/biosci/biag006

Barrios M, Teixeira de Mello F (2022) Urbanization impacts water quality and the use of microhabitats by fish in subtropical agricultural streams. Environmental Conservation 49 (3): 155–163. https://doi.org/10.1017/S0376892922000200

Begnini F, Wolff LL, Lopes TM, Teixeira de Mello F, Delariva RL (2025) Immediate effects of channelization on fish assemblages in two urban tropical streams. Neotropical Ichthyology 23 (4): e250075. https://doi.org/10.1590/1982-0224-2025-0075

Birindelli JLO, Sidlauskas BL (2018) Neotropical ichthyology preface: how far has Neotropical ichthyology progressed in twenty years? Neotropical Ichthyology 16 (3): e180128. https://doi.org/10.1590/1982-0224-20180128

Bohus A, Gál B, Barta, B, Szivák, I, Karádi-Kovács, K Boda P, Padisák J, Schmera D (2023) Effects of urbanization-induced local alterations on the diversity and assemblage structure of macroinvertebrates in low-order streams. Hydrobiologia 850: 881–899. https://doi.org/10.1007/s10750-022-05130-1

Boon JS, Keith SA, Exton DA, Field R (2023) The role of refuges in biological invasions: a systematic review. Global Ecology and Biogeography 32: 1244–1271. https://doi.org/10.1111/geb.13701

Cavalcanti LPG, Pontes RJS, Regazzi ACF, de Paula Júnior FJ, Frutuoso RL, Sousa EP, Filho FFD, Lima JWO (2007) Efficacy of fish as predators of *Aedes aegypti* larvae, under laboratory conditions. Revista De Saude Publica 41 (4): 638–644. https://doi.org/10.1590/s0034-89102006005000041

Cavalli D, Frota A, Lira AD, Gubiani ÉA, Margarido VP, da Graça WJ (2018) Update on the ichthyofauna of the Piquiri river basin, Paraná, Brazil: a conservation priority area. Biota Neotropica 18 (2): e20170350. https://doi.org/10.1590/1676-0611-bn-2017-0350

Chen K, Olden JD (2020) Threshold responses of riverine fish communities to land use conversion across regions of the world. Global Change Biology 26: 4952–4965. https://doi.org/10.1111/gcb.15251

Cruz LC, Pompeu PS (2020) Drivers of fish assemblage structures in a Neotropical urban watershed. Urban Ecosystems 23: 819–829. https://doi.org/10.1007/s11252-020-00968-6

Daga V, Debona T, Abilhoa V, Gubiani ÉA, Vitule J (2016) Non-native fish invasions of a Neotropical ecoregion with high endemism: a review of the Iguaçu River. Aquatic Invasions 11 (2): 209–223. https://doi.org/10.3391/ai.2016.11.2.10

Daga VS, Gubiani ÉA, Cunico AM, Baumgartner G (2012) Effects of abiotic variables on the distribution of fish assemblages in streams with different anthropogenic activities in southern Brazil. Neotropical Ichthyology 10 (3): 643–652. https://doi.org/10.1590/s1679-62252012000300018

Dagosta FCP, Monção MS, Nagamatsu BA, Pavanelli CS, Carvalho FR, Lima FCT, Langeani F, Dutra GM, Ota RR, Seren TJ, Tagliacollo V, Menezes NA, Britski HA, de Pinna M (2024) Fishes of the upper rio Paraná basin: diversity, biogeography and conservation. Neotropical Ichthyology 22 (1): e230066. https://doi.org/10.1590/1982-0224-2023-0066

Garcia DAZ, Ribeiro LA (2026) Educação ambiental na prevenção de invasões biológicas no Brasil: Desafios e perspectivas para a conservação. Research, Society and Development 15: e2415551058–e2415551058. https://doi.org/10.33448/rsd-v15i5.51058

dos Reis RB, Frota A, Deprá GC, Ota RR, da Graça WJ (2020) Freshwater fishes from Paraná state, Brazil: an annotated list, with comments on biogeographic patterns, threats, and future perspectives. Zootaxa 4868: 451–494. https://doi.org/10.11646/zootaxa.4868.4.1

dos Reis RB, Roloff MZ, Pedroso TH, Reinas GCZ, Stabile BH, Martins IC, Carvalho FR (2025) First record of the non-native species *Bryconops* aff *melanurus* (Bloch, 1794) (Characiformes: Iguanodectidae) in the Paraná River floodplain, upper Paraná river basin, Brazil. Journal of Fish Biology 107: 302–307. https://doi.org/10.1111/jfb.70008

Esmaeili HR, Gholamifard A, Teimori A, Baghbani S, Coad BW (2010) *Xiphophorus hellerii* Heckel, 1848 (Cyprinodontiformes, Poeciliidae), a newly introduced fish recorded from natural freshwaters of Iran. Journal of Applied Ichthyology 26: 937–938. https://doi.org/10.1111/j.1439-0426.2010.01515.x

Faria L, Carvalho BM, Carneiro L, Miiller NOR, Pedroso CR, Occhi TVT, Tonella LH,Vitule JRS (2023) Invasive species policy in Brazil: a review and critical analysis Environmental Conservation 50 (1): 67–72. https://doi.org/10.1017/S0376892922000406

Fricke R, Eschmeyer WN, van der Laan R (2025) Catalog of Fishes: Genera, Species, References. http://researcharchivecalacademyorg/research/ichthyology/catalog/fishcatmainasp. Accessed on: 2025-08-27.

Ganassin MJM, Frota A, Muniz CM, Baumgartner MT, Hahn NS (2020) Urbanisation affects the diet and feeding selectivity of the invasive guppy *Poecilia reticulata*. Ecology of Freshwater Fish 29: 252–265. https://doi.org/10.1111/eff.12511

Garcia DAZ, Britton JR, Vidotto-Magnoni AP, Orsi ML (2018) Introductions of non-native fishes into a heavily modified river: rates, patterns and management issues in the Paranapanema River (Upper Paraná ecoregion, Brazil). Biological Invasions 20: 1229–1241. https://doi.org/10.1007/s10530-017-1623-x

Garcia DAZ, Pelicice FM, Brito MFGD, Orsi ML, Magalhães ALB (2021) peixes não-nativos em riachos no brasil: estado da arte, lacunas de conhecimento e perspectivas. Oecologia Australis 25 (2): 565–587. https://doi.org/10.4257/oeco.2021.2502.21

GBIF.org (2025) GBIF occurrence download. https://doi.org/10.15468/dl.3r8byf. Accessed on: 2025-07-08.

Iacarella J, Dick J, Alexander M, Ricciardi A (2015) Ecological impacts of invasive alien species along temperature gradients: testing the role of environmental matching. Ecological Applications 25: 706–716. https://doi.org/10.1890/14-0545.1

Islamy RA, Valen F, Alfian R, Hasan V (2025) First Record of *Xiphophorus hellerii* (Heckel, 1848) (Cyprinodontiformes: Poeciliidae) from the Bangka Island, Indonesia Egyptian. Journal of Aquatic Biology and Fisheries 29 (2): 1055–1065.

Jacobson CR (2011) Identification and quantification of the hydrological impacts of imperviousness in urban catchments: a review. Journal of Environmental Management 92 (6): 1438–1448. https://doi.org/10.1016/j.jenvman.2011.01.018

Júlio Júnior HF, Dei Tós C, Agostinho ÂA, Pavanelli CS (2009) A massive invasion of fish species after eliminating a natural barrier in the upper rio Paraná basin. Neotropical Ichthyology 7 (4): 709–718. https://doi.org/10.1590/S1679-62252009000400021

Lamothe KA, Alofs KM, Jackson DA, Somers KM (2018) Functional diversity and redundancy of freshwater fish communities across biogeographic and environmental gradients. Diversity and Distributions 24: 1612–1626. https://doi.org/10.1111/ddi.12812

Lampert KP, Schartl M (2008) The origin and evolution of a unisexual hybrid: *Poecilia formosa*. Philosophical Transactions of the Royal Society B: Biological Sciences 363 (1505): 2901–2909. https://doi.org/10.1098/rstb.2008.0040

Larentis C, Baldasso MC, Kliemann BCK, Neves MP, Zavaski AG, Sandri LM, Ribeiro AC, Xavier DP, Costa G, Delariva RL (2019) First record of the non-native *Xiphophorus hellerii* (Cyprinodontiformes: Poeciliidae), in the Iguazu river basin, Paraná, Brazil. Journal of Applied Ichthyology 35: 1164–1168. https://doi.org/10.1111/jai.13965

Larentis C, Kliemann BCK, Neves MP, Delariva RL (2022b) Effects of human disturbance on habitat and fish diversity in Neotropical streams. PLoS ONE 17 (9): e0274191. https://doi.org/10.1371/journal.pone.0274191

Larentis C, Lopes T, Oliveira AG, Message H, Delariva R, Pavanelli C (2025) Forest cover, local environmental conditions, and non-native species influence fish beta diversity in the Neotropical streams. Aquatic Ecology 59: 1413–1433. https://doi.org/10.1007/s10452-025-10233-7

Larentis C, Pavanelli CS, Delariva RL (2022a) Do environmental conditions modulated by land use drive fish functional diversity in streams? Hydrobiologia 849: 4465–4483. https://doi.org/10.1007/s10750-021-04756-x

Lau JK, Lauer TE, Weinman ML (2006) Impacts of channelization on stream habitats and associated fish assemblages in east central Indiana. The American Midland Naturalist 156: 319–330. http://www.jstor.org/stable/4094628

Lennox PA, Rasmussen JB (2016) Long-term effects of channelization on a cold-water stream community. Canadian Journal of Fisheries and Aquatic Sciences 73: 1530–1537. https://doi.org/10.1139/cjfas-2015-0561

Lucinda PHF (2005) Systematics of the genus *Cnesterodon* Garman, 1895 (Cyprinodontiformes: Poeciliidae: Poeciliinae). Neotropical Ichthyology 3: 259–270. https://doi.org/10.1590/s1679-62252008000200001

Lucinda PHF (2008) Systematics and biogeography of the genus *Phalloceros* Eigenmann, 1907 (Cyprinodontiformes: Poeciliidae: Poeciliinae), with the description of twenty-one new species. Neotropical Ichthyology 6: 113–158. https://doi.org/10.1590/S1679-62252008000200001

Lucinda PHF, Garavello JC (2000) Two new species of *Cnesterodon* Garman, 1895 (Cyprinodontiformes: Poeciliidae) from the upper rio Paraná drainage. Comunicação Museu Ciências PUCRS, Série Zoológica 13: 119–138.

Maddern M (2009) *Xiphophorus hellerii*. CABI compendium. https://doi.org/10.1079/cabicompendium.59751. Accessed on: 2025-07-08.

Magalhães ALB, Jacobi CM (2017) Colorful invasion in permissive Neotropical ecosystems: establishment of ornamental non-native poeciliids of the genera *Poecilia*/*Xiphophorus* (Cyprinodontiformes: Poeciliidae) and management alternatives. Neotropical Ichthyology 15: e160094. https://doi.org/101590/1982-0224-20160094

Magalhães A, Jacobi CM (2010) E-commerce of freshwater aquarium fishes: potential disseminator of exotic species in Brazil. Acta Scientiarum Biological Sciences 32: 243–248. https://doi.org/10.4025/actascibiolsci.v32i3.3919

McGrane SJ (2016) Impacts of urbanisation on hydrological and water quality dynamics, and urban water management: a review. Hydrological Sciences Journal 61: 2295–2311. https://doi.org/10.1080/02626667.2015.1128084

McKinney M (2009) Urbanization, Biodiversity, and Conservation: the impacts of urbanization on native species are poorly studied, but educating a highly urbanized human population about these impacts can greatly improve species conservation in all ecosystems. BioScience 52: 883–890. https://doi.org/10.1641/0006-3568(2002)052[0883:ubac]2.0.co;2

Meyer JL, Paul MJ, Taulbee WK (2005) Stream ecosystem function in urbanizing landscapes. Journal of the North American Benthological Society 24: 602–612. https://doi.org/10.1899/04-021.1

Miller RR (1948) The cyprinodont fishes of the Death Valley system of eastern California and southwestern Nevada. Miscellaneous publications Museum of Zoology, University of Michigan, 68: 7–111.

Miranda R, Bartrons M, Brucet S, Benejam L (2023) Long-term monitoring on a new channelized stream section: changes in mesohabitat, composition, and size structure of fish assemblages. Restoration Ecology 31: e13995. https://doi.org/10.1111/jfb.12392

Moi DA, Alves DC, Figueiredo BRS, Antiqueira PA, Teixeira de Mello F, Jeppesen E, Romero, GQ, Mormul RP, Bonecker CC (2021) Non-native fishes homogenize native fish communities and reduce ecosystem multifunctionality in tropical lakes over 16 years. Science of the Total Environment 769: 144524. https://doi.org/10.1016/j.scitotenv.2020.144524

Orsi ML, Britton JR (2014) Long-term changes in the fish assemblage of a Neotropical hydroelectric reservoir. Journal of Fish Biology 84: 1964–1970. https://doi.org/10.1111/jfb.12392

Ortega JCG, Bacani I, Dorado-Rodrigues TF, Strüssmann C, Fernandes IM, Morales J, Mateus L, da Silva HP, Penha J (2021) Effects of urbanization and environmental heterogeneity on fish assemblages in small streams. Neotropical Ichthyology 19: e210050. https://doi.org/10.1590/1982-0224-2021-0050

Ota RR, Deprá GC, da Graça WJ, Pavanelli CS (2018) Peixes da planície de inundação do alto rio Paraná e áreas adjacentes: revised, annotated and updated. Neotropical Ichthyology 16: 1–111. https://doi.org/10.1590/1982-0224-20170094

Pamplona LGC, Alencar CH, Lima JWO, Heukelbach J (2009) Reduced oviposition of *Aedes aegypti* gravid females in domestic containers with predatory fish. Tropical Medicine & International Health 14: 1347–1350. https://doi.org/10.1111/j.1365-3156.2009.02377.x

Parolin, M, Volkmer-Ribeiro C, Leandrini JA (2010) Abordagem ambiental interdisciplinar em bacias hidrográficas no estado do Paraná. FECILCAM, Campo Mourão, Brazil, 158 pp.

Pelicice FM, Agostinho AA (2009) Fish fauna destruction after the introduction of a non-native predator (*Cichla kelberi*) in a Neotropical reservoir. Biological Invasions 11: 1789–1801. https://doi.org/10.1007/s10530-008-9358-3

Pelicice FM, Bialetzki A, Camelier P, Carvalho FR, García-Berthou E, Pompeu PS, Teixeira de Mello FT, Pavanelli CS (2021) Human impacts and the loss of Neotropical freshwater fish diversity. Neotropical Ichthyology 19 (3): e210134. https://doi.org/10.1590/1982-0224-2021-0134

Pessoa LA, Baumgartner MT, Frota A, Garcia TD, Júnior MPS, Pessoa LGA, Goulart E (2025) Urban land use homogenizes fish assemblages across mesohabitats in headwater streams. Aquatic Sciences 87 (34): 1–12. https://doi.org/10.1007/s00027-025-01162-6

Ricciardi A (2007) Are modern biological invasions an unprecedented form of global change? Conservation Biology 21: 329–336. https://doi.org/10.1111/j.1523-1739.2006.00615.x

Ricciardi A (2013) Invasive Species. In: Leemans R (Eds.) Ecological systems: selected entries from the encyclopedia of sustainability science and technology. Springer, New York, USA, 161–178. https://doi.org/10.1007/978-1-4419-0851-3

Rosen DE (1960) Middle-American poeciliid fishes of the genus *Xiphophorus*. Bulletin of the Florida State Museum 5: 57–242. https://doi.org/10.58782/flmnh.bnrr6484

Rosen DE, Bailey RM (1963) The poeciliid fishes (Cyprinodontiformes): their structure, zoogeography, and systematics. Bulletin of the American Museum of Natural History 126: 1–176.

Roy AH, Capps KA, El-Sabaawi RW, Jones KL, Parr TB, Ramírez A, Smith RF, Walsh CJ, Wenger SJ (2016) Urbanization and stream ecology: diverse mechanisms of change. Freshwater Science 35: 272–277. https://doi.org/10.1086/685097

Souto-Santos ICA, Lucinda PHF, Buckup PA (2023) Bilateral genital asymmetry in livebearer fishes of the genus *Phalloceros* Eigenmann, 1907, with description of a new species from coastal drainages of southern Brazil (Cyprinodontiformes: Poeciliidae). Journal of Fish Biology 103: 91–102. https://doi.org/10.1111/jfb.15410

Souto-Santos ICA, Mejia E, Arcila D, Buckup PA (2025) Dismantling the *Phalloceros harpagos* species complex (Teleostei: Poeciliidae): description of a new species from Brazilian highlands based on morphological and molecular evidence. Neotropical Ichthyology 23: e240129. https://doi.org/10.1590/1982-0224-2024-0129

Stockwell CA, Henkanaththegedara SM (2011) Evolutionary conservation biology Ecology and evolution of Poeciliid fishes. In: Evans JP, Pilastro A, Schlupp I (Eds.) Evolutionary ecology of the livebearing fishes. University of Chicago Press, Chicago, USA, 128–141.

Stowe ES, Petersen KN, Rao S, Walther EJ, Freeman MC, Wenger SJ (2023) Stream restoration produces transitory, not permanent, changes to fish assemblages at compensatory mitigation sites. Restoration Ecology 31: e13903. https://doi.org/10.1111/rec.13903

Takács P, Abonyi A, Bánó B, Erős T (2021) Effect of non-native species on taxonomic and functional diversity of fish communities in different river types. Biodiversity and Conservation 30: 2511–2528. https://doi.org/10.1007/s10531-021-02207-6

Teixeira de Mello F, Sierra P, Moi DA, Alonso J, Lucas C, Suárez B, Alvareda E, Alvarez J, Andrade MS, Arimon L, Urtado L, Piperno A (2024) Effects of urbanization and accessibility to sanitation services on water quality in urban streams in Uruguay. Environmental Monitoring and Assessment 196: 185. https://doi.org/10.1007/s10661-024-12365-z

Vitule JRS (2009) Introdução de peixes em ecossistemas continentais brasileiros: revisão, comentários e sugestões de ações contra o inimigo quase invisível. Neotropical Biology and Conservation 4: 111–122. https://doi.org/0.4013/nbc.2009.42.07

Vitule JRS, Skóra F, Abilhoa V (2012) Homogenization of freshwater fish faunas after the elimination of a natural barrier by a dam in Neotropics. Diversity and Distributions 18: 111–120. https://doi.org/10.2307/41420102

Walsh CJ, Roy AH, Feminella JW, Cottingham PD, Groffman PM, Morgan RP (2005) The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society 24: 706–723. https://doi.org/10.1899/04-028.1

Zhang S, Zhan A, Zhao J, Yao M (2024) Metropolitan pressures: significant biodiversity declines and strong filtering of functional traits in fish assemblages. Science of the Total Environment 944: 173885. https://doi.org/10.1016/j.scitotenv.2024.173885

Appendix

Table A1. Collection locations and categorized streams for *Xiphophorus hellerii* in Brazil acquired through GBIF data.

State	Decimal latitude	Decimal longitude	Stream type	Institution code	Catalog number	Date collected
Rio de Janeiro	−22.5046	−042.9822	Forested	iNaturalist	286678508	2025-06-04
Rio de Janeiro	−22.5048	−042.9825	Forested	iNaturalist	286678654	2025-06-04
Amazonas	−03.0072	−059.9400	Urban	iNaturalist	106799295	2025-04-30
Rio de Janeiro	−22.5472	−043.0680	Forested	iNaturalist	129209188	2022-12-16
Minas Gerais	−19.9999	−043.9466	Urban	iNaturalist	256178591	2024-12-25
Rio de Janeiro	−23.0356	−043.5050	Forested	iNaturalist	77301425	2022-12-16
Santa Catarina	NA	NA	NA	CAJ	NA	NA
Santa Catarina	−20.4075	−041.7125	Rural	USP-RP	17514	2020-03-15
Santa Catarina	−20.4075	−041.7125	Rural	USP-RP	17513	2020-03-15
Santa Catarina	−20.4075	−041.7125	Rural	USP-RP	17506	2020-03-15
Rio de Janeiro	−21.3461	−043.3472	Rural	MZUSP	100820	NA
Minas Gerais	−19.1124	−043.2791	Rural	MZUSP	112243	NA
Minas Gerais	−19.1122	−043.2680	Rural	MZUSP	112238	NA
São Paulo	−23.8985	−046.1558	Forested	MZUSP	115455	NA
Rio de Janeiro	−22.9527	−044.1227	Forested	iNaturalist	189868123	2023-11-03
Paraná	−23.5256	−051.5603	Rural	UEL	6511	NA
São Paulo	NA	NA	NA	UEL	27081	NA
São Paulo	NA	NA	NA	UEL	27082	NA
Paraná	NA	NA	NA	UEL	17781	NA
Paraná	NA	NA	NA	UEL	17782	NA
Paraná	NA	NA	NA	UEL	17783	NA
Paraná	NA	NA	NA	UEL	17784	NA
Paraná	NA	NA	NA	UEL	17785	NA
Paraná	NA	NA	NA	UEL	17808	NA
Paraná	NA	NA	NA	UEL	17809	NA
Paraná	NA	NA	NA	UEL	17804	NA
Paraná	NA	NA	NA	UEL	17805	NA
Paraná	NA	NA	NA	UEL	17806	NA
Paraná	NA	NA	NA	UEL	17807	NA
Paraná	NA	NA	NA	UEL	17810	NA
Paraná	−23.2856	−051.0113	Rural	UEL	9023	NA
Paraná	−25.4592	−054.5275	Rural	UEM	23596	2019-02-27
Goiás	−17.7269	−048.5483	Rural	UEM	1101	1999-04-09
São Paulo	−23.9646	−047.5071	Urban	UEM	5515	2008-01-19
Paraná	−24.7069	−053.7522	Urban	UEM	8560	2008-04-18
Paraná	−24.7069	−053.7522	Urban	UEM	8571	2009-02-27
Paraná	−24.7186	−053.7533	Urban	UEM	7284	2009-04-28
Paraná	−23.3442	−051.9308	Rural	UEM	6030	2008-06-01
Paraná	−25.0797	−053.4008	Rural	UEM	21119	2018-08-28
Mato Grosso do Sul	−21.1354	−056.4864	Urban	UFMS	ZUFMS-PIS06897	NA
Pará	−6.0665	−050.0592	Urban	iNaturalist	59802285	2020-09-16
São Paulo	−23.3778	−046.0214	Forested	UNICAMP	10019	2014-12-23
São Paulo	−23.3808	−046.0192	Forested	UNICAMP	7650	2012-12-26
São Paulo	−23.3808	−046.0192	Forested	UNICAMP	7996	2013-12-27

State	Decimal latitude	Decimal longitude	Stream type	Institution code	Catalog number	Date collected
São Paulo	−23.3778	−046.0214	Forested	UNICAMP	12243	2015-12-30
Pará	−06.0640	−050.0594	Urban	iNaturalist	59802384	2020-09-16
Rio de Janeiro	NA	NA	Urban	UFRRJ	2313	NA
Rio de Janeiro	−22.5858	−043.0692	Rural	UFRRJ	962	NA
Rio de Janeiro	−22.6211	−043.2419	Urban	UFRRJ	961	NA
Mato Grosso do Sul	−21.1300	−056.4803	Urban	UFMS	ZUFMS-PIS04956	NA
Paraná	NA	NA	NA	UEL	16558	2005-09-21
Paraná	NA	NA	NA	UEL	16559	2005-09-21
Paraná	NA	NA	NA	UEL	16532	2005-03/21
Paraná	NA	NA	NA	UEL	16536	2005-11-24
Paraná	NA	NA	NA	UEL	16557	2005-11-23
Paraná	NA	NA	NA	UEL	16535	2005-06-22
Paraná	NA	NA	NA	UEL	16561	2005-11-24
Paraná	NA	NA	NA	UEL	16534	2005-06-22
Paraná	NA	NA	NA	UEL	16560	2005-09-21
Paraná	NA	NA	NA	UEL	16556	2005-06-21
Paraná	NA	NA	NA	UEL	16533	2005-03-22
Paraná	NA	NA	NA	UEL	16555	2005-09-21
Paraná	−23.2856	−051.0113	Rural	UEL	8606	NA
NA	NA	NA	NA	MNRJ	9477	1944-01-01
Rio de Janeiro	−22.9658	−043.4931	Rural	MNRJ	28954	2005-01-01
Rio de Janeiro	−22.9667	−043.5000	Forested	MNRJ	25905	2003-01-01
Rio de Janeiro	−22.9661	−043.4797	Forested	MNRJ	25863	2003-01-01
Rio de Janeiro	−23.0000	−043.4833	Rural	MNRJ	25619	2003-01-01
Rio de Janeiro	−22.9167	−043.4167	Urban	MNRJ	25405	2003-01-01
Rio de Janeiro	−22.9833	−043.5000	Urban	MNRJ	25397	2003-01-01
Rio de Janeiro	−22.9661	−043.4797	Forested	MNRJ	25339	2003-01-01
Rio de Janeiro	−22.9683	−043.4942	Urban	MNRJ	24122	2003-01-01
Minas Gerais	−20.2736	−043.7053	Urban	MNRJ	21565	2001-01-01
Rio de Janeiro	−21.9333	−042.1167	Forested	MNRJ	14325	2016-01-01
Paraná	−24.9417	−053.4071	Urban	UEM	26060	2024-01-15
Paraná	−23.3661	−051.1739	Urban	UEL	23714	2024-03-28