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RESEARCH ARTICLE

Impacts of land use and hydrological alterations on water quality and fish assemblage structure in headwater Pampean streams (Argentina)

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Abstract

Headwater streams play an essential role in catchment ecosystem processes and are particularly vulnerable to land use degradation and anthropic hydrological alterations. Considering that fish assemblages are reliable indicators of the ecological integrity of streams, we evaluated the ichthyofauna structure in four headwater Pampean streams located at catchment areas with contrasting land uses and hydrological man-made alterations. During autumn and spring of 2017 and 2018, fish assemblages were sampled and physicochemical and water table depth were measured. Specific richness, abundance, and diversity of fish assemblages were estimated. The degree of anthropic intervention of drainage networks and wetland coverage was estimated for each catchment. Statistical analyses assessed the differences between rural and periurban sampling sites in terms of environmental variables and fish assemblage structure, as well as the relationships between the biotic and abiotic variables. Results showed differences among headwater streams according to their land uses, hydrological alterations, and fish assemblage structure, evidencing a combined effect of land use and hydrologic alterations on fish assemblage. Changes in these communities can reflect not only alterations in water quality caused by local land use but also the influence of the catchment environmental integrity related to hydrological modifications and other phenomena that can occur simultaneously and at different geographical scales.

Keywords Groundwater · Wetland loss · Fish community · Anthropogenic disturbances · Synergic impacts · Lowland streams

Introduction

Streams and rivers have been recognized as the most threatened freshwater systems in the world (Strayer and Dudgeon 2010). Headwater streams are the major component of river networks and may contribute to more than three-quarters of the stream drainage network length (Leopold et al. 1995; Benda et al. 2005). These systems have received increased attention in recent years because of their functional and structural importance for the drainage network, exhibiting a critical role for the preservation of freshwater diversity (Meyer et al. 2007; Shao et al. 2019). Headwater streams perform biological, geochemical, and physical processes that are essential for ecosystem services throughout their catchment (Meyer and Wallace 2001; Colvin et al. 2019). Moreover, such reaches not only deliver sediments and organic material to downstream waters contributing to nutrient cycling and water quality, but also enhance flood protection and mitigation (Gomi et al. 2002; Hill et al. 2014; Cunha et al. 2020). Particularly for fish populations, headwaters play a fundamental role in providing spawning and nursery habitats (Biggs et al. 2017). However, despite their

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ecological importance, headwater courses represent the less researched part of riverine systems and remain often excluded from water resource management planning (Biggs et al. 2017).

A variety of aquatic organisms have been proposed as indicators of environmental quality (Domínguez et al. 2020). Fish assemblages emerge as reliable indicators of the ecological integrity of streams and rivers for many reasons. Life-history information is extensive for most fish species; they usually represent different trophic levels, reproductive strategies and tolerance to pollution being present in almost all water bodies, even in those with certain levels of contamination (Karr 1981). Moreover, fish that inhabit systems that maintain a baseflow tend to adapt to stable water regimes showing a variety of traits related to specific microhabitats, in contrast to hydrologically disturbed and unstable systems in which fish assemblages are composed of generalist and tolerant species (Blann et al. 2009).

Anthropogenic activities related to land use and hydrological alteration of watercourses cause ecological deterioration of streams and rivers (Strayer and Dudgeon 2010; Albert et al. 2021). For instance, agriculture promotes increased inputs of sediment, nutrients, and agrochemicals to streams (Allan 2004), whereas urban centers promote inputs of a wide range of pollutants (Paul and Meyer 2001). These can promote eutrophication and generate pulses of toxicity, impoverishing the water quality of riverine systems (Paul and Meyer 2001).

Modification of water regimes by channelization, dam construction, and stream-groundwater disconnection can result in extreme water flow pulses and recurrent drought of streams and riverine wetlands (Rheinhardt et al. 1999; Bunn and Arthington 2002; Hankock 2002), threatening the biotic integrity of fluvial ecosystems (Poff and Allan 1995). Headwater streams and their wetlands have historically been conceived as waterlogged sites that impede human development and should be eliminated without considering the ecosystem services they provide (Rodrigues Capítulo et al. 2020). In this context, the periurbanization of the sectors surrounding large cities has modified the natural physiography of watercourses (Tucci 2012; Paz et al. 2021), through a process driven by the engineering criterion of evacuating large volumes of water as quickly as possible, leaving aside the short-term problems caused by these interventions in the watershed (Tucci 2012). Likewise, the long-term problem and the role of groundwater in controlling the environmental flows necessary for the development of biotic communities has been underestimated for many years through the intensive water extraction for human supply in these periurban areas (Custodio 2010). Considering that the impact of land use on water quality and the hydrological dynamics can occur simultaneously and at different spatial and temporal scales, the study of the anthropic impacts from both perspectives could show the possible synergistic effects of these factors in the biotic component of ecosystems.

The aim of this study was to evaluate the responses of fish assemblages to changes in water quality and hydrology in four headwater Pampean streams located in rural and periurban areas. We analyzed the case of the development of La Plata city as an example of how the territorial management linked to urban centers growth alter the ecological quality of riverine ecosystems. We expect to find that streams running through periurban areas be characterized by simplified fish assemblages compared with rural streams, related to a water quality deterioration and hydrological alterations.

Materials and methods

Study area

The study streams are tributaries of the Río de la Plata estuary located in the Pampean plain near La Plata city, Argentina (Fig. 1a). The climate of the area is temperate and humid, with average monthly air temperatures ranging from 9.9 to 22.4 °C, and annual rainfall and evapotranspiration of 1060 mm and 783 mm, respectively (Laurencena et al. 2010). The watercourses lack a forested riparian zone and run through low-slope grassy meadows. Their bottoms are formed by fine sediments, and their waters are alkaline with high concentrations of suspended organic matter, nutrients and dissolved oxygen, compared to other lotic environments in the world (Feijoó et al. 1999; Rodrigues Capítulo et al. 2010). The contribution of groundwater to the stream base-level determines permanent or semi-permanent character to the water courses in the region (Kruse 2015), as well as the presence and permanence of wetlands associated with these watercourses (Rodrigues Capítulo et al. 2020). Thus, water flow does not vary substantially throughout the year, except in ephemeral and acute situations when flow peaks may occur after heavy rains (Feijoó et al. 1999; Rodrigues Capítulo et al. 2010).

Land use in the study area initially consisted of extensive livestock on grasslands, which was progressively replaced, after the foundation of La Plata city, by extensive crop cultivation, followed by undercover crop cultivation and finally by urbanization (López and Rotger 2020). Such changes occurred since the nineteenth century with expansive dynamics from the city to the peripheral areas, impacting watercourses through land use activities (Kruse et al. 2014) and drastic structural modifications of the drainage systems with the aim of increasing the productive area by reducing the residence time of water in the watersheds (Rodrigues Capítulo et al. 2020).

Four headwaters streams, two located in periurban areas—Carnaval (CNV) and Del Gato (GAT)—and two in

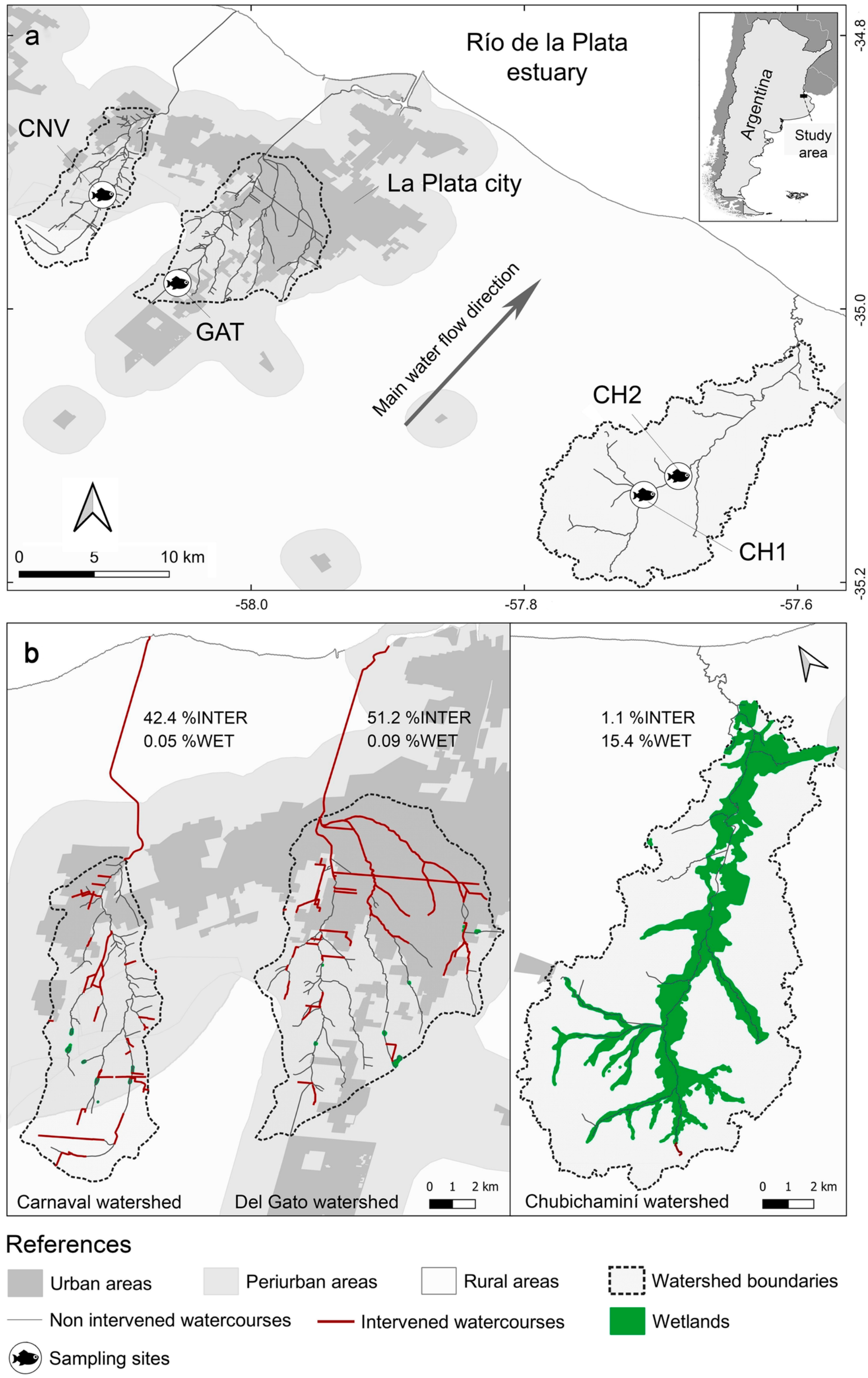


Fig. 1 **a** Sampling site locations in catchment areas of tributaries of the Río de la Plata estuary, indicating water flow direction, watershed boundaries, and mainly land uses near La Plata city, Argentina. Carnaval, CNV; Del Gato, GAT and Chubichaminí sites 1 and 2, CH1 and CH2. **b** Watershed hydrological alterations according to the percentage of watercourses intervention (%INTER) and the percentage of areas covered by wetlands (%WET) in watersheds with rural and periurban/urban land use

rural areas—Chubichaminí (CH1-CH2)—were selected as sampling sites in the surroundings of La Plata city, Argentina (Fig. 1a). Each site was sampled twice a year, in austral autumn and spring of 2017 and 2018. The Carnaval catchment is mainly covered by periurban land use represented by open field crop cultivation in the middle and upper sectors (surrounding CNV sampling site). Also, in upper sectors, this watershed exhibits rural land use, whereas it is highly urbanized at lower sectors. The land use in Del Gato catchment consists of periurban and urban land use. The first one is represented by undercover crop cultivation (horticulture) coupled with low-density urbanization patches in the upper sector (INDEC 2010) surrounding the GAT sampling site. The middle and lower sectors are highly urbanized by La Plata city. The Chubichaminí watershed is almost exclusively covered by rural land use with extensive livestock on grasslands (less than 0.7 cows ha^{-1}). The values of area and percentages of land uses in each watershed are detailed in Table 1.

Physicochemical survey

Dissolved oxygen, water temperature, pH, conductivity, and turbidity were measured in situ (HORIBA Multiparameter U-10). Water samples were collected at each sampling site and were transported to the laboratory in coolers to determine concentrations of nutrient (total phosphorous, soluble reactive phosphorous, ammonium, nitrates and nitrites), organic matter (biochemical and chemical oxygen demand) and total suspended solids. Samples were filtered through 0.45 µm Sartorius membrane filter and analyzed with standard methods (APHA 2012). Additionally, the concentration of *Escherichia coli* was analyzed as an indicator of fecal contamination collecting 100 ml of stream water in sterile containers, which were transported in coolers to the laboratory and analyzed according to standard methods (ISO 9308-1 2014). All the environmental variables were assessed by triplicate and averaged for each sample.

Hydrological survey

The hydrodynamic characterization of the first aquifer level was performed by measuring the phreatic levels at each sampling site. Since the water table was considerably deeper at CNV and GAT, measurements were conducted in existing monitoring drillings 35 m deep at these sites. Conversely, as water tables were shallower in CH1 and CH2, 5 m drillings were made manually using a "fish tail" shovel. The monitoring drillings were made with continuous, slot type PVC tubes, from the first 30 cm to the end of the hole and with an opening diameter of 0.1 cm. A water table probe (Solinst 107 TLC meter) was used to measure the water table depth (meter below ground level: m b.g.l.).

The intensity of drainage network intervention in each watershed was quantified by geoprocessing vectorial shapes obtained from government repositories (IGN 2010) using the software Quantum-GIS v3.10.11. The watercourse reaches of each catchment, were discriminated, and classified according to their anthropic modification in non-intervened (natural characteristics) and intervened (man-made, rectified, piped and/or impermeabilized). Based on the summation of non-intervened and intervened reaches lengths (km), the percentage of drainage network intervention (%INTER) was estimated for each watershed. Values of riverine wetlands coverage (km^2) were obtained and expressed as percentages from each watershed (%WET).

Table 1 Values of area and percentage of land use coverage in the Carnaval, Del Gato and Chubichaminí watersheds, near the city of La Plata, Argentina

	Carnaval	Del Gato	Chubichaminí
Sampling sites	CNV	GAT	CH1 and CH2
Area	55.7 km^2	43.2 km^2	97.5 km^2
Rural land use	27.6%	0.5%	99.8%
Periurban land use	55.3%	47.8%	0.1%
Urban land use	17.1%	51.7%	0.1%

Fish assemblage survey

Fish sampling was carried out applying the same effort at all sampling sites. The open water areas of the streams were surveyed with a seine net with the cod end (seine length 15 m; wing mesh size 10 mm distance between knots and height 1.45 m, cod end length 2 m and mesh size 5 mm) in a 20 m section of the stream. The microhabitats formed by vegetated sectors were sampled with a D-shaped net (0.36 m width by 0.46 m height, 1 × 1 mm mesh size and 2 m handle) pulled along one linear meter.

Easily identifiable fish species were recorded in the field and released. The other captured individuals were euthanized in an anesthetic solution (benzocaine) in excess, fixed in 10% formaldehyde solution, and then transferred to 70% ethanol for laboratory identification under a stereoscopic microscope. Species were identified following Azpelicueta and Braga (1991), Braga (1993), Říčan and Kullander (2008), Almirón et al. (2015) and Rosso et al. (2018). Updates in taxonomy were reviewed following Mirande and Koerber (2015) and Terán et al. (2020).

Specific richness (S) was recorded for each sample. Specific abundance for each sample was recorded by fishing gear, standardized according to the area covered efficiently by each fishing gear (m^2) and finally averaged to make a single abundance value per sample (N), expressed as individuals per square meter (ind m^{-2}). Since the D-shaped net was employed to study exclusively vegetated areas, abundance data were corrected according to the percentage of these areas in each stretch of the stream analyzed. From these data, the Shannon–Wiener diversity index (H') was calculated for each sample. The variables S, N, and H' will be referred to as attributes of fish assemblages henceforth.

Statistical analysis

Differences in physicochemical and hydrological variables between sampling sites were assessed by Kruskal–Wallis non-parametric analysis of variance using the software SigmaPlot 12. Also, Non-metric Multidimensional Scaling (NMDS) analysis was performed to assess the dissimilarity between periurban and rural sampling sites according to physicochemical and hydrological variables in an integrated way. Distance matrixes were built using the Euclidean distance method. The environmental variables (except for pH) were previously $\log10(x+1)$ transformed. Differences between periurban and rural sampling sites were tested through one-way ANOSIM. The method of point biserial correlation was performed to identify the variables that explain the differences associated with each land use. These analyses were carried out using the packages "vegan" (Oksanen et al. 2020) and "indicator species" (De Cáseres et al. 2010) from the software R Core Develop Team.

Differences in attributes of fish assemblages between sampling sites were assessed by Kruskal–Wallis analysis. The relevance of hydrological and physicochemical variables to S, N, and H' were evaluated by Generalized Linear Models (GLM). Gaussian distribution for response variable and identity function was used. The best set of explanatory variables was selected by the stepwise forward selection method. Model statistical significance and pseudo-R^2 coefficient were calculated by Chi-square ANOVA function and Nagelkerke index, respectively. These analyses were carried out using the package "MASS" (Ripley et al. 2013).

Non-metric Multidimensional Scaling (NMDS) analysis was performed to assess the similarity of sampling sites according to the fish assemblage structure. Bray–Curtis similarity index was calculated from abundance data previously $\log10(N+1)$ transformed. One-way ANOSIM analysis was carried out to test the significance between periurban and rural sample groups. The quantitative contribution of species to the similarity within each group was identified through the analysis of percentage similarity (SIMPER). The analyses were carried out using the package "vegan" (Oksanen et al. 2020) from the software R Core Develop Team.

Partitioned analysis of variance was performed to discriminate the influence of physicochemical and hydrological set of variables on the structure of fish assemblages. Species with total percentage abundance less than 0.5% in relation to the total catch were excluded from the analysis to reduce possible biases (Ter Braak and Smilauer 1998). Environmental variables were previously normalized and fish assemblage abundance data were $\log10(N+1)$ transformed. In the preliminary run, a Detrended Correspondence Analysis (DCA) was performed from fish assemblage data. Since the length of the gradient was 2.28 units of standard deviation for the first DCA axis, a lineal response model was used (Ter Braak and Smilauer 1998) and Redundancy Analyses (RDA) was performed. Physicochemical and hydrological variables were selected through forward selection method and Variance Inflation Factor (VIF $<$ 20) to avoid biases due to multicollinearity among variables (Ter Braak and Smilauer 1998). The analyses were carried out using the packages "vegan" (Oksanen et al. 2020) from the software R Core Develop Team. In all analyses, results with a p-value $<$ 0.05 were considered statistically significant.

Results

Environmental variables

Physicochemical variables showed that pH, conductivity, turbidity, Chemical Oxygen Demand (COD), Biochemical Oxygen Demand (BOD), and *E. coli* were significantly higher at rural sites. Soluble Reactive Phosphorus (SRP) and Total Phosphorus (Total P) were significantly highest at periurban sites (Table 2). Dissolved oxygen concentration (DO), suspended solids (SS), and nitrogen compounds (nitrites, nitrates, and ammonium) showed no significant differences between sites. Kruskal–Wallis results and values of average and standard deviation of environmental variables are shown in Table 2.

The intervention of watercourses evidenced a higher longitudinal hydrologic alteration in watersheds with mainly periurban land use, recording values of 42.4 and 51.2%INTER (CNV and GAT watersheds, respectively), in contrast with rural watercourses (CH1 and CH2 sites watershed) that showed values of 1.1%INTER (Fig. 1b). The variations in lateral hydrologic connectivity of the riverine system were observed in the area covered by wetlands; the periurban watersheds yielded values lower than 0.1%WET and the rural ones 15.4%WET (Fig. 1b).

The depth of the first phreatic level was significantly higher in periurban than rural sites (16.4 and 0.40 m b.g.l., respectively). This result evidence the influence of the first phreatic level on the stream base flow in rural sites, conversely to the periurban sites which are not linked to the water table and their water level depends only on the lateral and longitudinal flow of the surface water (rainfall and/or effluents). It is worth noting that the CNV sampling site dried up during the summer-autumn of 2018, evidencing the consequences of their longitudinal, lateral, and vertical hydrologic modifications.

The NMDS and ANOSIM results confirmed that rural and periurban sampling sites were statistically different in terms of their environmental variables ($R=0.94$, $p=0.001$, Fig. 2a). The point biserial correlation method identified that the variables %INTER, WT, and SRP were positively and significantly associated with periurban sites, while %WET,

Table 2 Average and standard deviations (±) of environmental variables and results of the statistical comparison between rural and periurban sampling sites near La Plata city, Argentina

Variable (unit)	Code	Rural sites	Periurban sites	Significance
Temperature (°C)	Temp	24.0 ± 1.7	21.5 ± 2.9	*
pH	pH	8.4 ± 0.3	7.9 ± 0.2	**
Conductivity (μS cm^{-1})	Cond	931.4 ± 90.9	599.7 ± 167.4	***
Dissolved oxygen (mg l^{-1})	DO	5.1 ± 2.2	6.0 ± 1.3	NS
Turbidity (NTU)	Turb	426.5 ± 237	112.4 ± 60.6	**
Nitrates (μg l^{-1})	NO_3^-	1350.8 ± 2332.1	2932.9 ± 4726.1	NS
Nitrites (μg l^{-1})	NO_2^-	37.2 ± 23.1	1611.2 ± 2311.3	NS
Ammonium (μg l^{-1})	NH_4^+	168.9 ± 54.5	105.3 ± 81.1	NS
Soluble reactive phosphorus (μg l^{-1})	SRP	212.8 ± 125.9	1323.7 ± 219.1	***
Total phosphorus (μg l^{-1})	Total P	470.8 ± 308.1	1444.6 ± 248.4	***
Chemical oxygen demand (mgO_2 l^{-1})	COD	67.2 ± 15.9	25.4 ± 8.5	***
Biochemical oxygen demand (mgO_2 l^{-1})	BOD	10.2 ± 1.6	6.8 ± 3.6	*
Suspended solids (mg l^{-1})	SS	80.8 ± 105.8	31.2 ± 27	NS
Escherichia coli MPN 100 ml^{-1}	*E. coli*	5446.1 ± 2824.2	895.1 ± 928.9	***
Water table depth (m b.g.l.)	WT	0.40 ± 0.01	16.4 ± 4.9	***

*$p < 0.05$, **$p < 0.01$, ***$p < 0.001$, *NS* no significant

COD, *E. coli*, conductivity, turbidity, and pH were significantly associated with rural sites (Fig. 2c).

Fish assemblages

A total of 3636 fishes were caught, corresponding to 32 species distributed in 13 families and within 5 orders (Table 3). Characiformes was the most represented order with a total of 19 species, followed by Siluriformes (6), Cyprinodontiformes (3), Cichliformes (2), and a single species for Synbranchiformes and Cypriniformes. *Cnesterodon decemmaculatus* was the dominant species in the periurban sampling sites (CNV and GAT) while *Cheirodon interruptus* dominated in the rural ones (CH1 and CH2).

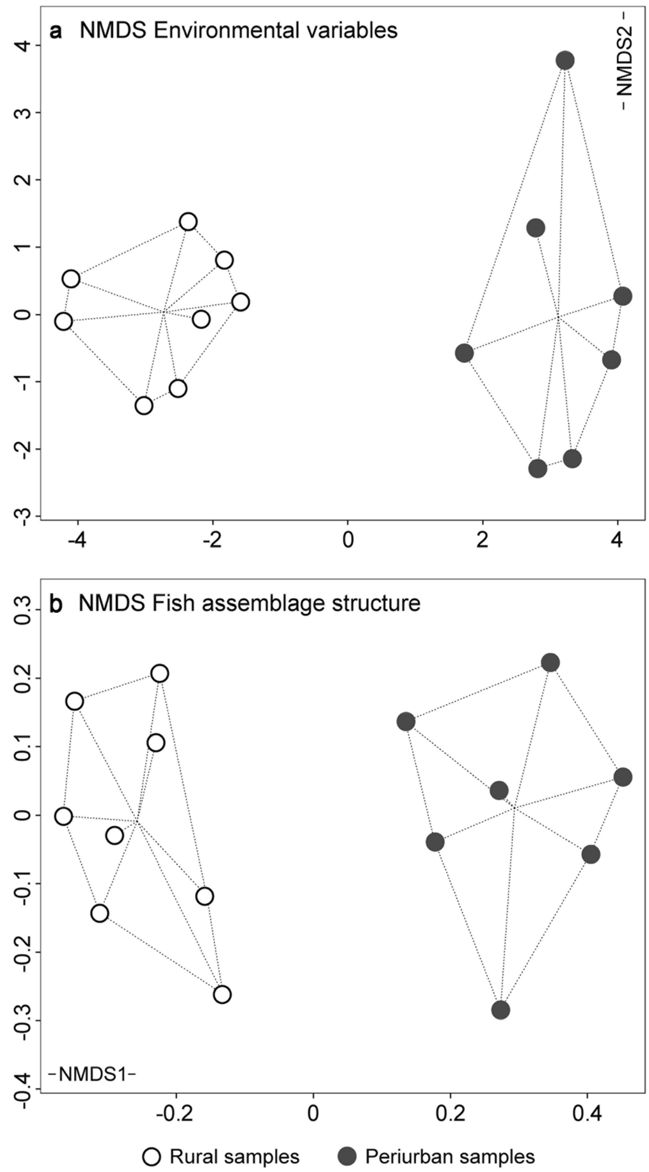
The fish assemblage attributes varied significantly between rural and periurban sites (Fig. 3a). The attributes of S and H' were significantly higher in rural sites compared to periurban ones ($p < 0.001$; Fig. 3a). Conversely, N was significantly greater in periurban sampling sites ($p < 0.001$; Fig. 3a). According to GLM results (Fig. 3b), S was negatively related to %INTER and positively related to DO and turbidity (pseudo-$R^2 = 0.92$; p-value < 0.001). In the case of N, a positive relationship was found with the SRP concentration (pseudo-$R^2 = 0.59$; p-value < 0.001), while H' was positively related to %WET (pseudo-$R^2 = 0.98$; p-value < 0.001).

Differences in the structure of the fish assemblage were evidenced by NMDS (Fig. 2b). Rural and periurban sampling sites were separated in two statistically different groups according to ANOSIM ($R = 0.93$, p-value = 0.001). SIMPER analysis (Fig. 2d) revealed a homogeneous contribution of fish species in the separation of rural and periurban groups. Among the most important species for this separation, *Cnesterodon decemmaculatus*, *Corydoras paleatus*, and *Phalloceros caudimaculatus* were the most representative of periurban sampling sites, while several species, such as *Diapoma terofali*, *Pseudocorynopoma doriae*, *Hyphessobrycon meridionalis*, *Psalidodon eigenmanniorum*, *Psalidodon rutilus*, *Charax stenopterus*, *Steindachnerina biornata,* and *Hypostomus commersoni* were representative of rural sampling sites.

The physicochemical partial RDA analysis identified the SRP, BOD, and conductivity as the best subset of variables explaining the fish assemblage structure. For partial RDA analysis hydrological variables, %INTER, and WT were identified as the best combination to explain the variation in fish assemblage structure (Fig. 4). The partitioned analysis of variance indicated that the structure of the fish assemblages was jointly explained by the physicochemical and hydrological variables, showing significant results in the analysis where both sets of variables were considered simultaneously (Fig. 4). The relationships between biotic structure with each set of environmental variables separately showed no significant results (Fig. 4).

Discussion

The analysis of physicochemical and hydrological variables indicated clear differences between the headwater streams located in catchments with different land uses. Considering the homogeneous climatic, geological, and topographical characteristics of the study region (Kruse et al. 2014), the differences recorded between sampling sites can be mainly attributed to the differential anthropogenic impacts in their catchments. Periurban sites exhibited the highest concentrations of SRP and Total P, as was



c Biserial point correlation results

Rural group		Periurban group	
Variable	**Rho**	**Variable**	**Rho**
%WET	0.99***	%INTER	0.99***
COD	0.87***	WT	0.99***
E. coli	0.78***	SRP	0.96***
Conductivity	0.76***		
Turbidity	0.72**		
pH	0.67**		

d Fish assemblage SIMPER results

Species	Rural av. abund	Periurban av. abund	Cumulated contr.
Cnesterodon decemmaculatus	0.91	1.86	0.11
Diapoma terofali	0.59	0	0.18
Pseusocorynopoma doriae	0.79	0.26	0.24
Hyphessobrycon meridionalis	0.54	0	0.3
Corydoras paleatus	0.15	0.64	0.36
Phalloceros caudimaculatus	0	0.43	0.41
Psalidodon eigenmanniorum	0.39	0	0.46
Psalidodon rutilus	0.38	0	0.5
Bryconamericus iheringii	0.44	0.3	0.54
Hypostomus commersoni	0.29	0	0.58
Cyphocharax voga	0.29	0.1	0.61
Steindachnerina biornata	0.3	0	0.65
Cheirodon interruptus	0.99	0.91	0.68
Australoheros facetus	0.26	0.19	0.71
Gymnogeophagus meridionalis	0.27	0.16	0.74
Charax stenopterus	0.29	0	0.78
Jenynsia lineata	0.17	0.08	0.8
Pimelodella laticeps	0.17	0.06	0.82
Characidium rachowii	0.16	0	0.84

Fig. 2 Sampling sites ordination by Non-metric Multidimensional Scaling (NMDS) according to **a** environmental variables and **b** structure of the fish assemblages, coupled with the detailed analysis of **c** the environmental variables and **d** fish species responsible for the grouping of the sampling sites. (*) $p<0.05$, (**) $p<0.01$, (***) $p<0.001$, (NS) no significant. The periurban group has one sample less than rural group because during the summer-autumn of 2018 the Carnaval sampling site dried up

observed in other Pampean catchments with similar land use (Solis et al. 2016; Arias et al. 2020; Paracampo et al. 2020) and other watersheds influenced by urbanization and agriculture elsewhere (Paul and Meyer 2001; Freeman et al. 2007). In rural sites, the highest values recorded for turbidity, suspended solids, COD, and *E. coli* can be associated with livestock activity according to several studies (Kondolf 1993; Gammon et al. 2003; Vidon et al. 2008). Also, the interaction between the stream and water table in rural areas increases the conductivity due to the high solute load characteristic of groundwater (Schmidt et al. 2012), revealing its contribution to the physicochemical and hydrological stability of streams. Conversely, the disconnection between groundwater and streams in periurban watercourses determines that physicochemical variables were mainly affected by land use activities at these sites. Indeed, under natural hydrological schemes, the streams and groundwater of this area have a vertical connection associated with the low phreatic depth, showing an effluent stream condition (Kruse et al. 2003; Deluchi et al. 2005; Laurencena et al. 2010). However, overexploitation of groundwater for urban supply and crop irrigation in the periurban area of La Plata city have deepened the water table and shifted the streams to an influent condition, in

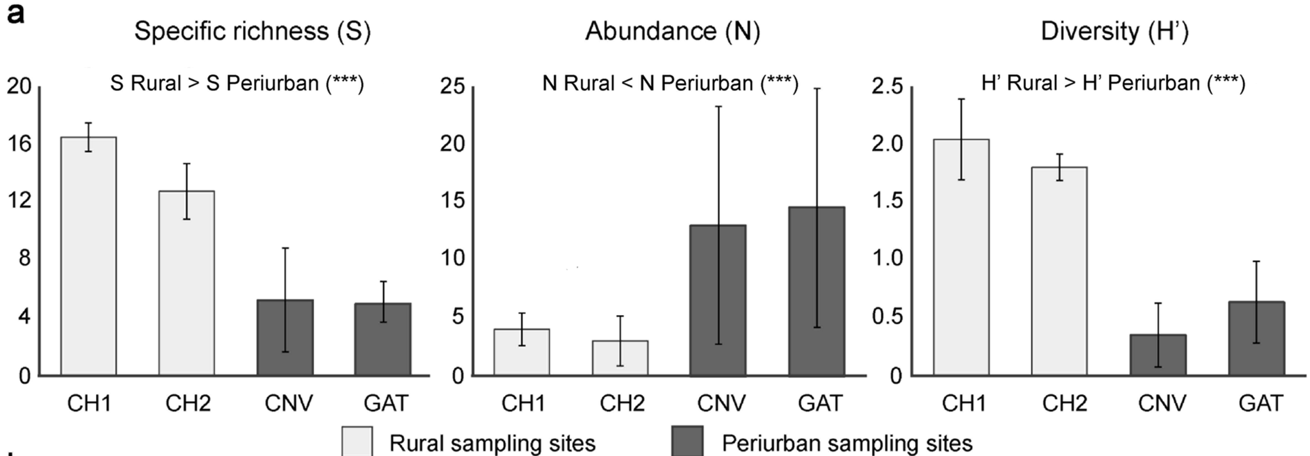
Table 3 Average and standard deviation (±) values of specific abundances (ind m^{-2}) of fishes recorded in rural (Chubichaminí: CH1–CH2 sites) and periurban sampling sites (Carnaval: CNV site, and Del Gato: GAT site) near La Plata city, Argentina

Systematic position	Code	Rural sites		Periurban sites	
		CH1	CH2	CNV	GAT
Cypriniformes					
Cyprinidae					
Cyprinus carpio Linnaeus 1758	*Cyp_ca*	0.08 ± 0.12	0.05 ± 0.07	–	0.08 ± 0.14
Characiformes					
Curimatidae					
Cyphocharax voga (Hensel, 1870)	*Cyp_vo*	0.01 ± 0.01	–	–	–
Steindachnerina biornata (Braga & Azpelicueta 1987)	*Ste_bi*	0.09 ± 0.10	0 ± 0.01	–	–
Prochilodontidae					
Prochilodus lineatus (Valenciennes 1837)	*Pro_li*	–	0.01 ± 0.01	–	–
Anostomidae					
Megaleporinus obtusidens (Valenciennes 1837)	*Meg_ob*	0.01 ± 0.01	0 ± 0.01	–	–
Erythrinidae					
Hoplias argentinensis Rosso, González-Castro, Bogan, Cardoso, Mabragaña, Delpiani & Díaz de Astarloa, 2018	*Hop_ar*	0.02 ± 0.02	0 ± 0.01	0.01 ± 0.02	0.03 ± 0.05
Characidae					
Andromakhe stenohalinus (Messner, 1962)	*And_st*	0.02 ± 0.02	–	–	–
Astyanax lacustris (Lütken 1875)	*Ast_la*	0.01 ± 0.01	–	–	–
Bryconamericus iheringii (Evermann & Kendall, 1906)	*Bry_ih*	0.09 ± 0.13	0.07 ± 0.07	0.24 ± 0.09	–
Charax stenopterus (Cope 1894)	*Cha_st*	0.11 ± 0.08	0.01 ± 0.03	–	–
Cheirodon interruptus (Jenyns, 1842)	*Che_in*	1.13 ± 0.72	0.99 ± 0.65	1.38 ± 1.66	1.25 ± 1.48
Diapoma terofali (Géry 1964)	*Dia_te*	0.17 ± 0.16	0.43 ± 0.29	–	–
Hyphessobrycon meridionalis Ringuelet, Miquelarena & Menni, 1978	*Hyp_me*	0.04 ± 0.05	0.03 ± 0.02	–	–
Oligosarcus jenynsii (Guenther, 1864)	*Oli_je*	0.02 ± 0.03	0.02 ± 0.03	–	–
Oligosarcus oligolepis (Steindachner 1867)	*Oli_ol*	0.02 ± 0.02	0 ± 0.01	–	–
Psalidodon anisitsi (Eigenmann, 1907)	*Psa_an*	0.02 ± 0.03	–	–	–
Psalidodon eigenmanniorum (Cope, 1894)	*Psa_ei*	0.12 ± 0.19	0.05 ± 0.05	–	–
Psalidodon rutilus (Jenyns, 1842)	*Psa_ru*	0.12 ± 0.08	0.02 ± 0.02	–	–
Pseudocorynopoma doriae Perugia, 1891	*Pse_do*	0.42 ± 0.31	0.52 ± 0.39	0.07 ± 0.06	0.07 ± 0.12
Crenuchidae					
Characidium rachovii Regan, 1913	*Cha_ra*	0.01 ± 0.02	0.02 ± 0.02	–	–
Siluriformes					
Heptapteridae					
Pimelodella laticeps Eigenmann, 1917	*Pim_la*	0.03 ± 0.05	0.01 ± 0.02	0.01 ± 0.02	–
Callichthyidae					
Corydoras longipinnis Knaack, 2007	*Cor_lo*	–	0.01 ± 0.02	0.05 ± 0.09	–
Corydoras paleatus (Jenyns, 1842)	*Cor_pa*	0.01 ± 0.01	0.02 ± 0.02	0.23 ± 0.27	0.82 ± 1.1
Loricariidae					
Loricariichthys anus (Valenciennes, 1836)	*Lor_an*	–	0.01 ± 0.01	–	–
Hypostomus commersoni Valenciennes, 1836	*Hyp_co*	0.49 ± 0.23	0.07 ± 0.04	–	–
Otocinclus arnoldi Regan, 1909	*Oto_ar*	0.04 ± 0.05	–	0.01 ± 0.02	–
Cyprinodontiformes					
Poeciliidae					
Cnesterodon decemmaculatus (Jenyns, 1842)	*Cne_de*	0.96 ± 1.16	0.79 ± 0.52	15.28 ± 5.26	10.15 ± 8.55
Phalloceros caudimaculatus (Hensel, 1868)	*Pha_ca*	–	–	–	1.34 ± 1.32
Anablepidae					
Jenynsia lineata (Jenyns, 1842)	*Jen_li*	0.02 ± 0.03	0.01 ± 0.02	0.04 ± 0.06	–

Table 3 (continued)

Systematic position	Code	Rural sites		Periurban sites	
		CH1	CH2	CNV	GAT
Synbranchiformes					
Synbranchidae					
Synbranchus marmoratus Bloch, 1795	*Syn_ma*	–	–	–	0.01 ± 0.02
Cichliformes					
Cichlidae					
Australoheros facetus (Jenyns, 1842)	*Aus_fa*	0.09 ± 0.07	–	0.04 ± 0.06	0.1 ± 0.17
Gymnogeophagus meridionalis Reis & Malabarba, 1988	*Gym_me*	0.17 ± 0.25	–	0.06 ± 0.06	–

a



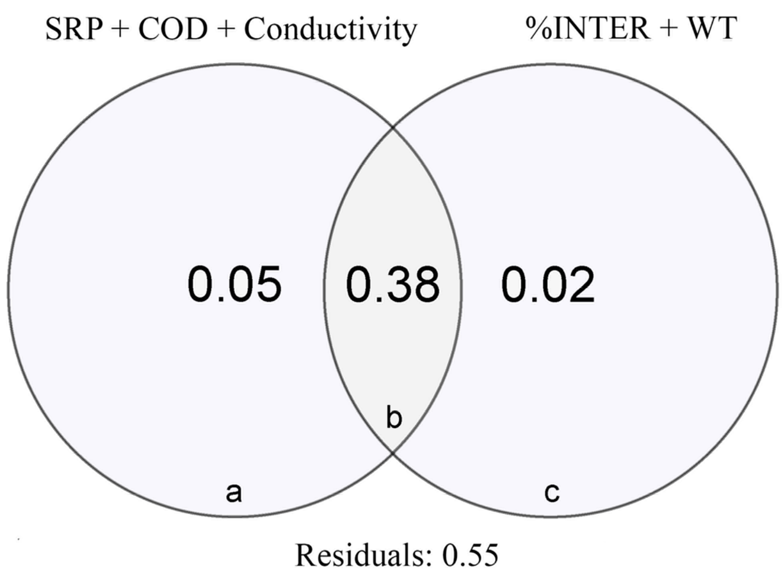
b

Biotic attribute	Variables	Estimate	St. Error	Significance	95% Lower	95%Upper	Pseudo-R^2
Specific richness (S)	Intercept	8.62	2.34	**	4.03	13.21	0.92
	DO	0.61	0.28	*	0.07	1.16	
	Turbidity	0.01	0.002	*	0.002	0.01	
	%INTER	-0.15	0.03	***	-0.21	-0.11	
Abundance (N)	Intercept	1.24	2.35	NS	-3.36	5.84	0.59
	SRP	10.89	2.52	***	5.96	15.83	
Diversity (H')	Intercept	0.56	0.11	***	0.35	0.76	0.98
	%WET	0.09	0.01	***	0.07	0.11	

Fig. 3 **a** Average, standard deviations, and results of the statistical comparison between the attributes of the fish assemblages from periurban (Carnaval: CNV and Del Gato: GAT) and rural (Chubichaminí 1 and 2: CH1 and CH2) sampling sites, near La Plata city, Argentina. **b** Relationships between the attributes of the fish assemblages and statistically significant environmental variables according to GLM results. (*) $p < 0.05$, (**) $p < 0.01$, (***) $p < 0.001$, (NS) no significant

relation to groundwater (Kruse et al. 2003; Deluchi et al. 2005; Laurencena et al. 2010).

Watersheds with mainly periurban land use showed the highest degree of drainage network intervention, as well as the lowest areas covered by wetlands. These alterations, in addition to water table disconnection, indicate high instability in flow speed and the reduction of water permanence (Tucci 2012). Several studies elsewhere have linked the loss of wetlands to alterations in groundwater and stream connectivity (Bernaldez et al. 1993; Serrano and Serrano 1996; Perkin et al. 2017), as well as to increases in watershed drainage (Erickson et al. 1979; Brandolin et al. 2013;



Variables sets	Significance
a + b	***
b + c	***
a + b + c	***
a	NS
c	NS

a) Physicochemical variables
b) Covariance between a and c
c) Hydrological variables

Fig. 4 Results of partitioned analysis of variance from periurban (Carnaval: CNV and Del Gato: GAT) and rural (Chubichaminí 1 and 2: CH1 and CH2) sampling sites, near La Plata city, Argentina. Variance of fish assemblage structure explained by **a** physicochemical variables, **c** hydrological variables, and **b** the covariance between both sets of variables. (*) $p<0.05$, (**) $p<0.01$, (***) $p<0.001$, (NS) no significant

Myers et al. 2013). Indeed, the fact that CNV site dried up during the summer-autumn of 2018 is a clear evidence of how changes in the longitudinal, lateral, and vertical stream connectivity modified the natural hydrologic dynamics of the studied periurban streams. Therefore, it is to be expected that biotic communities in these areas will be more adversely affected than those inhabiting the rural ones.

The 28 fish species recorded in rural sites can be considered a high richness for Pampean headwater streams according to reports from previous studies (Ringuelet 1975; Fernández et al. 2008; Di Marzio et al. 2003; Colautti et al. 2009). In contrast, periurban sites showed less than half of the specific richness recorded in such environments. According to GLM results, turbidity, dissolved oxygen, and %INTER were the variables that best explained the variation in this biotic attribute. Although turbidity is generally considered to have negative effects on fish assemblages (e.g. Waters 1995; Heitke et al. 2006), a positive relation with specific richness was recorded in agreement with other studies reporting the positive effect of turbidity on the complexity of fish assemblages (Turesson and Brönmark 2007; Dodrill et al. 2016). Considering that Pampean streams may be naturally turbid (Feijoó et al. 1999; Bauer et al. 2002), it is expected that a moderate increase in this variable due to extensive livestock activity will have little effect on fish communities, as they can inhabit environments with these characteristics. Therefore, this finding supports the hypothesis that turbidity may be important for maintaining the integrity of fish assemblages in prairie streams that have naturally high suspended sediment loads, as was previously reported by Bonner and Wilde (2002). Dissolved oxygen is widely recognized as a key variable in the structuring of fish assemblages (Franklin 2014). Although no differences were found for this variable between rural and periurban sites in this study, the positive relationship with richness could be reflecting that fluctuations in these variables are independent of the land use surrounding the studied headwater streams. The negative relationship between fish specific richness and %INTER can be linked to water flow instability and loss of permanent marginal habitats. These alterations cause the isolation of the remaining stream sectors that concentrate aquatic organisms (Datry et al. 2016). In this scheme of fragmented systems, the number of species depends on the local extinction and colonization rates, according to the framework of metacommunity theory (Leibold et al. 2004). The reduction of habitats increases the biotic and abiotic pressures, raising the local extinction of species related to predation, competition, and environmental filtering processes (Larned et al. 2010; Datry et al. 2016). In parallel, the loss of longitudinal connectivity within the watershed limits the colonization, resulting in the decrease of fish specific richness (Meyer et al. 2007; Shao et al. 2019). The relationship between the highest values of fish diversity and %WET in the rural watershed supports the concept that streams associated with riverine wetlands can develop and maintain a high diversity at local and watershed scale, given their extension, heterogeneity, variety of habitats and because they can represent ecological corridors that favor migration processes (Wantzen and Junk 2000; Wantzen et al. 2008).

The highest fish abundance recorded in periurban areas was related to SRP concentrations according to GLM results. In our study, the analysis of the structure of fish assemblages evidenced that higher abundances in periurban sites were caused by the high density of the small Poeciliidae *C. decemmaculatus*. Several studies recorded high densities of Poeciliidae species in disturbed environments, linked to their ability to tolerate hypoxic conditions or high pollutant concentrations (Araújo et al. 2009; Paracampo et al. 2020; Paredes del Puerto et al. 2021). Thus, the tolerant characteristics and opportunistic reproductive strategies (sensu Winemiller 1989) of *C. decemmaculatus*, in addition to the decrease in interspecific competition by the loss of species described above, may explain the success of this species in disturbed environments and, consequently, the differences in fish abundance recorded in this study. The detailed analysis of fish assemblage structure by SIMPER revealed that *C. decemmaculatus*, *C. paleatus,* and *P. caudimaculatus* were dominant in periurban sites. As was described for *C. decemmaculatus*, the other two species are considered also tolerant to pollution since they are adapted to survive hypoxic conditions (Kramer and Mehegan 1981; Plaul et al. 2016). In addition, the opportunistic and omnivore species *Cheirodon interruptus* and *Bryconamericus iheringii* (Menni and Almirón 1994; Ferriz et al. 2010; López van Oosterom et al. 2013; García et al. 2019) showed high abundances in periurban as well as in rural sites. A similar situation occurred with *Gymnogeophagus meridionalis* and *Australoheros facetus*, which have been recorded in environments with moderate disturbances in previous studies (Yorojo Moreno et al. 2017; Paredes del Puerto et al. 2020). On the contrary, most of the species that characterize rural sites have been associated with low impacted conditions (Remes Lenicov et al. 2005; Bertora et al. 2018; Paredes del Puerto et al. 2021). Many of these species display a wide variety of trophic habits such as the invertivorous, carnivorous, detritivorous, and omnivorous (Escalante 1987a, b; Menni 2004; Fernández et al. 2008; Machin 2012; González Sagrario and Ferrero 2013; Brancolini et al. 2014) and equilibrium or seasonal reproductive strategies (Winemiller 1989; Winemiller et al. 2008). Thus, fish assemblages in rural sites are similar to those associated with permanent, structured, and low impacted Pampean environments, as opposed to those characterizing periurban sites that show adaptations to live in impacted and hydrologically unstable environments.

Empirical studies have shown that the structure and functional organization of fish assemblages in lotic systems vary according to hydrological fluctuation (Aadland 1993; Frenzel and Swanson 1996; Mims and Olden 2013), as well as to water quality impairment by anthropic activities (Paul and Meyer 2001; Allan 2004; Paracampo et al. 2020). The partitioned analysis of variance showed that the differences found in fish assemblage structure were explained by the interaction of hydrological and physicochemical variables, indicating a synergistic effect of alterations in watershed hydrology and water quality deterioration by anthropic activities. Besides SPR and %INTER, the variables conductivity, COD and WT were the most important in this relation. High values of COD and conductivity have been associated with environmental pollution of aquatic systems (Daga et al. 2012; Atique et al. 2020); however, in our study highest values of these variables were recorded in rural streams. Although livestock activity may increase the COD values, under relatively undisturbed conditions, Pampean streams may have naturally high organic matter loads, mainly linked to the decomposition of macrophytes (Feijoó and Lombardo 2007), and high conductivity due to the influence of the water table (Kruse et al. 2003). Indeed, the highest values of fish species richness and diversity reported for rural Pampean streams were associated with conductivity and organic matter ranging within the maximum values observed in this study (Menni et al. 1996; Paracampo et al. 2020; Paredes del Puerto et al. 2021).

Changes in fish assemblages of Pampean headwater streams reflect not only alterations in water quality caused by local land use, but also the influence of environmental integrity of catchments caused by hydrological modifications, as well as other factors operating at broader scales such as the disruption in watershed-main rivers connectivity. Indeed, the presence of juveniles of migratory species such as *Prochilodus lineatus* and *Megaleporinus obtusidens* in rural sites is another noticeable difference with the periurban ones. Since these species breed in the main rivers of the Del Plata basin (Uruguay, Paraná, Paraguay and Río de la Plata) and use the associated shallow lakes, wetlands, and headwater streams as rearing sites (Baigún et al. 2003; Lozano et al. 2019) their absence in periurban sites reveals the loss of connectivity of these watercourses with the Rio de la Plata estuary by the impact caused by urban land use in middle and lower sectors of the watersheds. This interpretation is supported by studies documenting that urbanization produces drastic changes in river systems (Walsh et al. 2005), generating severe degradation of the fish assemblages when occupying above 15% of the catchment (Yoder et al. 1999), or their complete loss over 30% (Klein 1979), as was observed in lower stretches of CNV and GAT streams. In this sense, the ichthyofauna of periurban headwaters is isolated from main watercourses and, therefore, the colonization after droughts or contamination pulses is highly restricted.

In view of the ongoing human population growth and the evolution of land use in the region (López and Rotger 2020), the projection of a future ecological scenario may become alarming with respect to the integrity of Pampean streams. In this context, periurban streams will be integrated into the urban landscape following the "urban stream syndrome" (Walsh et al. 2005), meanwhile, the livestock catchments will change into agricultural systems due to the displacement of the periurban land use, transferring their impacts to the streams. Within this scenario, this research provides the necessary background for the generation of tools for sustainable management of river systems in lowland areas where urban growth planning must be addressed from an ecohydrological approach.

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Author contributions JMPP: investigation, conceptualization, methodology, formal analysis, resources, data curation, writing-original draft, writing—review and editing, and visualization. IG: resources, data curation, and writing—review and editing. TM: resources, data curation, and writing—review and editing. AP: resources, data curation, writing—review and editing. LRC: resources, data curation, writing—review and editing. JGS: resources, data curation, and writing—review and editing. MM: resources, data curation, and writing—review and editing. DC: investigation, conceptualization, methodology, formal analysis, resources, data curation, writing—review and editing, visualization, supervision, project administration, and funding acquisition.

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Data availability The datasets generated during and/or analyzed during the current study are available from the corresponding author on reasonable request.

Code availability Not applicable.

Declarations

Conflict of interest The authors declare that they have no known competing financial interests or personal relationships that may have influenced the work reported in this paper.

Ethics approval Care during collection and handling of fish for this study complied with the Buenos Aires Province (Argentina) Wildlife and Fisheries Authority guidelines and policies (Law 11,477). The collections for this study were not a part of faunal surveys and they did not employ any type of experimental procedure, surgery or chemical agents that would induce neuromuscular blockage or injury on the collected organisms. All fish collected were euthanized as humanely as possible by anesthetic overdose to prevent unnecessary suffering.

Consent to participate Not applicable.

Consent for publication Not applicable.

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