

ORIGINAL ARTICLE

Fish assemblage structure in a Neotropical urbanised prairie stream exposed to multiple natural and anthropic factors

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Abstract

A myriad of factors affect stream fish assemblages, and most of them are intimately regulated by surrounding land uses. One of the most pernicious impacts on streams is the release of urban effluents. Accordingly, understanding the fish assemblage patterns as well as the environmental drivers that modulate them in the context of urban streams which also are exposed to other land uses is mandatory. We evaluated the longitudinal patterns of fish assemblages in an urbanised Neotropical prairie stream exposed to downstream patches of different surrounding land uses. Several variables of water quality, in-stream habitat and riparian corridors were measured, and their relationships with fish metrics were explored. A fish metric selection protocol was implemented, and several multivariate analyses were used. Three main patterns of fish response were observed. Assemblage-level attributes as diversity, richness, biomass, number of families and trophic guilds and proportion of Characiformes and intolerant species were responsive to the natural gradient in habitat (pools) and riparian (riparian width) conditions expected in non-impacted lotic ecosystems. Conversely, a downstream continuum of recovery in water quality (dissolved oxygen, conductivity, NO₃:NH₄ and bacteriological loads) and habitat structure (sediment depth) would influence the distribution of fish species regarding their environmental tolerance and habitat preferences. Finally, local habitat aspects (nitrites, submerged macrophytes, bare soil cover and bank incisions by livestock) seemed relevant for the trophic structure (omnivorous species) of fish assemblages. A complex natural and anthropic longitudinal scenario together with local disruptions imposed by surrounding land uses were directive for fish.

KEYWORDS

ecological integrity, fish traits, in-stream habitat, riparian conditions, surrounding land uses, urban continuum network

1 | INTRODUCTION

Fluvial hydrosystems are known to show longitudinal patterns of variation in physical, geomorphologic and hydrological features from headwaters to mouth (Vannote et al., 1980). Following this

natural longitudinal gradient, freshwater biota show changes in assemblage organisation in terms of species abundance and composition, trophic and functional guilds. Nevertheless, freshwater ecosystems have been greatly altered worldwide through the modification of natural landscapes and land-use intensification (Albert

et al., 2020). In this scenario, lotic ecosystems flow through highly fragmented basins where patches of different surrounding land uses load large inputs of material and energy resulting in a rapid decline of their ecological integrity (Allan, 2004). This produces a marked deterioration of these environments due to the synergic negative effects between water quality degradation and changes in the fluvial habitat structure, riparian condition, macrophytes coverage, channel morphology and substrate composition (Platts, 1979; Walsh et al., 2005). Therefore, longitudinal variation of land uses may be interpreted as a disruptor of the continuum in the natural functioning of fluvial ecosystems (Poole, 2002). Since water quality, in-stream habitat structure and riparian condition are key factors for the sustainability of aquatic communities (Allan & Flecker, 1993), many freshwater biota of fragmented ecosystems (Clapcott et al., 2012; Grenouillet et al., 2008) have longitudinal patterns of assemblage attributes largely different from those expected in natural longitudinal variation.

Fish communities are particularly sensible to the alterations of aquatic ecosystems and especially to changes in land use of the drainage basin (Fausch et al., 1990). Indeed, the use of fish metrics (communities attributes which account for the richness, abundance, trophic composition and condition of the specimens) leads to a more accurate understanding and assessment of the ecological integrity of freshwater ecosystems (Karr, 1981). Several studies from disparate different lotic ecosystems worldwide have demonstrated that stream fish assemblages are affected by water quality, habitat heterogeneity and riparian vegetation, all factors highly influenced by surrounding land uses (Allan, 2004). For instance, in United States, the abundance of tolerant species has shown to increase as water turbidity, nutrients and chloride increase (McCormick et al., 2001). These authors also found that tolerant species are negatively related with various multimetric indices assessing channel, riparian and watershed quality. Degradation of water quality also caused a marked decline in fish abundance and species richness in populated basins of Europe (Boët et al., 1999) and Asia (Gagny et al., 2000). Respect to in-stream habitat, larger mean substrate size supports high assemblage diversity, abundance and proportion of specialists (Kautza & Sullivan, 2012) whereas channelisation leads to homogenisation of habitat conditions, resulting in a general decline in specialised species (Boët et al., 1999). Stream width and water depth are important habitat features for limnetic fish species (Li et al., 2012). Patches of riparian corridor devoid of vegetation cover were associated with decreased fish abundance and the presence of sediment-tolerant and sometimes invasive species (Jones et al., 1999).

In the Neotropical region, changes in water conductivity, concentrations of dissolved oxygen and total phosphorus affected the composition and structure of stream fish assemblages of urban streams from the upper Paraná River (Daga et al., 2012). Most impacted streams of the Pampa Plain, with high concentrations of phosphorus, showed impoverished fish assemblages, composed of species tolerant to environmental pollution (Paracampo et al., 2020). Similarly, increasing land use intensity produced that oligotrophic

streams of Uruguay were characterised by a higher evenness, larger individuals and at least 40% of sensitive species, while eutrophic reaches showed the opposite community features with less than 20% of sensitive species (Chalar et al., 2013). Overall, a reduction in water quality is coupled with less diverse assemblages and few dominant species (Bertora et al., 2018a). Higher species richness and assemblage equitability are associated with larger heterogeneity of available habitats (Ferreira & Casatti, 2006) and larger vegetation cover in preserved riparian corridors (Cetra & Petreire, 2007; Tibúrcio et al., 2016). A high proportion of piscivorous was collected in streams with low total suspended solids, high coverage of macrophytes (Granitto et al., 2016) and high riparian quality (Bertora et al., 2018a; Granitto et al., 2016).

All this evidence demonstrates that there are a myriad of factors affecting fish fauna in streams and that most of them are intimately regulated by surrounding land uses. In this respect, one of the most pernicious impacts on fluvial hydrosystems is perhaps the release of urban and industrial effluents. Urban reaches of fluvial ecosystems are characterised by a deterioration of water quality due to the increase of nutrients, organic matter, metals, pollutants, solids, chlorides and bacteriological loads from surface runoff and the discharge of effluents (McGrane, 2016; Walsh et al., 2005). This water quality degradation usually recovers downstream (Paul & Meyer, 2001) given by a continuum of fluxes and transformations of carbon, contaminants, energy and nutrients (Kaushal & Belt, 2012). However, urbanisation on stream does not operate in isolation. Downstream from urban point source effluents streams receive large amounts of material and energy from the interaction with surrounding land uses resulting in a rapid decline of their ecological integrity (Allan, 2004). This makes urban streams particularly complex ecosystems (Walsh et al., 2005). Despite general knowledge about urbanisation impacts on streams, the drivers for fish assemblages in such contexts are not totally understood due to their biological diversity and dynamicity (Furlan et al., 2013; Jackson et al., 2001). Accordingly, understanding the fish assemblage patterns and the environmental drivers that modulate them in the context of urban streams which also are exposed to other land uses need to be explored further. Therefore, in order to better understand the functioning of urban streams in highly fragmented basins, the longitudinal gradient imposed by urban effluents and its continuum of recovery should be considered but also the local effects of downstream changes imposed by different patches of surrounding land uses.

In this context, the main objective of this work was to evaluate the longitudinal patterns of fish assemblages in a Neotropical prairie stream exposed to upstream urban effluents and downstream patches of different surrounding land uses. Particularly, we evaluated whether fish traits respond to a natural or anthropic continuum of variation or, instead, local aspects of in-stream habitat and riparian corridors modulated by surrounding land uses are able to regulate them. With this research, we aim to contribute to the understanding of the dynamics of fish in Neotropical prairie streams in the context of increasing urbanisation and changing land uses.

2 | MATERIALS AND METHODS

2.1 | Study area

The Pampa Plain is a large and highly populated region in central Argentina with suitable lands for crop and livestock production. During the last decades, drastic changes in land uses have occurred and a considerable agricultural expansion replacing the natural grasslands was observed. Indeed, water quality of prairie streams of the Pampa Plain is intimately aligned with the prevailing land use in the surrounding landscape (Amuchástegui et al., 2015; Rosso & Fernandez Cirelli, 2013). Nearby agricultural land uses, urbanisation and industries also influence the integrity of riparian vegetation, aquatic macrophytes and in-stream habitat structure (Cortelezzi et al., 2013; Rosso & Fernandez Cirelli, 2013; Cochero et al., 2016; Arocena et al., 2018).

Prairie streams in the Pampa Plain are eutrophic to hypereutrophic systems, with low velocity, herbaceous riparian vegetation, fine substrates and autochthonous macrophytes as main primary producers (Feijoó et al., 1999; Giorgi et al., 2005). The Langueyú stream (LS) belongs to the homonymous basin which has an area of approximately 600 km² and is located in southeast of Buenos Aires province, Argentina (Figure 1). At the Tandilia hills, three first-order steeped tributaries flows to converge in the LS. Downstream, shortly after receiving urban effluents from Tandil City, the LS flows throughout lowlands in a southwest to northeast direction, converges with the Channel 1 and ends into the Atlantic Ocean. High agricultural productivity is observed in the rural area (cropland and livestock), and several industrial developments are located near urbanisations. In the last decades, urban and agricultural areas increased by 81.6% and 39.4%, respectively, and areas for grazing were reduced by 22.8% in a period of 20 years (Vázquez & Zulaica, 2011). In urban and peri-urban sectors, the stream receives pluvial, wastewater and industries effluents (Ruiz de Galarreta et al., 2013). Vacuum truck illegal discharges are also reported (Ruiz de Galarreta et al., 2013).

At these latitudes, the Neotropical fish fauna represents a pauperised subset (Ringuelet, 1975) of the highly diversified assemblages observed at lower latitudes (Malabarba et al., 1998). A check list of the fish fauna of the LS was recently provided (Bertora et al., 2018b). A 15 species-rich community with many tolerant species and one exotic was reported.

2.2 | Sampling sites

Four reaches of 100 m long with contrasting land uses at their immediate landscapes (i.e. lateral, upstream and downstream areas; at least 2-km surrounding land) were selected in the LS (Figure 1). All these sampling reaches were located at the main stem of LS in lowlands, in order to avoid biases introduced by headwaters steeped tributaries. There, the selected reaches represent the different environmental scenarios imposed by the surrounding land uses to which the Pampa Plain streams are exposed. The uppermost reach

represents a strongly impacted condition by urbanisation (urban site = U, 37°16'24" S, 59°07'35" W). Downstream, two sites account for the influence of cropland (annual crops) activity (cropland site = C, 37°11'15" S, 59°08' W) and unrestricted cattle access to the stream (livestock site = L, 37°05'47" S, 59°06'29" W). The last reach represents the less disturbed conditions to which the stream is exposed (natural grassland site = N, 36°55'39" S, 58°56'9" W). Three bimonthly sampling events during three consecutive spring-summer periods (2016/17, 2017/18, 2018/19) were conducted at each sampling site. All samplings campaigns were performed under mean hydrological conditions and avoiding recent rainfall events.

2.3 | Environmental conditions: water quality, in-stream habitat structure and riparian conditions

Several physical, chemical and bacteriological variables were measured to characterise the water quality of different sampling sites. Variables such as temperature, pH, water conductivity, salinity and total dissolved solids were measured *in situ* using a multiparametric probe and dissolved oxygen and percentage oxygen saturation by means of an oxymeter. Water samples were collected at mid-depth and midstream and were kept cold until arriving to the laboratory. Measurements were performed using standardised methods (APHA-AWWA-WEF, 2017): chloride (SM 4500-Cl B), total suspended solids (SM 2540 D), nitrites (SM 4500-NO₂ B), ammonium (SM 4500-NH₃ C), total phosphorous (SM 4500-P E), chemical oxygen demand (SM 5220 C), viable mesophiles bacteria (SM 9215 B), total coliforms bacteria (SM 9221 B) and faecal coliforms *Escherichia coli* (SM 9221 E). Nitrates were measured using the spectrophotometric method by reduction with hydrazine sulphate. The nitrate/ammonium ratio (NO₃:NH₄) was calculated. These variables were measured in all sampling dates ($n = 36$).

The characterisation of in-stream habitat structure was performed using five equidistant transects perpendicular to the course and covering the entire reach (transects at 0, 25, 50, 75 and 100 m). The area of different hydrological microhabitats (percentage of pools, riffles and runs), the number of backwaters zones and woody debris were quantified in the four segments delimited by the five transects. On each transect, the wet channel width, the relative cover of different types of macrophytes (submerged, emergent and floating) and the substrate composition (bedrock, boulder (250–65 mm), gravel (65–2 mm) and sand (<2 mm) adapted from Barbour et al., 1999) were measured. The linear distances across each transect that were covered by each type of macrophytes or substrates were quantified and the proportion of the stream width accounted for each type was calculated (Fletcher et al., 2000). At four equidistant points on each transect, water and sediment depths were measured. For subsequent analyses, a mean of each variable in the five transects and the four inter-transect segments was calculated. Immediately upstream to the first transect, water velocity and discharge were quantified by slug additions of a solution (500g/4L) of conservative solute tracer (chloride as sodium chloride, Elozegi et al.,

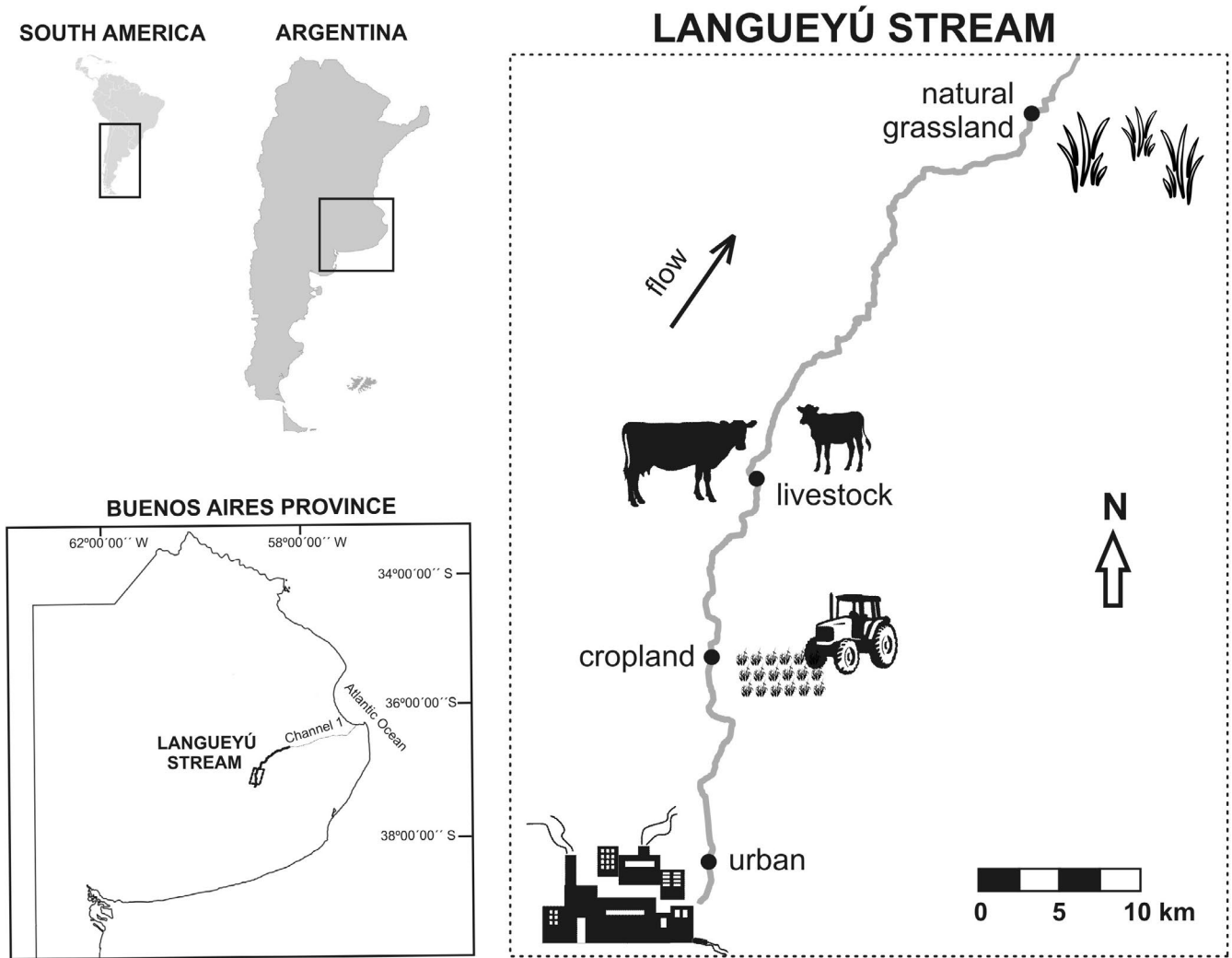


FIGURE 1 Location of sampling sites in the Langueyú stream, Pampa Plain, Buenos Aires, Argentina

2009), as an integral measure of the hydrological conditions of the entire reach. These measurements were performed in all sampling dates ($n = 36$).

The condition of riparian corridor was evaluated including aspects of soil surface cover, vegetation structure, the degree of bank alteration and in-stream canopy. The riparian width (average and minimum) was measured at both margins of each transect. In order to characterise the vegetation structure, the proportion of woody (trees and shrub), herbaceous and bare soil cover, as well as the amount of trees and shrubs were quantified at both margins and within the four inter-transect segments. The number of bank incisions by livestock and the proportion of bank stability were measured at both margins of the four segments delimited by transects. Bank stability was computed as the proportion of banks covered by macrophytes and roots regarding the total bank length (Rosso & Fernandez Cirelli, 2013). The in-stream irradiance of the light (expressed as percentage open area) within each segment delimited by two consecutive transects was visually quantified with sighting tubes (Johnson & Covich, 1997). For further analyses, an overall site

mean for each variable was calculated. These variables were measured at the beginning of each sample period ($n = 12$, four sites, three sampling periods).

2.4 | Fish assemblages

Fish sampling was performed using different nets and fishing devices: trammel nets (12 m), beach seine net (12 m) and artisanal traps. Trammel nets were constructed with an outer mesh size of 80 mm between knots and an inner mesh size of 15 mm between knots. Beach seine net was constructed with 12 m long wings (10 mm mesh) and a 2 m long bag (5 mm mesh). The artisanal traps were constructed with a 20 L plastic container with a cone-shaped entrance. Trammel nets and artisanal traps were left overnight and 2–4 seining averaging 50 m long were performed at each site during each sampling date. Field samplings were conducted under scientific fishing permit granted by the Ministry of Agroindustry of the Buenos Aires province (Resolution N° 409).

Fish sampling and handling protocols followed during the course of our surveys were evaluated and approved by the Ethics Comitee of the Facultad de Ciencias Exactas y Naturales, Universidad Nacional de Mar del Plata (RD-2018-126). Fish species were identified following Rosso (2006). Reference specimens of collected species were deposited at the fish collection of the Instituto de Investigaciones Marinas y Costeras (IIMyC, CONICET-UNMDP) in Mar del Plata, Argentina. Fish sampling was performed in all sampling dates ($n = 36$).

2.5 | Data analyses

2.5.1 | Environmental conditions

Environmental variables were unified to $n = 12$ by averaging values of their variables in each sampling site ($n = 4$) for each sampling period ($n = 3$). To test for redundancy, Spearman rank correlation coefficients were calculated among them. Those variables with high and significant ($\rho \geq .7$, p value $< .05$) correlation were removed from analysis matrices according to an ecological criterion. A principal component analysis (PCA) was conducted with environmental variables to explore spatial ordination of sampled sites. To perform this analysis, data were standardised and correlation matrix was used.

2.5.2 | Design and calculation of fish metrics

We analysed fish metrics from different multimetric indexes, including the Biotic Integrity Index (IBI, Karr, 1981) and its diverse adaptations. Those pertinent to the evaluation of Pampa streams were directly used, others required adaptations and new ones were designed. For this, collected species were classified according to their taxonomy, biogeographic origin (native or exotic), environmental tolerance, habitat preferences, feeding and parental cares using pertinent bibliography (Bistoni et al., 1999; Bozzetti & Schulz, 2004, Chalar et al., 2013; Fernández et al., 2012; Hued & Bistoni, 2005; Rosso, 2006; Teixeira de Mello, 2007). In turn, different ecological indexes were proposed as metrics: Shannon-Weaver diversity, Pielou evenness and Berger-Parker dominance. In addition, the species and total capture per unit of effort in abundance (CPUEa) and biomass (CPUEb) were calculated. The unit of effort was set at 3 seining of 50 metre long with beach seine net and overnight deployment of a trammel net and three artisanal traps. The total CPUE values (abundance and biomass) were scaled to their respective maximum values.

2.5.3 | Selection of fish metrics

Few studies conduct a rigorous screening of fish attributes from a larger list of candidate metrics and those using objective criteria and statistical procedures are scarce (McCormick et al., 2001). Here, a

metric selection analysis was performed using a formal protocol developed by McCormick et al. (2001) to identify the most powerful metrics to conduct the following analyses. This protocol allows each metric to be evaluated for its range of variation, variability, sensitivity to environmental gradient and redundancy with other metrics. Metrics were rejected if they presented between 0 and 2 species, number of families or trophic categories, or range of variation $< 10\%$; signal: noise ratio (variation between sites: variation between samples) < 2 ; absence of significant correlations (Spearman's correlation coefficient, p value $< .05$) with nonredundant environmental variables, and nonsignificant differences between the sampled sites (Kruskal-Wallis test, p value $< .05$); and they showed high correlation (Spearman correlation coefficient, $\rho \geq .75$) with other metrics. Selected fish metrics were unified to $n = 12$ by averaging values of their variables in each sampling site ($n = 4$) for each sampling period ($n = 3$).

2.5.4 | Response of biological and environmental matrices to the longitudinal continuum

To evaluate whether the observed changes in environmental conditions (water quality, in-stream habitat structure and riparian conditions) and fish assemblages respond to a longitudinal upstream-downstream continuum, RELATE routines (Clarke & Gorley, 2015) were used. This analysis determines the level of association between the serial model matrix (linear distance between sampled sites: U-C 12.2 km; C-L 13.7 km and L-N 32.5 km) and the Euclidean distance matrix of selected environmental variables or fish metrics. Environmental variables and fish metrics were standardised to zero mean and unit variance. To perform this analysis, Spearman correlation and 9999 random permutations were used.

2.5.5 | Spatial patterns of fish assemblages in relation to different environmental conditions

In order to visually interpret the fish assemblage's similarities between different environmental conditions, an analysis of nonmetric multidimensional scaling (nMDS; Clarke & Green, 1988) was performed. Euclidean distance measures of standardised fish metrics were used to quantify the similarity among sites. A stress coefficient less than 0.15 indicates a good representation of the data (Clarke, 1993). To test the differences in fish assemblages in different environmental conditions, a one-way ANOSIM analysis were performed using sites as factors. This analysis was conducted on the Euclidean similarity matrix generated with standardised fish metrics. Subsequently, the similarity-percentage analysis (SIMPER, Clarke, 1993) was used to determine the level of dissimilarity of the fish assemblages between pairs of sites and the relative contribution of each fish metric to discriminate pairs of fish assemblage under different environmental conditions.

2.5.6 | Relationships between fish assemblage and environmental conditions

The empirical relationships between fish and the environmental conditions were explored by means of Spearman correlation coefficients between PCA scores of the environmental scenario and selected fish metrics.

All statistical analyses were performed with software PRIMER.5 (Plymouth Routines In Multivariate Ecological Research) and PAST 4.01 (Paleontological Statistics Software Package for Education and Data Analysis).

3 | RESULTS

3.1 | Environmental conditions

Environmental conditions of sampled sites in the LS are summarised in Table 1. The first two axes of the PCA cumulatively explained 60% of the total variation in environmental conditions of sampling sites of this prairie stream (Figure 2). The first component loaded towards its positive end those sites whose waters showed high total suspended solids, total phosphorous and bacteriological loads (mesophiles and *E. coli*), together with high sediment depth (Table 1). The negative extreme of the first component characterised environments with more oxygenated waters, broader riparian corridors dominated by herbaceous vegetation and the absence of canopy cover. With a lesser magnitude than former variables, the first component also represents a gradient in the proportion of pools and substrate dominated by exposed bedrock (high at negative end). All samples from the urban site were grouped at the positive end of the first component, a position characterised by the worse environmental scenario. Samples from cropland, livestock and natural grassland were positioned towards the negative end and were further discriminated by the second component. This component grouped towards its positive end samples with nitrite-rich waters (cropland and livestock) whereas samples whose waters presented higher NO₃:NH₄ ratio loaded high at the negative end (natural grasslands). Samples with a riparian corridor with large surfaces of bare soil (livestock) loaded high towards the positive end of the second component. The second component also roughly separated samples with high bank stability and proportion of boulders in substrates (natural grasslands) from those where bank incisions by cattle (livestock) and high proportion of submerged macrophytes (cropland and livestock) were present.

Approximately, only half of the variation in water quality ($\rho = .401$, p value = .012), in-stream habitat structure ($\rho = .577$, p value < .001) and riparian conditions ($\rho = .482$, p value = .004) was explained by the longitudinal position of sites in the stream network. A subset of variables from the three explored matrices accounted for the longitudinal component of variation by means of either a monotonically downstream increase or decrease (Table 1). Longitudinal aspects of water quality were supported by a downstream increased in DO, NO₃:NH₄ and decrease in water conductivity and

bacteriological loads. In-stream habitat variables accounting for the longitudinal component of variation included proportion of pool habitats (increasing) and sediment depth (decreasing). Only mean riparian width (increasing) closely responded to the longitudinal gradient of variation in riparian conditions. Other water quality variables that showed an initial improvement from urban to cropland reach were disrupted in their continuum of recovery at the livestock site. The initial reduction in total phosphorous, total suspended solids and chemical oxygen demand was interrupted in livestock site showing an increase at (TP) or downstream (TSS, COD) from this site. In line with these tendencies, nitrite concentrations dropped downstream to the livestock site to the same value recorded in urban reach. Some habitat structure variables also showed a disrupted pattern by the unrestricted access of cattle to the stream. At this reach, the channel width roughly increased reaching extreme maximums (up to 11.5 m), and consequently, the lowest water velocity was recorded. Together with the cropland reach, both sites presented submerged macrophytes mats in their courses. At livestock site, the minimum riparian width was zero due to a nil development (zero metres) of riparian corridor found at some transects. In this reach, the lowest bank stability was also observed. In urban reach, an arboreal riparian corridor was reported.

3.2 | Fish assemblages

A total of 2134 specimens belonging to 16 species, 11 families and 6 orders were collected in different sampling sites of LS (Table 2). Characiformes and Siluriformes were the most represented orders with 3 families, while Characidae was the most represented family with 4 species. A single exotic species was found, the common carp (*Cyprinus carpio*). A large number of tolerant species, with omnivorous feeding and without parental care was collected (Table 3). Only three species showed a widespread distribution, being collected in three sampling sites along the study stream. The tolerant peppered catfish *Corydoras paleatus* was the unique species collected at the urban reach. This species was also found downstream to urban site at the cropland and livestock reaches. *Cheirodon interruptus* and *Rhamdia quelen* were collected from cropland to natural grassland site (Table 2). Seven species were exclusively collected at the more distant and less impacted reach.

Downstream from the urban reach, where fish assemblage was exclusively composed by *C. paleatus*, the cropland and livestock reaches presented a more equitable fish community (Table 4). At these reaches, fish assemblages were composed by higher number of families, high proportion of tolerant species and omnivorous individuals. In addition, fish assemblages were dominated by those individuals preferring shallow environments. In turn, the lees perturbed site exposed to natural grasslands showed a fish assemblage with the highest richness (12 species), diversity, total biomass, number of families and trophic guilds (omnivorous, piscivorous and detritivorous- algivorous). The exotic species, *C. carpio*, was exclusively collected at this site. A high proportion of intolerant species and

TABLE 1 Environmental conditions (water quality, in-stream habitat structure and riparian condition; standard deviation = SD) of sampling sites in the Langueyú stream (U, urban; C, cropland; L, livestock; N, natural grassland)

Variable	Code	U		C		L		N	
		Mean (SD)	Mean (SD)	Mean (SD)	Mean (SD)	Mean (SD)	Mean (SD)	PC1	PC2
Water quality									
Temperature (°C)	T	23.28 (3.16)	21.86 (3.91)	23.14 (3.2)	23.45 (2.81)	0.085	-0.281		
pH	pH	8.31 (0.3)	8.89 (0.59)	8.71 (0.46)	9.15 (0.46)				
Dissolved oxygen (mg/L)	DO	1.44 (1.8)	9.33 (4.3)	11 (3.25)	13.02 (5.13)	-0.907	-0.261		
Dissolved oxygen saturation (%)	DO%	16.58 (18.65)	110.84 (51.4)	134.3 (40.13)	158.37 (66.19)				
Water conductivity (µS/cm)	K	1167 (108.98)	1077.56 (93.81)	1049.67 (98.45)	912.44 (177.67)				
Salinity (ppt)	S	539.22 (31.92)	498.56 (22.78)	485.89 (23.3)	415.89 (70.5)				
Total dissolved solids (ppm)	TDS	826 (79.06)	764.67 (67.29)	744.33 (69.99)	641 (121.63)				
Total suspended Solids (mg/L)	TSS	74.79 (30.83)	11.92 (10.75)	9.54 (10.81)	44.81 (36.99)	0.762	-0.495		
Chloride (mg/L)	Cl	91.25 (16.84)	79.75 (12.27)	79.13 (14.14)	70.75 (16.43)				
Nitrates (mgN-NO3/L)	NO3	1.56 (0.91)	2.16 (0.73)	2.73 (1.53)	3.83 (1.39)				
Nitrites (mgN-NO2/L)	NO2	0.36 (0.46)	0.60 (0.29)	0.77 (0.36)	0.36 (0.23)	-0.453	0.556		
Ammonium (mgN-NH4/L)	NH4	15.45 (4.17)	12.43 (4.61)	8.87 (3.71)	0.48 (0.59)				
Nitrate/ammonium ratio	NO3:NH4	0.13 (0.12)	0.22 (0.15)	0.49 (0.64)	14.04 (9.01)	-0.329	-0.852		
Total phosphorus (mg/L)	TP	2.77 (0.88)	2 (0.64)	2.21 (0.59)	1.68 (0.44)	0.611	0.328		
Chemical oxygen demand	COD	183.34 (90.58)	83.01 (34.48)	59.28 (40.53)	65.27 (38.09)				
Mesophiles (UFC/ml)	Meso	170717.11 (155026.68)	56042.89 (46443.08)	7953.78 (6567.78)	7005.56 (13064.8)	0.802	0.156		
Total coliforms (colif/100ml)	Tcol	7055.56 (9616.02)	1892.22 (785.84)	1370.22 (1046.19)	1086.33 (863.33)				
<i>E. coli</i> (<i>E. coli</i> /100 ml)	Ecol	6766.67 (9788.9)	2032.67 (3496.66)	584.33 (816.47)	596.11 (810.33)	0.719	0.182		
In-stream habitat structure									
Pools (%)	pools	0.01 (0.02)	0.11 (0.19)	0.25 (0.22)	0.46 (0.13)	-0.626	-0.445		
Riffles (%)	riffles	0.24 (0.12)	0.15 (0.08)	0	0				
Runs (%)	runs	0.75 (0.14)	0.74 (0.11)	0.75 (0.22)	0.54 (0.13)				
Backwaters (n)	backwa	10.67 (2.52)	10 (6.56)	8.33 (1.53)	7.33 (6.51)	0.301	0.339		
Woody debris (n)	woodyde	1.33 (2.31)	0.67 (1.15)	1 (1)	1 (1)	0.018	-0.166		
Submerged macrophytes (%)	submacro	0	0.24 (0.15)	0.39 (0.17)	0.04 (0.05)	-0.564	0.671		
Emergent macrophytes (%)	emermacro	0	0	0	0.01 (0.01)	-0.22	-0.438		
Floating macrophytes (%)	floatmacro	0.01 (0.01)	0	0	0.01 (0.01)				
Substrate bedrock (%)	subbedrock	0.15 (0.08)	0.77 (0.08)	0.39 (0.23)	0.66 (0.1)	-0.599	-0.143		

(Continues)

TABLE 1 (Continued)

Variable	Code	U		C		L		N		
		Mean (SD)	Mean (SD)	Mean (SD)	Mean (SD)	Mean (SD)	Mean (SD)	PC1	PC2	
Substrate boulder (%)	suboul	0.03 (0.03)	0.01 (0.01)	0.07(0.03)	0.21 (0.13)	-0.326	-0.707			
Substrate gravel (%)	subgrav	0.7 (0.1)	0.04 (0.04)	0.29 (0.27)	0.04 (0.05)					
Substrate sand (%)	subsand	0.13 (0.09)	0.18 (0.1)	0.25 (0.11)	0.1 (0.05)					
Sediments depth average (cm)	sdep avg	4.94 (1.1)	2.11 (0.91)	1.12 (0.43)	1.01 (0.69)	0.928	0.104			
Water depth average (cm)	wdep avg	36.73 (3.13)	35.53 (5.02)	54.78 (16.07)	60.07 (4.87)					
Channel width average (m)	cwid avg	5.12 (0.15)	7.32 (0.97)	9.53 (2.03)	9.37 (0.43)					
Water discharge (L/s)	wdis	358.54 (111.92)	267.75 (78.48)	389.33 (193.99)	502.57 (218.66)					
Water velocity (m/s)	wveloc	0.76 (0.23)	0.42 (0.15)	0.34 (0.1)	0.37(0.1)					
Riparian condition										
Riparian width min (m)	riwid min	9.24 (0)	9.73 (0.64)	0	21.5 (0)	-0.878	-0.314			
Riparian width average (m)	riwid avg	11.2 (0.05)	17.05 (1.18)	23.48 (0)	26.28 (0)					
Woody cover (%)	wcov	0.22 (0.02)	0.02 (0.01)	0	0.07 (0.004)					
Herbaceous cover (%)	hcov	0.78 (0.02)	0.98 (0.01)	1 (0)	0.93 (0.004)	-0.924	0.323			
Bare soil (%)	bscov	0.11 (0.04)	0.18 (0.16)	0.21 (0.01)	0	0.043	0.907			
Trees (n)	tress	23.17 (2.75)	2.17 (0.29)	0	11 (2.6)					
Shrubs (n)	shrubs	0	0.33 (0.29)	0	3.33 (2.89)					
Bank incisions by livestock (n)	bincilive	0	0	2 (0)	0	-0.479	0.577			
Bank stability (%)	bstabil	0.84 (0.1)	0.74 (0.09)	0.57 (0.04)	0.84 (0.12)	0.349	-0.674			
Open area (%)	openarea	0.52 (0.07)	0.96 (0.01)	1 (0)	0.83 (0.1)	-0.853	0.391			

Note: Nonredundant variables are highlighted. Spearman correlation coefficients between the first two principal components and environmental variables were included.

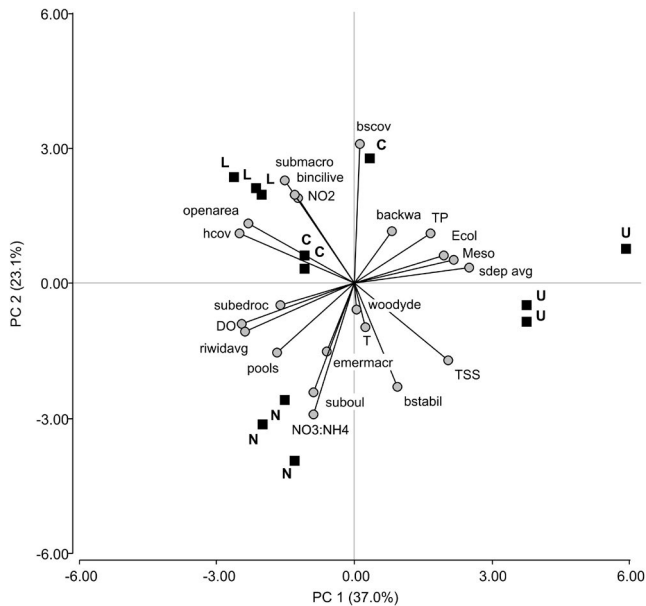


FIGURE 2 Biplot of the first two principal components analysis axes based on nonredundant environmental conditions (water quality, in-stream habitat and riparian corridor) of sampling sites of the Langueyú stream (U, urban; C, cropland; L, livestock; N, natural grassland). Variable codes as Table 1

their individuals, Characiformes and those species and individuals preferring deep environments were observed. Some intolerant taxa as *Australoheros facetus*, *Oligosarcus jenynsii* and *Psalidodon pampa* were exclusively collected at this reach.

Overall, a significant percentage of variation ($\rho = .64$, p value $=.001$) in fish assemblage structure was explained by the longitudinal position of sites in the basin.

A total of 57 metrics were calculated for fish assemblages of the LS (Table 4), but only 11 were selected for further analyses after the screening protocol (Figure 3). Six metrics were rejected due to their limited ranges of variation, 29 by presenting a low signal-to-noise ratio (variation between sites vs. variation between samples), only one due to its lack of sensitivity to environmental gradient and 10 by being redundant with other metrics (Appendix 1).

The two-dimensional scaling of fish assemblages under different environmental conditions showed a consistent pattern (stress coefficient = 0.082, Figure 4). There were evident particular areas in ordinations exclusive for each land use. In fact, the fish assemblage showed significant differences between sites ($R = .591$, p value $<.001$). Fish assemblage of the urban site was the most divergent, being these differences largest with the natural grassland site. In turn, the fish assemblages from cropland and livestock reaches partially overlapped. As observed in the nMDS, maximum dissimilarity of the fish assemblage structure between pairs of sites was found between urban and natural grassland sites (Table 5). Conversely, cropland and livestock sites showed the minimum dissimilarity. Fish metrics such as the number of trophic guilds and families, *C. paleatus* abundance and richness were the metrics that contributed most to explain the biological differences between the studies sites, accumulating more than 56% of the differences (Table 5).

3.3 | Relationships between fish assemblage and environmental conditions

The proportion of tolerant species to hypoxia, benthic species and *C. paleatus* abundance showed a high, positive and significant correlation with PC1 scores (Table 6), while richness, diversity, total biomass, proportion of intolerant and Characiformes species, and number of trophic guilds and families were negatively related. In turn, the proportion of omnivorous was highly, positively and significantly related with PC2 scores. These correlations showed that the detrimental environmental conditions characterised by total suspended solids, phosphorous-rich waters and high bacteriological loads and sediment depth, as found in urban reach, were positive and significant related to a fish assemblage absolutely dominated by *C. paleatus*. In turn, it also represents a tautological explanation for the high proportion of tolerant individuals to hypoxia and benthic species observed at this reach. Conversely, in an environment with well oxygenated waters, high proportion of pools, substrates dominated by bedrock and maximum values of riparian width with herbaceous cover and absence of canopy cover, as was found at the natural grassland less impacted site, a fish assemblage with high richness, diversity, biomass, number of trophic guilds and families and proportion of intolerant and Characiformes species was observed. Environmental conditions with nitrite-rich waters, submerged macrophytes mats and a riparian corridor with a high proportion of bare soil cover and bank incisions by livestock, those that characterised agricultural exposed reaches (cropland and livestock sites), were related to a higher proportion of omnivorous species in fish assemblages.

4 | DISCUSSION

The longitudinal patterns of fish assemblages in a Neotropical urbanised prairie stream exposed to contrasting land uses and associated changes in major environmental conditions were surveyed.

Roughly, one half of the variation in water quality, in-stream habitat and riparian matrices was explained by the longitudinal position of sampling sites in the basin. Nevertheless, the nature of this longitudinal behaviour of explored matrices was not as straightforward as should be expected in lotic ecosystems. Major attributes of lotic ecosystems change as water progress downstream. Particularly, water quality is drastically affected downstream since streams receive large amounts of energy and material throughout their interactions with nearby land uses (Allan, 2004). Similarly, longitudinal geomorphologic and hydrological changes in lotic ecosystems commonly determine that downstream reaches present wider and deeper environments with increasing proportion of pool habitats and low-sized substrates (Vannote et al., 1980). However, in the urbanised LS, longitudinal patterns in water quality aspects were characterised by a downstream improvement (water conductivity and bacteriological loads decreased whereas dissolved oxygen and $\text{NO}_3:\text{NH}_4$ increased) instead of deterioration. Similarly, opposite to what should

TABLE 2 Taxonomic classification of fish species collected in different sampling sites of Langueyú stream (U, urban; C, cropland; L, livestock; N, natural grassland)

Order	Family	Species	U	C	L	N	Collection code
Characiformes	Characidae	<i>Bryconamericus iheringii</i> (Boulenger, 1887)			X	X	UNMDP 4854
		<i>Cheirodon interruptus</i> (Jenyns, 1842)		X	X	X	UNMDP 4855
		<i>Oligosarcus jenynsii</i> (Günther, 1864)				X	UNMDP 4857
		<i>Psalidodon pampa</i> (Casciotta, Almirón & Azpelicueta, 2005)				X	UNMDP 4853
	Curimatidae	<i>Cyphocharax voga</i> (Hensel, 1870)				X	UNMDP 4856
	Erythrinidae	<i>Hoplias argentinensis</i> Rosso, González-Castro, Bogan, Cardoso, Mabrugaña, Delpiani, & Díaz de Astarloa, 2018				X	UNMDP 4965
Cichliformes	Cichlidae	<i>Australoheros facetus</i> (Jenyns, 1842)				X	UNMDP 4858
Cypriniformes	Cyprinidae	<i>Cyprinus carpio</i> Linnaeus, 1758				X	UNMDP 4859
Cyprinodontiformes	Anablepidae	<i>Jenynsia multidentata</i> (Jenyns, 1842)		X	X		UNMDP 4860
	Poeciliidae	<i>Cnesterodon decemmaculatus</i> (Jenyns, 1842)		X	X		UNMDP 4861
Siluriformes	Callichthyidae	<i>Corydoras paleatus</i> (Jenyns, 1842)	X	X	X		UNMDP 4862
	Heptapteridae	<i>Pimelodella laticeps</i> Eigenmann, 1917			X	X	UNMDP 4863
		<i>Rhamdia quelen</i> (Quoy & Gaimard, 1824)		X	X	X	UNMDP 4864
	Loricariidae	<i>Hypostomus commersoni</i> Valenciennes, 1836				X	UNMDP 4865
		<i>Loricariichthys anus</i> (Valenciennes, 1836)		X		X	UNMDP 4866
Synbranchiformes	Synbranchidae	<i>Synbranchus marmoratus</i> Bloch, 1795		X	X		UNMDP 4867

be expected, some aspects of in-stream habitat as sediment depth decreased downstream. The most parsimonious patterns resembling those conditions of nonimpacted lotic ecosystems were observed for some in-stream habitat and riparian characteristics. For instance, the proportion of pool habitat and riparian width increased downstream. Whereas some spatial aspects of water quality, in-stream habitat and riparian condition in the LS could be explained by longitudinal patterns (either natural or anthropic) others could not. Particularly, local aspects related with urbanisation and cattle intrusion seemed to be relevant for the onset of some detrimental conditions of water quality, in-stream habitat and riparian conditions.

Intimately aligned with all these particular patterns in water quality, in-stream habitat and riparian conditions, marked changes in fish assemblage attributes were observed. Our results suggest that fish were only partially responsive to the longitudinal position in the watershed. Fish assemblage attributes were also related to the discontinuities imposed by urbanisation and downstream patches of surrounding land uses. Indeed, selected fish metrics were intimately associated with different key environmental variables of water quality, in-stream habitat structure and riparian condition. Understanding patterns and key factors that influence species occurrence, richness and distribution are among the main goals of community ecology and conservation biology (Ricklefs, 1987). Fish assemblages of LS were composed by 16 species. All species previously reported for this stream (Bertora et al., 2018b) were collected plus the wolf fish *Hoplias argentinensis*.

About 64% of the total variation in fish assemblages was explained by the longitudinal upstream-downstream spatial

arrangement of sampled sites. Longitudinal variation in fish communities is a common phenomenon in streams (Matthews, 1998) as response to the structural changes and physical habitat variations along the longitudinal axis of fluvial ecosystems such as gradual increase in habitat complexity and the physical and chemical changes of water (Gorman & Karr, 1978; Vannote et al., 1980). Along these longitudinal gradients, an increase in species richness, diversity and biomass is expected (Alexandre et al., 2010; Furlan et al., 2013). The downstream increase in richness, diversity, biomass, number of families and trophic guilds observed in the LS mimics this natural longitudinal pattern observed in lotic ecosystems. Indeed, the number of trophic guilds and families together with species richness were the fish metrics that contributed most to differentiate fish assemblages between sites. The downstream increase in fish biomass could be explained by the increase in the availability of pool habitats, since in deeper areas, larger fish become more abundant (Welcomme, 1985). In fact, total fish biomass was positively related to relative pool area in Amazon headwater streams, indicating that space might be a limiting factor for total fish biomass (Bojsen & Barriga, 2002). Similarly, as riparian habitat expanded downstream, fish assemblage were characterised by a higher richness, diversity, number of trophic guilds and families and proportion of intolerant and Characiformes species. Overall, more preserved riparian corridors support fish assemblages with high abundance (Gowns et al., 2003), richness (Cetra & Petrere, 2007) and proportion of piscivorous specimens (Bertora et al., 2018a; Granitto et al., 2016). Most Characiformes are water column species, active pelagic swimmers that feed at the surface or mid-water level. More preserved environments allow species

TABLE 3 Classification of the fish species collected in the Languyú stream

Species	Origin	Tolerance to envir. degradation	Tolerance to hypoxia	Habitat	Feeding	Parental cares
<i>Bryconamericus iheringii</i>	Native	Intolerant ^{2,3,4,5}	No ⁸	Shallow environments ⁶	Omnivorous ^{5,6,7}	No ⁸
<i>Cheirodon interruptus</i>	Native	Intolerant ^{2,3,5}	No ⁸	Shallow environments ⁶	Omnivorous ^{5,6,7}	No ⁸
<i>Oligosarcus jenynsii</i>	Native	Intolerant ^{2,5}	No ⁸	Deep environments ^{2,6}	Piscivorous ^{5,6,7}	No ⁶
<i>Psalidodon pampa</i>	Native	Intolerant ⁵	No ⁸	Deep environments ⁶	Omnivorous ⁵	No ⁸
<i>Cyphocharax voga</i>	Native	Tolerant ^{1,5}	No ⁸	Benthic/deep environments ⁶	Detritivorous-algivoros ^{1,6,7}	No ⁸
<i>Hoplias argentinensis</i>	Native	Tolerant ⁶	Yes ⁶	Deep environments	Piscivorous ⁶	Yes ⁶
<i>Australoheros facetus</i>	Native	Intolerant ⁵	No ⁸	Deep environments ⁶	Omnivorous ^{5,6}	Yes ⁶
<i>Cyprinus carpio</i>	Exotic	Tolerant ⁷	Yes ⁸	Benthic ⁷ /deep environments ⁸	Omnivorous ⁶	No ⁶
<i>Jenynsia multidentata</i>	Native	Tolerant ^{2,4,6}	No ⁸	Shallow environments ²	Omnivorous ^{6,7}	Yes ⁶
<i>Cnesterodon decemmaculatus</i>	Native	Tolerant ^{1,2,3,4}	No ⁸	Shallow environments ⁸	Omnivorous ^{6,7}	Yes ⁶
<i>Corydoras paleatus</i>	Native	Tolerant ^{1,2,3,4,5,6}	Yes ^{1,4,5}	Benthic ⁵ / shallow environments ^{6,7}	Omnivorous ^{5,6,7}	No ⁸
<i>Pimelodella laticeps</i>	Native	Intolerant ²	No ⁸	Benthic ⁷ /deep environments ⁸	Omnivorous ^{6,7}	No ⁶
<i>Rhamdia quelen</i>	Native	Tolerant ^{3,4,5,6,7}	Yes ⁸	Benthic ^{5,6} / deep environments ⁸	Piscivorous ⁷	No ⁶
<i>Hypostomus commersoni</i>	Native	Tolerant ^{3,5}	Yes ⁸	Benthic ⁵ / deep environments ^{6,7}	Detritivorous-algivoros ^{6,7}	Yes ⁶
<i>Loricariichthys anus</i>	Native	Intolerant ⁸	Yes ⁸	Benthic ⁷ / deep environments ⁸	Detritivorous-algivoros ⁶	Yes ⁶
<i>Synbranchus marmoratus</i>	Native	Tolerant ^{5,6}	Yes ^{4,8}	Shallow environments ^{6,8}	Piscivorous ^{6,7}	Yes ⁶

Note: References: 1: Chalar et al. (2013), 2: Hued and Bistoni (2005), 3: Teixeira de Mello (2007), 4: Bistoni et al. (1999), 5: Bozzetti and Schulz (2004), 6: Rosso (2006), 7: Fernández et al. (2012), 8: Expert opinions.

Abbreviation: Envir., environmental.

of Characiformes to explore more efficiently, with positive effects on the structure of the population (Orsi et al., 2004). In fact, low occurrence and density of individuals of the order Characiformes evidenced effects of urban pollution on fish assemblages (Cunico et al., 2006). The diversity of these nektonic species is positively influenced by the heterogeneity of available habitat (Ferreira & Casatti, 2006; Tibúrcio et al., 2016). Combined effects of available pool habitats and preserved riparian conditions at the more distant downstream reach in the LS would also contributed to the exclusively occurrence of the piscivorous Characiformes *H. argentinensis*, the largest freshwater fish predator of the Pampa Plain.

Longitudinal variation in fish assemblages of the LS did not solely obey to natural gradients of environmental matrices. Urbanisation would sharply impact on fish assemblages of the LS through its effects on water quality but also on stream habitat. The anoxic and nutrient-bacteriological-rich conditions of the upstream urban reach, likely favoured the occurrence of highly tolerant species, with the ability of using atmospheric oxygen and thrive under eutrophic, contaminated conditions. Aspects of habitat structure as the highest depth of sediment column, that also presented a specular condition from that expected in natural environments, could also be of importance. Altogether, these conditions could help to explain the

presence of benthic tolerant organisms as food sources and benthophagous fish species preying upon them (Bertora et al., in press). In addition, local aspects of riparian conditions, as the presence of canopy cover might also contributed to regulate fish assemblage at this impacted reach. Shading by canopy at urban reach certainly further contributes to reduce the development of macrophytes that would favour a more balanced fish community by offering habitat and trophic opportunities for more fish species (Dala-Corte et al., 2016). All these environmental conditions would seem directive for the occurrence of the monotypic fish assemblage at the impacted urban reach exclusively composed by *C. paleatus*.

Corydoras paleatus is a typical species of fish communities in freshwaters ecosystems of the Pampa Plain (Rosso, 2006). Particularly, this species is highly tolerant to hypoxia because the use of the caudal portion of the intestine as an accessory air-breathing organ (Plaul et al., 2016). Among water quality variables, dissolved oxygen is perhaps the most influential aspect for fish assemblages in impacted ecosystems (Boët et al., 1999; Daga et al., 2012; Paracampo et al., 2020). Fish species tolerant to hypoxia became abundant in oxygen-poor waters by means of their special behaviours or physiologic adaptations (Ferreira & Casatti, 2006). For those species lacking such traits, anoxic conditions are lethal. Indeed, according to our results,

TABLE 4 Fish metrics (SD = standard deviation) of sampling sites in Langueyú stream (U, urban; C, cropland; L, livestock; N, natural grassland)

Metrics	Code	U		C		L		N	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD
Richness	rich	1	0	3.222	0.972	3.889	1.167	5.444	1.81
Diversity	diver	0	0	0.773	0.34	0.864	0.224	1.182	0.364
Equitability		0	0	0.668	0.197	0.68	0.203	0.744	0.182
Dominance		1	0	0.659	0.213	0.622	0.132	0.533	0.184
CPUEa/CPUEa max		0.048	0.062	0.197	0.19	0.321	0.337	0.164	0.217
CPUEb/CPUEb max	CPUEb	0.006	0.009	0.02	0.02	0.019	0.02	0.4	0.329
Prop. of exotic spp.		0	0	0	0	0	0	0.028	0.083
Prop. of exotic indiv.		0	0	0	0	0	0	0.005	0.016
Prop. of native spp.		1	0	1	0	1	0	0.972	0.083
Prop. of native indiv.		1	0	1	0	1	0	0.995	0.016
Prop. of intolerant spp.	intoler	0	0	0.328	0.165	0.339	0.044	0.701	0.167
Prop. of intolerant indiv.		0	0	0.29	0.28	0.176	0.175	0.8	0.143
Prop. of tolerant spp.		1	0	0.672	0.165	0.661	0.044	0.298	0.166
Prop. of tolerant indiv.		1	0	0.71	0.28	0.824	0.175	0.2	0.143
Prop. of tolerant to hypoxia spp.	tolhipox	1	0	0.437	0.103	0.317	0.058	0.3	0.156
Prop. of tolerant to hypoxia indiv.		1	0	0.408	0.29	0.37	0.283	0.214	0.142
Prop. of benthic spp.	benthic	1	0	0.415	0.084	0.313	0.091	0.393	0.134
Prop. of benthic indiv.		1	0	0.404	0.292	0.371	0.289	0.228	0.136
Prop. of deep env. spp.		0	0	0.078	0.118	0.037	0.111	0.765	0.138
Prop. of indiv. from deep env.		0	0	0.007	0.014	0.003	0.009	0.535	0.307
Prop. of shallow env. spp.		1	0	0.9	0.156	0.941	0.122	0.235	0.138
Prop. of indiv. from shallow env.		1	0	0.99	0.018	0.996	0.01	0.465	0.307
Number of trophic guilds	trophic	1	0	1.333	0.5	1.222	0.441	2.111	0.928
Prop. of omnivorous spp.	omniv	1	0	0.9	0.156	0.959	0.081	0.545	0.132
Prop. of omnivorous indiv.		1	0	0.99	0.018	0.998	0.004	0.673	0.169
Prop. of piscivorous spp.		0	0	0.072	0.148	0.041	0.081	0.22	0.14
Prop. of piscivorous indiv.		0	0	0.006	0.014	0.002	0.004	0.149	0.093
Prop. of detritivorous-algivoros spp.		0	0	0.028	0.083	0	0	0.235	0.161
Prop. of detritivorous-algivoros indiv.		0	0	0.005	0.014	0	0	0.13	0.177
Number of families	fami	1	0	3.222	0.833	3.444	0.726	3.111	0.601
Number of families/number of spp.		1	0	1	0	0.937	0.095	0.594	0.2
Prop. of Characiformes spp.	Characi	0	0	0.3	0.153	0.32	0.073	0.536	0.221
Prop. of Characiformes indiv.		0	0	0.286	0.277	0.173	0.177	0.678	0.292
Prop. of Siluriformes spp.		1	0	0.415	0.084	0.313	0.091	0.3	0.125
Prop. of Siluriformes indiv.		1	0	0.404	0.292	0.371	0.289	0.211	0.126
Prop. of Cyprinodontiformes spp.		0	0	0.263	0.2	0.344	0.078	0	0
Prop. of Cyprinodontiformes indiv.		0	0	0.307	0.313	0.454	0.308	0	0

(Continues)

TABLE 4 (Continued)

Metrics	Code	U		C		L		N	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD
Prop. of mojarra indiv.		0	0	0.286	0.277	0.173	0.177	0.559	0.281
CPUEa <i>A. facetus</i>		0	0	0	0	0	0	0.101	0.228
CPUEa <i>B. iheringii</i>		0	0	0	0	0.066	0.161	0.357	0.3
CPUEa <i>C. interruptus</i>		0	0	0.286	0.277	0.107	0.117	0.107	0.186
CPUEa <i>C. decemmaculatus</i>		0	0	0.304	0.307	0.407	0.279	0	0
CPUEa <i>C. paleatus</i>	C.palea	1	0	0.397	0.291	0.368	0.284	0	0
CPUEa <i>C. voga</i>		0	0	0	0	0	0	0.01	0.015
CPUEa <i>C. carpio</i>		0	0	0	0	0	0	0.005	0.016
CPUEa <i>H. commersoni</i>		0	0	0	0	0	0	0.143	0.14
CPUEa <i>J. multidentata</i>		0	0	0.003	0.009	0.047	0.089	0	0
CPUEa <i>L. anus</i>		0	0	0.005	0.014	0	0	0.024	0.051
CPUEa <i>O. jenynsii</i>		0	0	0	0	0	0	0.108	0.082
CPUEa <i>P. laticeps</i>		0	0	0	0	0.002	0.007	0.008	0.016
CPUEa <i>P. pampa</i>		0	0	0	0	0	0	0.094	0.148
CPUEa <i>R. quelen</i>		0	0	0.003	0.005	0.001	0.002	0.038	0.099
CPUEa <i>S. marmoratus</i>		0	0	0.003	0.009	0.001	0.004	0	0
CPUEa <i>H. argentinensis</i>		0	0	0	0	0	0	0.004	0.011
Prop. of parental care spp.		0	0	0.313	0.215	0.367	0.058	0.312	0.167
Prop. of parental care indiv.		0	0	0.314	0.311	0.455	0.31	0.265	0.226
Prop. of indiv. with DELT		0	0	0.029	0.087	0.007	0.021	0.033	0.051

Note: Selected metrics are highlighted.

Abbreviations: CPUEa, capture per unit of effort in abundance, CPUEb, capture per unit of effort in biomass, indiv., individuals, spp., species, env., environments, DELT, deformities, eroded fins, lesions and tumours.

a gradient in dissolved oxygen concentration would be highly relevant to partially explain the longitudinal variation of the fish assemblages in the LS. Biological traits other than tolerance to hypoxia could be relevant for the success of *C. paleatus* at urban reaches. This peppered catfish is a tolerant, nonvisual, benthic feeder species (Bistoni et al., 1999; Bozzetti and Schulz, 2004; Chalar et al., 2013). The nonvisual and benthic behaviour seem to be positive attributes for thrive in the urban reach with high total suspended solids and a well-developed column of sediments. All these biological traits could help to explain the occurrence of *C. paleatus* as the only species that could colonise the hostile urban conditions of the LS. In fact, the abundance of this species was one of the most important metrics characterising the differences between fish assemblages from contrasting environmental conditions. Similarly, this species was also dominant in regional urban ecosystems, the Suquia River near Córdoba city (Hued & Bistoni, 2005) and Pampa streams from the populated coastal drainages of the Río de la Plata (Paracampo et al., 2020). The relative abundance of tolerant species increases with urbanisation (Paul & Meyer, 2001). Particularly, the abundance of tolerant species are positively related with chemical variables associated with increasing turbidity, nutrients and general human activity, and negatively correlated with various indices of channel, riparian and watershed quality (McCormick et al., 2001). For that reason, tolerant species increase in number with chemical and physical

degradation, are the last to leave with degradation and the first to re-appear with recovering (Teixeira Pinto & Araújo, 2007).

Accordingly with the uniqueness of *C. paleatus*, fish assemblage at urban conditions presented minimum values of richness, diversity, biomass, number of families and trophic guilds. Indeed, the effects of urbanisation on fish are generally characterised by a decline in richness, diversity, total abundance, biomass and abundance of intolerant species, an increase in abundance of introduced, tolerant, dominant species and, ultimately, influence the integrity of fish assemblages (Paul & Meyer, 2001). These changes are strongly associated with the drastic impacts that cities and industries impose on different aspects of the ecological integrity of these vulnerable fluvial ecosystems. This is a global pattern (Boët et al., 1999; Gagny et al., 2000; Jackson et al., 2001) also observed in Neotropical urban streams (Alexandre et al., 2010; Cunico et al., 2006; Daga et al., 2012) and, recently, in Pampa streams (Paracampo et al., 2020). Worldwide, the total urban land use occupies low extensions, but still have disproportionately large effects on the biota (Paul & Meyer, 2001) and particularly in the fish community (Fausch et al., 1990). For instance, a low increase in the urbanisation level has changed the composition and structure of Neotropical stream fish assemblages mainly due to changes in conductivity and concentrations of dissolved oxygen and total phosphorus (Daga et al., 2012). Particularly, a decrease in diversity, abundance, specialists and nektonic species

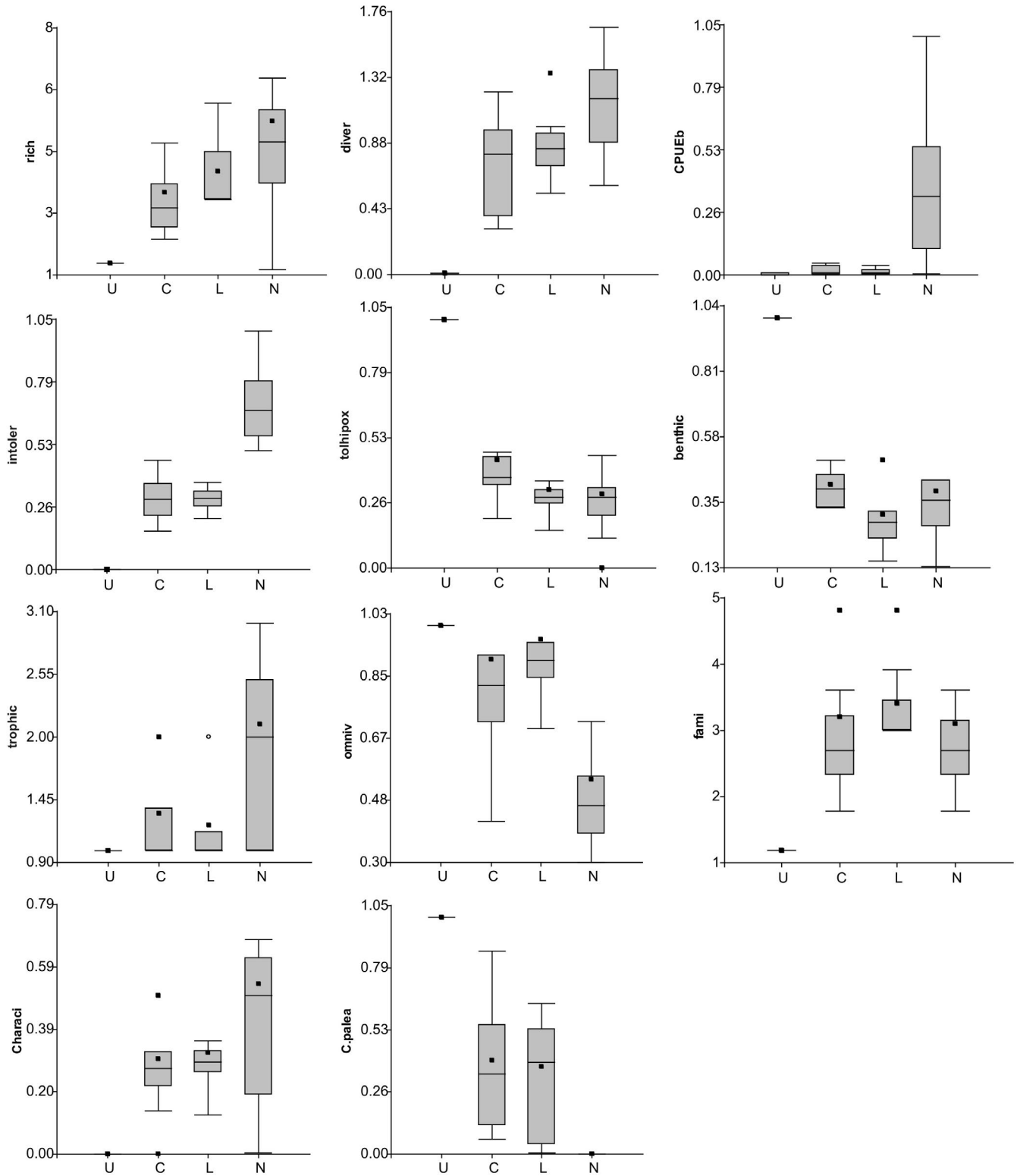


FIGURE 3 Box-whisker plots of selected fish metrics in different sampling sites of the Langueyú stream (U, urban; C, cropland; L, livestock; N, natural grassland). Metrics codes as Table 4

with an increase of generalist and tolerant species are observed at the downstream most impacted reaches receiving urban effluents (Da Silveira et al., 2018). Interestingly, our survey showed that urbanisation at upstream reaches could reverse this longitudinal

pattern, forcing intolerant species to thrive at the downstream less impacted reaches and favouring tolerant, air breathing species to dominate the upstream impacted reach. The presence of intolerant species is an important metric since they are the first to decline with

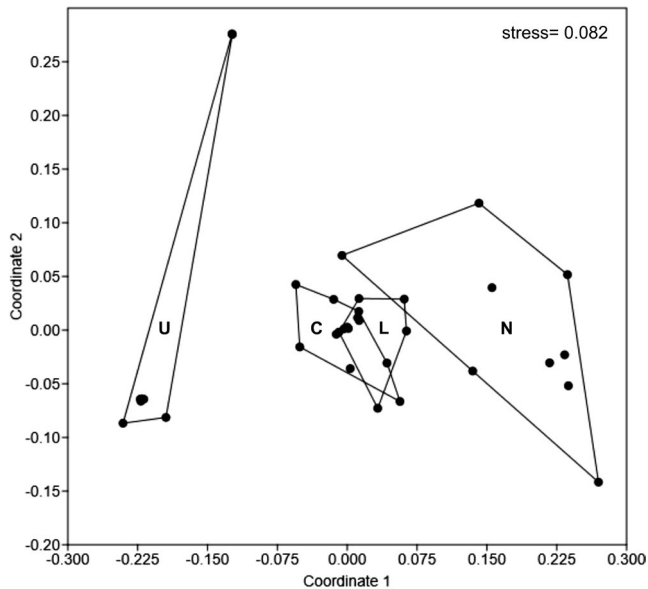


FIGURE 4 Nonmetric multidimensional scaling (nMDS) of fish assemblages under different environmental conditions of Langueyú stream (U, urban; C, cropland; L, livestock; N, natural grassland). Some sampling points are superimposed on others

TABLE 6 Spearman correlation coefficients between fish metrics and PC1 and PC2 scores

Fish metrics	Environmental conditions			
	PC1 scores		PC2 scores	
	ρ	p value	ρ	p value
rich	-0.68	0.015	-0.355	0.258
diver	-0.661	0.019	-0.392	0.208
CPUEb	-0.579	0.049	-0.539	0.07
intoler	-0.603	0.038	-0.204	0.524
tolhipox	0.817	0.001	0.039	0.905
benthic	0.798	0.002	-0.285	0.369
trophic	-0.572	0.052	-0.237	0.458
omniv	-0.107	0.74	0.727	0.007
fami	-0.623	0.03	0.110	0.735
Characi	-0.575	0.051	-0.155	0.631
C. palea	0.638	0.026	0.331	0.293

Note: Bold numbers highlight statistically significant relationships (p value ≤ 0.05). Metrics codes as Table 4.

TABLE 5 Average dissimilarity and percentage contribution of each item fish metric to discriminate the fish assemblage between pairs of sites (U, urban; C, cropland; L, livestock; N, natural grassland)

Fish metrics	U-C	U-L	U-N	C-L	C-N	L-N
rich	6.44	8.99	10.89	11.34	10.21	8.36
diver	10.51	10.47	10.76	11.43	7.35	5.41
CPUEb	0.05	0.04	8.74	0.23	22.43	24.34
intoler	6.67	5.16	12.51	6.52	12.29	11.27
tolhipox	12.92	14.78	8.59	5.71	2.67	1.44
benthic	14.31	16.07	7.31	5.48	1.32	1.89
trophic	4.77	3.24	8.67	13.45	13.56	16.01
omniv	10.12	8.99	5.55	7.36	9.97	12.82
fami	15.97	16.04	6.61	14.15	3.06	3.15
Characi	7.87	6.67	11.15	8.97	10.84	9.29
C.palea	10.37	9.55	9.23	15.35	6.29	6.02
Average dissimilarity	25.49	28.98	53.88	5.19	20.15	18.52

Note: Metrics codes as Table 4. Bold metrics were those that contributed most to discriminate between sites.

increasing impact by human activities. Declines may be due to water quality degradation, habitat degradation, or a combination of both (Karr, 1981).

In addition to the observed changes in fish assemblages imposed by both natural and anthropic specular continuums, some particular attributes of fish assemblages would be associated with local aspects of environmental conditions. Particularly, cropland and livestock sites displayed fish assemblages characterised by a high proportion of omnivorous species. Consequently, these sites showed the minimum dissimilarity in their fish assemblages. These reaches also displayed the presence of large mats of submerged

macrophytes. The combination of full sunlight incidence, low water velocity and total suspended solids may help to explain the presence of macrophytes at these reaches as well as their absence at urban (low sunlight incidence, high water velocity and high suspended solids) and grassland (good sunlight incidence and low water velocity but high suspended solids) sites. Macrophyte mats in Pampa Plain streams are favoured by the autochthonous lack of canopy cover, low current velocities and high nutrient concentrations (Rodrigues Capítulo et al., 2010). In Neotropical streams exposed to agricultural land use (cropland and livestock), Dala-Corte et al. (2016) reported richer fish assemblages associated with macrophyte cover where

macrophyte-associated nektonic fish replaced benthic and lithophilic species. Submerged vegetation plays an important structuring role in Pampa Plain streams by regulating and modifying the physicochemical and biological characteristics of these ecosystems (Giorgi et al., 2005), and particularly provides refuge and reproduction areas for fish as well as food availability. Furthermore, specimens of the order Cyprinodontiformes were exclusively collected at sites with macrophytes, where *J. lineata* and *C. decemmaculatus* represented almost 50% of the total fish abundance in the livestock reach. These small fish are favoured by the refuge provided by macrophytes mats (Mazzeo et al., 2003) and, particularly, by shallow environments produced by channel widening generated by cattle intrusion on the stream (O'Callaghan et al., 2018).

Overall, fish assemblages in the LS were exposed to a complex longitudinal scenario where the natural variation expected in nonimpacted ecosystems is overlapped with another longitudinal gradient imposed by the continuum of recovery in water quality downstream from the urban effluents. On the other hand, local disruptions imposed by surrounding land uses, particularly cattle intrusion, represented a local, nonlongitudinal source of environmental changes. Fish assemblages showed forceful empirical relationships to the observed environmental conditions imposed by multiple natural and anthropic factors in this urbanised prairie stream. Particularly, three main patterns of fish responses were observed. Assemblage-level attributes as diversity, richness, fish biomass, number of families and trophic guilds, and proportion of Characiformes and intolerant species would respond to a natural gradient in habitat and riparian conditions expected in nonimpacted lotic ecosystems. Conversely, a specular continuum in water quality and habitat structure would largely influence the distribution of fish species regarding their environmental tolerance, including air breathing adaptation, benthic habits and, particularly, *C. paleatus* abundance. Finally, local habitat aspects of cropland and livestock reaches seemed relevant for the trophic structure (omnivorous species) and abundance of Cyprinodontiformes of fish assemblages.

Our results represent a case study in a single stream but still produce a good piece of evidence about empirical patterns in Neotropical fish assemblages in relation to environmental variation in an urbanised prairie stream exposed to contrasting land uses. In this respect, the use of metrics accounting for the diversity, biomass, habitat preferences, species' tolerance, trophic composition, taxonomy and specific abundance allowed a holistic view of fish assemblage organisation under contrasting environmental conditions. These results may be relevant for conservation and management purposes. Fishes have been historically proposed to assess the ecological integrity of fluvial ecosystems. However, most notorious patterns of changes in fish assemblages in relation to environmental conditions in studies assessing the integrity of aquatic ecosystems are supported by speciose fish fauna where sensitive species are excluded from impacted reaches (Karr, 1981; McCormick et al., 2001; Bozzetti & Schulz, 2004; Chalar et al., 2013). Our results showed that a pauperised Neotropical fish fauna compose of many tolerant species would still respond to habitat, riparian and water

quality degradation imposed by urbanisation and surrounding land uses. These results reinforce the arguments for the use of fish as a biological criterion in the assessment of the ecological integrity of streams at temperate Neotropical ecosystems, where sensitive, easily responsive species are not present.

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AUTHORS CONTRIBUTIONS

All authors contributed to the study conception, design and performed data collection. AB and JJR performed the data analysis and wrote the first draft of the manuscript. All authors commented on previous versions of the manuscript and approved the final manuscript.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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APPENDIX 1

Rejected metrics along the four steps of selection protocol proposed by McCormick et al. (2001). Fish metrics codes as Table 4

Range	Variability	Sensitivity	Redundancy
Prop. of exotic indiv.	Dominance	CPUEa <i>R. quelen</i>	Equitability
CPUEa <i>C. voga</i>	CPUEa/CPUEa max		Prop. of intolerant indiv.
CPUEa <i>C. carpio</i>	Prop. of exotic spp.		Prop. of tolerant spp.
CPUEa <i>P. laticeps</i>	Prop. of native spp.		Prop. of deep env. spp.
CPUEa <i>S. marmoratus</i>	Prop. of native indiv.		Prop. of indiv. from deep env.
CPUEa <i>H. argentinensis</i>	Prop. of tolerant indiv.		Prop. of shallow env. spp.
	Prop. of tolerant to hipoxia indiv.		Prop. of omnivorous indiv.
	Prop. of benthic indiv.		Prop. of piscivorous indiv.
	Prop. of indiv. from shallow env.		n° of families/n° spp.
	Prop. of piscivorous spp.		Prop. of siluriformes spp.
	Prop. of detritivorous-algivorous spp.		
	Prop. of detritivorous-algivorous indiv.		
	Prop. of Characiformes indiv.		
	Prop. of Siluriformes indiv.		
	Prop. of Cyprinodontiformes spp.		
	Prop. of Cyprinodontiformes indiv.		
	Prop. of mojarra indiv.		
	CPUEa <i>A. facetus</i>		
	CPUEa <i>B. iheringii</i>		
	CPUEa <i>C. interruptus</i>		
	CPUEa <i>C. decemmaculatus</i>		
	CPUEa <i>H. commersoni</i>		
	CPUEa <i>J. multidentata</i>		
	CPUEa <i>L. anus</i>		
	CPUEa <i>O. jenynsii</i>		
	CPUEa <i>P. pampa</i>		
	Prop. of parental care spp.		
	Prop. of parental care indiv.		
	Prop. of indiv. with DELT		