Knowl. Manag. Aquat. Ecosyst. 2022, 423, 15
© A. Bertora *et al.*, Published by EDP Sciences 2022
https://doi.org/10.1051/kmae/2022015

www.kmae-journal.org

Knowledge & Management of Aquatic Ecosystems

Journal fully supported by Office français de la biodiversité

RESEARCH PAPER OPEN ACCESS

Combined effects of urbanization and longitudinal disruptions in riparian and in-stream habitat on water quality of a prairie stream

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Received: 4 November 2021 / Accepted: 20 May 2022

Abstract – Local habitat and riparian modifications imposed by surrounding land use drastically impact the water quality of streams. However, whether these effects could still be discernible when the watercourse also receives urbanization effluents has not been fully explored. We evaluated the water quality of a Neotropical prairie stream exposed to urbanization and explored the role of downstream patches of different surrounding land uses (cropland and livestock) in further regulating water quality. Forty-two variables of water quality, habitat structure and riparian condition were measured at four reaches of the Langueyú stream. Significant differences in water quality were observed. Water conductivity, dissolved oxygen, salinity, dissolved solids, chloride, inorganic nitrogen and bacteriological loads displayed a continuum of recovery from the urban reach. Indeed, almost 24 percent of the total variation in water quality was explained by the longitudinal arrangement of sites. Alternatively, pH, phosphorous, suspended solids and chemical oxygen demand showed a disruption in this continuum of recovery and were highly related with local aspects of habitat structure and riparian conditions imposed by cropland and livestock. Key aspects of effluent treatment, riparian integrity and in-stream habitat must be addressed within a comprehensive social context in order to design sustainable management of fluvial urbanised ecosystems.

Keywords: Urban continuum framework / land use / cropland / livestock / multimetric indexes

1 Introduction

Over the next 30 years, the global population is projected to increase from 7.7 billion to as many as 9.7 billion people, with rapid growth and urbanization in less developed areas of the world (United Nations, 2019). Following population expansion, the demand of land cover for crop and animal husbandry is expected to increase considerably (Tilman *et al.*, 2011; Fragkias *et al.*, 2013) together with the incentive for intensification in agricultural practices (Godfray *et al.*, 2010). Modified landscapes to meet crop and animal production represent a major threat to the ecological integrity of fluvial ecosystems (Strayer *et al.*, 2003). In recent years, a rapid decline in the availability of usable freshwater in terms of water quality and quantity due to unsustainable land use practices was observed (Giri and Qiu, 2016). In streams, this scenario is worsening by the indirect degradation of water quality caused by urbanization (Paul and Meyer, 2001; Walsh *et al.*, 2005), agriculture (Moss, 2008; Arocena *et al.*, 2018) and livestock (Platts, 1979; O'Callaghan *et al.*, 2018) through changes in the fluvial habitat structure, riparian condition, macrophytes coverage, channel morphology and substrate composition. Accordingly, understanding how urban and agricultural practices affect water quality and how key ecosystem structures, such as riparian vegetation and stream habitats, may further modulate the impact of these activities is essential to ensure human welfare while reducing its environmental costs (Ramião *et al.*, 2020).

Urban land use is commonly a low percentage of total catchment areas, yet it exerts a disproportionately large influence in streams both proximately and over distance (Paul and Meyer, 2001). Urban point source effluents largely increase water nutrients (Meyer *et al.*, 2005; Walsh *et al.*, 2005), metals (McGrane, 2016) and solids (Walsh *et al.*, 2005). In addition,

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accumulation of organic matter promotes microbial growth, leading to oxygen depletion (Sirota *et al.*, 2013).

Instead, agricultural land use degrades water quality of streams by increasing non-point inputs of nutrients (Omernik *et al.*, 1981; Cunha *et al.*, 2019), sediments (Omernik *et al.*, 1981; Moss, 2008), organic matter (Cooper, 1993; Moss, 2008) and pesticides (Cooper, 1993). Streams in highly agricultural landscapes with impacted riparian vegetation tend to have poor habitat quality, low bank stability (Richards *et al.*, 1996; Wang *et al.*, 1997) and large depositions of sediments on the streambed which entails siltation (Molina *et al.*, 2017).

Livestock congregate along rivers and streams for shade, more succulent vegetation and drinking water, where animals deposit fresh fecal matter near shade and water (O'Callaghan *et al.*, 2018). In consequence, water quality is degraded through large increases of nutrients, total suspended solids, turbidity and bacteria (Vidon *et al.*, 2008; Horak *et al.*, 2020). The sloughing-off and collapse of banks, deterioration of riparian vegetation, sediment accumulation, widening and shallowing of channels, and alteration in substrate composition are among the main habitat changes imposed by livestock (Platts, 1979; O'Callaghan *et al.*, 2018).

The influence of land uses on fluvial ecosystems is given through multiple pathways and mechanisms operating at different spatial scales (Allan *et al.*, 1997). However, several studies demonstrated that riparian zone and land use close to streams are more important to water quality than the landscape pattern of the entire catchment (Dodds and Oakes, 2008). As fluvial systems typically flow over different land uses along their continuum, spatial studies focusing on water quality along the longitudinal axis become a great challenge and, in turn, would lead to a more accurate understanding of this complex of ecosystems (Ramião *et al.*, 2020). More integrated assessments of river systems are needed to propose efficient and sustainable river management guidelines (Jungwirth *et al.*, 2000).

Urban reaches of lotic ecosystems show a marked deterioration of water quality which depending of some key aspects as nature, volume and frequency of effluents (Ellis and Hvitved-Jacobsen, 1996) and the presence of tributaries (Kelly *et al.*, 2010) usually recovers downstream. The urban watershed continuum framework recognizes a continuum of downstream fluxes and transformations of carbon, contaminants, energy and nutrients (Kaushal and Belt, 2012). In addition, it proposes that there are longitudinal downstream pulses in material and energy exports that are amplified by interactive land-use and hydrologic variability. Along with this interaction with surrounding land uses, lotic ecosystems also receive large imports of material and energy (Allan, 2004). Therefore, downstream patches of contrasting land uses could be considered as sources of disruption in the continuum of the functioning of lotic ecosystems (Poole, 2002). Particularly, the role of longitudinal downstream changes imposed by surrounding land uses on the dynamics of water quality variables along an urban continuum framework could be explored to understand the response of fluvial systems in scenarios of multiple antrophic impacts.

In this context, the main objective of this work was to evaluate the water quality of a Neotropical prairie stream exposed to urbanization and to explore the role of downstream patches of different surrounding land uses in further regulating water quality. Particularly, we evaluated whether different aspects of water quality present a continuum of recovery imposed by the urbanization at upstream reaches or, alternatively, downstream changes in land uses and associated riparian and in-stream habitat conditions disrupt them being able to modulate some aspects of water quality. We predict that urban footprint on stream water quality will erase the role of riparian land use and local stream habitat in regulating water quality. With this research, we aim to contribute to understanding of the dynamics of water quality in prairie streams in the context of increasing urbanization and changing land uses. Considering the role of longitudinal variation in land uses on water quality, more comprehensive management (including restoration actions) and conservation guidelines for these threatened ecosystems could be proposed.

2 Materials and methods

2.1 Study area

The Pampa Plain covers a large region in central Argentina with suitable lands for crop and livestock production (Viglizzo *et al.*, 2001) where a considerable agricultural expansion at the expense of natural grasslands was observed (Viglizzo *et al.*, 2011). This region also hosts the largest amount of inhabitants of Argentina, which entails a continuous expansion of cities and industries. Prairie streams in the Pampa Plain are eutrophic to hypereutrophic systems, with low velocity, herbaceous riparian vegetation, bottom-dominated by fine substrates and autochthonous macrophytes as main primary producers (Feijoó *et al.*, 1999). The high phosphorous levels in these ecosystems are due to phosphorus-rich volcanic material transported from Los Andes range and deposited in the region during the Quaternary (Morrás, 1999). The water quality of these ecosystems has shown to be intimately aligned with the prevailing land use in the surrounding landscape (Vilches *et al.*, 2011; Rosso and Fernández Cirelli, 2013; Amuchástegui *et al.*, 2015). Other components of these ecosystems, such as the riparian vegetation, macrophytes and habitat structure are also influenced by cropland (Cortelezzi *et al.*, 2013; Rosso and Fernández Cirelli, 2013; Arocena *et al.*, 2018) and livestock uses (Cortelezzi *et al.*, 2013; Rosso and Fernández Cirelli, 2013; Giorgi *et al.*, 2014). The consequences of urban and industrial developments on the water quality of Pampa Plain streams have been recently investigated (Cortelezzi *et al.*, 2013; Cochero *et al.*, 2016).

The Langueyú stream is located in the Pampa Plain, in southeast of Buenos Aires province, Argentina (Fig. 1). The Langueyú stream belongs to the homonymous basin which has an area of approximately 600 km^2. At the Tandilia hills, three first-order steeped tributaries flows to converge in the Langueyú stream. It flows in a southwest to northeast direction, runs through Tandil, Rauch and Ayacucho cities and flows into the Atlantic Ocean by means of a man-made channel (Channel 1). Langueyú stream has almost 131 km long and an average annual flow of 2.18 m^3/s (Hernandez *et al.*, 2002). Tandil city has a sub-damp to damp climate (Thornthwaite and Mather, 1957) with annual average rainfall of 838 mm and average temperature of 13.8 °C. The Langueyú stream basin is characterized by high agricultural productivity in the rural area (cropland and livestock) and industrial

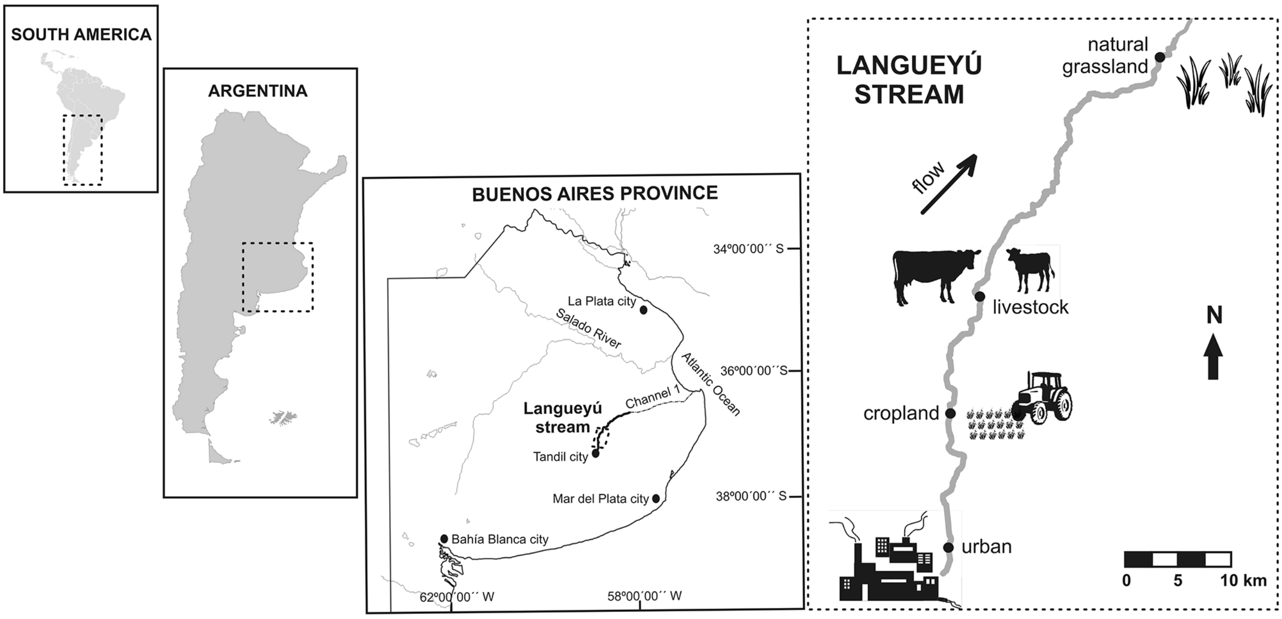

Fig. 1. Location of sampling sites in the Langueyú stream, Pampa Plain, Buenos Aires, Argentina.

developments in urban and surroundings areas (Ruiz de Galarreta *et al.*, 2010). However, in the last decades, an agricultural expansion and changes in land uses were observed. Particularly, agricultural areas increased by 61.3%, areas for grazing showed a reduction of approximately 50% and urban areas increased by 169.8% in a period of 30 years (Somoza *et al.*, 2021). In urban and peri-urban sectors, the stream receives stormwater, wastewater and industrial (mainly metallurgical and food industries) effluents (Cortelezzi *et al.*, 2019). Many of these effluents have deficient treatment (OPDS, 2009). Vacuum truck illegal discharges are also reported (Banda Noriega and Díaz, 2010; Ruiz de Galarreta *et al.*, 2013). In recent years, Tandil city has undergone significant demographic growth, above the Argentine average, and currently has around 150,000 inhabitants (123,343 inhabitants in the last survey from INDEC, 2010).

2.2 Sampling

2.2.1 Sampling sites

The choice of sites followed the need for by evaluating of the role of downstream reaches from urbanization with contrasting land uses in their immediate (*i.e.* lateral, upstream and downstream areas; at least 2-km surrounding land) landscapes and the respective in-stream and riparian structure on water quality. Four reaches of 100 meters long were selected in the Langueyú stream (Fig. 1). Sampling reaches were located at the main stem of Langueyú stream in lowlands (less than 2 cm/m), in order to avoid biases introduced by headwater steeped tributaries. In this area, the stream lacks tributaries. The uppermost reach represents a strongly impacted condition by Tandil city and its industrial development through the discharge of stormwater, wastewater and industrial effluents (urban site = U, 37°16′24″ S, 59°07′35″ W). Downstream, two reaches account for the influence of cropland (cropland site = C, 37°11′15″ S, 59°08′ W) and livestock (livestock site = L, 37°05′47″ S, 59°06′29″ W) activities. Annual crops implanted up to few meters of the stream reach and unrestricted cattle access characterize these sites. 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). The linear distances between sampled sites were as follows: U-C 12.2 km; C-L 13.7 km and L-N 32.5 km.

At each of the 4 sites, three sampling visits were performed during each of the three consecutive spring-summer periods (2016/17, 2017/18, 2018/19) surveyed. Therefore, 9 samples from each site were collected, reaching a total of 36 samples for the entire stream. Sampling scheme was concentrated during spring-summer period in order to avoid temporal biases introduced by seasonal variations typical of temperate latitudes. Field samplings were carried out avoiding recent (96 hs.) rainfall events.

2.2.2 Water quality (WQ)

A total of 16 physical, chemical and bacteriological water variables were measured.

Temperature, pH, water conductivity, salinity and total dissolved solids were measured *in situ* using a multiparametric probe (Oakton PCSTestr 35) and dissolved oxygen and percentage oxygen saturation with an oxymeter (HACH sension 156). Water samples were taken at mid-depth and midstream and were kept cold until reaching the laboratory for processing. Ammonium (SM 4500-NH3 C), total phosphorous (SM 4500-P E), chloride (SM 4500-Cl B), total suspended solids (SM 2540 D), chemical oxygen demand (SM 5220 C), viable mesophiles bacteria (SM 9215 B), total coliforms bacteria (SM 9221 B) and fecal coliforms *Escherichia coli* (SM 9221 E) were quantified according to standard methods (APHA-AWWA-WEF, 2017). Nitrate was measured through the spectrophotometric method by reduction with hydrazine sulfate. In turn, the nitrate/ammonium ratio (NO_3:NH_4) and total inorganic nitrogen (NO_3+NH_4) were calculated. These measurements were performed on all sampling dates (n = 36).

2.2.3 In-stream habitat structure (HS)

The characterization of habitat structure was performed by assessing the condition of 17 variables measured along five equidistant transects perpendicular to the stream and covering the entire reach under study (transects at 0, 25, 50, 75 and 100 m). On each transect, the wet channel width, the relative cover of different forms of macrophytes growth (floating, submersed and emergent) 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 measured and the proportion of the stream width accounted for each type was calculated (Fletcher *et al.*, 2000). Water and sediment depths were measured at four equidistant points on each transect. In the four segments delimited by the five transects, the area covered (as percentage) by different hydrological microhabitats (pools, riffles and runs), the number of backwaters zones and woody debris were quantified. For further analyses an overall site mean of each variable was calculated by averaging the values recorded along the five transects and the four segments delimited by the five transects. Water velocity and discharge were assessed immediately upstream to the transect 0, as a measure of the hydrological conditions "entering" into the sampled reach. These hydrological variables were quantified following Elosegi *et al.* (2009) by a coup addition of a solution (500 g/4 L) of salt (conservative solute tracer, chloride as sodium chloride). These variables were measured in all sampling dates (n = 36).

2.2.4 Riparian condition (RC)

The condition of riparian corridor was evaluated by means of 9 variables covering aspects of soil surface cover, vegetation structure, in-stream canopy and the degree of bank alteration. The riparian width was measured at both margins of each transect. The proportion of woody (trees and shrub), herbaceous and bare soil cover, as well as the number of trees and shrubs were quantified. These measurements were performed at both margins and within the four segments delimited by the five stream transects. The in-stream canopy area (expressed as percentage of canopy cover) within each segment delimited by two consecutive transects was visually quantified with sighting tubes (Johnson and Covich, 1997). The number of bank incisions by livestock and the proportion of bank stability were quantified in both margins of the segments delimited by transects. Bank stability was computed as the ratio between the linear length of banks covered by vegetation and its roots over the total bank length (Rosso and Fernández Cirelli, 2013). For further analyses an overall site mean for each variable was calculated, by averaging the values recorded at both margins and in the five transects or in the four segments delimited by the five transects. These variables were measured at the beginning of each sample period (n = 12, four sites, three sampling periods).

2.3 Data analyses

Variables such as wet channel width, riparian width, water depth and sediment column depth were analyzed with their average and dispersion values (minimum and maximum). All data was standardized to zero mean and unit variance. Prior to analyses, Spearman rank correlation coefficients were calculated within each set of environmental variables (WQ, HS and RC) to test for redundancy. Variables with high (rho ≥ 0.7) and significant ($p < 0.05$) correlation were removed from analysis matrices and those finally entering into the analyses were selected according to an ecological criterion.

2.3.1 Multimetric indexes

In order to assess multivariate aspects of water quality, different regional water quality indexes (WQI) were calculated. These were designed to different basins in Pampa Plain, such as, Matanza-Riachuelo (Berón, 1984), Reconquista River (Basílico *et al.*, 2015), Salado River (Moscuzza *et al.*, 2007) in Argentina and Santa Lucía basin in Uruguay (Arocena *et al.*, 2008). To gain consensus about empirical patterns observed in the multivariate response of WQ, three additional international water quality indexes were also calculated (Queralt, 1982 from Catalonia, Spain; Cude, 2001 from Oregon, United States; Debels *et al.*, 2005 from Chillán, Chile). Biological oxygen demand was substituted by chemical oxygen demand in those indexes that include the former variable. The chemical oxygen demand is a more accurate variable with a lower cost (Debels *et al.*, 2005). In addition, two habitat quality indexes (HQI) were calculated, one widely used worldwide (Barbour *et al.*, 1999) and another designed for urban streams of the Pampa Plain, including the Langueyú stream (Cochero *et al.*, 2016). In turn, data of riparian condition was used to calculate three riparian quality indexes (RQI), two designed for Pampa Plain streams (Rosso and Fernández Cirelli, 2013; Basílico *et al.*, 2015) and one for fluvial ecosystems of Spain adapted to different regions of the world (Munné *et al.*, 2003).

2.3.2 Water quality gradient

Water quality variables were standardized to cero mean and unit variance. A Principal Component Analysis (PCA) was conducted on WQ variables. To perform this analysis a correlation matrix was used. Differences in water quality between sampling sites were evaluated by a non-parametric multivariate analysis of the variance (PERMANOVA). Euclidean similarity index and 9999 permutations were used to perform this analysis. In turn, univariate differences in WQ variables among sampling sites were tested by means of a non-parametric analysis of the variance by the Kruskal-Wallis test. To test whether the observed changes in water quality variables respond to a likely longitudinal upstream-downstream continuum in the recovery of water quality aspects, a RELATE routine (Clarke and Gorley, 2015) was performed. This analysis determines the level of association between two resemblance matrices, in this case, the serial model matrix (linear distance between sampled sites expressed in km) and the WQ Euclidean distance matrix. Spearman correlation was used and the permutation tests were performed with 9999 random permutations.

2.3.3 Water quality in the context of local aspects of HS and RC

To evaluate the WQ in the context of local aspects of HS but also RC, WQ and HS matrices were unified to n = 12 to match the sample size of RC. To this end, we averaged values of their variables for each spring-summer period in all sampling sites (3 averaged sampling periods for each of the four sites = 12 samples). Spearman rank correlation coefficients were estimated to explore the empirical relationships of WQ with HS and RC variables at different level of analysis. Correlation coefficients were calculated between single WQ, HS and RC variables, and between their multimetric indexes. Two Distance-based Linear Models (DistLMs) were performed to achieve a direct quantitative partitioning of the multivariate variability of WQ that could be explained by HS and RC variables. DistLM is a routine for analyzing and modeling the relationship between a response multivariate data cloud and one or more predictor variables (Anderson *et al.*, 2008). BEST procedure was used to generate models including all possible combinations of predictor variables and modified Akaike's Information Criterion (AICc) was used to identify the best model. Those models with the lowest AICc were considered the most parsimonious. DistLMs were run with 9999 permutations. The difference between the AICc value of the best model and each of the other models (ΔAICc) was calculated and the Akaike weights of models (Burnham and Anderson, 2002) with values of these differences less than 2 were estimated. In these models (ΔAICc < 2), the relative importance of each predictor variable (Wi, predictor weight, Symonds and Moussalli, 2011) was calculated. For each predictor variable, the Akaike weights of all the models containing that variable were summed. Those predictor variables that frequently occur in the most likely models (ΔAICc < 2) have an Akaike weight close to 1 whereas variables that are absent from or are only present in less likely models (high AICc values) have an Akaike weight close to 0.

All statistical analyses were performed with PRIMER.5 (Plymouth Routines In Multivariate Ecological Research) with the add-on package PERMANOVA+, and PAST 4.01 (Paleontological Statistics Software Package for Education and Data Analysis).

3 Results

3.1 Local conditions of water quality, in-stream habitat structure and riparian conditions

Water quality attributes, in-stream habitat structure and riparian conditions of sampled sites are summarized in Tables 1–3. The uppermost reach exposed to urban conditions presented waters with comparatively lower pH (always above 7.8), dissolved oxygen and nitrate, as well as, a higher conductivity, salinity, dissolved and suspended solids, chlorides, ammonium, total phosphorous, chemical oxygen demand and bacteriological loads (mesophiles, coliforms and *E. coli*). The in-stream habitat structure in this uppermost urban reach was characterized by fast currents with a dominance of runs and riffles (Fig. 2). A narrow and shallow channel, high sediment depth, substrate dominated by gravel and almost a total absence of macrophytes were also observed. The riparian corridor was narrow with a large number of trees and high bank stability. Downstream, in the site exposed to cropland, most aspects of water quality improved, being more noticeable in the bacteriological loads. In addition, a sharply decreased of total suspended solids was recorded. In-stream habitat structure presented a shallow channel with a high proportion of runs, substrate dominated by bedrock, and submerged macrophytes' development (Fig. 2). Riparian corridor was mainly herbaceous with bare soil patches and lacking canopy cover. Further downstream, in the livestock-exposed reach, some aspects of water quality improved further, whereas other, as total phosphorus concentration increased, pH decreased and total suspended solids remained constant, being fairly similar to the immediate upper reach. This reach showed a broad channel also dominated by runs but with 25% of pools (Fig. 2). A lowest maximum sediment depth, substrate with an even composition of different sizes and the development of submerged macrophytes also characterized the in-stream habitat structure in this site. The riparian corridor presented the largest surfaces of bare soil and the vegetation cover was fully herbaceous. In addition, at some transects the riparian development was nil whereas the worse bank stability was observed. In natural grasslands, water presented the lowest concentrations of ions, salts and solids (water conductivity, salinity, dissolved solids, chlorides), ammonium, total phosphorous and bacteriological load. This site also showed high pH, dissolved oxygen and nitrate. Maximum values of NO_3:NH_4 ratio was observed. This reach had higher proportion of pools (almost 50%), an increase in the maximum sediment depth, substrate dominated by bedrock and boulders and very low coverage of macrophytes (Fig. 2). The riparian width showed maximum values, a dominance of herbaceous cover with isolated trees and shrubs, the absence of bare soil zones and high bank stability.

Overall, sampling sites showed significant differences in water quality ($F = 7.022$, $p = 0.0001$). All WQ variables but temperature were significantly different between sites (p value < 0.05, Tab. 1). The in-stream habitat structure quality was lowest in the urban reach according to both indexes calculated (Tab. 2). Riparian conditions quality (Tab. 3) were maximum in natural grasslands (according to all RQI indexes calculated) and minimum in the livestock reach (according Basílico *et al.* and Munné *et al.* indexes).

3.1.1 Water quality gradient

In the multivariate PCA ordination the first two axes cumulatively explained 57.37% of the total variation in water quality of sampling sites of the Langueyú stream (Fig. 3). Chemical oxygen demand (rho = 0.777), mesophiles (rho = 0.766), water conductivity (rho = 0.755), total phosphorous (rho = 0.742) and fecal coliforms *E. coli* (rho = 0.711) showed a positive correlation with the first component, while dissolved oxygen (rho = –0.765) and pH (rho = –0.598) were negatively related. On the other hand, NO_3:NH_4 (rho = 0.701), temperature (rho = 0.646) and total suspended solids (rho = 0.544) were positively correlated with the second principal component. All samples of the urban site and a few samples of cropland, the two uppermost reaches, were grouped in the positive extreme of the first component. The remaining samples of cropland together with those of the livestock and natural grassland were positioned towards the negative end. Natural grassland and urban samples were mainly confined in the positive end of the second principal component, while cropland and livestock samples were mainly in the negative zone.

Table 1. Water quality characterization (mean and standard deviation) of sampling sites in the Langueyú stream. The redundancy test shows the non-redundant variables (pass). Non-redundant variables are highlighted. The numbers in redundant variables represent the selected informative variables accounting for their information. The statistics of the Kruskal-Wallis test were included. WQI = water quality index.

N°	Variable	Code	Urban	Cropland	Livestock	Natural grassland	Redundancy Test	Kruskal-Wallis Test H (*p* value)
			Mean (SD)	Mean (SD)	Mean (SD)	Mean (SD)		
1	Temperature (°C)	T	23.28 (3.16)	21.86 (3.91)	23.14 (3.2)	23.45 (2.81)	Pass	3.09 (0.377)
2	pH	pH	8.31 (0.3)	8.89 (0.59)	8.71 (0.46)	9.15 (0.46)	Pass	12.26 (0.007)
3	Dissolved oxygen (mg/L)	DO	1.44 (1.8)	9.33 (4.3)	11 (3.25)	13.02 (5.13)	Pass	21.4 (<0.001)
4	Dissolved oxygen saturation (%)	DO%	16.58 (18.65)	110.84 (51.4)	134.3 (40.13)	158.37 (66.19)	3	
5	Water conductivity (µS/cm)	K	1167 (108.98)	1077.56 (93.81)	1049.67 (98.45)	912.44 (177.67)	Pass	12.29 (0.006)
6	Salinity (ppt)	S	539.22 (31.92)	498.56 (22.78)	485.89 (23.3)	415.89 (70.5)	5	
7	Total dissolved solids (ppm)	TDS	826 (79.06)	764.67 (67.29)	744.33 (69.99)	641 (121.63)	5	
8	Total suspended solids (mg/L)	TSS	74.79 (30.83)	11.92 (10.75)	9.54 (10.81)	44.81 (36.99)	Pass	24.41 (<0.001)
9	Nitrate (mgN-NO3/L)	NO_3	1.56 (0.91)	2.16 (0.73)	2.73 (1.53)	3.83 (1.39)	11	
10	Ammonium (mgN-NH4/L)	NH_4	15.45 (4.17)	12.43 (4.61)	8.87 (3.71)	0.48 (0.59)	11	
11	Nitrate/ammonium ratio	NO_3:NH_4	0.13 (0.12)	0.22 (0.15)	0.49 (0.64)	14.04 (9.01)	Pass	23.66 (<0.001)
12	Total inorganic nitrogen	NO_3+NH_4	17.01 (3.59)	14.59 (4.41)	11.61 (2.65)	4.35 (1.58)	11	
13	Total phosphorus (mg/L)	TP	2.77 (0.88)	2 (0.64)	2.21 (0.59)	1.68 (0.44)	Pass	9.56 (0.023)
14	Chemical oxygen demand	COD	183.34 (90.58)	83.01 (34.48)	59.28 (40.53)	65.27 (38.09)	Pass	16.18 (0.001)
15	Chloride (mg/L)	Cl	91.25 (16.84)	79.75 (12.27)	79.13 (14.14)	70.75 (16.43)	5	
16	Mesophiles (UFC/mL)	Meso	170717.11 (155026.68)	56042.89 (46443.08)	7953.78 (6567.78)	7005.56 (13064.8)	Pass	17.99 (<0.001)
17	Total coliforms (colif/100 mL)	Tcol	7055.56 (9616.02)	1892.22 (785.84)	1370.22 (1046.19)	1086.33 (863.33)	18	
18	*E. coli* (E. coli/100 mL)	Ecol	6766.67 (9788.9)	2032.67 (3496.66)	584.33 (816.47)	596.11 (810.33)	Pass	10.86 (0.009)
	WQI Berón (1984)	WQI Ber	1.49 (0.58)	2.78 (0.44)	2.94 (0.28)	5.22 (0.58)	–	
	WQI Basílico *et al.* (2015)	WQI Bas	1.22 (1.02)	3.71 (0.74)	4.03 (0.53)	5.43 (0.93)	–	
	WQI Moscuzza *et al.* (2007)	WQI Mos	36.27 (7.59)	58.5 (6)	64.58 (5.84)	68.04 (5.04)	–	
	WQI Arocena *et al.* (2008)	WQI Aro	5.8 (5.09)	33.71 (14.75)	41.98 (11.11)	39.83 (8.17)	–	
	WQI Queralt (1982)	WQI Que	24.94 (7.57)	51.75 (7.21)	59.22 (10.5)	54.12 (9.5)	–	
	WQI Cude (2001)	WQI Cud	11.65 (0.55)	12.88 (0.8)	13.22 (0.92)	13.36 (0.87)	–	
	WQI Debels *et al.* (2005)	WQI Deb	31.75 (8.35)	52.66 (6.1)	56.67 (5.03)	56.56 (4.16)	–	

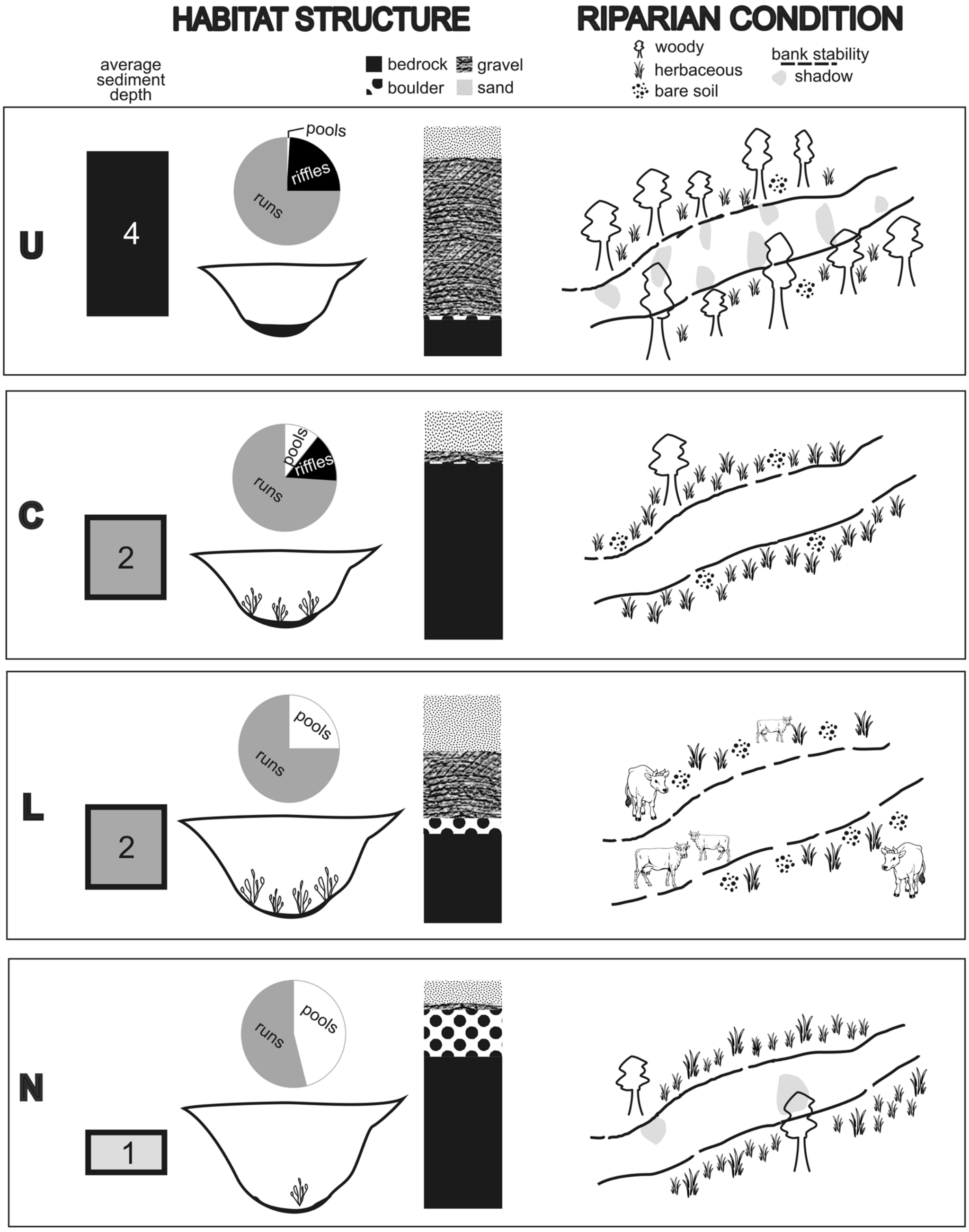

Fig. 2. Graphical summary of the variations of in-stream habitat structure and riparian conditions of the sampling sites in the Langueyú stream (U = urban, C = cropland, L = livestock, N = natural grassland). Increasing numbers and darkness used as an ordinal code for ease the interpretation of average sediment column depth. In riparian condition, the larger the dashed line, the lower the bank stability. Pie and bar charts of microhabitats and substrate composition respectively, present actual percentage values.

The poorest conditions in water quality were observed in the uppermost urban site (Tab. 1, Fig. 4). Indeed, multimetric indexes categorized the water quality in this site as "very poor", "contaminated" and "similar to raw septic sewage". All WQ indexes detected a marked improvement from urban to the downstream cropland reach, where water quality was categorized as "poor", "high pollution", "lightly contaminated" and "deteriorated". Further downstream, some WQ indexes detected another marked improvement from livestock reach to natural grasslands whereas other did not. Indeed, almost 24% (rho = 0.232) of the total variation in water quality was significantly ($p = 0.005$) explained by the longitudinal upstream-downstream spatial arrangement of sampled sites (RELATE routine). Similarly to multimetric indexes, single WQ variables also presented different patterns of downstream longitudinal variation. For instance, dissolved oxygen and nitrate showed a progressive increase downstream from urban site, while water conductivity, salinity, total dissolved solids, chloride, ammonium and bacteriological loads showed an inverse pattern

Table 2. In-stream habitat structure characterization (mean and standard deviation) of sampling sites in the Langueyú stream. The redundancy test shows the non-redundant variables (pass). Non-redundant variables are highlighted. The numbers in redundant variables represent the selected informative variables accounting for their information. HQI = habitat quality index.

N°	Variable	Code	Urban	Cropland	Livestock	Natural grassland	Redundancy Test
			Mean (SD)	Mean (SD)	Mean (SD)	Mean (SD)	
1	Pools (%)	Pools	0.01 (0.02)	0.11 (0.19)	0.25 (0.22)	0.46 (0.13)	Pass
2	Riffles (%)	Riffles	0.24 (0.12)	0.15 (0.08)	0	0	1
3	Runs (%)	Runs	0.75 (0.14)	0.74 (0.11)	0.75 (0.22)	0.54 (0.13)	1
4	Backwaters (n)	Backwa	10.67 (2.52)	10 (6.56)	8.33 (1.53)	7.33 (6.51)	Pass
5	Woody debris (n)	Woodyde	1.33 (2.31)	0.67 (1.15)	1 (1)	1 (1)	Pass
6	Floating macrophytes (%)	Floatmacro	0.01 (0.01)	0	0	0.01 (0.01)	Pass
7	Submerged macrophytes (%)	Submacro	0	0.24 (0.15)	0.39 (0.17)	0.04 (0.05)	Pass
8	Emergent macrophytes (%)	Emermacro	0	0	0	0.01 (0.01)	Pass
9	Sediments depth min (cm)	Sdep min	0	0	0	0	10
10	Sediments depth average (cm)	Sdep avg	4.94 (1.1)	2.11 (0.91)	1.12 (0.43)	1.01 (0.69)	Pass
11	Sediments depth max (cm)	Sdep max	17.67 (3.48)	13.33 (5.24)	8.33 (1.53)	10.67 (8.17)	10
12	Substrate bedrock (%)	Subedrock	0.15 (0.08)	0.77 (0.08)	0.39 (0.23)	0.66 (0.1)	Pass
13	Substrate boulder (%)	Suboul	0.03 (0.03)	0.01 (0.01)	0.07(0.03)	0.21 (0.13)	Pass
14	Substrate gravel (%)	Subgrav	0.7 (0.1)	0.04 (0.04)	0.29 (0.27)	0.04 (0.05)	Pass
15	Substrate sand (%)	Subsand	0.13 (0.09)	0.18 (0.1)	0.25 (0.11)	0.1 (0.05)	Pass
16	Water discharge (L/s)	Wdis	358.54 (111.92)	267.75 (78.48)	389.33 (193.99)	502.57 (218.66)	22
17	Water velocity (m/s)	Wveloc	0.76 (0.23)	0.42 (0.15)	0.34 (0.1)	0.37(0.1)	19
18	Channel width min (m)	Cwid min	4.45 (0.42)	6.06 (0.3)	6.04 (0.6)	7.57 (0.35)	19
19	Channel width average (m)	Cwid avg	5.12 (0.15)	7.32 (0.97)	9.53 (2.03)	9.37 (0.43)	Pass
20	Channel width max (m)	Cwid max	6.05 (0.59)	8.69 (1.95)	14.5 (8.67)	11.5 (0.87)	19
21	Water depth min (cm)	Wdep min	17 (4)	13 (6)	18 (5)	29 (6)	22
22	Water depth average (cm)	Wdep avg	36.73 (3.13)	35.53 (5.02)	54.78 (16.07)	60.07 (4.87)	Pass
23	Water depth max (cm)	Wdep max	71.78 (7.6)	58 (11.53)	98.11 (25.62)	93.33 (5.89)	22
	HQI Cochero *et al.* (2016)	HQI Coc	3.81 (0.04)	6.7 (0.12)	7.15 (0.07)	6.87 (0.37)	–
	HQI Barbour *et al.* (2019)	HQI Bar	92.33 (1.53)	126.67 (8.62)	118 (9.54)	145 (8.19)	–

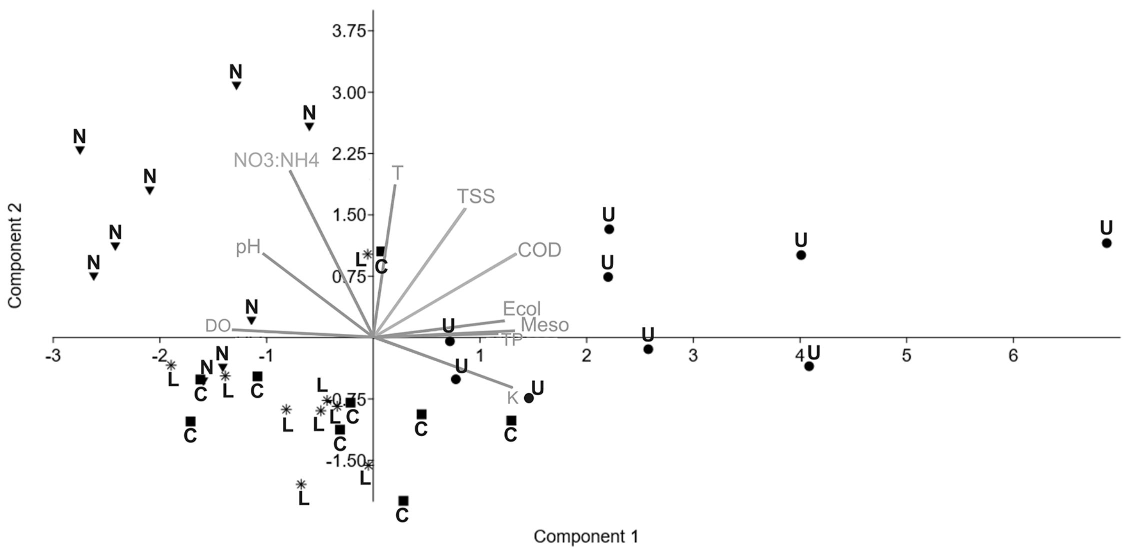

Fig. 3. Biplot of the first two PCA axes based on water quality of the Langueyú stream (9 samples of U = urban, C = cropland, L = livestock and N = natural grassland). Variable codes as Table 1.

(progressive decrease, Tab. 1, Fig. 5). Conversely, other WQ variables displayed a disruption in this longitudinal recovery. A decrease of pH and an increase of TP were recorded at livestock site, while total suspended solids and chemical oxygen demand increased downstream to cattle intrusion, at natural grassland reach. Indeed, some water quality indexes detected a disruption in the continuum of recovery from livestock to natural grassland reach.

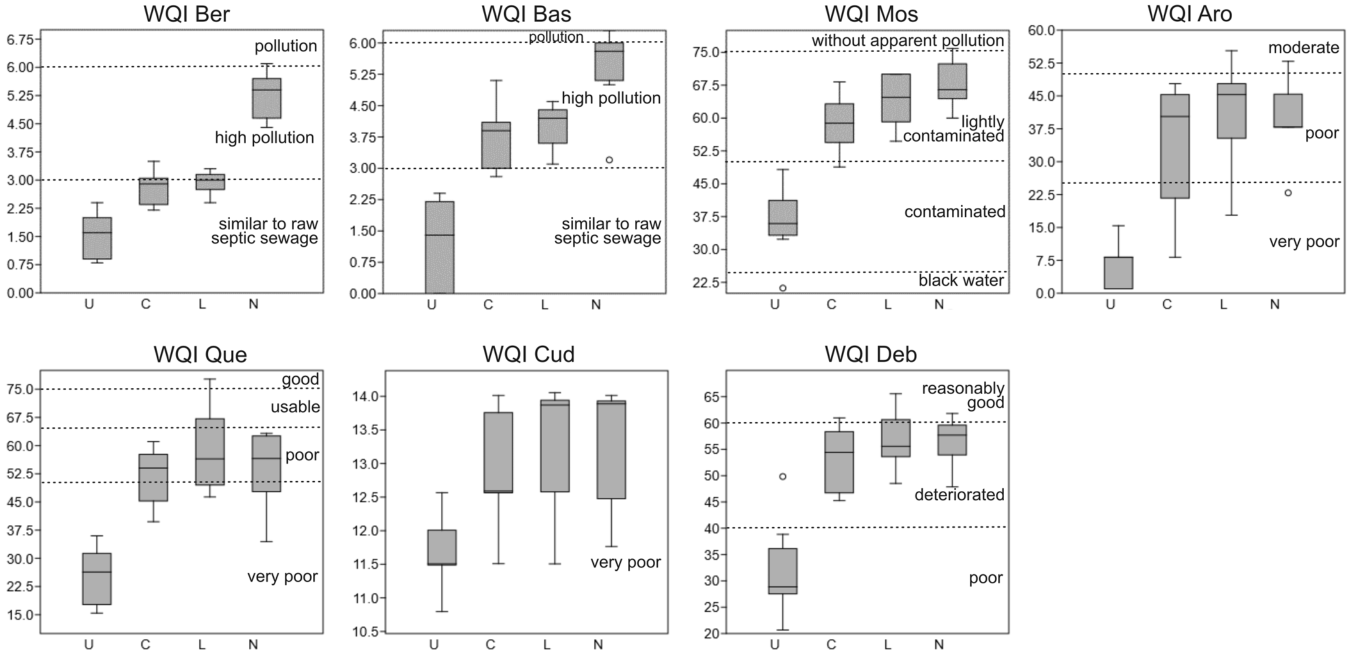

Fig. 4. Box-whisker plots of Water Quality Indexes (WQI) in different sampling sites of the Langueyú stream (U = urban, C = cropland, L = livestock, N = natural grassland). Variable codes as Table 1. The thresholds for the water quality categories are indicated for each index.

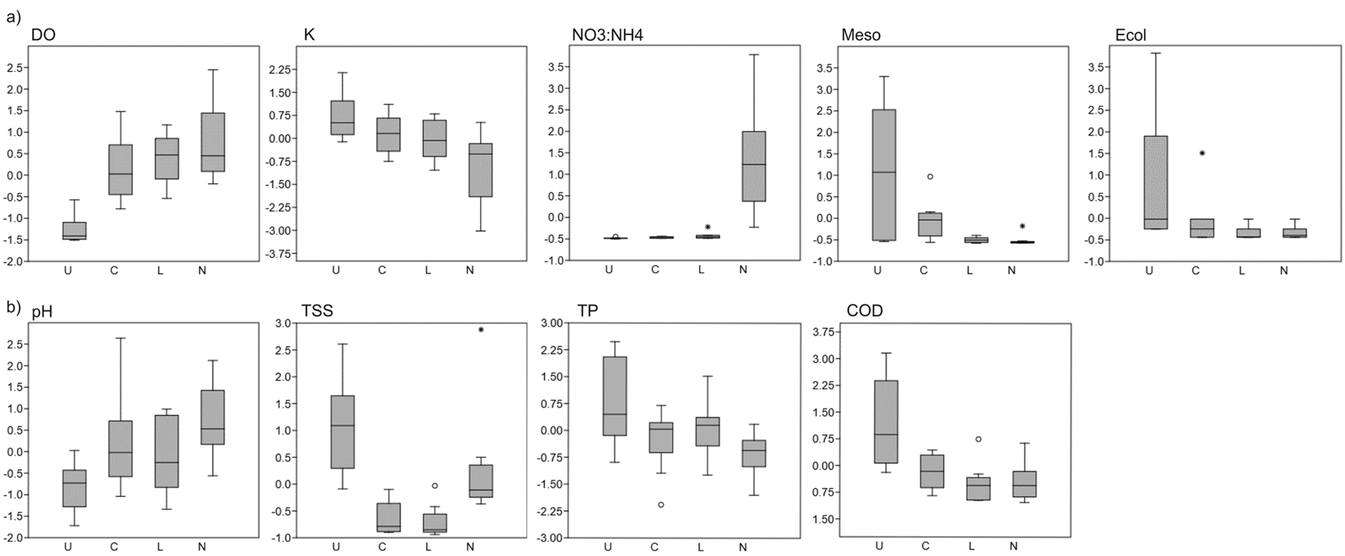

Fig. 5. Box-whisker plots of non-redundant standardized water quality variables at sampling sites of Langueyú stream (U = urban, C = cropland, L = livestock, N = natural grassland). (a) Variables with a continuum of recovery. (b) Variables with a disruption in the continuum of recovery. Variable codes as Table 1.

3.1.2 Water quality in the context of longitudinal variation of HS and RC

The water quality of the Langueyú stream was highly and significant related with several local aspects of in-stream habitat structure and riparian conditions (Tab. 4). Water quality variables and the associated multimetric indexes were more frequently related with the in-stream habitat structure than with riparian conditions. Dissolved oxygen and NO_3:NH_4 ratio increased as the mean channel width and proportion of pools increased. Contrary, the water conductivity, chemical oxygen demand and bacteriological loads showed an inverse pattern with these local habitat structure variables. The riparian width and its herbaceous cover were the most relevant aspects of the riparian corridor for water quality. Mean riparian width was significant and positively related with pH, dissolved oxygen and NO_3:NH_4, and negatively with water conductivity, chemical oxygen demand and bacteriological loads. As

Table 3. Riparian condition characterization (mean and standard deviation) of sampling sites in the Langueyú stream. The redundancy test shows the non-redundant variables (pass). Non-redundant variables are highlighted. The numbers in redundant variables represent the selected informative variables accounting for their information. RQI = riparian quality index.

N°	Variable	Code	Urban	Cropland	Livestock	Natural grassland	Redundancy Test
			Mean (SD)	Mean (SD)	Mean (SD)	Mean (SD)	
1	Riparian width min (m)	Riwid min	9.24 (0)	9.73 (0.64)	0	21.5 (0)	Pass
2	Riparian width average (m)	Riwid avg	11.2 (0.05)	17.05 (1.18)	23.48 (0)	26.28 (0)	Pass
3	Riparian width max (m)	Riwid max	15.13 (0.6)	21.88 (0.46)	33.8 (0)	32.3 (0)	2
4	Woody cover (%)	Wcov	0.22 (0.02)	0.02 (0.01)	0	0.07 (0.004)	5
5	Herbaceous cover (%)	Hcov	0.78 (0.02)	0.98 (0.01)	1 (0)	0.93 (0.004)	Pass
6	Bare soil (%)	Bscov	0.11 (0.04)	0.18 (0.16)	0.21 (0.01)	0	Pass
7	Shrubs (n)	Shrubs	0	0.33 (0.29)	0	3.33 (2.89)	5
8	Trees (n)	Tress	23.17 (2.75)	2.17 (0.29)	0	11 (2.6)	5
9	Bank stability(%)	Bstabil	0.84 (0.1)	0.74 (0.09)	0.57 (0.04)	0.84 (0.12)	Pass
10	Bank incisions by livestock (n)	Bincilive	0	0	2 (0)	0	Pass
11	Canopy cover (%)	Canocov	0.16 (0.02)	0	0	0.01 (0.01)	Pass
	RQI Basílico *et al.* (2015)	RQI Bas	80.17 (1.44)	85.5 (4.33)	70.5 (0)	87.17 (1.44)	–
	RQI Munné *et al.* (2003)	RQI Mun	50 (0)	40 (8.66)	30 (0)	60 (0)	–
	RQI Rosso and Fernández Cirelli (2013)	RQI Ros	1.48 (0.09)	1.41 (0.14)	1.46 (0.04)	1.91 (0.13)	–

Table 4. Spearman correlation coefficients between water quality (WQ) and in-stream habitat structure (HS) variables and their indexes (left), and between water quality and riparian condition (RC) variables and their indexes (right). Only high (rho≥ 0.7) and significant (*p* value< 0.05) correlations are shown. Variable codes as Tables 1, 2 and 3.

WQ	HS	rho	*p* value	WQ	RC	rho	*p* value
T	Subsand	–0.78	0.003	pH	Riwid avg	0.71	0.009
pH	Subgrav	–0.65	0.031	DO	Riwid avg	0.77	0.004
DO	Cwid avg	0.71	0.018	K	Riwid avg	–0.85	0.001
DO	Pools	0.74	0.006	TSS	Hcov	–0.84	0.001
DO	Subgrav	–0.76	0.012	TSS	Canocov	0.81	0.001
K	Cwid avg	–0.74	0.014	NO_3:NH_4	Riwid avg	0.85	0.001
K	Wdep avg	–0.7	0.011	COD	Riwid avg	–0.78	0.003
K	Sdep avg	0.85	0.005	COD	Hcov	–0.65	0.022
TSS	Submacro	–0.76	0.004	Meso	Riwid avg	–0.75	0.005
NO_3:NH_4	Cwid avg	0.67	0.018	Ecol	Riwid avg	–0.72	0.008
NO_3:NH_4	Pools	0.69	0.014	WQI Bas	RI Ross	0.69	0.013
NO_3:NH_4	Suboul	0.69	0.013	WQI Ber	RI Ross	0.67	0.018
COD	Cwid avg	–0.89	0.003				
COD	Suboul	–0.72	0.009				
Meso	Pools	–0.86	0.0004				
Meso	Suboul	–0.71	0.009				
Ecol	Cwid avg	–0.8	0.002				
Ecol	Wdep avg	–0.75	0.005				
Ecol	Pools	–0.73	0.008				
Ecol	Backwa	0.73	0.007				
Ecol	Subgrav	0.67	0.017				
WQI Bas	HI Coc	0.68	0.025				
WQI Bas	HI Bar	0.89	0.0001				
WQI Ber	HI Coc	0.71	0.009				
WQI Ber	HI Bar	0.9	0.0001				
WQI Mos	HI Bar	0.74	0.006				
WQI Que	HI Coc	0.87	0.004				
WQI Deb	HI Coc	0.7	0.02				

Tab. 5. First ten best models found for water quality of the Langueyú stream considering the in-stream habitat structure and riparian condition as explanatory variables. N = number of variables. Variable codes as Table 2 and 3.

Water quality (In-stream habitat structure)				Water quality (Riparian condition)			
N	Variables	AICc	R^2	N	Variables	AICc	R^2
1	Sdep avg	24.711	0.451	2	Hcov, Bscov	23.208	0.643
2	Sdep avg, Suboul	24.991	0.586	2	Riwid avg, Hcov	25.399	0.572
2	Sdep avg, Subsand	25.188	0.579	2	Riwid min, Hcov	25.441	0.570
2	Submacro, Sdep avg	25.311	0.575	1	Riwid avg	25.526	0.413
3	Sdep avg, Subuol, Subsand	25.746	0.703	3	Hcov, Bscov, Canocov	25.87	0.699
1	Cwid	26.545	0.361	2	Riwid avg, Canocov	26.063	0.548
3	Submacro, Sdep avg, Subsand	26.666	0.679	3	Hcov, Bscov, Bstabil	26.193	0.691
2	Pools, Sdep avg	26.67	0.524	3	Riwid avg, Hcov, Bscov	26.268	0.689
2	Floatmacro, Sdep avg	26.717	0.522	3	Riwid min, Hcov, Bscov	26.297	0.689
1	Subgrav	26.795	0.347	1	Hcov	26.395	0.369

herbaceous cover increased, the total suspended solids and chemical oxygen demand decreased. Regional WQ indexes were strongly and positively related with both in-stream HS and RC indexes.

The best explanatory model (lowest value of AICc) for water quality considering all non-redundant local habitat structure variables incorporated one variable, the average of sediment depth (Tab. 5). Given their frequencies of occurrence in models proposed and the weight of these models, sediment depth (W_i = 0.922), percentage of sand (W_i = 0.342), percentage of boulders (W_i = 0.285) and submerged macrophytes (W_i = 0.217) were the most important predictor variables for the water quality considering the in-stream habitat structure. Instead, the variation in water quality explained by the riparian conditions was best described by a model with herbaceous and bare soil cover. In addition, these variables (W_i = 1) were the most frequently riparian condition variables selected by the best water quality explanatory models.

4 Discussion

Our results showed that urbanization together with local in-stream habitat and riparian conditions intimately aligned with contrasting surrounding land uses modulated key aspects of water quality in a prairie stream. Several aspects of water quality displayed a continuum of recovery, with a progressive reduction (water conductivity, salinity, total dissolved solids, chloride, ammonium and bacteriological loads) or increase (dissolved oxygen, nitrate) of selected variables downstream from the urban reach. Alternatively and contrary to our predictions, key aspects of water quality (pH, TP, TSS and COD) showed a disruption in this continuum of recovery when contrasting riparian and in-stream habitat features were assessed.

4.1 Effects of urbanization on water quality

Urban and industrial development strongly impacted on the water quality of this prairie stream. Upstream to this impacted site, before the discharge of the effluents, the water quality is known to be markedly better. Particularly, a higher dissolved oxygen concentration and lower water conductivity, NH4 and COD were reported (Cortelezzi *et al.*, 2019). Deterioration of water quality in urban streams occurs due to the increase of nutrients, metals, pollutants and solids from the discharge of stormwater, wastewater and industrial effluents, and from the non-specific contributions as a result of increase in impervious areas and surface runoff (McGrane, 2016). Consequently, a high oxygen demand is generated by the increase of decomposer organisms generating an abrupt decrease in dissolved oxygen (Sirota *et al.*, 2013). The magnitude of the impact on water quality depends, among other variables, on the level of urbanization of the city and the capacity to treatment wastewater and industrial effluents (Walsh *et al.*, 2005). The Tandil city has a continuous population growth (123,343 inhabitants, INDEC 2010) but only a 60% of sewage network coverage (Cortelezzi *et al.*, 2019). The operation of the wastewater treatment plants that discharge in Langueyú stream is altered by clandestine connections of industrial and sewage effluents to the storm-water drainage (Banda Noriega *et al.*, 2010; Cortelezzi *et al.*, 2019). The failing of treatment plants due to illegally connected sewer pipes and industrial effluents has been the main cause of contamination in urban streams (Jewell, 2001).

4.2 Water quality gradient

A marked gradient in water quality was observed in the Langueyú stream downstream from the impacted reach by urbanization. The urban watershed continuum framework predicts a continuum of downstream transport and transformation of carbon, contaminants, energy and nutrients (Kaushal and Belt, 2012). Overall, stream self-purification comprises different mechanisms such as dilution, sedimentation, reaeration, adsorption, absorption, and chemical (acid–base, redox or precipitation reactions, coagulation, flocculation) and biological reactions (bacterial degradation, macrophytes assimilation) that occur simultaneously, allowing the recovery of the natural state of a stream over a certain distance (Vagnetti *et al.*, 2003). For instance, the progressive decrease in water conductivity, salinity, chloride and total dissolved solids have been attributed to the sedimentation and adsorption reactions in the unsaturated zones as well as dilution of effluents along the longitudinal axis of lotic ecosystems (Kaushal and Belt, 2012). In turn, the progressive increase in dissolved oxygen can be given by the aeration (oxygen diffusion) generated by the movement, mixture and turbulence of water along its course downstream. The ammonium decreased downstream is considered normal in more oxygenated waters, and it was observed in self-purification studies (Elosegi *et al.*, 1995; Jing *et al.*, 2001). It is caused by several mechanisms, including oxidation (nitrification) and biological assimilation. In parallel to an ammonium drop, an increase of nitrate is usually expected. In general, water inputs from wastewater treatment plants increase nutrient availability and, in the case of the nitrogen, shift the dominant form in transport, from nitrate to ammonium (Marti *et al.*, 2004). These authors found that in streams which nutrient concentrations consistently decline along the reach, the stream was acting as a net sink for nutrients and the sources of wastewater effluents no exceed assimilation capacity of these fluvial ecosystems. The bacteriological loads in fluvial hydrosystems sharply drop downstream due to sedimentation and natural decay, but an important abatement factor is also the filtration and adsorption by aquatic plants during their phases of growth (Elosegi *et al.*, 1995; Kleinheinz *et al.*, 2009).

4.3 Water quality in the context of longitudinal disruptions in riparian and in-stream habitat

Conversely to those variables showing a longitudinal downstream transformation in a clear evidence of water quality recovery, other aspects of water quality did not present such longitudinal behavior. This disruption was evident at the reach exposed to unrestricted cattle access. There are a variety of reasons why cattle are allowed to access the watercourses and riparian margins, including the provision of a cheap, low-maintenance source of water (O'Callaghan *et al.*, 2018). The degradation of water quality in streams exposed to livestock is generated by increases in nutrients, total suspended solids, turbidity and bacteria (Vidon *et al.*, 2008; Horak *et al.*, 2020) derived from grazing and deposit of fecal matter in water or on the stream banks (O'Callaghan *et al.*, 2018). Accordingly, streams impacted by cattle grazing have been shown to exhibit poorer water quality than streams where cattle access is restricted (Line, 2003).

Nutrient concentrations in Pampa Plain stream water are relatively high compared to other lotic systems of the world (Omernik, 1977; Binkley *et al.*, 2004) but high levels of nutrients have been also reported for some rural landscapes in Europe (Muller *et al.*, 2015). In addition to the naturally high levels of phosphorous of these ecosystems, cattle breeding has been suggested as an important anthropogenic source of phosphorus to streams (Mugni *et al.*, 2005). The introduction of excretes (fecal and urine) into the stream increases nutrient levels in water and particularly, fecal deposition of cattle increases the water phosphorus loads (James *et al.*, 2007). After cattle exclusion, these authors reported a reduction of the in-stream deposition of fecal phosphorus by 32%. Unrestricted cattle access to lotic ecosystems also negatively affects bank stability. The stream bank deterioration has been linked to high phosphorus sediment losses and poor overall water quality (Sekely *et al.*, 2002). The introduction of excretes into the stream and bank deterioration also increases the organic matter loads (O'Callaghan *et al.*, 2018). Consequently, an increase in oxygen demand and a reduction in pH are expected during respiration processes. In addition, the role of unstable margins and deteriorated riverbanks as sources of humic and fulvic acids which could further drop the pH must not be ruled out.

The delivery of sediments to lotic ecosystems is a natural phenomenon. However, there has been an increasing concern about the enhancement of sediment loadings as a result of anthropogenic activities (Jones *et al.*, 2011). These activities, including livestock and agriculture practices trigger sediment loads by disturbances in riverbank structure (Shields *et al.*, 2010) and riparian conditions (O'Callaghan *et al.*, 2018). A sharply decreased in TSS was observed downstream to urban reach at cropland and livestock sites. This pattern could be explained by the presence of macrophytes at both sites, which showed a significant and inverse relation with total suspended solids. A complex relationship exists between macrophytes and fine sediments: macrophytes affect the conveyance of fine sediment and are, in turn, affected by the sediment loading (Jones *et al.*, 2011). Macrophytes create water flow resistance, dissipate the turbulent energy, creating areas of low velocity that encourages deposition of fine organic and inorganic particles. A substantial amount of material can be retained within stands of plants. This accumulation of fine sediment results in changes in bed morphology (Corenblit *et al.*, 2007). In fact, these two reaches with the presence of macrophytes had a higher proportion of fine substrate. The development of dense macrophytes communities in Pampa Plain streams is favored by the low current velocity, good light reception due to the absence of riparian trees and the high concentrations of nutrients (Rodrigues Capítulo *et al.*, 2010). These two reaches showed high concentrations of total phosphorus, as well as a lack of canopy cover due to the almost total absence of trees. Downstream to livestock site, the TSS markedly increased. The combined effect of sediment loading from cattle intrusion together with the lack of macrophytes in the natural grassland reach may help to explain this disruption in the longitudinal recovery of this WQ variable. These results suggest that macrophytes may play a central role in regulating some aspects of water quality. 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). Indeed, submerged macrophytes together with percentage of sand and sediment depth were highly relevant to explain the variability of the observed water quality in the Langueyú stream. Furthermore, high and significant associations were also found between the indexes of water quality and in-stream habitat structure.

In-stream habitat factors other than macrophytes also seemed relevant for water quality of the Langueyú stream. Particularly, the proportion of pools and the mean channel width were the most frequent habitat variables significantly related with water quality aspects. At first glance, this could be easily attributable to the spatial juxtaposition of the continuum of recovery in WQ with the natural downstream increase of channel width and pool habitats in lotic ecosystems. Nevertheless, the longitudinal spatial scale of our analysis only accounted for a small (but significant) fraction of the observed variation in WQ. Interestingly, the largest values of mean and maximum channel width were registered at livestock reach where the disruption in the continuum of recovery of several WQ aspects was observed. Unrestricted cattle access to streams and riparian corridors can cause the sloughing-off and collapse of banks which results in channels becoming wider and shallower (Ranganath *et al.*, 2009). In fact, Magilligan and McDowell (1997) reported a 10 to 20% decrease in channel width in a stream where cattle had been excluded for 14 years.

Water quality also seemed to be affected by local riparian conditions at contrasting surrounding land uses. Although typically a small area within a watershed, riparian zones often have a disproportionate influence on water and solute fluxes to streams waters and often mitigate the impact of upland sources of contaminants on water quality (Vidon *et al.*, 2010). In our survey, a disturbed riparian corridor was recorded in the stream reach exposed to livestock. Livestock grazing can affect the riparian environment by changing, reducing or eliminating vegetation (Platts, 1979). Groundcover vegetation has reported to be two times greater in livestock exclusion reaches than in grazed reaches (Ranganath *et al.*, 2009). In addition, grazed riparian areas had approximately five times more bare ground than areas with livestock exclusion (Schulz and Leininger, 1990). Livestock can also impact water quality by compaction of riparian soil, which prevents water infiltration and alters biogeochemical cycling in the surrounding riparian zone and inputs into stream networks (*e.g.*, Reisinger *et al.*, 2013). In the Pampa Plain, a recovery of the structure of the riparian vegetation stopped the collapse and flattening of the riverbank after the first year of cattle exclusion (Giorgi *et al.*, 2014). Worldwide, the lateral linkages associated with the width, extent and composition of riparian zones, mediate the water quality thought the retention of sediments, nutrients, and materials into stream channels (Richards *et al.*, 1996). Particularly, in the Pampa Plain, the water quality and riparian conditions are intimately associated with the prevalent land use in the surrounding landscape (Rosso and Fernández Cirelli, 2013; Granitto *et al.*, 2016). The effectiveness of stream buffers is moderated by very local variations in vegetative type and aspect (Rabeni and Smale, 1995). In fact, our results showed positive and significant association between indexes of water quality and riparian condition, significant relationships between the extension of the riparian vegetation and the herbaceous cover with water quality variables, and highlighted the importance of herbaceous and bare soil cover as the best predictor variables to explain the variation in water quality.

4.4 Management implications

These results have strong implications for ecosystem management. Particularly, stream restoration projects should include multiple actions such as, the recovery of in-stream structural heterogeneity, channel morphology, bank stability and riparian cover (Palmer *et al.*, 2010), which allow for self-sustaining stream function (Ceneviva-Bastos *et al.*, 2017). In this scenario, riparian corridors are a key element, not only because they physically act as a connection between terrestrial and aquatic biomes, but also because of their multiple functions (Naiman and Décamps, 1997). In this way, as a first management measure, protection of their integrity should be considered, both in terms of vegetation cover and species composition (Johnson *et al.*, 2007). Nevertheless, due to the local reach scale to which many of these riparian actions are implemented (Nakamura *et al.*, 2005), the conservation and restoration of riparian corridor usually generates longitudinal discontinuities. However, there is evidence suggesting that management intervention at riverbanks, even when spatially restricted, significantly improve hydrological retention and reduce the nutrient load in agricultural streams (Weigelhofer *et al.*, 2012).

In addition, the correct treatment of sewage and industrial effluents is a key management practice to mitigate impacts on water quality and the ecological integrity of urban streams (Walsh *et al.*, 2005; Carey and Migliaccio, 2009). Many cities do not have a sewage system that includes the entire population and/or effluents do not receive adequate treatment (Wu *et al.*, 1999; da Cruz e Sousa and Ríos-Touma, 2018). Therefore, the role of policy makers is essential for regulating both the wastewater treatment plants and the industries aiming to reduce the quantity and increase the quality of the discharges. In fact, an improvement in the operation of 1822 wastewater treatment plants in China during 2006 to 2016, showed a considerable improvement in the discharges quality in nutrients (total phosphorus, ammonia, total nitrogen) and chemical and biological demands of oxygen (Qi *et al.*, 2020). A complementary management practice that would help reduce the urban impact on the streams would be the use of artificial wetlands. In general, constructed wetlands can be designed to remove from 65% to 83% of chemical oxygen demand, 55% to 72% of total nitrogen and 30% from 84% of total phosphorous from wastewaters (Rodriguez-Dominguez *et al.*, 2020). The improvement of the quality of the water is produced jointly by the action of the macrophytes, substrate and their associated organisms.

The problem associated with the impacts of urbanization and longitudinal disruptions imposed by other surrounding land uses in prairie urbanised streams largely exceeds the scientific approach of the academy. Therefore, for an effective management plan, a comprehensive and heuristic view including interested stakeholders in the analysis is mandatory. The need then arises for an integrative socio-ecological perspective that considers ecological, social and political aspects to design approaches for the management of urbanized prairie streams (Naiman, 2013).

5 Conclusions

Water quality of a prairie stream exposed to urbanization was responsive not only to the longitudinal downstream continuum of recovery, but also to the disruptions in habitat structure and riparian conditions intimately associated with surrounding land uses. This was supported by the high and significant empirical associations between the in-stream habitat and riparian conditions and the attributes of the water quality. Several aspects of water quality as water conductivity, dissolved oxygen, salinity, total dissolved solids, chloride, total inorganic nitrogen and bacteriological loads displayed a continuum of recovery downstream from the urban reach. Alternatively, key variables of water quality as pH, phosphorous, total suspended solids and chemical oxygen demand showed a disruption in this continuum of recovery. Our results suggest that modifications of riparian habitat and in-stream habitat by nearby land uses may be responsible for these disruptions. Overall, these results suggest that an integral management of water quality in urbanized prairie streams should address not only issues related to urbanization, but also the heterogeneity of the in-stream habitat and the conservation of riparian vegetation.

Acknowledgements. Andrea Bertora would like to acknowledge the financial support of a doctoral fellowship from the Comisión de Investigaciones Científicas de la Provincia de Buenos Aires (CIC) and the Consejo Nacional de Investigaciones Científicas y Técnicas (CONICET). JJR acknowledges the support provided by CONICET. The authors thank the owners and managers of the study sites for allowing access. Authors are indebted to Viviana Colasurdo (Facultad de Ingeniería, Universidad Nacional del Centro de la Provincia de Buenos Aires UNCPBA) for several chemical analyses of water samples and Anahí Tabera (Facultad de Ciencias Veterinarias, UNCPBA) for microbiological analysis of water samples.

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Cite this article as: Bertora A, Grosman F, Sanzano P, Rosso JJ. 2022. Combined effects of urbanization and longitudinal disruptions in riparian and in-stream habitat on water quality of a prairie stream. *Knowl. Manag. Aquat. Ecosyst.*, 423, 15.