

Evaluating environmental indicators and invertebrate based indices for diagnosing urban impacts in large river wetlands

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Research Article

Keywords: Biomonitoring, Wetlands, Land use, Macroinvertebrate, Biologic metrics, Biologic index

Posted Date: August 20th, 2026

DOI: <https://doi.org/10.21203/rs.3.rs-10515748/v1>

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Abstract

Urban expansion is causing the degradation of aquatic ecosystems, especially in urban wetlands. This study aimed to determine which physicochemical parameters, biological variables, and multimetric indices are most effective for diagnosing environmental quality in wetlands impacted by urbanization within a large river basin. Historical data and new sampling were used. All sites were categorized according to their land use. Comparative analyses of physicochemical variables were performed, and the Water Quality Index was applied, along with the evaluation of 74 benthic macroinvertebrate metrics (richness, density, tolerance, trophic functional groups, and biotic indices). The results showed that the reference wetlands exhibited good environmental conditions, with lower pH and conductivity values and greater transparency. In contrast, the urban wetlands exhibited fair to poor water quality, with higher conductivity values, lower transparency, and high levels of chlorophyll *a*, indicating eutrophic conditions. A total of 14 macroinvertebrate metrics were selected because they best discriminated between land uses. During high water, Ecological Quality of Macroinvertebrates (EQR_{macro}), Macroinvertebrate Index in Pampean Rivers (IMRP), and the density of Ephemeroptera, Mollusca, and Chironomidae (Tanypodinae) were key indicators. During low water, EQR_{macro}, Biological Monitoring Working Party (BMWP), IMRP, and Naididae density were key indicators. This study provides a useful tool for the management and monitoring of urban wetlands.

Introduction

Wetlands are essential aquatic environments that provide numerous ecosystem services to human society, including water quality protection, climate change mitigation, flood control, drought prevention, and serve as an important reservoir of flora and fauna.

Currently, the proportion of landscape covered by urban development has increased considerably worldwide (United Nations 2019; Behnisch et al. 2022) and is a major cause of biodiversity loss (McDonald et al. 2020). The widespread increase in urban land cover has led to the fragmentation of wildlife habitats across the landscape, as well as a reduction in species richness (McKinney 2006).

In this sense, urban wetlands are severely degraded and have poor water quality due to the effluent they receive from many types of human activities. These include the discharge of agrochemicals, industrial effluents, and sewage, turning these wetlands into sinks for various pollutants (Häder et al. 2020; Alikhani et al. 2021).

The expansion of urbanization into rural areas is observed in South America (Jordan 2017; United Nations 2019), including the Pampas plains. In the Pampas plains, wetlands form through natural processes along the Paraná River floodplain. This plain begins near the city of Corrientes and ends at the Río de la Plata, with the first 600 km forming the middle section (Iriondo 1972). Approximately 40% of the flooded area are wetlands, where benthic macroinvertebrates are essential components of these ecosystems. They constitute a functionally and taxonomically diverse group that serves as food for fish,

amphibians, and waterfowl, and is involved in the breakdown of organic matter and nutrients. Furthermore, they can indicate environmental changes (compiled in Zilli et al. 2022).

The increasing deterioration of these wetlands has necessitated the development of methodologies to determine their degree of alteration. In this context, the use of macroinvertebrates as indicators is a useful tool for detecting alterations that occur in aquatic environments (Barbour et al. 1999). Their advantages as bioindicators include their general ubiquity, abundance, diversity, and sensitivity to different environmental disturbances (Resh 2008; Abbasi and Abbasi 2012). Several methodological approaches have been developed to assess the biological quality of water, using metrics based on the taxonomic properties of the community (e.g., richness, diversity, relative composition of different taxa, density, and biotic indices), on functional traits (e.g., feeding functional groups), or both (Birk et al. 2012). Given the large number of available metrics and indices, it is necessary to select those most sensitive to the ecoregion under study and the gradient being measured. In this regard, the methodology of Barbour et al. (1996) has been internationally validated for macroinvertebrates.

The diagnosis of aquatic ecosystems through the use of biotic metrics and indices has been widely developed in North America, Europe, and Asia (Cai et al. 2019). In South America, while there is a growing body of research on river assessment and, to a lesser extent, on lake assessment, the design of these methodologies is more recent and still limited (Cortelezzi et al. 2022).

Argentina has a significant diversity of urban wetlands, and studies have focused primarily on epidemiological aspects (Faggi and Martínez-Carretero 2013; Juárez et al. 2018) and, to a lesser extent, on urban effects (Gómez et al. 2021). The cities of Santa Fe and Santo Tomé, located in the Paraná River basin, experienced strong urban development in the last century, resulting in new forms of land use and the creation of urban wetlands (Pallud 2010; Rausch et al. 2019). The increase in population density in these cities generated a greater amount of effluents discharged into water bodies (on time and without prior treatment), establishing a stressful social and ecological scenario, by exerting great pressure on the wetlands (Caino 2016; Frau et al. 2018; Vaschetto et al. 2021; Arias et al. 2022).

Although benthic macroinvertebrates have been extensively analyzed in the wetlands and channels of the Middle Paraná River floodplain (e.g., Marchese 2008; Pavé and Marchese 2005), knowledge regarding biological indicators in urban wetlands of this region remains scarce and has only recently begun to be addressed (Arias et al. 2022). The objective is to determine which physicochemical parameters, biological variables, and multimetric indices are most effective for diagnosing environmental quality in urban-impacted lakes within a large river basin. To this end, a historical database was used, and a new sampling was conducted at specific sites. The study includes a comparison of wetlands using different taxonomic metrics (richness, density, composition, tolerance, diversity, dominance), functional traits (feeding groups), and biotic indices. By addressing these objectives, we hope to add further empirical data to the growing field of biomonitoring studies, ultimately contributing to the development of a multimetric macroinvertebrate-based index for assessing the water quality of wetlands in the region.

Methodology

Study area

The study was conducted at 9 sites during both high and low water periods, as the hydrological phase can have a great influence on the biota and environmental variables (Fig. 1, Supplementary Information 1). Available data were taken from the Institute National of Limnology (Benthic Laboratory) database for 4 reference lakes (i.e., Ref1, Ref2, Ref3, and Ref4; Zilli, 2010) and 2 urban lakes (i.e., Urb1 and Urb2; Arias, 2019). Additionally, sampling was conducted during the high (November 2017) and low (July 2018) water periods in 3 urban lakes located in the city of Santa Fe (Urb3, Urb4, and Urb5). Specific details of the sampling design, type of sampler used, sampling dates, and associated bibliographic sources are available in Supplementary Information 1.

Land use

In the study area, there are a variety of lakes surrounded by different land uses. Thus, land use was categorized by the predominant land use in an area with a radius of 500 meters from the edge of the wetlands, using the Land Cover Classification System (LCCS) (Di Gregorio and Jansen 1998; Di Gregorio 2005, with a spatial resolution of: xMin, yMin – 62.8797, -34.3836 and xMax, yMax – 58.8808, -27.9994). The analyses were performed using QGIS 2.18.24 software. To identify land use in each site, data sources from different government offices and websites were used (source: Provincial Institute of Statistics and Censuses [IPEC] - Santa Fe Delegation; Spatial Data Infrastructure of Santa Fe [IDESF]). The classification was based primarily on 3 land uses: 1) %Urban (UR); 2) %Natural herbaceous (NH); 3) %Floodable area (FA). In this way, the land use types for the wetlands were assigned: reference sites (100% NH) (n = 4), and urban sites (98% UR and 2% FA) (n = 5).

All the analyses below were run separately for both hydrometric phases (low and high water).

Physicochemical variables

At each sampling date and site, the main physicochemical variables were recorded in the center of the waterbody: water transparency (cm) using a Secchi disk, depth (cm) using portable probes, pH, conductivity (CD) ($\mu\text{S}/\text{cm}$), temperature ($^{\circ}\text{C}$), and dissolved oxygen (DO) ($\text{mg O}_2/\text{L}$) using a Hach HQ40D multiparameter device. Nitrites (ppm NO_3^-), nitrates (ppm NO_2^-), ammonium (ppm NH_3^-), and total phosphorus (ppm P) were analyzed using the EDTA titration method according to the Standard Method adapted by Hach (DREL 2000). For the determination of chlorophyll a (Cl a), 2 L of water were filtered using Whatman GF/C filters. The acetone extract (90% acetone) was obtained according to standardized methods (APHA et al. 1988; APHA et al. 1992), and the concentration of Cl a was determined by absorbance: HACH DREL 2000 spectrophotometer for Ref1, Ref2, Ref3, Ref4, and HACH DR5000 for Urb1, Urb2, Urb3, Urb4, Urb5.

The Kruskal-Wallis test was used to evaluate the existence of significant differences between the sites for each of the environmental parameters studied. Among those that obtained significant results ($p < 0.05$), a Dunn test was applied to identify significant differences in pairwise comparisons ($p < 0.05$).

Macroinvertebrates

The macroinvertebrate sampling procedure consisted of sampling the available habitats (e.g., vegetated coastlines, open waters). Thus, in most of the lakes, vegetated mesohabitats and open waters were sampled, except for Urb1 and Urb2, where only vegetated habitats existed (Arias 2019). At each sampling point, three samples (replicates) were taken according to the depth and substrate characteristics of the sampling site (Supplementary Information 1): with an Ekman grab (225 cm²) (Zilli 2010; Arias 2019), a Rigosha grab (100 cm²) (Arias 2019; Capeletti et al. 2021), a Tamura grab (322 cm²), and a D-net (100 cm²) (Capeletti et al. 2021). In all cases, the density unit was individuals per square meter (ind/m²). All samples were sieved through a 200 µm mesh, fixed in 10% formaldehyde solution, and stained with erythrosine. The fixed and stained samples were transported to the laboratory and stored in plastic containers. In the laboratory, invertebrates were separated from the sediments under a stereomicroscope and preserved in a 70% ethanol solution. Identification and counting were performed using stereomicroscopes or optical microscopes for different taxa. Invertebrates were identified to the lowest possible taxonomic level using the keys available in Lopretto and Tell (1995), Brinkhurst and Marchese (1988), Domínguez and Fernández (2009), and Trivinho-Strixino (2011). The taxa were assigned to their functional feeding group (FFG) according to Merritt et al. (2008) and information obtained by the research team.

Water Quality Index

The subjective Water Quality Index (WQI) proposed by Pesce and Wunderlin (2000) was applied:

$$WQI_{sub} = k \frac{\sum i c_i x P_i}{\sum i P_i}$$

where k is a subjective water quality constant with values between: 1 if the water is uncontaminated (clear or with naturally occurring suspended solids); 0.75 if the water is slightly contaminated by unnatural light color, foam, or slight turbidity; 0.50 if the water is contaminated, indicated by unnatural color, high (unnatural) turbidity, suspended organic solids, etc.; and 0.25 for highly contaminated water, indicated by blackish color, strong odor, visible fermentation, etc. C_i is the value assigned to each parameter after normalization, and P_i ranges from 1 to 4, representing a scale of importance for each parameter to aquatic life, where 1 is the least important and 4 the most important. The parameters used to calculate this index were pH, conductivity, temperature, and dissolved oxygen.

Calculation and selection of metrics

A total of 74 metrics were estimated to assess the quality of the wetlands and the impact of different land uses (reference vs. urban) (Supplementary Information 2). Taxonomic and biological trait metrics were calculated and categorized into 6 groups: richness ($\approx 24\%$), density ($\approx 42\%$), tolerance ($\approx 12\%$), trophic functional groups ($\approx 12\%$), and biotic indices ($\approx 10\%$). The calculated biotic indices were based on the BMWP (Biological Monitoring Working Party) and ASPT (Average Score Per Taxon) methods (Armitage et al. 1983) and their adaptations, BMWP' (Biological Monitoring Working Party, Alba Tercedor and Sánchez-Ortega, 1988) and BMWP'' (Biological Monitoring Working Party, Loyola, 2000). The IMRP (Macroinvertebrate Index in Pampean Rivers, Rodrigues Capitulo 1999), EQR IndMacro (Ecological Quality of Macroinvertebrates) (Catalan et al. 2003), and the multimetric indices ICBrio (Index of Benthic Communities in Rivers, Kuhlmann et al. 2012) and IBIS (Index of Benthic Invertebrates in Saline Rivers, Capeletti et al. 2021) were also calculated. It was decided to calculate the indices used for lotic environments since some of them have been calculated in other studies for lentic environments and were able to discriminate between wetlands according to anthropogenic impact (Arias 2019; Arias et al. 2022).

Metrics with low ranges or with mostly identical values are unable to differentiate the most impacted sites from those that are less impacted (Stoddard et al. 2008). Therefore, richness metrics with a range < 6 , percentage metrics with a range $< 10\%$, and metrics with $> 90\%$ of values equal to zero were eliminated (Klemm et al. 2003, Macedo et al. 2016). Then, to assess the sensitivity level of each metric, the methodology proposed by Barbour et al. (1996) was used, which evaluates the sensitivity of metrics by assessing means and interquartile ranges: type 3 sensitivity (no overlap of interquartile ranges), type 2 (low overlap of ranges and medians outside the overlap), type 1 (moderate overlap of interquartile ranges and medians outside the overlap), and type 0 (high overlap of ranges and median). For this study, we retained the metrics that showed high sensitivity levels (type 3 and 2).

After applying the method of Barbour et al. (1996), the selected metrics were reanalyzed according to the following criteria: 1) no redundancy among metrics, 2) showing the predicted response to water quality variables, 3) showing the greatest sensitivity to degradation, and 4) ease of calculation/use for future monitoring purposes (by scientists, management agencies, etc.). Redundancy among metrics was tested using Pearson correlation analysis. A metric was considered redundant if the correlation coefficient was greater than 0.70 and the significance value was less than 0.05.

Results

Physicochemical variables

The physicochemical variables of the studied wetlands are shown in Table 1. In general, during high water, the pH in the urban sites was alkaline, and the water CD was relatively high. In contrast, in the reference sites, the pH was more neutral (close to 7), and the CD was lower compared to the urban sites. Transparency was greater in the reference wetlands than in the urban ones, and Cl *a* levels indicated eutrophic conditions (above 30 $\mu\text{g/L}$). During low water, the pH in the urban sites was neutral (with

greater variation in the urban wetlands), the temperature was relatively lower than in the reference sites, and the CD was higher. In the reference sites, the pH was also neutral, and as during high water, transparency was greater in the reference wetlands than in the urban ones.

Table 1

Physicochemical variables of the reference and urban lagoons of the Paraná River Valley and transitional ecoregion, mean, standard deviation, and minimum and maximum values (in parentheses). Variables that showed significant differences between land uses ($p < 0.05$) in the Kruskal-Wallis test are differentiated with the letters a and b (^{a,b}).

Site	High water	Low water		
	Reference (n = 4)	Urban (n = 5)	Reference (n = 4)	Urban (n = 5)
pH	7.23 ± 0.1 (7.1–7.3)	8.02 ± 0.65 (7.01–8.68)	7.5 ± 0.08 (7.4–7.6)	7.55 ± 0.71 (6.9–8.39)
Conductivity (uS/cm)	98.25 ± 13.72 (88–117) ^a	776.6 ± 545.78 (244–1515) ^b	85.5 ± 10.85 (70–95) ^a	620.08 ± 741.68 (0.88–1691.5) ^b
Temperature (°C)	25.75 ± 0.5 (25–26)	23.36 ± 1.64 (21.6–25.7)	16 ± 1.15 (15–17)	12.12 ± 1.58 (9.8–13.85)
Dissolved oxygen (mg/L)	8.55 ± 2.23 (5.4–10.53)	7.678 ± 3.47 (1.74–10.22)	7.025 ± 1.38 (6–9.05)	8.47 ± 1.60 (5.86–10.26)
Secchi (m)	0.61 ± 0.27 (0.43–1) ^a	0.343 ± 0.19 (0.145–0.53) ^b	0.61 ± 0.25 (0.33–0.94) ^a	0.24 ± 0.07 (0.17–0.35) ^b
Depth (m)	2.6 ± 0.79 (1.9–3.7)	1.042 ± 0.78 (0.56–2.4)	2.9 ± 0.62 (2.4–3.8)	1.76 ± 2.39 (0.45–6)
Nitrate (ppm)	0.29 ± 0.15 (0.15–0.5)	0.24 ± 0.23 (0.1–0.5)	0.29 ± 0.03 (0.25–0.32)	0.17 ± 0.12 (0.1–0.3)
Nitrite (ppm)	0.003 ± 0.0005 (0.002–0.003)	0.01 ± 0.001 (0.005–0.006)	0.004 ± 0.001 (0.002–0.005)	0.006 ± 0.007 (0.001–0.014)
Ammonium (ppm)	0.12 ± 0.028 (0.09 – 0.155)	1.773 ± 1.49 (0.21–3.17)	0.29 ± 0.03 (0.25–0.32)	1.37 ± 1.51 (0.34–3.1)

Site	High water		Low water	
	Reference (n = 4)	Urban (n = 5)	Reference (n = 4)	Urban (n = 5)
Total phosphorus (ppm)	0.47 ± 0.39 (0.205–1.05)	2.26 ± 0.81 (1.34–2.83)	0.59 ± 0.61 (0.25–1.5)	2.66 ± 2.24 (0.07–4)
Chlorophyll a (ug/L)	3.53 ± 0.90 (2.45–4.6)	88.10 ± 90.08 (29.81–191.85)	1.51 ± 1.23 (0.11–3.09)	61.27 ± 56.18 (2.4–114.3)

The wetlands were relatively shallow (with an average depth of 2 meters). DO levels were high in all wetlands. Nutrient levels were higher in all urban wetlands than in the reference wetlands, although the differences were not statistically significant ($p < 0.05$).

Regarding the comparison between urban and reference sites, significant differences were observed in water conductivity and transparency (Kruskal-Wallis test, $p < 0.05$, $n = 15$) during high and low water periods. Water conductivity values were higher in the urban sites, while transparency was greater in the reference wetlands.

WQI

The k value for wetlands Ref1, Ref3, Ref4, and Ref2 was 1; for Urb3 it was: 0.75; for Urb4 and Urb1 and Urb2: 0.50; and for Urb5: 0.25.

The WQI_{sub} values for the urban wetlands (both during high and low water levels) were low, ranging from 16 to 51, indicating fair to poor water quality. In contrast, the WQI values for the reference wetlands (during high and low water levels) were much higher (from 71 to 92), indicating good water quality.

Macroinvertebrates analysis

A database consisting of 122 taxa, identified mostly at the species or genus level, was obtained (Supplementary Information 3). As a result of applying the defined criteria for all metrics (Supplementary Information 3) (74), 14 were selected to evaluate different land uses based on their sensitivity (types 3 and 2), redundancy, and estimated response to water quality variables. During high water, 10 metrics (EQRmacro, IMRP, Ephemeroptera density, Mollusca density, Chironomidae (Tanypodinae) density, Ephemeroptera (Polymitarcyidae) density, O/C ratio, total number of taxa, number of genera, and number of Oligochaeta taxa) best discriminated between wetlands with different land uses (reference vs. urban) (Fig. 2). On the other hand, during low water, 4 metrics (EQRmacro, BMWP, IMRP, and Naididae density) best distinguished urban wetlands from reference wetlands (Fig. 3).

Regarding the correlations between individual metrics, several significant responses were obtained. The correlation during high water between richness and density metrics showed that some metrics were

redundant, with all correlations being positive ($r > 0.70$). The density of Ephemeroptera, Chironomidae (Tanypodinae), and Ephemeroptera (Polymitarcydae) was significantly correlated with the total number of taxa and the total number of genera; the density of Ephemeroptera with the density of Tanypodinae; the density of Mollusca with the O/C ratio and the number of Oligochaeta. Furthermore, the density of Tanypodinae was significantly correlated with the density of Polymitarcydae, O/C with the number of Oligochaeta, and the number of Oligochaeta with the number of genera. Regarding low water levels, a correlation test was not performed since only one metric was selected using the method of Barbour et al. (1986), and three indices were not suitable for this correlation process.

On the other hand, several significant relationships were found between the metrics and the environmental variables. During high water, the density of Ephemeroptera, Mollusca, Tanypodinae, and Polymitarcydae decreased as the water content in the wetlands increased. Conversely, with increasing temperature, the density of Ephemeroptera, Mollusca, and Polymitarcydae increased. The water content ratio and genus richness were also positively correlated with wetlands depth. During low water, the density of Naididae was positively correlated with the water content and negatively correlated with depth.

Discussion

Physicochemical indicators

The reference wetlands showed environmental conditions aligned with Allan (2004) and Yates and Bailey (2010), characterized by low anthropogenic impact, the lowest pH and CD values, and good water quality (WQI, Pesce and Wunderlin 2000). In contrast, urban wetlands had over 25% urban land use, indicating - per Luck and Wu (2002) and Allan (2004) - the onset of biological deterioration, with WQI indicating fair to poor water quality (Pesce and Wunderlin 2000).

The environmental variables that showed significant differences between the urban and reference wetlands were conductivity and transparency, consistent with studies conducted on urban wetlands by Juárez et al. (2018) and Taborda et al. (2018). This variation was observed equally during high and low water levels. Furthermore, according to several authors (summarized in Teubner et al. 2020), water transparency can be considered a key indicator for evaluating wetlands subjected to urban land use. This is primarily because low levels of transparency are associated with high $Cl a$ values (coinciding with the values in the urban wetlands). $Cl a$ is the main light-harvesting pigment of primary producers and is commonly used as a rough estimate of phytoplankton biomass (Teubner et al. 2020).

CD has been proposed as a marker of pollution from urban land use and is considered useful for pollution detection (Bondarenko et al. 2016), consistent with our study, where urban wetlands exhibited high conductivity values. On the other hand, in our study, dissolved oxygen values did not differentiate wetlands with different land uses, a result similar to that found by Fontanarroza et al. (2013) in the urban wetlands of Parque Palermo (Buenos Aires, Argentina), where, despite being polluted, these wetlands

presented high dissolved oxygen values (between 3 and 8 mg/L DO). This pattern may be explained by high levels of nutrient pollution (nitrates and phosphates); the proliferation of algae and aquatic plants drives intensive daytime photosynthesis, thereby sustaining elevated oxygen concentrations despite the anthropogenic impact (Vermonden et al. 2009).

Indicators based on macroinvertebrates

During both high and low water, the EQRmacro and IMRP biotic indices clearly distinguished wetlands under different land uses. Both indices incorporate macroinvertebrate taxonomic richness, indicating that richness can effectively discriminate a degradation gradient in urban wetlands. Although EQRmacro had previously only been applied in Mediterranean regions (Sánchez-Montoya 2008), this study constitutes its first recorded application in neotropical wetlands. Additionally, the BMWP index also differentiated wetlands by impact levels during low water, consistent with observations in lotic environments in the region (Capeletti et al. 2021).

Though the IMRP and BMWP were designed for rivers, they were used for lentic environments and yielded very good results, showing differences between reference environments and urban environments (Capeletti et al. 2021; Arias et al. 2022). This could be because both use lower-resolution taxonomic categories (subfamily and genus) in their calculations, thus accounting for greater heterogeneity in macroinvertebrate structure and exhibiting greater sensitivity to detect subtle changes in water quality.

The performance of the biotic indices aligned with Oligochaeta richness metrics during high-water periods, showing higher genus and species richness in reference wetlands than in urban ones. Although genus and species richness of Oligochaeta are commonly used, finer taxonomic resolution is recommended because taxa respond differently to pollution (Prat et al. 2009). Reference wetlands presented greater taxa richness and sensitive taxa, while urban wetlands contained only tolerant taxa. These results are consistent with studies reporting low taxa richness in urban wetlands both regionally (Fontanarrosa et al. 2013; Taborda et al. 2018; Gallardo et al. 2019) and globally (Hill et al. 2015; Rivera Usme et al. 2013).

Regarding density metrics (Ephemeroptera Polymitarcyidae, Mollusca and Tanypodinae), all showed the highest values in reference wetlands, consistent with the studies cited below. The order Ephemeroptera is considered sensitive to anthropogenic influence, and therefore, many environmental assessment studies relied on the presence of certain species and the increase in their density (Buckup et al. 2007; Crisci-Bispo et al. 2007). In our study, Ephemeroptera density in reference wetlands was substantially higher than in urban wetlands, aligning with regional studies that classify this order as sensitive to pollution (Marchese et al. 2020). Within this group, *Campsurus* nymphs were dominant among the mayflies. However, special attention should be given to some species of this genus, as they exhibit a higher level of tolerance compared to other Ephemeroptera species. In our region, *Campsurus violaceus* and *Campsurus notatus* are found (Molineri et al. 2015). In a water quality monitoring study conducted by the Environmental Agency of the State of São Paulo (CETESB, in Portuguese, Kuhlmann et al. 2012), the high abundance of *Campsurus truncatus* in eutrophic environments suggests that this species should

not be considered a sensitive taxon in qualitative indices (Kuhlmann et al. 2020). This highlights the need for tolerance trials for our species, since, based on the results obtained, they could develop higher population densities in less impacted environments, potentially indicating better environmental conditions in lentic systems.

Regarding Mollusca in Argentina, the ecology of many freshwater species within this group has not been extensively studied, and their ecological requirements remain largely unknown in many cases (De Francesco and Hassan, 2009). It should be noted that in lentic environments, mollusk density increases, as they thrive at higher densities, particularly in shallow, vegetated water bodies (Dillon 2000; De Francesco and Hassan 2009). In urban wetlands, poor water quality is associated with reduced aquatic vegetation cover, according to studies by Salcedo et al. (2018). This vegetation represents an important food resource for aquatic organisms (Thomaz and Ribeiro da Cunha 2010). It also serves as an indirect food resource by increasing periphyton biomass, which is an important food source for snails. Therefore, the decline of mollusks observed in urban wetlands could be attributed to habitat loss and reduced food availability. Further studies are needed to determine, within this taxon, the most appropriate taxonomic level to use as a bioindicator based on its tolerance or sensitivity to pollution.

Finally, the presence of Tanypodinae (*Ablabesmyia*, *Coelotanypus*, *Procladius*, *Alotanypus*, *Tanypus*, *Larsia*, and *Denopelopia*) in reference wetlands serves as a good indicator of water quality, as this subfamily is sensitive to high pollution levels from wastewater discharge (Molineri et al. 2020, for northwest of Argentina). According to Fontanarrosa et al. (2013), the removal of aquatic vegetation may hinder both the establishment and persistence of aquatic insects. For Tanypodinae specifically, vegetation provides habitat and food resources for their prey (Cremona et al. 2008). Therefore, the reduction of vegetation in urban wetlands may explain the decrease in density or absence of Tanypodinae in these environments.

During low water periods, only Naididae density allowed for the differentiation of wetlands. Authors such as Rodríguez and Reynoldson (2011) and Marchese et al. (2020) consider Naididae to be highly tolerant of pollution. In the region, this family (which includes Naidinae, Pristininae, Tubificinae, and Rhyacodrilinae) is very well represented with a great richness of genera (*Pristina*, *Dero*, *Allonais*, *Slavina*, *Chaetogaster*, *Nais*, *Stephensoniana*, *Amphichaeta*, *Paranais*, *Stylaria*, *Bratislavia*, *Limnodrilus*, *Paranadrilus*, *Tubifex*, *Bothrioneurum*, *Branchiura*) (Marchese et al. 2020). Furthermore, compared to insects and mollusks, it is a much more sessile group.

The significant correlations between metrics and environmental variables aligned with findings from previous studies (Buckup et al. 2007; Crisci-Bispo et al. 2007; Bondarenko et al. 2016; Molineri et al. 2020). These studies showed that pollution-sensitive taxa - Ephemeroptera, Polymitarcyidae, Mollusca, and Tanypodinae - declined as conductivity increased, while the less sensitive group Naididae increased under the same conditions.

Conclusion

In the urban wetlands of the Paraná River, high CD and low transparency were the main impacts on water quality in these systems. The WQI was a good indicator of the poor quality of the urban wetlands, distinguishing them from the reference wetlands. The density of insect groups (Ephemeroptera and Chironomidae) and the total richness of taxa and genera would be the candidate metrics to evaluate differences in land use in the region's wetlands. The reference wetlands show very low anthropogenic impact, and the richness of sensitive taxa increases in these higher-quality environments. It is worth highlighting the Oligochaeta group, organisms with a wide distribution in our region, which allows us to distinguish urban wetlands from reference sites using the number of Oligochaeta taxa during the high water period and the density of Naididae during the low water period.

We need more data to create or adapt a multimetric index for the region's wetlands. Based on our results, we suggest using the EQRmacro and IMRP indices (which allowed us to discriminate between different sites) until more sites can be evaluated and the metrics defined more robustly. On the other hand, it should be noted that the adjustment of metrics corresponding to different taxa at different phases, especially those related to abundance, corresponds to differences in seasonality and life cycles, since low water levels coincide with winter and high water levels with summer, when different taxa have peak abundance. This underscores the importance of considering seasonality when designing biomonitoring systems.

Given the natural resources these wetlands provide and the need for a deeper understanding of their evolution over time, it is essential to monitor them and implement environmental management measures aimed at restoring water quality. Conservation policies are also needed to maintain the health of urban freshwater ecosystems, recognizing cities as socio-ecological systems and incorporating the values, attitudes, and behaviors of their inhabitants.

Declarations

Funding

This research was financially supported by Fondo para la Investigación Científica y Tecnológica (FONCyT) (PICT 2018 3447).

Competing Interests

The authors have no relevant financial or non-financial interests to disclose.

Author Contributions

All authors contributed to the study conception and design. Material preparation, data collection and analysis were performed by Capeletti Julieta, Alberto Diana, Facelli Florencia, Marchese Mercedes and Zilli Florencia. The first draft of the manuscript was written by Capeletti Julieta and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Data Availability

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

Acknowledgements

We are grateful indebted to Esteban Creus and Marcelo Piacenza for field assistance.

References

1. Abbasi T, Abbasi SA (2012) Water quality indices. Elsevier, Ámsterdam
2. Alba-Tercedor J, Sánchez-Ortega A (1988) Un método rápido y simple para evaluar la calidad biológica de las aguas corrientes basado en el de Hellawell (1978). *Limnetica* 4:51–56. <https://doi.org/10.3145/epi.2008.mar.18>
3. Alikhani S, Nummi P, Ojala A (2021) Urban wetlands: A review on ecological and cultural values. *Water* 13:3301. <https://doi.org/10.3390/w13223301>
4. Allan JD (2004) Landscapes and riverscapes: The influence of land use on stream ecosystems. *Annu Rev Ecol Evol Syst* 35:257–284. <https://doi.org/10.1146/annurev.ecolsys.35.120202.110122>
5. APHA, AWWA & WEF (1988) Standard Methods for the Examination of Water and Wastewater. 17th Edition. American Public Health Association, Washington
6. APHA, AWWA & WEF (1992) Standard Methods for the Examination of Water and Wastewater. 18th Edition. American Public Health Association, Washington
7. Arias J (2019) Análisis de invertebrados indicadores de calidad de sistemas acuáticos urbanos de la ciudad de Santa Fe. Thesis, University of Litoral
8. Arias MJ, Vaschetto PA, Marchese M, Regaldo L, Gagneten AM (2022) Benthic Macroinvertebrates and Zooplankton Communities as Ecological Indicators in Urban Wetlands of Argentina. *Sustainability* 14:1–21. <https://doi.org/10.3390/su14074045>
9. Armitage PD, Moss D, Wright JF, Furse MT (1983) The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running-water sites. *Water Res* 17:333–347. [https://doi.org/10.1016/0043-1354\(83\)90188-4](https://doi.org/10.1016/0043-1354(83)90188-4)
10. Barbour MT, Gerritsen J, Griffith GE, Frydenborg R, Mccarron E, White JS, Bastian ML (1996) A framework for biological criteria for Florida streams using benthic macroinvertebrates. *J North Am Benthol Soc* 15:185–211. <https://doi.org/10.2307/1467948>
11. Barbour MT, Gerritsen J, Blaine SD, Stribling JB (1999) Rapid Bioassessment Protocols For Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates, and Fish, Second Edition. Environmental Protection Agency, Washington
12. Behnisch M, Krüger T, Jaeger JA (2022) Rapid rise in urban sprawl: Global hotspots and trends since 1990. *PLOS Sustain Transform* 1:e0000034

13. Birk S, Willby NJ, Kelly MG, Bonne W, Borja A, Poikane S, Bund WVD (2012) Intercalibrating classifications of ecological status: Europe's quest for common management objectives for aquatic ecosystems. *Sci Total Environ* 454–455:490–499. <https://doi.org/10.1016/j.scitotenv.2013.03.037>
14. Bondarenko EA, Ilina KV, Andrianova MJ, Chusov AN (2016) Main inorganic ions and electric conductivity of polluted urban streams. *Mag Civ Eng* 68:37–44. <https://doi.org/10.5862/MCE.68.4>
15. Brinkhurst RO, Marchese M (1988) A contribution to the taxonomy of the aquatic Oligochaeta (Haplotaxidae, Phreodrilidae, Tubificidae) of South America. *Can J Zool* 65:3154–3165
16. Buckup L, Bueno AAP, Bond-Buckup G, Casagrande M, Majolo F (2007) The benthic macroinvertebrate fauna of highland streams in southern Brazil: Composition, diversity and structure. *Rev Bras Zool* 24:294–301. <https://doi.org/10.1590/s0101-81752007000200005>
17. Cai Y, Xu J, Zhang M, Wang J, Heino J (2019) Different roles for geography, energy and environment in determining three facets of freshwater molluscan beta diversity at broad spatial scales. *Sci Total Environ* 659:451–462. <https://doi.org/10.1016/j.scitotenv.2018.12.373>
18. Caino A (2016) Alternativas de gestión de una laguna urbana eutrofizada de Santa Fe (Argentina) utilizando el fitoplancton como indicador de calidad trófica. Thesis, University of Litoral
19. Capeletti J, Marchese MR, Zilli FL (2021) Evaluating macroinvertebrate metrics for ecological assessment of large saline rivers (Argentina). *Environ Sci Pollut Res* 28:66464–66476. <https://doi.org/10.1007/s11356-021-16559-7>
20. Catalan J, Ventura M, Munné A, Godé L (2003) Desenvolupament d'un index integral de qualitat ecológica i regionalització ambiental dels sistemes lacustres de Catalunya. Agència Catalana del Agua, Generalitat de Catalunya
21. Cortelezzi A, Paz LE (2022) Macroinvertebrate biomonitoring in Latin America: Progress and challenges. *Freshw Sci* 42:204–213
22. Cremona F, Planas D, Lucotte M (2008) Biomass and composition of macroinvertebrate communities associated with different types of macrophyte architectures and habitats in a large fluvial lake. *Fundam Appl Limnol* 171:119
23. Crisci-Bispo VL, Bispo PC, Froehlich CG (2007) Ephemeroptera, Plecoptera and Trichoptera assemblages in two Atlantic Rainforest streams, Southeastern Brazil Vera. *Rev Bras Zool* 24:545–551
24. De Francesco CG, Hassan GS (2009) The significance of molluscs as paleoecological indicators of freshwater systems in central-western Argentina. *Palaeogeogr Palaeoclimatol Palaeoecol* 274:105–113
25. Di Gregorio A (2005) Land cover classification system: classification concepts and user manual: LCCS, vol 2. Food & Agriculture Organization, Rome
26. Di Gregorio A, Jansen LJ (1998) A new concept for a land cover classification system. *Land* 2:55–65
27. Dillon RT (2000) *The Ecology of Freshwater Molluscs*. Cambridge University Press, Cambridge

28. Dominguez E, Fernández H (2009) Macroinvertebrados bentónicos sudamericanos. Sistemática y biología. Fundación Miguel Lillo, Tucumán
29. Faggi A, Martínez-Carretero E (2013) Argentina. In: Fors IM (ed) Ecología urbana experiencias en América Latina. Limusa, Ciudad de Mexico, pp 11–25
30. Zilli FL, Marchese MR, Arias MJ, Capeletti J, Facelli F, Saigo M (2022) Macroinvertebrados acuáticos. In: Marchese M, Gagneten AM (2022) Ambientes acuáticos de la provincia de Santa Fe: protocolos de monitoreo con perspectiva socioecológica. University of Litoral, Santa Fe, pp 108–131
31. Fontanarrosa MS, Collantes MB, Bachmann AO (2013) Aquatic Insect Assemblages of Man-Made Permanent Ponds, Buenos Aires City, Argentina. Neotrop Entomol 42:22–31.
<https://doi.org/10.1007/s13744-012-0093-1>
32. Frau D, de Tezanos Pinto P, Mayora G (2018) Are cyanobacteria total, specific and trait abundance regulated by the same environmental variables? Ann Limnol Int J Lim 54:3
33. Gallardo LI, Coronel JM, Poi ASG (2019) Urban rain-fed lakes: macro-invertebrate assemblages associated with *Egeria najas* as indicators of biological integrity in wetlands of Corrientes Province (Argentina). Biodivers Conserv 28:1549–1568
34. Gómez N, Siri A, Capítulo LR, Colautti DC, Alcalde L, Capítulo AR et al (2021) Effects of urban demand for food and water on physicochemicals and biotic structure of riverine wetlands in the Pampean plain. Ecohydrol Hydrobiol 22:355–369
35. Häder DP, Banaszak AT, Villafañe VE, Narvarte MA, González RA, Helbling EW (2020) Anthropogenic pollution of aquatic ecosystems: Emerging problems with global implications. Sci Total Environ 713:136586
36. Hill MJ, Mathers KL, Wood PJ (2015) The aquatic macroinvertebrate biodiversity of urban ponds in a medium-sized European town (Loughborough, UK). Hydrobiology 760:225–238
37. Iriando M (1972) Mapa geomorfológico de la llanura aluvial del río Paraná desde Helvecia hasta San Nicolás. República Argentina. Rev Asoc Geol Argent (27 – 2, pp 155–160
38. Jordán R, Rizzo L, Prado A (2017) Desarrollo Sostenible, Urbanización y Desigualdad en América Latina y el Caribe: Dinámicas y Desafíos para el Cambio Estructural. CEPAL, Santiago, Chile
39. Juárez I, Crettaz Minaglia MC, Gianello D, Rodríguez MS, San Millán F, Chaves E, Juárez RA (2018) Diagnóstico ambiental de la cuenca media-baja del arroyo Santa Bárbara (Gualeguaychú, Entre Ríos). Sci Interfluvius 9
40. Klemm DJ, Blocksom KA, Fulk FA, Herlihy AT, Hughes RM, Kaufmann PR et al (2003) Development and evaluation of a Macroinvertebrate Biotic Integrity Index (MBII) for regionally assessing Mid-Atlantic Highlands streams. Environ Manage 31:656–669. <https://doi.org/10.1007/s00267-002-2945-7>
41. Kuhlmann ML, Imbimbo HV, Watanabe HM (2020) *Campsurus truncatus* Ulmer, 1920 (Polymitarcyidae): an Ephemeroptera in eutrophic waters. Acta Limnol Bras 32:e17

42. Kuhlmann M, Imbimbo HV, Ogura LL (2012) Protocolo para o biomonitoramento com as comunidades bentônicas de rios e reservatórios do estado de São Paulo. Environmental Company of the State of São Paulo, São Paulo
43. Lopretto E, Tell G (1995) Ecosistemas de aguas continentales. Rev Soc Entomol Argent 55:1–4
44. Loyola RGN (2000) Atual estágio do IAP no uso de índices biológicos de qualidade. Anais do V Simpósio de Ecossistemas Brasileiros: Conservação. UFES, Vitória–ES 10:46–52
45. Macedo DR, Hughes RM, Ferreira WR, Firmiano KR, Silva DRO, Ligeiro R, Callisto M (2016) Development of a benthic macroinvertebrate multimetric index (MMI) for Neotropical Savanna headwater streams. Ecol Indic 64:132–141. <https://doi.org/10.1016/j.ecolind.2015.12.019>
46. Marchese MR, Gagneten AM, Montalto L, Gallardo LI, Damborsky MP, Poi ASG (2020) Aplicación de indicadores biológicos en el nordeste argentino. In: Dominguez E (eds) La bioindicación en el monitoreo y evaluación de los sistemas fluviales de la Argentina: bases para el análisis de la integridad ecológica. Eudeba, Buenos Aires pp 9-122
47. Marchese MR, Rodriguez AR, Pave PJ, Carignano MR (2008) Benthic invertebrates structure in wetlands of a tributary of the middle Parana River (Argentina) affected by hydrologic and anthropogenic disturbances. J Environ Biol 29:343–348
48. McDonald RI, Mansur AV, Ascensão F, Colbert ML, Crossman K, Elmqvist T et al (2020) Research gaps in knowledge of the impact of urban growth on biodiversity. Nat Sustain 3:16–24
49. McKinney ML (2006) Urbanization as a major cause of biotic homogenization. Biol Conserv 127:247–260
50. Merritt RW, Cummins KW, Berg MB (2008) An Introduction to the Aquatic Insects of North America. Kendall/Hunt Publishing Company, Iowa
51. Molineri C, Salles FF, Emmerich D (2015) Revision of Campsurus violaceus species group (Ephemeroptera: Polymitarcyidae) with new synonymies and nomina dubia in Campsurus Eaton, 1868
52. Molineri C, Tejerina EG, Torrejón SE, Pero EJI, Hankel GE (2020) Indicative value of different taxonomic levels of Chironomidae for assessing the water quality. Ecol Indic 108:105703. <https://doi.org/10.1016/j.ecolind.2019.105703>
53. Pallud P (2010) Las Urbanizaciones Privadas Cerradas en Santo Tomé. Santa Fe Pampa 6:37–58
54. Pave PJ, Marchese M (2005) Invertebrados bentónicos como indicadores de calidad del agua en ríos urbanos (Paraná-Entre Ríos, Argentina). Ecol Austr 15:183–197
55. Pesce SF, Wunderlin DA (2000) Use of water quality indices to verify the impact of Cordoba City (Argentina) on Suquia River. Water Res 34:2915–2926. [https://doi.org/10.1016/S0043-1354\(00\)00036-1](https://doi.org/10.1016/S0043-1354(00)00036-1)
56. Prat N, Ríos B, Acosta R, Rieradevall M (2009) Los macroinvertebrados como indicadores de la calidad del agua. In: Dominguez E, Fernández HR (eds) Macroinvertebrados bentónicos sudamericanos. Sistemática y Biología. Fundación Miguel Lillo, Tucumán, pp 631–654

57. Rausch GA, Martínez I, Nardelli ML, Szupiany E (2019) Concentración/dispersión en Santa Fe, Argentina: problemáticas e interrogantes sobre los procesos urbanos contemporáneos en una ciudad media de América Latina. *Cuad de Geografía: Revista Colombiana de Geografía* 28:66–88. <https://doi.org/10.15446/rcdg.v28n1.68757>
58. Resh VH (2008) Which group is best? Attributes of different biological assemblages used in freshwater biomonitoring programs. *Environ Monit Assess* 138:131–138
59. Rivera Usme JJ, Pinilla Agudelo G, Camacho Pinzón DL (2013) Grupos tróficos de macroinvertebrados acuáticos en un humedal urbano andino de Colombia. *Acta Biológica Colombiana* 18:279–292
60. Rodrigues Capítulo A (1999) Los macroinvertebrados como indicadores de calidad de ambientes lóticos en el área pampeana. *Revista Sociedad Entomológica Argentina* 58:208–217
61. Rodriguez P, Reynoldson TB (2011) *The Pollution Biology of Aquatic Oligochaetes*. Springer Science & Business Media, Berlín
62. Sánchez-Montoya MM (2008) Estado ecológico de los ríos mediterráneos: tipología, condiciones de referencia y establecimiento de clases. Thesis, University of Murcia
63. Stoddard JL, Herlihy AT, Peck DV, Hughes RM, Whittier TR, Tarquinio E (2008) A process for creating multimetric indices for large-scale aquatic surveys. *J North Am Benthological Soc* 27:878–891. <https://doi.org/10.1899/08-053.1>
64. Taborda VJ, Crettaz Minaglia MC, Apartin C, Natale GS, Rodrigues Capítulo A (2018) Aplicación de un índice de integridad biótica para humedales en dos lagunas urbanas pampeanas. Thesis, University of Entre Ríos
65. Teubner K, Teubner I, Pall K, Kabas W, Tolotti M, Ofenböck T, Dokulil MT (2020) New emphasis on water transparency as socio-ecological indicator for urban water: Bridging ecosystem service supply and sustainable ecosystem Health. *Front Environ Sci* 8:1–22. <https://doi.org/10.3389/fenvs.2020.573724>
66. Thomaz SM, Ribeiro da Cunha E (2010) The role of macrophytes in habitat structuring in aquatic ecosystems: methods of measurement, causes and consequences on animal assemblages' composition and biodiversity. *Acta Limnol Bras* 22:218–236
67. Trivinho-Strixino S (2011) *Larvas de Chironomidae: guia de identificação*. São Carlos: UFSCar 1(2)
68. Vaschetto P, Regaldo L, Polla W, Andrade V, Gervasio S, Gagneten AM (2021) Plankton community responses to anthropogenic pollution in an Argentinian urban reserve. *Water Air Soil Pollut* 232:427
69. Vermonden K, Leuven RS, van der Velde G, van Katwijk MM, Roelofs JG, Hendriks AJ (2009) Urban drainage systems: An undervalued habitat for aquatic macroinvertebrates. *Biol Conserv* 142:1105–1115
70. Yates AG, Bailey RC (2010) Selecting objectively defined reference sites for stream bioassessment programs. *Environ Monit Assess* 170:129–140
71. Zilli F (2010) Bentos en ambientes leníticos con diferente grado de conectividad en la llanura aluvial del río paraná medio. Thesis, University of La Plata

Figures

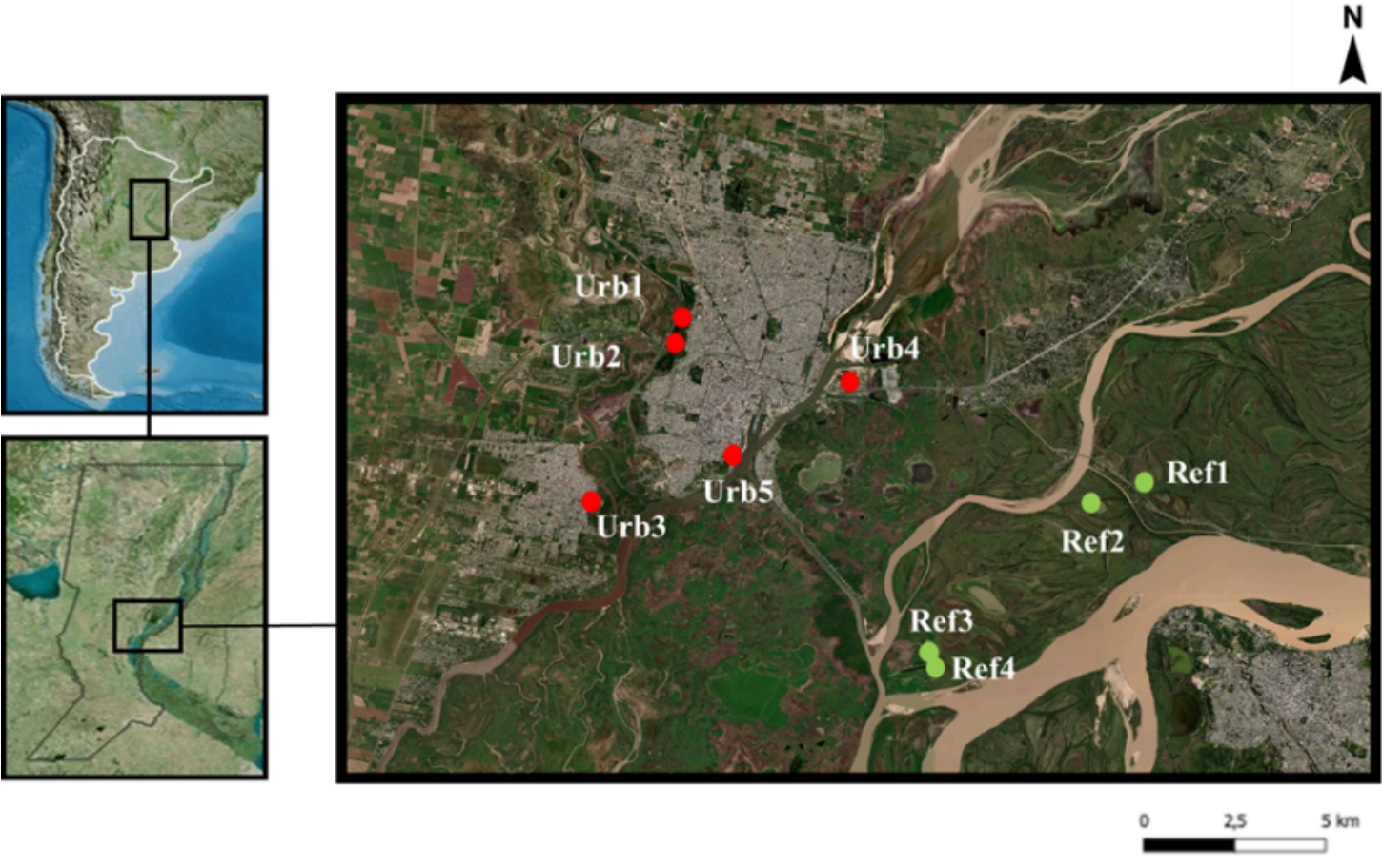


Figure 1

Map of the study area and the sampling sites located in Santa Fe province, Argentina. Ref1 (Laguna Irupé), Ref2 (Laguna La Mira), Ref3 (Vuelta Irigoyen), Ref4 (El Puesto), Urb1 (Reserva Natural Urbana del Oeste 1), Urb2 (Reserva Natural Urbana del Oeste 2), Urb3 (Laguna Bedetti), Urb4 (Reserva de la Ciudad Universitaria) and Urb5 (Lago Parque del Sur General Manuel Belgrano).

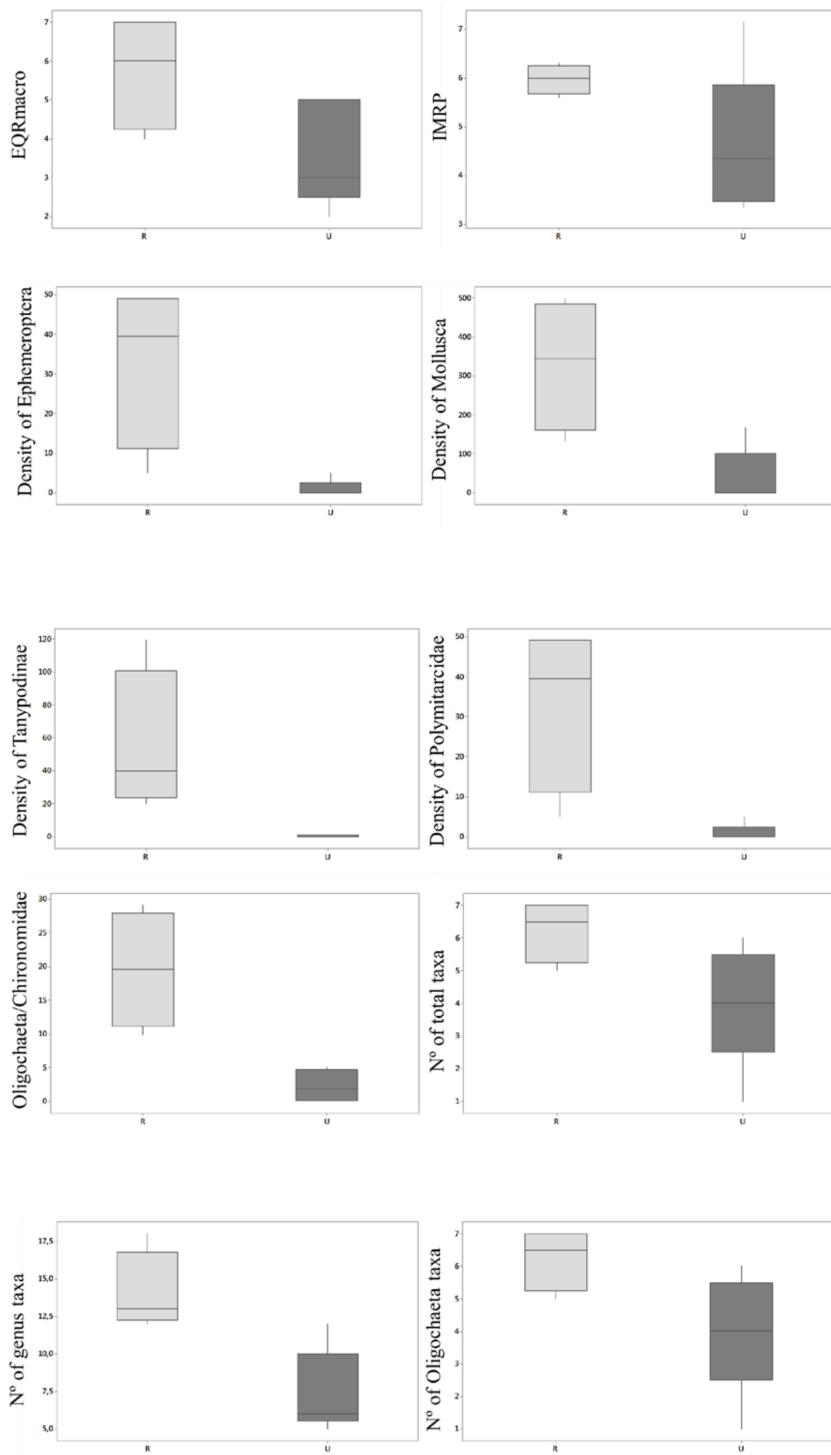


Figure 2

Distribution of values for candidate metrics of invertebrates with high sensitivity in the reference (R) and urban (U) wetlands during periods of high water. The range bars show the maximum and minimum of non-outlier data; the boxes are interquartile ranges (25th percentile to 75th percentile) and the dividing line is medians.

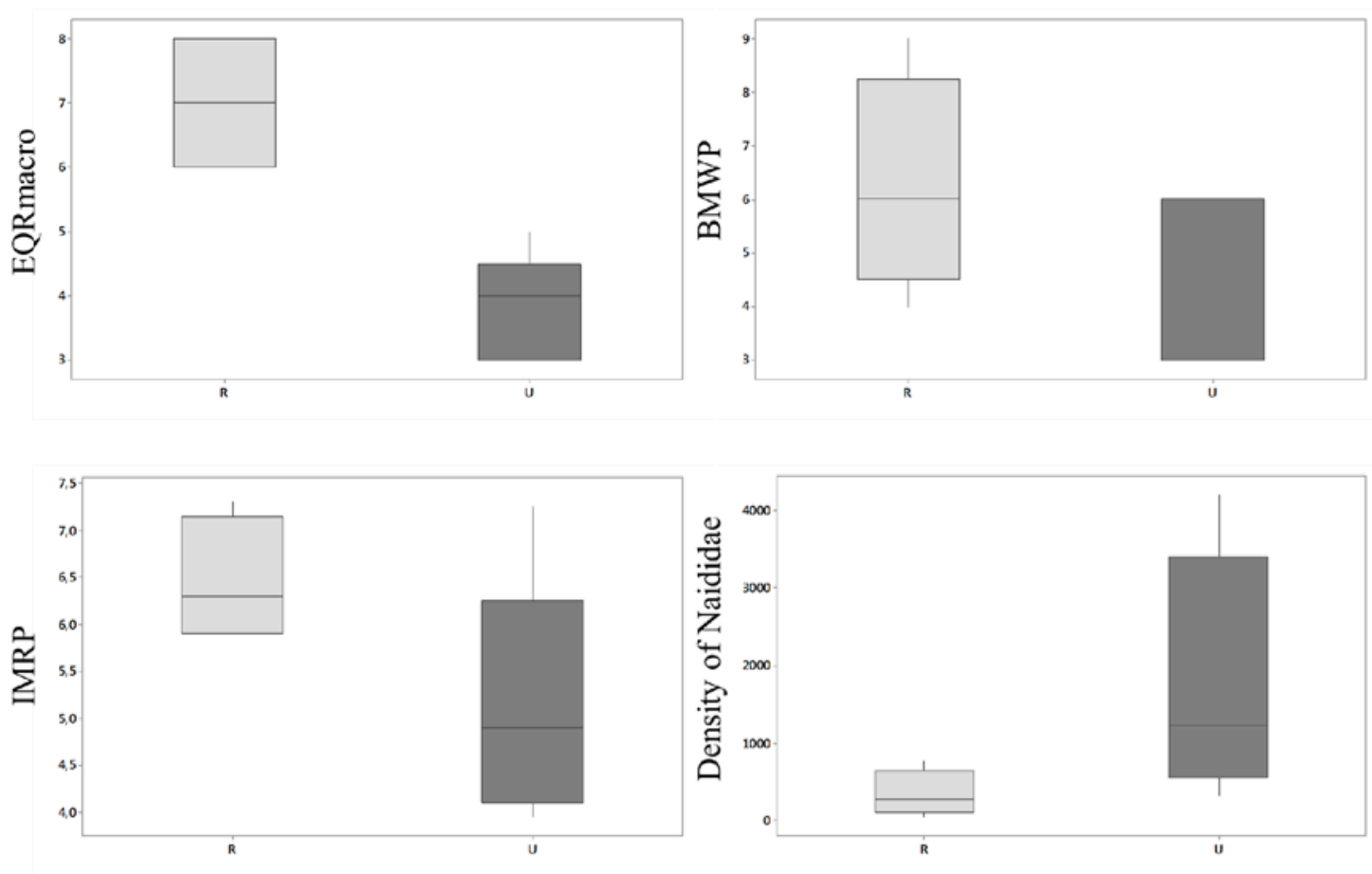


Figure 3

Distribution of values for candidate metrics of invertebrates with high sensitivity in the reference (R) and urban (U) wetlands during periods of low water. The range bars show the maximum and minimum of non-outlier data; the boxes are interquartile ranges (25th percentile to 75th percentile) and the line dividing the bars are medians.

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