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Nutrient levels, trophic status and land-use influences on streams, rivers and lakes in a protected floodplain of Uruguay

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ABSTRACT

Land-use intensification impacts freshwater ecosystems in many agricultural landscapes, including the Rio de la Plata grasslands in Southeastern South America. We evaluated water chemistry of rivers, streams and lakes associated with a protected river-floodplain in the Queguay River basin in Uruguay. From 2019–2021, we measured phosphorus (P) and nitrogen (N) forms, dissolved oxygen (DO), turbidity and other chemical parameters; basin-scale land-use via Sentinel satellite imagery; and trophic status based on nutrient concentrations and Principal Components Analysis (PCA). Results highlighted low nutrient concentrations in the Queguay rivers, indicating this watershed as a national reference for evaluating eutrophication in Uruguay. Levels of total nitrogen (TN) in rivers averaged 430–510 μg/L and total phosphorus (TP) averaged 44–51 μg/L, classifying them as oligotrophic-mesotrophic systems. Streams ranged in TP from 51 to 97 μg/L and in TN from 545 to 701 μg/L, suggesting meso- to eutrophic states according to TP levels, and potential anthropogenic eutrophication in smaller basins with crop cover > 25%. Floodplain lakes are a conservation target within the protected area and had TP averaging 74–109 μg/L, suggesting eutrophic-hypereutrophic states based on TP and potential external nutrient inputs. The positive correlation between basin-scale crop cover and mean TP and soluble reactive phosphorus (SRP) concentrations suggested that land-cover plays a role in nutrient levels of fluvial and lotic systems. This study establishes a baseline for water chemistry in lakes and fluvial ecosystems in the Queguay River basin within an extensive agricultural region of the La Plata River Grasslands. By providing the first time series dataset of water chemistry for this area, we fill a major geographical gap regarding the limnological attributes of freshwater ecosystems in northwestern Uruguay. Results highlight the value of the Queguay basin for freshwater habitat conservation, and the vulnerability of these systems to land cover changes and nutrient loading from external sources.

1. Introduction

Freshwater ecosystems in South American river basins comprise a network of aquatic habitats (Ward, 1998; Ward et al., 1999), that together support the highest diversity of freshwater fauna on earth (Reis et al., 2016). These environments are subject to multiple anthropogenic impacts, including agricultural intensification, land-use conversion and hydrological modification, leading to declines in water quality and populations of aquatic species, as well as biodiversity loss (Coutinho et al., 2009; Salazar et al., 2015; Schneider et al., 2017). Many of these land-use activities are associated with increased nutrient loading in aquatic environments, thus potentially leading to processes of eutrophication (Allan, 2004; Arbuckle and Downing, 2001; Jordan et al., 1997). As nitrogen and phosphorus affect aquatic biomass and primary

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productivity, understanding the relative contributions of these essential nutrients is a key for evaluating potential threats to the structure and function of freshwater ecosystems (Hecky and Kilham, 1988; Vollenweider, 1968).

Trophic status is a fundamental property of aquatic ecosystems, inextricably linked to the biotic integrity and chemistry of lakes and streams (Dodds and Cole, 2007). Incorporating nutrient concentrations and other physical or biological parameters, trophic status serves as an indicator of eutrophication, classifying aquatic systems according to biomass (Brugnoli et al., 2019; Chalar et al., 2011; Dodds and Cole, 2007; Pacheco et al., 2012). The trophic status of freshwater systems can be susceptible to the impacts of agricultural intensification and land-use conversion (Coutinho et al., 2009; McDowell et al., 2004), ultimately disturbing aquatic biodiversity and ecosystem productivity. Changes in trophic status – related to changes in temperature, dissolved oxygen, and turbidity – have direct impacts on habitat quality for aquatic biodiversity, as well as reproduction, survival and growth at multiple life stages (Pusey and Arthington, 2003). While there is substantial data to infer the trophic status of temperate systems, more information on water chemistry and trophic status of tropical and subtropical freshwater ecosystems is required to understand the ecology of these systems (Cunha et al., 2013, 2021; Huszar et al., 2006).

The La Plata Grasslands is one of the most extensive areas of natural grasslands in the world over at 700,000 km^2 (Soriano, 1991), and is currently experiencing distinct changes in land-cover from grassland to intensive agriculture and afforestation (Baeza and Paruelo, 2020). Within this regional agricultural landscape, eutrophication is a threat to water quality (Alonso et al., 2019; Chalar, 2009), as evidenced by large blooms of toxic cyanobacteria in the principal rivers, reservoirs, and coastal lagoons (Bonilla et al., 2015; de la Escalera et al., 2017; González-Piana et al., 2011; Vidal and Britos, 2012). Recent large-scale algal blooms in the la Plata estuary, initiated by processes of eutrophication upstream, exacerbated by increased water residence time, rainfall and flooding (Aubriot et al., 2020; Kruk et al., 2021), highlight the urgent need for understanding nutrient dynamics in major tributaries, lakes and manmade reservoirs. Pressures on these aquatic systems associated with agriculture and land-use conversion, include nutrient-loading, sedimentation and carbon inputs, coupled with climate variability, and alterations in both trophic dynamics and diversity (Barreto et al., 2017; Chalar et al., 2017; Goyenola et al., 2015; Graeber et al., 2015). Floodplain lakes contribute to the overall biodiversity in riparian floodplain landscapes (Saunders et al., 2002), but can be particularly susceptible to nutrient loading in South American riparian ecosystems (Agostinho et al., 2018; Thomaz et al., 2007). In particular, increases in total phosphorus are associated with the conversion of native grasslands to intensive agriculture (Gorgoglione et al., 2020; Goyenola et al., 2015), as well as urban inputs (Alvareda et al., 2020).

We introduce the first baseline of water chemistry and land-use data in fluvial and lacustrine systems in a large, protected freshwater floodplain of the Queguay River Basin - a watershed that plays an important role for conserving biodiversity and protecting aquatic resources within the landscape of the La Plata grassland region of Southeastern South America. We evaluate 1) water chemistry and nutrient concentrations in fluvial and lake systems over time, 2) gradients of trophic status of streams and marginal lakes and a discrimination of natural vs. anthropogenic eutrophication, and 3) potential threats of upstream land-use activities to water chemistry of fluvial and lake systems designated as conservation targets in a protected area. We used the following water chemistry parameters as indicators for habitat quality, including dissolved oxygen, saturation, turbidity, electrical conductivity, total dissolved solids, oxidation-reduction potential, nitrate, ammonium, soluble reduced phosphate, total nitrogen total phosphorus, and total N:P over two years in a network of streams and lakes of the Queguay River Forest Reserve in Uruguay. We classified land-use into seven categories to associate water chemistry with land cover at basin scale. Finally, we used nutrient levels and multivariate analysis to identify trophic gradients and evaluate potential natural and anthropogenic drivers of eutrophication.

2. Material and methods

2.1. Study system

The Pampa biome in Southeastern South America is characterized by extensive grasslands and land-use conversion of grassland or prairie to crops and afforestation (Fig. 1; Brazeiro et al., 2020; de Oliveira et al., 2017). Within this biome, the Queguay River Basin, extending 8551 km^2, is a tributary of the lower Uruguay River and holds one of Uruguay's largest riparian forest areas, "*Montes del Queguay*" [translated here as: Queguay River Forest Reserve], integrated into the National System of Protected Areas (SNAP) in 2014. The confluence of two large rivers and multiple streams form a freshwater floodplain that constitutes a Protected Area (PA) of 19.969 km^2. The Queguay Basin provides habitat for at least 98 fish species (Paullier et al., 2019), and the recent description of a new annual species *Austrolebias queguay* Serra, Loureiro, based on the holotype specimen from the Queguay wetlands (Serra and Loureiro, 2018) suggests the area's importance for aquatic species diversity. The PA is surrounded by a matrix of land-uses, including *Eucalyptus* spp. plantations, soybean, and other crops, as well as grasslands used as pasture. The PA aims to protect fluvial ecosystems and lakes, for which increasing nutrient levels from agrochemicals has been reported as a potential threat to these systems (DINAMA, 2012).

We selected as fluvial sites, the six largest watersheds that fed directly into the PA: two rivers, the Queguay Grande (QAP) and Queguay Chico (QCH), and four streams, Buricayupí (BUR), Guayabo Grande (GGR), Ñacurutú (NAC) and Santana (SAN; Table 1, Fig. 1). We also sampled 5.4 km upstream of SAN at Santana – Rta. 4 (SAN4), immediately upstream of a 5.0575 ha afforestation nursery that produced 15 million seedlings in 2019 (UPM 2020), to control for potential point sources that could confound land-use impacts from diffuse sources of nutrients.

Lakes were identified using the national Geographic Information System (GIS) layer for open waters (MVOTMA, 2019) and local expertize of authors. Their geomorphometric form and location in the floodplain suggest that some of these lakes are abandoned meanders of the main river channel (Ward et al., 1999). We chose the term "marginal lakes", as used for those in the Paraná River basin (Roberto et al., 2009; Thomaz et al., 2007). Given their episodic connectivity with the floodplain multiple times a year during flooding events, these marginal lakes may be classified as plesiopotomal habitats (Ward et al., 1999). We identified a total of 22 permanent lakes in the PA that ranged in size from 0.5 to 14.6 ha, of which four of them were selected based on accessibility, area, and conservation and management value (Table 2). They included the three largest lakes in the PA in terms of surface area (LAG1, LAG2 and LAG3), and a < 1 ha lake (LAG4) isolated from the hydrological network within an extensive forested area in the PA (Fig. 1). Area and perimeter were calculated from Google Earth images, and thus may be underestimated due to canopy cover (Table 2). Shoreline development (D) described morphometric differences among lakes, where P = lake perimeter and A = lake area (Table 1).

$$\text{Shoreline development}: D = P/(2\sqrt{\pi A})(\text{Aronow1982}) \quad (1)$$

For fluvial sites, basin form factor, where L = basin length and A = basin area was used to describe morphometric parameters of the study watersheds (Table 1).

$$\text{Basin Form Factor: } A/(L^2) \quad (2)$$

Drainage basins for all sites were delineated using Digital Elevation Models ASTER GDEM 2 (NASA/METI/AIST/Japan Spacesystems [GIS layer] and U.S./Japan ASTER Science Team, 2001). Lake drainage

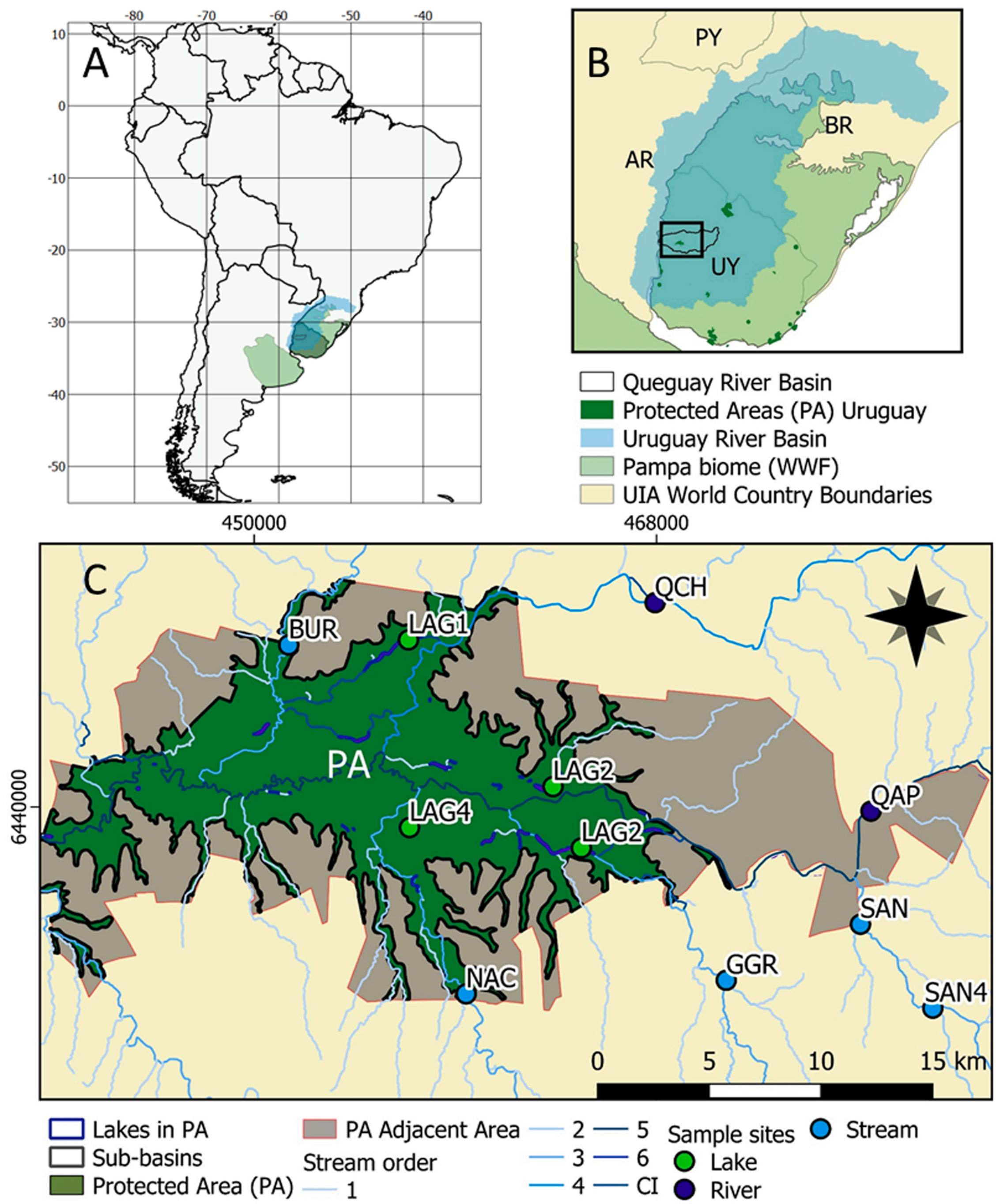

Fig. 1. Map of A) South America, B) study region in the Queguay River Basin, within the Uruguay River Basin. C) Water chemistry sampling sites (colored by type; labeled by code name) in the Protected Area (green) and official "Adjacent Area" (gray) in the Queguay River Basin.

Table 1
Site characteristics of sampled streams and rivers in the Queguay River Basin, where E = Elevation *sensu* DEM, A = Basin Area (km^2), L = total length, W = width of basin and Rf = Form factor. In site names, R = River, Ao = Arroyo (Stream).

Sitio	*Code*	*E (masl)*	*A (km^2)*	*L (km)*	*W_{max} (km)*	*Rf*	*Depth (m)*	*Stream Order*
Streams & Rivers								
Ao. Buricayupi	**BUR**	33.0	425	42	18	1.31	0.80	3
Ao. Guayabo Gr.	**GGR**	40.9	287	34	19	0.80	0.35	3
Ao. Nacurutu	**NAC**	40.8	248	32	12	1.72	0.65	3
Ao. Santana	**SAN**	41.9	235	27	18	0.73	0.50	3
Ao. Santana – Rt. 4	**SAN4**	52.0	172	18	15	0.53	2.5	3
R. Queguay Gr.	**QAP**	35.3	3277	122	49	1.36	1.65	5
R. Queguay Ch.	**QCH**	40.4	1385	91	34	1.20	1.0	4

Table 2
Site characteristics of marginal lakes, where E = Elevation, P = Perimeter, L = length, CO = Cardinal orientation, W_{max} = maximum width and D_L = shoreline development. Depth (m) to the nearest 0.05 m in streams and lakes was the average maximum depth. Number of fluvial inputs is the number of inlets/outlets that connect to the lakes *sensu* the national GIS layer for water courses in Uruguay.

Lakes	***Code***	*E (m)*	*Lake area (ha)*	*Area basin (km^2)*	*P (km)*	*L (km)*	*CO*	*W_{max} (m)*	*D_L*	*Depth (m)**	*No. fluvial inputs*
L. Amarilla	**LAG1**	33.0	14.0	1793	6.61	3.56	SW-NE	102	4.00	4	2
L. del Amarillo	**LAG2**	35.7	13.8	52	3.75	2.18	E-W	192	2.94	4.2	1
L. del Burro	**LAG3**	34.4	14.7	35	6.39	3.72	E-W	91	3.82	4	3
L. Agua dulce	**LAG4**	34.9	0.91	4003	1.12	0.66	SW-NE	31	3.34	2	0

basins were drawn based on the fluvial inputs directly into the lake, except for LAG4, which due to its isolation from the fluvial network, was considered to hold a drainage basin that comprised all rivers and streams flowing into the Queguay River parallel to the lake (Supplementary Material). Ecoregions of Uruguay were obtained from the national ecoregion GIS layer (Brazeiro et al., 2015).

2.2. Water chemistry monitoring

We monitored 14 physicochemical parameters every 2–3 months from Jan-2019 to Feb-2021 in streams and rivers, and from Mar-2019 to Feb-2021 in lakes. Temperature (T), pH, turbidity (NTU), electrical conductivity (Cond), Oxidation-Reduction Potential (ORP), dissolved oxygen (DO), oxygen saturation (SO2), and total dissolved solids (TDS), were measured with a multiparametric sensor HORIBA U-50. For nitrate (NO3), ammonium (NH4), total nitrogen (TN), total phosphorus (TP), soluble reactive phosphorus (SRP) and TN:TP ratios we collected 1 L water samples at each site. Water chemistry sampling in all four lakes was conducted within the top 0.50 cm of the water column in the center of each lake using a kayak to avoid edge effects. Samples were preserved at < 4 °C and sent to the Limnology Laboratory, Facultad de Ciencias, Universidad de la República - Montevideo, Uruguay, for analysis. Dissolved nutrients were analytically determined from filtered samples using Munktell Micro-glass fiber filter MGF. SRP was estimated using the technique described by Murphy and Riley (1962), NH4 following Koroleff (1970), and NO3 by the method of Müller and Wiedemann (1955). TN and TP were determined using unfiltered water samples by the simultaneous analytical technique of Valderrama (1981).

Trophic status is the rate of carbon input into these systems and is related to nitrogen (N) and phosphorus (P) input rates (Schindler, 1977; Smith, 1998; Vollenweider, 1968). We evaluated site trophic status based on TN and TP concentrations (Dodds et al., 1998; Nürnberg, 1996; Smith, 1998), using TN:TP as a reference for P- vs. N-limited systems. We were unable to measure chlorophyll-a. Ordination analysis was performed to determine the relative distribution of lake and fluvial sites along a trophic gradient (Primpas et al., 2010). Given the lack of biological parameters (e.g., chlorophyll-a), we evaluated streams and rivers along a gradient from oligo-mesotrophic to eutrophic, and lakes along a gradient from oligotrophic to hypertrophic, using nitrogen and phosphorus concentrations as well as Principal Component Analysis (PCA; Knoll et al., 2015).

To contextualize these results on a national scale, we compared mean nutrient levels in Queguay rivers and streams to those of other agricultural watersheds in Uruguay. The Negro River basin is the largest tributary to the Uruguay River, with a total area of 68,000 km^2, that contains three hydroelectric dams. The Negro River is a national focus regarding water quality and quantity (Aubriot et al., 2020), and the impacts of afforestation on streamflow (Silveira et al., 2016). Water chemistry sampling in Negro River sites had also been conducted with HORIBA U-50 and YSI multiparametric sensors and similar analytic methods for nutrient concentrations in the same laboratory (Chalar et al., 2015). Thus, we considered mean values for these watersheds highly comparable, despite potential differences in ecoregion, soils, and geology. Nutrient concentrations for the Cuareim and Santa Lucia River Basins were obtained from the publicly available dataset of the Uruguayan Ministry of Environment (OAN-MA, 2021). The Santa Lucia is a national focus for water chemistry monitoring (Barreto et al., 2017; Gorgoglione et al., 2020). The Cuareim River is an agricultural watershed within the same Basalt ecoregion as the upper Queguay River Basin.

2.3. Land use land cover (LULC) classification

A Land-use Land cover (LULC) Map was made using a supervised classification of Sentinel 2 MultiSpectral Instrument (S2/MSI) images. We selected S2 Level-2A Surface Reflectance images between two periods (18-Sep-2019–06-Oct-2019 for austral spring, and 26-Jan-2020–05-Feb-2020 for summer) to capture phenological differences among LULC classes. For each period we constructed a new image computing the median value of each band based on all available images with $< 1\%$ cloud cover. We used all reflective bands ($N = 10$; Supplementary Material, Table A1) and computed two additional indexes for vegetation function, i.e., the Normalized Difference Vegetation Index (Rouse et al., 1974) and the Water Index (Gao, 1996). All sensor bands with spatial resolution of 10×10 m were resampled at 20×20 m. The LULC classification discriminated nine categories: Winter crops, Summer crops, Double crops, Afforestation, Native forest, Natural grassland, Bare soil, Wetlands, and Water (including natural and artificial water bodies). All crops were lumped, resulting in seven LULC classes (Supplementary Material, Appendix A).

Based on visual interpretation of images from both periods and the support of high-resolution images, at least 40 polygons per LULC category were selected. In each LULC category we randomly selected 70% of polygons for training classification algorithms and the remaining 30% to evaluate classification processes. Additionally, inside each polygon subset, we randomly selected 500 and 200 pixels per category for training and evaluation process respectively. We performed a supervised classification with a random forest classifier (Breiman, 2001) using the selected pixels of training subsets and values of all 24 bands (10 reflective bands plus two indexes per period) as predictor variables. Random forest classification was performed with 50 iterations and seven variables per tree split. The evaluation process was performed with a contingency matrix between classification results and the reserved evaluation subset and computed overall, user and producer accuracy. All S2/MSI images were obtained from the Google Earth Engine (GEE) platform (Gorelick et al., 2017). All remote sensing analyses were performed in GEE; all maps were created in QGIS v. 3.14 (QGIS Development Team, 2020). LULC classifications during the 2019–2020 water chemistry survey were then used to evaluate differences in trophic status among sites and discriminate sources of natural and anthropogenic eutrophication.

2.4. Statistical analyses

To compare mean water chemistry parameters among lakes, rivers and streams, we used nonparametric Kruskal-Wallis (KW) tests and *posthoc* Dunn tests in the *PMCMCplus* package (Pohlert, 2021). To evaluate differences in variability by system type, we performed the nonparametric Fligner-Killeen test for equal variances. A correlation matrix was constructed to select independent variables. The PCA, comprising nine chemical parameters (SO2, PH, ORP, Cond, NO3, NH4, TN, TP), was conducted using the *prcomp* command in base R language (R Core Team, 2020). These parameters are recommended by the Mediterranean eutrophication monitoring program (UNEP, 2003), with biological parameters not measured here, chlorophyll-a and phytoplankton (Kitsiou and Karydis, 2011). We used linear regression to evaluate the relationship between percentage crop cover and site-based averages for nine water chemistry parameters. All statistical analyses were conducted in R version 4.0.2 (R Core Team, 2020).

3. Results

3.1. Water chemistry

Characteristic of lentic systems, lakes showed lower DO than rivers and streams and lower electrical conductivity in comparison to streams (KW values in Table 3, Fig. 2). DO levels in rivers averaged 8.7 ± 1.4 mg/L, while in lakes OD was 5.0 ± 2.3 mg/L at the surface (Table 4, Fig. 2). Lakes were more turbid than streams, but not rivers, with an NTU of 28.0 ± 26 influenced by the high turbidity of the smaller and shallower LAG4 (NTU 62 ± 27, Fig. 3). Lakes exhibited lower electrical conductivity (174 ± 47 μS/cm), as well as lower TDS (113 ± 31 mg/L) compared to streams and rivers (Table 4, Fig. 3). The ORP of lakes was lower than that of streams and rivers (KW = 7.3, p = 0.007), showing that fluvial systems held a higher oxidative potential than lakes.

Total phosphorus concentrations were, on average, higher in lake systems at 85 ± 26 μg/L than rivers at 47 ± 7 μg/L and streams at 75 ± 29 μg/L, influenced by high levels of TP > 100 μg/L in LAG3. LAG2 and 3, which were fed by small streams with > 25% crop cover, showed relatively high average TP (Table 5). LAG3, which was highly connected to the fluvial network and fed directly by the del Burro Stream, showed the highest levels of TP and SRP, the lowest levels of NH4 and the lowest TN:TP ratio among all lakes. In contrast, LAG1 had relatively low crop cover and relatively low PT of 74 ± 22 μg/L (Table 5). The isolated lake (LAG4) displayed the highest levels of TN and NTU, the lowest levels of TP and the highest TN:TP ratios among all lakes (Table 5). Among all sites, the lowest average levels of PT (44–51 μg/L) were observed in the two river sites, QCH and QAP, which influenced the overall averages among habitat type. Lakes were higher in TN than rivers (602 ± 110 vs 474 ± 124 μg/L), but on average slightly lower in TN than streams (625 ± 166 μg/L). We found exceptionally lower TN values in QAP and QCH of 433 μg/L and 510 μg/L, respectively, and higher TN in the isolated LAG4 surrounded by native forest. TN among streams and lakes LAG1, 2 and 3 were similar (Table 5). Lakes and streams had lower TN:TP (more N-limited) than rivers (p < 0.05, KW tests in Table 3). Among streams, higher NO3 concentrations were observed in BUR, SAN and SAN4 (Fig. 3); while high SRP and TP concentrations were observed only in SAN and BUR, reaching levels TP > 150 μg/L in Dec.–2019 and Jun-2020, respectively (Supplementary Material, Fig. 2B.).

The variability of nutrients in these systems reflected the spatial and temporal dynamics of nutrient concentrations and their potential relationship with basin-scale land cover (Fig. 5). Regarding temporal variability, fluvial systems were more variable than lakes in terms of TDS, Conductivity, Turbidity, TN, and Nitrate (Fig. 4, Table 3). The absolute lowest TN values were 295 μg/L in QAP, meanwhile maximum values > 1000 μg/L were observed in LAG2 (1121 μg/L) and streams GGR (1055 μg/L) and SAN (1055 μg/L). TP values varied from a minimum of 29.2 μg/L in QAP to 168 μg/L in LAG2 and maxima of 156–160 μg/L in streams BUR and SAN as well as in LAG3 (Fig. 4).

3.2. Land use land cover (LULC) classification

The most modified sub-basins in terms of LULC were streams NAC and GGR, with > 50% land cover representing crops and forest plantations (Fig. 5). The basins of the rivers, QCH and QAP, were the least transformed basins feeding into the PA, with a natural grassland cover of 80% in both basins (Fig. 5). The drainage basin of lake fluvial inlets, LAG2 and 3 were directly fed by small watersheds of 51.8 km^2 and 35.3 km^2, respectively, with 28–48% crop cover and 7–12% afforestation. In contrast, LAG1 was directly fed by two inputs from a relatively large basin (1792 km^2) comprising 70% natural grassland and 14% crop cover, including the entire Queguay Chico River watershed. LAG4 was an isolated lake in the central Queguay River floodplain, thus the watershed was delineated as the Queguay River basin from this point upstream, as a source of external inputs during flooding. This large watershed also exhibited relatively low crop cover 14% and high grassland cover (70%), encompassing the basins QCH and QAP. The contingency matrix showed good results for the classification process, with an overall accuracy of 96.9%, with low and equally distributed commission and omission errors (Supplementary Material, Tables A2 and A3). Percentage crop cover was correlated positively with increases in average TP and SRP ($F = 5.6$, $p = 0.042$ and $F = 5.2$, $p = 0.049$, respectively; Fig. 6).

3.3. Trophic status

We considered both N and P as potentially limiting nutrients of trophic status, given a range of average TN:TP of 6.7–13.8 among sites, except for the N-limited LAG4 (TN:TP = 5.5). There was a positive correlation between TP and SRP ($r = 0.86$, $p < 0.001$) and Cond and TDS ($r = 1.00$, $p < 0.001$), thus SRP and TDS were not included in the PCA. The first two principal components accounted for 69.7% of the variance (Fig. 7). Along the first axis there was loading of TP (0.39), crop cover (0.44), grassland cover (−0.44) and NO3 (−0.35), where more oligo-mesotrophic states were observed for large rivers, QAP and QCH, with low TN and SRP and larger relative grassland cover. Mean nutrient levels in QAP and QCH suggested oligo-mesotrophic conditions (Table 5). Eutrophic conditions were represented by LAG2 and LAG3, both displaying higher TN and SRP, and a relatively higher percentage of crop cover (Fig. 5). These conditions correspond to eutrophic conditions according to suggested boundaries for TP > 75 μg/L in streams and TP > 30 μg/L in lakes (Smith et al., 1999) but could be classified as oligotrophic according to NT > 700 μg/L (Table 5). LAG 1 was located on one end of the trophic gradient, corresponding to lower SRP, lower TN and higher TN:TP ratio among lakes (Fig. 7). The higher levels of TN in the isolated LAG4, located at the bottom-center of the PCA was a potential indicator of natural eutrophication. Along the vertical axis (Dim2), streams GGR, NAC, and SAN were associated with higher electrical conductivity and higher SO2, and, at the opposite extreme, LAG4 with high turbidity (NTU) and higher NH4 (Fig. 7, PCA Loadings in Supplementary Material, Table C1).

3.4. Regional comparison

Comparison of site-specific mean nutrient concentrations with

Table 3

Tests for differences in mean (Kruskal-Wallis – KW) and variance (Fligner-Killeen – FK) of water chemistry parameters among system type (rivers, streams and lagoons) over the study period. Significant differences indicated as, – p < 0.1, * p < 0.05, ** p < 0.01, *** p < 0.001.

Test	*T*	*SO2*	*DO*	*NTU*	*Cond*	*ORP*	*TDS*	*pH*	*NO3*	*NH4*	*SRP*	*TN*	*TP*	*TN:TP*
KW	0.001	**35.7** ***	**27.0** ***	**8.2** *	**52.4** ***	**10.5** **	**36.9** ***	3.7	**8.0** *	2.3	**15.1** ***	**15.0** ***	**28.6** ***	**10.0** **
FK	0.07	5.5·	3.6	**6.9** *	**15.9** ***	3.7	**17.3** ***	0.88	3.7	1.1	**12.2** **	4.4	**20.6** ***	0.28

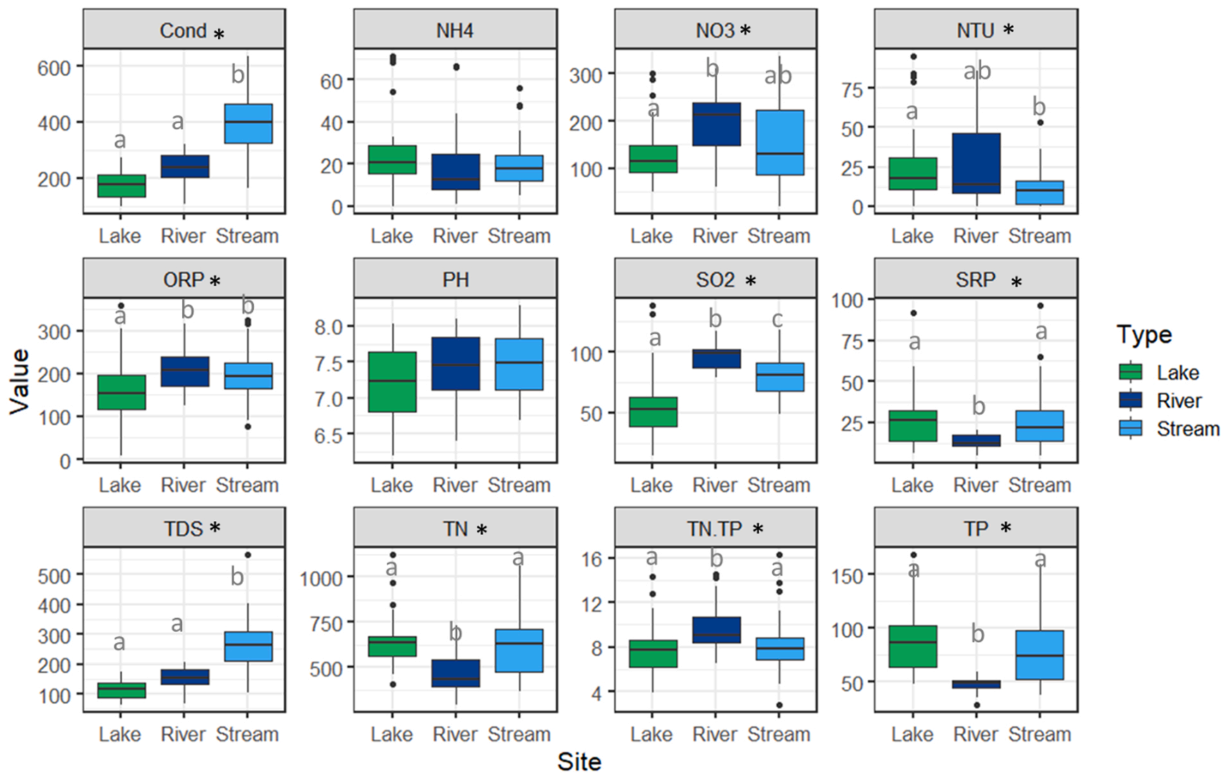

Fig. 2. Box-whisker plot comparing chemical parameters, excluding T and DO, in lakes (n = 4), rivers (n = 2) and streams (n = 4). Units for all nutrient forms in μg/L; Cond in μS/cm; ORP in mV as in Tables 4 and 5. Asterisks indicate significant differences according to Kruskal-Wallis tests ($p < 0.05$); letters indicate significant differences between Type *sensu* Dunn tests ($p < 0.05$).

Table 4
Mean and SD values for water chemistry parameters, Temperature (°Celsius), Dissolved oxygen (mg/L), Percent saturation (SO2), pH, Oxidation Reduction Potential (ORP), Conductivity (μS/cm), Total Dissolved Solids (TDS mg/L), and Turbidity (NTU) in each study site.

Site	*T (°C)*	*DO mg/L*	*SO2 (%)*	*pH*	*ORP mV*	*Cond μS/cm*	*TDS mg/L*	*NTU*
Streams								
BUR	19.9 ± 6.1	7.6 ± 1.6	85 ± 10	7.60 ± 0.31	197 ± 58	334 ± 52	217 ± 34	18.4 ± 7.6
GGR	20.2 ± 7.4	7.1 ± 2.1	76 ± 15	7.57 ± 0.33	193 ± 55	469 ± 144	300 ± 89	21.9 ± 13.8
NAC	18.8 ± 5.8	6.4 ± 2.4	68 ± 20	7.33 ± 0.44	176 ± 81	411 ± 72	267 ± 46	2.7 ± 3.8
SAN	19.0 ± 5.6	7.4 ± 1.6	80 ± 11	7.59 ± 0.55	185 ± 68	367 ± 98	233 ± 64	6.8 ± 9.2
SAN4	19.5 ± 5.8	6.7 ± 2.6	73 ± 23	7.57 ± 0.61	193 ± 41	390 ± 107	253 ± 70	12.2 ± 7.9
Rivers								
QAP	20.3 ± 5.7	8.5 ± 1.5	95 ± 10	7.45 ± 0.56	203 ± 45	203 ± 67	132 ± 44	25.2 ± 28.4
QCH	18.4 ± 5.0	8.7 ± 1.3	94 ± 8	7.48 ± 0.41	213 ± 48	256 ± 57	167 ± 37	20.8 ± 18.9
Marginal Lakes								
LAG1	20.8 ± 5.8	6.1 ± 2.7	70 ± 32	7.40 ± 0.41	154 ± 83	155 ± 40	101 ± 26	20.4 ± 7.8
LAG2	19.4 ± 5.5	4.5 ± 1.1	50 ± 10	7.22 ± 0.51	135 ± 86	214 ± 34	140 ± 22	21.1 ± 28.7
LAG3	18.2 ± 6.0	4.8 ± 2.9	54 ± 29	7.31 ± 0.34	115 ± 76	210 ± 19	137 ± 13	13.3 ± 9.5
LAG4	17.1 ± 5.8	4.2 ± 1.8	44 ± 18	7.26 ± 0.52	178 ± 87	123 ± 11	80 ± 6.5	58.2 ± 26.9

agricultural sub-basins in Uruguay showed that the QAP and QCH Rivers upstream of the PA displayed among the lowest values of PT, SRP, NH4 and NT (Table 5; Fig. 8). Given the size of the Negro River basins from approximately 1700 to 16,000 km^2, nutrient values were more appropriately compared with QCH and QAP (basin size ~1400–3200 km^2) rather than Queguay Basin streams (234–425 km^2). All TP values in the Rio Negro Basin, with the single exception of the SAL were higher than those of Queguay River sites, with TP means of 126–269 μg/L (Table 5). The Negro River sub-basins TN:TP ratios ranged from 4.3 to 6.6, except for SAL at 10.7 (Table 5). Within the same Basaltic ecoregion as the Upper Queguay and Salsipuedes Rivers, the Cuareim River sub-basins displayed mean SRP 2–4.5 times that of the QAP and QCH, except for the Tres Cruces Stream with levels below detectability in 2019–2021. All mean TP and TN in the Cuareim River Basins were greater than QAP and QCH, but TN:TP ratios were similar at values of 9–10 (Table 5). In the Santa Lucia Basin to the south, TP in rivers with drainage basins > 1000 km^2 was 2–13 times greater than TP in the QCH and QAP rivers. Similarly, SRP was 5.5–25 times greater in Santa Lucia rivers than in the Queguay Rivers studied here (Table 5).

4. Discussion

We provide an ecological baseline for water chemistry of freshwater lakes and fluvial systems in an agricultural region of subtropical southeastern South America. Given the regional conversion of grasslands to crops, afforestation and other intensive land-uses in the last four decades, the need for baseline values and reference systems is fundamental for regional and national planning. We report some of the lowest average TP and TN values for rivers in Uruguay, as well as demonstrate high nutrient concentrations in protected floodplain lakes. Results

Table 5
Nutrient concentrations (mean ± SD) and TN:TP in the Queguay, Negro, Santa Lucia, and Cuareim River Basins in Uruguay. Basin drainage area (Area in km^2) was estimated according to basin shapefiles (DINAGUA, 2020). Minimum mean values highlighted in bold. Trophic status (TS) *sensu* Smith et al. (1999) indicated as blue = oligotrophic, light green = meso, yellow = eutrophic, red = hypertrophic (lakes).

Water body	Code	Area (km^2)	NO3 (μg/l)	NH4 (μg/l)	TN (μg/l)	SRP (μg/l)	TP (μg/l)	TN:TP	TS x TP	TS x TN
Queguay Basin										
Ao. Buricayupí	*BUR*	425	156 ±101	25 ± 14	602 ± 118	24 ± 7	90 ± 13	6.7		
Ao. Guayabo Gr.	GGR	287	112 ± 58	22 ± 12	638 ± 195	20 ± 14	70 ± 20	9.2		
Ao. Nacurutú	NAC	247	114 ± 55	**15 ± 6**	545 ± 148	19 ± 10	66 ± 22	8.6		
Ao. Santana	SAN	234	190 ± 99	25 ± 25	701 ± 197	36 ± 17	97 ± 41	8.4		
Ao. Santana – Rt.4	SAN4	172	187 ± 96	27 ± 32	643 ± 136	14 ± 12	51 ± 20	13.8		
R. Queguay Gr. - AP	QAP	3227	170 ± 69	18 ± 19	**433 ± 124**	**13 ± 4**	**44 ± 7**	10.0		
R. Queguay Chico	QCH	1385	225 ± 53	20 ± 20	510 ± 117	**13 ± 5**	51 ± 5	10.1		
L. Amarilla	LAG1	1793	150 ± 73	19 ± 9	561 ± 127	20 ± 11	74 ± 22	8.2		
L. del Amarillo	LAG2	52	124 ± 74	31 ± 22	588 ± 100	23 ± 11	82 ± 25	7.4		
L. del Burro	LAG3	35	108 ± 34	16 ± 14	582 ± 70	44 ± 25	109 ± 27	5.5		
L. Agua Dulce	LAG4	4003	111 ± 25	28 ± 21	691 ± 101	22 ± 12	77 ± 21	9.5		
Negro Basin										
R. Tacuarembó (Lag)	TAC	16076	172 ± 78	40 ± 29	839 ± 307	64 ± 24	126 ± 57	6.6		
R. Yi (Bote)	YI	12715	184 ± 87	43 ± 39	755 ± 281	88 ± 33	134 ± 46	5.6		
R. Negro – (Pereira)	NEG	11.516	167 ± 73	53 ± 65	789 ± 250	76 ± 34	127 ± 42	6.2		
Ao Grande del Sur (Rt.3)	GRS	3294	340 ± 201	97 ± 208	1154 ± 328	233 ± 61	269 ± 51	4.3		
R. Salsipuedes (Rt.20)	SAL	1697	178 ± 167	25 ± 31	704 ± 279	31 ± 17	66 ± 38	10.7		
Santa Lucia Basin										
R. Santa Lucia Chico	SLC01	257	181 ± 133	84 ± 78	1170 ± 326	44 ± 17	87 ± 33	13.3		
Ao. Canelones Grande	CG02	319	459 ± 121	56 ± 27	1540 ± 147	589 ± 123	616 ± 168	2.5		
Santa Lucia	SL02	883	114 ± 11	39 ± 11	646 ±197	33 ± 27	72 ± 34	9.0		
Santa Lucia	SL01	1104	597 ± 434	94 ± 155	1004 ± 253	71 ± 37	106 ± 44	9.4		
R. Santa Lucia Chico	PS02	2570	517 ± 142	84 ± 75	1580 ± 367	327 ±52	672 ± 247	**2.4**		
R. San José	SJ04	3567	605 ± 240	74 ± 21	1423 ± 372	285 ± 110	317 ± 102	4.5		
R. Santa Lucia	SL04	5171	280 ± 130	32 ± 11	711 ± 402	121 ± 60	155 ± 59	4.6		
Cuareim Basin										
Ao. Tres Cruces	RC3C70	1473	**90, <LD**	45 ± 32	612 ± 312	**<LD**	67 ± 25	9.2		
R. Cuareim	RC50	8271	190 ± 43	75 ± 54	862 ± 250	24 ± 1*	94 ± 20	9.2		
Ao. Yucutuya	RCYU80	1077	137 ± 64	37 ± 10	902 ± 291	54 ± 26*	114 ± 40	7.9		

<LD = Lower than detectable
* Contains 1 value <LD, mean based on 5 values above detected levels.

suggest that external pressures from agricultural activities play an important role for nutrient levels in lakes and streams, particularly TP and SRP concentrations. The relatively low nutrient concentrations of the Queguay waterbodies were evident when compared to tributaries of the Negro River Basin, providing a reference for evaluating regional standards for water quality in the Lower La Plata River basins.

4.1. Nutrient concentrations in streams and rivers

The Queguay River study sites showed relatively low levels of all forms of nitrogen and phosphorus when compared to other agricultural watersheds in the region. For example, we detected low levels of NH4, SRP, TP and TN in the QAP and QCH Rivers, which drain over 4500 km^2 of what is largely native grasslands, the dominant native land cover of the Pampa biome (Baeza and Paruelo, 2020). These large rivers exhibited TN:TP values that suggest slight P-limitation, which is relatively uncommon for many other agricultural watersheds in the region. When comparing the nutrient levels of the Queguay Rivers to tributaries of the adjacent Negro River basins, we observed mean SRP levels of 13 μg/L, less than half of the lowest value reported for the Negro Basin in the reference Salsipuedes Stream. Similar results of exceptionally low nutrient concentrations in the Queguay Basin were also found for TP and TN (Table 5). Comparisons in nutrient concentrations can be confounded by differences in drainage area and geography, nonetheless, they provide a critical set of reference values on a regional scale to understand what can be considered acceptable levels of nutrients in agricultural watersheds. Water quality in the river and reservoirs within the Negro River is a national health concern as basins are affected by a

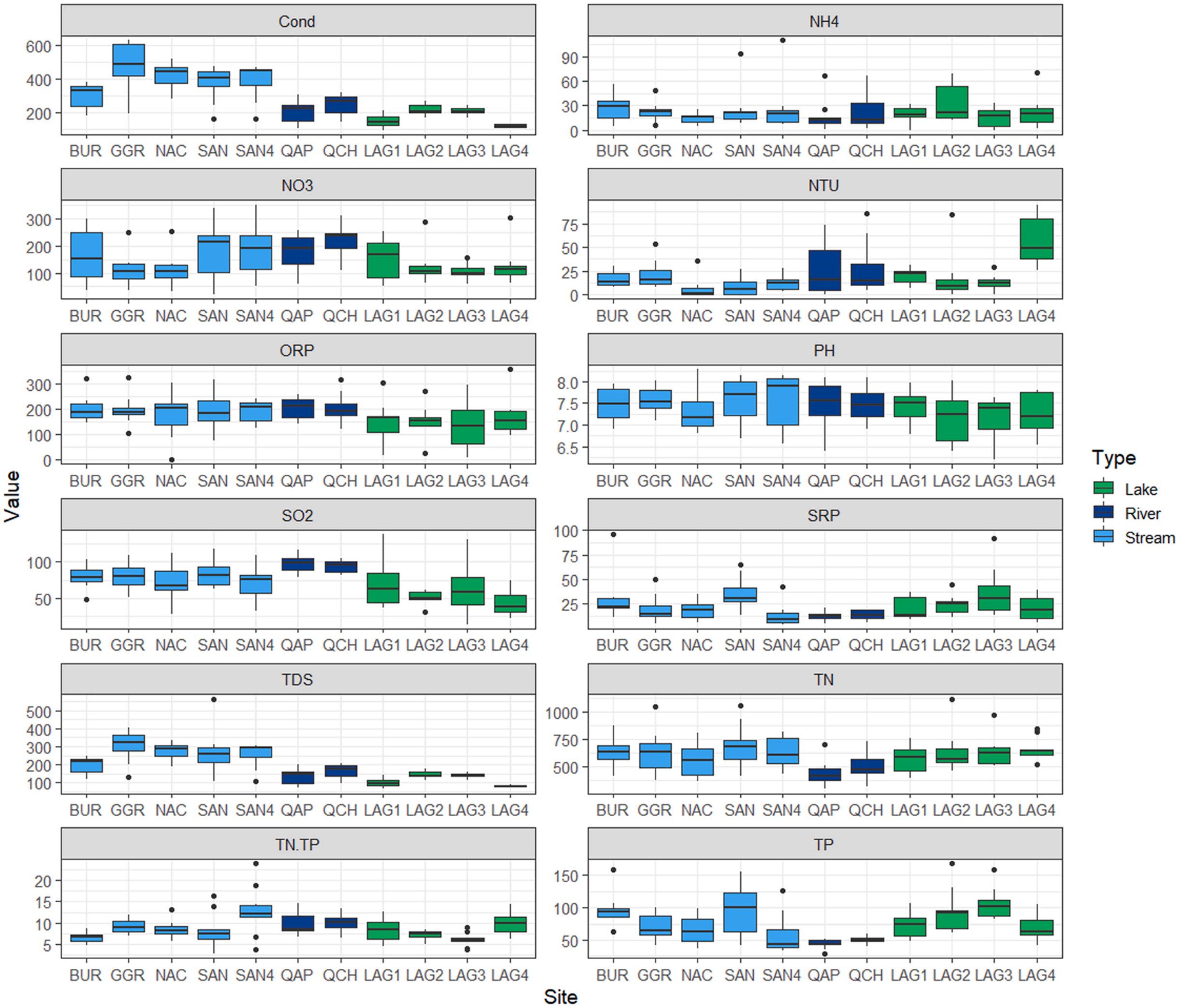

Fig. 3. Box-whisker plot comparing twelve chemical parameters in stream, river, and lake sites. Units for all nutrient forms in μg/L; Cond in μS/cm as in Tables 4 and 5.

combination of extensive livestock ranching, agriculture, and a growing afforestation program for *Eucalyptus* spp. tree plantations since 1997 (Silveira and Alonso, 2009). Considering the lack of large hydroelectric dams, afforestation, and intensive agriculture in the upper Queguay Grande and Queguay Chico rivers, these river-basin systems provide a critical reference for relatively low nutrient levels in low-impact agricultural watersheds in the northern Pampa biome.

The relative status and ecological value of aquatic habitats in the Queguay Basin is also illuminated when compared to more heavily impacted streams that dominate water chemistry research in the region. The Santa Lucia River provides drinking water for 60% of Uruguay's population (MVOTMA, 2018), and has some of the highest nutrient levels for agricultural watersheds in Uruguay (Chalar et al., 2011). Intensive phosphorus sampling over two years in micro-catchments (< 20 km^2) in the upper Santa Lucia Basin showed TP levels in low-impact basins with LULC grassland and low-intensity cattle grazing at of median of 100 μg P/L (Goyenola et al., 2015). This watershed is emblematic of some of the current problems with water quality in heavily populated areas (Barreto et al., 2017). In the largest tributary to the Uruguay River in Argentinean territory, the Gualeguaychú River Basin has relatively high levels of TP at a median of 80–320 μg/L in streams of comparable drainage area to those studied here (Juárez et al., 2016). These same streams have up to 15-fold the highest TN values for streams in the Queguay (Juárez et al., 2016). Authors cite the combination of land-use impacts together with the degradation of riparian vegetation could lead to relatively high nutrient levels in these agricultural watersheds (Juárez et al., 2016).

Currently, the national standards for permitted total phosphorus levels in Uruguayan fluvial systems establish a maximum of 25 μg/L in national waters designated for preservation of fish and other aquatic flora and fauna (DINAGUA, 1979). None of the Queguay sites displayed mean TP values below 44 μg/L, and the absolute minimum TP record was 29.2 μg/L. Currently, there is a proposal to raise the binational standard to 100 μg/L in the Lower Uruguay River between Uruguay and Argentina (CARU, 2019). Previous studies in the Lower Uruguay River largely focus on chemical and biological indicators of eutrophication in the main channel (Llorente et al., 2017; O'Farrell and Izaguirre, 2014), and in the 783 km^2 reservoir at the Salto Grande Dam at the Uruguay-Argentina border (Chalar, 2006, 2009; Chalar Marquisá et al., 2002; Conde et al., 1996; De León and Chalar, 2003). Downstream of the dam, the Uruguay River averages 77 μg/L TP, indicating mesotrophic to eutrophic status depending upon the season and river hydrology (Ferrari, 2020). Given an estimated 68% of total phosphorus inputs from agricultural sources (Chalar, 2006), as opposed to industry and urban runoff, quantification of phosphorus levels in major tributaries of the Lower Uruguay River is important for managing and mitigating impacts on aquatic systems and resources. Data from relatively low-impact tributaries of the Uruguay River Basin, such as the Queguay River,

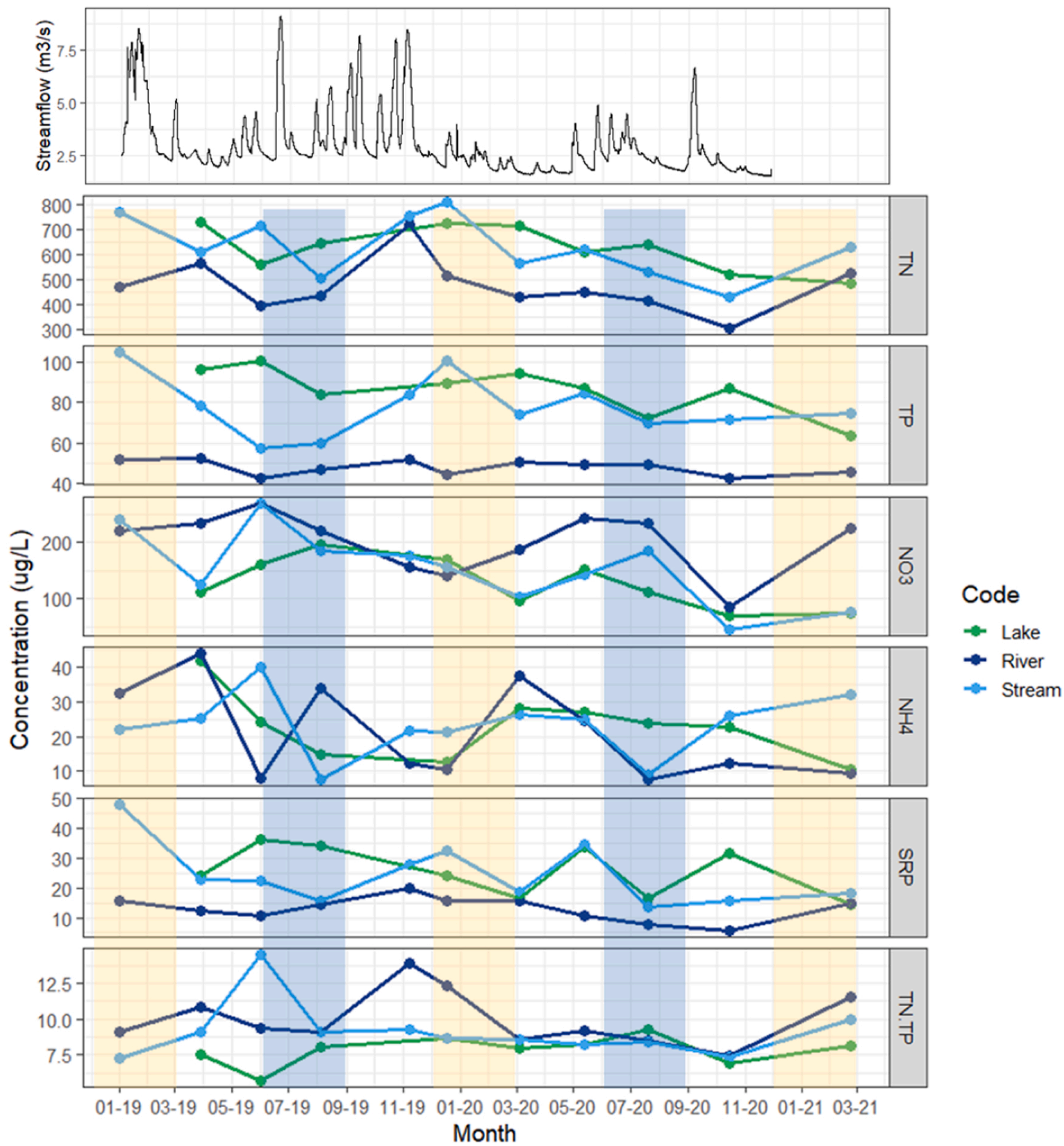

Fig. 4. Time series of mean nutrient concentrations and TN:TP for lakes, streams and rivers of the Queguay Basin, showing months of austral summer (yellow) and winter (blue), as well as daily streamflow downstream of the study area at Queguay – Rt. 3 (−32.1352°S; −57.9387°W).

will play a critical role in reevaluating adequate regional standards for water quality in this agricultural region.

4.2. *Potential sources of eutrophication in marginal lakes*

Marginal lakes in the riparian zone of the Queguay River are potentially susceptible to nutrient loading from both external land-uses as well as inherent properties of these lakes in this dynamic floodplain. All lakes were subject to potential sources of diffuse contamination by allochthonous sources of nutrients from agricultural sources, but LAG3 stood out from the others with the lowest TN:TP for all sites and TP and SRP almost double that of the "reference" lake, LAG4. LAG3 likely receives inputs of soluble phosphorus from upstream allochthonous sources outside the PA, including the Burro Stream Basin. In contrast, the high TN and low nitrate levels in LAG4 suggested high organic inputs from surrounding woody vegetation, and potentially lower turnover of ammonium to oxygenated forms of nitrogen. This lake could represent a later phase of the natural process of lake isolation and natural eutrophication due to the slow buildup of organic material. The potential anthropogenic eutrophication of lakes such as LAG3 is likely related to periodic connectivity to the floodplain during hydrological flood pulses throughout the year (Thomaz et al., 2007). Studies of marginal lakes in the Upper Parana River Basin show that isolated lakes exhibit high TN related to the accumulation of organic material (Roberto et al., 2009; Supplementary Material). This would also explain the relatively high TN levels in LAG4 as compared to all other sites. In small Parana River lakes, an increase in connection to the floodplain was associated with an increase in TP in lakes (Roberto et al., 2009). Phosphorous concentrations in marginal lakes vary over time with hydrology and connectivity to the main river (Bovo-Scomparin and Train, 2008; De Oliveira and Calheiros, 2000). We were unable to evaluate the timing and duration of lake connectivity to the Queguay Grande River, which likely plays a key role in the observed variability in nutrient levels over time.

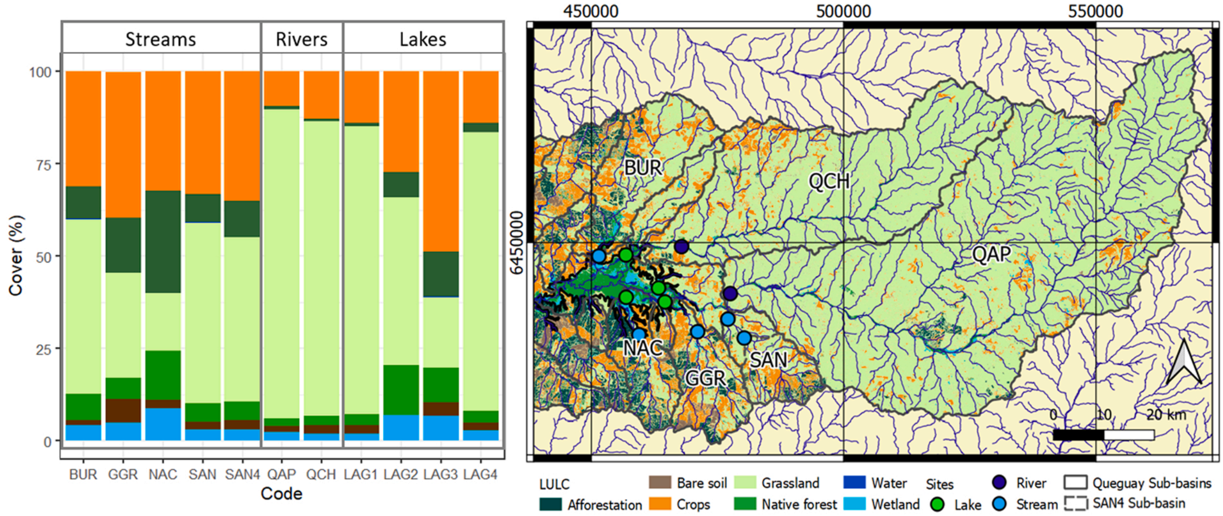

Fig. 5. A) Percentage cover for each of seven classes, based on the B) supervised LULC classification for 2019–2020 in the designated drainage basin of each sample site, listed in order of streams (BUR, GGR, NAC, SAN, SAN4), rivers (QAP, QCH) and lakes (LAG1 to 4). Water cover was < 1% in all basins. Values in Supplementary Material.

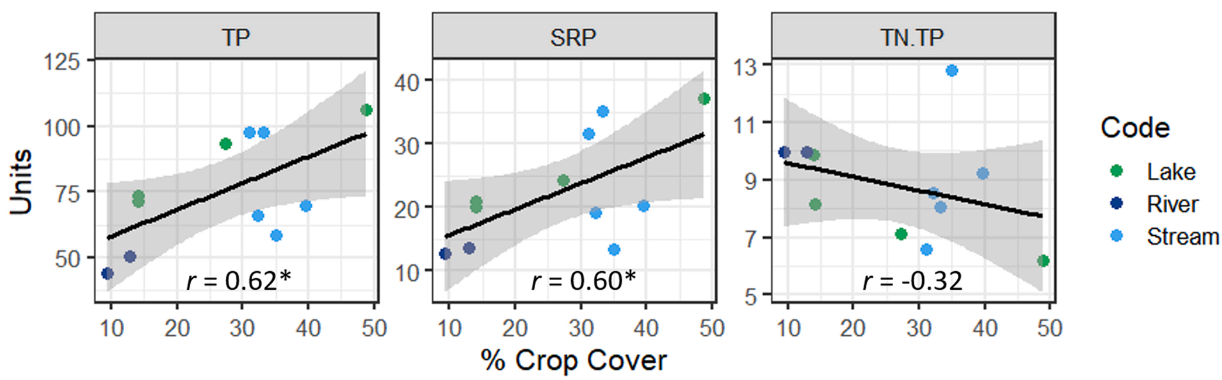

Fig. 6. Preliminary linear regression of percentage crop cover (2019–2020) and mean concentrations of TP, PO4 and TN:TP among sites. Units for all nutrient forms in µg/L. Pearson correlation coefficients (*r*) shown for each regression with significance value (* $p < 0.05$,·$p < 0.1$).

4.3. Water chemistry and land-use

External land-uses upstream of study sites likely play an important role for water chemistry. Intense agricultural activities involving use of phosphate fertilizers enrich water bodies downstream and thus reduce TN:TP ratios (Arbuckle and Downing, 2001). Both natural soil composition and anthropogenic and grazing activities that favor soil erosion and transport of dissolved materials and sediments to water bodies could affect the water chemistry of streams. Potential nutrient loading in streams SAN and BUR were evidenced by TP > 90 µg/L, respectively. SAN was the only stream with TN > 700 µg/L, considered a borderline value for oligo-mesotrophic systems (Smith et al., 1999). The low concentrations of NO3, SRP, and TN and TP were observed in QAP and QCH and lakes LAG1 and LAG4, all fed by watersheds with extensive areas of natural grassland ≥ 80% and relatively low cover of intensive land-uses. The larger QCH and QAP rivers originated in eutric leptosoles (FAO, 1990) with limited areas of intensive anthropic land uses (13% and 9% crop cover, respectively; 1% afforestation in both basins). Although native forest covered only 2–3% of these large watersheds, their location in the riparian zone as forest corridors along the main river channels may also play a role in diminishing nutrient inputs from diffuse sources in the basin. These river sites represented more oligo-mesotrophic conditions for fluvial systems in the region. Further research on algal biomass, macrophyte growth and other biological indicators would contribute a more complete evaluation of their relative trophic status.

One of the driving concerns behind this research was the susceptibility for eutrophication of lakes, which plays a potentially important but poorly understood role in aquatic biodiversity, fish reproduction, and habitat heterogeneity. These lakes are identified as a conservation target of the PA but with no data on the quality of these habitats and their ecological function in the floodplain. Here we found that lake LAG3 - and possibly LAG2 -were subject to anthropogenic sources of eutrophication, based on relatively high phosphorus levels and low TN:TP (Table 5), as well as a relatively high proportion of crops and afforestation in inlet drainage basins. Agricultural intensification outside the PA could contribute to the deterioration of water chemistry and the loss of ecological integrity in both lakes. Nonetheless, connectivity with the floodplain via episodic flooding is fundamental for homogenizing nutrient concentrations and returning lakes to lower levels (Güntzel et al., 2010; Thomaz et al., 2007). We expected the small, isolated lake surrounded by extensive areas of riparian forest to provide a potential "reference" system for other lakes in the floodplain. Nonetheless, the data suggest that this lake, a fraction of the size of the others, may be undergoing its own process of natural eutrophication. LAG4 stood out from the other lakes with lower levels of dissolved oxygen and higher turbidity and high concentrations of ammonium and TN. These characteristics may be linked to the predominance of heterotrophic over autotrophic processes, driven by the allochthonous contributions of organic matter from riparian vegetation leading to natural processes of lake eutrophication.

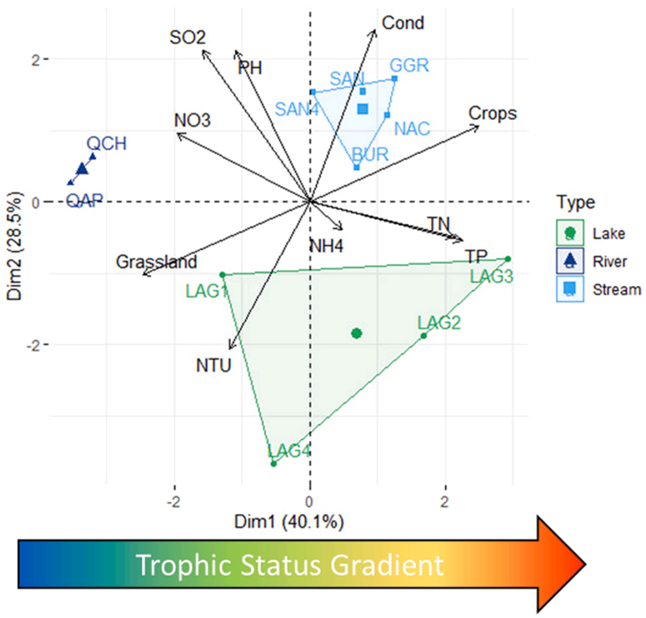

Fig. 7. PCA diagram based on water chemistry and percent cover of two dominant land-cover classes, Crops and Grassland for each study site. The first dimension (Dim1) along the x-axis was interpreted as a trophic status gradient from oligotrophic to eutrophic and hypereutrophic states (left to right), as shown in Table 5.

With regards to the potential drivers of eutrophication in Uruguayan rivers and lakes (Alcántara et al., 2021; Goyenola et al., 2021), this study makes a few key contributions. First, the Queguay Rivers provide a reference watershed comprised almost entirely of natural prairie (80%), the dominant natural land-cover class in this Pampean landscape. We provide current data on phosphorus and nitrogen concentrations, dissolved oxygen, turbidity among other variables related to eutrophication processes. Second, we demonstrate that the Queguay Rivers show the lowest average TP among the Cuareim, Negro and Santa Lucia River sites, at least for 2019–2020, establishing some of the lowest values recorded for TP in Uruguayan rivers. Finally, we created a land-use map for the Queguay River basin and found a positive, albeit preliminary, correlation between percent crop cover and phosphorus concentrations. Many studies in the region cite the role that land-cover plays for nutrient levels and eutrophication (Alcántara et al., 2021; Aubriot et al., 2020; Chalar et al., 2017; Gorgoglione et al., 2020; Goyenola et al., 2020, 2021, 2015). We intend for these novel data from the Queguay watershed to be incorporated into more complex models that further investigate the role that land-cover plays on nutrient levels, and ultimately on eutrophication in river ecosystems of the Pampa biome.

4.4. Implications for conservation and management

The basaltic region of northern Uruguay is one of the least-transformed regions within the La Plata Grasslands (Baeza and Paruelo, 2020). Here, we show that it also has some of the lowest nutrient levels in Uruguayan watersheds. Nonetheless, few management or conservation efforts aim to maintain the large areas of grassland that likely contribute to the observed low levels of phosphorus forms and consistently high levels of DO in the study area. The national system of protected areas in Uruguay aims to integrate PAs within a national

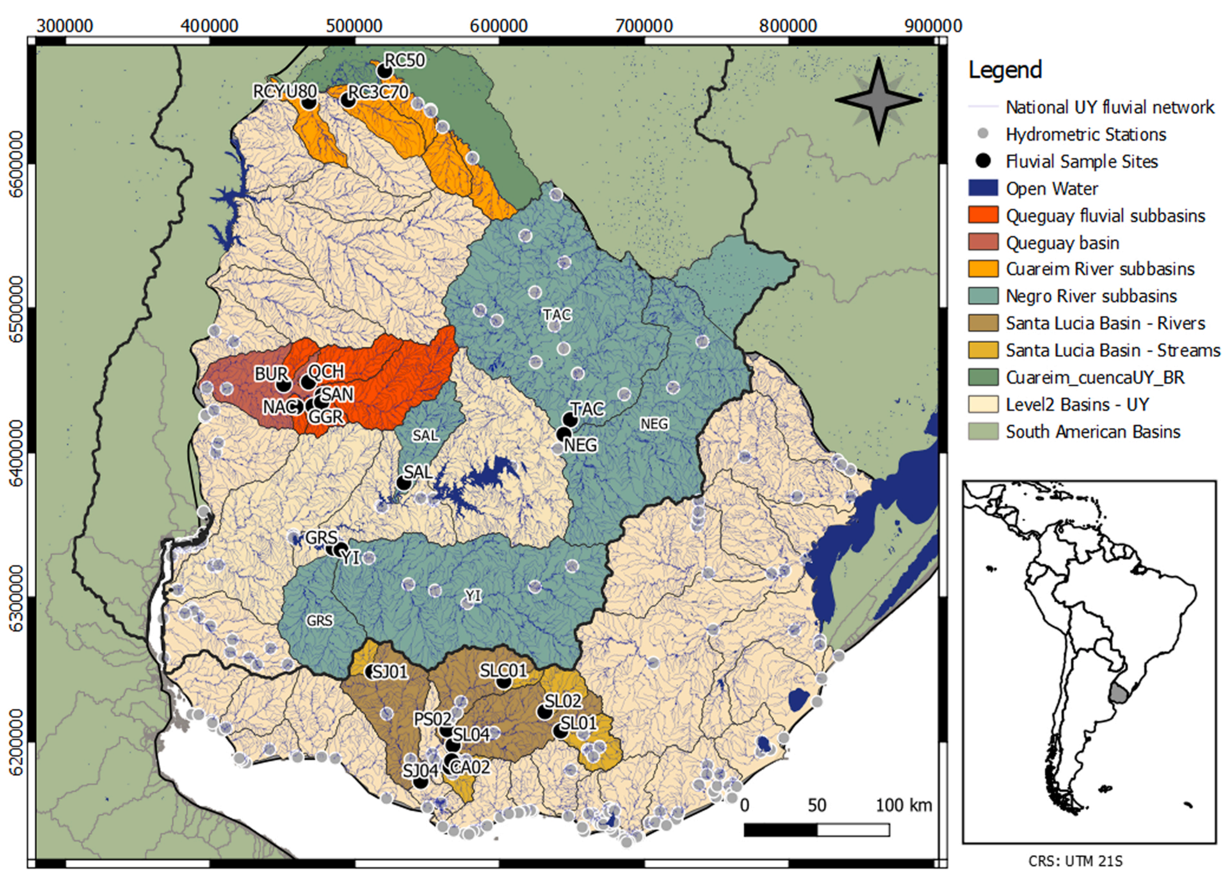

Fig. 8. Map of stream and river sample sites and respective basins in Uruguay, comparing the Queguay River Basin (red) with data from sites in the Negro River Basin (green); Cuareim River Basin (orange); and Santa Lucia River Basin (gold; Table 5).

policy for land-use planning within a sustainable development framework (MVOTMA, 2015). Among aquatic resources with large conservation value are emblematic fish species and related socioeconomic activities that sustain local populations. PAs aim to protect these target species as well as the artisanal and sport fisheries that constitute sustainable economic activities (Begossi et al., 1999; Castello et al., 2009; Santana et al., 2020). Changes in water chemistry (e.g., pH, DO, Electrical Conductivity, Temperature) as well as nutrient loading and anthropogenic eutrophication can compromise biodiversity and the life cycle of aquatic species, affecting growth, survival, movement ecology and reproduction. Within this context, water quality monitoring in PAs designed to protect aquatic species can contribute to zoning for the conservation of water resources, including focal areas for intervention, buffer zones and protected headwater regions (Hermoso et al., 2015).

5. Conclusions

This study establishes a baseline for water chemistry in marginal lakes and fluvial ecosystems in the Queguay River basin, within an extensive agricultural region of the La Plata River Grasslands. By providing the first time series dataset of water chemistry for this area, we fill a major geographical gap regarding the limnological attributes of critical freshwater ecosystems in northwestern Uruguay. We demonstrate the major differences that set lakes apart from fluvial systems in a large freshwater floodplain, highlighting the water chemistry parameters most relevant to the conservation and management of natural resources. The hypereutrophic state of one marginal lagoon suggests that these protected lentic systems are subject to external inputs, but also that lakes could play a role in nutrient retention and buffering external impacts on the river itself. The preliminary association between land-use and water chemistry highlights the importance of basin-scale management to avoid PA isolation and water quality deterioration within the PA, as observed in the Everglades (Fan et al., 2011). Through this analysis we identified the Queguay Chico and Grande Rivers as reference systems in comparison to other streams and rivers in the country Uruguay, including the Negro and Santa Lucia rivers. We also identified streams with relatively high phosphorus and phosphate levels, potentially serving as an alarm system for anthropogenic eutrophication in smaller basins with higher crop cover. Finally, we identify some characteristics of a lake undergoing natural eutrophication, including high total nitrogen, as part of the dynamics of this internal floodplain. This study contributes to the understanding of the broader relationship between land-use and nutrient concentrations in Uruguay's fluvial ecosystems and provides a first assessment of water quality and land-use for the Queguay River Basin.

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CRediT authorship contribution statement

Lucas and Crossa conducted pilot research and established monitoring sites with Brum. Lucas and Brum conducted the field work. Chalar oriented integration of nutrient analyses and participated in data analysis and interpretation; Chalar also oriented chemical analysis in the lab by De Giacomi. Baeza oriented Ibarguren in the land cover classification and interpretation of results, as well as in developing LULC maps; Alvareda and Paradiso participated in interpretation of physicochemical properties of water; Mejia participated in applications to conservation and management. Lucas coordinated the overall investigation process and conducted formal analysis of data. All coauthors contributed to the writing of the manuscript.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.limno.2022.125966.

References

Agostinho, A.A., Thomaz, S.M., Gomes, L.C., 2018. Threats for Biodiversity in the Floodplain of the Upper Paraná River: Effects of Hydrological Regulation by Dams.
Alcántara, I., Somma, A., Chalar, G., Fabre, A., Segura, A., Achkar, M., Arocena, R., Aubriot, L., Baladán, C., Barrios, M., 2021. A reply to "Relevant factors in the eutrophication of the Uruguay River and the Río Negro". Sci. Total Environ., 151854
Allan, J.D., 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annu. Rev. Ecol. Evol. Syst. 35, 257–284.
Alonso, J., Quintans, F., Taks, J., Conde, D., Chalar, G., Bonilla, S., Arocena, R., Haakonsson, S., Aubriot, L., Goyenola, G., 2019. Water Quality in Uruguay: Current Status and Challenges. Water Quality in the Americas. Risks and Opportunities. Mexico, IANAS.
Alvareda, E., Lucas, C., Paradiso, M., Piperno, A., Gamazo, P., Erasun, V., Russo, P., Saracho, A., Banega, R., Sapriza, G., 2020. Water quality evaluation of two urban streams in Northwest Uruguay: are national regulations for urban stream quality sufficient? Environ. Monit. Assess. 192, 1–22.
Arbuckle, K.E., Downing, J.A., 2001. The influence of watershed land use on lake N: P in a predominantly agricultural landscape. Limnol. Oceanogr. 46, 970–975.
Aubriot, L., Zabaleta, B., Bordet, F., Sienra, D., Risso, J., Achkar, M., Somma, A., 2020. Assessing the origin of a massive cyanobacterial bloom in the Río de la Plata (2019): towards an early warning system. Water Res. 181, 115944.
Baeza, S., Paruelo, J.M., 2020. Land use/land cover change (2000–2014) in the Rio de la Plata grasslands: an analysis based on MODIS NDVI time series. Remote Sens. 12, 381.
Barreto, P., Dogliotti, S., Perdomo, C., 2017. Surface water quality of intensive farming areas within the Santa Lucia River basin of Uruguay. Air Soil Water Res. 10, 1178622117715446.
Begossi, A., Silvano, R., Do Amaral, B., Oyakawa, O., 1999. Uses of fish and game by inhabitants of an extractive reserve (Upper Juruá, Acre, Brazil). Environ. Dev. Sustain. 1, 73–93.
Bonilla, S., Haakonsson, S., Somma, A., Gravier, A., Britos, A., Vidal, L., De León, L., Brena, M., Pírez, M., Piccini, C., 2015. Cianobacterias y cianotoxinas en ecosistemas límnicos de Uruguay. Innotec 9–22.
Bovo-Scomparin, V.M., Train, S., 2008. Long-term variability of the phytoplankton community in an isolated floodplain lake of the Ivinhema River State Park, Brazil. Hydrobiologia 610, 331–344.
Brazeiro, A., Achkar, M., Toranza, C., Bartesaghi, L., 2020. Agricultural expansion in Uruguayan grasslands and priority areas for vertebrate and woody plant conservation. Ecol. Soc. 25.
Brazeiro, A., Panario, D., Soutullo, A., Gutiérrez, O., Segura, A., Mai, P., 2015. Identificación y delimitación de eco-regiones de Uruguay. Eco-Regiones de Uruguay: Biodiversidad Presiones y Conservación. Aportes a la Estratega Nacional de Biodiversidad. Facultad de Ciencias, CIEDUR, VS-Uruguay, SZU, Montevideo, pp. 46–59.
Breiman, L., 2001. Random forests. Mach. Learn. 45, 5–32.
Brugnoli, E., Muniz, P., Venturini, N., Brena, B., Rodríguez, A., García-Rodríguez, F., 2019. Assessing multimetric trophic state variability during an ENSO event in a large estuary (Río de la Plata, South America). Reg. Stud. Mar. Sci. 28, 100565.
CARU, 2019. Digesto Sobre El Uso Y Aprovechamiento Del Río Uruguay. Aprobado por Resolución CARU No. 28/19, de 5 de diciembre de 2019. In: (CARU), C.A.d.R.U. (Ed.). ⟨https://www.caru.org.uy/web/pdfs_publicaciones/DIGESTO/DIGESTO%20SOBRE%20EL%20USO%20Y%20APROVECHAMIENTO%20DEL%20RIO%20URUGUAY%20APROBADO%20POR%20RESOLUCION%20CARU%20NRO%2028-19%20DE%205-12-2019.pdf⟩.
Castello, L., Viana, J.P., Watkins, G., Pinedo-Vasquez, M., Luzadis, V.A., 2009. Lessons from integrating fishers of arapaima in small-scale fisheries management at the Mamirauá Reserve, Amazon. Environ. Manag. 43, 197–209.

Chalar, G., 2009. The use of phytoplankton patterns of diversity for algal bloom management. Limnologica 39, 200–208.
Chalar, G., Arocena, R., Pacheco, J.P., Fabián, D., 2011. Trophic assessment of streams in Uruguay: a trophic State Index for Benthic Invertebrates (TSI-BI). Ecol. Indic. 11, 362–369.
Chalar, G., Garcia-Pesenti, P., Silva-Pablo, M., Perdomo, C., Olivero, V., Arocena, R., 2017. Weighting the impacts to stream water quality in small basins devoted to forage crops, dairy and beef cow production. Limnologica 65, 76–84.
Chalar Marquisá, G., Brugnoli, E., Clemente Soto, J.M., Hernández, L., Paradiso Giles, M. M., 2002. Antecedentes y nuevos aportes al conocimiento de la estructura y dinámica del Embalse Salto Grande. In: Fernández Cirelli, A., Chalar, G. (Eds.), El agua en Sudamérica, de la Limnología a la Gestión en Sudamérica. CYTED XVII Aprovechamiento y Gestión de los Recursos Hídricos, pp. 123–142.
Chalar, G., 2006. Dinámica de la eutrofización a diferentes escalas temporales: Embalse Salto Grande (Argentina – Uruguay). In: Tundisi, J.G., Tundisi, T.M., Galli, C.S. (Eds.), Eutrofização na América do Sur: Causas, conseqüências e tecnologias de gerenciamento e controle. InterAcademy Panel on International Issues, InterAmerican Network of Academies of Sciences. Instituto Internacional de Ecologia, Instituto Internacional de Ecologia e Gerenciamento Ambiental, Academia Brasileira de Ciências, Conselho Nacional de Desenvolvimento Científico e Tecnológico, pp. 87–101 ⟨http://www.dedicaciontotal.udelar.edu.uy/adjuntos/produccion/338_academicas_academicaarchivo.pdf⟩.
Chalar, G., Fabián, D., Mauricio González-Piana, M., Piccardo, A., 2015. Informe Interanual: Estado y evolución de la calidad de agua de los tres embalses del río negro, Convenio UTE – Facultad de Ciencias Período Setiembre 2011 – Marzo 2015 Sección Limnología, Facultad de Ciencias, UdelaR Montevideo. ⟨http://limno.fcien.edu.uy/pactuales/EUTROFIZACION-Y-CALIDAD-DE-AGUA-DE-LOS-EMBALSES-DEL-RIO-NEGRO-2011-2015.pdf⟩.
Conde, D., Pintos, W., Gorga, J., De León, R., Chalar, G., Sommaruga, R., 1996. The main factors inducing chemical spatial heterogeneity in the Salto Grande, a reservoir on the Uruguay River. Large Rivers 571–578.
Coutinho, H.L., Noellemeyer, E., Jobbagy, E., Jonathan, M., Paruelo, J., 2009. Impacts of land use change on ecosystems and society in the Rio de La Plata Basin. Appl. Ecol. Knowl. Land. Decis. 56, 65.
Cunha, D.G.F., do Carmo Calijuri, M., Lamparelli, M.C., 2013. A trophic state index for tropical/subtropical reservoirs (TSItsr). Ecol. Eng. 60, 126–134.
Cunha, D.G.F., Finkler, N.R., Lamparelli, M.C., Calijuri, Md.C., Dodds, W.K., Carlson, R. E., 2021. Characterizing trophic state in tropical/subtropical reservoirs: deviations among indexes in the lower latitudes. Environ. Manag. 68, 491–504.
De León, L., Chalar, G., 2003. Abundancia y diversidad del fitoplancton en el Embalse de Salto Grande (Argentina-Uruguay). Ciclo estacional y distribución espacial. Limnetica 22, 103–113.
De Oliveira, M.D., Calheiros, D.F., 2000. Flood pulse influence on phytoplankton communities of the south Pantanal floodplain, Brazil. Hydrobiologia 427, 101–112.
DINAGUA, 1979. Decreto No. 253/979. Aprobación de normativa para prevenir la contaminación ambiental, a través del control de las aguas. In: Uruguay, D.N.d.A. D.–. (Ed.).
DINAGUA, Gd, 2020. Watershed Delineations of Uruguay, Levels 1–5.
DINAMA, 2012. Proyecto de Selección y Delimitación del área Montes del Queguay al Sistema Nacional de Áreas Protegidas,Versión 19 diciembre 2012. Dirección Nacional de Medio Ambiente (DINAMA). Available to the public: ⟨https://www.gub.uy/ministerio-ambiente/comunicacion/publicaciones/proyecto-ingreso-del-area-protegida-recursos-manejados-montes-del⟩.
Dodds, W.K., Cole, J.J., 2007. Expanding the concept of trophic state in aquatic ecosystems: it's not just the autotrophs. Aquat. Sci. 69, 427–439.
Dodds, W.K., Jones, J.R., Welch, E.B., 1998. Suggested classification of stream trophic state: distributions of temperate stream types by chlorophyll, total nitrogen, and phosphorus. Water Res. 32, 1455–1462.
de la Escalera, G.M., Kruk, C., Segura, A.M., Nogueira, L., Alcántara, I., Piccini, C., 2017. Dynamics of toxic genotypes of Microcystis aeruginosa complex (MAC) through a wide freshwater to marine environmental gradient. Harmful Algae 62, 73–83.
Fan, X., Gu, B., Hanlon, E.A., Li, Y., Migliaccio, K., Dreschel, T.W., 2011. Investigation of long-term trends in selected physical and chemical parameters of inflows to Everglades National Park, 1977–2005. Environ. Monit. Assess. 178, 525–536.
FAO, G.l., 1990. Suelos de Uruguay.
Ferrari, G.M., 2020. Water flow and temperature as main factors that regulate phytoplankton and cyanobacterial blooms in a large subtropical river. Innotec 30–66.
Gao, B.-C., 1996. NDWI—a normalized difference water index for remote sensing of vegetation liquid water from space. Remote Sens. Environ. 58, 257–266.
González-Piana, M., Fabián, D., Delbene, L., Chalar, G., 2011. Toxics blooms of Microcystis aeruginosa in three Rio Negro reservoirs, Uruguay. Harmful Algae News 43, 16–17.
Gorelick, N., Hancher, M., Dixon, M., Ilyushchenko, S., Thau, D., Moore, R., 2017. Google Earth Engine: planetary-scale geospatial analysis for everyone. Remote Sens. Environ. 202, 18–27.
Gorgoglione, A., Gregorio, J., Ríos, A., Alonso, J., Chreties, C., Fossati, M., 2020. Influence of land use/land cover on surface-water quality of Santa Lucía river, Uruguay. Sustainability 12, 4692.
Goyenola, G., Meerhoff, M., Teixeira-de Mello, F., González-Bergonzoni, I., Graeber, D., Fosalba, C., Vidal, N., Mazzeo, N., Ovesen, N., Jeppesen, E., 2015. Monitoring strategies of stream phosphorus under contrasting climate-driven flow regimes. Hydrol. Earth Syst. Sci. 19, 4099–4111.
Goyenola, G., Graeber, D., Meerhoff, M., Jeppesen, E., Teixeira-de Mello, F., Vidal, N., Fosalba, C., Ovesen, N.B., Gelbrecht, J., Mazzeo, N., 2020. Influence of farming intensity and climate on lowland stream nitrogen. Water 12, 1021.
Goyenola, G., Kruk, C., Mazzeo, N., Nario, A., Perdomo, C., Piccini, C., Meerhoff, M., 2021. Production, nutrients, eutrophication and cyanobacteria blooms in Uruguay: putting puzzle pieces together. INNOTEC e558–e558.
Graeber, D., Goyenola, G., Meerhoff, M., Zwirnmann, E., Ovesen, N.B., Glendell, M., Gelbrecht, J., Teixeira de Mello, F., González-Bergonzoni, I., Jeppesen, E., 2015. Interacting effects of climate and agriculture on fluvial DOM in temperate and subtropical catchments. Hydrol. Earth Syst. Sci. 19, 2377–2394.
Güntzel, A.M., Panarelli, E.A., da Silva, W.M., Roche, K.F., 2010. Influence of connectivity on Cladocera diversity in oxbow lakes in the Taquari River floodplain (MS, Brazil). Acta Limnol. Bras. 22, 93–101.
Hecky, R., Kilham, P., 1988. Nutrient limitation of phytoplankton in freshwater and marine environments: a review of recent evidence on the effects of enrichment 1. Limnol. Oceanogr. 33, 796–822.
Hermoso, V., Cattarino, L., Kennard, M.J., Watts, M., Linke, S., 2015. Catchment zoning for freshwater conservation: refining plans to enhance action on the ground. J. Appl. Ecol. 52, 940–949.
Huszar, V.L., Caraco, N.F., Roland, F., Cole, J., 2006. Nutrient-chlorophyll relationships in tropical-subtropical lakes: do temperate models fit? Nitrogen Cycling in the Americas: Natural and Anthropogenic Influences and Controls. Springer, pp. 239–250.
Jordan, T.E., Correll, D.L., Weller, D.E., 1997. Relating nutrient discharges from watersheds to land use and streamflow variability. Water Resour. Res. 33, 2579–2590.
Juárez, R., Crettaz Minaglia, M.C., Aguer, I., Juárez, I., Gianello, D., Ávila, E., Roldán, C., 2016. Aplicación de índices bióticos de calidad de agua en cuatro arroyos de la cuenca del río Gualeguaychú (entre Ríos, Argentina).
Kitsiou, D., Karydis, M., 2011. Coastal marine eutrophication assessment: a review on data analysis. Environ. Int. 37, 778–801.
Knoll, L.B., Hagenbuch, E.J., Stevens, M.H., Vanni, M.J., Renwick, W.H., Denlinger, J.C., Hale, R.S., González, M.J., 2015. Predicting eutrophication status in reservoirs at large spatial scales using landscape and morphometric variables. Inland Waters 5, 203–214.
Koroleff, F., 1970. Direct determination of ammonia in natural waters as indophenol blue. Information on Techniques and Methods for Seawater Analysis, pp. 19–22.
Kruk, C., Martínez, A., de la Escalera, G.M., Trinchin, R., Manta, G., Segura, A.M., Piccini, C., Brena, B., Yannicelli, B., Fabiano, G., 2021. Rapid freshwater discharge on the coastal ocean as a mean of long distance spreading of an unprecedented toxic cyanobacteria bloom. Sci. Total Environ. 754, 142362.
Llorente, C.G., Molina, D.A., Zorzoli, P.A., Volpedo, A.V., 2017. Influencia de las descargas industriales y domésticas de los asentamientos urbanos sobre el Río Uruguay entre los años 1998 y 2004. AUGMDOMUS 5.
McDowell, R., Biggs, B., Sharpley, A., Nguyen, L., 2004. Connecting phosphorus loss from agricultural landscapes to surface water quality. Chem. Ecol. 20, 1–40.
Müller, R., Wiedemann, O., 1955. Die Bestimmung des Nitrations in Wasser. Vom Wasser 22, 247–271.
Murphy, J., Riley, J.P., 1962. A modified single solution method for the determination of phosphate in natural waters. Anal. Chim. Acta 27, 31–36.
MVOTMA, 2015. Plan estratégico 2015–2020. Sistema Nacional de Áreas Protegidas de Uruguay. Ministerio de Vivienda, Ordenamiento Territorial y Medio Ambiente (MVOTMA), Uruguay.
MVOTMA, 2018. Plan Nacional de Aguas, Ministerio de Vivienda, Ordenamiento Territorial y Medio Ambiente (MVOTMA), Uruguay. ⟨https://www.gub.uy/ministerio-ambiente/politicas-y-gestion/planes/plan-nacional-aguas⟩.
Mvotma, G.l., 2019. Geoservicios – GIS Shapefile, Water Body Layer [Espejos de agua] Publicly Available at: ⟨https://www.dinama.gub.uy/geoservicios/⟩.
NASA/METI/AIST/Japan Spacesystems [GIS layer], U.S./Japan ASTER Science Team, 2001. ASTER DEM Product [ASTER GDEM 2]. NASA EOSDIS Land Processes DAAC. Accessed 2021-03-19 from ⟨https://doi.org/10.5067/ASTER/AST14DEM.003⟩.
Nürnberg, G.K., 1996. Trophic state of clear and colored, soft-and hardwater lakes with special consideration of nutrients, anoxia, phytoplankton and fish. Lake Reserv. Manag. 12, 432–447.
O'Farrell, I., Izaguirre, I., 2014. Phytoplankton of the middle and lower stretches of the Uruguay River. Adv. Limnol. 65, 113–126.
OAN-MA, D., 2021. Observatorio Ambiental Nacional (OAN) – Ministerio de Ambiente (MA) – Open Database of Water Quality in Uruguay. ⟨https://www.ambiente.gub.uy/oan/datos-abiertos/⟩.
de Oliveira, T.E., de Freitas, D.S., Gianezini, M., Ruviaro, C.F., Zago, D., Mércio, T.Z., Dias, E.A., do Nascimento Lampert, V., Barcellos, J.O.J., 2017. Agricultural land use change in the Brazilian Pampa Biome: the reduction of natural grasslands. Land Use Policy 63, 394–400.
Pacheco, J.P., Arocena, R., Chalar, G., García, P., González Piana, M., Fabian, D., Silva, M., Olivero, V., 2012. Evaluación del estado trófico de arroyos de la cuenca de Paso Severino (Florida, Uruguay) mediante la utilización del índice biótico TSI-BI. Augmdomus 4.
Paullier, S., Bessonart, J., Brum, E., Loureiro, M., 2019. List of fish species of the Queguay River Basin, Uruguay River Low. Bol. Soc. Zool. Urug. 28, 66–78.
Pohlert, T., 2021. PMCMRplus: Calculate Pairwise Multiple Comparisons of Mean Rank Sums Extended, R Package Version 1.9.3. ⟨https://CRAN.R-project.org/package=PMCMRplus⟩.
Primpas, I., Tsirtsis, G., Karydis, M., Kokkoris, G.D., 2010. Principal component analysis: development of a multivariate index for assessing eutrophication according to the European water framework directive. Ecol. Indic. 10, 178–183.
Pusey, B.J., Arthington, A.H., 2003. Importance of the riparian zone to the conservation and management of freshwater fish: a review. Mar. Freshw. Res. 54, 1–16.

QGIS Development Team, 2020. QGIS Development Team, 2020. QGIS Geographic Information System. Open Source Geospatial Foundation Project. ⟨http://qgis.osgeo.org⟩.
R Core Team, 2020. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. URL: ⟨https://www.R-project.org⟩.
Reis, R., Albert, J., Di Dario, F., Mincarone, M., Petry, P., Rocha, L., 2016. Fish biodiversity and conservation in South America. J. Fish Biol. 89, 12–47.
Roberto, M., Santana, N., Thomaz, S., 2009. Limnology in the Upper Paraná River floodplain: large-scale spatial and temporal patterns, and the influence of reservoirs. Braz. J. Biol. 69, 717–725.
Rouse, J.W., Haas, R.H., Schell, J.A., Deering, D.W., 1974. Monitoring Vegetation Systems in the Great Plains with ERTS. NASA Special Publication, p. 309.
Salazar, A., Baldi, G., Hirota, M., Syktus, J., McAlpine, C., 2015. Land use and land cover change impacts on the regional climate of non-Amazonian South America: a review. Glob. Planet. Chang. 128, 103–119.
Santana, E.A., Oliveira, E.Fd, Balbino, Nd.S., Gurgel, H., 2020. Management of Pirarucu (Arapaima gigas, Teleostei, Osteoglossidae) in sustainable use units as a proposal for the restoration of aquatic ecosystems. Acta Limnol. Bras. 32.
Saunders, D.L., Meeuwig, J.J., Vincent, A.C., 2002. Freshwater protected areas: strategies for conservation. Conserv. Biol. 16, 30–41.
Schindler, D.W., 1977. Evolution of phosphorus limitation in lakes: natural mechanisms compensate for deficiencies of nitrogen and carbon in eutrophied lakes. Science 195, 260–262.
Schneider, C., Flörke, M., Stefano, L.D., Petersen-Perlman, J.D., 2017. Hydrological threats to riparian wetlands of international importance–a global quantitative and qualitative analysis. Hydrol. Earth Syst. Sci. 21, 2799–2815.
Serra, W.S., Loureiro, M., 2018. Austrolebias queguay (Cyprinodontiformes, Rivulidae), a new species of annual killifish endemic to the lower Uruguay river basin. Zoosyst. Evol. 94, 547.
Silveira, L., Alonso, J., 2009. Runoff modifications due to the conversion of natural grasslands to forests in a large basin in Uruguay. Hydrol. Process. Int. J. 23, 320–329.
Silveira, L., Gamazo, P., Alonso, J., Martínez, L., 2016. Effects of afforestation on groundwater recharge and water budgets in the western region of Uruguay. Hydrol. Process. 30, 3596–3608.
Smith, V.H., 1998. Cultural eutrophication of inland, estuarine, and coastal waters. Successes, Limitations, and Frontiers in Ecosystem Science. Springer, pp. 7–49.
Smith, V.H., Tilman, G.D., Nekola, J.C., 1999. Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. Environ. Pollut. 100, 179–196.
Soriano, A., 1991. Temperate subhumid grasslands of South America. In: Coupland, R.T. (Ed.), Temperate Subhumid Grasslands. Ecosystems of the World, Volume 8A, Natural Grasslands. Elsevier Scientific Publishing Company, Amsterdam, pp. 367–407.
Thomaz, S.M., Bini, L.M., Bozelli, R.L., 2007. Floods increase similarity among aquatic habitats in river-floodplain systems. Hydrobiologia 579, 1–13.
UNEP, 2003. Eutrophication Monitoring Strategy of MED POL, Meeting of the MED POL National Coordinators. UNEP(DEC)/MED WG.231/14. UNEP, Sangemini, Italy.
Valderrama, J.C., 1981. The simultaneous analysis of total nitrogen and total phosphorus in natural waters. Mar. Chem. 10, 109–122.
Vidal, L., Britos, A., 2012. Uruguay: Occurrence, Toxicity and Regulation of Cyanobacteria. Current Approaches to Cyanotoxin Risk Assessment, Risk Management and Regulations in Different Countries, p. 130.
Vollenweider, R., 1968. Scientific Fundamentals of Lake and Stream Eutrophication, with Particular Reference to Phosphorus and Nitrogen as Eutrophication Factors, Technical Report DAS/DSI/68.27). OECD, Paris, France.
Ward, J.V., 1998. Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. Biol. Conserv. 83, 269–278.
Ward, J.V., Tockner, K., Schiemer, F., 1999. Biodiversity of floodplain river ecosystems: ecotones and connectivity. Regul. Rivers Res. Manag. 15, 125–139.