Effects of intensive agriculture and hydrological changes on macrophyte and macroinvertebrate assemblages in lowland riverine wetlands

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Abstract

The aim of this study was to investigate the effects of agricultural land use and periods of hydrological variability on the environmental variables, as well as macrophyte and macroinvertebrate assemblages in lowland riverine wetlands. We compared two wetlands with intensive agricultural land use against two others with extensive livestock considered references for the region during a normal and a dry flow period. Nutrient concentrations were significantly higher in agricultural riverine wetlands. These wetlands exhibited higher relative coverage of floating anchored macrophytes and the absence of submerged vegetation. They showed significantly lower taxonomic richness and density of macroinvertebrates and a higher relative abundance of scrapers and predators. Wetlands of both land uses had a lower total density of macroinvertebrates and a higher proportion of tolerant desiccation taxa in the dry period. Particular differences between land uses, such as lower dissolved oxygen concentrations and lower macroinvertebrate diversity in agricultural wetlands, were found during the dry period. These findings indicate that the differences between land uses increased during the aforementioned period. This study provides evidence of the effects of the surrounding landscape and hydrologic periods in the environmental characteristics as well as the macrophyte and macroinvertebrate assemblages of the riverine wetlands studied.

Introduction

Wetlands are globally recognized as essential due to the many ecological functions and services that they provide to human society (Dixon and Wood 2007). They provide water quality protection (Verhoeven et al. 2006; Daneshvar et al. 2017), climate change mitigation, flood control, and drought prevention (Mitsch and Gosselink 2000; Zedler and Kercher, 2005). They are also an important reservoir of flora and fauna (Sileshi et al. 2020; Gupta et al. 2020). Around the world, the wetlands are subjected to several types of threats, both anthropogenic and natural. Agriculture is currently considered the main driver of wetland loss or degradation globally (Davidson 2014; Everard and Wood 2017).

The replacement of natural ecosystems by agricultural land use produces a homogenization that threatens biodiversity (Allan 2004; Dornelas et al. 2009). The fertilizers applied indiscriminately in intensive agriculture activities are one of the most important sources of contamination by nutrient loading in freshwater ecosystems (Grashof-Bokdam and van Langevelde 2005; Stokal et al. 2016). The excess of these nutrients, such as phosphorus and nitrogen, transported to watercourses reduce water quality due to the increased growth of undesirable algae and low oxygen concentrations (Blann et al. 2009; Woodward et al. 2012). This has several negative effects on freshwater ecosystem biodiversity, like replacement and loss of species in their assemblages (Gustafson and Wang 2002; Craft et al. 2007; Chalar et al. 2013; Lange et al. 2014; Lateral et al. 2018). Furthermore, drainage and water management for agriculture has significantly altered the hydrology of wetlands affecting also its biodiversity (Blann et al. 2009; Lacoul and Freedman 2006; Green et al. 2017). Water withdrawals for agriculture have been found to markedly reduce streamflow, causing a loss of connection between the surface and the

groundwater systems (Postel 2000) and resulting in a loss of habitat for aquatic biota (McKay and King 2006).

Riverine wetlands (RWs) are recognized as a major component of biodiversity in fluvial ecosystems (Amoros et al. 2000; Tockner et al. 2002). They are areas where streams overflow and connect with the surrounding terrestrial environment (Ringuelet, 1962; Brinson et al. 2002) and where macrophyte assemblages are highly developed (Amoros et al. 2000). As it is known, macrophytes are key components of wetlands ecosystems (Rejmánková 2011). They remove a variety of pollutants from the water (Bonanno and Vymazal 2017), and increase the habitat complexity and heterogeneity of these ecosystems (Thomaz and Cunha 2010). Also, they provide several resources for other organisms, like microhabitats (Dudley, 1988; Warfe et al. 2008), shelter (Heck and Crowder, 1991; Thomaz and Cunha 2010) and food (Dvořák, 1996; Díaz-Valenzuela et al. 2016).

Within the wetlands biota, macroinvertebrates play an important role in the overall functioning of these ecosystems as they occupy a central position in the food web and in organic matter cycling and energy flow (Batzler et al. 1999). The reduction in water quality caused by agricultural land use also leads to decreases in macroinvertebrate richness and density, as well as the decrease in the abundance of sensitive groups (Genito et al. 2002; Lange et al. 2014; Malacarne et al. 2016). In these sense, the macroinvertebrate assemblage is widely used as indicator of in-stream biotic integrity of the land use (Bunn et al. 2010), and the comprehensive analysis of their composition can provide an overview of wetland environmental status (Balcombe et al. 2005; Mereta et al. 2013).

The knowledge of biodiversity of RWs and the effects of land use and hydrological periods is necessary for ecosystem management and conservation plans. Our goal was to investigate the effects of intensive agricultural land use on the environmental variables and the macrophyte and macroinvertebrate assemblages of Pampean lowland RWs of Argentina, in time periods characterized by different hydrological conditions. We hypothesized that i) intensive agricultural land use induces changes in physical and chemical variables and in the structural and functional responses of macrophyte and macroinvertebrate assemblages; ii) an extraordinary dry period leads to variations in the assemblages and increases differences between land uses.

Materials And Methods

Study area

The study was conducted in the Pampean ecoregion of Buenos Aires, Argentina (Fig. 1). This region is a vast grassy plain that covers central Argentina. It has a humid and temperate climate, mean annual precipitation between 1000 and 1200 mm, and a mean annual temperature of 16°C (Hurtado et al. 2006). Riverine wetlands of this region are threaten by agriculture that affects the water quality and alters the natural habitat (Gómez et al. 2016). In particular, RWs located in the Pampean plain of Argentina represent the low depressions of Pampean stream basins, characterized by abundant and diverse aquatic

vegetation. The streams mentioned are characterized by an absence of riparian forest, low current velocities, and high nutrient levels (Rodrigues Capítulo et al. 2001; Feijoó and Lombardo 2007).

We selected two periurban RWs with intensive agricultural land use and other two with extensive livestock located on tributary streams of the freshwater section of the Río de la Plata estuary (Fig. 1, Online Resource: Table S1). The selection of these RWs was based on geohydrological and land use analysis (Rodrigues Capítulo et al. 2020; Gómez et al. in press). The four of them are located in characteristic Pampean streams and despite of their land use preserve their wetlands. Del Gato and the Carnaval RWs are in the agricultural area surrounding La Plata, the capital of the Province of Buenos Aires. This area is one of the main horticultural regions in the country (Baldini et al. 2019) with a high productivity based on intensive use of fertilizers and pesticides along with greenhouse technology (Arias et al. 2020). Downstream, the Carnaval and Del Gato streams run through densely populated areas and their channels have been heavily impacted by anthropic action. The over-extraction for crop irrigation and drinking water supply of the city of La Plata and its surroundings has caused the local deepening of the water tables (Gómez et al. in press). This has contributed to the loss of groundwater supply to the base flow of the streams where these RWs occur (Deluchi et al. 2005; García and Zanandrea 2017). The other RWs, Cajaravillas and Chubichaminí, are located in Magdalena, a rural area with extensive livestock grazing on natural grassland (0.7 cows per ha., Gómez et al. in press). Freshwater systems surrounded by these grasslands are considered the reference for the region (Solis et al.; 2018; Arias et al. 2020; Paracampo et al. 2020). These RWs receive groundwater and surface water, with a greater volume being supplied by the former. Both RWs flow downstream through rural areas that do not hinder their natural drainage to the Río de la Plata estuary.

To know the effects of hydrological variation, four samplings were performed on the RWs. Two samplings were carried out during a period of normal flow, which we call “normal period” (September and October, mean monthly cumulative precipitation of 80 mm and 61 mm respectively), and two samplings in a dry period (February and early March). The monthly accumulative precipitations of this period were 2 mm in February and 92 mm in March, with 82 mm accumulated in the five days after the sampling date. The average precipitation recorded was lower than the historical mean monthly accumulated precipitation for the last ten years (National Meteorological Service: February 177 mm and March 123 mm). This intense drought coincides with the “La Niña” phase of the ENSO phenomenon (Gómez et al. in press).

In each RW we selected a section of 50m in the center of the channel, depending on the ease of access, where the sampling was carried out. In that section, in each sampling occasion, physical and chemical variables and macrophyte assemblage descriptors were measured and macroinvertebrates were sampled (see below). During the dry period, the RWs studied were characterized by lower width, lower water velocity, and lower flow in the four wetlands studied (Online Resource: Table S1). The Carnaval, one of the agricultural RWs, was completely dry during the February campaign.

Site characterization

The pH, temperature, dissolved oxygen concentration (DO), conductivity, and total dissolved solids were measured in situ in triplicate with a HORIBA Multiparameter U-10. Besides, on each sampling occasion, a water sample was collected for the analysis of nutrient concentrations (Total Nitrogen, Nitrates [N-NO₃], Nitrites [N-NO₂], Ammonia [N-NH₄], Total Phosphorus and Phosphate [P-PO₄]) and oxygen demands (biochemical and chemical, BOD and COD, respectively) under standardized protocols (APHA, 1998).

Macrophytes

The macrophytes species, along with their coverage and life forms, were recorded. Five evenly-spaced transects were performed perpendicular to the channel in the section of 50m in each wetland to calculate the macrophyte coverage. On each transect, the total coverage of each patch and species were estimated by measuring the length of the transect covered by the water and by each species (Feijoó and Menéndez 2009). In addition, total macrophyte coverage was calculated as the sum of total coverage values of present species. For this study, we classified the species as emergent, submerged, floating-leaved anchored, and free-floating (Cabrera, 1948; Lahitte et al. 2004).

Macroinvertebrates

Triplicate samples of the macroinvertebrates were taken from the present vegetation with a sieve of 500-um mesh in a Plexiglas square (625 cm²) (Cortelezzi et al. 2013). The material was fixed in situ with 5% (v/v) aqueous formaldehyde. In the laboratory, the invertebrates were separated under a stereomicroscope, counted, and identified to the lowest possible taxonomic level through standard keys (Lopretto and Tell, 1995; Merritt et al. 2008; Domínguez and Fernández 2009).

Data analysis and statistical approach

The RWs along the physical and chemical variables were characterized by principal-component analysis (PCA). Before the analysis, the variables were standardized. The variables that presented little contribution, those that present low correlation with the first and the second component, were removed to simplify analysis (Kassambara 2017). Differences in physical and chemical variables between land uses and hydrological periods and their interaction were assessed by two-way ANOVA with 'RW' nested within 'land use'. The variables that did not fit with a normal distribution (temperature, conductivity, DO, Total Nitrogen and Total Phosphorus) were log-transformed. Pairwise comparisons on the main fixed factors were performed using Tukey's post hoc tests. Model residuals were tested for normality using a Shapiro-Wilk test.

The total coverage of macrophytes and the relative coverage of each life form expressed as percentages were calculated to characterize the macrophyte assemblage. Also, the richness was estimated as the number of taxa present, and diversity was estimated by the Shannon-Wiener Index (Shannon and Weaver, 1949). In addition, to characterize the macroinvertebrate assemblage, density was expressed as the average number of individuals per square meter, and richness and diversity were estimated. We assigned each taxon to a functional feeding group (FFG) using available references (Cummins et al.

2005; Allan and Castillo 2007; Merritt et al. 2008). The relative abundance values of each FFG were calculated for each site and sampling occasion using macroinvertebrate densities (ind. m⁻²).

Macrophyte coverage, diversity of both assemblages, as well as density and FFG of macroinvertebrates, were compared between land uses and hydrological periods by two-way ANOVA with 'RW' nested within 'land use'. The interaction between 'land use' and 'hydrological period' was also assessed. Density of macroinvertebrates data were log-transformed, whereas coverage of macrophytes and FFGs of macroinvertebrates were arcsine transformed to fit with the normal distribution. Pairwise comparisons on the main fixed factors were performed using Tukey's post hoc tests. Model residuals were tested for normality using a Shapiro–Wilk test. For count data (richness), we used generalized linear models (GLMs) with Poisson error distribution (link: log) for the same factors.

A Permutational Multivariate Analysis of Variance (PERMANOVA) was used to test differences in macroinvertebrate taxonomic composition between 'land use' (agriculture and livestock) and 'hydrological period' (normal and dry), with 'RWs' nested within 'land use'. The PERMANOVA was applied on a Bray–Curtis dissimilarity matrix calculated from the abundance data of macroinvertebrates. We also used a similarity percentage analysis (SIMPER) to identify taxa separating different land use categories and hydrological periods and to quantify the contribution of individual taxa to each category.

All statistical analysis were performed using the language environment R version 3.6.3 (R Development Core Team 2020) and the RStudio Team (2015). The packages used for the analysis were FactoMineR (Lê et al. 2008), vegan (Oksanen et al. 2013), biodiversityR (Kindt and Coe 2005), and stats (R Development Core Team 2020).

Results

Sites characterization

The first two PCA axes explained 64.1% of the overall variance (Fig. 2). The first axis (43.5%) illustrated the environmental differences between land uses. Agricultural RWs were characterized by higher nutrient concentration (total phosphorus, total nitrogen, soluble reactive phosphorus, and to a lesser extent, ammonium). By contrast, livestock RWs were characterized by higher values of pH, conductivity, and TDS and lower values of nutrient concentrations (Online Resource: Table S2, correlations between axis and variables). The second axis (20.6% of the total variance) was positively correlated with temperature, nitrate, and nitrite whereas was negatively correlated with flow and dissolved oxygen. This axis showed the difference between hydrological periods in livestock RWs, with respect to temperature, DO and flow. The dry period was characterized by higher temperature, lower flow and DO in comparison with the normal period.

Significant differences were found in water physical and chemical characteristics between land uses (Table 1). Agricultural RWs exhibited higher values of total phosphorus ($F_{1,6} = 34.50$, $p = 0.001$), total nitrogen ($F_{1,6} = 116.94$, $p < 0.001$), and P-PO₄ ($F_{1,6} = 44.84$, $p < 0.001$). By contrast, livestock RWs had

higher values of conductivity ($F_{1,34} = 250.33$, $p < 0.001$), TDS ($F_{1,34} = 147.72$, $p < 0.001$) and pH ($F_{1,34} = 169.51$, $p < 0.001$) in both hydrological periods. Dissolved oxygen concentration was lower in the dry period in comparison with normal period in RWs of both land uses and also, was lower in agricultural RWs than in livestock RWs in the dry period ($F_{1,34} = 4.480$, $p = 0.042$). In contrast, temperature was higher in the dry period in both land uses ($F_{1,34} = 51.67$, $p < 0.001$). Biochemical oxygen demand showed differences between hydrological periods, with higher values in the dry period, only in agricultural RWs ($F_{1,6} = 16.98$, $p = 0.006$).

Macrophytes

We recorded 15 species during the study, and the total coverage of macrophytes was always greater than 60% in the RWs studied (Table 2). No significant differences were found in richness (estimate = 0.18, z -value = 0.43, $p = 0.670$; estimate = 0.29, z -value = 0.65, $p = 0.51$), diversity ($F_{1,7} = 0.10$, $p = 0.763$; $F_{1,7} = 1.72$, $p = 0.231$) or total coverage ($F_{1,7} = 0.00$, $p = 0.956$; $F_{1,7} = 0.94$, $p = 0.346$) between land uses and hydrological periods, respectively. Coverage of floating-anchored species was higher in agricultural RWs than in livestock RWs ($F_{1,7} = 637.14$, $p < 0.001$), where coverage of emergent macrophytes was higher ($F_{1,7} = 82.41$, $p < 0.001$). Submerged macrophytes were not recorded in agricultural RWs. *Typha latifolia*, *H. ranunculoides*, *A. philoxeroides*, and *L. peploides* were the dominant species in the agricultural RWs, whereas *I. pseudacorus* and *S. californicus* were the dominant in livestock RWs. Unlike the normal period characterized by a higher coverage of emergent macrophytes ($F_{1,7} = 19.70$, $p = 0.003$), the dry period was characterized by a higher coverage of a free-floating species (*L. gibba*, $F_{1,7} = 10.73$, $p = 0.014$). With respect to floating-anchored macrophytes they respond differently in each land use, decreasing their coverage in the dry period in agricultural RWs and increasing in that period in livestock ones (Online Resource: Table S3).

Macroinvertebrates

A total of 63 taxa of macroinvertebrates were collected in the RWs studied (Online Resource: Table S4). Taxa richness differed significantly between land uses (estimate = 0.39, z -value = 2.68, $p = 0.007$), with an average of 13 taxa in agricultural RWs and an average of 20 taxa in livestock RWs (Fig. 3). Mean density also differed between land uses ($F_{1,37} = 31.26$, $p < 0.001$), with agricultural RWs showing half the density of livestock RWs (Fig. 3). In addition, the density found during the dry period for the two land uses was half that recorded for the normal period ($F_{1,37} = 34.40$, $p < 0.001$, Fig. 3). Diversity only showed significant differences between land uses in the dry period ($F_{1,37} = 5.48$, $p = 0.025$), with higher values in livestock RWs than in agricultural RW (Online Resource: Table S5).

Regarding FFGs, 26 taxa were identified as predators, 23 as collector-gatherers, 7 as scrapers, 5 as collector-filterers, and 2 as shredders (Online Resource: Table S4). Free-living aquatic nematodes were not included in the FFG analysis due to the controversies in the FFG classification (Moens et al. 2006; López van Oosterom et al. 2013). In the comparison between land uses, agricultural RWs showed higher relative

abundance of predators ($F_{1,37} = 5.87$, $p = 0.020$) and scrapers ($F_{1,37} = 6.15$, $p = 0.012$, Fig. 4). By contrast, livestock RWs exhibited a higher relative abundance of collector-gatherers in both periods ($F_{1,37} = 6.50$, $p = 0.015$) and a higher relative abundance of collector-filterers in the dry period, coinciding with a reduction of this FFG in the agricultural RWs (significant interaction, $F_{1,37} = 10.567$, $p = 0.002$; Fig. 4). Differences between hydrological periods were also found: the dry period exhibited higher relative abundance of predators ($F_{1,37} = 28.63$, $p < 0.001$) than the normal period, which had a significantly higher relative abundance of shredders ($F_{1,37} = 26.39$, $p < 0.001$, Fig. 4).

The assemblage compositions differed significantly between land uses (PERMANOVA: pseudo- $F_{1,37} = 16.46$, $P = 0.001$) and hydrological periods (PERMANOVA: pseudo- $F_{1,37} = 12.39$, $P = 0.001$). Based on SIMPER analysis results, the dissimilarity between land uses in the normal period was 48%, whereas the dissimilarity increased to 67% in the dry period. The taxa that contributed the most to differences between agriculture and livestock were *Hyaella curvispina* (Shoemaker, 1942) (5.80%), Oligochaeta (5.33%), *Uncancylus concentricus* (dOrbigny, 1835) (5.29%), Cyclopoida (5.18%), Cladocera (4.97%), Chironomidae (4.79%), and *Caenis* sp. (4.64%) (with higher abundance in livestock than agricultural RWs) and DugesIIDae (4.68%) and Entomobryidae (3.47%) with higher abundance in agricultural RWs. The differences between hydrological periods were attributed to the decreased abundance of *H. curvispina* (7.92% in livestock, 7.48% in agriculture), DugesIIDae (5.29% in livestock, 10.42% in agriculture), and *Caenis* sp. (4.88% in livestock) in dry periods and the increased abundance of particular taxa such as Nematoda (4.00% in livestock, 4.09% in agriculture), Cyclopoida (3.79% in livestock, 5.73% in agriculture), and Ceratopogonidae (4.43% in livestock).

Discussion

Our results suggest that agricultural land use had a strong influence on the physical and chemical parameters, macrophyte structure, and macroinvertebrate metrics of the Pampean lowland RWs studied. The physical and chemical characteristics found for agricultural RWs denote water quality degradation of these wetlands in comparison with livestock RWs according to the first studies of the region (Tarda et al. 2019; Cocherio et al. 2020; Gómez et al. in press).

The high concentration of nutrients in RWs with agriculture in their catchment is typical of freshwater systems in agricultural areas because the high amount of fertilizer used in crops increases the concentration of nutrients in the surrounding freshwater systems (Carpenter and Bennett 2011; Egler et al. 2012; Mugni et al. 2013; Ruiz-Picos et al. 2016). In contrast, the higher conductivity, TDS, and pH found in livestock RWs agreed with the characterization performed by Molina et al. (2017) for aquatic environments associated with pastures, herbaceous, and shrub vegetation. In particular, in pasture catchments like those studied, cattle have free access to wetlands for watering, which can increase the total dissolved solids and conductivity due to the constant disturbance and resuspension of the substrate (Gary et al. 1983; Roche et al. 2013). Also, the higher conductivity of the livestock RWs could be produced by the groundwater contribution (Caruso 2002).

In accordance with our results, changes in macrophyte species resulting from agricultural land use were also recorded for USA wetlands (Gustafson and Wang 2002). Submerged macrophyte were absent and floating macrophyte coverage increased in agricultural lands, as documented in the context of increased nutrient load (Rasmussen and Anderson 2005). Species composition could change in relation to agricultural land use as well. In our work, we found *T. latifolia* and *H. ranunculoides* as the dominant species in agricultural sites. *Typha* spp. are considered to increase their dominance in nutrient-enriched wetlands (Cooper et al. 2006; Craft et al. 2007; Rejmánková 2011). These species can benefit from agricultural nutrient pulses by taking up nutrients rapidly and funneling them into growing tissues (Zedler and Kercher 2004). Similarly, *H. ranunculoides* is known to increase its growth and biomass with increasing nutrient availability in laboratory conditions (Hussner and Lösch 2007). But, not only the surrounding land use can determine the macrophyte composition, also the physical attributes and the hydrologic disturbance affects the relative abundance of macrophyte species (Lacoul and Freedman 2006). The higher concentration of TDS and the shallow and wide characteristic of the livestock RWs allowed the development of a higher coverage of emergent macrophytes in agreement with previous studies (Feijoó et al. 1999; Egertson et al. 2004). Despite the land use, macrophyte coverage was high in the four RWs studied, giving these wetlands a high habitat complexity (Thomaz and Cunha 2010). Regarding hydrological conditions, the high coverage of *L. gibba* found in the dry period correlates with the increase in the coverage of floating macrophytes documented at low flows by other authors (Cao et al. 2018; Fraaije et al. 2018). Lower flow velocities lead to a predominance of more typically lentic plant communities, such as free-floating species (Fraaije et al. 2018).

Although the use of macroinvertebrates as indicators of agricultural wetland land use is still discussed (Batzner 2013; Gleason and Rooney 2017), we found that this assemblage respond to the effect of agriculture on the studied RWs, and can be considered good indicators of land use in this wetlands. In agreement with the results found in other types of wetlands around the world, lower richness and density were observed in wetlands with intensive agricultural land use. That is the case for example for some Mediterranean littoral wetlands (Ortega et al. 2004) and ponds (Della Bella and Mancini 2009), North American floodplain wetlands (Chipps et al. 2006) and Chinese wetlands (Wu et al. 2019) impacted by intensive agricultural activities. In line with our results for RWs, Arias et al. (2020) and Solis et al. (2017) also found that mean density was twice as high between streams of the Pampean region with extensive livestock and intensive agriculture in their surroundings.

The reduction in taxonomic richness is usually related to the loss of sensitive taxa in agricultural land use (Gerth and Giannico 2017). High percentages of agricultural land cover reduced the number of sensitive macroinvertebrate taxa and produced a macroinvertebrate community composition that reflected an altered habitat (Genito et al. 2002). In coincidence with Mugni et al. (2013) and Solis et al. (2017, 2018), who studied streams in the Pampean region, our results in RWs suggest that the deterioration in water quality generated by intensive agriculture could cause a decrease on the abundance of sensitive macroinvertebrate taxa such as *H. curvispina* and *Caenis* sp. Cooper et al. (2006) also found less abundance of a *Hyalella* species (*Hyalella azteca*) in wetlands adjacent to agricultural land. In addition, we found that tolerant taxa such as Dugesidae and Entomobryidae characterized agricultural RWs, in

coincidence with the results of Solis et al. (2018). We also recorded the presence of some sensitive invertebrates (Rodrigues Capítulo et al. 2001) that were found only in livestock RWs such as *Lestes*, *Rhionaeschna*, and individuals of the Simuliidae family.

Feeding strategies of macroinvertebrates could also reflect the adaptation of species to stressors and form part of a unified measure across communities differing in taxonomic composition (Tomanova et al. 2006). The effect of agricultural land use was reflected in a greater relative abundance of scrapers and predators in RWs. The increase in the frequency of scraper feeding habits is expected to occur in nutrient-enriched environments (Dolédéc et al. 2006; Statzner and Bêche 2010). On the other hand, predator abundance was associated with the higher density of Dugesiidae and *Helobdella* sp., considered tolerant taxa by other authors (Rodrigues Capítulo et al. 2001; Solis et al. 2019). The lowest proportion of collector-filterers in agricultural RWs in the dry period was in accordance with Gebrehiwot et al. (2017), who found a decrease in this FFG at organic polluted sites of Ethiopian wetlands in a dry period. Also, the relative abundance of this FFG is considered to be a useful metric for characterizing the ecological condition of river-associated wetlands in Eastern Africa (Mereta et al. 2013).

The presence of different life forms of macrophytes can also drive the composition of invertebrate FFG by determining the availability and type of food resources (Cremona et al. 2008; de Souza Rezende et al. 2019). In this sense, submerged macrophytes are usually related to a greater abundance of collectors-gatherers (Cremona et al. 2008; Peiró et al. 2015; de Souza Rezende et al. 2019), because its architecture allow the retention of more detritus (Sand-Jensen, 1998). In our study, this life form was absent in agricultural RWs in concordance with the lower proportion of this FFG. On the other hand, the highest proportion of predators in agricultural RWs and in the dry period could be related to the higher coverage of floating-leaved macrophytes. This life form provides little habitat in the vertical dimension that promotes a higher exposure to visual predators compared to the other macrophyte life forms (Gosselain et al. 2005). In this sense, the land use and hydrological period effects observed in the macrophyte assemblages could also contribute to the differences in the proportion of FFGs.

Much of the current knowledge about the ecological response of macroinvertebrates to droughts is related to the effects of largely predictable seasonal droughts rather than supra-seasonal events (Lake 2003). Species inhabiting temporal ecosystems that suffer predictable droughts have different resistance mechanisms, including tolerance to the deterioration of water quality conditions and the presence of desiccation-resistant life history stages (Bogan et al. 2017). Also, present resilience mechanisms as dispersal to rewetted habitats from refugia (Boulton and Lake 2008). On the other hand, faunal recovery from supra-seasonal droughts varies from one case to another (Lake 2003). Generally, species are vulnerable to these events as they are not adapted and cannot escape the disturbance events in time (Boulton 2003). The unusual absence of precipitation, the high evapotranspiration in summer, and the loss of connection with the groundwater in agricultural areas (Rodrigues Capítulo et al. 2020) led the Carnaval RW to be dry in the February campaign. The lower oxygen concentration in RWs surrounded by agricultural land indicated a higher effect of the dry period on the water characteristics of these RWs. According to Robinson et al. (2004), reduced flow commonly leads to decreases in dissolved-oxygen

content, which is critical to the survival of many aquatic species and can also affect their distribution and abundance. This condition was associated with the lower macroinvertebrate diversity, the wide difference in taxa richness, and the higher dissimilarity in macroinvertebrate composition found in agricultural RWs in comparison with livestock RWs during the dry period.

In line with our findings on the RWs studied, it is known that natural low flows cause decreases in invertebrate densities (Wood and Armitage 2004). Some authors suggested that this decrease occurs in response to changes in competition and predation because habitat area decreases and food quality and quantity are altered by flow reduction (Cowx et al. 1984; Wood et al. 2000). Furthermore, invertebrate community composition often changes in response to low or reduced flow in streams (Gore et al. 2001; Suren et al. 2003) and wetlands (Sim et al. 2013). The increase in the density of particular taxa associated with the dry period in macroinvertebrate assemblages was similar to that reported by other authors. Larned et al. (2007) also found Nematoda and Copepoda to be resistant to desiccation. Desiccation-resistant stages are well-known in copepods (Dahms, 1995), and aquatic nematodes are believed to survive extended dry periods in a state of anhydrobiosis (Drummond et al. 2015). Moreover, Ceratopogonidae larvae appear almost immediately when surface flows resume, suggesting that they use the hyporheic zone as a refuge from surface drying (Stanley et al. 1994; Stubbington 2012). Conversely, *H. curvispina* and *Caenis* sp. decrease their abundance in the dry period. There are no previous reports on changes in the abundances of these taxa under drought conditions. However, other related species have been studied. Ladle and Bass (1981) and Wood and Armitage (2004) recorded similar responses of *Gammarus pulex* (L.) to dry conditions. Regarding *Caenis* sp., the reduced flow and decreased water velocity were documented to be detrimental to other Ephemeroptera taxa (Calapez et al. 2017). With regards to FFGs, in agreement with our results, the effects of a flow reduction in streams and wetlands were associated with an increase in the prevalence of predators and a decrease in shredders (Bêche et al. 2006; Statzner and Bêche 2010; Gebrehiwot et al. 2017). Reduced velocities could favor invertebrate predators by removing velocity-mediated predation refugia (Dewson et al. 2007) and affect shredder abundance by modifying the quality of resources (Statzner and Bêche 2010).

The results showed a synergetic effects of nutrient loading and hydrological disturbances in the RWs studied. Similar results were mentioned by Green et al. (2017) for the Doñana wetlands in Spain, a more complex system of wetlands than the studied. As mention these authors, decisions from informed policy makers can promote ecosystem resilience to global threats through local measures. The correct management of agricultural activities in the land surrounding a river, as part of integrated watershed management, is of paramount importance to the conservation of associated wetland water quality (Wang 2001). Therefore, unsustainable agricultural practices could be replaced with environmentally-friendly, ecological agriculture to preserve wetland ecosystems (Zou et al. 2018).

Conclusions

The findings provide strong evidence of the importance that the surrounding landscape has in the environmental characteristics and macrophyte and macroinvertebrate assemblages of lowland RWs. We

conclude that the effects of intensive agricultural land use on physico-chemical variables, mainly nutrient enrichment, contributed to the differences found in macrophyte life form coverage and macroinvertebrate assemblages. The differences between land uses were greater during a drought period, which indicates a higher sensitivity to different hydrological conditions in those wetlands surrounded by intensive agricultural systems in comparison with those surrounding by extensive livestock considered as reference for the region.

Despite the land use, the great macrophyte coverage in the RWs studied indicates the importance of these systems in the basin and the value of their conservation. However, the differences found in macroinvertebrate FFGs could also be influenced by the composition of macrophyte assemblage, indicating the role and importance of this assemblage in lowland RWs. In this sense, the combined use of macrophytes and macroinvertebrate assemblages were a powerful tool for describing and assessing the studied riverine wetlands. The metrics evaluated were useful for studying the land use of the catchment and the hydrological conditions. This study provides valuable information to future conservation and management projects and to the scarce knowledge of this type of wetlands.

Declarations

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Code availability (software application or custom code): not applicable.

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Tables

Table 1. Mean and standard error of physico-chemical variables in the riverine wetlands of agricultural and livestock land use in the two hydrological periods. Significantly higher values between land uses ($p < 0.05$) are underlined, ^a significant difference between hydrological periods, ^b significant differences in the interaction.

	Agriculture		Livestock	
	normal	dry	normal	dry
Temperature (T°, °C)	18.78 ± 0.23	20.44 ± 0.13 ^a	17.61 ± 0.35	21.27 ± 0.29 ^a
pH	7.18 ± 0.07	6.83 ± 0.02	<u>8.16 ± 0.04</u>	<u>8.2 ± 0.01</u>
Conductivity (Cond, ms/cm)	0.35 ± 0.02	0.29 ± 0.01	<u>0.74 ± 0.01</u>	<u>1.02 ± 0.03</u>
Dissolved oxygen (DO, mg/l)	6.01 ± 0.26	2.75 ± 0.03 ^b	6.97 ± 0.26	4.6 ± 0.31 ^b
Total dissolved solids (TDS, g/l)	0.23 ± 0.01	0.19 ± 0.01	<u>0.47 ± 0.01</u>	<u>0.65 ± 0.02</u>
Nitrates (N-NO ₃ , mg N/l)	0.05 ± 0.01	0.03 ± 0	0.03 ± 0.01	0.05 ± 0.01
Nitrites (N-NO ₂ , mg N/l)	0.03 ± 0.01	0.03 ± 0	0.02 ± 0	0.04 ± 0
Ammonia (N-NH ₄ , mg N/l)	0.12 ± 0.02	0.18 ± 0.06	0.11 ± 0.03	0.06 ± 0.01
Phosphate (P-PO ₄ , mg P/litro)	<u>1.04 ± 0.1</u>	<u>1.15 ± 0.09</u>	0.1 ± 0.01	0.26 ± 0.03
Total Phosphorus (Total P, mg P/l)	<u>1.54 ± 0.21</u>	<u>1.48 ± 0.19</u>	0.5 ± 0.09	0.41 ± 0.01
Total Nitrogen (Total N, mg N/l)	<u>3.37 ± 0.19</u>	<u>12.18 ± 0.41</u>	2.92 ± 0.18	2.22 ± 0.17
Biochemical oxygen demand (BOD, mgO ₂ /l)	8.75 ± 1.48	28 ± 1.5 ^a	14.25 ± 0.69	10.58 ± 1.07
Chemical oxygen demand (COD, mgO ₂ /l)	47.5 ± 2.58	81.33 ± 9.72	75.5 ± 2.2	49.42 ± 2.86

Table 2. Macrophyte species coverage (* <5 %; ** 5–25 %; *** >25 %) at the agricultural and livestock RWs in the two hydrological periods. Also, total coverage and life forms are given: E, Emergent; F-A, Floating anchored; F-F, Free-floating; S, Submerged.

Species	Life forms	Agriculture		Livestock	
		normal	dry	normal	dry
<i>Typha latifolia</i> L.	E	**	***	*	
<i>Sagitaria montevidensis</i> CHAM. et SCHLTDL.	E	*	*		
<i>Eleocharis montana</i> (H.B.K.) ROEM. Et Schult.	E	*		**	
<i>Polygonum punctatum</i> Elliot	E	*	**	*	
<i>Iris pseudacorus</i> L.	E			***	*
<i>Gymnocoronis spilanthoides</i> (Don) DC.	E	*		**	**
<i>Schoenoplectus californicus</i> (C. A. Meyer) Soják	E			***	***
<i>Lilaeopsis</i> sp. Greene	E	*			
<i>Hydrocotyle ranunculoides</i> L.	F-L	***	*	*	**
<i>Althernantera philoxeroides</i> (Mart.) Griseb.	F-L	**	**		
<i>Ludwigia peploides</i> (Kunth) P. H. Raven	F-L	**	**		*
<i>Lemna gibba</i> L.	F-F		**		**
<i>Myriophyllum aquaticum</i> (Vell.) Verdc.	S			*	*
<i>Stuckenia pectinata</i> (L.) Börner [= <i>Potamogeton pectinatus</i> (L.)]	S			*	
<i>Ceratophyllum demersum</i> L.	S				**
Total coverage (%)		67	87	85	84

Figures

Figure 1

Map of the study area and the sampling sites located in Buenos Aires province, Argentina. Land use areas modified from Solis et al. (2019).

Figure 2

Principal Component Analysis of the analyzed physico-chemical variables in the four riverine wetland studied in the sampled period. Circles: agricultural land use; squares: livestock land use. Closed symbols: normal period; open symbols: dry period. See Table 1 for abbreviations.

Figure 3

Mean and standard error of macroinvertebrate taxa richness, density and Shannon diversity in the agricultural and livestock RWs in the two hydrological periods.

Figure 4

Relative abundance of macroinvertebrate functional feeding groups in the agricultural and livestock RWs in the two hydrological periods.

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