Longitudinal patterns in distribution of native and non-native fish species in a regulated temperate Neotropical river

Padrões longitudinais na distribuição de espécies nativas e não-nativas de peixes em um rio Neotropical temperado regulado

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Abstract: Aim: We evaluated the longitudinal patterns in distribution of native and non-native fish species in a hydrologically fragmented and environmentally variable lowland temperate river. Methods: Four sites representing contrasting habitat and environmental conditions were sampled: a clear water reservoir, a turbid water lagoon and two river reaches with clear and turbid waters each. Environmental variables were measured in situ and in the laboratory. Fishes were sampled using trammel and beach seine nets. Results: Two main environmental scenarios were identified: the upstream reaches, with colder, clearer and nutrient-oxygen poor waters (reservoir and its downstream river) and the downstream reaches, where turbidity, dissolved oxygen, water temperature, conductivity and nutrients largely increased (lagoon and its downstream river). Fourteen species with a high non-native/native (4:10) ratio were collected. Non-native species (NNS) were confined to lentic conditions, where the silverside Odontesthes bonariensis dominated. Native species (NS) better thrive in lotic conditions where the turbid scenario further favored tolerant species. Environmental conditions also seemed to influence the distribution of NNS. Fish assemblage structure considering either, all species, NNS or NS significantly differed among sampled reaches and habitat (lentic-lotic) conditions. Total fish abundance was higher in lentic reaches. Species richness and diversity were favored by the turbid scenario. Beta diversity was mostly explained by the replacement component revealing the substitution of species as the main pattern of variation. Water conductivity, nitrates and dissolved oxygen were the most important predictor variables in the best and most frequent explanatory models of fish assemblage structures. Conclusions: Our results revealed that a low diversified Neotropical fish fauna is disrupted by habitat fragmentation due to the creation of artificial impoundments and the introduction of NNS. Environmental conditions further modulate the fish assemblage structure by affecting the distribution of species where tolerant species were favored by turbid, nutrient-rich waters with higher conductivity and pH.

Keywords: reservoir; eutrophication; non-native species; fishes; Sauce Grande river.
In river ecosystems, habitat fragmentation ranges from local incisions of riparian banks to large reservoirs. According to the Serial Discontinuity Concept (Ward & Stanford, 1983), dams result in upstream-downstream shifts in biotic and abiotic patterns and processes. Dam construction alters the natural dynamics of rivers, changes the environmental characteristics (lotic into a lentic environment), decreases nutrient concentrations (after filling process) and increases water transparency (Agostinho et al., 2007). These changes drastically impact on fish communities (Agostinho et al., 2016; Turgeon et al., 2019). Particularly, this scenario favors an abundance increase of limnetic native fish species (Johnson et al., 2008; Gubiani et al., 2018), obstructs the dispersal and migration of organisms (Penczak & Kruk, 2000) and can cause shifts in community structure (Liew et al., 2016; Smith et al., 2018). The disruption of natural scenario imposed by large dams and reservoirs also enhances the establishment non-native species (Johnson et al., 2008; Liew et al., 2016). Overall, a declining abundance of native endangered and threatened species and increasing the abundance of non-native species are expected in regulated rivers (Guenthier & Spacie, 2006).

Non-native species are causing dramatic changes in many ecological systems worldwide, and are profoundly altering the natural communities (Cucherousset & Olden, 2011; Pyšek et al., 2020). Although not all introduced fishes become established, many exert significant ecological, evolutionary, and economic impacts (Jeschke & Strayer, 2005). Ecological effects have been shown to be severe and range from behavioral shifts of native species in the presence of invaders to changes in food webs and the extirpation of entire faunas.
The Sauce Grande River belongs to a large hydrographic basin located in the south of Buenos Aires province, originated by small order streams that head in the Sierra de la Ventana, at 500 m above sea level. Shortly beyond its source, the river is artificially impounded by the Paso de las Piedras Reservoir. With a surface area rounding 3,600 hectares, this reservoir averages 8.2 m of depth with a maximum of 28 m (Fernández et al., 2009). Its main use is to provide drinking tap water to Bahía Blanca (around 300,000 inhabitants) and Punta Alta cities (around 60,000 inhabitants) and the industries of “Ingeniero White”. This system is also used for recreational fishing of the inland silverside Odontesthes bonariensis, certainly, combined impacts of habitat fragmentation, eutrophication and non-native fish species on fish assemblages of the Pampa Plain represent a major threat to native fish fauna. The lowland rivers of this region are therefore a good opportunity to improve our understanding about patterns of native fish fauna under multiple stressors and the potential ecological impacts of non-native fishes. In this paper, a survey of fish assemblages in a hydrologically fragmented and environmentally variable temperate Neotropical river of the Pampa Plain, Argentina, was conducted. Particularly, the aim of this paper was to evaluate patterns in abundance and distribution of native and non-native fish species and its relationship with key limnological variables along several reaches of this river with contrasting habitat and environmental conditions.

It could be anticipated that habitat fragmentation imposed by dams will drastically affect fish abundance and distribution, particularly favoring non-native fishes in these environments. On the contrary, native reophilic species would hardly thrive in these modified conditions. Longitudinal variation in key environmental variables is also expected to affect fish assemblages. The magnitude and direction of these relationships are not easily anticipated other than the most tolerant fishes would be favored by the most detrimental environmental conditions. Overall, understanding the impact of multiple stressors on the fish assemblages would lead to thoughtful management and conservation actions that necessarily must be taken on strong ecological grounds about the target species.

2. Material and Methods

2.1. Study area

The Sauce Grande River is part of a large hydrographic basin located in the south of Buenos Aires province, originated by small order streams that head in the Sierra de la Ventana, at 500 m above sea level. Shortly beyond its source, the river is artificially impounded by the Paso de las Piedras Reservoir. With a surface area rounding 3,600 hectares, this reservoir averages 8.2 m of depth with a maximum of 28 m (Fernández et al., 2009). Its main use is to provide drinking tap water to Bahía Blanca (around 300,000 inhabitants) and Punta Alta cities (around 60,000 inhabitants) and the industries of “Ingeniero White”. This system is also used for recreational fishing of the inland silverside Odontesthes bonariensis (ADA, 2017). Downstream, river
flows with oligo-mesotrophic, hipohaline and clear waters (Cony, 2018) until it is naturally impounded by coastal sand dunes creating the Sauce Grande Lagoon. This lagoon has an area of 21.55 km², an average depth of 1.4 m and a maximum depth of 1.8 m (Fornerón et al., 2010). This Lagoon was defined as a eutrophic, oligohaline and turbid environment dominated by phytoplankton (Cony, 2018). As in the reservoir, the Sauce Grande Lagoon is periodically stocked with inland silversides to support recreational fisheries. Downstream to the Sauce Grande Lagoon, the river flows into the Atlantic Ocean after a meandering course of 30 km.

2.2. Field and laboratory activities

Samplings were conducted in four sites along the Sauce Grande River aimed to represent the contrasting habitat and environmental conditions to which fishes are exposed in this ecosystem. Sampling locations were placed in Paso de las Piedras Reservoir, an artificial impoundment with clear waters (labeled CwLe=clear water lentic conditions; 38°24’16.25” S, 61°44’38.87” W), Sauce Grande Lagoon, a natural impoundment with turbid waters (labeled TwLe=turbid water lentic conditions; 38°56’18.51” S, 61°22’56.02” W) and two reaches of Sauce Grande River, downstream to Paso de las Piedras Reservoir, a river reach with clear waters (labeled CwLo=clear water lotic conditions; 38°45’47.00” S, 61°42’43.09” W) and downstream to Sauce Grande Lagoon, a river reach with turbid waters (labeled TwLo=turbid water lotic conditions; 38°57’5.94” S, 61°13’46.63” W) (Figure 1). Each sampling site was visited three times during spring (October), summer (March) and autumn (May) months.

Environmental variables were measured in situ including water temperature (°C), dissolved oxygen (mg/L), water conductivity (µS/cm), pH and turbidity (NTU) using a multiparameter probe (Horiba U-53G). Nitrate and total phosphorus (mg/L) were quantified after processing water samples in the laboratory according to standard methods (APHA, 2012). In Paso de las Piedras Reservoir, all these variables were gathered from the Autoridad del Agua de la Provincia de Buenos Aires (ADA, 2018). Materials and methods for field and laboratory water analysis by ADA are the same as those employed in this study.

Fish sampling was performed using trammel and beach seine nets. Trammel nets were constructed with an outer mesh size of 80 mm between knots and an inner mesh size of 15 mm between knots. A total of 12 m were deployed at each river reach and 100 m in both lentic environments. Beach seine net was constructed with 15 m long wings (10 mm mesh) and a 2 m long bag (5 mm mesh). These gear devices were probed to successfully collect all fish species intended to be collected in this biogeographic region (Bertora et al., 2018a). Trammel nets were left overnight and 2 to 4 seining were performed at each site during each sampling.

Figure 1. Location of sampling sites in the Sauce Grande River. CwLe = clear water lentic conditions at Paso de las Piedras Reservoir; CwLo = clear water lotic conditions at reach of Sauce Grande River downstream of the reservoir; TwLe = turbid water lentic conditions at Sauce Grande Lagoon; TwLo = turbid water lotic conditions at reach of Sauce Grande River downstream of the lagoon.
date. No additional species was caught beyond the third seining. Fish sampling and handling protocols followed during the course of our surveys were evaluated and approved by the Ethics Comitee of the ‘Facultad de Ciencias Exactas y Naturales’ of the ‘Universidad Nacional de Mar del Plata’ (RD-2018-126).Fish captured were euthanized by an overdose in benzocaine solution as suggested by international guidelines (Barker et al., 2002). Fish species were identified following Ringuette et al. (1967) and Rosso (2006).

2.3. Data analyses

2.3.1. Spatial autocorrelation test

In ecological studies, where the sampling sites are arranged along the same longitudinal gradient, the spatial auto-correlation of the data must be tested. Therefore the influence of the longitudinal position of sites along the main stem of the river on environmental variables, fish species abundance and assemblage attributes were analyzed to identify if our results reflect the patterns of studied variables rather than geographic distance between sampled reaches. So, two RELATE routines (Clarke & Gorley, 2015) were performed to verify the correlation between the serial model and the faunal (abundances and attributes) and environmental dissimilarity matrices. The faunal matrix was constructed using Bray-Curtis similarity and the environmental matrix was constructed using Euclidean distance. This analysis determines the level of association between two resemblance matrices, in this case, the serial model matrix (linear distance between sampled sites in km) and the faunal Bray-Curtis similarity matrix or the environmental Euclidean distance matrix. The linear distances between sampled sites were as follows: CwLe-CwLo 40 km, CwLo-TwLe 35 km and TwLe-TwLo 11 km. All data were previously standardized (to zero mean and unit variance). Spearman correlation was used and the permutation tests were performed with 9999 random permutations. The null hypothesis postulating that there is no spatial autocorrelation of the data was tested, that is no tendency for variables to be similar at nearby localities.

2.3.2. Environmental conditions

A Principal Component Analysis (PCA) was conducted with all environmental variables measured at each sampling site. This analysis orders and represents continuous multivariate data in a smaller dimension (standardized orthogonal linear combinations of the variables) that explain the data variability (proportion of total variability explained). The biplot allows visualizing observations and variables in the same scatterplot, thus it is possible to identify associations between observations, between variables and between variables and observations. To perform this analysis data were standardized and correlation matrix was used. Statistical significance in longitudinal and habitat (lentic vs. lotic) variation of environmental conditions were tested by means of an ANOSIM procedure (Clarke & Green, 1988) conducted on the Euclidean similarity matrix of measured variables.

2.3.3. Fish assemblages

Fish species abundances were expressed as capture per unit of effort (CPUE total). The unit of effort was standardized to an overnight deployment of 12 m of trammel nets and one seining of beach seine net averaging 10 m long. Relative specific abundances were transformed to square root to reduce the weight of the dominant species. In addition, different fish assemblage attributes were estimated. Abundance of non-native species (CPUE NNS) was calculated including exotic (C. carpio) and allochthonous (O. bonariensis, Cyphocharax voga and P. trucha) species. Exotic species were those considered phylogenetically and biogeographical distant that are introduced, typically from other continents. Conversely, allochthonous species were Neotropical or Austral fish fauna without evidence of being naturally present in the studied basin. Abundance of native species (CPUE NS) was estimated including the remaining captured species. Species richness and Shannon-Weaver diversity were calculated for each site. A one way Kruskal-Wallis ANOVA and Mann-Whitney tests were used to test for significance in differences of fish assemblages attributes (CPUE NNS, CPUE NS, species richness and Shannon-Weaver diversity) among sites and between habitats (lentic vs. lotic) respectively. In order to determine if fish assemblage structure in contrasting river reaches significantly differ considering the abundances of all fishes, abundances of NS or abundances of NNS, three separate one-way ANOSIM analyses were performed, using different sites as factors. These analyses were conducted on the Bray-Curtis similarity matrix generated with the specific abundances. ANOSIM analyses were also used to evaluate the differences in fish assemblage structure between lotic (CwLo and TwLo) and lentic conditions (CwLe and TwLe), using the habitat as a factor. All the similarity matrices generated in ANOSIM tests
were used to perform the analysis of non-metric multidimensional scaling (nMDS; Clarke & Green, 1988) in order to visually inspected dissimilarities. To further explore structural resemblance between fish assemblages among different environmental and habitat conditions, a similarity percentages analysis (SIMPER; Clarke, 1993) was conducted. This test highlights those species that contribute most to explain the dissimilarities between observed assemblages. In addition, beta diversity (β diversity) was calculated to explore patterns in composition of the fish communities along the longitudinal gradient (Carvalho et al., 2012). The total beta diversity based on Jaccard dissimilarity coefficient was partitioned into the replacement (β repl.) and richness difference (β rich.) components (Baselga, 2010).

2.3.4. Relationships between environmental conditions and fish assemblages

The PCA scores were used as an integration of the environmental variables to determinate possible associations between this environmental scenario and the fish community. Spearman rank correlation coefficients were estimated to explore the empirical relationships between fish assemblage attributes and specific abundances with PCA scores of environmental variables.

Distance-based linear models (DistLM) were performed to achieve a direct quantitative partitioning of the multivariate variability that is explained by each of several environmental variables. DistLM is a routine for analyzing and modeling the relationship between a response multivariate data cloud (biological variables) and one or more predictor variables (environmental variables) (Anderson et al., 2008). This tool does a partitioning of variation in a data cloud described by a resemblance matrix according to a regression model. We used the same response matrices included in the nMDS-ANOSIM protocol: the abundances of all fish species combined, abundances of NS and abundances of NNS. Abundance data was square-root transformed prior to analyses. Draftsman plots were performed to check for skewness and redundancy among predictor variables. Then, environmental variables were log transformed and the turbidity was excluded from the analysis, as it was highly correlated with other variables. Models including all possible combinations of predictor variables were generated using the BEST procedure. Modified Akaike’s Information Criterion (AICc) was used to identify the best model. Models with the lowest AICc are considered the most parsimonious. All DistLMs were run with 9999 permutations. The difference between the AICc value of the best model and each of the other models (ΔAICc) was calculated and the Akaike weights of models (Burnham & Anderson, 2002) with values of these differences less than 2 were estimated. Burnham & Anderson (2002) suggested that models having AICc values within 2 units of the best model should be examined more closely to see if they differ from the best model by 1 variable. Thus, those models that showed a ΔAICc <2, differed from the best model by a single variable and obtained an increase of $R^2$ were selected to complement the best model in explaining the relationships between environmental variables and fish assemblages. In these models (ΔAICc <2), the relative importance of each predictor variable (Wi, predictor weight, Symonds & Moussalli, 2011) was calculated. For each predictor variable, the Akaike weights of all the models containing that predictor were summed. Those predictors that frequently occur in the most likely models (ΔAICc <2) have an Akaike weight close to 1 whereas variables that are absent from most likely models or are only present in less likely models (high AICc values) have an Akaike weight close to 0.

Statistical analyses were performed with PRIMER.5 (Plymouth Routines In Multivariate Ecological Research) with the add-on package PERMANOVA+, PAST 4.01 (Paleontological Statistics software package for education and data analysis) and InfoStat packages.

3. Results

3.1. Spatial autocorrelation test

The spatial autocorrelation analyses showed that there was no tendency for fish assemblage ($R= 0.062, p$ value= 0.473) nor environmental variables ($R= 0.031, p$ value= 0.572) to be more similar at nearby localities in the Sauce Grande River.

3.2. Environmental conditions

Sampled sites showed significant differences in their main environmental attributes (ANOSIM: $R= 0.756, p$ value= 0.0002, Table 1). Along the longitudinal gradient of the river, different reaches displayed a progressive increase in dissolved oxygen and temperature. Although all sampled sites had alkaline waters, river reaches with more turbid waters had the highest pH values. In addition to
their higher turbidity, these sites also showed higher conductivity and nutrient concentrations. No significant differences were found in environmental variables according to the habitat condition of the sites (lentic vs. lotic, ANOSIM: R= 0.059, p value= 0.226).

The first two axes of the Principal Component Analysis cumulatively explained 86.8% of the total variation in environmental conditions along the sampling sites of the Sauce Grande River (Figure 2). This analysis showed a marked spatial ordination of samples along the first component. The CwLe and CwLo samplings were closely grouped in the negative end of the first component, while TwLo samples were located around the centroid of the ordination. The TwLe samples were confined to the positive extreme of the first component. This spatial arrangement allowed discriminating sites along a gradient of progressive increase in pH, turbidity, conductivity and nutrient concentrations (Table 1). In the second principal component, sites were located from negative to positive end according to a progressive increase in water temperature and dissolved oxygen concentration. Clear water reaches (CwLe and CwLo) with less nutrient concentrations and water conductivity (negative end of CP1) also showed colder and less oxygenated waters. The warmer and more oxygenated waters (positive end of CP2) were only observed under turbid and nutrient-rich conditions, a situation more variable in lentic turbid reaches.

Table 1. Environmental variables measured in different sampling sites of the Sauce Grande River.

| Environmental variables | Code | CwLe | CwLo | TwLe | TwLo | PC1 | PC2 |
|-------------------------|------|------|------|------|------|-----|-----|
| dissolved oxygen (mg/L) | DO   | 6.58 | 9.8  | 6.88 | 0.29 | 8.61 | 2.34 |
| nitrate (mg/L)          | NO3  | 1.14 | 0.41 | 0.63 | 0.12 | 3.7  | 0.73 |
| pH                      | temp | 8.21 | 0.27 | 8.56 | 0.51 | 9.97 | 0.04 |
| temperature (°C)        | temp | 15.06| 5.39 | 18.53| 1.23 | 20.87| 3.8 |
| total phosphorus (mg/L) | TP   | 0.23 | 0.16 | 0.87 | 0.12 | 3.43 | 0.99 |
| turbidity (NTU)         | turb | 8.15 | 1.1  | 5.33 | 1.53 | 453.33| 159.48|
| water conductivity (µS/cm) | WC   | 411.78| 22.28| 1680| 216.56| 7143.33| 442.3 |

CwLe = clear water lentic conditions; CwLo = clear water lotic conditions; TwLe = turbid water lentic conditions; TwLo = turbid water lotic conditions; avg = average; SD = standard deviation; PC1 and PC2: correlation coefficients between environmental variables and the first two principal components.

Figure 2. Biplot of the first two PCA axes based on environmental data of the Sauce Grande River (CwLe = clear water lentic conditions; CwLo = clear water lotic conditions; TwLe = turbid water lentic conditions; TwLo = turbid water lotic conditions; DO = dissolved oxygen; temp = temperature; NO3 = nitrate; WC = water conductivity; TP = total phosphorus; turb = turbidity; CP1 = First Principal Component; CP2 = Second Principal Component).
3.3. Fish assemblages

The samplings yielded 14 species belonging to 7 orders and 10 families (Table 2). Characiformes and Siluriformes were the most represented orders with 5 and 3 species respectively. All families but Characidae and Heptapteridae presented one single species. Non-native species (NNS) like *C. carpio*, *C. voga*, *O. bonariensis* and *P. trucha* were recorded. Overall, the most abundant species were *O. bonariensis*, *O. jenynsii*, *C. interruptus* and *J. lineata* (Table 2). Eight species showed a widespread distribution, being collected in all sampling sites along the study river: *B. iheringii*, *C. interruptus*, *C. paleatus*, *J. lineata*, *O. jenynsii*, *P. laticeps*, *P. pampa* and *R. quelen* (Table 2). However, their spatial patterns in abundance distribution were highly variable (Figure 3). The small piscivorous *O. jenynsii* showed similar abundances in different sites, while the abundance distribution of *P. pampa* and *P. laticeps* was highly skewed to lotic conditions and *J. lineata* mostly occurred in lentic conditions. Species like the small characids *B. iheringii* and *C. interruptus* and three Siluriformes, *C. paleatus*, *P. laticeps* and *R. quelen*, presented a minimum abundance in the reservoir (CwLe). In turn, the two more distant downstream reaches accounted for the largest abundances of *B. iheringii*, *C. interruptus* and *C. paleatus* (Table 2, Figure 3). In contrast to the widespread species, there were some species confined to a single sampling site, two NNS, *P. trucha* and *C. carpio* in CwLe and the cichlid *A. facetus* in TwLo. The inland silverside *O. bonariensis* although it was collected in all sites except CwLo, it was noticeably more abundant in lentic reaches. The same was observed for the detritivorous *C. voga*, a species only found in lentic environments, being most abundant in the lagoon. *C. decemmaculatus* was the only species being preferentially collected in the lotic reach downstream of the reservoir (CwLe).

Due to the spatially variable abundance and distribution of fish species along sampling sites, significant differences were found in the structure of fish assemblages in different reaches of the Sauce Grande River (ANOSIM: *R* = 0.614, *p* value = 0.001, Figure 4a). In the reservoir (CwLe) the fish community was dominated by the inland silverside *O. bonariensis*, followed by the small piscivore *O. jenynsii* and the one-sided livebearer *J. lineata* (Figure 4b). However, in lotic reaches downstream to the reservoir (CwLo and TwLo), the proportion of the inland silverside sharply decreased and the contribution of *C. interruptus*, *B. iheringii*, *C. decemmaculatus*, *C. paleatus* and *P. pampa* to the fish community structure was more noticeably. In the lagoon (TwLe), *O. bonariensis* was again the most represented species and the contributions of the remaining species closely resembled those observed in lotic reaches.

Significant differences were also observed in the structure of fish assemblages (Figure 4a) between the studied sites if the abundances

![Figure 3](https://via.placeholder.com/150)

**Figure 3.** Bar chart showing the distribution of total fish collected of each species within the Sauce Grande River (CwLe = clear water lentic conditions; CwLo = clear water lotic conditions; TwLe = turbid water lentic conditions; TwLo = turbid water lotic conditions). Species are intentionally sorted by means of their spatial distributions to ease the interpretation. Species codes as listed in Table 2.
Table 2. Site-specific and cumulative (Total) abundance (in CPUE, SD = standard deviation) of fish species collected, richness and diversity of sampled sites in Sauce Grande River.

| Order          | Family                  | Species                  | Code | CwLe total mean SD | CwLo total mean SD | TwLe total mean SD | TwLo total mean SD | Total |
|----------------|-------------------------|--------------------------|------|--------------------|--------------------|--------------------|--------------------|-------|
| Atheriniformes | Atherinopsidae          | Odontesthes bonariensis* | OB   | 123.61 41.2 43.94  | 39 13 2.6          | 2.5 0.83 0.76      | 165.11             |
| Characiformes  | Characidae              | Psalidodon pampa         | PP   | 0.04 0.01 0.02     | 12 4 2.18          | 4 1.33 1.15        | 9.5 3.17 1.04      | 25.54 |
|                | Bryconamericus theringii|                         | BI   | 6.33 2.1 2.8       | 9 3 1              | 18.5 6.17 0.76     | 11.5 3.83 1.61     | 45.33 |
|                | Cheirodon interruptus   |                         | CI   | 0.04 0.01 0.02     | 16.5 5.5 2.18      | 25 8.33 2.08       | 28.5 9.5 3.04      | 70.04 |
|                | Oligosarcus jenynsi     |                         | OJ   | 20.43 6.8 7.1      | 14.5 4.83 0.76     | 20.5 6.83 1.76     | 17.5 5.83 2.57     | 72.93 |
|                | Cyphocharax voga*       |                         | CV   | 0.35 0.12 0.07     | 10 3.33 1.53       | 10 3.33 1.53       |                    | 3.33  |
| Cichliformes   | Cichlidae               | Australoheros facetus   | AF   |                    |                    |                    |                    | 3     |
| Cypriniformes  | Cyprinidae              | Cyprinus carpio*        | CC   | 0.16 0.05 0.05     |                    |                    |                    | 0.16  |
| Cyprinodontiformes | Anablepidae           | Jenynsia lineata       | JL   | 19.33 6.44 10.03   | 5 1.67 1.53        | 26.5 8.83 1.26     | 13.5 4.5 1.32      | 64.33 |
| Poeciliidae    | Cnestherodon decemmaculatus |                   | CD   | 13 4.33 1.26       | 5 1.67 1.53        | 4.5 1.5 1.8        |                    | 22.5  |
| Perciformes    | Percichthyidae          | Percichthys trucha*     | PT   | 0.08 0.03 0.02     |                    |                    |                    | 0.08  |
| Siluriformes   | Callichthyidae          | Corydoras paleatus     | CP   | 0.42 0.1 0.2       | 5.5 1.83 1.61      | 6.5 2.17 1.53      | 17 5.67 0.29       | 29.42 |
| Heptapteridae  | Pimelodella laticeps    |                         | PL   | 0.04 0.01 0.02     | 5.5 1.83 1.26      | 3 1 1              | 7.5 2.5 1.8        | 16.04 |
|                | Rhamdia queien          |                         | RQ   | 0.2 0.07 0.03      | 3 1 1              | 5 1.67 1.53        | 3 1 0              | 11.2  |
|                |                         |                         |      |                    | 8.33 0.58          | 8 1                | 9.67 0.58          | 10 1  |
|                |                         |                         |      |                    | 0.65 0.14          | 1.95 0.12          | 2.05 0.05          | 2.05  |

CwLe = clear water lentic conditions; CwLo = clear water lotic conditions; TwLe = turbid water lentic conditions; TwLo = turbid water lotic conditions; *non-native species.
Figure 4. (a) Non-metric multidimensional scaling of the abundances of all fish species (ALL), only native species (NS) and only non-native species (NNS) in sampled sites (CwLe = clear water lentic conditions; CwLo = clear water lotic conditions; TwLe = turbid water lentic conditions; TwLo = turbid water lotic conditions) of the Sauce Grande River (lentic sites: squares, lotic sites: dots; CwLe = gray square; CwLo = gray dot; TwLe = black square; TwLo = black dot); (b) Percentage contribution of specific abundances for the same three data sets (ALL, NS and NNS). The species codes as listed in Table 2. “Others” category in abundances of all fish species (ALL) corresponds to species whose abundance was less than 6.5% at all sites; (c) Mean abundances and standard deviations of fish species for the three data sets.
of native species (ANOSIM: $R = 0.3858$, $p$ value $= 0.0006$) are discriminated from those of non-native species (ANOSIM: $R = 0.591$, $p$ value $= 0.0029$). Moreover, if sites are compared in terms of habitat (lentic versus lotic) conditions, the structure of the fish community showed significant differences (Figure 4a), considering all the collected species (ANOSIM: $R = 0.5481$, $p$ value $= 0.0029$), only the native ones (ANOSIM: $R = 0.2389$, $p$ value $= 0.0049$) and only non-native ones (ANOSIM: $R = 0.5907$, $p$ value $= 0.002$). Lotic reaches were almost devoid of NNS, except for the record of some specimens of *O. bonariensis* collected in TwLo (Figure 4b). All NNS inhabiting the Sauce Grande River were collected in CwLe (reservoir) and two of them, *C. voga* and *O. bonariensis*, were also found (Table 2, Figure 3) in TwLe (lagoon). In CwLe the combined abundance of NNS accounted for the 75% of the overall fish abundance, while in TwLe it slightly surpassed the 25% of total fish abundance (Figure 4b). Overall, the species that contributed most to explain these dissimilarities in different habitat and environmental conditions were the inland silverside *O. bonariensis*, *C. decemmaculatus*, *C. interruptus*, *C. paleatus* and *J. lineata* (Table 3). The same patterns were observed when native and non-native species were considered separately.

Regarding the attributes of the fish community some general patterns were observed (Table 2, Figure 4c). Overall, abundances of NS (Kruskal-Wallis $H = 8.88$, $p$ value $= 0.0267$), species richness (Kruskal-Wallis $H = 6.67$, $p$ value $= 0.0683$) and assemblage diversity (Kruskal-Wallis $H = 7.31$, $p$ value $= 0.0627$) were significantly (some marginally) different among sites. The abundances of NS and diversity showed a certain longitudinal tendency, being highest in the most downstream reaches, TwLe and TwLo. The reservoir showed the less diversified community and the species richness was higher in sites with turbid, nutrient-rich waters. Total fish abundance was considerably higher in lentic reaches albeit the wide dispersion of values observed in the reservoir precludes a significance in tested differences (Mann-Whitney $W = 49$, $p$ value $= 0.1212$). Instead, when considering only the NNS, abundance was significantly higher in lentic reaches (Mann-Whitney $W = 57$, $p$ value $= 0.0022$).

The analysis of beta diversity showed that lotic reaches were fairly similar ($\beta$ diversity average CwLo-TwLo $= 0.25$, Figure 5) whereas the largest difference was found between CwLe and the lotic reach downstream to reservoir ($\beta$ diversity average CwLo-CwLe $= 0.67$). The similarity found between the lotic reaches and the lagoon (CwLo-TwLe $= 0.39$, TwLo-TwLe $= 0.26$) is greater than that found with the reservoir (CwLo-CwLe $= 0.67$, TwLo-CwLe $= 0.58$). Their beta diversity patterns were mostly explained by the replacement component. However, assemblage dissimilarities between lotic reaches were mainly due to the richness differences component (absolute gain or loss of species).

### Table 3. Contribution percentage of specific abundances to different sampling sites in the Sauce Grande River using SIMPER analysis.

| Species       | CwLe vs. CwLo | CwLe vs. TwLe | CwLe vs. TwLo | CwLo vs. TwLe | CwLo vs. TwLo | TwLe vs. TwLo |
|---------------|---------------|---------------|---------------|---------------|---------------|---------------|
| A. facetus    | <1            | 3.87          | 0             | 9.46          | 6.41          |
| B. iheringii  | 5.62          | 5.67          | 5.66          | 4.95          | 4.75          |
| C. carpio     | <1            | 1.06          | <1            | 0             | 0             |
| C. decemmaculatus | 11.03   | 4.99          | 7.55          | 13.63         | 7.14          |
| C. interruptus | 12.02         | 15.24         | 4.76          | 9.65          | 3.81          |
| C. paleatus   | 5.24          | 10.45         | 5.59          | 16.02         | 8.17          |
| C. voga       | 1.7           | 1.61          | 13.33         | 0             | 14.8          |
| J. lineata    | 9.4           | 9.1           | 14.45         | 13.22         | 7.17          |
| O. bonariensis | 27.65         | 22.75         | 26.76         | 8.74          | 23.8          |
| O. jenynsii   | 5.02          | 5.46          | 3.2           | 5.91          | 4             |
| P. laticeps   | 6.56          | 6.96          | 5.25          | 7.14          | 7.37          |
| P. pampa      | 10.3          | 8.57          | 7.63          | 5.58          | 6.7           |
| P. trucha     | <1            | <1            | 0             | 0             | 0             |
| R. quelen     | 3.83          | 3.77          | 5.81          | 5.7           | 5.89          |

CwLe = clear water lentic conditions; CwLo = clear water lotic conditions; TwLe = turbid water lentic conditions; TwLo = turbid water lotic conditions.
3.4. Relationships between environmental conditions and fish assemblages

Different fish assemblage attributes and specific abundances were intimately associated with particular environmental conditions (Table 4). The total fish and NS abundances, species richness, assemblage diversity and the abundance of some species such as B. iberingii and C. interruptus seemed to be favored by environments with high pH, turbidity, conductivity and nutrient concentrations, as summarized by the first component. In turn, A. facetus, C. paleatus and P. pampa were most abundant in warmer, oxygen-rich waters.

For the fish assemblage structure considering all fish species, the best explanatory model (lowest value of AICc) had two environmental variables: nitrates and water conductivity (Table 5). With the inclusion of dissolved oxygen, the second best model substantially improved its explanation power (47.9 to 62.7%) with a minimum increase (0.71) in the AICc. These three variables also composed the best model (72.8%) for the fish assemblages considering the abundances of NNS. Instead, when only NS abundances were considered, the best model incorporated water conductivity as the unique variable with a small R² (29.7%). With the inclusion of total phosphorous, its explanatory power accounted for 45.8% of the observed variation. Given its frequency of occurrence in models proposed and the weight of these models, the nitrate ($W_i^{all species} = 0.595$, $W_i^{NNS} = 1$) and water conductivity ($W_i^{all species} = 0.733$, $W_i^{NNS} = 0.680$) were the most important predictor variables for the fish assemblages considering the abundance of all fish species, adding dissolved oxygen ($W_i = 0.727$) for the abundances of NNS. For fish assemblages of NS abundances, the water conductivity ($W_i^{NS} = 0.675$) was the most frequently predictor variable occurring in the models.

4. Discussion

4.1. Environmental conditions

Environmental conditions significantly differed among sampled reaches. In this context, two main environmental scenarios can be roughly identified in the Sauce Grande River. One is composed by the
upstream, less alkaline, clearer, colder and nutrient-oxygen-poor waters represented by the reservoir and its downstream lotic reach. The other, is composed by the more distant downstream Sauce Grande Lagoon and its downstream lotic reach, where turbidity, nutrients, oxygen, water conductivity and temperature largely increases. Indeed, reaches with colder, clearer and nutrient-oxygen poor waters grouped tightly at one extreme of the ordination irrespective of their lentic or lotic conditions. The lower values of conductivity, turbidity and nutrient concentrations in the reservoir closely resemble a pattern recurrently observed in these artificial water bodies (Agostinho et al., 2007). Similar

abiotic conditions in the river reach downstream to the reservoir were observed. About this, Baxter (1977) postulated that heat, silt and nutrients retained in reservoirs are lost to the streams. On the other hand, the enhanced productivity commonly incorporated by floodplain lagoons to lowland river ecosystems (Junk et al., 1989) may largely explain the eutrophic conditions observed in downstream reaches. The more eutrophic downstream reaches of this river were also characterized by higher dissolved oxygen concentration, pH, turbidity and water conductivity. The eutrophication is an inevitable process given in a course from the source to mouth caused by drainage from cultivated and inhabited districts (Butcher, 1947). In this respect, changes in land-uses could be considered as one of the main drivers for the observed deterioration of water quality in Argentina (Rosso & Fernández Cirelli, 2013; Amuchástegui et al., 2016) as in many parts of the world (Allan, 2004). The impact on the thermal condition and oxygenation of waters seemed evident in the reservoir and its downstream river reach, where the lowest temperatures and dissolved oxygen concentrations were recorded. Under natural conditions, the small volume of water in a river reach and turbulent mixing ensure that river water responds to changes in the meteorological conditions and incorporate atmospheric oxygen (McCartney, 2009). In contrast, large masses of still water in reservoirs allow thermal stratification and the occurrence of cold, deoxygenated waters in the hypolimnion (Beutel & Horne, 1999). Nevertheless, dissolved oxygen concentration in the lower layer of Paso de las Piedras Reservoir is close to the saturation value almost throughout all the year (Estrada et al., 2011) due to the lack of stratification (Intartaglia & Sala, 1989). Therefore, reservoir-derived reduction of dissolved oxygen is less likely and its increase by enhanced primary

| Table 4. Spearman rank correlation coefficients between fish assemblage attributes and specific abundances with PCA scores of environmental variables. |
|-----------------|-----------------|-----------------|
|                | PCA axis 1      | PCA axis 2      |
| AF abundance   | 0.25            | 0.57*           |
| BI abundance   | 0.81*           | 0.13            |
| CC abundance   | -0.46           | -0.15           |
| CD abundance   | 0.11            | -0.16           |
| CI abundance   | 0.71*           | 0.02            |
| CP abundance   | 0.51            | 0.66*           |
| CV abundance   | 0.37            | -0.47           |
| JL abundance   | 0.47            | -0.2            |
| OB abundance   | 0.08            | -0.23           |
| OJ abundance   | 0.51            | 0.39            |
| PL abundance   | 0.32            | 0.12            |
| PP abundance   | 0.2             | 0.57*           |
| PT abundance   | -0.45           | -0.13           |
| RQ abundance   | 0.43            | 0.41            |
| total abundance| 0.62*           | 0.11            |
| NS abundance   | 0.89*           | 0.25            |
| NNS abundance  | 0.08            | -0.23           |
| richness       | 0.78*           | 0.23            |
| diversity      | 0.74*           | 0.32            |

*p value<0.05. prop. NNS = proportion of non-native species. Species codes as listed in Table 2.

| Table 5. First 10 best overall models found for fish assemblages of Sauce Grande River using the AICc criterion. |
|---------------------------------------------------------------|
| N | Variables | AICc | R²   | N | variables | AICc | R²   | N | variables | AICc | R²   |
|---|-----------|------|------|---|-----------|------|------|---|-----------|------|------|
| 2 | NO₃, WC   | 84.48| 0.479| 1 | WC        | 82.83| 0.297| 3 | NO₃, DO, WC| 93.18| 0.728|
| 3 | NO₃, DO, WC| 85.19| 0.627| 1 | TP        | 83.16| 0.278| 2 | NO₃, WC    | 93.98| 0.569|
| 1 | WC        | 85.56| 0.226| 2 | TP, WC    | 83.38| 0.458| 2 | NO₃, DO    | 94.99| 0.531|
| 1 | TP        | 85.67| 0.219| 2 | NO₃, WC   | 83.53| 0.451| 3 | NO₃, TP, DO| 95.09| 0.68 |
| 2 | pH, WC    | 85.85| 0.416| 1 | temp      | 83.88| 0.233| 4 | temp, NO₃, DO, WC| 95.39| 0.806|
| 3 | NO₃, TP, WC| 85.94| 0.603| 2 | pH, WC    | 84.35| 0.412| 2 | NO₃, TP    | 95.39| 0.515|
| 4 | NO₃, TP, DO, WC| 86.31| 0.757| 2 | temp, WC  | 84.48| 0.406| 1 | NO₃      | 95.4 | 0.341|
| 2 | NO₃, TP   | 86.35| 0.391| 3 | NO₃, TP, WC| 84.53| 0.597| 3 | NO₃, pH, WC| 96.14| 0.652|
| 1 | Temp      | 86.36| 0.173| 2 | NO₃, TP   | 84.73| 0.393| 4 | NO₃, TP, DO, WC| 97.41| 0.771|
| 4 | temp, NO₃, DO, WC| 86.97| 0.744| 3 | TP, pH, WC| 85.24| 0.573| 4 | temp, NO₃, TP, DO| 97.62| 0.767|

N = number of variables.
production in downstream reaches is perhaps a more plausible reason for observed differences.

4.2. Fish assemblages in the context of a fragmented habitat and variable environmental conditions

The fish fauna inhabiting the Sauce Grande River represents one of the southernmost occurrences of Brazilian taxa in the continent and it is composed by a very impoverished fraction of the paranaoplatense fish fauna (Ringuelet, 1961). All species previously known from the Sauce Grande River basin (Casciotta et al., 1999; Grosman et al., 2017) were collected in our surveys. Four non-native species were collected but only O. bonariensis developed well established populations (Grosman et al., 2017).

Some general patterns in fish assemblage attributes were observed. Total fish abundance and abundance of NNS were higher in lentic conditions when compared with lotic reaches. In reservoirs, fish abundance is usually higher than that observed in the river before the dam was constructed (Agostinho et al., 2016). This is mostly explained by the development of lacustrine species that take the opportunity of the new created lentic conditions (Johnson et al., 2008; Smith et al., 2018). On the other hand, a lacustrine environment created by a lagoon within the river network enhances the primary productivity (Junk et al., 1989) which certainly may support larger fish populations (Petrere, 1983), including the non-native species. Fish assemblage diversity was extremely low (a third of the lowest value in other reaches) in the reservoir. Artificially impounded habitats worldwide typically develop disparate divergent biological communities from those that originally occurred in the river (Freedman et al., 2014; Perônico et al., 2020). The effects brought by dams, change according to the inherent adaptability of fish communities to respond to the physico-chemical and biological changes (Rosenberg et al., 1997). Neotropical fishes evolved predominantly in flowing waters, and only few species have adaptations to inhabit in lentic environments, such as those created by impoundments (Gomes & Miranda, 2001). Such a lack of traits and plasticity may partly explain the decrease in richness and diversity observed in Sauce Grande reservoir. Similar findings were reported for reservoirs in temperate regions (Turgeon et al., 2019) and in the main river basins in Brazil (Agostinho et al., 2007; Perônico et al., 2020). Total fish abundance, species richness, the abundance of NS and assemblage diversity also seemed to be influenced by environmental conditions since they were all high, positive and significantly related with a gradient of increasing turbidity, water conductivity and nutrient concentrations. An increase in assemblage diversity and species richness is expected in fluvial ecosystems as habitat diversity (Lowe-McConnell, 1975) but also eutrophication (Butcher, 1947) increases. Given the nature of the Pampean lotic ecosystems, with high nutrients concentration and sediment loads (Feijoó et al., 1999), it is expected that the native fish species are adapted and favored by these conditions which could be detrimental for non-native fish species. Indeed, our results revealed that both NS and NNS were influenced by major environmental gradients where water conductivity, nitrates and dissolved oxygen turned to be the most important predictor variables in the best and most frequent explanatory models of the fish assemblage structures.

Spatial patterns in fish species distribution were reflected in the beta diversity analysis. The similarity found between both lotic reaches and the lagoon was greater than that found with the reservoir. This suggests that the marked changes imposed by the reservoir against the natural conditions of the lagoon and lotic reaches, generated drastic changes in the fish communities. Indeed, the highest dissimilarity in fish assemblage composition was found between the reservoir and the lotic reach downstream to the dam. As the similarity between reservoir and its downstream reach regarding environmental conditions was high, the observed differences in beta diversity patterns can be mostly attributed to the fragmented habitat conditions imposed by the dam. In our study, beta diversity patterns were mostly explained by species replacement along the longitudinal river axis, a pattern also observed in several tributaries of the upper Paraná River (Peláez et al., 2017). Species replacement refers to the simultaneous gain and loss of species along ecological gradients, when species tend to replace each other (Legendre, 2014). In regulated rivers it is expected that damming-induced changes affect beta diversity components downstream of the reservoir, contributing to the dominance of species replacement component (Lansac-Tôha et al., 2019). In this context, the increase of the abundance of non-native species is also a well-known consequence of reservoirs in the Neotropic region (Guenther & Spacie, 2006; Liew et al., 2016) and worldwide (Han et al., 2008; Jellyman & Harding, 2012).

The Sauce Grande River showed a high non-native/native species ratio, more noticeably in lentic reaches. Non-native species such as C. carpio and...
P. trucha were only collected in the reservoir. The low number of specimens of these species collected in the Sauce Grande, together with their narrow distribution suggests a recent entrance of these species in this basin. Paradoxically, the presence of both species in the Pampa Plain is known from many decades ago. The exotic Cyprinus carpio was introduced in Argentina in 1925 (Baigún & Quiró, 1985) and its distribution has expanded significantly in the last three decades (Maiztegui et al., 2016). Similarly, P. trucha was intentionally stocked in the Sauce Grande Basin, more than five decades ago (Ringuelet, 1961). This historical context suggests that some local constraints could be hampering the successful establishment of these NNS in this river ecosystem. Cyprinus carpio has a broad tolerance to different environmental factors, including eutrophication and turbidity (Zambrano et al., 1998) that characterize the downstream reaches of the Sauce Grande River. In contrast, the restriction of P. trucha in clear waters would probably also obey to the fact that this environment more closely resembles the natural habitats (low water conductivity and turbidity, low amounts of nutrients) of this species in freshwater ecosystems of Patagonia (Macchi et al., 2007).

Non-native Neotropical allochthonous species, C. voga and O. bonariensis were largely confined to lentic reaches, but their local abundances in these environments were contrasting. Whereas the inland silverside presented its largest population in the reservoir, the detritivorous C. voga was more abundant in the turbid, eutrophic conditions lagoon. The trophic ecology of these species together with the preferential route of energy and food webs in each habitat could help to explain the observed patterns. C. voga is an iliophagous detritivorous fish species (Corrêa & Piedras, 2013). Turbid and productive lakes are more likely to sustain a detritivore assemblage of consumers (Vanni et al., 2006). In contrast, the preferential food web dynamic in reservoirs derived from pelagic nutrients (Freedman et al., 2014) would explain the outstanding abundance of O. bonariensis which fed on the zooplankton of Paso de las Piedras Reservoir (Fritz, 2018).

Whereas some native species were widely and equally distributed along the different sampled reaches, others where restricted or skewed to a particular habitat (lotic or lentic reaches) or environmental (clear or turbid waters) conditions. O. jenynsii was less responsive to environmental and habitat factors, a pattern also observed for this species in another riverine-lentic ecosystem of the Pampa Plain (Rosso & Quiró, 2009). Riverine habitat fragmentation likely more markedly influenced A. facetus since it was only found in lotic conditions. Nevertheless, some major patterns in other NS respective to habitat were observed. Species preferably collected in riverine natural conditions of the Sauce Grande River, as P. pampa, C. decemmaculatus and the small Siluriformes P. laticeps, are considered intolerant to environmental degradation (Hued & Bistoni, 2005; Bertora et al., 2018b). The number of intolerant species is considered an exclusive metric of the fish assemblages living in healthy riverine conditions (de Freitas Terra & Araújo, 2011). C. decemmaculatus is the most frequent and abundant Cyprinodontiformes in lotic ecosystems of the Pampa Plain being comparatively less common in lentic conditions (Rosso, 2006). Conversely to these species, the one-sided livebearer J. lineata was largely confined to lentic conditions. This species is commonly found as one of the most abundant accompanying species of O. bonariensis in shallow lakes (Rosso, 2006).

Native species responsive to environmental gradient were B. iheringii, C. interruptus, A. facetus, C. paleatus and P. pampa. Turbid reaches, with nutrient-rich waters and higher conductivity were dominated by eurioic species as B. iheringii and C. interruptus. Eutrophication is as a strong driver for fish assemblage structure in freshwater ecosystems (Van de Bund & Van Donk, 2002; Granzotti et al., 2018). Generally, it could be expected that vision oriented fishes prevail in clear oligotrophic waters, as most Characiformes, whereas chemo or tactile-oriented fishes, as most Siluriformes, domain in turbid eutrophic conditions (Pouilly & Rodriguez, 2004; Rosso et al., 2010). Both B. iheringii and C. interruptus are small invertivorous Characiformes usually consumed by larger fish. The food intake of small invertivorous fishes decreased in turbid water due to anti-predator behavior resulting in higher prey survival (Figueiredo et al., 2016). This may partially explain the positive association of small Characiformes and water turbidity in the Sauce Grande Basin. A similar result was observed for Serrapinnus notomelas, other small invertivorous Characiformes inhabiting the upper Paraná River (Piana et al., 2006). Concomitantly, the enhanced primary productivity that can be associated to
turbid, eutrophic reaches of this system, also would favor the establishment of large populations of these small Characiformes. *Corydoras paleatus* presents a wide environmental plasticity which explains its wide distribution in South America (Tencatt et al., 2016). In our study, the abundance of this small armored catfish was higher in turbid conditions and further favored by warmer waters with higher concentrations of dissolved oxygen. A positive relationship between abundance of Siluriformes species and water turbidity was also found in temperate shallow lakes of Argentina (Rosso et al., 2010) and in several freshwater ecosystems of Bolivia, Venezuela and Brazil (Pouilly & Rodríguez, 2004).

4.3. Aspects of management and conservation

Reservoirs affect the natural river conditions, causing deep changes in biological communities, so management actions are usually not simples. In this respect, efficient management and conservation of Neotropical fish in reservoirs are currently constrained by the incipient knowledge about the Neotropical fish fauna, the absence or inadequate monitoring of the results of the implemented actions, insufficient knowledge about the problems to be solved, and the superfluous management of impoundments impacts (Agostinho et al., 2016). In addition, reservoirs are also linked with the establishment of many non-native fish species in Neotropical river ecosystems (Daga et al., 2015). It is therefore necessary to prevent the arrival of potentially non-native species, the timely management of incursions, effective management of those already established and promote a deep social awareness engaging volunteers in surveillance and monitoring (key role of citizen science; Pyšek et al., 2020). The absence of non-native species is the first step towards conservation of freshwater ecosystems as, ideally, freshwater protected areas should contain no non-native species (Saunders et al., 2002). This claims for a thoughtful and more careful management of surface waters in Neotropical ecosystems regarding stocking programs for non-native fish farming, an inconvenient common practice in many tropical reservoirs of South America (Ortega et al., 2015).

5. Conclusions

Our results revealed that a low diversified Neotropical fish fauna is modulated both, by habitat fragmentation imposed by the creation of artificial impoundments and by major environmental variables. Water temperature seemed to be further affected by the presence of the reservoir. In consequence, reservoir affected fish assemblages directly by habitat fragmentation and the enhancement of non-native species and indirectly by its effects on environmental variables as water temperature. In this scenario, native reophilic species better thrive in lotic reaches but once there, environmental conditions were important in regulating the expression of fish assemblage structure. Tolerant species were favored by turbid, nutrient-rich waters with higher conductivity, pH and dissolved oxygen. Altogether, habitat disruption and limnological variables related with eutrophication and water quality (water conductivity, nutrient concentrations, pH, dissolved oxygen and turbidity) as well as those affected by disruption of natural conditions imposed by the reservoir (water temperature) shaped observed differences between fish assemblage structures in the spatially fragmented and environmentally variable Sauce Grande River basin.

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References

AGOSTINHO, A.A., GOMES, L.C. and PELICICE, F.M. *Ecologia e manejo de recursos pesqueiros em reservatórios do Brasil*. Maringá: Eduem, 2007.

AGOSTINHO, A.A., GOMES, L.C., SANTOS, N.C., ORTEGA, J.C. and PELICICE, F.M. Fish assemblages in Neotropical reservoirs: colonization patterns, impacts and management. *Fisheries Research*, 2016, 173, 26-36. http://dx.doi.org/10.1016/j.fishres.2015.04.006.

ALLAN, J.D. Landscapes and riverscapes: the influence of land use on stream ecosystems. *Annual Review of Ecology Evolution and Systematics*, 2004, 35(1),
257-284. http://dx.doi.org/10.1146/annurev. ecolsys.35.120202.110122.

AMERICAN PUBLIC HEALTH – APHA. American Water Works Association – AWWA. Water Environment Federation – WEF. Standard methods for examination of water and wastewater. Washington: APHA, 2012.

AMUCHÁSTEGUI, G., DI FRANCO, L. and FEIJÓO, C.S. Catchment morphometric characteristics, land use and water chemistry in Pampean streams: a regional approach. *Hydrobiologia*, 2016, 767(1), 65-79. http://dx.doi.org/10.1007/s10750-015-2478-8.

ANDERSON, M.J., GORLEY, R.N. and CLARKE, K.R. PERMANOVA+ for PRIMER: guide to software and statistical methods. Plymouth: PRIMER-E, 2008.

ANSARI, A.A., GILL, S.S. and KHAN, F.A. Eutrophication: threat to aquatic ecosystems. In: ANSARI, A.A., SARVAJEET, S.G., LANZA, G.R. and RAST, W., eds. Eutrophication: causes. consequences and control. Berlin: Springer, 2010, pp. 143-170. http://dx.doi.org/10.1007/978-90-481-9625-8_7

AUTORIDAD DEL AGUA – ADA. Resolución N°395. La Plata: Gobierno de la Provincia de Buenos Aires, 2017.

AUTORIDAD DEL AGUA – ADA. Datos físico-químicos, nutrientes y monitoreo de fitoplancton [online]. La Plata, 2018 [viewed 10 Mar. 2018]. Available from: http://www.adagba.gov.ar/institucional/monitoreo.php

BAIGÚN, C.R.M. and QUIRÓS, R. Introducción de peces exóticos en la República Argentina. Mar del Plata: Instituto Nacional de Investigación y Desarrollo Pesquero, 1985.

BALDI, G., GUERSCHMAN, J.P. and PARUELO, J.M. Characterizing fragmentation in temperate South America grasslands. *Agriculture, Ecosystems & Environment*, 2006, 116(3-4), 197-208. http://dx.doi.org/10.1016/j.agee.2006.02.009.

BARTER, D., ALLAN, G.L., ROWLAND, S.J. and PICKLES, J.M. A guide to acceptable procedures and practices for aquaculture and fisheries research. New South Wales: NSW Fisher, 2002.

BASELGA, A. Partitioning the turnover and nestedness components of beta diversity. *Global Ecology and Biogeography*, 2010, 19(1), 134-143. http://dx.doi. org/10.1111/j.1466-8238.2009.00490.x

BAXTER, R.M. Environmental effects of dams and impoundments. *Annual Review of Ecology and Systematics*, 1977, 8(1), 255-283. http://dx.doi.org/10.1146/annurev.es.08.110177.001351.

BERTORA, A., GROSMAN, F., SANZANO, P. and ROSSO, J.J. Fish fauna from the Languéyú basin, Argentina: a prairie stream in a heavily modified landscape. *Check List*, 2018a, 14(2), 461-470. http://dx.doi.org/10.15560/14.2.461.

BERTORA, A., GROSMAN, F., SANZANO, P. and ROSSO, J.J. Composición y estructura de los ensambles de peces en un arroyo pampeano con uso del suelo contrastante. *Revista del Museo Argentino de Ciencias Naturales Nueva Serie*, 2018b, 20(1), 11-22. http://dx.doi.org/10.22179/REVMACN.20.545.

BEUTEL, M.W. and HORNE, A.J. A review of the effects of hypolimnetic oxygenation on lake and reservoir water quality. *Lake and Reservoir Management*, 1999, 15(4), 285-297. http://dx.doi.org/10.1080/074349909354124.

BURNHAM, K.P. and ANDERSON, D.R. Model selection and multimodel inference. New York: Springer, 2002.

BURTON, R.W. Studies in the ecology of rivers: VII. The algae of organically enriched waters. *Journal of Ecology*, 1947, 35(1/2), 186-191. http://dx.doi. org/10.2307/2256507.

CARVALHO, J.C., CARDOSO, P. and GOMES, P. Determining the relative roles of species replacement and species richness differences in generating beta-diversity patterns. *Global Ecology and Biogeography*, 2012, 21(7), 760-771. http://dx.doi.org/10.1111/ j.1466-8238.2011.00694.x

CASSIOOTTA, J., ALMIRÓN, A., CIONE, A. and AZPELICUETA, M. Brazilian freshwater fish assemblages from southern Pampean area, Argentina. *Biogeographica*, 1999, 75, 67-78.

CHAMBERS, P.A., DEWREDEE, R.E., IRLANDI, E.A. and VANDERMEULEN, H. Management issues in aquatic macrophyte ecology: a Canadian perspective. *Canadian Journal of Botany*, 1999, 77(4), 471-487. http://dx.doi.org/10.1139/b99-092.

CLARKE, K.R. and GORLEY, R.N. Getting Started with PRIMER v7. Plymouth: Plymouth Marine Laboratory, 2015.

CLARKE, K.R. and GREEN, R.H. Statistical design and analysis for a ‘biological effects’ study. *Marine Ecology Progress Series*, 1988, 46, 213-226. http://dx.doi.org/10.3354/meps046213.

CLARKE, K.R. Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology*, 1993, 18(1), 117-143. http://dx.doi.org/10.1111/j.1442-9993.1993.tb00438.x

CONY, N.L. Aspectos biológicos, ecológicos y ambientales de la laguna pampeana Sauce Grande y la cuenca media del río de influencia. Bahía Blanca: Universidad Nacional del Sur, 2018.

CORRÊA, F. and PIEDRAS, S.R.N. Alimentação de *Cyphocharax vox* (Hensel. 1869) (Characiformes. Curimatidae) no arroio Correntes, Pelotas, Rio Grande do Sul, Brasil. *Biotemas*, 2013, 21(4), 117-122. http://dx.doi.org/10.5007/2175-7925.2008v21n4p117.

CUCHEROUSSET, J. and OLDEEN, J.D. Ecological impacts of nonnative freshwater fishes. *Fisheries,*
Bertora, A. et al.

2011, 36(5), 215-230. http://dx.doi.org/10.1080/03632415.2011.574578.

DAGA, V.S., SKÓRA, F., PEDIAL, A.A., ABILHOA, V., GUBIANI, É.A. and VITULE, J.R.S. Homogenization dynamics of the fish assemblages in Neotropical reservoirs: comparing the roles of introduced species and their vectors. *Hydrobiologia*, 2015, 746(1), 327-347. http://dx.doi.org/10.1007/s10750-014-2032-0.

ESTRADA, V., DI MAGGIO, J. and DIAZ, M.S. Water sustainability: a systems engineering approach to restoration of eutrophic Lakes. *Computers & Chemical Engineering*, 2011, 35(8), 1598-1613. http://dx.doi.org/10.1016/j.compchemeng.2011.03.003.

FEIJÓO, C.S., GIORGI, A., GARCIÁ, M.E. and MOMET, F. Temporal and spatial variability in streams of a pampean basin. *Hydrobiologia*, 1999, 394, 41-52. http://dx.doi.org/10.1023/A:1003583418401.

FERNÁNDEZ, C., PARODI, E.R. and CÁCERES, E.J. Limnological characteristics and trophic state of Paso de las Piedras Reservoir: an inland reservoir in Argentina. *Lakes and Reservoirs: Research and Management*, 2009, 14(1), 85-101. http://dx.doi.org/10.1111/j.1440-1770.2009.00393.x.

FIGUEIREDO, B.R., MORMUL, R.P., CHAPMAN, B.B., LOLIS, L.A., FIORI, L.E. and BENEDITO, E. Turbidity amplifies the non-lethal effects of predation and affects the foraging success of characid fish shoals. *Freshwater Biology*, 2016, 61(3), 293-300. http://dx.doi.org/10.1111/fwb.12703.

FORNERÓN, C.F., PICCOLO, M.C. and CARBONE, M.E. Análisis morfométrico de la laguna Sauce Grande (Argentina). *Huellas*, 2010, 14, 11-30.

FREEDMAN, J.A., LORSON, B.D., TAYLOR, R.B., CARLINE, R.F. and STAUUFFER JUNIOR, J.R. River of the dammed: longitudinal changes in fish assemblages in response to dams. *Hydrobiologia*, 2014, 727(1), 19-33. http://dx.doi.org/10.1007/s10750-013-1780-6.

FREITAS TERRA, B. and ARAÚJO, F.G. A preliminary fish assemblage index for a transitional river-reservoir system in southeastern Brazil. *Ecological Indicators*, 2011, 11(3), 874-881. http://dx.doi.org/10.1016/j.ecolind.2010.11.006.

Fritz, L.J. *Rol del mezo zooplancton en la trama trífica pelágica del Embalse Paso de las Piedras: integración en un modelo de restauración de cuerpos de agua eutroficos*. Bahía Blanca: Universidad Nacional del Sur, 2018.

GODINHO, A.L., FONSECA, M.T. and ARAÚJO, M.L. The ecology of predator fish introductions: the case of Rio Doce valley lakes. In: R.M. PINTO-COELHO, A. GIANI and E. VON SPERLING, eds. *Ecology and human impact on lakes and reservoirs in Minas Gerais with special reference to future development and management strategies*. Belo Horizonte: SEGRAC, 1994, pp. 77-83.

GOMES, L.C. and MIRANDA, L.E. Riverine characteristics dictate composition of fish assemblages and limit fisheries in reservoirs of the Upper Parana River Basin. *Regulated Rivers: Research and Management*, 2001, 17(1), 67-76. http://dx.doi.org/10.1002/1099-1646(200101/02)17:1<67::AID-RRR615>3.0.CO;2-P.

GRANZOTTI, R.V., MIRANDA, L.E., AGOSTINHO, A.A. and GOMES, L.C. Downstream impacts of dams: shifts in benthic invertevous fish assemblages. *Aquatic Sciences*, 2018, 80(3), 28. http://dx.doi.org/10.1007/s00027-018-0579-y.

GROSMA, F., SANZANO, P., BERTORA, A., COLASURSO, V., FRITZ, L., ESTRADA, V. and DI MAGGIO, G. Ictiología del Dique Paso de las Piedras, Provincia de Buenos Aires. *Biología Acuática*, 2017, 32(Suppl.), 75.

GUBIANI, É.A., RUARO, R., RIBEIRO, V.R., EICHBERGER, A.C.A., BOGONI, R.F., LIRA, A.D., CAVALLI, D., PIANA, P.A. and DA GRAÇA, W.J. Non-native fish species in Neotropical freshwaters: how did they arrive, and where did they come from? *Hydrobiologia*, 2018, 817(1), 57-69. http://dx.doi.org/10.1007/s10750-018-3617-9.

GUENTHER, C.B. and SPACIE, A. Changes in Fish Assemblage Structure Upstream of Impoundments within the Upper Wabash River Basin. *Indiana*. *Transactions of the American Fisheries Society*, 2006, 135(3), 570-583. http://dx.doi.org/10.1577/T05-0311.1.

HAN, M., FUKUSHIMA, M., KAMEYAMA, S., FUKUSHIMA, T. and MATSUSHITA, B. How do dams affect freshwater fish distributions in Japan? Statistical analysis of native and nonnative species with various life histories. *Ecological Research*, 2008, 23(4), 735-743. http://dx.doi.org/10.1007/s11284-007-0432-6.

HUED, A.C. and BISTONI, M.A. Development and validation of a Biotic Index for evaluation of environmental quality in the central region of Argentina. *Hydrobiologia*, 2005, 543(1), 279-298. http://dx.doi.org/10.1007/s10750-004-7893-1.

INTARTAGLIA, C. and SALA, S.E. Variación estacional del fitoplancton en un lago no estratificado: Embalse Paso de las Piedras, Argentina. *Revista Brasileira de Biologia*, 1989, 49, 873-882.

JARIE, H.P., NEAL, C., WILLIAMS, R.J., NEAL, M., WICKHAM, H.D., HILL, L.K., WADE, A.J., WARWICK, A. and WHITE, J. Phosphorus sources, speciation and dynamics in the lowland eutrophic River Kennet, UK. *The Science of the Total Environment*, 2002, 282-283, 175-203. http://dx.doi.org/10.1016/S0048-9697(01)00951-2. PMid:11846070.

JELLYMAN, P.G. and HARDING, J.S. The role of dams in altering freshwater fish communities in New Zealand. *New Zealand Journal of Marine and
Longitudinal patterns in distribution...

Freshwater Research, 2012, 46(4), 475-489. http://dx.doi.org/10.1080/00288330.2012.708664.

JESCHKE, J.M. and STRAYER, D.L. Invasion success of vertebrates in Europe and North America. Proceedings of the National Academy of Sciences of the United States of America, 2005, 102(20), 7198-7202. http://dx.doi.org/10.1073/pnas.0501271102. PMid:15849267.

JOHNSON, P.T.J., OLDEN, J.D. and VANDER ZAPP, M.J. Dam invaders: impoundments facilitate biological invasions into freshwaters. Frontiers in Ecology and the Environment, 2008, 6(7), 357-363. http://dx.doi.org/10.1890/070156.

JUNK, W.J., BAYLEY, P.B. and SPARKS, R.E. The flood pulse concept in river-floodplain systems. Canadian Journal of Fisheries and Aquatic Sciences, 1989, 106(1), 110-127.

LANSAC-TÔHA, E.M., HEINO, J., QUIRINO, B.A., MORESCO, G.A., PELÁEZ, O., MEIRA, B.R., RODRIGUES, L.C., JATI, S., LANSAC-TÔHA, E.A. and VELO, I.EM. Differently dispersing organism groups show contrasting beta diversity patterns in a dammed subtropical river basin. The Science of the Total Environment, 2019, 691, 1271-1281. http://dx.doi.org/10.1016/j.scitotenv.2019.07.236. PMid:31466207.

LEGENDRE, P. Interpreting the replacement and richness difference components of beta diversity. Global Ecology and Biogeography, 2014, 23(11), 1324-1334. http://dx.doi.org/10.1111/geb.12207.

LIEW, J.H., TAN, H.H. and YEO, D.C.J. Dammed rivers: impoundments facilitate fish invasions. Freshwater Biology, 2016, 61(9), 1421. http://dx.doi. org/10.1111/fwb.12781.

LOWE-MCCONNELL, R.H. Fish communities in tropical freshwaters: their distribution, ecology and evolution. London: Longman, 1975.

MACCHI, P.J., PASCUAL, M.A. and VIGLIANO, P.H. Differential piscivory of the native Percichthys trucha and exotic salmonids upon the native forage fish Galaxias maculatus in Patagonian Andean lakes. Limnologica, 2007, 37(1), 76-87. http://dx.doi.org/10.1016/j.limno.2006.09.004.

MAIZTEGUI, T., BAIGÚN, C.R.M., GARCÍA DE SOUZA, J.R., MINOTTI, P. and COLAUTTI, D.C. Invasion status of the common carp Cyprinus carpio in land waters of Argentina. Journal of Fish Biology, 2016, 89(1), 417-430. http://dx.doi.org/10.1111/jfb.13014. PMid:27241358.

MANCINI, M., GROSMAN, F., DYER, B., GARCÍA, G., DELPONTI, O., SANZANO, P. and SALINAS, V. Pejerreyes del sur de América. Río Cuarto: UniRío, 2016.

MCCARTNEY, M. Living with dams: managing the environmental impacts. Water Policy, 2009, 11(S1), 121-139. http://dx.doi.org/10.2166/wp.2009.108.

ORMEROD, S.J., DOBSON, M., HILDREW, A.G. and TOWNSEND, C. Multiple stressors in freshwater ecosystems. Freshwater Biology, 2010, 55, 1-4. http://dx.doi.org/10.1111/j.1365-2427.2009.02395.x.

ORTEGA, J.C., JÚLIO JÚNIOR, H.E., GOMES, L.C. and AGOSTINHO, A.A. Fish farming as the main driver of fish introductions in Neotropical reservoirs. Hydrobiologia, 2015, 746(1), 147-158. http://dx.doi.org/10.1007/s10750-014-2025-z.

PELÁEZ, O.E., AZEVEDO, F.M. and PAVANELLI, C.S. Environmental heterogeneity explains species turnover but not nestedness in fish assemblages of a Neotropical basin. Acta Limnologica Brasiliensis, 2017, 29(0), e117. http://dx.doi.org/10.1590/ s179-975x8616.

PENCZAK, T. and KRUK, A. Threatened obligatory riverine fishes in human modified Polish rivers. Ecology Freshwater Fish, 2000, 9(1-2), 109-117. http://dx.doi.org/10.1034/j.1600-0633.2000.90113.x.

PERÓNICO, P.B., AGOSTINHO, C.S., FERNANDES, R. and PELICICE, F.M. Community reassembly after river regulation: rapid loss of fish diversity and the emergence of a new state. Hydrobiologia, 2020, 847(2), 519-533. http://dx.doi.org/10.1007/s10750-019-04117-9.

PETRETE, J.R.M. Relationships among catches, fishing effort and river morphology for eight rivers in Amazonas State (Brazil), during 1976-1978. Amazoniana: Limnologia et Oecologia Regionalis Systematis Fluminis Amazonas, 1983, 8(2), 281-296.

PIANA, P.A., GOMES, L.C. and CORTEZ, E.M. Factors influencing Serrapinnus notomelas (Characiformes: Characidae) populations in upper Paraná river floodplain lagoons. Neotropical Ichthyology, 2006, 4(1), 81-86. http://dx.doi.org/10.1590/S1679-62252006000100008.

POUILLY, M. and RODRÍGUEZ, M.A. Determinism of fish assemblage structure in Neotropical floodplain lakes: influence of whole-lake and supra-lake conditions. In: Proceedings of the 2nd International Symposium on the Management of Large Rivers for Fisheries. Rome: FAO, 2004, pp. 243-265.

PYŠEK, P., HULME, P.E., SIMBERLOFF, D., BACHER, S., BLACKBURN, T.M., CARLTON, J.T., DAWSON, W., ESSL, F., FOXCROFT, L.C., GENOVA, P., JESCHKE, J.M., KÜHN, L., LIEBOLD, A.M., MANDRAK, N.E., MEYERSON, L.A., PAUCHARD, A., PERGL, J., ROY, H.E., SEEBEENS, H., VAN KLEUNEN, M., VILÁ, M., WINGFIELD, M.J. and RICHARDSON, D.M. Scientists’ warning on invasive alien species. Biological Reviews of the Cambridge Philosophical Society, 2020, 95(6), 1511-1534. http://dx.doi.org/10.1111/brv.12627. PMid:32588508.
RINGUELET, R.A. Rasgos Fundamentales de la Zoogeografía Argentina. *Physis*, 1961, 22(63), 151-170.

RINGUELET, R.A., ARÁMBURU, R.H. and ALONSO DE ARÁMBURU, A. Los peces argentinos de agua dulce. La Plata: Comisión de Investigaciones Científicas, 1967.

ROSENBERG, D.M., BERKES, F., BODALY, R.A., HECKY, R.E., KELLY, C.A. and RUDD, J.W. Large-scale impacts of hydroelectric development. *Environmental Reviews*, 1997, 5(1), 27-54. http://dx.doi.org/10.1139/a97-001.

ROSSO, J.J. and FERNÁNDEZ CIRELLI, A. Effects of land use on environmental conditions and macrophytes in prairie lotic ecosystems. *Limnologica*, 2013, 43(1), 18-26. http://dx.doi.org/10.1016/j.limno.2012.06.001.

ROSSO, J.J. and QUIRÓS, R. Interactive effects of abiotic, hydrological and anthropogenic factors on fish abundance and distribution in natural run-of-the-river shallow lakes. *River Research and Applications*, 2009, 25(6), 713-733. http://dx.doi.org/10.1002/rra.1185.

ROSSO, J.J. *Peces pampeanos: guía y ecología*. Ciudad Autónoma de Buenos Aires: Literature of Latin America, 2006.

ROSSO, J.J., SOSNOVSKY, A., RENNELLA, A. and QUIRÓS, R. Relationships between fish species abundances and water transparency in hypertrophic turbid waters of temperate shallow lakes. *International Review of Hydrobiology*, 2010, 95(2), 142-155. http://dx.doi.org/10.1002/iroh.200911187.

SAUNDERS, D.A., HOBBS, R.J. and MARGULES, C.R. Biological consequences of ecosystem fragmentation: a review. *Conservation Biology*, 1991, 5(1), 18-32. http://dx.doi.org/10.1111/j.1523-1739.1991.tb00384.x.

SAUNDERS, D.L., MEEUWIG, J.J. and VINCENT, A.C.J. Freshwater protected areas: strategies for conservation. *Conservation Biology*, 2002, 16(1), 30-41. http://dx.doi.org/10.1046/j.1523-1739.2002.99562.x.

SMITH, V.H. and SCHINDLER, D.W. Eutrophication science: where do we go from here? *Trends in Ecology & Evolution*, 2009, 24(4), 201-207. http://dx.doi.org/10.1016/j.tree.2008.11.009. PMid:19246117.

SMITH, W.S., STEFANI, M.S., ESPÍNDOLA, E.L.G. and ROCHA, O. Changes in fish species composition in the middle and lower Tietê River (São Paulo, Brazil) throughout the centuries, emphasizing rheophilic and introduced species. *Acta Limnologica Brasiliensis*, 2018, 30(0), e310. http://dx.doi.org/10.1590/s2179-975x0118.

SYMONDS, M.R.E. and MOUSSALLI, A. A brief guide to model selection, multimodel inference and model averaging in behavioural ecology using Akaike’s information criterion. *Behavioral Ecology and Sociobiology*, 2011, 65(1), 13-21. http://dx.doi.org/10.1007/s00265-010-1037-6.

TENCATT, L.M.C., BRITTO, M.R. and PAVANELLI, C.S. Revisionary study of the armored catfish *Corydoras paleatus* (Jenyns. 1842) (Siluriformes: Callichthyidae) over 180 years after its discovery by Darwin, with description of a new species. *Neotropical Ichthyology*, 2016, 14(1), 75-94. http://dx.doi.org/10.1590/1982-0224-20150089.

TURGEON, K., TURPIN, C. and GREGORY-EAVES, I. Dams have varying impacts on fish communities across latitudes: A quantitative synthesis. *Ecology Letters*, 2019, 22(9), 1501-1516. http://dx.doi.org/10.1111/ele.13283. PMid:31112010.

VAN DE BUND, W.J. and VAN DONK, E. Short-term and long-term effects of zooplanktivorous fish removal in a shallow lake: a synthesis of 15 years of data from Lake Zwemlust. *Freshwater Biology*, 2002, 47(12), 2380-2387. http://dx.doi.org/10.1046/j.1565-2427.2002.01006.x.

VANNI, M., BOWLING, A., DICKMAN, E., HALE, R., HIGGINS, K., HORGAN, M., KNOLL, L., RENWICK, W. and STEIN, R. Nutrient cycling by fish supports relatively more primary production as lake productivity increases. *Ecology*, 2006, 87(7), 1696-1709. http://dx.doi.org/10.1890/01-9658(2006)87[1696:NCFBFSR]2.0.CO;2. PMid:16922320.

VANNOTE, R.L., MINSHALL, G.W., CUMMINS, K.W., SEDELL, J.R. and CUSHING, C.E. The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences*, 1980, 37(1), 130-137. http://dx.doi.org/10.1139/f80-017.

VIGLIZZO, E.F., LÉRTORA, F.A., PORDOMINGO, A.J., BERNADOS, J.N., ROBERTO, Z.E. and DEL VALLE, H. Ecological lessons and applications from one century of low external-input farming in the pampas of Argentina. *Agriculture, Ecosystems & Environment*, 2001, 83(1-2), 65-81. http://dx.doi.org/10.1016/S0167-8809(00)00155-9.

WARD, J.W. and STANFORD, J.A. Intermediate-disturbance hypothesis: an explanation for biotic diversity patterns in lotic ecosystems. In: T.D. FONTAINE and S.M. BARTELL, eds. *Dynamics of lotic systems*. Ann Arbor: Ann Arbor Science, 1983, pp. 347-356.

ZAMBRANO, L., PERROW, M.R., MACÍAS-GARC, C. and AGUIRRE-HIDALGO, V. Impact of introduced carp (*Cyprinus carpio*) in subtropical shallow ponds in Central Mexico. *Journal of Aquatic Ecosystem Stress and Recovery*, 1998, 6(4), 281-288. http://dx.doi.org/10.1023/A:1009958914016.

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