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RESEARCH ARTICLE

Assessment of the environmental quality of two urbanized lotic systems using multiple indicators

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Abstract

Urbanization impacts ecosystems through loss and fragmentation of habitat, loss of diversity, increase in runoff, and contaminant discharges, and the invasion of exotic species. Potrero de los Funes Village (San Luis, Argentina) is experiencing not only a population increase during summer months due to tourism but also an accelerated growth of its permanent population. In order to evaluate the potential effects of urbanization, the environmental quality of Potrero de los Funes River and Las Chacras Stream was assessed, using physical–chemical and biological indicators. Macroinvertebrates, through the application of the Biotic Index for San Luis Sierras (BISLS), anuran amphibian richness and relative abundance, and riparian vegetation were used as bioindicators. While the Simplified Index of Water Quality (SIWQ) was used to characterize the physical–chemical quality of water at each site. SIWQ and BISLS scores were significantly different between sites ($F_{7,28} = 9.88$, $p < .001$ and $F_{7,28} = 24.18$, $p < .001$, respectively). SIWQ was significantly correlated with BISLS (Spearman correlation coefficient = 0.8, $p < .001$). Four anuran species were registered along Potrero River, with no significant differences in the intensity of vocalizations between sites. No species were detected in Las Chacras Stream. Average total plant species richness, exotic plants richness, and vegetation cover were higher at the most impaired system, Las Chacras Stream. The principal component analysis showed that the first two principal components (PCs) explained 76.3% of the total variance. PC1, with strong loadings of SIWQ, BISLS, and amphibian richness, was principally driven by chemical water quality and biological conditions. PC2 was mainly determined by plant richness. The chemical and biological water quality of Potrero and Las Chacras is somewhat impaired, being the upper Las Chacras Stream the most compromised area. This study provides information that will certainly be used to manage future impacts of urbanization on aquatic resources.

KEYWORDS

Argentina, bioindicators, environmental quality, lotic systems, urbanization

1 | INTRODUCTION

The increase in human population has resulted in the development of new urban areas. This landscape transformation is a complex process that involves the creation of new land cover and new biotic assemblages driven by changes in the chemical, physical, and ecological conditions (Hamer & McDonnell, 2008). Urbanization may cause loss and fragmentation of habitats, loss of species diversity, increased runoff, greater discharges of point and nonpoint source pollution, and the promotion of exotic plants and animal species (Faulkner, 2004; Longing & Haggard, 2010; McKinney, 2002, 2006, 2008; Ridley et al., 2005; Sutula & Stein, 2003). The impacts of urbanization are currently one of the most prevalent causes of natural ecosystem alteration.

Chemical water quality is a key element in the assessment of environmental quality of aquatic ecosystems. However, the use of biological indicators represents an important integrating tool (Zampella, Bunnell, Laidig, & Procopio, 2006). Because chemical monitoring cannot provide a holistic assessment of the health of an aquatic system, biological monitoring has been recognized as one of the most useful tools in assessing water quality. It provides an integrated framework for addressing cumulative and synergistic environmental impacts (Adams & Greeley, 2000; Bode & Novak, 1995; Metcalfe-Smith, 1994; Muenz et al., 2006; Golladay, Vellidis, & Smith, 2006; Oscoz et al., 2007).

A bioindicator is a species or group of species that reflect the abiotic or biotic state of an environment and represents the impact of environmental changes on a habitat, community, or ecosystem (Hodkinson & Jackson, 2005). A bioindicator must be sensitive enough to exhibit changes in response to a stressor (Burger, 2006; Carignan & Villard, 2002). Valuable bioindicators provide early warning to environmental impacts, allow continuous assessment over a wide range and intensity of stresses, are affordable to measure and can be accurately estimated (Burger, 2006; Carignan & Villard, 2002).

Macroinvertebrates are among the most accurate bioindicators to assess and monitor health in aquatic environments because they are less mobile than fish and therefore unable to avoid discharges. Macroinvertebrates are directly and constantly affected by the physical and chemical conditions of the stream; they possess a wide range of sensitivity to pollutants and respond to both short-term episodic events and longer-term cumulative effects (Longing & Haggard, 2010; Stribling, Jessup, White, Boward, & Hurd, 1998). Urbanization of watersheds has been associated with higher and more frequent peak flows and elevated concentrations of pollutants and sediments that may affect the species richness and abundance of sensitive macroinvertebrates (Davidson-Bennett, 2011). Some quality indicators based on macroinvetebrates metrics have been developed in order to reduce taxonomic resolution, produce a rapid turn-around of results, and assess a wide range of water qualities (Canter & Atkinson, 2011; Ector & Rimet, 2005; Longing & Haggard, 2010; Metcalfe-Smith, 1994; Muenz et al., 2006; Resh & Unzicker, 1975; Wallace, Grubaugh, & Whiles, 1996).

Habitat loss, fragmentation, and degradation are among the greatest threats to amphibian populations (Hamer & McDonnell, 2008). Amphibians may be good indicators of environmental conditions (Houlahan, Findlay, Schmidt, Meyer, & Kuzmin, 2000), and their decline could be used as an indicator of environmental degradation (Muenz et al., 2006; Sewell & Griffiths, 2009). Vegetation is also often an excellent indicator of the physical and chemical condition of wetlands (Lopez & Fennessy, 2002; Mensing, Galatowitsch, & Tester, 1998). The composition of riparian vegetative zones is a key element to assess the environmental quality of a river, because they are directly related to the diversification of the landscape, the regulation of water temperature, inputs of organic material, and nutrients, and the creation of habitat used by different organisms (Capon & Dowe, 2007; Hood & Naiman, 2000).

Potrero de los Funes Village (San Luis, Argentina) constitutes one of the main tourist destinations of the region. Potrero de los Funes River, together with Las Chacras Stream, is used as recreational swimming places and as drinking water resources. The permanent population of the village has experienced accelerated growth. According to the Instituto Nacional de Estadística y Censos (2010), the total population of the town was approximately 410 in 1991, 944 in 2001, and 1698 in 2010. This is equivalent to an average increase of 105% every 10 years.

Previous studies (Almeida, Quintar, Gonzalez, & Mallea, 2007; Oliva González, Almeida, Quintar, Mallea, & Gonzalez, 2011) assessed physical–chemical and bacteriological water quality of upper, middle, and downstream sections of the Potrero River (PR). Upper sites had low interference from anthropogenic activities and represented the "environmental base line" of the watercourse; middle sites were within fully urbanized areas near wastewater drainages from residential buildings and hotels, and downstream sites were located at the end of the urban area, before draining into the Potrero de los Funes reservoir. Water quality of PR was deteriorated downstream from urbanized areas with an increase of organic load and nutrients (phosphorous and nitrogen). This loss of quality had a temporal pattern linked to the increase of tourist activities during the summer (Almeida et al., 2007; Oliva González, Almeida, Calderón, Mallea, & González, 2014; Oliva González et al., 2011).

The goal of this study was to assess the environmental conditions of two urban lotic systems in Potrero de los Funes Village (San Luis, Argentina) using a physical–chemical index of water quality and macroinvertebrates, anuran amphibians, and riparian vegetation as bioindicators.

2 | METHODS

2.1 | Study site

Potrero de los Funes River has a length of 2700 m with an average flow of 7.5 m^3/s and a basin area of 42 km^2. It rises from the confluence of two tributaries, Los Molles Creek and La Bolsa Creek, at 1003 m above sea level in the Central Sierras of San Luis, Argentina. After the confluence, the PR flows through the hilly village of Potrero de los Funes to finally drain into the Potrero de los Funes reservoir. The spillway of this reservoir forms Las Chacras Stream that runs through the Quebrada de los Cóndores area (Figure 1).

Four sampling sites along PR and four sampling sites along Las Chacras Stream (QS) were chosen for the study (Figure 1). Site PR1 was considered as the reference site. Previous studies assessed PR1 as the area of environmental base line of water quality on the basis of physical–chemical and bacteriological analysis (Oliva González et al., 2011). PR1 also represents the initial stretch of PR, where its two tributaries streams converge. Considering the impact of urbanization as percent of urban land use (buildings, roads, and recreational infrastructure) and total plant cover (%) in a radius of 150 m around the sampling point, PR1 is the least impacted site: 10% of urban land use and 90% plant cover. PR2 (60% plant cover—30% urban land use) and PR3 (40% plant cover —50% urban land use) were located downstream from PR1 in fully urbanized areas, near wastewater drainages from residential buildings and hotels. PR4 (10% plant cover—70% urban land use) was located 2,600 m from PR1, at the end of the urban area and 618 m upstream of the river mouth discharge into Potrero de los Funes reservoir.

Sites at Las Chacras Stream were: QS1 (50% plant cover—50% urban land use), located 10 m downstream from the dam spillway; QS2 (60% plant cover—30% urban land use): located 100 m downstream from QS1 in an area that receives the discharge of poorly treated domestic waste from a five star hotel; QS3 (80% plant cover

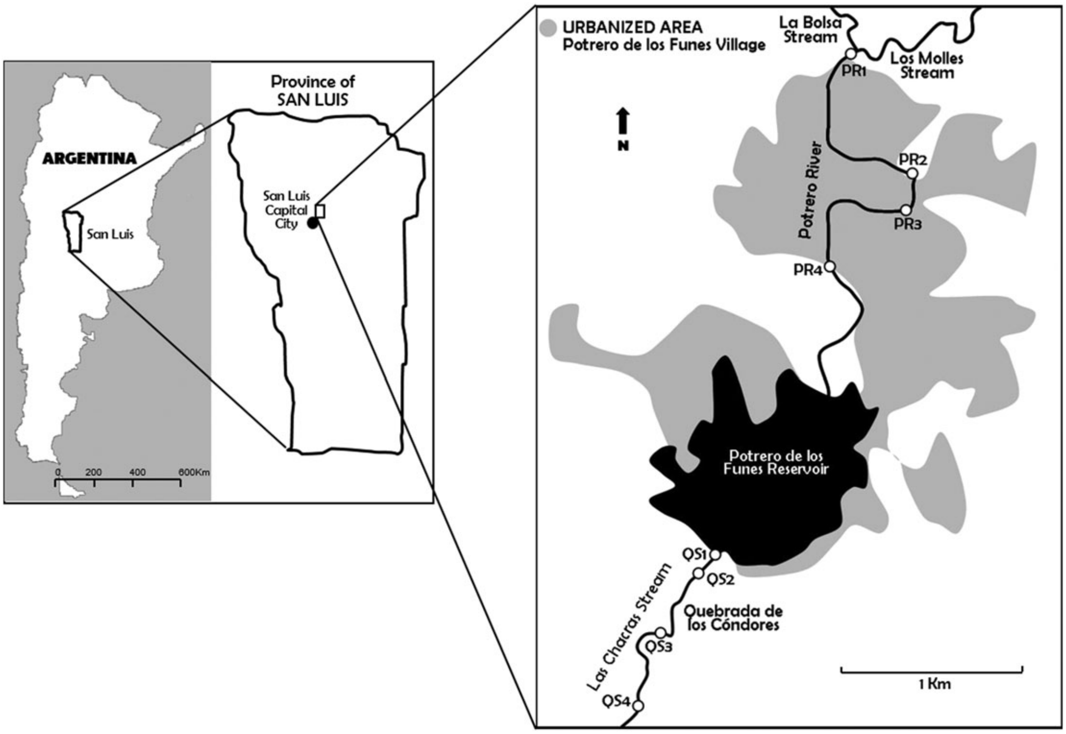

FIGURE 1 Localization of sampling sites (white dots) at Potrero River (PR) and Las Chacras Stream along Quebrada de los Cóndores (QS)

—10% urban land use): located 900 m from QS1, in a zone characterized by a low urban development likewise the QS4 (70% plant cover—20% urban land use) located at 1,200 m from QS1. Sites were also selected on the basis of their similar physical–morphological conditions. Substrate size, flow velocity, depth, and riffle/pool ratios are uniform between sites into PR and Las Chacras Stream.

2.2 | Physical–chemical sampling, analytical methods and water quality

A Simplified Index of Water Quality (SIWQ) was used to estimate the water quality of the sites (Queralt, 1982). SIWQ uses five physical–chemical parameters that includes water temperature (°C) and electrical conductivity (µS/cm), which were tested in the field using a portable meter, total suspended solids (mg/L), dissolved oxygen (mg/L O_2) and chemical oxygen demand (mg L O_2), which were measured in laboratory, following standard protocols (APHA, 2005). Analytical quality of data was ensured through careful standardization, procedural blank measurements, and spiked and duplicate samples. The ionic charge balance of each sample was within $\pm 5\%$.

2.3 | Biological indicators

Benthic macroinvertebrates and water samples for physical–chemical determinations were collected during high-flow (December–March) and low-flow (May–August) seasons between 2009 and 2011. Multihabitat samples were collected at each site using a 0.09-m^2 area and 300-µm meshed Surber net. In both lotic systems, the selected microhabitats were one riffle and one pool at every site. The two samples were combined for comparison between the different sites.

All macroinvertebrate collections were preserved in 70% ethanol and returned to the laboratory for sorting and identification. The specimens were identified in the laboratory, down to the lowest taxonomic level required by the Biotic Index of San Luis Sierras (BISLS), family for some classes (i.e., Coleoptera, Diptera) and genus for others (i.e., *Ephemeroptera*, *Trichoptera*; Vallania, Garelis, Trípole, & Gil, 1996).

Species richness and relative abundance of amphibians were estimated from calling and visual encounter surveys. All sites were visited during dusk, 3 times during the months of maximum activity (October, November, and December) of 2009 and 2010. Anuran calls were recorded for 5 min on each site. Two visual encounter surveys plots (100 m × 5 m) were installed one in each side of the channel. After the calling registration and recording, observers searched plots while walking at a standard pace using as much time as was needed to examine each area thoroughly. In accordance with North American Amphibian Monitoring Program protocol, surveys were conducted at least 0.5 hr after dusk and completed by 01:00 (Weir & Mossman, 2005).

In every site (PR1 to Q4), vegetation was characterized in one plot (100 m × 3 m) in each bank of the channel. The Zürich–Montpellier (Braun-Blanquet, 1928, 1932) method was used to generate phytosociological inventories. Richness of native and exotic vascular plants was recorded, and the tree and herbaceous abundance-cover was estimated. The species that could not be identified in the field were taken to the laboratory for taxonomic identification using dichotomous keys and in accordance with Instituto de Botánica Darwinion (2014).

2.4 | Data analysis

In order to evaluate the physical–chemical water quality, an SIWQ was used (Queralt, 1982). This index uses a simple algorithm that requires evaluation of five chemical parameters: temperature, chemical oxygen demand, total suspended solids, dissolved oxygen, and electrical conductivity (Colman Broggi & Bellagamba, 2006). The selection of the parameters used by the index was made by taking into account their representativeness in the water quality characteristics, and the cost effective analytical methods to estimate them (Queralt, 1982). The final score is calculated from the formula: ISQA = $T. (A + B + C + D)$. The term T depends on the water temperature and it has values between 0.8 and 1. The variable A fluctuates as a function of chemical oxygen demand, with values between 0 and 30, and it is a measure of natural or artificial organic content, either biodegradable or not. B is a function of total suspended solids, it ranges from 0 to 25 and quantifies filterable particles, from organic, inorganic, industrial, and urban sources. The value C is a function of dissolved oxygen; it varies between 0 and 25 and is related to oxidability and to biodegradable organic matter content. D depends on electrical conductivity; it can take values between 0 and 20 and it is related to inorganic salts concentration, mainly chloride and sulfate. The outcome from the equation, an overall water quality dimensionless value, can range from 100 to 0. SIWQ qualifies water for different uses, from the best to the poorest quality: all uses (100–85), swimming (85–75), fishing (75–60), navigation (60–45), crop irrigation (45–30) forestry irrigation (30–15), and no use allowed (15–0).

A macroinvertebrates index, the BISLS, has been generated to evaluate biological water quality of rivers in the central zone of San Luis province (Vallania et al., 1996). BISLS combines taxonomic richness, and the sensitivity of each taxa presents in the sample to estimate the biological water quality of the site. It assumes that the presence of more sensitive taxa (Ephemeroptera, Plecoptera, Trichoptera) indicates better water quality. This index can have values from 1 for extremely contaminated environments to 12 for environments with no contamination.

The call index, proposed by Pillsbury and Miller (2008), was used as an indicator of relative abundance of amphibians, it follows: 0, *no individuals of a given species heard*; 1, *one individual heard*; 2, *multiple individuals with no overlap in calls*; 3, *full chorus*.

One way analysis of variance was used to compare amphibian maximum monthly call index value, BISLS values, number of invertebrate taxa, and SIWQ values between sites. A paired *t* test comparing BISSL, number of taxa, and SIWQ between sampling seasons (high flow and low flow) was performed for each site before running the analyses of variance, to account for within sites individual variability. Independent sample *t* test was used to compare BISLS values, number of macroinvertebrate taxa, SIWQ values and plant variables between PR and Las Chacras Stream. Spearman correlation analysis was performed to analyze the relation of SIWQ with BISLS, the number of macroinvertebrate taxa and amphibian parameters. Principal component (PC) analysis of the normalized variables (SIWQ, BISLS, amphibian richness, amphibian relative abundance (call index), plant richness, ratio native/exotic plant richness, and plant cover) was performed to extract significant PCs and to further reduce the contribution of variables with minor significance (Bagur, Morales, & López-Chicano, 2009; Zhou, Huang, Guo, Zhang, & Hao, 2007).

3 | RESULTS

3.1 | Physical–chemical water quality (SIWQ)

Values of SIWQ varied between 98.56 (at PR1) and 67.35 (at QS1), with a total average of 87.42 ± 9.33. Differences in SIWQ between sampling seasons (low flow and high flow) were not significant for all the sites sampled (independent samples *t* test, all *p*'s > .05). SIWQ values were significantly different between sites ($F_{7,28} = 9.88$, $p < .001$; Figure 2). PR had SIWQ values significantly higher than Las Chacras Stream (independent samples *t* test: $t = 6.51$, $df = 26$, $p < .001$; Table 1). SIWQ values were significantly correlated with BISLS values (Correlation coefficient = .8, $p < .001$; Figure 3) and with the number of macroinvertebrate taxa (Correlation coefficient = .6, $p < .001$).

3.2 | Biological indicators

Forty-two taxa were identified in the benthic macroinvertebrates samples (Table 2). The most common taxa were Chironomidae (Diptera), Naididae (Oligochaeta), Hydroptilidae (Trichoptera), Elmidae (Coleóptera), Simulidae/Ceratopogonidae (Diptera), and Caenidae (Ephemeroptera). BISLS values ranged from 11 (*not contaminated*; at PR2) to 5 (*very contaminated*; at QS2). The number of invertebrate taxa varied from 22 (at RP2) to 6 (at QS2). BISLS and the number of invertebrate taxa were not significantly different between sampling seasons (low flow and high flow), for all the studied sites (independent samples *t* test, all *p*'s > .05). Average values of BISLS ($F_{7,28} = 24.18$, $p < .001$) and number of invertebrate taxa ($F_{7,28} = 5.32$, $p < .001$) for all the sampled dates were significantly different between sites (Figure 4). Values of BISLS and the number of macroinvertebrate taxa of PR were significantly higher than Las Chacras Stream (independent samples *t* test: $t = 5.67$, $df = 26$, $p < .001$ for BISLS and $t = 3.9$, $df = 26$, $p < .001$ for number of taxa).

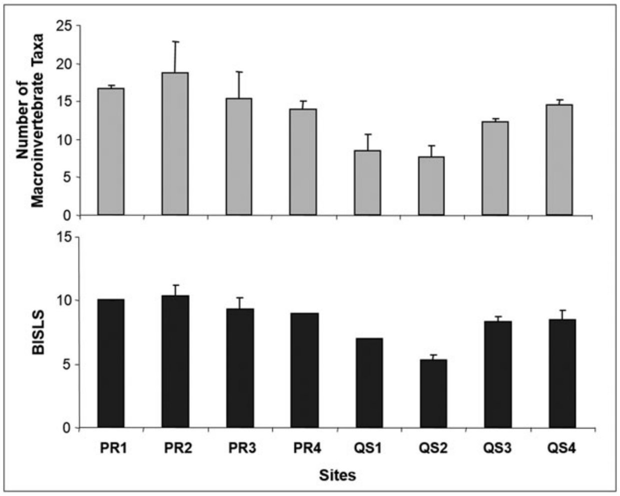

FIGURE 2 Values of the Simplified Index of Water Quality (SIWQ) calculated for water samples from Potrero River and Las Chacras Stream

TABLE 1 Values of physical–chemical variables and SIWQ (average ± standard error) at sites in Potrero River (PR) and Las Chacras Stream along Quebrada de los Cóndores (QS)

Variables	PR1 (*n* = 4)	PR2 (*n* = 4)	PR3 (*n* = 4)	PR4 (*n* = 4)	QS1 (*n* = 4)	QS2 (*n* = 4)	QS3 (*n* = 4)	QS4 (*n* = 4)
Temperature (°C)	13.25 ± 1.96	16.55 ± 0.82	17.05 ± 1.05	18.85 ± 1.42	17.43 ± 0.53	17.90 ± 0.72	19.00 ± 1.22	18.43 ± 1.58
Chemical oxygen demand (mg/L O_2)	1.25 ± 0.59	2.02 ± 1.03	1.67 ± 0.78	1.34 ± 0.62	5.34 ± 4.12	5.05 ± 4.26	1.25 ± 0.44	2.30 ± 0.77
Total suspended solids (mg/L)	1.29 ± 0.34	8.45 ± 0.52	3.79 ± 0.90	3.93 ± 0.57	17.73 ± 0.78	20.03 ± 1.14	6.03 ± 0.61	3.27 ± 0.15
Dissolved oxygen (mg/L O_2)	8.80 ± 0.40	7.73 ± 0.53	8.57 ± 1.71	9.58 ± 1.53	6.32 ± 0.80	4.30 ± 0.98	7.01 ± 0.07	7.25 ± 0.51
Conductivity (µS/cm)	146.50 ± 24.83	208.75 ± 20.17	212.75 ± 34.03	216.50 ± 27.59	371.00 ± 69.85	406.67 ± 27.24	413.00 ± 34.04	400.00 ± 38.93
SIWQ	97.83 ± 0.32	90.83 ± 2.56	92.20 ± 1.40	93.81 ± 1.48	78.45 ± 7. 08	72.58 ± 2.88	79.71 ± 6.96	85.60 ± 0.79

SIWQ = Simplified Index of Water Quality.

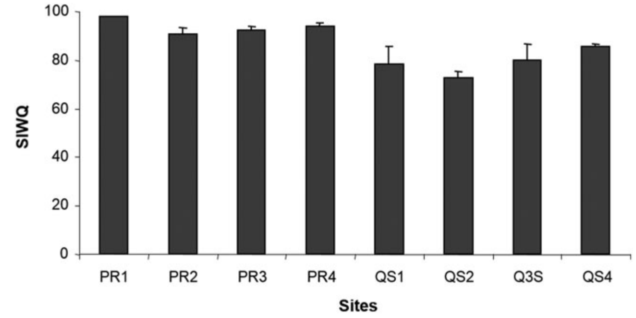

FIGURE 3 Correlation between scores of the Biotic Index of San Luis sierras (BISLS) and the Simplified Index of Water Quality (SIWQ) values calculated for sites at Potrero River and Las Chacras Stream

Four anuran amphibian species were registered during call and visual encounter surveys: *Leptodactylus mystacinus*, *Hypsiboas pulchellus*, *Rhinella arenarum* and *Odontoprhynus americanus*. A maximum of two species per site were detected at PR1, PR2, and PR4, whereas only one species was active at PR3. The higher call index (3) was estimated at PR4. There were no significant differences in the maximum intensity of vocalizations between sites where amphibian were present ($F_{3,16}$ = 0.46 *p* = .71). Activity of anurans was absent along Las Chacras Stream at Quebrada de los Cóndores area. No significant correlation was found between amphibian metrics and SIWQ or BISLS.

The average number of plant species per site was 31.21, with a maximum of 50 at QS3 and a minimum of 18 at PR3. The ratio between native and exotic plant species was 1.69 on average and ranged from 4.8 at PR1 to 0.64 at QS1 and QS2. Plant cover was maximum (100%) at PR2 and minimum at PR4 (25%). Total plant species richness, exotic plant richness, and vegetation cover were on average higher at Las Chacras Stream; the rest of plant variables (native plant richness and relationship native/exotic plants) were higher at PR, but the observed differences were not significant (independent samples *t* test *p*'s > .05).

The PC analysis showed that the first two PCs explained 76.3% of the total variance in PR and Las Chacras Stream environmental quality data according to the eigenvalue-one criterion (variances greater than 1; Table 3). PC1, which explained 56.6% of the variance, had strong loadings (>0.8) of SIWQ, BISLS, and amphibian richness, so it is principally driven by physical–chemical and biological water quality. PC2, explaining 19.7% of the variance, was mainly determined by plant richness (Figure 5).

A summary of the results for PR and Las Chacras Stream is presented in Table 4.

4 | DISCUSSION

The chemical water quality of PR and Las Chacras Stream is somewhat impaired; however, all SIWQ scores were above 60, which is considered acceptable (Queralt, 1982). Water quality along PR vqualified for all possible uses with scores from 87.02 to 98.56, while all sites along Las Chacras Stream showed some degree of impairment

TABLE 2 Macroinvertebrate taxa identified in benthic samples from sites in Potrero River (PR) and Las Chacras Stream (QS)

Phyllum	Class	Order	Family	Genus
Arthropoda	Insecta	Coleoptera	Elmidae	
			Staphylinidae	
			Girinidae	
		Diptera	Blepharoceridae	
			Ceratopogonidae	
			Chironomidae	
			Dolicopodidae	
			Empididae	
			Eprhydidae	
			Muscidae	
			Psicopodidae	
			Psychodidae	
			Simulidae	
			Stratiomidae	
			Syrphydae	
			Tabanidae	
			Tipulidae	
		Ephemeroptera	Caenidae	*Caenis*
			Baetidae	*Baetis*
				Baetodes
				Dactylobaetis
			Tricorythidae	*Leptohyphes*
				Tricorythodes
		Hemiptera	Naurocoridae	
			Gerridae	
		Odonata	Gomphidae	
			Zygoptera	
		Trichoptera	Odontoceridae	
			Hydroptilidae	
			Hydropsychidae	
			Leptoceridae	
			Phylopotamidae	
			Polycentropodidae	
		Lepidóptera		
Annelida	Oligochaeta	Tubificida	Tubificidae	
			Naididae	
	Hirudinea			
Mollusca	Gastropoda	Basommatophora	Phisydae	*Physa*
			Lymnaeidae	*Lymnae*
			Planorbidae	*Bionphalarya*
			Ancylidae	*Uncacylus*
	Bivalvia			
Crustacea		Decápoda		

with some sites presenting water quality not compatible with recreational (swimming) and fishing uses (SIWQ values from 67.35 to 85.01). The most relevant parameters in the calculus of SIWQ were total suspended solids, with the highest values at sites QS1 and QS2 and the lowest at PR1, and conductivity, with a clear pattern of higher values at all QS sites. The amount of suspended solids is related to the increase of impermeable surfaces due to urbanization that contributes to nonpoint source pollution (Carle, Halpin, & Stow, 2005). The construction of streets, buildings, industries, and spaces for parking generates large quantities of sediment input to the streams and may also indicate the presence of other pollutants that are carried on the surfaces of sediment in suspension (Stephens, Reid, & Graham, 2002). The contribution of variations in dissolved oxygen and temperature was minor but significant. Previous studies have shown a deterioration of surface waters downstream from PR2, coincides spatially with the most urbanized part of the town and temporally with the

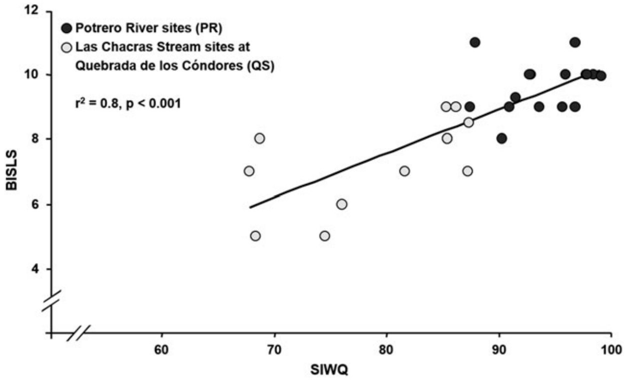

FIGURE 4 Biotic Index of San Luis Sierras (BISLS) and number of macroinvertebrate taxa registered in water samples from Potrero River and Las Chacras Stream

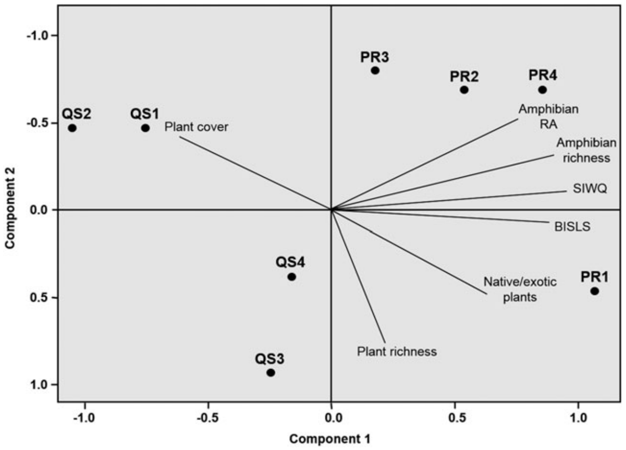

FIGURE 5 Principal components analysis using measured variables across sites at Potrero River (PR) and Las Chacras Stream along Quebrada de los Cóndores (QS)

increase of tourist activities during the summer season (Oliva González et al., 2011). Almeida et al. (2007) applied a General Quality Index calculated from 11 parameters, including bacteriological measurements and likewise reported a decline of water quality in zones of the river located in the urbanized area of Potrero de los Funes town. Our results for Las Chacras Stream match the ones reported by Oliva González et al. (2014) who found the same pattern in water quality.

On the basis of BISLS values, PR may be categorized as not contaminated (sites PR1 and PR2) to slightly contaminated (sites PR3 and PR4), while Las Chacras Stream has sites classified as heavily contaminated (QS2), contaminated (QS1) and slightly contaminated (QS3 and QS4). These results agree with previous studies, where QS1 and QS2 were assessed as points with very poor physical–chemical water quality and a very low concentration of dissolved oxygen, due to the presence of a wastewater discharges (Oliva González et al., 2014). Even though tolerances of macroinvertebrates to hypoxic conditions are taxonomically specific and, in some ecosystems, they are highly tolerant of seasonally low-dissolved oxygen conditions (Kaller & Kelso, 2007); in general, invertebrate communities are largely influenced by temperature and dissolved oxygen levels (Tarr, Baber, & Babbitt, 2005). Taxa such as Ephemeroptera are highly sensitive to low oxygen conditions, while Chironomidae are the least sensitive taxa (Connolly, Crossland, & Pearson, 2004). In our study, the site QS2 showed the lowest concentrations of dissolved oxygen (45%) and the lowest number of macroinvertebrate taxa and BISLS. This was paired with a high abundance of Chironomids and the absence of mayflies (Ephemeropteras). The improvement in physical–chemical quality of water, observed towards points QS3 and QS4 (and also evident from BISLS values), may be explained by the increase in the velocity of the stream, the dissolved oxygen, and the self-purification capacity of the stream (Oliva González et al., 2014). A further improvement of water quality has been observed in sites downstream of Q4, before it is impacted by Juana Koslay City (Calderón, González, Moglia, González, & Jofré, 2014).

TABLE 3 Eigenvalues. Variance explained and variable loadings of principal component analysis (PCA)

	Component 1	Component 2
Eigenvalue	3.959	1.380
% of variance explained	56.557	19.716
% of cumulative variance	56.557	76.273
Variable	Loadings	
SIWQ	0.964	0.102
Amphibian richness	0.908	0.324
BISLS	0.892	−0.054
Amphibian call index	0.761	0.511
Native/exotic plant richness ratio	0.628	−0.481
Total plant cover	−0.620	0.410
Plant richness	0.228	−0.775

BISLS = Biotic Index for San Luis Sierras; SIWQ = Simplified Index of Water Quality.

A significant correlation between SIWQ and BISLS was found. This agrees with other authors that stated the relevance of biological indexes as an alternative and/or integrative tool in the assessment of the water quality of aquatic ecosystems (Canter & Atkinson, 2011; Longing & Haggard, 2010; Muenz et al., 2006; Oscoz et al., 2007).

Anuran amphibian species registered in this study have already been cited for the central area of the San Luis sierras. *R. arenarum* is recognized as being characteristic of urban environments (Cei, 1980; Leynaud, Pelegrin, & Lescano, 2006; Sanabria, Quiroga, & Acosta, 2005). *H. pulchellus* was recorded for the first time in the province in 2011, inhabiting urbanized areas of Chorrillos River in the cities of Juana Koslay and San Luis (Calderon & Jofré, 2011). Previously, Agüero, Moglia, and Jofré (2010) documented *L. mystacinus* and *R. arenarum* in urban permanent and temporary water bodies in urban parks and temporary puddles in San Luis City. Vocalization activity of *L. mystacinus*, *H. pulchellus*, *R. arenarum*, and *O. americanus* was documented in tributaries (Las Chacras Stream and Cuchi Corral Stream) and the main channel of Chorrillos River but no significant correlations between urbanization and amphibian richness and relative abundance were detected (Calderón et al., 2014). A similar pattern was found in our study, with no significant differences found in the intensity of vocalizations between sites. The higher call index was estimated at site PR4, which is highly modified and impacted by urbanization and tourist activities. Although richness and presence/absence of amphibians is in general

TABLE 4 Summary of quality assessment (BISLS and SIWQ) and amphibian and plant metrics for Potrero River (PR) and Las Chacras Stream (QS)

	Water quality		Amphibian metrics		Plant metrics		
SITE	BISLS	ISQA	Richness[a]	Relative abundance[a]	Total richnes	Native/exotic richness	Total cover
RP1	Uncontaminated (10)	Uncontaminated—Suitable for all uses (97.41)	2	1	35	4.80	40
RP2	Uncontaminated (10.3)	Uncontaminated—Suitable for all uses (87.22)	2	2	37	1.05	100
RP3	Slightly contaminated (9.3)	Uncontaminated—Suitable for all uses (91.78)	1	1	18	2.00	90
RP4	Slightly contaminated (9)	Uncontaminated—Suitable for all uses (91.89)	2	3	29	1.07	25
QS1	Contaminated (7)	Moderately contaminated—Unsuitable for swimming and fishing (67.95)	0	0	23	0.64	95
QS2	Very contaminated (5.3)	Moderately contaminated—Unsuitable for swimming and fishing (67.3)	0	0	23	0.64	95
QS3	Slightly contaminated (8.3)	Uncontaminated—Suitable for all uses (84.97)	0	0	50	1.63	60
QS4	Slightly contaminated (8.5)	Uncontaminated—Suitable for all uses (86.87)	0	0	35	1.69	60

BISLS = Biotic Index for San Luis Sierras; SIWQ = Simplified Index of Water Quality.

[a] Maximum value for the site.

negatively related to urbanization, they have broad habitat requirements that may allow adaptation to urban environments, especially when the alteration of the landscape involves establishment of suitable habitats (Hamer & McDonnell, 2008).

Anurans were detected along PR but were absent along Las Chacras Stream at Quebrada de los Cóndores, where water quality was significantly lower than in the river. However, no correlation was found between water quality and amphibian metrics, which implies that many other variables may influence amphibian community composition, distribution, abundance, and vocalization activity such as hydrologic period, presence of predators, and nearby noise and plant related variables (Burne & Griffin, 2005; Reading & Jofré, 2003; Sun & Narins, 2005; Urbina-Cardona, Olivares-Perez, & Reynoso, 2006).

A reduction in the number of native plant species was observed from the less impacted sites to the most altered sites within PR (PR1-PR2 to PR3-PR4) and Las Chacras Stream, reaching a minimum of nine native species at sites QS1 and QS2, and recovering to higher values at sites QS3 and QS4. The number of native species was higher than the number of exotic species in all sites except QS1 and QS2. The maximum difference between native and exotic plant species was at the less altered site PR1, where the number of native plants was 5 times higher than the number of exotic plants. Urban and suburban areas are significant spots for the dispersion of introduced plant species, and in riparian forests, particularly in hot and semiarid regions, urbanization stimulates the progress of the invasion by causing hydrologic drought (Duguay, Eigenbrod, & Fahrig, 2007; Sung, Li, Rogers, Volder, & Wang, 2011).

The proportion of exotic species for temperate systems worldwide is around 10% (Macdonald & Fame, 1988). The values found in PR riparian zones (17–61%, average 43%) were higher than other wetlands areas of Argentina (Kalesnik, Cagnoni, Bertolini, Quintana, & Madanes, 2005; Kalesnik & Malvárez, 2004; Márquez & Dalmasso, 2003). The same pattern was found in riparian habitats of Chorrillos River (Calderón et al., 2014). Plant cover did not have a clear pattern related to urbanization/water quality. The most impacted site at the river (PR4) had the lowest cover, as a result of the clearing of shores for camping areas, while the site with the best water quality and better representation of native flora, PR1, had low cover, and finally, highly modified sites (QS1 and QS2) had high values of plant cover.

Principal component analysis separated sites along the river with better biological and chemical quality, from those along the stream at Quebrada de los Cóndores, and clearly discriminated between QS1–QS2 and QS3–QS4. Sites PR1 and PR4 had the highest values for chemical and biological quality; the cause of the separation of PR1 along component 2 being PR1's higher number of native species and richness, because both sites had low plant cover. Sites with intermediate impacts on biological quality were PR2, PR3, QS3, and QS4, separated between them principally due to the absence of amphibians and lower values of SIWQ and BISLS in sites on Las Chacras Stream. Even with the high scores in plant cover, sites QS1 and QS2 were the most impacted. While the poor water quality of QS1 and QS2 seems to present an autodepuration effect downstream, accompanied by an improvement in the relation native/exotic plant species, these sites may be a potential focus for the spreading of invasive plants.

The authors would like to include some recommendations for the improvement in the management of water quality of PR and Las Chacras Stream. The control of urban runoff and poorly treated wastewater is considered two of the most important aspects in the rehabilitation of urban aquatic ecosystems (Findlay & Taylor, 2006). Stormwater runoff control can be achieved by the implementation of water sensitive designs, sustainable drainage systems, and/or low-impact development technologies (Ahiablame, Engel, & Chaubey, 2012). Low-impact development practices are microscale control practices used to bring the natural hydrology of a site close to that of its predevelopment conditions (Coffman, 2002). Analyzing the time, costs, and maintenance of the different control practices, the implementation of rain barrels for water retention would be the most effective practice to diminish the effect of runoff on water quality and quantity of PR. On the other hand, Las Chacras Stream is hugely affected by the untreated wastewater input released by the five star hotel located upstream. The implementation of a wastewater treatment plant is highly recommended to improve the water quality of this system.

In conclusion, chemical and biological water quality of PR and Las Chacras Stream is in good condition. However, an important impairment on biological and chemical quality, that should be monitored, was detected in upper Las Chacras Stream. This study is intended to help river scientists and managers understand the extent of the impact of urban development and the release of poorly treated wastewater on natural aquatic ecosystems. This research provides information that can be used in the design of strategies to minimize the impacts of urbanization on aquatic resources.

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REFERENCES

Adams, S. M., & Greeley, M. S. (2000). Ecotoxicological indicators of water quality: Using multi-response indicators to assess the health of aquatic ecosystems. *Water, Air, and Soil Pollut*, *123*(1–4), 103–115.

Agüero, N. S., Moglia, M. M., & Jofré, M. B. (2010). Are the pattern of abundance and distribution of anuran related ,with the structure of plant communities in aquatic habitats of San Luis City (Argentina)? *Neotropical Biology and Conservation*, *5*(2), 63–76.

Ahiablame, L. M., Engel, B. A., & Chaubey, I. (2012). Effectiveness of low impact development practices: Literature review and suggestions for future research. *Water, Air, & Soil Pollution*, *223*(7), 4253–4273.

Almeida, C. A., Quintar, S., Gonzalez, S. P., & Mallea, M. A. (2007). Influence of urbanization and tourist activities on the water quality of the Potrero de los Funes River (San Luis—Argentina). *Environmental Monitoring and Assessment*, *133*(1–3), 459–465.

APHA. American Public Health Association (2005). *Standard Methods for the Examination of Water and Wastewater* (21st ed.). Washington DC, USA.

Bagur, M. G., Morales, S., & López-Chicano, M. (2009). Evaluation of the environmental contamination at an abandoned mining site using multivariate statistical techniques. The Rodalquilar (southern Spain) mining district. *Talanta*, *80*(1), 377–384.

Bode, R. W., & Novak, M. A. (1995). Development and application of biological impairment criteria for rivers and streams in New York State. In W. S. Davis, & T. P. Simon (Eds.), *Biological assessment and criteria: Tools for water resource planning and decision making* (pp. 97–107). Boca Ratón, Florida: Lewis Publishers.

Braun-Blanquet, J. (1928). *Pflanzensoziologie* (1st ed.). Berlin: Springer-Verlag.

Braun-Blanquet, J. (1932). *Plant sociology: The study of plant communities*. New York and London: Macgraw Hill.

Burger, J. (2006). Bioindicators: Types, development, and use in ecological assessment and research. *Environmental Bioindicators*, *1*(1), 22–39.

Burne, M. R., & Griffin, C. R. (2005). Habitat associations of pool-breeding amphibians in eastern Massachusetts, USA. *Wetlands Ecology and Management*, *13*(3), 247–259.

Calderon, M. R., & Jofré, M. B. (2011). *Hypsiboas pulchellus* (Montevideo Treefrog, Ranita del Zarzal). Argentina: San Luis Province. *Herpetological Review*, *42*(3), 385.

Calderón, M. R., González, P., Moglia, M., González, S. O., & Jofré, M. (2014). Use of multiple indicators to assess the environmental quality of urbanized aquatic environments in San Luis, Argentina. *Environmental Monitoring and Assessment*, *186*(7), 4411–4422.

Canter, L. W., & Atkinson, S. F. (2011). Multiple uses of indicators and indices in cumulative effects assessment and management. *Environmental Impact Assessment Review*, *31*(5), 491–501.

Capon, S. J., & Dowe, J. L. (2007). Diversity and dynamics of riparian vegetation. In *Principles for riparian lands management* (pp. 3–33). Australia: Land and water.

Carignan, V., & Villard, M. A. (2002). Selecting indicator species to monitor ecological integrity: A review. *Environmental Monitoring and Assessment*, *78*(1), 45–61.

Carle, M. B., Halpin, P. N., & Stow, C. A. (2005). Patterns of watershed urbanization and impacts on water quality. *Journal of the American Water Resources Association*, *41*(3), 693–708.

Cei, J. M. (1980). Amphibians of Argentina. N.S. In *Monografía 2*, Firenze, Italy: Monitore Zoologico Italiano.

Coffman, L. S. (2002). Low-impact development: An alternative stormwater management technology. In R. L. France (Ed.), *Handbook of Water Sensitive Planning and Design* (pp. 97e124). Washington, D.C.: Lewis.

Colman Broggi, G., & Bellagamba, J. (2006). Calidad de agua de cursos en el Uruguay y análisis de normativa vigente. In *Congreso Interamericano de Ingeniería Sanitaria y Ambiental* Punta del Este, Uruguay. (Vol. 30) (pp. 1–11). AIDIS.

Connolly, N. M., Crossland, M. R., & Pearson, R. G. (2004). Effect of low dissolved oxygen on survival, emergence, and drift of tropical stream macroinvertebrates. *Journal of the North American Benthological Society*, *23*(2), 251–270.

Davidson-Bennett K. M. (2011). Watershed urbanization impacts to headwater streams in northeastern Ohio. Thesis, Master of Science, Ohio State University, USA.

Duguay, S., Eigenbrod, F., & Fahrig, L. (2007). Effects of surrounding urbanization on non-native flora in small forest patches. *Landscape Ecology*, *22*(4), 589–599.

Ector, L., & Rimet, F. (2005). Using bioindicators to assess rivers in Europe: An overview. In S. Lek, M. Scardi, P. Verdonschot, J. Descy, & S. Park (Eds.), *Modelling community structure in freshwater ecosystems* (pp. 7–19). Berlin: Springer-Verlag.

Faulkner, S. (2004). Urbanization impacts on the structure and function of forested wetlands. *Urban Ecosystems*, *7*(2), 89–106.

Findlay, S. J., & Taylor, M. P. (2006). Why rehabilitate urban river systems? *Area*, *38*(3), 312–325.

Hamer, A. J., & McDonnell, M. J. (2008). Amphibian ecology and conservation in the urbanising world: A review. *Biological Conservation*, *141*(10), 2432–2449.

Hodkinson, I. D., & Jackson, J. K. (2005). Terrestrial and aquatic invertebrates as bioindicators for environmental monitoring, with particular reference to mountain ecosystems. *Environmental Management*, *35*(5), 649–666.

Hood, W. G., & Naiman, R. J. (2000). Vulnerability of riparian zones to invasion by exotic vascular plants. *Plant Ecology*, *148*(1), 105–114.

Houlahan, J. E., Findlay, C. S., Schmidt, B. R., Meyer, A. H., & Kuzmin, S. L. (2000). Quantitative evidence for global amphibian population declines. *Nature*, *404*(6779), 752–755.

Instituto de Botánica Darwinion. (2014). Catálogo de las plantas vasculares de Argentina. Available at http://www2.darwin.edu.ar/Proyectos/FloraArgentina/FA.asp. Acceded on March 28, 2014.

Instituto Nacional de Estadística y Censos (INDEC) 2010. Resultados definitivos censo2010. http://www.censo2010.indec.gov.ar/resultadosdefinitivos.asp. Acceded on 10 March 2012

Kalesnik, A., & Malvárez, A. I. (2004). Las especies exóticas invasoras en los sistemas de humedales. *El caso del Delta inferior del Río Paraná. INSUGEO. Miscelánea*, *12*, 131–138.

Kalesnik F., Cagnoni M., Bertolini P., Quintana R. D., Madanes N. (2005). La vegetación del Refugio Educativo de la Ribera Norte, partido de San Isidro, Provincia de Buenos Aires, Argentina. Invasión de especies exóticas. Miscelánea 14 :140.

Kaller, M. D., & Kelso, W. E. (2007). Association of macroinvertebrate assemblages with dissolved oxygen concentration and wood surface area in selected subtropical streams of the southeastern USA. *Aquatic Ecology*, *41*(1), 95–110.

Leynaud, G. C., Pelegrin, N., & Lescano, J. N. (2006). Anfibios y Reptiles. In E. H. Bucher (Ed.), *Bañados del río Dulce y Laguna Mar Chiquita (Córdoba, Argentina)* (pp. 219–235). Córdoba: Academia Nacional de Ciencias.

Longing, S. D., & Haggard, B. E. (2010). Biological assessment to support ecological recovery of a degraded headwater system. *Environmental Management*, *46*(3), 459–470.

Lopez, R. D., & Fennessy, M. S. (2002). Testing the floristic quality assessment index as an indicator of wetland condition. *Ecological Applications*, *12*(2), 487–497.

Macdonald, I. A. W., & Fame, G. W. (1988). Invasion of introduced species into nature reserves in tropical savannas and dry woodlands. *Biological Conservation*, *44*(1–2), 67–93.

Márquez, J., & Dalmasso, A. D. (2003). Las comunidades de ambientes húmedos del Parque Nacional El Leoncito, San Juan, Argentina. *Multequina*, *12*(2), 55–67.

McKinney, M. L. (2002). Urbanization, biodiversity, and conservation the impacts of urbanization on native species are poorly studied, but educating a highly urbanized human population about these impacts can greatly improve species conservation in all ecosystems. *Bioscience*, *52*(10), 883–890.

McKinney, M. L. (2006). Urbanization as a major cause of biotic homogenization. *Biological Conservation*, *127*(3), 247–260.

McKinney, M. L. (2008). Effects of urbanization on species richness: A review of plants and animals. *Urban Ecosystems*, *11*(2), 161–176.

Mensing, D. M., Galatowitsch, S. M., & Tester, J. R. (1998). Anthropogenic effects on the biodiversity of riparian wetlands of a northern temperate landscape. *Journal of Environmental Management*, *53*(4), 349–377.

Metcalfe-Smith J. L. (1994). Biological water-quality assessment of rivers: Use of macroinvertebrate community. In: Calow P, Petts GE (eds.) *The Rivers Hanbook: Hydrological and Ecological Principles (II)*. Blackwell Science: Oxford, UK, pp 144–170.

Muenz, T. K., Golladay, S. W., Vellidis, G., & Smith, L. L. (2006). Stream buffer effectiveness in an agriculturally influenced area, southwestern Georgia. *Journal of Environmental Quality*, *35*(5), 1924–1938.

Oliva González, S., Almeida, C. A., Quintar, S., Mallea, M. A., & Gonzalez, P. S. (2011). Application of multivariate statistical techniques to evaluate organic pollution on a river in Argentina. *Ambiente e Agua-An Interdisciplinary Journal of Applied Science*, *6*(3), 27–42.

Oliva González, S., Almeida, C. A., Calderón, M. R., Mallea, M. A., & González, P. (2014). Assessment of the water quality on a river affected by organic pollution: Application of chemometrics in spatial and temporal variations. *Environmental Science and Pollution Research*, *21*(18), 10583–10593.

Oscoz, J., Gomá, J., Ector, L., Cambra, J., Pardos, M., & Durán, C. (2007). Estudio comparativo del estado ecológico de los ríos de la cuenca del Ebro mediante macroinvertebrados y diatomeas. *Limnetica*, *26*(1), 143–158.

Pillsbury, F., & Miller, J. (2008). Habitat and landscape characteristics underlying anuran community structure along an urban–rural gradient. *Ecological Applications*, *18*(5), 1107–1118.

Queralt, R. (1982). La calidad de las aguas en los ríos. *Tecnología del Agua*, *4*, 49–57.

Reading, C. J., & Jofré, G. M. (2003). Reproduction in the nest building vizcacheras frog *Leptodactylus bufonius* in central Argentina. *Amphibia-Reptilia*, *24*(4), 415–427.

Resh, V. H., & Unzicker, J. D. (1975). Water quality monitoring and aquatic organisms: The importance of species identification. *Journal Water Pollution Control Federation*, *47*(1), 9–19.

Ridley, S. P. D., Busteed, G. T., Kats, L. B., Vandergon, T. L., Lee, L. F., Dagit, R. G., … Sauvajot, R. M. (2005). Effects of urbanization on the distribution and abundance of amphibians and invasive species in southern California streams. *Conservation Biology*, *19*(6), 1894–1907.

Sanabria, E. A., Quiroga, L. B., & Acosta, J. C. (2005). Patrones de actividad temporal estacional y uso de microhábitat de una población de adultos de *Bufo arenarum*, en los humedales de Zonda, San Juan, Argentina. *Boletín de la Sociedad Herpetológica Mexicana*, *13*(2), 61–65.

Sewell, D., & Griffiths, R. (2009). Can a single amphibian species be a good biodiversity indicator? *Diversity*, *1*(2), 102–117.

Stephens, K. A., Reid, D., & Graham, P. (2002). *Stormwaters planning: A guide for British Columbia*. British Columbia Ministry of Water, Land and Air Protection Environment Canada. Greater Vancouver Regional District, Vancouver, BC.

Stribling, J. B., Jessup, B. K., White, J. S., Boward, D., & Hurd, M. (1998). *Development of a benthic index of biotic integrity for Maryland streams. Report CBWP-EA-98-3*. Annapolis, Maryland: Maryland Department of Natural Resources.

Sun, J. W., & Narins, P. M. (2005). Anthropogenic sounds differentially affect amphibian call rate. *Biological Conservation*, *121*(3), 419–427.

Sung, C. Y., Li, M. H., Rogers, G. O., Volder, A., & Wang, Z. (2011). Investigating alien plant invasion in urban riparian forests in a hot and semi-arid region. *Landscape and Urban Planning*, *100*(3), 278–286.

Sutula M., Stein E. (2003). Habitat value of natural and constructed wetlands used to treat urban runoff: A literature review. Technical Report 388. Southern California Coastal Water Research Project. Westminster, CA.

Tarr, T. L., Baber, M. J., & Babbitt, K. J. (2005). Macroinvertebrate community structure across a wetland hydroperiod gradient in southern New Hampshire, USA. *Wetlands Ecology and Management*, *13*(3), 321–334.

Urbina-Cardona, J. N., Olivares-Perez, M., & Reynoso, V. H. (2006). Herpetofauna diversity and microenvironment correlates across a pasture-edge-interior ecotone in tropical rainforest fragments in the Los Tuxtlas Biosphere Reserve of Veracruz. *Biological Conservation*, *132*(1), 61–75.

Vallania, E. A., Garelis, P. A., Trípole, E. S., & Gil, M. A. (1996). Un índice biótico para las Sierras de San Luis (Argentina). *Rev UNRC*, *16*(2), 129–136.

Wallace, J. B., Grubaugh, J. W., & Whiles, M. R. (1996). Biotic indices and stream ecosystem processes: Results from an experimental study. *Ecological Applications*, *6*(1), 140–151.

Weir, L. A., & Mossman, M. J. (2005). North American Amphibian Monitoring Program (NAAMP). In M. J. Lannoo (Ed.), *Amphibian* *declines: Conservation status of United States species* (pp. 307–313). Berkeley, California, USA: University of California Press.

Zampella, R. A., Bunnell, J. F., Laidig, K. J., & Procopio, N. A. (2006). Using multiple indicators to evaluate the ecological integrity of a coastal plain stream system. *Ecological Indicators*, *6*(4), 644–663.

Zhou, F., Huang, G. H., Guo, H., Zhang, W., & Hao, Z. (2007). Spatio-temporal patternsand source apportionmentofcoastal water pollution in eastern Hong Kong. *Water Research*, *41*(15), 3429–3439.

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