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Agrochemicals' effects on functional feeding groups of macroinvertebrates in Pampas streams

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A B S T R A C T

The use of agrochemicals in Argentina has increased over the last decades and may represent an environmental risk for adjacent water bodies. In this work we study the invertebrate assemblages in nine streams sampled at 11 sites with different land use on the adjacent plots in the years 2011–2014. Four sites were located inside a biosphere reserve, 4 sites were located adjacent to livestock plots and the other 3 sites were adjacent to cropped plots. The taxa composition was assessed and sorted into functional feeding groups: shredders, gatherers, filterers, scrapers and predators. Significant differences were detected among the functional feeding groups according to the use given to the adjacent land. Total density and taxonomic richness were significantly higher in the streams next to the reserve and livestock sites than in those adjacent to cropped plots; there were no differences between the first two. Gatherers and shredders density was significantly higher in the reserve and livestock than in the cropped sites. Scrapers and predators were the best represented at the cropped sites. Nutrient concentrations in water and endosulfan concentrations in the bottom sediments were also higher at the cultivated sites, suggesting that agrochemical loads from land cultivation caused the observed differences in composition.

1. Introduction

Crop production has increased in Argentina over the last decades due to agricultural intensification. Traditionally, a mixed system of livestock raising and crop production, mainly of wheat and corn, was the main land use. Soy was not a traditional crop. Genetically modified, glyphosate-resistant soy was introduced on the Argentine market in 1996 and steadily increased in production, having since then become the main crop (MAGPyA, 2016). Wheat and soy varieties with a short growing period have allowed two harvests per year, wheat followed by soy. Pesticide consumption increased from 6 million kilograms in 1992 (Pengue, 2000) to 32 million kilograms in 2012 (CASAFE, 2013). Fertilizer application increased from 0.22 million tons in the early 1990 s to 3.2 million tons in 2013 (CIAFA, 2016). Endosulfan has been the most widely used pesticide in Argentina along with cypermethrin (Astoviza et al., 2015), followed by chlorpyrifos. Among them, endosulfan persists the longest in the environment (Castro et al., 2002; Pramanic et al., 2012). All of these have been detected in the bottom sediments of Pampas streams (Hunt et al., 2016). In the present study endosulfan was chosen as an overall indicator of insecticide loads from the adjacent crops.

Nonpoint source contamination with agrochemicals is increasingly recognized as a cause of water quality deterioration in inland waters (Schulz, 2004). Pesticides are incorporated into aquatic habitats mainly through runoff (Jergentz et al., 2004). Downstream transport results in ephemeral toxicity pulses. It has only rarely been possible to sample peak pesticide concentrations. Chemical data are instantaneous in nature and therefore require large numbers of measurements for an accurate assessment. By contrast, living organisms incorporate environmental conditions over long periods of time. Because the invertebrate community is formed by species with different sensitivity, the immediate exposure effect would be a change in the relative abundance of the different community components. Furthermore, pesticide exposure may lead to changes within the composition by altering species interactions. Therefore, the aquatic biota structure and function is expected to change as a result of agrochemical contamination and may potentially be used to evaluate environmental impacts (Segnini, 2003).

Macroinvertebrates show different specializations with regard to food obtainment, allowing classifications based on different functional feeding groups (FFGs). Wallace and Webster (1996) and Merritt et al. (2008) established 5 different functional groups: gatherers, shredders, filterers, scrapers and predators. Feeding strategies represent typical traits reflecting the adaptation of species to environmental conditions.

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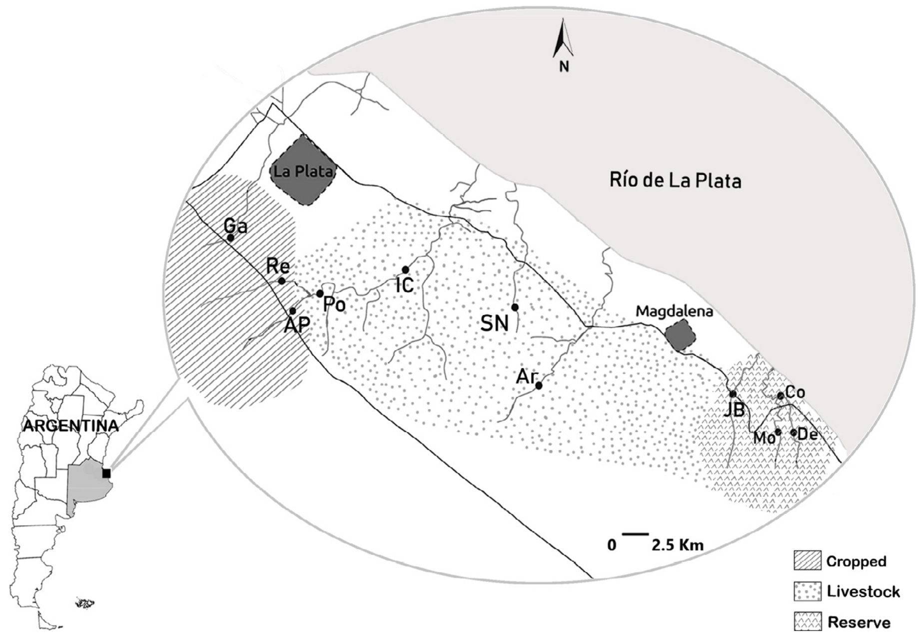

Fig. 1. Studied area and sampling sites. Abbreviations: Ga: Gato, Re: Remes, AP: Afluente Pescado, Po: Poblet, IC: Ignacio Correa, SN: Sin Nombre, Ar: Arregui, JB: Juan Blanco, Co: Confluencia, Mo: Morales, De: Destino.

The structure of functional feeding groups may form part of a unified measurement across communities differing in taxonomic composition. Hurd et al. (1996) reported changes in the functional feeding composition of a community in response to pesticide exposure. In South America, information about the FFG classification of invertebrates in streams is almost absent (Tomanova et al., 2006). The effect of agrochemical applications on the functional group composition in streams on the Pampas remains unreported. The objective of the present work is to compare the composition and structure of invertebrate functional groups at stream sites with different land uses on adjacent plots in Pampas streams, Buenos Aires province, Argentina.

2. Materials and methods

2.1. Study area

The study area belongs to the eastern border of the Pampas plain, on the River Plate coastal strip, Buenos Aires province, Argentina (Fig. 1). The Pampas are a vast plain area covered by sediments originating in volcanic ash contributed by the Andes mountain chain during the Holocene (Hurtado et al., 2006). The studied area has a small west-to-east slope of about 0.1%, and the streams run roughly parallel. The maximum height is 25 m above sea level (Hurtado et al., 2006). Soil and bottom sediment texture is quite uniform and dominated by the silt fraction (43%) while sand and clay account for 29% and 28%, respectively. Mean sediment organic matter is 4.5% (Marrochi, 2018). The climate is mild and humid; mean monthly temperatures range from 9.9 °C in July to 22.4 °C in January. Mean annual rainfall is 1060 mm, without a dry period; slightly lower precipitations occur in winter (196 mm) and higher ones in summer (289 mm) (Hurtado et al., 2006; Fucks et al., 2017).

Nine streams were studied at 11 sites with different land use in the surrounding plots (Fig. 1). Furthest from La Plata city, four streams were sampled within the "Parque Costero Sur", UNESCO Biosphere Reserve (Athor, 2009). These were Confluencia (Co) (35°7′53.10″S; 57°24′1.47″W), Destino (De) (35°8′15.35″S; 57°23′40.2″W), Morales (Mo) (35°8′17.54″S; 57° 23′57.7″W) and Juan Blanco (JB) (35°8′30.23″S; 57°26′23.98″W). The landscape is grassland with small patches of forest. Four sampling sites were surrounded by livestock plots: one each on the streams Arregui (Ar) (35°7′22.1″S; 57°41′11.6″W) and Sin Nombre (SN) (35°2′22.8″S; 57°42′40.5″W), and two sites along the Pescado stream: Poblet (Po) (35°2′2.45″ S; 57°56′ 34.3″W), and Ignacio Correa (IC) (35°1′23.97″S; 57°51′27.42″W). The Pescado headwaters lie in a cropped area; the Poblet and Ignacio Correa sampling sites were located 3.8 and 8 km downstream, respectively, from the cropped area (Fig. 1). Livestock on natural pastures without fertilization was the main land use at these four sites. Three streams: Remes (Re) (35°1′31.87″S; 57°59′39.6″W), Afluente Pescado (AP) (35°3′13.08″S; 57°58′35.3″W) and Gato (Ga) (34°58′53.8″S; 58°3′12.1″W) drain basins under cultivation. The Remes and Afluente Pescado streams are located close to each other and join together to form the Pescado stream (Fig. 1). Main crops were soy and corn in the Remes and Afluente Pescado basins and tomatoes along the Gato. The most used insecticides in the area were endosulfan, cypermethrin and chlorpyrifos. Pesticides are usually applied some three times per growing period, in spring and summer (October-March). Fertilizers are applied during the sowing, mainly in October and November. Three different sampling periods were studied during three successive crop growing seasons (Table 1).

The studied streams lack forested borders. Conspicuous macrophyte growth is a common feature. Macrophyte dominance was not related to land use in the adjacent plots. Strongest growth was attained at the

Table 1
Sampling sites in each land use area in the three successive sampling periods.

Period	Number of samplings	Cropped	Livestock	Reserve
Dec. 2011–Apr. 2012	4	Remes	Poblet Ignacio Correa Arregui	Blanco Destino
Dec. 2012–Apr. 2013	4	Remes Gato	Arregui Sin Nombre	Confluencia Destino Morales
Dec. 2013–Mar. 2014	4	Remes Gato Afluente Pescado	Arregui Sin Nombre Morales	Confluencia Destino

Remes cropped site and the Sin Nombre livestock site. At Remes, the emergent macrophyte *Typha domingensis* completely covered the water surface. The Gato cropped site also showed *T. domingensis* growth, though attaining lower densities, accompanied by *Ludwigia peploides*. The Destino and Morales reserve streams also showed conspicuous macrophyte growth; *L. peploides* and *Myriophyllum aquaticum* were dominant in the Destino stream and *Azolla filiculoides* and *M. aquaticumin* in the Morales. By contrast, the Ignacio Correa, Poblet and Arregui livestock streams and the Juan Blanco reserve stream showed a sparse fringe of the emergent *Schoenoplectus californicus* along the stream borders, attaining lower cover. The Confluencia reserve stream and the Arregui livestock stream also showed the dominance of *S. californicus*, with higher biomass. In the Confluencia reserve stream *S. californicus* was accompanied by *L. peploides* and *A. filiculoides*. The Afluente Pescado contained the smallest macrophyte cover, consisting in the emergent macrophyte *G. spinlantoides* sparsely covering the water surface.

2.2. Environmental variables

Stream width and depth were assessed with a measuring tape. Water temperature (T°C), conductivity, pH and dissolved oxygen (DO) were measured with a multiparameter device (Yellow Springs Instruments SI 556). Surface water samples were carried in coolers to the laboratory and filtered through Whatman GF/C filters. Dissolved nutrients were determined in the filtrate following APHA (2012). Soluble reactive phosphorus (SRP) was determined by the reaction with molybdate-ascorbic acid, nitrite by diazotation and nitrate by reduction with hydrazine followed by diazotation, and ammonium by the reaction with indophenol blue.

2.3. Pesticide analysis

Sediment samples were collected with a stainless steel scoop from the top two centimeters, and placed in amber glass jars. The samples were kept on ice in coolers until arrival at the laboratory and stored refrigerated until extraction, performed within the following couple of days. Sediments were extracted with a mixture of acetone and methylene chloride following You et al. (2004). Cleanup procedures were carried out using Florisil solid phase extraction (SPE) cartridges (USEPA, 2007). Endosulfan was measured by gas chromatography as reported by Solis et al. (2016, 2017, 2018)), with a detection limit of 0.5 ng/g dw.

2.4. Macroinvertebrate sampling

Samples of macroinvertebrates were collected from emerging vegetation by means of a D-net with a 30 cm diameter and 500 µm mesh size. The same sampling procedure was performed, covering an area of 1 m^2, for each sample (Ramirez, 2010). Three replications were taken at each sampling site. Collected samples were passed through a screen to eliminate surplus water and then placed in plastic bags with 80% alcohol and erythrosin B for later separation, identification and counting (Barbour et al., 1999; Merritt et al., 2008). The macroinvertebrates were identified under a stereoscopic microscope using the taxonomic keys of Domínguez and Fernández (2009) and Merritt et al. (2008), to the taxonomic family level as a minimum. Then the taxa were classified according to functional feeding groups (FFGs), following Cummins and Klug (1979), Wallace and Webster (1996) and Merritt et al. (2008), into gatherers, which feed on fine particulate organic matter (FPOM; < 1 mm in diameter); shredders, which shred large pieces of macrophyte vascular tissue or decomposing wood (CPOM; > 1 mm); filterers, which present specialized anatomical structures such as setae or oral brushes with silk secretions; scrapers, adapted to scrape the films developed on mineral or organic substrates, and predators, which feed on animal tissues.

2.5. Statistical analysis

Sampling sites were sorted into three groups according to land use in the adjacent plots: reserve (Juan Blanco, Confluencia, Destino and Morales streams), cropped (Remes, Afluente Pescado and Gato streams) and livestock (Poblet, Ignacio Correa, Arregui and Sin Nombre streams). Environmental measurements at each land use plot were compared by means of analysis of variance (ANOVA) followed by the Tukey test at a significance level of $p \leq 0.05$. Statistical assumptions of homogeneity (Levene) and normality (Shapiro-Wilk) were assessed; whenever the variables did not meet the corresponding assumptions for the use of a parametric model, the Kruskal–Wallis, nonparametric ANOVA on ranks test was performed, followed by Dunn's test. Whenever endosulfan concentrations turned out not to be detectable, the quantification limit was used in the statistical comparison. The analyses were performed using the SigmaPlot 12.0 program.

Those variables that showed significant differences among the different land uses were chosen to test their effect on the invertebrate assemblage composition by means of a multivariate analysis using the CANOCO version 4.53 program (ter Braak and Smilauer, 1998). We worked on the total abundance matrix; only those taxa whose abundance represented a contribution higher than 0.5% of the total abundance, and endosulfan concentrations above quantification limits, were used. The biological variables were transformed to $\log_{10}(x + 1)$ and chemical parameters were standardized. A detrended Correspondence Analysis (DCA) was used to find the response model (linear or unimodal) for the choice of subsequent direct gradient analysis. The results indicated a linear model, so a redundancy RDA analysis would better describe the data (ter Braak and Smilauer, 1998; Leps and Smilauer, 2003). A Monte Carlo permutation test was used to verify the significance of the model (ter Braak and Smilauer, 1998; Leps and Smilauer, 2003).

3. Results

3.1. Environmental variables

The landscape within the studied area is fairly uniform and the limnological variables emerged as similar in all streams and sites. No significant differences were detected in the measured environmental variables (Table 2) among different land uses. Conductivity showed an extensive variation range. Higher values were measured in coincidence with dry periods. Oxygen concentrations were generally high at all sites. However, low oxygen concentrations were occasionally measured in coincidence with high temperatures at the sites with larger macrophyte cover: Remes, Sin Nombre, Destino and Morales.

Nitrate, ammonium and SRP concentrations were significantly higher at the sites adjacent to cropped plots than at the livestock and reserve sites ($p \leq 0.05$), without significant differences between the latter (Fig. 2), except for SRP which was higher at the livestock than at the reserve sites. Nitrite concentrations were not significantly different for the different land uses.

Table 2
Environmental variables measured at the studied sites, means (variation ranges).

	Reserve	Livestock	Cropped
Temp. (°C)	21 (13–34)	24 (16–31)	22 (15–26)
OD (mg/l)	4 (0.5–10)	6 (0.4–11)	4 (1–9)
Conduct. (µS/cm)	624 (134–3930)	543 (62–1928)	300 (131–579)
pH	7.4 (6.6–8.7)	7.5 (6.5–8.8)	7.4 (6.3–8.3)
Width (m)	11 (5–25)	10 (4–25)	6 (1–11)
Depth (m)	0.5 (0.1–1)	0.65 (0.1–2)	0.43 (0.10–0.80)

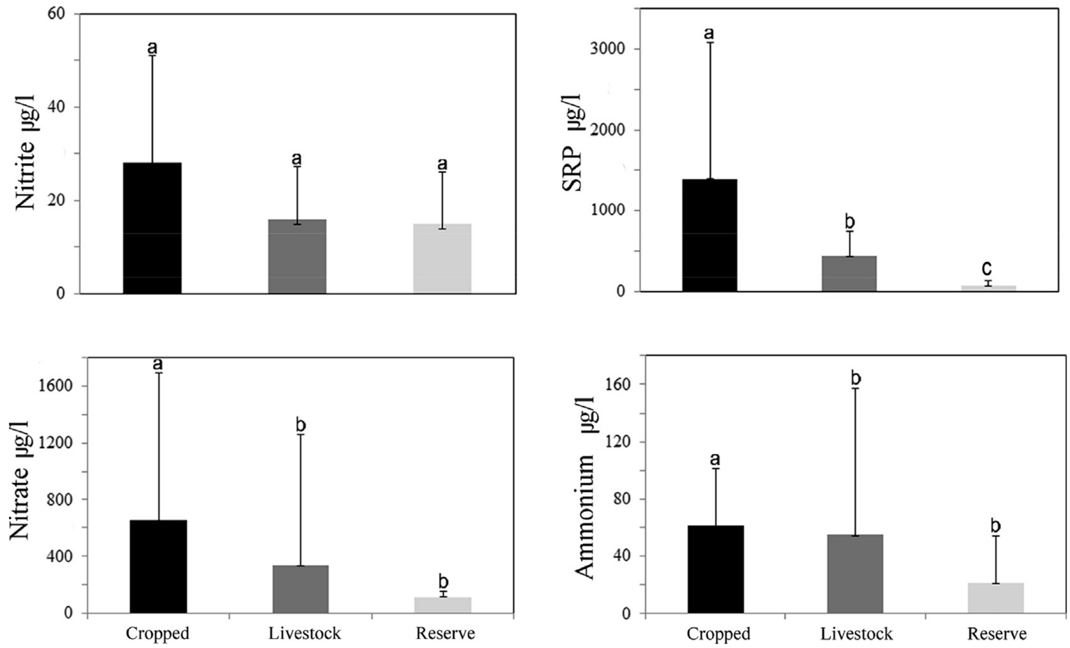

Fig. 2. Nutrient concentrations at stream sites adjacent to plots with different land uses (mean and standard deviation). Significant differences are marked with different letters ($p < 0.05$).

3.2. Insecticides

Endosulfan concentrations were commonly below detection limits at the reserve and livestock sites; detection frequency was significantly lower (9 and 18%, respectively) than in the cropped streams (75%). Concentrations were significantly higher at the sites adjacent to crops (8.7 ± 12.1 ng/g dry weight) than at the reserve (1.3 ± 3.7 ng/g dry weight) and livestock (2.7 ± 6 ng/g dry weight) sites, without significant differences between the latter.

3.3. Macroinvertebrate assemblages

Macroinvertebrate density and taxonomic richness were significantly higher at the reserve and livestock than at the cultivation sites ($p \leq 0.05$). Fifty-three and 54 different taxa were at the reserve and livestock sites, respectively, against 43 at the cropped sites. Twenty-one taxa were recorded at either reserve or livestock sites without being recorded at the cultivation streams.

Shredder and gatherer density was significantly higher at the reserve and livestock than at the cropped sites, without significant differences between the first two (Table 3). Shredders decreased from 1004–1856 ind/m^2 at the reserve and livestock sites to 116 ind/m^2 at the cropped sites. *Hyalella curvispina,* the most abundant shredder taxon, decreased from 907 to 1773 ind/m^2 at the reserve and livestock sites to 92 ind/m^2 at the cropped sites. Predator density was also significantly higher at the livestock than at the cropped sites. Scraper density was not significantly different among different land uses. Filterers represented a minor component of the invertebrate assemblages, amounting to roughly 1% of the total density at the reserve and livestock sites while representing 6% at the cultivated sites (Fig. 3). At the reserve and livestock sites, scrapers, shredders and gatherers were significantly more abundant than filterers. Scrapers and predators were the groups best represented at the cropped sites. Within the cropped sites, predator density was significantly higher than that of shredders, gatherers and filterers.

The multivariate analysis (RDA) performed to assess the contribution of the selected variables (endosulfan, SRP, nitrate and ammonium) in the overall variation of the community composition and site differentiation showed that it was significantly explained by the 4 selected variables. The first axis accounted for 63% and the second for 24% of the total variability. SRP weighted heavily to the left of the first canonical axis, followed by ammonium, endosulfan and nitrate, while ammonium weighted heavily in the second canonical axis. The cropped sites were related in the analysis to higher endosulfan, nitrate and SRP concentrations. Likewise, several of the livestock sites were related to ammonium. By contrast, the reserve sites were located on the opposite side. The scraper *P. canaliculata* and the predators Enochrus, Dugesiidae and Glossiphoniidae were related to a gradient of environmental deterioration, evidenced by the higher concentrations of endosulfan and nutrients. Most other taxa gathered together on the opposite side of the gradient (Fig. 4).

4. Discussion

The high similarity in the environmental variables in all streams and sites is indicative of landscape homogeneity within the studied area. Higher nutrient concentrations in the streams adjacent to cropped plots suggest the contribution of fertilizer applications in the surrounding crops. The highest SRP concentration in Remes was coincident with the runoff events following crop fertilization (Mugni, 2009). Mugni et al. (2013) reported higher SRP concentrations in a first order stream when the adjacent crop was fertilized than the previous year when the same crop was not fertilized. Similarly, Hart et al. (2004) reported higher SRP concentrations in a first order stream when the surrounding pastures were fertilized. Increased nutrient concentrations in streams draining cultivated basins have repeatedly been reported in the literature and interpreted as the result of fertilizer applications (Boyer and Pasquarell, 1995; Johnson et al., 1997; Meybeck, 1998; Mugni et al., 2005).

Significantly higher endosulfan concentrations in the cropped basins suggest the contribution of pesticide applications to the adjacent crops. Endosulfan detection frequencies and measured concentrations were consistently found to offer no significant differences between the livestock and reserve sites, where no insecticide applications took place, thus providing further evidence that applications to crops were the main source of endosulfan to the Pampas streams. Occasional endosulfan detection at low concentrations in the reserve and livestock streams is not surprising. Endosulfan is known to attain long range atmospheric transportation from the application sites (Weber et al.,

Table 3
Total density (ind/m²) of each taxon for different land uses during the studied period.

		Reserve	Livestock	Cropped
FILTERERS		55	45	83
Culicidae	*Culicidae*	55	45	83
GATHERERS		1947	937	50
Baetidae	*Callibaetis*	14	0	0
Baetidae	*Baetidae*	209	306	14
Caenidae	*Caenidae*	25	40	2
Elmidae	*Elmidae*	6	4	0
Chironomidae	*Chironomidae*	1098	366	11
Stratiomyidae	*Stratiomyidae*	523	53	13
Ephydridae	*Ephydridae*	11	25	2
Ostracoda	*Ostracoda*	61	143	8
SHREDDERS		1004	1856	116
Curculionidae	*Curculionidae*	89	49	10
Haliplidae	*Haliplus*	0	5	0
Chrysomelidae	*Chrysomelidae*	2	8	11
Hydrochidae	*Hydrochus*	0	2	3
Tipulidae	*Tipulidae*	1	0	0
Palaemonidae	*Palaemonetes argentinus*	0	19	0
Hyalellidae	*Hyalella curvispina*	907	1773	92
Lepidoptera	*Lepidoptera*	5	0	0
SCRAPERS		2157	1571	556
Sphaeriidae	*Sphaeriidae*	0	2	20
Ancylidae	*Gundlachia*	30	9	14
Planorbidae	*Biomphalaria peregrina*	1988	1519	311
Ampullaridae	*Pomacea canaliculata*	133	40	143
Physidae	*Stenephysa marmorata*	1	0	15
Scirtidae	*Scirtidae*	5	1	53
PREDATORS		1078	2431	524
Hirudinea	*Hirudinea*	38	184	73
Dugesiidae	*Dugesiidae*	0	147	107
Aeshnidae	*Aeshnidae*	45	27	25
Libellulidae	*Libellulidae*	16	29	25
Anisoptera	*Anisoptera*	1	2	0
Coenagrionidae	*Coenagrionidae*	1	23	0
Lestidae	*Lestidae*	0	9	0
Zygoptera	*Zygoptera*	235	419	51
Corixidae	*Corixini*	20	39	5
	Sigara	3	7	2
Belostonatidae	*Belostomatinae*	4	14	0
	Belostoma	110	91	34
Notonectidae	*Notonecta*	100	58	10
	Notonectidae	1	1	0
Pleidae	*Neoplea*	33	22	2
Nepidae	*Curicta*	1	5	0
Aphidoidea	*Aphidoidea*	0	1	4
Hebridae	*Lipogomphus*	10	5	1
Staphylinidae	*Staphylinidae*	1	4	5
Noteridae	*Hydrocanthus*	69	42	6
	Suphis	2	2	0
Hydrophilidae	*Tropisternus*	44	153	23
	Berosus	13	26	5
	Derallus	9	18	4
	Enochrus	31	128	53
	Anacaenini ind	20	105	18
	Hydrobiomorpha	14	4	0
	Helochares	1	2	0
	Hydrophilidae ind	1	11	11
Dytiscidae	*Bidessini*	118	436	41
	Thermonectus	1	0	0
	Laccophilus	55	81	7
	Laccodytes	32	75	7
	Lancetes	1	0	0
	Megadytes	1	0	0
Tabanidae	*Tabanidae*	10	4	4
Ceratopogonidae	*Ceratopogonidae*	37	256	1
Muscidae	*Muscidae*	0	1	0

2010). Astoviza et al. (2015) studied endosulfan concentrations in the air over an extended area including the tracts studied in the present work. Maximum aerial concentrations were measured over a cropped site between the Remes and Gato streams, where the highest concentrations in bottom sediments were measured in the present study, while the minimum was reported over a livestock site, in Magdalena, close to the reserve (Fig. 1), where the minimum sediment concentrations were measured in the present study.

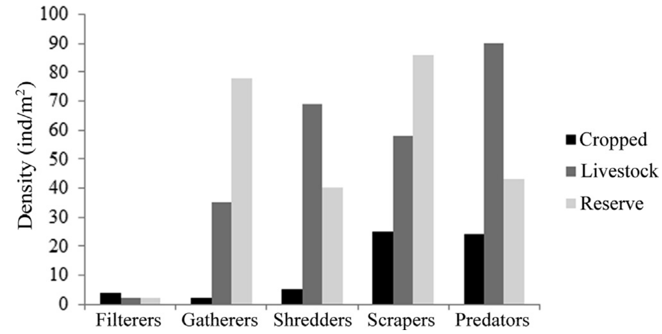

Fig. 3. Density (ind/m²) of each functional group for the different land uses.

Several authors reported the effect of land use on stream invertebrate assemblages (Subramanian et al., 2005; Farrell et al., 2015; Fierro et al., 2017; Villada-Bedoya and Ospina-Bautista 2017). The harmful effect of agriculture has repeatedly been reported. Although the quoted authors referred to the potential effect of agrochemical applications, none of these studies measured insecticide concentrations and therefore paid more attention to discussing the effect of other processes linked to agricultural land use, such as, for example, the deterioration of riparian vegetation at the cropped sites. Macrophyte complexity has been associated with high abundance and richness of macroinvertebrates in Pampas streams (Ferreiro et al., 2014). Macrophyte growth forms with highly branched and dissected leaves support larger numbers of macroinvertebrates than macrophytes with firm, undissected stalks (Walker et al., 2013). In the present study the vegetation structure in each stream did not show relationship with land use in adjacent plots. The largest biomass occurred at the Remes cropped site and in the Sin Nombre livestock stream. Three of the four livestock sites (Poblet, Ignacio Correa, Arregui) and the Juan Blanco stream at the Reserve contained a sparse border strip of the thin and narrow *Scirpus californicus* emergent macrophyte. The observed relationships among the agrochemical-related variables (nutrients and endosulfan) and the assemblage composition were attained in spite of the potential variability introduced by other, unquantified variables such as vegetation structure.

Fu et al. (2015) studied the impacts of land use (forest, agricultural and urban) on macroinvertebrate functional feeding groups in the Dongjiang River Basin, China. In results similar to ours, Fu et al. (2015) reported that the abundance of shredders was significantly higher in forest than in cropped and urban streams. In our study, the multivariate analysis determined a gradient of environmental deterioration that significantly affects the assemblage composition. The Pampas streams lack forested borders but contain conspicuous riparian macrophyte cover. Macroinvertebrate food supply consists mainly of leaves and litter contributed by the surrounding environment. Shredders play an important role in the process of leaf litter decomposition (Jonsson et al., 2001). The large decrease in shredders' abundance at the cultivated sites seems related to pesticide exposure. Mugni et al. (2011) reported toxicity to *H. curvispina* in runoff and stream water at the Remes cropped site in coincidence with the runoff episodes following insecticide application to the adjacent crops. Toxicity to *H. curvispina* in runoff water after endosulfan application to experimental soy plots, followed by rain simulation events, was documented by Paracampo et al. (2012). The pesticides most applied in the area (cypermethrin, chlorpyrifos, endosulfan) are largely hydrophobic. The insecticides contributed by runoff are mainly adsorbed on the particulate fraction (Jergentz et al., 2004), likely to be leaves and litter, because the crop canopy intercepts the applied insecticide spray, consistent with the larger effect observed on the shredder and gatherer feeding groups as

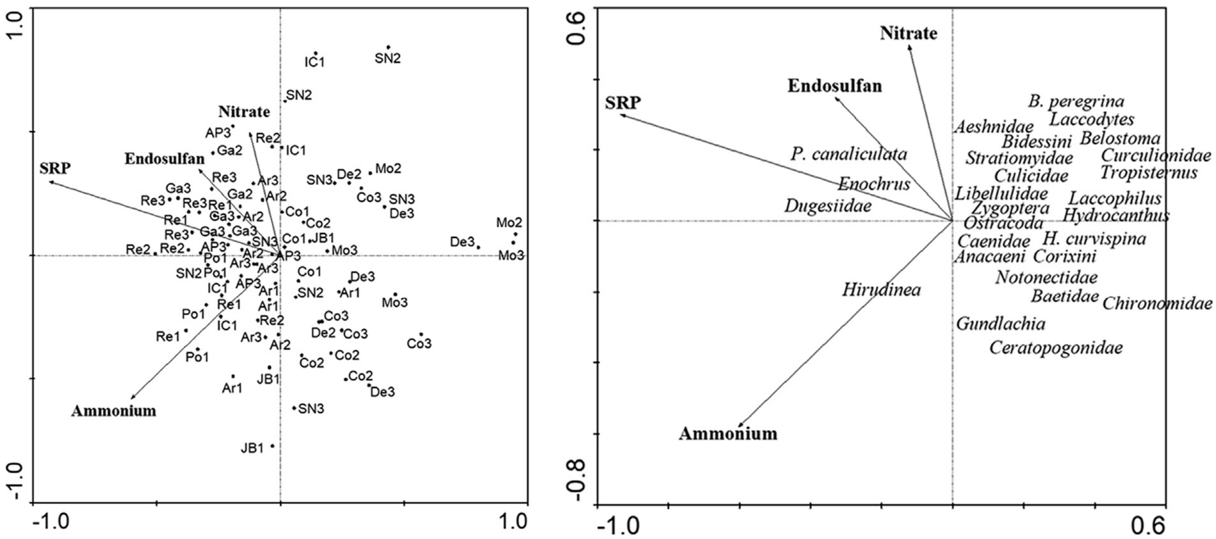

Fig. 4. Biplots of species (left graph) and samples (right graph) versus environmental variables in Pampas streams. Ga: Gato, Re: Remes, AP: Afluente Pescado, Mo: Morales, De: Destino, Co: Confluencia, JB: Juan Blanco, Ar: Arregui, SN: Sin Nombre, Po: Poblet, IC: Ignacio Correa. Numbers 1–3 refer to the different sampling periods (Table 1).

observed in the present study. Hurd et al. (1996) studied the effect on stream macroinvertebrate assemblages of aerial diflubenzuron application to forests. Shredders, the dominant functional feeding group, showed reduced densities in the treated watersheds. Rodrigues et al. (2018) studied the effect of exposure to the insecticide chlorantraniliprole on the structure and function of benthic macroinvertebrate communities in stream mesocosms and reported that the abundance of shredders and grazers was strongly reduced, decreasing leaf decomposition and increasing primary production.

Amphipods in general and *Hyalella* in particular are considered to be shredders (Cummins and Klug, 1979; Zilli et al., 2008; Principe et al., 2010; Patrick, 2013; Baudy et al., 2016; Jourdan et al., 2016). However, Misserendino (2007) considered *Hyalella* to be a collector-gatherer. The same taxon, with the same mouthpart morphology, can behave as a shredder or collector-gatherer depending on different environmental circumstances and the relative abundance of different food supplies (Tomanova et al., 2006). Both gatherers and shredders process allochtonous organic matter, both were strongly reduced in the cropped sites in the present study, and both are expected to be affected by insecticides adsorbed to particulate organic matter.

Predators and scrapers were the groups best represented at the cropped sites. The scraper *P. canaliculata* and the predators Enochrus, Dugesiidae and Glossiphoniidae have consistently been reported in the literature as comparatively tolerant to pesticide exposure (Levy and Miller, 1978; Lapkina and Arkhipova, 2000; Liess and Von der Ohe, 2005). Wogram and Liess (2001) developed a classification of invertebrates according to their specific relative sensitivity to toxic substances including insecticides. Also, Neumann and Dudgeon (2002) estimated relative physiological tolerances by family. Both studies are in agreement with our results, identifying Hyalellidae as a sensitive group and Ampullariidae (*P. canaliculata)*, Hydrophilidae (Enochrus), Dugesiidae and Glossiphoniidae as tolerant organisms. Tolerant scrapers and predators might profit from increased available resources at cropped sites (Farrell et al., 2015). Higher nutrient concentrations at the cultivated sites might increase the phytoplankton and periphyton biomass, thus favoring scrapers and filterers, functional feeding groups which increased their proportion of the total abundance at the cultivated sites. Among the predators, several insects develop terrestrial adult stages, allowing faster recolonization after toxicity pulses (Wallace et al., 1986; Wallace, 1990), consistent with the comparatively higher proportion of predators observed at the cultivated sites.

Livestock and reserve sites, where agrochemicals were not applied, did not show significant differences in nutrients and endosulfan concentrations and assemblage composition. Extensive livestock breeding on natural pastures represents an activity of low impact on the water quality and the resident fauna. Moreover, it is interesting to note that Poblet did not differ from the other livestock and reserve sites in spite of being located 3 km downstream from the Remes cropped site, indicating that the stream recovered from agrochemical impact over that distance. Therefore, leaving a strip of livestock land along the stream borders seems an adequate preservation measure to attain a sustainable production in the Pampas.

5. Conclusion

The composition of macroinvertebrate assemblages was significantly different in the streams adjacent to cropped plots from that in streams running next to livestock and reserve fields. The general response of macroinvertebrates to the disturbance caused by agriculture was a significant decrease in total macroinvertebrate abundance and taxa richness. Macroinvertebrate functional feeding group composition was also significantly different. Shredders and gatherers were feeding group well represented at the reserve and livestock sites and were the groups most significantly reduced in the cropped streams.

Multivariate analysis showed that most taxa present in the reserve and livestock streams were significantly decreased at the cropped sites in coincidence with increased nutrient and endosulfan concentrations, while a few tolerant taxa increased with them. It is thus suggested that the large decrease in abundance and richness in the cultivated streams is caused by insecticide exposure.

Scrapers and predators were the most abundant functional feeding groups at the cultivated sites. Higher nutrient concentrations at the cultivated sites presumably increased available resources for the tolerant scrapers.

Present evidence suggests that the ongoing agricultural intensification in the Pampas region is causing a large scale change in the macroinvertebrate functional feeding groups' composition, dominance shifting from shredders and gatherers to scrapers and predators. Shredders could be used for monitoring and assessing land use impacts on stream water quality.

Extensive livestock breeding on natural pastures did not produce any discernible effect on the macroinvertebrate composition, and therefore could be suggested as a land use along stream borders that attenuates the agrochemical impact.

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References

APHA, 2012. Standard Methods for the Examination of Water and Wastewater, 19th ed. American Public Health Association, Washington, DC.

Astoviza, M.J., Cappelletti, N., Bilos, C., Migoya, M.C., Colombo, J.C., 2015. Massive airborne Endosulfan inputs related to intensive agriculture in Argentina's Pampa. Chemosphere 144, 1459–1466.

Athor, J. (Ed.), 2009. Parque Costero del Sur – Naturaleza, Conservación y Patrimonio Cultural. Fundación de Historia Natural "Félix de Azara", Buenos Aires, pp. 528.

Barbour, M.T., Gerritsen, J., Snyder, B.D., Stribling, J.B., 1999. Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish, second edition.

Baudy, P., Zubrod, J.P., Konschak, M., Weil, M., Schulz, R., Bundschuh, M., 2016. Does long-term fungicide exposure affect the reproductive performance of leaf-shredders? A partial life-cycle study using *Hyalella azteca*. Environ. Pollut. 222, 458–464.

Boyer, D., Pasquarell, G., 1995. Nitrate concentrations in karst springs in an extensively grazed area. Water Resour. Bull. 31, 729–736.

CASAFE (Cámara de Sanidad Agropecuaria y Fertilizantes, Buenos Aires, Argentina), 2013. < http://www.casafe.org/sobrelaindustria.htm > . (Accessed on November 2013).

Castro, J., Sánchez-Brunete, C., Rodríguez, J., Tadeo, J., 2002. Persistence of chlorpyrifos and endosulfan in soil. Fresenius Environ. Bull.

CIAFA (Cámara de la Industria Argentina de Fertilizantes y Agroquímicos), 2016. < http://www.ciafa.org.ar > . (Accessed November 2016).

Cummins, K.W., Klug, M.J., 1979. Feeding ecology of stream invertebrates. Ann. Rev. Ecol Syst. 10, 147–172.

Domínguez, E., Fernández, H.R. (Eds.), 2009. Macroinvertebrados Bentónicos. Sistemática y Biología, first ed. Fundación Miguel Lillo, Tucumán, pp. 656.

Farrell, K., van Vuren, J.H.J., Ferreira, M., 2015. Do aquatic macroinvertebrate communities respond to land-use effects in the Wilge River, Mpumalanga, South Africa? Afr. J. Aquat. Sci. https://doi.org/10.2989/16085914.2015.1025220.

Ferreiro, N., Feijoó, C., Giorgi, A., Rosso, J., 2014. Macroinvertebrates select complex macrophytes independently of their body size and fish predation risk in a Pampean stream. Hydrobiologia. https://doi.org/10.1007/s10750-014-1953-y.

Fierro, P., Bertrán, C., Tapia, J., Hauenstein, E., Peña-Cortés, F., Vergara, C., Cerna, C., Vargas-Chacoff, L., 2017. Effects of local – use on riparian vegetation, water quality, and the functional organization of Macroinvertebrate assemblages. Sci. Total Environ. 609, 724–734.

Fu, L., Jiang, Y., Ding, J., Liu, Q., Zhi Peng, Q., Yi Kang, M., 2015. Impacts of land use and environmental factors on macroinvertebrate functional feeding groups in the Dongjiang River basin, southeast China. J. Freshwater Ecol. https://doi.org/10.1080/02705060.2015.1017847.

Fucks, E., D'Amico, G., Pisano, M., Nuccetelli, G., 2017. Evolución geomorfológica de la región del gran la plata y su relación con eventos catastróficos. Rev. l Asoc. Geol. Argent. 74, 141–154.

Hart, M.R., Quin, B.F., Nguyen, M.L., 2004. Phosphorus runoff from agricultural land and direct fertilizer effects: a review. J. Environ. Qual. 33, 1954–1972.

Hunt, L., Bonetto, C., Resh, V.H., Forsin Buss, D., Fanelli, S., Marrochi, N., Lydy, M.J., 2016. Insecticide concentrations in stream sediments of soy production regions of South America. Sci. Total Environ. 547, 114–124.

Hurd, M., Perry, S., Perry, W., 1996. Nontarget effects of a test application of diflubenzuron to the forest canopy on stream macroinvertebrates. Environ. Toxicol. Chem. 15, 1344–1351.

Hurtado, M.A., Giménez, J.E., Cabral, M.G. (Eds.), 2006. Análisis Ambiental Del Partido de La Plata: Aportes al Ordenamiento Territorial, first ed. Consejo Federal de Inversiones, Buenos Aires, pp. 125.

Jergentz, S., Pessacq, P., Mugni, H., Bonetto, C., Schulz, R., 2004. Linking in situ bioassays and dynamics of macroinvertebrates to assess agricultural contamination in streams of the Argentine Pampa. Ecotoxicol. Environ. Saf. 59, 133–141.

Johnson, L., Richards, C., Host, G., Arthur, J., 1997. Landscape influences on water chemistry in Midwestern stream ecosystems. Freshwater Biol. 37, 193–208.

Jonsson, M., Malmqvist, B., Hoffsten, P.O., 2001. Leaf litter breakdown rates in boreal streams: does shredder species richness matter? Freshwater Biol. 46, 161–217.

Jourdan, J., Westerwald, B., Kiechle, A., Chen, W., Streit, B., Klaus, S., Oetken, M., Plath, M., 2016. Biol. Invasions 18, 763–774.

Lapkina, L.N., Arkhipova, N.R., 2000. Comparative analysis of tolerance to pesticides in Annelids. Russ. J. Ecol. 31, 336–340.

Leps, J., Smilauer, P., 2003. Multivariate Analysis of Ecological Data using CANOCO. Cambridge University Press, Cambridge.

Levy, R., Miller, T.W., 1978. Tolerance of the planarian *Dugesia dorotocephala* to high concentrations of pesticides and growth regulators. Entomophaga 23, 31–34.

Liess, M., Von der Ohe, P., 2005. Analyzing effects of pesticides on invertebrate communities in streams. Environ. Toxicol. Chem. 24, 954–965.

MAGPyA (Ministerio de agroindustria ganadería y pesca), 2016. < http://www.siia.gov.ar/_apps/siia/estimaciones/estima2.php > . (Accessed on February 2016).

Marrochi, M.N., 2018. Impacto de agroquímicos sobre los ensambles de macroinvertebrados en arroyos rurales. Ph.D. dissertation. La Plata University, pp. 150.

Merritt, R.W., Cummins, K.W., Berg, M.B. (Eds.), 2008. An introduction to the aquatic insects of North America. Kendall-Hunt, Dubuque, pp. 1158.

Meybeck, M., 1998. Man and river interface: multiple impacts on water and particulates chemistry illustrated in the Seine river basin. Hydrobiologia 373 (374), 1–20.

Misserendino, M.L., 2007. Macroinvertebrate functional organization and water quality in a large arid river from Patagonia (Argentina). Annal. Limnol. 43, 133–145.

Mugni, H., Jergentz, S., schulz, R., Maine, A., Bonetto, C., 2005. Phosphate and nitrogen compounds in streams of Pampean Plain areas under intensive cultivation (Buenos Aires, Argentine). Phosphates in Sediments, 4th International Symposium, pp. 163-170.

Mugni, H.D., 2009. Concentración de Nutrientes y Toxicidad de Pesticidas en Aguas Superficiales de Cuencas Rurales. Ph.D. dissertation. La Plata University, pp. 140.

Mugni, H., Ronco, A., Bonetto, C., 2011. Insecticide toxicity to Hyalella curvispina in runoff and stream water within a soybean farm (Buenos Aires, Argentina). Ecotoxicol. Environ. Saf. 74, 350–354.

Mugni, H., Paracampo, A., Bonetto, C., 2013. Nutrient concentrations in a pampasic first order stream with different land uses in the surrounding plots (Buenos Aires, Argentina). Bull. Environ. Contam. Toxicol. 91, 391–395.

Neumann, M., Dudgeon, D., 2002. The impact of agricultural runoff on stream benthos in Hong Kong, China. Water Res. 36, 3103–3109.

Paracampo, A., Mugni, H., Demetrio, P., Pardi, M., Bulus, G., Asborno, M., Bonetto, C., 2012. Toxicity persistence in runoff from experimental soybean plots following pesticide applications. J. Environ. Sci. nd Health, Part B 47, 761–768. https://doi.org/10.1080/03601234.2012.676358.

Patrick, C.J., 2013. The effect of shredder community composition on the production and quality of fine particulate organic matter. Freshwater Sci. 32, 1026–1035.

Pengue, W., 2000. Cultivos Transgénicos ¿Hacia Dónde Vamos? Lugar Editorial S. A, Buenos Aires, pp. 190.

Pramanic, S., Saha, P., Bhattacharyya, A., Dhuri, A., 2012. Studies on the fate of ready-mix formulation of endosulfan and cypermethrin in tea. Bull. Environ. Contam. Toxicol. 89, 1016–1020.

Principe, R.E., Gualdoni, C., Oberto, A., Raffaini, G., Corigliano, M., 2010. Spatial-temporal patterns of functional feeding groups in mountain streams of Córdoba, Argentina. Ecol. Austral 20, 257–268.

Ramirez, A., 2010. Métodos de recolección. Rev. Biol. Trop. 58, 41–50.

Rodrigues, A., Machado, A., Bordalo, M., Saro, L., Simao, F., Rocha, R., Golovko, O., Zlábek, V., Barata, C., Soares, A., Pestana, J., 2018. Invasive species mediate insecticide effects on community and ecosystem functioning. Environ. Sci. Technol. 52, 4889–4900.

Schulz, R., 2004. Field studies on exposure, effects, and risk mitigation of aquatic non-point-source insecticide pollution: a review. J. Environ. Qual. 33, 419–448.

Segnini, S., 2003. El uso de los macroinvertebrados bentónicos como indicadores de la condición ecológica de los cuerpos de agua corriente. Ecotropicos 16, 45–63.

Solis, M., Mugni, H., Hunt, L., Marrochi, N., Fanelli, S., Bonetto, C., 2016. . Land use effect on invertebrate assemblages in Pampasic streams (Buenos Aires, Argentina). Environ. Monit. Assess. https://doi.org/10.1007/s10661-016-5545-3.

Solis, M., Mugni, H., Fanelli, S., Bonetto, S., 2017. Effect of agrochemicals on macroinvertebrate assemblages in Pampasic streams, Buenos Aires, Argentina. Environ. Earth Sci. https://doi.org/10.1007/s12665-017-6476-1.

Solis, M., Bonett, C., Marrochi, N., Paracampo, A., Mugni, H., 2018. Aquatic macroinvertebrate assemblages are affected by insecticide applications on the Argentine Pampas. Ecotoxicol. Environ. Saf. 148, 11–16.

Subramanian, K.A., Sivaramakrishnan, K.G., Gadgil, M., 2005. Impact of riparian land use on stream insects of Kudremukh National Park, Karnataka state, India. J. Insect Sci. 1536–2442 ISSN: 1536-2442.

ter Braak, C., Smilauer, P., 1998. CANOCO Reference Manual and User's Guide to Canoco for Windows: Software for Canonical Community Ordination (ver. 4). Microcomputer Power, Ithaca, NY.

Tomanova, S., Goitia, E., Helesic, J., 2006. Trophic levels and functional feeding groups of macroinvertebrates in neotropical streams. Hydrobiologia 556, 251–264.

USEPA US Environmental Protection Agency, 2007. Report on the Environment: Science Report. < https://cfpub.epa.gov/ncea/risk/recordisplay.cfm?Deid = 140917 > .

Villada-Bedoya, S., Ospina-Bautista, F., G-Dias, L., Varón, J., 2017. Biol. Trop. 65, 1635–1659.

Walker, P.D., Wijnhoven, S., van der Velde, G., 2013. Macrophyte presence and growth form influence macroinvertebrate community structure. Aquat. Bot. 104, 80–87.

Wallace, J.B., Vogel, D.S., Cuffney, T.F., 1986. Recovery of a headwater stream from an insecticide-induced community disturbance. J. N. Am. Benthol. Soc. 5, 115–126.

Wallace, J.B., 1990. Recovery of lotic macroinvertebrate communities from disturbance. Environ. Manage. 14, 606–620.

Wallace, J.B., Webster, J.R., 1996. The role of macroinvertebrates in stream ecosystem function. Ann. Rev. Entomol. 41, 115–139.

Weber, J., Halsall, C.J., Muir, D., Teixeira, C., Small, J., Solomon, K., Hermanson, M., Hung, H., Bidleman, T., 2010. Endosulfan, a global pesticide: a review of its fate in the environment and occurrence in the Arctic. Sci. Total Environ. 408, 2966–2984.

Wogram, J., Lies, M., 2001. Rank ordering of macroinvertebrate species sensitivity to toxic compounds by comparison with that of daphnia magna. Bull. Environ. Contam. Toxicol. 67, 360–367.

You, J., Weston, D.P., Lydy, M.J., 2004. A sonication extraction method for the analysis of pyrethroid, organophosphate, and organochlorine pesticides from sediment by gas chromatography with electron-capture detection. Arch. Environ. Contam. Toxicol. 47 (2), 141–147.

Zilli, F., Montalto, L., Marchese, M., 2008. Benthic invertebrate assemblages and functional feeding groups in the Paraná River floodplain (Argentina). Limnologica 38, 159–171.