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Neotropical Entomology

ISSN 1519-566X
Volume 42
Number 1

Neotrop Entomol (2013) 42:22-31
DOI 10.1007/s13744-012-0093-1

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ECOLOGY, BEHAVIOR AND BIONOMICS

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Keywords

Aquatic insect, artificial ponds, richness estimation, urban environment, urban management

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Edited by Fernando B Noll – UNESP

Received 13 December 2011 and accepted 1 September 2012
Published online 2 November 2012

Abstract

Freshwater habitats are important elements within urban green space and they are endangered by various types of human activity. With the aim to increase the knowledge about species biodiversity in urban ecosystems, we characterised the assemblages of aquatic insects in four permanent man-made ponds in Buenos Aires city (Argentina) during a 1-year period. We recorded 32 species with *Sigara* spp. (Hemiptera) as the most abundant. The removal of aquatic vegetation from the studied ponds may have affected both the establishment and permanence of the insect community. Swimmers were the dominant group in the studied sites, followed by burrowers and sprawlers, and only a few strictly climbers were collected. Therefore, all sampled ponds were dominated by collectors (principally gatherers), secondarily by predators and only few shredders were detected, which was much affected by the removal of macrophytes. Non-parametric abundance indexes estimated a number of species very close to the observed number in each site. Conversely, the incidence indexes estimated more species because there were many more taxa present only in one sample than those represented by few individual in a sample. Our data provides some insights on the community of man-made ponds that can improve the management of these aquatic urban habitats. Considering that macrophytes affect animal assemblages due to their role as physical structures that increase the complexity or heterogeneity of habitats, they should not be removed by authorities in order to promote biodiversity.

Introduction

Urban environments are heterogeneous mosaics of residential dwellings, commercial properties, parks, and other land-use types that provide an array of habitats for arthropods (McIntyre 2000). In terms of biological diversity, urban habitats are usually regarded as poorer due to their high anthropogenic degradation as compared with agriculture or forestry neighbouring areas (Koperski 2010). Urbanisation leads to ecosystem destruction, habitat fragmentation, and species extinction (Vermonden *et al* 2009); hence, restoration, preservation, and enhancement of biodiversity in urban areas are becoming increasingly important (Savard *et al* 2000). The consequences of urbanisation include changes in the richness, composition, and individual species abundance of animal and plant assemblages (Smith *et al* 2006). Particularly, the loss of urban green space jeopardises the overall biodiversity of urban and suburban areas, and forces us to consider the importance of existing urban nature more carefully in the planning process (Yli-Pelkonen & Niemelä 2005). Niemelä (1999) pointed out several reasons to consider the value of urban settings in ecological studies. First, as most people live in urban areas (50% worldwide, 80% in industrialised countries), ecological knowledge of the effects that humans have on urban ecosystems is imperative to create healthy and pleasant environments. Second, in urban areas, ecological processes are comparable to those outside them. Third, the considerable variation in urban habitats and their species diversity has been poorly documented and finding explanations for the phenomena involved and predicting changes as urbanisation proceeds are challenges for urban ecologists.

Important elements within urban green space are freshwater habitats, and they are endangered by various types of human activities (Gledhill *et al* 2008). Most research on specific water body types has focused on rivers, streams, and lakes with little data describing other smaller natural or man-made habitats such as ditches, ponds, headwater streams, springs, and flushes (Williams *et al* 2003). Particularly, with a few notable exceptions, the wider ecology of ponds had been almost entirely neglected and, as a habitat, ponds were largely ignored by freshwater biologists and policy makers (Biggs *et al* 2005). Fortunately, it has become increasingly clear that research activity focusing upon ponds and other small water bodies has risen significantly over the past few years (Oertli *et al* 2009). A study conducted in an anthropogenic disturbed region revealed that ponds can contribute highly to freshwater biodiversity at a regional level, with recent evidence showing that they often support considerably more species, more unique species and more scarce species than other water body types (Williams *et al* 2003).

There are relatively few general studies on arthropods in urban environments excluding those related to pest control or epidemiology (McIntyre 2000). Insects are good bioindicators of environmental changes, air and water pollution and quality of environment, and it is well known that studies on their communities may be applied in urban planning, design, and management (Zapparoli 1997). Therefore, we aimed to describe the structure of aquatic insect assemblages in man-made permanent ponds and to analyse their functional aspects considering functional feeding groups and habits in the Buenos Aires city, Argentina.

Material and Methods

Study area

Buenos Aires city (34°35′S, 58°29′W) is located at 25 m asl, on the right margin of the Río de la Plata River, an estuary of approximately 50 km of width in this section. The diameter of this almost circular city is approximately 16 km and its surface area covers ca. 204 km^2. Its population of 2.9 million inhabitants constitutes the core of a huge megalopolis of 12.8 million people (INDEC 2010). The climate is humid temperate, with four definite seasons, mean annual precipitation of 1,076 mm and mean annual temperature of 17.4°C (Instituto Geográfico Militar 1998).

Sampling sites were located in one of the largest (≈80 ha) green spaces of the city, known as "Parques de Palermo" (Fig 1). This parkland is daily used as a recreational area and frequently is cleaned by the customary collection of garbage and lawn cutting. Four man-made permanent freshwater habitats were selected for the present study. These habitats were classified as ponds according to the following definitions: (a) water bodies between 1 m^2 and 2 ha in area, which may be permanent or seasonal, including both man-made and natural water bodies (Biggs *et al* 2005) and (b) a water body with a maximum depth of no more than 8 m, offering water plants the possibility to colonise almost the entire area of the pond (Oertli *et al* 2000). The main characteristics of the four studied ponds are summarised in Table 1. Water chemical values recorded on January 2004 were provided by the city authorities. These urban ponds are eutrophic, with differences on their chemical variables. P1, usually shadowing, showed the highest conductivity, chlorides, hardness and the lowest value of dissolved oxygen. Conversely, P4 was the most oxygenated may be due to the great photosynthetic activity, with the highest value of pH, total phosphorus and ammonium. In these ponds, no free-floating plant was observed and the removal of submerged vegetation by the park authorities was observed on each sample date. The surface water temperature of each water body was measured on every sample date with a digital thermometer. The surface areas of water bodies were calculated from aerial photographs provided by the Instituto Geográfico Militar (1998; scale 1:15,000) using GIS-Arc View 3.1.

Sampling and laboratory methods

Ponds were sampled monthly from October 2003 through September 2004. Capture stations were established every 200 steps along the whole perimeter of each pond; i.e. the number of stations was proportional to the size of the water body. Insect samples were collected with rectangular hand nets, 10×7.5 cm frame, 350 μm mesh size, during 3 min per station. Samples were immediately fixed in situ, in 80% ethanol to avoid predation. All the specimens collected were identified to the lowest taxonomic level as possible and also identifying their developmental stages (larva, pupa, and adult), due to the fact that some groups and young larvae are difficult to identify to species level. Taxonomic identifications were performed using the appropriate systematic keys and specialised literature on the local fauna: general insects (Merritt & Cummins 1984, Lopretto & Tell 1995), Coleoptera (Archangelsky 1997, Oliva *et al* 2002), Diptera (Paggi 2001), Ephemeroptera (Domínguez *et al* 2001), Hemiptera (Schnack

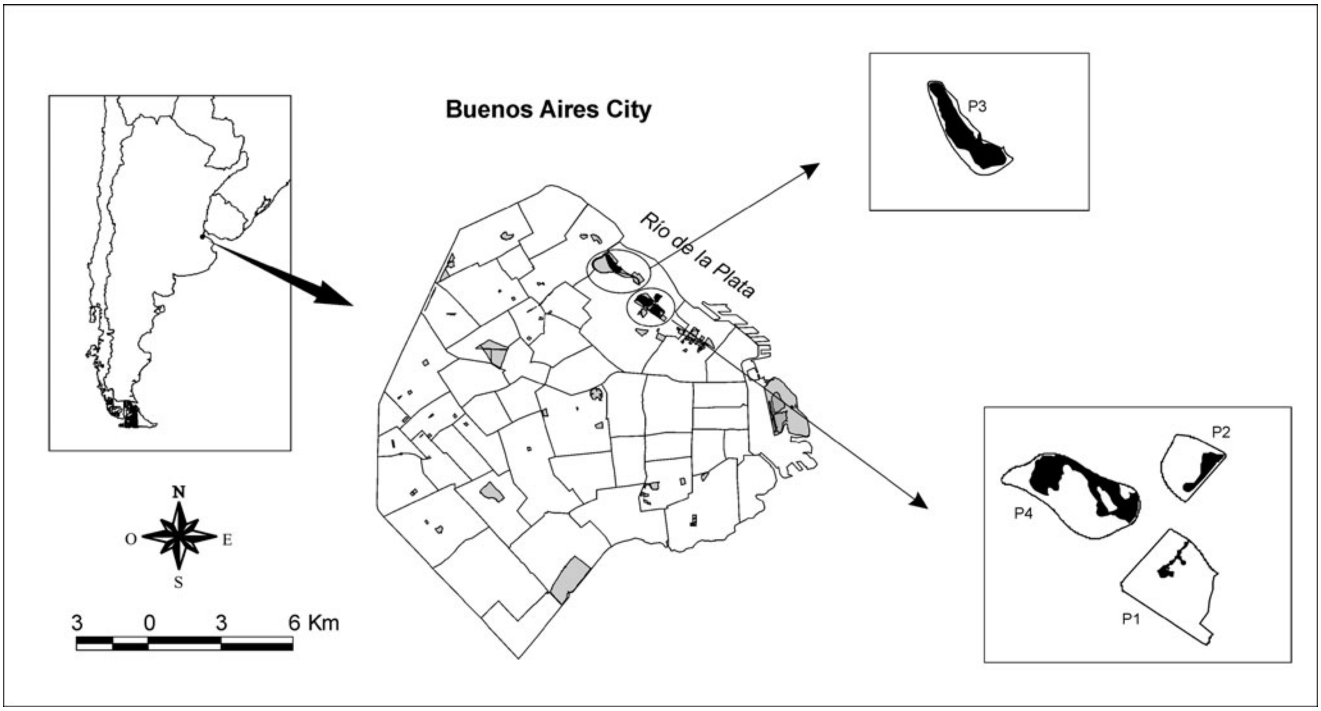

Fig 1 Geographic location of the study area showing green spaces (*grey*) and the studied ponds (*black*).

1976, Bachmann 1981, Bachmann & López Ruf 1994), and Odonata (Rodrigues Capítulo 1992).

Data analyses

The specimens collected were assigned to different Functional Feeding Groups (FFG) and habits following Merritt & Cummins (1984) and the mentioned literature on local fauna. The percentages of individual abundances of each FFG and habit were described. Several non-parametric estimators of richness were calculated for each pond: Chao1, Chao2, abundance-based coverage estimator (ACE), incidence-based coverage estimator (ICE), and Jack 2 (a second-order jackknife). Each sample date represents all of the data collected every 200 steps grouping. The EstimateS 8.2 (Colwell 2009) was used to calculate all non-parametric estimators over 500 randomised iterations of the species accumulation. To evaluate the similarity in species composition among ponds, the classic Sørensen (incidence based) and Morisita–Horn (abundance based) sample similarity indices were computed (Magurran 2004). In addition, the Chao's Sørensen abundance-based similarity index, which considers unseen species, was computed (Chao *et al* 2005). These indexes are designed to equal 1 in cases of complete similarity and 0 if the sites are totally dissimilar. Also Chao's shared species estimator for all pairs of sites was used to estimate the number of species shared by them. The EstimateS 8.2 (Colwell 2009) was used to compute all the applied indexes.

Results

The overall species richness recorded was 32. A total of 3,082 individuals belonging to five insect orders were identified as follows: 12 morphospecies in four families of Coleoptera, 11 in five families of Hemiptera, five in three families of Diptera, two families of Odonata, and two genera of Ephemeroptera (Table 2). Considering all ponds together, Hemiptera and Ephemeroptera were the most abundant orders in the insect assemblages throughout the study period. Among Hemiptera, nymphs and adults of Corixidae constituted 96% of this group, and was mainly represented by *Sigara chrostowskii* Jaczewski and *Sigara platensis* Bachmann. In Ephemeroptera, 99% was represented by larvae of *Caenis* sp., principally recorded on P3. Nevertheless, considering each pond individually, only P3 was characterised by the above mentioned taxa; in the other ponds (P1–P2–P4) *Sigara* spp. and Chironominae represented 72%, 97%, and 63%, respectively, of the insect assemblage. The lowest observed taxa richness was detected in P1 (7 taxa), the highest in P4 (23 taxa), and intermediate value in P2 and P3 (14 and 15 taxa).

The analysis of FFG showed a high dominance of collectors with lower abundances of predators and scrapers (Fig 2A). The collector–gatherers group was mainly represented by *Sigara* spp. and Ephemeroptera, and collector–filterers by Chironominae. Considered the habits categorization, swimmers (mobile forms in the water column) dominated the pond communities (Fig 2B), following by

Table 1 Main characteristics of four ponds in Buenos Aires city (Argentina), water chemical values were recorded on January 2004.

	P1	P2	P3	P4
Area (m^2)	4,700	9,500	99,700	49,400
Max. depth (m)	≈ 1	≈1	≈5	≈1
Substrate	Natural	Natural	Natural	Natural
Shoreline	Natural	Concrete and granite (sloping)	Concrete and granite (sloping)	Concrete (vertical)
Aquatic vegetation	Poorly represent and frequently removed	Poorly represent and frequently removed	Poorly represent and frequently removed	Poorly represent and frequently removed
Annual water temperature (°C) mean (min–max)	20.30 (13.6–27)	21.2 (13.9–28.5)	21.75 (14.3–29.2)	19.95 (12.8–27.1)
pH	8.33	8.53	7.40	9.86
Hardness (mg $CaCo_3$/l)	425	124	10.4	72.6
Conductivity (µS/cm)	3,820	2,520	229	1,610
Chloride (Cl^- mg/l)	1110	497	37.2	277
Total phosphorus (mg/l)	0.06	0.12	0.04	0.19
Ammonium (NH_3 mg/l)	<0.01	<0.01	0.11	0.56
January water temperature (°C)	24.40	27.20	28	27.30
Dissolved oxygen (mg/l)	2.90	5	6.30	8.14
Turbidity—Secchi (m)	bottom	bottom	1.60	0.20
Fishes	*Cnesterodon decemmaculatus* (Jenyns)	*C. decemmaculatus*	*C. decemmaculatus*, *Hoplias malabaricus* (Bloch), *Australoheros facetus* (Jenyns), *Cyprinus carpio* Linnaeus, *Prochilodus lineatus* (Valenciennes), *Gymnoyeophagus* sp. (Eigenmann)	*C. decemmaculatus*, *A. facetus*

Min minimum, *Max* maximum

Table 2 Insect taxa collected from the different ponds (P1–4).

Taxa	P1	P2	P3	P4	FFG	Hab
Ephemeroptera						
Baetidae						
Callibaetis sp. (l)			+	+	CG	SW-CL
Caenidae						
Caenis sp. (l)	+	+	+		CG	SP
Odonata						
Anisoptera (l)						
Libellulidae (l)		+			PR	SP
Zygoptera (l)						
Coenagrionidae (l)		+	+	+	PR	CL
Heteroptera						
Belostomatidae						
Belostoma sp. (l)		+		+	PR	CL-SW
B. elegans (Mayr) (a)	+				PR	CL-SW
Corixidae						
Sigara argentiniensis Hungerford (a)	+		+		CG-PR	SW-CL
S. chrostowskii Jaczewski (a+l)		+	+	+	CG-PR	SW-CL
S. denseconscripta (Breddin) (a)		+			CG-PR	SW-CL
S. platensis Bachmann (a+l)	+	+	+	+	CG-PR	SW-CL
S. rubyae (Hungerford) (a+l)		+		+	CG-PR	SW-CL
Nepidae						
Ranatra sjostedti Montandon (a)				+	PR	CL
Notonectidae						
Buenoa sp. (l)			+	+	PR	SW
Notonecta sp. (l)		+			PR	SW
Pleidae						
Neoplea maculosa (Berg) (a)	+			+	PR	SW-CL
Coleoptera						
Dryopidae						
Pelonomus sp. (a)				+	SC	SW-CL
Dytiscidae						
Brachyvatus acuminatus (Steinheil) (a)				+	PR	SW-CL
Desmopachria concolor Sharp (a)				+	PR	SW-CL
Liodessus sp. (a)			+	+	PR	SW-CL
Hydrophilidae						
Berosus sp. (a+l)			+	+	SC(a)-PR(l)	SW-DI
Paracymus sp. (a)				+	SC	SW-DI
Tropisternus burmeisteri Fernández & Bachmann (a)				+	SC	SW-DI
T. ignoratus Knisch (a)				+	SC	SW-DI
T. lateralis (Brullé) (a)				+	SC	SW-DI
T. setiger (Germar) (a)	+			+	SC	SW-DI
Noteridae						
Hydrocanthus sp. (a)			+		PR	CL
Suphisellus sp. (a)			+		PR	CL
Diptera						
Chironomidae						
Chironominae (l,p)		+	+	+	CF	BU-CL
Orthocladiinae (l,p)			+	+	CG	BU
Tanypodinae (l,p)		+		+	PR	SP-SW
Ephydridae (l,p)		+	+		?	BU-SP
Muscidae (l,p)		+	+	+	PR	SP

(a) adults, *(l)* larva, *(p)* pupa, *FFG* functional feeding groups assigned to taxa collected, *CF* collectors–filterers, *CG* collector–gatherers, *PR* predators, *SC* scrapers, *Hab* habits assigned, *BU* burrowers, *SP* sprawlers, *SW* swimmers, *CL* climbers, and *DI* divers

burrowers (sedentary forms), except in P3 where the sprawlers was the second group in importance (inhabiting fine sediments). A clear seasonal pattern of richness and abundance was observed in all ponds. From January to July, all the ponds recorded the lowest richness, represented by less than 10 individuals per site. Conversely, the highest richness and abundance were recorded during the spring months (Fig 3).

According to all non-parametric richness indexes (except Chao 2 for P4), no increment was observed when all the samples have been accumulated (Fig 4). The abundance indexes (Chao 1 and ACE) and Jack 2 estimated a number of species very close to the observed number in each site. On the contrary, the incidence indexes estimated more species than the abundance ones. A comparison of Chao 1 with Chao 2 and ACE with ICE reveals that the insect assemblages were heterogeneous. This pattern arises because there were many more unique (i.e. species represented only in one sample) than singleton species (i.e. species represented by one individual in a sample).

The analysis of similarity indexes (Table 3) suggested opposing patterns. According to the binary similarity Sørensen index, P3 and P4 were the most similar ponds (in these ponds were detected the greatest number of species) and P1 and P4 showed the major dissimilarity (in which the extremes numbers of species were recorded). Conversely, when quantitative indexes were considered, P3 and P4 showed the greatest dissimilarity. Particularly, Chao's Sørensen Abundance-based similarity index expressed a great similitude between P2 and P3 habitats, following P2–P4 and P1–P3.

Discussion

The habitats included in the present study were man-made ornamental ponds placed in one of the largest green space

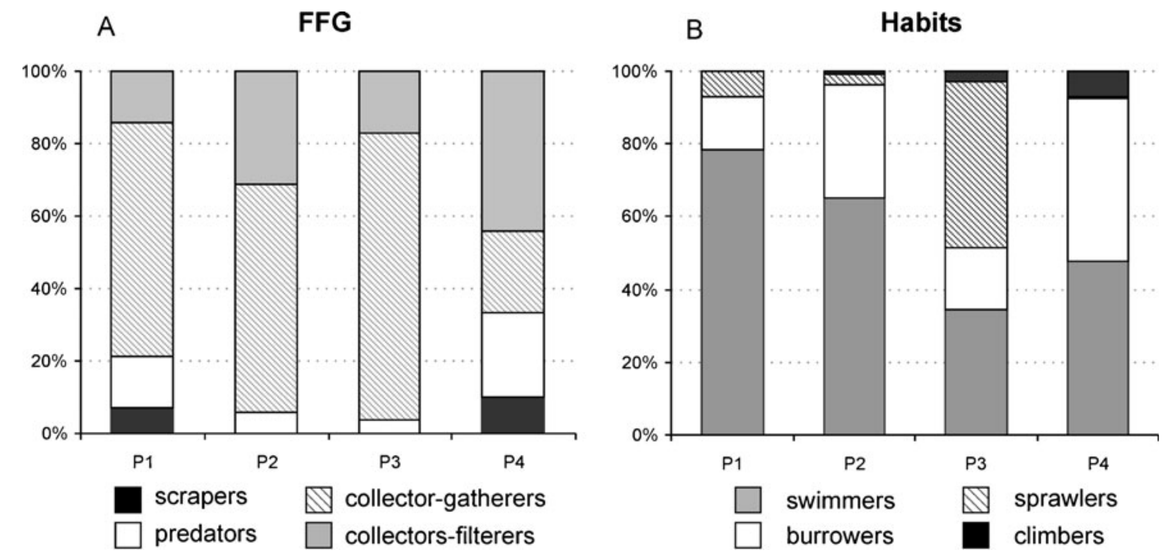

Fig 2 Percentage of abundances of individuals according to **A** functional feeding groups (FFG) and **B** habits of the collected aquatic insects in the study ponds (P1–4).

of Buenos Aires City. The number of insect taxa recorded in our survey suggests that these permanent ponds support a poor insect community in comparison with temporary ponds of the city (Fischer *et al* 2000, Fontanarrosa *et al* 2004, 2009) or natural ponds situated in an ecological reserve on the margin of the city (Fontanarrosa *et al* 2004). Almost all the taxa recorded here were common species previously detected in other ponds of the city (Fischer *et al* 2000, Fontanarrosa *et al* 2004, 2009). The only exception was the larvae of *Caenis* sp., which was detected for first time in ponds of Buenos Aires City.

The information available on macroinvertebrates assemblage in urban ponds is really scarce, or focuses on a specific group. Gledhill *et al* (2005) also found a low number of aquatic invertebrate species (mean=6 species, range between 0 and 19) in 10 urban ponds in the Northwest of England. Another work, dealing with eight urban ponds of Texas (Wolf *et al* 1999), reported a great number of

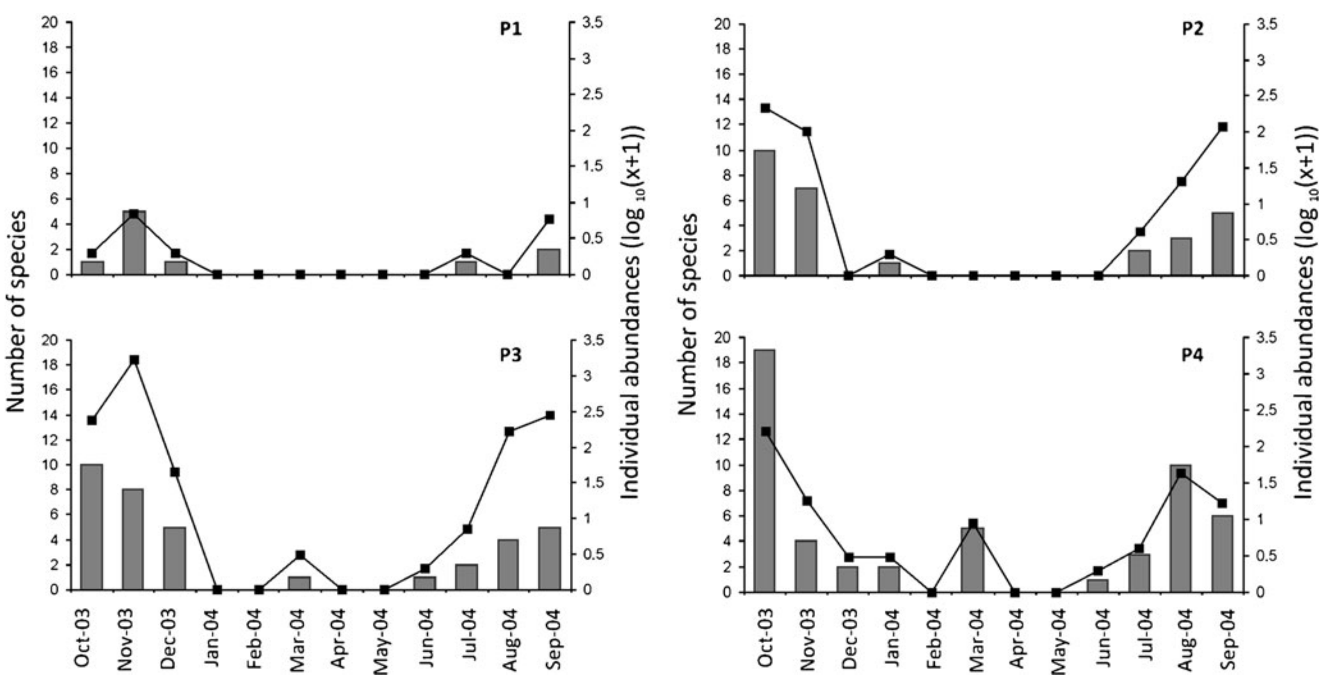

Fig 3 Monthly patterns of the number of species (*bars*) and individual abundances (*lines*) recorded in the study man-made ponds (P1–4); period Oct 2003–Sep 2004.

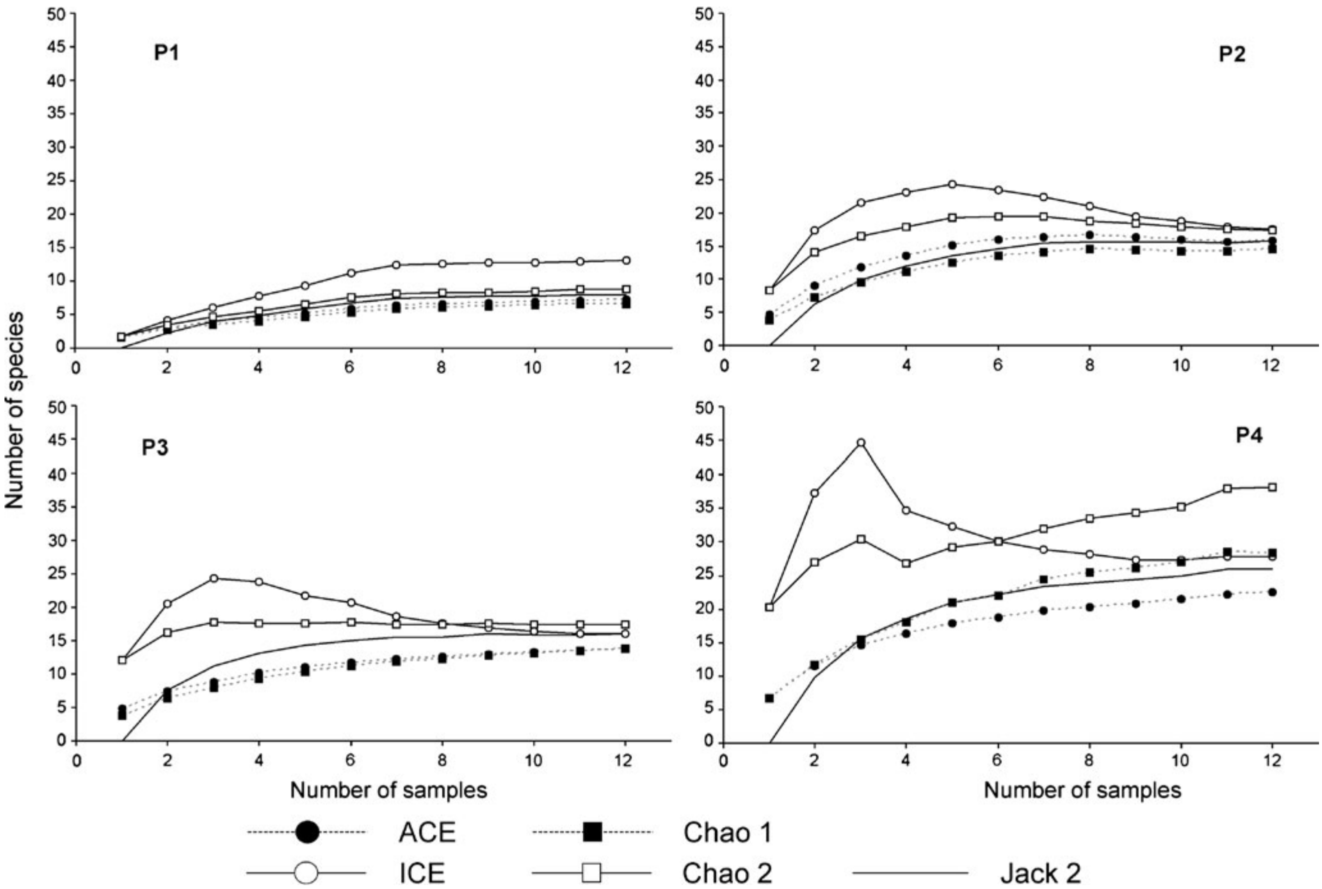

Fig 4 Performance of the five richness estimator indexes used on each pond (P1–4). Abundance-based indexes: Chao 1 and ACE; incidence-based indexes: Chao 2, ICE and Jack 2.

macroinvertebrate taxa (94), ranging from 15 to 49 per pond. Nevertheless, we had to consider that these ponds had a natural origin and were larger than our man-made ponds. The authors reported some differences in macroinvertebrate diversity among pond related to general habitat features and found few relationships between water quality and species composition of the communities. They recorded great macroinvertebrate abundances and diversity in ponds with extensive macrophyte assemblages, and conversely, the ones with no littoral vegetation had the lowest number of insect taxa and lowest overall diversity of macroinvertebrates. Moreover, studies on urban water focused on chironomid assemblages recorded the same three subfamilies observed in our study (Freimuth & Bass 1994, Hamerlik & Brodersen 2010). Chironominae was the most abundant subfamilies in our permanent ponds, in concordance with other studies, suggesting that this subfamily prefers permanent habitats according to their survival strategies, their large body size and huge number of progeny (Bazzanti *et al* 1997, 2003, Hamerlík *et al* 2011).

Table 3 Values of observed species richness (S_{obs}), observed number of shared species among ponds, estimated number of shared species among ponds (Chao shared estimated), and estimated values of different similarity indexes among ponds (*Søren* Sørensen Classic Index, *M–H* Morisita–Horn, *Chao–Søren* Chao–Sørensen, *Est* abundance based).

1st Pond	2nd pond	S_{obs} 1st pond	S_{obs} 2nd pond	Shared S_{obs}	Chao shared estimated	Søren	M–H	Chao–Sørensen
P1	P2	7	14	5	12	0.48	0.52	0.73
P1	P3	7	15	4	4	0.36	0.44	0.86
P1	P4	7	23	5	7	0.33	0.49	0.79
P2	P3	14	15	7	7	0.48	0.55	0.98
P2	P4	14	23	9	13	0.49	0.68	0.87
P3	P4	15	23	10	12	0.53	0.39	0.65

The relative few species recorded may be due to diverse factors, and between them we have to highlight that in the studied ponds the aquatic plants are frequently removed. The aquatic vegetation acts in different ways for aquatic insects, including oviposition site, food source, substrate, refuge to prey species, and ambush site to predator species (Merritt & Cummins 1984, Cremona *et al* 2008). Species richness and abundance of aquatic vegetation influence macroinvertebrates assemblage in temporary and permanent ponds, providing more diversified and suitable habitats, which in turn increase both taxonomic and functional (feeding mechanisms and habits) macroinvertebrates diversity (Bazzanti & Della Bella 2004, Cremona *et al* 2008). The removal of aquatic vegetation from the study ponds may affect both the establishment and permanence of the insect community. Therefore, swimmers were the dominant group, followed by burrowers and sprawlers, only a few strictly climbers were observed. Accordingly, all the ponds were dominated by collectors (principally gatherers), secondarily by predators and only few shredders were detected, which was much affected by the extraction of macrophytes. Other studies which considered the macroinvertebrates distribution within ponds, observed fewer number of macrofaunal species in sediments without vegetation compared with vegetated areas (Della Bella *et al* 2005, Arocena 2007, Bazzanti *et al* 2010), supporting our reasoning. The authors attributed the difference in composition and structure of the macroinvertebrate community to the high habitat heterogeneity provided by aquatic plants.

The presence of fishes also influences insect communities. Juveniles and adults of almost all the species collected in these ponds included insects in their diets (Ringuelet *et al* 1967, Froese & Pauly 2008). Again, the lack of aquatic plants reduces refuge availability for prey species (Peckarsky 1984), increasing the pressure of predation by fish. In a recent study, Vermonden *et al* (2009) concluded that to optimise biodiversity in urban ponds, water management should aim at stimulating vegetation (diversity of habitat structure), lowering nutrient levels and increasing transparency, which are key factors for macroinvertebrate diversity. Moreover, Bazzanti *et al* (2010) confirmed the role of submerged and emergent vegetation in maintaining high biodiversity and suggested that all microhabitats should be considered to provide both an exhaustive collection of species for pond management and conservation and basic insights into the functioning of pond communities.

Regarding the seasonality, to our knowledge there is no study assessing seasonal fluctuation of the richness or abundance of aquatic insects from man-made permanent ponds in temperate South America. Some studies from this region focused on natural permanent ponds found highest values in spring and autumn and the lowest in winter and summer (von Ellenrieder & Fenández 2000, von Ellenrieder & Perez Goodwyn 2000). On the other hand, studies on temporary ponds of the region registered different seasonal patterns mainly associated to hydroperiod records (von Ellenrieder & Fenández 2000, von Ellenrieder & Perez Goodwyn 2000, Fischer *et al* 2000, Fontanarrosa *et al* 2009). In our study, we observed a clear seasonal pattern of richness and abundance but with lowest values during summer and autumn in all the ponds. As hydroperiod has no effect on artificial permanent ponds, our results could be related to human intervention in urbanised areas (e.g. chemical or physical alterations). Further research is needed to investigate the reasons behind observed patterns in our man-made ponds.

Community evenness, sampling intensity, and the true level of species richness indirectly influence the performance of non-parametric estimators via the fraction of all the species found in a sample or sample coverage (Brose *et al* 2003), and therefore, different estimators should be considered. The richness observed in each pond was apparently well estimated compared with the abundance species richness indexes used, and underestimated when it was compared with the incidence indexes. The later estimated more species because there were many more taxa in only one sample than those represented by few individual in a sample. Our investigation was not aimed at evaluating the performance of the aforementioned estimators in the studied environments, but a previous study (Foggo *et al* 2003) indicated that the both Chao and ICE may represent significant improvements over the non-parametric estimators, with ICE in fact proving to be the most reliable and accurate estimator among the tested ones. These results also indicate that counting individuals beyond the abundance of two yields, adds little information if the aim is simply to quantify the maximum species richness in a pond. The main application of these techniques might be an improvement of the accuracy of rapid biodiversity assessments, where the ranking of habitats on observed richness is used to determine conservation priorities and richness estimates could perhaps best be used in conjunction with the identity of the species encountered to improve the quality of such studies (Foggo *et al* 2003).

The assessment of similarity among study ponds produced contradictory results. The classic Sørensen index estimates a mayor coincidence between P3–P4 whereas the quantitative indexes indicated an extreme dissimilarity. Due to the presence of many taxa in one sample (rare species) and the difference in sample effort (proportional to the pond area), we consider that quantitative indexes are the most appropriate to characterise our ponds. The Chao's Sørensen abundance-based index estimated more similarity between P2–P3, ponds with very dissimilar area and depth but similar sloping margins of concrete and granite. The extreme dissimilar ponds (P3–P4) had comparable area but differ in depth and margin structure. Brauns *et al* (2007) demonstrated that, independently of the lake type, the shoreline characteristic (e.g. natural, beaches, retaining walls, or ripraps) affects eulittoral macroinvertebrates richness through the availability of more complex habitat for species establishment. Even more, the authors suggested that artificial enhancement of habitat complexity may offer a promising strategy in urban lakes that are subjected to several types of human shoreline development. A further study is necessary to assess which variables may determine the difference observed among our ponds. It is noteworthy to mention that the ecological value of a single pond may be less important than that of the network or "pondscape" as a whole (Boothby 2000). Gledhill *et al* (2008) in a study carried out on 37 urban ponds reported that while factors such as water chemistry, aquatic vegetation structure, and shade may determine the precise composition of the invertebrate and plant communities within a pond or cluster of ponds, the pond density appears as a greater determinant of species richness. These findings have significance for those involved in planning and managing urban environments, further strengthening the need for functional ecological connectivity in urban areas.

Oertli *et al* (2005) have emphasised that although the value of a pond as a freshwater habitat begins to be acknowledged, studies on these habitats in urban environments are still scarce. These authors stressed that the political recognition of ponds as entities and as important parts of the water environment remains insignificant throughout Europe. On the other hand, enhancement of biodiversity in urban ecosystems can have a positive impact on the quality of life and education of urban dwellers and thus facilitate the preservation of biodiversity in natural ecosystems (Savard *et al* 2000). In South America, studies carried out on urban ponds are not common, particularly those focused on artificial ponds. The present study contributes with some knowledge that can improve the management of these urban habitats. It is widely recognised that macrophytes affect animal assemblages and promote biodiversity due to their role of as physical structures that increase the complexity or heterogeneity of habitats (Thomaz & Ribeiro de Cunha 2010). In this sense, there is no doubt that authorities' actions on these urban ponds have been unfavourable to the aquatic insect community and biodiversity conservation.

Acknowledgments The authors wish to acknowledge the authority of the government of Buenos Aires City, Dirección General de Control de la Calidad Ambiental, for the data of chemical variables provided. This work was supported by Consejo Nacional de Investigaciones Científicas y Tecnológicas (CONICET, PIP 02541).

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