

Summer assessment of zooplankton biodiversity and environmental control in urban waterbodies on the Island of Montréal

EL-AMINE MIMOUNI,^{1,2,3,†} BERNADETTE PINEL-ALLOUL,^{1,2,3} BEATRIX E. BEISNER,^{2,3,4} AND PIERRE LEGENDRE^{1,2,3}

¹*Département de Sciences Biologiques, Université de Montréal, C.P. 6128, Succ. Centre-Ville, Montréal, Québec H3C 3J7 Canada*

²*Groupe de Recherche Interuniversitaire en Limnologie et Environnement Aquatique (GRIL), Université de Montréal, C.P. 6128, Succ. Centre-Ville, Montréal, Québec H3C 3J7 Canada*

³*Centre de la Science de la Biodiversité du Québec (CSBQ), Stewart Biology, McGill University, 1205 Avenue du Docteur-Penfield S3/18, Montréal, Québec H3A 1B1 Canada*

⁴*Department of Biological Sciences, University of Québec at Montréal, C.P. 8888, Succ. Centre-Ville, Montréal, Québec H3C 3P8 Canada*

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Abstract. Waterbodies in urban environments are usually built or maintained to serve socioeconomic functions. However, they also represent ecosystems that can contribute notably to urban biodiversity. To better understand contribution to biodiversity of urban ecosystems, the variation in zooplankton community composition in 19 waterbodies across the Island of Montréal (Québec, Canada) was monitored across three summer months. Communities were dissimilar between and within waterbodies with species richness differences and replacement patterns playing equal parts in shaping the observed variation. Within each waterbody, notable differences were detected between months, which can affect biodiversity estimation or community composition assessment. Zooplankton species richness was especially well explained by macrophyte cover, which had a positive effect. Compositional differences were also explained by macrophyte cover and by waterbody emptying. Partitioning the beta diversity revealed that only richness difference patterns were explained by macrophyte cover, as species replacement patterns were not explained by any of the measured environmental variables.

Key words: biodiversity; conservation management; feeding groups; richness difference; species replacement; urban waterbodies; zooplankton communities.

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† **E-mail:** el-amine.mimouni@umontreal.ca

INTRODUCTION

Although often overlooked historically, small lakes and ponds are now seen as valuable ecosystems providing important services such as water distribution, sedimentation, and carbon cycling (Downing 2010). Furthermore, because they can harbor several rare or unique species (Oertli et al. 2002, Wood et al. 2003, Williams et al. 2004) and are extremely abundant at the global scale (Downing and Duarte 2006, but see Seekell and Pace 2011 and McDonald et al. 2012),

small waterbodies can represent non-negligible sources of biodiversity. Additionally, public perception of urban wetlands is often positive and they are usually considered an asset because they can provide ecological services (Kaplowitz and Kerr 2003, Manuel 2003, Johnson and Pflugh 2008) and favor human health and well-being (White et al. 2010, Völker and Kistemann 2011, 2013). Likewise, evidence suggests that they are also useful ecosystems to advance the knowledge of pond and small lake ecology, as well as for the development of suitable biodiversity

conservation measures in novel habitats of urban regions.

However, the richness and biodiversity of small waterbody communities are at risk (Oertli et al. 2005), especially in urban areas, where aquatic macroinvertebrate and plant communities may be considered to have low ecological values (Noble and Hassall 2014). Small waterbodies are sometimes termed “ponds,” which comprise a class of waterbodies usually defined as being less than 2–5 ha in surface (Biggs et al. 2005). However, such size classes are arbitrary, as the distinction between pond and lake represents a gradient in ecological conditions (De Meester et al. 2005, Søndergaard et al. 2005). The 2–5 ha definition is also hampered by the fact that active legislations to protect waterbodies, such as the Water Framework Directive in Europe, only apply to waterbodies larger than 50 ha (Nicolet et al. 2007, E.P.C.N. 2008, Miracle et al. 2010, Möckel 2013). Consequently, small ponds are inadequately protected because their small size usually excludes them from formal protection plans and their conservation must compete with other priorities, often leading to infilling during land reallocation.

To date, despite many records of important pond losses (Hassall 2014), few assessments of their contribution to urban biodiversity exist. Zooplankton has been strongly advocated as a biological indicator of ecological quality in aquatic ecosystems (Jeppesen 2011). Indeed, it occupies a central position in aquatic food webs, impacting adjacent trophic levels and consequently the overall cycling of materials and energy (McQueen et al. 1986, Ghadouani et al. 2006, Finlay et al. 2007). Zooplankton also affects ecosystem services, including the control of undesirable phytoplankton biomass (e.g., cyanobacterial blooms), leading to cleaner and clearer waters (Peretyatko et al. 2009, Teissier et al. 2011), or acts as a biological tool to assess water quality or contamination by waterborne parasites (Gannon and Stemberger 1978, Walseng et al. 2003, Nowosad et al. 2006). Despite all these advantages, the biodiversity and ecology of urban zooplankton communities remains largely undescribed, and thus, its conservation value is mostly unknown and is most likely underestimated.

In this study, zooplankton biodiversity was examined through the summer growing season across nineteen urban waterbodies located on the

Island of Montréal, Canada. We described zooplankton biodiversity and community composition, focusing on species richness difference and replacement patterns. We identify the major sources of variation, focusing on the temporal changes among summer months in these highly dynamic communities. Finally, we determine the most decisive environmental factors related to community variation. From our results, we suggest conservation and management practices that could be implemented to preserve heterogeneity in zooplankton biodiversity and community composition in urban freshwater ecosystems.

METHODS

Study sites and sampling design

Nineteen waterbodies of various types (including permanent and temporary ponds, small lakes, and three wetlands), distributed over the Island of Montréal (Québec, Canada; 45.46°–45.69° N, 73.50–73.90° W; Fig. 1), were sampled during the last two weeks of June, July, and August 2011. Depth and water transparency were measured using a Secchi disk at three pelagic sampling points, and the results were averaged to obtain a single waterbody estimate. The waterbodies sampled during this study are regularly surveyed by the City of Montréal water-quality monitoring program (Réseau de Suivi du Milieu Aquatique: <http://ville.montreal.qc.ca>). To assess nutrient enrichment in waterbodies, we used total phosphorus concentrations (TP) values estimated as close as possible to our zooplankton sampling dates and averaged them into one summer value.

Zooplankton communities in the pelagic (i.e., center) and the littoral (i.e., edge) habitats were sampled separately by randomly choosing three sampling points in each zone, paired across pelagic and littoral habitats, for a total of six sampling points per waterbody. Zooplankton was sampled from a small anchored inflatable boat using a 3-L bucket dipped ten times at arm's length. The 30 L volume of sample water was filtered through a 54 µm mesh size plankton net. Organisms were narcotized with carbonated water and fixed with ~5 mL of pure formaldehyde (4% final solution) in the field.

Vegetation cover was estimated from visual observations in 2011 and a survey of the

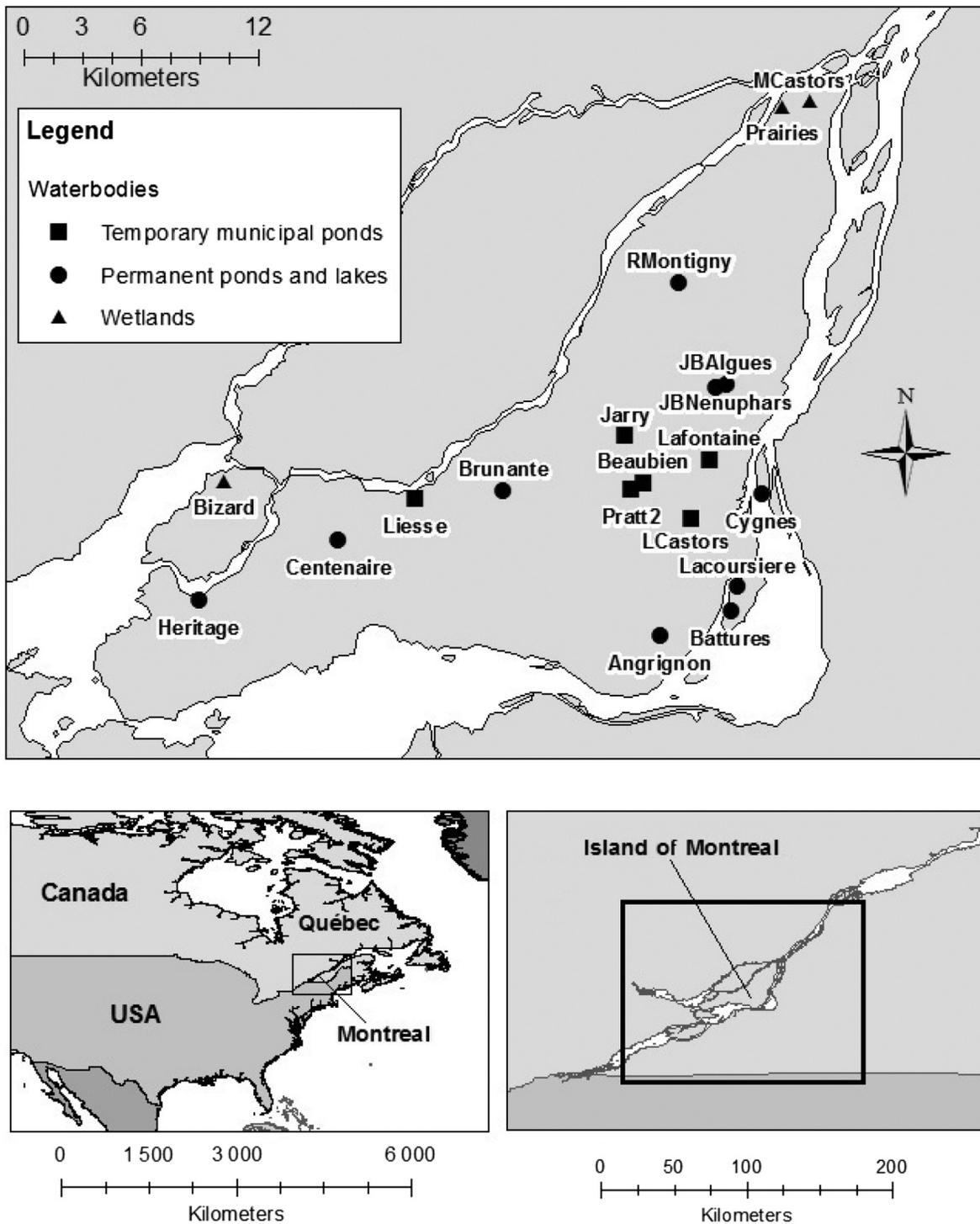


Fig. 1. Geographic location and distribution of the 19 waterbodies sampled during the summer of 2011 on the Island of Montréal (Québec, Canada). Modified from Mimouni et al. (2015).

vegetation dominance in 2012. The presence of fish communities was confirmed by city managers, visual observations, and occasional captures during sampling. Through contacts with city managers, waterbodies that were emptied and cleaned before winter were noted.

Zooplankton taxonomic composition and feeding groups

Zooplankton samples were fixed in formaldehyde for approximately six months and then transferred to a 75% ethanol and 5% glycerol solution. Except when organic matter and detritus were too abundant or when densities were too high, a quarter of each well-mixed zooplankton sampling unit was taken using a large-mouthed pipette and transferred to a Ward counting wheel (Ward 1955). Taxa were sorted and counted under a Leica Wild M3B stereomicroscope and identified with a Wild Heerbrugg microscope to the finest possible taxonomic resolution (genus and species) using taxonomic keys for Rotifera (Edmondson 1960, Stemberger 1979, Nogrady et al. 1996, Haney 2010), Cladocera (Brooks 1959, Hebert 1995, Haney 2010), and Copepoda (Smith and Fernando 1978, Hudson and Lesko 2003). Zooplankton taxa were also classified based on their feeding ecology. Cladoceran feeding groups were established based on foraging mode, as in Declerck et al. (2007) and Barnett et al. (2007). Cladocera feeding on periphyton or detritus on substrates or macrophytes were considered substrate-grazing cladocerans (SGCladocera). Cladoceran taxa that actively filter the water column for phytoplankton or particulate organic matter were considered seston-filtering cladocerans (SFCladocera). The cladoceran taxa *Polyphemus pediculus* and *Leptodora kindtii*, which are predatory and obtain their food differently from other cladoceran taxa (Young and Taylor 1988, Browman et al. 1989), formed a separate feeding group of carnivorous cladocerans (CCLadocera). Copepods were separated into raptorial cyclopoids (RCopepoda) and stationary and suspension-feeding calanoids (SSFCopepoda) as per Barnett et al. (2007). Rotifers were assigned to feeding groups (Obertegger and Manca 2011) as either microphagous taxa (MRotifera), which simultaneously collect a variety of food items, or raptorial taxa (RRotifera), which actively grasp, pierce, or pump to catch prey.

Zooplankton diversity patterns

Beta diversity of zooplankton in each waterbody was expressed as the total variance of the community species composition table (Pélissier et al. 2003, Legendre et al. 2005, Anderson et al. 2006). The total variance was calculated from a matrix of Jaccard (1908) dissimilarities, as it has several desirable properties and is often used in beta diversity studies (Whittaker 1972, Qian and Ricklefs 2007, Tuomisto 2010, Podani and Schmera 2011, Legendre and De Cáceres 2013). The Jaccard dissimilarity matrix was then partitioned in two components, namely species richness differences ($Rich_{Jac}$) and species replacements ($Repl_{Jac}$) (Carvalho et al. 2011, 2012, Podani and Schmera 2011). To simultaneously evaluate the importance of species richness and replacement for all pairs of waterbodies, triangular graphs (simplices) were used to represent the pairwise indices of the $S_{Jac} = (1 - D_{Jac})$, $Rich_{Jac}$, and $Repl_{Jac}$ triplets, as suggested by Podani and Schmera (2011) and Podani et al. (2013). These plots were computed for each month as well as for the whole summer (all monthly data combined).

Variation of zooplankton community composition and feeding groups

MANOVA was used to investigate the effects of the main factors (site, zone, and month) and their interactions on community composition and feeding group's dominance. MANOVAs were computed using the principal coordinate axes of a principal coordinate analysis (PCoA; Gower 1966) of the square-rooted Jaccard dissimilarity matrix to test for differences in species composition (presence-absence data), and of the Hellinger-transformed (Legendre and Gallagher 2001) feeding group abundance matrix to test for differences in dominance. Owing to a significant three-way interaction among the factors ($p\text{-perm} < 0.05$) for community composition and two out of the three two-way interactions being significant for feeding group abundances ($p\text{-perm} < 0.05$), we conducted a two-way MANOVA for each site separately, with factors month (June, July, or August) and waterbody zone (pelagic or littoral) coded as Helmert contrasts. One sampling unit of pond *Battures* was lost during sorting, and this waterbody was thus excluded from these analyses to preserve the balance of the groups in the analyses. To account for multiple testing, the P -values of the

MANOVAs obtained using the full model (i.e., all variables in the model matrix) were adjusted using Holm's (1979) correction for multiple tests. Where significant MANOVA P -values were obtained, the importance of the temporal variation was further assessed using variation partitioning (Borcard et al. 1992, Borcard and Legendre 1994, Peres-Neto et al. 2006) by partitioning between the factors month and zone.

To determine how many feeding group patterns exist in urban waterbodies, k -means clustering was used on the Hellinger-transformed feeding group abundances. Three groups were identified using the Caliński and Harabasz (1974; $k = 3$) and interpreted using distance principal components analysis (PCA) biplots of the zooplankton communities constrained on the k -means groups.

Relationships between zooplankton communities and environmental features

To examine relationships between environmental variables and the zooplankton communities, multiple regression models were used. Environmental variables were individually transformed to reduce skewness: Mean depth (m), surface (m^2), and total phosphorus concentrations ($\mu\text{g/L}$) were \log_e -transformed, and macrophyte cover (%) was square-root-transformed. Emptying of waterbodies by city employees before winter and the presence of fish were coded as Helmert contrasts. To exclude temporal variation, all species observed across all samples collected during the summer in each waterbody were considered.

Multiple regressions between the species richness of the various zooplankton taxonomic assemblages and feeding groups with the environmental variables were done using forward selection (Blanchet et al. 2008), and standardized regression coefficients are reported. To account for multiple testing, the p -values of the linear models were adjusted using Holm's (1979) correction for multiple tests.

Distance-based redundancy analysis (db-RDA; Legendre and Anderson 1999, McArdle and Anderson 2001) was applied to each beta diversity component dissimilarity matrix (D_{Jac} , Rich_{Jac} , Repl_{Jac}) to examine community composition. First, an exact significance test was computed for each matrix and the environmental variables using permutations of the raw data (McArdle and

Anderson 2001) with the environmental variables identified using forward selection (Blanchet et al. 2008) on the eigenvectors of a PCoA of transformation of the Jaccard-based matrices. For these analyses, the Jaccard-based matrices were transformed to make them Euclidean, as db-RDA works best when the dissimilarity is Euclidean (Legendre and Anderson 1999, Legendre 2014). For the case of D_{Jac} , taking the square root of the dissimilarity was enough to make it Euclidean (Gower and Legendre 1986). The square roots of the Rich_{Jac} and Repl_{Jac} dissimilarity matrices were taken, as well as Lingoes' (1971) correction for negative eigenvalues, and the validity of these subsets was confirmed by carrying out the exact test on the Jaccard-based indices using only the subset of environmental variables on the non-transformed dissimilarity matrix.

For graphical outputs, biplots of db-RDA based on the transformed Jaccard-based dissimilarity matrices were performed, with the significance of canonical axes tested using the forward testing method of axes (ter Braak 1990, Legendre et al. 2010). Taxa that were not well explained by the model were not included in the biplot based on an approximate R^2 statistic (aR^2) calculated on the fitted and the residual sums of squares for each taxon (*vegan* function *capscale()*; see also Legendre and Legendre 2012). Only taxa with an $aR^2 \geq 33\%$ were considered and plotted. Using the significant environmental variables, variation partitioning was used to assess the relative importance of these variables. Significant differences between the fractions attributable to the retained variables were tested by using the bootstrap method of Peres-Neto et al. (2006) as implemented in the MATLAB program VARCAN (Peres-Neto 2006).

All other analyses were done in R 2.15.2 (R Development Core Team 2012) using the packages MASS (Venables and Ripley 2002), APE (Paradis et al. 2004), ade4 (Dray and Dufour 2007), rdaTest (Legendre and Durand 2010), vegan (Oksanen et al. 2012), and adespatial (Dray et al. 2016).

RESULTS

Waterbody environmental characteristics

The studied waterbodies covered a broad range of morphometric and environmental characteristics (Table 1; see also Appendix S1 for

Table 1. General characteristics of environmental and morphometric variables for the 19 urban waterbodies (WB) sampled.

Quantitative variables	Mean	Standard deviation	Minimum	Median	Maximum
Surface (m ²)	27,592	34,332	392	11,395	114,467
Depth (m)	1.9	2.0	0.2	1.4	9.4
Secchi depth/depth	0.2	0.3	0.7	0.8	1.0
Total phosphorus (µg/L)	64.4	68.8	9.5	34.7	265.3
Macrophyte cover (%)	44.5	37.4	0	40	100
Binary variables		Yes		No	
WB emptied and cleaned		5		14	
Fish presence		16		3	

environmental conditions in each waterbody). Surface area was variable, and even though most waterbodies had an area of less than 2 ha, putting them in the category of ponds, some waterbodies had a higher surface area, and thus could be considered small lakes. Waterbodies were shallow (most <2 m deep) with one exception (*Cygnés*), which had a deep pelagic zone (>9 m). Secchi depth was variable, and in some shallow ponds, light reached the sediments even in the open water zone. There were three groups of waterbodies based on the ratio of Secchi depth/waterbody depth from 0 to 0.5 (six waterbodies), between 0.5 and 0.8 (four waterbodies), and between 0.8 and 1.0 (nine waterbodies). Total phosphorus concentrations ranged from oligo-mesotrophic (9.5 µg/L) to hypereutrophic (265.3 µg/L). Macrophyte cover also varied from none (bare concrete ponds without vegetation) to wetlands almost completely covered with dense vegetation. Fourteen waterbodies had aquatic vegetation in the littoral zone including mainly emergent (*Phragmites*, *Typha*, *Scirpus*, *Lythrum*, *Equisetum*, *Sparganium*, *Pontederia*, *Butomus*, *Alisma*), floating (*Nymphaea*, *Nymphoides*, *Lemna*, *Wolffia*), and submerged (*Potamogeton*, *Ceratophyllum*, *Anacharis* (*Elodea*), *Utricularia*, *Myriophyllum*, *Valisneria*, *Najas*) plants and some Characeae algae (*Nitella flexilis*, *Chara vulgaris*; see Appendix S2). Five waterbodies were regularly emptied and cleaned before winter. The presence of fish communities was confirmed in 16 waterbodies.

Zooplankton composition and diversity patterns

A total of 90 zooplankton taxa (Appendix S3) were recorded across all sites and time periods. Of these taxa, 60 were rotifers, 24 were

cladocerans, and 6 were copepods (3 cyclopoids and 3 calanoids). Across all months, species richness of each waterbody was quite variable. The difference between the richest (53 taxa) and the poorest site (12 taxa) was approximately four-fold, and sites contained on average 32 taxa (Appendix S3). Monthly estimates of waterbody species richness underestimated total species richness, containing only around two-thirds of all recorded species. These values ranged from 47% to 88% of total species richness in June, from 32% to 84% in July, and from 28% to 83% in August. Taxa also had quite heterogeneous occurrence patterns, with isolated occurrences of certain taxa. Indeed, 12 taxa were present at only three sites, 12 others were present at only two sites, and 12 other taxa were present at a single site (Appendix S3). On the other hand, some taxa were observed at almost every site: *Keratella* sp. was found in all 19 waterbodies; *Chydorus* sp. and *Lecane* (*Monostyla*) sp. in 18 waterbodies; Bosminidae spp. and *Polyarthra* spp. in 17 waterbodies; and *Euchlanis* spp. in 16 waterbodies (Appendix S3).

Most of the waterbodies showed a diversity of feeding groups: Of the seven possible feeding group types, 12 waterbodies had six types represented and only one waterbody had all seven types. Two waterbodies had five feeding groups, three had four, and one had three. In every waterbody, several taxa made up each feeding group. The feeding group composed of the largest number of representative taxa was microphagous rotifers (39 taxa), followed by raptorial rotifers (21), substrate-grazing cladocerans (13), seston-filtering cladocerans (9), raptorial cyclopoids (3), stationary and suspension-feeding calanoids (3), and carnivorous cladocerans (2).

Table 2. Mean values of the Rich_{Jac} , Repl_{Jac} , and S_{Jac} components of beta diversity and matrix fill of zooplankton communities in the 19 sampled waterbodies for each month and for the summer period together.

Month	$100 \times \overline{\text{Rich}}_{\text{Jac}}$	$100 \times \overline{\text{Repl}}_{\text{Jac}}$	$100 \times \overline{\text{S}}_{\text{Jac}}$	$100 \times \overline{\beta}_{\text{Jac}}$	M_{Fill}
June	32.29	40.12	27.59	72.41	25.56
July	37.63	34.09	28.28	71.72	24.27
August	41.01	35.21	23.77	76.23	22.51
Summer	33.84	30.93	35.22	64.78	35.61

Note: Values have been multiplied by 100.

Communities had more or less equal portions of their average Jaccard dissimilarity accounted for by richness differences and replacement, with both proportions being close to one-half of the average Jaccard dissimilarity (Table 2). Likewise, the triplets of summer averages for the three statistics ($\overline{\text{Repl}}_{\text{Jac}}$, $\overline{\text{Rich}}_{\text{Jac}}$, $\overline{\text{S}}_{\text{Jac}}$) were close to $(\frac{1}{3}, \frac{1}{3}, \frac{1}{3})$ (Table 2 and Fig. 2). This is consistent with the fact that the means for each month were fairly close to the simplex centroid. These patterns were observed for each individual month. That said, the distribution of points did show some structure within the simplex. Most site pairs fell into the left and upper portions of the simplex plot for each month, indicating low similarity, but with varying influences of richness differences and replacements of taxa. Over time, the values of Rich_{Jac} seemed to increase (going from 0.32 to 0.41) whereas those of Repl_{Jac} decreased (going from 0.40 to 0.35), while values of S_{Jac} stayed the same (around 0.24–0.28). This may indicate that, over time, differences in waterbodies were more due to richness differences than to species replacements. However, the scatter of points was high, especially in August. In addition, the importance of the number of pairs of sites showing complete nestedness (points that fall on the $\text{Rich}_{\text{Jac}}\text{--}\text{S}_{\text{Jac}}$ segment) increased through time (0 in June, 1 in July, 7 in August).

Sources of variation in zooplankton communities and feeding groups

For community composition variation, MANOVA models for all waterbodies, except *Centenaire*, were significant even after Holm's correction (Table 3). For variation in feeding group abundances, three waterbodies (*Heritage*, *Cygnés*, and *RMontigny*) were not significant after Holm's correction (Table 3). When considering community composition, thirteen waterbodies showed differences between months, four

showed differences between zones, four showed a significant Month*Zone interaction, and MANOVA could not be considered for one waterbody (*Centenaire*) due to the absence of homogeneity of variance–covariance matrices. When considering the feeding groups, fifteen waterbodies showed significant differences between months, five waterbodies showed significant differences between zones, no waterbody showed a significant Month*Zone interaction, and MANOVA could not be considered for three waterbodies (*Heritage*, *Cygnés*, and *RMontigny*) due to the non-significance of the model p-value after Holm's correction.

The polygons for each group of sampling units were well defined in the case of species feeding groups (Fig. 3). The constrained PCA biplot showed three well-defined groups, with each having one or two defining feeding groups. The first axis mainly distinguished between groups that had elevated rotifer dominance (groups 1 and 2), and group 3 which had greater cladoceran dominance. Further differences in feeding groups were depicted along the second axis. Group 1 had higher relative abundance of microphagous rotifers (MRotifera), whereas group 2 had more raptorial rotifers (RRotifera). Group 3 was dominated by seston-filtering cladocerans (SFCladocera), with some substrate-grazers (SGCladocera). The dominant taxa in group 1 were (in ascending order of dominance) *Keratella* sp., *Keratella tecta*, *Brachionus angularis*, *Lecane* sp., *Platyonus patulus*, *Lecane (Monostyla)* sp., *Lepadella patella*, *Lecane (Monostyla) quadridentata*, and *Euchlanis* spp. The dominant taxa in group 2 were *Polyarthra* spp. and *Ploesoma* sp. The dominant taxa in group 3 were *Ceriodaphnia* sp., *Bosminidae* spp., *Diaphanosoma* sp., *Scapholeberis* sp., *Simocephalus* sp., and *Daphnia* spp. (SFCladocera), as well as *Chydorus* spp. and *Alona* spp. (SGCladocera).

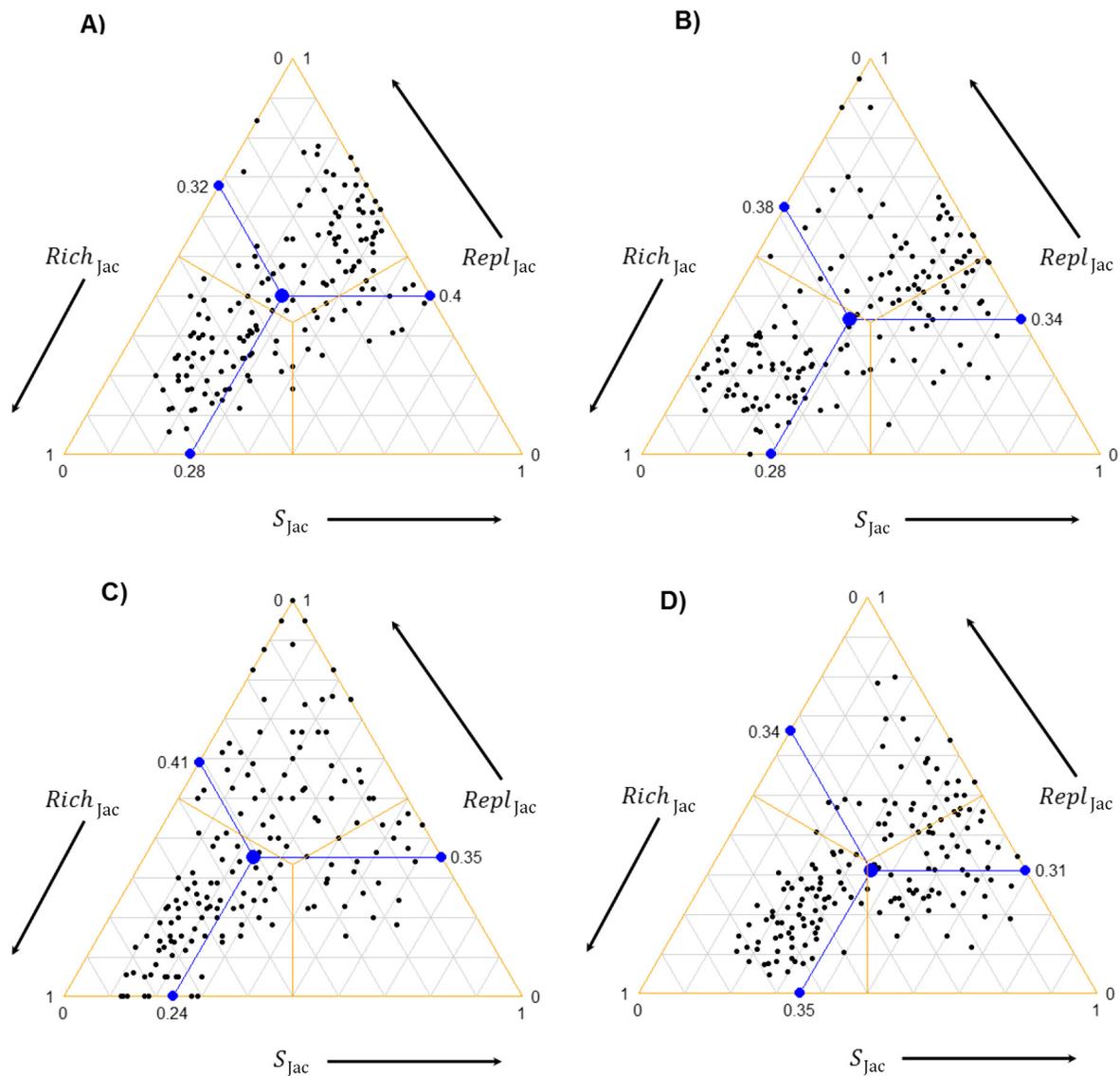


Fig. 2. Simplices showing community dynamics relationships between the similarity, richness differences, and species replacement axes. The panels show these patterns for zooplankton communities for the months of June (A), July (B), and August (C) and for all months combined (D). Each individual black point represents a pair of waterbodies whose triplet of values must sum up to 1. Blue dots on the edges represent average values for the three additive components of diversity: richness differences ($Rich_{Jac}$), species replacement ($Repl_{Jac}$), and Jaccard similarity (S_{Jac}), with its corresponding position of this triplet within the simplex added.

Relationships between zooplankton communities and environmental features

Significant relationships between zooplankton species richness and the environmental variables were observed for two taxonomic groups and one feeding group (Table 4). Environment–richness relationships were significant for entire

zooplankton, rotifers, and raptorial rotifers, which all responded positively only to macrophyte cover (model P -value < 0.05 in all cases).

Zooplankton biodiversity was also affected by environmental variables, with selected variables related to particular biodiversity component (D_{Jac} , $Rich_{Jac}$ or $Repl_{Jac}$). Macrophyte cover and

Table 3. Fractions of variation accounted for by the factors month and zone, as well as their possible two-way interaction on the Jaccard (presence–absence) and Hellinger-transformed (abundance) community data in each waterbody.

Waterbody	Species composition Square-rooted Jaccard dissimilarity			Feeding groups Hellinger distance		
	Month	Zone	Interaction	Month	Zone	Interaction
Pratt2	*	*	9.24	81.57	–	–
Beaubien	32.53	–	–	25.19	–	–
Heritage	*	*	11.96	NA	NA	NA
Lafontaine	32.70	–	–	86.74	–	–
Centenaire	NA	NA	NA	72.51	–	–
LCastors	28.70	–	–	59.11	–	–
Angrignon	*	*	14.65	41.57	13.27	–
Jarry	33.47	–	–	69.95	–	–
Cygnés	19.34	9.19	–	NA	NA	NA
Brunante	28.83	7.30	–	70.25	–	–
Bizard	22.33	–	–	63.91	6.01	–
Liesse	*	*	15.43	48.57	–	–
Lacoursiere	18.27	10.48	–	29.48	13.31	–
JBNenuphars	25.23	–	–	67.42	11.34	–
JBAlgues	23.09	–	–	52.87	–	–
RMontigny	28.05	–	–	NA	NA	NA
Prairies	24.55	–	–	54.44	–	–
MCastors	23.38	8.27	–	62.49	6.88	–

Notes: Only significant fractions are reported (non-significant fractions are denoted by “–,” fractions that cannot be estimated appropriately due to a significant interaction are denoted by “*,” and combinations that cannot be tested due to absence of homogeneity of variance–covariance matrices or to the non-significance of the model 0050-value after Holm’s correction are denoted by “NA”).

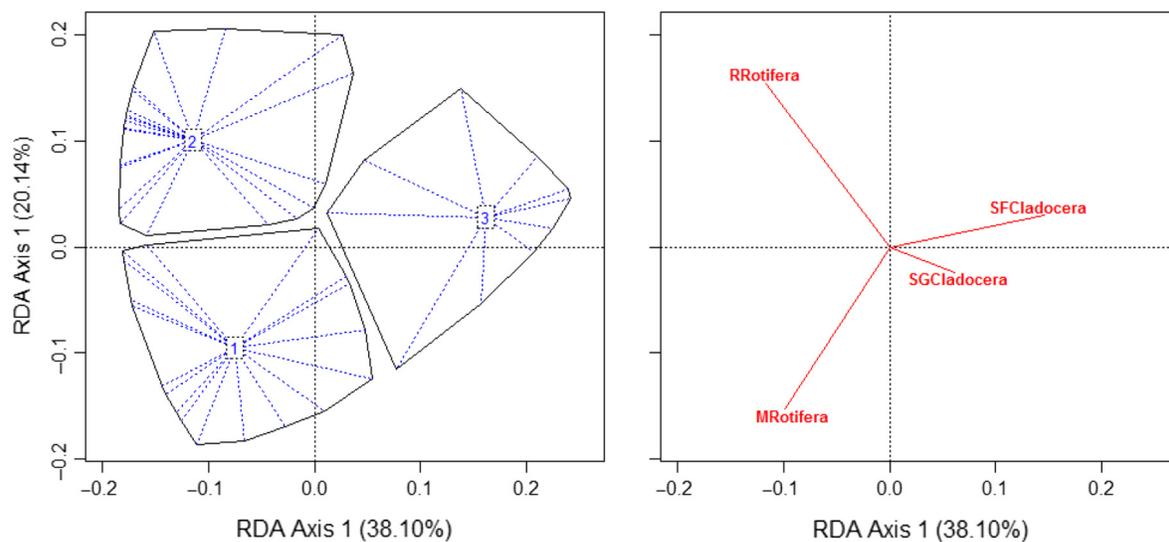


Fig. 3. Distance biplot based on the ordination of the zooplankton communities constrained by the groups, as determined by k -means clustering. To improve legibility, the vectors for the feeding groups CCladocera, RCopepoda, and SSFCopepoda, which had small loadings on the canonical axes, were not drawn.

Table 4. Significant variables, standardized regression coefficients (b), and adjusted R^2 (R^2_{adj}) based on multiple regressions using either all the variables or only variables retained after forward selection, for the different zooplankton taxonomic groups and feeding groups.

Assemblage	b_{Area}	b_{Depth}	b_{Phos}	b_{Macr}	b_{Empty}	b_{Fish}	R^2_{adj}
Zooplankton	–	0.35	–0.30	0.76	–	–	68.83
Rotifera	–	–	–	0.73	–	–	58.44
Cladocera	–	–0.65	–0.36	–	–0.62	–	63.06
RRotifera	0.33	–	–	0.72	–	–	59.58

Notes: Only significant models are reported. Area: surface (m^2); S_{Depth} : ratio between mean depth (m) and Secchi depth (m); TP: total phosphorus ($\mu g/L$); Macr: macrophyte cover (%).

emptying were retained for D_{Jac} (p-perm < 0.05; $R^2_{adj} = 15.53\%$) and macrophyte cover for $Rich_{Jac}$ (p-perm < 0.05; $R^2_{adj} = 40.65\%$). No variables were retained for $Repl_{Jac}$ (p-perm > 0.05).

Only the first canonical axis was significant for D_{Jac} (p-perm < 0.05). The first axis mainly differentiated communities along a gradient of macrophyte cover and waterbody emptying (Fig. 4). Indeed, negative scores were associated with low macrophyte cover and waterbody emptying, whereas high values were associated with high

macrophyte cover and waterbody permanence. Twenty-nine taxa over the 90 taxa were well explained by the selected environmental variables (a R^2 value $\geq 33\%$), and the majority (25) of these were positively related to the macrophyte cover gradient showing strong positive correlations with the first axis. Regularly emptied waterbodies before winter showed impoverished communities with fewer taxa than undisturbed waterbodies. For $Rich_{Jac}$, the first and only canonical axis was significant (p-perm < 0.05), though only a single canonical axis was considered. This first axis was positively correlated with macrophyte cover (Fig. 5). Relative to the first RDA, twice that number of taxa (50 taxa) were well explained. Most of these taxa were microphagous rotifers (*Lecane (Monostyla) sp.*, *Platonus patulus*, *Lecane sp.*, *Lepadella patella*) as well as seston-filtering cladocerans (*Diaphanosoma sp.*, *Simocephalus sp.*) and substrate-grazing cladocerans (*Chydorus spp.*, *Alona spp.*). Only the rotifers *Brachionus sp.* and *Keratella tecta* showed negative correlations with the first axis (see Appendix S3 for species names).

Variation partitioning showed that an appreciable amount of the variation in zooplankton richness was attributable to macrophyte cover, which uniquely accounted for 63.34% of the total explained variation, but also by waterbody area and total phosphorus, which respectively accounted for 11.99% and 8.45% of the variation (Fig. 6A). All three fractions were significant, but were not significantly different from each other (P -value > 0.05, 9999 bootstrap samples). For Jaccard community composition, macrophyte cover (6.72%), followed by waterbody emptying (3.23%), explained the most variation (Fig. 6B) with both individual fractions significant (p-perm < 0.05), but not from each other (P -value

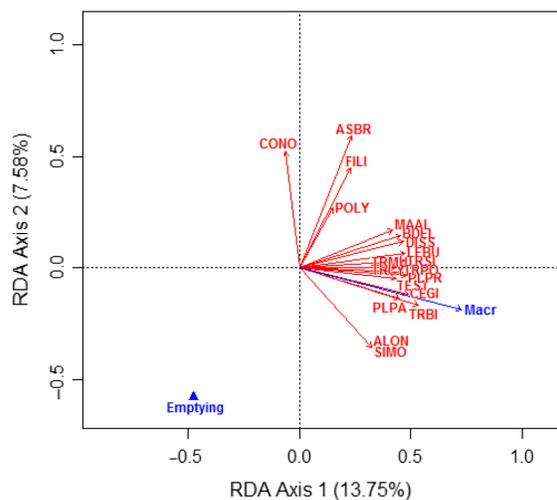


Fig. 4. RDA correlation biplot showing the relationship between zooplankton communities and the environmental variables for the model explaining the D_{Jac} component of diversity. The values associated with each canonical axis are percentages of (explained) variation in the D_{Jac} component of diversity. In order to retain only important taxa and improve legibility, only taxa that were well explained by the analysis (taxa for which a $R^2 \geq 30\%$) are represented. See Appendix S1 for the full names of the taxa.

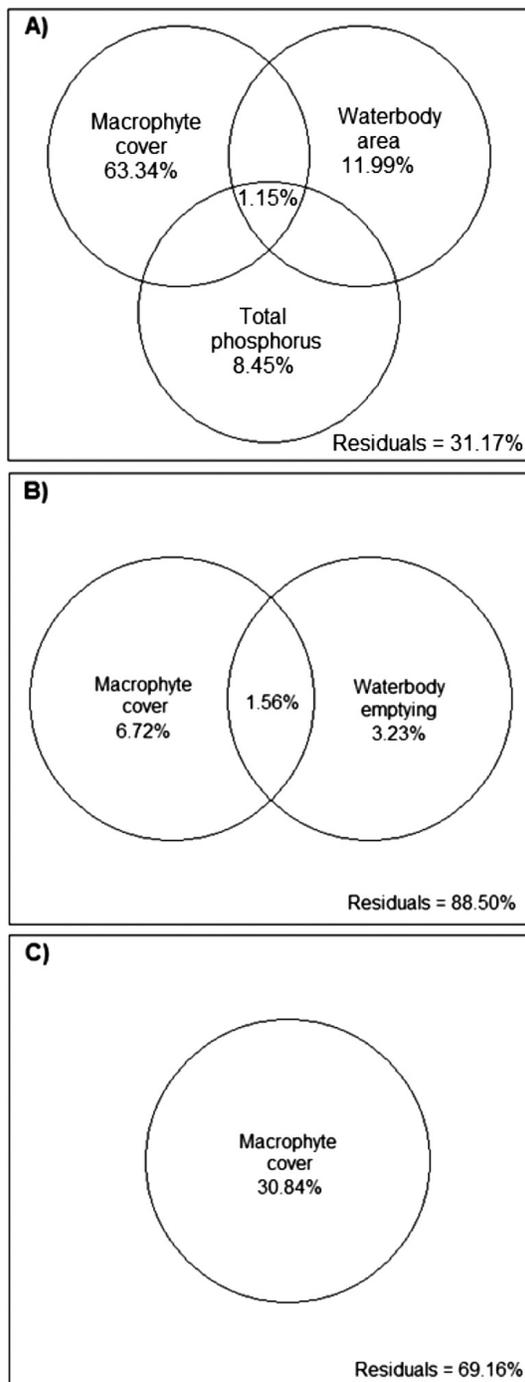


Fig. 6. Venn diagram showing the partitioning of the variation in zooplankton species richness (A), the Jaccard dissimilarity (B), and the richness difference components (C) between the environmental variables retained by forward selection. The values reported are adjusted coefficients of multiple determinations (R^2_{adj}).

priority effects (Scheffer et al. 2006). However, we note that, through time, the scatter around the A-simplex increased, thus weakening the initially strong structure detected in June. Both species replacement and similarity values fluctuated with no clear pattern. The only value that consistently increased with time was richness differences. This increase indicated that, by August, richness differences play an important role in shaping zooplankton communities across the landscape.

An important aspect of biodiversity conservation is that the type of biodiversity to be conserved should be made clear from the start (Game et al. 2013). We can identify three possible endpoints that conservation strategies should maximize in urban waterbodies, which can also be considered three levels of biodiversity: (1) local within-site species richness (α diversity), (2) between-site (or β) diversity, and (3) regional richness (γ diversity). Different practices and prioritization schemes can be put into place depending on which of these aims is chosen.

1. If within-site zooplankton richness is to be maximized, then we can identify from our results a key variable, macrophyte cover, which positively affected within-site richness and can be acted upon by conservation programs. Increasing macrophyte cover would increase species richness within each waterbody, but would not guarantee which species fill these spots.
2. If between-site diversity is to be maximized, then macrophyte cover might also be focused on, but its effect might be less important than on local richness. Indeed, maximizing between-site differences is complicated by the fact that differences between sites lie along a gradient between richness differences and replacement patterns (Podani and Schmera 2011, Podani et al. 2013, Legendre 2014). Therefore, one could maximize among-site differences by pushing toward a richness difference gradient or toward a replacement gradient, and arguments for either of these choices could be made.
3. If regional diversity is to be maximized, then perhaps combinations of sites might be sought, rather than acting on a few critical variables. Several studies have suggested that

conservation policies aimed at preserving regional diversity should consider all waterbody size ranges (Oertli et al. 2002), flow, size, and permanence regimes (De Bie et al. 2010), as well as early and late successional stages of ponds on the landscape (Hassall 2014).

Variation of zooplankton communities and feeding groups

Despite being ecologically distinct from larger lakes (Oertli et al. 2002, Søndergaard et al. 2005, Scheffer et al. 2006, Meerhoff and Jeppesen 2009), small waterbodies should still show some level of seasonal and environmental predictability, to which zooplankton communities could respond, partially determining their succession through time. There were two lines of evidence for temporal variability in the studied waterbodies. Firstly, single monthly surveys tended to considerably underestimate species richness. Secondly, significant differences in community composition were detected between months for several waterbodies for both community composition and feeding group abundances. For waterbodies like *Pratt2*, these changes could be associated with the fact that the waterbody was emptied and completely cleaned (by manually removing all organic matter) during the sampling period, between June and July, as opposed to before winter like the other waterbodies that were subjected to this practice. However, other waterbodies also had important fractions of variation attributable to the month factor (e.g., *Centenaire*, *Jarry*, and *Heritage*), indicating that communities naturally changed over the summer months, without the influence of obvious anthropic stressors. Various factors such as changing environmental conditions or differences in hatching phenologies among the species could be at cause. This result is somewhat similar to that of Burdíková et al. (2012), who also found strong seasonal differences in testate amoebae communities in Czech urban ponds. However, their study considered protists, which, while considered to be microzooplankton, are an extremely dynamic component owing to their smaller size and asexual reproduction, operating even faster than other asexual groups such as rotifers and cladocerans. We also report that similar processes occur for larger fractions of zooplankton including micro-, meso-, and macrozooplankton communities.

The within-waterbody spatial zone factor had a smaller effect than others, and this is not surprising for a number of reasons. First, most of the waterbodies studied would be classified as ponds (Oertli et al. 2005), having surface areas <2 ha and maximum depths <8 m, so that macrophytes can potentially colonize the entire area. Thus, because ponds are essentially entirely a littoral environment, differentiation of littoral and pelagic zones is not as clearly defined as it would be in larger lakes. Second, to reduce sampling bias, the pelagic and littoral zones were sampled in the same way. However, greater differences could have been detected had we used a more appropriate sampling method for the littoral zones (e.g., scrubbing plants or rocks, collecting sediment). Indeed, studies have shown that rotifer species richness in littoral habitats and psammon habitats of Polish lakes does not saturate even with a very large number of samples taken using a 5- or 1-L sampler in the open water and aquatic plants (Muirhead and MacIsaac 2006). Finally, it could be that the influence of the littoral zone for structuring zooplankton communities has been overemphasized in some cases. The relative importance of habitat structure and macrophyte cover versus other environmental variables in shaping zooplankton communities is still not yet fully known, and some studies have found that environmental variables such as pH can have a stronger influence than habitat structure (Kenlan et al. 1984, Walseng et al. 2008).

Similar dominance patterns in species composition over the summer months may hide shifts in community functional traits. As zooplankton species vary considerably in their ecological traits, the analysis of these can reveal insights into ecosystem processes (Barnett et al. 2007, Obertegger et al. 2010, Obertegger and Manca 2011, Litchman et al. 2013, Hébert et al. 2015). Changes in species abundances could be non-random and the result of the presence of a strong selection for species composition based on the ecological traits that are favored in urban environments. Using feeding groups, we found no selection for a single one across all the studied factors. Rather, communities shifted between the various feeding groups through time, although certain feeding groups, such as the carnivorous cladocerans and the copepods, were consistently

less abundant than others, as is also the case in lakes (Pinto-Coelho et al. 2005).

Most assemblages were dominated by a single feeding group consisting of one of raptorial rotifers, microphagous rotifers, or seston-filtering cladocerans (the latter included a moderate amount of substrate-grazing cladocerans), and with the main axis of differentiation occurring between either cladoceran or rotifer dominance. Such an inverse relationship between cladoceran and rotifer abundances has been reported from other aquatic ecosystems such as large lakes, shallow lakes, and river backwaters (Adalsteinsson 1979, Gilbert 1988, Lampert and Rothhaupt 1991) and is likely a result of competitive interactions between them (Gilbert and Stemberger 1985, MacIsaac and Gilbert 1989, 1991). Furthermore, an additional distinction can be made between assemblages dominated by raptorial versus microphagous rotifers, which formed two well-defined clusters. Such differences in rotifer communities can be associated with perturbations such as changes in trophic state (Oberegger and Manca 2011, Spoljar 2013). Thus, urban waterbodies display monthly shifts between zooplankton feeding groups, likely reflecting biotic interactions and possibly changes in trophic state, rather than only environmental filtering selecting for feeding group dominance.

Relationships between zooplankton community and environmental features

Zooplankton species richness was affected by the environmental conditions of the waterbodies, with slightly different responses depending on which taxonomic or feeding group was considered. However, one variable that was consistently positively related to the richness of all groups which had significant models was macrophyte cover. In a study of shallow waterbodies across Europe, Declerck (2005) also found that macrophyte cover was an important variable for the species richness of several aquatic organisms (bacteria, ciliates, phytoplankton, zooplankton, fish, macroinvertebrates, and water plants). Likewise, Noble and Hassall (2014) associated the poor ecological values of aquatic macroinvertebrate and plant communities of urban waterbodies with poor management of macrophyte communities. These results are consistent with

the notion that, for shallow lakes and ponds, macrophytes are a key element for increasing invertebrate diversity (Scheffer et al. 2006).

Studies have reported relationships between zooplankton species richness and morphometric variables such as depth (Keller and Conlon 1994) or waterbody surface area (Dodson 1992, Allen 1999, Dodson et al. 2000). However, mean depth contributed significantly only to total zooplankton species richness and waterbody area only to raptorial rotifer species richness. For total zooplankton, the scaled regression coefficients for waterbody depth were appreciably large, which is in line with studies that have reported waterbodies as being associated with greater species richness (Keller and Conlon 1994, Cottenie and De Meester 2003). Consequently, waterbody depth is as important regarding species richness as macrophyte cover in shallow lakes (Scheffer 1998).

Waterbody surface area was a significant variable only for raptorial rotifer species richness. The fact that this result was not more widespread, influencing other taxa, was unexpected for a number of reasons: Several studies have found more diverse communities in larger lakes (Fryer 1985); others describe multiple regression models for richness that include waterbody area (Allen 1999, Dodson et al. 2000, De Meester et al. 2005); and others have focused on the relationship between richness and area (Browne 1981, Dodson 1991, 1992). The absence of significant relationships in our study could be due to a stronger relationship between species richness and habitat diversity, as offered by macrophytes. Indeed, waterbodies with more macrophyte cover would offer a higher diversity of habitats for zooplankton communities. Consequently, if small urban waterbodies are to be considered as islands, then it appears that macrophyte cover is a key variable.

However, species richness offers only a partial view of the processes at play regarding beta diversity, as dissimilarities between communities are the result of both species replacement and richness differences (Harrison et al. 1992, Williams 1996, Lennon et al. 2001, Legendre 2014). Urban waterbodies seem to show equal amounts of both processes, making the research of the factors affecting each type of patterns especially meaningful. Theoretically, macrophytes could act on either of these processes either by offering a

structurally complex habitat with a higher number of ecological niches or by acting as an ecological gradient (e.g., of food availability, predator visibility) that would act as a filter for taxa. However, we find that macrophyte effects were observable solely on richness difference patterns and not on species replacement patterns, which could not be attributed to any of the measured environmental variables. These results are consistent with the fact that macrophytes have often been considered a structural component of aquatic ecosystems, affecting habitat complexity (Thomaz and da Cunha 2010, Kovalenko et al. 2011, Bolduc et al. 2016), allowing for the establishment of a more diverse community, rather than representing an environmental filter or gradient with the potential to affect species replacement. Overall, while the measured environmental variables may enable prediction of richness differences, they are unable to determine which taxa actually fill these spots. To obtain such knowledge, further studies focusing on the species replacement component and an expanded set of explanatory variables are necessary.

CONCLUSION

Although urban waterbodies may be common in some urban landscapes, only recently have they begun to receive scientific attention regarding their biodiversity potential (Hassall 2014). We are only beginning to understand species distribution patterns and ecological relationships in these environments. We found urban waterbodies to be reservoirs of biodiversity that change over both space and time. We found that environmental variables reported to be important for aquatic biodiversity (Jeppesen et al. 2000, Declerck 2005) remain good predictors of zooplankton community patterns, but that they affect only richness difference patterns and not replacement patterns. However, given that both richness differences and replacement played equal roles in shaping our urban zooplankton communities, an important part of the ecology of these communities remains unknown. Future studies should seek to elucidate how zooplankton communities in urban waterbodies assemble, and determine the main drivers of zooplankton replacement patterns.

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SUPPORTING INFORMATION

Additional Supporting Information may be found online at: <http://onlinelibrary.wiley.com/doi/10.1002/ecs2.2277/full>

Ecosphere

Summer assessment of zooplankton biodiversity and environmental control in urban waterbodies on the Island of Montréal

El-Amine Mimouni, Bernadette Pinel-Alloul, Beatrix E. Beisner, and Pierre Legendre

Appendix S1

Table S1: Environnemental characteristics of each waterbody

	Area	Depth	Secchi	TP	Macrophyte	Fish	Emptying
Pratt2	1790.04	0.23	0.23	9.50	0	0	1
Beaubien	720.19	0.83	0.58	66.00	0	0	1
Heritage	8354.79	2.58	1.87	119.83	10	1	0
Lafontaine	19063.44	1.34	1.34	20.00	0	1	1
Centenaire	114466.65	2.12	0.64	54.00	0	1	0
LCastors	18954.68	1.36	1.35	20.67	80	1	0
Angrignon	49103.79	2.16	1.68	20.00	75	1	0
Jarry	10494.74	0.77	0.77	14.00	60	0	1
Cygnés	34757.81	9.43	4.30	15.00	30	1	0
Brunante	17519.43	2.45	0.66	34.67	40	1	0
Bizard	113486.50	0.74	0.74	24.00	80	1	0
Liesse	392.12	0.52	0.52	107.33	40	1	1
Lacoursiere	11395.41	1.46	1.46	21.33	60	1	0
Battures	48741.49	3.08	1.20	52.00	10	1	0
JBNenuphars	6308.65	1.02	1.02	44.33	80	1	0
JBAlgues	7982.23	1.04	1.04	30.00	80	1	0
RMontigny	46362.80	2.33	0.48	107.67	0	1	0
Prairies	8801.33	0.51	0.29	265.33	100	1	0
MCastors	5560.12	2.27	0.96	198.00	100	1	0

Table S2: Macrophytes cover (%) and ranks of dominance in each waterbody

Waterbody	Mcover (in %)	Dominance rank							
		1	2	3	4	5	6	7	8
Brunante	40	Myriophyllum exalbescens	Nymphaea tuberosa	Scirpus validus	Butomus umbellatus	Lythrum salicaria	Phalaris arundinacea	Phragmites	
Liesse	40	Lemna minor	Pontederia cordata	Nymphaea tuberosa	Ceratophyllum demersum				
Bizard	80	Ceratophyllum demersum	Potamogeton foliosus	Typha angustifolia	Nymphoides cordatum	Scirpus validus	Phragmites communis	Heteranthera dubia	
Heritage	10	Myriophyllum exalbescens	Pontederia cordata	Nymphaea tuberosa	Butomus umbellatus	Scirpus fluviatilis	Typha latifolia		
LCastors	80	Myriophyllum exalbescens	Phragmites	Anacharis canadensis (Elodea)					
Angrignon	75	Myriophyllum exalbescens	Nymphaea tuberosa		Typha latifolia	Scirpus fluviatilis	Phragmites communis		
Battures	10	Valisneria americana	Myriophyllum exalbescens	Phragmites					
Lacoursiere	60	Nymphaea tuberosa	Myriophyllum exalbescens	Typha	Phragmites	Sagittaria cuneata	Pontederia cordata	Butomus umbellatus	Potamogeton gramineum
Cyignes	30	Myriophyllum exalbescens	Typha angustifolia	Nymphaea tuberosa					
JBAgues	80	Myriophyllum exalbescens	Chara vulgaris	Typha	Nymphaea tuberosa	Scirpus fluviatilis	Scirpus validis	Phragmites	Equisetum fluviatile
JBNenuphars	80	Nymphaea tuberosa	Equisetum fluviatile	Utricularia intermedia	Riccia	Phragmites			
Jarry	60	Chara vulgaris	Nitella flexilis	Najas flexilis	Scirpus fluviatilis	Butomus umbellatus	Typha	Phragmites	
Prairies	100	Lemna minor	Ceratophyllum demersum	Wolfia punctata	Typha angustifolia	Phalaris arundinacea	Phragmites	Lithrum salicaria	
MCCastors	100	Wolfia punctata	Ceratophyllum demersum	Sparganium americanum	Phragmites maximus	Butomus umbellatus	Alisma gramineum	Sagittaria latifolia	Lithrum salicaria

Table S3: List of recorded zooplankton taxa with their number of occurrences and their associated feeding group

Family	Code	Taxa	OccSumme r	OccJun e	OccJul y	OccAugust	Feeding group
Rotifera							
Asplanchnidae Eckstein, 1883	ASPL	<i>Asplanchna cf. brightwelli</i> Gosse, 1850	11	6	8	6	RRotifera
Atrochidae Harring, 1913	CUVO	<i>Cupelopagis vorax</i> Leidy, 1857	1	0	1	0	RRotifera
Bdelloidea Hudson, 1884	BDEL	<i>Bdelloidea</i> spp. Hudson, 1884	12	5	10	6	MRotifera
Brachionidae Ehrenberg, 1838	BRAC	<i>Brachionus</i> sp. Pallas, 1766	5	3	3	2	MRotifera
	BRAN	<i>Brachionus angularis</i> Gosse 1851	6	3	5	4	MRotifera
	BRCA	<i>Brachionus caudatus</i> Barrois and Daday, 1894	1	1	0	1	MRotifera
	BRQU	<i>Brachionus quadridentatus</i> Hermann, 1783	6	4	5	5	MRotifera
	KELO	<i>Kellicottia longispina</i> Kellicott, 1879	2	2	0	0	MRotifera
	KERA	<i>Keratella</i> sp. Bory de St. Vincent, 1822	19	18	17	16	MRotifera
	KEFA	<i>Keratella cochlearis faluta</i> Ahlstrom, 1943	2	1	0	1	MRotifera
	KEHI	<i>Keratella hiemalis</i> Carlin, 1943	3	3	0	0	MRotifera
	KETE	<i>Keratella tecta</i> Gosse, 1851	3	3	2	2	MRotifera
	NOAC	<i>Notholca acuminata</i> Ehrenberg, 1832	3	3	0	0	MRotifera
	NOEX	<i>Notholca acuminata extensa</i> Ehrenberg, 1832	1	1	0	0	MRotifera
	PLPA	<i>Platytia patulus</i> Müller, 1786	12	7	9	10	MRotifera
	PLQU	<i>Platytia quadricornis</i> Ehrenberg, 1832	4	3	1	2	MRotifera
	Conochilidae Harring, 1913	CONO	<i>Conochiloides</i> sp. Hlava, 1904	6	2	2	4
Dicranophoridae Harring, 1913	DICR	<i>Dicranophorus</i> sp. Nitzsch, 1827	9	2	5	8	RRotifera
Euchlanidae Ehrenberg, 1838	EUCH	<i>Euchlanis</i> spp. De Beauchamp, 1910	16	12	13	12	MRotifera
Filiniidae Harring and Myers, 1926	FILI	<i>Filinia</i> sp. Bory de St. Vincent, 1824	9	3	7	6	MRotifera
Gastropodidae Harring, 1913	ASEC	<i>Ascomorpha ecaudis</i> Perty, 1850	11	6	9	5	RRotifera
	GAHY	<i>Gastropus cf. hyptopus</i> Ehrenberg, 1838	2	2	1	1	RRotifera
Hexarthridae Bartos, 1959	HEMI	<i>Hexarthra mira</i> Hudson, 1871	6	2	5	5	MRotifera
Lecanidae Remane, 1933	LECA	<i>Lecane</i> sp. Nitsch, 1827	14	7	11	11	MRotifera
	LEMO	<i>Lecane (Monostyla)</i> sp. Nitzsch, 1827	18	16	15	12	MRotifera
	LEBU	<i>Lecane (Monostyla) bulla</i> Gosse, 1851	15	11	9	10	MRotifera
	LECR	<i>Lecane crepida</i> Harring, 1914	2	0	0	2	MRotifera
	LELE	<i>Lecane leontina</i> Turner, 1892	2	1	1	1	MRotifera
	LELU	<i>Lecane ludwigi</i> Eckstein, 1883	3	1	1	2	MRotifera
	LEOH	<i>Lecane ohioensis</i> Herrick, 1885	6	4	4	2	MRotifera
	LEQU	<i>Lecane (Monostyla) quadridentata</i> Ehrenberg, 1832	10	7	9	8	MRotifera
	LEST	<i>Lecane stokesi</i> Pell, 1890	2	2	0	1	MRotifera

Lepadellidae Harring, 1913	COLU	<i>Colurella</i> sp. Bory de St. Vincent, 1824	7	5	5	7	MRotifera
	LEPA	<i>Lepadella patella</i> Bory de St. Vincent, 1826	13	11	9	8	MRotifera
	LEEH	<i>Lepadella ehrenbergi</i> Perty 1850	3	0	2	3	MRotifera
	PARA	<i>Paracolurella</i> sp. Myers, 1936	1	1	0	0	MRotifera
Mytilinidae Harring, 1913	LOPH	<i>Lophocharis</i> sp. Ehrenberg, 1838	3	3	0	0	MRotifera
	MYTI	<i>Mytilina</i> sp. Bory de St. Vincent, 1826	11	9	10	5	MRotifera
Nothomattidae Hudson and Gosse, 1886	NOTH	Nothomattidae spp. Hudson and Gosse, 1886	2	1	1	0	RRotifera
	CEGI	<i>Cephalodella gibba</i> Ehrenberg, 1832	9	3	5	7	RRotifera
	MON O	<i>Monommata</i> sp. Bartsch, 1870	3	2	0	2	RRotifera
Philodinidae Ehrenberg, 1838	DISS	<i>Dissotrocha</i> sp. Bryce, 1910	8	7	7	7	MRotifera
Scaridiidae Manfredi, 1927	SCAR	<i>Scaridium</i> sp. Ehrenberg, 1830	5	4	3	4	RRotifera
Synchaetidae Hudson and Gosse, 1886	PLOE	<i>Ploesoma</i> sp. Herrick, 1885	3	1	2	3	RRotifera
	POLY	<i>Polyarthra</i> spp. Ehrenberg, 1834	17	13	16	15	RRotifera
	POEU	<i>Polyarthra</i> cf. <i>euryptera</i> Wierzejski, 1891	1	1	0	0	RRotifera
	SYN	<i>Synchaeta</i> spp. Ehrenberg, 1832	10	6	4	7	RRotifera
	POSU	<i>Pompholyx sulcata</i> Hudson, 1885	1	1	0	1	MRotifera
Testudinellidae Harring, 1913	TEST	<i>Testudinella</i> sp. Bory de St. Vincent, 1826	13	9	11	8	MRotifera
	TRIC	<i>Trichocerca</i> sp. Lamarck, 1801	1	0	1	0	RRotifera
Trichocercidae Harring, 1913	TRBI	<i>Trichocerca bicristata</i> Gosse, 1887	9	4	5	7	RRotifera
	TRCY	<i>Trichocerca cylindrica</i> Imhof, 1891	6	6	4	4	RRotifera
	TRLA	<i>Trichocerca lata</i> Jennings, 1894	2	0	2	2	RRotifera
	TRMU	<i>Trichocerca mucosa</i> Stokes, 1896	6	5	2	2	RRotifera
	TRMC	<i>Trichocerca multicrinis</i> Kellicott, 1897	4	1	1	4	RRotifera
	TRRA	<i>Trichocerca rattus</i> Müller, 1776	2	2	0	0	RRotifera
	TRSI	<i>Trichocerca similis</i> Wierzejski, 1893	7	4	5	5	RRotifera
	Trichotriidae Harring, 1913	MACR	<i>Macrochaetus</i> sp. Perty, 1850	1	1	0	0
TRPO		<i>Trichotria pocillum</i> Müller, 1776	10	7	5	7	MRotifera
TRTE		<i>Trichotria tetractis</i> Ehrenberg, 1830	7	5	5	2	MRotifera

Cladocera

Bosminidae Baird, 1845	BOSM	Bosminidae spp. Baird, 1845	17	16	15	13	SFCladocera
Chydoridae Stebbing, 1902	ACHA	<i>Acroperus harpae</i> Baird, 1834	1	1	1	1	SGCladocera
	ALON	<i>Alona</i> spp. Baird, 1850	15	14	13	13	SGCladocera
	CARE	<i>Camptocercus rectirostris</i> Schödler, 1862	6	6	4	2	SGCladocera
	CHYD	<i>Chydorus</i> spp. Leach, 1843	18	18	15	13	SGCladocera
	EURY	<i>Eurycercus</i> sp. Baird, 1843	2	2	0	0	SGCladocera
	GRTE	<i>Graptoleberis testudinaria</i> Fischer, 1848	3	3	2	2	SGCladocera
	KULA	<i>Kurzia</i> cf. <i>latissima</i> Kurz, 1874	5	3	1	1	SGCladocera
	LEAC	<i>Leydigia</i> cf. <i>acanthocercoides</i> Fischer, 1853	2	1	1	0	SGCladocera
	PLDE	<i>Pleuroxus denticulatus</i> Birge, 1879	10	8	7	5	SGCladocera
	PLPR	<i>Pleuroxus procurvus</i> Birge, 1879	10	9	5	6	SGCladocera

Daphniidae Staus, 1820	CERI	<i>Ceriodaphnia</i> sp. Dana, 1853	15	13	12	11	SFCladocera
	DAA M	<i>Daphnia ambigua</i> Scourfield, 1947	1	1	0	0	SFCladocera
	DAGA	<i>Daphnia galeata mendotae</i> Birge, 1918	3	3	3	2	SFCladocera
	DAPU	<i>Daphnia</i> spp. Dana, 1853	4	4	1	0	SFCladocera
	SCAP	<i>Scapholeberis</i> sp. Dumont and Pensaert, 1983	15	14	11	9	SFCladocera
	SIMO	<i>Simocephalus</i> sp. Schoedler, 1858	15	14	10	10	SFCladocera
Ilyocryptidae Smirnov, 1992	ILYO	<i>Ilyocryptus</i> sp. Sars, 1862	3	2	1	0	SGCladocera
Leptodoridae Lilljeborg, 1861	LEKI	<i>Leptodora kindtii</i> Focke, 1844	1	1	0	0	CCLadocera
Macrothricidae Norman and Brady, 1867	MACR	<i>Macrothrix</i> sp. Baird, 1843	7	4	4	2	SGCladocera
	OPGR	<i>Ophryoxus gracilis</i> Sars, 1861	1	0	0	1	SGCladocera
Polyphemidae Baird, 1845	POPE	<i>Polyphemus pediculus</i> Linnaeus, 1761	2	2	1	1	CCLadocera
Sididae Baird, 1850	DIAP	<i>Diaphanosoma</i> sp. Fischer, 1850	15	13	15	12	SFCladocera
	SICR	<i>Sida crystallina</i> Müller, 1776	8	6	4	5	SGCladocera
Copepoda							
Cyclopidae Dana, 1846	EUPE	<i>Eucyclops</i> cf. <i>pectinifer</i> Cragin, 1883	10	6	8	8	RCopepoda
	MAAL	<i>Macrocyclus albidus</i> Jurine, 1820	8	6	6	2	RCopepoda
	MICR	<i>Microcyclus</i> sp. Claus, 1893	11	4	8	8	RCopepoda
Diaptomidae Baird, 1850	SKRO	<i>Skistodiaptomus oregonensis</i> Lilljeborg, 1889	3	3	3	2	SSFCopepoda
	SKRE	<i>Skistodiaptomus reighardii</i> Marsh, 1895	5	2	5	1	SSFCopepoda
	ONBI	<i>Onychodiaptomus birgei</i> Marsh, 1894	6	5	1	0	SSFCopepoda

SGCladocera: Substrate-grazing cladocera

SFCladocera: Seston-filtering cladocera

CCLadocera: Carnivorous cladocera

RCopepoda: Raptorial cyclopoids

SSFCopepoda: Stationary and suspension-feeding calanoids

MRotifera: Microphagous rotifera

RRotifera: Raptorial rotifera

Table S4. Number of taxa of each zooplankton assemblage (Rotifera, Cladocera, Copepoda, Zooplankton) and in the 19 sampled waterbodies for each month (June, July, August) and the total summer survey.

June				
	Rotifera	Cladocera	Copepoda	Zooplankton
Pratt2	6	4	0	10
Beaubien	2	6	0	8
Heritage	16	8	3	27
Lafontaine	2	8	0	10
Centenaire	10	6	0	16
LCastors	15	12	3	29
Angrignon	13	11	1	24
Jarry	22	7	1	29
Cygnés	18	14	3	34
Brunante	20	8	1	29
Bizard	18	12	3	32
Liesse	6	7	0	13
Lacoursiere	24	9	3	36
Battures	8	7	1	16
JBNenuphars	18	13	4	34
JBAlgues	18	12	3	33
RMontigny	9	2	0	11
Prairies	10	6	2	18
MCastors	18	6	4	28
July				
	Rotifera	Cladocera	Copepoda	Zooplankton
Pratt2	3	2	0	5
Beaubien	7	7	0	14
Heritage	16	9	0	25
Lafontaine	7	5	1	13
Centenaire	6	1	0	7
LCastors	17	4	3	24
Angrignon	16	9	1	26
Jarry	15	6	2	23
Cygnés	19	11	3	33
Brunante	15	8	4	26

Bizard	25	9	4	37
Liesse	2	7	0	9
Lacoursiere	22	10	3	34
Battures	4	4	0	8
JBNenuphars	19	12	4	34
JBAlgues	19	9	3	31
RMontigny	7	2	1	10
Prairies	16	6	3	25
MCastors	23	5	3	31
August				
	Rotifera	Cladocera	Copepoda	Zooplankton
Pratt2	3	4	0	7
Beaubien	3	7	0	10
Heritage	7	2	1	10
Lafontaine	3	2	0	5
Centenaire	10	0	0	10
LCastors	14	3	1	18
Angrignon	13	4	0	17
Jarry	19	7	2	27
Cygnés	22	14	4	39
Brunante	8	5	2	14
Bizard	25	9	2	36
Liesse	2	6	1	9
Lacoursiere	26	8	3	36
Battures	4	7	1	12
JBNenuphars	27	11	3	40
JBAlgues	22	8	2	32
RMontigny	7	2	0	9
Prairies	16	6	2	24
MCastors	24	4	2	30
Total				
	Rotifera	Cladocera	Copepoda	Zooplankton
Pratt2	6	6	0	12
Beaubien	8	9	0	17
Heritage	22	10	4	36
Lafontaine	8	8	1	17
Centenaire	16	6	0	22
LCastors	26	12	4	41
Angrignon	20	12	2	33
Jarry	29	8	3	39

Cygnés	33	16	5	53
Brunante	21	9	4	33
Bizard	31	13	5	48
Liesse	6	8	1	15
Lacoursiere	35	12	5	51
Battures	9	8	2	19
JBNenuphars	31	14	4	48
JBAlgues	27	13	3	43
RMontigny	11	2	1	14
Prairies	21	7	3	31
MCastors	27	6	4	37
