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Les macroinvertébrés benthiques littoraux :

Bioindicateurs de la qualité écologique des milieux humides en zone urbaine

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Les macroinvertébrés benthiques littoraux :
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Abstract

Aquatic ecosystems in urban landscapes are now recognized as good islets of biodiversity and valuable environments providing ecological services. However, more knowledge is needed to assess their ecological quality. Benthic macroinvertebrates are widely used as bioindicators, but rarely for urban ecosystems. In this study, we used macroinvertebrates to evaluate the ecological quality of urban ponds and lakes in the island of Montreal. We collected macroinvertebrates during summer 2011 in the littoral zone of 20 waterbodies varying in urban and limnological characteristics. We evaluated spatio-temporal variation in several diversity and biotic indices and multiple metrics based on taxonomic composition and functional traits. We investigated if macroinvertebrate metrics responded to variation in urban land-use, pond management and limnological conditions. Our study showed that small waterbodies, as ponds, lakes and marshes are important resources for sustaining aquatic biodiversity in urban landscapes. Natural wetlands and artifical permanent ponds had higher ecological quality and supported more diverse and abundant macroinvertebrates than artificial managed temporary ponds in municipal parks. Vegetation cover, nutrient and organic contents, and algal biomass were the most important factors explaining spatial variation in macroinvertebrate metrics based on taxonomy and functional traits. Pond management, urban density, and water permanence were also influencial factors. Some univariate metrics also had potential to assess the responses of macroinvertebrates to environmental features. We discussed the implications of our study for management and quality assessment of urban ponds.

Keywords: Urban ecology, littoral macroinvertebrates, taxonomic composition, functional traits, biodiversity, bioindicators, urban ponds, ecological quality.

Résumé

Les milieux aquatiques en zone urbaine sont reconnus comme des îlots de biodiversité qui offrent de nombreux services écologiques. Dans cette étude, nous avons utilisé les macroinvertébrés comme bioindicateurs de la qualité écologique des étangs, petits lacs et marais de l'Île de Montréal. Les macroinvertébrés ont été récoltés durant l'été 2011 dans la zone littorale de 20 sites variant par leur urbanisation et leurs caractéristiques limnologiques. Nous avons évalué la variation dans la richesse en taxa, les indices de diversité et plusieurs métriques basées sur la composition taxonomique ou les traits fonctionnels. Nous avons déterminé la réponse des métriques aux changements dans l'urbanisation, l'aménagement et les conditions des plans d'eau. Notre étude montre que les étangs, marécages et petits lacs constituent des réserves importantes de biodiversité en zone urbaine. Les marécages naturels et les étangs et lacs permaments avaient une meilleure qualité écologique et supportaient des communautés de macroinvertébrés plus diverses et abondantes que les petits étangs temporaires aménagés des parcs. Le couvert de végétation aquatique, l'enrichissement en nutriments et en matière organique ainsi que la biomasse des algues expliquaient le plus de variation dans les macroinvertébrés. Les aménagements, la densité urbaine et la permanence de l'eau avaient aussi une influence. Certaines métriques univariées avaient aussi le potentiel d'évaluer la réponse des macroinvertébrés aux facteurs environnementaux. Nous avons discuté les implications de notre étude pour le suivi environnemental de la biodiversité et la qualité écologique des milieux aquatiques en zone urbaine.

Mots-clés : Écologie urbaine, macroinvertébrés littoraux, composition taxonomique, traits fonctionnels, biodiversité, bioindicateurs, étangs urbains, qualité écologique.

Table des matières

Liste des tableaux

Liste des figures

Liste des abréviations

%COH	Abondance Relative de Chironomidae, Oligochaeta et Hirudinea / *Relative abundance of Chironomidae, Oligochaeta and Hirudinea*
%ET	Abondance Relative d'Ephemeroptera et Trichoptera / *Relative abundance of Ephemeroptera and Trichoptera*
°C	Degrés Celcius / *Degree Celsius*
°N	Degrés Nord / *Degree North*
°W	Degrés Ouest / *Degree West*
$\mu g \cdot L^{-1}$	Microgramme par Litre / *Microgram per Liter*
A	Origine, Utilisation Urbaine / *Origin, urban land-use*
B	Couverture végétale, morphométrie, qualité de l'eau et biomasse algale / *Vegetation cover, morphometry, water quality and algal biomass*
C	Présence/absence de poissons / *Fish presence/absence*
Chla	Biomasse de Chlorophylle / *Chlorophyll Biomass*
cm	Centimètres / *Centimeters*
ET	Nombre de taxa d'Ephemeroptera et de Trichoptera / *Number of taxa of Ephemeroptera and Trichoptera*
ET/COH	Abondance d'Ephemeroptera et de Trichoptera sur l'abondance de Chironomidae, Oligochaeta et Hirudinea / *Abundance of Ephemeroptera and Trichoptera on abundance of Chrironomidae, Oligochaeta and Hirudinea*
FBI	Indice Biotique basé sur le Familles / *Family Biotic Index*
FDis	Dispersion Fonctionnelle / *Functional Dispersion*
FEve	Équitabilité Fonctionnelle / *Functional Eveness*
Fric	Richesse Fonctionnelle / *Functional Richness*
GPS	*Global Positioning System*
Ha	Hectare / *Hectare*
K	Potassium / *Potassium*
km^2	Kilomètre Carré / *Kilometer Square*
LCBD	Contribution Locale à la Diversité Beta / *Local Contribution to Beta Diversity*

m	Mètre / *Meter*
$mg \cdot L^{-1}$	Milligramme par Litre / *Milligram per Liter*
Mm	Millimètre / *Millimeter*
NH_4	Ammonium / *Ammonium*
PCA	Analyse en Composantes Principales / *Principal Components Analysis*
R^2	Coefficient de Régression / *Regression Coefficient*
RDA	Analyse Canonique de Redondance / *Redundancy Analysis*
SCBD	Contribution des Espèces à la Diversité Beta / *Species Contribution to Beta Diversity*
TOC	Carbone Organique Total / *Total Organic Carbon*
TP	Phosphore Total / *Total Phosphorus*
α	*Alpha*
β	*Beta*
γ	*Gamma*
μm	Micromètre / *Micrometer*
$\mu S \cdot cm^{-1}$	MicroSiemens par Centimètre / *MicroSiemens per Centimeter*

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Chapitre 1 : Introduction générale

I. Écologie urbaine

L'écologie est une science encore relativement jeune qui s'est surtout focalisée au cours du 20ième siècle sur les écosystèmes naturels tels que les forêts, les lacs et les cours d'eau. Ce n'est que récemment que les écologistes ont commencé à reconnaître l'importance d'étudier les écosystèmes anthropisés que l'on rencontre en région urbaine. L'écologie urbaine est maintenant reconnue comme un nouveau champ de recherche avec ses propres paradigmes intégrant les interactions complexes entre l'environnement et la société. (Pickett *et al.,* 1997; 2001; Marzluff *et al.,* 2008). L'étude de l'écologie des milieux urbains est d'autant plus importante qu'au cours du 21ième siècle, plus des deux-tiers de la population humaine sera concentrée dans de très grands centres urbains où la fragmentation des habitats, la perte des espaces naturels et le développement résidentiel et industriel sont des causes importantes de perte de biodiversité et d'intégrité écologique des écosystèmes (Niemelä, 1999; Grimm *et al.,* 2008).

Au Québec, l'écologie urbaine est devenue un sujet d'actualité suite au Sommet sur la biodiversité et le verdissement de Montréal qui a eu lieu en avril 2010 (http://www.cremtl.qc.ca/mots-clefs/sommet-biodiversite-montreal). Un des constats de ce sommet est le manque d'études sur les milieux aquatiques comparativement aux travaux faits sur les espaces et corridors verts, même si on prend de plus en plus conscience de l'importance des milieux humides dans les villes. Ils sont non-seulement une source et des réservoirs d'eau douce plus grands qu'on le croirait à première vue, mais aussi des bons îlots pour promouvoir la biodiversité aquatique et un espace nature de très haute qualité (Hassall, 2014).

II. Importance des petits lacs et étangs en limnologie et en écologie

Les petits lacs et les étangs constituent une proportion importante des écosystèmes d'eau douce au niveau mondial (Downing, 2006). En effet, les millions de petits lacs de moins d'un hectare (0,001-0,01 km^2) occupent autant de surface sur les continents que les quelques très grands lacs de 10 000 à 100 000 km^2 (Fig. 1A) (Downing, 2010). Ces petits lacs, au nombre de 304 millions, recouvrent une aire de 4,2 millions de km^2. Mais, cet estimé ne tient pas compte des très petits lacs et étangs de 100 à 1000 m^2 (0.01 à 0.1 ha), permanents ou temporaires, qui pourraient être au nombre de 3,2 milliards et recouvrir une aire totale de 0,8 milliards de km^2. Ces milieux d'origine naturelle ou artificielle se retrouvent surtout dans des regions agricoles et urbaines où leur nombre surpasse celui des lacs naturels par un ratio de 100:1 (Oertli *et al.,* 2005a). Ils couvrent une surface de 76 830 km^2 au niveau mondial et 21 600 km^2 au États-Unis. Depuis 2000, plusieurs initiatives de recherche (BIOPOOL – EuroDIVERSITY; PONDSCAPE; European Pond Workshops) ont été lancées en Europe pour évaluer les caractéristiques limnologiques et écologiques de ces petits plans d'eau et leur rôle pour la conservation de la biodiversité aquatique dans les régions agricoles et urbaines (Oertli *et al.,* 2005b; Biggs *et al.,* 2005; Céréghino *et al.,* 2008a). En comparaison, les recherches portant sur l'écologie urbaine en Amérique du Nord, en particulier sur les milieux humides, sont encore en émergence. Elles se concentrent surtout sur les cours d'eau (Paul et Meyer, 2001; Walsh *et al.,* 2005) et les petits étangs des regions agricoles et périurbaines de l'est des États-Unis (Dodson, 2008; Drenner *et al.,* 2009). Aucune étude n'a jusqu`alors porté attention aux étangs dans les grandes villes du Canada, pour déterminer leur valeur pour la conservation de la biodiversité en région urbaine.

III. Valeur des étangs pour préserver la biodiversité et la qualité écologique en région urbaine

Recemment, un article de synthèse (Downing, 2010) a mis l'emphase sur l'importance des étangs et petits lacs au niveau des cycles globlaux du carbone et de l'azote mais également pour la conservation de la biodiversité aquatique au niveau mondial. Les petits lacs et étangs de moins d'un hectare (0.01 km^2) supportent une plus grande biodiversité de plancton, de poissons et de macrophytes que les grands lacs de 100 à 1000 km^2 (Fig. 1B). Ces milieux supportent aussi une grande diversité d'oiseaux aquatiques, d'amphibiens et d'invertébrés à cause de la présence des herbiers aquatiques et de l'absence de poissons prédateurs dans les milieux temporaires (Scheffer *et al.,* 2006). En region urbaine, les étangs et petits lacs constituent une énorme diversité d'habitats ayant des conditions abiotiques et biotiques très différentes (Hassall, 2014). On y retrouve des milieux de taille très variables (10 m^2 à 1 km^2) incluant des marais ou des lacs dans les parcs ou réserves naturelles, des réservoirs de drainage des eaux pluviales ou des étangs de retention des eaux industrielles, des petits lacs et étangs dans les parcs municipaux, et de très petits étangs dans les jardins résidentiels. Bien, qu'individuellement, ils peuvent contenir une faible diversité d'espèces, la mosaïque d'habitats qu'ils constituent, crée une grande hétérogéneité environnementale offrant une grande variété de niches écologiques au niveau du paysage urbain. De plus, les étangs répartis dans le paysage urbain servent de points de dispersion des espèces vers d'autres étangs, via les réseaux hydrographiques ou le transport par le vent ou les oiseaux des stades dormants ou résistants de plantes et d'invertébrés. Ce phénomène de dispersion augmente la diversité régionale, malgré une biodiversité locale parfois faible à l'échelle d'un étang unique (Hassall, 2014). En milieu urbain, ce phénomène de dispersion est d'autant plus important qu'il permet de coloniser les nouveaux étangs artificiels par des peuplements pionniers. Finalement, les milieux humides créés ou entretenus dans les villes fournissent de nombreux services écologiques, en particulier des espaces plus naturels aptes à améliorer le bien-être et la santé des résidents et des lieux éducatifs d'intérêt pour le public (Hassall, 2014).

Les étangs et petits lacs sont maintenant reconnus comme des systèmes modèles très intéressants pour les études en biologie de la conservation, en écologie des communautés et en évolution (De Meester *et al.,* 2005). L'étude de ces systèmes permet de mieux comprendre la théorie des métacommunautés (Pillai *et al.,* 2011) et les patrons de distribution de la biodiversité, deux éléments essentiels pour définir des pratiques de gestion des milieux humides visant à accroître la diversité biologique au niveau des paysages agricoles et urbanisés (Céréghino *et al.,* 2008a).

Plusieures métriques permettant de décrire les patrons de variation de la diversité biologique selon une approche hiérarchique allant de l'échelle locale à l'échelle régionale, peuvent s'appliquer aux milieux humides en zone urbaine. La diversité alpha (α) représente la diversité à l'échelle locale (Gaston et Spicer, 2004). Elle correspond au nombre de taxons présents au sein d'un milieu unique et peut aussi être exprimée par un indice de diversité basé sur le nombre et la fréquence de distribution des taxons (Indices de Shannon ou de Simpson). La diversité beta (β) représente le changement dans la diversité entre les milieux et sert à déterminer la contribution des sites ou des taxons à la variation spatiale de la biodiversité au niveau régional (Legendre *et al.,* 2005; Legendre et Càceres, 2013). Finalement, la diversité gamma (γ) représente le nombre total de taxons cumulés au niveau régional dans l'ensemble des milieux. Plusieurs études ont démontré l'existence d'une corrélation positive entre l'importance des changements de la composition en espèces (diversité β) de macroinvertébrés et l'hétérogénéité des habitats en rivières (Townsend, 1989; Townsend et Hildrew, 1994). En contrepartie, l'homogénéisation des sites en zone urbaine due à des stress ou des aménagements anthropiques (par exemple, faucardage de la végétation, altération de l'habitat par le remblayage ou l'érosion, introduction d'espèces exotiques au détriment des espèces indigènes) devrait entraîner une baisse de la diversité β des macroinvertébrés. Toutefois, les études donnent parfois des résultats contradictoires. Ainsi, l'augmentation des pratiques agricoles dans les bassins versants favorise l'homogénéisation des communautés de macroinvertébrés dans les ruisseaux des régions de piémont ou de plaine aux USA (Maloney *et al.,* 2011), tandis que d'autres perturbations environnementales en augmentent l'hétérogénéité biologique et la diversité β (Hawkins *et al.,* 2014). Aucune étude n'a évalué les relations entre le niveau de perturbations anthropiques et la diversité des macroinvertébrés benthiques des milieux urbains, bien qu'il soit nécessaire d'avoir une bonne compréhension des patrons de diversité aux niveaux local et régional pour établir des programmes de gestion environnementale. Ainsi, un étang qui a une grande diversité localement pourrait ne pas avoir une grande valeur de conservation s'il contient uniquement des espèces communes retrouvées dans tous les étangs de la région. Par contre, un étang ayant peu d'espèces mais des espèces endémiques ou rares aurait une meilleure valeur de conservation car il contribue plus aux changements dans la diversité β au niveau régional. La gestion des étangs urbains devrait donc viser à maximiser la diversité β et γ au niveau des peuplements de macroinvertébrés en conservant des étangs représentant un continuum de stades de succession allant de peuplements pionniers à des peuplements stabilisés (Hassall *et al.,* 2012).

IV. Les macroinvertébrés comme bioindicateurs de qualité environnementale

Les macroinvertébrés benthiques sont déjà reconnus comme de très bons bioindicateurs de qualité écologique des lacs et rivières car ils sont en contact étroit avec leur environnement durant toute leur durée de vie (Pinel-Alloul *et al.,* 1996; Tessier *et al.,* 2008; Tall *et al.,* 2008). Le suivi environnemental basé sur les macroinvertébrés est très efficace parce que ceux-ci ne répondent pas seulement aux polluants, mais aussi aux changements dans l'utilisation des terres dans le bassin versant et aux changements physiques et biologiques de l'habitat, qui sont difficiles à évaluer avec un suivi classique de toxicologie ou de chimie (Rosenberg et Resh, 1993). Au Canada, les macroinvertébrés benthiques ont été utilisés dans les grands programmes de suivi environnemental des milieux d'eau douce (Rosenberg et Resh, 1993; Bailey *et al.,* 2004; programme CABIN : Environnement Canada 2008, 2010, programme ESEE 2012). Plus particulièrement, ils ont été utilisés pour évaluer l'intégrité écologique des milieux humides dans les Grands Lacs (Kashian et Burton, 2000) et les grands fleuves comme le Saint-Laurent et la rivière Fraser (Reynoldson *et al.,* 1997, 2001; Tall *et al.,* 2008). Toutefois, leur application comme bioindicateurs pour évaluer la qualité écologique des milieux humides en zone urbaine ou agricole est encore peu développée. En Europe, les macroinvertébrés benthiques, dont les familles d'insectes telles que les Odonates et les Éphémères, ont servis pour l'évaluation de la qualité des petits étangs en zone agricole en Suisse (Oertli *et al.,* 2005b), en France (Céréghino *et al.,* 2008b) et en Angleterre (Moss *et al.,* 2003). Au Canada, les macroinvertébrés benthiques sont encore très peu utilisés pour évaluer la qualité écologique des milieux humiques en zone urbaine. Plus particulièrement dans la ville de Montréal, on ne recense actuellement que quelques études préliminaires sur les communautés de macroinvertébrés des ruisseaux (Barfoud, 2011) et des étangs (Dedieu 2010).

V. Analyses de la structure des communautés de macroinvertébrés aquatiques

i. Niveau d'identification taxonomique

Plusieurs études ou programmes de suivi environnemental basés sur l'occurrence (présence-absence) ou l'abondance des macroinvertébrés ont montré qu'une analyse taxonomique au niveau de la famille peut donner des informations similaires à celles obtenues avec une analyse plus fine au niveau du genre, même si au niveau du genre plus de taxons indicateurs sont retrouvés (Reynoldson *et al.,* 2001; Environment Canada, 2002 and 2010; Feio *et al.,* 2006; Chessman *et al.,* 2007; Jones, 2008; Masson *et al.,* 2010; Neeson *et al.,* 2013). Il a été démontré qu'une résolution taxonomique plus élevée au niveau du genre n'affecte pas beaucoup la définition des traits fonctionnels des communautés de macroinvertébrés (Dolédec *et al.,* 2000; Gayraud *et al.,* 2003). De plus, l'analyse taxonomique au niveau de la famille permet de trier les échantillons plus rapidement et avec moins de main d'œuvre. Pour ces raisons, l'identification des macroinvertébrés a été faite au niveau de la famille dans le cadre de notre étude sur les étangs urbains.

ii. Approche taxonomique

Classiquement, l'analyse taxonomique des communautés de macroinvertébrés se base sur la détermination de la richesse en taxons et sur leur occurrence (présence-absence) ou abondance (nombre d'individus). Cette méthode permet d'estimer différentes métriques pour évaluer la qualité écologique des écosystèmes.

Premièrement, on peut calculer des métriques univariées très simples comme la richesse en taxons et les indices de diversité de Shannon et de Simpson. L'indice de Shannon (Shannon, 1948) tient compte du nombre de taxons et de l'abondance des individus de chacun de ces taxons. Une communauté dominée par un seul taxon aura un indice plus petit qu'une communauté comprenant plusieurs taxons ayant des abondances similaires. La valeur de l'indice varie de 0 (un seul taxon ou un taxon dominant les autres) à log S (tous les taxons ont la même abondance). L'indice de diversité de Shannon est la métrique la plus utilisée pour évaluer les impacts environnementaux sur les écosystèmes. L'indice de Shannon est souvent associé à l'indice d'équitabilité de Pielou (Pielou, 1966). L'indice d'équitabilité calcule la répartition des individus dans les différents taxons, sans tenir compte de la richesse en taxons. Sa valeur varie de 0 (un taxon dominant) à 1 (répartition égale des individus entre les taxons). Ces deux indices dépendent de la taille des échantillons et du type d'habitat. Il est donc difficile de les utiliser comme descripteurs de l'état d'un milieu, sauf si on détermine des valeurs seuil pour chaque type d'habitat et pour une surface donnée. L'indice de diversité de Simpson (Simpson, 1949) calcule la probabilité que deux individus pris au hasard appartiennent au même taxon (ou famille). L'indice de Simpson est inversement proportionnel à la diversité. Une autre formulation existe pour établir un indice représentant directement la diversité en taxons. Il suffit de soustraire l'indice de Simpson de 1 et donc cet indice varie de 0 (diversité minimum) à 1 (diversité maximum).

L'approche taxonomique permet aussi de calculer divers indices biotiques indicateurs de l'intégrité biologique des sites. Par exemple, l'indice d'Hilsenhoff (Hilsenhoff, 1987) se base sur l'abondance relative des différents taxons classés en fonction de leur niveau de tolérance à l'enrichissement organique ou la pollution. Un indice élevé signifie qu'il y a un stress dans l'environnement. D'autres indices biotiques sont basés sur les ratios entre l'abondance des familles tolérantes ou sensibles à la pollution. Un des plus connus est le ratio ET:COH basé sur les abondances (absolues ou relatives) de taxons sensibles (Éphémères et Trichoptères: ET) et de taxons tolérants (Diptères Chironomides, Oligochètes et sangsues Hirudinées: COH).

Finalement, l'approche taxonomique permet aussi de calculer des métriques multivariées sur la base des matrices d'assemblages des taxons dans chacun des sites. Ces métriques peuvent être calculées sur les données d'occurrence (présence-absence) à l'aide de l'indice de Jaccard (Jaccard 1908) ou les données d'abondance avec l'indice de Hellinger (Rao 1995). Ces différentes métriques permettent d'estimer la diversité locale (α: richesse en taxon dans un site), les changements dans la diversité entre les sites (β : variation de la richesse entre les sites) et la diversité régionale (γ : dans l'ensemble des sites). Elles se distribuent selon un continuum allant d'une simple mesure de la richesse en taxons (indices de diversité) à des mesures qui tiennent compte de l'occurrence des taxons (indice de Jaccard) ou de leur abondance relative (indice de Hellinger). Ces diverses méthodes permettent d'estimer la contribution des sites ou des taxons à la diversité β, soit aux variations de la composition et de la richesse en taxons entre les sites.

L'utilisation d'une combinaison de métriques donne des informations complémentaires qui permettent une compréhension plus complète de la structure des écosystèmes et de leurs réponses aux changements environnementaux (Clarke et Warwick, 2001). Toutefois, les métriques basées sur la composition taxonomique ne permettent pas de prédire les effets des changements environnementaux sur le fonctionnement des communautés et des écosystèmes ni de faire des comparaisons entre les réponses des écosystèmes de différentes écorégions dont les assemblages de taxons diffèrent.

iii. Approche par traits fonctionnels

Pour relier les changements dans les communautés de macroinvertébrés au fonctionnement des écosystèmes, on préconise actuellement de déterminer la diversité fonctionnelle des macroinvertébrés sur la base de leurs traits bioécologiques (Usseglio-Polatera et al. 2000a and 2000b). Il a été démontré que la diversité fonctionnelle est la métrique la plus polyvalente puisqu'elle donne une indication non seulement du nombre d'espèce et de la dominance, mais aussi de leur rôle dans la communauté. Elle est indépendante de l'effort d'échantillonnage, ne demande pas plus d'effort d'identification qu'une approche basée uniquement sur la taxonomie, est facile à calculer et permet de faire des comparaisons entre des sites ayant un pool d'espèces différentes, toutes des caractéristiques désirées pour faire un suivi sur des communautés dans différentes échelles spatiales et/ou temporelles (Mouillot et al. 2006; Bady et al. 2005; Van den Brink et al., 2011). Évaluer les caractéristiques fonctionnelles d'une communauté en regroupant les individus en des groupes non-taxonomiques sur la base de leur traits biologiques et écologiques est une méthode qui dépend moins de la variabilité saisonnière, ce qui fait que la réponse de la communauté aux conditions environnementales est plus informative. Cela permet une comparaison sur une plus grande échelle biogéographique (Pinto et al. 2009; Dolédec et al. 1999, 2006; Baird et al. 2008).

Les traits fonctionnels sont définis comme des caractéristiques qui reflètent l'adaptation d'une espèce à son environnement (Menezes *et al.,* 2010). La liste des traits fonctionnels couramment utilisés pour les macroinvertébrés aquatiques a été établie par Poff *et al.* (2006) sur la base de 4 catégories de traits bioécologiques (histoire de vie, mobilité, morphologie, écologie). Selon Dolédec *et al.* (2006), les traits ayant rapport au cycle de vie (nombres de cycles de reproduction, durée de vie, mode de position des œufs, et les comportements parentaux) expliquent le plus de variance dans les profils de traits bioécologiques. Le choix des traits fonctionnels est primordial pour s'assurer que leurs variations soient liées aux changements de l'environnement. Il faut donc prioriser les traits qui sont sélectionnés en fonction des gradients environnementaux, soit ceux qui sont moins reliés à la phylogénie et faire attention à la redondance des traits choisis (Van den Brink et al., 2011). La structure de la communauté basée sur les traits est stable dans les mêmes types d'habitats, ce qui n'est pas le cas de la structure de la communauté basée sur la taxonomie qui varie significativement avec les différences géologiques et altitudinales (Charvet et al. 2000; Menezes et al. 2010; Archaimbault et al. 2005). De plus le nombre de réplicats qui doivent être analysés arrive plus vite à un plateau quand on utilise les traits fonctionnels (Schmera et al. 2009; Bady et al. 2005). Finalement, selon Dolédec et al. (2006), les traits des espèces peuvent mieux identifier les caractéristiques biologiques liées au fonctionnement de l'écosystème pouvant permettre le développement d'actions de gestions ciblées.

Pour quantifier la diversité des communautés dans un espace fonctionnel multidimensionnel, Villéger *et al.* (2008) ont proposé une série de 3 indices : la richesse fonctionnelle (le volume de l'espace fonctionnel occupé par les espèces composant la communauté), l'équitabilité fonctionnelle (la régularité de la distribution de traits fonctionnels dans ce volume en fonction de l'abondance des espèces) et la divergence fonctionnelle (divergence dans la distribution des traits fonctionnels entre les espèces dans ce volume). Par contre, il a été montré que la richesse fonctionnelle est sensible aux valeurs extrêmes et ne peut intégrer l'information portant sur l'abondance relative des espèces. En conséquence, les espèces rares avec des valeurs de traits extrêmes vont provoquer l'inflation de la richesse fonctionnelle, ce qui peut être une propriété indésirable selon les applications. L'équitabilité et la divergence fonctionnelle tiennent compte de l'abondance relative, mais n'estiment pas la dispersion des espèces dans l'espace des traits comme la richesse fonctionnelle. C'est pourquoi un autre indice qui estime la dispersion fonctionnelle a été développé qui considère l'abondance relative des espèces (Laliberté et Legendre, 2010).

VI. Objectifs

Récemment, on a estimé les sources de variation de la biodiversité des communautés de zooplancton dans la région urbaine de Montréal (Mimouni *et al.,* 2015; Pinel-Alloul et Mimouni 2013) mais aucune étude s'est interessée aux communautés de macroinvertébrés benthiques.

Ce projet a pour objectifs 1) de déterminer l'importance de la variation temporelle et la variation spatiale des communautés de macroinvertébrés benthiques des étangs de la ville de Montréal sur la base de la composition taxonomique 2) de déterminer l'influence de différents types de variables (origine et aménagement, urbanisation, qualité des eaux, couvert de végétation, présence de poissons) sur la structure taxonomique et fonctionnelle des communautés de macroinvertébrés benthiques en milieu urbain et 3) d'évaluer la pertinence d'utiliser des métriques univariées basées sur les indices de diversité et des indices biotiques, ou des métriques multivariées basées sur la composition taxonomique et fonctionnelle des macroinvertébrés benthiques, pour le suivi environnemental des étangs urbains. Finalement, le dernier objectif est de comparer le potentiel de l'approche taxonomique classique ou de l'approche par traits fonctionnels pour le suivi de la qualité des plans d'eau en zone urbaine.

VII. Hypothèses

Notre étude aborde plusieurs questions et/ou hypothèses de recherche. Nous pensons que la variation spatiale (20 sites) des communautés de macroinvertébrés sera plus importante que la variation temporelle durant l'été (juin, juillet, août 2011) puisque le choix des sites est assez diversifié (incluant des milieux naturels, et des milieux artificiel permanents et temporaires), et qu'il n'y aura pas d'interaction de l'espace et du temps puisque les sites d'études sont assez rapprochés (sur la même île) et que le phénomène de succession saisonnière des peuplements est limité à cause des aménagements et de la vidange de certains plans d'eau en hiver. Nous pensons aussi que différentes variables environnementales (aménagement anthropique, morphométrie, qualité de l'eau et présence de poissons) auront des effets significatifs sur la structure des communautés de macroinvertébrés (probablement que les macrophytes, la présence de poissons qui ont un contrôle top-down et la permanence de l'eau seront les plus importantes si on se fie à la littérature sur les divers réseaux d'étangs en Europe). Les différents types de variables environnementales (aménagement anthropique, morphométrie, qualité de l'eau et présence de poissons) auront des niveaux d'effets différents; l'effet de la végétation et de la qualité de l'eau sera le plus important puisque les nutriments et les macrophytes sont reliés à la diversité des macroinvertébrés dans plusieurs types de milieux. L'aménagement des plans d'eau aura aussi une grande importante parce que certaines mesures sont assez drastiques (vidange, faucardage). On testera en particulier si les assemblages de macroinvertébrés seront différents entre les milieux naturels et artificiels, vidangés ou non, avec des macrophytes ou non, avec des poissons ou non, et selon le niveau trophique (Phosphore total, Chlorophylle-a). On évaluera les différences dans la richesse en espèces, la diversité taxonomique, les groupes dominants et la diversité fonctionnelle. Il y aura un lien significatif entre certaines variables environnementales (probablement les variables reliées au niveau d'eutrophisation comme le phosphore, la chlorophylle-a et la concentration en cyanobactéries) et les valeurs des indices de diversité et des indices biotiques. Nous nous attendons en particulier à une baisse des indices biotiques dans les milieux les plus eutrophes, les plus riches en cyanobactéries filamenteuses. Par contre, nous pensons qu'il y aura une augmentation de la diversité et des indices biotiques dans les milieux les plus riches en macrophytes. Certains indices de diversité ou biotiques se démarqueront comme étant les meilleurs indicateurs de la qualité environnementale des plans d'eau et de leur valeur pour le maintien de la biodiversité aquatique en milieu urbain.

VIII. Figures

Figure IA. Nombre et surface cumulative (km^2) au niveau mondial des lacs naturels ou construits par l'homme en fonction de leur taille (km^2) (Downing, 2010)

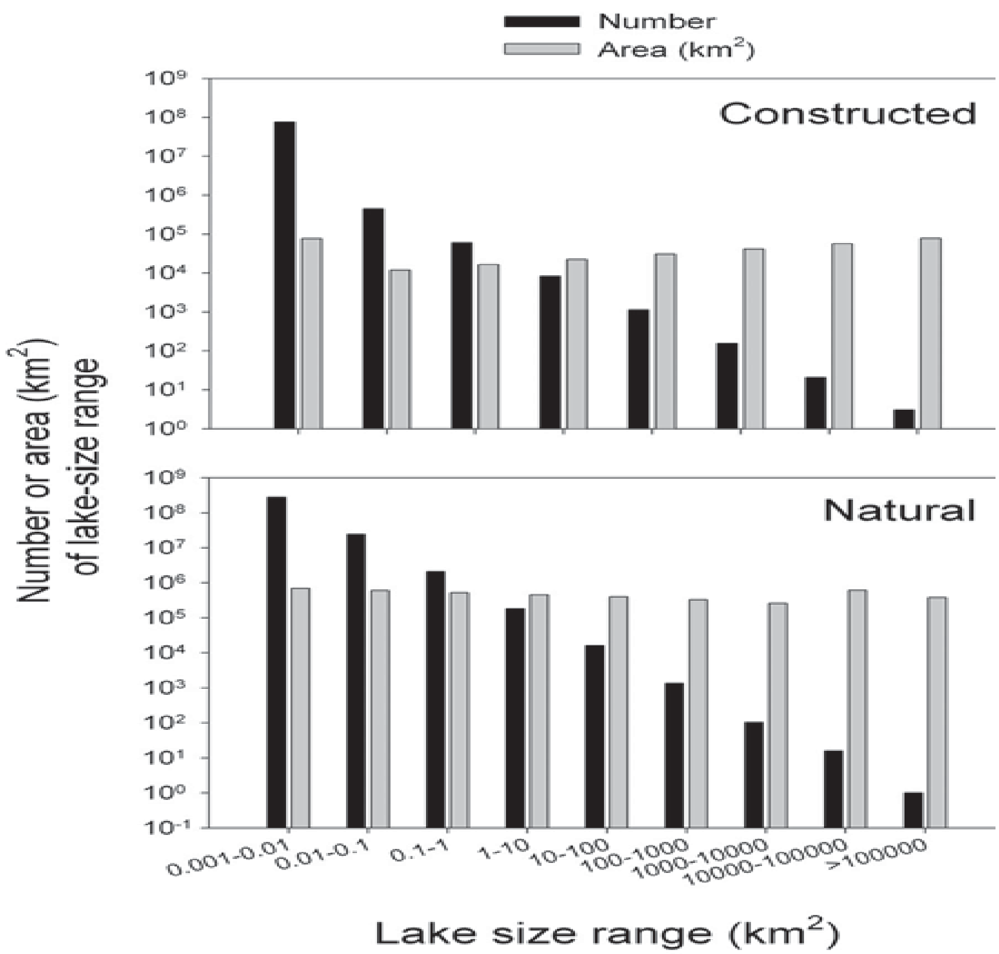
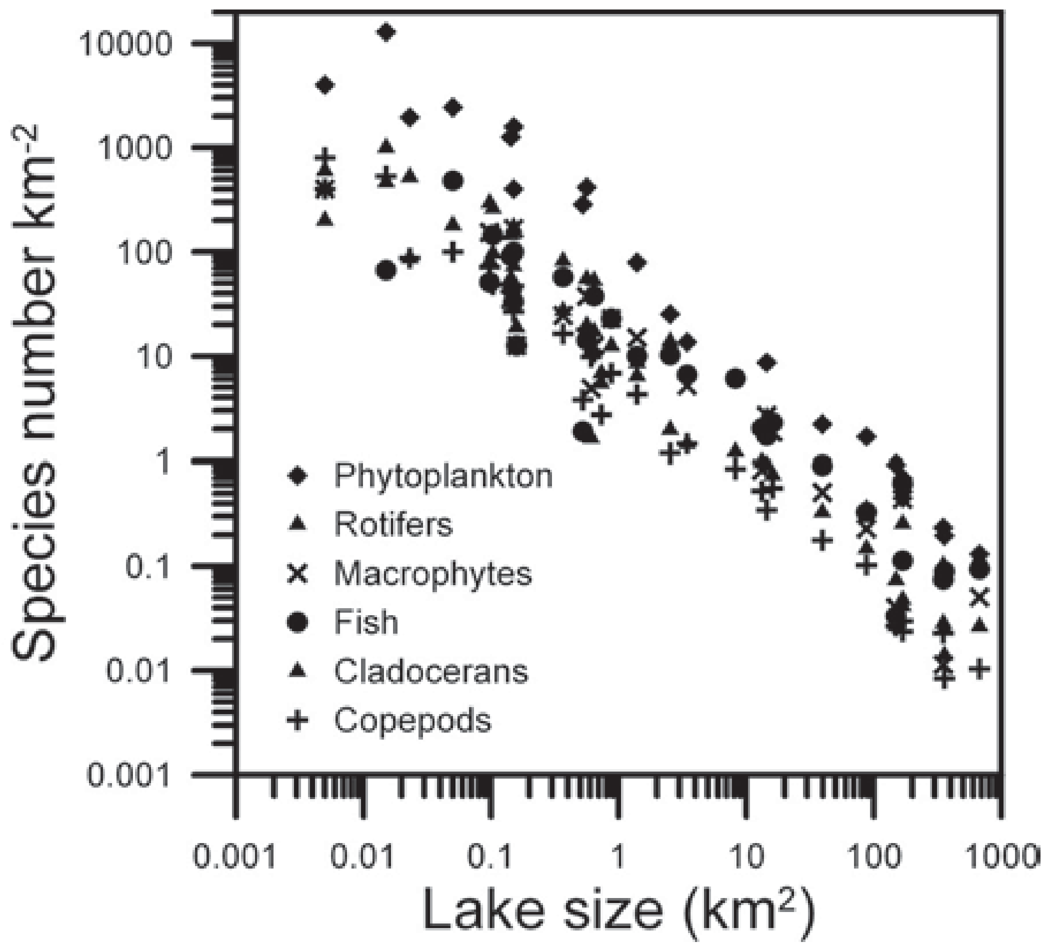


Figure IB. Nombre cumulatif d'espèces par km^2 de différents groupes taxonomiques en fonction de la superficie des lacs (Downing, 2010)



Chapitre 2 : Article

Assessment of biodiversity and ecological quality of urban ponds and lakes based on macroinvertebrate multimetric indices

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Abstract

Biological indicators, as benthic macroinvertebrates, are widely used for bioassessment of lakes and rivers, but more rarely for urban waterbodies. We used multiple metrics based on macroinvertebrates to evaluate the biodiversity and ecological quality of urban unstratified ponds and small lakes. We collected macroinvertebrates at three times (June, July, August) during summer 2011 in the littoral zone of 17 ponds and shallow lakes, and 3 marshes in the large metropolitan area of Montréal (Quebec, Canada). Four types of waterbodies sharing similar environmental features were identified using k-means partitioning. We selected univariate metrics based on taxa richness, taxonomic and functional diversity, and biotic indices, as well as multivariate metrics based on taxa assemblages and functional trait profiles to investigate spatio-temporal variation in macroinvertebrate communities. As temporal variation among months was low compared to spatial variation among waterbodies, we pooled summer data and determined spatial variation in macroinvertebrate metrics using cluster and principal component analyses. To determine the effects of urban land-use, management and environmental conditions, we constrained macroinvertebrate metrics with the environmental variables using redundancy analysis and considered the amount of variation they explained.

We found that artificial temporary ponds in municipal parks without fish and vegetation supported less diverse and abundant macroinvertebrates, and showed lower ecological quality, as indicated by low diversity, biotic and functional indices, than natural wetlands and artifical permanent ponds and lakes with fish and vegetation cover. However, macroinvertebrate communities were highly variable across waterbodies in the urban region and even within waterbody types. Environmental features explained between 44 and 46% of spatial patterns in macroinvertebrate taxa assemblages and functional traits. Local limnological conditions (vegetation cover, nutrients, organic content, and algal biomass) were the most important factors related to spatial change in macroinvertebrate metrics (21-28%). Pond management, urban density, and water permanence were also influencial factors (9-13%), but fish presence/absence had minor influence (1%). There were important interactions between environmental factors (0-7%). Some univariate metrics based on macroinvertebrate indices (number of taxa, number of taxa of Ephemeroptera and Trichoptera, Sannon's index and functionnal richness) also had the potential to assess the responses of macroinvertebrates to environmental features (such as macrophyte cover, different algae concentrations, Secchi transparency, etc) at the scale of the urban region of the city of Montreal (adjusted R^2 ranging from 0,17 to 0,77 for different metrics). The macroinvertebrate-environment model based on functional traits had a better explanatory potential than the model based on taxonomy but only by 2%. We discussed the implications of our study for management and quality assessment of urban ponds, and recommend to conserve various types of temporary and permanent waterbodies, these latter with littoral vegetation, and fish, to promote aquatic biodiversity in urban region.

Keywords: Macroinvertebrates, bioassessment metrics, biodiversity, taxonomic composition, functional traits, biotic indices, urban ponds and lakes.

Introduction

Urban ecology is a recent research field that has attracted much interest in the last decades because current projections of land-use suggest an important increase in the extent of urban areas at global scale posing a serious threat to biodiversity and ecosystem health (Grimm et al., 2000; Marzluff et al., 2008). Within the field of "urban ecology", we seek a better understanding of patterns in biodiversity, structure and function of communities in urban landscapes to assess their responses to environmental changes and human stressors (Pickett et al., 1997; McDonnell & Hahs, 2013). Urban waterbodies represent a set of habitats that have important ecological functions, as well as social and economic uses (Hassall, 2014). However, they may suffer greater biodiversity threats than terrestrial habitats due to catchment disturbances, increased nutrient loading, contamination, species invasion, residential and industrial land-use and mismanagement (Brönmark & Hansson, 2002; Hassall, 2014).

Small ponds and lakes (the lakes being stratified in opposition to ponds) account for a large proportion of inland waters at global scale (Downing et al., 2006). Small ponds and lakes (< 0.1 km^2) represent an estimated number of 3.2 billion waterbodies, and cover an area of 0.8 billion km^2 similar to the area occupied by large lakes (> 10 000 km^2). At regional scale, they form heterogeneous small habitats that sustain a greater diversity of plankton, plants, invertebrates and fish than more homogeneous large lakes (Downing, 2010). Thus, ponds and shallow lakes are now viewed as very interesting model systems to study conservation biology, ecology and evolution because they respond rapidly to anthropogenic disturbances and changes in environmental conditions and climate (De Meester et al., 2005; Jeppesen et al., 2014). Moreover, ponds represent an enormous diversity of habitats, which provide refuge and dispersion for freshwater organisms contributing greatly to regional diversity in urban and agricultural landscapes (Hassall, 2014; Céréghino et al. 2008b; Biggs et al., 2005). However, ponds are still a neglected component of research in limnology and ecology (Céréghino et al. 2008a). For all these reasons, ponds and small lakes are now recognized as priority habitats for monitoring and conservation in urban environments.

In Europe, bioassessment of surface water resources has been promoted by the Water Framework Directive (WFD: Irmer, 2000; ECOFRAME: Moss et al., 2003) in order to develop multimetric approaches for assessing ecological quality of running waters, lakes and ponds (Trigal et al. 2009; Gallardo et al. 2011). This framework has recently been applied for pond conservation in Britain, France and Switzerland, and helped defining good management to preserve and enhance biological diversity in peri-urban and agricultural landscapes (European Pond Conservation Network: Oertli et al., 2002 and 2005a; Williams et al., 2004; Biggs et al., 2005; Céréghino et al., 2008a). In comparison, bioassessment of freshwater ecosystems in urban regions of North America is still in infancy and needs further investigations (Marzluff et al., 2008; Purcell et al., 2009).

In Canada, macroinvertebrates have been widely used as biological indicators in national bioassessment programs (Rosenberg & Resh, 1993; Bailey et al., 2004; Environment Canada, 2008). They served for assessing the ecological integrity of coastal wetlands of the Great Lakes (Kashian & Burton, 2000), fluvial lakes of the St. Lawrence River (Pinel-Alloul et al., 1996; Tall et al., 2008; Masson et al., 2010; Tall et al., 2015), and the River Fraser (Reynoldson et al., 2001). In Europe, macroinvertebrates have been recognized as good indicators of ecological integrity and anthropogenic impacts in ponds in many agricultural, mountain and coastal regions (Fuentes-Rodriguez et al., 2013; Solimini et al., 2008; Oertli et al., 2005b; Della Bella et al., 2005; Cereghino et al., 2008b; Trigal et al., 2009). In North America, they also served to measure the impact of urbanization on streams (Purcell et al., 2009) but not yet on urban ponds and lakes.

In bioassessment studies, a multi-faceted approach combining univariate and multivariate metrics has been proved as the most informative method increasing the knowledge of the structure and function of macroinvertebrate communities and their responses to environmental gradients and anthropogenic disturbances (Gallardo et al., 2011; Melo et al., 2015). Traditional metrics such as diversity indices, taxa richness, and biotic indices based on sensitive or tolerant taxa were first used for measuring the effect of eutrophication, pollution and organic disturbances in streams, rivers, and lakes (Hilsenhoff, 1987 and 1988; Pinel-Alloul et al., 1996; Heino & Soininen, 2007; Masson et al. 2010; Lunde & Resh, 2012). However, these metrics provided little understanding on how environmental factors and anthropogenic disturbances respectively influence the structure and function of macroinvertebrates, and their ability to assess human impacts on ecosystems has been questioned (Mouillot et al., 2006). Consequently, alternative metrics based on taxa functional traits that consider not only taxa richness and dominance, but also their ecological function, trophic relationships or evolutionary relatedness have gained attention in bioassessment studies; they are now recognized as better indicators of ecological integrity of ecosystems (Dolédec & Statzner, 2008; Van den Brink et al., 2011). Functional traits are defined as taxa ecological attributes that reflect adaptation to their environments (Menezes et al., 2010). Thus, using a functional traits approach adds value to a traditional taxonomic approach because traits are universal descriptors throughout different types of communities, enabling comparisons at any spatial or temporal scale, even in different geographical regions (Baird et al., 2008). In

Europe, macroinvertebrate functional trait-based metrics have been used to evaluate ecological integrity of running waters (Archaimbault et al., 2010; Bady et al., 2005, Dolédec et al., 2006, Usseglio-Polaterra et al., 2001) but more rarely in lakes (Heino, 2008). However, despite the important effort devoted for integrating a functional approach into freshwater biomonitoring and ecological risk assessment (Van den Brink et al., 2011), further research is needed to improve the application of functional traits to assess health and integrity of aquatic ecosystems facing human disturbances such as urban ponds and lakes.

The main goals of this study are (i) to evaluate spatial (among waterbodies) and temporal (during summer) variations in littoral macroinvertebrate communities of 17 artificial ponds and shallow lakes and 3 marshes representing the common types of freshwater habitats found in the metropolitan island of the city of Montréal (Quebec, Canada), (ii) to test if different metrics based on macroinvertebrate diversity, taxa composition, functional traits or indices of tolerance to organic pollution are valuable tools for assessing biodiversity patterns and ecological quality of ponds and lakes in urban regions, (iii) to examine how macroinvertebrate diversity and biotic indices, taxa composition, and functional traits respond to multiple environmental factors related to urban land-use, pond features and management, and changes in pond environmental conditions, and finally (iv) to test which approach (taxa assemblages or functional traits) is the most powerful and useful for monitoring changes in environmental quality and biodiversity in urban freshwater ecosystems.

To structure the research, we defined four sets of questions and/or hypotheses: first (i), the selected metrics developed using macroinvertebrates will reflect changes in environmental features and ecological quality of ponds and lakes, and the multivariate metrics based on taxa assemblages and functional traits would be the most valuable tools, compared to the univariate metrics based on diversity and biotic indices, for assessing environmental changes among urban ponds and lakes as noticed in streams in agricultural regions (Dolédec et al., 2006) and large rivers (Bady et al., 2005). Secondly (ii), among the environmental variables, urban density, temporary or permanent status of ponds as well as vegetation cover and fish presence will be the most important drivers of spatial variation in macroinvertebrate metrics as observed in small ponds and shallow lakes in Europe (Fuentes-Rodriguez et al., 2013, Céréghino et al., 2008c, Gascón et al., 2008). Third (iii), we expect to observe less temporal variation during summer than spatial variation among waterbodies in macroinvertebrate communities due to high environmental heterogeneity among urban ponds as observed in streams (Stark & Phillips, 2009). Fourth (iv), we expect that the functional traits approach would have a better potential than the taxonomic one for biomonitoring urban aquatic systems. Finally, we will discuss the relevance and the implications of the study results for management and conservation of urban ponds and lakes.

Methods

Study sites and environmental features

The study was carried out during summer 2011 in 20 waterbodies distributed throughout the Island of Montréal (Québec, Canada) (45.46-45.69ºN; 73.50-73.90ºW) (Fig. 1). Montreal is the second largest urban area in Canada, located in the southwest of the province of Quebec at the confluence of the St. Lawrence and Ottawa Rivers. Its climate is humid continental or hemiboreal (Köppen-Geiger climate classification: McKnight and Hess 2000).

Over the last decades (1971-2000), air temperature was on average -8.9°C in winter and 22.3°C in summer, and mean precipitation was of 1062.5 mm per year (Environment Canada, 2012). The urban landscape of the Island of Montreal is typically dominated by residential and industrial zones and supports a population of 1 886 481 habitants in 2011 (City of Montréal, 2012).

Our survey included 17 artificial ponds and small lakes constructed during the last century in municipal parks and residential zones for water retention and recreation, and 3 marshes formed naturally by the hydrological network in large recreational parks. These waterbodies represent the normal typology of freshwater systems found in North America's large cities. They covered a large range of conditions represented by three sets of environmental features: (A) natural or artificial origin, permanent or temporary water, urban land-use and management, (B) vegetation cover, morphometry, water quality and algal biomass, and (C) fish presence/absence (Table 1). All studied sites were included in the water quality monitoring program of the city of Montreal (RMSA, Réseau de suivi du milieu aquatique: http://ville.montreal.qc.ca/). City managers provided us with data on residential density, pond origin and management, water quality (conductivity, concentrations of potassium (K), total phosphorus (TP), ammonium (NH_4), and total organic carbon (TOC)), and fish presence/absence. We completed the information provided by managers on fish presence/absence on site by visual observations, and coarse identification of fish captured with the kick net used for sampling macroinvertebrates. Maximum depth was estimated on site by probing at three points in the deepest zone in each waterbody. Transparency was measured with a Secchi disk two times per visit and averaged. Algal biomass and coarse composition were also estimated on site by measuring total chlorophyll biomass (Chl*a*) and the relative contribution of four spectral groups of algae (Greens, Blue-greens-Cyanobacteria, Diatoms, Crytophyta-Picocyanobacteria) using a fluoroprobe probe (Beutler et al., 2002).

We estimated qualitatively the vegetation cover using six classes based on the percentage of the pond area occupied by emergent, floating and submerged macrophytes (0: no vegetation, 1: 1-20%, 2: 20-40%, 3: 40-60%, 4: 60-80%, 5: 80-100%). Dominant macrophyte species were collected and identified using Fassett (2006).

Studied waterbodies in the urban island of Montréal meet the typology of urban ponds based on their primary function, as defined by Hassall (2014). Our survey included (i) ornamental lakes and ponds of small to medium size, with artificial substratum (concrete or polethylène membrane) constructed in municipal parks that are heavily managed for meeting aesthetic sensitivies of residents (Pratt1, Pratt2, Beaubien, Lafontaine, Liesse), (ii) ornamental lakes constructed in new residential districts (Battures, Brunante, Heritage, Lacoursière), which offer aquatic landscape to enhance the well-being of residents; some were subjected to vegetation removal or Asian carp introduction, (iii) artificial lakes in large municipal parks (Angrignon, Jarry, Lac des Castors, JBN, JBA, Cygne) and (iv) marshes in nature reserves (Bizard, Prairies) which provide natural and more protected recreational areas to the public, and finally (v) drainage and retention ponds (Marais des castors, RMontigny, Centenaire) to collect ground and pluvial waters. Together, these waterbodies represent contrasted conditions in water permanence, trophic status, vegetation cover, fish presence/absence, management, and disturbance levels, providing a large variety of habitats for macroinvertebrates and sustaining aquatic biodiversity in cities. In our survey, trophic states (Wetzel 2001) ranged from oligo-mesotrophic in temporary small ponds to hypertrophic conditions in marshes and retention ponds. Excessive inputs of minerals and nutrients due to urban land-use for infrastructure and roads are also characteristic disturbances of the waterbodies of the urban region of Montreal.

Sampling and analysis of macroinvertebrates

We sampled macroinvertebrates in the littoral zone because it provided more microhabitats than the profundal zone. On our first visit, we chose three littoral sites in each waterbody representing characteristic habitats at a depth less than 1m, and we georeferenced them using a portable GPS (Magellan RoadMate 1470). Since Boix et al. (2005) recommended 2 to 4 replicates for sampling macroinvertebrates in ponds with area smaller than 64 ha, we collected 3 replicates in each of our studied waterbodies which had areas < 12 ha. In total, we collected 9 samples in each waterbody during summer 2011. Macroinvertebrates were collected using a kick net (46 x 23 cm opening, 500-µm mesh size) that was dragged always by the same person over about 1.5 m of bottom trough sediment at a depth of ~ 2 cm and vegetation, if present. Sweep netting performed relatively well in ponds within dense stands of vegetation, with some limitation to capture mobile adult insects in heavily vegetated sites where a combination of pond netting and activity traps would yield a more complete estimate of taxa richness (Becerra Jurado et al., 2008; García-Criado & Trigal, 2005; Oertli et al., 2005b). On site, collected macroinvertebrates were extracted from sediment and vegetation and screened into 1-mm and 500-µm metal sieves. Both size fractions were pooled in a large bucket and preserved in 75% ethanol solution. Macroinvertebrates were then stained with rose Bengal to facilitate taxa sorting.

In the laboratory, all macroinvertebrates were sorted from the entire sample under a stereomicroscope Leica WILD M3B at 64X, 160X or 400X magnification, and counted according to taxonomic groups. Insects and molluscs were identified at the family level using Merritt et al. (2008) and Clarke (1981). Oligochaeta, Nematoda, Hydracarina, Nemertea, Planaria, Hydra and Hirudinea were identified to the order level using Moisan & Pelletier (2008). Previous studies showed that coarse resolution of macroinvertebrate composition at the family level is as efficient as a finer resolution at the genus level both for developing taxonomic and functional metrics for macroinvertebrates in large rivers (Gayraud et al., 2003, Haybach et al., 2004, Masson et al., 2010) and streams (Dolédec et al., 2000, Heino & Soininen, 2007).

Data analyses

Environmental data were transformed prior to analysis to reduce skewness. Geographic coordinates were transformed into Cartesian coordinates. Secchi transparency, depth, green algae, diatoms, total Chl*a* concentrations, and conductivity were ln-transformed whereas cryptophytes and blue-green algae concentrations were 4^{th} root transformed. pH, TOC, NH_4, K and TP concentrations could not be normalized by any transformation, but most of the tests used are robust enough to overcome this problem. We performed a principal components analysis (PCA) on the matrix of environmental data of sampling sites to determine the most reliable environmental variables discriminating groups of sites, i.e the ones that are longer than the circle of equilibrium contribution in scaling 1 (Borcard et al., 2011). A k-means partitioning based on the ln-transformed environmental data served to identify groups of sites with similar environmental features based on the local maximum of the Calinski-Harabasz criterion (Caliński & Harabasz, 1974).

We applied Hellinger's transformation to the macroinvertebrate abundance matrix, which contained many zeros, as recommended by Legendre & Gallagher (2001). To evaluate spatio-temporal variation in macroinvertebrate communities among waterbodies and among sampling months, we used the method (based on distance-based Moran's eigenvector maps) of Legendre et al. (2010) to test the space-time interaction in the absence of replication; we also determined the significance and the relative importance of temporal and spatial variations (Legendre & Legendre 2012).

To describe spatial patterns in macroinvertebrate biodiversity, we calculated α diversity (taxa richness in each waterbody), β diversity (change in taxa richness among waterbodies) and γ diversity (total number of taxa in the urban region) metrics. Alpha (α) diversity was calculated as the number of taxa found in each of the 59 samples. Beta (β) diversity was calculated as the variance in macroinvertebrate data matrix across sites (Legendre et al. 2005) and partitioned into local contributions of sites (LCBD) and taxa contributions (SCBD) by calculating the marginal sums of squares of the Hellinger-transformed data matrix (Legendre & Càceres, 2013). Gamma (γ) diversity in the urban region was estimated by accumulating the family-order taxa richness of all 59 samples (Gaston & Spicer, 2004). These biodiversity metrics were also calculated separately for the groups of sites with similar environmental features (waterbody types) determined by K-means cluster analysis.

To determine spatial variation in macroinvertebrate coarse community structure, we calculated a variety of univariate metrics based on richness and abundances of all taxa or of specific taxonomic groups known to respond differently to environmental disturbances (Mandeville, 2002). As biodiversity metrics, we selected the Shannon diversity index (Shannon, 1948), the Pielou's evenness index (Pielou, 1966) and the Simpson diversity index (Simpson, 1949). As metrics related to taxa tolerance to organic pollution, we chose the Family Biotic Index (FBI) developed by Hilsenhoff (1987, 1988), the percentage of Ephemeroptera and Trichoptera (%ET), the number of taxa of Ephemeroptera and Trichoptera (ET), the percentage of Chironomidae, Oligochaeta and Hirudinea (%COH) and the ratio of ET/COH abundances.

Multivariate metrics were derived from taxa composition and functional trait profiles of macroinvertebrates in each waterbody. Functional traits were established using the biological and ecological traits of freshwater benthic fauna in North America (Poff et al., 2006; Viera et al., 2006) and Europe (Usseglio-Polatera et al., 2000a, 2000b and 2001; Pinto et al., 2009; Tachet et al., 2010), and selected traits defined for macroinvertebrates of the St. Lawrence River Plain and Laurentian Great Lakes ecoregions (Desrosiers et al. 2015, unpublished; Poff et al., 2006). We used 17 functional traits adapted for the macroinvertebrate taxa found in our urban ponds and lakes; each trait was described by several modalities (2-5), summing a total of 53 modalities across the 17 traits (Table 2). They correspond to different patterns in life history (i.e. voltinism, life span, aquatic stages living in the water, ability to survive desiccation, ability to exit the system, general resilience; the last three were used especially for this study on ponds. We used a fuzzy coding procedure (Chevene et al., 1994) to describe the relative affinity of a given taxon for the different modalities of a given trait (1 point being distributed between the modalities of a given trait). Fuzzy coding was used to get a more representative trait based at the family level. We calculated three distance-based functional univariate metrics: functional richness, functional evenness (Villéger et al., 2008) and functional dispersion (Laliberté & Legendre, 2010). Functional richness (FRic) is the volume of functional space occupied by all taxa in the community; functional evenness (FEve) is the regularity of the distribution of taxa abundances in this volume. FRic is sensitive to extreme values and cannot integrate information on relative abundances of taxa whereas FEve considers relative abundances but does not estimate species dispersion in trait space. Consequently, the metric of functional dispersion (FDis) is proven more useful since it considers relative abundances of taxa and calculates their dispersion at the same time. Functional trait profiles for a given site were established by weighting taxa contribution for each modality of the traits based on their ln-transformed taxa abundances at the site. The sums of weighted scores (one per trait modality) were then expressed as relative abundance distribution of taxa (within a trait), giving the site trait profile (Usseglio-Polatera et al., 2000b and 2001; Lecerf et al., 2006).

Principal component analysis (PCA) was performed on the matrix of univariate metrics by sites (diversity, biotic and functional indices) to describe spatial variation in macroinvertebrate metrics among sites, and determined the most reliable indices as the ones that are longer than the circle of equilibrium contribution in scaling 1 (Borcard et al., 2011).

Finally, we applied the Indval method (Dufrêne & Legendre, 1997) on the matrices of taxa abundances and functional traits profiles by sites/months to determine the taxa and traits indicators of macroinvertebrate communities of the four groups of sites with similar environmental features representing different waterbody types.

To evaluate the relationships between macroinvertebrate metrics and the environmental features, we applied a redundancy analysis (RDA) on 2 data matrices: taxa assemblages by sites and functional trait profiles by sites. We performed variation partitioning on those two models (before stepwise selection which is not necessary before this procedure) based on the three sets of explanatory environmental variables (Table 1). We then used stepwise selection with the double stopping criterion (Blanchet et al., 2008) to select, among the environmental variables, the best drivers related to spatial variation in macroinvertebrate metrics. The relationships of each biotic index with retained environmental variables (after a stepwise selection procedure) were also tested using multiple regression analysis.

Statistical analyses were performed by the R software version 2.15.3 (R Development Core Team, 2013) with different functions from the packages *mice* (Van Buuren & Groothuis-Oudshoorn, 2011), *FD* (Laliberté & Shipley, 2011), *PCNM* (Legendre et al., 2012), and *vegan* (Oksanen et al., 2012). Also we used the codes from Borcard et al. (2011) for their cleanplot.pca function and the code from Legendre (2013) for their beta.div function.

Results

Environmental heterogeneity among waterbodies in the urban landscape

Waterbodies were distributed equally among low, medium and high urban residential areas (Table 1). Seven of the ponds were temporary, being emptied before winter and refilled in spring with municipal water while the other thirteen were permanent. Four ponds were treated with copper sulfate to control algae bloom, and in five others, aquatic vegetation was mechanically removed. Environmental features in each type of waterbodies with similar environmental features are presented in supplemental information (Table S1). Studied waterbodies covered a large range of size and trophic status (Table 1). On average, waterbodies were relatively small (mean area: 2.6 ha) and shallow (mean depth: 2 m). Surface area varied from 0.03 to 11.5 ha, perimeter from 0.07 to 2.5 km and maximum depth from 0.2 to 10 m. Secchi water transparency was 1 m on average but varied greatly from 0.2 to 4.4 m across waterbodies; in several shallow ponds, water transparency reached the maximum depth. Total chlorophyll biomass averaged 5 $\mu g \cdot L^{-1}$ and varied from 1.4 to 12 $\mu g \cdot L^{-1}$ reflecting a gradient in nutrient enrichment as shown by total phosphorus (8-260 $\mu g \cdot L^{-1}$). Green algae were the most abundant group followed by the diatoms and the blue-green algae (cyanobacteria). Water quality also varied among waterbodies (Table 1). Waters were generally alkaline with pH close or higher than 8, and well mineralized as indicated by high conductivity (mean: 444 $\mu S \cdot cm^{-1}$; maximum: 850 $\mu S \cdot cm^{-1}$). TOC, K and NH_4 concentrations were also high (means: 7 $mg \cdot L^{-1}$, 2600 $\mu g \cdot L^{-1}$ and 25 $\mu g \cdot L^{-1}$, respectively), and variable among waterbodies.

Aquatic vegetation was present in the littoral zone of fourteen waterbodies; macrophyte cover varied from low (10-40%) in four waterbodies, medium (40-60%) in three others to high (75-100%) in seven others including two marshes (Marais des castors, Prairies) totally covered by vegetation. Vegetation was composed of emergent, floating and submerged macrophytes (see Table S2, supplemental material, for macrophyte cover and dominant species in each waterbody). Four of the temporary ponds did not have fish; the others were inhabited by small fish, including indigenous planktivorous sticklebacks in two waterbodies (Bizzard, Lacoursière) and introduced Asian carps in one (Battures) (Table 1; Table S1).

PCA analysis on environmental features detected two major gradients (Fig. S1, supplemental material) and the first two axes explained 39% of the total environmental heterogeneity among waterbodies. Axis I (26%) reflected the productivity gradient with higher nutrient and algal concentrations in natural marshes. Axis 2 (13%) represented a gradient in size and depth, opposing the permanent and deeper ponds and lakes with fish and vegetation to the temporary and small ponds without fish, and generally without vegetation.

K-mean partitioning (Fig. S2, supplemental material) revealed 4 groups of sites having distinct environmental features, representing different types of waterbodies (Fig. 1, Table 3). Group 1 was composed of the 3 marshes: Bizard, Marais des castors, and Prairies. These permanent waterbodies of natural origin supported fish populations, were generally covered with extensive vegetation beds and had the highest concentrations of TP and TOC and the highest algal biomass. Group 2 was composed of Battures, Brunante, Centenaire, Cygnes, Héritage, Lacoursière and RMontigny. This group represented deep and permanent artificial waterbodies with high conductivity, relatively good transparency, and low to medium vegetation cover (except for Centenaire and RMontigny, which were turbid and without vegetation). They were located in residential districts, embedded in urban infrastructures, and had intermediate levels of nutrients and algal biomass. All of them supported fish and two (Brunante, Lacoursière) were subjected to macrophyte removal and Asian carp introduction. Group 3 was composed of Angrignon, JBA, JBN, Lac des castors and Lafontaine. They were artificial ponds of medium size located in large municipal parks, within a forested and natural environment. Generally, they had clear water and extensive vegetation cover, except for Lafontaine which has concrete walls. Three of them were permanent waterbodies while two were temporary (Lafontaine and Lac des castors). They were not very productive with low nutrient concentrations and algal biomass. Three of them were subjected to macrophyte removal and four were treated with copper sulphate to suppress algal growth. Group 4 was composed of Beaubien, Jarry, Liesse, Pratt1 and Pratt2. They were artificial temporary ponds of very small size (except Jarry), located in municipal parks, which were drained of water during winter. Some of them (Jarry, Liesse) supported vegetation beds while the others (Beaubien, Platt 1 and 2) with concrete walls or with bottom sediments covered by a polyethylene membrane did not have vegetation. More details on the environmental characteristics of each type of waterbodies are shown in supplemental material (Table S1).

Spatio-temporal variations in macroinvertebrate communities

Overall, a total of 68 macroinvertebrate taxa represented by 108 984 individuals were collected and counted across sites and sampling months (58 943 in June, 25 336 in July, 24705 in August). The space-time interaction test on macroinvertebrate data matrix (20 sites x 3 sampling months) indicated significant variation across space and time (P=0.001 after 999 permutations); however, the space-time interaction was not significant (P=0.42 after 999 permutations). Spatial variation among waterbodies was much more important (R^2=0.655) than temporal variation during summer (R^2=0.047). Thus, we only evaluated spatial variation in macroinvertebrate communities among waterbodies and used the whole summer data for further descriptions of biodiversity and community structure patterns, and for modeling relationships with environmental features. Raw data of total abundances, percentages, and occurrences of the 68 taxa collected in the 20 waterbodies are shown in supplemental material (Table S3A). Mean values of the abundances of major taxonomic groups in each waterbody and in the groups of sites with similar environmental features are also shown in supplemental material (Table S3B).

Macroinvertebrate diversity metrics

The accumulation curve based on all 59 macroinvertebrate samples collected during summer 2011 showed that gamma (γ) diversity in the urban area of Montréal reached 68 taxa (families or higher taxonomic groups) (Fig. 2). Most of the regional taxon pool of macroinvertebrates was reached with only 10 samples, indicating some homogenization and similarity in dominance patterns of macroinvertebrates across the urban region and waterbody types. When considering each group of sites with similar environmental features, γ diversity varied from 60 taxa in Group 3, 55 taxa in Group 2, 50 taxa in Group 1, and only 41 taxa in Group 4 (Table 4). For each group, most of the taxon pool was obtained with less than 10 samples, as for the whole set of waterbodies.

Overall, mean local (α) diversity (including rare taxa) during summer was 20 taxa but varied greatly from 8 to 42 taxa among waterbodies (Fig. 3A, Table S3A). Eight waterbodies (Bizzard, Marais des castors, Héritage, Lacoursière, Angrignon, JBA, JBN, Lac des castors) showed high taxa richness (> 25 taxa). Six waterbodies (Prairies, RMontigny, Battures, Brunante, Jarry, Liesse) had medium taxon richness (15-25 taxa) whereas five others (Centenaire, Cygnes, Lafontaine, Beaubien, Pratt2) showed low taxon richness (< 15 taxa). In average, sites from Groups 1 and 3 were more diversified (27-25 taxa) than sites from Groups 2 and 4 (12-18 taxa) (Table 4). However, local richness was not clearly linked to groups of sites with similar environmental features, due to high variation in taxon richness within several groups (Groups 1-3) (Fig. 3A).

Total beta (β) diversity across the studied waterbodies was 0.40 with each group of sites accounting for 18 to 32 % of the total β diversity across the urban region (Table 4). On average, Group 2 including 7 sites showed the highest β diversity whereas Group 1 with only 3 sites had the lowest β diversity. Local contribution of sites to β diversity (LCBD) was also variable across the urban region (between < 1% to 4%: Fig. 3B). Six waterbodies (Prairies, RMontigny, Battures, Lafontaine, Beaubien, Pratt2) contributed highly to β diversity, whereas 4 others (Brunante, Cygnes, JBA, Lac des castors) contributed the least to β diversity; the others shared intermediate contributions. On average, sites from Groups 2 and 3 showed lower contributions to β diversity than sites from Groups 1 and 4 (Table 4).

There was no correlation (p-value < 0.05) between local taxon richness (α diversity) and site contributions (LCBD) to β diversity because sites having the highest local taxon richness (Fig. 3A) were not the ones contributing the most to β diversity (Fig. 3B). For instance, some sites (Prairies in Group 1, Lafontaine in Group 3 and Beaubien and Pratt2 in Group 4) had high contributions to β diversity (LCBD) while having low α diversity. In contrast, some sites had high α diversity (Bizard, Marais des castors in Group 1, Héritage and Lacoursière in Group 2, Angrignon, JBA, JBN, Lac des castors in Group 3) but did not contribute much to β diversity.

Macroinvertebrate community structure and taxa contribution to beta diversity

Macroinvertebrate communities were dominated by two major groups, the Diptera (mainly Chironomidae) and the Oligochaeta accounting respectively for 37% and 22% of total abundance (Fig. 4; Table S3-A). Other important groups (3-6% of total abundance) were the Nematoda, the Mollusca Pulmonata (Planorbidae), the Ephemeroptera (Caenidae), the Ostracoda, and the Hydracarina (as others). Groups contributing between 1-3% of total abundance were the Odonata (Coenagrionidae), the Mollusca Pulmonata (Lymnaeidae, Physidae) and Prosobrancha (Bithynidae), the Amphipoda (Talitridae/Doglienotidae), the Ephemeroptera (Baetidae), and the Hemiptera (Corixidae, Pleidae). We observed important variation in macroinvertebrate composition among waterbodies but no clear relationship with waterbody types (Fig. 4; Table S3-B). Diptera Chironomidae accounted for almost the total number of organisms found in artificial and temporary small ponds of municipal parks (Pratt1, Pratt2: Group 4). They also accounted for the majority of taxa in eutrophic permanent lakes (Battures, Brunante and Centenaire: Group 2) and in eutrophic marshes (Bizzard, Prairies: Group 1). Oligochaeta were the most abundant taxa in very disturbed waterbodies as the retention reservoir Montigny. Pulmonata Planorbidae were very abundant in Prairies and Lacoursière with extensive submerged vegetation, but also on the concrete walls of Lafontaine.

Overall, macroinvertebrate taxa contributing the most to beta diversity (SCBD) were the Annelida (Oligochaeta), Mollusca Pulmonata (Physidae, Planorbidae, Lymnaeidae), Diptera (Chironomidae, Ceratopogonidae), Hemiptera (Corixidae, Pleidae), Ephemeroptera (Caenidae, Baetidae), other Mollusca Prosobrancha (Bithyniidae), Nematoda, Odonata

(Coenagrionidae), Amphipoda (Talitridae/Doglienotidae), Trichoptera (Hydroptilidae), and others (Hydracarina, Ostracoda) (Fig. 5; Table S3-A).

Indicator taxa for each group of sites are presented in Table 4. The marshes (Group 1) were the most diverse with several indicator taxa of aquatic Coleoptera (Curculionidae, Haliplidae), Hemiptera (Belostomatidae, Mesoveliidae, Pleidae), Diptera (Stratiomyidae, Ceratopogonidae), Trichoptera (Phrygaenidae), and Lepidoptera (Pyralidae). Group 1 was also characterized by indicator taxa of Mollusca Gastropoda Pulmonata (Lymnaeidae, Planorbidae) and Bivalva (Sphaeridae). Mollusca Gastropoda Pulmonata (Ancylidae) was the only indicator taxon for the sites of Group 2, being abundant in Brunante, Heritage, and RMontigny. Group 3 was characterized by sensitive aquatic insects as the Trichoptera (Leptoceridae, Hydroptilidae), the Ephemeroptera (Caenidae), and the Ostracoda. Finally, we could not distinguish indicator taxa for the Group 4 due to low richness of common taxa (Chironomidae).

Macroinvertebrate univariate metrics

Raw data of univariate diversity metrics in each waterbody are presented in Table S4 (Supplemental material). Shannon diversity index was on average 2.43 (± 0.72 standard deviation), but highly variable (range: 0.65-3.44); Pielou evenness index averaged 0.59 (± 0.11 standard deviation) and ranged from 0.35 to 0.73; Simpson diversity index was 0.69 (± 0.17 standard deviation) in average and ranged from 0.25 to 0.87. In general, Shannon and Simpson diversity indices and Pielou's evenness were the highest in Group 1, intermediate in Groups 2 and 3, and the lowest in Group 4 (Table 4).

There is no clear distinction in other biotic indices between groups of sites with distinct environmental features, due to important variation within each group (Table S4, supplemental material). Total abundance of macroinvertebrates and total taxa richness tended to be higher in Groups 1 and 3 than in Groups 2 and 4 (Table 4). Generally, there were less than 4 taxa of Ephemeroptera and Trichoptera (ET) in all sites and groups; however, ET percentage increased in Groups 3 and 4. Group 2 sites showed the highest percentage of worms (COH), followed by Groups 4 and 3. ET/COH ratios were higher in Groups 3 and 4 than in Groups 1 and 2. Finally FBI index was generally around 7 in all sites and groups (Table 4).

Raw data on the functional diversity metrics in each waterbody are presented in Table S4 (Supplemental material). On average, functional richness (FRic) was 1.20, functional evenness (FEve) 0.54, and functional dispersion (FDis) 0.26. FRic was 50% higher in Groups 1, 2, and 3 (1.21-1.38) than in Group 4 (0.88). In opposite, FEve was slightly higher in Group 4 (0.62) and Group 2 (0.56) than in Groups 1 and 3 (0.46-0.51). Finally FDis was slightly lower in Group 4 (0.22) than in Groups 1-3 (0.26-0.30). Differences observed in Group 4 were mostly due to lower FRic and FDis, and higher FEve in two temporary waterbodies (Platt 1 and 2) than in the other waterbodies, which shared relatively similar values (Table S4).

The first two axes of the PCA analysis based on the univariate metrics of each site (Fig. S3, supplemental material) explained 70% of total variation in the metrics. Along PC1 Axis (42% of explained variation), we detected an inverse gradient between Shannon and Simpson diversity, and functional dispersion (FDis) (negative side), and FBI index and dominance of COH (positive side). Total taxa richness, and ET taxa richness were positively correlated with functional richness (FRic) and inversely related with functional evenness

(FEve), the percentage of ET, and the ratio ET/COH along PC2 (28% of explained variation). It is also interesting to note that functional evenness was not related to Pielou evenness.

Significant indicators of functional traits (p-value <0.05) for the 4 groups of waterbodies are presented in Table 4. Although it was difficult to directly relate specific functional traits with groups of sites, PCA analyses of the matrix of macroinvertebrate taxa and functional traits (Figure S4, supplemental material) enabled us to discriminate 4 main groups of taxa associated with specific functional traits. First, insect larvae with long life span (life3), aerial respiration (Resp3), ability to survive desiccation (Desi1), ability to exit the system (Exit1), and overall good resilience (Res1) were represented by Coleoptera (Dytiscidae) and Hemiptera (Belostomatidae, Corixidae, Gerridae, Velidae). Secondly, mollusc gastropods were represented by well armoured pulmonates (Planorbidae, Physidae, Valvaltidae) or prosobranchs (Bithynidae, Viviparidae) that are multivoltine (Volt3) and well armoured (Arm4). Finally, insect larvae of Ephemeroptera (Baetidae, Caenidae), Trichoptera (Leptoceridae, Hydroptilidae), Lepidoptera (Pyralidae), Odonata (Caenogrionidae, Libellulidae, Aeshnidae, Corduliidae), and Diptera (Ceratopogonidae) formed the third group of collector-gatherer (Trop1) breathing with gills (Resp2) and poorly armoured (Arm2) organisms that have less resilience (Res0), and do not survive dessication (Des0).

Relationships between macroinvertebrate metrics and environmental features

In order to compare the performance of different approaches, we developed RDA models between environmental features and macroinvertebrates based on multivariate metrics (taxonomic assemblages and functional traits profiles). The biplots (environment–metrics) of the 2 RDA are presented in Figures 6 and 7. We tested the simple and combined effects of the three sets of environmental variables (A: Origin, urban land-use, management; B: vegetation cover, morphometry, water quality and algal biomass; C: fish presence/absence) using variance partitioning modelling (Table 5).

Macroinvertebrate taxonomic assemblages – environment RDA model

Environmental features explained 44% of variation in macroinvertebrate taxonomic composition among waterbodies (RDA model: $R^2 = 0.439$) (Fig. 6, Table 5). The most important variables kept in the model were macrophyte cover, urban residential density, TOC, fish presence, NH_4 and K concentrations, Secchi transparency, and Emptying, which represented the temporary small ponds emptied during winter. This model detected two major environmental gradients along the RDA axes. Axis 1 (28%) reflected the opposition between temporary ponds without fish and vegetation (left side), which supported low populations of Diptera (Chironomidae) and Ephemeroptera (Baetidae), and the permanent productive waterbodies, with fish, low to medium vegetation cover, which supported an abundant population of Diptera and were the richest in Oligochaeta (right side). Axis 2 (23%) reflected the vegetation cover gradient, associated with high abundance of Mollusca Pulmonata (Planorbidae), and Amphipoda in waterbodies totally covered with submerged vegetation (lower side).

Variation partitioning of the RDA model based on taxonomic composition showed that spatial heterogeneity in local limnological conditions (B: vegetation cover, morphometry, water quality and algal biomass) accounted for 21% of the explained variation (Table 5).

Differences in pond origin and management, and in urban residential density (A) accounted for another 9%. Fish presence/absence (C) had minor effect (1%) but was strongly interrelated with A and B. Combined effects between two sets of environmental features ranged from 3 to 7%, and were only 1% for the 3 sets of environmental variables.

Macroinvertebrate functional traits – environment RDA model

Environmental features explained 46% of variation in macroinvertebrate functional trait profiles among waterbodies (RDA model: $R^2 = 0.455$) (Fig. 7). The most important variables kept in the model were macrophyte cover, fish presence, urban residential use, K and Chl*a* concentrations, and copper sulphate treatment. This model detected also two major gradients along the RDA axes. Axis 1 (29%) reflected the inverse gradient associated with fish presence, high total organic carbon, and high algal biomass (positive side), and with K concentrations (negative side). Axis 2 (14%) was indicative of the vegetation cover gradient. These gradients were associated with different functional traits. Axis 1 opposed taxa not armored (Arm1), with short life (Life1), and small size (Size1) in fish-less small ponds (Diptera Chironomidae, Ephemeroptera, Fig. S4) to large or medium size taxa (size 2-3), well armored (Arm4), with long-life (life3), and good adherence (Habi2) in fish-inhabited productive waterbodies (Mollusca Gastropoda, Fig. S4). Along Axis 2, we found macroinvertebrate taxa with short life (Life2), carnivore feeding (Trop4) having low resilience (Res0) and dispersal potentials (Disp0), and weak abilities to exit the systems (Exit0) (Trichoptera, Odonata and Ephemeroptera) opposed to organisms able to exit the systems (Exit1) (Heteroptera and Coleptera), Fig. S4).

Variation partitioning of the RDA model based on functional traits showed that spatial heterogeneity in local limnological conditions (B: vegetation cover, morphometry, water quality and algal biomass) accounted for 28% of the explained variation in macroinvertebrate functional traits among waterbodies (Table 5). Differences in pond origin and management, and urban residential density (A) accounted for another 13% while fish presence/absence (C) had minor effect (1%). The variation explained by two sets of explanatory variables ranged from 0 to 3%, and 7% of the variation was explained by the combined 3 types of environmental variables.

Macroinvertebrate univariate metrics- environment models

The multiple linear regressions between the different univariate metrics and the selected environmental factors had multiple $R^2 > 0.50$ for four metrics. The adjusted R^2 ranged from 0.17 (Pielou's index) to 0.77 (number of taxa). The other good models were found for the Shannon index (adjusted R^2=0.55), Fric (adjusted R^2=0.67), number of taxa ET (adjusted R^2=0.68). For these four models, macrophyte cover was always included in the model. Other variables differed between the models, but most of them had a link with water enrichment (Secchi transparency, chlorophyll-a concentration, bluegreen algae concentration, P, TOC, etc). Management variables were also included in three of the models but at the rate of one per model. For example, the origin was included in the model for Shannon index, macrophytes removal in the model for Fric, and copper sulfate treatment in the model for the number of taxa ET.

Discussion

Studies comparing the aquatic biodiversity of different types of waterbodies in urban regions of North America are still very scarce, in comparison to Europe (Hassall 2014; Biggs et al. 2005). A recent study on biodiversity patterns of zooplankton in urban waterbodies of the region of Montreal (Québec) showed that they could sustain as much diversity of microorganisms as reference lakes and ponds in unperturbed regions (Pinel-Alloul & Mimouni, 2013; Mimouni et al. 2015). However, studies have not yet addressed macroinvertebrate biodiversity and community structure within urban regions in North America. This study is the first to investigate the potential of waterbodies of the urban region of Montréal (Québec) to sustain aquatic biodiversity using macroinvertebrate taxonomic and functional multimetrics.

Macroinvertebrate biodiversity patterns

Our study supports the new paradigm, which recognizes the important role of small ponds and lakes for sustaining aquatic biodiversity in urban landscapes, as seen in periurban and agricultural regions in Europe. Montreal's waterbodies sustained high diversity of macroinvertebrates at the regional scale. Gamma (γ) diversity at the family-order level (68 taxa) reported in our survey equals that of macroinvertebrates in farm ponds of agricultural regions of Spain (68 family taxa) (Fuentes-Rodríguez et al. 2013) and France (52 family-genus taxa) (Céréghino et al. 2008b). It was also in the same range as that estimated when using higher taxonomic resolution (family-genus-species level) in field ponds (144 taxa) and garden ponds (44 taxa) along an urban–rural gradient surrounding the town of Loughborough in England (Hill & Wood, 2014). We acknowledge that our estimate of the regional pool may be underestimated when using family-order level. If taxa were identified at the genus-species level, regional taxa pool in urban waterbodies of Montreal might have reached one hundred taxa, as found in field ponds in England (Hill & Wood, 2014).

Accumulation curve of macroinvertebrate taxa across space and time reached a plateau close to 60 taxa after 20 samples, and to 50 taxa with only 10 samples. This finding points out that our survey encompassed most of the regional diversity accounted by the dominant macroinvertebrate families commonly found in the studied waterbodies. However, taxa with low occurrence will be recorded only with a complete survey in all types of waterbodies.

Mean local (α) diversity (20 taxa) and its range of variation (8-42 taxa) in Montreal's urban waterbodies were comparable to taxon richness (mean: 20 taxa; range: 7-26 taxa) estimated in ephemeral waterbodies with submerged and emergent vegetation in agricultural karstic regions of Ireland (Porst & Irvine 2009). However, local diversity in Montreal's waterbodies was higher than that reported in urban ponds of poor ecological quality in northern England (range: 4-13 taxa: Noble & Hassall 2014) and in pristine alpine ponds in Switzerland (mean 11 taxa; range: 6-24: Oertli et al. 2008). As observed in urban ponds of northern England (Noble & Hassall 2015), temporary small ponds of city parks, with artificial substratum (concrete walls), and without vegetation and fish, had the least taxa. In contrast, permanent and more productive marshes and lakes with natural substratum, extensive vegetation beds and fish, exhibited the highest local taxon richness. Fuentes-Rodriguez et al. (2013) also found that natural and artificial substratum ponds in agricultural area of Andalusia

(Spain) differed significantly in α diversity with the lowest diversity being observed in ponds lined with polyethylene lined or with concrete bottom.

There is not yet any estimate of β diversity in urban waterbodies. Our first estimate of 0.40 in Montreal's waterbodies is lower than values (0.65-0.68) reported in farm ponds of Andalusia and temporary ponds in wetlands of Doñana National Park in Spain (Fuentes-Rodriguez et al., 2013; Florencio et al., 2014). However, because these studies are in different continents, it is premature to conclude that this difference is due to an higher homogenization of taxonomic assemblages in urban than in agricultural regions, Site contribution to β diversity was highly variable (< 1% to 4%) among Montreal urban waterbody types. Temporary ponds in municipal parks with artificial substratum (Lafontaine, Beaubien, Pratt2) and one wetland fully covered with vegetation (Prairies) showed higher contributions to β diversity than permanent lakes (mainly in Groups 2 and 3) and marshes (Bizzard, Marais des castors) with natural substratum. Similar observation was made in Mediterranean farm ponds (Fuentes-Rodríguez et al., 2013).

The lack of agreement between taxon richness (α diversity) and contributions to β diversity was already observed when assessing biodiversity patterns of zooplankton communities within the same set of urban waterbodies (Mimouni et al. 2015). High contribution to β diversity (LCDB) in some sites could be due to the dominance of one specific taxon (Pulmonata in Prairies, Oligochaeta in RMontigny, Diptera in Pratt 1-2) or to the presence of rare taxa (Ostracoda and/or Amphipoda in Lafontaine, Beaubien and Jarry). In sites with low LCBD, communities are more diversified (high α diversity) and composed of common taxa (Bizard, MaraisCastors). On one hand, this observation highlights the conservation value of the species-poor temporary ponds in municipal parks, which contributed to increase the total diversity across the urban landscape. Being emptied during winter and refilled at spring, they provided new habitat to be colonized by insects with good dispersal capacities, such as Diptera and Ephemeroptera, which emerge as flying adults, reproduce and deposit eggs in ponds at spring. On the other hand, species-rich permanent lakes and marshes also deserve conservation priority because they contributed the most to the regional pool of taxa (γ diversity). Usually, permanent ponds were deep and well covered by vegetation providing diversified and food-rich habitats, which sustained a perennial and diverse macroinvertebrate fauna.

Dominance patterns in macroinvertebrate community structure

Diptera (Chironomidae) and Oligochaeta were numerically dominant in most Montreal urban ponds, as previously observed in European ponds (Solimini et al. 2005; Oertli et al. 2008; Porst & Irvine 2009). Oligochaeta, which are tolerant of organic pollution, were dominant in the highly degraded retention reservoir of Montigny with turbid (Secchi: 0.48 m) and enriched (TP: 111 $\mu g.L^{-1}$) water. Diptera (Chironomidae), another tolerant taxon, accounted for more than 50% of the total macroinvertebrate abundance in the most eutrophic or degraded waterbodies of residential areas (Brunante, Battures, Centenaire), and in emptied temporary small ponds (Pratt1, Pratt2). Oligochaetes and chironomids are gathering collectors, which feed on organic matter that accumulates in sediments both in vegetated and unvegetated habitats (Solimini et al., 2005 and 2008). Only few taxa were differentially dominant in some of our waterbodies. For example, Gastropoda Plumonata (Planorbidae, Lymnaeidae) were large-sized scrapers dominant in Prairies, a waterbody not well oxygenated and totally covered by submerged macrophytes (*Ceratophyllum demersum*), likely because they can breathe at the water surface and feed on periphyton. More sensitive gastropods (Prosobranchia: Bithynidae), with branchial respiration, were encountered in higher abundance on floating and submerged plants that were covered with periphyton in Lacoursière with artificial aeration. In Lafontaine, the absence of fish and high water transparency likely allowed gastropods to colonize the concrete walls covered with periphyton. Spatial variation among waterbodies in the abundance of these major macroinvertebrate taxa accounted for changes in community composition (β diversity) across Montreal urban region. The macroinvertebrate fauna of marshes (Group 1) was characterized by several indicator taxa of insect larvae, and molluscs, but was more homogenous, as indicated by lower contribution to β diversity. In contrast, the three other groups of waterbodies did not support numerous indicator taxa because they were less homogeneous in community structure. Permanent ponds of recreational parks (Group 3) were characterized by the presence, albeit in small number, of sensitive taxa (Trichoptera and Ephemeroptera) indicating good environmental quality.

Macroinvertebrate metrics' performance

The selection of appropriate univariate metrics to assess ecological integrity of aquatic ecosystems is difficult because their performance varies considerably in different waterbodies (Gallardo et al., 2011). The ability of traditional metrics, as species richness and diversity indices (Heino et al., 2007), to assess human impacts is not clear because they cannot discriminate between natural and human related stressors (Mouillot et al., 2006). Thus, alternative methods based on functional diversity that consider not only number and dominance of taxa but also their ecological function have gained attention (Dolédec & Stazner 2008). Our study indicated that univariate taxonomic and functional indices are redundant and complementary for discriminating types of urban waterbodies in Montreal. Indeed, taxonomic (taxon richness, Shannon/Simpson indices) and functional (FRic, FDis) diversity metrics were positively related, as reported in streams and rivers (Heino, 2008). They also exhibited inverse relationships with biotic indices based on taxon tolerance to organic pollution (FBI, COH). In general, natural marshes (Group 1) and less enriched permanent ponds in recreational parks (Group 3) supported macroinvertebrate communities with higher abundance and higher taxa diversity than permanent waterbodies in residential areas (Group 2) and managed temporary ponds (group 4). Accordingly with values of the family-level biotic index (FBI) (Hilsenhoff 1988), which were always close or higher than 7, most waterbodies could be considered in relatively poor to very poor conditions. This may be due to important inputs of organic matter and nutrients from the urban landscape, as indicated by the rarity of very sensitive taxa (Plecoptera, Ephemeroptera, and Trichoptera). However, the FBI index was developed for streams and its appropriateness is questionable when biomonitoring urban lentic waterbodies where these taxa are naturally infrequent. Functional diversity indices also ranked waterbodies along a gradient of decreasing ecological quality. Overall and whatever the univariate metric used, natural marshes can be considered of good quality, permanent artificial lakes in municipal and recreational parks of intermediate quality and temporary artificial small ponds of poor quality, as observed in urban ponds of northern England (Noble & Hassall 2015). Although univariate metrics responded to gradient in human and environmental stressors, the multiple regression models showed that they had the lowest performance for biomonitoring urban systems.

Macroinvertebrate communities in Montréal urban waterbodies were also characterized with multivariate metrics based on taxonomic composition or functional traits. Recently, these approaches were used to assess spatial patterns of macroinvertebrate communities in lakes, rivers, and streams (Archaimbault et al. 2010; Heino et al., 2013, Dolédec et al., 2000 and 2006), but rarely on small ponds (Céréghino et al. 2008c). Our study is the first to contrast taxonomic and functional responses of littoral macroinvertebrates across urban waterbodies in North America. Our study supports the hypothesis that taxonomic and functional multivariate metrics have better performance than univariate metrics for biomonitoring macroinvertebrate communities across urban waterbodies. RDA models based on taxonomic composition and functional traits had adjusted R^2 of 44-46% respectively. Multiple regression models based on univariate metrics had adjusted R^2 of 17-77%, but only four metrics were useful ($R^2 > 0,5$). Furthermore, the new approach based on functional traits was slightly more powerful (46%) than the taxonomic approach (44%). However, there are still important limitations for the achievement of accurate estimation of functional traits of macroinvertebrates for biomonitoring urban waterbodies (Solimini et al., 2008). Our study gave a preliminary assessment of contrasted functional profiles based on life history, mobility, morphological and ecological traits that could be useful for future assessment of ecological quality of urban waterbodies but further progress should be made on the definition of macroinvertebrate functional traits in human-stressed waterbodies.

Environmental control of macroinvertebrate community in urban landscape

Ponds are an important biodiversity resource both in natural and human-impacted landscapes. Although considerable progress has been made in characterizing pond ecosystem and biota in Europe (Biggs et al. 2005), little is known on how they are affected by environmental and human factors. In our study, we assessed the influence of three sets of factors related to pond environmental characteristics, urban land-use, and management practices. We found that local limnological conditions related to vegetation cover, organic and nutrient enrichment, water transparency, and algal abundance were the most important environmental factors explaining spatial variation in macroinvertebrate communities in Montreal's urban waterbodies, whatever the metrics used. They accounted for 21 to 28% of explained variation in macroinvertebrate multivariate metrics. Stepwise selection procedure selected macrophyte cover as one of the most important variables associated with high enrichment in organic carbon and algal biomass. Increased macrophyte cover in the most enriched marshes and residential lakes was associated with high taxon richness and high functional diversity and dispersion of macroinvertebrates. In contrast, unvegetated artificial small ponds showed low taxon richness and functional diversity. The important role played by aquatic vegetation cover in sustaining macroinvertebrate taxon richness and functional diversity has been pointed out in disturbed and protected regions in Europe. In protected coastal regions of Italy, vegetated habitats in temporary and permanent ponds displayed the highest species richness of Coleoptera, Ephemeroptera, Odonata and Hemiptera by providing structural complexity, food resources, and refuge against predation (Della Bella et al. 2005). In agricultural regions, increase in vegetation cover in farm ponds was coupled with a rise in taxon richness (Bazzanti et al. 2010; Fuentes-Rodriguez et al. 2013). Moreover, the structure and age of the vegetation play a role in selecting biological traits of macroinvertebrates favouring long-live, large size univoltine taxa (Céréghino et al. 2008a). Our study also suggests that high vegetation cover in urban ponds is related with high diversity and abundance of long-lived insect larvae and of large size molluscs. The highest abundance of molluscs was observed in marshes with maximum vegetation cover. However, it is difficult to disentangle the relative effects of increasing inputs of nutrient and organic matter from macrophyte cover because they were all related. In permanent ponds at different altitudinal vegetation belts (foothill, montane, subalpine, alpine) of Switzerland, excessive nutrient levels in hypertrophic ponds resulted in decreasing taxon richness, especially of sensitive taxa of Coleoptera, and changes in biological/ecological traits of macroinvertebrates (Oertli et al., 2008).

Pond origin, urban density and management practices also influence significantly the macroinvertebrate communities in urban ponds. Our study showed that management practices, such as winter draining or fish removal, are the main characteristics linked with the low diversity and density of macroinvertebrates in small artificial ponds of city parks. In fish ponds of Czech Republic, sediment removal resulted in a significant decrease in both abundance and diversity richness of macroinvertebrates (Sychra & Adamek 2011). Mayfly larvae were the dominant invertebrates before restoration and sediment removal, while chironomid larvae and oligochaetes dominated after sediment removal, as observed in our heavily managed small ponds of city parks. In Mediterranean ponds, water permanence and hydroperiod length have strong effect on macroinvertebrate communities (Della Bella et al., 2005; Gascon et al. 2008). In Poland managed ponds stocked with common carp, macroinvertebrate biodiversity and abundance were the greatest in ponds with small-size carps and positively related with emergent macrophyte cover. The presence of small carps and vegetation in ponds favored the mollusc gastropods, herbivorous Coleoptera, and predatory Odonata (Nieoczym & Kloskowski 2015). In accordance to those findings, in our study, the residential pond (Brunante) containing introduced Asian carps had high water turbidity and relatively low taxon richness with dominance of Diptera Chironomidae and Oligochaeta.

Relevance and implications for waterbodies management in urban region

Although recent research, including this study, has demonstrated that urban ponds and lakes contribute a great deal to aquatic biodiversity in urban and peri-urban regions, there is still little knowledge about the way urban waterbodies function and respond to multiple environmental and anthropogenic stressors, and on how they should be managed (Hassall 2014; Biggs et al. 2005). This study emphasizes the valuable contribution of small waterbodies, as ponds, lakes and marshes, for sustaining aquatic biodiversity of macroinvertebrates in urban landscapes, as shown for zooplankton (Pinel-Alloul & Mimouni, 2013; Mimouni et al., 2015). These waterbodies could play a strategic role in conservation and management of urban freshwater biodiversity, which provides ecological services to cities and people.

A review on the importance of spatial and temporal variations of macroinvertebrates for European pond conservation (Jeffries 2005) concluded that the ecological integrity of ponds and their sustainable development under changing urban and agricultural landscapes should rely on macro-management (across ponds) rather than on micro-management (within ponds). Our study gives some support to this recommendation. We showed that both permanent and temporary waterbodies accounted for regional species pool (γ diversity) and spatial variation in local diversity (β diversity) of macroinvertebrates. Thus the conservation of a variety of pond types and habitats is essential because it will provide sufficient environmental heterogeneity to allow taxa to track finely-tuned habitats across landscapes, and then adapt to local environment or disperse in case of unsuitable environment. Our study also highlights that temporal variation during summer is unlikely to confound interpretation of spatial changes among types of waterbodies, and does not need to be considered in biomonitoring, as previously suggested for streams (Stark & Phillips 2009). Our study suggests that permanent and temporary ponds have a good potential to sustain both regional taxonomic pool and heterogeneous communities (β diversity). Management of these ponds requires to incorporate landscape spatial scale (across different types of waterbodies) and long-term timescale (on specific types of waterbodies) studies. Conservation and biomonitoring strategies at the regional scale rather than local management of each waterbody should be used by city managers and environmental agencies (parks, urban management planning, etc).

Considering the most appropriate approaches to incorporate in biomonitoring of aquatic biodiversity in urban waterbodies, our study showed that a single approach based on univariate indices alone is not a suitable surrogate to approaches based on multivariate metrics (Menetrey et al., 2011; Moya et al. 2011). Our study highlights the importance of using macroinvertebrate metrics based on taxonomic assemblages (directly or with functional traits data) to assess ecological quality of natural and man-made ponds. However, due to lack of knowledge and the difficulty for managers to apply and interpret functional traits in urban waterbodies, metrics based on taxonomic assemblages and indicators may have the best potential for biomonitoring.

Finally, management practices as winter draining, macrophyte and sediment removal, and chemical treatment should be used with caution in order to preserve ecological integrity of urban waterbodies. We have seen that management practices could lead to high beta diversity, but in most cases this increase was due to a large proportion of the same taxon (for example Chironomidae). On the other hand, natural marshes not only contributed largely to beta diversity, but also had a high alpha diversity. A change in management practices to a more biodiversity oriented strategy should lead to an increased effort in conservation of aquatic vegetation, which favor high transparency, water oxygenation, and biodiversity (all those factors are present in natural systems). Attention should also be given to educate the public on the importance, attractiveness, and conservation value of maintaining ponds in urban settings in their natural state.

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Table 1. Environmental features of the 20 studied waterbodies based on three sets of variables: A: Origin, urban land-use and management practices, B: Vegetation, morphometry, water quality, algal biomass, C: Fish presence/absence.

A: Origin, urban land-use and management practices			
Origin	Natural (3)		Artificial (17)
Winter Emptying	Temporary (7)		Permanent (13)
Urban Density	Low (7)	Medium(6)	High (7)
Copper Sulfate Treatment	Yes (4)		No (16)
Macrophyte Removal	Yes (5)		No (15)

B: Vegetation, morphometry, water quality, algal biomass		
	Mean ± Sd	Min - Max
Vegetation	Presence (14)	Absence (6)
Surface (ha)	2.56 ± 3.42	0.01 – 11.45
Depth (m)	1.97 ± 2.13	0.24 – 9.64
Perimeter (km)	0.91 ± 0.77	0.03 – 2.49
Secchi (m)	1.08 ± 0.91	0.24 – 4.41
Total Phosphorus ($\mu g \cdot L^{-1}$)	59,28 ± 74.76	8.00 – 260.50
Total Chlorophyll ($\mu g \cdot L^{-1}$)	4.85 ± 3.06	1.36 – 11.92
Green Algae ($\mu g \cdot L^{-1}$)	2.25 ± 1.75	0.50 – 6.75
Bluegreen Algae ($\mu g \cdot L^{-1}$)	1.04 ± 0.49	0.46 – 2.19
Diatom ($\mu g \cdot L^{-1}$)	1.10 ± 0.95	0.26 – 4.24
Cryptophyta ($\mu g \cdot L^{-1}$)	0.50 ± 0.30	0.14 – 1.05
Ph	8.13 ± 0.24	7.54 – 8.45
Conductivity ($\mu S \cdot cm^{-1}$)	443.87 ± 206.85	159.89 – 849.44
Total Organic Carbon ($mg \cdot L^{-1}$)	6.62 ± 4.09	2.00 – 17.30
K ($\mu g \cdot L^{-1}$)	2596.48 ± 1222.60	687.73 – 5675.00
NH4 ($\mu g \cdot L^{-1}$)	25.43 ± 8.74	20.00 – 53.60

C: Fish presence/absence		
Fish	Presence (16)	Absence (4)

Table 2. List of functional traits and modalities applied to the macroinvertebrate taxa found in the 20 studied water bodies (adapted from Poff et al. 2006 and Desrosiers et al. 2015, in preparation)

Life history Traits		
Voltinism (number of reproduction cycle per year)	Semivoltine (< 1cycle/year)	Volt1
	Univoltine (1 cycle/year)	Volt2
	Multivoltine (> 1 cycle/year)	Volt3
Desiccation	Does not survive desiccation	Des0
	Survives desiccation	Des1
Exit	Cannot exit the system	Exit0
	Ability to exit the system	Exit1
Resilience	Yes	Res1
	No	Res0
Life span	Very short (<1 week)	Life1
	Short (≤ 1 month)	Life2
	Long (> 1 month)	Life3
Aquatic stages	Eggs	Sta1
	Larva/nymph	Sta2
	Pupa	Sta3
	Adult	Sta4
Mobility Traits		
Dispersal or dissemination potential	Low (<1km flight before laying eggs)	Disp0
	High (>1km flight before laying eggs)	Disp1
Locomotion and/or substrate relation	Swimmer/sprawler	Hab1
	Crawler/climber	Hab2
	Burrower	Hab3
	Attached	Hab4
	Skate	Hab5
Substrate specific attachment	None	Att1
	Case, silk-net	Att2
	Soft tube	Att3
	Hooks, Suckers, byssus, filament	Att4

Morphological Traits		
Body form	Spherical, conical, humped, spirally	Form1
	Cylindrical, tubular, vermiform	Form2
	Streamlined, flattened	Form3
Armouring	None	Arm1
	Poor	Arm2
	Good	Arm3
	Total (e.g. shell)	Arm4
Maximal potential size	Small (<9mm)	Size1
	Medium (9-16mm)	Size2
	Large (>16mm)	Size3
Physiological Traits		
Respiration	Aquatic (tegument)	Resp1
	Aquatic (gill)	Resp2
	Aerial	Resp3
Reproduction techniques	Asexual	Rep1
	Sexual	Rep2
Reproduction strategies	With parental care (ovoviviparity, marsupium on parent)	Ovo1
	Eggs/clutches, free	Ovo2
	Eggs/clutches, fixed	Ovo3
Feeding habits	Collector-gatherer	Trop1
	Filterer	Trop2
	Scraper, chewer, grazer	Trop3
	Predator, parasite	Trop4
	Shredder	Trop5
Ecological Traits		
Organic matter sensitivity	High	Org1
	Intermediate	Org2
	Low	Org3

Table 3. Groups of sites with distinct environmental features based on K-means partitioning and cluster analysis. Values represent mean ± standard deviation.

Groups	**1**	**2**	**3**	**4**
Sites	Bizard Marais des castors Prairies	Battures Brunante Centenaire Cygnes Héritage Lacoursière RMontigny	Angrignon JBA JBN Lac des castors Lafontaine	Beaubien Jarry Liesse Pratt1 Pratt2
Origin	Natural	Artificial	Artificial	Artificial
Status	Permanent	Permanent	Permanent (3) Temporary (2)	Temporary
Management		Vegetation removal (2)	Vegetation removal (3)	
Depth (m)	2.07 ± 2.45	3.37 ± 2.81	1.39 ± 0.47	0.53 ± 0.25
Surface (ha)	4.17 ± 6.23	3.96 ± 3.68	1.94 ± 1.75	0.26 ± 0.42
Perimeter (km)	0.96 ± 0.85	1.31 ± 0.66	1.06 ± 0.86	0.19 ± 0.18
Transparency (m)	0.67 ± 0.34	1.53 ± 1.36	1.29 ± 0.27	0.53 ± 0.25
pH	8.11 ± 0.23	8.09 ± 0.16	8.20 ± 0.38	8.12 ± 0.26
Conduct. ($\mu S \cdot cm^{-1}$)	346 ± 51	609 ± 161	262 ± 18	416 ± 251
TP ($\mu g \cdot L^{-1}$)	182 ± 136	40 ± 34	28 ± 10	44 ± 45
TOC ($mg \cdot L^{-1}$)	15 ± 2.5	5.2 ± 1.5	5.6 ± 1.3	4.2 ± 2.1
NH_4 ($\mu g \cdot L^{-1}$)	29 ± 7	28 ± 13	22 ± 3	22 ± 4
K ($\mu g \cdot L^{-1}$)	2837 ± 2564	3154 ± 612	1431 ± 338	2836 ± 883
Chl*a* ($\mu g \cdot L^{-1}$)	7.85 ± 1.77	5.84 ± 3.85	2.74 ± 1.12	3.80 ± 1.96
Greens ($\mu g \cdot L^{-1}$)	3.79 ± 1.34	2.76 ± 2.28	1.13 ± 0.69	1.50 ± 0.77
Diatoms ($\mu g \cdot L^{-1}$)	1.71 ± 0.24	1.39 ± 1.39	0.60 ± 0.38	0.80 ± 0.57
Cyanos ($\mu g \cdot L^{-1}$)	1.49 ± 0.35	1.16 ± 0.54	0.61 ± 0.11	1.06 ± 0.51
Cryptos ($\mu g \cdot L^{-1}$)	0.91 ± 0.19	0.54 ± 0.31	0.28 ± 0.10	0.42 ± 0.26
Vegetation cover	5 ± 0	1.3 ± 1.1	3.6 ± 2.1	1.2 ± 1.6
Fish	Yes	Yes	Yes	No (except 1)

Table 4. Macroinvertebrate metrics (mean summer values ±standard deviation) of diversity, biotic, and functional metrics, taxon and functional traits indicators in the 4 groups of sites with distinct environmental features identified by K-means partitioning and cluster analysis.

Groups	**1**	**2**	**3**	**4**
Sites	Bizard Marais des castors Prairies	Battures Brunante Centenaire Cygnes Héritage Lacoursière RMontigny	Angrignon JBA JBN Lac des castors Lafontaine	Beaubien Jarry Liesse Pratt1 Pratt2
α diversity	27.2 ± 8.6	18.6 ± 7.6	25.3 ± 8.5	12.4 ± 5.8
β diversity	0.179	0.317	0.248	0.256
γ diversity	50	55	60	41
Shannon index	3.03 ± 0.25	2.38 ± 0.61	2.75 ± 0.54	1.83 ± 0.87
Pielou evenness	0.65 ± 0.04	0.58 ± 0.11	0.61 ± 0.12	0.54 ± 0.11
Simpson index	0.80 ± 0.04	0.67 ± 0.15	0.77 ± 0.07	0.55 ± 0.22
Mean abundance	2633 ± 1674	1173 ± 1016	3438 ± 2522	746 ± 768
Nb taxa total	27 ± 8	19 ± 7	26 ± 7	12 ± 6
Nb taxa ET	3 ± 2	3 ± 1	3 ± 1	2 ± 1
pET (%)	2.39 ± 2.45	4.00 ± 2.00	12.92 ± 8.25	19.77 ± 11.41
pCOH (%)	34.01 ± 15.13	60.82 ± 18.53	43.80 ± 22.56	55.05 ± 20.56
ET/COH	0.08 ± 0.08	0.09 ± 0.06	0.48 ± 0.37	0.98 ± 0.65
FBI Hilsenhof	6.95 ± 0.48	7.31 ± 0.30	7.16 ± 0.26	6.87 ± 0.48
FRic	1.38 ± 0.32	1.21 ± 0.28	1.38 ± 0.35	0.92 ± 0.46
FEve	0.46 ± 0.08	0.56 ± 0.12	0.51 ± 0.12	0.61 ± 0.15
FDis	0.28 ± 0.06	0.26 ± 0.06	0.30 ± 0.03	0.23 ± 0.10
Indicator taxa	Curculionidae Phrygaenidae Belostomatidae Stratiomyidae Mesoveliidae Pyralidae Sphaeridae Haliplidae Pleidae Ceratopogonidae Planorbidae	Ancylidae	Leptoceridae Hydroptilidae Ostracoda Caenidae Valvatidae	
Indicator traits	Form1 Hab3 Resp3 Life3 Ovo1 Disp0 Trop2 Size3 Trop4 Org2	Resp1 Att3 Rep1	Size2	Sta2 Arm1 Sta3 Ovo2 Form2 Trop1 Hab1 Org1 Life1 Size1

Table 5. Variance partionning of RDA models relating the macroinvertebrate taxonomic assemblages and functional traits to simple and interactive effects of three sets of environmental features: A: Origin, urban land-use and management practices, B: Vegetation, morphometry, water quality and algal biomass, C: fish presence/absence.

Environmental sets Simple - Interactive Effects	**Multivariate metrics Taxon assemblages**	**Multivariate metrics Functional traits**
A (%)	9	13
B (%)	21	28
C (%)	1	1
A/B (%)	7	-
A/C (%)	4	3
B/C (%)	3	2
A/B/C (%)	1	7
Explained variation (%)	46	51

Figure 1. Localisation of the 20 studied waterbodies within the Island of Montreal. Symbols correspond to groups of sites identified using k-mean partitioning of environmental features (see Table 1). Group 1: Bizard, Marais des castors, Prairies; Group 2: Battures, Brunante, Centenaire, Cygnes, Héritage, Lacoursière and Montigny; Group 3: Angrignon, JBA, JBN, Lac des castors and Lafontaine; Group 4: Beaubien, Jarry, Liesse, Pratt1 and Pratt2.

Figure 2. Accumulation curve of taxon richness for the 59 macroinvertebrate samples collected during summer 2011.

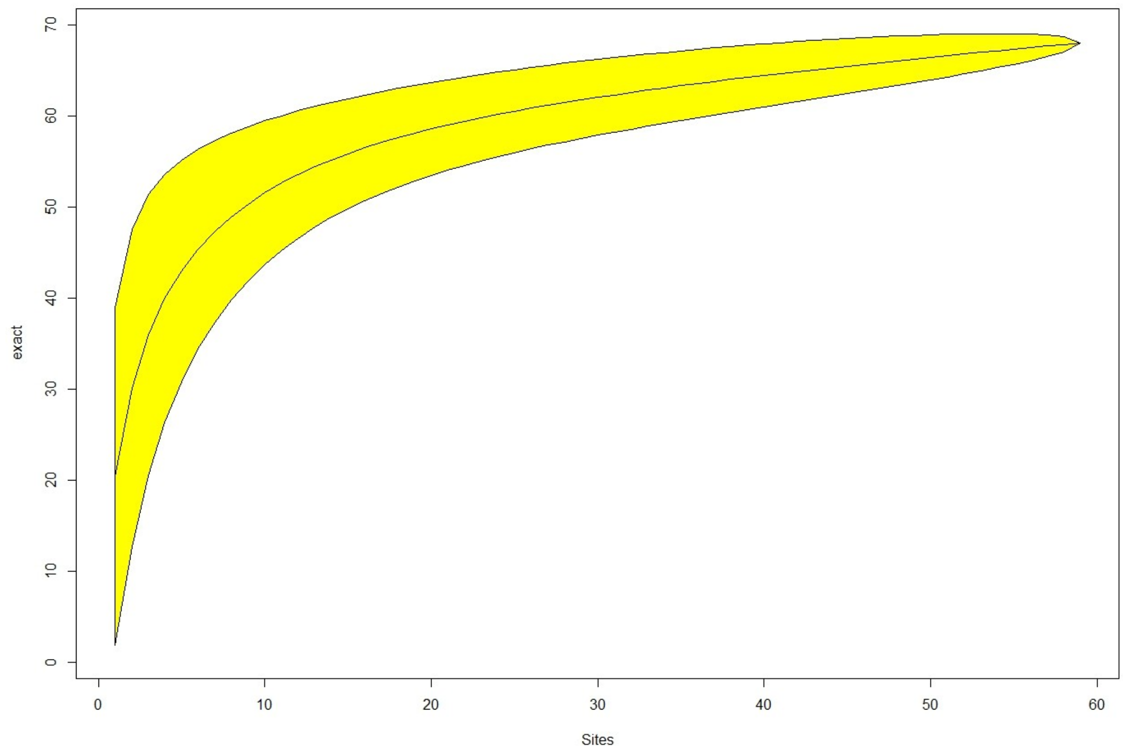


Figure 3A. Diversity alpha (taxa richness) in 19 sampled sites. Boxplots present the quartiles of the three combined replicates in each site. Pratt 1 was not included because it was not sampled in June. Sites are presented in order of the 4 groups determined by the k-means analysis on environmental data.

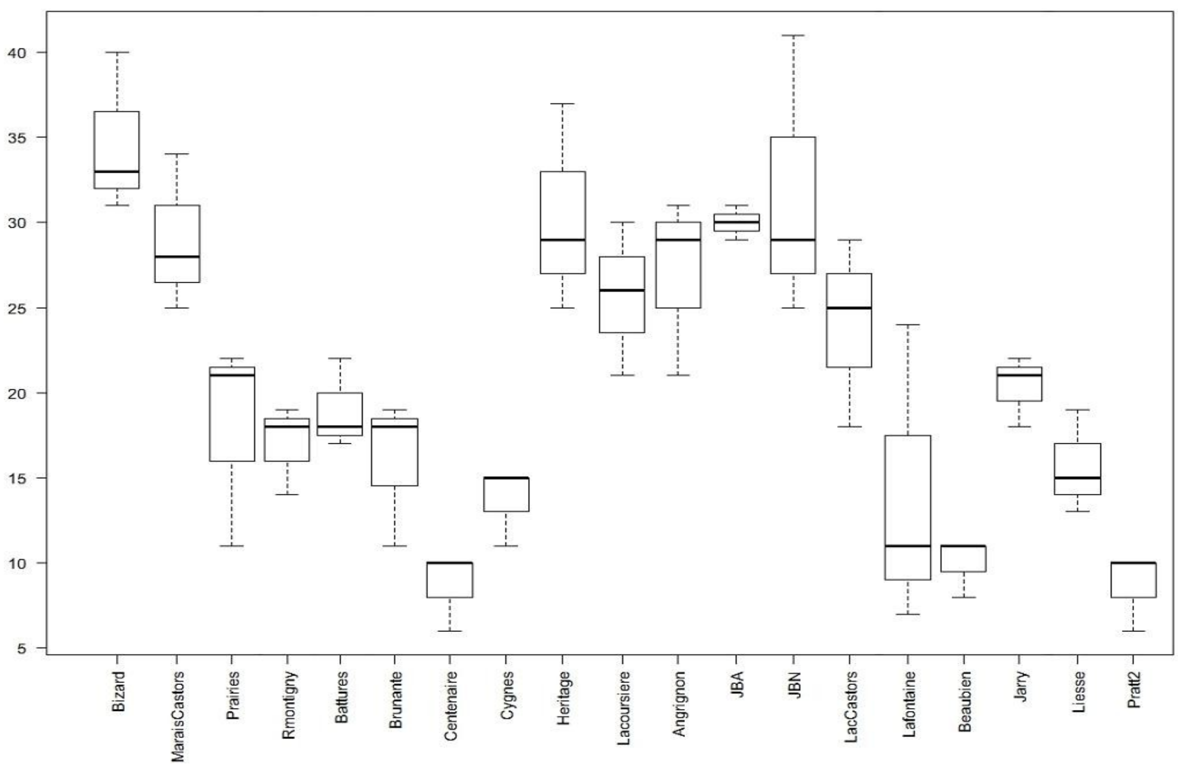


Figure 3B. Beta-diversity (LCBD) in 19 sampled sites. Boxplots present the quartiles of the three combined replicates in each site. Pratt 1 was not included because it was not sampled in June. Sites are presented in order of the 4 groups determined by the k-means analysis on environmental data.

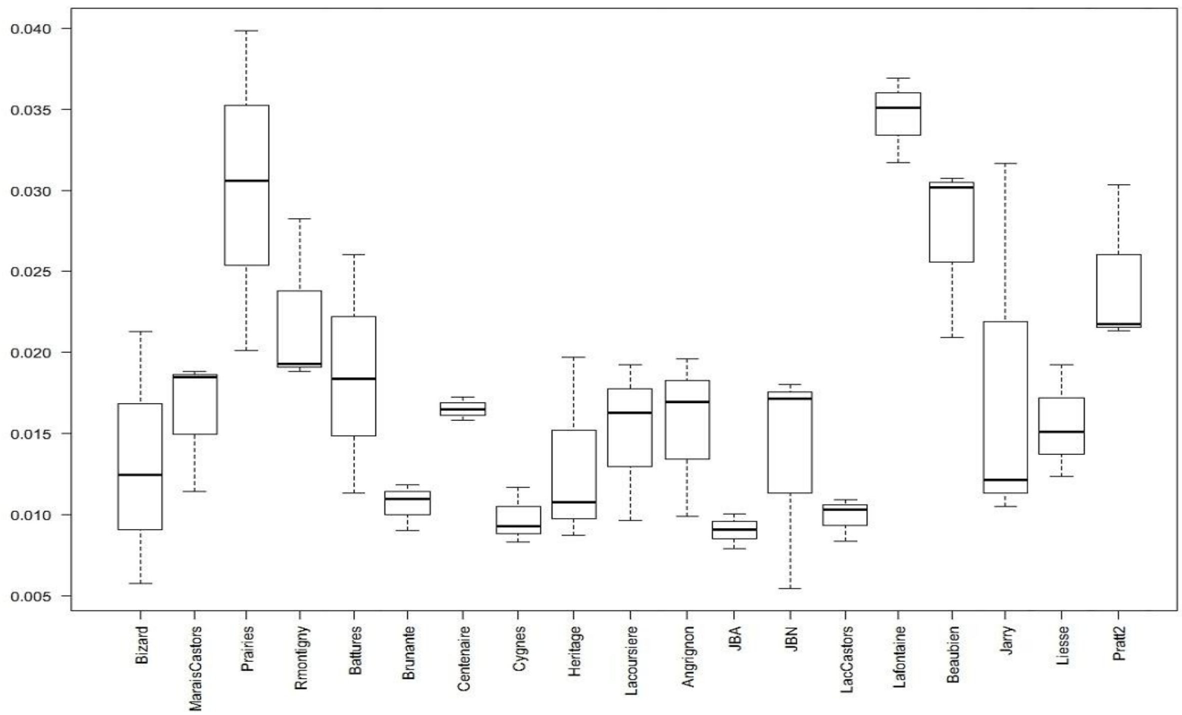


Figure 4. Composition of macroinvertebrate communities in each waterbody based on the relative occurrence of the principal taxonomic groups recorded across summer 2011 (June, July and August).

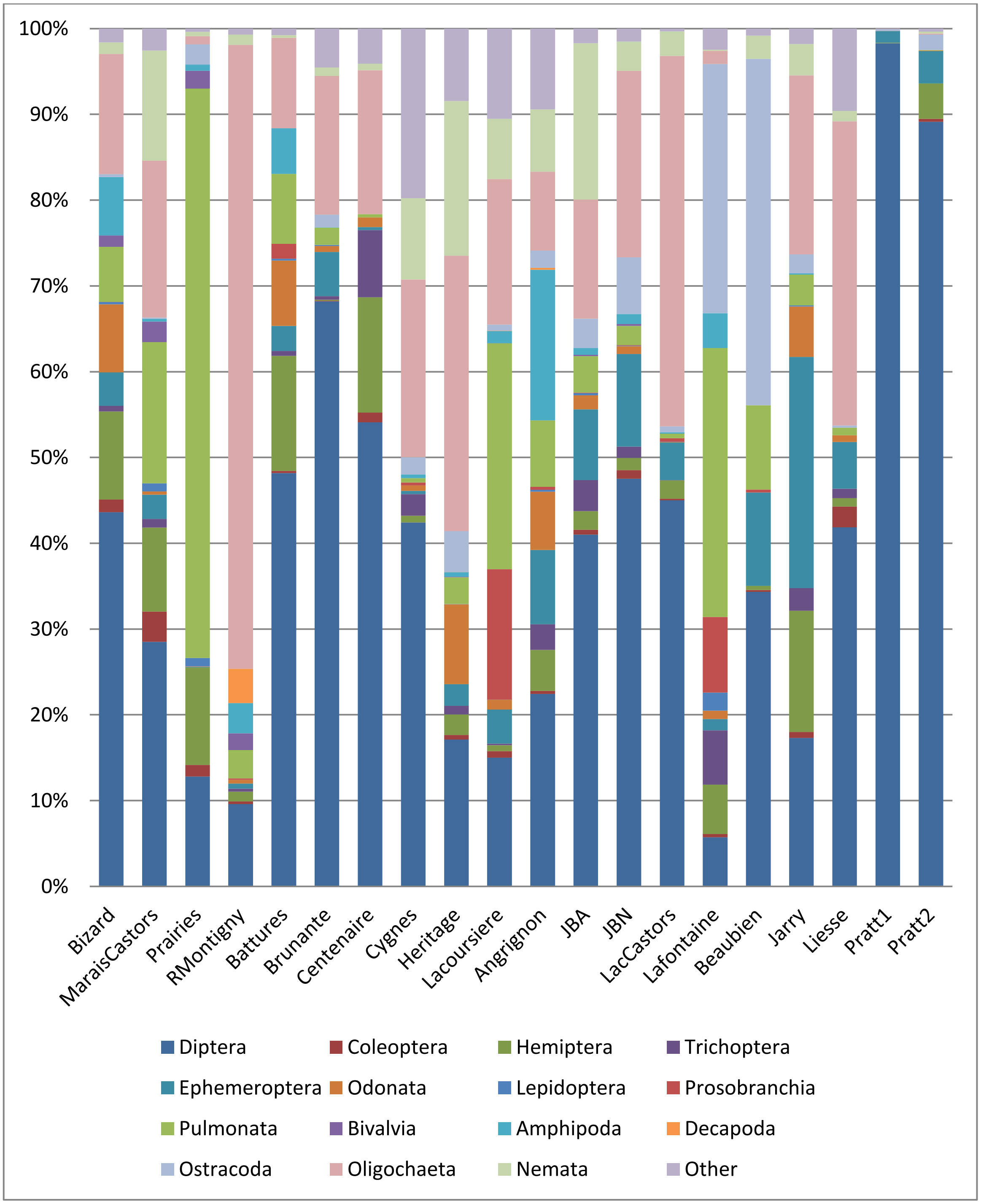


Figure 5. Contribution to beta diversity (SCBD) of the main taxa across the urban area

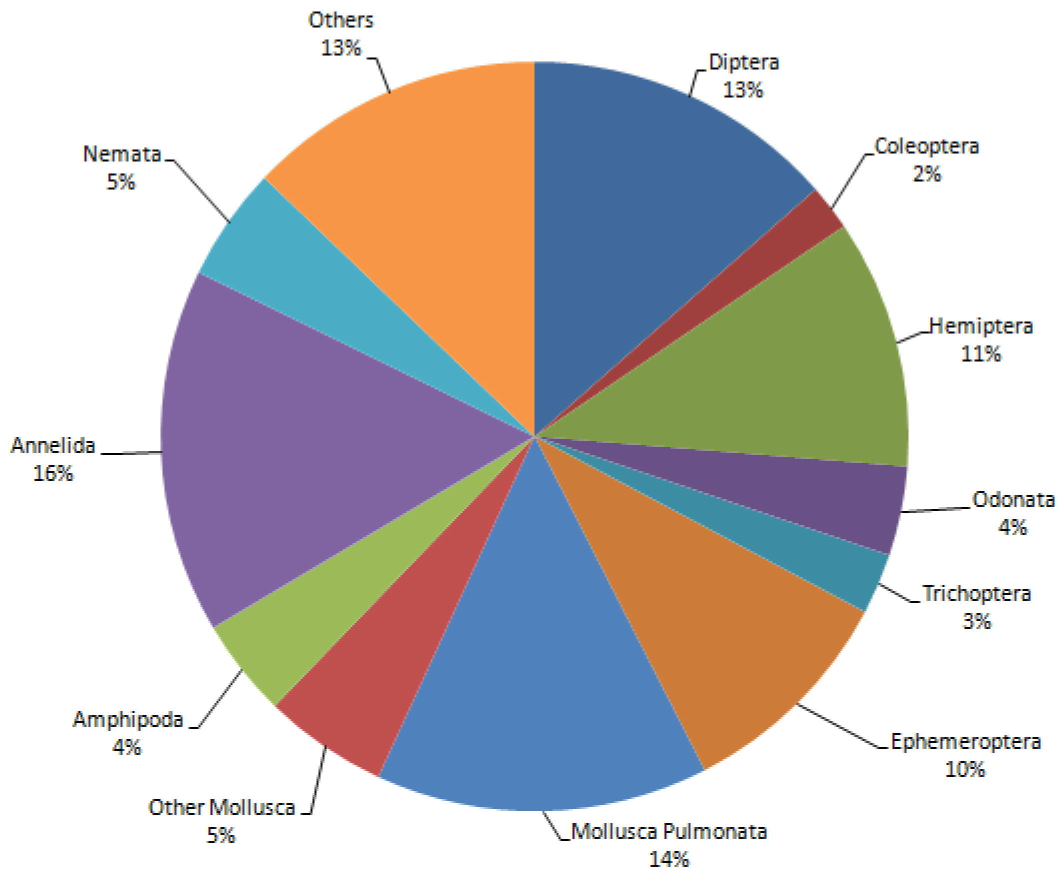


Figure 6. RDA model of the macroinvertebrate taxonomic composition constrained by environmental variables in the studied waterbodies

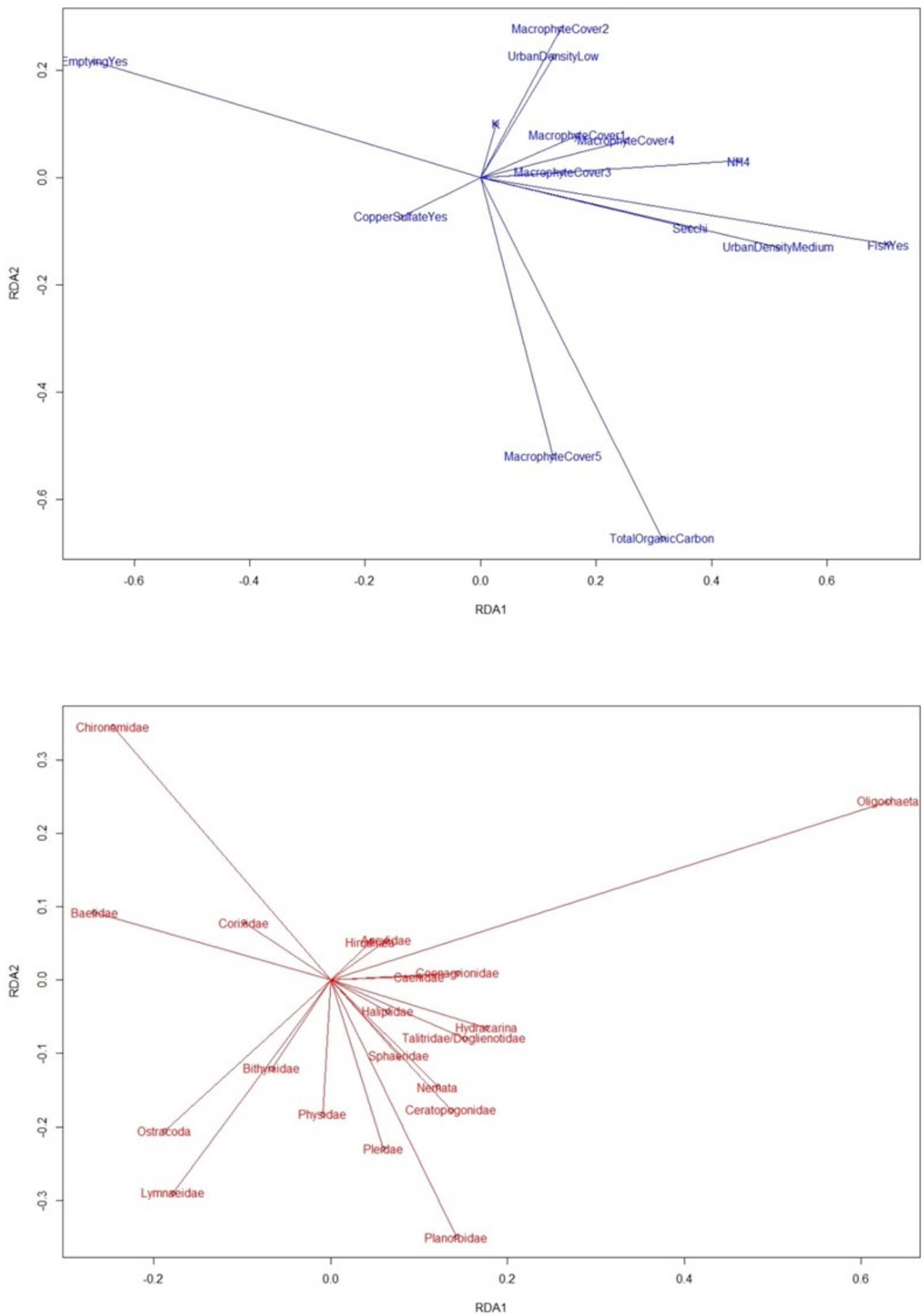
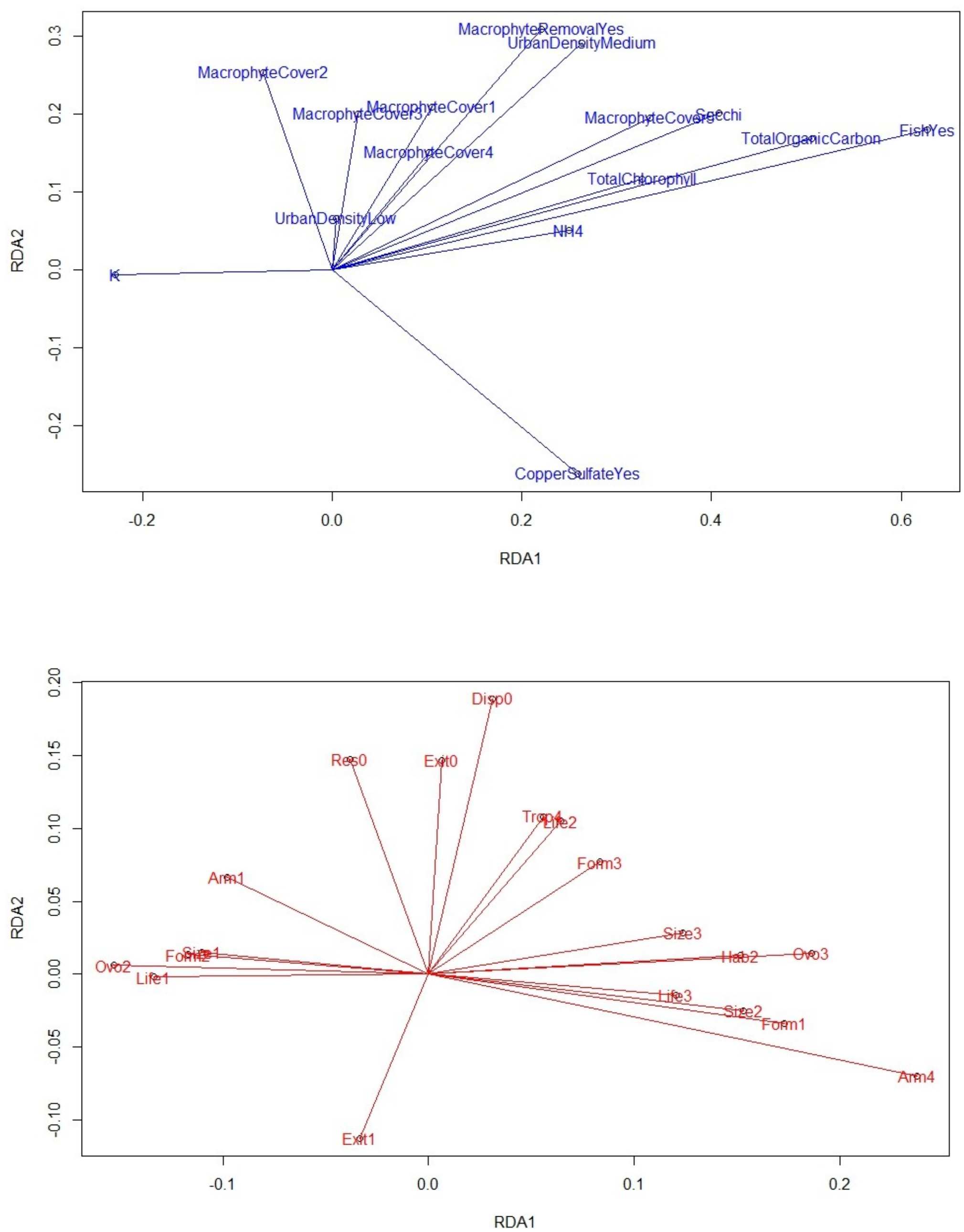


Figure 7. RDA model of the macroinvertebrate functional trait profiles constrained by environmental variables in the studied waterbodies



Chapitre 3: Conclusion générale

Rappel des objectifs

Ce projet avait pour premier objectif de déterminer pour une première fois la biodiversité aquatique des macroinvertébrés dans les étangs urbains d'une grande ville d'Amérique du Nord. Notre étude a permis de démontrer que les marais naturels, les étangs et petits lacs permanents des sites résidentiels et les petits étangs temporaires des parcs municipaux de l'île de Montréal supportent une diversité de macroinvertébrés aussi importante que celle retrouvée dans d'autres petits plans d'eau en zones agricole et périurbaine en Europe (Céréghino et al. 2008a; Oertli et al. 2002).

Premièrement, on a évalué si la variation temporelle était plus faible que la variation spatiale dans la composition en macroinvertébrés benthiques des étangs de la ville de Montréal. Nous avons trouvé que la variation spatiale avait beaucoup plus d'importance que la variation temporelle pour la composition en macroinvertébrés benthique des étangs. Donc, nous avons surtout évalué l'importance de la variation spatiale de plusieurs métriques basées sur la diversité et la composition des macroinvertébrés.

Ensuite, nous avons déterminé la pertinence d'utiliser des métriques univariées basées sur des indices de diversité ou des indices biotiques liées à la tolérance à la pollution organique, en comparaison à des métriques multivariées basées sur la composition taxonomique ou fonctionnelle des macroinvertébrés benthiques. Nous avons démontré que les estimés de diversité et les indices biotiques étaient des métriques qui étaient à la fois redondantes mais aussi complémentaires pour le suivi environnemental de la biodiversité aquatique dans les étangs urbains. Par exemple, la diversité de Shannon, la diversité de Simpson et la dispersion fonctionnelle sont très fortement corrélées entre elles et aussi avec le nombre total de taxa et d'Ephémères et de Trichoptères (ET). De plus, l'indice biotique FBI est inversement relié au pourcentage de taxa sensibles (%ET) et positivement au pourcentage de taxa tolérants (%COH). Donc dans un paysage urbain, on recommande d'utiliser plusieurs métriques afin de bien évaluer les changements des communautés entre les plans d'eau et leur évolution suite à des changements environnementaux et des modifications des pratiques d'aménagement.

Par ailleurs, nous avons déterminé l'influence de différents types de variables environnementales (A : origine et aménagement, densité urbaine; B : couvert de végétation, qualité des eaux, morphométrie; C : présence de poissons) sur la structure taxonomique et fonctionnelle des communautés de macroinvertébrés benthiques en milieu urbain et sur les réponses des métriques de diversité ou des indices biotiques. Nous avons trouvé que les variables reliées au couvert de végétation aquatique, à la qualité des eaux (nutriments, matière organique) et à la biomasse algale étaient toujours les facteurs les plus importants pour expliquer la variation spatiale des communautés de macroinvertébrés, quelles que soient les métriques utilisées, soit univariées (indices biotiques, diversité taxonomique ou fonctionnelle) ou multivariées (assemblages des taxa et profils des traits fonctionnels).

Finalement, nous avons évalué la valeur de l'approche taxonomique classique en comparaison avec l'approche par traits fonctionnels pour faire le suivi de la qualité environnementale et de la composition et la diversité en macroinvertébrés benthiques dans les plans d'eau douce de la région urbaine étudiée. Nous avons constaté que le modèle utilisant les traits fonctionnels explique seulement 2% de plus de la variance que le modèle utilisant la composition en taxa. Étant donné qu'il est encore difficile de trouver les informations sur les traits fonctionnels des macroinvertébrés dans la littérature, il nous semble préférable de se baser actuellement sur la taxonomie classique qui est plus facile à appliquer dans le cadre d'un suivi environnemental et qui répond de façon comparable à l'environnement.

Directions futures

Notre étude démontre qu'il y a encore beaucoup de lacunes dans les connaissances et de développement à faire pour appliquer un suivi optimal de la biodiversité aquatique en milieu urbain. Par exemple, il existe très peu de données sur les Hémiptères qui sont des organismes plus présents dans les étangs que dans d'autres milieux aquatiques (rivières). Il serait nécessaire de tester à nouveau les deux approches (taxonomique et fonctionnelle) pour voir si les traits expliqueraient une plus grande partie du modèle reliant la structure des macroinvertébrés avec l'environnement en utilisant une base de données plus complète. L'approche par traits fonctionnels s'avérerait peut-être plus efficace pour évaluer les changements dans l'environnement ou distinguer les différents mésohabitats d'un milieu. Il serait aussi intéressant de regarder si la variation spatiale et temporelle serait différente en regardant la composition en traits fonctionnels plutôt que la composition taxonomique.

Au niveau des pratiques d'aménagement, vider les étangs pour l'hiver, faucarder la végétation et enlever les sédiments apparaissent comme des pratiques qui limitent la biodiversité aquatique. Elles devraient être utilisées avec modération et en dernier recours pour conserver l'intégrité écologique des étangs urbains. On devrait donc se tourner vers l'aménagement à grande échelle plutôt que d'aménager chaque étangs individuellement puisque la valeur pour la biodiversité est plus élevée à l'échelle du paysage où l'on retrouve plusieurs types d'étangs correspondant souvent à différents stades de vieillissement des plans d'eau. Il faudrait de plus éduquer le public sur l'importance de garder les étangs dans un état le plus naturel possible et de conserver la végétation aquatique.

Finalement, il serait intéressant d'utiliser une approche utilisant divers types d'indicateurs biotiques réunis ensemble, comme la méthode PLOCH le fait déjà en Europe (végétaux, amphibiens, insectes, etc.) (Oertli et al., 2005b). Une première intégration des études effectuées sur trois types de communautés (phytoplancton, zooplancton, macroinvertébrés) (Marinescu, 2015) a permis récemment de classer les plans d'eau selon un gradient de conservation de la biodiversité aquatique. Cette approche multi-indicateurs semble avoir un bon potentiel pour transmettre les résultats de la recherche aux agences impliquées et conseiller les municipalités au niveau de la gestion de l'aménagement des étangs urbains.

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Annexes

Table S1. Environmental features of the 20 studied waterbodies and each group of sites revealed by k-means partitioning and cluster analysis.

2011	Copper Sulphate Treatment	Waterbody emptying	Macrophyte removal	Urban Density	Depth (m)	Secchi depth (m)	Surface (ha)	Perimeter (km)	pH	TP	NH4	K	TOC	DO	Cond	Trophic level	Chla (µg.L^{-1})	Browns (µg.L^{-1})	Cryptos (µg.L^{-1})	Cyanos (µg.L^{-1})	Greens (µg.L^{-1})	Fish	Stickleback	Origin	Macrophyte cover
Moyenne					1,97	1,08	2,56	0,91	8,13	59,28	25,43	2596,48	6,62	123,24	443,87		4,85	1,10	0,50	1,04	2,25				2,40
Écart type					2,13	0,91	3,41	0,76	0,24	74,76	8,74	1222,60	4,09	37,60	206,85		3,06	0,95	0,30	0,48	1,75				2,04
Minimum					0,24	0,24	0,03	0,07	7,54	8,00	20,00	687,73	2,00	58,39	159,89		1,36	0,26	0,14	0,46	0,50				0,00
Maximum					9,64	4,41	11,45	2,49	8,45	260,50	53,60	5675,00	17,30	237,98	848,44		11,92	4,24	1,05	2,19	6,75				5,00
GROUP 1																									
Bizard	No	No	No	Low	0,75	0,75	11,35	1,88	7,96	24,33	28,03	687,73	12,33	130,81	324,44	Eutrophic	9,74	1,93	0,70	1,89	5,23	Yes	Yes	Natural	5
Prairies	No	No	No	Low	0,56	0,29	1,01	0,81	8,38	260,50	23,15	2147,00	17,30	130,74	404,67	Eutrophic	6,22	1,45	0,99	1,24	2,59	Yes	No	Natural	5
MaraisCastors	No	No	No	Medium	4,89	0,96	0,16	0,20	8,00	260,00	36,50	5675,00	15,25	58,39	309,44	Eutrophic	7,60	1,74	1,05	1,33	3,56	Yes	No	Natural	5
Moyenne					2,07	0,67	4,17	0,96	8,11	181,61	29,23	2836,58	14,96	106,65	346,18		7,85	1,71	0,91	1,49	3,79				5,00
Ecart-type					2,45	0,34	6,23	0,85	0,23	136,21	6,75	2564,15	2,50	41,79	51,20		1,77	0,24	0,19	0,35	1,34				0,00
GROUP 2																									
Centenaire	No	No	No	Low	2,12	0,64	11,45	1,84	8,26	54,00	20,00	2725,33	3,93	130,22	849,44	Eutrophic	7,81	1,44	0,69	0,88	4,80	Yes	No	Artificial	0
Brunante	No	No	Yes	Medium	2,45	0,66	1,75	1,29	8,07	34,67	23,33	3438,00	4.43	109,52	610,56	Meso-eutrophic	3,81	0,56	0,31	1.08	1,86	Yes	No	Artificial	2
Heritage	No	No	No	High	2,49	1,86	0,78	0,58	7,99	20,50	20,00	4184,00	5,65	125,83	787,44	Meso-eutrophic	2,47	0,59	0,26	0,66	0,96	Yes	No	Artificial	1
Lacoursiere	No	No	Yes	Medium	1,46	1,46	1,12	0,57	8,06	23,00	20,00	2668,30	6,70	105,19	630,22	Meso-eutrophic	3,12	0,53	0,35	1,07	1,17	Yes	Yes	Artificial	3
Battures	No	No	No	Low	3,08	1,20	4,87	1,26	8,25	27,50	20,00	3492,00	6,80	126,98	485,00	Eutrophic	9,42	4,24	0,93	1,18	3,09	Yes	No	Artificial	1
Cygnes	No	No	No	Low	9,64	4,41	3,09	1,20	8,18	12,67	39,70	3148,73	3,13	118,16	444,11	Meso-eutrophic	2,30	0,37	0,29	0,95	0,68	Yes	No	Artificial	2
RMontigny	No	No	No	Medium	2,33	0,48	4,64	2,43	7,82	111,00	53,60	2404,00	5,25	137,41	455,78	Hypereutrophic	11,92	2,03	0,95	2,19	6,75	Yes	No	Artificial	0
Moyenne					3,37	1,53	3,96	1,31	8,09	40,48	28,09	3154,34	5,24	121,90	608,94		5,84	1,39	0,54	1,16	2,76				1,29
Ecart-type					2,81	1,36	3,68	0,66	0,16	33,75	13,35	612,34	1,48	11,53	161,32		3,85	1,39	0,31	0,54	2,28				1,11
GROUP 3																									
Angrignon	No	No	Yes	Medium	2,18	1,67	4,91	2,49	8,39	20,00	20,00	1239,33	6,63	130,50	331.33	Meso-eutrophic	3,34	0,55	0,33	0,74	1.72	Yes	No	Artificial	4
Lafontaine	Yes	Yes	No	High	1,34	1,34	1,91	1,03	8,45	20,50	26,35	1777,20	3,85	129,78	274,67	Oligo-mesotrophic	1,73	0,27	0,21	0,67	0,57	Yes	No	Artificial	0
JBNenuphars	Yes	No	No	High	1,02	1,02	0,58	0,33	7,54	42,00	26,15	1251,10	6,65	84,07	234,44	Meso-eutrophic	3,36	0,77	0,38	0,60	1,62	Yes	No	Artificial	5
JBAlgues	Yes	No	Yes	High	1,04	1,04	0,79	0,44	8,37	36,00	20,00	1078,10	6,50	142,89	241,11	Meso-eutrophic	3,89	1,17	0,34	0,56	1,82	Yes	No	Artificial	5
LacCastors	Yes	Yes	Yes	Low	1,36	1,36	1,5	1,00	8,25	21,00	20,00	1811,2	4,55	118,64	279,11	Oligo-mesotrophic	1,36	0,26	0,14	0,46	0,50	Yes	No	Artificial	4
Moyenne					1,39	1,29	1,94	1,06	8,20	27,90	22,50	1431,39	5,64	121,18	262,33		2,74	0,60	0,28	0,61	1,13				3,60
Ecart-type					0,47	0,27	1,75	0,86	0,38	10,36	3,42	338,40	1,34	22,45	17,76		1,12	0,38	0,10	0,11	0,69				2,07
GROUP 4																									
Pratt 1	No	Yes	No	High	0,33	0,33	0,10	0,11	7,88	13,00	20	3352,00	2,00	237,98	757,63	Oligo-mesotrophic	2,08	0,26	0,25	0,92	0,66	No	No	Artificial	0
Pratt 2	No	Yes	No	High	0,24	0,24	0,10	0,11	8,24	8,00	20,00	3468,00	2,30	120,02	601,11	Meso-eutrophic	2,91	0,46	0,28	0,85	1,32	No	No	Artificial	0
Beaubien	No	Yes	No	High	0,80	0,58	0,06	0,13	7,85	76,50	20,00	2801,00	4,80	94,17	300,00	Meso-eutrophic	4,14	1,42	0,32	0,67	1,74	No	No	Artificial	0
Liesse	No	Yes	No	Low	0,51	0,51	0,03	0,07	8,45	107,33	22,77	3238,00	6,93	98,46	159,89	Eutrophic	7,04	1,42	0,88	1,95	2,68	Yes	No	Artificial	3
Jarry	No	Yes	No	Medium	0,77	0,77	1,01	0,51	8,20	13,00	28,95	1323,60	5,20	115,03	264,56	Meso-eutrophic	2,81	0,46	0,35	0,92	1,08	No	No	Artificial	3
Moyenne					0,53	0,49	0,26	0,19	8,12	43,57	22,34	2836,52	4,25	137,13	416,64		3,80	0,80	0,42	1,06	1,50				1,20
Ecart-type					0,25	0,21	0,42	0,18	0,26	45,51	3,88	882,59	2,08	68,42	251,47		1,96	0,57	0,26	0,51	0,77				1,64

Table S2. Macrophyte cover and dominant species (if present) in the 20 studied waterbodies

Waterbody	Macrophyte Cover		Dominant Species							
	(%)	(Class)	1	2	3	4	5	6	7	8
Angrignon	75	4	*Myriophyllum exalbescens*	*Nymphaea tuberosa*	*Anacharis canadensis*	*Typha latifolia*	*Scirpus fluviatilis*	*Phragmites*		
Battures	10	1	*Valisneria americana*	*Myriophyllum exalbescens*	*Phragmites*					
Beaubien	0	0								
Bizard	> 80	5	*Ceratophyllum demersum*	*Potamogeton foliosus*	*Typha angustifolia*	*Nymphoïdes cordatum*	*Scirpus validus*	*Phragmites communis*	*Heteranthera dubia*	
Brunante	40	2	*Myriophyllum exalbescens*	*Nymphaea tuberosa*	*Scirpus validus*	*Butomus umbellatus*	*Lythrum sallicaria*	*Phalaris arondinacea*	*Phragmites*	
Centenaire	0	0								
Cygnes	30	2	*Myriophyllum exalbescens*	*Typha angustifolia*	*Nymphaea tuberosa*					
Heritage	10	1	*Myriophyllum exalbescens*	*Pontederia cordata*	*Nymphaea tuberosa*	*Butomus umbellatus*	*Scirpus fluviatilis*	*Typha latifolia*		
Jarry	60	3	*Chara vulgaris*	*Nitella flexilis*	*Najas flexilis*	*Scirpus fluviatilis*	*Butomus umbellatus*	*Typha*	*Phragmites*	
JBA	> 80	5	*Myriophyllum exalbescens*	*Chara vulgaris*	*Typha*	*Nymphaea tuberosa*	*Scirpus fluviatilis*	*Scirpus validis*	*Phragmites*	*Equisetum fluviatile*
JBN	> 80	5	*Nymphaea tuberosa*	*Equisetum fluviatile*	*Utricularia intermedia*	*Riccia*	*Phragmites*			
Lac Castors	80	4	*Myriophyllum exalbescens*	*Phragmites*						
Lacoursière	60	3	*Nymphaea tuberosa*	*Myriophyllum exalbescens*	*Typha*	*Phragmites*	*Sagittaria cuneata*	*Pontederia cordata*	*Butomus umbellatus*	*Potamogeton gramineus*
Lafontaine	0	0								
Liesse	40	3	*Lemna minor*	*Pontederia cordata*	*Nymphaea tuberosa*	*Ceratophyllum demersum*				
Marais Castors	100	5	*Wolfia punctata*	*Ceratophyllum demersum*	*Sparganium americanum*	*Phragmites maximus*	*Butomus umbellatus*	*Alisma gramineum*	*Sagittaria latifolia*	*Lithrum salicaria*
Prairies	100	5	*Lemna minor*	*Ceratophyllum demersum*	*Wolfia punctata*	*Typha angustifolia*	*Phalaris arondinacea*	*Phragmites*	*Lithrum salicaria*	
Pratt1	0	0								
Pratt2	0	0								
R-Montigny	0	0								

Table S3-A. Abundance, occurrence, and relative importance of macroinvertebrate taxa in the 20 studied waterbodies

Famille	Angrignon	Battures	Beaubien	Bizard	Brunante	Centenaire	Cygnes	Heritage	Jarry	JBA	JBN	LacCastors	Lacoursiere	Lafontaine	Lisse	MaraisCastors	Prairies	Pratt1	Pratt2	Rmontigny	somme	occurence	%
Amphizoidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	0,003
Hydroscaphidae	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	1	0,003
Ptiliidae	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0,003
Staphylinidae	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0,003
Nepidae	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0,003
Elmidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	1	0,003
Hebridae	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	2	1	0,005
Hydrometridae	0	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	2	0,003
Lepidostomatidae	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	2	1	0,005
Polycentropodidae	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	1	0,004
Hydropsychidae	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	1	0,005
Noteridae	0	0	0	17	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	17	1	0,045
Macroveliidae	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0,004
Chrysomelidae	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	4	0	0	0	0	5	2	0,014
Empydidae	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6	1	0,017
Phrygaenidae	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	8	2	0,022
Tipulidae	0	0	0	1	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	3	3	0,008
Platyhelminthes	0	0	0	0	2	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0	9	2	0,025
Ephydridae	0	0	0	0	0	0	0	0	0	9	0	0	0	0	0	0	1	0	0	0	10	2	0,028
Culicidae	0	0	0	1	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	3	5	0,009
Viviparidae	0	0	0	0	0	0	0	0	0	1	2	1	5	0	0	0	0	0	0	1	10	5	0,026
Valvatidae	5	2	0	1	0	0	0	0	0	1	0	6	0	0	0	0	0	0	0	0	14	5	0,038
Cambaridae	3	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	22	27	3	0,073
Tabanidae	0	0	0	0	0	0	0	1	0	1	0	0	0	0	5	0	0	0	0	1	8	4	0,022
Belostomatidae	0	0	0	5	0	0	0	0	0	0	0	0	1	0	0	2	0	0	0	0	8	3	0,022
Asellidae	0	0	0	0	0	1	0	0	0	0	4	4	0	0	0	0	0	0	0	0	10	3	0,027
Sminthuridae	0	0	0	1	5	0	0	1	0	0	3	0	0	0	0	1	0	0	0	0	11	5	0,031
Hydridae	6	3	0	0	1	2	0	1	0	0	0	0	0	9	0	0	0	0	0	0	21	5	0,058
Curculionidae	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	0	4	2	0,011
Lestidae	1	1	0	0	0	0	0	0	0	4	0	0	0	0	1	0	0	0	0	0	7	4	0,02
Poduridae	0	0	0	0	1	0	0	0	0	0	2	1	3	0	1	0	0	0	0	0	8	5	0,022
Chaoboridae	0	0	0	1	0	0	0	0	1	0	0	0	0	0	5	0	0	0	2	0	10	4	0,028
Stratiomyidae	1	0	0	4	0	0	0	2	0	2	1	0	0	0	0	10	4	0	0	0	23	7	0,063
Gammaridae	1	11	0	0	0	0	1	10	0	0	0	0	5	50	0	0	0	0	0	20	98	7	0,264
Nemertea	2	0	0	0	0	0	0	59	1	13	0	0	23	9	0	0	0	0	0	0	108	6	0,291
Bithyniidae	0	7	1	0	0	0	2	0	0	0	0	23	356	108	0	0	0	0	0	0	497	6	1,343
Hydrophilidae	1	0	0	1	1	0	0	1	0	0	0	0	1	1	0	1	0	0	0	0	7	7	0,02
Aeshnidae	3	0	0	2	0	0	0	3	1	3	1	0	0	0	1	2	0	0	0	0	18	8	0,049
Gerridae	4	0	0	17	0	0	0	0	0	3	1	0	2	0	0	0	0	0	0	1	28	6	0,075
Sphaeridae	0	0	0	51	0	0	0	1	0	6	7	0	0	0	0	79	15	0	0	11	171	7	0,462
Gyrinidae	0	0	0	2	0	0	0	2	0	0	4	2	0	0	0	4	2	0	0	0	17	6	0,047
Corduliidae	1	3	0	7	0	0	0	9	0	0	2	0	0	0	0	3	0	1	0	0	27	7	0,073
Libellulidae	3	0	0	69	0	0	0	14	1	17	4	1	6	0	1	1	0	0	0	0	117	10	0,315
Pleidae	0	0	0	186	0	0	0	3	0	3	1	1	4	0	0	299	75	0	0	0	573	8	1,548
Ancylidae	2	5	0	6	25	0	0	31	0	2	2	2	0	0	0	0	0	0	0	11	88	9	0,237
Veliidae	31	16	0	30	0	0	1	28	0	17	3	0	0	0	0	2	2	0	0	0	131	9	0,354
Mesoveliidae	1	2	0	41	2	0	0	18	0	10	2	1	1	0	0	5	4	0	0	1	87	13	0,234
Pyralidae	3	1	0	11	2	0	0	1	1	9	3	1	0	26	0	31	7	0	0	0	96	13	0,259
Physidae	9	0	0	38	0	0	0	13	6	18	17	0	53	156	0	46	48	0	0	2	405	11	1,094
Talitridae/Doglienotida	259	17	0	266	0	0	1	4	0	27	42	10	28	0	0	10	5	0	0	0	670	11	1,81
Lymnaeidae	0	1	17	4	0	0	0	1	2	36	13	19	29	226	2	246	111	0	0	0	708	13	1,911
Dytiscidae	3	0	1	21	0	0	0	4	1	12	15	6	0	2	2	2	1	0	1	0	72	13	0,194
Leptoceridae	33	0	0	15	0	1	0	2	8	66	14	2	2	25	4	22	0	0	0	2	197	13	0,531
Hydroptilidae	10	3	0	4	7	6	12	24	4	56	33	1	2	52	0	9	0	0	0	0	224	14	0,604
Haliplidae	1	1	0	16	0	0	0	8	1	7	14	4	17	1	7	104	5	0	0	1	188	14	0,507
Coenagrionidae	92	35	0	230	10	1	3	218	26	32	26	4	22	12	0	6	0	0	0	2	718	15	1,938
Caenidae	125	11	1	146	63	0	1	52	37	261	236	301	88	0	0	89	0	0	1	3	1413	15	3,814
Hirudinea	2	1	0	9	5	0	2	2	5	9	1	17	8	9	34	9	1	0	1	2	116	17	0,312
Ostracoda	30	0	114	13	23	0	9	125	11	117	234	53	18	354	1	4	17	5	8	0	1138	17	3,072
Planorbidae	104	36	10	203	7	0	2	36	10	90	47	20	543	0	1	244	331	0	0	5	1688	16	4,558
Notonectidae	6	4	1	88	0	1	1	4	2	32	32	34	1	1	0	1	1	0	3	3	214	17	0,578
Corixidae	29	47	1	35	0	11	1	8	66	8	11	123	7	69	3	10	2	2	17	1	451	19	1,216
Baetidae	3	4	30	7	19	0	1	14	93	23	145	28	6	16	21	4	1	29	17	1	461	19	1,245
Hydracarina	130	0	2	53	59	1	90	151	3	37	43	3	215	2	2	74	1	1	0	2	869	18	2,347
Nemata	108	2	8	53	16	1	44	472	18	626	121	215	166	1	5	418	4	0	1	7	2285	19	6,168
Oligochaeta	136	54	0	547	258	15	96	840	100	477	767	3248	402	19	137	596	7	1	1	406	8107	19	21,89
Ceratopogonidae	82	3	1	181	4	0	17	93	7	180	68	15	84	1	4	314	41	1	0	2	1100	18	2,969
Chironomidae	244	245	96	1516	1084	48	181	351	75	1216	1607	3370	272	69	146	604	48	2076	416	51	13714	20	37,03
Total abundance	1481	515	282	3906	1596	90	465	2618	481	3436	3533	7523	2372	1221	385	3256	738	2114	470	558	37039		
Total number of taxa	38	25	13	42	20	12	18	40	24	40	41	31	31	23	20	35	26	8	12	23	68		100

Table S3-B. Abundance of macroinvertebrate taxa in the 20 studied waterbodies and in each group of sites revealed by k-means partitioning and cluster analysis.

Moyenne	Diptera	Coleoptera	Hemiptera	Trichoptera	Ephemeropt	Odonata	Lepidoptera	Prosobranch	Pulmonées	Bivalvia	Amphipoda	Decapoda	Ostracoda	Oligochaeta	Nemata	Autres	Total
GROUP 1																	
Bizard	1703	58	401	26	152	309	11	1	250	51	266	0	13	547	53	63	3905
Prairies	94	10	84	0	1	0	7	0	489	15	5	0	17	7	4	3	737
MaraisCastors	928	115	319	32	93	11	31	0	536	79	10	0	4	596	418	83	3256
Moyenne	**909**	**61**	**268**	**19**	**82**	**107**	**16**	**0**	**425**	**48**	**94**	**0**	**12**	**383**	**158**	**50**	**2633**
Ecart type	**805**	**53**	**165**	**17**	**76**	**175**	**13**	**0**	**153**	**32**	**149**	**0**	**7**	**327**	**226**	**42**	**1674**
GROUP 2																	
Lacoursiere	356	18	17	4	94	28	0	361	625	0	33	1	18	402	166	250	2372
Cygnes	197	0	4	12	2	3	0	2	2	0	2	0	9	96	44	92	465
Heritage	448	14	62	26	66	244	1	0	82	1	14	0	125	840	472	221	2618
Rmontigny	54	2	6	2	3	2	0	1	18	11	20	22	0	406	7	4	558
Battures	248	1	69	3	15	39	1	9	42	0	27	0	0	54	2	4	515
Centenaire	48	1	12	7	0	1	0	0	0	0	0	0	0	15	1	4	89
Brunante	1088	1	2	7	82	11	2	0	32	0	0	0	23	258	16	72	1596
Moyenne	**349**	**5**	**25**	**9**	**38**	**47**	**1**	**53**	**114**	**2**	**14**	**3**	**25**	**296**	**101**	**92**	**1173**
Ecart type	**357**	**7**	**29**	**8**	**42**	**88**	**1**	**136**	**227**	**4**	**14**	**8**	**45**	**288**	**174**	**104**	**1016**
GROUP 3																	
Lafontaine	70	4	71	77	16	12	26	108	383	0	50	0	354	19	1	30	1221
Angrignon	332	5	71	44	128	100	3	5	115	0	260	3	30	136	108	139	1480
JBA	1409	19	74	124	284	56	9	2	146	6	27	0	117	477	626	59	3435
JBN	1678	35	49	47	380	32	3	2	79	7	42	0	234	767	121	53	3531
LacCastors	3386	13	160	3	329	5	1	31	42	0	10	0	53	3248	215	25	7523
Moyenne	**1375**	**15**	**85**	**59**	**228**	**41**	**8**	**29**	**153**	**3**	**78**	**1**	**158**	**929**	**214**	**61**	**3438**
Ecart type	**1316**	**13**	**43**	**45**	**151**	**39**	**10**	**45**	**134**	**4**	**103**	**1**	**135**	**1329**	**242**	**46**	**2522**
GROUP 4																	
Beaubien	97	1	1	0	31	0	0	1	28	0	0	0	114	0	8	2	282
Pratt1	2077	0	2	0	29	1	0	0	0	0	0	0	5	1	0	1	2114
Pratt2	418	2	19	0	18	0	0	0	0	0	0	0	8	1	1	2	469
Jarry	83	3	68	13	130	28	1	0	17	0	1	0	11	100	18	9	481
Liesse	161	9	4	4	21	3	0	0	3	0	0	0	1	137	5	37	385
Moyenne	**567**	**3**	**19**	**3**	**46**	**6**	**0**	**0**	**10**	**0**	**0**	**0**	**28**	**48**	**6**	**10**	**746**
Ecart type	**855**	**4**	**29**	**6**	**47**	**12**	**0**	**0**	**12**	**0**	**0**	**0**	**48**	**66**	**7**	**15**	**768**
Moyenne GROUP 1,2,3,4	**744**	**16**	**75**	**22**	**94**	**44**	**5**	**26**	**144**	**9**	**38**	**1**	**57**	**405**	**114**	**58**	
Ecart type GROUP 1,2,3,4	**895**	**27**	**107**	**32**	**114**	**84**	**9**	**82**	**200**	**20**	**78**	**5**	**93**	**722**	**183**	**72**	
Minimum GROUP 1,2,3,4	**48**	**0**	**1**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1**	
Maximum GROUP 1,2,3,4	**3386**	**115**	**401**	**124**	**380**	**309**	**31**	**361**	**625**	**79**	**266**	**22**	**354**	**3248**	**626**	**250**	
Somme GROUP 1,2,3,4	**14878**	**312**	**1495**	**432**	**1874**	**887**	**96**	**521**	**2889**	**171**	**768**	**27**	**1138**	**8107**	**2285**	**1153**	**37033**
% GROUP 1,2,3,4	**40**	**1**	**4**	**1**	**5**	**2**	**0**	**1**	**8**	**0**	**2**	**0**	**3**	**22**	**6**	**3**	

Table S4. Values of metrics based on diversity and biotic indices, and functional diversity indices in each waterbody and for each group of sites determined using k-means partitioning and cluster analysis.

	Somme	Nb taxa	Nb taxa ET	SommeET	SommeCOH	%ET	%COH	ETsurCOH	FBI	Shannon	Pielou	Simpson	Fric	Feve	Fdis
GROUP 1															
moyBizard1	3905	35	4	179	2072	4,94	48,05	0,17	7,50	3,09	0,60	0,78	1,643	0,447	0,293
moyMaraisCastors1	3256	29	3	125	1209	2,19	35,99	0,07	6,61	3,24	0,67	0,85	1,513	0,420	0,303
moyPrairies1	737	18	0	1	56	0,04	17,99	0,01	6,75	2,76	0,68	0,78	0,997	0,527	0,237
moyenne	2633	27	3	101	1112	2,39	34,01	0,08	6,95	3,03	0,65	0,80	1,38	0,46	0,28
ecart -type	1673,59	8,44	2,08	91,31	1011,12	2,45	15,13	0,08	0,48	0,25	0,04	0,04	0,34	0,06	0,04
GROUP 2															
moyBattures1	515	19	3	18	300	4,30	44,68	0,16	7,20	2,67	0,63	0,73	1,297	0,590	0,300
moyBrunante1	1596	16	2	89	1348	5,05	78,73	0,06	7,79	2,05	0,52	0,61	1,347	0,567	0,260
moyCentenaire1	89	9	1	7	63	8,76	70,52	0,15	7,15	1,94	0,61	0,61	0,840	0,570	0,263
moyCygnes1	465	14	2	13	279	2,41	63,93	0,04	7,25	2,43	0,65	0,74	1,000	0,560	0,300
moyHeritage1	2618	31	4	92	1193	3,77	49,51	0,09	7,14	2,89	0,59	0,78	1,573	0,453	0,283
moyLacoursiere1	2372	26	3	98	682	3,67	34,28	0,12	6,97	3,23	0,69	0,85	1,200	0,480	0,280
moyRmontigny1	558	17	2	5	458	0,96	84,12	0,01	7,64	1,45	0,35	0,39	1,233	0,667	0,157
moyenne	1173	19	3	46	618	4	60,82	0,09	7,31	2,38	0,58	0,67	1,21	0,56	0,26
écart-type	1015,73	7,35	0,77	44,21	485,94	2,44	18,53	0,06	0,30	0,61	0,11	0,15	0,24	0,07	0,05
GROUP 3															
moyAngrignon1	1480	28	4	173	382	10,40	29,99	0,44	7,09	3,44	0,72	0,87	1,420	0,440	0,333
moyJBA1	3435	31	4	408	1702	14,32	50,93	0,33	7,06	2,88	0,58	0,76	1,453	0,557	0,293
moyJBN1	3531	32	4	428	2375	25,45	40,50	1,13	7,06	2,88	0,59	0,77	1,773	0,493	0,290
moyLacCastors1	7523	24	3	333	6635	11,75	78,23	0,17	7,63	1,95	0,43	0,66	1,330	0,477	0,290
moyLafontaine1	1221	14	1	93	97	2,65	19,35	0,37	6,98	2,61	0,73	0,78	0,930	0,570	0,270
moyenne	3438	26	3	287	2238	12,92	43,80	0,48	7,16	2,75	0,61	0,77	1,38	0,51	0,30
écart-type	2522,01	7,27	1,44	147,60	2629,97	8,25	22,56	0,37	0,26	0,54	0,12	0,07	0,30	0,05	0,02
GROUP 4															
moyBeaubien1	282	10	1	31	96	13,72	34,65	0,92	7,03	2,01	0,60	0,64	0,637	0,523	0,280
moyJarry1	481	20	4	142	180	26,76	34,23	1,44	6,38	2,96	0,68	0,80	1,323	0,537	0,293
moyLiesse1	385	16	2	25	317	6,69	81,87	0,08	7,51	2,17	0,55	0,64	1,240	0,617	0,283
moyPratt1.1	2114	6	1	29	2077	35,59	59,22	1,76	6,42	0,65	0,39	0,25	0,320	0,700	0,100
moyPratt2.1	469	9	1	18	418	16,08	65,29	0,70	7,03	1,38	0,46	0,42	0,867	0,703	0,163
moyenne	746	12	2	49	617	19,77	55,05	0,98	6,87	1,83	0,54	0,55	0,88	0,62	0,22
écart-type	768,49	5,93	1,07	52,46	825,06	11,41	20,56	0,65	0,48	0,87	0,11	0,22	0,42	0,09	0,09
moyenne GROUP 1,2,3,4	1851,63	20,19	2,57	115,31	1096,84	9,97	51,10	0,41	7,11	2,43	0,59	0,69	1,20	0,54	0,26
écart-type GROUP 1,2,3,4	1825,80	8,79	1,26	131,69	1506,50	9,66	20,68	0,51	0,38	0,72	0,11	0,17	0,36	0,08	0,06

Figure S1. Principal component analysis (PCA) on the matrix of environmental variables of the 59 sampling sites in the 20 waterbodies in scaling 1.

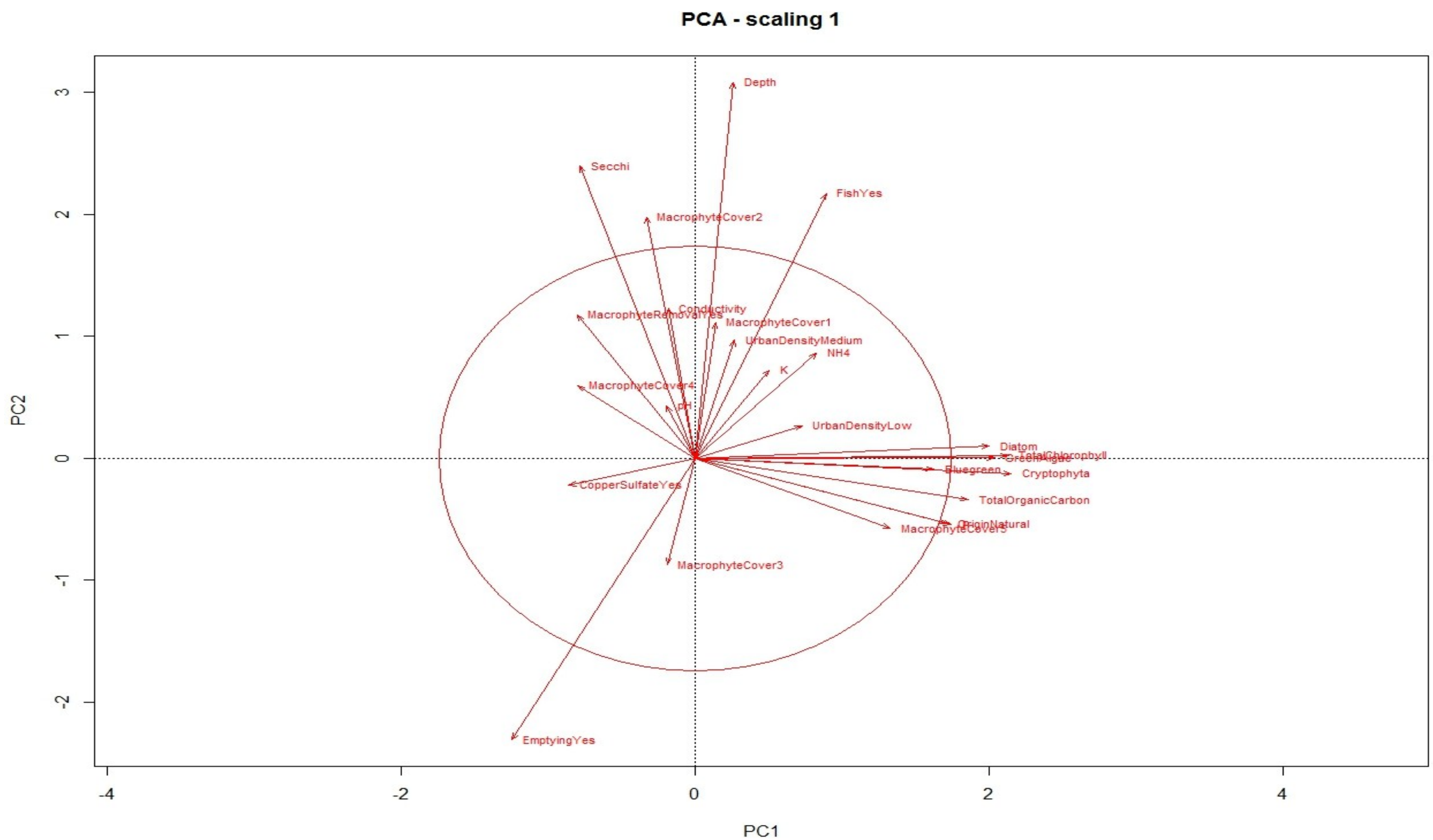


Figure S2. K-means partitioning based on environmental variables using the local maximum of the Calinski-Harabasz criterion (maximum established for 4 groups).

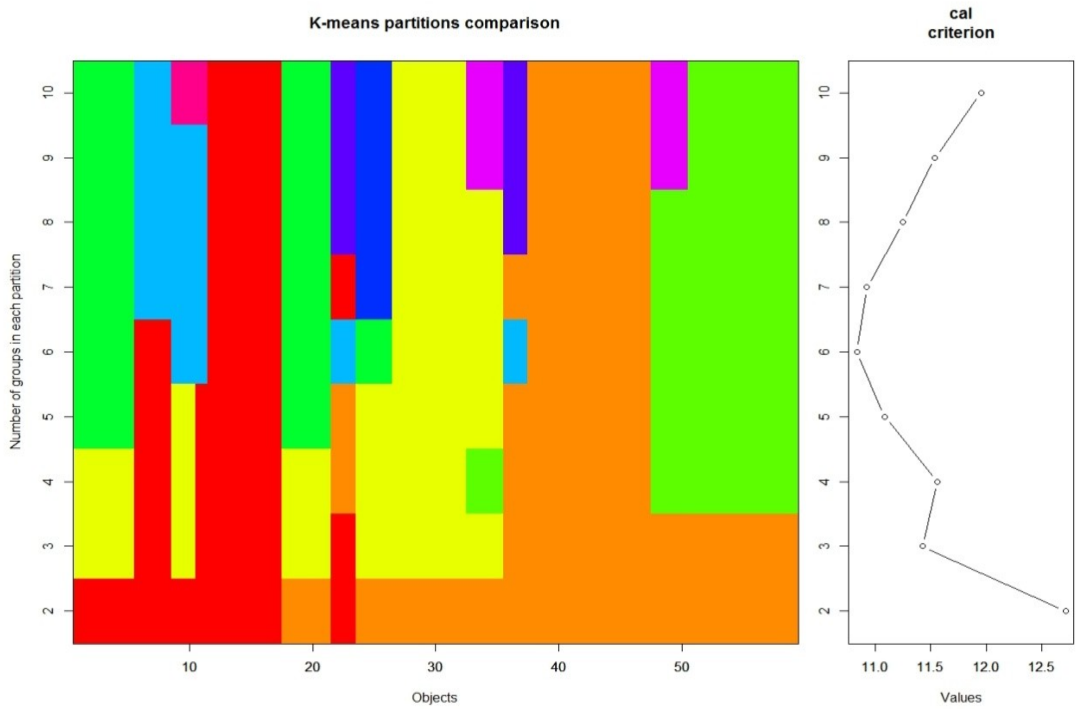


Figure S3. Principal component analysis (PCA) on the matrix of macroinvertebrate univariate metrics based on diversity, biotic, and functional indices of the 59 sampling sites of the 20 waterbodies in scaling 1.

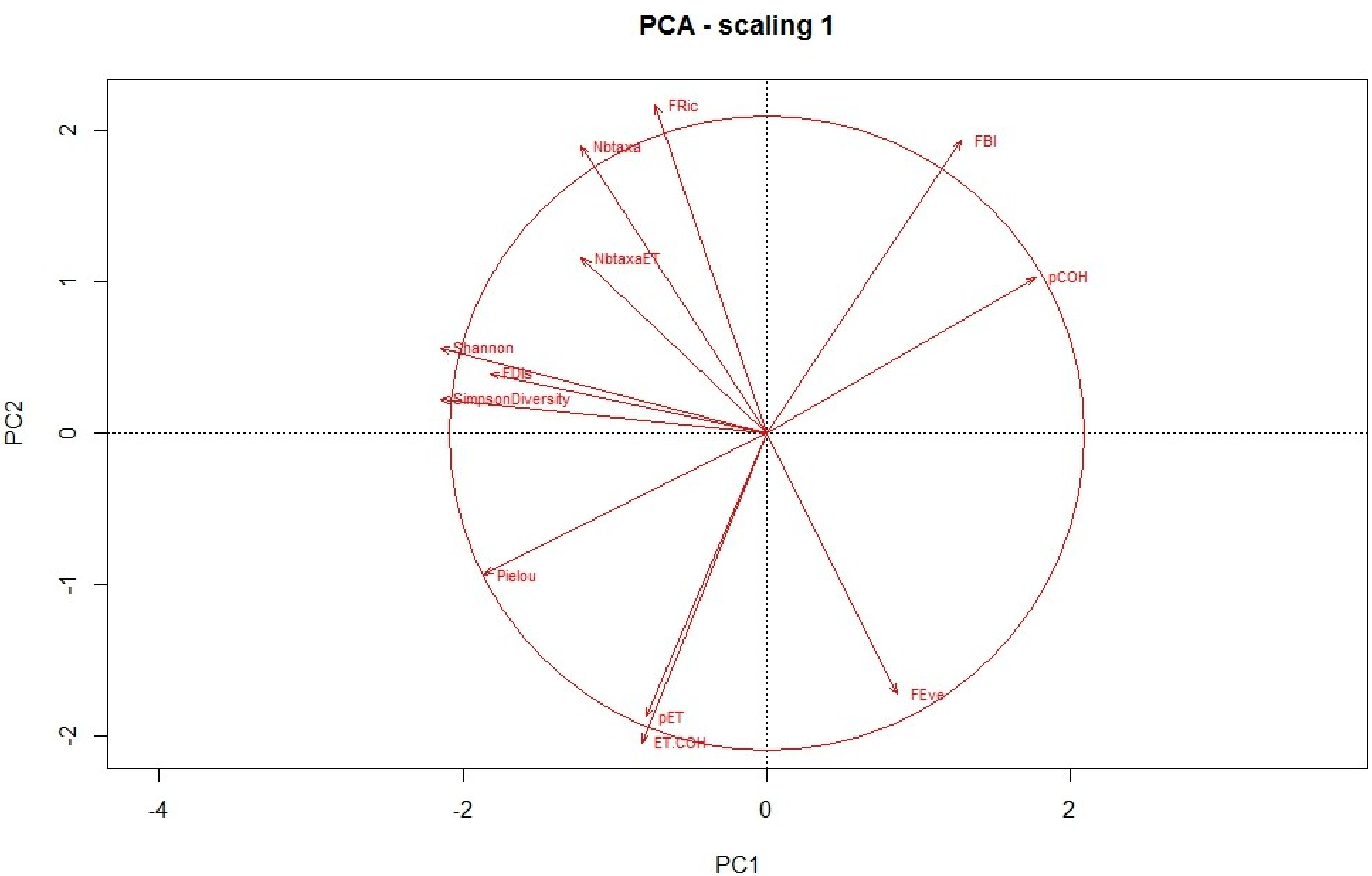
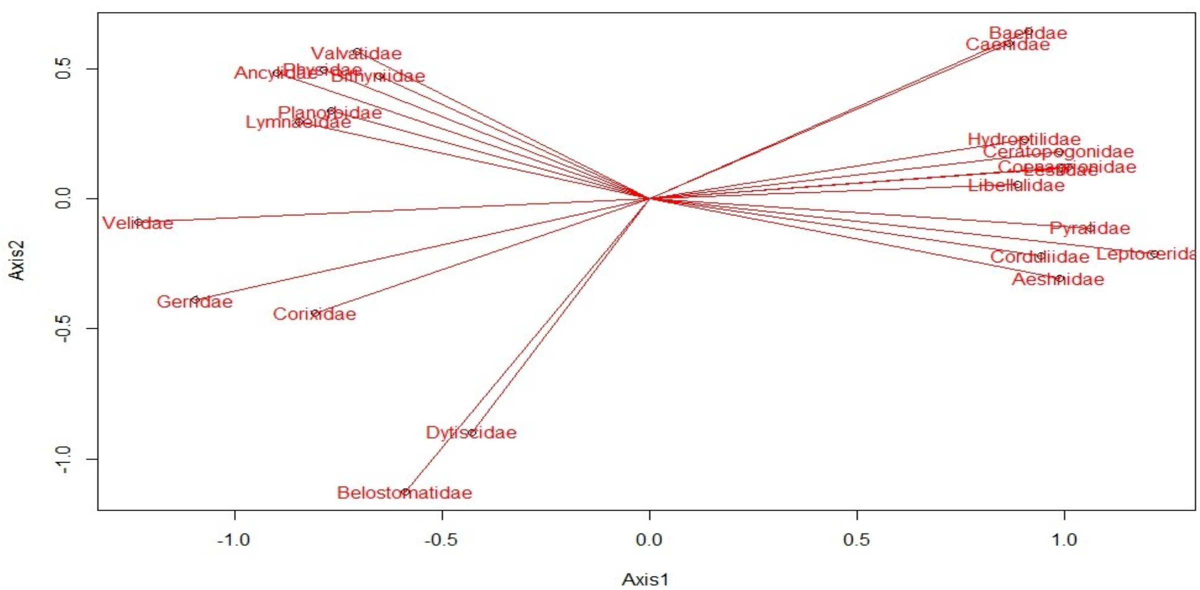


Figure S4. Principal component analysis on the matrix of macroinvertebrate families (A) and functional traits of each family (B) (axis 1 explained 36% of total variation and axis 2 explained 16% of total variation).

A



B