The aquatic macroinvertebrate biodiversity of urban ponds in an European town (Loughborough, UK)

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Abstract

Urbanization is one of the greatest threats to freshwater biodiversity, with the area of land covered by towns and cities predicted to increase significantly in the future. Ponds are common features in the urban landscape and have been created for a variety of reasons ranging from ornamental/amenity purposes through to the detention of urban runoff and pollution. This paper aims to quantify the aquatic macroinvertebrate biodiversity associated with garden, ornamental and other urban ponds in Leicestershire, UK. We examined the macroinvertebrate biodiversity of 41 urban ponds (13 garden, 12 park and 16 other urban ponds) within the town of Loughborough, UK. Park ponds supported greater macroinvertebrate richness than garden or other urban ponds. Garden ponds were the most taxon poor. Pond size was strongly correlated with macroinvertebrate diversity. Collectively, urban ponds were found to be physically and biologically heterogeneous and were characterised by high community dissimilarity. Urban ponds provide a diverse range of habitats for a mixture of common and rare aquatic macroinvertebrate taxa and represent a valuable biodiversity resource within anthropogenically dominated landscapes. Recognition of the significant contribution of ponds to urban freshwater biodiversity is important for future aquatic conservation within anthropogenically dominated landscapes.

Key Words: urbanization, small lentic waterbodies, artificial habitat, anthropogenic ecosystems, macroinvertebrate, pond management

Introduction

The proportion of landscape covered by urban development has significantly increased globally in the last century and human populations are becoming more concentrated within cities (United Nations, 2014). Urbanized land is projected to increase up to 185% from current levels by 2030 (Seto et al., 2012) and 66% of global population is predicted to live in urban areas by 2050 (United Nations, 2014). The threat of urbanization to biodiversity has been well documented (McKinney, 2002; Shochat et al., 2010) and the widespread increase in urban land-cover has resulted in the fragmentation of wildlife habitats in the wider landscape (Goddard et al., 2010), and a reduction in species richness (McKinney, 2002; McKinney, 2008). Urbanization has been widely reported to result in habitat and biotic homogenization, increased disturbance frequency and implicated in the successful introduction and proliferation of non-native taxa (Grimm et al., 2008; Niinemets & Penuelas, 2008). However, floral richness has been identified to peak at moderate levels of urbanization (McKinney, 2008).

Urban spaces can be largely characterized into two groups; 1) highly developed landscapes such as residential, commercial and industrial spaces and; 2) open areas including parks, wasteland and domestic gardens (Goertzen & Suhling, 2013). As the population within urban areas increases, compact developments with high density commercial and residential areas (buildings and urban infrastructure) will expand at the expense of open ‘green’ spaces (Dallimer et al., 2011). This is likely to put further pressure on urban biodiversity, potentially leading to increased local extinction rates (Sushinsky et al., 2013).

The growing need for the protection and conservation of freshwater biodiversity has been raised on the international political agenda in recent years. The United Nations launched and supported an international decade (2005-2015) for action on ‘water for life’ with a special emphasis on highly modified and fragmented landscapes (Dudgeon et al., 2006). Many freshwater waterbodies located within urban landscapes are under significant pressure from anthropogenic disturbances (Urban et al., 2006) including pollution (Paul & Mayer, 2001; Dudgeon et al., 2006) and habitat modification / loss

(Gopal, 2013). Conservation and management of biodiversity within urban areas currently relies heavily on the designation of areas protected by planning or regulatory outcomes (Mcdonald et al., 2008; Chester & Robson, 2013). However, the sustainable management of biodiversity should not be expected to depend exclusively on protected sites as growing urban populations and land-cover change are projected to threaten many existing designated areas (Chester & Robson, 2013; Guneralp & Seto, 2013). Research addressing the biodiversity of urban regions has historically focused on terrestrial and lotic habitats (e.g., Paul & Mayer, 2001; Fontana et al., 2011), with research specifically quantifying aquatic macroinvertebrate biodiversity within urban ponds being limited (e.g., Gledhill et al., 2008; Goertzen & Suhling, 2013; Briers, 2014; Noble & Hassall, 2014; Hassall & Anderson, 2015). The increase in urban land-cover highlights the pressing need to examine urban biodiversity of small lentic waterbodies (ponds) in an increasingly challenging environment. Ponds within urban landscapes have been created for a variety of reasons including: i) ornamental and amenity purposes (e.g., garden, urban park and school ponds); ii) to reduce urban runoff and / or improve urban water quality (e.g., sustainable urban drainage ponds and detention ponds) (Heal et al., 2006; Williams et al., 2013; Hassall, 2014) and; iii) industrial ponds (remnant and current e.g., mill ponds) that were created historically and persist, although their original function may be redundant (Wood et al., 2001). It is widely recognised that ponds are abundant in urban landscapes (Goertzen & Suhling, 2013) and many persist due to their amenity value (Wood et al., 2003). As urban landscape features, ponds are typically aesthetically pleasing to the general public, small in size and as a result perceived to be relatively easy to manage (Biggs et al., 1994), and thus may provide a relatively easy but effective means to conserve aquatic biodiversity.

Ponds are acknowledged to support high aquatic faunal and floral diversity at a regional scale and provide important habitats for a number of rare and endemic species in the urban and wider environment (Williams et al., 2003; Biggs et al., 2005; Colding et al., 2009; Goertzen and Suhling, 2013). Their physicochemical heterogeneity affords a wide range of niches for aquatic macroinvertebrate taxa (Williams et al., 2003), making them suitable habitats for a wide range of taxa. Many macroinvertebrate species are active colonisers and can migrate between ponds, with wellconnected pond networks typically supporting very high macroinvertebrate biodiversity (Gledhill et al., 2008; Williams et al., 2008).

This paper specifically aims to quantify the aquatic macroinvertebrate biodiversity associated with different types of urban pond. A wide variety of pond types are present in town and city regions ranging from those located in urban parks, urban drainage detention ponds and garden ponds (Hassall, 2014). We sought to test the following research questions: i) Is aquatic macroinvertebrate biodiversity related to pond size and location in the urban landscape (garden, parkland and other urban settings)? ii) Does macroinvertebrate community heterogeneity (β diversity) vary among urban pond types as a result of microhabitat availability and physicochemical variability?

Materials and methods

Study sites

The town of Loughborough (Leicestershire, UK) covers an area of approximately 35 km^2 and has a total population of approximately 60,000. A total of 41 ponds were sampled in the town of Loughborough and the surrounding urban environment. Three types of urban ponds were examined in detail: i) 13 garden ponds - typically small water bodies located within the boundary of an urban domestic residential building; ii) 12 park ponds - waterbodies of varying size located within public spaces (e.g., parks), serving as amenity features and typically with easy access to some or all of the pond perimeter (Hassall, 2014) and; iii) 16 other urban ponds - those that did not fit into a category but varied in size and were located in high density, compact developments often on private land with controlled access (these comprised 9 urban drainage ponds - 5 of which were ephemeral in nature and dried at least once during the survey period, 4 located within school grounds and used as wildlife / education tools and 3 ponds surrounded by high density commercial developments). Urban ponds of all types frequently have a concrete / synthetic substrate / base, steep bank sides and may hold water on a permanent or temporary basis.

Macroinvertebrate sampling

Aquatic macroinvertebrate sampling was undertaken using a standard pond net (mesh size, 250µm) with the total time (maximum 3 minutes; Biggs et al., 1998) used to sample each pond being proportional to its surface area (Hinden et al., 2005). Ponds with a surface area greater than $50m^2$ were sampled for a total of three minutes; for smaller ponds 30 seconds of sampling for every $10m^2$ surface area was employed. In addition, a manual inspection of any hard surfaces or larger substrates (e.g., rocks) was undertaken for up to 60 seconds to ensure that all available microhabitats and surfaces were examined. This method recognises both the size of the waterbody and the habitat diversity contained within it. However, other sampling methods could yields greater diversity of particular macroinvertebrate taxa (e.g., Odonata) such as sampling for 15 to 20 minutes (Ruggiero et al., 2008). Macroinvertebrate samples were taken on three occasions coinciding with the spring, summer and autumn seasons (high, intermediate and low water levels respectively) in each pond. For each pond the presence and proportion (% of surface area) of aquatic microhabitats were recorded within the following categories; i) open water, ii) submerged vegetation, iii) emergent vegetation and; iv) floating vegetation. The total sampling time at each pond was divided equally between the microhabitats present. However, if an individual pond was dominated by a single microhabitat, sample time was divided further to reflect this (Biggs et al., 1998). In the laboratory, invertebrate samples from each habitat were processed separately and preserved in 70% industrial methylated spirits prior to identification. The majority of invertebrate fauna was identified to species level, although Diptera larvae, Planariidae, Collembola and Hydrachnidiae were identified to the lowest possible taxonomic level.

Environmental data collection

At each pond site, surface area (m^2), mean water depth (cm), hydroperiod (duration during the 12-month study period that the pond contained water or was dry), the percentage of the pond margin and pond surface shaded by overhanging vegetation, substratum composition (percentage gravel, sand and silt), bank structure (percentage natural earth, wood, synthetic concrete and stone), the presence of fish and evidence of pollution were recorded. Conductivity (μS cm^{-1}), pH and water temperature, were recorded in the field using a Hanna conductivity meter (HI198311) and a Hanna pH meter (HI98127). Dissolved oxygen (DO mg l^{-1}) was recorded at each pond site using a Mettler Toledo Dissolved Oxygen Meter (SG6). Pond connectivity, the number of hydrological connections to other waterbodies, and pond isolation, the number of other waterbodies within 500m (Waterkeyn et al., 2008), were recorded using GIS software (ArcMap 10.1). Every attempt was made to record all waterbodies within 500m of each pond site, however garden ponds associated with small domestic dwellings were particularly difficult to identify as they were not recorded on national maps (OS MasterMap) and difficult to observe using aerial images provided by Google Earth software (Google Earth, 2015). As a result it is acknowledged that a small number of garden ponds may have been missed.

Statistical analyses

Community abundance (total number of individuals per site), taxon richness and alpha diversity indices (Shannon Wiener Diversity index and the Berger Parker Dominance index) were calculated for each pond site and microhabitat using the Species Diversity and Richness IV software (Pisces Conservation, 2008). To achieve this all species abundance data for individual ponds - for each season (total diversity), and for the individual microhabitats within them (e.g., diversity within the submerged macrophyte habitat during spring, summer and autumn), were combined in the final analysis to provide a measure of diversity within each pond and microhabitat respectively. Prior to statistical analysis the data were examined to ensure that they complied with the underlying assumptions of parametric statistical tests (e.g., normal distribution). Where these assumptions were violated (e.g., for community abundance data) they were $\log_{10}$ transformed. Differences in faunal biodiversity among the pond types and microhabitats (open water, emergent macrophytes, submerged macrophytes and floating macrophytes) were examined using a nested analysis of variance (nested ANOVA) with subsequent use of Tukey *post hoc* tests to determine where significant differences between groups occurred (Van de Meutter *et al.,* 2005). Pond type and microhabitat were included as fixed effects and site was nested within pond type as a random effect. One-way analysis of variance was used to examine differences in physicochemical parameters among the ponds. Pearson correlation coefficients between environmental parameters and invertebrate community metrics were calculated and scatter plots inspected. Analyses were undertaken in IBM SPSS Statistics (version 21, IBM Corporation, New York). Beta (β) diversity was measured using Jaccard's Coefficient of Similarity (Cj: using total pond macroinvertebrate data) in the Community Analysis Package 3.0 program (Pisces Conservation, 2004). Differences in Jaccard's Coefficient of Similarity among the ponds were assessed using one-way analysis of variance (ANOVA). Heterogeneity of macroinvertebrate communities between urban pond sites were assessed using Analysis of Similarity (ANOSIM) in PRIMER v6 (Clarke & Gorley, 2006). Faunal abundance data was log $(X + 1)$ transformed prior to ANOSIM.

The relationship between macroinvertebrate assemblages and physicochemical gradients was examined using Redundancy Analysis (RDA) in CANOCO 4.5 (Leps & Smilauer, 2003). Due to natural seasonal variability in community composition, seasonal data from individual pond sites were combined and mean values of environmental parameters derived. Environmental parameters were $\log_{10}$ transformed prior to analysis to reduce the influence of skew and eliminate their physical units (Legendre & Birks, 2012). Faunal abundance data was Hellinger transformed following the approach of Legendre & Gallagher (2001). The statistical significance of associations between each of the environmental variables and the RDA axes were determined using a forward selection procedure, employing a random Monte-Carlo permutations test (999 random permutations) with Bonferroni correction. Only the physical, chemical and spatial parameters significantly influencing the faunal distribution ($p<0.05$) were included in the final model.

Results

Physicochemistry

Physicochemical conditions varied widely among all urban ponds. Dissolved oxygen, pH, conductivity, percentage of pond margin / water surface area shaded, submerged macrophytes and emergent macrophytes were not significantly different between urban pond types (all ANOVA $p>0.05$). Where submerged macrophytes were present there was often a diverse range of taxa including Characeae, *Elodea* spp. and Haloragaceae. Emergent macrophyte microhabitats were dominated by sedges and reeds, whilst floating macrophyte microhabitats were dominated by *Lemna* spp. *Post hoc* analysis indicated that park ponds had a significantly greater mean surface area (ANOVA $F_{2, 40} = 43.282$; $p<0.001$) and depth (ANOVA $F_{2, 40} = 15.963$; $p<0.001$) than other urban and garden ponds when all sampling dates were considered. Other urban ponds were significantly larger than garden ponds. The proportion of the surface area covered by floating macrophytes was significantly higher in garden ponds than the other pond types (ANOVA $F_{2, 40} = 9.960$; $p<0.001$).

Macroinvertebrate biodiversity

The mean abundance of aquatic invertebrates recorded from all urban ponds was 1880, with the lowest recorded within garden ponds (mean: 863, range: 45-2379), followed by other urban (mean: 2013, range: 39-6766) and park ponds (mean: 2804, range: 303-6628). A total of 170 taxa were recorded from 18 orders and 60 families from the garden (total: 44 taxa, range: 2-24), other urban (total: 91 taxa, range: 3-42) and park ponds (total: 149 taxa, range: 4-61). The invertebrate taxa most widely distributed across the urban pond sites were: Chironomidae (40 ponds), Oligochaeta (40 ponds), *Asellus aquaticus* (Isopoda: Asellidae) (27 ponds) and *Crangonyx pseudogracilis* (Amphipoda: Gammaridae) (25 ponds). Four nationally scarce or nationally notable Coleoptera were recorded within the urban ponds; *Helochares punctatus* (Hydrophilidae), *Agabus uliginosus* (Dytiscidae) and *Gyrinus distinctus* (Gyrinidae) were all recorded once within 3 separate park ponds and *Helophorus strigifrons* (Hydrophilidae) was recorded from 1 other urban pond and 1 park pond.

Two non-native taxa were recorded from the urban ponds examined. The most widespread non-native taxon was the amphipod *Crangonyx pseudogracilis* recorded from 61% (25) of the ponds (5 garden ponds, 9 other urban ponds and 11 park ponds) and was abundant (accounting for up to 68% of community abundance) in many of the sites where it occurred. The non-native Gastropoda, *Potamopyrgus antipodarum* (Gastropoda: Hydrobiidae) was recorded from 22% of the ponds (4 other urban ponds and 5 park ponds). Examination of community indices in association with physicochemical and spatial characteristics indicated that the most significant correlations were recorded between water surface area ($\log_{10}$) and community abundance ($\log_{10}$), taxon richness, Shannon Wiener diversity, and the Berger Parker Dominance index (Fig. 1). The area of the pond margin shaded and pond connectivity to other waterbodies also displayed a number of significant correlations with community parameters (Table 1).

A significant difference in community abundance among the three urban pond types was recorded (ANOVA $F_{38, 143} = 3.627$; $p<0.001$) (Fig. 2). *Post hoc* analysis indicated that community abundance was significantly greater in park and other urban ponds than garden ponds ($p<0.05$). The greatest number of taxa recorded was from a park pond (61 taxa) located within an urban green space. Park ponds supported significantly higher taxon richness (ANOVA $F_{38, 143} = 2.917$; $p<0.001$) than the other two pond types (Fig. 2b). The Shannon Wiener Diversity index was significantly higher within park than ‘other’ urban ponds and garden ponds (ANOVA $F_{38, 143} = 3.945$; $p<0.001$) (Fig. 2c), and was also recorded to be significantly higher in other urban ponds compared to garden ponds. The Berger Parker Dominance index indicated that garden ponds were dominated by a small number of macroinvertebrates compared to other urban or park ponds (ANOVA $F_{38, 143} = 2.551$; $p<0.001$) (Fig.2d).

When individual microhabitats (open water, emergent, submerged and floating leaved macrophytes) were examined, a significant difference in the number of taxa (ANOVA $F_{3, 143} = 9.988$; $p<0.001$) and community abundance (ANOVA $F_{3, 143} = 6.621$; $p<0.001$) was observed among urban ponds. Faunal abundance was typically greater in submerged and floating leaved macrophytes, with the exception of garden ponds where floating leaved macrophytes contained fewer individuals (Fig. 2a). Taxon richness was higher within submerged macrophytes and emergent macrophytes than open water and floating macrophytes across all pond types (Fig. 2b). The Berger Parker Dominance index and Shannon Wiener Diversity Index did not differ among microhabitats ($p>0.05$) (Fig. 2c and Fig. 2d).

Community heterogeneity

A significant difference in community composition was recorded between garden and park ponds (ANOSIM $p<0.001$). Discrete groups of park and garden ponds were observed within the RDA ordination plot (Fig. 3) suggesting that garden and park ponds supported distinct invertebrate communities. Other urban ponds were widely dispersed and overlapped all other pond types in the RDA biplot. Other urban ponds had significantly lower Jaccard's Coefficient of Similarity value (C_j = 0.19) than garden pond sites (C_j = 0.27) and park pond sites (C_j = 0.24) (ANOVA $F_{2, 263} = 10.897$ $p<0.001$). When all urban ponds were considered urban pond macroinvertebrate community composition was highly heterogeneous (C_j = 0.18) indicating that there was marked heterogeneity (β diversity) in the macroinvertebrate communities recorded within individual urban ponds.

Macroinvertebrate community - environmental parameters relationships

Redundancy Analysis (RDA) of the pond macroinvertebrate community data and environmental parameters highlighted clear differences between the three urban pond types. The RDA axes were highly significant (Monte Carlo significance test: $F=2.038$ $p<0.002$) with the first four axes explaining 26.1% of the variation in species data (axis 1: 11.4%, axis 2: 6.8%, axis 3: 5.7% and axis 4: 2.2%) and 96% of the taxa-environment relationship (axis 1: 42.1%, axis 2: 24.7, axis 3: 21.1% and axis 4: 8.1). Forward selection identified five significant physicochemical variables correlated with the first two RDA axes: water surface area, pH, emergent macrophytes (all $p<0.005$), submerged macrophytes and the dry phase ($p<0.05$) (Fig. 3a). Pond connectivity and pond isolation did not significantly influence urban pond macroinvertebrate community composition in the analysis.

When the invertebrate assemblages of the three pond types were examined in relation to environmental variables, park and garden pond invertebrate communities were separated on the first and second axes along gradients associated with pond surface area and emergent and submerged macrophyte cover (Fig. 3a). Park ponds were characterised by a greater water surface area, and emergent and submerged macrophyte cover, whilst garden ponds were characterised by smaller surface areas and less emergent and submerged macrophytes (Fig. 3a). Other urban ponds had highly variable environmental characteristics but were associated with greater proportions of emergent macrophyte cover, a small surface area and a dry phase; a small number were ephemeral (5 ponds) and dried during one survey, and plotted at the positive end of axis 2 (Fig. 3a). Urban ponds with the greatest taxon richness and Shannon Wiener diversity were typically associated with greater water surface area, greater submerged macrophytes and emergent macrophytes (Fig. 3b and Fig. 3c).

The RDA faunal plot indicated several species of Odonata (e.g., *Aeshna mixta* and *Erythromma najas*), Hemiptera (e.g., *Sigara dorsalis*, *Notonecta glauca* and *Corixa punctata*), Coleoptera (e.g., *Haliplus confinis* and *Noterus clavicornis*) and Gastropoda (*Lymnaea stagnalis* and *Segmentina complanata*) were associated with ponds with larger surface areas (Fig. 4). In addition, a number of Coleoptera (*Hydrobius fuscipes*, *Agabus sturmii*, Dytiscidae larvae and *Hydroporus pubescens*), and Gastropoda (e.g., *Lymnaea peregra* and *Potamopygrus antipodarum*) were associated with ponds that had high emergent macrophyte cover (Fig. 4). Taxa associated with submerged macrophytes included several species of Hemiptera (e.g., *Sigara falleni*, *Sigara distincta* and *Callicorixa praeusta*) and Trichoptera (e.g., *Limnephilus marmoratus* and *Phryganea bipunctata*). Relatively high abundances of Diptera larvae (Chironomidae, Simuliidae and Culicidae) were typically recorded within garden ponds (Fig. 4). Psychodidae, Scirtidae larvae, Collembola and Stratiomyidae were associated with ephemeral ponds and plotted at the positive end of RDA-axis 1 (Fig. 4).

Discussion

Macroinvertebrate diversity

Park ponds in this study were recorded to support the greatest invertebrate richness whilst garden ponds were found to support the lowest diversity among the three urban pond types and were frequently dominated by dipteran larvae. However, it is important to acknowledge that Diptera were only identified to family level in this study and it is likely that garden ponds may support high dipteran species diversity that has not been quantified in this study. Garden ponds in Sheffield, UK, were also dominated by Diptera larvae and supported a limited number of invertebrate taxa (Gaston et al., 2005a). The high macroinvertebrate diversity recorded from park ponds in this study was greater than that recorded from urban ponds in Halton (Lancashire, UK): total = 119 taxa, and urban drainage ponds in Dunfermline (Fife, UK): total = 66 taxa (Gledhill et al., 2008; Briers, 2014). A significant proportion of the regional macroinvertebrate species pool (228 taxa, Hill, 2015) was represented within the urban ponds (170 taxa). High regional invertebrate diversity has also been recorded within aquatic urban systems in the Netherlands (Vermonden et al., 2009) and for other organisms such as waterbirds (Santoul et al., 2009) and amphibians (Brand & Snodgrass, 2009).

However, at the scale of the individual pond, macroinvertebrate diversity within ponds was variable, ranging from 2 to 61 taxa. This almost certainly reflects both the physical and chemical heterogeneity of the ponds, but also their location within structurally complex and highly fragmented anthropogenic settings. Many of the most taxon rich park ponds were located in 'green spaces' which may have acted as a buffer zone protecting aquatic taxa from runoff from anthropogenic surfaces and disturbances. The importance of buffer zones in the conservation of amphibian populations has been highlighted (Semlitsch & Bodie, 2003), although there has been limited research assessing their effectiveness in relation to macroinvertebrate biodiversity within ponds (Langley et al., 1995).

A positive association was observed between macroinvertebrate diversity and pond surface area in this study and is a pattern that has been documented in some (e.g., Biggs et al., 2005; Nilsson & Svensson, 1995; Ruggiero et al., 2008) but not all pond biodiversity studies (Scheffer et al., 2006; Nakanishi et al., 2014). The significant relationship between diversity and urban pond size in this study may be the result of a reduction in predator density and greater habitat diversity in larger ponds, and greater disturbance and stochastic events in smaller ponds (Whittaker & Fernandez-Palacios, 2007). However, Oertli et al. (2002) demonstrated that the influence of pond size can vary depending on the macroinvertebrate group; Odonata had a relatively strong correlation with pond size, whilst Coleoptera, Sphaeriidae and overall faunal richness displayed a weak association with pond size. In addition, research has identified that a series of smaller ponds can support a greater diversity than a single larger pond (Oertli et al., 2002). The small catchment area of ponds can enable quite different environmental conditions to develop (reflecting local microsite conditions and stochastic effects (Scheffer et al., 2006)) even in ponds that are in close geographical proximity to each other (Davies et al., 2008). This high physicochemical heterogeneity provides a wide range of conditions/niches for flora and fauna to colonize and at a regional scale ponds have been demonstrated to support greater aquatic macrophyte and macroinvertebrate diversity than larger ponds and other waterbodies (Oertli et al., 2002; Williams et al., 2003; Biggs et al., 2005).

The proximity of urban ponds to other waterbodies was not identified as a significant influence on macroinvertebrate community composition in this study. Urban ponds, especially those in domestic gardens are often surrounded by walls, fences or buildings (barriers), typical of urban landscapes. These physical barriers may significantly reduce pond connectivity and the ability of invertebrate taxa to disperse or colonise new habitats, even if they are in close geographical proximity. Urban ponds with greater hydrological connectivity to other waterbodies were significantly correlated with macroinvertebrate diversity and highlight the importance of even ephemeral linkages in the landscape (e.g., ditches, ephemeral channel and/or connectivity resulting from flooding) for the dispersal and colonisation of taxa between urban ponds. The small size of garden ponds (typically around $10m^2$), their management practices (e.g., maintenance of open water, reduced macrophyte cover and/or actively managed to prevent succession) and their high turn-over, due to changes in house ownership and garden management fashions, may significantly limit the ability of garden ponds to replicate the habitat diversity of ponds within the wider urban and rural landscape (Gaston et al., 2005b). However, despite these limitations and low alpha diversity, garden ponds make an important contribution to the regional urban species pool (Gaston et al., 2005a; Gledhill et al., 2008). Given the high abundances of garden ponds, estimated to be between 2.5 - 3.5 million in the UK (Davies et al., 2009), future research is required to examine their potential to serve as refugia for macroinvertebrate communities.

Greater public awareness and guidance regarding the best management practices may also enhance the biodiversity value of garden ponds in the future.

Macroinvertebrate community heterogeneity

Substantial macroinvertebrate community heterogeneity was observed within and between urban pond types. The high community dissimilarity recorded demonstrates that urban ponds provide a range of habitats / niches for invertebrate taxa to utilise. Other urban ponds in this study were shown to have a high dissimilarity in community composition. This reflects the varying pond successional stages, the diverse physicochemical characteristics and management practices (there were a wide range of pond types in the other urban pond group, e.g., stormwater retention ponds, golf course ponds and school ponds) observed among the other urban ponds (Biggs et al., 1994; Williams et al., 2003; Nicolet et al., 2004; Biggs et al., 2005). This inter-pond, spatial dissimilarity also reflects the different levels of management that urban ponds are subject to; ranging from regular active management through to an absence of intervention. Macroinvertebrate communities from ponds in the wider landscape have also been shown to display significant temporal heterogeneity and turnover of species, which can cause temporal variation in the conservation value of pond habitats (Jeffries, 2011; Hassall et al., 2012). Future research is required to examine the nature of temporal heterogeneity of urban pond communities and the implications for the conservation of urban biodiversity.

Urban pond conservation and management

Despite their largely anthropogenic origin and the presence of several non-native taxa (*C. pseudogracilis* and *P. antipodarum*) the results clearly demonstrate that many urban ponds can support species rich invertebrate communities including taxa of conservation interest (Hassall & Anderson, 2015). Urban ponds potentially have a vital role to play in reducing aquatic habitat fragmentation and serving as stepping stones in anthropogenic / disturbed landscapes (Chester & Robson, 2013). Ponds not only contribute to ongoing conservation efforts but may actively enhance freshwater biodiversity in the urban region (Le Viol et al., 2009; Vermonden et al., 2009; Briers, 2014). In common with other studies of urban ponds this research indicated that the majority supported generalist taxa (Gledhill et al., 2008; Goertzen & Suhling, 2013; Briers, 2014). However, urban ponds may be particularly important habitat for motile taxa such as Coleoptera and Odonata which can opportunistically colonise available habitat aerially (Scher & Thiery, 2005; Goertzen & Suhling, 2013). A number of these active colonisers with an aerial adult life-stage were well represented and among the most species rich groups recorded (Coleoptera, Trichoptera and Hemiptera) in the urban ponds studied.

However, this study, and others (Noble & Hassall, 2014) have also demonstrated that a large number of urban ponds are species poor and of a low conservation value. Poor quality urban ponds are often not reported as they are considered uninteresting (Hassall, 2014). It has been identified that approximately 80% of ponds in England and Wales are of a poor or very poor quality (Williams et al., 2010). Lower quality, degraded urban ponds may act as ecological traps for macroinvertebrate taxa unable to detect anthropogenic disturbance prior to colonising. Participation of the general public in urban pond conservation / management, such as pond warden schemes, may help ensure ponds are preserved in the urban environment and could augment urban aquatic biodiversity (Boothby, 1995; DCPWA, 2014). Pond warden schemes, currently in operation in a small number of UK cities, allow a larger number of urban ponds to be monitored and managed in a more strategic manner and could greatly improve the ecological quality of degraded urban ponds at a national scale. It is also important to recognise that ponds located in urban spaces (e.g., school, or public park) provide an opportunity for the general public to interact with freshwater ecosystems and may help to engage the non-scientific community in biological conservation and raise awareness of the importance and management needs of small freshwater habitats (Hassall, 2014).

A number of observations and considerations regarding urban pond management can be made from this study. First, the creation of new urban ponds should be encouraged wherever it is practical and possible. The design of the pond should reflect its primary purpose; for example an ornamental plant pond may have vertical sides to ensure a constant water depth while a pond to support amphibians should include shallow shelves to allow individuals to leave the pond (Smith & Sutherland, 2014). In other instances, the requirements of the target floral and faunal group may need to be specifically considered (e.g., Odonata - Goertzen & Suhling, 2013) or for biodiversity more generally (Bardsley, 2012). Second, the results of this research demonstrated that the greatest macroinvertebrate diversity was recorded within the emergent and submerged macrophyte habitats. Management practices to support macroinvertebrate biodiversity should therefore aim to maintain or enhance submerged and emergent macrophyte habitats; although other habitats including areas of open water should also be maintained to support open water specialist taxa. Third, the area and immediate landuse surrounding a pond should be considered since the presence of a buffer-zone may enhance the aesthetic value, as well as mitigating the effects of anthropogenic disturbances and large urban structures. Physical barriers (e.g., high walls / fences and surrounding buildings) will influence colonisation and dispersal routes for organisms. Finally, urban ponds should be considered as part of the wider landscape / pondscape wherever possible. Pond warden schemes may help facilitate this and ensure that appropriate and sensitive management operations are undertaken but also that a wide range of pond types (including ephemeral ponds and those at both early and late successional stages) exist to ensure that both urban and wider landscape biodiversity is maximised wherever possible.

Conclusion

Ponds are common and abundant features in the urban landscape, many of which have anthropogenic origins and were built for a range of purposes including public amenity, flood reduction and water treatment. Urban ponds can support rich and diverse macroinvertebrate communities and in this study, park ponds supported the highest macroinvertebrate diversity whilst domestic garden ponds were the most taxa poor. Pond size was found to be strongly associated with macroinvertebrate diversity and the high beta diversity recorded demonstrates that individual ponds support different communities and that they potentially make an important contribution to regional biodiversity. A number of practical management recommendations have been suggested to increase aquatic macroinvertebrate biodiversity in anthropogenically dominated urban areas. Irrespective of their biodiversity and conservation value, it is important to recognise that urban ponds serve a number of societal functions and provide an opportunity for public engagement with freshwater habitats in addition to supporting biodiversity. Recognition of the significant contribution that ponds make to urban freshwater biodiversity is therefore important for the future conservation and management of urban ponds and other artificial waterbodies. This is vital for the ongoing protection of sites and biota from further habitat fragmentation in urban landscapes.

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Tables

Table 1 - Summary of Pearson's Correlation Coefficients between environmental parameters and biotic indices.

	Taxon Number	Shannon Wiener Diversity index	Berger Parker Dominance index	Log_{10} Abundance
Log_{10} Area	0.823**	0.704**	-0.600**	0.432**
Log_{10} Depth	0.601**	0.313*	-0.121	0.445**
Log_{10} Water Surface Shaded	0.000	0.106	-0.224	0.359*
Log_{10} Pond Margin Shaded	0.297	0.348*	-0.387*	0.399**
Log_{10} Emergent Macrophyte	0.178	0.284	-0.292	0.015
Log_{10} Submerged Macrophyte	0.304	0.198	-0.069	0.359*
Log_{10} Floating Macrophyte	-0.347*	-0.318	0.291	0.126
Log_{10} Conductivity	-0.006	0.156	-0.245	-0.095
Log_{10} Dissolved Oxygen	0.346*	0.267	-0.175	0.160
pH	0.006	0.015	0.072	-0.208
Pond Proximity	0.253	0.293	-0.314*	-0.022
Pond Connectivity	0.375*	0.414**	-0.418**	0.019

* P<0.05

** P<0.01

Figure Captions

Fig. 1 Scatter plots of the relationship between pond surface area (log_{10}) and macroinvertebrate community indices; (a) abundance (log_{10}), (b) taxon richness, (c) Shannon Wiener Diversity index and (d) Berger Parker Dominance index. Open symbols = park ponds, grey symbols = other urban ponds and solid black symbols = garden ponds.

Fig. 2 Macroinvertebrate diversity indices (mean +/- 1SE) recorded within different microhabitats (open water, emergent macrophytes, submerged macrophytes and floating macrophytes) for garden, other urban and park ponds: (a) abundance (log_{10}), (b) taxon richness, (c) Shannon Wiener Diversity index and (d) Berger Parker Dominance index.

Fig. 3 RDA site plots of macroinvertebrate communities recorded from the three urban pond types (garden, other urban and park ponds) studied around the town of Loughborough (Leicestershire, UK): (a) site plot with significant environmental vectors (SM= submerged macrophytes, EM= emergent macrophytes), note - only significant environmental parameters are presented; (b) taxa richness bubble plot; and (c) Shannon Wiener Diversity index bubble plot. Note: the size of each bubble is proportional to the absolute values represented.

Fig. 4 RDA taxon plot for garden, other urban and park pond macroinvertebrate assemblages with significant environmental vectors data (SM= submerged macrophytes, EM= emergent macrophytes). Note - only significant environmental parameters are presented.

Fig.1

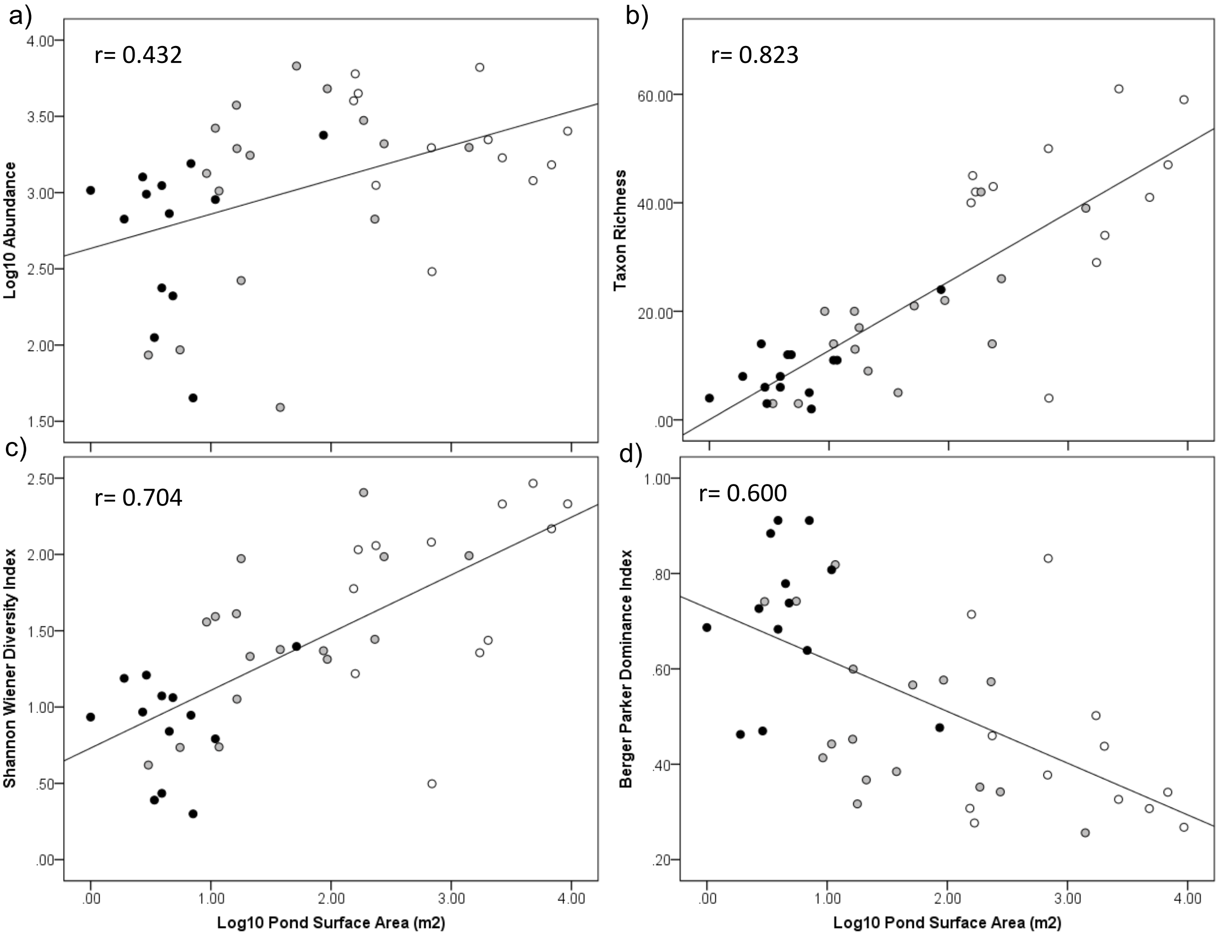

a)
r= 0.432
Log10 Abundance
b)
r= 0.823
Taxon Richness
c)
r= 0.704
Shannon Wiener Diversity Index
d)
r= 0.600
Berger Parker Dominance Index
Log10 Pond Surface Area (m2)
Log10 Pond Surface Area (m2)

Fig. 2

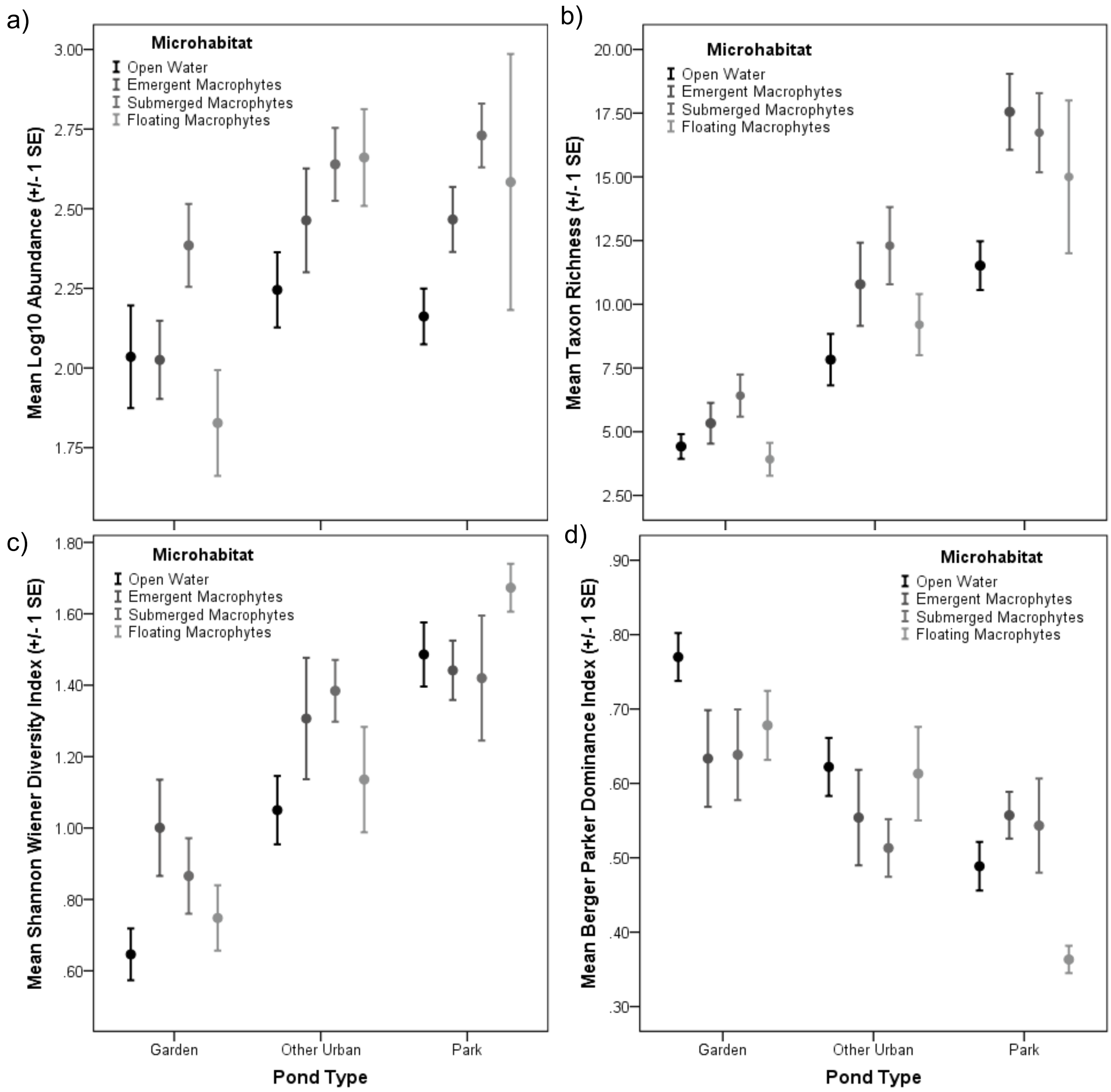
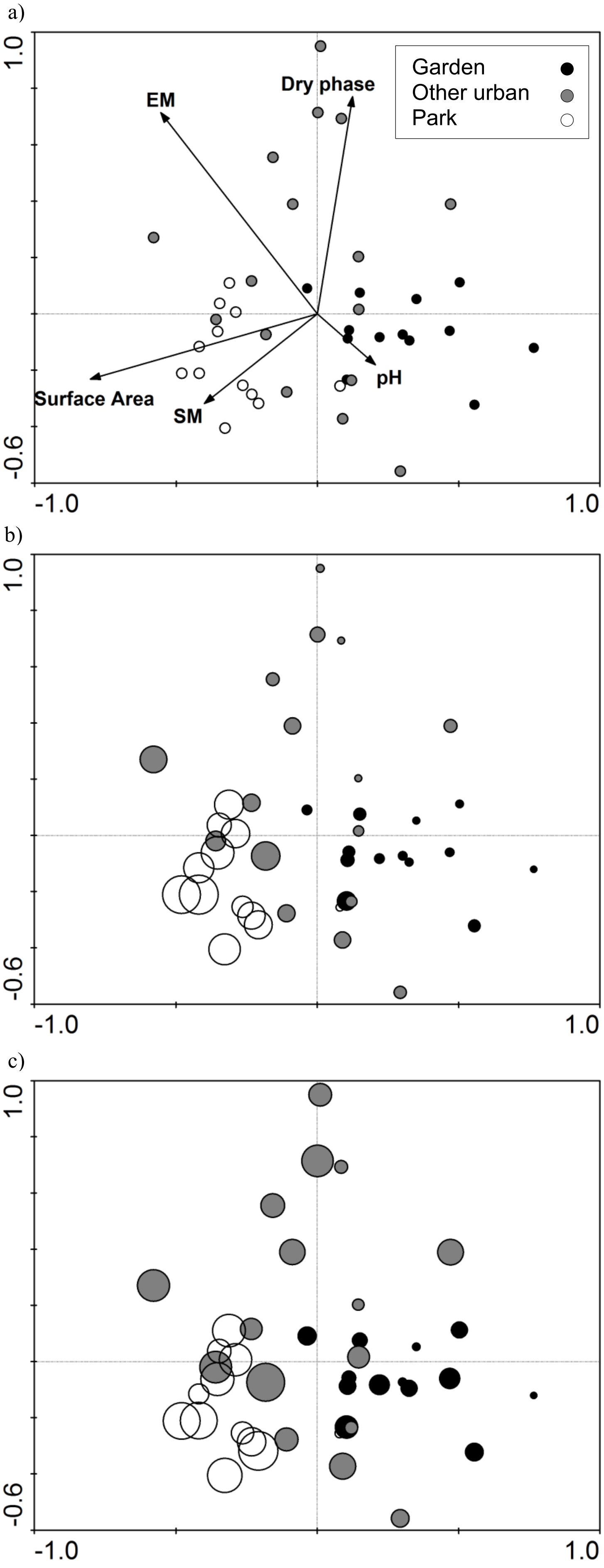
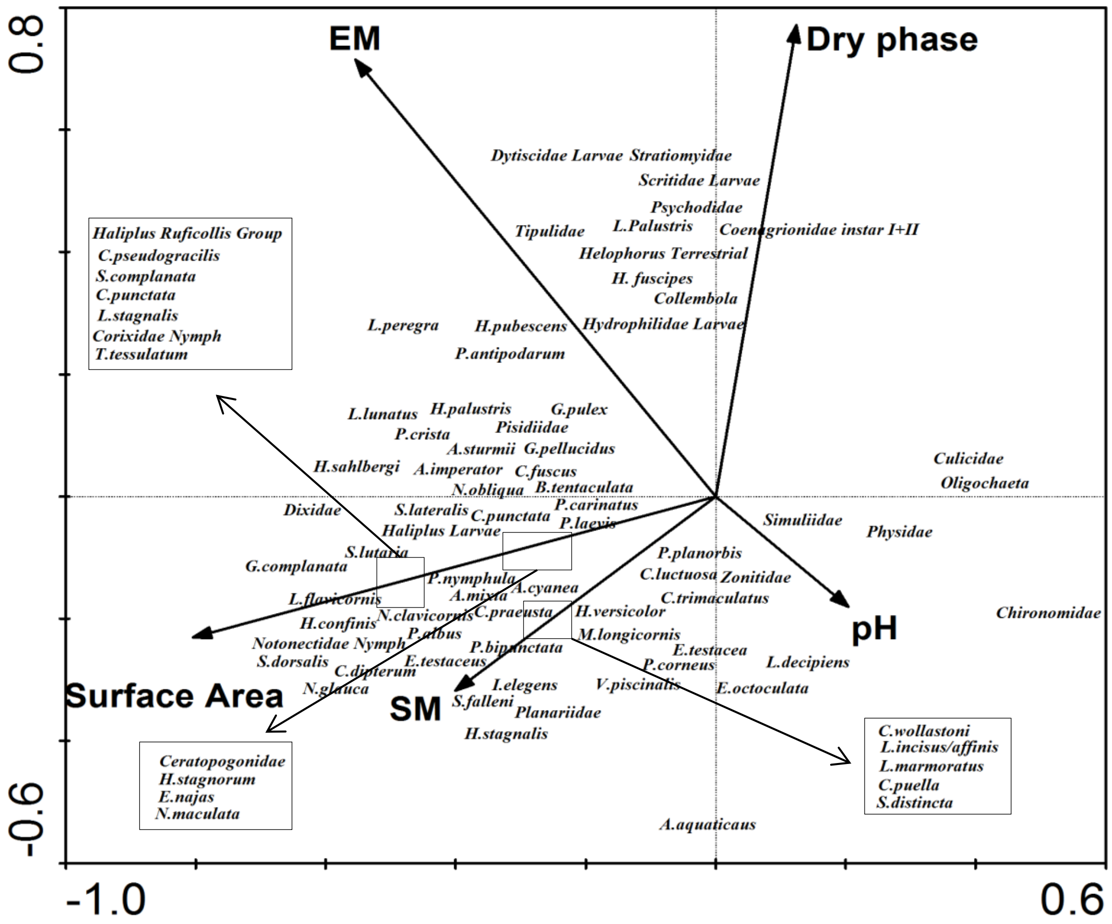

a)
b)
c)
d)
Microhabitat
Open Water
Emergent Macrophytes
Submerged Macrophytes
Floating Macrophytes
Mean Log10 Abundance (+/- 1 SE)
Mean Taxon Richness (+/- 1 SE)
Mean Shannon Wiener Diversity Index (+/- 1 SE)
Mean Berger Parker Dominance Index (+/- 1 SE)
Garden
Other Urban
Park
Pond Type

Fig. 3



a)
Garden
Other urban
Park
EM
Dry phase
Surface Area
SM
pH
1.0
-0.6
-1.0
1.0
b)
c)

Fig. 4


0.8
-0.6
-1.0
0.6
EM
Dry phase
pH
SM
Surface Area
Dytiscidae Larvae Stratiomyidae
Scritidae Larvae
Psychodidae
Tipulidae
L.Palustris
Coenagrionidae instar I+II
Helophorus Terrestrial
H. fuscipes
Collembola
L.peregra
H.pubescens
Hydrophilidae Larvae
P.antipodarum
Haliplus Ruficollis Group
C.pseudogracilis
S.complanata
C.punctata
L.stagnalis
Corixidae Nymph
T.tessulatum
L.lunatus
H.palustris
G.pulex
P.crista
Pisidiidae
A.sturmii
G.pellucidus
H.sahlbergi
A.imperator
C.fuscus
N.obliqua
B.tentaculata
Culicidae
Oligochaeta
Dixidae
S.lateralis
C.punctata
P.carinatus
P.laevis
Haliplus Larvae
Simuliidae
Physidae
S.lutaria
P.planorbis
G.complanata
C.luctuosa
Zonitidae
P.nymphula
A.cyanea
A.mixta
C.trimaculatus
L.flavicornis
N.clavicornis
C.praeusta
H.versicolor
Chironomidae
H.confinis
P.albus
M.longicornis
Notonectidae Nymph
P.bipunctata
E.testacea
S.dorsalis
E.testaceus
P.corneus
L.decipiens
C.dipterum
N.glauca
I.elegens
V.piscinalis
E.octoculata
S.falleni
Planariidae
H.stagnalis
C.wollastoni
L.incisus/affinis
L.marmoratus
C.puella
S.distincta
Ceratopogonidae
H.stagnorum
E.najas
N.maculata
A.aquaticaus