Garden pond diversity: opportunities for urban freshwater conservation

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Abstract

Urbanisation is increasing globally, degrading terrestrial and freshwater habitats and reducing faunal and floral richness. Whilst the potential for garden ponds to serve as important biodiversity resources in urban areas has been documented in a limited number of studies, quantifying the contribution of garden ponds to urban freshwater diversity has been largely neglected. This study aims to quantify the taxonomic richness, community composition and conservation value of aquatic macroinvertebrates in domestic garden and non-urban ponds. Taxonomic richness was significantly lower in garden ponds than non-urban ponds at an alpha and gamma scale. A greater richness of Odonata, Coleoptera, Gastropoda and Hemiptera were recorded in non-urban ponds. Garden ponds were found to support compositionally different macroinvertebrate communities compared to non-urban ponds, influenced by variation in water depth and conductivity. A total of 23 taxa were recorded from garden ponds only. Non-urban ponds had a significantly higher conservation value compared to garden ponds (87% of garden ponds were of low or moderate conservation value, while only 35% of non-urban ponds were in these categories). Although urban garden ponds currently support limited macroinvertebrate diversity and have lower conservation value, they contribute to the regional species pool and their potential to limit future urban biodiversity loss is significant. Given their high abundance and popularity within the urban landscape, clear guidance is required for pond-owners on how to best manage garden ponds to support and sustain biodiversity. For this to be achieved, research is required to increase fundamental understanding of urban pond ecology, and the development of evidence led garden pond management practices.

Keywords: anthropocene, biodiversity**,** conservation value, macroinvertebrate, taxonomic richness, urbanisation, urban ponds

Introduction

We live in a world dominated by human-modified ecosystems, and it is urban areas that have transformed the physical and biological environment extensively (Grimm et al., 2008). It is estimated that urban global landcover will increase by 1.2 million km^2 by 2030 (Seto et al., 2012), and 68% of the human population will reside in urban areas by 2050, with the most urbanised regions being North America (82%), Latin America (81%) and Europe (74% - United Nations, 2018). Increasing urbanisation and the conversion of 'natural land' to an increasingly artificial, homogeneous landscape is degrading and threatening ecosystems often far beyond the urban boundary (Mcdonald et al., 2009). Urbanisation has been demonstrated to simplify habitat diversity and complexity (McKinney 2006), reduce air, soil and water quality (Power et al., 2018), alter geomorphological and hydrological processes (McGrane, 2016), increase anthropogenic disturbance of residual habitats (McKinney, 2008), facilitate the colonisation of non-native fauna and flora (Gaertner et al., 2017) and increasingly fragment remaining natural habitat patches (Liu et al., 2016).

Urban habitat degradation has typically resulted in a reduction in taxonomic richness and abundance of riverine and terrestrial taxa, including mammals (Tait et al., 2005), fish (Morgan & Cushman, 2005; Weijters et al., 2009), birds (Sol et al., 2014), amphibians (Hamer & McDonnell, 2008; Parris, 2006) and macroinvertebrates (McKinney, 2008; Wang et al., 2012; Martinson & Raupp, 2013). Although, a greater richness of floral species has been recorded at intermediate levels of urbanisation (McKinney, 2008), and some species of terrestrial invertebrate have recorded a higher richness in urban areas (Magura, et al., 2010). Urbanisation has also been shown to be a primary driver of biotic homogenization (Knop, 2016), with the replacement of native species sensitive to change by stress-tolerant taxa that are able to exploit the urban landscape (McKinney, 2006). While the detrimental effects of urbanisation (lower taxonomic richness and biotic homogenisation) have been well documented among terrestrial and lotic systems, the opposite pattern has been recorded among ponds (defined here as lentic waterbodies between 1 m^2 and 2 ha: Hill et al., 2018a). Urban ponds have been found to have comparable diversity to non-urban ponds, provide significant ecological heterogeneity

(Hill et al., 2018; Hill et al., 2017), and may serve as refuges and stepping-stones for aquatic taxa in otherwise largely inhospitable landscape settings (Hassall 2014). Although, some studies have also reported lower taxonomic richness in ponds associated with increasing urban land cover (Noble & Hassall, 2014).

Despite the significant contribution of ponds to urban freshwater biodiversity, urbanisation (alongside other factors, including agricultural intensification) has caused a decline of pond numbers across the globe, particularly during the 20th century (Gledhill et al., 2008; Wood et al., 2003). However, in recent years, the UK has witnessed a growing number of ponds, increasing at a rate of 1.4% per annum (Williams et al., 2010). Almost all ponds occur in networks, and pond density is a key factor driving urban pond diversity (Gledhill et al., 2008). Therefore, pond loss poses a particular risk to the functioning of pond networks, given the substantial physical barriers (e.g., high rise commercial and domestic buildings) present in urban landscapes (Hassall, 2014). In urban environments where space is limited, garden ponds could therefore make a significant contribution to the ecological diversity of urban pond networks (filling the gap where ponds have been lost to development),

Garden ponds are a popular garden feature, with an estimated 2.5 to 3.5 million in the UK, covering 349 hectares (Davies et al., 2009). Garden ponds are built for a variety of reasons, including to support ornamental fish, for their amenity and aesthetic value, and to support biodiversity. In recent years, the potential of garden ponds has been recognised, with the media and environmental charities encouraging the construction of ponds in gardens to support wildlife (RHS, 2020; RSPB, 2020). Despite the high abundance and popularity of garden ponds, there has been little research examining the environmental and ecological conditions of ponds, with most research in gardens focussed on terrestrial flora and fauna (Paker et al., 2014; Matteson et al., 2008). The limited studies undertaken on garden ponds have largely focussed on amphibians (Cayuela et al., 2020; Hamer & Parris, 2011), with limited research examining macroinvertebrate communities (but see Hill & Wood (2014) and Gaston et al. (2005)). Existing evidence currently available regarding the contribution of garden ponds to macroinvertebrate biodiversity is either at a relatively coarse taxonomic resolution (Gaston et al. 2005), resulting in a significant proportion of the biodiversity being unaccounted for or derived from a comparatively small dataset in one urban area (13 garden and 13 non-urban ponds – Hill & Wood 2014) so that the wider generalisation of the results is unknown.

Given the high abundance and popularity of garden ponds, their potential to offset reductions in diversity due to wider pond habitat loss, and the ongoing lack of understanding of the environmental and ecological condition of garden ponds (Hill & Wood 2014), there is a pressing need to examine their aquatic biodiversity and conservation value. This study provides a comparative analysis of environmental conditions and macroinvertebrate diversity (taxonomic richness and community heterogeneity) among garden and non-urban ponds (located in an agricultural landscape), to facilitate the quantification of the wider conservation value of garden pond habitats and to assess the generalisability of the results from previous research (Hill & Wood 2014) using a more extensive dataset.

Material and methods

Site selection

Garden ponds. Sites were selected based on local personal contacts of the authors in the town of Abingdon, Oxfordshire, UK (Fig. 1). Although the selection was not random, the sites were chosen to be typical of garden ponds and included both those maintained by garden wildlife enthusiasts (managed for biodiversity) and more traditional ornamental garden fish ponds (typically managed for ornamental fish or amenity purposes). Sites were distributed across an area of approximately 8 km^2, all of which were within the suburban matrix of the town. All but one of the 30 garden ponds surveyed were in suburban areas. The remaining site was located in the grounds of a school, although in practice the surroundings of this site (lawns, scattered trees, nearby buildings) were similar to those of domestic gardens. The garden ponds were small, ranging in area from 0.6 to 20.0 m^2 in area (mean: 5.0 m^2).

Non-urban ponds. In total, 20 non-urban ponds were selected using a stratified random approach, surveyed as part of a wider study of the nature of freshwater habitats in lowland England, within predominantly farmed countryside. None of the sites were located in designated nature reserves or protected areas and, as has been confirmed by subsequent studies, were affected by the generally high levels of water pollution and other impacts seen in modern intensive European rural landscapes (Davies et al., 2007). The characteristics of the rural ponds are described in detail in Williams et al. (2004). None of the ponds in the rural area were in suburban areas. Non-urban ponds were typically ten times larger than those of the gardens, with a mean area of 550 m^2. Non-urban ponds were distributed across an area of c100 km^2 in Oxfordshire, UK (Fig. 1)

Macroinvertebrate sampling

Macroinvertebrates were sampled in both garden and rural locations following the procedures of the UK National Pond Survey (Biggs et al., 1998). To summarise, the garden and non-urban ponds were sampled for a total of three minutes using a standard 30 x 30 cm (1 mm mesh) pond hand net. Sampling time was allocated evenly between distinct mesohabitats (e.g., submerged macrophytes, woody debris, emergent macrophytes, open water, gravel substrates) present in each pond. Typically, a mesohabitat was sampled by sweeping with a net for a few seconds and the material retained in a bucket before moving on to the next mesohabitat, with the process repeated until 180s of sampling had been completed. In the non-urban pond sites, the sampling procedure was modified slightly to cover a standard area of 75 m^2 so that different waterbody types could be consistently compared. All samples were returned to the laboratory live for sorting within 2-3 days of collection (stored in a cold room whilst awaiting sorting). Specimens were preserved for identification in 70% industrial methylated spirits, except for those that could be confidently identified immediately to species level during sample sorting (e.g. distinctive molluscs, water bugs and water beetles) or those that must be identified live (e.g., flatworms Tricladida). Where necessary, uncommon taxa of conservation significance were verified by national recorders for the groups concerned. Groups identified to species level were flatworms (Tricladida), leeches (Hirudinea), gastropod molluscs (Gastropoda), large crustaceans (Malacostraca), mayflies (Ephemeroptera), stoneflies (Plecoptera), dragonflies and damselflies (Odonata), water bugs (Heteroptera), water beetles (Coleoptera), alder flies (Megaloptera) and caddisflies (Trichoptera). Note that species identification has been retained at the taxonomic levels which were in use at the time of the surveys. Non-urban pond surveys were undertaken in 2000 and garden pond surveys in 2010. Although there was a 10-year gap in sampling dates, when total richness was considered across both datasets, 70% of taxa recorded in the garden ponds were also recorded from non-urban ponds, indicating that communities form garden and non-urban ponds sites for part of the same regional species pool and are comparable. However, despite the high percentage of shared species between garden and non-urban ponds, we cannot entirely rule out that the gap in sampling time between the two pond groups may have influenced the differences in biodiversity recorded between garden and non-urban ponds (see Outhwaite et al. 2020).

Environmental data collection

Environmental data collection followed the procedures of the National Pond Survey (Biggs et al., 1998). Altitude, the percentage of water overhung by trees and vegetation, water depth, depth of silt, pond margin complexity (provides an estimate of the complexity of the shape of the pond – see Biggs et al. 1998 for more details), pH and conductivity were recorded for both garden and non-urban ponds. Using standard laboratory procedures, Nitrogen and phosphorus, measured as NO_3-N and Total Phosphorus and Biological Oxygen Demand (BOD) were recorded for both garden and non-urban ponds. Conductivity and pH were measured in the field with portable meters.

Data analysis

All statistical analyses were undertaken in the R environment (R Development Core team, 2019). Gamma diversity was calculated as the total number of aquatic macroinvertebrate taxa recorded among all pond study sites. Estimated gamma diversity was quantified using the Chao2 estimator in the *vegan* package, which uses the number of uncommonly occurring taxa in a sample to estimate the number of undiscovered species (see Oksanen et al., 2019). Differences in estimated gamma diversity between garden and non-urban ponds were considered significant if there was no overlap in the 95% confidence intervals. Alpha diversity was defined as the taxonomic richness within individual pond sites. Differences in faunal alpha diversity (taxonomic richness) between garden and non-urban ponds was examined using Mann–Whitney U tests. Similarly, differences in environmental conditions between garden and non-urban ponds were statistically examined using Mann-Whitney U tests.

Differences in aquatic macroinvertebrate compositions and environmental conditions between garden and non-urban ponds were examined using a ‘permutational analysis of variance’ (PERMANOVA) with the function *adonis* in the vegan package and visualised using NMDS (based on the Sorenson dissimilarity for invertebrate data and Euclidean distances for environmental data) using the *metaMDS* function in the vegan package. To examine compositional variation within each pond type the homogeneity of multivariate dispersions was calculated for the environmental and macroinvertebrate data using the *betadisper* function in vegan and compared using an ‘Analysis of Variance’ (ANOVA). Indicator Value Analysis was undertaken using the function *multipatt* in the labdsv package, to identify those taxa associated (indicator taxa) with garden or non-urban ponds (Dufrene & Legendre, 1997). The contribution of species turnover and nestedness to total beta-diversity (defined as macroinvertebrate community heterogeneity among ponds) within garden and non-urban ponds, and for all ponds across the study area was examined. Beta-diversity (using the Sørensen dissimilarity metric) was partitioned into species turnover and nestedness-resultant components using the *beta.multi* function from the betapart package (Baselga et al., 2018).

The associations between the nine environmental variables (pH, conductivity, water depth, silt depth, nitrate, pond margin complexity, total phosphorus, BOD, percentage water overhung) and macroinvertebrate communities was assessed using redundancy analysis (RDA), using the function 'ordiR2step' in vegan to identify the significant ($p<0.05$) environmental variables influencing macroinvertebrate assemblages. OrdiR2step is a forward selection model that has three stopping rules: 1) once the adjusted R^2 starts to decrease; 2) when the full model adjusted R^2 is exceeded, and 3) if the preselected permutational significance level ($p<0.05$) is exceeded (Oksanen et al., 2019).

The conservation value of garden and non-urban ponds was calculated using the Species Rarity Index (SRI). The rarity value assigned to each macroinvertebrate is based on the UK Joint Nature Conservation Committee (JNCC) designations (see Williams et al., 2004). The SRI was calculated by summing the rarity/threat value assigned to each macroinvertebrate taxon in a pond and then dividing by the number of species recorded in that pond sample (see Biggs, 2005 for a detailed methodology to calculate SRI).

Results

Environmental characteristics

Altitude ($W=0$, $p<0.01$), the percentage of water overhung by trees and vegetation ($W=201$, $p<0.05$), depth ($W=89$, $p<0.01$), silt depth ($W=95.5$, $p<0.01$) and conductivity ($W=51$, $p<0.01$) were significantly higher for non-urban ponds than garden ponds (Table 1). Pond margin complexity ($W=405$, $p<0.05$) and total phosphorus ($W=437$, $p<0.01$) were significantly higher for garden ponds than non-urban ponds. There was no significant difference in pH, Nitrate and BOD among the two pond types.

Environmental characteristics among garden and non-urban ponds were relatively distinct in the NMDS biplot, and PERMANOVA identified a significant statistical difference between environmental characteristics (R^2=0.36 p<0.001; Fig. 2A). Multivariate dispersion of environmental characteristics was not significantly different (F=2.77 p=0.10) between garden (distance to centroid: 129.2) and non-urban ponds (distance to centroid: 176.7, Fig. 2B).

Local and regional diversity

A total of 202 macroinvertebrate taxa were recorded across the garden and non-urban pond sites. A greater number of taxa were recorded among non-urban ponds (observed: 172 taxa) compared to garden ponds (observed: 99 taxa; Fig. 3A). Estimated gamma diversity (based on the Chao 2 estimator) was significantly (p<0.05) higher for the non-urban ponds (estimated gamma: 219.65, 95% CI: 185.7-253.6) than the garden ponds (estimated gamma: 121.35, 95% CI: 101.65-141.05; Fig. 2A). The most widespread species recorded in this study included *Asellus aquaticus* (present in 77% of garden ponds and 45% of non-urban ponds), *Lymnaea stagnalis* (garden: 40% non-urban: 15%), *Pyrrhosoma nymphula* (garden: 46% non-urban: 25%), Oligochaeta (garden: 56% non-urban: 90%), *Crangonyx pseudogracilis* (garden: 46% non-urban: 60%) and *Cloeon dipterum* (garden: 53% non-urban: 65%; Fig. 3B). See Appendix A: Table S1 for the prevalence of all taxa recorded in this study.

At the alpha scale, taxonomic richness was significantly higher (W=74, p<0.01) in non-urban ponds (mean: 35.5, median: 32.5) compared to garden ponds (mean: 11.93, median: 7.5; Table 1, Fig. 4A). Higher taxonomic richness in non-urban ponds was driven by a greater richness of Coleoptera, Odonata, Hemiptera, Trichoptera and Gastropoda when compared to garden ponds (Fig. 4B). The greatest number of invertebrate taxa recorded was from a non-urban pond (71 taxa), with 9 of the 10 most taxa rich being non-urban.

Community composition

PERMANOVA demonstrated that there was a significant difference in macroinvertebrate communities among garden and non-urban ponds (R^2=0.14 p<0.001), which is shown by the separation between garden and non-urban ponds in the NMDS biplot (Fig. 5A). Multivariate dispersion was found to be similar for macroinvertebrate composition among non-urban (distance to centroid: 0.49) and garden ponds (distance to centroid: 0.53, $F=2.51$ $p=0.12$, Fig. 5B). Both garden and non-urban ponds demonstrated high beta-diversity, based on the Sorenson dissimilarity metric (garden ponds: 0.94, non-urban ponds: 0.9), and when all ponds were considered (0.96). Almost all variation in community assembly could be explained by species turnover rather than nestedness when all sites were considered together (turnover: 94%, nestedness: 6%); and when garden (turnover: 93%, nestedness: 7%) and non-urban ponds (turnover: 91%, nestedness: 9%) were assessed separately. When macroinvertebrate communities among each pond group were compared, garden ponds on average supported less taxa than non-urban ponds, and 70% of taxa recorded in the garden ponds were also recorded from non-urban ponds, indicating that garden pond sites formed nested subsets of non-urban sites.

A total of 23 taxa were only recorded from garden ponds, while 102 taxa were recorded only from the non-urban ponds (see Appendix A: Table S2 for full list of unique taxa). Only three taxa, Culicidae, *Planorbis carinatus* and *A. aquaticus*, were identified as significant indicators of garden ponds. A total of 48 taxa were identified as indicator taxa for non-urban ponds including *Corixa punctata*, *Hydraena riparia*, Ceratopogonidae, *Glossiphonia complanata and Lymnaea truncatula* (Table 2). See Appendix A: Table S3 for the full list of statistically significant indicator taxa for garden and non-urban ponds.

Redundancy analysis demonstrated that garden and non-urban pond macroinvertebrate communities were separated on the first and second axes along gradients associated with conductivity and water depth (both $p<0.05$, Fig. 6). The RDA model was significant ($F=1.50$ $p<0.011$), explaining 10% of the variation in macroinvertebrate community composition on all constrained axes, based on the adjusted R^2 value (adjusted $R^2=0.10$). Garden ponds were associated with lower electrical conductivity and water depth, while non-urban ponds were characterised by greater electrical conductivity and water depth (Fig. 6).

Conservation value

Significantly greater Species Rarity Index (SRI) scores were recorded within non-urban ponds (mean SRI: 1.2, median:1.17, SE: 0.039) compared to garden ponds (mean SRI: 1.03, median: 1, SE: 0.009) ($W=85$, $p<0.001$, Table 3). Based on the taxonomic richness and the SRI, macroinvertebrate communities within seven non-urban ponds were of very high conservation value and eight ponds were of high conservation value (See Biggs (2005) Table 8 for the classification of conservation value). Only one garden pond supported communities with a high conservation value; although no garden ponds were found to be of very high conservation value, based on the taxonomic richness and SRI scores (Table 3). A total of 96% of garden ponds were found to be of a low or moderate conservation value, while 25% of non-urban ponds supported communities of low or moderate conservation value. A total of 6 Coleoptera species with a conservation designation were recorded within the ponds examined: *Agabus conspersus*, *Hygrotus nigrolineatus*, *Hygrotus decoratus*, *Scarodytes halensis*, *Nebrioporus depressus* and *Oulimnius major*. Only one of these species, *S. halensis*, was recorded from a single garden pond, while all other macroinvertebrate species with a conservation designation were recorded only from non-urban ponds.

Discussion

Macroinvertebrate richness

While there has been increasing research interest in urban pond ecology (Hassall, 2014), there have been few studies that have considered garden pond diversity (but see Hill & Wood 2014; Gaston et al., 2005), an important omission given the very large number of ponds in gardens (Davies et al. 2009). This study demonstrated that macroinvertebrate richness in garden ponds was significantly lower than non-urban ponds at a regional (gamma) and local (alpha) scale. Similar findings were recorded by Hill and Wood (2014), reporting a lower macroinvertebrate diversity in garden ponds than field ponds but from a much smaller subset of ponds. Similarly, Gaston et al. (2005) reported that garden ponds supported limited macroinvertebrate richness (at a coarse taxonomic resolution), dominated by Diptera. However, these findings contrast with other research focussed on urban ponds more widely, which indicated that they could support comparable diversity to non-urban ponds when larger geographical areas and multiple urban centres were considered (Hill et al., 2017; Hassall & Anderson, 2015).

The lower taxonomic richness recorded among garden ponds in this study may reflect their reduced environmental variability and their small size compared to non-urban ponds, limiting the range of conditions which macroinvertebrate taxa were able to exploit. The importance of local environmental variability for regional pond richness among non-urban ponds has been well documented (Biggs et al., 2005; Williams et al., 2004), allowing species to track wide environmental gradients across the pond network (Hill et al., 2017a; Cottenie et al., 2005). Despite having a higher morphological (shape) complexity than non-urban ponds, garden ponds also commonly lack habitats important for these groups: for example, most garden ponds have poor shallow marginal edge structure, important to water beetles, and often lack good stands of aquatic plants needed by Odonata as egg-laying sites and larval habitat (Williams et al 2020a). Garden pond floral diversity can be high, but some of this diversity is frequently comprised of non-native taxa, that may be unsuitable for aquatic macroinvertebrates (Thompson et al., 2003).

Connectivity has also been demonstrated to be an important factor influencing the richness and composition of pond communities. Several studies have found urban density (e.g., the number of buildings around a pond or the percentage of land covered by built up areas around ponds) had a negative effect on urban pond macroinvertebrate richness (Blicharska et al., 2017; Heino et al., 2017). Garden ponds frequently have a limited connectivity compared to non-urban ponds or other urban waterbodies, partially driven by urban pond loss and the structural complexity of the urban landscape

(Vehkaoja et al., 2020; Biggs et al., 2005). Garden ponds may be surrounded by numerous physical barriers such as roads, domestic and commercial buildings and being entirely enclosed by fences or walls may limit the ability of taxa to disperse to garden ponds, especially among garden ponds that may be entirely enclosed by fences or walls (Thornhill et al., 2018). However, the non-urban ponds examined in this study were well connected as the agricultural landscape in the study area was largely open with few barriers, which may have contributed to the enhanced diversity recorded in this study. More widely, other urban ponds may be located in school grounds or urban parks, surrounded by a greater amount of green space and fewer urban barriers which may in part reflect why they have recorded comparable macroinvertebrate diversity to non-urban ponds across the UK (Hill et al. 2017). Ornamental fish are frequently heavily stocked in garden ponds (Patoka et al., 2017) and are often the primary motivational factor to build a garden pond. Although not measured in this study, high densities of fish have been recorded to influence macroinvertebrate composition and richness in ponds through predation (Nurminen et al., 2017; Foltz & Dodson, 2009; Fairchild et al., 2000; Diehl, 1992). Future research is clearly required to quantify the impact of non-native floral species, the effect of fish density and the interactive effect of habitat complexity and fish density on garden pond communities.

Macroinvertebrate community composition

Significant differences in macroinvertebrate community composition were recorded among garden and non-urban ponds in this study, related to greater water depths and conductivities in non-urban ponds. These findings support those reported by Hill and Wood (2014) on a smaller dataset, where gradients in water depth, conductivity, overhanging vegetation and floating vegetation were found to be driving compositional differences between urban and field ponds and suggests that these patterns are more widely applicable to garden ponds. Deeper ponds are likely to support more heterogeneous environmental conditions and may be less likely to dry during summer months (Heino et al., 2017). However, the RDA model recorded a relatively low explanatory power (~10%), indicating that the most important drivers of compositional variation were not included and/or that compositional patterns are driven by stochastic factors (Hassall & Anderson, 2015).

Across the study region and among the individual pond types, community heterogeneity (beta-diversity) could almost entirely be explained by species turnover rather than nestedness. This reflects the pattern reported in previous research that has examined patterns in beta-diversity in ponds (Hill et al., 2017a; Vad et al., 2017). The high contribution of species turnover to beta-diversity among garden ponds may reflect (1) macroinvertebrate taxa tracking the limited environmental gradients recorded among garden ponds; (2) the wide range, and more frequently employed management practices that garden ponds are subject to (e.g., removal of fine sediment and adjacent vegetation). In this study, garden ponds were subject to dredging, removal of emergent, submerged and/or submerged vegetation and the cutting back or mowing of surrounding vegetation, all of which are likely cause the short-term disturbance to macroinvertebrate communities and; (3) garden pond isolation (as a result of urban structural complexity), which may limit opportunities for dispersal and increase the level of ecological uniqueness of garden ponds (Thornhill et al., 2018). In less intensively managed, and more connected non-urban pond networks in agricultural landscapes, there may be other mechanisms influencing the high turnover recorded, such as such as variability in shading (Sayer et al. 2012), pollution (Biggs et al. 2007), pond margin poaching / trampling by livestock (van den Broeck et al 2019), and other local environmental variables (Gioria et al. 2010). However, when comparing the total richness recorded in garden ponds with non-urban ponds, 70% of the macroinvertebrates recorded in the garden ponds were also recorded among the non-urban ponds, demonstrating that garden ponds were in part nested subsets of non-urban ponds, driven by the loss, rather than gain of macroinvertebrate species within garden ponds (Legendre, 2014).

Implications for urban conservation planning

The suggestion that garden ponds have the potential to provide an important biodiversity resource in urban areas is not new (Thornhill et al., 2018; Hassall, 2014), although the present study suggests that currently their conservation value and contribution to macroinvertebrate diversity is modest, supporting the findings of Hill and Wood (2014). However, garden ponds supported 23 taxa (11% of total taxonomic richness) that were not recorded from non-urban ponds, highlighting that garden ponds make an important contribution to regional (gamma) diversity. Given that garden ponds support a total of 99 taxa, they may provide a refuge site or act as stepping-stones through the urban landscape for a wide range of macroinvertebrate taxa, although physical urban barriers (e.g., solid vertical walls and fences) may reduce their efficacy to act as stepping stones. Garden ponds also have the potential to provide a relatively unpolluted aquatic habitat, a resource which is increasingly scarce in the urban environment (McGoff et al., 2017). More widely, garden ponds have been documented to provide an important refuge site for amphibians (Beebee, 1979). Future research is needed to increase our fundamental understanding of the ecology of these poorly studied systems, to further refine the role they can play in conservation / management strategies for freshwater biodiversity. For example, when supplied with unpolluted water (e.g. rainwater) ponds may be able to simulate the small seasonal pools that historically would have been abundant in pre-drainage landscapes. There seems little doubt they can provide this habitat for amphibians, and it is likely they could also do this usefully for a range of macroinvertebrates including water beetles (Coleoptera), aquatic snails (Gastropoda / Mollusca), dragonfly and damselfly (Odonata) and caddis flies (Trichoptera), all of which can be abundant in small shallow high-quality ponds (Williams et al. 2020).

With an estimated 2.5-3.5 million garden ponds in the UK (Davies et al., 2009), their potential to support a rich urban freshwater biodiversity resource is significant. The greatest taxonomic richness recorded from a garden pond in this study was comparable to most non-urban ponds, suggesting that garden ponds can reach biodiversity levels recorded among ponds in more rural / less urbanised landscapes. In the UK, there are approximately 22.3 million dwellings (Ministry of Housing, Communities and Local Government, 2019), of which 87% have a garden (Buck, 2016). Utilising some of this abundant space for garden pond development provides an opportunity to increase the density and connectivity of urban freshwater networks and may be the only option offsetting wider urban pond loss (Gledhill et al., 2008). However, to ensure that garden pond creation is successful, and the maximum biodiversity potential of existing and new garden ponds is reached, clearer management guidance and advice for pond owners is required, and the goal of garden pond creation should include biodiversity alongside the wider amenity value. In addition, current management guidance is largely focussed at the individual pond scale and maintaining relatively homogeneous early successional stages in garden ponds that is unlikely to reflect the heterogeneity of environmental conditions observed within non-urban ponds (Gaston et al., 2005). Clearly, there is therefore a need to consider garden pond management at both the individual pond and landscape scale to re-create the natural conditions of ponds (Goertzen et al. 2013) and sufficient environmental heterogeneity. For example, Sinclair et al. (2020) highlighted that variability in vegetation management (and associated changes in environmental conditions) across an urban pond network maximised compositional variation in aquatic macroinvertebrates. Garden ponds need to be considered in the wider management framework of urban freshwater networks, to ensure that there is a holistic and integrated approach to urban freshwater management and conservation.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at XXXXX."

References

Baselga, A., Orme, D., Villeger, S., De Bortoli, J., Leprieur, F., Logez, M., & Henriques-Silva, R. (2018). Partitioning beta diversity into turnover and nestedness components. R package version 1.5.1.

Beebee T.J.C. (1979). Habitats of the British amphibians (2): suburban parks and gardens. *Biological Conservation*, 15, 241-257.

Biggs, J., Fox, G., Whitfield, M., & Williams. P. 1998. A guide to the methods of the National Pond Survey. Pond Action, Oxford.

Biggs, J. (2005). Ponds, pools and lochans: guidance on good practice in the management and creation of small waterbodies in Scotland.

Biggs, J., Williams, P., Whitfield, M., Nicolet, P., & Weatherby, A., (2005). 15 years of pond assessment in Britain: results and lessons learned from the work of Pond Conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 15, 693-714.

Biggs, J., Williams, P., Whitfield, M., Nicolet, P., Brown, C., Hollis, J., Arnold, D. & Pepper, T. (2007). The freshwater biota of British agricultural landscapes and their sensitivity to pesticides. *Agriculture, Ecosystems & Environment*, 122, 137-148.

Blicharska, M., Andersson, J., Bergsten, J., Bjelke, U., Hilding-Rydevik, T., Thomsson, M., Östh, J., & Johansson, F. (2017). Is there a relationship between socio-economic factors and biodiversity in urban ponds? A study in the city of Stockholm. *Urban Ecosystems*, 20, 1209-1220.

Buck, D. (2016). Gardens and health: Implications for policy and practice. King's Fund, London.

Cayuela, H., Valenzuela-Sánchez, A., Teulier, L., Martínez-Solano, Í., Léna, J.P., Merilä, J., Muths, E., Shine, R., Quay, L., Denoël, M., & Clobert, J. (2020). Determinants and consequences of dispersal in vertebrates with complex life cycles: a review of pond-breeding amphibians. *The Quarterly Review of Biology*, 95, 1-36.

Cottenie, K. (2005). Integrating environmental and spatial processes in ecological community dynamics. *Ecology Letters*, 8, 1175-1182.

Davies, B.R., Biggs, J., Williams, P.J., Lee, J.T. & Thompson, S. (2007). A comparison of the catchment sizes of rivers, streams, ponds, ditches and lakes: implications for protecting aquatic biodiversity in an agricultural landscape. In Pond Conservation in Europe (pp. 7-17). Springer, Dordrecht.

Davies, Z.G., Fuller, R.A., Loram, A., Irvine, K.N., Sims, V., & Gaston, K.J. (2009). A national scale inventory of resource provision for biodiversity within domestic gardens. *Biological Conservation*, 142, 761-771.

Diehl, S. (1992). Fish predation and benthic community structure: the role of omnivory and habitat complexity. *Ecology*, 73, 1646-1661.

Dufrene, M., & Legendre, P. (1997). Species assemblages and indicator species: the need for a flexible asymmetrical approach. *Ecological Monographs*, 67, 345–366.

Foltz, S.J. & Dodson, S.I. (2009). Aquatic Hemiptera community structure in stormwater retention ponds: a watershed land cover approach. *Hydrobiologia*, 621, 49-62.

Gaertner, M., Wilson, J.R., Cadotte, M.W., MacIvor, J.S., Zenni, R.D. & Richardson, D.M., (2017). Non-native species in urban environments: patterns, processes, impacts and challenges. *Biological Invasions*, 19, 3461–3469.

Gaston, K, Smith, R, Thompson, K., & Warren, P. (2005). Urban domestic gardens (II): experimental tests of methods for increasing biodiversity. *Biodiversity and Conservation*, 14, 395-413.

Gioria, M., Bacaro, G. & Feehan, J., (2010). Identifying the drivers of pond biodiversity: the agony of model selection. Community Ecology, 11, 179-186.

Gledhill, D.G., James, P. & Davies, D.H. (2008). Pond density as a determinant of aquatic species richness in an urban landscape. *Landscape Ecology*, 23, 1219-1230.

Grimm, N.B., Faeth, S.H., Golubiewski, N.E., Redman, C.L., Wu, J., Bai, X. & Briggs, J.M. (2008). Global change and the ecology of cities. *Science*, 319, 756-760.

Hassall, C. (2014). The ecology and biodiversity of urban ponds. *Wiley Interdisciplinary Reviews: Water*, 1, 187-206.

Hassall, C. & Anderson, S. (2015). Stormwater ponds can contain comparable biodiversity to unmanaged wetlands in urban areas. *Hydrobiologia*, 745, 137-149.

Hamer, A.J. & McDonnell, M.J. (2008). Amphibian ecology and conservation in the urbanising world: a review. *Biological conservation*, 141, 2432-2449.

Hamer, A.J., & Parris, K.M. (2011). Local and landscape determinants of amphibian communities in urban ponds. *Ecological Applications*, 21, 378–390.

Heino, J., Bini, L.M., Andersson, J., Bergsten, J., Bjelke, U. & Johansson, F. (2017). Unravelling the correlates of species richness and ecological uniqueness in a metacommunity of urban pond insects. *Ecological Indicators*, 73, 422-431.

Hill, M.J., Biggs, J., Thornhill, I., Briers, R.A., Gledhill, D.G., White, J.C., Wood, P.J. & Hassall, C. (2017). Urban ponds as an aquatic biodiversity resource in modified landscapes. Global Change Biology, 23, 986-999.

Hill, M.J., Heino, J., Thornhill, I., Ryves, D.B. & Wood, P.J. (2017a). Effects of dispersal mode on the environmental and spatial correlates of nestedness and species turnover in pond communities. *Oikos*, 126, 1575-1585.

Hill, M.J., Biggs, J., Thornhill, I., Briers, R.A., Ledger, M., Gledhill, D.G., Wood, P.J. & Hassall, C. (2018). Community heterogeneity of aquatic macroinvertebrates in urban ponds at a multi-city scale. *Landscape Ecology*, 33, 389-405.

Hill, M.J., Hassall, C., Oertli, B., Fahrig, L., Robson, B.J., Biggs, J., Samways, M.J., Usio, N., Takamura, N., Krishnaswamy, J. & Wood, P.J. (2018a). New policy directions for global pond conservation. *Conservation Letters*, 11, p.e12447.

Hill, M.J. & Wood, P.J. (2014). The macroinvertebrate biodiversity and conservation value of garden and field ponds along a rural-urban gradient. *Fundamental and Applied Limnology*, 185, 107-119.

Knop, E. (2016). Biotic homogenization of three insect groups due to urbanization. *Global Change Biology*, 22, 228–236.

Legendre, P. (2014). Interpreting the replacement and richness difference components of beta diversity. *Global Ecology and Biogeography*, 23, 1324-1334.

Liu, Z., He, C. & Wu, J. (2016). The relationship between habitat loss and fragmentation during urbanization: an empirical evaluation from 16 world cities. *PLoS One*, 11.

Lundkvist, E., Landin, J. & Karlsson, F. (2002). Dispersing diving beetles (Dytiscidae) in agricultural and urban landscapes in south-eastern Sweden. *Annales Zoologici Fennicii*, 39, 109-123.

Magura, T., Horváth, R. & Tóthmérész, B. (2010). Effects of urbanization on ground-dwelling spiders in forest patches, in Hungary. *Landscape Ecology*, 25, 621-629.

Martinson, H.M. & Raupp, M.J. (2013). A meta-analysis of the effects of urbanization on ground beetle communities. *Ecosphere*, 4, 1-24.

Matteson, K.C., Ascher, J.S. & Langellotto, G.A. (2008). Bee richness and abundance in New York City urban gardens. *Annals of the Entomological Society of America*, 101, 140-150.

Mcdonald, R.I., Forman, R.T., Kareiva, P., Neugarten, R., Salzer, D. & Fisher, J. (2009). Urban effects, distance, and protected areas in an urbanizing world. *Landscape and Urban Planning*, 93, 63-75.

McGrane, S.J. (2016). Impacts of urbanisation on hydrological and water quality dynamics, and urban water management: a review. *Hydrological Sciences Journal*, 61, 2295-2311.

McKinney, M.L. (2006). Urbanization as a major cause of biotic homogenization. *Biological Conservation*, 127, 247-260.

McKinney, M.L. (2008). Effects of urbanization of species richness: a review of plants and animals. *Urban Ecosystems*, 11, 161–176.

Ministry of Housing Communities and Local Government. (2019). English Housing Survey Headline Report, 2017-18. Ministry of Housing Communities and Local Government. London, UK.

Morgan, R.P. & Cushman, S.F. (2005). Urbanization effects on stream fish assemblages in Maryland, USA. *Journal of the North American Benthological Society*, 24, 643-655.

Noble, A., & Hassall, C. (2014). Poor ecological quality of urban ponds in northern England: causes and consequences. *Urban Ecosystems*, 18, 1–14.

Nurminen, L., Hellén, N., Olin, M., Tiainen, J., Vinni, M., Grönroos, M., Estlander, S., Horppila, J., Rask, M. & Lehtonen, H. (2018). Fishing-induced changes in predation pressure by perch (*Perca fluviatilis*) regulate littoral benthic macroinvertebrate biomass, density, and community structure. *Aquatic Ecology*, 52, 1-16.

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., Henry, M., Stevens, H., Szoecs, E., & Wagner, H. (2019). Vegan: Community ecology package. R package version 2.5-6.

Outhwaite, C.L., Gregory, R.D., Chandler, R.E., Collen, B. & Isaac, N.J. (2020). Complex long-term biodiversity change among invertebrates, bryophytes and lichens. *Nature Ecology & Evolution*, 4, 384-392.

Paker, Y., Yom-Tov, Y., Alon-Mozes, T. & Barnea, A. (2014). The effect of plant richness and urban garden structure on bird species richness, diversity and community structure. *Landscape and Urban Planning*, 122, 186-195.

Parris, K.M. (2006). Urban amphibian assemblages as metacommunities. *Journal of Animal Ecology*, 75, 757-764.

Patoka, J., Bláha, M., Kalous, L. & Kouba, A. (2017). Irresponsible vendors: Non-native, invasive and threatened animals offered for garden pond stocking. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 27, 692-697.

Power, A.L., Tennant, R.K., Jones, R.T., Tang, Y., Du, J., Worsley, A.T. & Love, J. (2018). Monitoring impacts of urbanisation and industrialisation on air quality in the Anthropocene using urban pond sediments. *Frontiers in Earth Science*, 6, 131.

R Development Core Team. (2019). R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna.

RHS. (2020). Wildlife ponds. Available at https://www.rhs.org.uk/advice/profile?PID=622 [last accessed: 01/04/2020].

RSPB. (2020). Planning a pond. Available at: https://www.rspb.org.uk/birds-and-wildlife/advice/gardening-for-wildlife/water-for-wildlife/planning-a-pond/ [last accessed: 01/04/2020].

Seto, K.C., Guneralp, B., & Hutyra, L.R. (2012). Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *PNAS*, 109, 16083–16088.

Sol, D., González-Lagos, C., Moreira, D., Maspons, J. & Lapiedra, O. (2014). Urbanisation tolerance and the loss of avian diversity. *Ecology letters*, 17, 942-950.

Tait, C.J., Daniels, C.B. & Hill, R.S. (2005). Changes in species assemblages within the Adelaide metropolitan area, Australia, 1836–2002. *Ecological Applications*, 15, 346-359.

Thompson, K., Austin, K.C., Smith, R.M., Warren, P.H., Angold, P.G., & Gaston, K.J. (2003). Urban domestic gardens (I): Putting small-scale plant diversity in context. *Journal of Vegetation Science*, 14, 71-78.

Thornhill, I., Batty, L., Hewitt, M., Friberg, N.R. & Ledger, M.E. (2018). The application of graph theory and percolation analysis for assessing change in the spatial configuration of pond networks. *Urban Ecosystems*, 21, 213-225.

United Nations. (2018). World Urbanization Prospects: The 2018 Revision. United Nations, New York.

Vad, C.F., Péntek, A.L., Cozma, N.J., Földi, A., Tóth, A., Tóth, B., Böde, N.A., Móra, A., Ptacnik, R., Ács, É. & Zsuga, K. (2017). Wartime scars or reservoirs of biodiversity? The value of bomb crater ponds in aquatic conservation. *Biological conservation*, 209, 253-262.

Van den Broeck, M., Rhazi, L., Waterkeyn, A., El Madihi, M., Grillas, P., Kneitel, J.M. & Brendonck, L., (2019). Livestock disturbances in Mediterranean temporary ponds: A mesocosm experiment with sheep manure and simulated trampling. *Freshwater Biology*, 64, 856-869.

Vehkaoja, M., Niemi, M. and Väänänen, V.M. (2020). Effects of urban infrastructure on aquatic invertebrate diversity. *Urban Ecosystems*, 1-10.

Wang, B., Liu, D., Liu, S., Zhang, Y., Lu, D. & Wang, L. (2012). Impacts of urbanization on stream habitats and macroinvertebrate communities in the tributaries of Qiangtang River, China. *Hydrobiologia*, 680, 39-51.

Weijters, M.J. Janse, J.H., Alkemade, R. & Verhoeven, J.T. (2009). Quantifying the effect of catchment land use and water nutrient concentrations on freshwater river and stream biodiversity. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 19, 104-112.

Williams, P., Biggs, J., Whitfield, M., Thorne A., Bryant, S., Fox, G. & Nicolet, P. (2020a). The Pond Book: A guide to the management and creation of ponds (3rd Edition). The Ponds Conservation Trust: Oxfordshire.

Williams, P., Biggs, J., Stoate, C., Szczur, J., Brown, C. & Bonney, S., (2020). Nature based measures increase freshwater biodiversity in agricultural catchments. Biological Conservation, 244, p.108515.

Williams, P., Whitfield, M., Biggs, J., Bray, S., Fox, G., Nicolet, P. & Sear, D. (2004). Comparative biodiversity of rivers, streams, ditches and ponds in an agricultural landscape in Southern England. *Biological conservation*, 115, 329-341.

Wood, P.J., Greenwood, M.T. & Agnew, M.D. (2003). Pond biodiversity and habitat loss in the UK. *Area*, 35, 206-216.

Tables

Table 1. Summary table of environmental and ecological characteristics for garden and non-urban ponds. PMC: pond margin complexity, SWS: pond surface water shaded, COND: conductivity (in microS cm−1), TP: Total phosphorus, N: Nitrate, BOD: biological oxygen demand.

		Altitude	PMC	SWS (%)	Depth (cm)	Silt depth (cm)	COND	TP	N	BOD	pH	Taxa Richness
Garden Ponds (n=30)	Mean	59.87	2.55	18.3	27.36	5.29	330.36	24.53	0.38	6.56	8.09	11.93
	Standard Error	0.51	0.19	5.63	3.3	1.32	29.38	2.09	0.06	1.25	0.12	1.68
	Min	52	0	0	5.2	0	57	6	0.21	1.37	6.84	1
	Max	70	5	100	78	31	699	43	1.85	29.6	9.58	35
Non-urban Ponds (n=20)	Mean	93.5	1.9	33.62	71.45	16.2	667.38	13.55	3.17	7.35	8.02	35.5
	Standard Error	3.75	0.19	7.85	9.41	3.29	44.37	2.68	1.9	3.56	0.08	4.41
	Min	73	1	0	15	3	335	1	0.1	2	7.12	8
	Max	125	3	95	160	63.3	1029	41	38.3	73	8.88	71

Table 2. Top 5 aquatic macroinvertebrate taxa identified as indicator taxa for garden or non-urban ponds. *=p<0.05, **=p<0.01, ***=p<0.01

Garden ponds	Stat	Non-urban ponds	Stat
Culicidae**	0.42	*Corixa punctata****	0.62
*Planorbis carinatus**	0.36	*Hydraena riparia****	0.58
*Asellus aquaticus**	0.32	Ceratopogonidae***	0.57
		*Glossiphonia complanata****	0.54
		*Lymnaea truncatula****	0.54

Table 3. The number of garden and non-urban ponds identified as *very high*, *high*, *moderate* and *low* conservation value (based on the taxonomic richness and SRI scores)

	Low conservation value (0-5)	Moderate conservation value (>5-10)	High conservation value (>15-20)	Very high conservation value (>20)
Garden ponds	16	13	1	0
Non-urban ponds	0	5	8	7

Figure captions

Fig. 1. Location of the 30 garden ponds and 20 non-urban ponds, across Oxfordshire, and its location in relation to England and Wales (inset). Grey circles = non-urban ponds, black circles = garden ponds.

Fig. 2. NMDS plots of dissimilarity in (A) environmental conditions (Euclidean distance; black circles = garden ponds, grey circles = non-urban ponds) and boxplots of multivariate dispersion distances for (B) environmental conditions from garden and non-urban ponds.

Fig. 3. Species accumulation curves of taxonomic richness (dark grey area = garden ponds, light grey area = non-urban ponds) (A) and prevalence of aquatic macroinvertebrate taxa (B) in garden and non-urban ponds.

Fig. 4. Macroinvertebrate richness (boxes show 25th, 50th and 75th percentiles, and whiskers show 5th and 95th percentiles) (A) and the total number of taxa within the main macroinvertebrate groups (B) recorded from the garden and non-urban ponds.

Fig. 5. NMDS plots of dissimilarity in (A) macroinvertebrate communities (Sørenson dissimilarity; black circles = garden ponds, grey circles = non-urban ponds), and boxplots of multivariate dispersion distances for macroinvertebrate communities (B) from garden and non-urban ponds.

Fig. 6. RDA site plots for garden and non-urban pond macroinvertebrate communities. Significant environmental parameters are presented. Grey circles = garden ponds, Black circles = non-urban ponds

Fig. 1

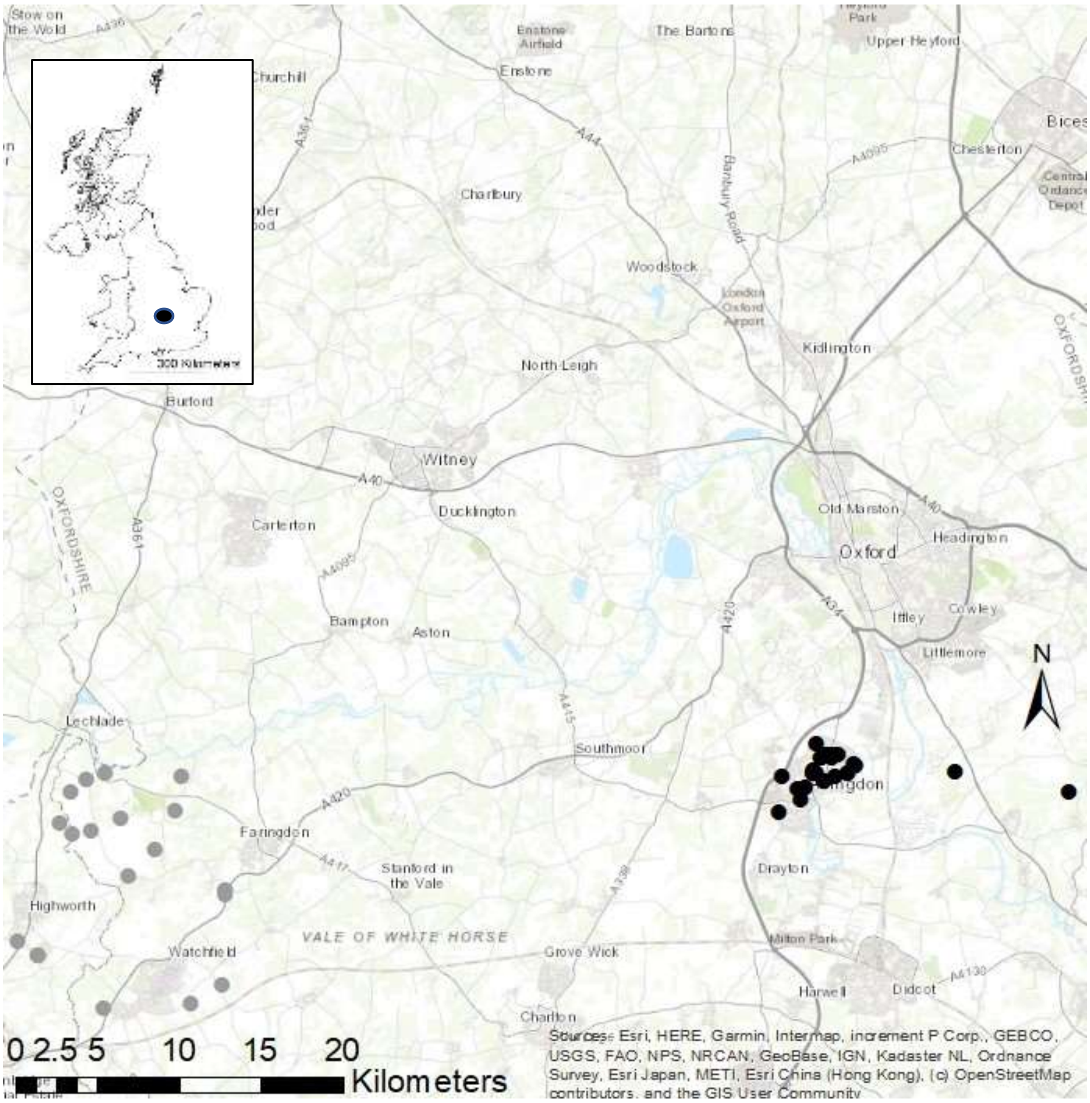
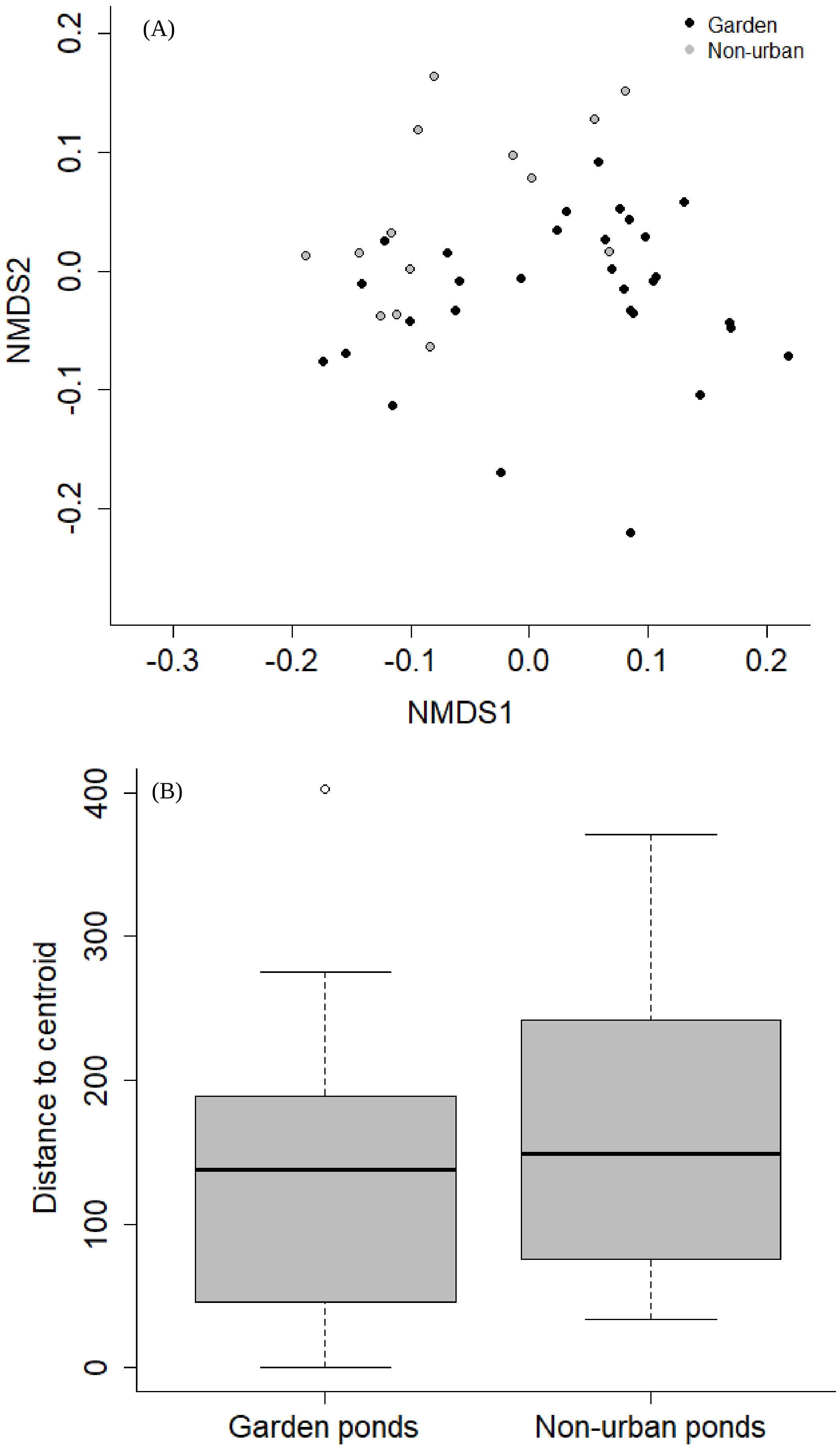
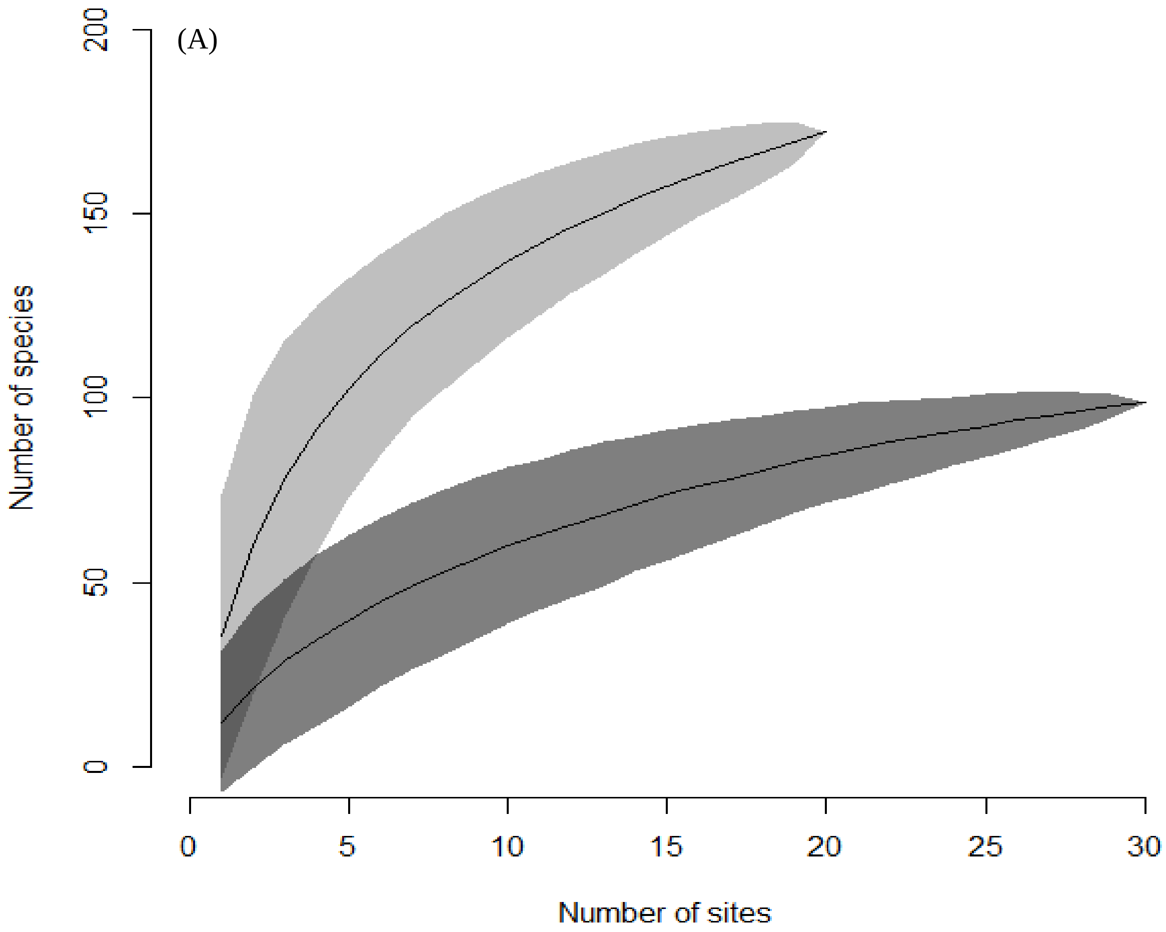

Stow on the Wold
Churchill
Enstone Airfield
Enstone
The Bartons
Upper Heyford
Chesterton
Charlbury
Banbury Road
Woodstock
London Oxford Airport
Kidlington
North Leigh
Burford
Witney
Ducklington
Carterton
OXFORDSHIRE
Old Marston
Headington
Oxford
Iffley
Cowley
Littlemore
Bampton
Aston
Southmoor
Lechlade
Faringdon
Stanford in the Vale
Drayton
Highworth
VALE OF WHITE HORSE
Watchfield
Grove Wick
Milton Park
Harwell
Didcot
Charlton
N
300 Kilometers
0 2.5 5 10 15 20
Kilometers
Sources: Esri, HERE, Garmin, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), (c) OpenStreetMap contributors, and the GIS User Community

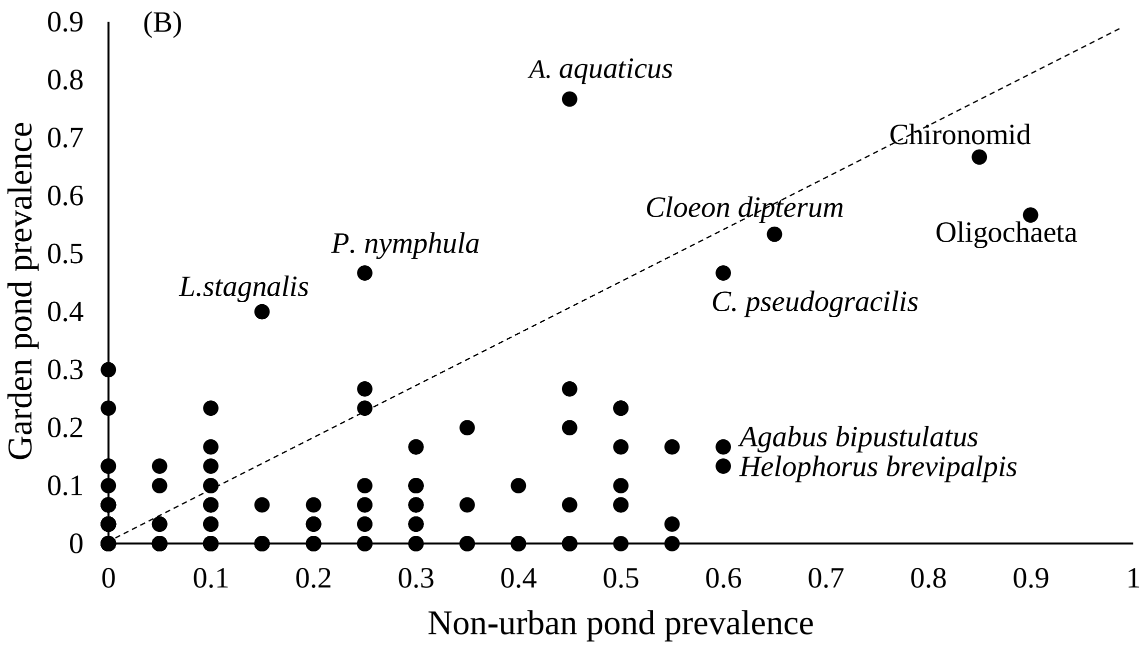
Fig. 2


(A)
Garden
Non-urban
NMDS2
0.2
0.1
0.0
-0.1
-0.2
-0.3
NMDS1
(B)
Distance to centroid
400
300
200
100
0
Garden ponds
Non-urban ponds

645 **Fig. 3**



(A)
Number of species
0
50
100
150
200
0
5
10
15
20
25
30
Number of sites



(B)
Garden pond prevalence
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
A. aquaticus
Chironomid
Cloeon dipterum
Oligochaeta
P. nymphula
L.stagnalis
C. pseudogracilis
Agabus bipustulatus
Helophorus brevipalpis
0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1
Non-urban pond prevalence

Fig. 4

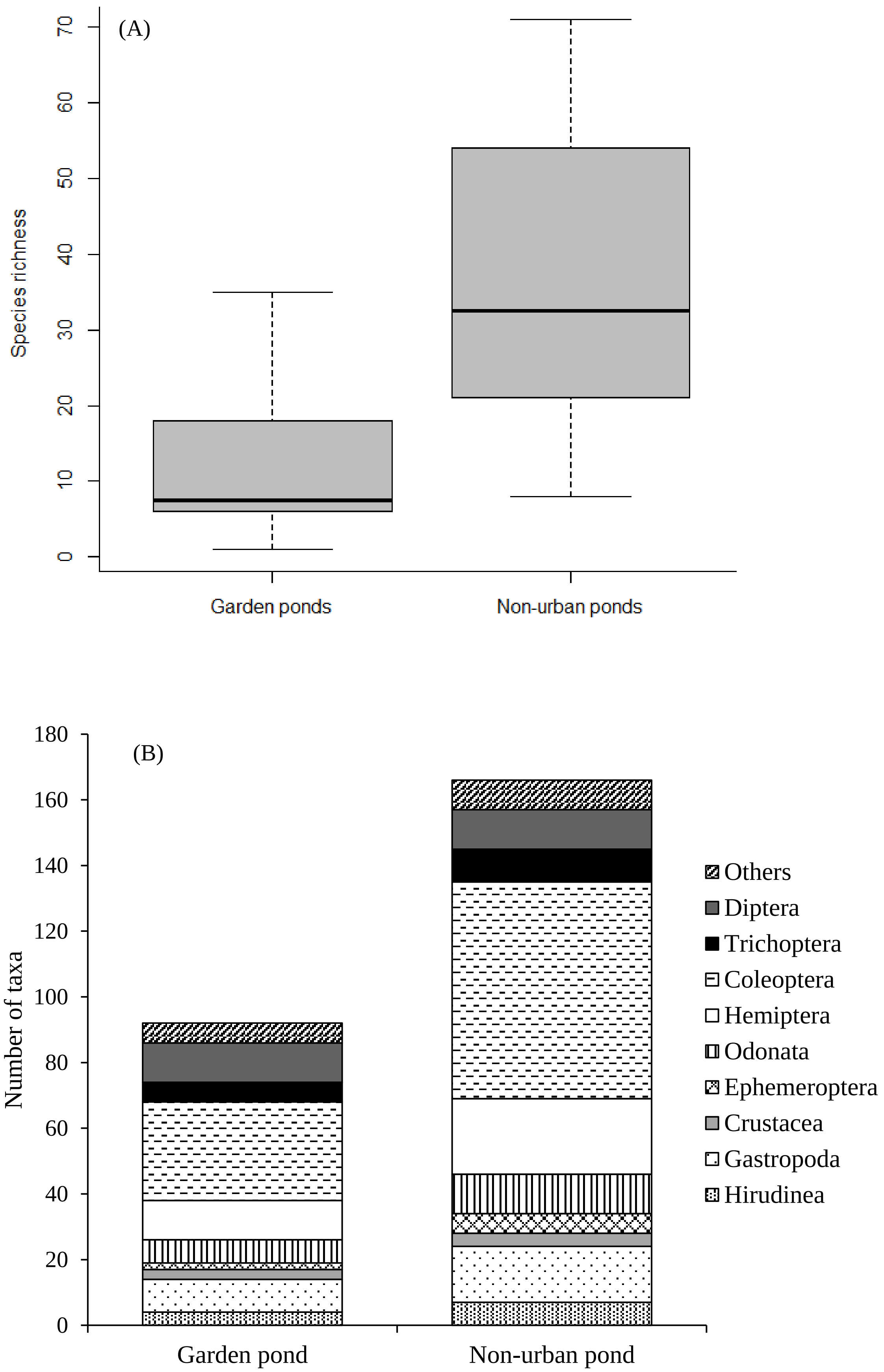

(A)
Species richness
0
10
20
30
40
50
60
70
Garden ponds
Non-urban ponds
(B)
Number of taxa
0
20
40
60
80
100
120
140
160
180
Garden pond
Non-urban pond
Others
Diptera
Trichoptera
Coleoptera
Hemiptera
Odonata
Ephemeroptera
Crustacea
Gastropoda
Hirudinea

Fig. 5

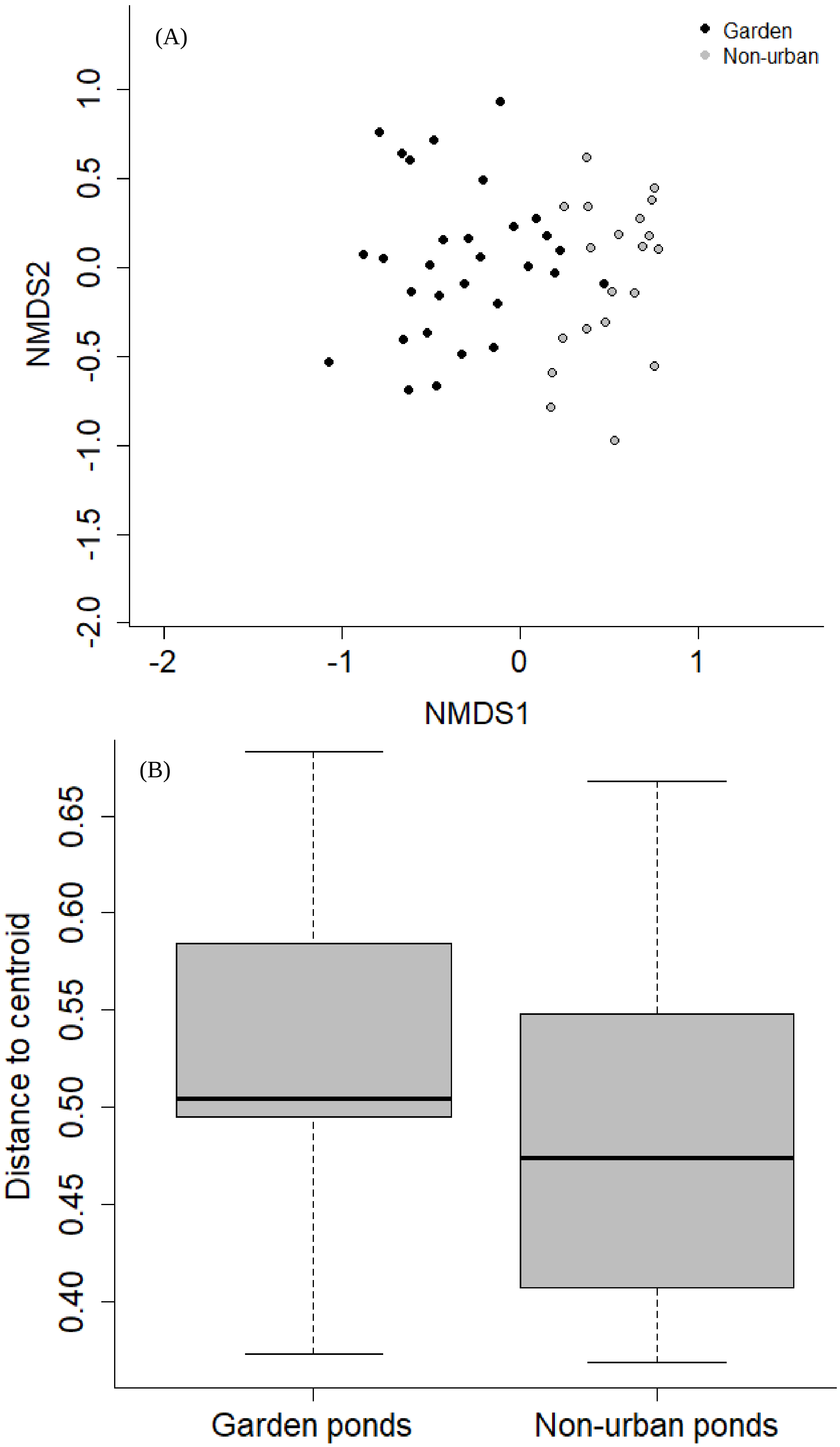

(A)
Garden
Non-urban
NMDS2
1.0
0.5
0.0
-0.5
-1.0
-1.5
-2.0
-2
-1
0
1
NMDS1
(B)
Distance to centroid
0.65
0.60
0.55
0.50
0.45
0.40
Garden ponds
Non-urban ponds

Fig. 6