Macroinvertebrate diversity in urban and rural ponds: implications for freshwater biodiversity conservation

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Abstract

Ponds are among the most biodiverse freshwater ecosystems, yet face significant threats from removal, habitat degradation and a lack of legislative protection globally. Information regarding the habitat quality and biodiversity of ponds across a range of land uses is vital for the long term conservation and management of ecological resources. In this study we examine the biodiversity and conservation value of macroinvertebrates from 91 lowland ponds across 3 land use types (35 floodplain meadow, 15 arable and 41 urban ponds). A total of 224 macroinvertebrate taxa were recorded across all ponds, with urban ponds and floodplain ponds supporting a greater richness than arable ponds at the landscape scale. However, at the alpha scale, urban ponds supported lower faunal diversity (mean: 22 taxa) than floodplain (mean: 32 taxa) or arable ponds (mean: 30 taxa). Floodplain ponds were found to support taxonomically distinct communities compared to arable and urban ponds. A total of 13 macroinvertebrate taxa with a national conservation designation were recorded across the study area and 12 ponds (11 floodplain and 1 arable pond) supported assemblages of high or very high conservation value. Pond conservation currently relies on the designation of individual ponds based on very high biodiversity or the presence of taxa with specific conservation designations. However, this site specific approach fails to acknowledge the contribution of ponds to freshwater biodiversity at the landscape scale. Ponds are highly appropriate sites outside of protected areas (urban/arable), with which the general public are already familiar, for local and landscape scale conservation of freshwater habitats.

Key words: Conservation value, landscape scale, reconciliation ecology, small lentic waterbodies, taxonomic richness

1. Introduction

Freshwaters support some of the most biologically rich and diverse habitats yet include some of the most threatened ecosystems at a global scale (Dudgeon et al., 2006; Gioria et al., 2010). The threats to freshwater biodiversity have been recognised at a policy level, including the over exploitation of their physical (e.g., water) and biological resources (e.g., fisheries), pollution, modification of the hydrological regime, degradation in habitat quality and colonisation by non-native species. As a result, freshwater ecosystems have been a key conservation priority over the last decade following the adoption of resolution 58/217 by the United Nations determining 2005-2015 as the international decade for action on 'water for life' (Dudgeon et al., 2006).

Over the last two decades, research centred on the conservation of pond flora and fauna has increased significantly, with the number of primary research papers published within academic journals addressing *pond biodiversity* tripling in the last decade (Cereghino et al., 2014). Previous research has demonstrated that ponds (standing waterbody between 25 m^2 and 2 ha in size; Williams et al., 2010) have the capacity to support a greater biodiversity of aquatic macroinvertebrates and macrophytes, as well as higher proportions of rare and endemic species than other freshwater habitats (Williams et al., 2003; Davies et al., 2008). This contribution to biodiversity may become particularly important in anthropogenically-dominated urban landscapes and intensive agricultural areas, where ponds may represent biodiversity hotspots and islands of aquatic habitat in otherwise ecologically poor environments (Sayer et al., 2012; Cereghino et al., 2014). Moreover, ponds provide a range of ecosystem services including; 1) environmentally sustainable solutions to water management - water storage (flood alleviation), nutrient and sediment retention, and; 2) local scale carbon storage/sequestration and mitigation for urban heat island effects (Downing et al., 2008; Coutts et al., 2012; Cereghino et al., 2014; Hassall, 2014).

Despite the wider importance of ponds to society and biological communities, freshwater conservation efforts globally have been primarily focussed on lotic and larger lentic waterbodies, whilst small freshwater bodies have been largely ignored (Williams et al., 2003; Oertli et al., 2009).

International legislation in relation to freshwater resources and ecosystems falls into two broad categories; 1) *pollution and water resources* - legislation focussed on improving the quality of freshwater and; 2) *nature conservation* - legislation orientated towards the protection of habitats that are under significant threat and species with specific designations (Hassall et al., 2016). At a European scale these two categories form the basis for the EU Water Framework Directive (WFD; *pollution and water resources*) and the EU Habitats Directive (*nature conservation*) which have been incorporated into national legislation across the 28 EU member states (Hassall et al., 2016). However, the WFD only affords protection to larger lentic systems (lakes >50ha), despite its key objective to improve the quality of all freshwater habitats (EC, 2000; Sayer, 2014). More recently national and international nature conservation agencies have highlighted the value of ponds more readily than those responsible for water resources and as a result, nature conservation legislation has afforded greater (but still significantly limited) protection to pond habitats and their biodiversity (Hassall et al., 2016). A limited number of pond types (e.g., Mediterranean temporary ponds) and species associated with them (e.g., the Great Crested Newt, *Triturus cristatus*) are recognised under the EU Habitats Directive (Oertli et al., 2005). However, in the absence of statutory routine (regular) monitoring of ponds across most of Europe, it is likely many ponds which meet the requirements to be afforded protection have been overlooked (Biggs et al., 2005). As a result of the lack of legislative protection, many ponds have been lost to infilling/drainage due to agricultural intensification or urban development, which has led to increasingly fragmented and isolated pond networks (Hull, 1997; Wood et al., 2003; Zacharias et al., 2007; Davies et al., 2009). In addition, many ponds suffer from poor habitat and water quality due to nutrient enrichment (chemical and organic) and the introduction of non-native species (Biggs et al., 2007; Williams et al., 2010).

While designated areas remain important to protect species and habitats, there is a need to consider biodiversity conservation outside of protected areas as the small land coverage of nature reserves is likely to be insufficient to protect the majority of biodiversity (Le Viol et al., 2009). Ponds are a common landscape features globally (Downing et al., 2006), and may provide suitable habitats and important refuges for aquatic and riparian flora and fauna in anthropogenically-dominated landscapes

(Chester and Robson, 2013; Hassall and Anderson, 2015) yet comparatively little is known about their wider value.

Given the potentially high ecological value of ponds, information regarding their biological quality is vital to the long term conservation and management of freshwater biodiversity (Gioria et al., 2010). Pond biodiversity research at larger scales has typically focussed on invertebrate diversity within a particular landscape setting (Céréghino et al., 2008; Gledhill et al., 2008; Usio et al., 2013). This is the first study to our knowledge which has considered the regional macroinvertebrate biodiversity of ponds across a range of lowland land use types. The current investigation of lowland ponds within a mixed urban and agricultural landscape setting specifically sought to: (1) quantify the macroinvertebrate diversity associated with floodplain, agricultural arable and urban ponds; (2) characterise the heterogeneity of faunal communities between and among floodplain, agricultural arable and urban ponds and; (3) examine the importance of ponds to landscape-scale biodiversity conservation.

2. Materials and Methods

2.1 Site Selection

A total of 91 ponds were examined (67 perennial, 24 ephemeral), close to the town of Loughborough (Leicestershire, UK; Fig. 1). The study area has a temperate climate with an average annual minimum temperature of 6.1 ºC, an average annual maximum temperature of 13.9 ºC and mean annual precipitation of 620.2 mm (1981-2010, data provided by the Met Office; Met Office, 2015). An exhaustive survey of pond habitats was undertaken using maps and aerial images using Google Earth software (Google Earth, 2015) to identify ponds in the study area. The ponds were located in three common land-use types typical of lowland landscapes in Europe; (i) floodplain ponds (35) located on floodplain meadows which are protected for nature conservation (Nature Reserves) and were naturally inundated by water from the River Soar during the winter and early spring; (ii) arable ponds (15) - located on intensively cultivated land – predominantly rapeseed or wheat crops; and (iii) urban ponds (41) - located within residential gardens, public spaces (parks), school grounds (used as educational tools) and high density commercial developments (urban drainage ponds; industrial, roadside and town centre locations; Hill et al., 2015). It is widely acknowledged that there are large numbers of urban ponds (Hassall, 2014) and floodplain ponds across the UK, whilst agricultural pond numbers have been in consistent decline for many decades (Wood et al., 2003). In addition, difficulties surrounding access to agricultural land when in crop resulted in the number of arable ponds surveyed being lower than urban and floodplain ponds.

2.2 Macroinvertebrate sampling

Each pond was sampled for aquatic macroinvertebrates on three occasions corresponding to spring (March), summer (June) and autumn (September) in 2012. Full details and rationale of field and laboratory sampling procedures are presented in Hill et al. (2015) and summarized here. The length of time allocated to sample aquatic macroinvertebrates in each pond was proportional to its surface area (Hinden et al., 2005) up to a maximum of three minutes (Biggs et al., 1998). A total of three minutes sampling time was assigned to ponds greater than 50 m^2; for smaller ponds 30 seconds of sampling for every 10 m^2 surface area was employed. Sampling time allocated to each pond was divided equally between the mesohabitats present (e.g., submerged macrophytes, emergent macrophytes, floating macrophytes and open water) although, if a single mesohabitat dominated the pond, sampling time was divided further to reflect this (Biggs et al., 1998). An inspection of any larger substrates (e.g., rocks) that could not be sampled with a pond net was undertaken for up to 60 seconds to ensure that all available habitats were sampled. Aquatic macroinvertebrate samples were processed in the laboratory and preserved in 70% industrial methylated spirits. Macroinvertebrate taxa were identified to species level wherever possible, although Diptera larvae, Planariidae and Physidae were identified to family level and Collembola, Hydrachnidiae and Oligochaeta were identified as such.

2.3 Environmental data collection

At each sample site a range of environmental characteristics were recorded including; surface area (m^2), mean water depth (cm), dry phase (duration during the 12-month study period that the pond was dry), the percentage of the pond margin that was shaded, conductivity (microS cm^{-1}: recorded using a

Hanna conductivity meter: HI198311), pH (recorded using a Hanna pH meter: HI98127), water temperature, (recorded using a Hanna pH meter: HI98127), surface (<20 cm depth) dissolved oxygen (DO mg l^{-1}: recorded using a Mettler Toledo Dissolved Oxygen Meter) and percentage of pond surface covered by submerged macrophytes, emergent macrophytes, floating macrophytes and open water. Pond connectivity (the number of waterbodies hydrologically connected to the sample site) and pond isolation (the number of other waterbodies within 500 m: Waterkeyn et al., 2008) were recorded using aerial imagery (Google Earth 2015) or maps and through field observations (extensively walking around each sample site during each season to identify any nearby waterbodies). Every effort was made to record all waterbodies within 500m of each pond site, however ephemeral ponds and garden ponds were particularly difficult to identify as many have never been recorded on national maps (OS MasterMap) and are not always visually apparent through inspection of aerial images via Google Earth software, particularly when overgrown or covered by overhanging vegetation. It is therefore acknowledged that some ephemeral and garden ponds will have been omitted.

2.4 Statistical Analysis

Macroinvertebrate species-abundance data from each season for individual ponds were pooled in the final analysis to provide a measure of alpha diversity within each pond. Rarefaction (Hulbert, 1971) was undertaken in PRIMER 6 to estimate species richness for each pond site based on a given number of individuals drawn randomly from a sample (McCabe and Gotelli, 2000). The least abundant pond study site had 41 individuals and as a result we randomly sampled 41 individuals from each replicate and recorded the rarefied species richness. Such analyses allow for comparisons of species richness based on specific numbers of individuals and as a result avoids biases associated with comparing different sample sizes (Ning and Nielsen, 2011). Before any statistical analyses were undertaken 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 not observed (e.g., for macroinvertebrate abundance data) the data were transformed ($\log_{10}$). Differences in faunal diversity (abundance and richness: alpha diversity) and environmental variables between floodplain, arable and urban ponds was examined using One-Way Analysis of Variance (ANOVA) and *post hoc* Tukey tests in IBM SPSS Statistics (version 21, IBM Corporation, New York) to quantify where differences among different pond types occurred. Gamma diversity was calculated as the total number of aquatic macroinvertebrate taxa recorded among all pond study sites. In addition, estimated gamma diversity was calculated using the Chao1 estimator in PRIMER 6.

Differences in environmental conditions and aquatic macroinvertebrate communities between pond types were visualised using NMDS with the *metaMDS* function in the vegan package in R (Okansen et al., 2015) and examined statistically by analysis of similarity (ANOSIM) in PRIMER v6 (Clarke and Gorley, 2006). SIMPER analysis was undertaken in PRIMER 6 to identify those taxa which contributed most to the statistical differences in macroinvertebrate community composition between floodplain, agricultural and urban ponds. Faunal abundance and environmental data were log transformed prior to ANOSIM, SIMPER and NMDS analysis. To examine the heterogeneity of environmental conditions and faunal composition among pond types, analysis of homogeneity of multivariate dispersions (PERMDISP) was undertaken using the vegan package (Okansen et al., 2015) and compared using One-way Analysis of Variance. Bray–Curtis dissimilarity was used for the macroinvertebrate taxa data and Euclidean distance was used for the environmental data for NMDS, ANOSIM and PERMDISP analysis. Redundancy Analysis (RDA) was employed to examine the relationship between macroinvertebrate composition and environmental variables. Prior to analysis, species-abundance data was Hellinger transformed (Legendre & Gallagher, 2001) and environmental parameters were $\log_{10}$ transformed (to reduce the influence of skew and overcome the effect of their physical units; Legendre & Birks, 2012). A stepwise selection procedure (forward and backward selection) using permutation-based significance tests (999 permutation) was used to determine the environmental variables that significantly ($p<0.05$) explained the variance in pond community composition. Only environmental parameters identified to significantly influence the macroinvertebrate assemblage were included in the final model. RDA was undertaken using the *ordistep* function in vegan.

The conservation value of each pond was examined using the Species Rarity Index (SRI) and the Community Conservation Index (CCI). The rarity value assigned to each macroinvertebrate for the CCI and SRI is based on the UK Joint Nature Conservation Committee (JNCC) designations (see Chadd and Extence, 2004 Appendix 1 and Williams et al., 2003). To calculate SRI, the rarity/threat value assigned to each macroinvertebrate taxa in the pond assemblage is summed and then divided by the number of species recorded in the pond sample (Williams et al., 2003; Rosset et al., 2013). CCI incorporates both the rarity of macroinvertebrate species at a national scale (conservation scores based on published sources and expert opinion) and the community richness (see Chadd and Extence, 2004). CCI can provide the basis for the development for conservation strategies when used in conjunction with knowledge of the habitat requirements of target organisms and communities (Chadd and Extence, 2004; Armitage et al., 2012).

3. Results

3.1 Environmental characteristics

The percentage of surface water shaded (ANOVA $F_{2, 90}$=6.94; p<0.01) and the percentage of floating macrophyte coverage (ANOVA $F_{2, 90}$=8.08; p<0.001) was significantly lower for floodplain ponds than arable or urban ponds (Table 1). Conductivity was significantly higher in arable ponds compared to urban ponds (ANOVA $F_{2, 90}$=3.59; p<0.05; Table 1). Pond isolation (ANOVA $F_{2, 90}$=74.19; p<0.001) and connectivity (ANOVA $F_{2, 90}$=26.09; p<0.001) were significantly higher for floodplain ponds than urban or arable ponds (Table 1). There was no significant difference in pond area, pond depth, percentage of the pond covered by emergent or submerged macrophytes, pH or dissolved oxygen among the three pond types examined.

3.2 Macroinvertebrate diversity

A total of 224 macroinvertebrate taxa were recorded from 21 orders and 68 families (see Supplementary Material Appendix 1) from floodplain (total: 175, range: 5-73), arable (total: 131, range: 9-51) and urban (total: 170, range: 2-61) ponds. Estimated gamma diversity (based on the Chao 1 estimator) was higher in floodplain (estimated 205 taxa) and urban ponds (estimated 194 taxa) than in arable ponds (estimated 142 taxa). On average, coleopteran taxa constituted a much greater proportion of taxonomic richness recorded in floodplain ponds (27%) compared to arable (12%) and urban ponds (11%; Fig. 2). Similarly, 16% of macroinvertebrate taxa recorded from floodplain ponds were hemipteran taxa compared to 9% in urban ponds and 1% in arable ponds. Within urban ponds, Diptera larvae formed, a greater proportion of the taxa richness (25%) than the other two pond types (floodplain: 12%, arable: 7%) whilst Ephemeroptera and Hirudinea constituted a greater proportion of taxonomic richness in arable ponds compared to floodplain and urban ponds (Fig. 2).

Floodplain ponds (mean taxon richness: 39.2) supported significantly greater macroinvertebrate richness (ANOVA $F_{2, 90}$=8.69; $p<0.001$) and rarefied species diversity (ANOVA $F_{2, 90}$ = 11.75; $p<0.001$) when compared to urban ponds (mean richness: 21.7; Fig. 3a). There was no significant difference in mean macroinvertebrate richness between arable ponds (mean richness: 30.9) and floodplain or urban ponds; although floodplain and urban ponds displayed greater variation in taxonomic richness (Fig. 3a). A total of 69% of floodplain ponds (24 ponds) and 53% of arable ponds (8 ponds) supported >30 taxa, whereas only 29% of urban ponds (12 ponds) recorded >30 taxa. The greatest taxonomic richness was recorded from a floodplain pond (73 taxa) and all 5 ponds with the greatest alpha macroinvertebrate richness were located on floodplains. No significant difference in the abundance of macroinvertebrates was recorded among floodplain, arable and urban ponds.

3.3 Faunal heterogeneity

A clear distinction between aquatic macroinvertebrate assemblages in floodplain, urban and arable ponds was observed within the NMDS ordination (Fig. 4a). Floodplain ponds supported significantly different macroinvertebrate assemblages compared to arable and urban ponds (ANOSIM $p<0.01$ r= 0.19). There was no significant difference in the macroinvertebrate assemblages recorded from urban and arable ponds. The top four macroinvertebrate taxa (identified by SIMPER analysis) driving the difference in differences in community composition between floodplain ponds and arable were Chironomidae (contributing 6.81% to the dissimilarity), Culicidae (4.96%) and Chaoboridae (4.64%) which were recorded in higher abundance in arable ponds and *Crangonyx pseudogracilis* (4.06%) which recorded a higher abundance in floodplain ponds. Greater abundances of Chironomidae

(6.84%), *C. pseudogracilis* (6.03%), *Asellus aquaticus* (4.96%) and Oligochaeta in urban ponds were identified by SIMPER as the top 4 macroinvertebrate taxa driving the community heterogeneity between floodplain ponds and urban ponds. The average median distance to the group centroid based on aquatic macroinvertebrate community dissimilarity (faunal multivariate dispersion) was similar among floodplain (0.57), arable (0.52) and urban (0.54) ponds (ANOVA $F_{2, 88}$=0.91; p=0.4; Fig. 4c) indicating that faunal communities in the three pond types showed similar levels of variation in faunal community composition. Environmental characteristics among floodplain, arable and urban ponds overlapped in the NMDS biplot and ANOSIM did not identify any statistical differences between environmental characteristics for the three pond types (ANOSIM r=0.041 p=0.07; Fig. 4b). The average median distance to the group centroid based on environmental dissimilarity was greater for urban ponds (917.6) than floodplain (479.2) and arable (539.4) ponds, although this was not statistically significant (ANOVA $F_{2, 88}$=0.99; p=0.38; Fig. 4d).

Redundancy analysis identified six significant environmental parameters correlated with the first two RDA axes: connectivity, pond dry months, pH, pond area (all p<0.005), percentage pond margin shaded and percentage pond coverage of emergent macrophytes (p<0.05) (Fig. 5). The RDA axes were highly significant (F=3.477 p<0.001), explaining 26% of macroinvertebrate community variation on all constrained axes, based on the adjusted R^2 values (Adjusted R^2=0.26). Floodplain ponds were separated from urban and agricultural ponds on the first and second axes along gradients associated with connectivity and the number of months the pond dried (Fig. 5). Floodplain ponds were characterized by a greater connectivity, area and ephemerality, whilst urban and agricultural ponds were associated with a greater percentage of the pond margin shaded, greater emergent macrophyte coverage but reduced connectivity and ephemerality (Fig. 5).

3.4 Conservation value

A total of 13 macroinvertebrate species with a conservation designation were recorded within the ponds examined (Table 2). In all, 23 ponds (24% of total sample sites) supported one or more invertebrate species with a conservation designation (13 floodplain ponds, 5 urban ponds and 5 arable ponds; Table 2). Floodplain ponds supported assemblages with significantly higher Species Rarity Index (SRI) values than urban ponds (ANOVA $F_{2, 90} = 6.02$ p>0.01; Table 2). Communities within floodplain ponds had significantly greater Community Conservation Index (CCI) scores than arable and urban ponds (ANOVA $F_{2, 90} = 12.87$ p>0.001; Table 2). Macroinvertebrate communities within 6 pond sites were of *very high conservation value* (5 floodplain ponds and 1 arable pond) based on their CCI scores (Fig. 6). In addition, 6 ponds were of *high conservation value* (6 floodplain ponds). No urban ponds were found to have a high or very high conservation value (Fig. 6). A total of 60% of ponds across the study region (34% of floodplain ponds, 60% of arable ponds and 76% of urban ponds) supported communities of low or moderate conservation value based on the CCI scores.

4. Discussion

This study has demonstrated that ponds support rich faunal communities of potentially high conservation value in rural and urban settings. Yet operationally, pond conservation remains a significant issue across Europe as a result of the lack of legislative power to protect pond habitats and their associated flora and fauna (Hassall et al. 2016). In Europe, the conservation of ponds currently relies heavily on the presence of rare taxa or records of very high biodiversity in order to designate individual ponds (Hassall et al., 2012). The current system of individual site designation remains an important mechanism for pond conservation as the process can protect species-rich habitats and rare taxa (BRIG 2011). However, the scale at which the current designation of ponds is applied is quite different to the scale at which ponds contribute most towards aquatic biodiversity. This study has demonstrated that faunal richness and conservation value at the alpha scale was highly variable (2-73 taxa) but ponds made a significant contribution to biodiversity at the landscape scale. Similar findings were recorded elsewhere in the UK by Williams et al. (2003) and Davies et al. (2008) who found that ponds supported significantly higher macroinvertebrate taxonomic richness at a landscape scale than rivers, lakes and ditches. The small, discrete surface catchments of ponds can result in a wide range of habitats/conditions for macroinvertebrate taxa to colonise and the development of highly diverse and heterogeneous communities at a landscape scale (Williams et al., 2003; Davies et al., 2008); as demonstrated by the high multivariate dispersion observed among pond types in this study. High macroinvertebrate community heterogeneity can be further attributed to the increased influence of stochastic events (related to dispersal limitation or priority effects) on small water bodies (Scheffer et al., 2006). As a result, pond conservation strategies need to be developed and applied at the landscape-scale to provide the greatest potential benefit to aquatic biodiversity (Davies et al., 2008; Sayer, 2014). Temporal studies of pond biodiversity have also demonstrated that the conservation value of individual ponds fluctuates over time as rare taxa present during one year may be absent the next (Greenwood and Wood., 2003; Hassall et al., 2012). This further suggests moving away from the designation of individual ponds towards the conservation of pond clusters and 'pondscapes' to provide the greatest long term conservation benefit for biodiversity (Hassall et al., 2012).

Floodplain ponds supported heterogeneous communities and were of a significantly higher conservation value compared to urban ponds in this study. This probably reflects floodplain ponds location in semi-natural landscapes (nature reserves), the resulting management practices (designed to benefit biodiversity), reduced shading (Sayer et al., 2012), their high connectivity to other waterbodies and reduced anthropogenic disturbances. In contrast, urban ponds are located in structurally complex and fragmented urban landscapes with lower connectivity (Noble and Hassall, 2014). When combined with the high levels of anthropogenic disturbance (e.g., urban runoff/pollution) and management practices (for purposes other than biodiversity: Briers, 2014), this can result in very different macroinvertebrate communities to floodplain and arable ponds. Floodplain pond communities typically had good water quality and high coverage of emergent and submerged macrophytes, providing suitable conditions for taxa of high conservation value and a dominance of Coleoptera and Hemiptera taxa, while high connectivity to other waterbodies also promoted easy dispersal between them. Urban ponds were dominated by Diptera larvae, which have been recorded to colonise isolated urban ponds (Gaston et al., 2005) and many have broad tolerances to adverse environmental conditions (Carew et al., 2007; Serra et al., 2016). Although environmental conditions were widely dispersed in the NMDS biplot they were not found to be statistically different between floodplain, agricultural and urban ponds. This most likely reflects the variability in environmental conditions across all three ponds types but may also reflect the limited number of environmental variables recorded. Further detailed examination of hydrochemical data, substrate type and bank type would have added greater information regarding environmental conditions within the ponds examined and the key environmental variables driving pond community composition to (26% of variation was explained by the RDA, indicating that other unmeasured abiotic variables influence community structure) and should be considered in future investigations.

Biodiversity conservation at a landscape scale commonly relies on designated areas or reserves to protect individual species and habitats (Briers, 2002; Mcdonald et al., 2008). In this study, the ponds of greatest biodiversity and conservation value were located on floodplain meadows specifically identified as nature conservation areas providing protection from anthropogenic disturbance. Nature reserves can help deliver landscape-scale (pondscape) conservation, especially on lowland floodplains, providing a highly connected freshwater landscape (incorporating rivers, lakes, ponds, ditches and wetlands) supporting high numbers of rare taxa and allowing organisms to disperse widely and colonise different aquatic habitats (Cottenie, 2005; Williams et al., 2008; Sayer, 2014). However, increasing anthropogenic land cover is projected to threaten the flora and fauna within many of these protected areas (Guneralp and Seto, 2013). The conservation of species or habitats should not depend exclusively on designated sites (Chester and Robson, 2013; Baudron and Giller, 2014), and biodiversity conservation should be opportunistically enhanced wherever possible. Many ponds provide rich and diverse habitats outside of protected areas (as demonstrated in this study by urban ponds similar diversity to floodplain ponds at the landscape scale) suitable for freshwater landscape-scale conservation. In the UK, the Wildlife Trusts are incorporating a ‘living landscape approach’ which provides landscape-scale conservation outside of conservation areas, restoring links and corridors through the creation of meadows, hedges and ponds between wildlife sites in urban and rural landscapes to reconnect large areas of land separated in the last 100-200 years to enhance biodiversity and create ‘wildlife-friendly’ environments (The Wildlife Trusts, 2014).

Whilst there is consensus regarding the value of undertaking pond conservation at the network/landscape scale, there is debate about how best to achieve this (Sayer et al., 2012). Currently the focus is on the building of new high quality ponds in response to pond loss and to increase pond connectivity. For example, the Million Ponds Project is a 50-year project which seeks to create a network of 500,000 (in addition to the existing 500,000 ponds in the UK) new clean water ponds across the UK (Freshwater Habitats Trust, 2014). However, management and restoration can provide a complimentary conservation strategy alongside pond creation to mitigate the impact of urbanisation and land use intensification and restore and improve aquatic biodiversity of the existing pond resource (Oertli et al., 2005; Sayer et al., 2013; Hassall, 2014). Agri-environment schemes (AES) provide financial compensation to farmers who incorporate measures which promote and benefit biodiversity, including maintaining pond habitats on agricultural land (Kleijn and Sutherland, 2003; Davies et al., 2008). Despite this, farmland pond numbers continue to decline and many agricultural/arable ponds are typically left unmanaged resulting in degraded ponds with poor habitat quality (e.g., high levels of pond shading), which over time can fill with sediment (Sayer et al., 2012). Active management such as sediment, tree and scrub removal is required in many agricultural areas to improve the condition of the resource for biodiversity and potentially create a culture of care and pride in relation to agricultural ponds (Sayer et al., 2012; Riordan et al., 2015). The agricultural ponds in this study had lower landscape-scale diversity than the other two pond types, reflecting their lack of management (most were at a late successional stage) and location in a homogenous, intensively farmed landscape (Boothby, 2003; Sayer et al., 2012). Agricultural conservation initiatives (such as AES) may be most beneficial when undertaken at smaller spatial scales (pond clusters) than larger scales, as the most effective locations can be targeted which will provide the maximum diversity for the economic and effort input (Davies et al., 2009).

For ponds located in agricultural or urban landscapes where their primary function is not for biodiversity, the application of reconciliation ecology (Rosenzweig, 2003) as a management/conservation tool may be the most beneficial way to improve biodiversity at larger geographical scales. Reconciliation ecology suggests ways to modify and diversify anthropogenically-created habitats to improve their biological conditions whilst maintaining the effectiveness of their primary function (Rosenzweig, 2003). Previous research has shown that only small changes to current management techniques for freshwaters in urban and agricultural landscapes is likely to significantly improve faunal richness in these anthropogenically-dominated landscapes (Twisk et al., 2000; Twisk et al, 2003; Hill et al., 2015). Reconciliation ecology as a management/conservation strategy for ponds has the potential to meet the needs of humans (e.g., flood alleviation, water storage) and support the conservation of biological diversity in landscapes subject to anthropogenic processes associated with urbanisation (Chester and Robson, 2013; Moyle, 2014). In addition, raising awareness of the contribution of urban ponds to biodiversity may also play a key role in influencing and shaping the perceptions of land owners, local government and general public regarding 1) the importance of ponds for freshwater conservation, 2) the urban-rural landscape as a functional interconnected system and 3) the wider conservation agenda. However, complications surrounding land ownership, increasing development on urban green space and the economic value of urban land may make landscape scale conservation in urban and peri-urban areas difficult to navigate and implement for policy makers.

4.1 Conclusion

This study has demonstrated that floodplain ponds supported the greatest macroinvertebrate diversity of the three land uses examined. However, ponds associated with arable and urban land uses also provide habitats of rich macroinvertebrate diversity and high conservation value. Ponds contribute significantly to biodiversity at a landscape scale and focussing conservation efforts at this scale is likely to be the most ecologically beneficial and sustainable way to conserve pond networks, promote regional biodiversity across rural and urban landscapes and increase the connectivity between ponds and other freshwater habitats. While specially designated areas for conservation remain an important strategy for biodiversity conservation, ponds provide aquatic habitat outside of protected areas suitable for freshwater landscape scale conservation. Pond conservation at the landscape scale may be best served by a combination of pond management and the creation of new ponds, which will greatly increase the numbers of high quality pond habitats and provide a range of pond types and environmental conditions suitable for a wide range of flora and fauna. Ponds need to be incorporated in more detail into freshwater conservation legislation. In particular, there is a need for an integrated approach to freshwater conservation incorporating ponds with other freshwaters to provide an efficient and sustainable way of protecting freshwater biological diversity.

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Tables

Table 1 - Summary table of environmental characteristics for urban, floodplain and arable ponds. SWS: pond surface area shaded, EM: emergent macrophytes, SM: submerged macrophytes, FM: floating macrophytes, COND: conductivity (in microS cm^{-1}), Iso: pond isolation and Connect: pond connectivity.

		Area (m^2)	Depth (cm)	SWS (%)	EM (%)	SM (%)	FM (%)	pH	COND	Iso	Connect
Urban (n = 41)	Mean	780.3	67.5	17.5	23.0	21.1	15.8	7.8	501.3	4	0.5
	Standard Error	301.3	10.3	4.5	4.6	3.7	4.1	0.1	43.8	0.4	0.2
	Min	0.8	4	0	0	0	0	6.3	63.7	0	0
	Max	9309	>200	100	100	90	96.7	9.8	1322	9	3
Floodplain (n = 35)	Mean	376.8	52.5	6.1	21.5	29.1	2.1	8	613.7	16	6
	Standard Error	154	6.5	3.3	4.4	4.5	1	0.1	50.7	1.1	1
	Min	10.3	8	0	0	0	0	6.4	80	7	0
	Max	5256	>200	93.3	86.7	100	30.3	9.1	1494	30	14
Arable (n = 15)	Mean	432.5	71.6	22.4	29.4	13.8	10.1	7.9	728.3	6	0
	Standard Error	295.8	15.1	8.5	7.5	3.2	3.8	0.1	78.6	0.7	0.1
	Min	24.4	12	0	0	0	0	7.4	205.0	0	0
	Max	4566	>100	100	86.7	37.3	55.0	8.3	1326.7	9	2
Region (n = 91)	Mean	567.8	62.4	13.9	23.6	23.0	9.6	7.9	582.0	9	3
	Standard Error	155.8	5.8	2.8	2.8	2.5	2.1	0.1	31.5	7.1	4.8
	Min	0.8	4	0	0	0	0	6.3	63.7	0	0
	Max	9309	>100	100	100	100	96.7	9.8	1494	30	14

Table 2 - Mean macroinvertebrate Community Conservation Index (CCI) scores, mean Species Rarity Index Scores (SRI) and the aquatic macroinvertebrate taxa with a conservation designation recorded from floodplain, arable and urban ponds.

	Floodplain	Arable	Urban
Mean CCI	13.14	8.97	6.20
Mean SRI	1.093	1.067	1.039
Number of ponds supporting at least one taxa with a conservation designation (/ total)	13 (/35)	5 (/15)	5 (/41)
Taxa with conservation designation	*Berosus luridus* *Ilybius subaeneus* *Agabus conspersus* *Hygrotus nigrolineatus* *Rhantus frontalis* *Helophorus dorsalis* *Paracymus scutellaris*	*Sisyra terminalis* *Agabus conspersus* *Rhantus frontalis* *Helophorus dorsalis* *Helophorus strigifrons*	*Coenagrion pulchellum* *Gyrinus distinctus* *Agabus uliginosus* *Helochares punctatus* *Helophorus strigifrons*

Figures

Figure 1

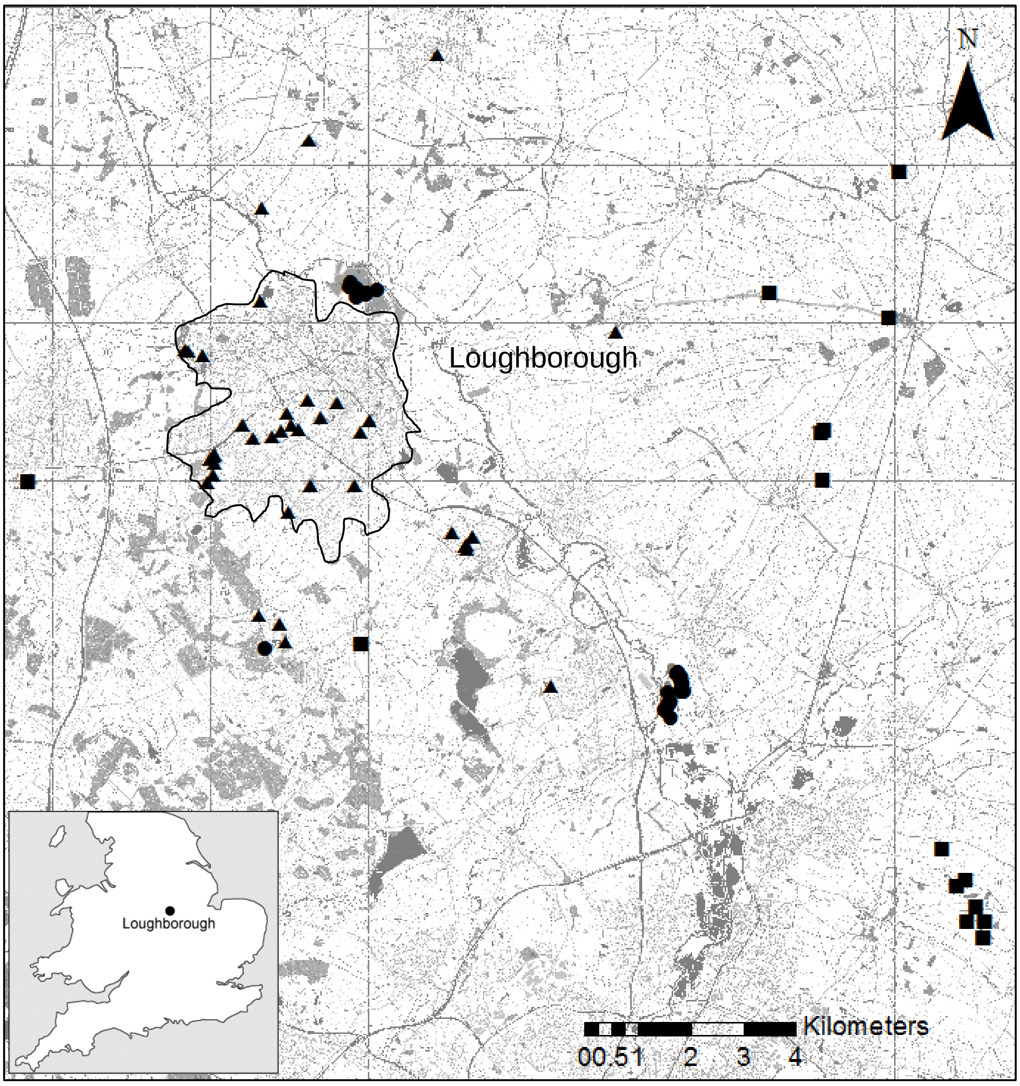

Figure 1 - Location of the 91 ponds (35 floodplain, 41 urban and 15 agricultural ponds) examined in Leicestershire, UK and its location in relation to England and Wales (inset). Triangles = urban ponds, circles = floodplain ponds and squares = agricultural ponds.

Figure 2

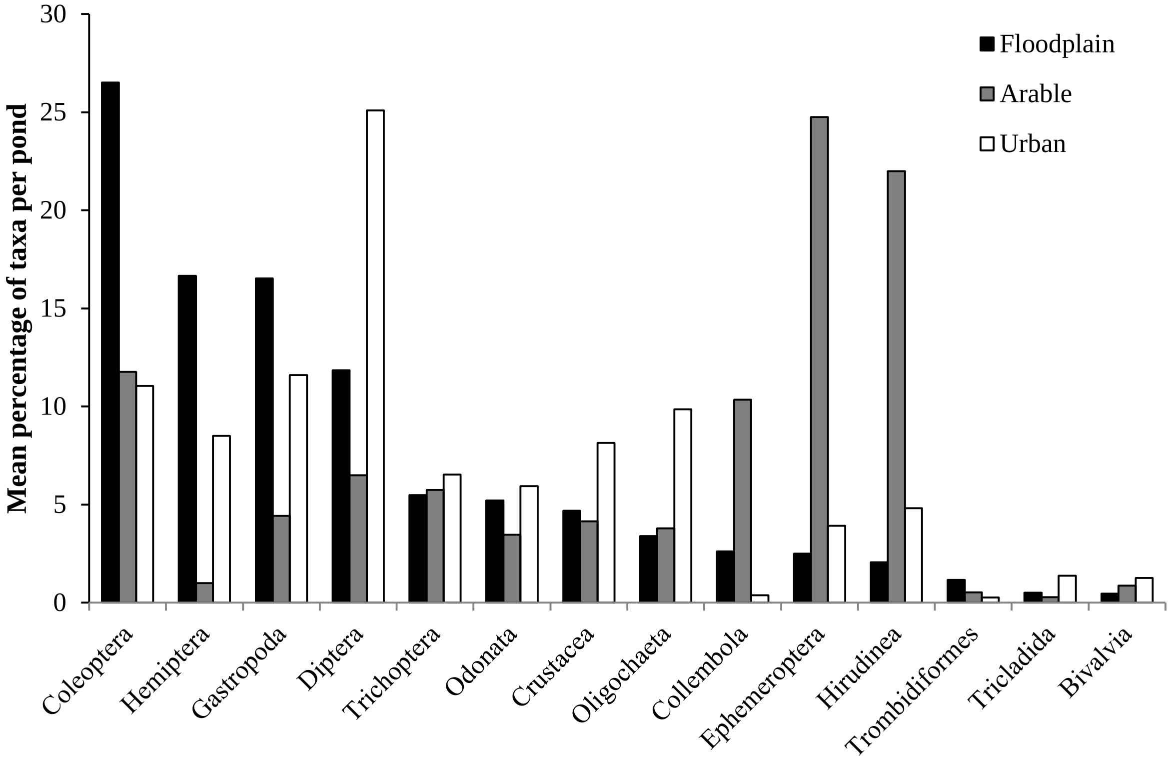

Figure 2 - Mean percentage of taxa per pond for selected macroinvertebrate groups in floodplain, arable and urban ponds.

Figure 3

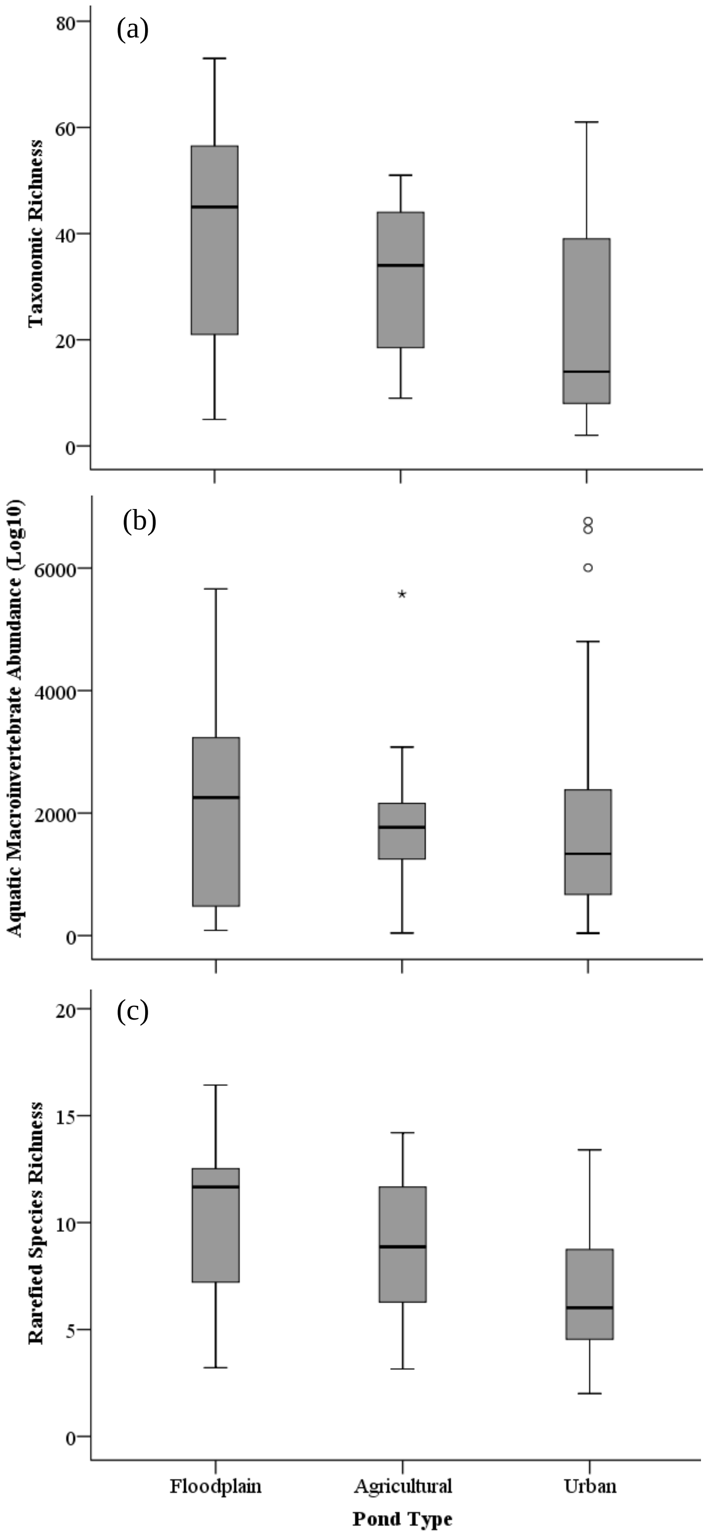

Figure 3 - Abundance (a), taxonomic richness (b) and rarefied species richness (c) of macroinvertebrates recorded from floodplain, arable and urban ponds. Open circle = outlier defined on the basis of being greater than 1.5 times the interquartile range, open square = outlier defined on the basis of being greater than 3 times the interquartile range.

Figure 4

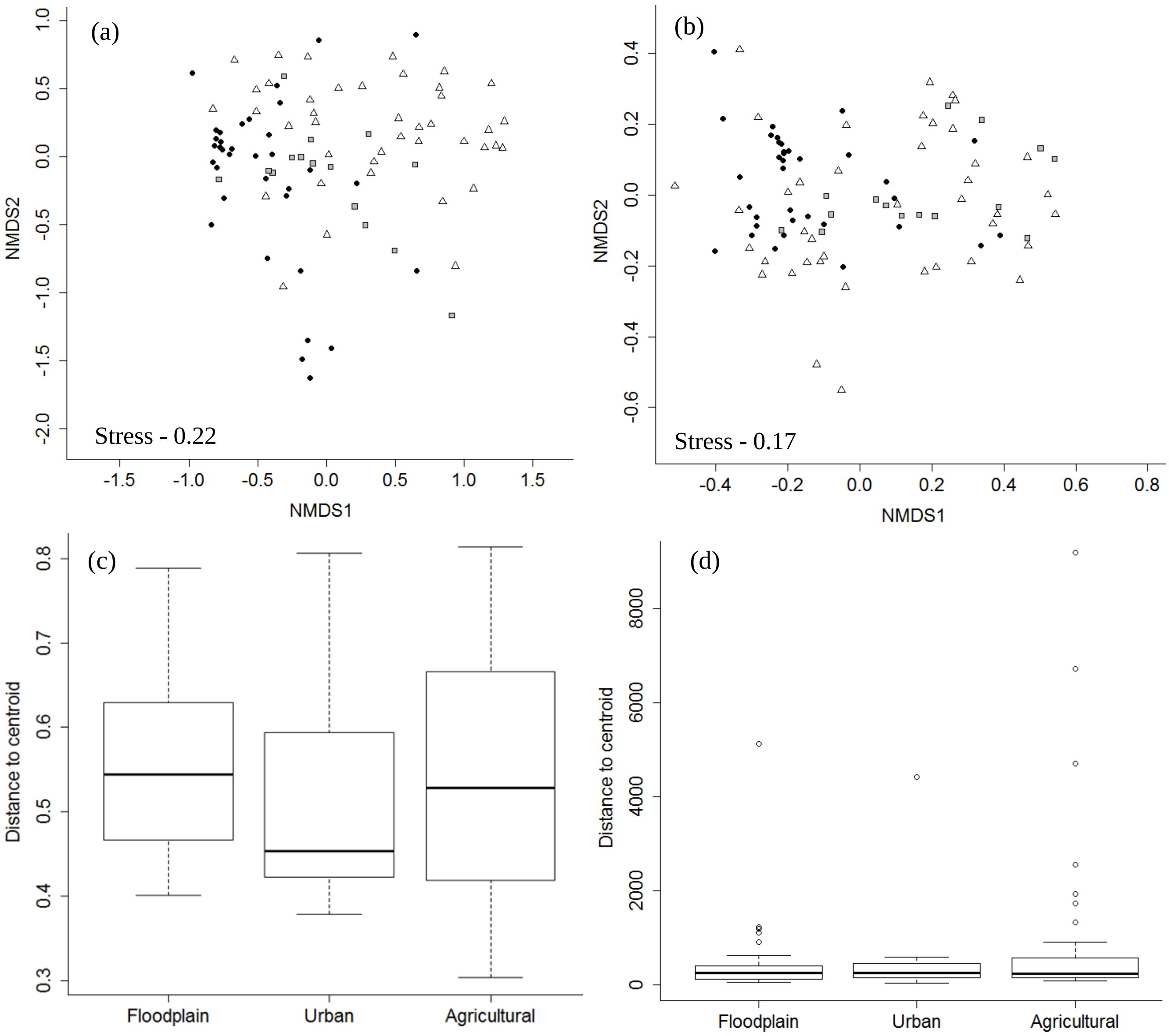

Figure 4 - Non-Metric Multidimensional scaling plots of variation in (a) macroinvertebrate communities and (b) environmental characteristics (black symbols - urban ponds, grey squares - arable ponds and open triangles - floodplain ponds) and boxplots of multivariate dispersion distances for (c) macroinvertebrate communities and (d) environmental conditions from the three pond types.

Figure 5

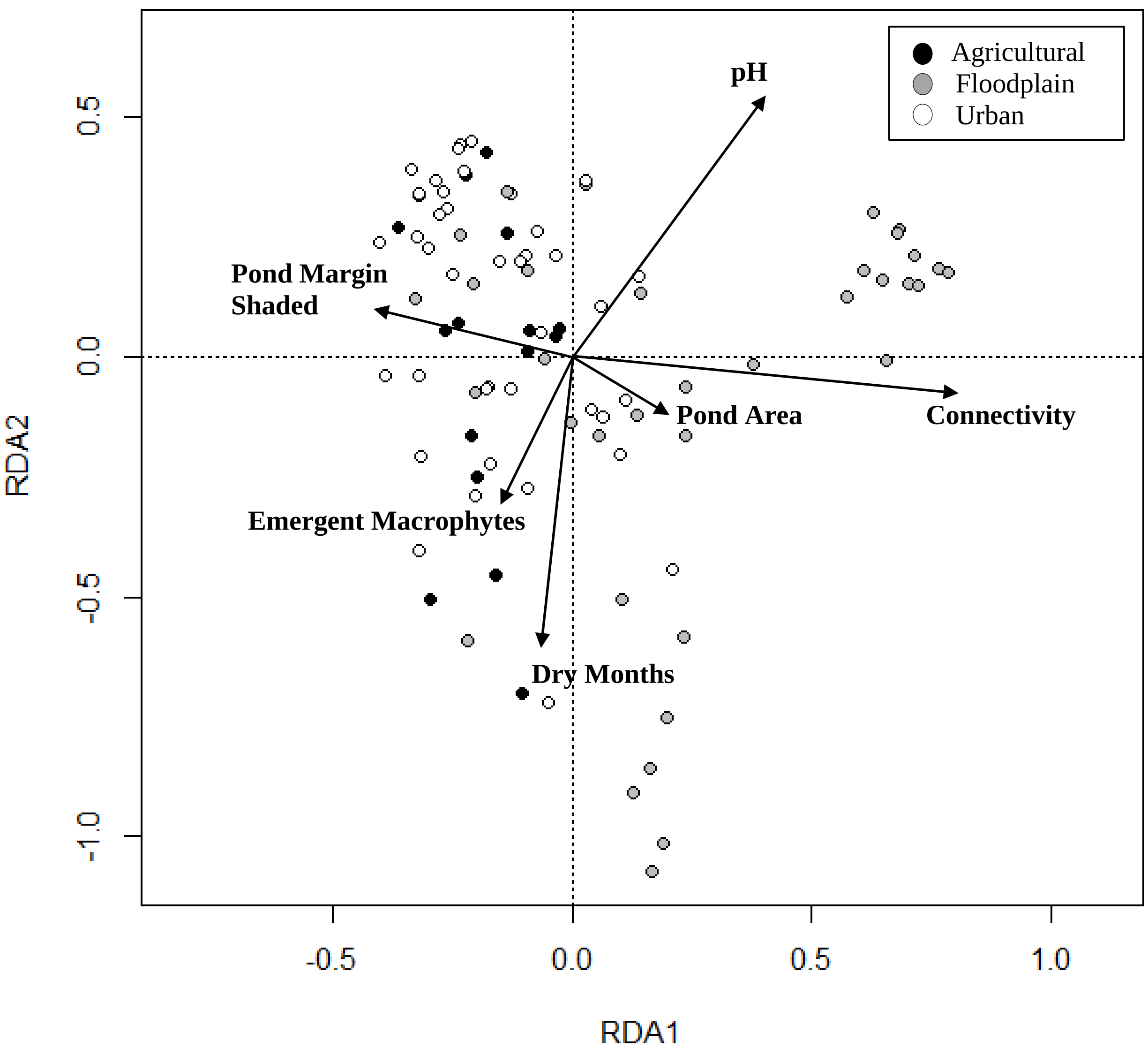

Figure 5 - RDA ordination of site plots for floodplain, agricultural and urban pond macroinvertebrate communities. Only significant environmental parameters are presented.

Figure 6

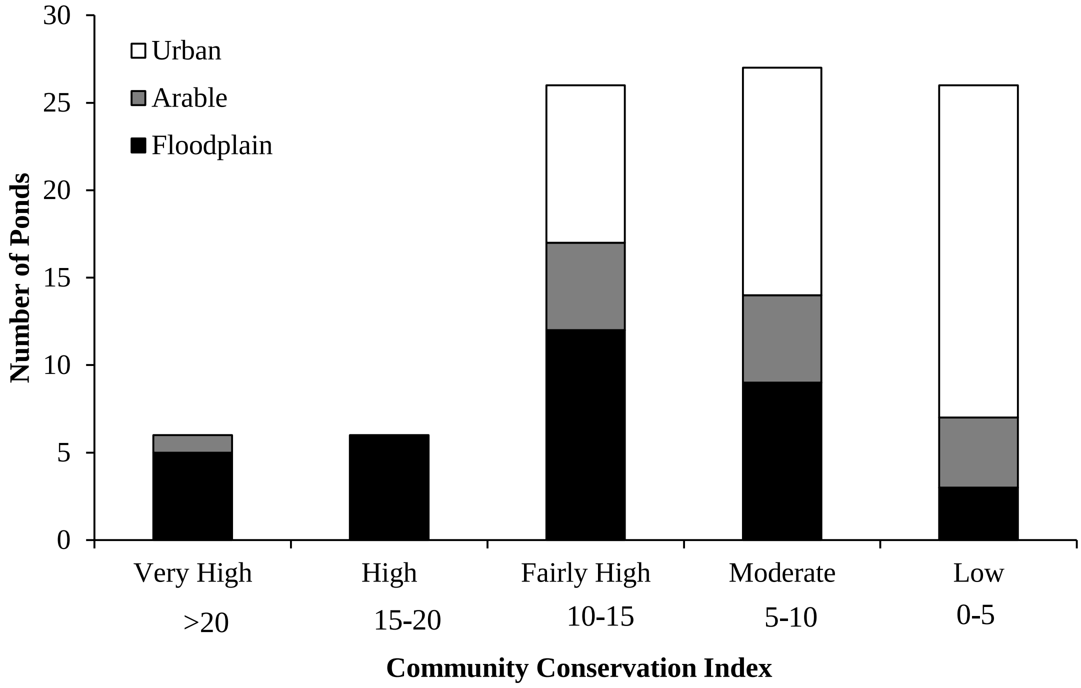

Figure 6 - The number of ponds determined as very high, high, fairly high, moderate and low conservation value based on the Community Conservation Index (Chadd and Extence, 2004).