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**The macroinvertebrate biodiversity and conservation value of garden and field ponds along a rural - urban gradient.**

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# **The macroinvertebrate biodiversity and conservation value of garden and field ponds along a rural - urban gradient**

**M. J. Hill and P. J. Wood<sup>1</sup>**

## **Abstract**

The biodiversity and conservation value of semi-natural and field ponds in rural locations are widely acknowledged to be high compared to other freshwater habitats. However, the wider value of urban ponds, and especially garden ponds, has been largely neglected in comparison. This study examines the biodiversity and conservation value of aquatic macroinvertebrates in ponds along an urban–rural continuum over three seasons. Macroinvertebrate faunal richness and diversity of garden ponds (in both urban and sub-urban locations) was markedly lower than that associated with field ponds. The fauna recorded in garden ponds were largely a subset of the taxa recorded in the wider landscape. A total of 146 taxa were recorded from the 26 ponds examined (135 taxa from field ponds and 44 taxa from garden ponds); although only 10 taxa were unique to garden ponds. Garden ponds were frequently managed (macrophytes removed or sediment dredged) and contained artificial fountains or flowing water features which allowed a number of flowing water (lotic) taxa to colonise and persist. Despite the relatively limited faunal diversity and reduced conservation value of garden ponds they have the potential to serve as refugia for some taxa, especially Odonata with highly mobile adults. At the landscape scale, garden ponds potentially provide a diverse and abundant range of freshwater habitats that could play an important role in conserving macroinvertebrate biodiversity. However, for this to be achieved there is a need to provide guidance to home-owners on how this potentially valuable resource can help support freshwater biodiversity.

**Keywords:** garden pond, urban ponds, invertebrates, ornamental, taxa richness, conservation value.

## INTRODUCTION

The potential pressures of increasing urbanisation on landscape biodiversity have been widely acknowledged (e.g. Dudgeon et al., 2006; Gopal, 2013; Mckinney, 2002). Urban developments have resulted in significant modification to the character of the landscape in many regions and major changes to the structure of biotic communities (Davies et al., 2009; Vermonden et al., 2009). These changes reflect habitat fragmentation (Goddard et al., 2010), reduced species richness (Mckinney, 2008), biotic homogenization (Mckinney, 2006; Grimm et al., 2008) and increased opportunities for non-native/invasive species (Niinemets and Penuelas, 2008) reported in many urban ecosystems. On-going urban developments across the globe have resulted in significant pressures on anthropogenic and urban freshwater ecosystems (Gledhill et al., 2008; Chester and Robson, 2013) associated with habitat loss and a reduction in its quality (Goulder 2008; Oertli et al., 2009; Vermonden et al., 2009 Williams et al., 2010).

Ponds are widely recognised as supporting greater regional (gamma) invertebrate diversity than most other freshwater ecosystems in the UK and across Europe (Davies et al., 2008; Williams et al., 2003). However, in many areas of the globe, pond numbers have declined significantly over the last 150 years due to land clearance, drainage and urban development (Hull, 1997; Wood et al., 2003; Cèrèghino et al., 2008; Oertli et al., 2009). Despite this trend of decline, the total number of ponds recorded in England increased between 1998 and 2007 (at a rate of 1.4% per annum), although many sites were reported to be in a poor condition (Williams et al., 2010). Most studies on the ecology and management of pond habitats have centred on rural and semi-natural water bodies in lowland (e.g., Sayer et al., 2012) and to a lesser extent upland settings (e.g., Oertli et al., 2008). In marked contrast, those located in the urban landscape (e.g., municipal parks, schools, gardens and urban conservation areas) have been significantly under-represented, and the wider value of urban ponds as potential biodiversity refuges has not been fully addressed (Gledhill *et al.*, 2008, Chester and Robson, 2013). In addition, there has been limited research and conservation of urban ponds compared to those located in rural and semi-natural areas (Langton et al., 1995; Wong and Young, 1997; Gledhill et al., 2005; Tanner and Gange, 2005; Gledhill et al., 2012; Colding et al., 2009).

Between 2000 and 2010 the urban landscape in the United Kingdom increased in area by 141,000 hectares (Khan, 2013) and over 60% of the population now resides in urban regions (Pateman, 2011). In 2008, there were 22.2 million dwellings in England, 85% of which included a private plot of land and 78% had gardens at both the front and back of the property (Department for Communities and Local Government, 2010). Even in highly urbanised cities, between 22-27% of the total urban area may comprise domestic gardens (Department for Communities and Local Government, 2010). The UK government has recognised the potential value of urban habitats and has encouraged wildlife gardening to potentially address some of the widely perceived negative effects of urbanization (Davies *et al.*, 2009). It has been estimated that between 2.5 and 3.5 million garden ponds exist in the UK covering 349 hectares (Davies *et al.*, 2009). The density and connectivity of ponds is a major determinant of floral and faunal diversity in urban locations (Gledhill *et al.*, 2008), although high density developments may act as physical barriers to flora and fauna dispersal and migration (Boothby, 1998).

Growing urban sprawl into the wider landscape and an increasing density of developments (Dallimer *et al.*, 2011) illustrates the potential importance garden ponds may play in supporting urban biodiversity. In addition, while many pre-existing urban ponds have been lost as a result of re-development (Wood *et al.*, 2003; Gledhill *et al.*, 2008; 2012), new urban and garden ponds have been created. Given the popularity of anthropogenic water features (Titchmarsh, 2013), garden ponds have the potential to offset, or at least mitigate, some of this decline in biodiversity recorded in urban and suburban areas (Chester and Robson, 2013; Gledhill *et al.*, 2012).

Research addressing the biodiversity and conservation value of private gardens has thus far largely focused on terrestrial flora and fauna (Davies *et al.*, 2009; Loram *et al.*, 2007; Cannon *et al.*, 2005; Chamberlain *et al.*, 2004; Goddard *et al.*, 2010; Smith *et al.*, 2006; Gaston *et al.*, 2005; Loram *et al.*, 2008). Only a relatively small proportion of this research has focused on freshwater bodies, with the majority centred on amphibian conservation (Latham *et al.*, 1994; Bebee, 1979; Parris, 2006; Hamer and McDonnell, 2008; Hamer and Parris, 2011). To date there has been limited research on the aquatic invertebrates inhabiting garden ponds (Gaston *et al.*, 2005; Monckay and Shine, 2003), in part due to the difficulties of gaining access from

householders (Wood et al., 2003). As result there is a pressing need to examine the aquatic invertebrate biodiversity and conservation value of garden ponds (in both urban and sub-urban locations) compared to semi-natural field ponds in rural locations.

This paper examines the biodiversity and conservation value of aquatic macroinvertebrates in ponds along an urban–rural continuum. We hypothesised that: i) aquatic macroinvertebrate biodiversity within garden ponds (in both urban and sub-urban locations) would be lower than that recorded in field ponds; ii) the conservation value of field ponds would be greater than that of garden ponds; iii) the fauna recorded in garden ponds would be a subset of the taxa recorded in the wider pond landscape; and iv) garden ponds serve as a refuge / reservoir for taxa within the wider pondscape.

## **METHODS**

### **Sampling programme**

A total of 26 ponds (13 garden ponds and 13 field ponds) were selected for study along an urban–rural continuum surrounding the town of Loughborough (Leicestershire, UK) (Figure 1). The garden ponds comprised 10 within the urban centre of Loughborough and 3 within suburban villages in the surrounding area. The field ponds sampled comprised 6 within agricultural fields or pasture and 7 in nature conservation areas; although livestock grazing also occurred at most sites. Each pond was sampled on three occasions during 2012 corresponding to spring, summer and autumn seasons (high, intermediate and low water levels respectively). At each pond, conductivity ( $\mu\text{S cm}^{-1}$ ), pH and water temperature were recorded using a Hanna conductivity meter (HI198311) and a Hanna pH meter (HI98127). Dissolved oxygen ( $\text{DO mg l}^{-1}$ ) was recorded at each pond site using a Mettler Toldedo Dissolved Oxygen Meter (SG6). Mean water depth (cm), water surface area (m), the percentage of the pond surface shaded by overhanging vegetation and the composition of the substratum (percentage gravel, sand and silt based on visual examination) was recorded.

Macroinvertebrate samples were collected using a standard pond net (mesh size, 250 $\mu$ m) employing an equal intensity methodology (Friday, 1987) with the total time used to sample each pond being proportional to its surface area. A maximum of three minutes was used to sample the largest ponds (Biggs et al., 1998) where the area was greater than 50m<sup>2</sup>; for smaller ponds 30 second sampling for every 10m<sup>2</sup> of surface area was used. At each site habitat characteristics and distribution were recorded and the available aquatic habitat assigned to one of the following three groups: i) open water, ii) emergent vegetation and iii) submerged vegetation. The total sampling time at each pond was divided equally between the microhabitats present. If the pond was dominated by a particular habitat the time was further divided to represent this (Biggs et al., 1998). Samples from each habitat were preserved in the field and stored separately.

In the laboratory, invertebrate samples from each habitat were processed separately and stored in 70% industrial methylated spirits prior to identification. The majority of insect fauna were identified to species or genus level with the exception of Diptera which were identified to family level. Other non-insect faunal groups were recorded at order or family level: Planariidae, Hydrachnidiae, Oligochaeta and Collembola. Meiofaunal groups (including Cladocera and Ostracoda) were recorded and counted but have been excluded from the statistical analysis presented.

### **Statistical Analyses**

To compare the faunal diversity among the ponds sampled, one-way analysis of variance (ANOVA) was used to examine three specific aspects of macroinvertebrate biodiversity: i) pond type (garden versus field), ii) seasonal differences (spring, summer and autumn) and iii) microhabitat variability (differences associated with open water, emergent vegetation and submerged vegetation habitats). All univariate analyses were performed in IBM SPSS Statistics (version 21, IBM Corporation, New York). Species richness and diversity indices were characterised using the Shannon Wiener diversity index and the Berger Parker dominance Index and were calculated using the Species Diversity and Richness IV program (Pisces Conservation Ltd., 2008). Alpha diversity was calculated as the invertebrate biodiversity at individual sites, beta diversity was measured using Jaccard's coefficient of similarity ( $C_j$ ) using

the Community Analysis Package 3.0 program (Pisces Conservation Ltd., 2004) and gamma ( $\gamma$ ) diversity was calculated as the total number of species in field and garden ponds. Species rarity was assessed using the species rarity index (SRI) following the methodology outlined by Williams et al., (2003). A conservation value was calculated for each pond based on the SRI and the season during which the greatest number of taxa were recorded using the classification outlined in Biggs et al. (2000).

The combined faunal and environmental datasets were ordinated using Canonical Correspondence Analysis (CCA) in the program CANOCO, Version 4.5 (ter Braak and Smilauer, 1998). To account for seasonal variability in community composition data from individual sites were combined for all sampling periods and mean values of environmental parameters determined. Abundance data were  $\log(x+1)$  transformed prior to analysis to reduce the influence of skewness and dominant taxa. The statistical significance of the environmental variables and the canonical axes were determined using the 'forward selection' procedure ( $P \leq 0.05$  after Bonferonni correction) based on a random Monte Carlo permutations test (999 random permutations). Only those environmental parameters identified as significantly influencing the faunal distribution were included in the final analysis.

## **Results**

### *Physiochemical Data*

The physical and chemical characteristics of the urban and field ponds were highly variable. There was no significant difference recorded between the pH, or dissolved oxygen concentration ( $\text{mg l}^{-1}$ ) among garden and field ponds (all ANOVA  $p > 0.05$ ); although field ponds typically displayed a greater range than garden ponds. Conductivity ( $\mu\text{S cm}^{-1}$ ) was significantly higher (ANOVA:  $F_{1, 25} = 5.04$ ;  $p = < 0.05$ ) in garden pond than field ponds when all sampling dates were considered. Pond size was significantly greater for field ponds than for garden ponds (ANOVA: area,  $F_{1, 25} = 40.83$ ;  $p = < 0.001$ ). Mean water depth did not differ between garden and field ponds although there was greater variability of depth for field ponds (water depth range = 0.09 - 1.50 m) than for garden ponds (range = 0.14 - 0.70 m).

## *Faunal Biodiversity*

A total of 146 taxa representing 19 orders and 52 families were recorded from the field (133 taxa: range 19-62) and garden ponds (44 taxa: range 2-23). There were 104 macroinvertebrate taxa which were only recorded in the field ponds but just 10 taxa were unique to garden ponds. Five of the taxa were unique to garden ponds; caddisfly larvae (*Hydropsyche angustipennis* (Trichoptera: Hydropsychidae), *Limnephilus lunatus* (Trichoptera: Limnephilidae), *Limnephilus rhombicus* (Trichoptera: Limnephilidae), *Beraea pullata* (Trichoptera: Beraeidae) and *Mystacides longicornis* (Trichoptera: Leptoceridae)) more commonly associated with lotic systems. The remaining taxa comprised three caddisfly larvae (*Cyrnus trimaculatus* (Trichoptera: Polycentropodidae), *Holocentropus picicornis* (Trichoptera: Polycentropodidae) and *Ceraclea fulva* (Trichoptera: Leptoceridae)) and two molluscs (*Gyraulus albus* (Mollusca: Planorbidae) and *Acroloxus lacustris* (Mollusca: Acroloxidae)) that are widely distributed in lentic waterbodies. Field ponds were characterised by higher species diversity of Coleoptera (49 taxa), Hemiptera (23 taxa), Trichoptera (12 taxa) and Odonata (Anisoptera - 5 taxa and Zygoptera - 7 taxa). The most widely distributed taxa, occurring in both garden and field ponds were the damselfly larvae, *Ischnura elegans* (Zygoptera: Coenagrionidae), mayfly larvae, *Cloeon dipterum* (Ephemeroptera: Baetidae), two crustaceans, *Asellus aquaticus* (Isopoda: Asellidae) and *Crangonyx pseudogracilis* (Amphipoda: Crangonyctidae) and two Diptera, Chironomidae and Culicidae. Five Coleoptera species with conservation designations (nationally scarce: occurring in 16-100 10km grids in the UK, or nationally notable b: occurring in 31-100 10 km grids in the UK) were recorded from field ponds; *Ilybius subaeneus* (Coleoptera: Dytiscidae), *Agabus conspersus* (Coleoptera: Dytiscidae), *Rhantus frontalis* (Coleoptera: Dytiscidae), *Helophorus dorsalis* (Coleoptera: Hydrophilidae) and *Helophorus strigifrons* (Coleoptera: Hydrophilidae).

The most taxa-rich garden pond (23 taxa) was only marginally richer than the poorest field pond (21 taxa). The richest field pond was located in a conservation area and contained 64 taxa; 3 other field ponds supported 50 taxa or more (Table 1). Several of the garden ponds were impoverished with only two taxa being recorded at the most species poor site and < 10 taxa were recorded from 8 ponds; there were no statistical differences in taxa richness between garden ponds within the urban and

suburban areas. The mean number of macroinvertebrate taxa recorded in field ponds (42 taxa) was over four times greater than that recorded in garden ponds (9 taxa). Macroinvertebrate taxa richness (ANOVA  $F_{1, 25} = 28.053$ ;  $P < 0.01$ ) and abundance (ANOVA  $F_{1, 25} = 17.705$   $P < 0.001$ ) were significantly higher in the field ponds than garden ponds.

The Species Rarity Index (SRI) did not differ significantly between garden ponds (SRI = 1.01) and field ponds (SRI = 1.07) and based on the raw scores the two pond types had moderate SRI scores and contained species largely considered to be common or 'local' species either: (i) confined to limited geographical areas, or (ii) of widespread distribution but relatively low population levels. However, when the community assemblage over a single season was also considered the overall conservation value of field ponds ranged between moderate (11-30 species) to high (31-50 species) while most garden ponds had low (0-10 species) or moderate (11-30 species) conservation value (Biggs et al., 2000). The Jaccard's coefficients of similarity for the 13 garden pond sites ( $C_j = 0.27$ ) and field ponds ( $C_j = 0.29$ ) were similar. However, when all sites were considered the Jaccard's coefficients of similarity was lower ( $C_j = 0.20$ ) indicating that there was a reduced similarity when all ponds were considered.

The Shannon Wiener diversity index of garden ponds was significantly lower than that of field ponds (ANOVA  $F_{1, 25} = 37.946$ ;  $P < 0.01$ ). The Berger Parker dominance index was significantly higher in garden ponds than in field ponds (ANOVA  $F_{1, 25} = 7.231$   $P < 0.01$ ) When garden ponds in the urban centre were compared with those from suburban areas there was no significant difference (Figure 2). Similar results were recorded for field ponds located on agricultural land and nature conservation areas (Figure 2).

When individual seasons were examined, the Shannon Wiener diversity index increased from spring to summer and from summer to autumn within garden ponds (Figure 3a). However, in field ponds Shannon Wiener diversity was similar in the spring and summer, but increased during the autumn season (Figure 3a). The Berger Parker dominance index displayed an inverse pattern, with dominance declining from spring to summer and from summer to autumn in garden ponds. Dominance was comparable during spring and summer within field ponds but was

lowest during the autumn (Figure 3b). Examination of the individual pond microhabitats indicated that the greatest Shannon Wiener diversity was recorded within vegetation (submerged and emergent) in both field and garden ponds (Figure 4a). The Berger Parker Dominance scores were highest in open water habitats but similar scores were recorded for submerged and emergent macrophytes in both garden and field ponds (Figure 4b).

### *Community Ordination*

Canonical Correspondence Analysis (CCA) indicated that there was a relatively clear separation of the field and garden ponds on the first axis. The first canonical axis explained 11.3 % of the variance in the invertebrate community data and 39.8 % of the taxa environment relationship. The second axis accounted for 7.7 % of the faunal variation and 27.3 % of the taxa environment relationship. Forward selection identified 4 environmental variables significantly correlated with the first two canonical axes: pond margin shaded by overhanging vegetation, area of floating vegetation habitat, water electrical conductivity ( $\mu\text{S cm}^{-1}$ ) (all p values <0.005) and water depth, (p <0.05) (Figure 5). When the distribution of the individual ponds was examined the garden ponds formed a relatively distinct group towards the positive end of Axis 1 and were associated with a greater volume of floating vegetation and shading from overhanging vegetation (Figure 5). The field ponds formed a more dispersed cluster due to greater variability in water depth and conductivity although there was some overlap with the garden pond cluster associated with aquatic macrophytes and overhanging vegetation (Figure 5).

## **Discussion**

Regional floral and faunal biodiversity associated with ponds has typically been reported to be greater than that of other freshwater bodies in the UK (Williams et al., 2003). However, most studies have centred on rural locations and those examining the biodiversity and conservation value of urban ponds, and especially garden ponds, have been limited to date. We hypothesised that the biodiversity of garden ponds would be lower than that of field ponds and found strong evidence to accept this hypothesis along the urban-rural continuum around Loughborough (Leicestershire),

UK. There were significantly more taxa and higher diversity indices recorded from field ponds than for the garden ponds examined in this study. The location of garden ponds in the urban centre or suburbs had no discernable effect on macroinvertebrate biodiversity, with low biodiversity characterising garden ponds anywhere along the urban-rural continuum.

The greatest garden pond macroinvertebrate taxa richness recorded was comparable to that of the least taxa rich field pond. These results are similar to those reported for Sheffield (UK) where the biodiversity of the aquatic invertebrate community was limited and dominated by dipteran larvae (Gaston et al., 2005). However, urban waterbodies in the Netherlands were shown to support comparable biodiversity and had a similar conservation value to rural canals and ditches (Vermonden et al., 2009). In addition, the macroinvertebrate biodiversity of a range of urban ponds in Halton in northwest England (Gledhill et al., 2008) were markedly higher (119 taxa) than that recorded in the garden ponds in this study (44 taxa), although total taxa richness was comparable to that recorded across all ponds (146 taxa).

The reduced number of taxa and biodiversity recorded in garden ponds probably reflect their limited connectivity and increased distance to other aquatic habitats. Habitat connectivity has been shown to have a strong influence on landscape ( $\gamma$ -diversity) biodiversity (Biggs et al., 2005; Boothby et al., 1995) and the physical structure of the urban environment (including buildings, roads and extensive impermeable surfaces) may lead to the further fragmentation and isolation of pre-existing or newly created pond networks in urban locations. These anthropogenic structures may also limit the ability of less mobile taxa to colonise or disperse in urban locations, especially garden ponds that may be completely surrounded by artificial fences or walls. In contrast, the high biodiversity of field ponds has been well documented across a range of spatial and temporal scales (Williams et al., 2003, Davies et al., 2008) and reflects their greater connectivity, heterogeneous physiochemical and habitat characteristics compared to the garden ponds examined in this study.

Although the influence of fish density on invertebrate communities within garden and field ponds was not assessed, fish have been shown to reduce macroinvertebrate

richness (Wood et al., 2001; Giles et al., 1990). Given the high incidence of fish communities in garden ponds they may be an important control on macroinvertebrate community structure and diversity, especially if macrophyte cover is limited. Future research is required to untangle the impact of fish density and feeding habits on garden pond macroinvertebrate diversity.

The importance of ponds to nature conservation in the wider landscape has been recognised (Williams et al., 2003; Wood et al., 2003; Williams et al., 1998). It has been shown that 150 of the 280 wetland invertebrates listed in the red data book utilise ponds as habitats (Drake, 1995) and 23 of the 38 freshwater and brackish water organisms given protection under section 5 and 8 of the Wildlife and Countryside act 1981 are associated with or regularly use pond habitats (Wood *et al.*, 2003). In addition, 31 of the 42 freshwater invertebrate species, excluding Diptera, categorised as endangered in the red data book list are associated with ponds (Gee *et al.*, 1994). Given that the conservation value of field ponds was greater than that of garden ponds using both the Species Rarity Index and the classification based on taxa richness, we found strong evidence to accept our second hypothesis that field ponds would have a greater conservation value than garden ponds. The SRI was slightly higher for field ponds compared to garden ponds and indicated that the majority of taxa recorded in both were regarded as locally confined to a limited geographical area or widely distributed; but all the five of the Coleoptera species recorded of conservation interest (nationally scarce or nationally notable) were only recorded within field ponds. The overall conservation value of garden and field ponds differed markedly due to significant differences in the number to taxa recorded among the ponds. The conservation value of most garden ponds was low (10 ponds) and only 3 ponds had moderate conservation value. In contrast, 7 of the field ponds had moderate conservation value (supporting 11-30 species) and 6 ponds had high conservation value supporting between 31-50 species during a single season (Biggs et al., 2000). These results provide further evidence of the wider biodiversity and conservation value of field ponds, many of which support taxa with specific conservation designations (Boothby, 1997; Sayer et al., 2012; Williams et al., 2003).

The majority of the taxa recorded in garden ponds were also recorded in field ponds in this study (39 taxa) providing evidence in support of our third hypothesis: that taxa recorded in garden ponds would be a subset of the taxa recorded in the wider pond

landscape. Only 10 taxa were unique to the garden ponds in this study and five of these were trichopteran larvae more typically associated with lotic environments (Edington and Hildrew, 1995; Wallace et al., 2003). Many of the garden ponds contained artificial flowing water features (fountains or re-circulating water) that were designed to be aesthetically pleasing, facilitate oxygenation of the water and/or to prevent algae/floating vegetation from covering the pond surface. These artificial water features powered by electrical pumps created a lotic environment in inflowing areas, which provided habitat for lotic trichopteran taxa.

Although garden ponds displayed reduced biodiversity and conservation value compared to field ponds we did find evidence in support of our fourth hypothesis: garden ponds serve as a refuge / reservoir of taxa in the wider landscape. Five Odonata taxa occurred extensively in both field and garden ponds. A key determinant of Odonata biodiversity within individual ponds is vegetation diversity with the surrounding landscape being less critical to this group due to high vagility (Goertzen and Suhling, 2013). *Ischnura elegans* was the most abundant damselfly within garden and field ponds. *I. elegans* was widely distributed and abundant in urban park ponds in Dortmund, Germany and appeared to thrive in locations that were frequently managed / disturbed (Goertzen and Suhling, 2013). It has also been shown to be tolerant to a wide range of water quality conditions typical of garden ponds (Somilini et al., 1997). However, for the majority of the other, less mobile, faunal groups there was limited evidence that garden ponds could serve as a refugium.

It has been estimated that 2.5-3.5 million garden ponds exist in UK (Davies et al., 2009) and that in the wider region of the study area, the city of Leicester may contain up to 8000 garden ponds (Latham et al., 1994). In comparison, the total number of lowland ponds nationally in the countryside was estimated to be round 478,000 in 2007 (Williams et al., 2010). Given the large number of garden ponds that exist, they could have an important role in sustaining aquatic biodiversity in the future. However, this may only be realistically achieved if appropriate measures are in place to enhance the connectivity of ponds in urban locations and home owners are provided with advice regarding the biodiversity and conservation potential of garden ponds. In many instances the development of new garden ponds may be the only available

option to compensate for the loss of ponds due to urban development (Gledhill et al., 2008).

The UK government scheme of 'wildlife gardening' appears to have had little impact on the type of garden pond created thus far. Only one of the garden ponds examined in this study was considered a 'wildlife' pond. The majority of garden ponds were constructed as ornamental features rather than for any biodiversity or conservation purpose. Habitat heterogeneity is frequently limited in garden ponds because vegetation and silt are frequently managed or removed (Davies et al., 2009). As a result garden ponds are typically kept at an early successional stage. Gaston et al., (2005) suggested that as a result of pond management activities in the garden, they were unlikely to replace the heterogeneity of ponds in the wider landscape and that they are never likely to support the biodiversity recorded in field ponds.

## **Conclusion**

The results of this research indicate that, for the ponds examined around the town of Loughborough (Leicestershire, UK), the biodiversity and conservation value of garden ponds was lower than that of field ponds. The fauna recorded in garden ponds were typically a subset of the taxa recorded in the wider landscape. There were a limited number of taxa (10) that were unique to garden ponds, but 5 of these (Trichoptera larvae) were more commonly associated with lotic environments and probably colonised the garden ponds due to the presence of artificial water-fountains or pump driven flowing water features. However, garden ponds may serve as temporary refugia for highly mobile taxa (e.g., Odonata, Coleoptera and some Hemiptera such as Corixidae) and for damselfly larvae (such as *Ischnura elegans* in this study) despite the relatively low faunal diversity recorded in urban ponds in general.

Garden ponds are common features in the urban and rural landscape and represent an abundant freshwater habitat that could play an important role in supporting macroinvertebrate biodiversity. Garden ponds may also play an important role in the conservation of floral and faunal communities (especially invertebrate and amphibians) at the landscape scale. However, if garden pond creation and

management is to be promoted as a means to enhance current biodiversity and conservation status it is important that home-owners / gardeners are provided with guidance regarding how this potentially valuable resource can help support freshwater biodiversity into the future.

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## List of Figures

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- Figure 5- Canonical Correspondence Analysis site plot for garden and field pond invertebrate and environmental data collected from sites around Loughborough, Leicestershire. Note – only significant environmental parameters (identified by Monte Carlo random permutations tests) influencing the invertebrate community are presented.

Figure 1

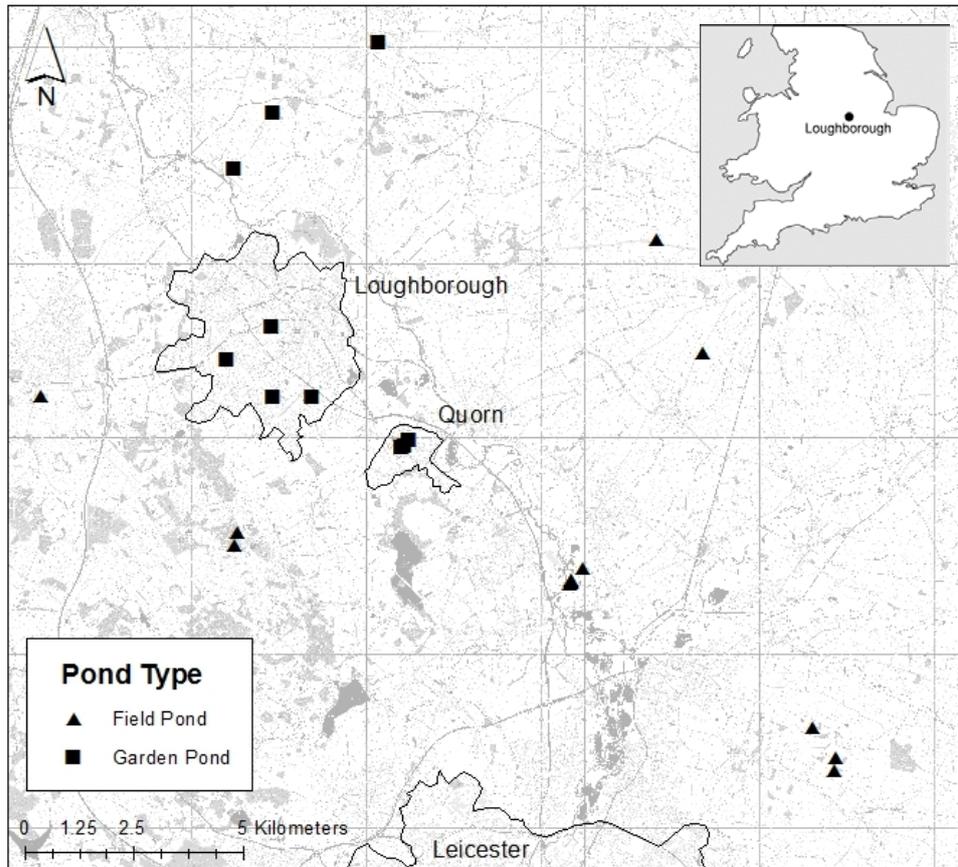
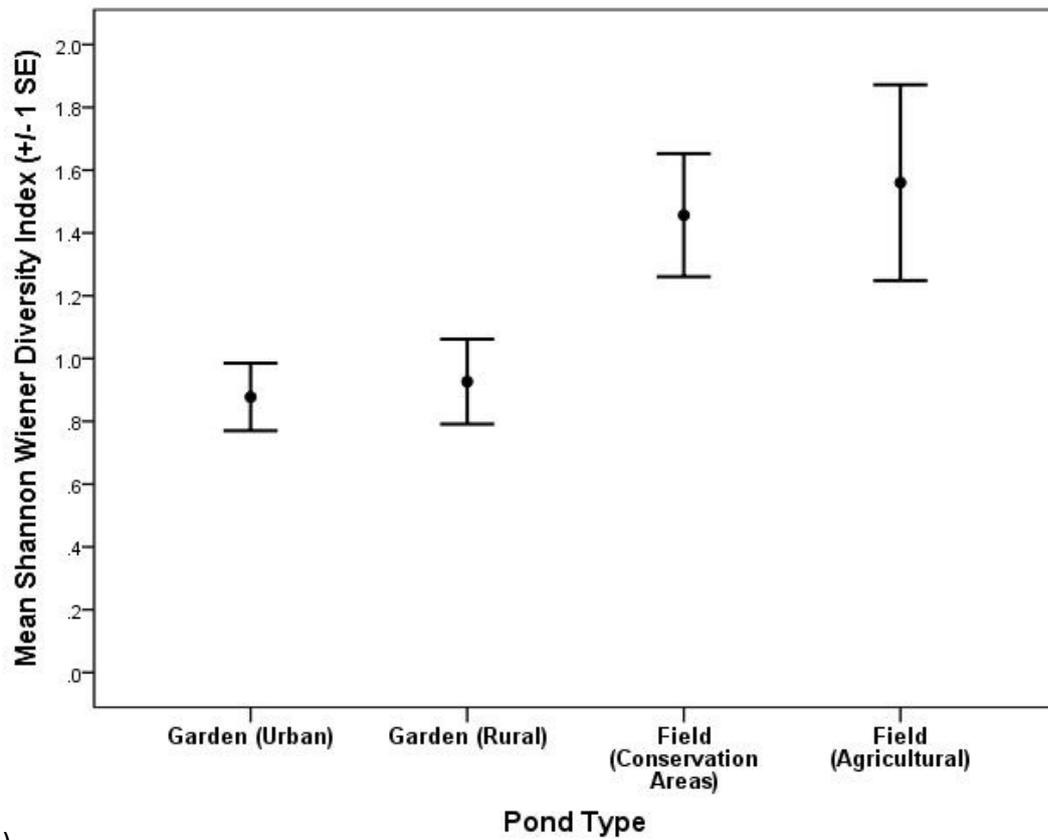


Figure 2

a)



b)

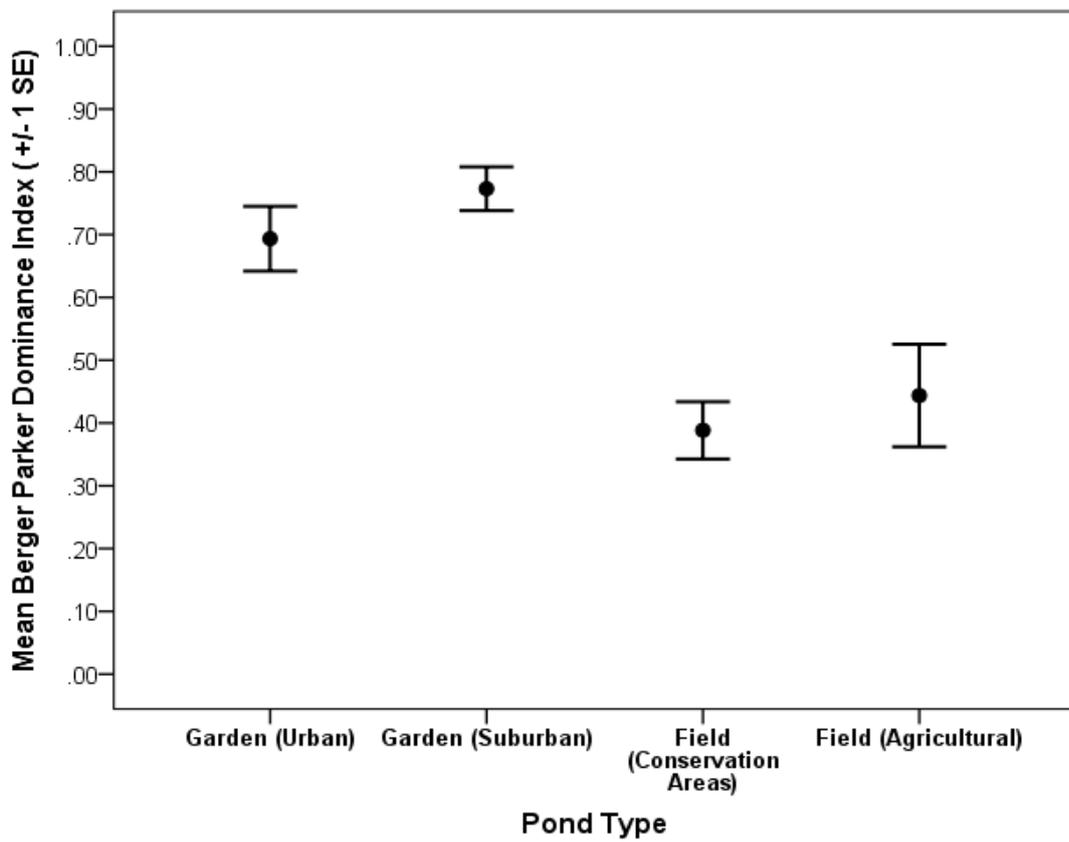
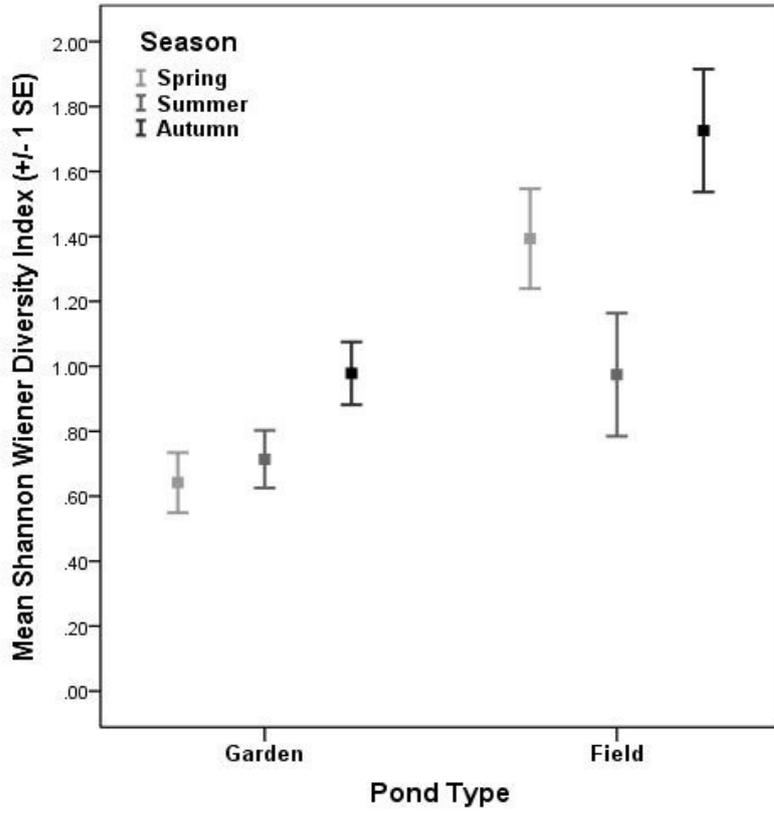


Figure 3

a)



b)

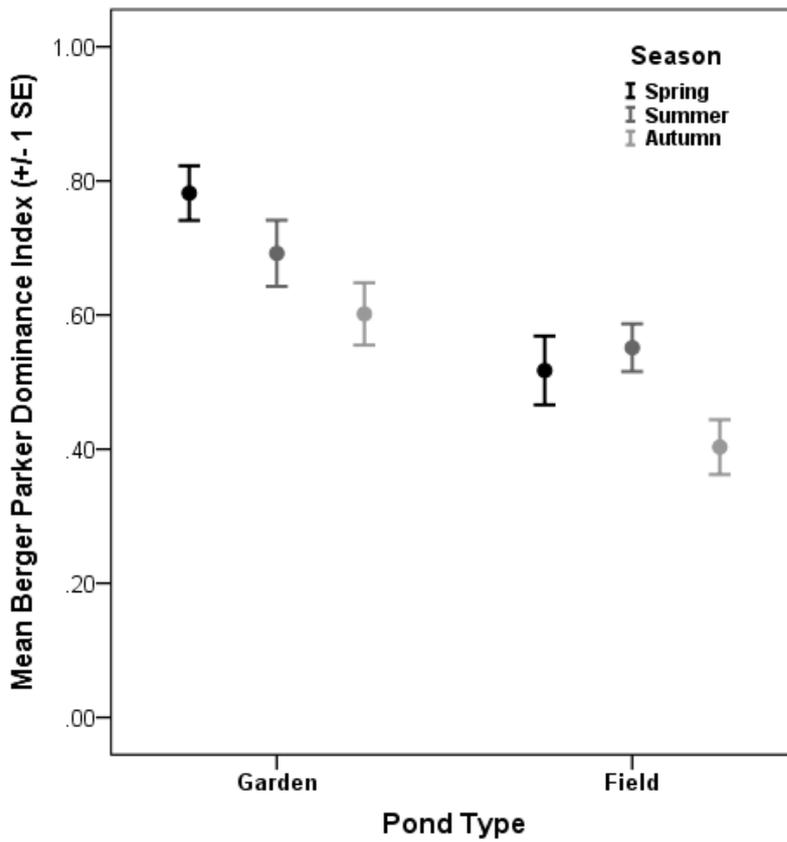
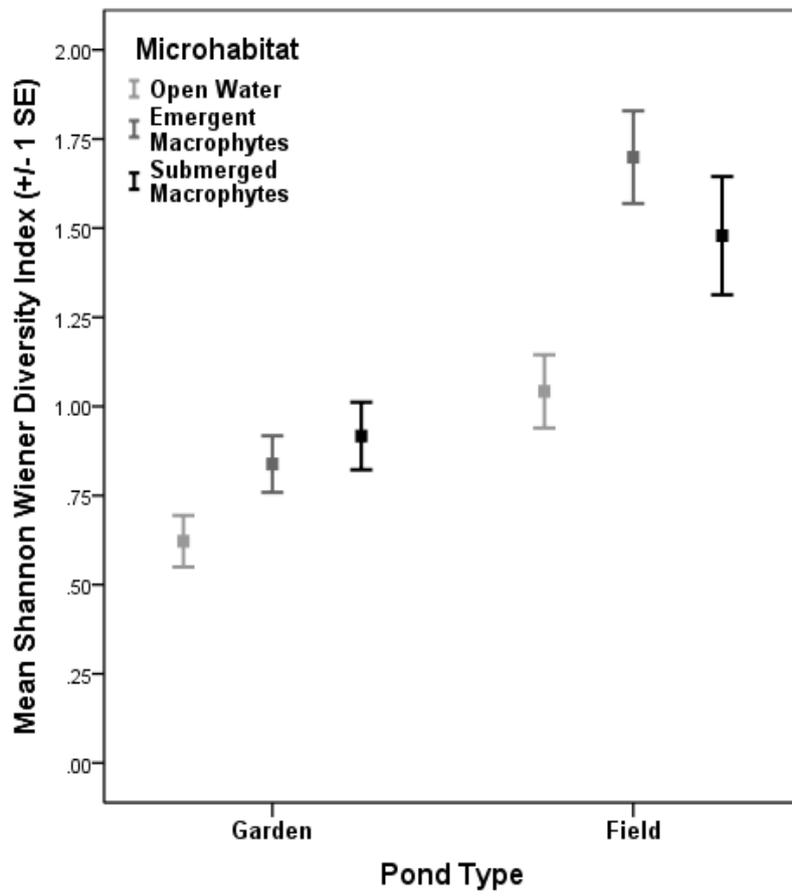


Figure 4

a)



b)

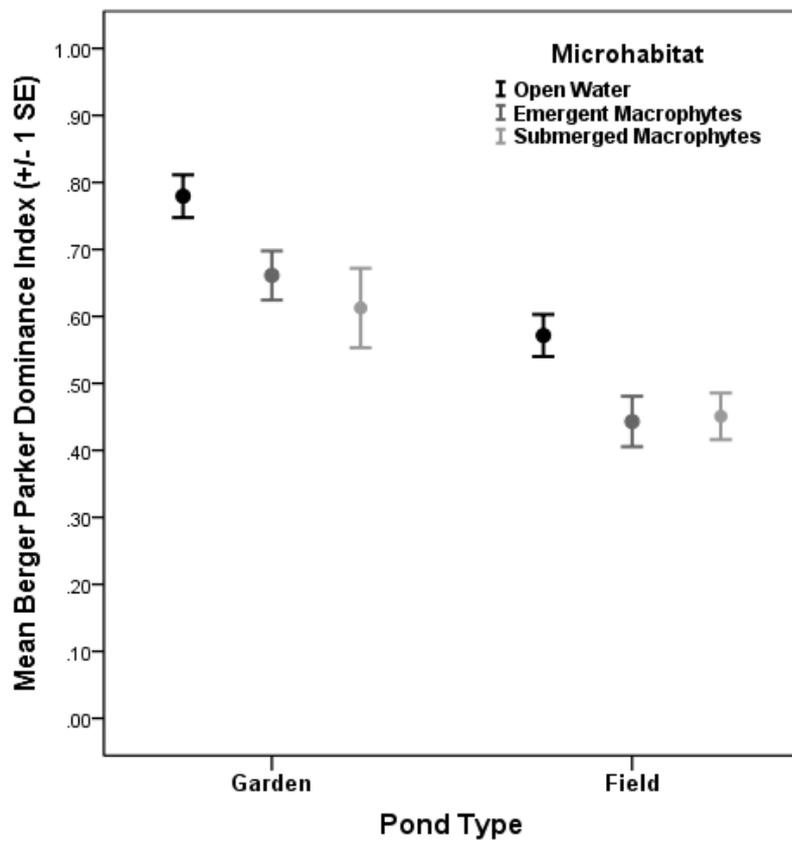


Figure 5

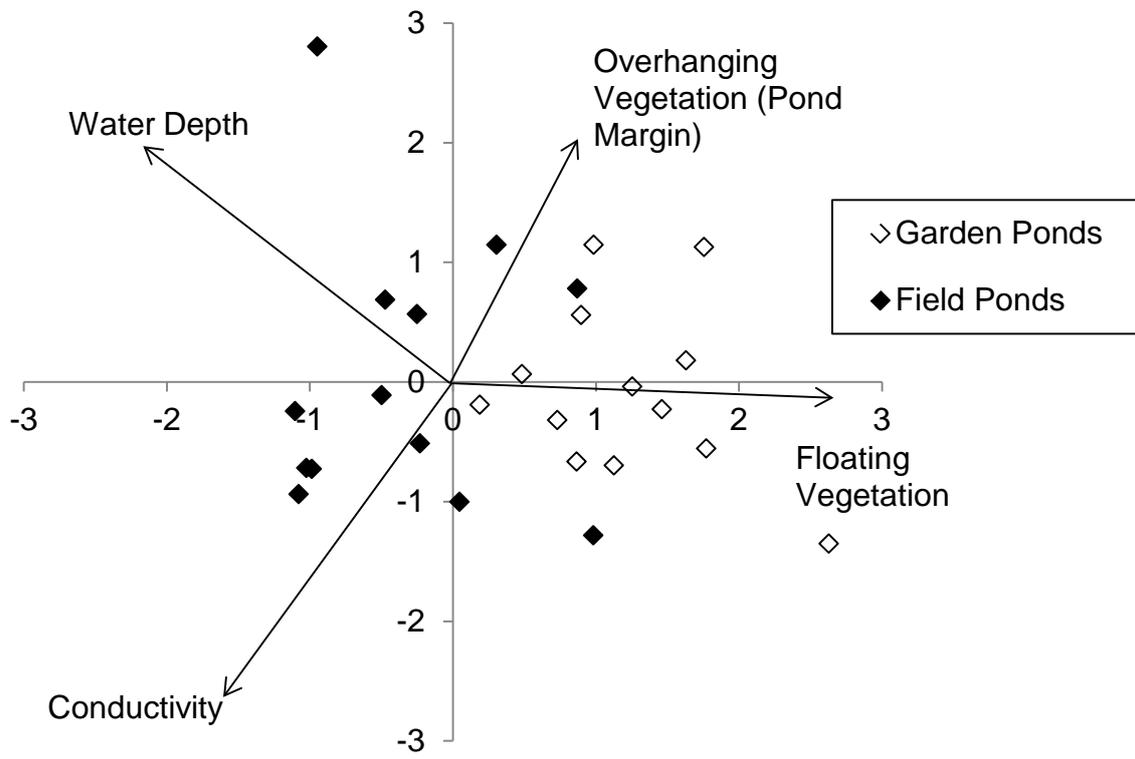


Table 1. Site and mean value of selected habitat (area, water depth and percentage water shaded by overhanging vegetation), invertebrate community measures (relative abundance, number of taxa, Shannon Wiener diversity index and Berger Parker dominance index) and conservation measures (Species Rarity Index (SRI) and overall conservation value based in the maximum number of taxa recorded in a single season) for the garden (GP) and field (FP) ponds examined in this study.

	Area (m <sup>2</sup> )	Depth (cm)	Shade (%)	Abundance	Taxa	Shannon Wiener Diversity Index	Berger Parker Dominance Index	SRI	Conservation Value
GP1	7.12	42.83	0	45	2	0.30	0.91	1	Low
GP2	4.51	60.00	0	728	12	0.84	0.78	1	Low
GP3	1.90	35.50	12	670	8	1.19	0.46	1	Low
GP4	2.70	56.33	0	1265	14	0.97	0.73	1.07	Moderate
GP5	3.94	17.23	0	1111	8	1.07	0.68	1	Low
GP6	4.85	35.47	0	210	12	1.06	0.74	1.08	Moderate
GP7	10.90	37.33	0	900	11	0.79	0.81	1	Low
GP8	3.37	16.50	0	112	3	0.39	0.88	1	Low
GP9	3.93	70.43	0	237	6	0.43	0.91	1	Low
GP10	0.80	27.33	33	1034	4	0.93	0.69	1	Low
GP11	2.94	39.33	50	977	6	1.21	0.47	1	Low
GP12	86.46	47.33	77	2379	24	1.37	0.48	1.04	Moderate
GP13	6.88	14.50	100	1550	5	0.95	0.64	1	Low
Mean	10.79 (+/-6.34)	38.47 (+/-4.77)	20.92 (+/-9.46)	862.92	8.85 (+/-1.64)	0.88 (+/-0.09)	0.70 (+/-0.44)	1.01(+/-0.02)	
FP1	171.00	166.67	23	2206	22	1.54	0.49	1.04	Moderate
FP2	38.02	37.33	68	2478	43	1.83	0.44	1.07	Moderate
FP3	108.07	81.00	0	2507	54	2.36	0.32	1.07	High
FP4	91.09	58.00	0	4125	62	2.92	0.20	1.08	High
FP5	97.61	59.00	0	3228	46	2.29	0.38	1.04	High
FP6	91.58	53.83	0	3758	47	2.15	0.37	1.13	High
FP7	107.90	83.00	0	3532	49	1.79	0.42	1.02	Moderate
FP8	102.78	35.00	5	633	27	1.59	0.59	1.04	Moderate
FP9	92.87	55.67	5	1974	29	1.26	0.67	1	Moderate
FP10	106.12	77.00	0	968	34	2.44	0.29	1.07	Moderate
FP11	146.93	77.33	7	2114	44	2.35	0.28	1	High
FP12	113.66	31.67	0	1003	51	2.75	0.24	1.21	High
FP13	131.75	9.33	93	2023	19	0.97	0.68	1.19	Moderate
Mean	107.64 (+/-8.73)	63.45 (+/- 10.55)	15.46 (+/-8.32)	9650.38	41.42 (+/-3.65)	1.51 (+/- 0.16)	0.57 (+/- 0.04)	1.07 (+/-0.01)	

