Water quality, biodiversity and ecosystem functioning in ponds across an urban land-use gradient in Birmingham, U.K.

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Abstract

The ecology of ponds is threatened by urbanisation and as cities expand pond habitats are disappearing at an alarming rate. Pond communities are structured by local (water quality, physical) and regional (land-use, connectivity) processes. Since *ca*1904 >80% of ponds in Birmingham, U.K., have been lost due to land-use intensification, resulting in an increasingly diffuse network. A survey of thirty urban ponds revealed high spatial and temporal variability in water quality, which frequently failed environmental standards. Most were eutrophic, although macrophyte-rich, well connected ponds supported macroinvertebrate assemblages of high conservation value. Statistically, local physical variables (e.g. shading) explained more variation, both in water quality and macroinvertebrate community composition than regional factors. Nonetheless, habitat availability within the wider landscape was important. Ecosystem functioning (leaf-litter breakdown) along a rural-urban gradient was confounded by habitat area, despite a decrease in functional redundancy.

Ponds are identified that promote network connectivity and management of land-use within 100m may buffer against diffuse pollution with reductions in riparian shading required to improve growth conditions for oxygenating vegetation and to reduce nutrient levels. The results indicate that many urban ponds are threatened habitats that require active management to protect and restore water quality, biodiversity and ecosystem functioning.

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> *Many drops make a bucket, many buckets make a pond, many ponds make a lake, and many lakes make an ocean*
>
> -Percy Ross

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Introduction

Chapter 1: Introduction

Urbanisation is a global anthropogenic phenomenon that dramatically alters the landscape (Grimm *et al.* 2008), generating impermeable surfaces and artificial structures (McDonnell and Hahs 2008) that result in habitat loss, fragmentation and disturbance (Wilby and Perry 2006). Over half of the world's population now lives in urban areas as does 79.6% of the United Kingdom's population; numbers that by 2050 are predicted to rise to 67.2% and to 85.9% respectively (UN 2012). Urbanisation results in air, water, noise and light pollution (Haughton and Hunter 1994) and an increased presence and persistence of non-native species (McKinney 2006). Land development during urban growth is typically long-lasting and intensifies over time (Scher and Thiery 2005; McKinney 2006), which has considerable implications for the ecology of a city (Dow 2000). Consequently, there is an urgent need to improve our understanding of the urbanisation process and its implications for biodiversity (McDonnell and Pickett 1990; McDonnell and Hahs 2008).

To date, much urban ecological research has centred on terrestrial rather than aquatic systems, examining the value of parks and public spaces for mammals (e.g. Croci *et al.* 2008), birds (e.g. Savard *et al.* 2000; Sandstrom *et al.* 2006b) and bees, with particular regard to ecosystem services (e.g. Bates *et al.* 2011; Pellissier *et al.* 2012). Frequently the result is localised increases (e.g. Angold *et al.* 2006; Bates *et al.* 2011; Kowarik 2011) but overall declines in diversity, which become more pronounced towards the urban core (McKinney 2002). However, the impact of moderate, suburban land-use is varied, with a number of researchers noting increases in diversity (Blair and Launer 1997; Leveau and Leveau 2005; Croci *et al.* 2008) whilst some indicate decreases (e.g. Sandstrom *et al.* 2006b; Hornung *et al.* 2007). Increases in diversity have been explained by the intermediate disturbance hypothesis (Cornell and Lawton 1992), whereby low levels of urbanisation increase habitat heterogeneity and food resources (Blair and Launer 1997). Moreover, the number of non-native species increases with urbanisation due to anthropogenic introductions, which can result in high local diversity (at the city-scale) but increasingly homogenous communities between cities (McKinney 2006).

Due to their position in the landscape, aquatic systems are more vulnerable to the impact of urban land-use than terrestrial systems as they can receive large loads of contaminants and pollution in run-off (Dudgeon *et al.* 2006) and may receive water from far beyond their wetted perimeter (Boothby and Hull 1997). Urban catchments are characterised by an increased coverage of impermeable surfaces such as roads, pavements and buildings (Walsh *et al.* 2001; Imberger *et al.* 2008; Brabec 2009), whilst the buffering capacity provided by vegetation and exposed soils is reduced (Hahs and McDonnell 2006). Consequently, catchment urbanisation creates flashier hydrographs, increases nutrient loads and other contaminants, and alters channel morphology, stability and the composition of flora and fauna (Paul and Meyer 2001; Roy *et al.* 2003; Allan 2004) i.e. the 'urban stream syndrome' (Walsh *et al.* 2003), which may be exacerbated by the infilling of other natural sink holes by development and/or natural processes (Williams *et al.* 1998b; Gledhill *et al.* 2008). Urban aquatic systems typically have lower abundance of native species, are dominated by tolerant taxa (Roy *et al.* 2003; Mancini *et al.* 2005) and non-native species are more prevalent (Ourso and Frenzel 2003). Whilst research on the impact of urbanisation on streams has advanced markedly over the past twenty years, research on ponds is lacking (but see Gledhill *et al.* 2008; Vermonden *et al.* 2009; Akasaka *et al.* 2010; Vermonden *et al.* 2010).

A pond is defined as a natural or artificial body of water that is between 1m^2 and 2ha in surface area that contains water for at least four months of the year (Biggs *et al.* 2005). They are ubiquitous landscape features (Downing *et al.* 2006), although their numbers have been in decline across the U.K. (Boothby 1999; Wood *et al.* 2003; Biggs *et al.* 2005), though recent evidence suggests a slight reversal in recent years (Williams *et al.* 2010; Jeffries 2012). There is global consensus that rural ponds can contribute a disproportionally large number of species to the regional pool relative to rivers, streams and lakes (Williams *et al.* 2004; Davies *et al.* 2008a; Della Bella *et al.* 2008; Markwell and Fellows 2008). This has been attributed to their relatively small catchments (Davies *et al.* 2008b), which lead to physico-chemical conditions that reflect local variations in land-use, habitat complexity, hydrology and climate (Biggs *et al.* 2005). Urban ponds may contribute significantly to regional biodiversity (Scher and Thiery 2005; Gledhill *et al.* 2008; Le Viol *et al.* 2009; Vermonden *et al.* 2009), but research is limited and these systems are threatened by contaminant inputs, including nutrients (Birch and McCaskie 1999; Stoianov *et al.* 2000; Peretyatko *et al.* 2009), heavy metals (Stolyar *et al.* 2008) and salt compounds (Marsalek 2003) (see Tixier *et al.* 2011 for a review). Furthermore, at the regional scale, habitat destruction leading to the loss of ponds (Wood *et al.* 2003; Huang *et al.* 2012; Jeffries 2012) may isolate populations in the remaining pond network, with largely unknown consequences for these increasingly disconnected communities. Although the proximity of ponds to neighbouring habitats is known to be important for local biodiversity (Brönmark 1985; Gledhill *et al.* 2008; Zealand and Jeffries 2009), the relative importance of regional factors (dispersal, external forcing), compared to local (abiotic and biotic) is not well established.

The composition of a local biological community reflects the interplay between local and regional processes. The local and regional aspect refers to the spatial scales at which ecological and biogeographic processes dominate (Cornell and Lawton 1992). Predation (top-down and bottom-up effects), disease, competitive exclusion and stochastic disturbances are examples of local processes, which for a long time were thought to be the main determinants of local diversity (Ricklefs 1987; Holt 1993). The extent to which regional processes can influence a local community is dependent upon the dispersal of species from neighbouring habitat patches, the configuration of the habitat network and obstacles, or opportunities, for dispersal across the terrestrial matrix (Marsh and Trenham 2001; Jeffries 2005; Fahrig 2007). Studying the relative influence of local and regional processes is important for predicting the future effects, such as habitat loss, upon species and biological communities of interest (Lindenmayer and Possingham 1996).

Urban-rural gradient studies provide a framework within which the relative effects of local and regional processes upon biological communities in an urban landscape can be assessed. Such studies were recognised in the early 1990s as being an unexploited opportunity to investigate ecological topics that not only have relevance in and urban setting, but also to the wider environment (McDonnell and Pickett 1990). However, whilst the concept has received much attention, little consensus exists for determining how urbanisation should be quantified, and the advent of geographical information systems (GIS) has increased the number of possible measures (Hahs and McDonnell 2006). Hahs and McDonnell (2006) performed a principal components analysis (PCA) of 17 commonly used urbanisation measures in order to identify those that captured most of the observed patterns of urbanisation. The 17 measures included basic physical variables such as the proportional coverage of impermeable surface (e.g. Sponseller and Benfield 2001; Ourso and Frenzel 2003), landscape metrics such as the proportional cover of urban land-use (e.g. Borgmann and Rodewald 2004) and demographic indicators such as household density (e.g. Tratalos *et al.* 2007).

The Water Framework Directive (WFD, European Commission 2000) requires that all surface waters attain an ecological ‘good’ status by 2015. Thus, the imminent implementation of the WFD presents clear imperative to improve the quality of pond habitats. However, despite their apparent importance for biodiversity at regional scales, ponds are generally excluded from WFD research programmes, such as STAR, AQEM and ECOFRAME (Cereghino *et al.* 2008) and are functionally different to lakes which have received more attention (Oertli *et al.* 2002; Fairchild *et al.* 2005; Sondergaard *et al.* 2005). In addition, monitoring and assessment of water bodies < 50ha under the WFD will not occur under, with the exception of those > 1ha that are located within protected areas, of which there are around 250 sites within England and Wales (Williams *et al.* 2010). Nonetheless, ponds can be ascribed statutory protection if they meet criteria that could qualify them as a Priority Habitats (JNCC 2008) under the Biodiversity Action Plan (U.K. BAP), such as supporting Red Data Book species or exceptional assemblages of key biotic groups (Appendix 1). Therefore, in order to conserve pond habitats research is needed to establish the main drivers of community dynamics at local to regional scales (Cereghino *et al.* 2008). This is particularly important in the face of climate change, which is likely to increase pressure upon aquatic systems (Brönmark and Hansson 2002; Dudgeon *et al.* 2006) that may need to accommodate more frequent dispersal events (Jeffries 2004).

1.1. Key ecological processes

1.1.1. Habitat loss

Pond numbers are in decline across the U.K. (Biggs *et al.* 2005), Europe (Indermuehle *et al.* 2008) and globally (Huang *et al.* 2012). Habitat loss, whether by natural or anthropogenic processes, is considered the greatest threat to global biodiversity (Brooks *et al.* 2002; Millenium Ecosystem Assessment 2005; Hanski 2011). Habitat loss is often used interchangeably with habitat fragmentation, although an important distinction is that the latter refers to the break-up of a single habitat patch into multiple components (Fahrig 1997). Where habitat is lost or detrimentally impacted, species populations become extinct *via* demographic, environmental stochastic (Fahrig 1997) and/or deterministic events (Marsh and Trenham 2001). Species response to habitat loss may occur within or between generations; the phenomenon whereby time lags in ecological responses to habitat loss mask the full extent of its impact on biodiversity has been dubbed 'extinction debt' by Tilman *et al.* (1994). In the longer term, habitat loss has profound genetic and evolutionary consequences for small populations, for example, *via* inbreeding (Cushman 2006).

1.1.2. Disturbance regimes

The structure of ecological communities is shaped by the frequency, magnitude and duration of natural disturbances (Rebele 1994; Hooper *et al.* 2005; McDonnell and Hahs 2008). Perhaps the most well known disturbance model, the intermediate disturbance hypothesis (Connell

1978) suggests that at moderate levels of disturbance, diversity is maximised because both competitive/late-successional (*k*-selected) and opportunistic/early-successional (*r*-selected) species can co-exist. Lower levels of disturbance, therefore, favour competitive species offering few vacant niches for opportunistic, whereas high levels of disturbance, favour early successional species. At high levels of disturbance ecosystems are suggested to be more vulnerable to invasion from non-native species which are able to exploit the niches left open by removed natives (D'Antonio and Meyerson 2002). Anthropogenic activities may alter natural disturbance regimes, recasting community composition to the benefit of some species and detriment of others. For example, channelisation of urban rivers and increased runoff from impermeable surfaces increases the frequency and magnitude of high flow events, also increasing nutrient and sediment loads (Paul and Meyer 2001; Allan 2004). In ponds, regular disturbance may also be caused by management practices such as dredging or vegetation removal for the purposes of fisheries management (Biggs *et al.* 2005).

1.1.3. Metacommunity concepts

Ponds are discrete aquatic habitats within a terrestrial matrix, such that dispersal between them is an important process for the organisms they support (Jeffries 1994; Nurnberger 1996; Briers and Warren 2000). Many ponds are thought to form metapopulations (Levins 1969; Hanski and Gilpin 1991), whereby species exist in a number of spatially-separated populations that are linked by dispersal (Bohonak and Jenkins 2003). Multiple metapopulation-forming species that inhabit the same habitat patch form a metacommunity (Wilson 1992).

The following four approaches to metacommunities have been proposed by Leibold *et al.* (2004), for which ponds provide an excellent opportunity for study (De Meester *et al.* 2005). First, the patch dynamic perspective views patches as homogenous, and colonisation is balanced by interspecific interactions (e.g. competition and colonisation abilities) (Leibold *et al.* 2004). The species-sorting hypothesis (Leibold *et al.* 2004) suggests that local communities differ in response to resource gradients, where dispersal facilitates the tracking of species to changing local conditions and subsequent opening or closing of niches to which they are adapted. By contrast, neutral theory (Hubbell 2001) suggests that local communities are influenced by variation in dispersal traits amongst species that are ecologically equivalent i.e. they have the same competitive ability and fitness. Finally, the mass-effect concept (Shmida and Wilson 1985) considers patches to be heterogeneous, but for colonisation rates to be sufficiently high so as to diminish interspecific interactions. Consequently, by establishing the relative influence of local and regional factors it is possible to indicate whether species-sorting mechanisms or, should dispersal limitation be evident, neutral theory concepts are more influential upon urban pond communities (Heino and Mykra 2008).

1.1.4. Ecosystem functioning

Urban ponds provide a novel system in which to assess ecosystem functioning. Urbanisation can decrease biodiversity in aquatic systems (Paul and Meyer 2001; Roy *et al.* 2003). Numerous studies have suggested that biodiversity loss can impair ecosystem functioning (Naeem and Li 1997; Tilman *et al.* 1997) and are founded on the premise that different species in a community fulfil functional roles that are important for the transformation of energy and matter cycling (Lawton 1994; Ghilarov 2000). An ecosystem high in functional redundancy contains several species that perform similar functional roles and may therefore be substitutable with little impact upon ecosystem processes (Lawton and Brown 1993). Consequently, high functional redundancy relates to high ecosystem reliability (Naeem and Li 1997), which improves the likelihood that a system will provide a consistently high level of performance (Naeem 1998). The extent to which the loss of species will impact upon ecosystem functioning, however, depends upon the ecosystem type, as well as the species abundance, functional role, contribution to functional processes and functional redundancy in the system (Hooper *et al.* 2005). It is not known whether urbanisation affects the functional resilience or ecosystem functioning of urban ponds.

1.1.5. Ecological resilience

Ecological resilience relates to the capacity of an ecosystem to recover from deleterious impacts to ecosystem functioning and maintain stability before shifting to an alternative stable state (Holling 1973). The concept of functional redundancy is one such example of ecological resilience whereby functional processes may be maintained despite species loss (e.g. Hladyz *et al.* 2011), although this is dependent upon the number of species that are substitutable within each functional group (Walker 1995).

Spatial resilience refers to the ability of connected ecosystems to persist on a regional level (Nystrom and Folke 2001). Improved connectivity between habitat patches, such as ponds, may improve population persistence by providing a larger, more abundant species pool from which an impacted habitat may source re-colonising individuals (Minor and Urban 2008) and in which genetic information can be exchanged (Bodin and Norberg 2007). Furthermore, increased heterogeneity of connected habitats has been argued to increase spatial aspects of resilience (Folke *et al.* 2004). Conversely, greater connectivity can facilitate the spread of some disturbances e.g. invasive species (Cumming 2002) or predators (Scheffer *et al.* 2006). New methodologies based around network (or graph) theory have been proposed for assessing the spatial resilience of fragmented habitats (Bodin and Norberg 2007; Urban *et al.* 2009; Saura *et al.* 2011) and could prove a useful tool for the maintenance of spatial resilience in pond networks.

1.2. Thesis aim

The aim of this research is to assess the conservation value of urban ponds and identify the local and regional factors that control their biodiversity and ecosystem functioning such that future threats to urban pond networks from anthropogenic disturbances can be minimised. This aim is met through four objectives which evaluate local and regional processes at several different scales:

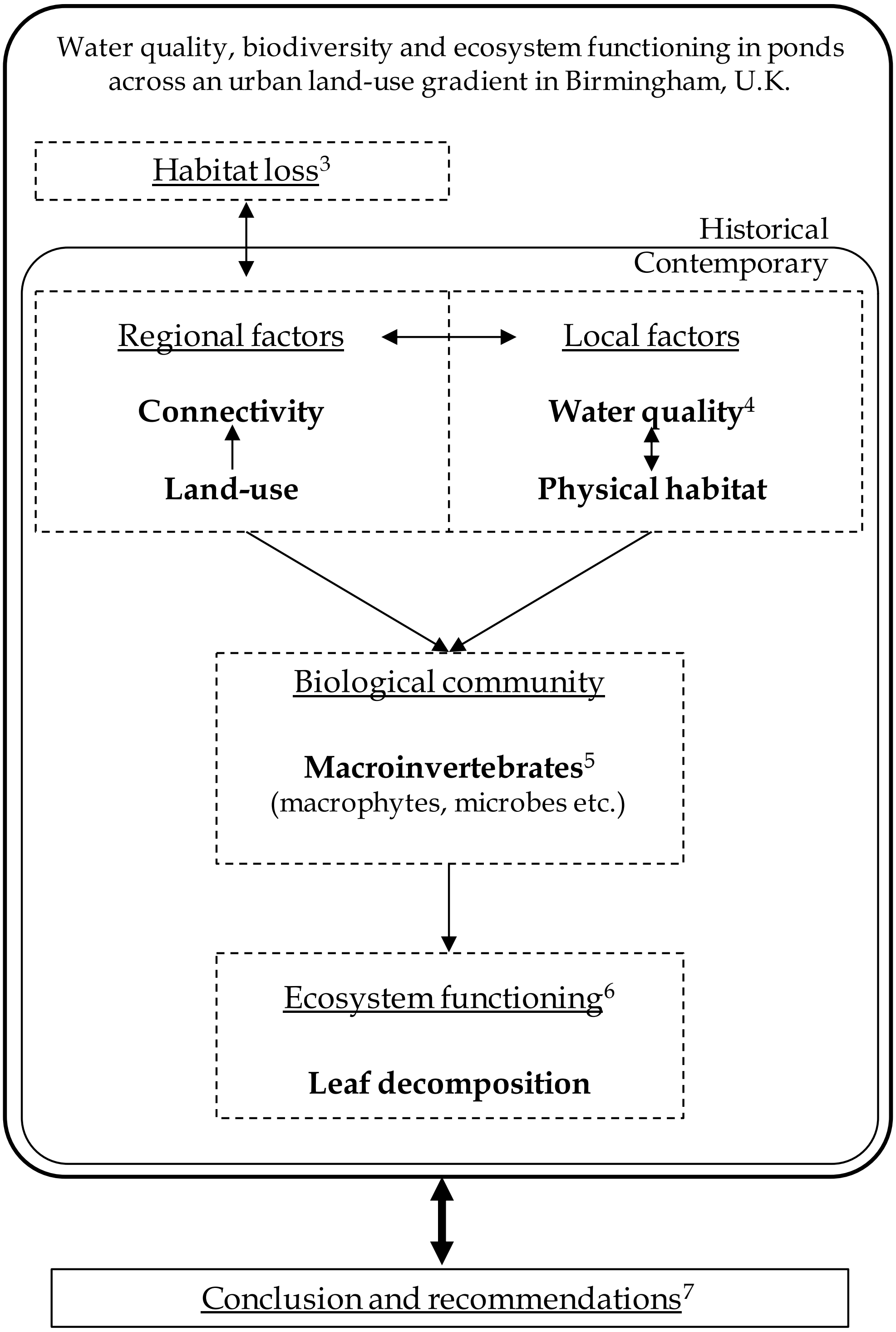
- To assess how habitat loss and gain has impacted upon the spatial resilience of the pond network and the potential implications this may have for species that persist in metapopulations
- To investigate the relative influence of local and regional factors upon water quality and its compliance with environmental guidelines
- To investigate the relative influence of local and regional factors upon the macroinvertebrate community and its conservation value
- To investigate the influence of urbanisation on ecosystem functioning and functional redundancy

1.3. Thesis outline

Chapter two details methods that apply to several chapters of this thesis and is included to avoid unnecessary repetition of methods in subsequent chapters. Importantly, chapter two describes the selection of 30 study ponds selected to span an urbanisation gradient that is referred to throughout the thesis. The subsequent five chapters interact within an analytical framework designed to meet the thesis aims (Figure 1.3.1).

Chapter three explores how the contemporary (2009) Birmingham pond network has varied between three time periods, ca1904, ca1962 and 2009 in order to facilitate an in depth analysis of pond habitat gain and loss. The chapter shows how changes in land-use affected pond densities over time, revealing both significant habitat loss since the turn of the 20th century, and surprising changes in the spatial configuration of the remaining pond network. The impact of changing pond density is contextualised using a network theory approach to explore the potential for insect dispersal and pond connectivity across the Birmingham conurbation. Analysis of the impact from past land-use change to pond densities,

Figure 1.3.1. Thesis research design and overview of the topics covered and their main linkages



* Superscript numbers indicate the chapter in which the subject forms the main focus

culminating in the contemporary pond network provides context for chapter four and spatial data which is incorporated into chapter five.

Chapter four investigates the trophic status and spatial and temporal variation in water quality across the urban ponds. Water quality is assessed for compliance with freshwater standards that are gathered from the Environment Agency chemical standards database (EA 2012). Urban ponds often have artificial catchments due to the nature of the surrounding landscape and here I identify a spatial extent within which ponds are most sensitive to changes in land-use. This is achieved by a sequence of ordinations of land-use attributes derived from a GIS at consecutive spatial extents, with the effect of local physical factors partialled-out.

In **chapter five**, macroinvertebrates are used as biological indicators of conservation value and environmental stress. The relative impacts of local and regional processes upon macroinvertebrate assemblages, which are informed by the work undertaken in chapters three and four, are evaluated in order to assess for the most pressing threats to biodiversity within urban ponds. Chapter five continues to consider differing controls upon taxa richness within five major macroinvertebrate orders that are characteristic of the pond environment. In doing so, the relevant landscape scale to taxa with contrasting dispersal traits is revealed.

In **chapter six**, ponds are classified into four groups that span the urbanisation gradient. Moving beyond structural indices of ecological integrity i.e. chapter five, a leaf decomposition experiment was used to assess for the impact of urbanisation upon ecosystem functioning. Important local (water quality, local physical and macroinvertebrate species) variables are identified using ordination and revealed confounding factors that masked trends in ecosystem functioning across the gradient. Furthermore, chapter six explores the diversity and ecosystem function relationship in order to assess the appropriateness of conservation for the improvement of biodiversity and a consideration of functional redundancy indicates potential risks to ecosystem resilience and stability.

Chapter seven summarises the research conclusions and uses the key findings to suggest strategies that may be employed by urban pond and landscape managers to improve the biodiversity, ecological resilience and conservation value of urban ponds in the future to the benefit of wildlife and people.

1.4. Study area

Birmingham and the Black Country (BBC) is an informal term for an area of the West Midlands conurbation comprising of the cities of Birmingham and Wolverhampton and the metropolitan boroughs of Sandwell, Walsall and Dudley. In all, BBC covers 62.5 km^2 of which around 55% is residential, 19% is urban or industrial and 13% is improved grassland, a further 3% is classified as either coniferous or deciduous woodland (data derived from Land Cover Map 2007). By population size, Birmingham is the second largest city in the UK after London (ONS 2011).

The underlying geology of BBC is broadly split into a western and eastern component. The west comprises of Lower and Middle Coal measures and the Etruria Marl, of Carboniferous age (~ 300 million years ago). There are also a few small but notable Silurian limestone outcrops. The eastern area is dominated by younger rocks from late Carboniferous formations to Sherwood Sandstone and Mercia Mudstone of the Triassic (~ 220 million years ago).

Historically, Birmingham and the Black Country was one of the most intensely industrialised regions within the U.K. and is considered to be the birthplace of the industrial revolution. Whilst the Black Country was known for coal mines and coal coking operations and for the vast number of iron foundries and steel mills, Birmingham was the centre of commerce for the sale and trade of locally manufactured goods. As a consequence, air pollution was considerable (Chitham 2009). Charles Dickens once described how the local factory chimneys

> *'...poured out their plague of smoke, obscured the light, and made foul the melancholy air'.*
>
> - Charles Dickens

However, industry in BBC declined rapidly during the latter half of the 21st century, particularly in Birmingham, and the modern day landscape is a reflection of land-use exploitation and comprises a complex land-use that includes brownfield, canals, remnant ancient woodland patches and parklands. In Birmingham, manufacturing industries have been replaced by tertiary industries such as shopping centres, bars and restaurants, whereas a considerable number of manufacturing industries persist on the periphery of urban centres in the Black Country.

2

General methods

Chapter 2: General methods

2.1. Site selection

Thirty ponds of contrasting surrounding land-use were selected from an estimated 1023 sites in Birmingham and the Black Country (BBC) (Figure 2.1.1). Ponds were drawn from a range of landscapes from rural villages to heavily urbanized city centres, characteristic of the West Midlands. In selecting the sites, the land-use categories of Owen *et al.* (2006) were used to guide site selection (Figure 2.1.1b). Owen *et al.* (2006) categorised each 1km^2 of the West Midlands, which includes BBC, into eight urban land-use classes. This was achieved by analysing the proportional cover of 25 land-use attributes such as open land, urban, villages and motorways within each 1km^2 using a principal components analysis (PCA) and cluster analysis. The resulting urban land classes were: villages/farms, suburban, light suburban, dense suburban, urban transport, urban, woodland/open land and light urban/open water.
A field survey of 111 ponds was undertaken to assess access and water chemistry e.g. pH, conductivity (not shown here). Selected from the initial 111, the final 30 study ponds were stratified across the urban land classes (Table 2.1) and selected on the basis of good access, geographical spread and representation of the water quality gradient. All subsequent analyses used new, more highly resolved, land-use coverage data derived by the author.

A GIS dataset of water bodies less than 2ha in surface area was derived from Ordnance Survey MasterMap (Ordnance Survey (GB) 2008), which was subsequently edited through a combination of expert knowledge made available through the Birmingham and Black Country Wildlife Trust, field visits (conducted between 18th February 2009 and 1st May 2009) and aerial imagery (Google Inc. 2009). Consequently, new ponds were added and infilled ponds removed. Similarly, the accuracy of pond outlines was also improved. The final dataset suggested a total of 1023 ponds within BBC, although this figure was unlikely to account for a number of temporary water bodies and garden ponds.

Further site detail and photographs is displayed in Appendix 2.

Figure 2.1.1 Study area and selection of study ponds across Birmingham and the Black Country (BBC) a) BBC, it's districts and ponds (blue), b) The 30 selected pond site locations and the Owen *et al.* (2006) urban land-use classifications used to guide study pond selection

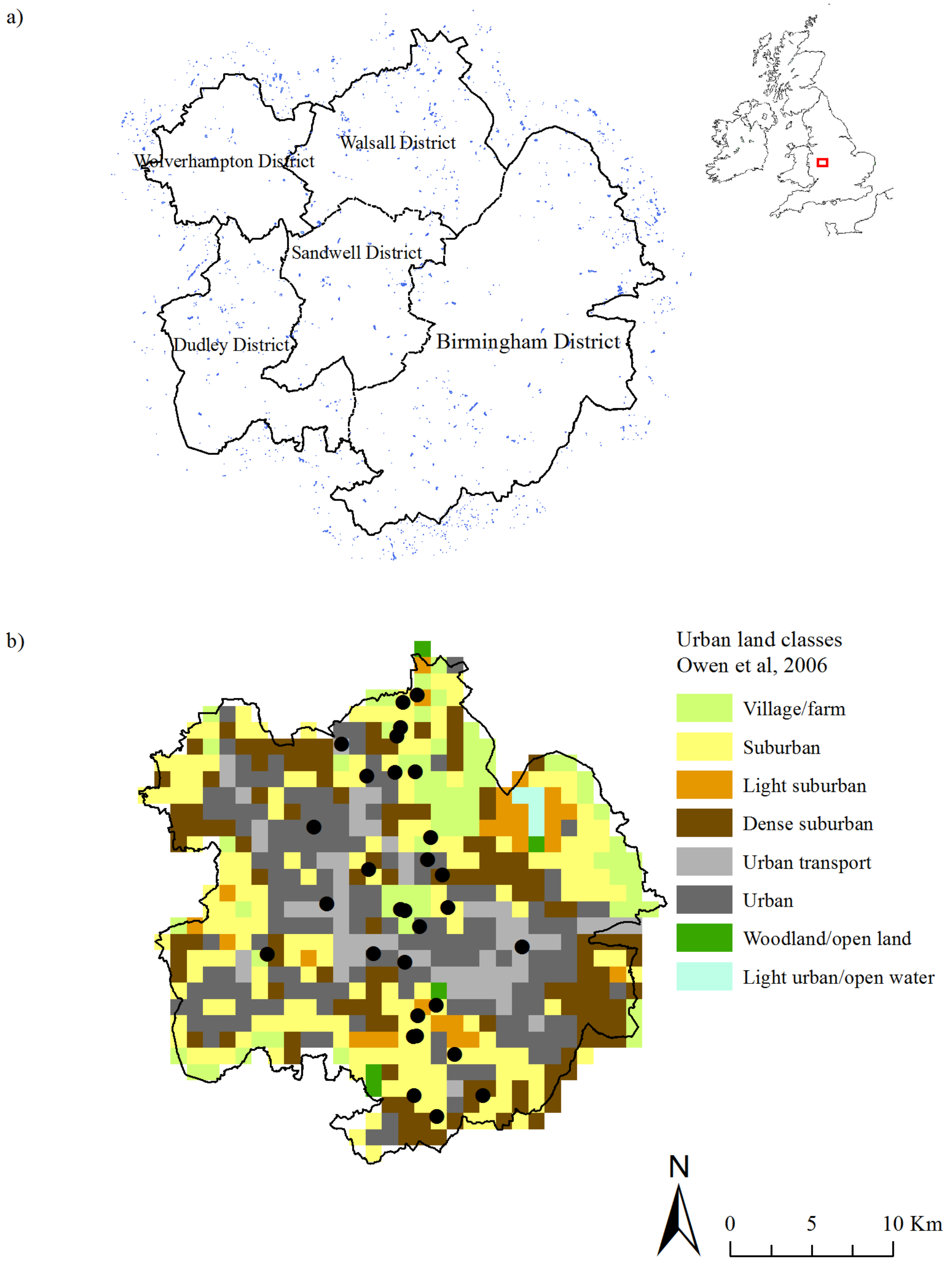


Table 2.1. Distribution of ponds within Urban Land Classes (Owen *et al.* 2006) and overall coverage of classes within Birmingham and the Black Country

Urban Land Class	% Coverage of total BBC area	Number of study ponds	% of study ponds
Village/ farm	11	6	20
Suburban	28	11	37
Light suburban	4	1	0.3
Dense suburban	20	2	0.7
Urban transport	10	4	13
Urban	25	6	20
Totals:	88	30	100

* 1km^2 classifications of woodland/open land and light urban/open water were too few to be incorporated into pond selection

2.1. Field sampling

Beginning in May 2009, samples were collected on a quarterly basis to represent seasonal differences in water quality, the last sample having been collected in February 2011 (Table 2.2). Macroinvertebrate assemblages were sampled twice, in late May/ early June and August 2009 i.e. just over two months apart as recommended by Biggs *et al.* (1998). This strategy gives adequate representation of the community and allow for species with differing phenology to be captured (Cayrou and Cereghino 2004). Therefore, the chances of catching species in later instars or adult form are improved, whilst allowing for recovery between samples.

The time taken for each sampling period was kept to a minimum to reduce within sample variance caused by changes in weather. Sampling of all 30 sites took up to 14 days where macroinvertebrate samples were collected alongside those for water quality, and up to four days where water sample collection was the sole objective.

2.2. Physico-chemical variables

Where practicable, a composite water sample was collected at each site from the pond margins near the inflow, outflow and at an approximately equidistant point between the two. In the absence of a clear inflow or outflow, samples were evenly spread around the pond perimeter. Contact with vegetation or the bottom substrate was carefully avoided during sampling, as was collection of surface debris. Once collected, samples were kept cool, filtered (GF/C, 1.2µm) and returned to the laboratory for determinations of major ions, trace metals, suspended solids and chlorophyll abundance. Samples were store in two 50ml

Table 2.2. Sampling periods and dates of sampling

Season and hydrological year	Sampling dates	Sample type
Spring 2009	20th May – 11th June 2009	Ma, WQ
Summer 2009	3rd August – 14th August 2009	Ma, WQ
Autumn 2009	3rd November – 17th November 2009	Ma*, WQ
Winter 2009	27th February – 3rd March 2010	WQ
Spring 2010	9th June – 11th June 2010	WQ
Summer 2010	18th August – 25th August 2010	WQ
Autumn 2010	10th November – 13th November 2011	WQ
Winter 2010	21st February – 23rd February 2011	WQ

Ma = Macroinvertebrate sample collected, WQ = Water quality assessed

* Macroinvertebrate sample collected, but not processed

uncontaminated vials, one for anion analysis and a second, acidified (2% nitric acid) for cation and heavy metals analysis.

2.2.1. *In situ* measurements

Dissolved oxygen (% saturation), electrical conductivity (µS/cm), temperature (^oC) and pH were measured *in situ* using a YSI 556 handheld multi-probe meter, calibrated daily before use (YSI, Yellow Springs, OH, USA). Triplicate measurements were taken around each pond perimeter in the same manner as for water sample collection.

2.2.2. Major ion analysis

Anions determined using a Dionex ICS2000 Chromatography System (Dionex Corporation, Sunnyvale, AC, USA) were Cl (chloride), NO_3 (nitrate) and SO_4 (sulphate). Cations measured using a Dionex DX500 (Dionex Corporation, Sunnyvale, AC, USA) were Na (sodium), NH_4 (ammonium), K (potassium), Mg (magnesium) and Ca (calcium). PO_4 (phosphate) was measured using a HI-93713 Phosphate Photometer (Hanna Instruments, Bedfordshire, U.K.) and alkalinity ($CaCO_3$) was determined by titration (to pH 4.5; HACH, Dusseldorf, Germany).

Samples were kept refrigerated at 5^oC until analysis. Following unsatisfactory results from the first two surveys (contemporary with macroinvertebrate sampling in 2009) the method of analysis for PO_4 analysis was changed from ion chromatography to colour photometry to reduce the delay between sample collection and analysis. Alkalinity was not analysed during the first survey. Consequently, data used for alkalinity and PO_4 are those recorded in the equivalent period(s) for the following hydrological year (2010) where they were not permissible in 2009. PO_4 and alkalinity were sebsequently measured within 24 hours of sample collection.

Calibration standards of 2.5, 5, 10, 25 and 100 mg/l were used for ion chromatography analyses. Minimum detection limits were calculated as three times the standard error of blank samples (deionised water) analysed concurrent with pond water samples collected from each sampling period (Appendix 3).

2.2.3. Suspended solids and chlorophyll

In order to analyse for suspended solids and chlorophyll *a*, *b* and *c*, sufficient volume of pond water was filtered such that suspended material was clearly visible on the filter paper (Whatman GF/C, 1.2µm pore size). Prior to use, the filter papers were dried at a temperature of 105°C to correct for weight attributable to moisture content and were consequently frozen after filtration (-20°C) until analysis. Upon analysis the filters were freeze dried to a constant mass and weighed to calculate suspended solids accordingly (Eq. 2.2.1).

$$SS(mg/l) = \frac{(f2 - f1) \times 1000}{V} \qquad \mathbf{2.2.1}$$

Where f_1 is the oven dried mass of the filter paper prior to use and f_2 is the freeze- dried mass of the filter after use. V is the volume of water filtered.

For chlorophyll analysis, the freeze-dried filters were folded and macerated for 30 seconds each in 2ml, 90% acetone. The solution was then consolidated to 10ml 90% acetone, mixed, and centrifuged at 1500 x g for 5 minutes. For measurement, the trichromatic method was employed (Jeffrey and Humphrey 1975). A ultra-violet spectrophotometer with a path length of 1cm and a 1.5nm spectral bandwidth was used (Jenway 6800, Bibby Scientific Ltd, Staffordshire, U.K.). The concentration of chlorophyll *a*, *b* and *c* was calculated according to the equations of Jeffrey and Humphrey (1975) (Eq's. 2.2.2 ,2.2.3, 2.2.4)

$$\textit{Chl-a} = 11.85(E_{664} - E_{750}) - 1.54(E_{647} - E_{750}) - 0.08(E_{664} - E_{750})) \times V_e/L \times V_f \qquad \mathbf{2.2.2}$$

$$\textit{Chl-b} = 5.43(E_{664} - E_{750}) + 21.03(E_{647} - E_{750}) - 2.66(E_{664} - E_{750})) \times V_e/L \times V_f \qquad \mathbf{2.2.3}$$

$$\textit{Chl-c} = 1.67(E_{664} - E_{750}) - 7.6(E_{647} - E_{750}) + 24.52(E_{664} - E_{750})) \times V_e/L \times V_f \qquad \mathbf{2.2.4}$$

Where L is the cuvette light path in centimetres, V_e is the extraction volume in litres and V_f is the filtered volume in millilitres. Chlorophyll concentrations are calculated as parts per billion or µg/l. Chlorophyll *a* and suspended solids were not analysed during the first sampling period. Consequently, data were used that were recorded in the equivalent period from the following hydrological year (2010). Abundance of chlorophyll *b* and *c* was typically below instrumental detection limits and was excluded from any subsequent statistical analyses.

2.2.4. Heavy metals

Heavy metal concentrations determined using a Perkin Elmer AA300 (Perkin Elmer, Massachusettes, USA) were Fe (iron), Mn (manganese), Ni (nickel), Cu (copper), Co (cobalt), Cr (chromium), Pb (lead) and Zn (zinc).

The dissolved metal fraction is defined as metal elements of a non-acidified sample that pass through a 0.45µm filter (Riley and Taylor 1968). Therefore, a sample filtered at 1.2µm (prior to acidification) may contain some non-bioavailable particulate metal. Consequently, heavy metal concentrations in water samples filtered at 0.2µ, 0.45µm and 1.2µm were compared to assess the biological relevance of the concentrations reported following 1.2µm filtration. No significant difference was found such that the concentrations reported give a good representation of the dissolved metal fraction (Appendix 4).

Calibration standards of 0.01, 0.1, 0.5, 1 mg/l were used for heavy metals analysis. Minimum detection limits were calculated as three times the standard error of blank samples (deionised water) analysed concurrent with pond water samples collected from each sampling period (Appendix 3).

2.2.5. Physical characteristics

Single measurements of pond surface area and percentages of surface area classified as open water, shaded, riparian vegetation and floating vegetation were all derived from a combination of digitised field notes and Normalised Difference Vegetation Index (NDVI) and photogrammetrically derived height GIS layers (Section 2.3.1.6) against digital pond outlines. The percentage of pond bank that was made from concrete was also calculated in a GIS from digitized field notes. Water sources (i.e. stream inflows, groundwater, surface run-off or building run-off) and fish presence information was gathered from stakeholder knowledge and visual inspection. A water level fluctuation index (WLFI) was calculated as the standard deviation in depth between surveys (taken quarterly between 2010 – 2011) measured at set points within the wetted-perimeter of each pond.

Macrophyte presence absence was recorded from within identified mesohabitats sampled for macroinvertebrates (Section 2.4) using Haslam *et al.* (1982) for identification.

2.3. Land use

Within a GIS, several physical and demographic datasets were incorporated to characterise land-use. Each dataset is discussed below and all of the measures derived from the GIS dataset are summarised in Table 2.3.

The artificial nature of urban pond inputs from infrastructure, for example surface drainage and sewer overflows, obscured delineation of catchments using traditional analyses of topology e.g. Digital Elevation Models (DEM). Consequently, land-use attributes were calculated within 13 cumulative spatial extents: 25m, 50m, 75m, 100m and then in 100m increments to 1000m. Greater resolution below 100m was included due to the known sensitivity of the pond environment to its immediate surroundings (Declerck *et al.* 2006; Akasaka *et al.* 2010). Land-use extracted from distances of more than 1000m from the pond edge was increasingly autocorrelated with those at preceding distances e.g. 1100m with 1000m. Similarly, increased correlation was found between land use surrounding different sites where site buffers began to overlap. Analysis of land-use was therefore capped at 1000m.

2.3.1.1. Ordnance Survey MasterMap

OS MasterMap is a close representation of real world objects (Figure 2.3.1a). The topographic features of the dataset include a breakdown of buildings, gardens, roads, pavements, railways, rivers, ponds and natural or manmade environments. The data used in this study was accessed in December 2008 (Ordnance Survey (GB) 2008) with the original MasterMap data amended to categorise road surfaces (Ordnance Survey (GB) 2009), accessed *via* EDINA Digimap (http://digimap.edina.ac.U.K.).

2.3.1.2. Photogrammetrically derived height data

The height of landscape features was derived using photogrammetry (Figure 2.3.1b), which is often more reliable than other techniques such as light detection and ranging (LiDAR) and interferometric synthetic aperture radar (IFSAR) in areas with steep or rapid changes in terrain such as built-up areas (Veneziano 2002). Photogrammetry is a technique for determining geometric properties from photographic images where the three-dimensional features of objects can be determined by measurements made in two or more photographs taken from different positions using the principles of triangulation. Data was available in 2m resolution (x and y direction), which was subsequently resampled in a GIS to 1m resolution (x, y and z directions), from aerial images taken on the 1st May and the 5th and 9th August 2007, accessed *via* the Landmap service (http://www.landmap.ac.U.K.) and made available by Bluesky (Bluesky International Limited, Leicestershire).

2.3.1.3. Land Cover Map 2007

Land Cover Map 2007 (LCM2007) uses 23 habitat classes to map the U.K. (Figure 2.3.1c), which are based on the U.K. Biodiversity Action Plan (BAP) Broad Habitats (Jackson 2000). Such habitats include arable land, coniferous or broadleaved woodland and rough grassland. LCM2007 is created by classifying summer-winter composite images captured by satellite sensors with 20 - 30m pixels (Morton *et al.* 2011). The dataset was built using the same generalised spatial framework as OS MasterMap and as such the two have a good level of compatibility (Figure 2.3.1c). LCM2007 was sourced through the Centre for Ecology and Hydrology (NERC (CEH) 2011).

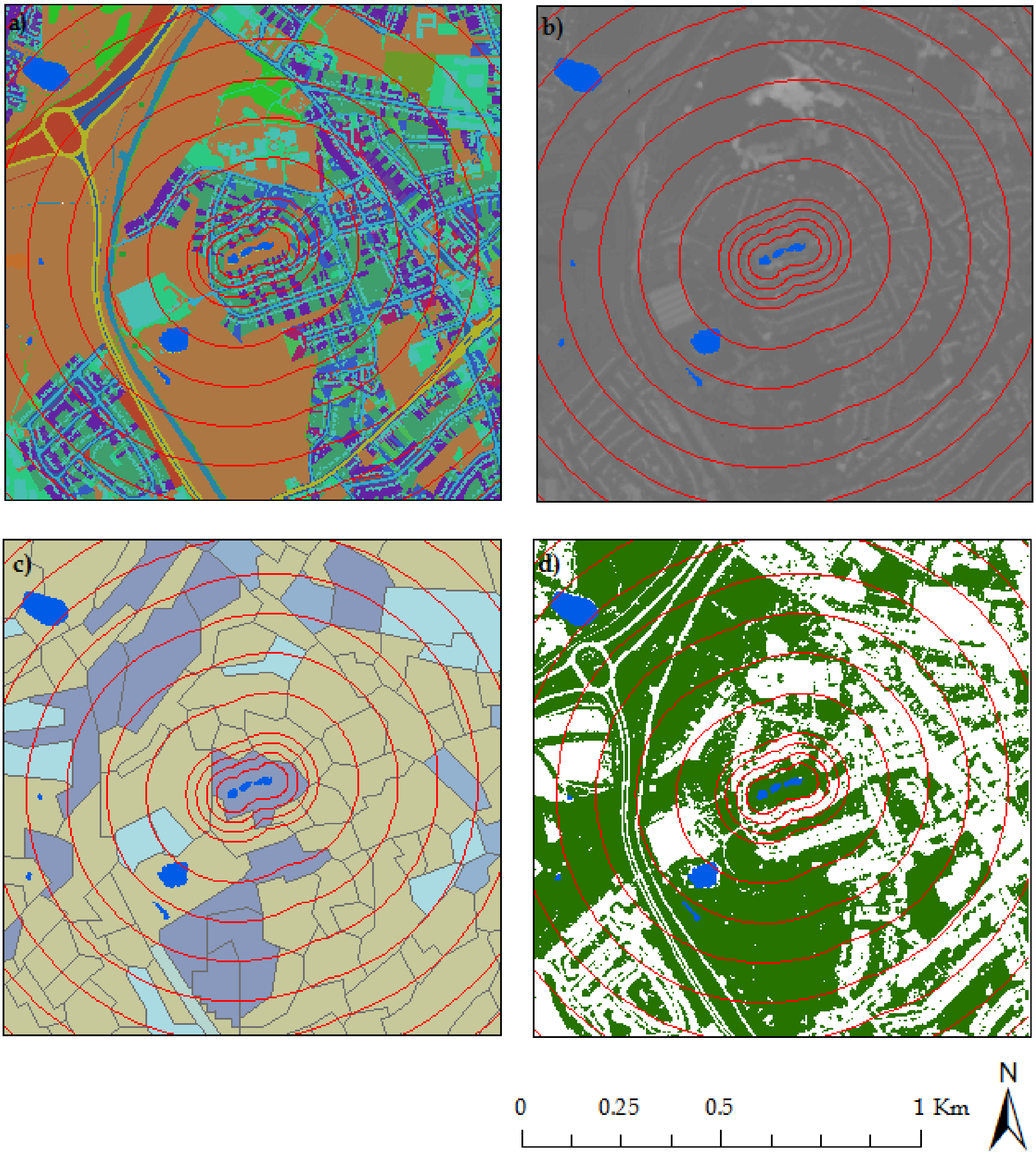
2.3.1.4. Normalised Difference Vegetation Index

The normalised difference vegetation index (NDVI) is a well established method that has been used to monitor plant growth, vegetation cover and biomass production from remotely sensed information (e.g. Zhou et al. 2001). First, the blue-band of (red, green, blue) aerial images was replaced by near infrared (NIR) which captures reflections of the sun's rays. NIR is reflected at the bottom of leaves after passing through, which allows for derivations of plant health and vigour. NDVI was subsequently calculated in a GIS by combining NIR and red bands (R); NDVI = (NIR – R) / (NIR + R). Data were available at 2m resolution, which was subsequently resampled in a GIS to 1m resolution in a 2-bit (binary) format from images taken on 12th May 2006 (Figure 2.3.1d). Aerial imagery and NIR data were accessed *via* the Landmap service (http://www.landmap.ac.U.K.) and made available by The GeoInformation Group (Fulbourn, Cambridge) and Bluesky (Bluesky International Limited, Leicestershire).

2.3.1.5. Census 2001

The 2001 U.K. Census of population was conducted on the 29th April 2001 (ONS 2011), although subtle changes from the 2001 dataset will have occurred it is likely to remain indicative of current demographic trends across the BBC. Data from the 2011 Census was taken on the 27th March 2011 and only piecemeal data were available at the time of writing. Key Statistics from the 2001 Census were obtained for postcode sectors in England and Wales and attributed to a GIS layer. Consequently the data could be extracted in the same manner as land-use data. Variables derived from the Census 2001 dataset were population density (number of people per hectare) and people aged 16-74 working in agriculture, hunting, forestry, fishing, mining and quarrying (rural employment). Household density was calculated by dividing the postcode sector area by the number of household spaces with residents, also available from the Census 2001 dataset (Table 2.3).

Figure 2.3.1. Screenshots of each of the main GIS land-use data layers
a) OS MasterMap b) Photogrammetrical height c) Land Cover Map 2007 d) NDVI vegetation, red lines indicate spatial extents at 25m, 75m, 100m and 100m increments to 1000m. Blue polygons are ponds.



2.3.1.6. Land-use variables derived from multiple GIS sources

Photogrammetry height was combined with buildings extracted from OS MasterMap in order to calculate a metric of building intensity (B_i) (Eq. 2.3.1).

$$B_i = \frac{\sum B}{\sum A} \times B_h \qquad \textbf{2.3.1}$$

Where $\sum B$ is the proportional coverage of buildings within a given spatial extent, $\sum A$ is the total area and B_h is mean building height. The building intensity metric is designed to give a more balanced representation of the degree of urban intensity than building height alone.

It was also possible to categorise NDVI derived vegetation cover by height. The categories consisted of vegetation at ground level at heights of ≤0m (i.e. grass, aquatic floating vegetation), low level vegetation between 0m and 3m (i.e. shrub, scrub and hedgerows), and mature trees between 3m and 60m. Errors were observed within the NDVI dataset where buildings with green roofs or green cars were present. Consequently a 4m buffer was projected around buildings (using OS MasterMap) and data was 'clipped' (removed) from the vegetation dataset. Nonetheless, due to the presence of wells and structures other than buildings some error remained and implied the need for a height limit. An upper limit for vegetation data of 60m was established as the tallest tree species in the United Kingdom, the Douglas Fir (*Pseudotsuga menziesii*). Heights < 0m were included in the ground level vegetation dataset and those above 60m excluded. It should be noted that given the nature of the NDVI methodology in conjunction with photogrammetry data, only the uppermost vegetation was recorded at any one location.

2.3.1.7. Establishing the degree of urbanisation

For this study, five landscape, three demographic and two physical metrics ('urbanisation indicators', Table 2.3) were used within a series of PCA ordinations, using data extracted from each of the 13 cumulative spatial extents at 25m, 50m, 75m, 100m and then in 100m increments to 1000m, in order to establish an urbanisation score for each site. Six of the ten metrics are directly referred to by the Hahs and McDonnell (2006) review, the remaining four (suburban cover, arable land, 'towns and villages' and building intensity) capture further subtle urbanisation patterns. The resulting PCA scores extracted from axes 1 and 2 are suggested to give a better account of 'environmental space' as opposed to geographic space (Hahs and McDonnell 2006) for the landscape surrounding each site.

Prior to use in the PCA, variables were transformed to improve normality and homogeneity of variance where necessary. Proportional variables were typically arcsine transformed, distance metrics were square-root transformed and building intensity was log(*n*+1) transformed. Data were then standardised to 0 mean and 1 standard deviation due to differences in scale. PCA ordinations were carried out using the function 'rda' for PCA in the 'vegan' package (Oksanen *et al.* 2012) for the R statistical program version 2.15.1 (R Core Team and contributors worldwide 2012). Normality and homogeneity of variance were tested by Shapiro-Wilk and Levene's tests carried out in IBM SPSS statistical package version 19 (IBM, Armonk, New York).

Between 64% and 75.7% of variation within the 10 urbanisation indicators was explained on the first two principal component axes (Table 2.4). Axis 1 (PCA 1) positively correlated to physical and landscape variables and to the proportion of impermeable surface in particular. Conversely, the proportion of arable land was negatively correlated to axis 1. Axis 2 (PCA 2) typically described demographic gradients. Variance explained by axis 2 correlated to a measure of either population or housing density and, by contrast, the proportion of people employed in rural occupations.

2.4. Macroinvertebrate sampling and identification

Macroinvertebrates were used to determine the importance of urban ponds for biodiversity and as biological indicators of habitat quality. Macroinvertebrates are useful in this respect because of their ubiquity, relative ease of sampling and the variety of tolerance levels exhibited by different species (Metcalfe 1989).

Odonata, Trichoptera, Megaloptera, Coleoptera, Malacostraca, Ephemeroptera, Hemiptera, Hirudinea, Mollusca, Tricladida were identified to species (Hammond and Merritt 1983; Elliott *et al.* 1988; Friday 1988; Savage 1989; Gledhill *et al.* 1993; Macan 1994; Edington and Hildrew 1995; Brooks 1997; Elliott and Mann 1998; Reynoldson and Young 2000; Wallace *et al.* 2003; Cham 2007; Cham 2009; Elliott 2009). Diptera, were identified to family level (Tachet *et al.* 2002) and Sphaeridae and Oligochaetes were counted as such. Some trichopteran species (*Oxyethira spp.* and *Hydroptila spp.*) were identified to genus level due to a lack of published information to enable identification to a lower taxonomic level.

Table 2.3. Summary of GIS derived measures. Measures in bold were used as an 'urbanisation indicator' for assessment of urbanisation

Measure	Scale or Unit	Source(s)
Landscape metrics		
Arable land	**Fraction (0 - 1)**	**LCM2007**
Broadleaved woodland	Fraction (0 - 1)	LCM2007
Coniferous woodland	Fraction (0 - 1)	LCM2007
Improved grassland	Fraction (0 - 1)	LCM2007
Suburban	**Fraction (0 - 1)**	**LCM2007**
Urban[1]	**Fraction (0 - 1)**	**LCM2007**
Rough grassland	Fraction (0 - 1)	LCM2007
Ground level vegetation (0m)	Fraction (0 - 1)	NDVI, photogrammetry
Low level vegetation (≤3m)	Fraction (0 - 1)	NDVI, photogrammetry
Mature vegetation (>3m ≤60m)	Fraction (0 - 1)	NDVI, photogrammetry
All vegetation	Fraction (0 - 1)	NDVI
Simpsons land-use diversity index	Fraction (0 - 1)	LCM2007
Demographic variables		
People in rural employment	**Fraction (0 - 1)**	**Census 2001**
Population density	**People ha^{-1}**	**Census 2001**
Housing density	**Households ha^{-1}**	**Census 2001**
Roads[2]	**Fraction (0 - 1)**	**OS MasterMap**
Towns and villages[3]	**Fraction (0 - 1)**	**OS MasterMap**
Physical variables		
Impervious surface	**Fraction (0 - 1)**	**OS MasterMap**
Other water[4]	Fraction (0 - 1)	OS MasterMap
Woodland[5]	Fraction (0 - 1)	OS MasterMap
Building intensity	**Unitless (min. 0, max. 2.64)**	**OS MasterMap, photogrammetry**
Distance to nearest town hall	m	GIS

[1] Urban cover is included as a physical and landscape metric

[2] Combined OS MasterMap data: Motorways, A, B and minor roads

[3] Combined OS MasterMap data: Buildings, gardens, developing land, steps, pavements, paths, man-made open space, man-made landforms and structures

[4] Combined OS MasterMap data: Ponds, lakes, canals, streams, rivers

[5] Combined OS MasterMap data: natural roadside areas with trees, mixed natural habitat with trees and natural railway features with trees

Table 2.4. Variance explained by, and variables correlated to axes 1 and 2 of a principal coordinates analysis (PCA) of 10 'urbanisation indicators' at different buffer distances (Pearson's correlation, r ≥ 0.30, unless stated in parenthesis).

Spatial extent	Variance explained		Explanatory variables	
	Axis 1	Axis 2	Axis 1	Axis 2
25m	39.9	24.1	+ve: Impervious surface, towns & villages, building intensity, suburban -ve: Rural employment	+ve: Urban (0.05)* -ve: Housing density, population density, rural employment, suburban
50m	42.7	23.1	+ve: Impervious surface, towns & villages, building intensity, suburban -ve: Rural employment	+ve: Urban (0.19)* -ve: Housing density, population density, rural employment
75m	43.9	23.7	+ve: Impervious surface, towns & villages, building intensity, suburban, roads -ve: Arable (0.28)*	+ve: Urban (0.22)* -ve: Housing density, population density, rural employment
100m	43.2	23.0	+ve: Towns & villages, impervious surface, suburban, building intensity, roads -ve: Arable	+ve: Population density, housing density, rural employment -ve: Urban (0.21)*
200m	44.3	20.6	+ve: Towns and & villages, impervious surface, suburban, building intensity, roads -ve: Arable	+ve: Urban -ve: Population density, housing density
300m	47.0	18.3	+ve: Towns and & villages, impervious surface, suburban, building intensity, roads -ve: Arable	+ve: Urban, rural employment -ve: Population density, housing density
400m	48.3	19.4	+ve: Impervious surface, towns & villages, suburban, road	+ve: Rural employment, urban -ve: Population density,

Spatial	Variance explained		Explanatory variables	
			surface, building intensity -ve: Arable	housing density
500m	51.5	20.3	+ve: Impervious surface, towns & villages, building intensity, roads, suburban -ve: Arable	+ve: Rural employment, Urban -ve: Population density, housing density
600m	52.7	21.0	+ve: Impervious surface, towns & villages, building intensity, roads, suburban -ve: Arable	+ve: Rural employment -ve: Population density, housing density
700m	53.5	20.7	+ve: Impervious surface, towns & villages, building intensity, roads, suburban -ve: Arable	+ve: Rural employment -ve: Population density, housing density
800m	53.9	20.8	+ve: Impervious surface, towns & villages, building intensity, roads, suburban -ve: Arable	+ve: Rural employment -ve: Population density, housing density
900m	53.8	21.1	+ve: Impervious surface, towns & villages, roads, building intensity, urban -ve: Arable	+ve: Rural employment -ve: Population density, housing density
1000m	54.2	21.5	+ve: Impervious surface, road surface, towns & villages, building intensity, urban +ve Arable	+ve: Rural employment -ve: Population density, housing density

3

A descriptive and historical overview of the Birmingham ‘pondscape’

Chapter 3: A descriptive and historical overview of the Birmingham 'pondscape'

3.1. Introduction

The loss of pond habitats has been reported across the U.K. (Biggs *et al.* 2005), Europe (Indermuehle *et al.* 2008) and globally (Huang *et al.* 2012). As land-use intensifies during urbanisation many ponds are no longer required for their original purpose and are lost to development (Boothby 1999). Furthermore, natural processes of pond creation are much reduced (Williams *et al.* 1998a; Indermuehle *et al.* 2008; Williams *et al.* 2010). Nevertheless, artificial habitats can form suitable surrogates where natural habitats are lost (Eversham *et al.* 1996; Bilton *et al.* 2001; Le Viol *et al.* 2009).

Biggs *et al.* (2005) described pond loss in the U.K. from 1880 to 2000, and revealed a mean annual decline in pond numbers of 0.27% per year i.e. a 32% loss of ponds over the period, although the rate of pond loss decreased in recent years. Other national studies have found annual pond loss to range from 0.78% to 1.23% (see Table 3.1), whilst regional studies report similar trends with losses ranging from 2.6% per year in Huddersfield (Wood and Barker 2000) to 0.04% in Edinburgh (Jeffries and Mills 1990). It is evident from the studies in Table 3.1 that this trend has occurred in both urban and rural areas, the largest loss occurring in London (>90% loss in numbers between 1870 and 1984). These estimates are also likely to mask a relatively high turnover of sites where ponds are lost and gained (Williams *et al.* 1998b). By contrast, pond numbers in the most recent U.K. countryside survey (Williams *et al.* 2010) are estimated to have increased by 12.5% between 1998 and 2007, with an 18000 loss and 70600 gain in pond numbers due to the creation of new ponds associated with leisure activities such as on golf courses. For similar reasons a net gain in pond numbers was estimated in south-east Northumberland over the last 20 years, despite considerable pond loss (30%) between the mid-nineteenth century and 1992 (Jeffries 2012). This study focuses on habitat loss in the pond network within the heavily urbanised city of Birmingham, U.K., between *ca*1904 and 2009 (105 years).

Table 3.1. Estimated pond loss within U.K. regions

Region	Period	Total loss (%)	Annual loss (%)	Change in numbers	Land-use	Source
Huddersfield	1985 – 1997	31	2.60	60 – 42	Urban/ industrial	Wood *et al.* 2003
North Leicestershire	1934 – 1979	60	1.33	958 – 370	Mostly pasture	Beresford and Wade 1982
Bedfordshire	1910 – 1981	82	1.15	Not given	Intensive arable	Beresford and Wade 1982
Sussex	1977 – 1996	21	1.10	33 – 26	Pasture (dewponds)	Beebee 1997
London region	1870 – 1984	>90	0.79	16000 – 1600	Mixed	Langton 1985
Huntingdonshire	1890 – 1980	56	0.68	Not given	Mixed	Beresford and Wade 1982
Cheshire	1870 – 1993	61	0.50	41564 – 16728	Rural and urban	Boothby and Hull 1997
Essex (selected areas)	1870 – 1989	55– 69	0.46 - 0.58	1366 to 423 - 616	Mixed	Heath and Whitehead 1992
Cambridgeshire	1840/90 – 1990	68	0.45 – 0.68	Not given	Intensive arable	Jeffries and Mills 1990
Leicestershire	1840/90 – 1990	60	0.40 – 0.60	Not given	Intensive arable	Jeffries and Mills 1990
Durham	1840/90 – 1990	41	0.27 – 0.41	Not given	Arable and pasture	Jeffries and Mills 1990
Clywd	1840/90 – 1990	32	0.21 – 0.32	Not given	Arable and pasture	Jeffries and Mills 1990
Midlothian	1840/90 – 1990	23	0.15 – 0.23	Not given	Arable and pasture	Jeffries and Mills 1990
Edinburgh	1840/90 – 1990	6	0.04 – 0.06	Not given	Urban	Jeffries and Mills 1990
SE Northumberland	1846/69 – 2005/2008	+15.8	+0.10 - 11.6	222 - 257	Mixed	Jeffries 2012
England and Wales	1880 – 1920	57.5	1.41	800000 – 340000	Mixed	Rackham 1986
United Kingdom	1990 – 1996	7.4	1.23	230600 – 228900	Mixed (lowland ponds)	Williams *et al.* 1998b
United Kingdom	1900 - 1990	75	0.78	1189200 – 297300	Mixed	Bailey-Watts *et al.* 2000
United Kingdom	1998 - 2007	+12.5	+1.4	425000 - 478000	Mixed	Williams *et al.* 2010

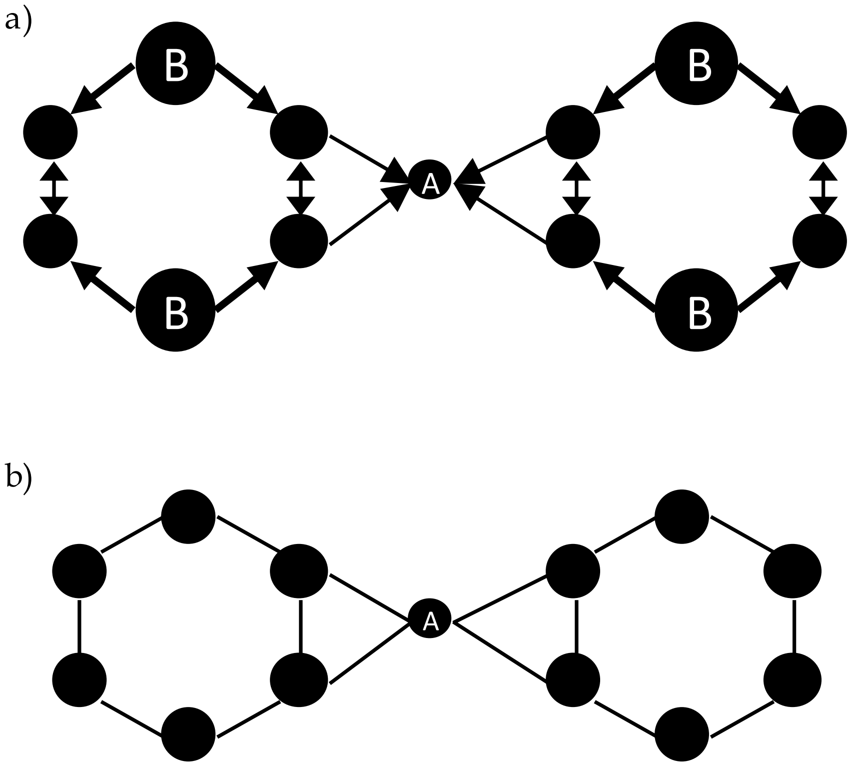
Table adapted from Wood *et al.* 2003

Ponds can be regarded as aquatic islands within a terrestrial landscape. The pond landscape, or 'pondscape' (Boothby 1996), therefore, is a naturally fragmented one. Pond-dwelling fauna need to disperse between habitat patches to acquire resources, avoid predators and other agents of mortality, avoid competition and seek out conspecifics for mating (Fahrig 2007). The spatial configuration of ponds, therefore, is a necessary consideration for the conservation of pond biodiversity (Biggs *et al.* 1994; Boothby and Hull 1997; Briers 2002; Lundkvist *et al.* 2002; Jeffries 2005) and is affected by pond loss and creation, which at any one time may be subjected to localised, rather than uniform changes due to anthropogenic activities i.e. it is not a static patchwork (Jeffries 2012). Nonetheless, the impact of pond loss on aquatic insect dispersal through the landscape is poorly understood (Rundle *et al.* 2002), not least because of the difficulty found in tracking small and short-lived animals (Bohonak and Jenkins 2003) over relatively large areas.

Many aquatic insect populations that are spatially distributed across the landscape function as metapopulations (Jeffries 1994; Briers and Warren 2000; Caudill 2003; Briers *et al.* 2004). The term metapopulation is used to describe a group of populations of a single species, subject to a temporally varying environment, linked by dispersal (Levins 1969); broadened to include any separate populations linked by dispersal (Hanski and Gilpin 1991). Habitat fragmentation reduces the likelihood of successful dispersal between habitats. Eventually, the requirements for successful dispersal between two neighbouring habitats outstrip the dispersal abilities of organisms present. Thus, multiple metapopulations result, affecting species with limited dispersal ability first. The extent of fragmentation will be contingent on the quality of the remaining pond habitat and that of the surrounding matrix (Jeffries 2005), such that ponds with low habitat quality support few species and an unfavourable intervening matrix increases resistance during dispersal. Isolated systems have fewer connected habitats from which to source recolonising organisms delaying recovery time from deterministic events (Briers and Warren 2000; Petersen *et al.* 2004; Caquet *et al.* 2007).

Two conceptual metapopulation models (Figure 3.1.1) are explicitly or implicitly invoked in a branch of mathematics called network or graph theory (Harary 1969), which is concerned explicitly with connectivity (Urban *et al.* 2009). The first is the source/sink model (Pulliam 1988), where sources are habitats in which the number of births outnumbers fatalities. The opposite trend occurs in sink habitats, which rely on immigration of individuals from source habitats to consolidate their populations. In the absence of data regarding pond productivity for historical landscapes, pond surface area may be used as a proxy as larger water bodies typically support a greater abundance and diversity of species,

Figure 3.1.1. Two metapopulation models invoked in graph theory as suggested by Urban *et al.* (2009; circles represent nodes (ponds) and the lines between nodes represent an exchange of individuals in a metapopulation. a) A digraph to represent source-sink dynamics after Pulliam (1988) where B nodes are net sources (births > deaths), hence larger nodes and node A a net sink (births < death) and therefore a smaller node. Arrows represent the direction of the net flow of individuals b) Spreading-of-risk (den Boer 1968) or long-distance rescue (Brown and Kodric-Brown 1977) where node A is vital for the maintenance of connectivity across the network



in particular macrophytes, which increases habitat complexity (Oertli *et al.* 2002; Sondergaard *et al.* 2005). Urban *et al.* (2009) refer the second model to the 'spreading of risk' (den Boer 1968) or 'long distance rescue' (Brown and Kodric-Brown 1977) models, which are more concerned with long range network connectivity in order to ensure disturbance risk is spread across a many habitat patches, from which an impacted site can source recolonizing individuals. The two models are not competitors (Urban *et al.* 2009) and can be complementary approaches to assess the functionality of a habitat network.

In network theory a graph conceptualizes the landscape as a set of points or 'nodes' connected by edges or 'links', considered to be functional connections. Ponds can thus be conceived of as nodes in the landscape linked by dispersal events that connect their spatially discrete populations. A pond, therefore, would be one node and the dispersal path to another pond a link. A directed graph, or *digraph*, is one whereby weight may be applied to each node (Figure 3.1.1a) e.g. habitat area. A graph in which each node can be reached *via* a link from any other node is connected, while an unconnected graph consists of multiple, separately connected components or subgraphs. The use of network theory in ecology has typically been associated in the formulation of food webs and the linkages between trophic levels (Krause *et al.* 2003; Luczkovich *et al.* 2003), but has found increasing applicability in the field of spatial ecology (Moilanen 2011).

Recently, network theory has been applied to examine patterns of connectivity and to identify key habitat patches and dispersal pathways through the development of several connectivity indices (Bunn *et al.* 2000; Urban and Keitt 2001; Pascual-Hortal and Saura 2006; Saura and Pascual-Hortal 2007; Urban *et al.* 2009). For example, Laita *et al.* (2011) used graph theory to evaluate the performance of legally protected woodlands for connectivity; Gurrutxaga *et al.* (2011) examined the disconnection caused by highways through forest area networks; Decout *et al.* (2012) examined *Rana temporaria* (Linnaeus 1758) distributions in a human-dominated environment and Eros *et al.* (2012) highlighted the potential use of graph theory within riverine systems. The potential, therefore, for network theory to inform landscape managers of key habitats for the maintenance of connectivity within their pond network is clear (although see Moilanen 2011 for limitations).

Many metrics can be calculated from network graphs, each representing a specific measure that captures a different aspect of connectivity (Laita *et al.* 2011). The different aspects of commonly used connectivity metrics are explored by Pascual-Hortal and Saura (2006) and Saura and Pascual-Hortal (2007). The 'integral index of connectivity' (IIC, Pascual-Hortal and Saura 2006; Saura and Pascual-Hortal 2007) integrates the amount of available habitat and the functional connections between habitats in a single measure for the entire network. Derived from the IIC are the metrics $\mathrm{BC}\frac{IIC}{k}$ (Bodin and Saura 2010), referred to in this study as $\mathrm{BC}_{\mathrm{IIC}}$ and dIICconnector (Saura and Rubio 2010), referred to here as $\mathrm{IIC}_{\mathrm{c}}$. Both $\mathrm{BC}_{\mathrm{IIC}}$ and $\mathrm{IIC}_{\mathrm{c}}$ have been shown to capture two unique aspects of connectivity provided by network nodes (Bodin and Saura 2010).

The $\mathrm{BC}_{\mathrm{IIC}}$ metric is a derivation of the better known betweeness centrality (BC) metric (Freeman 1979), but is weighted by node attributes (e.g. surface area) and distance between patches, as well as the number of shortest paths between any two nodes in the network that use the focal patch as a stepping stone (Bodin and Saura 2010). Therefore, the $\mathrm{BC}_{\mathrm{IIC}}$ metric incorporates source-sink (Pulliam 1988) dynamics as well as spatial position. Contrastingly, $\mathrm{IIC}_{\mathrm{c}}$ appears relevant to the importance of a node in spreading-of-risk (den Boer 1968) or long-distance rescue (Brown and Kodric-Brown 1977) models and corresponds to how much a focal habitat patch contributes to connectivity between other patches by serving as a stepping stone, that cannot be fully replaced by other patches in the network (Bodin and Saura 2010). As a non-weighted metric, IIC_c avoids the tendency of some metrics to simply attribute higher value to the largest nodes in the landscape (Ferrari *et al.* 2007). Therefore, two metrics can be used together to identify priority sites for the provision of connectivity (IIC_c) and ecological flux (BC_{IIC}).

The IIC metric considers the availability of functional connections i.e. links between habitat patches. A traditional measure of functional connectivity is a maximum dispersal threshold of a studied organism (Adriaensen *et al.* 2003). A distance threshold is applied which, if exceeded, two nodes are considered to be isolated from each other. Alhough ideal in principle, dispersal thresholds for many animals are poorly understood and difficult to establish as they require data that are prohibitively time-consuming or expensive to collect (Kupfer 2012). Estimations of maximum dispersal distance from mark-recapture studies may also be biased by the number of individuals collected and the size of the study area (Franzen and Nilsson 2007). Alternatively, several studies have applied a series of dispersal thresholds that cover a range of dispersal abilities present within an ecological community (Hinsley and Bellamy 2000; Gurrutxaga *et al.* 2011; Laita *et al.* 2011). In doing so, conservation initiatives can be informed according to potential functional connectivity of the studied landscape from a multi-species point of view (Hinsley and Bellamy 2000), rather than by the needs of a single species. A flexible approach such as this can highlight habitat patches that are essential for the maintenance of landscape connectivity for members of the ecological communities that vary in dispersal ability.

This study combines the data gathered from a historical audit of the Birmingham pond network to analyse how the potential connectivity of the pond network has changed due to a loss in pond numbers and habitat area within a graph theoretic framework. The land-uses attributable to habitat loss and those land-uses that support the highest density of ponds in the contemporary landscape are also assessed. Finally, network theory metrics are used to identify ponds key to the maintenance of the pond network.

3.1.1. Objectives and hypotheses

This chapter characterises changes in the Birmingham pond network over approximately 105 years, covering three periods in time. The decline in pond numbers, area of available habitat and the connectivity between habitat patches is analysed with three specific objectives and hypotheses.

1. To quantify the change in pond numbers and pond area between *ca*1904, *ca*1962 and 2009.

   Hypothesis 1

   Pond loss between ca1904 and 2009 is extensive. The majority of ponds will have been lost between ca1904 and ca1962 due to urban development during this period

2. To review the main drivers of pond creation and the impact of land-use types on pond creation and loss.

   Hypothesis 2

   The majority of ponds have been created for agricultural purposes and have since been lost due to urbanisation and changes in farming practices. The degree of urbanisation is a key determinant of pond density

3. To assess how pond losses affect connectivity and ecological landscape resilience for organisms with differing dispersal abilities, and, highlight key sites for the ecological resilience of the contemporary Birmingham pond network in relation to aquatic organisms with differing dispersal abilities.

   Hypothesis 3

   The Birmingham pond network is more disconnected than ever before, with the distance between ponds increasing to an extent that could affect aquatic macroinvertebrate dispersal potential, particularly for those with poor dispersal abilities

3.2. Methods

3.3. Digitisation of the historical pondscape

Data for historical analyses of the pondscape in the Birmingham administrative area (267.8km^2) were derived from two historical map layers, accessed and downloaded in individual 1km^2 raster tiles from EDINA Digimap and processed in ArcGIS 10 (ESRI 2011).

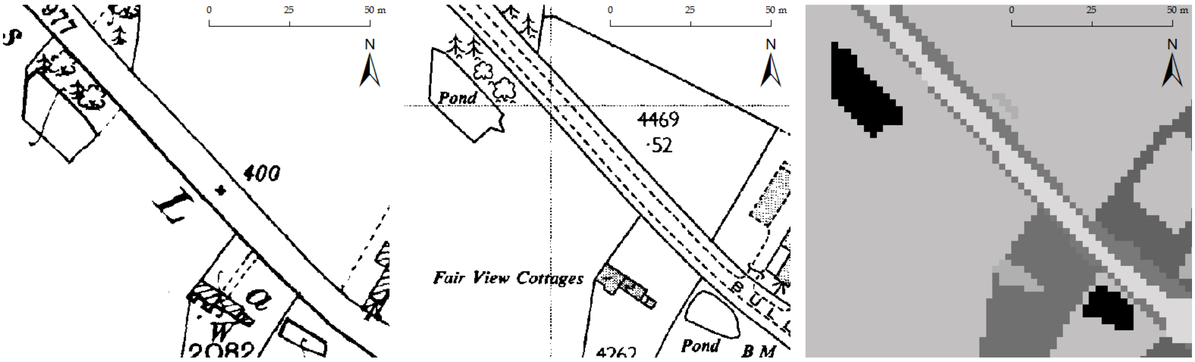
The County Series, 1st revision maps (EDINA Historic Digimap Service 2012b), within Birmingham, were published between 1904 ±2 years (Appendix 5) and are referred to as *ca*1904. The County Series maps were drawn at two scales, 1:10560 (6 inch to the mile) and 1:2500, of which the latter has been used, and mapped landscape features with an area of 16m^2 or larger, although smaller, isolated or significant features may also be mapped (Oliver 2005). Therefore, the majority of ponds should be recorded, although a single 1km^2 map tile was unavailable.

The National Grid overhaul and re-survey (Edition A) maps (National Grid EDINA Historic Digimap Service 2012a), at 1:2500 scale, were published between 1943 and 1995 and superseded those of the County Series after the Second World War. Within Birmingham, the maps were published between 1944 and 1977 (33 years), although the majority (97%) were published between 1951 and 1973 (22 years, Appendix 5). The mean year of publishing for these maps was 1962, thus they are referred to as *ca*1962.

The contemporary, 2009, dataset was derived from Ordnance Survey MasterMap (Ordnance Survey (GB) 2008), which was subsequently edited using a combination of expert knowledge made available through the Birmingham and Black Country Wildlife Trust, preliminary field visits (conducted between 18th February 2009 and 1st May 2009) and aerial imagery (Google Inc. 2009) (Section 2.1).

Each historical map layer was digitised using a GIS. Ponds on each map were drawn digitally as both a single point and polygon using ESRI's ArcScan tool pack extension. Water bodies were included that were annotated on the maps as 'ponds', 'fish ponds', 'mill ponds', 'lakes' and 'pools' and where water body area was < 2ha. Water features annotated as 'reservoirs, 'swimming pool' 'bathing pool' were not included unless annotated as disused and likely to have developed to a more naturalised state. Similarly, those within the grounds of a sewage treatment works, for example, 'sludge lagoons' and 'tanks' were ignored unless disused. Ponds present in the *ca*1904 maps were not always annotated; consequently, curvilinear features or those too small to constitute field margins were classed as ponds (e.g. Figure 3.3.1). In addition, where possible, locations of ponds within a given dataset were cross-checked with the other data layers to guide pond locations where uncertainty arose, for example, where a pond feature appeared to be drawn on the map, but not annotated. Similarly, map annotations were also used to record a broad description of the land use surrounding the pond.

Figure 3.3.1. An example of pond continuity throughout the study period and the historical maps utilised for the pond audit



a) *ca*1904* b) *ca*1962 c) 2009 (black)

*Note the lack of annotation in the *ca*1904 map

3.3.1. Land-use correlates to pond density

In order to test the relationship between land-use coverage and pond density within the contemporary landscape, the number of ponds within 1 km squares was related to the percentage coverage of different land-use types derived from MasterMap and LCM2007 coverage data and to population density (Section 2.2). Pond density per 1km square was quantified in ArcGIS 9.3 (ESRI 2008) using the package Hawth's Tools (Beyer 2004) points in polygon tool. The thematic raster summary tool in Hawth's Tools was used to extract the proportion of land-use classified as arable land, woodland, improved grassland, suburban, urban and impermeable surface within each 1 km square, as well as population density.

Due to the uncertainties over the shape of relationships between land-use cover and pond density, generalised additive models (GAMs) were used. GAMs use a smoothing curve to model the relationship between the explanatory variable and the response variable and allow for non-linear relationships (Zuur *et al.* 2007). GAMs were constructed in the package 'mgcv' (Wood 2011) in R statistical package version 2.15.1 (R Core Team and contributors worldwide 2012). As the seven land-use predictors were not normally distributed, cubic regression splines were fitted to each using a fixed amount of smoothing (k = 4) in order to capture trends with the least numbers of degrees of freedom (Fewster *et al.* 2000). This was achieved by producing multiple plots of the smoothing curve for different degrees of freedom and visually inspecting each curve (Zuur *et al.* 2009).

Where a 1 km square did not have any land-use coverage for those investigated, that data row was omitted from the analysis to avoid severe over-dispersion in the dataset due to the occurrence of zero values. GAMs were fitted to a negative-binomial distribution with log link. To improve the robustness of the analyses, the spatial coverage was extended to include Sandwell, Dudley, Wolverhampton and Walsall i.e. BBC, for which centroids (i.e. not polygons) of ponds in the contemporary pond network were available (Hewitt 2011).

3.3.2. Connectivity analysis of the historical pondscape and identification of priority ponds

The equivalent connected area index (EAC) (Saura *et al.* 2011) was used to assess whole scale pond network connectivity change over time and across different dispersal thresholds (discussed below). This can be interpreted as the effective amount of available habitat across a landscape with respect to connectivity. Here it was calculated based on binary inter-patch connections, i.e. patches are either connected or not, which is derived from the integral index of connectivity (IIC) (Pascual-Hortal and Saura 2006). The IIC is given by:

$$IIC = \frac{\sum_{i=1}^{n}\sum_{j=1}^{n}\frac{a_i . a_j}{1+nl_{ij}}}{A_L^2} = \frac{IICnum}{A_L^2} \qquad \textbf{3.3.1}$$

And EAC is given by:

$$ECA = \sqrt{\sum_{i=1}^{n}\sum_{j=1}^{n}\frac{a_i \times a_j}{1+nl_{ij}}} \qquad \textbf{3.3.2}$$

Where a_i and a_j are the areas of the individual habitat patches and nl_{ij} is the number of links comprising the shortest path between patches i and j. A_L is the study area, regardless of habitat. In this instance A_L is given as the area of the Birmingham administrative area, 267.79km^2. The total number of nodes in the landscape is given by n. When patches i and j are disconnected $nl_{ij} = \infty$, when $i = j$, $nl_{ij} = 0$. The IIC index ranges from 0 to 1 and increases as connectivity improves.

Two metrics were then used to establish the importance of individual nodes towards landscape connectivity. The first metric BC_{IIC} (Bodin and Saura 2010) is a generalised version of the well established betweeness centrality metric (BC_k) (Freeman 1979). BC_k is the sum of all shortest pathways between all pairs of nodes within a graph that pass through a given node as a proportion of all shortest paths. BC_k is given by:

$$BC_k = \sum_i \sum_j \frac{g_{ij}(k)}{g_{ij}} \qquad \textbf{3.3.3}$$

Where $g_{ij}(k)$ is the sum of all shortest paths that go through node k, and g_{ij} is the total number of shortest paths between all pairs of nodes. BC_k does not consider individual node attributes such as habitat quality. Surface area, which was made available through the historical pond auditing process, was used as a proxy for pond productivity due to its relationships with a number of ecological aspects (Oertli *et al.* 2002; Sondergaard *et al.* 2005),. The BC_{IIC} metric was proposed by Bodin and Saura (2010) in order to assess the ecological flux (flow of organisms), and not just dispersal possibility, through a patch. It is given by:

$$BC_{IIC} = \mathrm{BC}\genfrac{}{}{0pt}{}{IIC}{k} = \sum_i \sum_j a_i a_j \left(\frac{a_i \times a_j}{1 + nl_{ij}}\right) \{i, j \neq k \; and \; if \in nm^*\} \qquad \textbf{3.3.4}$$

Where nm^* represents the shortest pathways of i and j (i ≠ j) that utilise node k as a stepping stone in the landscape prior to the removal of node k i.e. those pairs of nodes whose connection is favoured by the presence of k (Bodin and Saura 2010). Consequently, the BC_{IIC} metric considers more central those nodes that serve as stepping stones between large habitat patches, it is furthermore weighted by the length of the path between the nodes, favouring shorter paths as more likely to be fulfilled by dispersing organisms.

The IIC metric (Eq. 3.3.1) can be used to inform the value of a node in the landscape by analysing the difference in IIC before and after the removal of a focal node. This is termed $dIIC_k$ and can be further broken down in to three components, dIICintra, dIICflux and dIICconnector (IIC_c), as described by Saura and Rubio (2010), where the d refers to the difference in the given component. The latter component IIC_c provides information regarding the role of individual nodes as irreplaceable providers of connectivity between other habitat areas (Gurrutxaga et al. 2011). A given habitat patch will only contribute to $dIIC_k$ through IIC_c when it is part of the shortest path for dispersal between any other two nodes.

3.3.2.1. Network analysis

First, minimum spanning trees of each historical pond network were established to delineate the skeleton of the network and the 'backbone' that provides most of the traversibility (Urban and Keitt 2001). A minimum spanning tree links all nodes with the lowest accumulated distance (or cost) that visits every node on a graph once only i.e. does not allow for cycles (refer to Table 3.2 for terms). This analysis provides basic information on the threshold at which the network becomes fully connected (the percolation threshold). This is the longest, shortest path between two nodes in the network. Minimum spanning trees were produced using the EDENetworks: Ecological and evolutionary networks (Kivela *et al.* 2011) program, freely available from http://becs.aalto.fi/edenetworks/. The program also calculates basic BC_k values for each node using geographic coordinates of the pond centroids.

Two dispersal distance thresholds were applied to the historical pond networks in order to assess the potential implications for metapopulations of organisms with differing dispersal capabilities. For aquatic insects, very few studies of dispersal abilities have been carried out. Despite spatial correlation in community structure existing between pond sites up to 13km apart (Briers and Biggs 2005), such community similarity may persist through the small percentile of aquatic insect populations that move considerably longer distances than the majority (e.g. Conrad *et al.* 1999; Angelibert and Giani 2003; Purse *et al.* 2003; Briers *et al.* 2004; Macneale *et al.* 2005). A more frequent exchange of individuals is required to recolonise ponds that have suffered a deterministic event or pulse disturbance. Therefore, the shorter 500m and longer 1500m dispersal distances are selected in light of those reported from previous studies listed in Table 3.3. The two dispersal thresholds are thought to represent the dispersal distances regularly achieved by the majority of aquatic insects with better and worse dispersal capabilities. Network connectivity metrics were calculated using the Conefor Sensinode 2.6 software package (Saura and Torne 2009), freely available from http://www.conefor.org/. The metrics calculated by Conefor Sensinode are wide-ranging, however, those used in this study are indicated in Table 3.2.

Finally, the two metrics BC_{IIC} and IIC_c were used in a complementary fashion to highlight those pond sites in the contemporary landscape which serve as important nodes for the connectivity of the pond network (IIC_c) and for the provision of ecological flux (BC_{IIC}). This was performed at eight different dispersal thresholds to analyse the connectivity of the landscape from the perspective of a range of organisms with differing dispersal abilities, i.e. from a multi-species point of view (Hinsley and Bellamy 2000). The dispersal distances investigated were 250m, 500m, 1000m, 1500m, 2000m, 2500m, 5000m and 10000m. These dispersal thresholds were considered to be widely applicable to the dispersal abilities exhibited by aquatic insects in previous studies (Table 3.4). The 20 highest scoring ponds for

BC_{IIC} and IIC_c were calculated for each dispersal distance and established as the candidate priority ponds that are integral to the network. From these, a sub-group of sites were selected that had markedly higher metric scores. These were considered priority ponds, although the approach may be used to highlight more or less sites as required.

3.3.3. Potential habitat quality

Single measurements of pond surface area and percentages of surface area classified as shaded or riparian vegetation were derived from a combination of Normalised Difference Vegetation Index (NDVI) and photogrammetrically derived height GIS layers (Section 2.3.1.6) against digital pond outlines.

Table 3.2. Definition of graph (or network) theory terms and metrics used here and associated references

Term	Definition
Component	A sub-graph within a network that is not fully connected
Cycle	A closed path of three or more nodes
Minimum spanning tree[1]	A spanning tree with the lowest accumulated distance (or cost) that visits every node on a graph
Path	A path is a walk in which no node is revisited
Spanning tree	A tree that visits every node on a graph
Tree	A tree is a path without cycles and with only one link between any pair of nodes

Metrics	Definition	Reference(s)
IIC	Integral index of connectivity. A habitat availability (reachability) index based on a binary network (unweighted links) as the underlying model of the fragmented landscape. This metric uses estimates of possibilities for dispersals between all pairs of patches. Hence, if direct dispersal between any two patches in the landscape is assessed as being possible, the link strength is set to unity. Otherwise, it is set to 0 (i.e. no link is assigned between the two patches).[2]	Pascual-Hortal and Saura (2006)
EAC	Equivalent connected area. The size of a single habitat patch (maximally connected) that would provide the same value of the *IIC* metric as the actual habitat pattern in the landscape.[2]	Saura *et al.* (2011)
BC_k	Betweenness centrality. The sum of all shortest pathways between all pairs of nodes within a graph that pass through a given node (*k*) i.e. how much it serves as an intermediate stepping stone.[2,1]	Freeman (1979)
$dIIC_k$	The loss of habitat availability (*IIC*) caused by the removal of patch *k*.[2]	Saura and Rubio (2010)
$BC\frac{IIC}{k}$ (BC_{IIC})	A metric that takes into account patch areas and geographical distances between patches instead of only the number of shortest paths as in BC_k. In this way, this generalized metric assigns more weight to the paths that are expected to carry larger flows of organisms and that connect bigger and therefore likely more ecologically important patches.[2]	Bodin and Saura (2010)
$dIICconnector_k$ (IIC_c)	The fraction of $dIIC_k$ (the loss of habitat availability caused by the removal of patch *k*) corresponding to how much patch k contributes to connectivity between other patches by serving as an intermediate stepping stone (connecting element) that cannot be fully replaced by other patches in the network. A high value implies that the loss of k would severely reduce the connectivity between other habitat patches.[2]	Saura and Rubio (2010)

Programs used to calculate metric: [1] EDENetworks (Kivela *et al.* 2011) [2] Sensinode Conefor 2.6 (Saura and Torne 2009)

Table 3.3. Summary of dispersal abilities of some lentic macroinvertebrates

Order	Family	Species	Method	Water body type and landscape	Dispersal distances	Source
Odonata	Coenagrionidae	*C. puella* *C. pulchellum* *E. cyathigerum* *I. elegans* *P. nymphula*	MRR	Marl pits; arable and livestock; U.K.	<1% > *ca* 600m (all species)	Conrad *et al.* 1999
	Lestidae	*L. sponsa*				
	Libellulidae	*S. sanguineum*			Max. 1.2km	
Odonata	Coenagrionidae	*C. puella* *C. scitulum*	MRR	Ponds; Arid, karstic;	0.9% >725m 1.5% >725m	Angelibert and Giani 2003
	Libellulidae	*L. depressa*		France		
Odonata	Coenagrionidae	*C. mercuriale*	MRR	Water meadow carriers & ditches; Meadow, agriculture, urban; U.K.	1.3% >500m 0.1% >1000m 65.7% <50m	Rouquette and Thompson 2007
Odonata	Coenagrionidae	*I. pumilio*	MRR	Springs & flushes; New Forest / Red River valley (rural-residential & agricultural); U.K.	67.9% <50m / 87.6% <50m Max. (♀) 575m /263m	Allen and Thompson 2010
Odonata	Coenagrionidae	*E. cyathigerum*	MRR	Not stated	27% >100m	Garrison 1978 (in Allen and Thompson 2010)
Hemiptera	Notonectidae	*N. maculata* *N. obliqua*	Exhaustive sweep-netting, seasonally	Dewponds; Peak District National Park U.K.	Max. >1.6km	Briers 1998
Diptera	Culicidae	Not stated	Not stated	Not stated	Max. <5km Max. <1km	Service 1997 (in Bohonak and Jenkins 2003)
Trichoptera	Hydropsychidae	*C. camplyal* *C. speciosa* *H. phalerata* *H. hageni* *M. zebratum*	Malaise traps	Lake St. Clair & Detroit River; Cropland; Canada	502m[1] 488m[1] 1.47km[1] 1.18km[1] 1.558km[1]	Kovats *et al.* 1996
Ephemeroptera	Ephemeridae	*Hexagenia spp.*			900m[1] Max. >5km	Kovats *et al.* (1996)

[1] Distance travelled by 50% of collected animals (median)

MRR – Mark-release-recapture NR – Not reported Max. – Study maximum

3.4. Results

3.4.1. Characterisation of the historical pondscape

Between *ca*1904 and 2009, there was an 82% reduction in the number of ponds within Birmingham (Table 3.4), leaving a total of 341 ponds in the contemporary (2009) landscape from a total of 1914 present in *ca*1904. Many of these ponds were lost before *ca*1962, by which time the number of ponds in Birmingham had fallen to 524, a 74% reduction. Pond density declined over the study period, from 7.1 ponds per km^2 *ca*1904, to 2.0 by *ca*1962 and 1.3 in 2009.

Over time, there has been an increase in the mean surface area of ponds, reflecting the loss of the smallest water bodies, especially between *ca*1904 and *ca*1962 (Mann-Whitney, $P < 0.05$, Figure 3.3.1, Table 3.4) and less pronounced between *ca*1962 and 2009 (Mann-Whitney, $P = 0.12$). The mean size of ponds retained in the landscape has increased over time (by 204% since *ca*1904). The total available pond habitat in Birmingham has declined (by 46% since *ca*1904); however, the retention of larger ponds has offset the rate of habitat reduction (Figure 3.4.1). As a proportion of the entire Birmingham administrative area, ponds cover 3.6%, 2.2% and 2.0% in *ca*1904, *ca*1962 and 2009 respectively.

As suggested by Williams *et al.* (1998b), the total number of ponds masks a considerable amount of turnover within the pond stock. The number of ponds that persisted throughout the study was 171, representing half (50.1%) of those in the contemporary landscape and 8.9% of the number present in *ca*1904. Of those ponds present in 2009, 173 (50.7%) were present in *ca*1904, (two were absent in *ca*1962), 58 (17.0%) were present by *ca*1962 and 110 (32.3%) created between *ca*1962 and 2009. Therefore, 49.9% (170) of the ponds present today have been created since *ca*1904. Although there was not a significant difference between the time periods (ANOVA, $P = 0.845$), the average surface area of the ponds that were present in each of the study periods decreased over time (Table 3.5). This may suggest natural successional processes or the reduction of pond size for developmental or management purposes. In the contemporary landscape, analysis of the proportion of pond surface area as vegetation >0m (i.e. non-floating) using NDVI data (Section 2.3.1.4) suggested that many of the 171 persistent ponds are at risk of filling-in. Forty-one percent (70) had vegetation coverage in excess of 75%, indicating that they are heavily shaded, the mean surface area of which was 269m^2. Fewer ponds created since *ca*1904 had >75% vegetation coverage (28.6%), however, considerably more had surface area ≤ 269m^2 (58.7%) than persistent ponds. Therefore, many of the older, persistent ponds are at risk of infilling and, despite less tree cover currently, the smaller size of ponds constructed since *ca*1904 may result in more rapid successional processes.

There is a clear relationship between pond surface area and proportion of tree coverage in those ponds that were present throughout the study ($r^2 = 0.43$, Figure 3.4.3a), although the pattern is still present in ponds that have been created since *ca*1904, the trend is much less clear ($r^2 = 0.06$, Figure 3.4.3b). This may furthermore suggest vegetation development over a longer period of time, for example, the majority of persistent ponds <2500m² appear to have high vegetation cover (Figure 3.4.3a), whereas the spread of vegetation coverage is much wider in ponds < 2500m² created since *ca*1904 (Figure 3.4.3b). It is also clear that larger size ponds in the contemporary pondscape are remnants of past pond networks, with only a handful of larger sites >1ha in surface area being created since *ca*1904.

Whilst it was not possible to discern the creational process or reason for construction of many of the ponds in Birmingham, it was possible to establish the origins of 334 ponds present at some point since *ca*1904 to 2009, including those that have since been lost (Table 3.6). Approximately one quarter (24.2%) originated from extraction activities such as brick and gravel workings and 19.9% were built for ornamental purposes, which were typically small (mean surface area 101m²). Ornamental ponds were those with distinctly geometric shapes in close proximity to buildings. The largest ponds were typically associated with industrial processes such as mill ponds and disused reservoirs and sludge lagoons. Very few ponds appeared to have been created through natural processes, however, 20 (6% of those with origins identified) were created from cut-off channels and stream backwaters.

Table 3.4. Declines in pond loss and pond density and changes in average pond surface area and total pond habitat availability within Birmingham

	*ca*1904	*ca*1962	*ca*2009
Number of ponds	1914	524	341
Ponds /km²	7.1	2.0	1.3
Mean surface area	508.7	1123.0	1546.0
Total surface area	982870	593894	528704

Table 3.5. Changes in mean pond surface area and total habitat of persistent ponds i.e. present throughout the study period (n = 171)

	*ca*1904	*ca*1962	*ca*2009
Mean surface area	2143.3	1934.6	1919.0
Total surface area	366500	330809	328157

Figure 3.4.1. Changes in surface area of ponds in Birmingham between *ca*1904 and 2009. Boxplots display minimum and maximum, 1st and 3rd quartile and median value for each period. Lettering denotes significant differences (Mann-Whitney, $P < 0.05$, used due to non-homogeneity of variances)

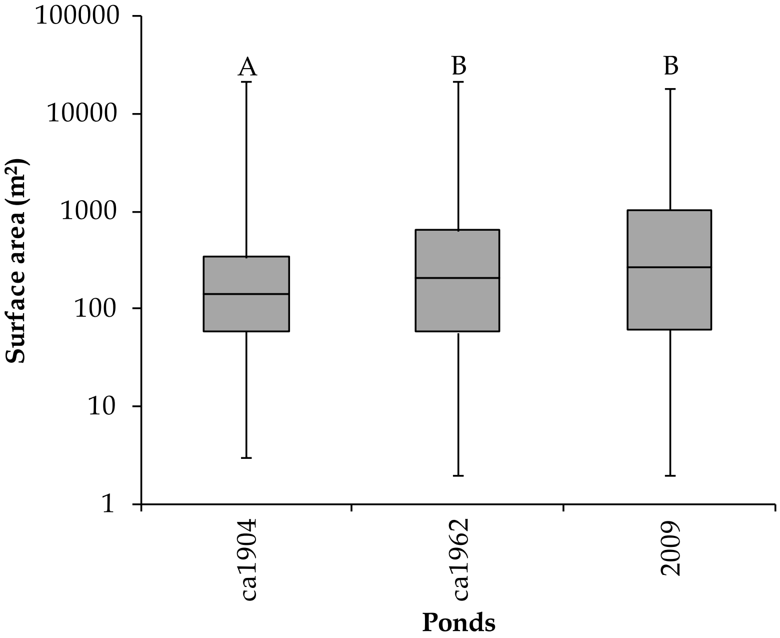


Figure 3.4.2. Decline in total available pond habitat and pond numbers between *ca*1904 and 2009 within Birmingham

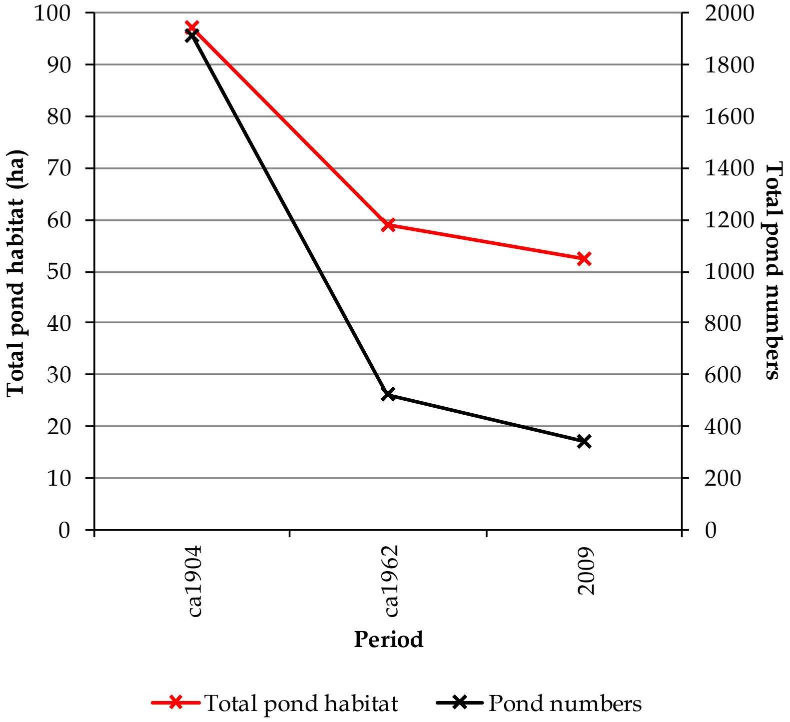
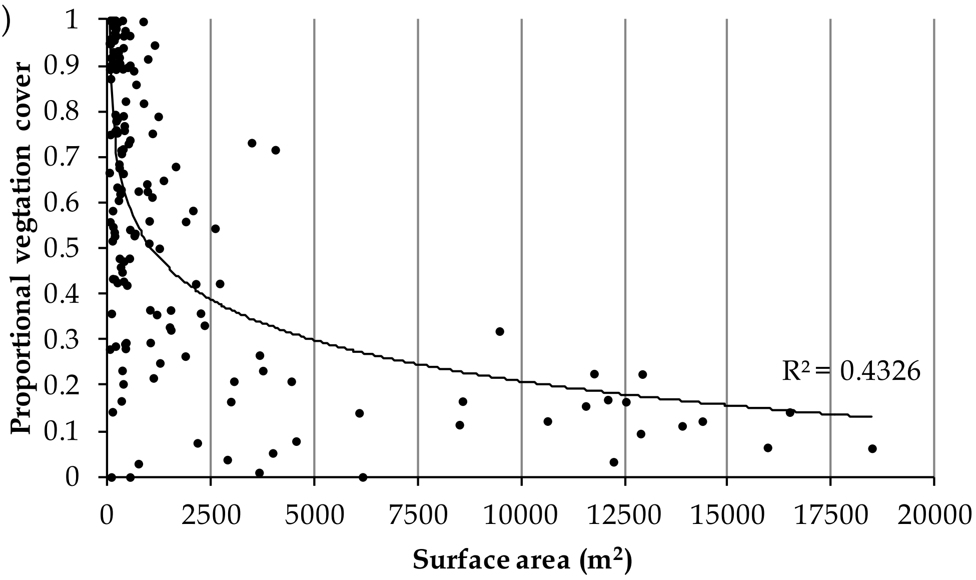
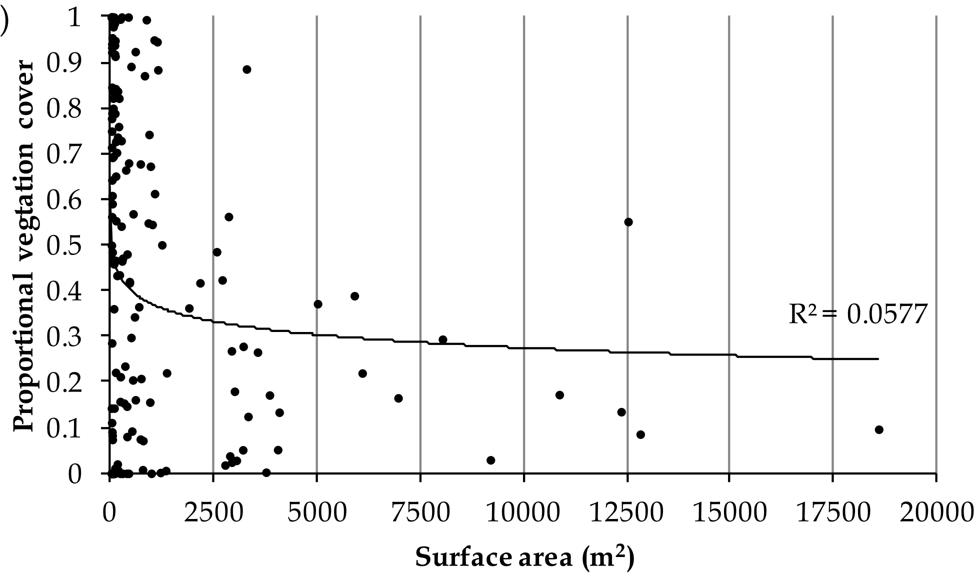


Figure 3.4.3. The relationship between pond surface area and vegetation >0m (i.e. non-floating) coverage in a) ponds present throughout the study, from *ca*1904 – 2009 b) pond present since *ca*1904

a)



b)



A categorisation of land-use associated with pond presence was gathered using land-use symbols present on the County (*ca*1904) and National Grid (*ca*1962) map series (Table 3.7). Many ponds were originally associated with farmland, however, the number of such ponds decreased rapidly from 71% of total pond numbers to just 13% in 2009. This is likely to reflect an overall change in land-use within Birmingham as the proportion of land in agricultural use has much reduced. Conversely, the number of ponds associated with or enveloped by suburban areas and areas of grassland (including public parkland and golf courses) increased by 2009 to account for 59% of those present, from 3% in *ca*1904. The number of ponds associated with urban / industrial land-use is consistently low, at no point accounting for more than 8.2% of ponds present. Land-use relationships with pond density within the 2009 landscape are further explored in section 3.4.2.

In *ca*1904, three distinct patches of high pond density were apparent within to the northeast, east and south of Birmingham centre (Figure 3.4.4). However, the numbers of ponds in these areas are much diminished by 2009, particularly to the east and represent areas of major pond loss (Figure 3.4.5). In some instances, pond losses are up to 30 ponds lost per 1km^2 (Figure 3.4.5c). Interestingly, although much overshadowed by the impact of urban expansion affecting the wider pondscape, several new ponds have been created as part of modern developments since *ca*1962 that marginally improve pond density within some central areas of Birmingham.

3.4.2. Land-use correlates to pond density

The relationships between land-use parameters and pond density in the contemporary BBC landscape were explored using GAMs (Figure 3.4.6). The explanatory power of the models ranged considerably between 6.1% deviance explained (urban) to 24.7% (impermeable surface) (Table 3.8).

Pond density peaks at approximately 40% and 30% coverage of arable land and woodland respectively before declining (Figure 3.4.6), although fewer 1 km^2 have coverages beyond 60% arable and 30% wooded land. Thus, at low levels of arable land and woodland, the influence of other land-uses such as impermeable surfaces, cause pond density to decline. At mid-range coverages, the land-use is less intense and more akin to open countryside, which allows for more natural processes of pond creation to occur as well as providing protection against development for ponds which are no longer required for their original purpose. At high coverage, agricultural intensification and dense woodland are more likely to cause infilling, causing pond density to decline.

Table 3.6. Pond origins (n=334)

	Origin										Mineral extraction						
	Boating lake	Cut-off channel	Disused reservoir	Disused sludge lagoon	Fish pond	Golf course	Mill pond/ stream	Moat	Ornamental	Others	Brickworks	Clay pit	Coalfields	Gravel pit	Marl pit	Sand pit	Unspecified quarry/ pit
n	6	20	4	27	48	6	21	39	67	15	42	4	1	15	2	5	12
% of total	1.8	6.0	1.2	8.1	14.3	1.8	6.5	11.9	19.9	4.5	12.5	1.2	0.3	4.5	0.6	1.5	3.6
Average m^2	3567	590	5318	3639	4246	1116	5287	1167	101	3683	663	684	72	5295	5287	293	331

Table 3.7. Land-use associated with ponds during each period

Land-use	*ca*1904	*ca*1962	2009*
Urban/ industrial	0.034 (66)	0.049 (26)	0.082 (28)
Suburban	0.026 (50)	0.126 (67)	0.342 (117)
Rural	0.090 (174)	0.128 (68)	-
Farmland	0.709 (1,366)	0.317 (170)	0.129 (44)
Allotments	0.004 (7)	0.034 (18)	-
Orchard	0.010 (19)	0.006 (3)	-
Woodland	0.089 (171)	0.149 (80)	0.178 (61)
Open land/ scrub	0.013 (26)	0.058 (31)	0.018 (6)
Grassland	0.020 (39)	0.123 (66)	0.251 (86)
Marsh	0.004 (7)	0.013 (7)	-

Number of ponds within each land-use are given in parenthesis
*Land-use classifications given by Land Cover Map 2007 in the contemporary landscape, which excludes rural, allotments, orchard and marsh

Figure 3.4.4. Number of ponds per km² within each period a) *ca*1904 (County series) b) *ca*1962 (National Grid series) c) 2009 (Contemporary)

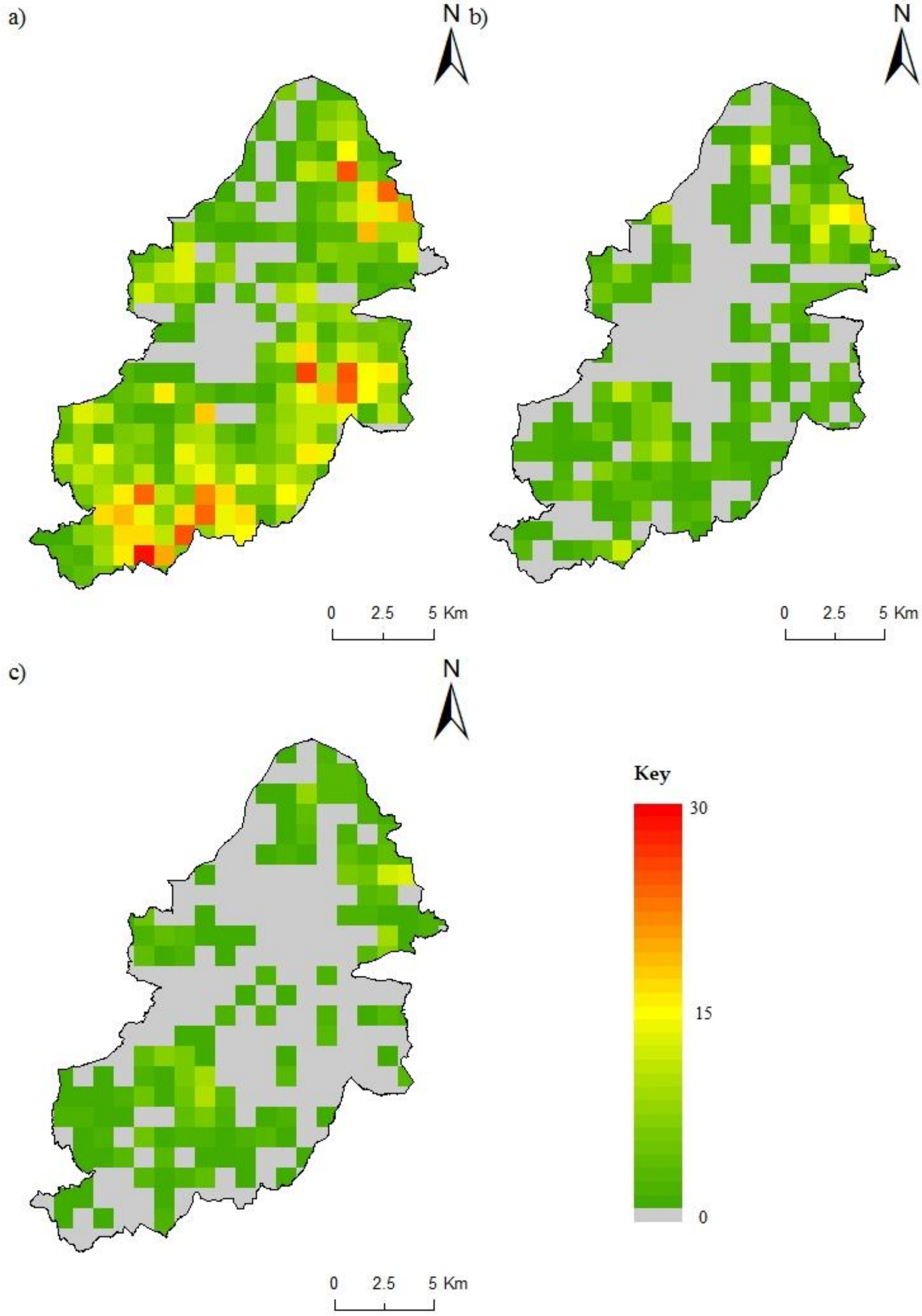


Figure 3.4.5. Change in pond numbers per km^2 between periods a) *ca*1904 to *ca*1962 b) *ca*1962 to 2009 c) *ca*1904 to 2009

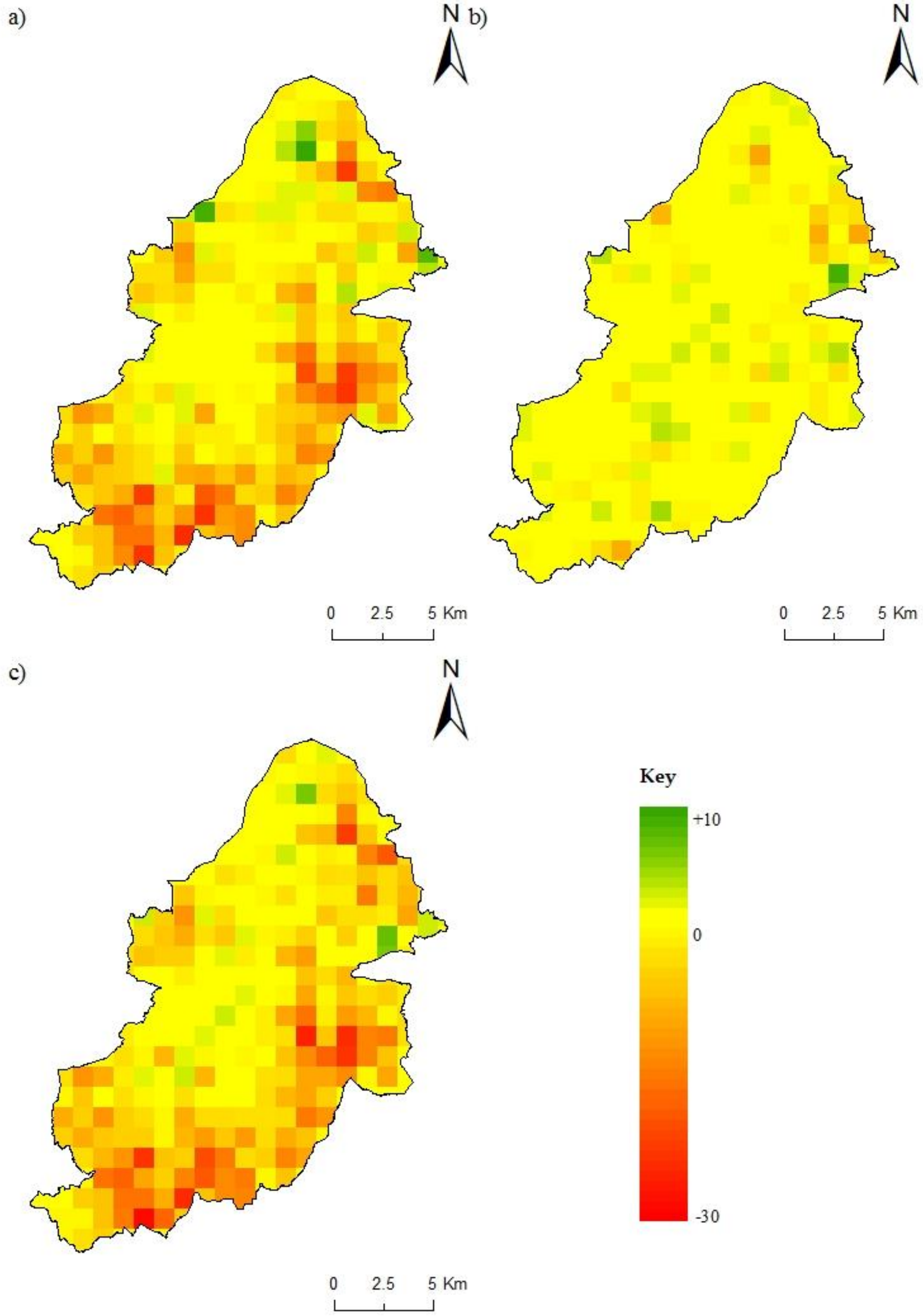


Table 3.8. Median (min and max) of predictor variables and results of generalised additive models (GAMs) describing pond density in terms of land-use and population density

Predictor	Median (min – max)	F statistic	Deviance explained (%)
Arable	0.12 (0.00 – 0.80)	9.0***	13.2
Woodland	0.02 (0.00 – 0.66)	11.6 ***	11.3
Improved grassland	0.10 (0.00 – 0.69)	22.7***	10.1
Suburban	0.59 (0.00 – 0.99)	42.7***	9.8
Urban	0.13 (0.00 – 0.97)	18.8***	6.1
Population density	33.5 (1.1 – 87.6)	10.7***	7.6
Impermeable surface	0.27 (0.00 – 0.69)	27.2***	24.7

Cases excluded list-wise where zero values occurred in predictor, *** $P < 0.001$

Approximately linear relationships exist between the remaining land-uses and pond density (Figure 3.4.6). The proportion of improved grassland is positively correlated to pond density, which includes public parks and golf courses, within which ponds are often built as features. The proportions of suburban, urban and impermeable surface have clear negative correlations with pond density. Therefore, in more developed areas pond numbers decline rapidly. Similarly, population density also has a negative relationship to pond density. The slight upturn at the tail end of the relationship between impermeable surface and pond density is likely reflective of the few newly created ponds within central areas such as those noted in Figure 3.4.5.

3.4.3. Connectivity analysis of the historical pondscape

3.4.3.1. Minimum spanning trees

A minimum spanning tree was created for each historical pond network, which is the spanning tree with the shortest total length of edges, which only allows single linkages i.e. one node to one node (Figure 3.4.7, for terms see Table 3.2). For the Birmingham pond network the minimum spanning tree historically routed outside of the central area of Birmingham through areas of higher pond density to the south and to the east (Figure 3.4.7a & b). However, due to the loss of ponds to the east, and a slight gain in of the centre of Birmingham (Figure 3.4.5), the 2009 minimum spanning tree re-routes through the central area (Figure 3.4.7c).

Figure 3.4.6. Fitted smoothing functions from generalised additive models (GAMs), showing the relationship between land-use coverage and pond density (ponds /km^2) within Birmingham and the Black Country GAMs fitted to a negative-binomial distribution, km squares with 0 coverage of land-use were omitted from the analyses. *Arable* Arable land, *Wood* Woodland (coniferous and broadleaved), *Improved* Improved grassland, *Suburban* Suburban land, *Urban* Urban land, *PopDens* Average population density, *ImpSurf* Impermeable surface. Shaded areas represent ±1 SE, *y* axis, e.g. s(*x*, 2.17) = smoothing estimator with 2.17 d.f.

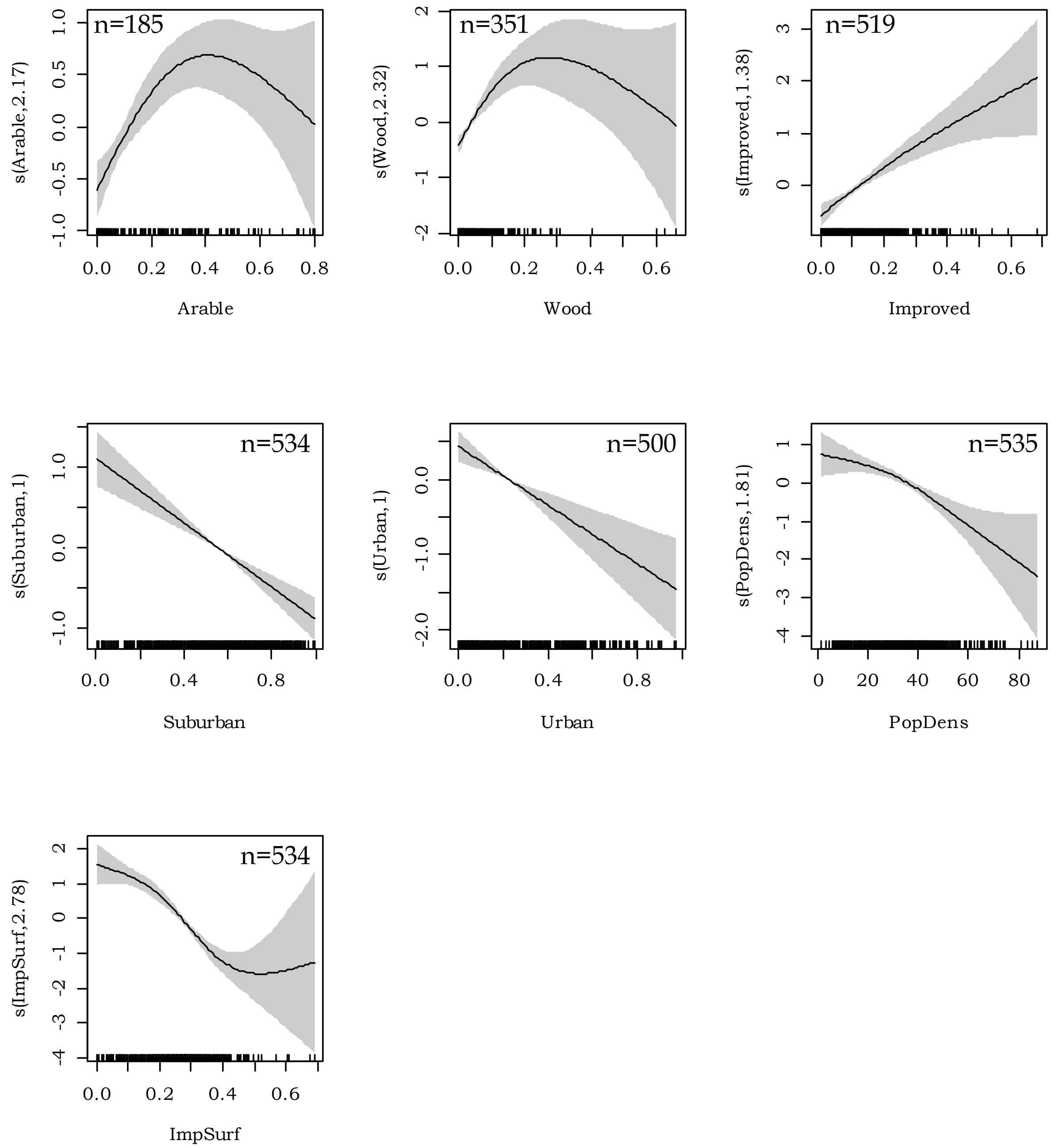
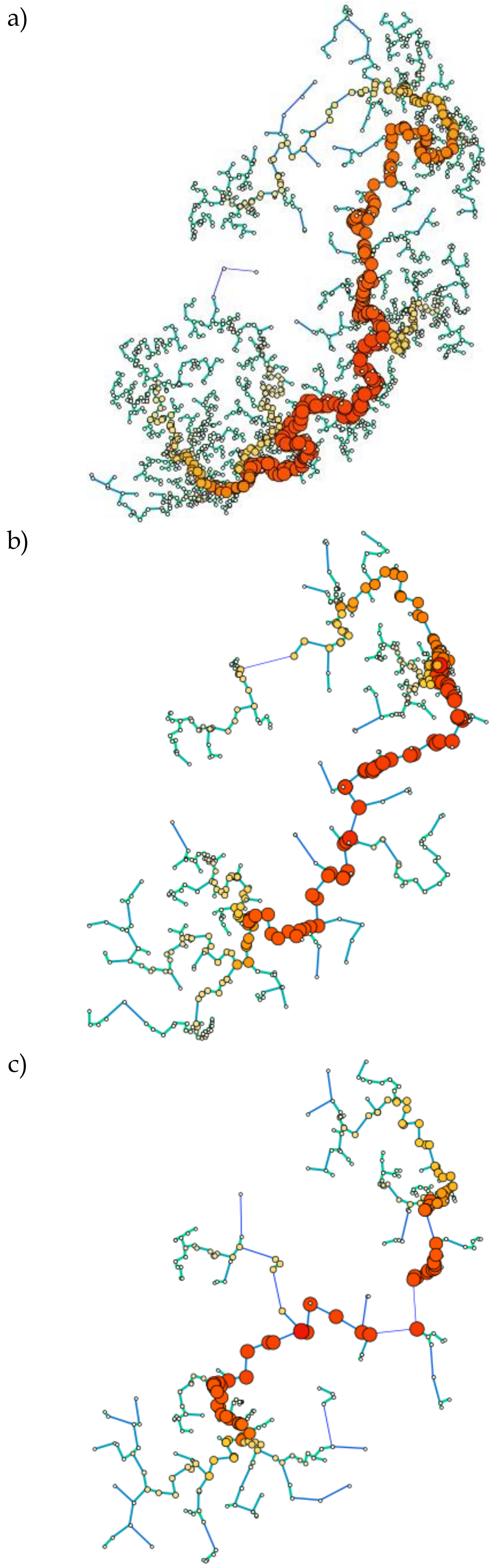


Figure 3.4.7. Minimum spanning trees of the changing Birmingham pond network, with nodes sized in proportion to their betweeness centrality. Larger nodes have higher centrality and highlight the pattern of flow across the landscape (after Bodin and Norberg 2007) a) *ca*1904 b) *ca*1962 c) 2009



The increase in average link length in the minimum spanning tree is indicative of reducing pond density over time, although the shorter distance required in the 2009 compared to *ca*1962 to uphold a fully connected network i.e. maximum edge distance, suggests a more even spread of pond sites (Table 3.9). Nonetheless, it is clear that fewer sites sustain the minimum spanning tree since *ca*1904 (Figure 3.4.7b and c), i.e. there are very few spatially redundant ponds in the contemporary network. An increasing reliance on individual ponds is also clear from the increase in mean and maximum BC node values when expressed as a proportion of total paths between nodes that the entire network supports (Table 3.9). The greater resilience within the *ca*1904 network is reflected in the steep slope of the percolation line (Figure 3.4.8); *ca*1962 and 2009 networks are similarly vulnerable below a threshold distance of approximately 1500m.

3.4.3.2. Threshold analysis

Two threshold distances were applied to each of the historical pond networks to represent habitat availability for species with weaker (500m) and stronger (1500m) dispersal abilities. The thresholds were based on the results of previous aquatic insect dispersal studies (Table 3.3). The equivalent connected area (EAC) decreased over time due to direct habitat loss and the loss of connecting nodes (stepping stones) across the network. Between the 1500m and 500m dispersal thresholds, EAC decreased by 47.5%, 55.5% and 59.6% in *ca*1904, *ca*1962 and 2009 respectively, suggesting less resilience to habitat loss (Table 3.10). The greater reliance of individual nodes as stepping stones is also shown by an increase in the BC_k metric when expressed as a proportion of the sum of all BC_k metrics ($\bar{x}$ BC_k %) and by an increase in the average IIC_c value suggesting that fewer nodes are spatially positioned to compensate for the loss of others. Consequently, the relative numbers of components increase over time, whilst the number of nodes per component and the size of the largest component decrease.

A dispersal threshold of 1500m maintains the pond network at a high level of connectivity, thus 99.9% of the ponds in the *ca*1904 landscape were connected in a single component at this threshold. Similarly, lowering of the dispersal threshold causes considerable disconnection within the subsequent networks, such that the number of components increase 24 fold in *ca*1962 (120) and 8 fold in 2009 (96) when the dispersal threshold is lowered to 500m. The degree of connectivity maintained with a dispersal threshold of 1500m helps to explain trends observed in mean BC_{IIC}, which over time decrease with a 500m threshold and increase with a 1500m threshold.

The BC_{IIC} metric combines source-sink dynamics as well as the importance of a pond as a stepping stone within a network (Table 3.2). At a 500m dispersal threshold, larger ponds that theoretically provide greater ecological flow i.e. habitat sources, become more isolated over time. This is indicated by ponds with above median surface area having a relatively lower BC_k metric over time in comparison to those with less than median surface area (Appendix 6). The opposite trend is apparent if the dispersal distance is set to 1500m such that larger sites remain connected and relatively more so than smaller sites. Therefore, the greater connectivity provided by a 1500m dispersal threshold maintains the connection with larger habitats such that they contribute to increase mean BC_{IIC} values i.e. at dispersal thresholds <1500m more large sites become isolated relative to the number of smaller sites over time.

3.4.4. Identifying key habitats for connectivity

Ponds were identified in the contemporary network that were important for connectivity and ecological flow. In order to do so, two metrics were used in a complementary fashion. The first, BC_{IIC}, represents the usage of a habitat patch as a stepping stone and is weighted by habitat area (as a proxy for productivity) and the second, IIC_c, considers the spatial position of the patch and the degree to which the loss of the patch can be compensated by others; no weight is attributed to the node. This analysis was carried out using a range of dispersal distances in order to cover a range of dispersal abilities belonging to members of the aquatic community. Dispersal thresholds were applied of 250m, 500m, 1000m, 1500m, 2000m, 2500m, 5000m and 10000m.

The contemporary pond network experienced considerable changes with decreasing threshold distance (Figure 3.4.10 & Figure 3.4.11). The networks largely comprised of isolated patches at the shortest distance (250m) and as a completely connected network beyond 2500m (i.e. a single component, Table 3.11). At 250m an average of 1.9 ponds made up each component and 118 (35% of total) were completely isolated. The largest improvement in overall landscape connectivity was evident when the threshold was increased from 500m to 1000m where the percentage increase in EAC peaked at 38%. This indicates that, in its current spatial configuration, a large proportion of ponds become functionally connected to a greater number of sites when the dispersal threshold is increased from 500m to 1000m.

The 20 highest scoring ponds for BC_{IIC} and IIC_c were calculated for each dispersal distance (Appendix 7), from these, a sub-group of sites were selected that had markedly higher metric scores and were considered priority ponds (Table 3.11), added weight was given to ponds which had both a high BC_{IIC} and IIC_c metric, which are listed in bold in Table 3.11. Between two and 10 ponds were selected as priority ponds at each threshold distance

Table 3.9. Minimum spanning tree network summaries (measured from pond centroids)

	County series	National Grid	Contemporary
Average edge distance	204.3	358.6	459.4
Maximum edge distance	1591.8	2661.6	2360.8
Average BC %	0.0005	0.0019	0.0029
Max BC %	0.0038	0.0096	0.0146

Figure 3.4.8. Network percolation; the relative vulnerability of historical pond networks to decreasing distance thresholds (measured from pond centroids), as indicated by the impact upon the greatest (functionally) connected component (GCC).

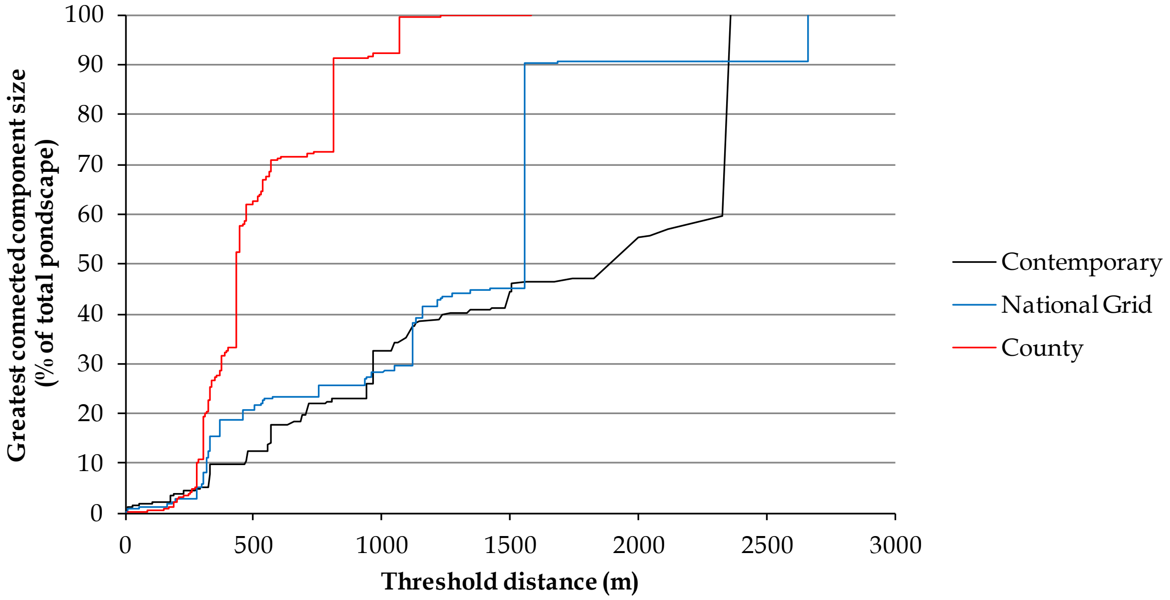
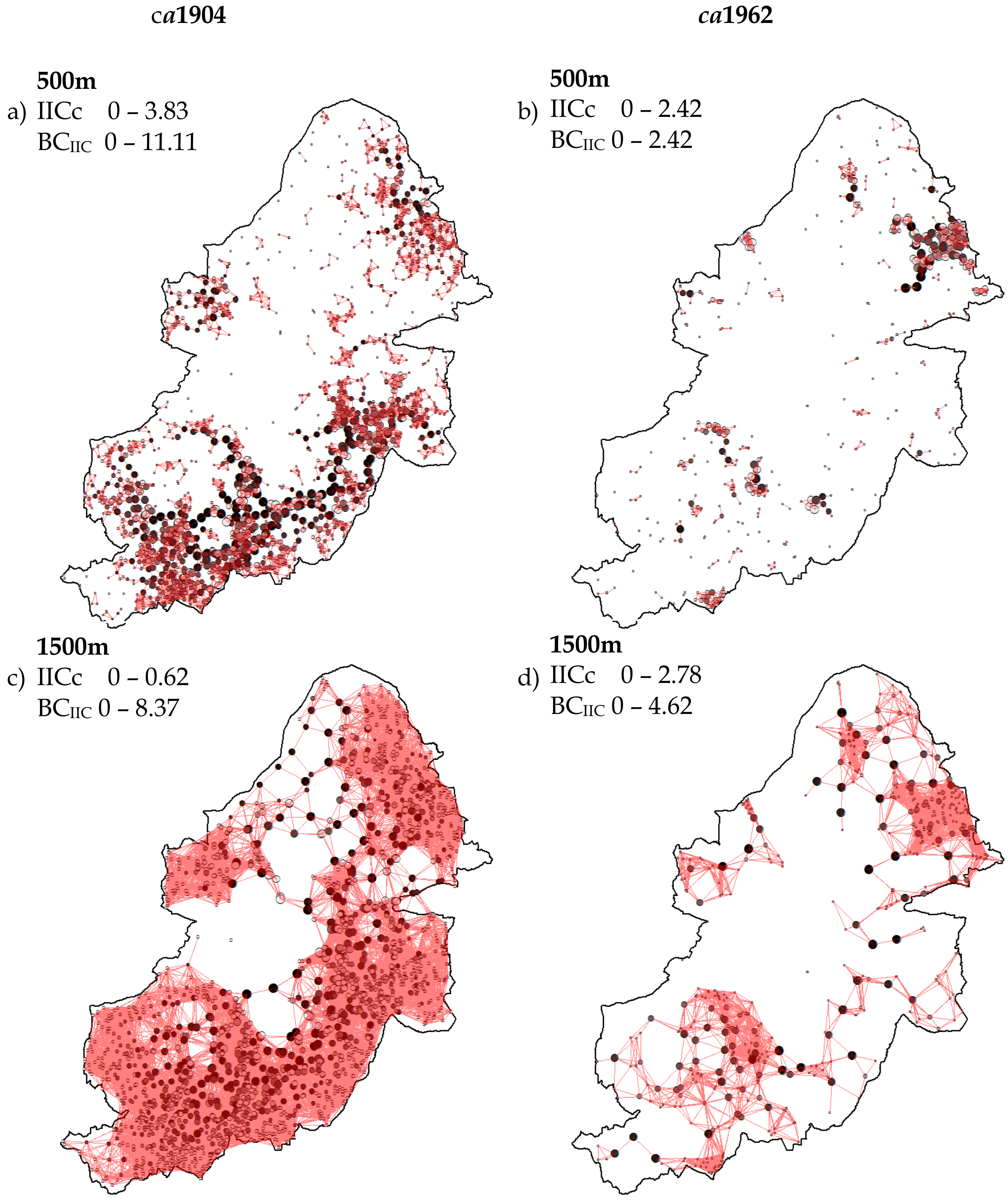


Table 3.10. Thresholded network summaries

	Shorter dispersal (500m)			Longer dispersal (1500m)		
	*ca*1904	*ca*1962	2009	*ca*1904	*ca*1962	2009
EAC	181550	100199	86392	382116	181905	144910
$\bar{x}$ BC$_k$ %	0.0000287	0.0019083	0.0029326	0.0005224	0.0019083	0.0029325
$\bar{x}$ IIC$_c$	0.003	0.016	0.021	0.003	0.019	0.080
$\bar{x}$ BC$_{IIC}$	0.407	0.057	0.048	0.229	0.367	0.508
Links	9795	1358	661	65234	6402	2915
Components (%)*	70 (3.7%)	125 (23.9%)	108 (31.7%)	2 (0.1%)	5 (1.0%)	12 (3.5%)
Nodes/component (%)*	27.3 (1.4)	4.2 (0.8)	3.2 (0.9)	957 (50.0)	104.8 (20.0)	28.4 (8.3)
Largest component (%)*	1230 (64.3)	115 (21.9)	42 (12.3)	1913 (100)	237 (69.5)	157 (46.0)

*Percentages given in parentheses are calculated relative to the total number of nodes i.e. the maximum possible number of components in a completely disconnected network

Figure 3.4.9. Historical pond networks of Birmingham. Ponds are shown as circles proportional to their BC_{IIC} value and are increasingly dark according to their dIICconnector (dIICc) value. Red lines represent links (functional connections) between nodes that are below a distance threshold for aquatic insects with shorter (500m) and longer (1500m) dispersal traits.



and between 0 and six were highlighted as having high BC_{IIC} and IIC_c metrics. Few ponds were a repeated priority across different dispersal thresholds, although ponds 43, 186, 218, 340, were. The use of the BC_{IIC} metric did not simply highlight larger ponds as the range of surface areas were ranked between 13 (12216m^2) to 340 (2m^2) from the 341 ponds. Area appeared to have the greatest influence on priority pond selection at the shortest dispersal threshold (250m) where most ponds were isolated. At 250m the selected pond areas ranged between rank 12 (12345m^2) to rank 75 (1320m^2). Beyond this, spatial location and the ecological flux of the adjoining links and nodes became more influential. The value of the IIC_c metric reduced beyond 2500m as the entire network became connected and, from a spatial configuration perspective, an increasing number of ponds were able to compensate for the loss of others. By 10000m, only 24 nodes (7%) had a IIC_c value > 0, suggesting a high degree of spatially redundant nodes.

Habitat within the priority ponds was briefly assessed using surface area and vegetation cover. Vegetation cover was split between 0m and 3m to represent low level riparian vegetation, and between 3m and 60m to represent mature tree cover. As established previously (section 3.4.1), larger ponds had reduced vegetation cover. Seven of the 37 priority ponds were below 100m^2 in surface area (Table 3.12), three of which had mature vegetation coverage of >75%, which suggests that, at the time of study, they may have supported few species, with the exception of certain specialist taxa. Nonetheless, mature vegetation cover was often low (mean 17.7%) and ranged considerably (minima 0, maxima, 92%). The potentially most valuable pond for connectivity of the pond network (site 340), as it was identified as a priority at dispersal distances between 1500m – 2500m, is a very small (2m^2) ornamental pond. At the opposite end of the scale three sites are in excess of 10000m in surface area, two are associated with public parks (sites 13, 17) and one with a disused wastewater treatment complex, now with extensive riparian growth (site 12).

Figure 3.4.10. The contemporary pond network of Birmingham. Ponds are shown as circles proportional to their BC_{IIC} (BC_{IIC}) value and are increasingly dark according to their dIICconnector (dIICc) value. Each graph is thresholded at different dispersal distances. Red lines represent links (functional connections) between nodes that are below the distance threshold a) 250m b) 500m c) 1000m d) 1500m

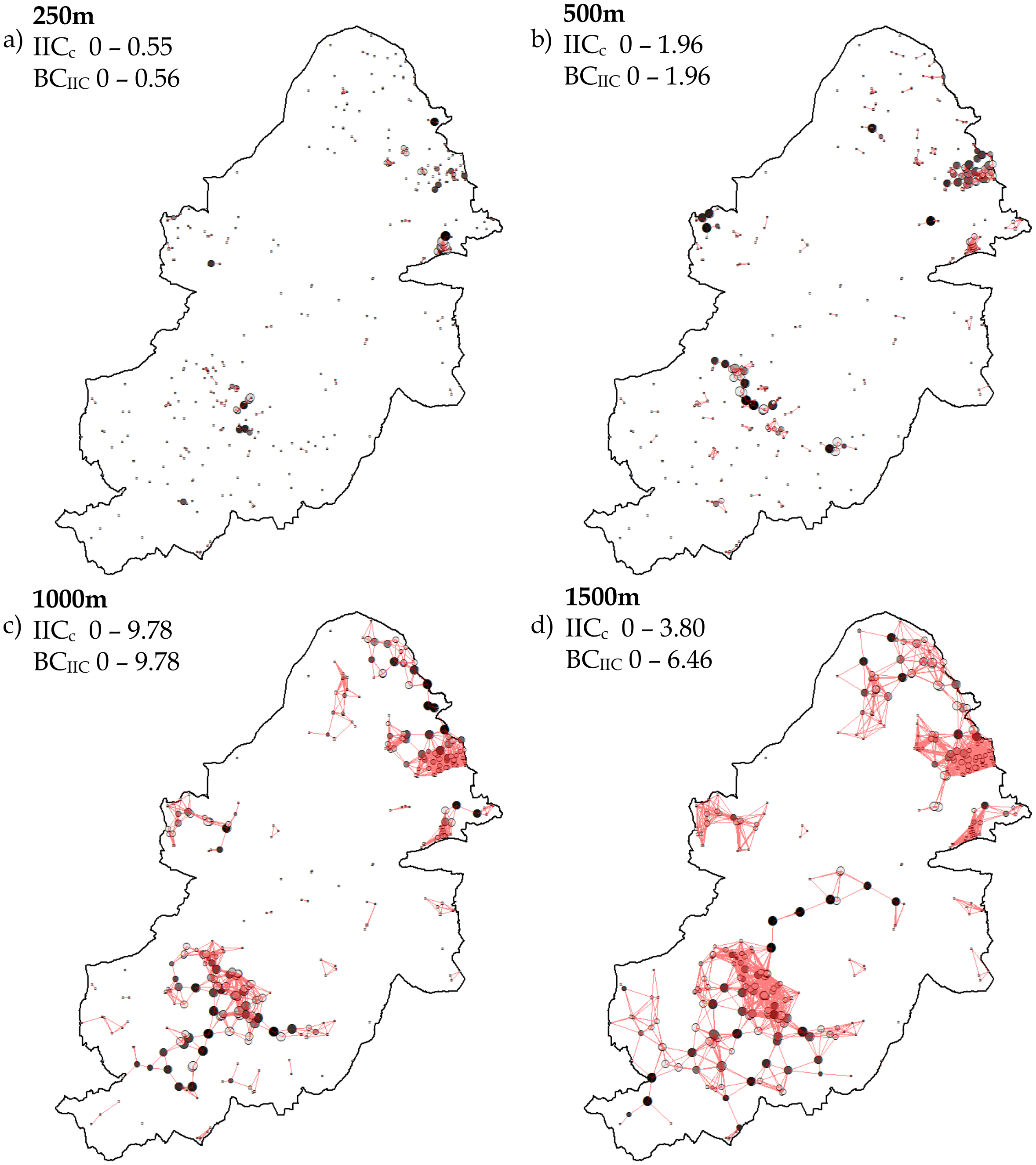


Figure 3.4.11. The contemporary pond network of Birmingham. Ponds are shown as circles proportional to their BC_{IIC} value and are increasingly dark according to their dIICconnector (dIIC) value. Each graph is thresholded at different dispersal distances. Red lines represent links (functional connections) between nodes that are below the distance threshold a) 2000m b) 2500m c) 5000m d) 10000m

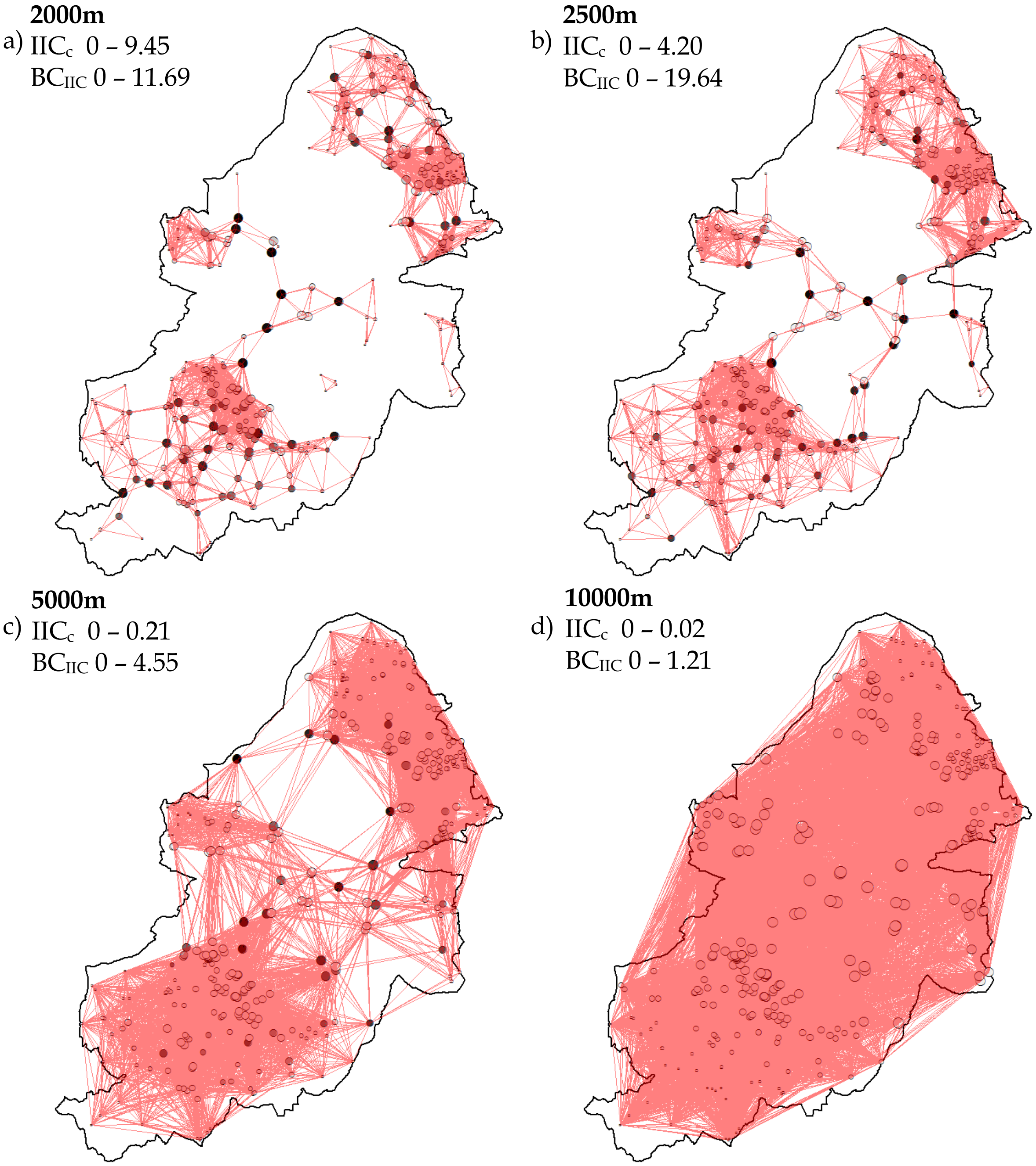


Table 3.11. Overall thresholded network summaries and priority ponds for the maintenance of connectivity and ecological flow across the Birmingham pond network

Threshold (m)	Comp.	Links	Nodes/ Comp.	Max. Comp.	EAC	% inc. EAC	Priority ponds by BC_{IIC}	Priority ponds by IIC_c
250m	309	184	1.9	18	80396	-	32, 36, **75**, 24, 48, 45, 12	75
500m	108	661	3.2	42	86392	7.5	**292**, **321**	292, 321
1000m	35	1663	9.7	111	118952	37.7	**43**, **276,** 201, 108	43, 276
1500m	12	2915	28	157	144910	21.8	276, 43, 93, **340**, **170**, 82, **245**, **161**, **248**, **184**	340, 170, 245, 161, 248, 184
2000m	4	4266	85	190	183768	26.8	340, 79, **220**, **186**, 50	220, 186
2500m	1	5781	341	341	236862	28.9	**17**, 179, 218, 340	17, 44
5000m	1	14601	341	341	297880	25.8	26, 13, 218, 216, 236, 61	1, 340, 170, 27
10000m	1	30086	341	341	341764	14.7	211, 172, 186, 106, 142	All ponds <0.025

Comp Number of components (interconnected sub-networks), *Links* The number of Euclidean paths between nodes below the threshold distance (functional connections), *Nodes/Comp* The average number of nodes per component *Max. Comp* The number of nodes in the largest component *EAC* The size of a single habitat path (maximally connected) that would provide the same value of the IIC metric as the actual habitat pattern in the landscape (Saura and Torné 2012), *% inc. EAC* The percentage increase in EAC as a function of increasing threshold distance. Priority ponds are established as those that are an important stepping stone between other pairs of nodes in the landscape, where more weight is assigned to the links that are expected to carry larger flows of organisms and that connect bigger and potentially more productive ponds (BC_{IIC}). Further prioritisation is given to ponds in **bold** that have a high BC_{IIC} and IIC_c, which indicates those ponds that are important stepping stones between other nodes, the positioning of which is not readily replaced by other nodes in the landscape. Ponds underlined are those that were selected as priority ponds across two or more different dispersal thresholds. See Appendix 7 for the top 20 ranked ponds for each metric at each threshold. Pond numbers represent their surface area rank with one being the largest.

Table 3.12. Habitat characteristic summary of priority ponds sorted by surface area

Priority pond number	Relevant dispersal threshold(s)	Surface area (m^2)	Proportional vegetation cover >3m	Proportional vegetation cover 0m >3m
12	250m	12345	0.000	0.135
13	5000m	12216	0.023	0.011
17	2500m	10848	0.139	0.033
24	250m	6945	0.057	0.109
26	5000m	6076	0.163	0.056
32	250m	4066	0.023	0.111
36	250m	3832	0.005	0.166
43	1000m, 1500m	3312	0.115	0.009
45	250m	3195	0.115	0.009
48	250m	3030	0.000	0.278
50	2000m	2956	0.000	0.028
61	5000m	2149	0.086	0.079
75	250m	1320	0.405	0.013
79	2000m	1190	0.000	0.006
82	1500m	1111	0.000	0.002
93	1500m	949	0.506	0.440
106	10000m	721	0.253	0.420
108	1000m	718	0.000	0.208
142	10000m	381	0.626	0.000
161	1500m	308	0.000	0.081
170	1500m	269	0.000	0.154
172	10000m	261	0.000	0.000
179	2500m	241	0.066	0.413
184	1500m	221	0.471	0.071
186	2000m, 10000m	216	0.189	0.023
201	1000m	171	0.000	0.000
211	10000m	148	0.000	0.006
216	5000m	136	0.000	0.020
218	2500m, 5000m	135	0.000	0.536
220	2000m	130	0.085	0.901
236	5000m	97	0.133	0.570
245	1500m	84	0.717	0.283
248	1500m	78	0.919	0.000
276	1000m, 1500m	39	0.808	0.178
292	500m	25	0.750	0.000
321	500m	12	0.061	0.424
340	1500m, 2000m, 2500m	2	0.000	0.000
	Mean:	2191	0.177	0.152

3.5. Discussion

Pond loss in Birmingham has been extensive. A rate of pond loss in Birmingham between *ca*1904 and 2009 of 0.78% per annum is comparable to that of London (0.79%) between 1870 and 1984 (Table 3.4, Langton 1985), and the 82% total loss in pond numbers between *ca*1904 and 2009 ranks Birmingham second highest, behind urban London and alongside an intensively agricultural landscape (Beresford and Wade 1982); although differences in survey methodologies are apparent (Wood *et al.* 2003). The analysis of three time periods revealed the extent of pond turnover, which may have been masked by a simple account of pond numbers (Williams *et al.* 1998b). Just over two thirds of the ponds present in the contemporary landscape are >50 years old and have considerable vegetation encroachment. Whilst some late stages of succession are likely to benefit specialist taxa, such as Coleoptera (Lundkvist *et al.* 2002), macroinvertebrate and macrophyte diversity has been shown to plateau around six years after pond creation (Williams *et al.* 2008), and younger ponds (six to 12 years old) can be more species rich than older (Williams *et al.* 1998b).

Since *ca*1962, the rate of annual pond loss has declined to 0.1%, consistent with the nationwide disappearance of ponds suggested by Biggs *et al.* (2005). The reduced rate of pond loss may be due to the retention of larger ponds, the longevity of which may be attributable to slow sedimentation rates (Moss 2010) and higher development costs due to their volume. Larger ponds typically have low tree cover and more macrophytes (Oertli *et al.* 2002; Sondergaard *et al.* 2005), which provide habitat for macroinvertebrates, amphibians and fish (Gilinsky 1984; Diehl 1992; Williams 1997; Parris 2006; Scheffer *et al.* 2006). Larger ponds may also support larger populations of aquatic insects, however, they are also increasingly likely to support fish populations (Sondergaard *et al.* 2005), which may constrain taxon richness (Wood *et al.* 2001), particularly among the predatory Coleoptera (Fairchild *et al.* 2000; Hinden *et al.* 2004; Hassall *et al.* 2011), Odonata (McPeek 1990) and Hemiptera (Schilling *et al.* 2009), although the extent of impact is contingent on refugia provided by macrophytes (Gilinsky 1984). However the retention of larger ponds over small, vegetation rich and fishless ponds may impact upon regional biodiversity as such sites can be highly diverse in the absence of fish predation pressure (Scheffer *et al.* 2006). That the smallest ponds are at risk of infilling is not surprising given the dominance of the littoral zone in these systems (by % area) (Declerck *et al.* 2006), which may succumb to vegetation encroachment. However, a large proportion of the pond network comprises of small ponds, in a late successional stage (indicated by proportional tree cover), such that for many species the connectivity and ecological value of the pond network may be compromised.

Traditionally the majority of ponds were associated with farmland (Table 3.7), for the watering of livestock and the extraction of marl for fertilizer (Miller 1991). The considerable decrease in the number of ponds associated with farmland reflects overall changes in land-use in BBC. Consequently, many old farm ponds now reside in a suburban landscape or have been lost due to urbanisation (e.g. Boothby and Hull 1997), as indicated by a negative correlation between pond density and impermeable surface, often used as a proxy for urbanisation (Walsh *et al.* 2001; Hahs and McDonnell 2006). Furthermore, increasing field sizes incumbent with agricultural intensification result in pond loss, consistent with Curado *et al.* (2011) and probably representative of an overall decline in habitat heterogeneity as observed in other studies (Dover and Sparks 2000; Hinsley and Bellamy 2000; Burel *et al.* 2004). At moderate levels of agriculture, however, pond density increases, representing overall land de-intensification as the pressure for urban or agricultural development is reduced and habitat heterogeneity improves. Pond density also increased with the coverage of improved grassland, which includes parkland and golf courses, and therefore ponds in amenity use, consistent with Gledhill *et al.* (2008), who found the highest pond densities were associated with greater provision of open access greenspace. Although improved grassland also includes land for livestock and may also reflect an increasing likelihood of pond presence for livestock watering.

The loss of ponds in the wider environment may be partially offset by the creation of small ponds within private gardens. A study by Davies *et al.* (2009), suggests that, on average, 10% of U.K. households have a garden pond, the average surface area of which is 0.99m^2. The mapping techniques in this study and those elsewhere, likely under represent such small water bodies. However, as recognised by Gledhill *et al.* (2008), although their quality may vary, garden ponds could provide vital stepping stones between those in the wider landscape, particularly as the proportion of households increase in urban areas as suggested by Swan and Oldham (1997) following the development of farmland in to residences. The 10% presence of ponds within households, taken at face value and modelled into the Birmingham landscape, show a great potential to act as compensatory habitats for the loss of ponds in the wider landscape (Appendix 8). However, much more field validation is required to understand this potential, particularly as garden pond presence is highly unlikely to be uniform across households (Loram *et al.* 2011).

As illustrated in the minimum spanning tree (Figure 3.4.7c), connectivity of the 2009 pond network relies on fewer, more isolated ponds to maintain connectivity. In addition, a potentially important role is played by a handful of newly created ponds located within central Birmingham. These sites are often ornamental ponds, built in association with new developments for aesthetic purposes. Whilst their spatial location places them as new and potentially valuable refuges for aquatic insect dispersal in an otherwise intensely developed environment, their ecological importance may be undermined by unsympathetic design e.g. concrete engineering and management regimes such has been found elsewhere (Hamerlik *et al.* 2011). Furthermore, resistance to aquatic insect movement within the urban landscape may compound their geographical isolation, such that if the ease of movement (i.e. effective distance) was incorporated the minimum spanning tree may once more re route around the edge of the city centre where the landscape presents fewer obstacles.

The changing spatial configuration of the pond network over time due to pond loss and gain affected the roles played by small and large ponds. At a short dispersal threshold (500m), more large sites became isolated relative to small sites, such that greater emphasis was placed on small sites for providing connectivity. By contrast, a longer dispersal threshold (1500m) maintained the connection of large sites to the network. This may mean that insects with good dispersal abilities are better able to utilise ponds with large surface areas whereas poor dispersers or species with high behavioural constraints to dispersal (e.g. Odonata, Conrad *et al.* 1999; Angelibert and Giani 2003) rely more on small sites to disperse in the modern landscape than they have in the past. This contrast in fortunes suggests a pinch point in the ecological resilience of the landscape between theoretical dispersal distances of 500m and 1500m that is further evidenced by a near ten-fold difference in the number of components created between the two thresholds. Further, future pond loss may increase the threshold distance required to maintain connection to larger sites such that the importance of smaller, more vulnerable ponds increases, posing greater risk to the connectivity of the pond network; although this is dependent upon changes in the spatial configuration of the network.

Management effort, therefore, should be aimed at the improvement of currently existing ponds as well as the creation of new ponds in optimum locations for the improvement of connectivity (Williams *et al.* 2008), or managing for different levels of connectivity across the network (Scheffer *et al.* 2006). To this end, network theory may once more prove useful as a decision-making tool (Gurrutxaga *et al.* 2011). Whilst in rural areas the creation of new ponds may take preference over the restoration of old ponds, with such initiatives as the 'Million Ponds Project' (Pond Conservation 2012). The restoration of ponds in an urban context may be a more viable alternative where land prices are high and resources for conservation are low (Sandstrom *et al.* 2006a). Traditionally, patch based analyses such as network theory have paid little attention to the internal habitat of the node (Urban and Keitt 2001), for example Bodin and Norberg (2007). Here the use of one exclusive, spatially oriented metric alongside a habitat area weighted metric has provided some, albeit limited inclusion of intra-habitat quality. The priority sites identified vary in their size and current habitat quality. In their current state, the ponds identified may not realistically represent sites of high ecological flux. However, the process has provided a sound basis upon which to make landscape scale decisions for the conservation of biodiversity. The restoration goals of these sites should be based within an urban context, taking into account human related values such that restorative action is publicly accepted (Ehrenfeld 1998; Sandstrom *et al.* 2006a).

3.6. Conclusion

Ponds are becoming increasingly scarce in the contemporary Birmingham landscape as sites are lost to agricultural intensification and urban development. The loss of ponds since *ca*1904 is considerable, but the rate of pond loss has shown slowed in recent times. This may be reflective of the types of ponds which are being retained as they are often in the public eye and form part of amenity parkland. It is apparent, however, that there is a critical threshold between 500m and 1500m at which many ponds become more isolated from their neighbouring habitats. If pond loss continues the distance of this critical threshold will increase, such that more ponds will become isolated by greater distances. With regard to the dispersal tendencies of many aquatic insects, the Birmingham pond network would appear at risk of becoming a disjointed network of components containing very few ponds each. Only a few individuals from each population will connect the sites through infrequent long distance dispersal events, much reducing the ecological resilience of the network. However, private garden ponds may play a pivotal role in maintaining connectivity between ponds in the public realm.

The identification of priority ponds for the connectivity of the pond network is the beginning of a more holistic strategy for the conservation of aquatic fauna and flora than traditional single site management. Nonetheless, although this study was performed at a landscape scale, the pond network was restricted by an administrative area. An extended analysis should be carried out to include the wider pond network such that study boundaries are reflective of natural boundaries. This would provide a better representation of the pond network, although in doing so, create a need for greater cross administration cooperation.

The biological relevance of the graph theoretic approach, as it has been applied here, can also be improved. Currently, there is no consideration of the inter-habitat landscape. In urban areas this is likely to form areas of high resistance to aquatic insects such that

Euclidean distance poorly represents dispersal potential. Incorporation of a landscape resistance matrix is likely to improve model realism, for example, calculation of least cost paths or attributing dispersal probabilities to graph links (e.g. Laita *et al.* 2010; Gurrutxaga *et al.* 2011). However, applications of least-cost paths are not without their own limitations (Sawyer *et al.* 2011). Similarly, whilst pond area provides some information with regard to intra-habitat quality, further measures to this end could be incorporated such as remotely sensed data. However, species distribution was not the aim of this study. This study highlights important nodes within the current pond stock to improve spatial resilience i.e. the priority sites are the first sites that landscape managers should consider improving or restoring to reduce the vulnerability of the pond network to habitat loss and an increasingly urbanised landscape.

Finally, the manner in which this study has been carried out is stepwise and intuitive and can be undertaken with freely available software, as with several other recent studies (e.g. Gurrutxaga *et al.* 2011), such that it may be readily repeated across regions and adapted for other landscapes.

4

The water quality of ponds across an urban land-use gradient

Chapter 4: The water quality of ponds across an urban land-use gradient

4.1. Introduction

Ponds are important contributors to regional biodiversity (Williams *et al.* 2004) and, as such, are a high priority for conservation. Many standing waters are threatened by human activities (Brönmark and Hansson 2002) and poor water quality, arising through chemical contamination (e.g. eutrophication), can severely impair pond communities (Roy *et al.* 2003; Mancini *et al.* 2005; Imberger *et al.* 2008). The environmental pollution of standing waters can increase susceptibility to invasion by exotic species (Vermonden *et al.* 2010) and also has implications for human health, especially as more people come into contact (e.g. via conservation, leisure activities), with standing waters (Birch and McCaskie 1999; Walsh 2000). In urban areas, where land prices are at a premium and land ownership over large areas is complex, ponds are a popular focus for conservation, offering tractable management options due to their size and small catchment areas (De Meester *et al.* 2005; Davies *et al.* 2008b). However, many ponds in urban areas are constructed without mind for conservation e.g. storm water retention (Tixier *et al.* 2011), and there is likely to be a trade-off between their intended purpose, water quality, and conservation value (Scher *et al.* 2004; Scher and Thiery 2005; Le Viol *et al.* 2009).

Ponds differ from other freshwater habitats due to their small volume to edge ratio (Declerck *et al.* 2006) and relatively large littoral zones (Palik *et al.* 2006). They have a high terrestrial-aquatic exchange of both organisms and matter than that of larger lakes and tend to receive water predominately through surface water run-off, as opposed to riverine inputs (Sondergaard *et al.* 2005). Pond sediments will tend to accumulate high concentrations of nutrients from run-off and organic matter, including leaf litter (Jeppesen *et al.* 1995). However, ponds have a smaller volume of water in which to buffer against internal or external sources of contaminants. Consequently, chemical exchanges between the sediment and water column can be dominant processes in ponds (Sondergaard *et al.* 2002; Tessier and

Woodruff 2002) and can continue long after management of external inputs have been controlled (Scheffer and van Nes 2007; Boros *et al.* 2009; Peretyatko *et al.* 2009).

Smaller systems are more likely to lack fish (De Meester *et al.* 2005; Sondergaard *et al.* 2005; Scheffer *et al.* 2006), however, the presence of benthic-feeding fish in particular (e.g. *Cyprinus carpio*) can enhance sediment-water chemical exchanges through bioturbation (Ritvo *et al.* 2004). The increase in suspended sediment load as a result of bioturbation can have a marked influence on macrophyte community composition (Wood *et al.* 2001), particularly submerged species, due to increased water turbidity and a consequent reduction in light availability (Matsuzaki *et al.* 2007). Therefore, the action of fish can directly or indirectly influence dissolved oxygen levels, suspended solids and sediment-water chemical exchanges, including nutrients. Furthermore, fish impact upon other biological interactions through predation e.g. zooplankton and phytoplankton (Peretyatko *et al.* 2009).

Macrophytes can reduce nutrient concentrations and improve water clarity. Nutrient uptake occurs through macrophyte root systems from the sediment and associated porewaters, and in the case of free-floating vegetation, from the water column (Henry-Silva *et al.* 2008). Water clarity can be improved by the reduction of phytoplankton biomass from macrophyte shading (Scheffer 1999). However, macrophyte diversity is generally lower in ponds than in larger water bodies (Oertli *et al.* 2002; Sondergaard *et al.* 2005), although a shallow water depth, which maximises light penetration potential, can enhance plant growth rates and over a larger proportion of their area (Declerck *et al.* 2006). Nonetheless, incident light availability varies as a function of shading by riparian trees (Lacoul and Freedman 2006), which can also provide wind protection reducing turbulence and possible sediment resuspension (Scheffer *et al.* 1993).

Anthropogenic activities affect the physical and chemical environment and the structure of the surrounding landscape (McDonnell and Pickett 1990). Surface waters in particular can be contaminated by human activities *via* point sources, such as sewage treatment discharge and by non-point, diffuse sources such as runoff from urban and agricultural areas (Sliva and Williams 2001). Negative impacts on water quality are frequently reported where urbanisation has occurred (Paul and Meyer 2001; Sonneman *et al.* 2001; Roy *et al.* 2003; Walsh *et al.* 2003; Mancini *et al.* 2005; Vermonden *et al.* 2009), especially where impermeable surfaces (or impervious cover) such as buildings and roads are extensive (Beasley and Kneale 2002; Mancini *et al.* 2005; Schueler *et al.* 2009). The way urbanisation is quantified, however, varies considerably depending on the subject matter and question being asked (Hahs and McDonnell 2006), but is typically selected from land-use indicators or demographic information, such as census data. The proportion of impermeable surface within a catchment has frequently been shown to have a low threshold of impact at around 30% coverage (Brabec 2009). Above this threshold the impact on water quality can intensify (Osborne and Wiley 1988; Walsh 2000; Sliva and Williams 2001). The impact of catchment urbanisation can be further confounded by the number of drainage connections, which effectively bypass the catchment altogether (Osborne and Wiley 1988; Hatt *et al.* 2004). In doing so, vegetation and other permeable surfaces are avoided, which are important for controlling run-off and can provide buffering of contaminants (Osborne and Kovacic 1993; Ourso and Frenzel 2003; Hatt *et al.* 2004).

The detection and control of diffuse pollution in ponds is an important focus for pond conservation (Biggs *et al.* 2005; Davies *et al.* 2008a) and there are many sources and pathways, which depend on surrounding land-use (Table 4.1). Both agricultural and urban activities can produce excessive N and P input to aquatic systems (Paul and Meyer 2001; Roy *et al.* 2003; Mancini *et al.* 2005). N and P is added to agricultural land as fertilizer, a net surplus of which frequently remains post harvest and can subsequently be exported to surface waters (Carpenter *et al.* 1998) directly through surface water drainage or through the atmosphere from the volatilization of NH_3 or N_2O and atmospheric deposition (Howarth 1988; Eichner 1990; Schlesinger and Hartley 1992). In a similar manner, fertilizers applied to manage green space such as lawns, public parks and amenity grassland can enter aquatic systems (Carpenter *et al.* 1998). Animal wastes (domestic and wild), for example, from Canada geese (*Branta canadensis*) (Stoianov *et al.* 2000; Chaichana *et al.* 2011), a considerable source of N and P (Manny *et al.* 1994) can be deposited on permeable and impermeable surfaces (Bryan-Ellis 2004). In addition, erosion on construction sites can be extremely high and cause a severely elevated sediment input to receiving waters including particle associated P (Cowen and Lee 1976).

Drainage misconnections are another important source of polluting runoff to ponds, especially in urban areas (Edmonds-Brown and Faulkner 1995; Dunk *et al.* 2008). This incorrect, and usually accidental connection of household drainage (e.g. from toilets, showers, sinks and dishwashers), to the surface water drainage system rather than the foul water drain result in water, rich in PO_4 and organic matter, being released in to surface waters.

Excessive N and *P* inputs can lead to eutrophication, which has been a recognised problem within lakes for decades (Brönmark and Hansson 2002). Eutrophication can have multiple negative effects upon freshwater ecosystems (see Table 2.2). Excess nutrients cause dense algal and aquatic weed growth that increases oxygen levels during daylight, but rapidly depletes them over night. When the plants senesce and decompose, microbial activity

Table 4.1: Diffuse pollutants, sources, pathways and associated land-use in the urban landscape

Land-use	Source(s)	Pathway(s)	Pollutant(s)
Impermeable surfaces (motorways, roads, pavements)	Engines, brakes, exhaust fumes, tyres, bodywork	Surface water drainage, atmospheric deposition, abandonment and dumping	**Poly-aromatic and mineral-oil hydrocarbons** **P_{tot}**, **NO_3,** NH_4 **Na**, Mg, **Ca**, **K**, SO_4, Cl **Cd**, Cu, **Pb**, Zn, **Cr**, Fe, **Ni**, Mn
	Road salts[1]	Surface water drainage	**Na**, **Cl**, P_{tot}, S, N, Cu, Zn, Ca
Landscaping	Gardens, grassed areas, cultivated land	Surface run-off, surface water drainage	Poly-aromatic and mineral-oil hydrocarbons P_{tot}, NO_3, NH_4 Na, Mg, Ca, K, SO_4, Cl Cd, Cu, Pb, Zn, Cr, Ni Pesticides, herbicides
Buildings	Roofing	Surface water drainage	Poly-aromatic and mineral-oil hydrocarbons P_{tot}, NO_3, **NH_4** Ca, Cl, **Mg**, **SO_4** Cd, **Zn**, **Cu**, Pb, Ni, Cr
	Residential housing	Foul water drainage misconnections	*(PO_4)*

Data on pollutants and sources after Beasley and Kneale (2002) and Gobel *et al.* (2007). Pollutants in bold refer to a potential main source

[1] Source: Marsalek 2003

Table 4.2. Adverse effects on freshwater ecosystems caused by eutrophication

Increased biomass of freshwater phytoplankton and periphyton
Shifts in phytoplankton species composition to taxa that may be toxic of inedible (e.g. bloom-forming cyanobacteria
Changes in vascular plant production, biomass, and species composition
Reduced water clarity
Decreases in the perceived aesthetic value of the water body
Taste, odour, and water supply filtration problems
Possible health risks in water supplies
Elevated pH and dissolved oxygen depletion in the water column
Increased fish production and harvest
Shifts in fish species composition towards less desirable species
Increased probability of fish kills

after Smith *et al.* (1999)

further reduces oxygenation of waters to levels at which fish kills can occur (Scheffer and van Nes 2007). High nutrient addition can shift ponds from a clear water macrophyte-dominated state to one of high turbidity where all but the most pollution tolerant macrophyte species are absent (Scheffer *et al.* 1993). As plant diversity is closely linked to macroinvertebrate diversity (Gledhill *et al.* 2008), a similar pattern in their diversity and abundance can be expected. Other trophic levels can also be affected by such cascading effects (Lodi *et al.* 2011). Eutrophication can interfere with the value of water for fisheries, recreation, industry, agriculture and drinking (Carpenter *et al.* 1998).

Trace concentrations of many heavy metals are an important component for physiological processes in animals, but at increased concentrations they can become detrimental to animal health (Brönmark and Hansson 2005). For example, Cu, Ni and Zn are critical micro-nutrients, but also the most commonly detected metals in urban runoff (Marsalek 1990). Increased road traffic is a key source of increased trace metals loads within urban water bodies (see Table 4.1), for example brakes, tyres, bodywork, are sources of solids, ranging from rapidly soluble, submicron particles to insoluble size aggregates (Sansalone and Buchberger 1997). Concentrations of heavy metals found within organisms are related to environmental variables of the water body and its watershed including pH, temperature and land-use (Chen *et al.* 2000) and the mobility of different trace metals within pond sediments varies with their sensitivity to changes in physicochemical conditions (Durand *et al.* 2004). Metal elements do not degrade in the environment and are persistent contaminants of lake, stream and river sediments (Power and Worsley 2009).

In the U.K., the Highways Agency has a duty to ensure, so far as reasonably practicable, that safe passage along a highway is not endangered by snow or ice (Railways and Transport Safety Act 2006). Birmingham City's opening rock salt stock is 7000 tonnes and road salting typically occurs 40-50 times a year (Birmingham City Council 2011). Rock salt consists primarily of Na and Cl together with impurities including P, S, N, Cu and Zn (Sliva and Williams 2001), which can represent up to 5% of the salt weight (Marsalek 2003). The application of road salt can subsequently increase the load of Na and Cl in water bodies receiving surface water, which can cause osmotic stress to organisms as well as increase concentrations of heavy metals in the aqueous phase (Mayer *et al.* 2008).

Traditional diffuse pollution management practices occur at the catchment level (e.g. Walsh *et al.* 2001; Allan 2004; Donohue *et al.* 2006). However, catchment delineation in heavily altered and artificial landscapes is a difficult and costly process due to drainage infrastructure (Akasaka *et al.* 2010) and the un-natural position of many urban ponds away from natural sink holes. Recently, studies have incorporated land-use information at different spatial extents, termed 'concentric ring analysis' (Pellet *et al.* 2004), in order to establish the most relevant to a study organism or habitat (Pellet *et al.* 2004; Declerck *et al.* 2006; Akasaka *et al.* 2010). The strongest influence of land-use on pond water quality has typically been found at close proximity to the sites. Akasaka *et al.* (2010) used generalized linear modelling (GLM) to relate land-use within increasingly distant spatial extents to a PCA score representative of turbidity and nutrient concentration and revealed that land-use nearest the ponds (i.e. 100m and 250m) explained most variation in pond water quality. Specifically, at the 100m extent, increasing proportions of urban land-use was correlated with higher nutrient concentrations and turbidity, whilst the coverage of freshwater habitat within 250m was associated with lower concentrations. Similarly, Declerck *et al.* (2006), in a non-urban landscape, found the effects of cropland and forest presence strongest up to 200m from the pond edge. Concentric ring analysis, therefore, is a valuable tool for establishing relevant spatial extents, within which ponds are most sensitive to change. This can provide a sound basis upon which to make effective management decisions.

In summary, pond water quality can be affected by local physical factors that occur within the wetted perimeter and within the water column and by spatial, land-use factors, which can affect diffuse pollution sources. The two are inextricably linked, however, the relative influence of one over the other upon water quality and the strength of the linkages between them can provide valuable information to inform management strategies. Variance partitioning (Borcard and Legendre 2002), has previously been used to assess the relative influence of multiple explanatory variable groupings such as water quality and habitat for macroinvertebrate community composition (e.g. Bechara 1996; Trigal *et al.* 2007) and climate, resource use and urbanisation on fossilised aquatic communities (Sweetman and Smol 2006). Few studies have applied the technique to aspects of water quality (but see Kernan and Helliwell 2001; Stendera and Johnson 2006).

This study investigates the spatial and temporal variation in water quality in ponds across an urbanisation gradient. Secondly, the water quality of urban ponds is compared to accepted environmental standards and contrasted to the results of pond studies within different regions and landscapes. Finally, the study combines concentric ring analysis and variance partitioning in a novel approach to assess the relative influence of land-use upon water quality at different spatial scales when the pure affects of local physical factors have been removed. In doing so the spatial extent at which pond water quality is most sensitive to alterations in land-use is identified.

4.1.1. Objectives and hypotheses

This chapter presents data to characterise and relate the water quality and contemporary land-use surrounding 30 ponds along an rural-urban gradient in the West Midlands over 22 months (May 2009 – February 2011). The following specific objectives were addressed:

1. To characterise spatial and temporal variation in pond water quality in the Birmingham and Black Country (BBC) region.

   Hypothesis 1

   Due to local differences in habitat and land-use, water quality in urban ponds is spatially and temporally varied, exhibiting wide ranges in water quality parameters.

2. Determine the spatial extent to which pond water quality is most sensitive to land-use composition along a gradient of urbanisation.

   Hypothesis 2

   Land-use exerts the largest influence on water quality at a relatively small spatial extent that reflects the catchment area of ponds.

3. Determine whether local features of ponds or surrounding land-use, or a combination of both control water quality.

   Hypothesis 3

   Local physical factors and land-use are both important for water quality. In terms of land-use, the degree of urbanisation is the dominant influence. Local physical factors and land-use are inextricably linked and share a large proportion of the explained variance in water quality.

4. Relate water quality with other studies and to environmental standards in order to assess how the water quality of urban ponds compares to water quality in other landscapes and aquatic systems.

   Hypothesis 4

Due to the pervasive nature of urbanisation, water quality frequently exceeds environmental guidelines. However, these values are comparable to other pond studies within anthropogenically disturbed environments.

4.2. Methods

4.2.1. Study site selection

Thirty ponds of contrasting surrounding land-use were selected from an estimated 1023 sites in Birmingham and The Black Country (BBC). For the full site selection process see Section 2.1.

4.2.2. Field campaign

Field surveys were carried out on a quarterly basis to represent seasonal differences in water quality between May 2009 and February 2011 (Table 4.3). On each visit a water sample was collected and observations were made with regard to the local physical conditions.

4.2.3. Local factors

4.2.3.1. Water quality

Once collected, samples were kept cool, filtered (GF/C, 1.2μm) and returned to the laboratory for determinations of major ions, trace metals, suspended solids and chlorophyll abundance. Triplicate *in situ* measurements of pH, electrical conductivity (μS/cm), temperature (°C) and dissolved oxygen (% saturation) were taken and the average measurements were calculated. Mean values of water quality determinands were used within ordinations. As the most contemporary, annual measurements from the 2010 hydrological year only (9th June – 23rd February, Table 4.3) were used to test compliance with environmental standards. Further detail on water quality sampling and analysis methods are discussed in Section 2.2.

4.2.3.2. Physical characteristics

Single measurements of pond surface area and percentages of surface area classified as open water, shaded, riparian vegetation and floating vegetation were all derived from a combination of digitised field notes and Normalised Difference Vegetation Index (NDVI) and photogrammetrically derived height GIS layers (Section 2.3.1.6) against digital pond outlines. The percentage of pond bank that was made from concrete was also calculated in a GIS from digitized field notes. Water sources (i.e. stream inflows, groundwater, surface run-off or building run-off) and fish presence information was gathered from stakeholder knowledge and visual inspection. A water level fluctuation index (WLFI) was calculated as the standard deviation in depth between surveys (taken quarterly between 2010 – 2011) measured at set points within the wetted-perimeter of each pond.

Macrophyte species presence absence was recorded from within identified mesohabitats sampled for macroinvertebrates (Section 2.4) using Haslam *et al.* (1982) for identification.

4.2.4. Regional factors

4.2.4.1. Land-use data

A GIS combining four land cover layers was used to generate detailed land-use data (Section 2.3). Land-use metrics were calculated for 13 cumulative spatial extents, measured from the pond edge in each case to 25m, 50m, 75m, 100m and then 100m increments to 1000m (Table 4.4).

Ten 'urbanisation indicators', selected from the land-use variables (indicated by ^{1} in Table 4.4) were incorporated into a PCA to create a score of urbanisation at each spatial extent (Section 2.3.1.7). The indicators chosen were those that were used commonly to measure urbanisation (see Hahs and McDonnell 2006) and closely linked to a general perception of urbanisation. PCA 1 was related to a gradient from a high proportion of arable land to a large degree of impermeable surface, PCA 2 is a gradient between high population density and an increased percentage of people in rural employment.

4.2.5. Statistical analyses

4.2.5.1. Unconstrained and constrained ordination of water quality

Water quality was examined using ordination. First, a principal components analysis (PCA) of measured water quality parameters was carried out to investigate spatial variability within the water quality dataset and to establish the underlying water quality gradients across the study sites. Second, a variance-partitioning approach (Borcard *et al.* 1992) using redundancy analysis (RDA), was used in order to examine the relative influence of local physical factors and land-use factors upon water quality. A set of RDAs were performed using land-use coverage data extracted at a series of distances of 25m, 50m and 75m and a further ten intervals of 100m from 100m to 1000m. Automatic forward stepwise model building was used for each RDA using the function 'ordistep' in the vegan R package, which

Table 4.3. Sampling periods and dates of sampling

Season and hydrological year	Sampling dates
Late spring/early summer 2009	20th May – 11th June 2009
Summer 2009	3rd August – 14th August 2009
Autumn 2009	3rd November – 17th November 2009
Winter 2009	27th February – 3rd March 2010
Late spring/early summer 2010	9th June – 11th June 2010
Summer 2010	18th August – 25th August 2010
Autumn 2010	10th November – 13th November 2011
Winter 2010	21st February – 23rd February 2011

Table 4.4. Brief summary of land-use types and their data sources calculated at 13 spatial extents (25m – 1,000m) to explain variance in water quality (also see Section 2.3).

	OS Mastermap	Land Cover Map	NDVI	Photogrammetry	Census 2001
Road[1]	▪				
'Towns & villages'[1]	▪				
Impermeable surface[1]	▪				
Water	▪				
Woodland	▪				
Private gardens	▪				
Green space	▪				
Suburban[1]		▪			
Urban[1]		▪			
Rough grassland		▪			
Arable[1]		▪			
Broadleaved woodland		▪			
Coniferous woodland		▪			
Improved grassland		▪			
Simpsons ($1/D_e$)		▪			
Vegetation >3m <60m			▪	▪	
Vegetation >0m < 3m			▪	▪	
Vegetation 0m			▪	▪	
All vegetation			▪	▪	
Building intensity[1]	▪			▪	
Rural employment %[1]					▪
Population density[1]					▪
Housing density[1]					▪

Those greyed out were removed from analyses due to high occurrence of zero values

[1] Land-use type used as an 'urbanisation indicator' to derive a gradient of urbanisation

uses permutation tests to obtain *P* values (<999 permutations). The first independent variable entered explains most of the variation in the dependent variables, the second one most of the remaining variation, and so forth. The procedure finishes when the next variable entered does not have a significant partial correlation, where $P > 0.05$. Using the variance-partitioning approach (Borcard *et al.* 1992), four RDAs were carried out at each spatial extent to establish the variance in water quality independently attributed to local physical and land-use factors and shared between the two where each RDA used; A) local physical factors alone (constant), B) land-use factors alone, C) local physical factors with the effect of land-use 'partialled out' and, D) land-use factors with the effect of local environmental factors partialled out. Shared variance was then calculated by either A – C or B – D. Unexplained variance is 100 - (C + D + (A - C or B - D)). Significance of ordination axes in each RDA was assessed by ANOVA (<999 permutations, $P < 0.05$).

Linear forms of ordination were employed following initial detrended correspondence analysis (DAC) using the 'decorana' function in the vegan R package, which revealed a short maximum axis length of 1.58 (Lepš and Šmilauer 2003). Ordinations were carried out using the functions 'rda' for PCA and RDA and 'decorana' for DAC in the 'vegan' package (Oksanen *et al.* 2012) for the R statistical program version 2.15.1 (R Core Team and contributors worldwide 2012). In the ordinations, mean water quality values from the whole field campaign were used and, if needed, transformed to improve normality and homogeneity of variance, as tested by Shapiro-Wilk and Levene's tests carried out in IBM SPSS statistical package version 19 (IBM, Armonk, New York). Data were standardised after transformation to 0 mean and 1 standard deviation due to different measurement scales. Data transformations were selected from log(n+1), square-root or arcsine transformations. Log(n+1) was used for most water quality parameters and arcsine for land-use data (proportional data). Variables were removed from the local physical and land-use datasets that were collinear. This was performed by removing one of any two variables that were greater than +/- 0.7 correlated (Pearson's correlation coefficients). The retained variable was the one considered most relevant and / or derived from the most contemporary land-use dataset (see Section 2.2 for dataset metadata).

4.2.5.2. Spatial autocorrelation

Spatial autocorrelation occurs when values at one locality are dependent on the values of another. For analysis of water quality this could be borne out of shared water sources, such as groundwater, urban drainage infrastructure, synchronous local climatic conditions or run-off from surrounding land-use. In freshwater habitats, fluctuations in the environment are more likely to be correlated at small spatial scales (Bohonak and Jenkins 2003). The study sites were tested for spatial independence at distance intervals using a Mantel correlogram using the function 'mantel.correlog' in the vegan package (Oksanen *et al.* 2012) for R statistical package version 2.15.1 (R Core Team and contributors worldwide 2012). The Mantel correlogram is an effective tool for detecting correlation between datasets (Borcard and Legendre 2012) and tests for community similarity using Bray-Curtis dissimilarity as a function of geographic distance classes (24 classes used). Mantel tests (Mantel 1967) assess the magnitude of the correlation between two or more symmetric distance matrices. Significance of the matrix correlation coefficient was tested by 999 random permutations with a Bonferroni correction. Violations of spatial independence between the study sites would indicate the presence of spatial autocorrelation within the dataset, which may ultimately lead to biased conclusions. Spatial autocorrelation analysis was performed on normalised multivariate water quality data (all parameters) to allow for Bray-Curtis computation.

4.2.5.3. Selection of environmental standards

The environmental standards referred to here, which have been typically established for rivers and lakes, have been used in the absence of pond equivalents. Consequently they are seen as the best alternative in the absence of guidelines specifically set for small standing waters. The most notable are those used in testing requirements of ecological 'good' status for all surface waters as outlined by the Water Framework Directive (WFD, European Commission 2000). The standards used and their sources are listed in Table 4.6.

Trophic State Index (TSI, Carlson 1977) was calculated for each site to assess their nutrient status using the following formula.

$$TSI = 30.6 + 9.81\ ln(Chl - a) \tag{4.2.1}$$

This is a measure of the trophic status of a body of water based on chlorophyll a (*Chl - a)* abundances (algal biomass). TSI ranges on a scale from 0-100 that is based upon relationships between secchi depth, surface water concentrations of algal biomass, and total phosphorus for a set of North American lakes (Carlson 1977). Sites closer to 100 are more eutrophic and each major division represents a doubling in algal biomass and can be interpreted according to more qualitative descriptions of trophic status (Table 4.5).

Table 4.5. Interpretations of the Trophic State Index for lakes (TSI, Carlson 1977) and potential implications for aquatic biology, after Carlson and Simpson (1996).

TSI	Attributes
<30	**Oligotrophy**: Clear water, oxygen throughout the year in the hypolimnion
30-40	Hypolimnia of shallower lakes may become anoxic
40-50	**Mesotrophy**: Water moderately clear, increasing probability of hypolimnetic anoxia during summer
50-60	**Eutrophy**: Anoxic hypolimnia, macrophyte problems possible
60-70	Blue-green algae dominate, algal scums and macrophyte problems
70-80	**Hypereutrophy**: (light limited productivity). Dense algal and macrophytes
>80	Algal scums, few macrophytes

Table 4.6: Environmental standards adopted from available freshwater guidance and legislation

Variable	Measure	Standard	Guidance / legislation
pH	$\geq 6 \leq 9$	5% and 95%ile values	WFD, all rivers
Conductivity	<1,000 µS/cm	Annual average	WFD[1], very shallow, high alkalinity lake
Temperature	<28°C	98%ile (good)	WFD, warm water (cyprinid fishery)
Dissolved oxygen	>60%	10%ile (good)	WFD[1], lowland, high alkalinity river
Cl	<250 mg/l	Annual average	Council Directive (EC) 76/464/EEC 1976
NO_3	<30 mg/l	Annual average (high[2])	GQA, National Rivers Authority 1994
SO_4	<400 mg/l	Annual average	Council Directive (EC) 76/464/EEC 1976
PO_4	<0.031	Geometric mean (good[1])	WFD[1], very shallow, high alkalinity lake
Na	<200 mg/l	Annual average	Council Directive (EC) 98/83/EEC 1998
NH_4	<0.6	90%ile (good[1])	WFD[1], lowland, high alkalinity river
Mn	<30 /300 µg/l	Annual average / max.	Council Directive (EC) 76/464/EEC 1976
Fe	<1,000 µg/l	95%ile	Council Directive (EC) 76/464/EEC 1976
Zn	<1,000 µg/l	Annual average	Council Directive (EC) 78/659/EEC 1978
Suspended solids	<25 mg/l	Annual average	Council Directive (EC) 98/83/EEC 1998
Chlorophyll *a*	>60[3]	TSI - C = 30.6 + 9.81 Ln [Chlor-*a*] (in µg/L)	Carlson (1977)
No appropriate standard for alkalinity ($CaCO_3$), Mg, Ca or K			

[1] Taken from the UK TAG 2008b and UK TAG 2008c on the Water Framework Directive, standard required for 'good' status for lakes with $CaCO_3$>50mg/l or rivers <80m altitude, >50mgl $CaCO_3$

[2] Value at which water is considered to have a 'high' NO^3 concentration under the GQA scheme

[3] Trophic State Index derived from chlorophyll *a* concentration at which lakes are considered eutrophic

4.3. Results

Ponds were slightly alkaline pH (7.61), with relatively low dissolved oxygen (61.1%) and elevated PO_4 (0.27 mg/l) and NH_4 (0.77mg/l) concentrations (Table 4.7). It was apparent that most sites experienced dynamic changes in chemical characteristics with large deviations over the course of the study (Appendix 9). Across the study, most parameters exhibited wide ranges between their minimum and maximum measurements.

4.3.1. Performance of water quality against environmental guidelines

Carlson's Trophic State Index (TSI, Carlson 1977 Figure 4.3.1) indicated that the majority of ponds studied (18/30 sites) exceeded the lower boundary at which they may be considered eutrophic (Carlson and Simpson 1996). Above this level there are likely to be more frequent problems associated with excessive macrophyte growth, a dominance of blue-green algae and an anoxic hypolimnion (in deeper lakes with stratification) (Table 4.5). Six sites (6, 8, 11, 16, 19 and 22) had a TSI > 70 and may be considered hypereutrophic, although two sites (9 and 12) had a TSI <30, which suggests oligotrophic conditions with clear water and high oxygen conditions throughout the year. PO_4, which frequently limits plant growth, and NH_4 concentrations were often in excess of or close to the proposed WFD standards for attaining 'good' ecological status (UK TAG 2008b; UK TAG 2008c) during 2010 (Figure 4.3.2h & k). In contrast, NO_3 was consistently measured in lower concentrations than its 30 mg/l standard (National Rivers Authority 1994, Figure 4.3.2g). Site 13 had the largest annual geometric mean of PO_4 in 2010 (1.96 mg/l) and is an ex-marl pit site fed solely from cemetery run-off and is almost entirely shaded. The next two sites with the highest annual geometric means for 2010, sites 4 and 19 (both 1.06 mg/l), were historically built as part of larger estates that have since become enveloped by modern residential housing and also have high levels of shading. Sites 4, 13 and 19 also have high 2010 mean concentrations of NH_4 2.23 mg/l, 1.48 mg/l and 1.83 mg/l respectively (Appendix 9).

Tenth percentile dissolved oxygen levels reflected the eutrophic status of most the study sites and were frequently below the 60% saturation threshold set (22/30 sites) for attaining 'good' status under the Water Framework Directive (UK TAG 2008b) (Figure 4.3.2d), suggesting that most sites experienced periods of low dissolved oxygen levels that may be harmful to aquatic life. Suspended solid loads were occasionally in excess of the 25 mg/l guideline for the protection of coarse fish (Council Directive (EC) 78/659/EEC 1978). High suspended sediment loads may also be indicative of high algal density, however, no significant correlation was found between suspended solids and chlorophyll *a* concentration (Pearson's correlation coefficient, $r = 0.21$, $P > 0.05$).

Sites 26 and 28 exceeded the 1000 µS/cm standard for conductivity proposed for use in the WFD (UK TAG 2008b). Site 26 is fed by a polluted brook whilst site 28 is very shallow and heavily shaded and therefore receives a large amount of leaf litter. Site 28 is also the sole site to exceed the standard for SO_4 set for regulating potentially polluting chemicals (Council Directive (EC) 76/464/EEC 1976) (Figure 4.3.2i), which may have resulted from leaf litter decomposition.

Concentrations of Ni, Cu, Co, Cr and Pb were typically below detection limits. Few, sites exceeded the standards for Fe (Figure 4.3.2o) and none for Zn (Figure 4.3.2q), however, the majority (26/30 sites) exceeded one or both of the standards set for Mn (Figure 4.3.2*P*). Study mean Mn concentrations were 328 µg/l, which is in excess of both aspects of the standard (annual mean < 30 µg/l and maxima < 300 µg/l, Council Directive (EC) 76/464/EEC 1976). An important source of dissolved manganese is anaerobic environments where particulate manganese oxides are reduced (Howe *et al.* 2004). Mn levels within the sediments against a geoaccumulation index found sediments within the study ponds to be uncontaminated to moderately contaminated with Mn (Cooper 2011), which suggests that some urban pond sediments have the potential to provide elevated dissolved Mn concentrations when dissolved oxygen levels are reduced. However, water samples were taken at the water surface and, although water depth at the sampling point was typically shallow i.e. <0.5m, the mechanism behind such elevated Mn concentrations is unclear as sediment derived Mn may not necessarily impact upon the whole water column. Nonetheless, site 28 had high concentrations of Fe, Mn and Zn, possibly due to its more ephemeral nature and therefore enhanced sediment-water exchange and lower dilution potential.

4.3.2. Spatial variation in water quality among ponds

Spatial variation in water quality was assessed using a PCA of mean water quality data from across the study. The first four axes of the PCA of water quality were significant (<999 permutations, $P < 0.05$) and cumulatively accounted for 67.7% of the overall variance. Those variables that were significantly related (Pearson's correlation coefficient, $P < 0.05$) to any of the first four axes are listed in Table 4.8. Axis 1 and 2 described 25.7% and 19% of the variation in water quality data respectively (Figure 4.3.3), revealing a strong gradient between high dissolved oxygen and pH to high PO_4 across the sites (Figure 4.3.3). Also on axis 1 are the variables conductivity, $CaCO_3$, Ca and Mg and SO_4, which are significantly inter-correlated ($r > 0.5$, $P < 0.05$) with the exception of $CaCO_3$ with SO_4 ($r = 0.29$, $P > 0.05$), but which group in a different area of the ordination space due to their relationship with axis 2. Sites 23 and 29, both well vegetated ponds within nature reserves, are sites with high dissolved oxygen (82.6% and 75.3% study means) and low PO_4 (both 0.09 mg/l study means). Related to the second axis are sites 26 and 28, with high conductivities (1100 µS/cm and 822 µS/cm study means). At the opposite end of the oxygen gradient are sites 4, 5, and 19 (35.6%, 33.7% and 28.9% study means), each of these sites also have considerable tree cover and limited macrophytic growth. Concentrations of Mn in particular, and to a lesser extent NH_4 and K, differentiate the first two axes. Sites 28 and 30, which are 100% shaded by mature vegetation and site 26 have high NH_4 concentrations, whilst the open waters of ponds 10, 14 and 24 have low NH_4 concentrations.

PCA axes 3 and 4 (Figure 4.3.4) are gradients in salinity (Na and Cl) and turbidity (as described by suspended solids and chlorophyll *a*) and describe 13.6% and 9.4% of the variation respectively. On axis 3, sites 2 and 17 are sites with high Na and Cl concentrations (> 50 mg/l Na, > 95 mg/l Cl study means). Site 2 is immediately adjacent to the M6 motorway and site 17 receives storm water; both receive run-off which likely includes salt compounds following de-icing activity. Sites 6 (139 µg/l) and 22 (127 µg/l) are those with high study mean chlorophyll *a* and suspended solids concentrations, 27 mg/l and 67 mg/l respectively. Site 6 is a heavily fished pond stocked with carp and other coarse fish where a large amount of bioturbation can be expected and site 22 is a very shallow and silted pond situated at a cross-road and is almost entirely choked with duckweed (*Lemna spp.*) and liverwort (*Riccia fluitans*). Conversely, site 5 is a heavily shaded pond that receives run-off and groundwater from the surrounding woodland (predominately English oak (*Quercus robur*) and has the lowest mean water temperature (11.2^{o}C), which inhibits algal growth.

Table 4.7. Summary of water quality parameters across the study sites (n=30). Mean values for each year and for the study duration, minimum and maximum given in parenthesis.

Variable	Unit	2009	2010	Both years
pH	-	7.59 (6.2-10.0)	7.63 (6.7-9.1)	7.61 (6.2-10.0)
Conductivity	µS/cm	501.8 (191-1526)	554.3 (45-2418)	528.0 (45-24181)
Temperature	°C	12.6 (2.7-27.5)	12.1 (3.9-23.8)	12.3 (2.7-27.5)
Alkalinity[1]	mg/l ($CaCO_3$)	125.4 (28-310)	139.0 (40-342)	134.4 (28-342)
Dissolved oxygen	% saturation	64.4 (6.0-185.2)	57.7 (6.5-157.3)	61.1 (6.0-185.2)
Cl	mg/l	41.7 (1.7-377.7)	54.3 (0.7-251.0)	47.8 (0.7-377.7)
NO_3	mg/l	5.96 (0.0-77.5)	3.32 (0.01-25.2)	4.58 (0.01-77.5)
PO_4[2]	mg/l	0.33 (0.01-2.63)	0.38 (0.00-2.63)	0.37 (0.00-2.63)
SO_4	mg/l	40.8 (0.93-183.3)	68.7 (1.54-1312)	54.7 (0.93-1312)
Na	mg/l	25.7 (4.24-209.3)	28.3 (5.93-142.2)	27.0 (4.24-209.3)
NH_4	mg/l	0.56 (0.0-6.55)	0.98 (0.03-8.21)	0.77 (0.03-8.21)
K	mg/l	4.91 (0.73-13.24)	6.10 (0.58-21.39)	5.50 (0.58-21.39)
Mg	mg/l	6.54 (1.56-28.14)	8.76 (2.94-41.36)	7.65 (1.56-41.36)
Ca	mg/l	50.3 (13.9-136.2)	56.4 (12.0-516.4)	53.3 (12.0-516.4)
Fe	µg/l	451 (0-8139)	284 (38-2181)	368 (38-8139)
Mn	µg/l	275 (0-3623)	381 (6-9211)	328 (6-9211)
Zn	µg/l	63 (0-803)	21 (12-57)	42 (12-803)
Chlorophyll *a*[2]	µg/l	89.6 (0.6-1495)	39.3 (0.0-604.1)	60.7 (0.0-1495)
Suspended solids[2]	mg/l	18.9 (0.0-109.8)	16.3 (0.0-77.8)	17.4 (0.0-109.8)

* Zero values represent concentrations below detection limits (Appendix 3)

[1] Data collected from autumn 2009, [2] Data collected from summer 2009

Figure 4.3.1. Trophic State Index (TSI) derived from chlorophyll *a* analysis (Carlson 1977). The value indicated represents the approximate lower boundary for eutrophic conditions, after Carlson and Simpson (1996).

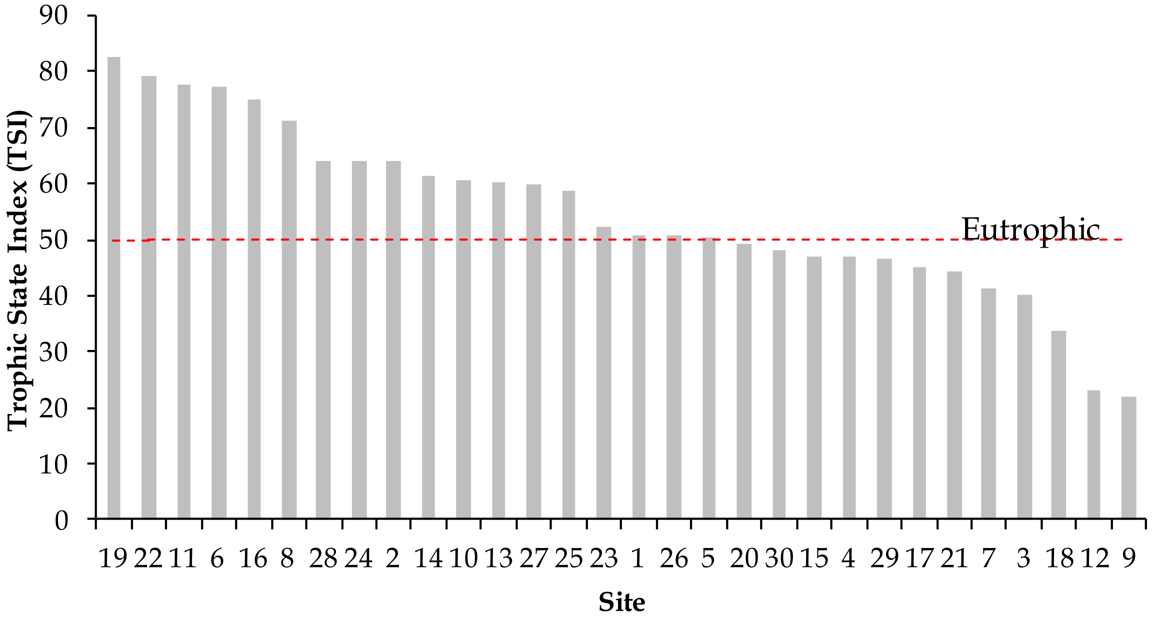


Figure 4.3.2. Individual site values for water quality, June 2010 – February 2011 against environmental standard boundaries and thresholds (indicated by red dashed line(s), refer to Table 4.6 for sources). Sites are ranked left to right, highest to lowest values. Dark grey bars indicates sites failing >5 different standards

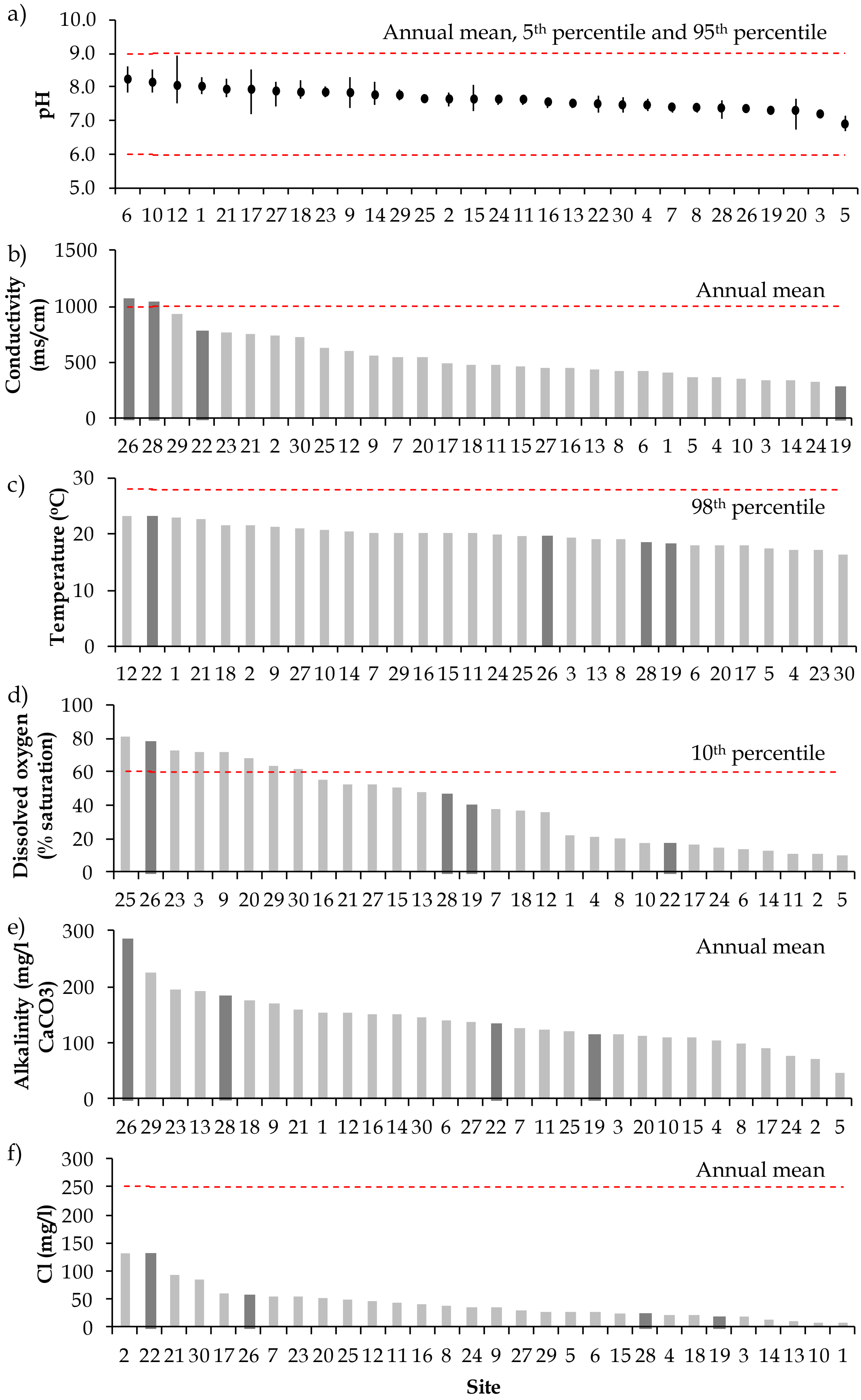


Figure 4.3.2 (cont) Individual site values for water quality, June 2010 – February 2011 against environmental standard boundaries and thresholds (indicated by red dashed line(s), refer to Table 4.6 for sources). Sites are ranked left to right, highest to lowest values. Dark grey bars indicates sites failing >5 different standards

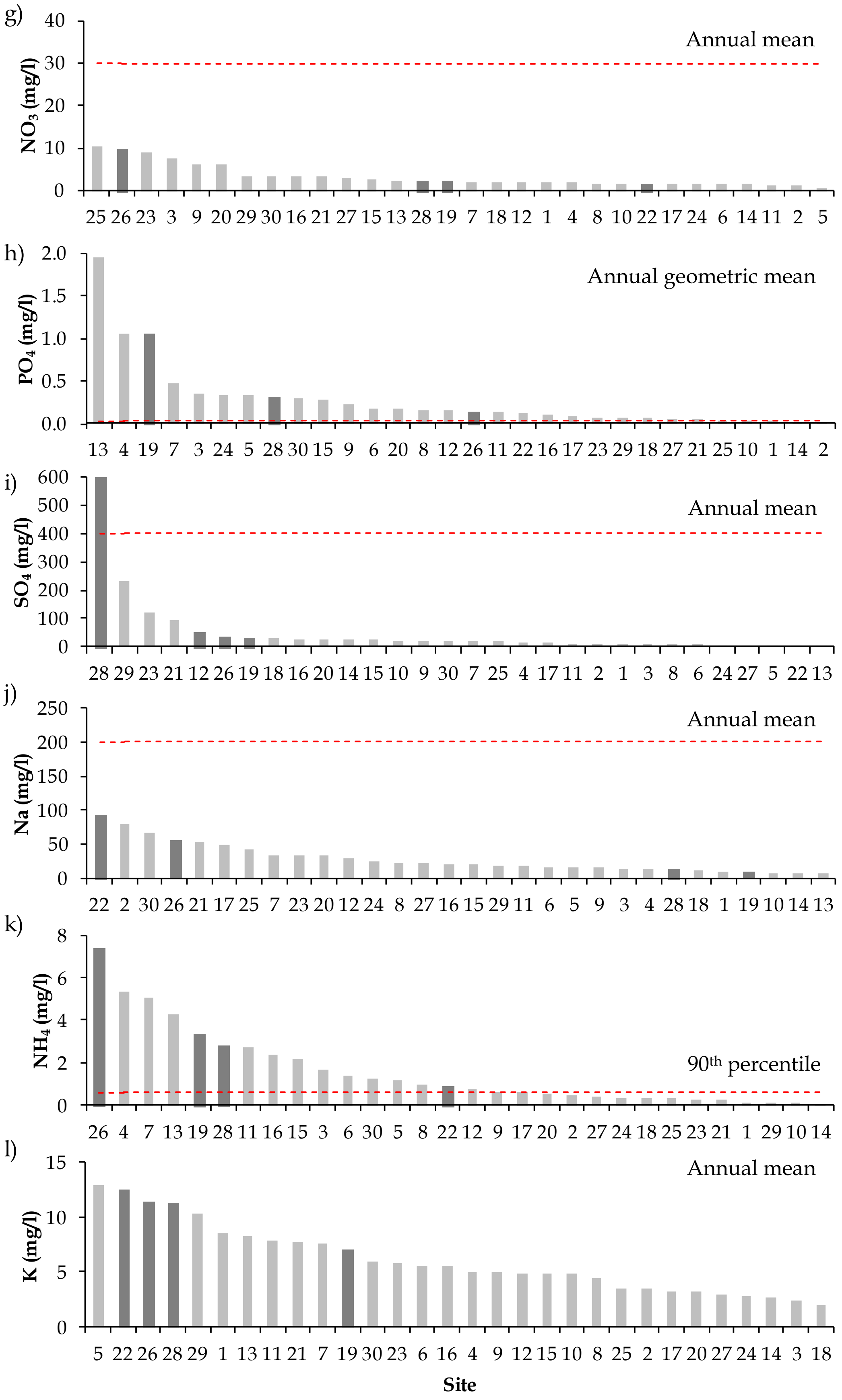


Figure 4.3.2 (cont) Individual site values for water quality, June 2010 – February 2011 against environmental standard boundaries and thresholds (indicated by red dashed line(s), refer to Table 4.6 for sources) Sites are ranked left to right, highest to lowest values. Dark grey bars indicates sites failing >5 different standards

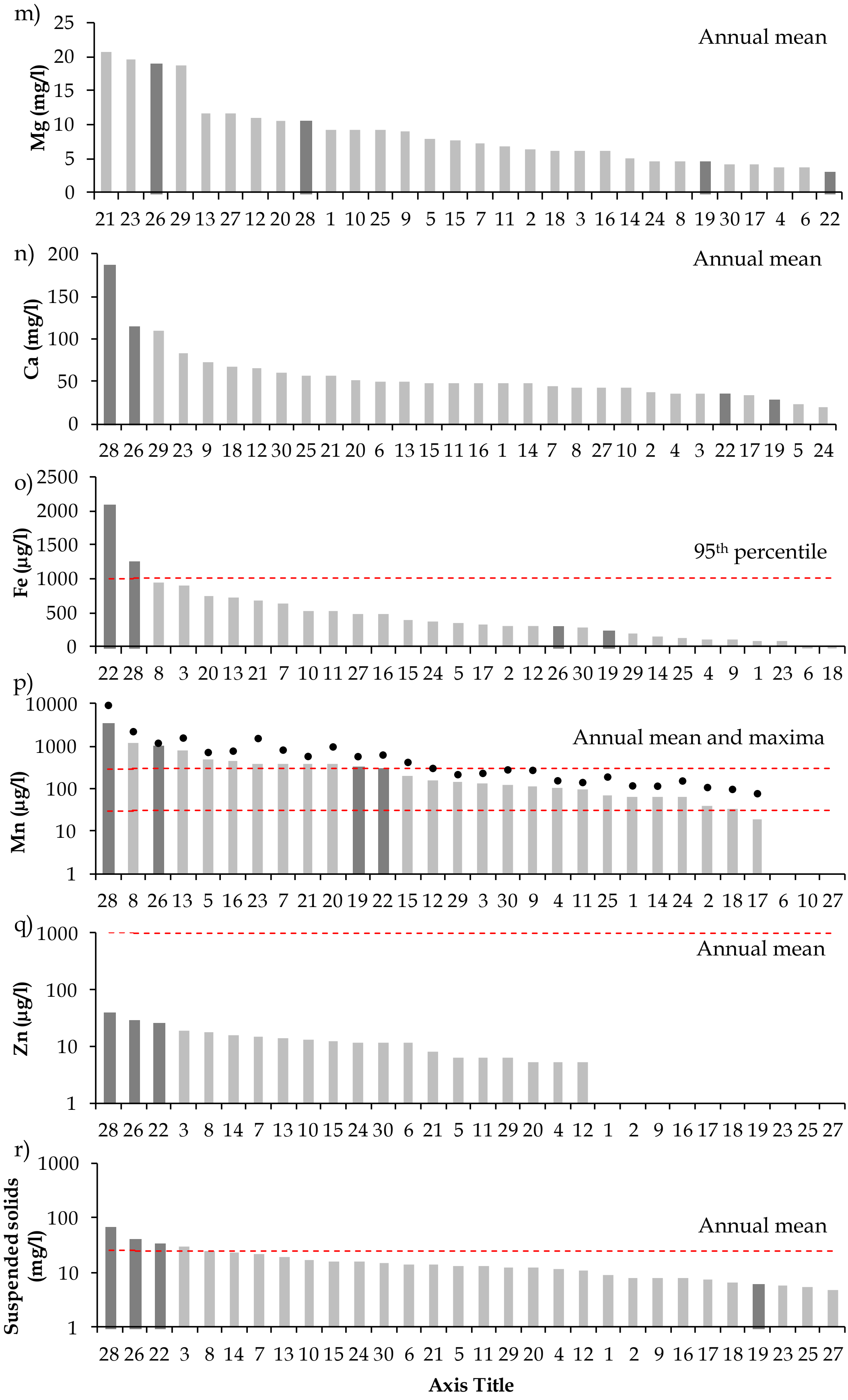


Table 4.8. Water quality variables significantly correlated to at least one of the first four PCA axes (Pearson's correlation coefficient)

Variable	Axis 1	Axis 2	Axis 3	Axis 4
Conductivity	**0.66**	**-0.63**		
Temperature	*0.51*			
pH	**0.63**	**0.59**		
Dissolved oxygen	**0.70**	*0.53*		
$CaCO_3$	**0.58**			
NH_4		**-0.57**		
Ca	**0.67**	*-0.50*		
Cl			***0.87***	
K		**-0.66**		
Mg	**0.68**			
Na			***0.84***	
PO_4	**-0.62**			
SO_4	**0.67**			
Mn		***-0.84***		
Zn	*-0.46*			
Suspended solids				***-0.87***
Chlorophyll *a*	*-0.53*			**-0.62**

Significant of relationship, *P* <0.1 = *I*, <0.05 = **I**, <0.01 = ***I***

Figure 4.3.3. Principal components analysis (PCA) biplot for axes 1 and 2 showing spatial variation in water quality parameters and pond sites. Vectors are shown for significant parameters as per Table 4.8.

Key:

DO – Dissolved oxygen
Temp – Temperature
NO_3 – Nitrate
Alk – Alkalinity
Mg – Magnesium
SO_4 – Sulphate
Ca – Calcium
EC – Electrical conductivity
K – Potassium
Mn – Manganese
NH_4 – Ammonium
PO_4 – Phosphate
Fe – Iron
Chla – Chlorophyll *a*
SS – Suspended Solids
Zn - Zinc

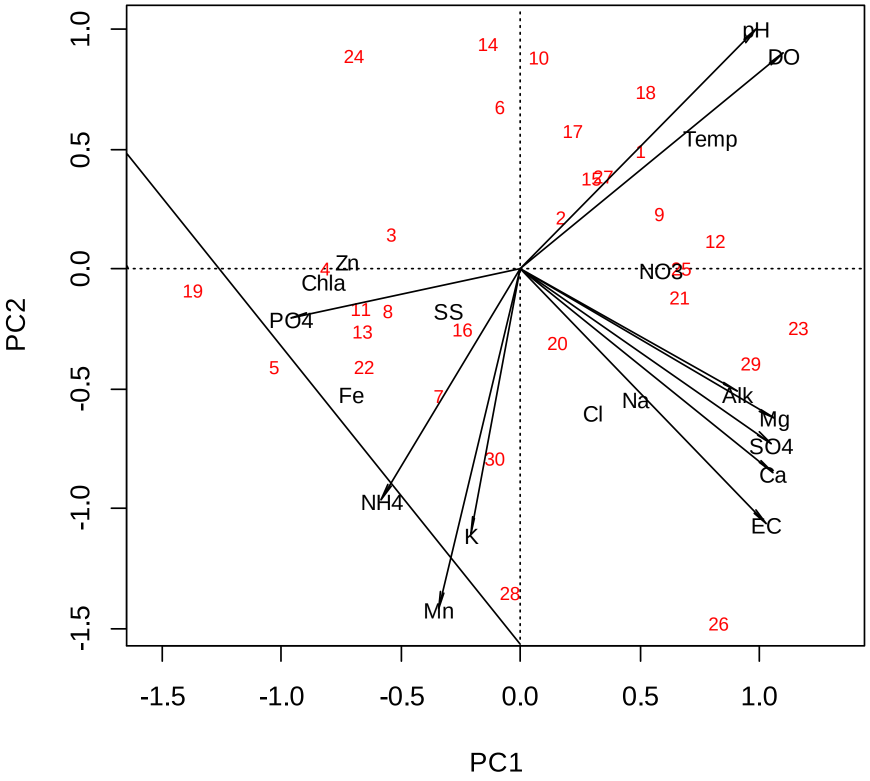


Figure 4.3.4. Principal components analysis (PCA) biplot for axes 3 and 4 showing spatial variation in water quality parameters and pond sites. Vectors are shown for significant parameters as per Table 4.8.

Key:

DO – Dissolved oxygen
Temp – Temperature
NO_3 – Nitrate
Alk – Alkalinity
Mg – Magnesium
SO_4 – Sulphate
Ca – Calcium
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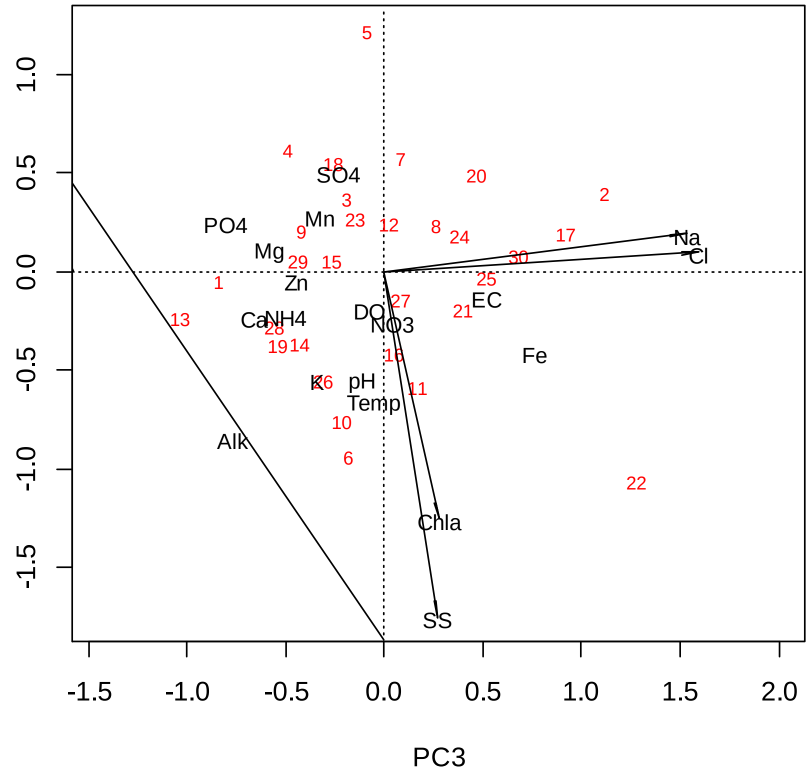
NH_4 – Ammonium
PO_4 – Phosphate
Fe – Iron
Chla – Chlorophyll *a*
SS – Suspended Solids
Zn - Zinc



4.3.3. Inter-seasonal water quality

4.3.3.1. *In situ* measurements

Site mean pH (range 6.2 – 10.0) and conductivity (45 – 2418 µS/cm) were relatively constant among seasons and years (Appendix 10a & b). Nevertheless, boxplots revealed strong temporal variation in pH and conductivity in some sites receiving storm water drainage (e.g. 17, range 44.5 µS/cm to 1526 µS/cm).

Water temperature varied seasonally, with summer maxima (2009 22.7°C, 2010 23.8°C) and winter minima (2009 2.7°C, 2010 3.9°C) and synchronicity between years (Appendix 10c). Dissolved oxygen concentrations were inversely related to water temperature i.e. lower in summer 2009 (mean 56.6%) and 2010 (mean 52.0%) highest in both the winter years when the temperature is at its lowest and biological uptake is reduced. Again, boxplots (Appendix 10d) revealed marked variation about the mean, especially in summer 2009 and 2010 when both severely depleted (7.8%) and super-saturated (185.2%) oxygen levels were recorded in some ponds. The greatest temporal variation in oxygen saturation was recorded at site 15 with an 11.9% minima (summer 2010) and 185.3% maxima (summer 2009) as a consequence of changing phytoplankton biomass.

Site alkalinity varied considerably (range 28 – 342 mg/l $CaCO_3$), although mean values remained relatively consistent (range 123 – 148 mg/l $CaCO_3$). There was no clear seasonal trend in alkalinity values (Appendix 10e).

4.3.3.2. Major anion analysis

PO_4 concentrations were consistently high across all seasons (Appendix 11c), 24 sites recorded PO_4 concentrations in excess of 0.1 mg/l and 9 greater than 1 mg/l. However, PO_4 concentrations showed winter lows (mean values 0.22 mg/l winter 2009 and 0.17 mg/l winter 2010). The peak season for PO_4 appeared to be spring. Although PO_4 concentrations were not available for spring 2009 (refer to Section 2.2.2), spring 2010 concentrations (mean 0.51 mg/l) were at least twice the concentration of winter 2009 or 2010. By contrast, NO_3 concentrations, although considerably lower than the 30 mg/l standard (National Rivers Authority 1994), tended to increase throughout the year to winter maxima (mean values 16.7 mg/l winter 2009 and 7.31 mg/l winter 2010, Appendix 11b). Spring had the lowest NO_3 concentrations (0.25 mg/l spring 2009, 1.15 mg/l spring 2010). Using the Redfield ratio (Redfield 1934), the majority of systems switched from being NO_3 limited during spring and summer to PO_4 limited during autumn and winter (Figure 4.3.5).

Cl values in each year showed winter peaks (mean 71.4 mg/l 2009 and 56.8 mg/l 2010). In both 2009 and 2010 winters site 30 recorded extremely high Cl concentrations (370.1 mg/l and 172.7 mg/l), which may suggest surface water drainage entering site 30 following period of road gritting (Appendix 11c).

SO_4 was consistent between years with the exception of summer 2010 values (mean 125 mg/l), which increased as consequence of several sites with very high concentrations (sites 21, 23, 28 and 29 >200 mg/l, Appendix 11d).

4.3.3.3. Major cation analysis

Site mean Na (range 4.2 mg/l – 209.3 mg/l) showed winter maxima (mean values 42.7 mg/l winter 2009 and 31.2 mg/l winter 2010), and reflected that shown by Cl (Section 4.3.3.2). Similarly, this is likely as a consequence of road gritting for the protection of road surfaces from ice, which has subsequently been received within surface water drainage. Site 30 had high concentrations of Na in both years during winter (201.6 mg/l 2009 and 94.2mg/l 2010) and 17 has the highest concentration in winter 2009 (209.3mg/l).

Elevated concentrations of Na, NH_4, K, Mg and Ca were recorded during summer 2010 (Appendix 12). However, there was no clear temporal trend in K, Mg or Ca across the study. Inter-quartile ranges in NH_4 were reduced during autumn and winter and expanded during summer. NH_4 is derived from the aerobic and anaerobic decay of organic material the highly variable concentrations during the summer sampling period (3rd – 19th August) coincided with the end of the growing season. The relative consistency of median concentrations across seasons, however, suggested that increases in NH_4 were restricted to a subset of sites.

4.3.3.4. Trace metals analysis

Very few ponds had concentrations of Co, Cr, Cu, Ni or Pb above the detection limits. Fe and Zn concentrations were consistent between the seasons, with the exception of relatively high concentrations of Zn during spring 2009 (median 282 μg/l and 803μg/l maxima). Fe concentrations were typically below 1000 μg/l and median values fluctuated very little between seasons (median range 162 μg/l – 333 μg/l) (Appendix 13a). The majority of measurements that exceeded 1000 μg/l were restricted to sites 11 and 22. Mn values were in the same range as Fe, although showed a greater deal of variability across the sites within each season (Appendix 13b). Mn concentrations were lowest in both winter seasons (mean 156 μg/l 2009 and 263 μg/l 2010), approximately half the peak mean concentration (578 μg/l summer 2010). Zn was rarely measured much above the detection limit (Appendix 13c) and average concentrations were an order of magnitude lower than Fe or Mn, with the exception of spring 2009.

4.3.3.5. Chlorophyll *a* and suspended solids

Suspended solids and chlorophyll *a* showed a similar overall trend to each other, although chlorophyll *a* was considerably more variable (Appendix 14a). The similar trend is likely due to the inclusion of phytoplankton biomass within the suspended solids analysis. Chlorophyll *a* abundances peaked during the summer (maxima 604 µg/l and median values 22.2 µg/l 2009 and 23.4 µg/l 2010) and winter, (maxima >1,000 µg/l and median values 84.1 µg/l 2009 and 27.4 µg/l 2010). Similarly, suspended solids concentrations peaked during summer (median values 14.1 mg/l 2009 and 12.0 mg/l 2010) and winter seasons (median values 13.5 mg/l 2009 and 15.2 mg/l 2010) and approximately doubled the concentrations measured in spring and autumn.

4.3.4. Analysis of local physical and land-use upon water quality

The variance partitioning approach suggested that, at all spatial extents, local physical factors were able to account for a greater proportion of variance in water quality than land use factors (Figure 4.3.6a). RDAs using land-use factors, with local physical factors partialled out revealed that an aspect of urbanisation i.e. PCA 1 (section 4.2.4.1), road surfaces or urban land-cover was important at each spatial extent from 25m up to 700m, with the exception of the model at 300m (Figure 4.3.6b). Overall, the difference in explanatory power of the models between the spatial extents is relatively small, the most powerful model being able to explain a further 11.2% of total water quality variance (explained + unexplained) than the least powerful. Nonetheless, taken in the context of explained variance only, the most powerful model (at 100m) is 37.2% more powerful than the least.

The relative importance of local physical characteristics against land-use factors is greatest up to 25m from the pond edge and greater than 800m from the pond edge explaining between 22.9% and 23.2% of water quality variability (Figure 4.3.6a). Between 50m and 700m the influence of the surrounding land use upon water quality increases and peaks at 100m where it accounts for 25.4% of water quality variation before reducing from 400m onwards. At 100m, the local physical model explains 16.2% with land-use factors partialled out, which in turn explain 16.1% of the variation. In total, the model accounts for 41.5% of variation in water quality (16.1 + 16.2 + 9.2 shared explained variance).

At 100m, the urbanisation gradient, PCA 1, derived from 10 'urbanisation indicators' was the most powerful explanatory variable followed by the proportion of mature vegetation (Figure 4.3.6b). Figure 4.3.7 displays the RDA for land-use at 100m alone where the amount of explained variance by land-use factors peaked. The first two axes were significant (<999 permutations, $P < 0.05$). 53.1% of constrained variance was explained on the first axis and 28.7% on the second axis. The first axis describes a gradient of urbanisation (Pearson's r = -0.90, $P < 0.001$) and the second axis describes a gradient of mature vegetation (Pearson's r = 0.83, $P < 0.001$). The water quality parameters significantly associated with each axis are listed in Table 4.9. Urbanisation had the strongest positive relationship with PO_4 and chlorophyll *a* and conversely with the inter-correlated variables of conductivity, Mg, alkalinity, Ca and SO_4. The proportion of mature vegetation was negatively correlated with temperature and dissolved oxygen, and to a lesser extent pH, whilst it was positively correlated with PO_4. Mn concentrations, although highly variable (Figure 4.3.3), were only weakly associated with an increase in mature vegetation, similarly, concentrations of NH_4 and K were not significantly related to any of the three axes.

The first two axes of the RDA of local physical factors were significant (999 permutations, $P < 0.05$) (Figure 4.3.8). 59.1% of constrained variance was explained on the first axis and 26.2% on the second axis. The first axis describes a gradient of shading as a consequence of increasing tree cover (Pearson's r = -0.92, $P < 0.001$), through to high macrophyte richness (and lack of shading, Pearson's, r = -0.64, $P <0.001$), and the second axis describes a transition between sites that do and do not receive surface run-off (Pearson's r = -0.84, $P < 0.001$). The water quality parameters significantly associated with each axis are listed in Table 4.10. Tree shading had a strong positive relationship with NH_4, PO_4 and Mn and a strong negative relationship with temperature, pH and dissolved oxygen. Dissolved oxygen in particular is related to macrophyte richness (Pearson's, r = 0.38, $P <0.05$). Tree coverage of 30% has previously been suggested to enhance macrophyte richness (Gee *et al.* 1997), although such a pattern was not clear here. The two most macrophyte diverse sites (sites 2, 23) had 38% and 41% tree cover respectively. A negative relationship exists between tree cover and macrophyte richness (Pearson's, r = -0.40, $P < 0.05$). In addition, a strong relationship between macrophyte richness and percentage concrete edge was found (Pearson's r = 0.59, $P < 0.001$). Surface run-off was strongly related to increased concentrations of Na and Cl, whilst it was only weakly negatively correlated to temperature, alkalinity, K and Zn.

The results of a Mantel correlogram suggested a lack of spatial autocorrelation in water quality composition between the study sites (Figure 4.3.9). At no distance was the relationship significant (Bonferroni correction, 999 permutations) and the Mantel correlation coefficients were low (maximum 0.065). Therefore, spatial independence amongst the study ponds was assumed.

Figure 4.3.5. Proportion of sites where plant growth is limited by PO_4 or NO_3 using the Redfield (Redfield 1934) ratio 16 NO_3 : 1 PO_4

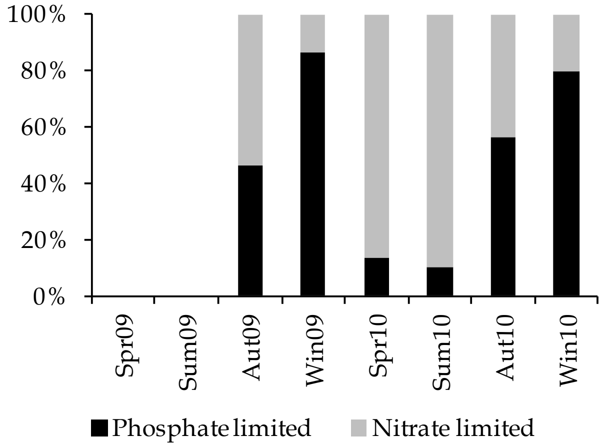
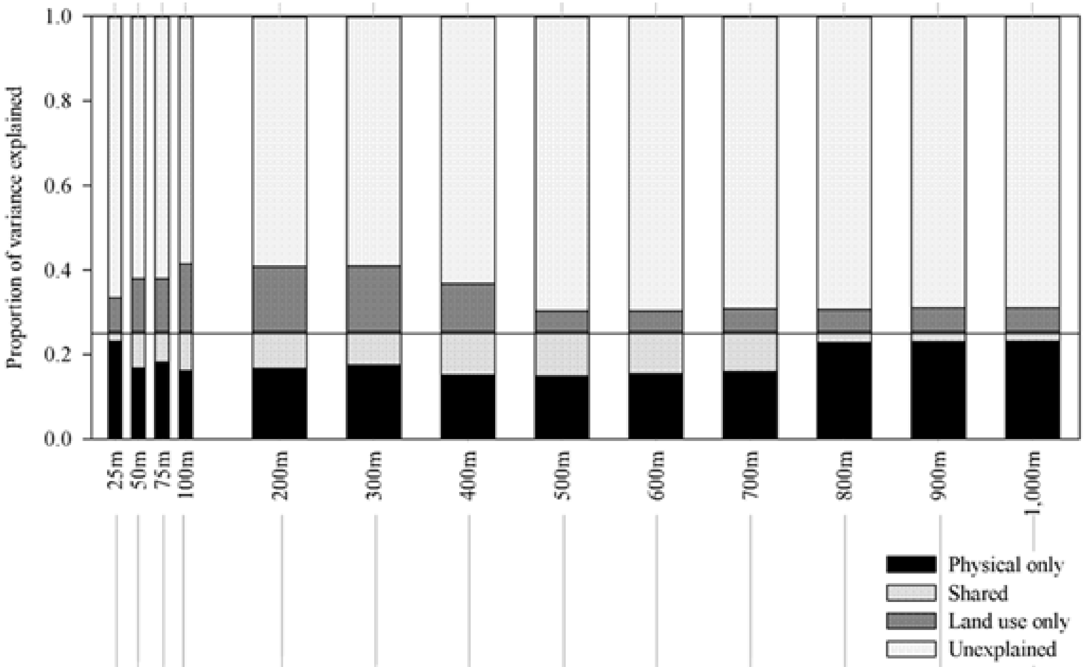


Figure 4.3.6. Transitions in the fitness of models for explaining water quality variation.
a) Stacked bar charts representing the partition of variance explained by the local physical factors alone, land-use at a given spatial extent and variance shared between both local physical and land-use factors b) Significant explanatory variables, in order of appearance within each stepwise RDA of land use explaining water quality variation at a given spatial extent

a)



b)

PCA 1	1	1	1	1									
Veg. >3m		2	2	2	3								
Simpsons				3	2								
Road surface					1								
Veg. 0-3m						1	2	1	1	1	1	1	1
Veg. 0m						2							
Broadleaved						3							
Gardens							1						
Urban							3	2	2	2			

Figure 4.3.7. RDA biplot of water quality constrained by land use at 100m only

Key:
DO – Dissolved oxygen
Temp – Temperature
NO_3 – Nitrate
Alk – Alkalinity
Mg – Magnesium
SO_4 – Sulphate
Ca – Calcium
EC – Electrical conductivity
K – Potassium
Mn – Manganese
NH_4 – Ammonium
PO_4 – Phosphate
Fe – Iron
Chla – Chlorophyll *a*
SS – Suspended Solids
Zn - Zinc

MatVg - Mature vegetation
PCA1 - Urbanisation
Simp - Simpsons index of land-use diversity

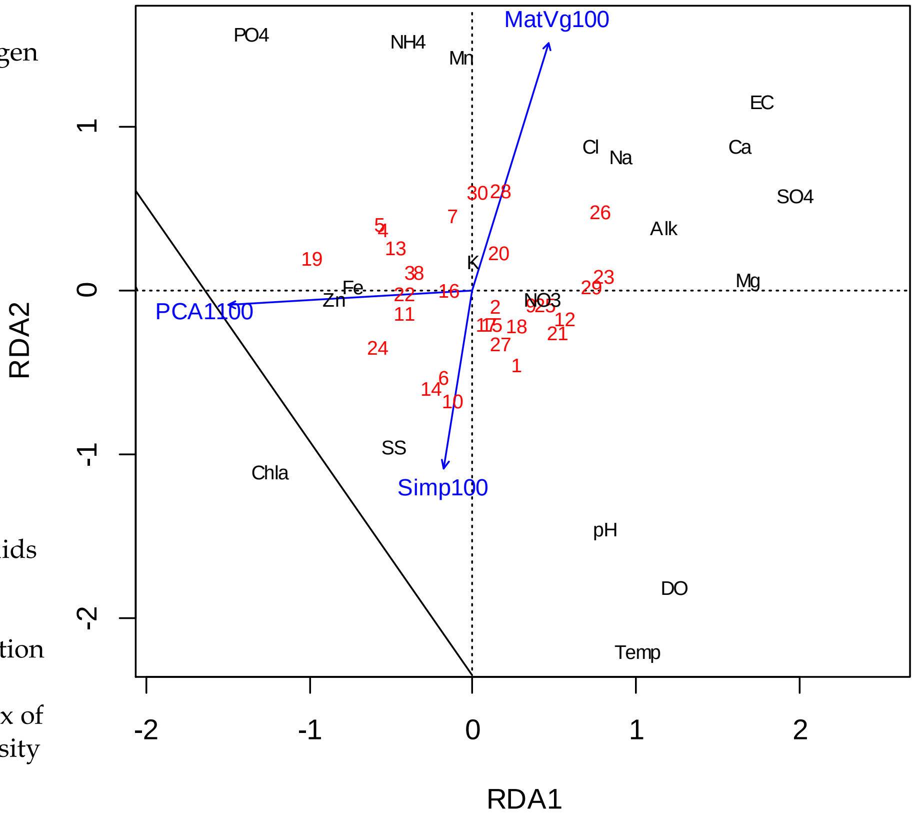


Figure 4.3.8. RDA biplot of water quality constrained by local physical factors only

Key:
DO – Dissolved oxygen
Temp – Temperature
NO_3 – Nitrate
Alk – Alkalinity
Mg – Magnesium
SO_4 – Sulphate
Ca – Calcium
EC – Electrical conductivity
K – Potassium
Mn – Manganese
NH_4 – Ammonium
PO_4 – Phosphate
Fe – Iron
Chla – Chlorophyll *a*
SS – Suspended Solids
Zn - Zinc

Mphyte - Macrophyte richness
RainRun - Rain run-off water source
TreeCov - Percentage tree cover

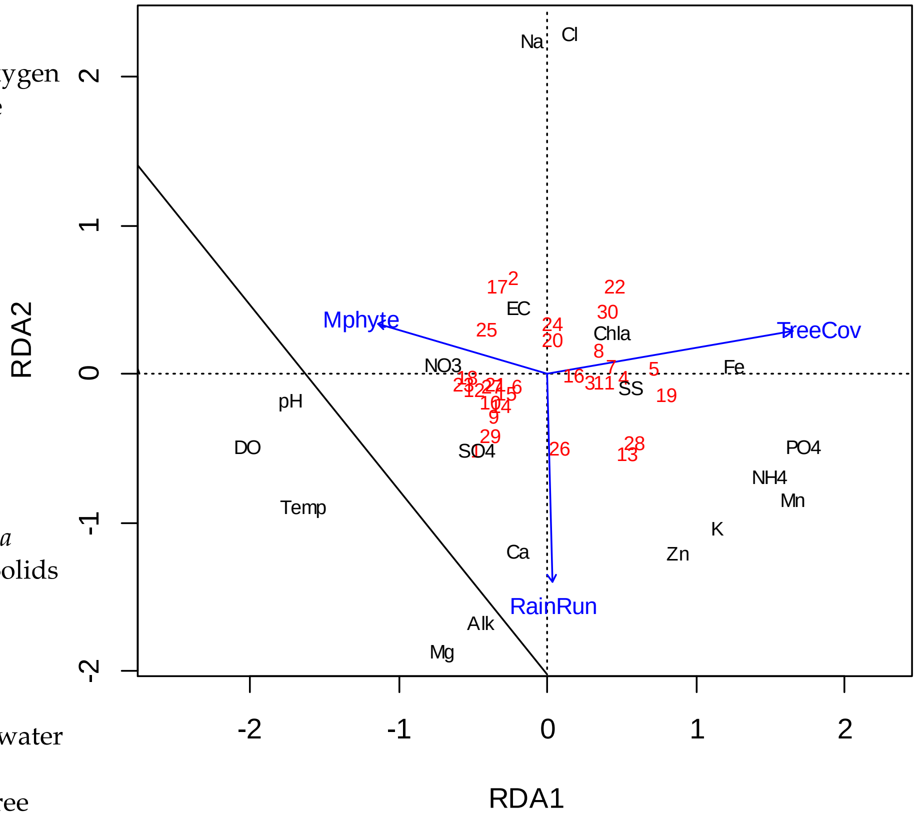


Table 4.9. Water quality variables significantly correlated to at least one of the two significant RDA axes (Pearson's correlation coefficient) using land-use at 100m

	EC	^oC	pH	DO	Alk	Ca	Cl	Mg	Na	NO_3	PO_4	SO_4	Fe	Mn	Zn	Chl*a*
Axis 1	***0.67***			**0.35**	**0.39**	*0.61*	*0.32*	***0.67***	**0.35**		***-0.50***	***0.79***			*-0.30*	***-0.50***
Axis 2		***-0.70***	**-0.37**	***-0.56***							**0.42**			**0.36**		

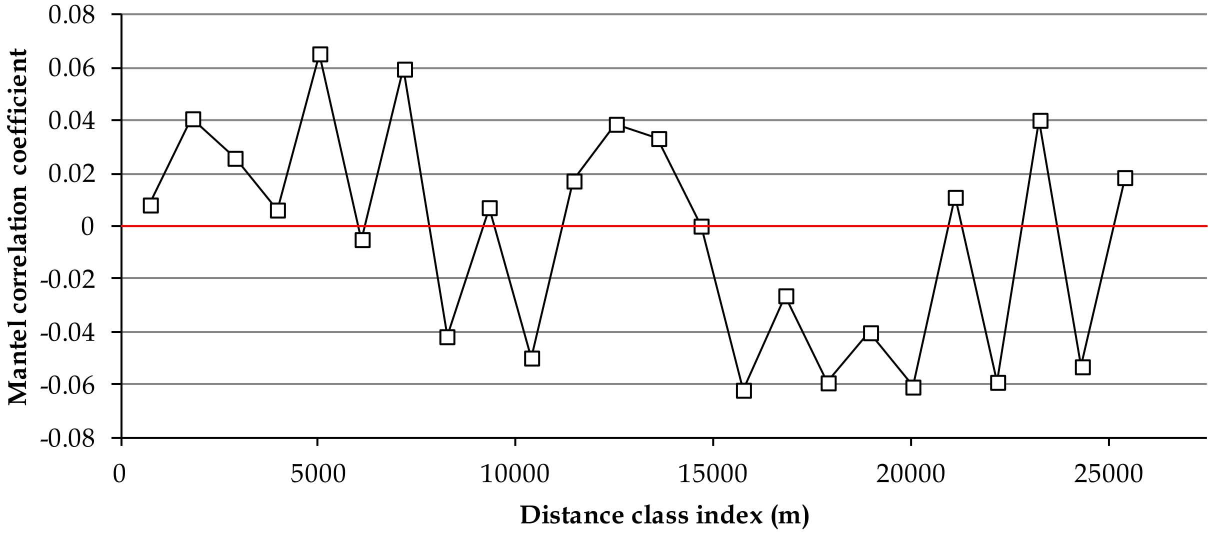
Significance of relationship, *P* < 0.1 = *I*, < 0.05 = **I**, <0.01 = ***I***

Table 4.10. Water quality variables significantly correlated to at least one of the two RDA axes (Pearson's correlation coefficient) using local physical factors

	^oC	pH	DO	Alk	NH_4	Cl	K	Na	Mg	NO_3	PO_4	Fe	Mn	Zn
Axis 1	***-0.75***	***-0.60***	***-0.78***		***0.51***		*0.34*		*0.30*	**-0.37**	***0.73***	*0.31*	***0.66***	**0.42**
Axis 2	**-0.39**			**-0.37**		***0.53***	*-0.31*	***0.61***						**-0.37**

Significance of relationship, *P* < 0.1 = *I*, < 0.05 = **I**, <0.01 = ***I***

Figure 4.3.9. Mantel correlogram for water quality composition across 30 pond sites. Distance class refers to the geographical distance between sites (24 bins). Solid squares show autocorrelations significant at Bonferonni corrected level (none).



4.4. Discussion

4.4.1. Water quality in the ponds of Birmingham and the Black Country

Water quality within urban ponds was extremely varied, possibly due to their small volume and catchment areas (Davies *et al.* 2008b). The ranges in water quality exhibited were more extensive than reported in other pond studies (Table 4.11), possibly due to increased complexity of land-use in BBC or accumulated contaminants from past pollution events (Durant *et al.* 2004; Graney and Eriksen 2004). Despite the difference in ranges, average values were consistent with those found in a similar study of urban drainage systems in the Netherlands (Vermonden *et al.* 2009). However, nutrient concentrations (NO_3 and PO_4) in our study were higher, and more similar to ponds in an agricultural landscape (Williams *et al.* 2004) and to those reported in a study of old industrial mill ponds (Wood and Barker 2000).

Urban ponds frequently failed to achieve environmental standards for water quality. The failings were particularly acute for PO_4, dissolved oxygen, Mn and NH_4. Against water quality standards for attaining 'good' ecological status under the WFD for PO_4 (for lakes), dissolved oxygen (for rivers) and NH_4 (for rivers) all the ponds sampled failed on at least one occasion in 2010. The 'good' ecological status of all surface waters by 2015 is a prerequisite of the WFD. However, the standards may be inappropriate for ponds, for example, average PO_4 concentration in 'minimally impaired' ponds was 0.07 mg/l (Table 4.11) and would classify as 'moderate' under the General Quality Assessment for rivers (National Rivers Authority 1994), suggesting relatively elevated levels in ponds are difficult to avoid. Eutrophication is a slow natural successional process in closed ponds over long periods of time (Khan and Ansari 2005) and in the absence of proactive management should be expected. Under the WFD, however, urban ponds such as studied here are likely to classify as artificial water bodies (AWB) as they exist through the creation of man where no water body previously existed (UK TAG 2008a). As such, AWBs will be required to achieve an alternate objective of at least 'good ecological potential' (GEP), which is assessed against both physical-chemical characteristics of the pond and the mitigation measures that may practicably be implemented.

Table 4.11. Comparison of average (minimum and maximum) values for water quality variables to 3 similar pond surveys

Variable	Unit	This study* (n=30)	Urban drainage systems[1] (n=36)	Agricultural landscape[2] (n=65)	Irrigation ponds[3] (n=55)
pH	-	7.6 (6.2-10)	7.6 (7.1-9.2)	8.1 (7.5-8.9)	7.2 (5.5 – 10.6)
Temperature	°C	12.3 (2.7-27.5)	-	-	-
Dissolved oxygen	% sat.	61.1 (6.0-185.2)	-	136 (63-255)	-
Conductivity	µS/cm	528 (45-2418)	527 (288-667)	654 (322-1265)	-
Cl	mg/l	47.8 (0.7-377.8)	44.7 (29.0-58.0)	_	-
NO_3	mg/l	5.2 (0.0-77.5)	3.4 (0.0-10.3)	3.7 (0.1-38.3)	-
PO_4	mg/l	0.32 (0.01-2.63)	0.11 (0.02-0.93)	0.27(0.00-2.49)[3]	0.14 (0.01 –1.28)
SO_4	mg/l	54.6 (0.9-1312)	-	-	-
Na	mg/l	26.7 (15.0-209.3)	28.4 (18.0-36.0)	-	-
NH_4	mg/l	0.98 (0.07-8.21)	-	-	-
K	mg/l	5.5 (0.6-17.1)	4.7 (0.8-10.6)	-	-
Mg	mg/l	7.6 (1.6-41.4)	8.9 (3.0-13.3)	-	-
Ca	mg/l	52.9 (12.0-413.1)	61.3 (14.0-88.0)	-	-
Suspended solids	mg/l	17.6 (0.9 –109.8)	-	73.0 (1.0-794)	17.4 (1.7 – 135)
Chlorophyll *a*	µg/l	60.8 (0.0 – 1494)	-	-	129 (3.1 – 577)
Fe	µg/l	480 (0-8139)	116 (25-653)	-	-
Mn	µg/l	473 (0-9211)	-	-	-
Zn	µg/l	80 (0-27)	31 (10-68)	-	-

[1] Vermonden *et al.* (2009), [2]Williams *et al.* (2004), [3]Akasaka *et al.* (2010)

[3]Total phosphorus

* Zero values represent concentrations below detection limits (Appendix 3)

Study mean NH_4 concentration was more than three times elevated when compared to a 'minimally-impacted' dataset (Biggs *et al.* 2005), although the values were similar to a study of two urban lakes in London (Birch and McCaskie 1999). In its own right NH_4 is largely harmless, aside from its potential contribution to nitrification processes, however, in higher pH conditions, >8, it can switch to the more toxic ammonia form (NH_3), which can be harmful to aquatic life. 50% of sites recorded pH values in excess of pH 8 at least once during the study, although the average was pH 7.6. High pH can be caused by photosynthesis in poorly buffered and nutrient-enriched lakes where carbon dioxide levels are reduced during photosynthesis (Lampert and Sommer 2007). The fluctuations in NO_3 concentrations, combined with NH_4 concentrations which generally reduced during autumn and winter may indicate a combination of reduced uptake by plant life and nitrification processes. Under oxic conditions NH_4 is first oxidised to nitrite (NO_2) and then to NO_3.

Low oxygen levels are frequently reported in shallow waters (Gee *et al.* 1997; Birch and McCaskie 1999; Angelibert *et al.* 2004) and typically occur during the summer months (Brönmark and Hansson 2005), the urban ponds studied here were no exception. Mean study dissolved oxygen levels were considerably lower than similar studies have reported, with the value from this study (61.1%) close to the minimum recorded in the study of ponds (Williams *et al.* 2004) within an agricultural region (63%, Table 4.11). Oxygen depletion during spring and summer is a consequence of a high density of primary producers, such as submerged macrophytes, phytoplankton and substrate-associated algae, which produce oxygen during the day and consume oxygen overnight. Many of the ponds studied here were nutrient enriched, which can encourage such conditions. The highest chlorophyll *a* concentrations were found on those sites with higher concentrations of PO_4. During the daytime this can cause super saturation and oxygen levels well in excess of 100%, overnight however, levels can decline rapidly and therefore account for the wide range of oxygen conditions sampled.

Chlorophyll *a* (algal biomass) did not appear to show a typical spring and summer maxima (Sommer 1986), instead, peak concentrations were recorded during the summer and winter months. On average, the study ponds had 30% tree coverage and 12 had coverage greater than 30%. During the summer and autumn months, high proportions of tree cover are likely to restrict light penetration to the water column and as a consequence, algal growth will be inhibited. As deciduous trees senesce, however, light exposure to the water surface will increase dramatically, and in nutrient rich environments dense stands of phytoplankton could be expected. Algal growth during winter periods may also be aided by PO_4 released during leaf litter breakdown and higher latent temperatures due to the effects of the urban heat island (UHI, Oke 1973). The reduction of PO_4 during winter may be attributed to uptake by phytoplankton, rather than an overall reduction in PO_4 availability. By comparison to an urban stream, Stoianov *et al.* (2000) found high summertime chlorophyll *a* concentrations between 100µg/l and 200µg/l. Chlorophyll *a* concentrations in this study are on average slightly lower, but considerably wider ranging, with a 1494µg/l maxima.

Mn was frequently recorded at concentrations that exceeded the environmental standard of 30 mg/l annual mean (Council Directive (EC) 76/464/EEC 1976). Mn is an essential nutrient for microorganisms, plants and animals. Dissolved Mn concentrations >0.8mg/l, as was the case in seven ponds, may cause fatality in some aquatic invertebrates, e.g. *Daphnia magna* (Straus 1820) (Reimer 1999). In greater concentrations, >1.9mg/l, as at two ponds, freshwater algae *Scenedesmus quadricauda* (Meyen 1829) may also be impacted (Fargasova *et al.* 1999). Freshwater molluscs and crustaceans are the most Mn sensitive freshwater macroinvertebrates (Howe *et al.* 2004), but would require concentrations well in excess of those recorded here to be impacted, for example, LC_{50} for *Asellus aquaticus* over 96 hours is 333mg/l (Martin and Holdich 1986).

Cl had a considerably wide range, as did Na (see Table 4.11). NaCl is the main ingredient of de-icing salts used on road surfaces during winter (Gobel *et al.* 2007) and previous studies have recorded high concentrations of Na and Cl in urban streams (e.g. Peters and Turk 1981; Jackson and Jobbagy 2005; Cunningham *et al.* 2009) and highway retention ponds (Le Viol *et al.* 2009). Winter 2009 was unusually cold and snowfall had left many roads snow-covered and some ponds frozen such that de-icing salts were distributed. The maximum concentrations recorded in this study, however, (Na 209mg/l and 201mg/l, Cl 378mg/l and 370mg/l) are unlikely to be high enough to be toxic to aquatic macroinvertebrates e.g. *Callibaetis fluctuans* (Walsh 1862) and *Physella integra* (Haldeman 1841) show considerable tolerance for increased Cl up to 5000 mg/l (Benbow and Merritt 2004), however, concentrations in the same magnitude as those found here (645 mg/l) in a controlled experiment reduced zooplankton abundance and consequently increase algal biomass (Van Meter *et al.* 2011). Moreover, the potential for complexation of other more toxic substances from sediments may be increased under higher Na and Cl concentrations (Le Viol *et al.* 2009). There has been little consideration for accumulated concentrations of Cl in freshwaters globally (Jackson and Jobbagy 2005).

Concentrations of suspended solids were generally lower and with a smaller range than that of agricultural ponds (Table 4.11) and comparable to that of the 'unimpaired' dataset (Biggs *et al.* 2005). Nonetheless, five sites were in excess of the 25mg/l standard (Council Directive (EC) 78/659/EEC 1978). Bioturbation by fish is a possible cause but, of these sites only one had fish presence and is therefore unlikely. However, the remaining four sites were typically unstable and had high wetland fluctuation index scores (WLFI), suggesting that the ponds received water from surface water run-off, which is likely to contain a high sediment load as particulate matter is washed off surrounding surfaces (Paul and Meyer 2001). Overall, a significant relationship was found between suspended solids and WLFI (Pearson's, r = 0.48, P = <0.01). Other potential sources of suspended solids may be wildfowl faeces or as a result of fish bioturbation (Ritvo *et al.* 2004).

4.4.2. Evaluation of the relevant spatial extent

Most variation was explained by models that included the physical urbanisation gradient derived from the 10 'urbanisation indicators' (PCA 1, Section 2.3.1.7). Peak explanatory power of the RDA process with variance partitioning was found at a 100m spatial extent (Figure 4.3.7). This result is consistent with others that have employed a similar analysis to the concentric ring approach here, albeit within agricultural landscapes (Declerck *et al.* 2006; Akasaka *et al.* 2010). Nonetheless, arable land-use is incorporated within the PCA of urbanisation indicators, represented by a low PCA 1 score. The urbanisation gradient, therefore, is a combination of increasing impermeable surface and decreasing proportion of arable land coverage. At close proximity from the pond edge (100m) a small amount of a land-use may exert a large influence within a small catchment (Davies *et al.* 2008b). This may indicates a need to reassess the landscape scale when considering ponds. Identification of the most meaningful spatial extent provides site managers information which can be incorporated in to management actions (Akasaka *et al.* 2010). However, the relatively small difference in variation explained by models between 100m and 300m suggests that individual sites may need further investigation within a range of spatial extents. It may also be necessary to conduct further study of relevant spatial extents if specific water quality variables are of concern, whereas this study considered the whole suite of water quality parameters.

4.4.3. The influence of local physical factors and land-use on water quality

The degree of urbanisation positively influenced nutrient concentrations, suspended solids, phytoplankton biomass and Fe and Zn concentrations (Figure 4.3.7). Impermeable surfaces are likely to form major sources of trace metals deposited by motorised vehicles (see Table 4.1). Nutrients can be washed off impermeable surfaces into freshwater systems that were deposited as fertilizer for green space (Carpenter *et al.* 1998), animal waste (Stoianov *et al.* 2000; Chaichana *et al.* 2011), or other organic matter e.g. leaf-litter. However, the impact of urbanisation may be twofold. More urban sites are likely to be under greater visitor pressure, which may result in more intensive green space management and will generate more waste in the form of litter, from domestic pets and through feeding of wildfowl which not only inputs nutrients directly, but also from attracting wildfowl in greater numbers.

Tree cover and macrophyte richness, which were inversely correlated (Figure 4.3.8), were found to be significant local physical factors in explaining variation in water quality; dissolved oxygen, pH, temperature, NH_4, PO_4 and Mn were all strongly interrelated (Table 4.10). Tree cover can influence water chemistry, organic matter input and shading (Walsh *et al.* 2003). At 30%, average tree cover was similar to that found in the Lowland Pond Survey (Williams *et al.* 1998b), which also found 20% of ponds to be >75% shaded and less than half <25% compared to 10% and 57% found here. The negative relationship between tree shading and macrophyte richness is consistent with that of Akasaka *et al.* (2010) who found reduced submerged macrophyte presence with broad-leaved trees along the pond edge. A peak macrophyte richness at a lower percentage tree cover than the suggested 30% by Gee *et al.* (1997) may be representative of the capacity for macrophytic growth as a function of pond engineering. Hard engineering has been found to hinder vegetation development directly (Parris 2006; Hamasaki *et al.* 2011) and indirectly by increasing wave action (Scheffer 2004). Tree cover is also a function of pond surface area, smaller ponds have a reduced volume to edge ratio, which equates to a higher proportion of littoral zone (Declerck *et al.* 2006) and therefore, a greater propensity for higher tree cover. However, the relative impact upon emergent plants in the riparian edge from tree-shading may remain consistent if considered as the percentage of pond *edge* shaded as pond size increases.

Tree cover can impact macrophyte growth by light exclusion, whether direct exclusion by tree canopy or through the accumulation of leaf litter. A lack of macrophytes can reduce oxygenation of the water column, as they produce oxygen through photosynthesis (Brönmark and Hansson 2005). Reduction of macrophytes from shading could cause a shift between macrophyte-rich clear water to a macrophyte-poor turbid state, dominated by floating plant species (Scheffer and van Nes 2007), though this is contingent on fish presence. Unlike Gee *et al.* (1997), who found no relationship between oxygenation and plant growth, dissolved oxygen concentrations were strongly related to macrophyte richness. Macrophyte species richness can increase along with pond surface area (Oertli *et al.* 2002; Sondergaard *et al.* 2005), here, peak macrophyte richness in a pond with surface area 4952m^2 is consistent with the results of Akasaka *et al.* (2010) where peak macrophyte species richness occurred at approximately 5000m^2. This may be explained by the presence of herbivorous fish in ponds of larger size (Scheffer *et al.* 2006), however, fish presence was not selected by the forward stepwise procedure for the RDA of local physical factors (Section 4.3.4).

Mature vegetation cover either directly over the pond or as land-use coverage within 100m was associated with lower oxygen and pH and increased nutrient concentrations, specifically NH_4 and PO_4. Allochthonous inputs of organic matter (e.g. leaf litter) are important basal resources in freshwater systems (Oertli 1993). Leaf litter in ponds may be sourced from trees directly overhanging the pond margins or sourced from their catchments in surface water drainage or wind action. Both NH_4 and PO_4 are derived from the decomposition of leaf litter (Sondergaard *et al.* 2002). The large amount of deposited organic material being broken down by bacteria not only increases the available PO_4 concentrations directly through nutrient recycling, but also by their oxygen consumption. As decomposing bacteria respire they deplete oxygen availability at the sediment surface. Under low oxygen conditions (low redox potential) complexation of trace metals, particularly Fe, with PO_4 does not occur as Fe is present in its soluble form (Fe^{2+}) rather than its non-soluble form (Fe^{3+}), which would otherwise complex with PO_4 and precipitate out of the water column. At the same time pH decreases with organic matter breakdown due to bacterial respiration which adds CO_2 to the system and releases H^+ ions. To further compound reduced oxygen levels, heavy shading prevents light from reaching the sediment, which inhibits oxygenating plant growth, which would reduce nutrient levels through uptake from both the sediment (Chambers *et al.* 1989) and the water column as with submerged vegetation (Cronk and Fennessy 2001). A reduced tree cover is likely to increase the abundance of algae upon organic sediments, which would produce oxygen and reduce PO_4 flow from the sediment by increasing redox potential. High concentrations of PO_4 in the urban ponds, therefore, could result from internal loading mechanisms, which can also release accumulated nutrients from the sediment that were as a result of past pollution episodes and leaf litter accumulation.

In high oxygen concentrations, the insoluble forms of Fe (Fe^{3+}) and Mn (Mn^{3+}) react with oxygen to form precipitates. Under low oxygen conditions, however, both are present in their soluble forms (Fe^{2+} and Mn^{2+}). Consequently, the relatively high concentration of both is indirectly attributable to tree cover, and hence the correlation observed with it, due to the respiration of decomposing bacteria on organic matter in the manner described above. Elevated winter oxygen levels resulted in more frequent Mn complexation reflected in the lower mean concentrations observed.

Na and Cl concentrations periodically increased, particularly during winter periods and were strong negative correlates to the presence of surface water run-off sources. In addition, a positive relationship with urbanisation, and therefore impermeable surfaces which may have provided Na and Cl sources did not arise. A confounding factor may be sites that receive storm water drainage from areas beyond the 100m scale. In this instance, a more detailed investigation into individual site infrastructure may be required.

Instream shading is widely reported to reduce summer temperatures in aquatic systems (Allan 2004 and references therein). Tree cover strongly reduced water temperatures, however, an insulating winter effect as has been found in forested streams (Gordon *et al.* 2004) where ponds with greater tree cover experience increased average temperatures, was not apparent. The presence of mature vegetation in the surrounding land-use, coupled with direct shading of the pond can create microclimates. Reduced temperatures can have implications for the physiological processes of aquatic organisms (Markarian 1980) and on the chemical composition of the pond environment due to its impact upon the oxygen capacity of the water i.e. lower temperatures increase oxygen capacity.

Finally, increased levels across the suite of water quality parameters were observed during summer 2010. August 2010, when the summer field survey took place, was the wettest month of 2010, with the nearest Met Office weather station recording 91mm of rainfall (Met Office 2012). It is possible that such wet conditions exposed the study ponds to increased amounts of run-off from the surrounding landscape. As a consequence, a larger input of pollutants may be expected from frequent storm events and subsequent surface run-off.

4.5. Conclusion

Water quality failed to attain environmental standards for several key variables, particularly those related to the WFD. Such failings pose significant concerns for the future of urban ponds, where they may be required to achieve 'good ecological potential' (GEP) by 2015 under the WFD. Although the environmental standards used here provide a useful proxy, improvements in the appropriateness of environmental standards for small ponds would provide more realistic aims within the context of the highly dynamic and in many cases, naturally eutrophic conditions.

Nonetheless, much of the variation in water quality can be explained by management of *in situ* local physical characteristics and land-use within close proximity of the ponds. Reduction in the amount of direct tree cover and the proportion of mature vegetation and impermeable surface within 100m is likely to result in a significant improvement in water quality, particularly in reducing PO_4 concentrations. Reduction in direct tree cover would allow better light conditions and improve the potential for plant growth which would, in turn, improve oxygen conditions and increase habitat complexity. Management recommendations have often focussed on an optimum 30% tree coverage (Biggs *et al.* 1994), however, the results of this study recommend a coverage of around 15-20% for maximising macrophyte growth. This figure may be dependent on the degree of hard engineering present within the site and the subsequent potential for macrophyte establishment.

Urbanisation was found to be a significant indicator of water quality, when considered at a small spatial extent. The most powerful model (<100m) in explaining water quality variability included a metric derived from 10 commonly used urbanisation metrics. However, a facet of urbanisation, for example, the proportion of urban areas (LCM 2007), remained in models up to 700m from the pond edge. Although the model that explained most variation in water quality was identified at 100m, a relatively small difference in explanatory power was found between 100m and 300m. As such, a conservative management approach would consider land-use up to 300m when identifying key land-use components.

At 100m, the amount of variance in water quality explained by local physical factors alone, and by land-use alone was very similar. However, regardless of spatial extent, local physical factors were more powerful than land-use. Consequently, management priorities should consider first what improvements to *in situ* characteristics such as the proportion of tree cover can be made. However, the amount of shared variance, which at 100m accounted for 22% of the total explained variance, indicates the intrinsic link between land-use and habitat. As such, pond management should not focus on either local physical factors or land-use in isolation.

This study has shown how the management of relatively small areas of land can improve conservation effort within urban areas with respect to aquatic resources. Improvement in the water quality of urban ponds is likely to improve biodiversity, amenity value and ecosystem services. The results here provide useful guidance to site managers. Individual sites will require further research in to water quality variation, using the implications of this study as a framework. This is particularly the case should the focus be on a particular variable or a few select variables which may be better controlled at longer or shorter spatial extents and by altering different components of the habitat. Even with appropriate management action, many ponds will remain susceptible to internal mechanisms for which more drastic action, such as dredging, may be required. Finally, management of land-use will be more relevant where much of the water source is from surface run-off. Pond sites receiving water from pipelines may render land-use largely irrelevant, in such circumstances investigation of infrastructure and drainage connections will be required to identify pollution sources.

5

Conservation value and determinants of macroinvertebrate community structure in urban ponds

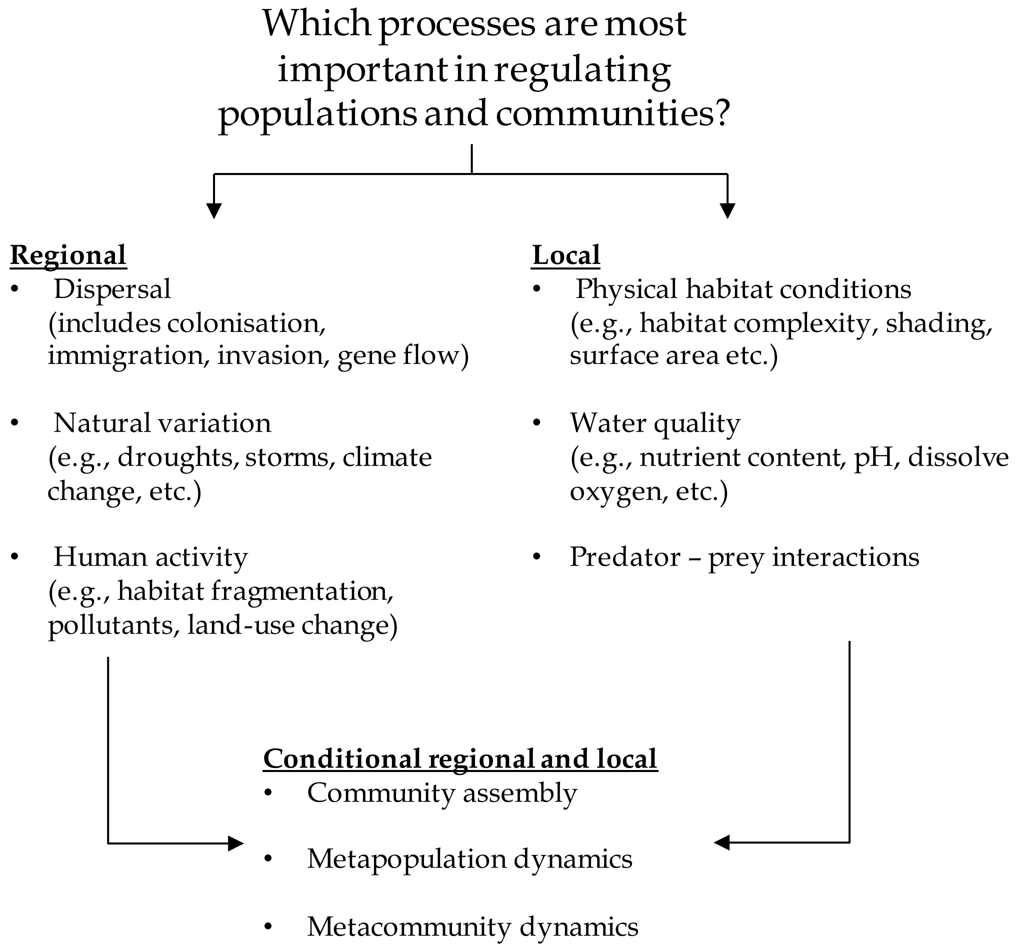
Chapter 5: Conservation value and determinants of macroinvertebrate community structure in urban ponds

5.1. Introduction

The urbanisation of catchments has profoundly altered streams and rivers, known as the 'urban stream syndrome' (Walsh *et al.* 2003), in which urban channels typically have flashy hydrographs, high concentrations of nutrients and other contaminants, altered channel morphology and low biodiversity (Wear *et al.* 1998; Paul and Meyer 2001; Roy *et al.* 2003; Walsh *et al.* 2003; Mancini *et al.* 2005). By comparison, knowledge of how urbanisation impacts upon ponds and their conservation value is poorly understood. In rural areas, ponds contribute more significantly to regional biodiversity than rivers and streams (Williams *et al.* 2004; Davies *et al.* 2008a). Ponds in urban areas are valued for their aesthetics and amenity use (Gledhill and James 2008; Sayer *et al.* 2008), however, their potential conservation value is often overlooked. The few studies that have investigated the conservation value of urban ponds have found that they are often undervalued and can support rare species and diversity comparable to ponds in rural areas (Wood and Barker 2000; Gledhill *et al.* 2008; Vermonden *et al.* 2009). Moreover, they stress the need to consider urban ponds as part of a wider pond network (Boothby and Hull 1997; Gledhill *et al.* 2008), particularly as pond numbers decline rapidly with urbanisation (Chapter 3). Furthermore, urban ponds offer a great potential focus for nature conservation, being logistically and economically relatively manageable by virtue of their small volume and limited catchment area (Biggs *et al.* 2005; De Meester *et al.* 2005; Davies *et al.* 2008b).

Ponds represent naturally isolated systems that rely on a frequent exchange of members of their ecological communities (Jeffries 1994; Briers and Warren 2000; Rundle *et al.* 2002), as a consequence their inhabitants are considered to exist in metacommunities (Leibold *et al.* 2004). The shape of a local community is contingent upon the relative impact of local and regional factors that include or exclude members of the metacommunity (Figure 5.1.1). Local factors refer to water chemistry, physical factors e.g. shading, habitat

Figure 5.1.1. Conceptual framework for studies of community structure assembly and regulation adapted from Bohonak and Jenkins (2003)



complexity, and predation. Regional factors refer to the availability of neighbouring habitats and the influence of surrounding land-use i.e. as a source of resistance to dispersal and of diffuse and point pollution.

Some important local factors that shape pond macroinvertebrate assemblages have been identified, including pH (Friday 1987; Biggs *et al.* 2005; Becerra Jurado *et al.* 2009; Vermonden *et al.* 2009), nutrients (Declerck *et al.* 2005; Vermonden *et al.* 2009; Hassall *et al.* 2011), shading (Gee *et al.* 1997; Lundkvist *et al.* 2002; Remsburg *et al.* 2008; Hassall *et al.* 2011), surface area (Brönmark 1985; Heino 2000; Biggs *et al.* 2005; Declerck *et al.* 2005), habitat complexity (Brönmark 1985; Wood *et al.* 2001; Bazzanti *et al.* 2003; Biggs *et al.* 2005; Declerck *et al.* 2005; Becerra Jurado *et al.* 2009) and fish presence (Fairchild *et al.* 2000; Wood *et al.* 2001; Scheffer *et al.* 2006). Interaction between local factors have also been highlighted, for example, the relationship between macroinvertebrate richness and nutrient concentration is likely mediated by the non-linear relationship between macrophyte richness and nutrient status (Declerck *et al.* 2005) such that the provision of habitat niches for macroinvertebrates peaks at mid-concentrations. Furthermore, as surface area increases so too does macrophyte species richness (and habitat complexity with the potential for macroinvertebrate diversity increase) and the probability of fish presence (Oertli *et al.* 2002; Sondergaard *et al.* 2005). Conversely, the degree of shading has an inverse relationship with surface area by virtue of a decreasing edge to volume ratio (Declerck *et al.* 2006). The relative influence of local factors can differ between systems, geographical regions and is dependent on the focal taxa. For example, no relationship between area and biodiversity was found in Alpine ponds (Hinden *et al.* 2004) and only for Odonata in Italian astatic ponds (Bazzanti *et al.* 2003). The presence of salmonid fish had no impact on macroinvertebrates (Gee *et al.* 1997) and certain species of coarse fish are known to preferentially target macroinvertebrate predators (Fairchild *et al.* 2000; Schilling *et al.* 2009). Frequently, however, studies of factors that shape aquatic communities are confined to local scale influences and do not take in to consideration regional effects (Cottenie and De Meester 2003).

Factors that impinge upon successful dispersal between communities are key to the balance between locally or regionally dominated metacommunities (Cottenie 2005). Yet, only recently have studies of pond macroinvertebrate communities begun to incorporate regional factors as determinants alongside local factors. Similarity in macroinvertebrate communities has been found to increase with decreasing spatial separation of sites (Oertli *et al.* 2008; Zealand and Jeffries 2009) and to exist up to 13km with the effect of local factors removed (Briers and Biggs 2005). Gledhill *et al.* (2008) found pond density within adjoining 1km grid squares to be a strong determinant of macroinvertebrate species richness in a study of urban ponds in north west U.K. suggesting dispersal limitation. Conversely, a mean distance of just 0.5km between ponds has been enough to shift the balance toward local factor dominance on macroinvertebrate community structure (Urban 2004) suggesting less dispersal limitation. Similarly, Heino and Mykra (2008) found a only very weak influence of spatial location on stream insect assemblages, whilst local factors were able to account for up to 31.7% of variation. Such differences in the influence of regional factors between studies may arise since the influence of neighbouring communities is mediated by the intra-habitat (patch) quality and inter-habitat distance (Jeffries 2005). Moreover, inter-habitat distance is affected by the intervening landscape (Leibold *et al.* 2004) i.e. different land-cover types present different levels of risk and benefit (Fahrig 2007). The inclusion of landscape resistance scores i.e. high scores for obstacles, facilitates the computation of 'effective distance' (Spear *et al.* 2010) between habitats. Least cost path analysis is one such method, within a geographical information system (GIS), by which this can be calculated. Least cost path analysis has already been used to identify optimum routes for the dispersal of a wide range of animals (Sutcliffe *et al.* 2003; Larkin *et al.* 2004; Desrochers *et al.* 2011), although use of least cost analysis has not yet been applied to aquatic communities due to the lack of empirical data on their terrestrial movement. However, the recent use of isotopic marking (Caudill 2003; Briers *et al.* 2004) and research into light pollution (e.g. Horvath *et al.* 2009; Kriska *et al.* 2009), which is particularly relevant in urban areas, are examples of recent research which can inform the application of landscape resistance scores to facilitate calculation of effective distance. Using a GIS in conjunction with sophisticated network analysis packages (e.g. EDENetworks, Kivela *et al.* 2011 and Conefor Sensinode, Saura and Torne 2009) it is now possible to calculate a suite of metrics that can incorporate effective distance and assess a variety of connectivity roles provided by a single pond within the wider pond network, as well as more conventional Euclidean distance and density based measures.

The relevance of regional factors to habitat, species or communities has been investigated at multiple spatial extents i.e. distances from the pond edge (Pellet *et al.* 2004; Declerck *et al.* 2006; Gledhill *et al.* 2008; Akasaka *et al.* 2010; Yu *et al.* 2012), termed 'concentric ring analysis' (Pellet *et al.* 2004). Similarly, several studies have applied a series of theoretical dispersal thresholds to assess the connectivity of current or proposed reserve networks (Hinsley and Bellamy 2000; Gurrutxaga *et al.* 2011; Laita *et al.* 2011). The relevant spatial extent for members of the aquatic community is likely to change with the focal organism(s) as a consequence of their individual dispersal traits and behavior during emergence and/ or dispersal. For example, Odonata are known to be highly philopatric and are unlikely to disperse far from their natal pond (Conrad *et al.* 1999), Coleoptera and Hemiptera, which are known to be strong fliers (Davy-Bowker 2002), disperse solely for the purposes of finding new habitat (Briers and Warren 2000; Lundkvist *et al.* 2002) and Gastropoda are passive dispersers (Kappes and Haase 2012). Further breakdown of the macroinvertebrate community into groups that are characteristic of the pond environment could provide greater detail upon which to base management decisions for the conservation of specific target taxa.

This study aims to provide information which can be readily incorporated to management practice. First, the conservation value of urban ponds is assessed and urban pond types that promote high conservation value and species richness are identified. Secondly, the most influential controls on overall macroinvertebrate community composition are considered and the relative importance of local (water quality and physical) and regional (land-use and connectivity) factors are established using variance-partitioning (Borcard *et al.* 1992). Finally, the local and regional drivers of taxon richness within major macroinvertebrate orders are identified.

5.1.1. Objectives and hypotheses

This chapter characterizes macroinvertebrate assemblages in 30 ponds across an urbanisation gradient, and relates assemblage taxonomic composition and taxon richness to a suite of variables that characterise water quality, habitat, spatial connectivity and land-use, to test the following objectives and hypotheses.

1. To characterize the macroinvertebrate communities of urban ponds and assess their conservation value.

   Hypothesis 1
   The conservation value of urban ponds is low due to the influence of urbanisation. Between-site diversity, however, is high due to local differences in habitat and land-use

2. To assess the importance of local (water quality, habitat) and spatial (connectivity, land-use) variables to community structure.

   Hypothesis 2
   Local factors are relatively more influential than regional due to the generally strong dispersal abilities of aquatic insects; however, the traversibility of the landscape and spatial configuration of pond habitat across the landscape is also important

3. To predict within-order richness of characteristic pond macroinvertebrate groups.

> Hypothesis 3
>
> *Important local and spatial factors differ between insect orders and are reflective of their dispersal traits and habitat requirements*

5.2. Methods

5.2.1. Study site selection

Thirty ponds of contrasting surrounding land-use were selected from an estimated 1023 sites in Birmingham and The Black Country (BBC). For the full site selection process see Section 2.1.

5.2.2. Field campaign

Field surveys were carried out in late spring/early summer 2009 and late summer 2009 (Table 4.3). On each visit macroinvertebrate and water samples were collected and observations were made with regard to the local physical conditions.

5.2.2.1. Macroinvertebrates

Macroinvertebrate sampling was based on the protocols of the National Pond Survey (Biggs *et al.* 1998). In each pond, sampling was conducted on a per unit effort basis (total 3-minute plunge/sweep sampling) with sampling time divided equally among mesohabitat patch types (e.g. 3 mesohabits identified = 60 seconds of sampling in each) in the pond. Within mesohabitats, sampling time was divided into replicated 10 second plunge/sweep samples, where each subsample was an initial 'plunge' of the net onto the sediment followed by 10 seconds of netting in a figure of eight motion just above the sediment surface. Each sample was collected and preserved in 70% Industrial Methylated Solution (IMS) for later sorting and identification.

In the laboratory, macroinvertebrates were sub-sampled from each plunge/sweep sample due to the large volume of material collected, with a minimum of 25 individuals collected for each 10 second plunge/sweep sample, or a minimum of 450 individuals per pond on each sampling occasion. Macroinvertebrates were identified to the lowest practicable taxonomic unit (usually species, Section 2.4). Data from each 10 second sample were pooled to generate a single dataset for each pond on each sampling occasion. Unless stated, use of macroinvertebrate data also refers to pooled data from the two sampling periods.

Table 5.1. Sampling periods and dates of sampling

Season and hydrological year	Sampling dates
Late spring/early summer 2009	20th May – 11th June 2009
Summer 2009	3rd August – 14th August 2009

5.2.3. Local factors

5.2.3.1. Water quality

Where practicable, a composite water sample was collected at each site from the pond margins near the inflow, outflow and at an approximately equidistant point between the two. In the absence of a clear inflow or outflow, samples were evenly spread around the pond perimeter. Contact with vegetation or the bottom substrate was carefully avoided during water sampling, as was collection of surface debris. Once collected, samples were kept cool, filtered (GF/C, 1.2µm) and returned to the laboratory for determination of major ions, trace metals, suspended solids and chlorophyll abundance. Triplicate *in situ* measurements of pH, electrical conductivity (µS/cm), temperature (°C) and dissolved oxygen (% saturation) were taken and the average measurements were calculated. For each determinand, data were averaged across the two sampling periods to provide a single datum point for each site. Further detail on water quality sampling and analysis methods are discussed in Section 2.2.

5.2.3.2. Physical characteristics

Single measurements of pond surface area and percentages of surface area classified as open water, shaded, riparian vegetation and floating vegetation were all derived from a combination of digitised field notes and Normalised Difference Vegetation Index (NDVI) and photogrammetrically derived height GIS layers (Section 2.3.1.6) against digital pond outlines. The percentage of pond bank that was made from concrete was also calculated in a GIS from digitized field notes. Water sources (i.e. stream inflows, groundwater, surface run-off or building run-off) and fish presence information was gathered from stakeholder knowledge and visual inspection. A water level fluctuation index (WLFI) was calculated as the standard deviation in depth between surveys (taken quarterly between 2010 – 2011) measured at set points within the wetted-perimeter of each pond.

Macrophyte presence absence was recorded from within identified mesohabitats sampled for macroinvertebrates (Section 2.4) using Haslam *et al.* (1982) for identification. A full macrophyte species list is included in Appendix

5.2.4. Regional factors

5.2.4.1. Land-use

A GIS combining four land cover layers was used to generate detailed land-use data (Section 2.3). Land-use metrics were calculated for 13 cumulative spatial extents, measured from the pond edge in each case to 25m, 50m, 75m, 100m and then 100m increments to 1000m.

Ten 'urbanisation indicators', selected from the land-use variables (indicated by ^{1} in Table 4.4) were incorporated into a PCA to create a score of urbanisation at each spatial extent (Section 2.3.1.7). The indicators chosen were those that were used commonly to measure urbanisation (see Hahs and McDonnell 2006 for a review) and closely linked to a general perception of urbanisation. PCA 1 was related to a gradient from a high proportion of arable land to a large degree of impermeable surface, PCA 2 is a gradient between high population density and an increased percentage of people in rural employment.

Land-use coverage was extracted from a GIS using the thematic raster summary tool within the Hawth's Tools (Beyer 2004) extension for ArcGIS 9.3.

5.2.4.2. Connectivity

In total, eight connectivity metrics were derived for each pond (Table 5.2). Four were calculated within the spatial extents 50m, 100, 250m, 500m, 1000m, 1500m, 2000m, 2500m. These were the number of ponds to which a focal pond is within a specified distance (degree centrality, DC), the number of times a focal pond is used as a stepping stone between any two others to maintain connectivity in a network (betweeness centrality, BC), and the proportion of pond and wetland (non-specific) habitats were calculated separately at different spatial extents. BC scores were limited to extents up to 500m, beyond which the dataset could not provide complete coverage of the potential pondscape.

Area-informed connectivity metrics have been shown to outperform distance metrics in ecological studies (Tischendorf et al. 2003). Therefore, in addition to the four metrics outlined above the area-informed metrics, area-proximity (AP) and effective distance-area (expressed as area-cost, AC) were derived that account for both the variability in inter-habitat distance and the relative contribution made to the regional species pool using surface area as a proxy for productivity. First, for AP, the surface area of the none-focal pond, or source (e.g. A1, Figure 5.2.1) is divided by the Euclidean distance between it and the focal pond (e.g. D1), which is calculated from pond edge to edge. The final metric is the sum of each of these separate calculations within a predefined radius i.e. AP = (D2/A2) + (D3/A3) +... (D7/A7) (Figure 5.2.1). For calculation of a metric that considered inter-habitat landscape resistance (AC), Euclidean distance (D) was substituted with effective distance (ED, Figure 5.2.2) derived by the formulation of a resistance landscape where each land-use classification was scored between one and 1000. For example, tall buildings scored 1000 and vegetated still water one (Appendix 15). Least-cost paths were calculated using PathMatrix (Ray 2005) for ArcGIS 3.2 (ESRI 1999). The least cost paths represent a proxy for landscape resistance between ponds rather than a single preferred dispersal route. Area-informed metrics were calculated for 1km and 2km only, due to the disparate nature of the pond network i.e. the longest minimum distance between a studied pond and another was exactly 1000m. Consequently, below this distance calculation of the metric was not possible for all 30 study sites. To compensate for this, the *AP* and *AC* metrics were also calculated for the nearest 1, 2, 3, 4, 5, 10, 25 and 50 neighbours.

A final two connectivity metrics were none area informed connectivity metrics. These were the average Euclidean and cost distance to the nearest 1, 2, 3, 4, 5, 10, 25 and 50 neighbours.

5.2.5. Statistical analyses

5.2.5.1. Characterizing urban pond types of high conservation value

First, Two Way Indicator Species Analysis (TWINSPAN, Hill 1979) was used to identify site characteristics that were associated with high conservation value. TWINSPAN uses correspondence analysis to iteratively divide a set of sites into clusters that have similar ecological communities, based on taxa relative abundance. At a group split, the species that are typical of one side of the dichotomy and not the other are considered good indicators of the ecological conditions in the cluster of sites at which they are present (Lepš and Šmilauer 2003). The predetermined identification of four TWINSPAN groups facilitated comparison of local and regional factors that were associated with the site clusters and contributed to shaping the macroinvertebrate communities. ANOVA was used to test for significant differences in local and regional factors between the TWINSPAN groups using the Tukey-Kramer *post-hoc* test, which allows for unequal group memberships. TWINSPAN was carried out using Community Analysis Package 1.4 (Pisces Conservation Ltd. 1999). ANOVAs were carried out in IBM SPSS 19.0 for Windows (SPSS Inc., Chicago, IL, USA).

Conservation value of the urban pond sites was assessed using the Community Conservation Index (CCI), which accounts for community richness, as well as the relative rarity of species present (Chadd and Extence 2004). Using this measure, taxonomically rich sites can obtain very high values of CCI which are comparative to those obtained from sites supporting nationally rare species. The CCI method assigns a conservation score to each species based on their relative rarity, which is averaged across the site and then multiplied by a community score, derived from either the rarest taxon present in the community, or the Biological Monitoring Working Party (BMWP) score (Chesters 1980); the latter is used here. CCI was calculated in SAFIS (Site Analysis for Freshwater Invertebrate Surveys) (Chalkley 2012). Taxon richness, species richness and Simpsons (E_{1-D}) evenness measure were also calculated using all individuals irrespective of the level of taxonomic identification.

The Simpson's (E_{1-D}) measure of evenness (Krebs 1999) is an extension of the original Simpson's diversity index (Simpson 1949) and measures community evenness on a scale between 0 and 1. As the community becomes more even, the Simpsons (E_{1-D}) increases (Magurran 2004). Rare species were identified as Red Data Book (1 – 3), Notable (A or B) or Regionally Notable (NR), the definitions of which are listed in Appendix 17, after Chadd and Extence (2004).

5.2.5.2. Separating out local and regional effects

Forward selected local (water quality and local physical) and regional (land-use and connectivity) factors from section 5.2.5.1 were used in variance partitioning (Borcard *et al.* 1992). For regional factors the procedure was carried out for parameters derived at each of the spatial extents. The model with the highest explanatory power was retained for use in variance partitioning. This allowed the proportion of explained variance in macroinvertebrate community that was attributable to each of the four variable groups (water quality, local physical, land-use and connectivity) both alone and in combination to be calculated. This was achieved using the 'varpart' procedure (for the full procedure, see Appendix 18) within the vegan package (Oksanen *et al.* 2012) for R statistical package version 2.15.1 (R Core Team and contributors worldwide 2012), by undertaking 15 direct CCA analyses and 27 CCA analyses with the effects of one or more of the variable sets (e.g. local or regional factors) partialled out. A further 11 fractions of explained variance are estimated by subtraction. Automatic forward stepwise model building was used for the CCAs using the function 'ordistep' in the vegan R package, which uses permutation tests to obtain *P* values (<999 permutations).

Table 5.2. Summary of connectivity metrics

Metric	Description	Relative complexity	Connectivity type	Package used
Degree centrality (*DC*)	Pond counts within selected spatial extent (pond density)	Low	Structural	Hawth's Tools (ArcGIS 9.3)
Water	Surface water as a proportion of spatial extent (irrespective of water boy type)	Low	Potential	Hawth's Tools (ArcGIS 9.3)
Pond	Total surface water of ponds as a proportion of spatial extent	Low	Potential	Hawth's Tools (ArcGIS 9.3)
Euclidean distance to neighbour(s)	Straight-line distance to nearest neighbour and average distance to nearest 1, 2, 3, 4, 5, 10, 25, 50 neighbours	Medium	Structural	R, Excel
Proximity/ area (*AP*) to neighbours	Straight-line distance to nearest neighbour and average distance to nearest 1, 2, 3, 4, 5, 10, 25, 50 neighbours weighted by habitat area and within 1 and 2 km.	Medium	Structural	R, Excel
Betweeness centrality (*BC*)	Count of the number of times focal pond is required as a stepping stone between any combination of 2 from the pondscape	High	Structural	EDENetworks
Effective distance to nearest neighbour (s) (*AC*)	Accumulated cost along least-cost distance path derived from resistance landscape to nearest neighbours and average effective distance to nearest 1, 2, 3, 4, 5, 10, 25 neighbours	High	Potential	PathMatrix[1] (ArcGIS 3.2), Excel
Effective distance/ area to neighbours	Accumulated cost along least-cost distance path derived from resistance landscape to nearest neighbours and average effective distance to nearest 1, 2, 3, 4, 5, 10, 25 neighbours weighted by habitat area and within 1 and 2 km	High	Potential	PathMatrix[1] (ArcGIS 3.2), Excel

[1] Ray (2005)

Table adapted from Magle *et al.* (2009)

Figure 5.2.1. Example calculation of area-proximity metric for a focal pond and neighbouring ponds within a 1km radius

Example

$A_1 / D_1 = 389 / 475 \quad = 0.82 +$

$A_2 / D_2 = 115 / 5 \quad = 23.0 +$

$A_3 / D_3 = 112 / 17 \quad = 6.59 +$

$A_4 / D_4 = 52 / 737 \quad = 0.07 +$

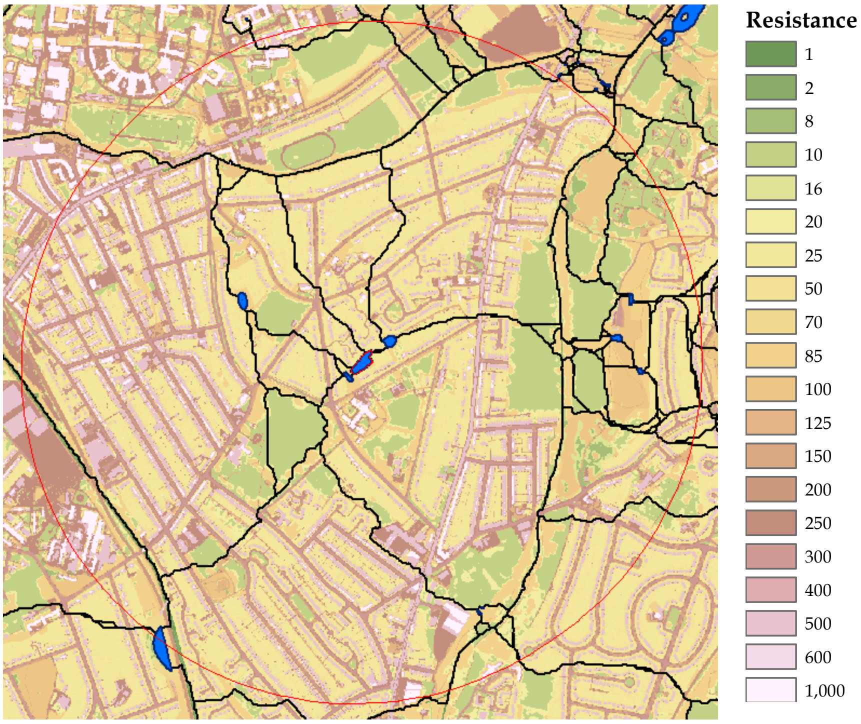
$A_5 / D_5 = 68 / 759 \quad = 0.09 +$

$A_6 / D_6 = 45 / 775 \quad = 0.06 +$

$A_7 / D_7 = 956 / 995 \quad = \underline{0.96}$

$AP1km \quad = 31.6$

Figure 5.2.2. Example result of least-cost paths and resistance landscape for a focal pond and neighbouring ponds within a 1km radius. Black lines indicate the least-cost paths (effective distance, *ED*) between any combinations of two ponds across the pond network



5.2.5.3. Key determinants of taxon richness within major macroinvertebrate orders

In order to explore the associations of local and regional factors with the taxa richness of major macroinvertebrate orders, generalized linear models (GLMs) were fitted to counts of species within each of the orders Gastropoda, Coleoptera, Trichoptera, Odonata and Hemiptera using an information-theoretic approach (Burnham and Anderson 2002). The approach compares the fits of a suite of candidate models using Akaike's information criterion (AIC), corrected for overdispersion and small sample sizes (AIC_c, Burnham and Anderson 2002). All combinations of independent variables were ran within models with no more than three parameters in order to improve signal to noise ratios. Use of AIC allows for models with different numbers of independent variables to be directly compared to each other. The best fitting model has the smallest AIC_c value, although the absolute size of AIC_c is unimportant. The models are ranked according to their AIC_c score and the difference between the smallest and any other model indicates the relative support for each (termed ΔAIC_c). To allow for model comparison, the Akaike weight was calculated for each ($wAIC_c$) (Burnham and Anderson 2002). The sum of Akaike weights is equal to one, such that they may have a probabilistic interpretation i.e. $wAIC_c$ indicates the probability that the model in question would be selected as the best fitting if data were collected again (Whittingham *et al.* 2005).

The number of models that comprise the 95% confidence set fitted to each data set is also reported. This is the number of candidate models for which the sum of their $wAIC_c$ is 0.95 and is therefore the set of models within which there is 95% confidence that the best approximating model to the true model is contained. It is the best *approximating* model as it is unknown whether the set contains the true model (Whittingham *et al.* 2005). The size of the confidence set, relative to the total number of candidate models further indicates the relative importance of the predictor variables i.e. if the confidence set is small fewer variables have greater control on the response.

A model averaging technique was applied in order to calculate the probability that a given variable would be selected within the best approximating model. Here, the sum of $wAIC_c$ for the models containing a given variable denotes the likelihood of its selection within the best approximating model relative to all other variables and weighted by model plausibility. Calculation of selection probabilities (SP) for each variable provides extra information over the selection of a single model (as in stepwise) as it is feasible for a variable to attain a high SP without being selected in the most parsimonious model, with the lowest AIC_c. Conversely, a review of a number of the top ranked models can reveal variables that without having being a significant parameter, by virtue of selection in the highest ranked model, has a high SP, despite a lack of inclusion in subsequent models.

GLMs were fitted using the 'glm' function implemented in the R stats package version 2.15.1 (R Core Team and contributors worldwide 2012), with a Poisson distribution and a log link function. For Odonata, due to the high occurrence of zero counts (37% of sites) a zero-inflated GLM model was fitted using the 'zeroinfl' function implemented in package 'pscl' (Zeileis *et al.* 2008). To correct for overdispersion detected in models for Coleoptera, Trichoptera, Hemiptera and taxa richness a negative-binomial model was subsequently used. Up to 317 candidate models were ran within each local and regional variable set, in total 8490 GLM models were constructed.

Prior to all statistical analyses, datasets were checked for outliers and statistical test assumption conformity according to Zuur *et al.* (2010). Macroinvertebrate species that were recorded in < 5% of the study sites were excluded from ordination and TWINSPAN analyses because rare species may confuse underlying patterns in community analysis (Faith and Norris 1989; Heino and Mykra 2008). Macroinvertebrate abundance data were transformed prior to CCA after Preston (1962): Preston class = ^{2}log (abundance + 1). Independent variables used in CCA and GLMs were either log(n+1), square-root or arcsine transformed to improve normality and homogeneity of variance where necessary as tested by Shapiro-Wilk and Levene's tests carried out in using IBM SPSS statistical package version 19 (IBM, Armonk, New York). Water quality variables were typically log(n+1) transformed, arcsine transformation for land-use coverages (proportional data) and distance measures were usually square-root transformed. Collinearity in independent variables was minimized by removing one of any two variables that were greater than +/- 0.7 correlated (Pearson's correlation coefficients). The retained variable was the one considered most ecologically relevant and / or, if land-use, derived from the most contemporary land-use dataset (Section 2.2 for dataset metadata). Prior to GLM construction, Variance Inflation Factors (VIFs) were also checked using the package 'nlme' (Pinheiro *et al.* 2012), with reference to the pairwise plots, variables were removed until all VIF < 3 (Zuur *et al.* 2010). Models were validated by the production of histograms and normality plots to assess the spread of residuals, which were also checked for overdispersion by dividing the deviance statistic by the degrees of freedom.

5.2.5.4. Spatial autocorrelation

Spatial autocorrelation occurs when values at one locality are dependent on the values at another. For analysis of macroinvertebrate communities this could be borne out of frequent exchanges of individuals by dispersal between two sites in close proximity. The study sites were tested for spatial independence at distance intervals using a Mantel correlogram using the function 'mantel.correlog' in the vegan package (Oksanen *et al.* 2012) for R statistical package version 2.15.1 (R Core Team and contributors worldwide 2012). The Mantel correlogram is an effective tool for detecting correlation between datasets (Borcard and Legendre 2012) and tests for community similarity using Bray-Curtis dissimilarity as a function of geographic distance classes (24 classes used). Mantel tests (Mantel 1967) assess the magnitude of the correlation between two or more symmetric distance matrices. Significance of the matrix correlation coefficient was tested by 999 random permutations with a Bonferroni correction. Violations of spatial independence between the study sites would indicate the presence of spatial autocorrelation within the dataset, which may ultimately lead to biased conclusions.

5.3. Results

A total of 193 taxa from 14 orders were found across the 30 ponds, with data pooled from spring and summer 2009. 157 were identified to species level (full species list in Appendix 19). The most species rich orders were Coleoptera (47 species), Hemiptera (31), Trichoptera (24), Gastropoda (17) and Odonata (15) (Table 5.3). Across ponds, taxon richness varied from 17 to 82 (median 48). Seventeen of the 30 sites studied contributed at least one unique taxon to the overall taxonomic community. Hemiptera richness and Coleoptera richness showed the greatest ranges from one to 17 and 0 to 16 species per site respectively. Simpsons (E_{1-D}) values (minima 0.05, maxima 0.24) suggested that most sites had an uneven community dominated by the high abundance of a few taxa.

5.3.1. Conservation value of urban ponds

Conservation value of the study sites was assessed using the Community Conservation Index (CCI), derived by Chadd and Extence (2004). Sites were generally of *'fairly high conservation value'* according to the average CCI scores for pooled data across the two seasons. CCI scores in this range are typical of sites supporting at least one uncommon species, or several species of restricted distribution and or a community of high taxon richness. One Red Data Book (RDB3 – Nationally rare) (Hyman and Parsons 1992), four Notable B and three Notable Regional species were recorded (Table 5.4, for descriptions refer to Appendix 17). Pond 21 was the only site to be of potentially *'very high conservation value'* (CCI = 26.4). Pond 21 has high taxon richness and supports the only RDB3 listed species found during the study *Hydrochus elongatus* (Schaller, 1783). *H. elongatus*, is a water scavenger beetle characteristic of ponds and drains and often is found amongst reed habitats, which were abundant at site 21.

Species classified as Notable B are *Hydrochus lividus* (Schaller 1783), a greenish-brown water scavenger beetle characteristic of ditches, fens and ponds, *Rhantus suturalis* (Macleay 1825), a diving beetle of silt and detritus pools, *Cercyon convexiusculus* (Stephens 1829), a water scavenger beetle and *Hydroglyphus geminus* (Fabricius 1792) a lesser diving beetle associated with new ponds, heath pools and mossy ditches. The three species listed as Notable Regional are *Mesovelia furcata* (Mulsant & Rey 1852), a pondweed bug often found amongst surface vegetation and algal mats, *Limnephilus decipiens* (Kolenati 1848), a cased caddis fly found in rich water with abundant vegetation and *Micronecta scholtzi* (Fieber 1860) a lesser water boatman characteristic of ponds and lakes with bare mineral (e.g. gravel) bottoms, which was found at a number of sites.

5.3.2. Spatial autocorrelation

Spatial autocorrelation of macroinvertebrate communities was tested by the production of a Mantel correlogram (Figure 5.3.1). At no distance was the Mantel correlation coefficient significant (999 permutations, Bonferroni corrected). This suggests variation in local and regional factors was influential enough upon the macroinvertebrate community such that violations of spatial independence of the study sites were not encountered.

Table 5.3. Distribution of taxa between major taxonomic orders (spring and summer pooled presence/absence data) and diversity scores

Site	Gastropoda	Coleoptera	Trichoptera	Odonata	Hemiptera	Ephemeroptera	Tricladida	Crustacea	Diptera[1]	Hirudinea	Others[2]	Total taxa	Total species	CCI	Simpsons (E_{1-D})
1	10	2	10	6	3	2	1	2	10	1	2	58	38	16.4	0.103
2	11	10	10	3	4	3	0	2	7	4	2	63	48	17.6	0.171
3	0	2	0	0	7	1	0	1	5	2	1	19	13	9.2	0.097
4	3	1	0	0	1	1	1	1	4	1	1	17	9	4.9	0.158
5	2	2	0	0	1	1	3	1	7	1	1	28	11	5.1	0.156
6	3	3	1	0	7	1	1	2	8	6	1	37	24	17.0	0.115
7	0	1	0	0	8	0	4	2	1	2	1	25	17	19.8	0.158
8	5	2	3	3	4	1	3	2	7	3	2	43	27	8.3	0.112
9	9	0	4	0	8	2	2	2	8	4	2	48	32	9.5	0.217
10	2	8	5	8	10	1	0	1	7	0	2	51	36	16.3	0.062
11	5	5	1	0	13	1	1	2	6	3	2	46	32	11.6	0.131
12	6	16	8	6	17	2	2	2	10	3	1	81	62	18.7	0.086
13	1	5	2	0	10	1	3	2	9	5	1	45	29	10.0	0.111
14	8	4	6	6	8	1	0	2	6	3	1	51	38	11.1	0.105
15	5	0	0	0	1	1	1	1	5	2	2	20	12	4.8	0.149
16	5	0	0	0	1	0	2	2	2	3	1	21	13	6.9	0.171
17	8	2	2	1	5	3	2	3	6	4	1	43	30	10.7	0.149
18	11	5	15	4	5	3	2	3	9	6	2	72	55	16.2	0.140
19	5	10	1	1	7	1	3	2	10	4	1	54	34	13.4	0.120
20	5	7	3	4	5	1	3	4	10	5	2	56	38	9.0	0.150
21	8	13	5	4	9	2	4	3	9	6	2	73	55	26.4	0.240
22	5	14	2	4	5	1	3	2	10	2	2	62	39	9.9	0.069
23	12	14	8	4	11	2	2	4	7	7	2	82	65	14.8	0.153
24	7	0	6	2	3	2	1	2	11	3	2	46	27	9.5	0.072
25	9	9	11	5	8	3	2	4	8	4	2	72	56	15.3	0.062
26	5	2	1	1	3	1	2	2	6	7	1	37	24	11.3	0.061
27	11	5	7	3	8	2	1	2	11	7	2	70	47	17.0	0.048
28	7	14	3	1	1	0	3	2	10	0	1	48	31	11.1	0.053
29	5	10	3	4	5	1	1	2	7	3	1	47	34	10.6	0.102
30	4	3	1	0	2	0	0	4	7	1	1	28	15	8.0	0.088
Average[3]:	5	5	3	2	5	1	2	2	7	3	2	48	32	$\bar{x}$ 12.3	$\bar{x}$ 0.120
Unique	17	47	24	15	31	4	5	4	15	8	2	-	-	-	-

[1] Members identified to family level only

[2] Oligochaeta and Megaloptera

[3] Median, unless otherwise stated

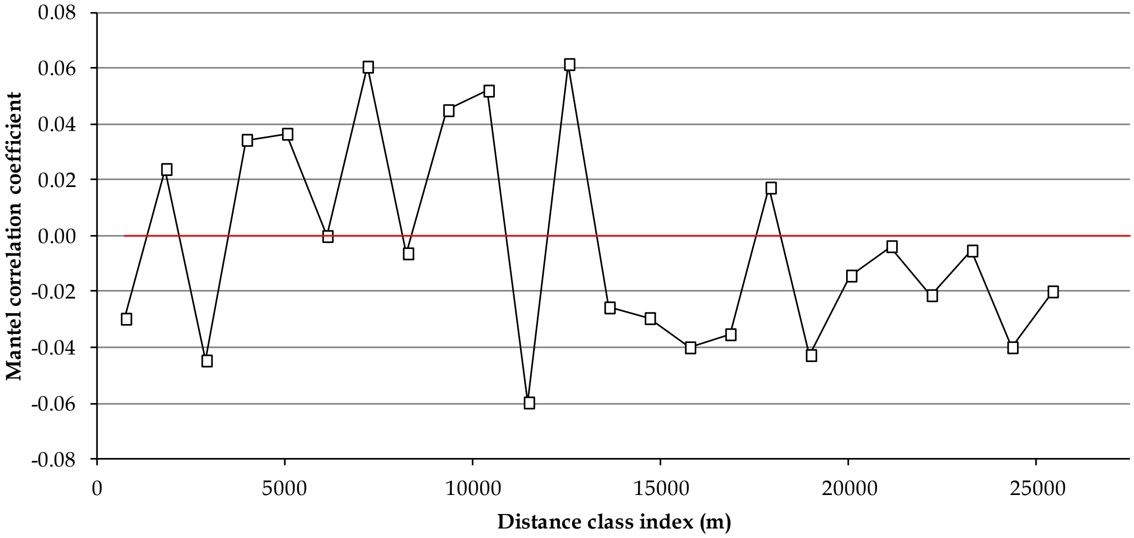
CCI – Community Conservation Index, values between 0 and >40, low to high conservation value

Table 5.4. Rare and notable species found during the study

Species	Order	Family	RDB 3	NB	NR	CCI	Site(s)
Hydrochus elongatus	Coleoptera	Hydrochidae	▪			8	21
Helochares lividus	Coleoptera	Hydrophilidae		▪		7	10, 21
Rhantus suturalis	Coleoptera	Dytiscidae		▪		7	28
Cercyon convexiusculus	Coleoptera	Sphaeridiidae		▪		7	1,6
Hydroglyphus geminus	Coleoptera	Dytiscidae		▪		7	7
Mesovelia furcata	Hemiptera	Mesoveliidae			▪	6	25
Limnephilus decipiens	Trichoptera	Limnephilidae			▪	6	2,18
Micronecta scholtzi	Hemiptera	Corixidae			▪	6	1,7,9,10,21, 23,24,25

RDB 3 – Red Data Book category 3 nationally rare
NB – Notable B; thought to occur in 31-100 10km squares within GB
NR – Notable Regional; Found in five or fewer GB localities

Figure 5.3.1. Mantel correlogram for macroinvertebrate community composition across 30 pond sites. Distance class refers to the geographical distance between sites (24 bins). Solid squares show autocorrelations significant at Bonferroni corrected level (*P* = 0.05/5 – 0.01).



5.3.3. Distinguishing urban pond macroinvertebrate assemblages and associated local and regional variables

Four macroinvertebrate communities were distinguished by TWINSPAN based on taxa and their abundances (Appendix 21). The first TWINSPAN division separated 12 sites from the 30. Indicative of the 12 sites were *Ischnura elegans* (Odonata, Vander Linden 1820) and *Radix balthica* (Gastropoda, Linnaeus 1758). The second TWINSPAN division further divided the 12 sites in to two groups comprised of four (Type 2, T2) and eight sites (Type 1, T1). *Hygrobia hermanni* (Coleoptera Fabricius 1781) and *Corixa panzeri* (Hemiptera, Fieber 1848) were indicative of T2 sites, whereas *Potamopyrgus antipodarum* (Gastropoda, J. E. Gray, 1843) was indicative of Type 1 sites. Of the remaining 18 sites, the second TWINSPAN division separated 11 sites (Type 3, T3) in which *Sigara lateralis* (Hemiptera, Leach 1817) was common and seven sites (Type 4, T4) indicative of Chaoboridae (Diptera) presence. Ordination of the pond types indicated good separation along the environmental gradients (Figure 5.3.2).

The forward stepwise CCA procedure within each of the local (water quality, local physical) factor sets identified significant relationships between macroinvertebrate community composition and PO_4 (mean 0.48 mg/l, range 0.01 to 2.62 mg/l, Appendix 22), dissolved oxygen (mean 65.9%, range 6.0 to 185.2%), fish presence or absence, percentage tree cover, macrophyte species richness (range 0 to 20 species) and suspended solids (mean 17.9 mg/l, range 2 to 110 mg/l). For regional factors, a series of forward stepwise CCAs using data from each spatial extent revealed that most variation in macroinvertebrate community composition was explained using land-use data within 100m and connectivity metrics at 1000m. At 100m, mature vegetation coverage and extent of urbanisation (PCA 1, see 5.2.4.1) were significant land-use variables. The proportion of aquatic habitat, irrespective of the type of wetland, within 1000m and the area weighted, effective cost to all neighbours within 2km (*AC*) were influential connectivity metrics. The combined effect of all significant local and regional factors in a single CCA explained 43.98% of the overall variance in macroinvertebrate community composition (Figure 5.3.2). All variables were tested for significant differences between the pond types (ANOVA, *post-hoc* Tukey-Kramer). Those variables with significant differences are shown in Table 5.5.

A small number of core taxa (> 1% total numbers) were common to all ponds, namely corixidae nypmhs, the crustaceans *Asellus aquaticus* (Linnaeus 1758) and *Crangonyx pseudogracilis* (Bousfield, 1958), the mayfly *Cloeon dipterum* (Linnaeus, 1761) and the dipterans of the Chironomidae and Chaoboridae (Table 5.6). *I. elegans,* a widespread and pollution tolerant odonate is the most frequently occurring odonate species in the dataset. The number of core taxa varied among pond types i.e. T1 > T2 >T3 > T4, reflecting increasing dominance by fewer taxa from ponds T1 to T4. The majority of unique taxa consisted of Coleoptera species, particularly in T1 and T4 ponds, although T1 ponds also supported seven unique Trichoptera (Table 5.7).

5.3.3.1. Type 1 (T1) ponds

T1 ponds were associated with fish presence, high macrophyte richness, little shading, high oxygen and low PO_4 concentration (Figure 5.3.3a). They were also well connected (within 1000m) with a limited degree of urbanisation within 100m (Figure 5.3.2, Table 5.5). The mean CCI value of 16.9, meant that T1 sites were typically of *high conservation value (*Chadd and Extence 2004*)*. The relatively intact communities were reflected by the number of core taxa (19), which was highest amongst the pond types, six of which were gastropods, including the TWINSPAN indicator taxon *P. antipodarum* (Table 5.6). The relative abundance of odonates was high in T1 ponds (2.3%) compared to T3 (1.8%) and T4 (0%). Trichoptera were well represented, with six of the eight unique species inhabiting more than one of the T1 ponds (Table 5.7). T1 ponds also supported the only unique Hirudinea, *Piscicola geometra,* a sanguivorous ectoparasite of freshwater fish and of highly oxygenized sites (Elliott and Mann 1998). Amongst the 22 unique taxa to T1 ponds i.e. those that were not found in ponds of any other type, was a Red Data Book 3 coleopteran species (nationally rare) *H. elongatus*. Six of the eight ponds classified as T1 are regularly angled and managed to that end. Average macrophyte species richness in ponds that were regularly angled was 6.1 compared to those that were not, 3.7, however, the difference was not significant (ANOVA, $P = 0.104$).

5.3.3.2. Type 2 (T2) ponds

T2 ponds were without fish, had relatively low tree cover, high macrophyte richness and relatively high levels of dissolved oxygen (Figure 5.3.2 & Figure 5.3.3b, Table 5.5). The mean CCI of 14.1 meant T2 ponds were of *fairly high conservation value* (Chadd and Extence 2004). Chaoboridae were abundant in T2 fishless ponds, and on average constituted 16.2% of the total abundance (Table 5.6). Similarly, a number of Hemiptera made up the core taxa, particularly Corixidae nymphs. In all, mean hemipteran relative abundance was 16.9%, the largest representation from any of the four types. The odonates *Lestes sponsa* (Hansemann 1823) and *Libellula depressa* (Linnaeus 1758), both unique to T2 ponds (Table 5.7), are often found in ponds without fish and with dense marginal vegetation, the latter tolerant of mildly polluted conditions (Brooks 1997). *H. hermanii*, which is known to prefer stagnant water, especially silt and detritus ponds and predates on oligochaetes (Holmen 1987), was found at three of the four T2 ponds. Mean odonate relative abundance was high in T2 (5.1%) ponds in comparison to T3 (1.8%) and T4 (0%). Four unique species of odonate were identified in T2 ponds: *Anax imperator* (Leach 1815), *L. sponsa*, *L. depressa* and *Sympetrum fonscolombii* (Sélys-Longchamps 1840), suggesting that they are particularly favourable for Odonata.

5.3.3.3. Type 3 (T3) ponds

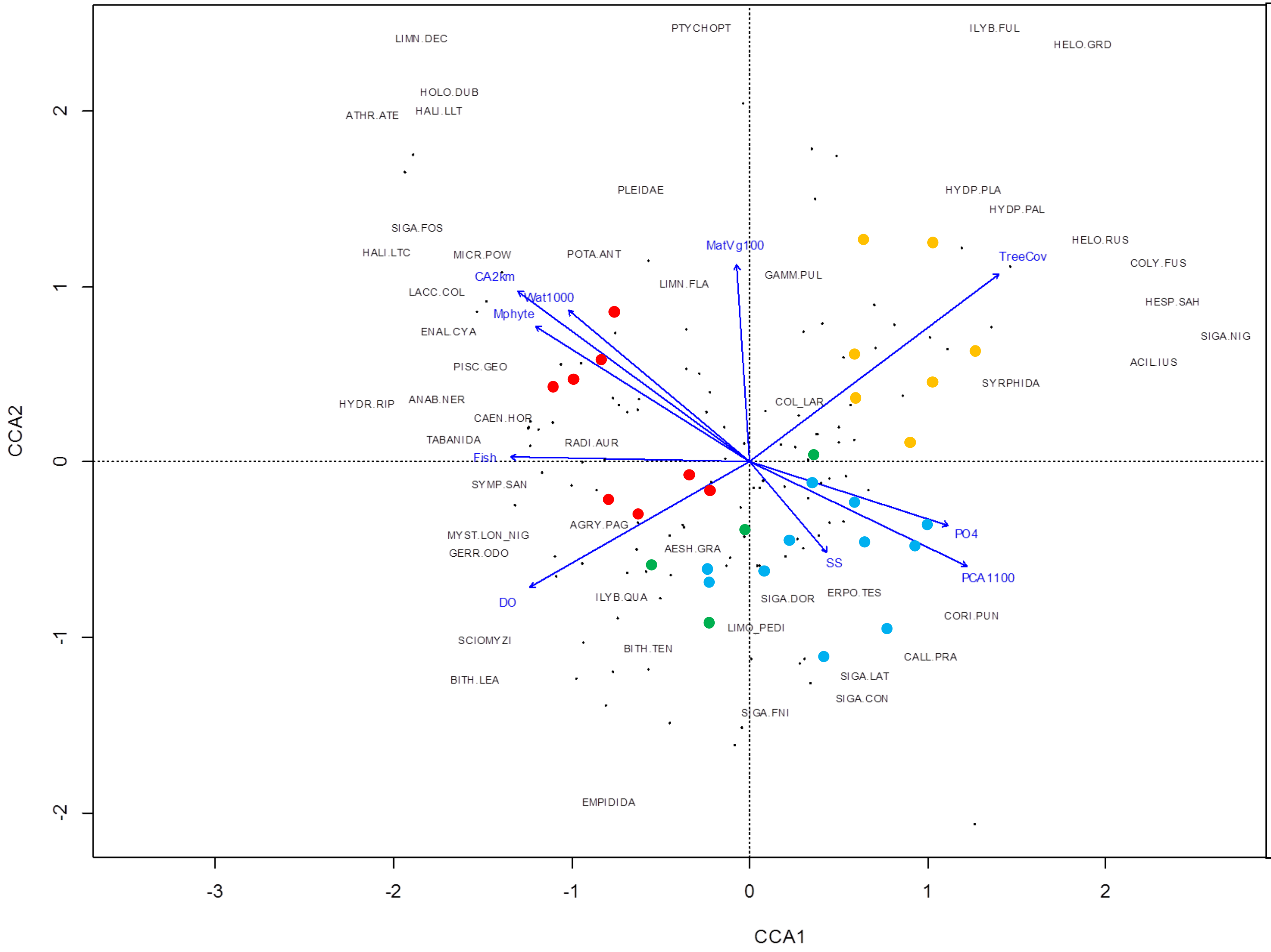
T3 ponds had larger surface areas, received surface water run-off, had more homogenous habitats and were more geographically isolated from other ponds (Figure 5.3.2 & Figure 5.3.3c, Table 5.5). Concentrations of PO_4 and suspended solids were generally high. Isolation of T3 sites was exacerbated by a high effective distance to neighboring habitats due to urbanisation within 100m.

A mean CCI of 10.7 meant that T3 ponds were typically of *low* to *moderate conservation value* (Chadd and Extence 2004). Over 25% of the macroinvertebrate assemblage in these ponds comprised of Chironomidae, the high abundance of which likely contributes to the higher mean relative abundance (4.7%) of *Helobdella stagnalis* (Hirudinea, Linnaeus, 1758), which are known to predate on Chironomidae (Table 5.6). The core taxa comprise of 13 taxa, the second lowest of the pond types, which suggests that the community is more uneven, although this was not reflected by mean Simpsons ($E_{1\text{-}D}$) values. Mean odonate relative abundance was low in T3 ponds (1.8%) in comparison to T1 and T2 ponds. During the study, five unique species were found in T3 ponds (Table 5.7), one in particular *Hydroglyphus pusillus* (Coleoptera, Fabricius 1792) is known to be an early colonizer of new reservoirs and lakes (Friday 1988).

5.3.3.4. Type 4 (T4) ponds

T4 sites were characterized by a high degree of shading, generally high PO_4 concentration and lower pH, dissolved oxygen and temperature (Figure 5.3.2 & Figure 5.3.3d, Table 5.5). Ponds of T4 typically had a small surface area, had no fish, and were more ephemeral (high WLFI). Like ponds of T3, more intense urbanisation within the surrounding landscape resulted in greater isolation. A mean CCI of 8.8 meant that T4 ponds were typically of *low* to *moderate conservation value* (Chadd and Extence 2004). *R. suturalis* is unique to pond T4 (Table 5.7), as is highest mean relative abundance of *A. aquaticus*, both species are associated with silt and detritus pools (Friday 1988). Also unique to T4 was *Oecetis ochracea* (Trichoptera, Curtis 1825), known to be an early colonizer of new reservoirs and lakes (Wallace *et al.* 2003). Despite the more ephemeral nature of ponds of T4, *Velia caprai* (Hemiptera, Tamanini 1947), which is associated with more stable sites (Savage 1989) is unique to pond T4. The characteristics of T4 ponds were poorly suited to supporting Odonata.

Figure 5.3.2. Canonical correspondence analysis of macroinvertebrate taxa.
Pond types are represented by different symbols (Type 1 = Red, Type 2 = Green, Type 3 = Blue, Type 4 = Orange. Black dots represent species



Key to species abbreviations

ACIL.IUS	*Acilius sulcatus*
AESH.GRA	*Aeshna grandis*
AGRY.APG	*Agrypnia pagetana*
ANAB.NER	*Anabolia nervosa*
ATHR.ATE	*Athripsodes aterrimus*
BITH.LEA	*Bithynia leachii*
BITH.TEN	*Bithynia tentaculata*
CAEN.HOR	*Caenis horaria*
CALL.PRA	*Callicorixa praeusta*
COL_LAR	Coleoptera larvae
COLY.FUS	*Colymbetes fuscus*
CORI.PUN	*Corixa punctata*
EMPIDIDA	*Empididae*
ENAL.CYA	*Enallagma cyathigerum*
ERPO.TES	*Erpobdella testacea*
GAMM.PUL	*Gammarus pulex*
GERR.ODO	*Gerris odontogaster*
HALI.LLT	*Haliplus lineolatus*
HALI.LTC	*Haliplus lineatocollis*
HELO.GRD	*Helophorus grandis*
HELO.RUS	*Helophorus spp.*
HESP.SAH	*Hesperocorixa sahlbergi*
HOLO.DUB	*Holocentropus dubius*
HYDP.APL	*Hydroporus palustris*
HYDP.PLA	*Hydroporus planus*
HYDR.RIP	*Hydraena riparia*
ILYB.FUL	*Ilybius fuliginosus*
ILYB.QUA	*Ilybius quadriguttatus*
LACC.COL	*Laccobius colon*
LIMN.DEC	*Limnephilus decipiens*
LIMN.FLA	*Limnephilus flavicornis*
LIMO_PED	*Limonidae/Pedicidae*
MICR.POW	*Micronecta poweri*
MYST.LON_NIG	*Mystacides longicornis/nigra*
PISC.GEO	*Piscicola geometra*
PLEIDAE	*Pleidae nymph*
POTA.ANT	*Potamopyrgus antipodarum*
PTYCHOPT	*Ptychopteridae*
RADI.AUR	*Radix auricularia*
SCIOMYZI	*Sciomyzidae*
SIGA.CON	*Sigara concinna*
SIGA.DOR	*Sigara dorsalis*
SIGA.FNI	*Sigara falleni*
SIGA.FOS	*Sigara fossarum*
SIGA.LAT	*Sigara lateralis*
SIGA.NIG	*Sigara nigrolineata*
SYMP.SAN	*Sympetrum sanguineum*
SYRPHIDA	*Syrphidae*
TABANIDA	*Tabanidae*

Table 5.5. Average (minimum and maximum) values of environmental variables that were significantly different between pond types (ANOVA, *post-hoc* Tukey-Kramer $P < 0.05$).

Water quality	Units	Type 1 (n = 8)	Type 2 (n = 4)	Type 3 (n = 11)	Type 4 (n = 7)
pH	-	7.8 (7.5-8.5)^{ab}	8.2 (7.8-8.6)^{a}	7.7 (7-9.1)^{ab}	7.2 (6.7-7.6)^{b}
Temperature	^{o}C	18.9 (17.6-22.1)^{a}	20.2 (17.5-24.2)^{a}	18.2 (15.7-20.1)^{ab}	16.1 (14.9-18.9)^{b}
Dissolved oxygen	% Sat.	79.1 (45.3-112.2)^{a}	88.9 (70.5-110.8)^{a}	72.7 (18.8-175.6)^{a}	26.9 (12.1-52.2)^{b}
PO_4^{3-}	mg l^{-1}	0.05 (0.02-0.13)^{a}	0.16 (0.02-0.33)^{ab}	0.77 (0.15-2.46)^{b}	0.7 (0.11-2.11)^{ab}
Mn^{1}	µg l^{-1}	175.6 (56-342.3)^{ab}	71.6 (56-131.8)^{a}	396.7 (56-1034)^{ab}	615.5 (65.5-1243.8)^{b}
Local physical					
Mesohabitats	Count	5 (4-7)^{a}	4 (4-6)^{ac}	3 (2-4)^{b}	3 (1-5)^{abc}
Surface area	m^{2}	3421 (721-6963)^{ab}	3846 (692-6423)^{ab}	5294 (1215-14967)^{a}	989 (299-2108)^{b}
Open water	%	0.62 (0.07-0.9)^{a}	0.73 (0.46-0.96)^{a}	0.61 (0.24-0.9)^{a}	0.15 (0-0.39)^{b}
Tree cover	%	0.07 (0-0.17)^{a}	0.02 (0-0.06)^{abc}	0.3 (0.04-0.65)^{b}	0.73 (0.27-1)^{c}
Floating cover	%	0.13 (0.01-0.47)^{a}	0.07 (0-0.16)^{ab}	0.01 (0-0.03)^{b}	0.01 (0-0.07)^{b}
Macrophyte richness	Count	7.6 (4-14)^{a}	7.1 (5-9.5)^{a}	1.3 (0-2.5)^{b}	4.3 (1-10)^{a}
Water Level Fluctuation Index	-	61.2 (4.1-168.8)^{ab}	193.6 (18.6-353.6)^{ab}	31.5 (6.4-119.4)^{a}	1245 (24-4698.4)^{b}
Spring source^{2}	Binary	0.63	0.25	0.36	0.57
Surface run-off source^{2}	Binary	0.38	0.5	0.82	0.57
Stream source^{2}	Binary	0.13	0	0.36	0
Fish presence^{2}	Binary	0.88	0	0.55	0
Regional connectivity					
Effective distance/area within 2km	-	22.15 (1.44-92.87)^{a}	5.77 (1.4-15.9)^{ab}	1.1 (0.18-4.1)^{b}	2.67 (0.51-9.37)^{b}
Water within 1000m	%	0.029 (0.014 - 0.049)^{a}	0.013 (0.01 - 0.016)^{ab}	0.008 (0.001 - 0.015)^{b}	0.018 (0.005 - 0.046)^{ab}
Regional land-use					
PCA axis 1 within 100m	-	-1.69 (-3.80 - 2.22)^{a}	-1.37 (-2.81 – 0.14)$_{ab}$	1.23 (-0.65 – 3.44)^{b}	0.77 (-0.84 – 3.11)$_{ab}$
Mature vegetation within 100m	%	0.30 (0.07 – 0.76)	0.26 (0.06 – 0.50)	0.25 (0.10 – 0.53)	0.38 (0.11 – 0.64)
Diversity metrics					
Species richness	-	67.6 (51 - 82)^{a}	55.5 (43 - 81)^{ab}	35.2 (19 - 48)^{b}	41.9 (17 - 62)^{b}
CCI	-	16.85 (11.1 - 26.4)^{a}	14.08 (10.6 - 18.7)^{ab}	10.72 (4.8 - 19.8)^{b}	8.76 (4.9 - 13.4)^{b}

^{a}, ^{b} and ^{c} indicate where significant differences occur

[1] Concentrations below limits of detection assigned limit of detection value (Appendix 3)

[2] Binary; figures represent proportion of sites including the variable (not statistically tested for differences)

Table 5.6. Mean relative abundance of core taxa (those >1% of total macroinvertebrate abundance) occurring within each pond type. Values in bold represent top three most common taxa within pond type

Core taxa	Type 1 (fish, low PO_4, high macrophyte richness)	Type 2 (no fish, low tree cover, high macrophyte richness)	Type 3 (high PO_4, isolated, low habitat diversity, large)	Type 4 (heavily shaded, no fish, low temp., small, high WLFI)
Gastropoda				
Anisus vortex	0.019			
Armiger crista	0.019			
Bithynia tentaculata			0.012	
Gyraulus albus	0.035	0.013		
Hippeutis complanatus			0.015	
Lymnaea stagnalis	0.038			
Physa fontinalis		0.035		
Planorbarius corneus				0.014
Potamopyrgus antipodarum	0.038			0.014
Radix balthica	0.022	0.047	0.039	0.052
Crustacea				
Asellus aquaticus	**0.145**	0.063	**0.113**	**0.245**
Crangonyx pseudogracilis	0.043	**0.111**	0.039	0.046
Hemiptera				
Cymatia coleoptrata		0.016		
Micronecta scholtzi		0.014		
Corixidae nymph	0.029	0.101	0.071	0.074
Notonectidae nymph		0.013		
Coleoptera				
Beetle larvae	0.012			0.013
Ephemeroptera				
Caenis horaria	0.012			
Cloeon dipterum	0.017	0.025	0.012	0.011
Hirudinea				
Helobdella stagnalis			0.047	
Odonata				
Ischnura elegans	0.012	0.016		
Anisoptera, immature		0.010		
Zygoptera, immature	0.011	0.025	0.018	
Diptera				
Ceratopogonidae	0.010			

Core taxa	Type 1 (fish, low PO_4, high macrophyte richness)	Type 2 (no fish, low tree cover, high macrophyte richness)	Type 3 (high PO_4, isolated, low habitat diversity, large)	Type 4 (heavily shaded, no fish, low temp., small, high WLFI)
Chaoboridae	0.010	**0.162**	0.038	0.086
Chironomidae	**0.182**	**0.145**	**0.261**	**0.202**
Bivalvia				
Sphaeriidae	0.022	0.024	0.045	0.044
Oligochaeta	**0.157**	0.056	**0.195**	**0.132**
Total:	**n = 19, ∑0.84**	**n = 17, ∑0.88**	**n = 13, ∑0.91**	**n = 12, ∑0.93**

Table 5.7. Unique taxa to each pond type i.e. those exclusive to each pond. Number of sites at which the taxon is present is given in parenthesis.

Type 1 (n = 8)	Type 2 (n = 4)	Type 3 (n = 11)	Type 4 (n = 7)
Coleoptera			
Cercyon convexiusculus (1)	*Hygrobia hermanni* (3)	*Anacaena globulus* (1)	*Helophorus grandis* (2)
Coelambus impressopunctatus (1)	*Suphrodytes dorsalis* (1)	*Gyrinus substriatus* (1)	*Hydroporus angustatus* (2)
Haliplus lineolatus (2)	*Ochthebius minimus* (1)	*Hydroglyphus pusillus* (2)	*Hydroporus pubescens*
Hydraena riparia (2)			*Ilybius fuliginosus* (2)
Hydrochus elongatus (1)			*Rhantus exsoletus* (1)
Hygrotus versicolor (1)			*Rhantus suturalis* (1)
Laccophilus hyalinus (1)			
Trichoptera			
Anabolia nervosa (2)	*Cyrnus trimaculatus* (1)		*Oecetis ochracea* (1)
Athripsodes aterrimus (2)			
Holocentropus dubius (2)			
Leptocerus tineiformis (4)			
Limnephilus decipiens (2)			
Lype reducta (1)			
Mystacides azurea (1)			
Cyrnus flavidus (4)			
Odonata			
Libellula quadrimaculata (1)	*Anax imperator* (1)		
Orthetrum cancellatum (1)	*Lestes sponsa* (1)		
	Libellula depressa (1)		
	Sympetrum fonscolombii (1)		
Hemiptera			
Mesovelia furcata (1)		*Notonecta maculata* (1)	*Callicorixa wollastoni* (1)
Plea leachi (1)			*Velia caprai* (2)
Hirudinea			
Piscicola geometra (5)			
Gastropoda			
Radix auricularia (2)		*Stagnicola palustris* (1)	
Ephemeroptera			
Caenis luctuosa (1)			

Figure 5.3.3. Four types of urban pond distinguished by TWINSPAN

a) Type 1:

b) Type 2

c) Type 3

d) Type 4

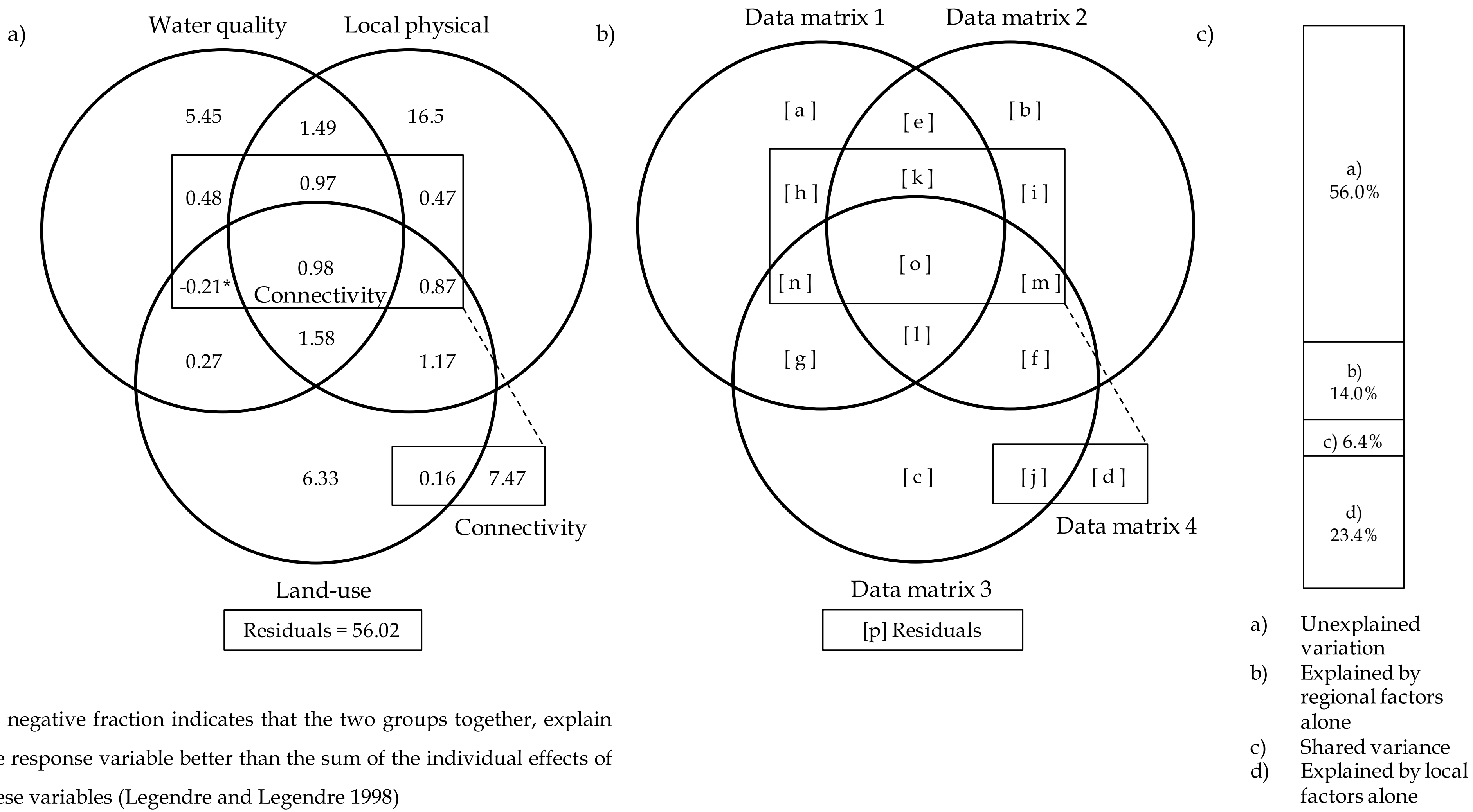
5.3.4. Separating out the effects of local and spatial influences on the macroinvertebrate communities of urban ponds

The forward selected variables from within each of the local (water quality, physical) and regional (land-use, connectivity) factors sets (section 5.3.3) were used within a variance partitioning procedure (Borcard *et al.* 1992) in order to isolate the amount of explained variance in macroinvertebrate composition that could be attributed to local and regional variables alone and in combination (Figure 5.3.4).

The isolated influence of local physical, water quality, land-use and connectivity factors explained 16.5%, 5.45%, 6.33% and 7.47% of variance in the macroinvertebrate community respectively (Figure 5.3.4a), such that their explanatory power ranked: local physical > connectivity > land-use > water quality (i.e. a – d Figure 5.3.4b). In all, 43.98% of variance within the macroinvertebrate community was explained using local and regional factors alone and in combination. Local physical variables were clearly the most influential, in addition to the 16.5% explained purely by local physical factors, a further 7.53% of explained variance was shared between local physical and at least one other factor set, such that total *potential* explained variance by local physical factors alone was 24.03%. For example, the greatest shared variance was found between local physical, water quality, and land-use variables (1.58%, i.e. l Figure 5.3.4b), although the majority of this variance (1.49%, i.e. e Figure 5.3.4b) was shared between local physical and water quality variables alone. Over half (50.4%) of variance explained in the direct CCA (non-partial) by water quality, was subsequently identified as shared variance.

Local factors alone were able to explain 23.4% of the total variance (Figure 5.3.4c), which equates to 53.3% of explainable variance, almost twice that explained by regional factors. The remaining 6.6% of explained total variance was shared amongst combinations of local and regional factors sets.

Figure 5.3.4. a) Venn diagram displaying the partitioning of explained variance between local (water quality, local physical factors) and regional (land-use and connectivity) factors. Numbers represent the percentage of variance attributable to each variable set. b) 15 CCA analyses were performed and 27 subtraction equations to estimate the fractions a - o (see Appendix 18) c) The variance attributable to local and regional factors combined.



*A negative fraction indicates that the two groups together, explain the response variable better than the sum of the individual effects of these variables (Legendre and Legendre 1998)

5.3.5. Predicting species richness within characteristic macroinvertebrate orders

Generalized Linear Models (GLMs) were used to explore patterns of taxon richness for the macroinvertebrate assemblage as a whole and that of five macroinvertebrate orders (Gastropoda, Coleoptera, Trichoptera, Odonata and Hemiptera).

Local physical factors provided the best fitting model for predicting richness in three of the macroinvertebrate orders (Gastropoda, Coleoptera and Odonata) and for taxa richness (Table 5.8), whereas there was little difference between water quality and local physical models for Trichoptera and land-use provided the best-fitting model for Hemiptera; although all models for Hemiptera richness performed poorly. Macrophyte species richness had strong support for inclusion in the best approximating model for all macroinvertebrate orders (selection probability (SP) >0.95, Table A19.2), with the exception of Hemiptera.

Of the connectivity metrics, many of the nearest neighbour metrics were collinear (Pearson's correlation coefficient > 0.7) and resulted in just three being retained for incorporation into the GLMs in addition to the amount of wetland and pond habitat within each spatial extent. These were the Euclidean distance to the nearest neighbour weighted by area (APNN), the effective distance to the nearest neighbour (CNN) and the cumulative cost to all neighbours within 2km weighted by area (AC2km). Of these, the latter was the most influential amongst macroinvertebrate orders, suggesting consideration of the wider pond network and the inter-habitat landscape is important. AC2km was also found to have a stronger relationship with taxa richness in all cases, with the exception of Odonata, than the non-landscape weighted AP2km metric (Appendix 23), thus was preferentially retained for model selection.

5.3.5.1. Gastropoda

Gastropod richness was best predicted by local physical variables ($r^2 = 0.7$, Table 5.8). Twelve models made up the 95% confidence set, each of which included macrophyte richness (SP > 0.99) and fish presence/absence (SP 0.97, Table A19.2) indicating strong support for their inclusion in the best approximating model. Interestingly, the relationship between fish presence and gastropod richness was positive, such that it is not likely to be a direct predator-prey relationship.

Thirteen models were included in the 95% confidence set for connectivity, 11 of which included the proportion of aquatic habitat within 250m (SP 0.89, Table A19.4). CNN, the next important connectivity metric had only moderate support (SP 0.55). The most parsimonious water quality model has a good probability of resembling the best approximating model

(wAIC$_c$ 0.42) from the 24 that made up the 95% confidence set, all of which included PO_4 and nine included both Fe (mean 818.5 µg/l, Appendix 21) and K (mean 4.8 mg/l). Consequently, PO_4 has very strong support for inclusion in the best approximating model for water quality (SP > 0.99). Support for Fe (SP 0.71) and K (SP 0.63) inclusion is somewhat lower (Table A19.1). Land-use variables were poor at predicting gastropod richness as demonstrated by a ΔAIC$_c$ of 16.7 (r^2 0.34). Nonetheless, the extent of urbanisation within 200m was the most important land-use variable (SP >0.99).

5.3.5.2. Coleoptera

Coleoptera richness was best predicted by local physical variables (r^2 0.56, Table 5.8). Thirty-one candidate models made up the 95% confidence set, all of which included macrophyte richness indicating strong support for its inclusion in the best approximating local physical model (SP >0.99, Table A19.2), 15 included fish presence or absence as predictor variables (SP 0.71).

There was a considerable difference in the model fit between the local physical model and the next most powerful, land-use (ΔAIC$_c$ 12.2, r^2 0.51). Nonetheless, the extent of urbanisation within 500m as indicated by PCA 1 (SP 0.80), as well as the proportion of tree cover (SP 0.95) were valuable predictors (Table A19.3), included in 18 and eight of the 21 models in the 95% confidence set respectively. The connectivity model was weaker than both the habitat and land-use models, as shown by a ΔAIC$_c$ of 14.1. The availability of pond habitat within 2500m was a good predictor and received strong support (SP > 0.99, Table A19.4) for inclusion in the best approximating model from the connectivity metrics. The water quality variables provided the poorest fitting model with a (ΔAIC$_c$ 23, r^2 0.16).

5.3.5.3. Trichoptera

For Trichoptera the best model fit was provided by water quality variables (Table 5.8), with a good probability it resembles the best approximating water quality model (wAIC$_c$ 0.71). Nine models made up the 95% confidence set, all of which included PO_4 and NH_4 resulting in strong support (both SP 0.98) for their inclusion in the best approximating water quality model (Table A19.1). Fe was only selected in the most parsimonious model of the 95% confidence set, undermining its relevance despite superficially strong support (SP 0.74). overall, the ΔAIC$_c$ between the most parsimonious water quality and local physical models was just 1.8 indicating that there was little difference in their model fit, in addition, the relationship between predicted and observed values was stronger using either local physical factors (r^2 0.59) or connectivity metrics (r^2 0.57) than for water quality. The 95% confidence set of local physical variable models comprised of a relatively large number of all possible model combinations (44/377) suggesting relatively small differences in their performance. Nonetheless, 36 included macrophyte richness (SP 0.95) and 12 included fish presence or absence (SP 0.78, Table A19.2). The percentage of tree cover, although appearing in the most parsimonious model, had weak overall support for inclusion in the best approximating local physical model (SP 0.35).

The proportion of other wetlands within 500m from the pond edge had strong support for inclusion in the best approximating connectivity model (SP 0.87, Table A19.4). *CNN* had moderate support (SP 0.60). Within 500m, the proportions of wetland and pond habitat were highly collinear (Pearson's correlation coefficient 0.81, $P < 0.001$). Urbanisation within 200m was the most powerful land-use predictor of Trichoptera richness (SP > 0.99, Table A19.3). However, the most parsimonious land-use model exhibited the poorest model fit (ΔAIC_c 13.9, r^2 0.47).

5.3.5.4. Odonata

Odonate richness was predicted best by local physical factors (AIC_c 88.0, r^2 0.77), although all local and regional factors sets of variables performed relatively well (r^2 0.60 to 0.77, Table 5.8). Thirty-seven models comprised the 95% confidence set for local physical variables, all of which included macrophyte richness (SP >0.99, Table A19.2). The most parsimonious model suggested that percentage tree cover was the only significant local physical predictor (<999 permutations, $P < 0.05$), however, a review of the 95% confidence set suggested less support as it only appeared in eight of the 37 models (SP 0.53).

Connectivity metrics provided the second best-fitting model (ΔAIC_c 7.9), with a good resemblance to the best approximating model ($wAIC_c$ 0.58). *AC2km* (SP >0.99, Table A19.4) and the proportion of wetland within 250m (SP 0.96) were selected in all four of the 95% confidence set of connectivity models. For water quality PO_4 (SP >0.99), which occurred in all 21 of the 95% confidence set, and NH_4 (mean 0.77 mg/l, Appendix 21, SP 0.62) were important (Table A19.1). Land-use variables had the lowest model fit (ΔAIC_c 20.4, r^2 0.60), nonetheless, the degree of urbanisation (SP >0.99) and proportion of mature vegetation within 100m (SP 0.98) received the greatest support, and the most parsimonious model likely represents the best approximating land-use model ($wAIC_c$ 0.80).

5.3.5.5. Hemiptera

The performance of the models for predicting Hemiptera richness was poor. This is reflected by low $wAIC_c$ scores, which suggests the lack of clear, powerful predictors within the explanatory variable groups (Table 5.8). Similarly, the percentage deviance explained and r^2 values are also small, the maximums being 35.1% and 0.32 respectively.

Little difference was found between the fit of land-use, local physical and water quality variables to hemipteran richness (ΔAIC_c, 1.3). Within their respective variable sets, $CaCO_3$ (SP 0.76, Table A19.1), percentage tree cover (SP 0.92, Table A19.2) and the proportion of tree cover within 25m (SP 0.95) received the strongest support. The number of models in the 95% confidence sets of models relative to the total number of models, for water quality (97/175) and local physical factors (128/377), further indicates the lack of outstanding model fits.

5.3.5.6. Taxon richness

Macroinvertebrate assemblage richness was best predicted by local physical factors (Table 5.8). All 63 of the 95% confidence models included macrophyte species richness (SP >0.99,Table A19.2), whilst all other habitat variables had low support for inclusion in a best approximating model (SP <0.36).

All four local and regional factor sets performed well for predicting taxa richness (r^2 >0.50). The proportion of tree coverage (SP 0.98) and degree of urbanisation within 200m (SP >0.99) were the most important land-use variables (Table A19.3), occurring in all six of the 95% confidence set of models. PO_4 (SP 0.96, Table A19.1) and NH_4 (SP 0.84) were the most important water quality variables, included in 30 and 13 of the 34, 95% confidence set. Relatively high $wAIC_c$ values for the most parsimonious water quality (0.48) and land-use models (0.61) suggests the models show a good resemblance to the best approximating model. The connectivity metrics of most relevance were representative of the wider pond network. The availability of pond habitat within the maximum spatial extent studied here (2500m) was included in the top seven of 12 95% confidence set models (SP 0.88, Table A19.4) and *AC2km* included nine of 12 (SP 0.76). *CNN* had moderate support (SP 0.43) despite not being included in the most parsimonious model.

Table 5.8. The four most parsimonious models predicting the relationship for species richness (response variable) for five macroinvertebrate orders and taxa richness from within each of the local factor (water quality, local physical) and regional factor (land-use, connectivity) datasets

Gastropoda	**Group**	**AIC_c**	**ΔAIC_c**	**$wAIC_c$**	**%DE**	**r^2**
Mphyte + Fish	Local phys.	134.1	0	0.193	58.0	0.70
Wat_250m + CNN	Connect.	139.8	5.7	0.255	49.2	0.61
PO_4 + Fe + K	Water	143.2	9.1	0.421	47.8	0.54
PCA1_200m	Land-use	150.8	16.7	0.133	28.9	0.34
Coleoptera						
Mphyte + Fish +%C	Local phys.	146.7	0	0.317	63.2	0.56
PCA1_500m + Tree +($E_{1/D}$)	Land-use	158.9	12.2	0.259	45.8	0.51
Pond_2,500m + (CNN)	Connect.	160.8	14.1	0.187	38.1	0.42
PO_4 + ($CaCO_3$)	Water	169.7	23.0	0.050	18.4	0.16
Trichoptera						
PO_4 + NH_4 + Fe	Water	126.0	0	0.713	65.9	0.51
Mphyte + Fish +%Tr	Local phys.	127.8	1.8	0.271	63.8	0.59
Wat_500m + AC2km	Connect.	138.9	12.9	0.226	44.3	0.57
PCA1_200m + (Tree_200m)	Land-use	139.9	13.9	0.142	41.9	0.47
Odonata						
%Tr + (Mphyte) + (Str.)	Local phys.	88.0	0	0.307	NA	0.77
AC2km + Pond_250m + (Wat_250m)	Connect.	95.9	7.9	0.577	NA	0.61
PO_4 + (NH_4) + ($CaCO_3$)	Water	99.5	11.4	0.260	NA	0.69
Tree_100m + (PCA1_100m) + (Grass_100m)	Land-use	105.4	20.4	0.800	NA	0.60
Hemiptera						
Tree_25m	Land-use	157.2	0	0.153	27.4	0.29
%Tr	Local phys.	157.8	0.6	0.058	26.1	0.24
$CaCO_3$ + DO + Ca	Water	158.5	1.3	0.113	35.1	0.32
(Pond_2,500m)	Connect.	163.4	6.2	0.253	11.7	0.11
Taxa richness						
Mphyte + %Tr + Spr.	Local phys.	238.0	0	0.098	66.3	0.66
PCA1_200m + Tree_200m + Grass_200m	Land-use	244.0	6.0	0.608	59.0	0.68
PO_4 + NH_4 + $CaCO_3$	Water	246.7	8.7	0.481	55.2	0.54
Pond_2,500m + AC2km	Connect.	251.0	13.0	0.305	44.0	0.50

AIC_c Corrected Akaike's Information Criterion, *ΔAIC_c* difference in AIC_c between a given model and the most parsimonious, *$wAIC_c$* the model selection probability from all candidate models, *%DE* percent deviance explained in the response variable by the model under consideration, *r^2* of the relationship between model predicted against observed values, *Mphyte* Macrophyte species richness, *Fish* Fish presence/absence, *%C* Percentage of concrete edge, *%Tr* Percentage tree Mean over pond, *Str.* Stream water source, *Spr.* Spring or groundwater source, *PO_4* Average phosphate concentration, *Fe* Mean iron concentration, *K* mean potassium concentration, *$CaCO_3$* Mean alkalinity, *NH_4* Mean ammonium concentration, *DO* Mean dissolved oxygen concentration, *Ca* Mean calcium concentration, *Wat* proportion of water within stated extent from the pond edge, *Pond* proportion of pond habitat within stated extent from the pond edge, *CNN* effective distance to the nearest neighbour, *AC2KM* Surface area of nearest neighbours divided by effective distance within 2km, *PCA1* urbanisation within stated extent from the pond edge, *Tree* Proportion of mature tree cover within stated extent from pond edge, *$E_{1/D}$* Simpsons index of land-use heterogeneity within stated extent from pond edge, *Grass* Proportion of ground level vegetation within stated extent from pond edge. Variables in parenthesis indicate a non-significant predictor ($P > 0.05$).

5.4. Discussion

5.4.1. The conservation value of urban ponds

Despite the heavily altered and urbanized Birmingham and Black Country (BBC) landscape, many of the ponds studied supported a wide variety of macroinvertebrate life. Thus, the results are in agreement with those of Vermonden *et al.* (2009) that, contrary to previous studies that have found lower macroinvertebrate diversity in urban streams (Lenat and Crawford 1994; Paul and Meyer 2001; Walsh *et al.* 2001; Roy *et al.* 2003; Mancini *et al.* 2005), conservation value in urban ponds can be relatively high; although it is possible that BBC may be relatively less degraded than the areas investigated in previous studies. The discrete nature of ponds and the small size of their catchments (Davies *et al.* 2008b) results in ponds within the same region differing in local influences upon their physical and chemical environment. These localized conditions result in a greater variety of physical and chemical characteristics between ponds, the result of which is a greater diversity of niches for organisms to exploit (Biggs *et al.* 2005). Furthermore, the discrete nature of ponds is in contrast to riverine systems, where a single upstream disturbance may impact upon the entire downstream system. Therefore, greater resilience to disturbance may be present across pond networks that are able to support high regional biodiversity despite locally impoverished conditions. To date, however, no studies have directly compared taxon richness in pond habitats to riverine systems in an urban context.

Pond studies in urban areas have emphasized their potential to support high regional diversity (Gledhill *et al.* 2008; Le Viol *et al.* 2009). In our study, a total of 193 macroinvertebrate taxa were identified (157 to species), considerably more than a study of 36 disused industrial mill ponds in Huddersfield, (taxa richness 124), although this may have been an underestimate due to the taxonomic level of identification (Wood and Barker 2000). A median species richness here of 32 is comparable to a study of 37 ponds in an urban northwest England landscape (median species richness = 28, Gledhill *et al.* 2008). Over half of the studied ponds supported one or more unique taxon, such that it was not only the most diverse and high quality habitats that contributed uncommon taxa. This strongly supports the proposal that a collection of ponds in different stages of succession (Williams *et al.* 1999; Biggs *et al.* 2005) and/ or maintaining a few in a turbid, pioneer state (Vermonden *et al.* 2009) increase s regional biodiversity.

Despite the clear value of ponds at a regional scale, many of those studied here also supported diverse local communities. Only two ponds scored a CCI < 5, which as a general guide are sites that support only common species and/or a community of low taxon richness and of low conservation value (Chadd and Extence 2004). The remaining sites were considered to be of moderate conservation value or above, largely as a consequence of the presence of at least one uncommon species. Site 21 had the highest conservation value (CCI 26.4) and is located in within a Site of Special Scientific Interest (SSSI) designated for its mosaic of wetland habitats (Natural England 2012a). The designation of site 21 supports the guidelines provided by Chadd and Extence (2004) that such a macroinvertebrate assemblage as that found may merit statutory protection. Furthermore, site 21 supported *H. elongatus*, the only nationally rare species found (two individuals). The closest recordings of *H. elongatus* are approximately 33km away in Coventry (Warwickshire BRC 2012). However, the site is also noted for being in decline due to increasing abundance of a non-native invasive plant species *Crassula helmsii* (Natural England 2012b).

5.4.2. Factors determining macroinvertebrate community composition

The combination of TWINSPAN and CCA described the main environmental gradients that separated out groups of ponds with similar macroinvertebrate communities. The deconstruction of the assemblage, as suggested by Thompson and Townsend (2006) further identified specific local and regional factors that showed greater relevance to the richness of different macroinvertebrate orders.

GLMs performed well for each of the macroinvertebrate groups examined with the exception of Hemiptera. Hemipterans are invariably strong dispersers (Briers and Warren 2000) and amongst the first to colonize new sites (Williams *et al.* 2008), it is feasible that sampled adults may not represent breeding populations reflective of the habitat. Alternatively, hemipterans show considerable variability in their functional traits, many of which are idiosyncratic by comparison to other macroinvertebrate orders, such that a further breakdown of Hemiptera to family level e.g. Corixidae, Notonectidae, may improve model fit.

5.4.2.1. Influence of local factors

Ponds that had the highest conservation value were associated with low nutrient status, high macrophyte complexity and little shading. Macrophyte richness was positively related to species richness in all the macroinvertebrate groups studied, with the exception of Hemiptera. Macrophyte complexity is well known to improve macroinvertebrate habitat by providing food and refuge from predation (Gilinsky 1984; Diehl 1992; Williams 1997) and positive relationships between macroinvertebrate and macrophyte richness are well known (Gledhill *et al.* 2008; Hassall *et al.* 2011). Macrophytes provide a substrate for epiphytic algal growth providing food for gastropods (Brönmark 1985) and a reduction in the risk of predation by molluscivorous fish (Covich and Kneževič 1978). Zealand and Jeffries (2009) show that differences between gastropod diversity in their study of ponds in Northumberland can be attributed to habitat similarity (macrophtyte richness). The pattern is likely to be similar for Trichoptera, the cased forms of which frequently require plant material to build their cases and caseless forms are often found attached to macrophytes (Samways and Steytler 1996; Schindler *et al.* 2003; Hassall *et al.* 2011). Many odonates require emergent vegetation to complete their life cycle, for predation opportunities (Lombardo 1997) and as perches in their adult form (Remsburg *et al.* 2008).

The most species poor sites had high nutrient concentrations, had fewer macrophytes, high levels of shading and were in a 'turbid' state (Scheffer *et al.* 1993). Overhanging trees impact macrophytic growth by excluding light from reaching the sediment directly or through the accumulation of leaf litter above the sediment. Fewer macrophytes reduce oxygenation of the water column, produced through photosynthesis (Brönmark and Hansson 2005), and nutrient uptake from the sediment (Chambers *et al.* 1989) and water column as with submerged vegetation (Cronk and Fennessy 2001). A reduction in macrophytes can encourage a shift between a macrophyte-rich clear water to a macrophyte-poor turbid state, dominated by floating vegetation by promoting algal biomass (Scheffer and van Nes 2007). Gee *et al.* (1997) found a peak in macrophyte species richness with approximately 30% shade over the pond margins, comparable to macrophyte richness here, which peaked between 38% and 41% tree cover. Thus, some tree shading may be beneficial to habitat complexity (Biggs *et al.* 1994; Gee *et al.* 1997), although ponds with the highest conservation value had less tree shading (0 – 17%), suggesting the influence of other factors on the macroinvertebrate assemblages beyond habitat complexity, for example, tree shading can affect habitat selection from the outset by affecting visual cues sought by daytime dispersing insects (Remsburg *et al.* 2008).

The effect of shading on nutrient status is multiplicative. Shading inhibits macrophyte growth, which would otherwise reduce nutrient concentrations and the decomposition of organic matter after deciduous trees senesce releases PO_4 into the water column. Nutrient enrichment (PO_4 concentration) was detrimental to species richness within each of the macroinvertebrate groups investigated, with the exception of Hemiptera, which showed little relationship with water quality. Although Diptera richness was not analysed, dominance and richness in this order is likely to increase with nutrient enrichment (Lenat and Crawford 1994; Hamerlik *et al.* 2011). The relevance of PO_4 concentration to macroinvertebrates is likely realised through the detrimental impacts eutrophic conditions have upon aquatic habitats (Smith *et al.* 1999).

Fish were present in pond types with both high (T1) and low (T3) macroinvertebrate and macrophyte diversity. Previous studies have found fish presence to have a negative impact on macroinvertebrate species richness (Wood *et al.* 2001) and on Coleoptera, Odonata and Hemiptera species richness (Schilling *et al.* 2009), and this is likely due to predation pressure. However, the pond habitats with fish presence and high macrophyte richness may benefit from fishery management. Much fishery management aims to keep areas of open water for fishermen and fine sediment is removed to maintain water depth in some places (Wood *et al.* 2001). Consequently, pond succession may be halted, which may otherwise increase the sediment to water depth ratio, nutrient content, produce a more homogenous macrophyte community as more tolerant species prevail and encourage a switch to a more turbid state (Scheffer *et al.* 1993). Ponds of low conservation value also supported fish but were typically associated with poor vegetation complexity and higher suspended sediment loads suggesting a switch to a turbid state has already occurred and was not halted by management intervention.

The influence of fish is not uniform across taxa, which agrees with the findings of Hassall *et al.* (2011). Where fish were absent, Chaoboridae abundance was high. Fish are well known to out-compete Chaoboridae for zooplankton prey, which are frequently found in greater abundance in the absence of fish (Sweetman and Smol 2006; Schilling *et al.* 2009). For Gastropoda and Trichoptera the relationship was positive and for Coleoptera, negative. Coleoptera, particularly predatory Dysticidae (Fairchild *et al.* 2000; Schilling *et al.* 2009), which are active in the water column are probably more vulnerable to fish predation than other macroinvertebrates (Heino 2000). Similarly, hemipteran abundance was reduced in ponds with fish, which is most likely due to their conspicuous, active nature, which makes them an obvious target for insectivorous fish (Cook and Streams 1984). The negative relationship between fish presence and Coleoptera richness has been noted in several further pond studies (Hinden *et al.* 2004; Hassall *et al.* 2011). Insectivorous fish have been shown to have a major impact upon some odonate species (McPeek 1990), although no relationship between odonate richness and fish was found. The strong positive relationships between fish presence and gastropod and trichopteran richness are unlikely to be direct. Where complex marginal habitats exist, predation pressure on macroinvertebrates can be ameliorated (Gilinsky 1984). Angled ponds tended to have higher macrophyte complexity, such that some regularly angled ponds may provide incidental benefits to gastropods and trichopterans by the maintenance of marginal vegetation complexity for the benefit of the fishery. Data on the composition of fish may also provide further insight, for example, benthic feeding fish such as Carp (*Cyprinus carpio*) and Bream (*Abramis brama*), both popular fishing quarry, can have a considerable impact upon macrophyte growth by increasing water turbidity and reducing light availability (Wetzel 2001) whereas impacts by salmonids, for example, may be negligible (Gee *et al.* 1997).

The GLM process was more subtle and highlighted taxa specific factors which were not influential on the macroinvertebrate community as a whole. Fe concentration was important for Gastropod species richness. Fe has been found to be an important control on overall macroinvertebrate composition in urban streams in concentrations > 440 μg/l (Beasley and Kneale 2002; Freund and Petty 2007). At the study median concentration (250 μg/l, Appendix 21), direct effects of Fe on aquatic macroinvertebrates such as those on vertebrates (Vuori 1995) are unlikely, although Fe concentration ranges are considerable (maxima site mean 7672 μg/l). Moreover, Fe can react with dissolved oxygen to produce iron hydroxides, which precipitate in the water column and settle on the sediment surface (e.g.Younger *et al.* 2002). Macroinvertebrate grazers, the feeding guild to which most gastropods are assigned (Cummins and Klug 1979), have been reported to be the first invertebrates eliminated by increasing Fe concentrations (Rasmussen and Lindegaard 1988).

NH_4 concentration was important for trichopteran and odonate richness and frequently exceeded the concentration related to ecological 'good' status under the Water Framework Directive (WFD) (Chapter 4 and Appendix 21). In its own right NH_4 is largely harmless, aside from its potential contribution to nitrification processes, however, in higher pH conditions, >8, it can switch to the more toxic ammonia form (NH_3), which can be harmful to aquatic life (Brönmark and Hansson 2005).

5.4.2.2. Influence of regional factors

The impact of urbanisation on the macroinvertebrate community as a whole at a relatively small spatial extent i.e. 100m (Figure 5.3.2), may be reflective of increasing numbers of diffuse and point pollution sources. Diffuse pollution may result from impermeable surfaces through surface water run-off and point sources from buildings, e.g. residential housing. The relevance of urbanisation within 100m may also be reflective of isolation in the wider environment inhibits he exchange of individuals between separate communities. The importance of effective distance, rather than Euclidean (straight-line), for macroinvertebrate dispersal has rarely been considered in studies of insect dispersal (Fahrig 2007). Yet, inter-habitat urbanisation can increase the influence of dispersal limitation upon macroinvertebrate communities above that of local environmental conditions (Urban *et al.* 2006). AC2km was an important connectivity metric to community composition and richness and performed better than the equivalent Euclidean metric (AP2km, Appendix 23). Urban areas in particular provide considerable resistance to dispersing aquatic insects. Polarized light pollution (Horvath *et al.* 2009), interferes with insects ability to locate aquatic habitat. Plastics, glass buildings, metal exteriors of cars, green metal roofs, oil lakes, and solar panels may all act as polarized ecological traps for a range of polarotactic insects including ephemeropterans, coleopterans and odonates (Ladócsy 1930; Horvath *et al.* 1998; Larson *et al.* 2000; Kriska *et al.* 2006b; Kriska *et al.* 2007; Stevens *et al.* 2007; Malik *et al.* 2008; Horvath *et al.* 2010; Malnas *et al.* 2011). The differences in spatial extent between the macroinvertebrate groups were more pronounced for connectivity metrics than for land-use (Table 5.9).

Aquatic gastropod dispersal is passive, as such they are largely incapable of dispersing themselves between habitats and rely on agents such as animal vectors (Bilton *et al.* 2001). For example, eggs and adults of the river limpet (*Ancylus fluviatilis*) have been found attached to the wing cases of a water beetle (*Acilius sulcatus*) (Davies *et al.* 1982). Dispersal therefore, is contingent on the dispersal abilities of the hosts, which may vary widely from amphibians to waterfowl (see Kappes and Haase 2012 for review). The value of connectivity to water bodies within 250m therefore (Table 5.9), is uncertain but may represent a combination of the average dispersal of the host and the survival of the snail in the terrestrial environment, although this may be possible for several days e.g. *Dreissena polymorpha* (Kappes and Haase 2012).

The strong dispersal abilities of Coleoptera and Hemiptera are reflected in the selection of the most distant spatial extent selected in the GLM models (2500m). Coleoptera and some families of Hemiptera (for example Corixidae and Notonectidae) are generally considered to be strong fliers, capable of dispersing several kilometers and among the first to colonize new sites (Briers and Warren 2000; Davy-Bowker 2002; Lundkvist *et al.* 2002; McAbendroth *et al.* 2005). Most flight by Hemiptera and Coleoptera is for the purpose of finding new habitat (Lundkvist *et al.* 2002), although this may be tempered by behavioral constraints, for example, *Agabus bipustulatus* (Linnaeus, 1767) did not disperse from ponds which had all but dried up (Davy-Bowker 2002). By reviewing isolation of ponds with *Notonecta spp.* present, Briers (1998) suggested that such species were able to disperse at distances greater than 1.6km.

Trichoptera species richness was more closely linked to connectivity metrics at 500m. Median dispersal distances of Trichoptera away from Lake St. Clair and the Detroit River, Canada, were recorded up to 1.558km (Kovats *et al.* 1996), suggesting considerable dispersal potential, especially in light of their relatively long adult life span (up to one month, Merritt and Cummins 1996). However, within streams at least, Trichoptera have typically exhibited a propensity to remain within the riparian zone (for example Collier and Smith 1998), as is the case for many aquatic insects (Merritt and Cummins 1996).

For odonates, connectivity metrics were most important at 250m distance, similarly, land-use, which is particularly relevant to odonate foraging during their adult form was most relevant within 100m. Odonates have a strictly terrestrial life-stage and display a high degree of philopatry (Angelibert and Giani 2003) meaning that they may move away to forage before returning to develop territories (Conrad *et al.* 1999). Although possessing long dispersal capabilities (e.g. > 1km, Rouquette and Thompson 2007) odonates do not, in general, fulfill their dispersal potential. This reflects behavioral constraints to dispersal (Macneale *et al.* 2005), which maintains the general consensus that long range dispersal in adult aquatic insects is a typically rare event (Bohonak and Jenkins 2003). Many recaptures of odonates in mark-recapture studies have occurred less than 100m from initial capture locations. For example, only 27% of marked *Enallagma cyathigerum* (Charpentier, 1840) moved a distance greater than 100m (Garrison 1978) and between 65.7% and 87.6% of *Coenagrion mercuriale* (Charpentier, 1840) and *Ischnura pumilio* (Charpentier, 1825) moved a distance less than 50m.

Table 5.9. Most relevant spatial extents for land-use and connectivity metrics for each of the macroinvertebrate groups studied as well as general dispersal traits

Taxa	Land-use	Connect.	Dispersal characteritics
Gastropoda	200m	250m	Passive, host dependent
Coleoptera	500m	2500m	High
Trichoptera	200m	500m	Medium
Odonata	100m	250m	Low
Hemiptera	25m	2500m*	High

* Model not significant

5.4.3. Separating out the effects of local and spatial factors determining macroinvertebrate community composition

Variance partitioning (Borcard *et al.* 1992) revealed the proportion of explained variance in macroinvertebrate composition that was shared between local (water quality and local physical factors) and regional (land-use and connectivity) factors. Urban pond macroinvertebrate assemblages tended to exhibit niche-based species-sorting as the driving structural mechanism (e.g. Leibold *et al.* 2004), although there was some support for dispersal limitation-related neutrality (e.g. Volkov *et al.* 2003). This result agrees well with a review of 158 datasets that found that most ecological communities had a significant environmental component and therefore, species sorting dynamics were often dominant (Cottenie 2005). In order for dispersal to show little to no relationship with community composition it must not be limited (Fuentes 2002). Although local factors were the most influential, the importance of the connectivity component indicated dispersal limitation was present for some members of the macroinvertebrate community.

Local physical factors were more relevant than water quality, explaining approximately three times as much variation in community assemblage. This suggests that the influence of water quality did not outweigh the effects of habitat factors. In effect, the restrictions to habitat development, for example, from light exclusion and hard engineering preceded the impact of water quality despite highly eutrophic conditions and widely fluctuating temporal water quality at many sites. Nonetheless, local physical factors and water quality are inextricably linked and dependent on each other, as suggested by their shared variance (also see Chapter 4).

5.5. Conclusion

The results of this study have important implications with respect to the management of urban ponds. First, urban ponds have the capacity to support highly diverse macroinvertebrate communities and to support rare species. Despite previous studies highlighting the negative effect of management for angling (Wood *et al.* 2001), careful management of sites used for amenity purposes such as angling, could also promote species richness by encouraging complex macrophyte growth. Macrophyte complexity, as well as providing good macroinvertebrate habitat (Williams *et al.* 1999) will provide food and protection for juvenile fish and consequently improve the quality of the fishery. The encouragement of macrophyte habitat will help to ameliorate the impact of nutrient enrichment. Improved vegetation complexity can delay the switch to a turbid state (Scheffer *et al.* 1993) and an associated decrease in flora and fauna diversity. Management of nutrient input in many instances requires immediate attention in the sites studied here. However, the overall taxonomic community richness found during the study is representative of the heterogeneity of the sites studied; even nutrient rich and macrophyte poor sites contributed unique taxa.

The study adds further weight to the need to consider the wider pond network and *beta* rather than *alpha* diversity (Williams *et al.* 2004). Although local factors were more influential on the macroinvertebrate community, spatial factors were important indicating some dispersal limitation is apparent. Urbanisation increases isolation, whether by increasing distances between pond habitats or increasing inter-habitat landscape resistance. Managers should, therefore, assess the availability of habitat beyond a single site and aim to increase ecological resilience within the landscape by improving the provision of quality pond habitats.

The spatial scale on which managers should focus depends on the taxa they wish to promote. The overall macroinvertebrate community is likely to benefit from considerations of land-use within 100m and availability of habitats within 1000m. A more complex, but potentially more beneficial strategy would be to better facilitate aquatic insect dispersal through the urban landscape, as well as consider the quality of neighbouring habitats (although only area was as used a proxy used here) within 2000m (as indicated by AC2km). Macroinvertebrates that are known to be strong dispersers are likely to be more sensitive to connectivity within the wider landscape, at least to 2500m, those that are poor dispersers or whose dispersal is behaviourally constrained (e.g. Odonata) are likely to benefit from a consideration of habitat availability within shorter distances (e.g. 250m).

6

Leaf litter breakdown as a measure of ecosystem function in ponds across an urban land-use gradient

Chapter 6: Leaf litter breakdown as a measure of ecosystem function in ponds across an urban land-use gradient

6.1. Introduction

Macroinvertebrate diversity declines with catchment urbanisation because polluted sites are dominated by tolerant species (e.g. Chironomidae and Oligochaeta) with few sensitive taxa (Roy *et al.* 2003; Walsh *et al.* 2003; Moore and Palmer 2005; Carroll and Jackson 2008). Furthermore, the biomass of tolerant taxa increases relative to more sensitive taxa of the orders Ephemeroptera, Plecoptera and Trichoptera (Paul and Meyer 2001) and similar trends have been noted for microbes (Lecerf and Chauvet 2008). Declining species diversity may have severe implications for the functional integrity of aquatic ecosystems (Chapin *et al.* 2000) since lost functional redundancy may constrain compensatory processing by the remaining tolerant species. Macroinvertebrates are categorised into functional feeding groups depending on their modality of feeding i.e. shredders, deposit and filter feeders, scrapers, piercers and predators (Usseglio-Polatera 1994; Tachet *et al.* 2002). Some urban streams have been found to be devoid of shredders (Paul *et al.* 2006), which are litter-feeding macroinvertebrates important for the decomposition process of organic matter (Cummins and Klug 1979; Tachet *et al.* 2000), and populations of scrapers, gatherers and filterers can be impoverished (Roy *et al.* 2003). In contrast, urbanisation frequently results in a greater abundance of non native species (McKinney 2006). Vermonden *et al.* (2010), for example, found five non-native Crustacea species with shredder traits in a study of urban drainage systems. In these systems, non-native Crustacea were the only species present in the most turbid, nutrient-rich water-bodies (Vermonden *et al.* 2010), consistent with the suggestion that highly disturbed ecosystems are more vulnerable to invasion (Elton 1958; Tilman 1999). Under such circumstances it is possible for pollution tolerant non-native species to functionally compensate for the loss of a native species.

Research addressing the impact of urbanisation on ecosystem functioning in aquatic systems is relatively new, emerging over the past decade (Wenger *et al.* 2009). Historically, structural metrics (e.g. taxon richness) were used to monitor freshwater pollution (Paul and Meyer 2001; Moore and Palmer 2005), but more recently leaf litter decomposition experiments have been promoted to assess the functional implications of anthropogenic impacts to streams and rivers (Gessner and Chauvet 2002; Pascoal *et al.* 2003; Chadwick *et al.* 2006; Wenger *et al.* 2009). Allochthonous inputs of leaf litter are an important basal resource in most freshwater systems (Gregory *et al.* 1991) and the rate at which leaf litter is broken down influence nutrient cycling and secondary production (Oertli 1993). The decomposition process is carried out by microorganisms (bacteria and fungi) and shredding macroinvertebrates (Gessner *et al.* 1999) and initial colonisation by microbial decomposers (e.g. hyphomycetes) promotes the palatability of leaf litter to macroinvertebrate shredders (Suberkropp and Wallace 1992). Consequently, the rate of leaf decomposition depends on the microbial and macroinvertebrate communities and the relationships between the two. The relative contribution made to leaf breakdown rates by each can be assessed by using leaf packs of differing mesh size to include or exclude macroinvertebrates (Boulton and Boon 1991).

Leaf litter decomposition is controlled by a variety of factors including water chemistry, hydrology (particularly in riverine systems) and the abundance and taxonomic composition of macroinvertebrate and microbial assemblages. Leaf breakdown rates in urban streams have been shown to increase (Collier and Winterbourn 1986; Walsh *et al.* 2003; Meyer *et al.* 2005) or show no detectable change with the degree of urbanisation (Sponseller and Benfield 2001; Huryn *et al.* 2002). Increased breakdown in urban streams may be attributable to flashy flow regimes and the physical fragmentation of leaf litter from more abrasive flow (Paul *et al.* 2006). Imberger *et al.* (2008), found that breakdown rates by microbes increased with increasing catchment impermeability and consequent elevated phosphate concentrations and water temperature, a result consistent with Pascoal *et al.* (2005). Microbial activity in these urban streams outstripped contributions to breakdown rates by macroinvertebrate shredders and abrasive flow. Microbial decomposition rates are thought to be contingent on nutrient concentrations (i.e. *P* and N) (Gulis and Suberkropp 2003; Pascoal *et al.* 2003; Pascoal *et al.* 2005; Goncalves *et al.* 2006). For example, aquatic hyphomycetes growing on leaves take up nutrients from the surrounding water (Suberkropp 1998). However, Pascoal *et al.* (2005) found that lower current velocity and increased sedimentation in one study stream masked the influence of nutrient enrichment. In contrast, Chadwick *et al.* (2006) found peak breakdown rates at intermediate levels of urbanisation (i.e. ~40% catchment impermeability) driven by abundant fungi and macroinvertebrates. Studies finding little difference between breakdown rates in areas of differing land-use may be confounded by agricultural run-off (Hagen *et al.* 2006; Paul *et al.* 2006), which can also increase stream nutrient concentrations, thereby masking the impact of catchment urbanisation.

Very few studies have undertaken leaf decomposition experiments in ponds or lakes. Ponds are small systems, in which aquatic-terrestrial linkages are more pronounced (Declerck *et al.* 2006), for example, in small streams, overhanging vegetation can shade a relatively large proportion of the aquatic habitat than in larger systems (Vannote *et al.* 1980). Consequently, riparian vegetation can reduce solar energy available for primary production (Hill *et al.* 1995) and provide a greater input of allochthonous material (e.g. leaves, woody debris). Littoral zones of ponds are often oxygen deficient, and is exacerbated by lower light conditions (Birch and McCaskie 1999; Fairchild *et al.* 2005). Bjelke (2005) studied the effect of low oxygen levels on the leaf processing efficiency of five lake-dwelling macroinvertebrate shredders, finding that at 1 mg/l O_2, none of the species fed and levels of 2 mg/l O_2 and normoxia (9 mg/l O_2) affected species to different extents (also see Bjelke and Herrmann 2005). Temporal niches of shredders, i.e. the timing of their growth to different periods of detritus input and composition has also been observed (Bjelke *et al.* 2005). Further effects to leaf processing rates within ponds are likely to arise from increased sedimentation due to the lack of flow, as in pools within river systems (Cummins *et al.* 1980) and similar to beaver ponds (Hodkinson 1975a), in which much of the allochthonous energy budget is accrued in the sediments (Hodkinson 1975b).

This study investigates leaf decomposition rates across an urbanisation gradient derived from a GIS at a spatial scale (100m from the pond edge) known to be relevant to both water quality (Chapter 4) and macroinvertebrate community composition (Chapter 5), consistent with several other studies (Declerck *et al.* 2006; Akasaka *et al.* 2010; Williams *et al.* 2010). Analysis of both structural indicators (macroinvertebrates) and a direct functional indicator (leaf decomposition rates) sheds light on the ecological integrity of urban pond systems, which have received scant attention to date.

6.1.1. Research questions

Research in this chapter investigates how the rates of an important ecosystem process, leaf litter decomposition, varies along an urbanisation gradient. The research posed four key questions:

1. Does urbanisation alter ecosystem functioning and what are the relative effects on leaf litter decomposition by microbes and macroinvertebrates in ponds?

2. How important are local environmental factors such as water quality and physical habitat in shaping leaf-litter breakdown rates in ponds?

3. Which macroinvertebrate species are important for leaf litter decomposition in urban ponds and how are they affected by urbanisation?

4. Are macroinvertebrate assemblages good indicators of ecosystem functioning in urban ponds?

6.1. Methods

6.1.1. Study site selection

Thirty ponds of contrasting surrounding land-use were selected from an estimated 1023 sites in Birmingham and The Black Country (BBC). For the full site selection process see Section 2.1.

Mature vegetation >3m in height within 10m of the study pond edges typically comprised of hawthorn (*Crataegus spp.*), silver birch (*Betula pendula*), crack (*Salix fragilis*) and grey willow (*Salix cinerea*), holly (*Ilex aquifolium*) and alder (*Alnus glutinosa*) (Appendix 25).

6.1.2. Experimental outline

A leaf breakdown experiment was conducted over four months in late summer 2010 (July - October). Abscised beech (*Fagus sylvatica*) leaves were collected from Winterbourne Botanical Gardens, Edgbaston, Birmingham once in December 2009. Water samples were collected from each pond in June and August 2010 (Table 6.1). Mean values were calculated from the two sampling periods for all water quality variables.

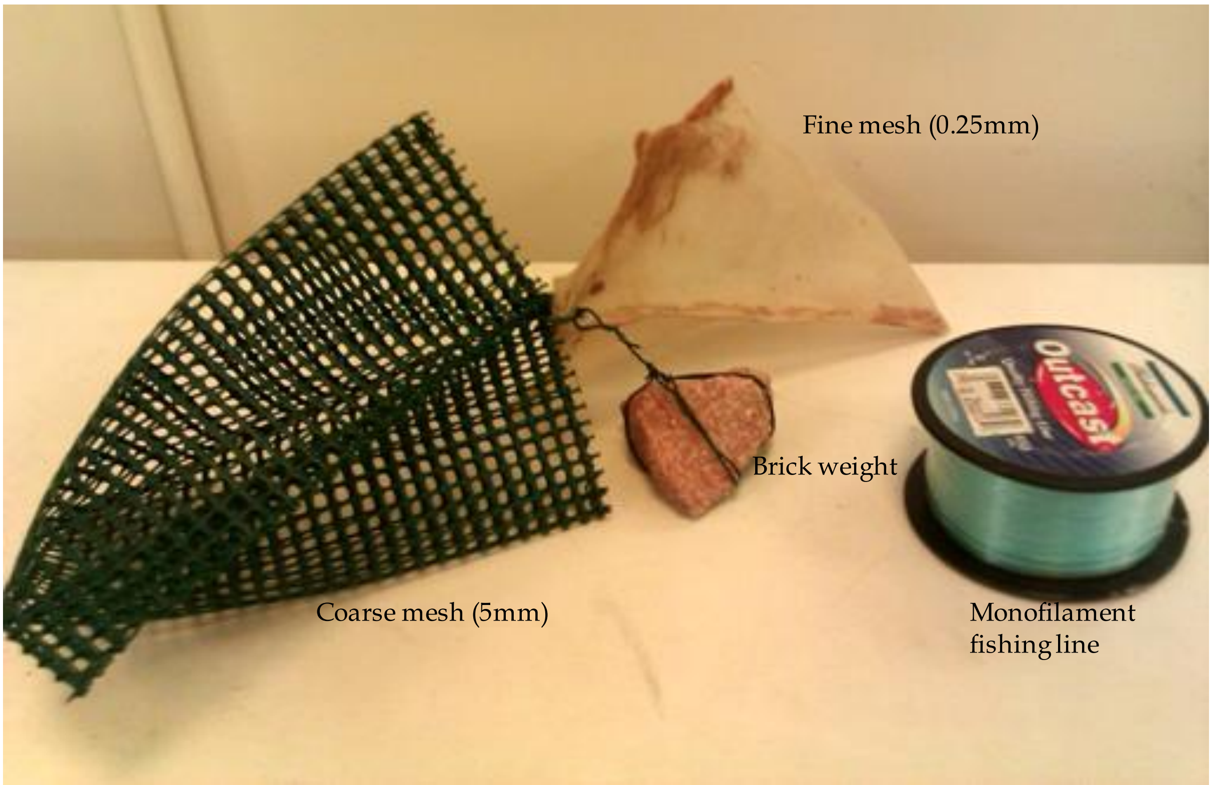
6.1.3. Leaf decomposition

Beech leaves were air-dried in laboratory conditions and 5g (± 0.05g) of leaf litter was placed in each leaf pack (15 cm x 15cm plastic mesh). The air-dried leaves were wetted with deionised water before being deposited in the pack in order to increase their flexibility and avoid leaf breakage. The mean air dry mass of a single leaf was 0.0065g (*n* = 76, 1 pack). Leaf packs of two mesh sizes were used. Coarse leaf packs and fine leaf packs had a 5mm and 0.25mm mesh respectively. The fine mesh packs excluded the vast majority of macroinvertebrates; however, a limited number of very small macroinvertebrates (e.g. Oligochaeta, chironomidae) gained access. The leaf packs were closed in a disphenoid-like shape in order to avoid artificial leaf compaction that may otherwise limit macroinvertebrate access (Figure 6.1.1). At each site, three pairs of leaf packs (one fine, one coarse) were anchored by a brick weight and, where necessary, fixed in position using a peg and monofilament line.

Table 6.1. Sampling periods and dates of sampling

Season and hydrological year	Sampling dates
Early summer 2010	9th June – 11th June 2010
Summer 2010	18th August – 20th August 2010

Figure 6.1.1 Example of leaf pack pair (without leaves), with brick weight attached and monofilament line



6.1.3.1. Conversion factors and leaching

In addition to the experimental leaf packs that were used in the study, four leaf packs were used to calculate the initial oven-dry mass of each leaf pack, by correcting for moisture content and leaching effects. To do so, two non-experimental air-dry leaf packs were oven dried (105°C, 48 hours) and individually weighed, to correct the air-dry masses of the experimental leaf packs for moisture content, using the following correction factor: m = mean oven-dry mass/mean air-dry mass. A further two bags were used in a laboratory test to correct for losses due to leaching (the loss of soluble organic and inorganic components), these leaf packs were submersed in deionised water for 72 hours, oven-dried (105°C, 48 hours) and then weighed to the nearest 0.01 g. The conversion factor for correcting leaf packs for leaching processes is: l = (mean post-leaching oven-dry mass) / (mean air-dry mass*m), where m corrects for moisture content as above.

To correct for the non-organic, ash component of leaves, one conversion factor was applied to the initial air-dry leaf mass and a second was applied to the leaf mass remaining after the study period. The first was achieved by reducing 5 x 0.5g (±0.025g) non-experimental samples of ground leaf in a muffle-furnace at 550°C for two hours and calculating the mean remaining weight. The second was achieved by reducing a subsample of 0.5g (±0.025g) of the remaining leaf litter from each experimental leaf pack post-submersion using the same method. The following conversion factor was then applied to the initial and post-study oven-dry leaf mass: a = (post furnace ash mass) / (oven dried subsample mass).

In total, the initial pre-submersion air-dry mass of leaf packs for the determination of breakdown rates was corrected from 5.0g (±0.05g) to 4.139g (±0.05g), a 17% reduction to account for moisture content, leaching effects and ash content. All values used throughout the chapter have been corrected for moisture content and leaching processes.

6.1.3.2. Calculation of leaf processing rate

Processing rates were calculated for each leaf bag using the exponential decay coefficient, 'k' (Petersen and Cummins 1974). This commonly used measure of decomposition rate assumes that for any amount of material at any time there is a constant fractional loss. Calculating k values is useful in that it allows processing rates to be comparable among different aquatic systems. k was calculated following the formula.

$$-k = \frac{ln\ ((dm_r \times a)/dm_i \times mla)}{d} \qquad \textbf{Eq. 6.1.1}$$

Alternatively written as:

$$-k = \frac{ln\ (AFDM_r/AFDM_i)}{d} \qquad \textbf{Eq. 6.1.2}$$

Where in Eq. 6.1.1, dm_i is the initial mass in the leaf pack in grams and dm_r is the mass remaining in grams of the leaf pack once removed from the pond. The operators m (moisture content), l (leaching) and a (ash component) are described in section 6.1.3.1. Time (d) was expressed as the number of days submersed. In Eq. 6.1.2, $AFDM_r$ is the ash-free dry mass of the leaf litter remaining post-experiment and $AFDMi$ is the ash-free dry mass prior to submersion. Breakdown rates were calculated for total breakdown (macroinvertebrate and microbial breakdown) in the coarse bags (k_{total}) and microbial only breakdown in the fine mesh bags ($k_{microbe}$). Macroinvertebrate only breakdown (k_{invert}) was determined by subtracting the coarse mesh breakdown rate by the fine mesh breakdown rate for each leaf pack pair ($k_{total} - k_{microbe}$).

6.1.4. Macroinvertebrates

All leaf packs were collected after a mean of 80 days incubation, transferred to plastic bags, and stored at -15°C. In the laboratory, leaves were removed from each pack and rinsed over a 0.5mm sieve, and any macroinvertebrates sorted from leaf debris identified and counted. Macroinvertebrates were later classified to functional feeding groups (FFG) *after* Tachet *et al.* (2002). Tachet *et al.* (2002) use fuzzy coding, such that macroinvertebrates may be classified to more than one FFG, here, the FFG was selected to which the taxa had highest affinity i.e. highest rank. Leaf material was oven-dried to constant mass at 105°C and weighed to the nearest 0.001g.

6.1.5. Water quality

Samples were kept cool, filtered (GF/C, 1.2μm) and returned to the laboratory for major ion analysis. Triplicate *in situ* measurements of pH, electrical conductivity (μS/cm), temperature (°C) and dissolved oxygen (% saturation) were taken and the average measurements were calculated. For each determinand, data were averaged for the two sampling periods to provide a single datum point for each site. Further detail on water quality sampling and analysis methods are discussed in Section 2.2.

6.1.6. Local physical factors

Single measurements of pond surface area and percentages of surface area classified as open water, shaded, riparian vegetation and floating vegetation were all derived from a combination of digitised field notes and Normalised Difference Vegetation Index (NDVI) and photogrammetrically derived height GIS layers against digital pond outlines. The percentage of pond bank that was made from concrete was also calculated in a GIS from digitized field notes. Water sources (i.e. stream inflows, groundwater, surface run-off or building run-off) and fish presence information was gathered from stakeholder knowledge and visual inspection. A water level fluctuation index (WLFI) was calculated as the standard deviation in depth between surveys (taken quarterly between 2010 – 2011) measured at set points within the wetted-perimeter of each pond. Mean values for suspended solids and chlorophyll *a* were calculated from the two sampling periods (Section 2.2.3). Further detail on the derivations of local physical factors is given in Section 2.2.

6.1.7. The urbanisation gradient

Chapters 4 and 5, as well as previous pond research (Declerck *et al.* 2006; Williams *et al.* 2010), have highlighted the 100m scale as being relevant to both water quality and macroinvertebrate community composition of ponds. Thus, land-use, and thus urbanisation, at the 100m scale was considered relevant to urban ponds as a catchment proxy, and it is at this scale that the impact of urbanisation is tested here. Using land-use data within 100m, a PCA of ‘urbanisation indicators ‘(see Section 2.3.1.7) explained 43.3% of variance on axis one (PCA 1), which was positively correlated with the proportional coverage of towns and villages, impermeable surfaces, suburban, building intensity and roads, and negatively correlated to the proportion of arable land-use (**Figure** 6.1.2). Axis 2 (PCA 2) explained 23% of the variance and was related to social indicators such as housing and population density. Four groups of ponds were identified using the PCA 1 scores as follows: rural (n = 7), suburban (n = 10), dense suburban (n = 7) and urban (n = 6) (Figure 6.1.3).

6.1.8. Statistical analyses

Differences in k_{total}, $k_{microbe}$, k_{invert}, macroinvertebrate and macroinvertebrate shredder abundance and density were compared among ponds in the four urbanisation categories using 1-way analysis of variance (ANOVA) followed by Tukey or Tukey-Kramer *post-hoc* multiple comparison tests. The Tukey-Kramer *post-hoc* test is the equivalent Tukey test for comparisons between uneven group sizes.

Ordination was employed in order to analyse relationships between physical and chemical variables, functional feeding groups and shredder community composition with leaf breakdown rates. Macroinvertebrate species that were recorded in < 5% of the study sites were excluded from the shredder community ordination because rare species may

Figure 6.1.2 Principal Components Analysis of 10 ‘urbanisation’ indicators *TwnVill* ‘Towns & villages’, *IS* Impermeable surface, *BdInt* Building intensity, *Suburb* Suburban coverage, *Road* Road coverage, *HsDens* Housing density, *PopDens* Population density, *RurEmp* % people in rural employment, *Urban* Urban land coverage, *Arable* Arable land-use coverage

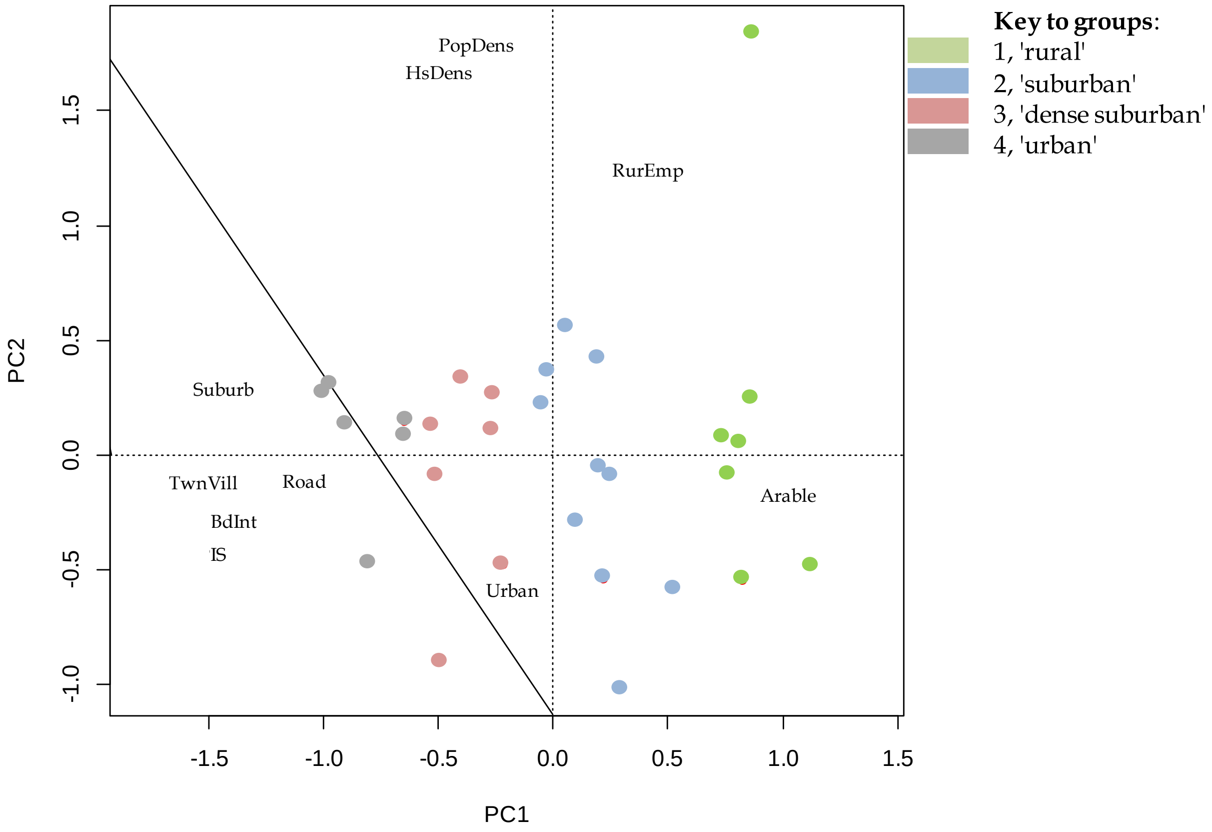
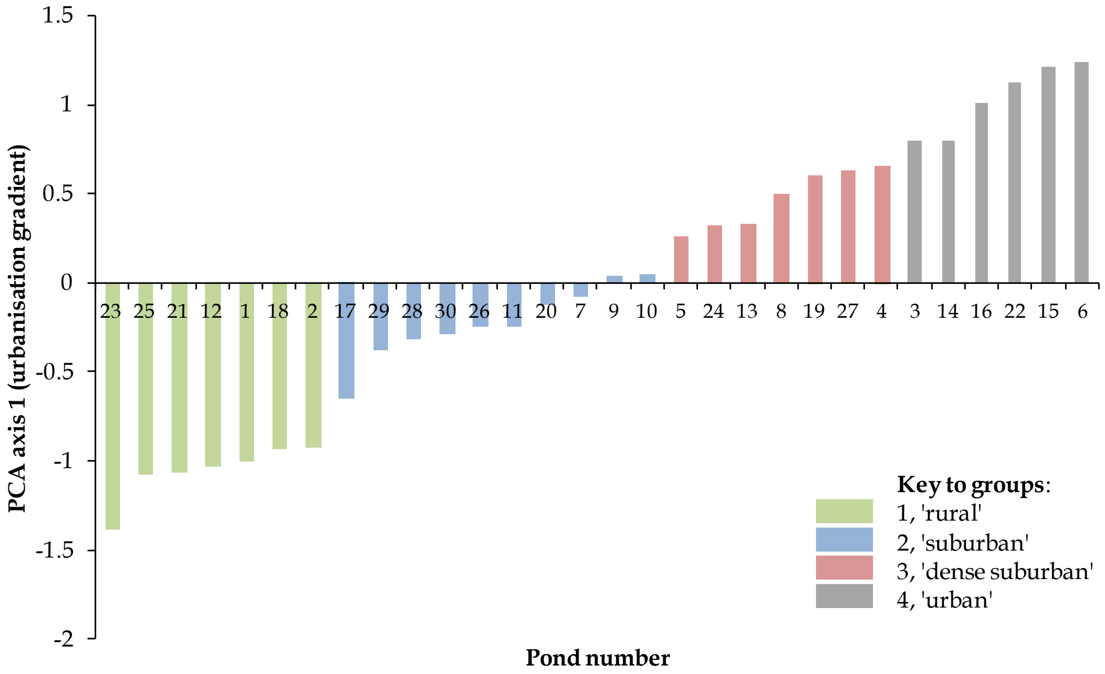


Figure 6.1.3. Pond groups along a gradient of urbanisation derived from axis 1 (PCA 1) of a principal components analysis of 10 ‘urbanisation indicators’ within a 100m buffer



confuse underlying patterns in community analysis (Faith and Norris 1989; Heino and Mykra 2008).

The linear form of ordination, redundancy analysis (RDA), was used rather than unimodal canonical correspondence analysis (CCA) because preliminary detrended correspondence analysis (DAC) indicated a short maximum gradient length (DAC axis 1 SD < 2, 0.85) (Lepš and Šmilauer 2003). In order to analyse the relationship between functional feeding groups with breakdown rates, the relative abundance of taxa belonging to each functional feeding group was summed for each pond. Significance of ordination axes in each RDA was assessed by ANOVA (<999 permutations, $P < 0.05$).

Prior to all analyses breakdown rates were square-root transformed to improve normality and homogeneity of variance for use in parametric tests, as tested by Shapiro-Wilk and Levene's tests carried out in using IBM SPSS statistical package version 19 (IBM, Armonk, New York). Similarly, water quality and local physical variables were square-root or log(n+1) transformed where necessary. Prior to ordination, physical and chemical variables were standardised to 0 mean and one standard deviation to account for different measurement units. Collinearity amongst independent variables was assessed by production of pairwise plots (Pearson's correlations), retaining one variable from any two where $r > 0.7$. Variance Inflation Factors (VIFs) were also checked using the package 'nlme' (Pinheiro *et al.* 2012). With reference to pairwise plots, variables were removed until all VIF < 5. Several threshold values for detecting multicollinearity with VIFs have been proposed, for example, 10 (Quinn and Keough 2002; Oksanen *et al.* 2012), 5 (Montgomery and Peck 1982; ter Braak and Šmilauer 2002) or 3 or lower (Zuur *et al.* 2010). The selection of a moderate threshold value means that variables with a VIF >5 have <20% of unique variance within a set of predictors.

Multiple comparison tests were carried out using IBM SPSS statistical package version 19 (IBM, Armonk, New York). VIFs were calculated in the package 'rms' (Harrell 2011) for R statistical program version 2.15.1 (R Core Team and contributors worldwide 2012). Ordinations were carried out within the R package 'vegan' (Oksanen *et al.* 2012) using the functions 'rda' for PCA and RDA and 'decorana' for DAC and tests of significance within ordinations were carried out using the 'anova' function.

6.2. Results

Mean k_{total} was 0.0020/d (range 0.0001 – 0.0077), whilst $k_{microbe}$ and k_{invert} both averaged 0.0010/d. Maximum $k_{microbe}$ and k_{invert} was 0.0036/d and 0.0055/d respectively. Breakdown rates did not differ significantly between $k_{microbe}$ and k_{invert} (ANOVA, *post-hoc* Tukey, Figure

6.2.1). Analysis of the relative contributions to total breakdown made by $k_{microbe}$ and k_{invert} revealed an equal split between the numbers of study sites at which each was the main driver of leaf decomposition. Mean percentage contribution to k_{total} by $k_{microbe}$ was 49.2% (range 5.6% - 100%) and mean percentage contribution to k_{total} by k_{invert} was 50.8% (range 0% - 94.4%). Consequently, neither macroinvertebrates nor microbes dominated breakdown processes across the study. Mean total mass loss due to the action of both macroinvertebrates and microbes was 13.8% (range 1.5% - 40.9%). Mean mass loss due the action of microbes only was 7.4% (range 0% - 24.8%). Mean mass loss by macroinvertebrates only was 6.4% (range 0% - 27%).

6.2.1. Urbanisation and ecosystem functioning

Significant differences in land-use composition between the land-use categories validated the grouping methodology (Table 6.2), in particular all four groups had statistically different (and increasing across the gradient) proportional coverage of impermeable surface within 100m. No statistically significant effect of urbanisation on k_{total} was found or to that attributable to k_{invert} or $k_{microbe}$ (ANOVA, $P > 0.05$, Figure 6.2.2). There was, however, a statistically significant effect of land-use on macroinvertebrates, both in terms of their total abundance within the leaf packs (ANOVA $P < 0.05$) and their total absolute density in leaf packs (ANOVA, $P < 0.01$). *Post-hoc* (Tukey-Kramer) tests revealed that macroinvertebrate numbers were greater in the urban group than in suburban ($P < 0.05$), reflecting increases in *A. aquaticus*, chironomids, oligochaetes and the gastropod, *Hippeutis complanatus* (Linnaeus 1758), but not between rural and dense suburban ($P > 0.05$). Mean total macroinvertebrate abundance was 2.8 and 2.5 times greater in the urban than the rural and suburban groups respectively. Nevertheless, there was no evidence that numbers of shredders varied among ponds of contrasting land-use (Figure 6.2.3).

6.2.2. The influence of macroinvertebrates on leaf breakdown rates

Macroinvertebrate abundance within the 5mm mesh leaf packs ranged between nine individuals to >1000 individuals. The core taxa (those comprising, on average, >1% of the macroinvertebrate community) comprised of the water hoglouse *Asellus aquaticus* (Linnaeus 1758), non-biting midges Chironomidae, a leech of macroinvertebrates *Helobdella stagnalis* (Linnaeus 1758), an introduced freshwater shrimp *Crangonyx pseudogracilis* (Bousfield 1958), two snail species; the flat ramshorn *Hippeutis complanata* (Linnaeus 1758) and *Bithynia tentaculata* (Linnaeus 1758) and the lake limpet *Acroloxus lacustris* (Linnaeus 1758). Six functional feeding groups (*after* Tachet *et al.* 2002) were identified; shredders, piercers, deposit feeders, predators, scrapers and filter feeders (Appendix 25).

Fourteen shredder taxa were collected from the 30 ponds and shredder density varied considerably (minima 0.25, maxima 196 ind./g AFDM). The most frequently occurring and abundant taxa were *A. aquaticus* (28 sites, mean 82 individuals) and *C. pseudogracilis* (16 sites, mean 24 individuals). *A. aquaticus* was the only shredder species to show an increase in numbers as the groups became more urban. All other shredder taxa were relatively uncommon and contributed <1% total shredder community; seven were present at single sites only. At single sites, uncommon shredder taxa contributed <23.7% to total shredder numbers (Tipulidae, site 28, suburban), although few contributed >2%.

Within each pond group, *A. aquaticus* was the most abundant shredder species, followed by *C. pseudogracilis*, which are both approximately an order of magnitude greater in mean relative abundance than the next most common (Figure 6.2.4). The richness of shredders occurring at more than one site decreases with increasing urbanisation from seven to three, and if extended to include those occurring at one site only, from 11 to three.

Two non-native shredder species are shared across the pond groups and one non-native predator was recorded (*Dugesia tigrina,* Girard, 1850) (Table 6.3). The native shrimp *G. pulex* was only found within the two most rural groups and at a total of just two sites (Figure 6.2.4), whereas *C. pseudogracilis* was present in 16, 62% of the study ponds. Of all the shredders, five were of the order Trichoptera, suggesting their functional importance within the shredder group, however, no Trichoptera were found in the most urban of sites.

The first two axes of an RDA of functional feeding group mean relative abundance against leaf breakdown rates were significant (ANOVA, $P < 0.05$, Figure 6.2.5). The first axis, to which shredders were strongly correlated (Pearson's correlation: $r = 0.87$, $P < 0.01$), explained 42.4% of variance. The second axis, to which filterer collectors were correlated (Pearson's correlation: $r = 0.78$, $P < 0.01$), explained a further 7.7%. Shredder density (/g AFDM) was correlated to k_{total} and k_{invert} (Figure 6.2.7a & b).

Further breakdown of species within the shredder group in a second RDA revealed important macroinvertebrate taxa for decomposition rates (Figure 6.2.6). The first two axes were significant (ANOVA, $P < 0.05$). The first axis, to which the mean relative abundance of *A. aquaticus* (Pearson's correlation: $r = 0.77$, $P < 0.01$) and *C. pseudogracilis* (Pearson's correlation: $r = 0.73$, $P < 0.01$) were strongly correlated, explained 52.4% of variance. The second, to which *G. pulex* (Pearson's correlation: $r = -0.45$, $P < 0.05$) and *P. antipodarum* (Pearson's correlation: $r = 0.39$, $P < 0.05$) were weakly correlated, explained a further 13.7%.

Figure 6.2.1 Total (k_{total}), microbe only ($k_{microbe}$), and macroinvertebrate only (k_{invert}), breakdown rate comparisons across the study sites (n = 59).

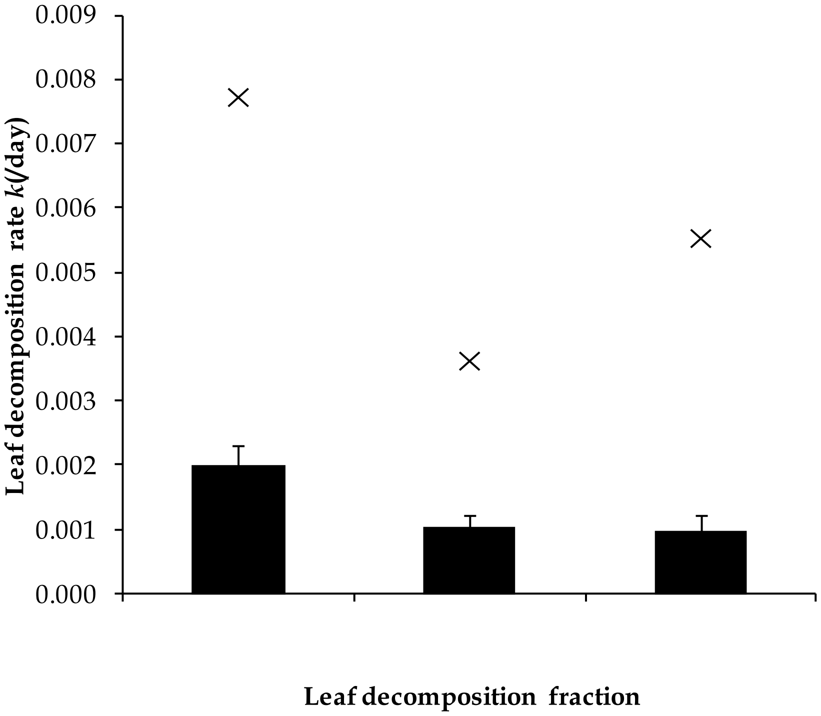


Table 6.2. Differences in land-use composition (proportional coverage) between 4 groups of ponds along a gradient of urbanisation within a 100m buffer. Bold type indicates an 'urbanisation indicator' used to formulate the urbanisation gradient. Lettering denotes significant differences between groups (ANOVA *post-hoc* Tukey Kramer, $P < 0.05$)

	1, 'rural'	2, 'suburban'	3, 'dense suburban'	4, 'urban'
Arable	0.32 ± 0.32^{a}	0.11 ± 0.19^{ab}	0.00 ± 0.00^{b}	2e^{-4} ± 5e^{-4b}
Suburban	0.05 ± 0.07^{a}	0.25 ± 0.22^{a}	0.62 ± 0.17^{b}	0.89 ± 0.24^{c}
Urban	0.01 ± 0.02	0.05 ± 0.09	0.003 ± 0.008	0.08 ± 0.20
% rural employment	0.07 ± 0.15	0.004 ± 0.003	0.007 ± 0.008	0.004 ± 0.002
Population density	33.2 ± 11.5	32.6 ± 13.3	32.8 ± 9.4	38.0 ± 2.7
Housing density	13.4 ± 4.0	13.3 ± 4.1	13.3 ± 4.4	16.1 ± 1.6
Road	0.003 ± 0.006^{a}	0.046 ± 0.046^{b}	0.063 ± 0.017^{b}	0.081 ± 0.037^{c}
'towns & villages'	0.71 ± 0.25^{a}	0.72 ± 0.15^{b}	0.56 ± 0.12^{c}	0.65 ± 0.21^{c}
Impermeable surface	0.06 ± 0.06^{a}	0.13 ± 0.12^{b}	0.15 ± 0.05^{c}	0.21 ± 0.07^{d}
Building intensity	0.01 ± 0.03^{a}	0.16 ± 0.12^{a}	0.46 ± 0.10^{b}	0.83 ± 0.46^{c}
Improved grassland	0.21 ± 0.30^{ab}	0.32 ± 0.27^{b}	0.26 ± 0.16^{b}	0.01 ± 0.03^{a}
Water	0.087 ± 0.055^{a}	0.044 ± 0.059^{ab}	0.007 ± 0.012^{b}	0.021 ± 0.046^{b}
Woodland	0.6e^{-3} ± 0.002^{a}	0.012 ± 0.017^{ab}	0.085 ± 0.145^{ab}	0.077 ± 0.092^{b}
Vegetation, 0m	0.16 ± 0.10	0.27 ± 0.22	0.24 ± 0.11	0.20 ± 0.08
Vegetation, 0-3m	0.42 ± 0.23	0.24 ± 0.13	0.32 ± 0.17	0.44 ± 0.16
Vegetation, 3-60m	0.42 ± 0.25	0.49 ± 0.26	0.45 ± 0.14	0.36 ± 0.12

*Proportional coverages do not equal 1 due to the presence of other land-uses aside from those listed

Figure 6.2.2 Comparison of mean (+1 SE) breakdown rates per day in ponds along an urban land-use gradient a) total breakdown b) microbial breakdown c) macroinvertebrate only and d) the relative contribution to total breakdown made by microbial activity

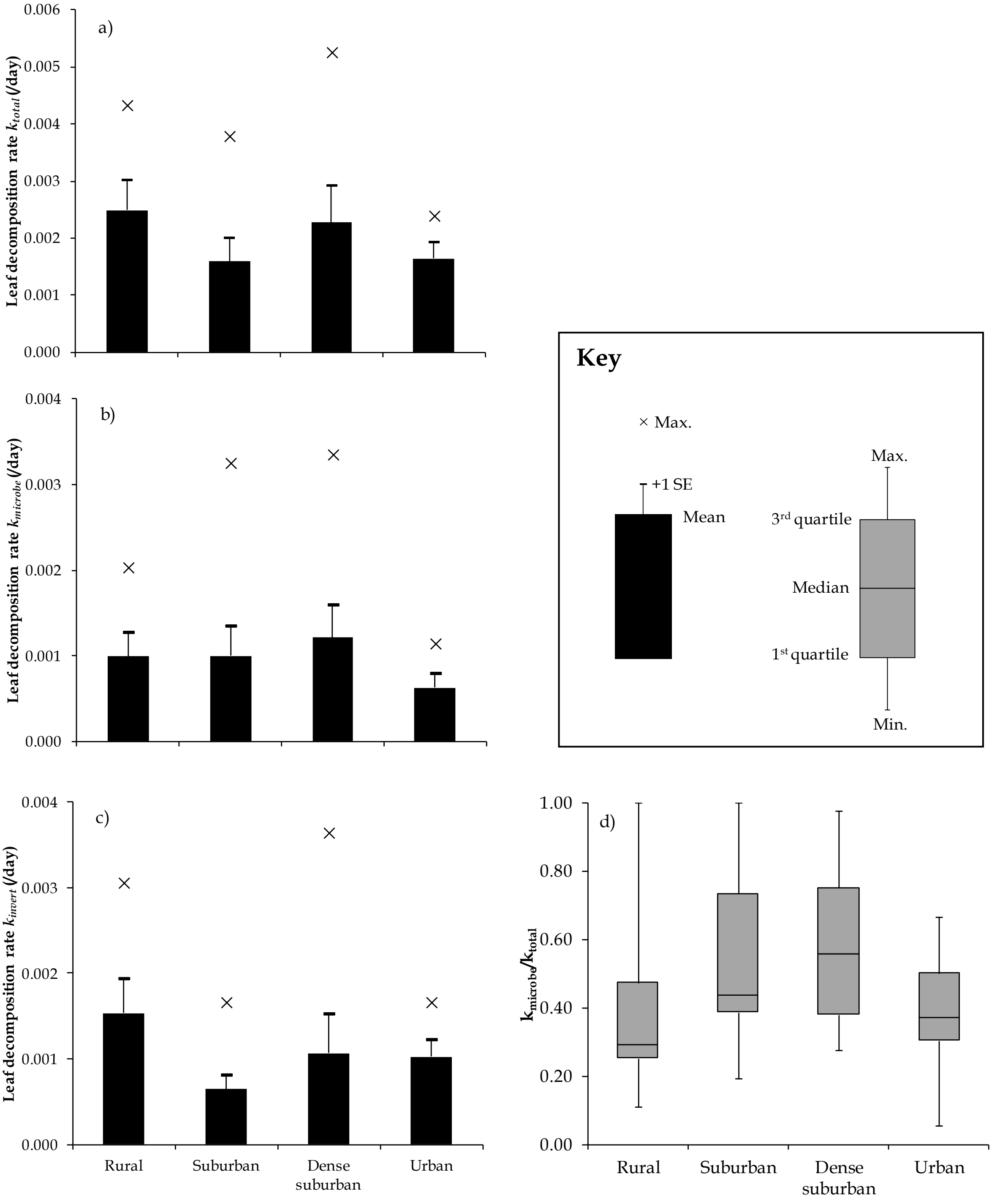


Figure 6.2.3. Comparison of mean (±1 SE) a) macroinvertebrate abundance, and b) density /g AFDM$_r$ and c) shredder abundance, and d) density /g AFDM$_r$. Different letters indicate significant differences amongst land-use categories (*post-hoc*, Tukey-Kramer)

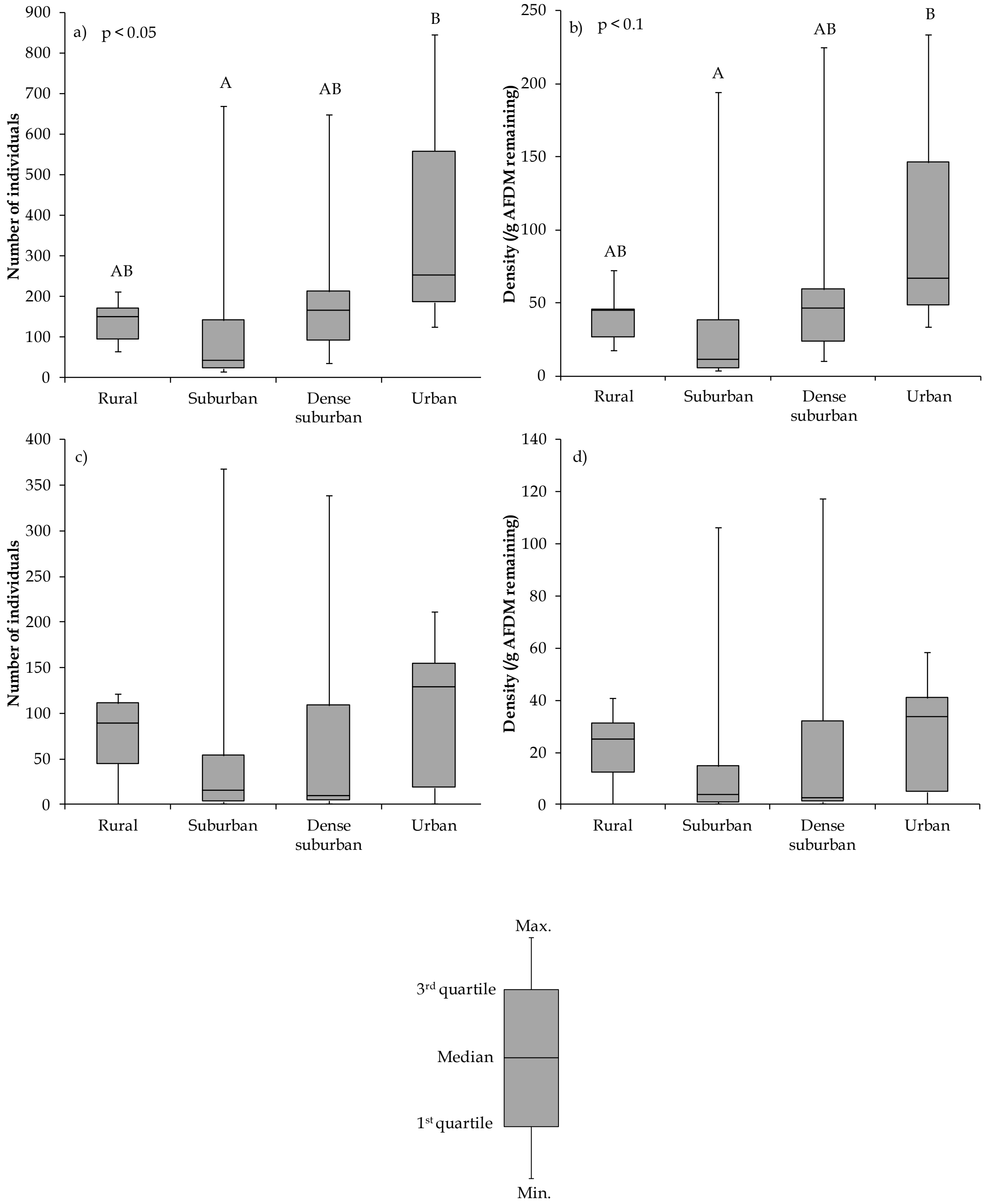


Table 6.3. Presence or absence of non-native species within land-use categories. Percentage of sites within each category is given in parenthesis.

Non-native species	1, 'rural'	2, 'suburban'	3, 'dense suburban'	4, 'urban'
Dugesia tigrina	0 (0)	1 (12.5)	0 (0)	0 (0)
Crangonyx pseudogracilis	5 (83.3)	6 (75.0)	4 (57.1)	1 (20.0)
Potamopyrgus antipodarum	3 (50.0)	1 (12.5)	1 (14.3)	1 (20.0)

For urban ponds, the taxon richness of shredders showed a strong positive relationship to k_{total} and weaker, but still significant, to k_{invert} (Figure 6.2.8a & c) within individual leaf packs and macroinvertebrate community taxon richness captured within the leaf packs was strongly correlated to leaf breakdown rates (Figure 6.2.8b).

Figure 6.2.4. Comparison of mean (±1 SE) shredder abundance (# individuals within leaf packs) for each urban land-use category (taxa occurring at 2 or more sites)

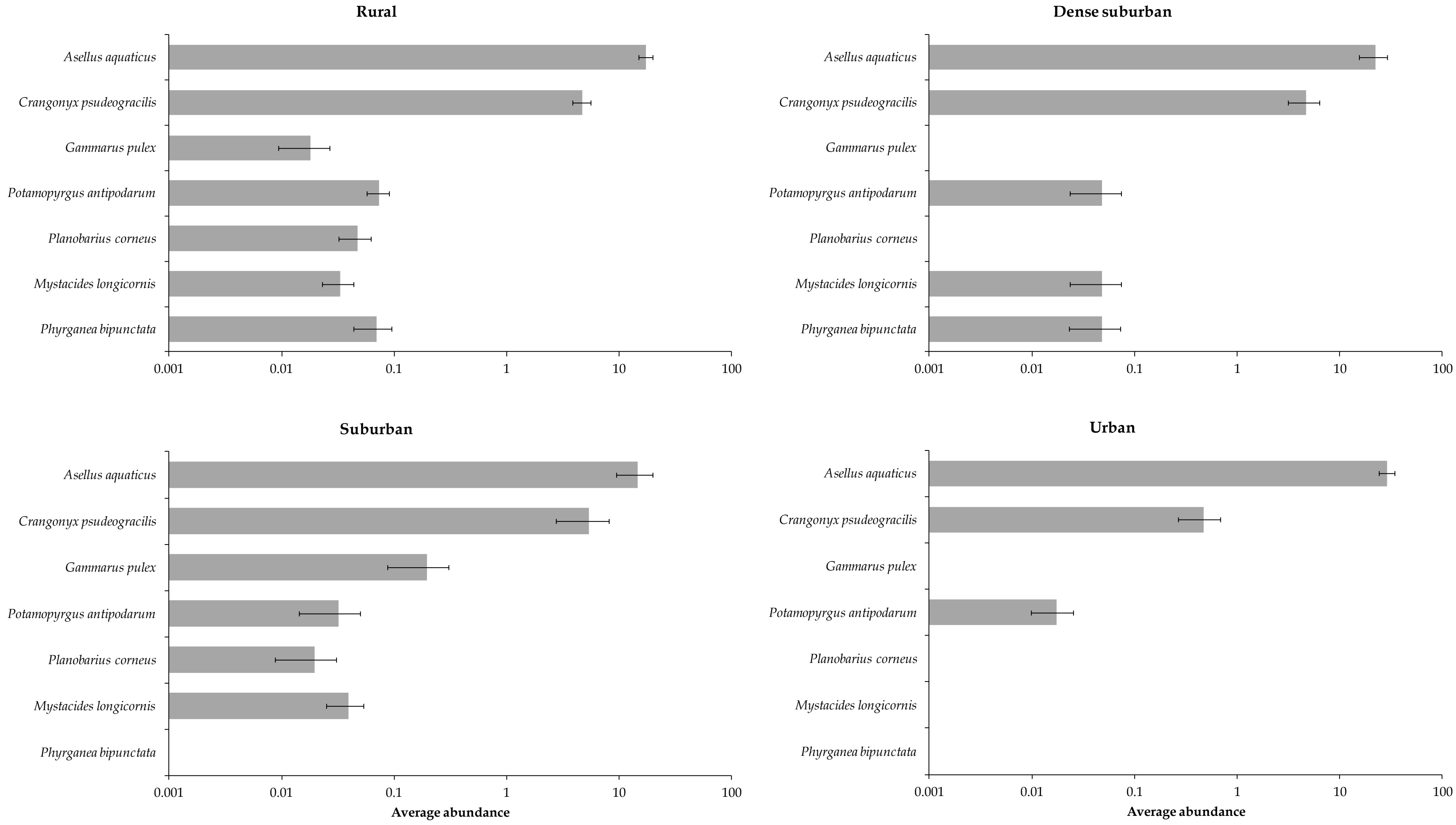


Figure 6.2.5: Redundancy analysis on functional group relative abundances against leaf breakdown rates. Circles represent ponds.

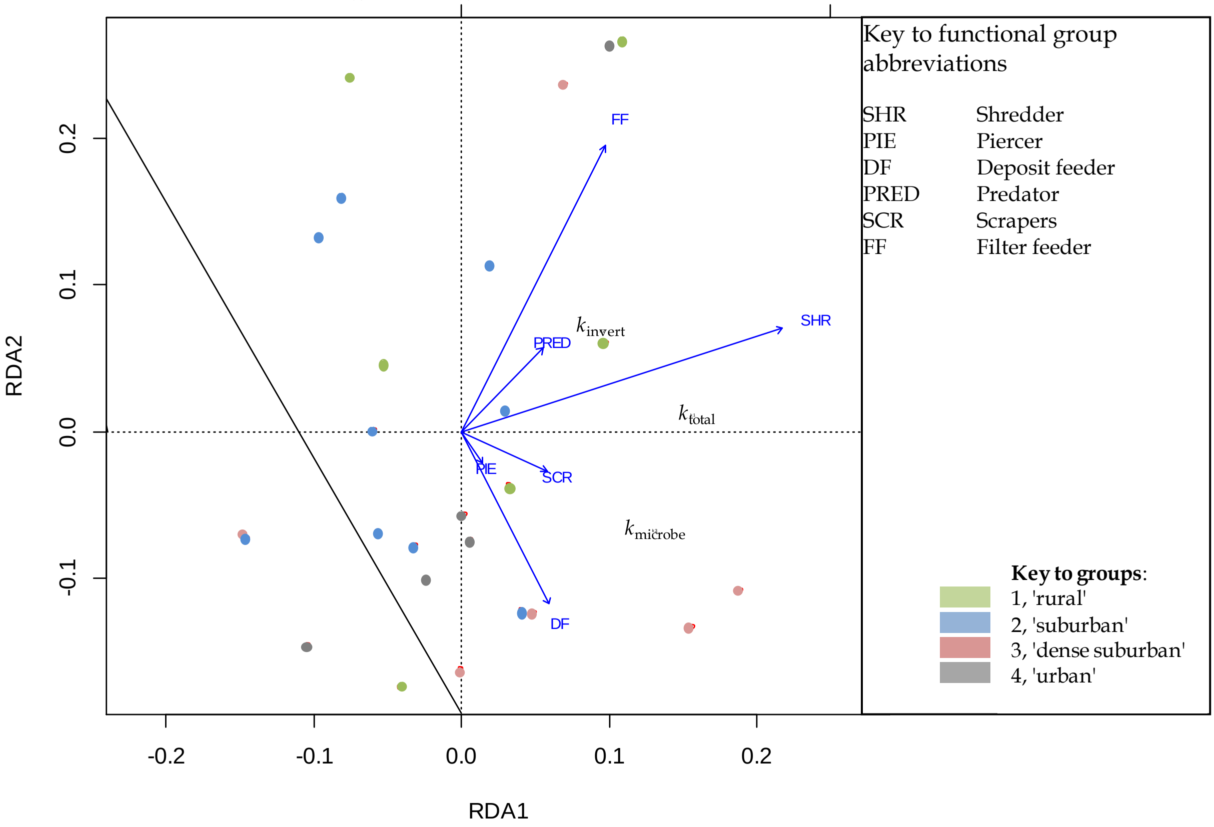


Figure 6.2.6: Redundancy analysis on shredder species relative abundance against leaf breakdown rates (taxa occurring at 2 or more sites). Circles represent ponds.

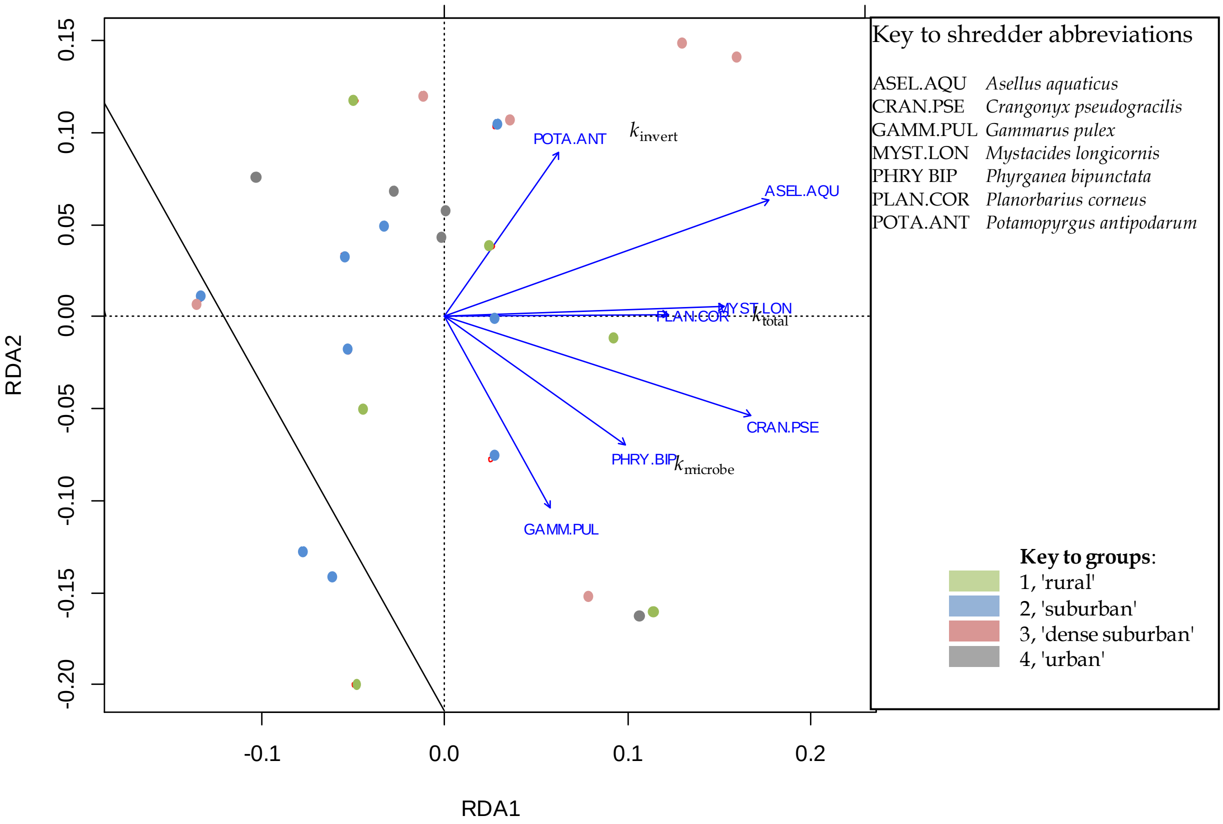


Figure 6.2.7. Relationship between leaf decomposition rates and a & c) shredder abundance (# individuals in leaf packs), b & d) shredder density (/g AFDM$_r$) in individual leaf packs (n = 59)

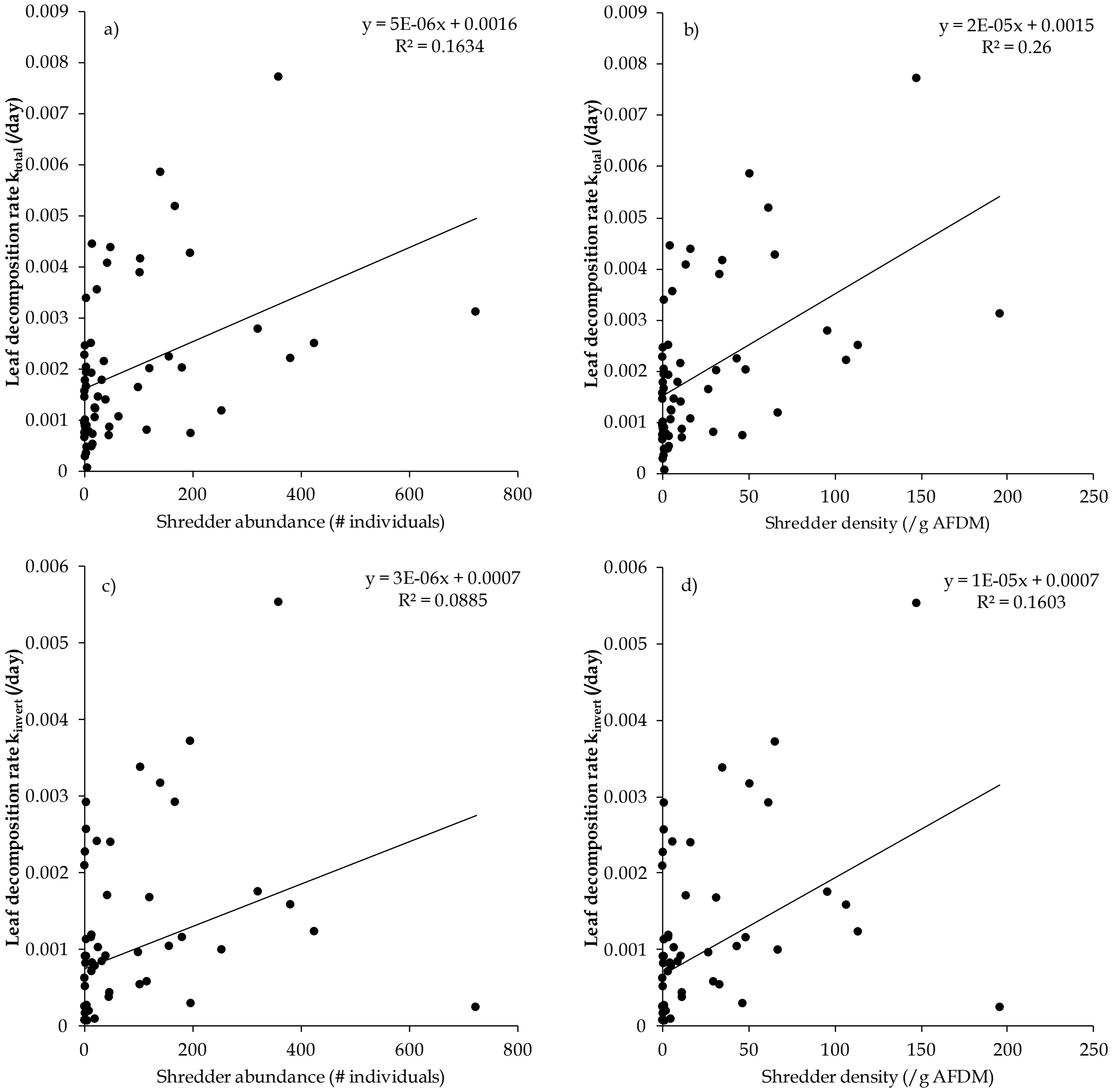
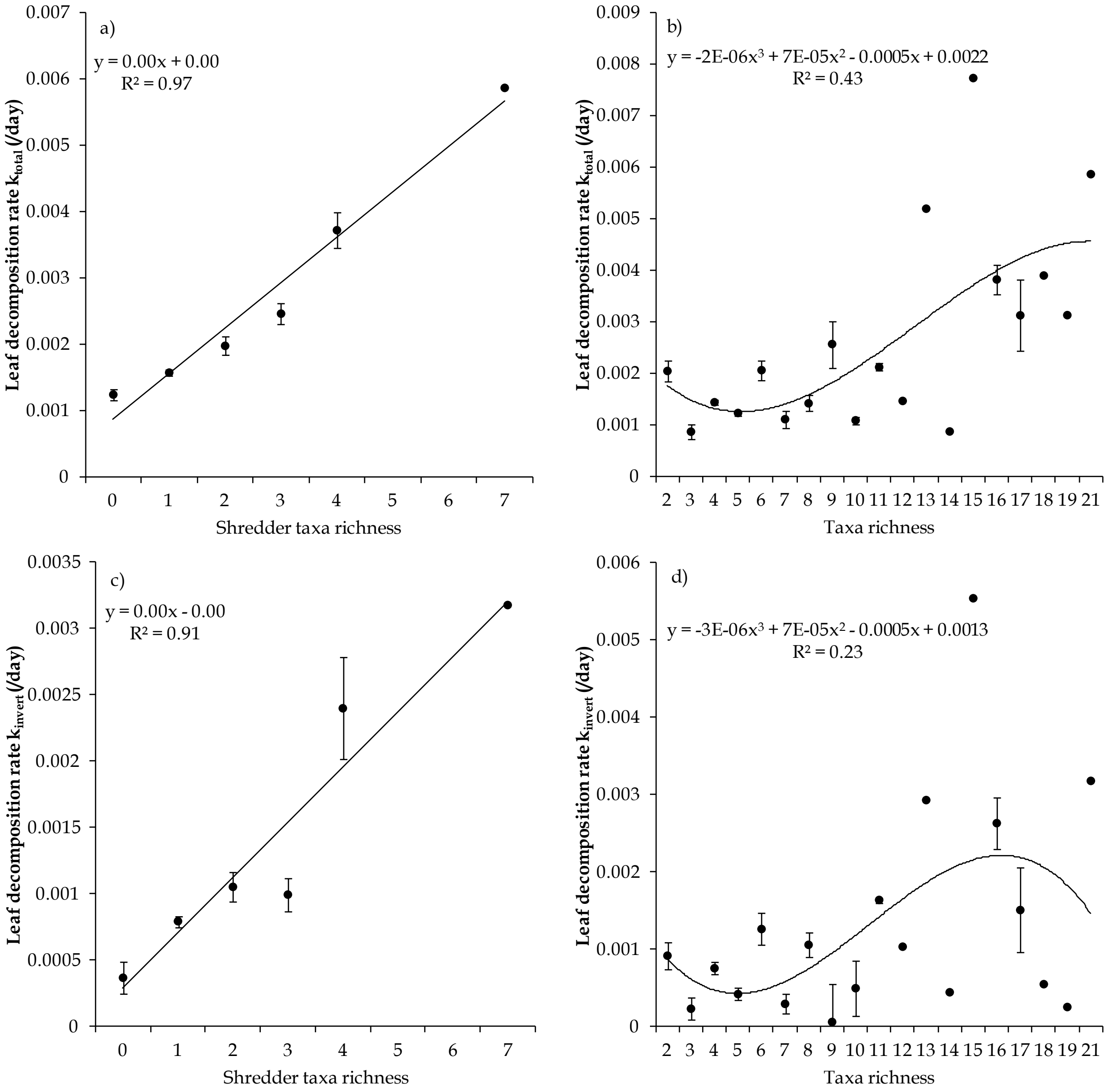


Figure 6.2.8. Relationship between leaf decomposition rates (± 1SE) and a & c) shredder taxa richness, b & d) taxa richness in individual leaf packs (n = 59)



6.2.3. The influence of local environmental factors on leaf breakdown rates

6.2.3.1. Local physical

The first two significant axes of an RDA (ANOVA, $P < 0.05$) (Figure 6.2.9) of local physical factors against leaf breakdown rates explained 49.3% and 16.0% of variance respectively. The first axis was most strongly correlated with surface area (Pearson's correlation: $r = 0.92$, $P < 0.01$), the second axis did not provide a clear gradient, but was most correlated to the presence of stream water inputs (Pearson's correlation: $r = 0.46$, $P < 0.05$).

Between the pond groups, surface area did not differ significantly (ANOVA, $P > 0.05$). Only the percentage of tree cover (i.e. shading) and the proportion of pond edge constructed from concrete statistically differed, where there was more tree cover in dense suburban sites than in rural (ANOVA, $P < 0.05$) and more concrete edge in urban than in rural (ANOVA, $P < 0.05$) (Table 6.4).

Significant positive correlations were observed between pond surface area and dissolved oxygen (Pearson's correlation: 0.61, $P < 0.01$) and negatively with tree cover (Pearson's correlation: -0.59, $P < 0.01$) and NH_4 (Pearson's correlation: -0.46, $P < 0.05$) (Appendix 26).

6.2.3.2. Water quality

The first and only significant axis of an RDA (ANOVA, $P < 0.05$) (Figure 6.2.6) of water quality against leaf breakdown rates explained 30.3% of variance and was strongly correlated to dissolved oxygen (Pearson's correlation: $r = -0.72$, $P < 0.01$).

Dissolved oxygen levels decreased from rural to urban groups and were statistically different between rural and both dense suburban and urban groups (ANOVA, $P < 0.05$) (Table 6.5). The rural group recorded the highest mean dissolved oxygen levels at any one site (118.5%, site 12) and dense suburban the lowest (10.9%, site 13). Conductivity and SO_4 concentrations were statistically significantly lower, and NH_4 and PO_4 concentrations higher in dense suburban sites than rural (ANOVA, $P < 0.05$). Mg concentration typically decreased across the groups from rural to urban, the difference between which was significantly different (ANOVA, $P < 0.05$).

Figure 6.2.9. Redundancy analysis on local physical variables against leaf breakdown rates. Circles represent ponds.

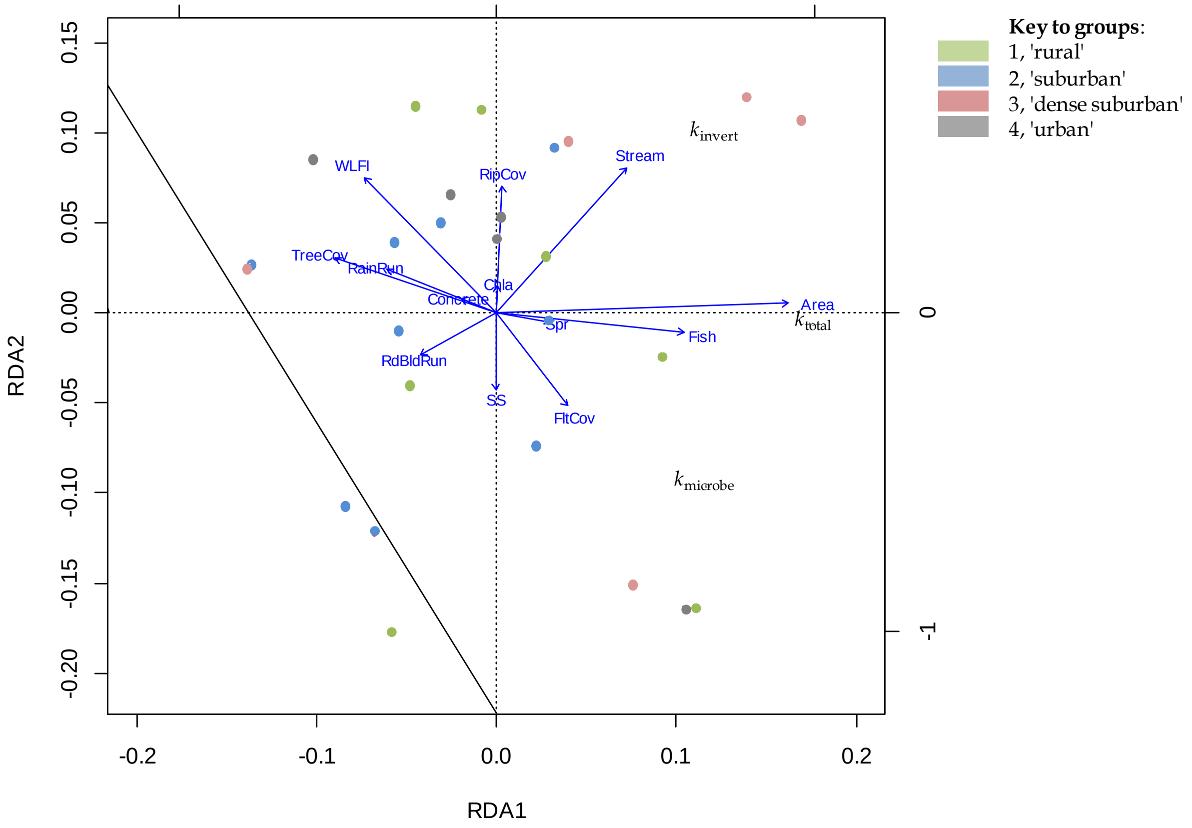


Table 6.4. Mean (min and max) values of local physical variables. Bold denotes variables for which significant differences were found between the groups, lettering denotes between which groups the differences occur (ANOVA, *post-hoc* Tukey-Kramer, $P < 0.05$)

	Unit	1, 'rural'	2, 'suburban'	3, 'Dense suburban'	4, 'Urban'
Area	m^2	3648 (1123-5992)	3536 (299-14923)	3122 (408-12078)	2291 (1010-5499)
Open	%	0.71 (0.49-0.9)	0.47 (0-0.96)	0.31 (0.07-0.76)	0.54 (0.32-0.75)
Concrete	**%**	**0 (0-0)^{a}**	**0.22 (0-1)^{ab}**	**0.11 (0-0.38)^{ab}**	**0.32 (0.19-0.58)^{b}**
TreeCov	**%**	**0.06 (0-0.17)^{a}**	**0.41 (0-1)^{ab}**	**0.51 (0.09-0.93)^{b}**	**0.32 (0.2-0.42)^{ab}**
RipCov	%	0.17 (0.02-0.37)	0.08 (0-0.32)	0.1 (0-0.37)	0.13 (0.01-0.41)
FltCov	%	0.06 (0.01-0.18)	0.04 (0-0.16)	0.08 (0-0.47)	0.01 (0-0.01)
WLFI	-	110.2 (18-354)	684 (12-4698)	42.9 (4-119)	735.7 (6-3558)
SS	mg/l	11.1 (4.7-25.8)	18.7 (5.8-53.9)	11.7 (2.8-24.1)	29.8 (5.3-64.9)
Chl*a*	µg/l	14.4 (0.3-44.4)	14 (0.5-44.5)	70.8 (8-302.6)	100 (3.4-176.7)
Spring	Binary	0.67 (0-1)	0.25 (0-1)	0.57 (0-1)	0.6 (0-1)
RainRun	Binary	0.33 (0-1)	0.75 (0-1)	0.43 (0-1)	0.8 (0-1)
RdBldRun	Binary	0.17 (0-1)	0.25 (0-1)	0.43 (0-1)	0.2 (0-1)
Stream	Binary	0.17 (0-1)	0 (0-0)	0.14 (0-1)	0.4 (0-1)
Fish	Binary	0.67 (0-1)	0.13 (0-1)	0.43 (0-1)	0.6 (0-1)

Figure 6.2.10. Redundancy analysis on water quality variables against leaf breakdown rates. Circles represent ponds.

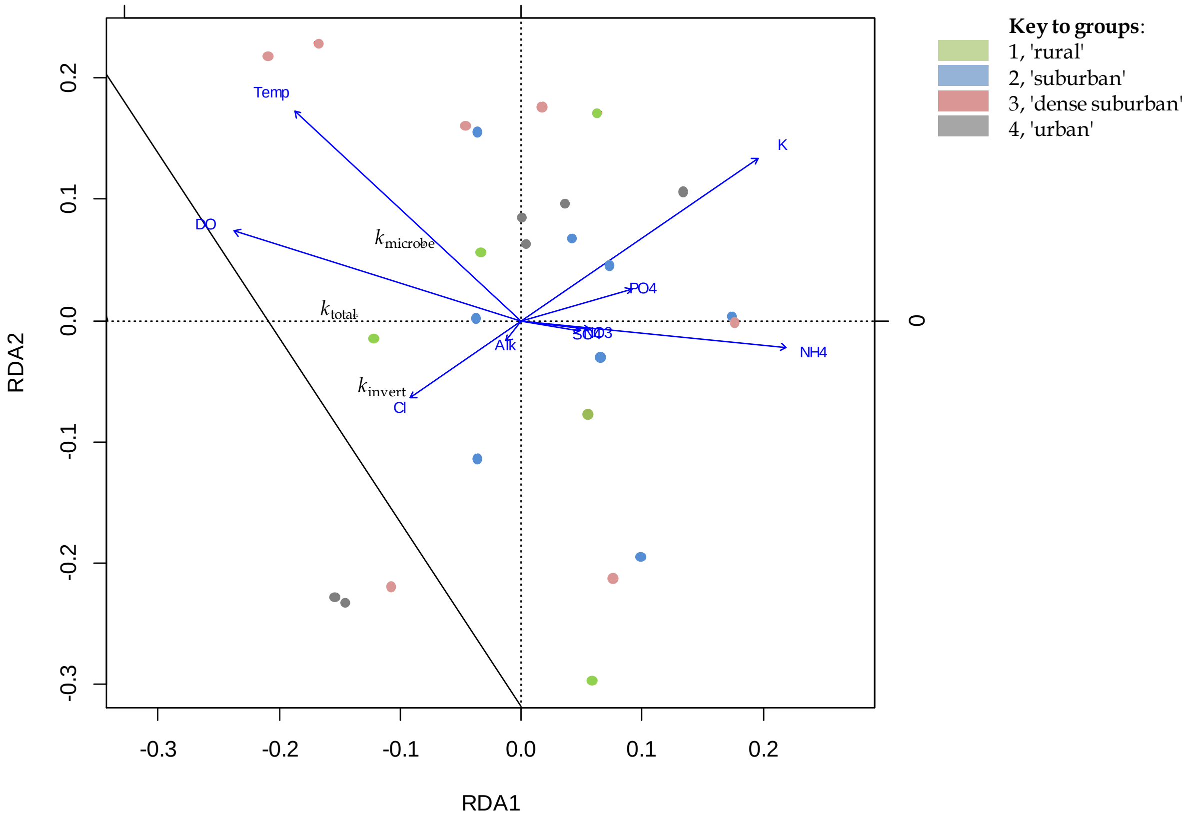


Table 6.5. Mean (min and max) values of water quality variables. Bold denotes variables for which significant differences were found between the groups, lettering denotes between which groups the differences occur (ANOVA, *post-hoc* Tukey-Kramer, $P < 0.05$)

	Unit	1, 'rural'	2, 'suburban'	3, 'Dense suburban'	4, 'Urban'
pH	-	7.98 (7.6-8.5)	7.84 (7.3-8.5)	7.44 (6.7-8.1)	7.6 (7.2-8.5)
EC	**µS/cm**	**653.7 (392-869)^{a}**	**654.1 (352-1445)^{ab}**	**364.4 (290-475)^{b}**	**455.7 (249-834)^{ab}**
°C	°C	18.8 (15.8-21.4)	17.4 (15-19.7)	17.3 (15.8-18.7)	18.1 (17.2-18.5)
DO%	**%**	**79.7 (49-119)^{a}**	**48.4 (21-98)^{ab}**	**28.1 (11-55)^{b}**	**34.3 (14-93)^{b}**
$CaCO_3$	mg/l	156.8 (76-236)	126.1 (72-168)	116.1 (48-210)	119.2 (94-142)
Na	mg/l	45.6 (9.7-82.7)	29.5 (9.6-62.3)	15.7 (7.2-22.6)	36.5 (12.1-114)
NH_4	**mg/l**	**0.12 (0-0.3)^{a}**	**0.8 (0-2.6)^{ab}**	**1.93 (0.1-4.1)^{b}**	**1.18 (0.5-2.1)^{ab}**
K	mg/l	5.9 (2.9-9.1)	6.1 (2-15.4)	6 (1.3-12.8)	5.9 (2.2-11)
Mg	**mg/l**	**15 (6.6-26.3)^{a}**	**9.5 (3.9-14.5)^{ab}**	**7.5 (3.6-13.6)^{ab}**	**4.8 (3.1-7.1)^{b}**
Ca	mg/l	58.5 (36.8-84.3)	85.7 (24.8-287.6)	35.2 (21.1-55.5)	38.9 (24.4-51.2)
Cl	mg/l	88.3 (11.7-189.1)	62.7 (14.3-126.5)	32.9 (15.3-52.1)	67.4 (17.5-200.1)
NO_3	mg/l	1.32 (0-4.2)	0.93 (0-6.7)	0.03 (0-0.2)	0.64 (0-2.5)
SO_4	**mg/l**	**109.3 (39-229)^{a}**	**161.8 (17-706)^{a}**	**25.6 (3-62)^{b}**	**23.2 (4-44)^{ab}**
PO_4	**mg/l**	**0.1 (0-0.3)^{a}**	**0.32 (0-0.8)^{ab}**	**1.1 (0.1-2.5)^{b}**	**0.66 (0.1-1.8)^{ab}**

* Zero values represent concentrations below detection limits (Appendix 3)

6.3. Discussion

Leaf breakdown rates did not suggest a reduction in ecosystem functioning across the rural-urban gradient as breakdown rates did not vary significantly, or with any clear trend among the land-use groups. Similarly, the relative contributions to breakdown by microbes or macroinvertebrates showed no clear pattern. Potential impacts to ecosystem functioning from urbanisation were masked by the presence of a habitat-area effect and changes in shredder community composition.

Surface dissolved oxygen levels were a function of surface area. Oxygen levels within the studied urban ponds frequently fall below the proposed annual values for attaining good ecological status under the Water Framework Directive (60% saturation, European Commission 2000; UK TAG 2008b) (Chapter 4). Leaf decomposition rates decrease under low oxygen conditions (Anderson and Sedell 1979; Piscart *et al.* 2011) and are likely as a result of lower abundances and activity of shredding macroinvertebrates (Bjelke 2005) and microbial decomposers (e.g. hyphomycetes, Medeiros *et al.* 2009). The influence of area on dissolved oxygen conditions stems from a greater exchange of organic matter between smaller ponds and their riparian zone than larger ponds (Declerck *et al.* 2006; Palik *et al.* 2006). Therefore, the same proportion of riparian shading will have a greater influence upon a small pond than a large one. Overhanging vegetation reduces light penetration to the sediment surface by direct blocking of sunlight and the accumulation of leaf litter. In doing so, growth of oxygenating macrophytes is inhibited (Gee *et al.* 1997; Jeffries 1998), as is the growth of phytoplankton (Scheffer 1999) and algae on the sediment surface, the establishment of which would provide an oxygenated layer above the sediment and reduce the potential for the release of nutrients from the sediment. Under shade, microbial activity further depletes oxygen levels, especially where leaf litter accumulation is high (Birch and McCaskie 1999).

Oxygen depletion and high PO_4 concentrations, which frequently exceeded the proposed annual values for attaining good ecological status under the Water Framework Directive (European Commission 2000; UK TAG 2008b) (Chapter 4) are indicative of eutrophic conditions. Thus, considerable stress is placed upon aquatic organisms and can lead to the increased dominance of a few tolerant taxa (Spieles and Mitsch 2003; Walsh *et al.* 2003). The increase in macroinvertebrate abundance from rural to urban pond groups was attributed to greater abundance of Chironomidae and the shredder *A. aquaticus*, which are highly tolerant to low oxygen levels, as indicated by Biological Monitoring Working Party (BMWP) scores of two and three respectively, which range up to 10 for the most sensitive taxa (Armitage *et al.* 1983). *A. aquaticus* is recognised as being omnivorous and is known to feed on microorganisms and living plant material (Gledhill *et al.* 1993) leading to its classification as a shredder, grazer and detritivore by some (Moog 2002). Consequently, *A. aquaticus* is a relatively inefficient shredder (Bjelke 2005). By contrast, two species of Crustacea, *G. pulex* and *C. pseudogr*acilis are obligate shredders, but decrease in number (*G. pulex* entirely) as urbanisation intensifies. Conversely, the increase in *A. aquaticus* abundance from rural to urban pond groups offsets numbers loss of other more efficient macroinvertebrate shredders. To further compound the issue, although *A. aquaticus* is often tolerant of conditions which exclude other crustaceans (Gledhill *et al.* 1993), laboratory tests have demonstrated that the species does not feed on leaf litter under low oxygen conditions (1 mg/l, Bjelke 2005), which are encountered in the BBC ponds. The cascading impact of habitat area upon physical and chemical characteristics and subsequent changes in the macroinvertebrate shredder community are consistent with Sangiorgio *et al.* (2010), who found similar impacts in a study of habitat area of freshwater springs on detritus processing.

As the pond groups became more urban, leaf decomposition was increasingly reliant upon fewer macroinvertebrate shredder taxa i.e. there was lower functional redundancy whereby multiple species perform similar roles in the community, and may therefore be substitutable with little impact on ecosystem processes (Lawton and Brown 1993). High levels of functional redundancy are important for ecosystem reliability (Naeem 1998). Therefore, ecosystem functioning would appear to be at higher risk in more urban ponds. However, an increased abundance of more tolerant shredder species (i.e. *A. aquaticus*), compensated for the loss of others such that leaf breakdown rates were not concurrent with a loss in shredder species number along the gradient, despite the observed correlation within the leaf packs (Figure 6.2.8). This is consistent with Downing and Leibold (2002), who found that composition can have an equal or more marked effect on ecosystem functioning than richness. Nonetheless, richness of the wider macroinvertebrate community within the leaf packs was a good indicator of breakdown rates. Biodiversity is frequently associated with ecosystems health and integrity (Loreau *et al.* 2001; Hooper *et al.* 2005) on the basis that functional redundancy is improved. Therefore, for urban ponds the safeguarding of biodiversity remains a prudent objective for the conservation and maintenance of ecological functioning (Loreau 2000; Balvanera *et al.* 2006).

Two non-native shredder species are present across the land-use categories, *C. pseudogracilis* and *P. antipodarum*. The crustacean, *C. pseudogracilis* is native to North America and was first detected by England in 1936 (Crawford 1937). It is now found throughout Europe, even in relatively isolated sites, which is attributed to intentional or unintentional introduction by anglers (Holland 1976). The gastropod *P. antipodarum* (Jenkin's spire shell), was first recorded in the River Thames in 1889 (Smith 1889) and originates from Australasia

(Ponder 1988). The increased abundance of exotic species in urban areas is well recognised (McKinney 2006). Several factors related to the disturbance of ecosystems from urbanisation can contribute to the success of non-native species such as more favourable habitat (Kowarik 1990), anthropogenic introduction (Ruesink *et al.* 2005), vacant niches resulting from reduced native species richness (Elton 1958), disturbance impacts upon the natural selection regime (Byers 2002) and low habitat heterogeneity (Vermonden *et al.* 2010). Several similarities are observed between this study and a study on the invasibility of urban drainage systems by Vermonden *et al.* (2010). First, native species were more abundant in ponds with lower nutrient concentrations, whilst in more turbid (indicated here as suspended solids, Table 6.5) and eutrophic ponds, non-native species dominanted. Finally, both studies share the non-native species *C. pseudogracilis* and *P. antipodarum*. These findings are consistent with those of Vermonden *et al.* (2010), who suggest that the function provided by non-native species may compensate for the loss of native species in highly disturbed systems.

Finally, leaves from plants within the family Fagaceae, such as oak (*Quercus spp.*) and beech (*Fagus spp.*) are classified as slow degraders, $k < 0.005$ (Petersen and Cummins 1974). This study agrees with the suggestion by Suberkropp and Chauvet (1995) that the decomposition rate of a given leaf species can vary greatly due to the influence of environmental factors such as water chemistry. The average k_{total} here (0.002/d) is slow by comparison to the Petersen and Cummins (1974) classifications and may immediately represent the impoverished condition within the ponds studied. However, few studies have assessed leaf litter breakdown rates within lentic environments and the lack of flow, which has been suggested as an influential factor in leaf decomposition experiments (Hodkinson 1975a; Benfield and Webster 1985; Paul *et al.* 2006), may be an obvious differential between breakdown rates in flowing and still water systems. In lotic conditions small leaf fragments will be consistently washed out that are created by shredders or broken off by current and/or abrasives after being softened following microbial activity (Webster and Benfield 1986). In addition, the rate of sedimentation in rivers will be lower, which might otherwise hinder colonisation of the leaf material by microbes or macroinvertebrates (Webster and Benfield 1986). The breakdown rates, however, remain broadly consistent with those reported elsewhere (Table 6.6) and to the average rates reported in a review of vascular plant breakdown by Webster and Benfield (1986), which included both lotic and lentic studies. Equivalent k_{total} rates were typically higher in riverine studies, although in the same order of magnitude (Lecerf *et al.* 2007; Sanpera-Calbet *et al.* 2009; Schindler and Gessner 2009; Swan *et al.* 2009), with the exception of acidified mountain streams (Dangles and Chauvet 2003).

Table 6.6. Breakdown rates (k) for leaf species of the family Fagaceae sourced from the scientific literature compared to this study

Species	System	Total length of study (days)	Breakdown rate (k)	Mesh size	Reference
F. sylvatica	Mountain streams (ph 6.4 – 7.1), France	154	≥0.00175	5mm	Dangles and Chauvet 2003
F. sylvatica	Mountain streams (ph 4.7 – 4.9), France		≤0.00100	“	
Q. rubra	Forested stream, Germany	55	0.0052	10mm	Schindler and Gessner 2009
			0.0031	0.5mm	
F. sylvatica	Forested stream, Germany	55	0.0037	10mm	
			0.0007	0.5mm	
F. sylvatica	Forested streams: Massif Central, France	31	16.3% Mass loss	10mm	Lecerf *et al.* 2007
	Carpathians, Romania	81	11.5% “	“	
Q. rubra	Forested streams: Massif Central, France	31	25.0% “	“	
	Carpathians, Romania	81	14.8% “	“	
Q. robur	Forested streams: Massif Central, France	31	41.6% “	“	
	Carpathians, Romania	81	22.9% “	“	
F. grandiflora	Spring-fed headwater tributary of	28	~0.005[1,2]	7 x 11mm	Swan *et al.* 2009
Q. prinus	Patapsco River, USA		~0.009[1,2]	“	
F. sylvatica	Forested stream, France	81	~0.0065[1,2]	10mm	Sanpera-Calbet *et al.* 2009
F. sylvatica	Temporary pond microcosms, Germany	70	16.0% Mass loss	12 x 12mm (over containers)	Schadler *et al.* 2005
Q. robur			~27.0%[1] “		
Fagaceaea	Literature review, lentic and lotic systems	NA	~0.00251	Various	Webster and Benfield 1986
F. sylvatica	Urban ponds	80	0.0020	5mm	This study
			0.0010	0.25mm	
			13.8% Mass loss	5mm	
			7.4% “	0.25mm	

Breakdown rates (k) represent the study mean unless otherwise stated

[1] by graphical interpretation, [2] when in a single species leaf litter mix

6.4. Conclusion

The results of the study imply that ecosystem functioning in urban ponds is maintained in larger ponds and could be improved by the removal of overhanging vegetation in smaller ponds. Improved light conditions will encourage aquatic vegetation growth, thereby reducing nutrient concentrations and increasing dissolved oxygen levels such that the abundance, diversity and efficiency of macroinvertebrate shredders and microbial decomposers increase. In some sites, more drastic management practices may be required, such as dredging, in order to reduce the ongoing threat of internal nutrient loading, which may recast sites to a more turbid state.

The effect of urbanisation on ecosystem functioning was confounded by the effect of habitat size and its physicochemical correlates such as shading and dissolved oxygen and changes in the composition of macroinvertebrate shredders. Decreasing dissolved oxygen levels from rural to urban pond groups, despite being the only significant water quality parameter, were not reflected in leaf breakdown rates across the groups. This suggests that the accumulated impacts of other correlates to habitat area and changes within the shredder community outweighed the influence of dissolved oxygen conditions. The influence of increased nutrient conditions across the urbanisation gradient, which has been noted in other freshwater rural-urban gradient studies (Gulis and Suberkropp 2003; Pascoal *et al.* 2003; Pascoal *et al.* 2005; Goncalves *et al.* 2006) was also unclear. Sampling at a greater resolution may have picked up on water quality that is likely to fluctuate widely within pond environments (Biggs *et al.* 2005). Similarly, 25% of leaf packs were not recovered and removed by members of the public, anglers or lost due to the action of wildfowl. In future, a different tactic for the installation of such an experiment in the public realm as well as a larger initial sample outlay would greatly improve the resolution of information gained.

Urbanisation did, however, appear to impact upon functional redundancy within the shredder functional feeding group. In the most urban sites leaf decomposition was increasingly reliant on a generalist species *A. aquaticus*, and lower abundances of a non-native species *C. pseudogracilis*. There is a suggestion that the presence of a non-native species has exploited a functional niche left open by the loss of native shredder species. Shredder taxon richness and community taxon richness was positively related to leaf decomposition rates in the individual leaf packs. Consequently, taxon richness remains a useful indicator of ecosystem functioning in these systems and should remain a conservation aim for pond managers. Macroinvertebrate richness, abundance and density all had a weaker relationship with macroinvertebrate only breakdown rate (k_{invert}), which may indicate at the reciprocal relationship between microorganisms and macroinvertebrates. Further investigation into microbial abundance, density and diversity within both the coarse and fine mesh leaf packs would contribute to a better understanding of this relationship.

The structure of beech leaves makes them less amenable to quick breakdown compared with other native tree leaf species (Sanpera-Calbet *et al.* 2009). Selection of a more labile leaf species, such as alder (*Alnus glutinosa*) may have provided a stronger indicator of the processes that took place during this study.

7

Conclusions

Chapter 7: Conclusions

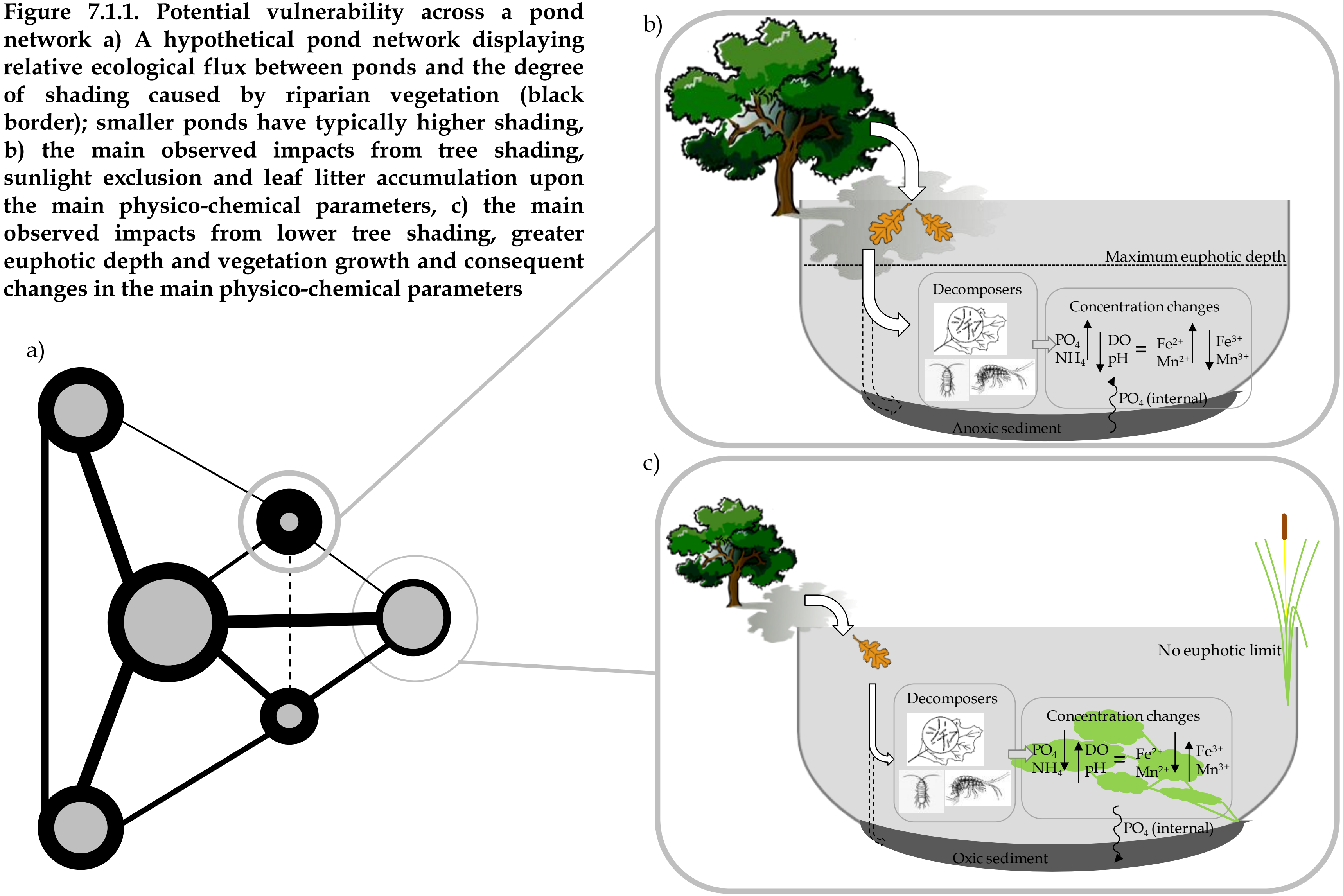
This research identified the major threats to the urban pond network and ecosystem. It achieved this by analysing local and spatial scale processes with respect to water quality, biodiversity and ecosystem functioning. Understanding how these ecosystems are being placed under increased pressure by habitat loss and urbanisation (and its incumbent physical effects) highlights possibilities to improve ecological resilience within pond networks which are applicable to future pond conservation strategies within urban landscapes and beyond.

Ponds have been lost from the Birmingham and Black Country (BBC) landscape at an alarming rate (Chapter 3). Pond loss was a consequence of urban expansion and agricultural intensification, which have left a diffuse habitat network comprising of larger remnant ponds. Water quality in those remaining ponds is varied, but is often degraded, being high in nutrients and low in oxygen, which is indicative of eutrophic conditions (Chapter 4). Nonetheless, provided an urban pond has good habitat complexity and is well connected to neighbouring ponds, they are able to support diverse macroinvertebrate communities and rare species of high conservation value (Chapter 5). Whilst some shading is beneficial, increased habitat complexity is likely to follow an improvement in light conditions brought about by the removal of overhanging vegetation, thereby stimulating aquatic vegetation growth. Leaf decomposition rates were contingent on shredder community composition and the combined effects of habitat area (and therefore shading) and lower dissolved oxygen levels rather than the degree of urbanisation (Chapter 6).

7.1. Urban ponds are vulnerable systems

The vulnerability of urban pond ecosystems was evident on several levels. Significant habitat loss between *ca*1904 and 2009 has reduced spatial resilience in the pond network and network analyses indicate that ecological flow across the contemporary pond network is increasingly reliant on fewer pond sites. The key threats to those ponds present in the contemporary network are highlighted in Figure 7.1.1. These environmental stressors compromise the capacity of the pond network to support diverse metacommunities.

Figure 7.1.1. Potential vulnerability across a pond network a) A hypothetical pond network displaying relative ecological flux between ponds and the degree of shading caused by riparian vegetation (black border); smaller ponds have typically higher shading, b) the main observed impacts from tree shading, sunlight exclusion and leaf litter accumulation upon the main physico-chemical parameters, c) the main observed impacts from lower tree shading, greater euphotic depth and vegetation growth and consequent changes in the main physico-chemical parameters



Cascading effects of a high degree of shading are particularly acute within small sites which make up a considerable percentage of the contemporary pond stock (Figure 7.1.1). Vegetation encroachment is more notable in ponds that have persisted throughout the last ~105 years and is likely to represent natural successional processes. Associated with shading was reduced habitat complexity, reflected by a decline in macrophyte richness, which was an important feature of sites with high conservation value, lower dissolved oxygen levels and higher nutrient concentrations (PO_4). Furthermore, ecosystem functioning was improved in larger ponds by a combination of reduced riparian shading and improved oxygen conditions, and consequently, a richer diversity of macroinvertebrates within the leaf litter, which in smaller sites was likely to be anoxic.

The process outlined contributed to all of the 30 studied ponds failing to attain at least one accepted environmental standard. All but two sites failed to achieve PO_4 concentrations recommended for attaining ecological 'good' status in lakes under the WFD. Nonetheless, ponds are too small to qualify for regular monitoring and assessment and do not fit the lake model upon which the WFD requirements are made (Boix *et al.* 2012). Moreover, urban ponds are likely to be identified as artificial water bodies (AWBs) under the WFD due to the artificial nature of their creation, (UK TAG 2004); thereby they would be subject to a less stringent set of standards. This research therefore adds further backing for the establishment of a new pond typology in order to delineate more appropriate targets for pond habitats (Boix *et al.* 2012). The high conservation value of some sites may qualify them for protection as Priority Habitats (BRIG 2008) under the U.K. Biodiversity Action Plan (UK BAP). However, ponds should not be viewed in isolation (Jeffries 2012). Local sites rely on a healthy exchange of individuals from the regional species pool, and many pond-dwelling organisms exist in metapopulations (Briers and Warren 2000; Jeffries 2005). The conservation value of one site is contingent upon its connectivity to the wider pond network.

Sites with the highest conservation value were those that had rich macrophyte complexity, little shading, lower nutrient concentrations and were well connected to the wider pond network (up to 2km), although this was mediated by the intervening terrestrial matrix. When sampling of taxon richness was not restricted to potentially anoxic sediments (as per the leaf pack experiment), the effect of habitat area was much reduced, such that small sites were just as likely to support macroinvertebrate assemblages of high conservation value as large. Land-use had most effect upon both water quality and macroinvertebrate community composition within 100m from the pond margin. In terms of water quality this represents an area within which site managers should investigate sources of diffuse pollution. For example, the number of impermeable surfaces should be reduced to allow infiltration of surface water into bare soils, concomitantly, encouraging vegetation growth will not only increase interception of surface water runoff, but also provide opportunities for emerged aquatic insects to forage (e.g. odonates, Remsburg *et al.* 2008), rest and find refuge from predation (from birds, bats). Although different macroinvertebrate orders were sensitive to different landscape scales in line with their dispersal traits, land-use within 100m is likely to give a reasonable representation of the wider landscape and should be a priority for land-use management. The effect of mature vegetation within 100m was not clear and although related to decreased dissolved oxygen and increased PO_4 it was also strongly correlated to overhanging vegetation such that the impact of mature vegetation coverage may, in effect, be replicated in the analysis. Nonetheless, greater habitat heterogeneity within the ponds and in the surrounding land should be encouraged.

Poor dispersers, such as odonates (behaviourally constrained) were more sensitive to habitat availability within short range, whereas strong dispersers (e.g. Coleoptera) were more sensitive to habitat availability in the wider landscape. However, the latter facet is somewhat counterintuitive because, if habitat quality and quantity is sufficient, strong dispersers should show little preference to landscape scale. Consequently, this result suggests that strong dispersers are more sensitive to the wider landscape as they are required to explore longer distances in order to find suitable habitat. For Coleoptera, this seems plausible given their often specific habitat requirements, which may not be readily met by habitats within a heavily disturbed urban environment (Lundkvist *et al.* 2002). Taxa that often exhibit poor dispersal ability may be more vulnerable to increased distance between neighbouring sites in the contemporary landscape. Therefore, odonates and to a lesser extent, trichopteran richness may be improved by the provision of clusters of ponds in relatively close proximity. Consequently, the landscape extent managers need to consider may be dependent upon the target taxa.

Urbanisation appeared to lower functional redundancy as shredder taxon richness decreased along the rural-urban gradient. In the most urban sites leaf decomposition was mostly reliant upon greater abundances of a single inefficient but pollution tolerant shredder species *A. aquaticus*. Thus, ecosystem functioning was increasingly vulnerable due to urbanisation. The main determinants of leaf decomposition rates were habitat area and dissolved oxygen, which were again linked in the manner described (Figure 7.1.1). The number of taxa within the leaf pack i.e. benthic taxon richness, was a good overall indicator of ecosystem functioning. Sites with high benthic taxon richness were typically larger and more open which likely improved oxygen conditions in the benthos. Improved oxygen conditions are also likely to improve the efficiency of shredders and decomposing micro organisms (Bjelke 2005; Medeiros *et al.* 2009).

7.2. Urban ponds as candidates for freshwater conservation in urban areas

- Despite considerable historical pond loss, the rate of pond loss has decreased since *ca*1962, with nearly a third of those in the contemporary landscape created since that period. Most notably, several new ponds have been constructed in the most urban landscapes and have to some extent offset the loss of ponds from more rural areas. The development of these new ponds for aesthetic value alongside new developments is an example of how pond numbers can be increased in the heart of cities because of the perceived benefits to people from exposure to wildlife. In urban areas this presents a good opportunity to reverse the trend of pond density decrease due to urbanisation. Furthermore, the potential role garden ponds could play as stepping stones should not be understated. Although the productivity of individual garden ponds may be small, their cumulative effect may be considerable and emphasized if public pond and private (garden) ponds are managed in conjunction as sub networks. Management of pond clusters (Gledhill *et al.* 2008; Jeffries 2012) may be a more feasible objective than whole scale 'pondscape' management and help avoid managed ponds conforming to a predefined pond image (Wood *et al.* 2003). Moreover, a diverse regional species pool is dependent upon habitat heterogeneity within the pond network, such that the numbers of environmental niches proliferate.

- Although their small volume and surface area make ponds more vulnerable to external and internal factors, it also makes them a tractable management option such that small changes can make a big impact (Davies *et al.* 2008b). Local physical factors were statistically more influential than regional scale factors for water quality and macroinvertebrate community composition. Consequently, physical habitat management should take priority over landscape scale factors in order to improve biodiversity potential. The research suggested that provided local physical habitat is good, sites can be colonised by a highly diverse regional species pool. At several sites this is already apparent where ponds are regularly angled, such that tree cover is reduced in order to improve bankside access and vegetation is managed to provide refugia for juvenile fish and to improve accessibility for fishing. With this in mind the presence of a large proportion of contemporary ponds within public parks and golf courses suggests that informed management action, rather than a lack of management, is the necessary step. Conversely, sites outside of public spaces may be suffering from neglect, with mixed consequences for their ecology, and without management will eventually succumb to natural successional processes, with increases in tree cover and lower macrophyte diversity and ultimately lost to sedimentation.

- This research suggests site managers should focus upon land-use within 100m of the pond edge, although urban infrastructure can convolute the influence of land-use. Within this distance, investigations of diffuse, or point, sources of pollution should fall under relatively simple land ownership. Difficulties may arise in more urban areas where the problems are more acute and may require closer liaison with homeowners and drainage personnel. Nonetheless, by comparison to stream catchments, the management of pollution entering ponds should be much more feasible; with the exception of those with a prominent stream water source.

- Compared to urban streams and rivers, the urban pond network may be more resilient and able to recover from frequent temporary disturbances such as storm water run-off and pollution incidents. Entire lengths of urban rivers may be impacted by a single upstream disturbance. In contrast, a deterministic event in one hydrologically isolated pond will have a minimal impact upon others in the network, save for a temporary reduction in the number of potential immigrants. In the ponds studied, wide ranges in spatial and temporal water quality and seasonal peaks in several determinands suggested high stochastic variability and regular deterministic events in some sites. However, provided the pond network is well connected, pond-dwelling organisms, evolved to have generally good dispersal abilities (Bilton *et al.* 2001) can quickly recover impacted ecosystems i.e. strength in numbers. In summary, urban pond habitats appear to be in a constant state of flux with habitats being consistently gained and lost, largely through the action of man, and with a range of habitat qualities at any one time. However, provided that a sufficient proportion is in good ecological health at any one time, the regional species pool remains intact.

7.3. Research limitations

This research has contributed to the understanding of local and regional stressors to the urban pond environment and has identified key issues which should be addressed to improve their ecological integrity. It has also highlighted opportunities for freshwater conservation in urban areas and is the first to assess ecosystem functioning in urban ponds. However, limitations were observed within the research, which should be taken into consideration prior to any further urban pond studies and application of the research findings:

- Analysis of the historical pond network unavoidably used maps that were derived using different methodologies. As such, the accuracy and detail may not be strictly comparable between datasets, for example, the earliest County Series dataset did not consistently annotate aquatic features. Furthermore, pond network analyses were restricted to the Birmingham administrative boundary due to computational constraints. Ideally, pond networks should be examined according to more natural boundaries e.g. a river catchment, or delineated from the wider pondscape according to a predefined distance threshold. Nonetheless, the use of an administrative area likely complies with the remit of biodiversity officers. Similar analyses carried out by neighbouring administrative units would be to the benefit of both and encourage cross-boundary cohesion between administrative teams.

- Habitat quality in the pond network was assessed using remotely sensed NDVI and photogrammetry data. Consequently, they provide a snapshot from the time of their respective aerial flights. Field derived data for all sites across the contemporary network (341 sites) was infeasible; however, across the 30 studied ponds this may have provided some further insights into the habitat complexity in terms of vegetation coverages. As it is, only the uppermost layer of vegetation can occupy any one location i.e. there was no account of emergent vegetation cover if it is obscured by mature vegetation. However, a record of macrophyte richness is likely to serve as a good indicator of vegetation coverage due to species-area relationships i.e. greater coverage is likely to result in higher species diversity.

- A multi-species approach was used in order to assess for potential impacts to pond network connectivity. By using this conservative approach a sequence of distance thresholds were applied as a representation of dispersal abilities possessed within the community. However, aquatic insect dispersal abilities are notoriously difficult to assess, and it is known that a small proportion of aquatic insect populations can travel considerable distances further than the population mean. In addition, a resistance landscape was derived from a limited amount of literature on potential obstacles to dispersal. Although it was shown that a cost-informed metric performed better than the same metric using Euclidean distance, there is much room for improvement.

- Several further factors could influence the physical and chemical composition of urban ponds that were not considered here. Geology in BBC is varied; however, this was not included in any analyses. The rationale for its omission is that many of the urban ponds studied are hard engineered and lined, such that the influence of groundwater is minimal. However, it was acknowledged that several sites did receive groundwater such that the potential influence of geology cannot be excluded and requires further examination. Water sources overall were not self-evident, and local stakeholder knowledge was often piecemeal, including local council drainage plans. Drainage in urban areas is at best complex, and considerable resources would be required to provide a complete account of water sources. Consequently, the effectiveness of land-use management as recommended by this study is contingent upon knowledge of water sources.

- The presence of internal loading mechanisms was suggested to be influential on water quality, particularly where dissolved oxygen levels were low and nutrient concentrations high. Although this may be so, chemical fluxes between the sediment and water column were not assessed. As above, the effectiveness of management strategies based upon this research may be compromised by internal loading mechanisms from sediment contamination, which may have accumulated over many years and has the potential to reset sites to a turbid state long after supposed successful management action. Nonetheless, improvements in oxygen conditions in particular, will help to lock contaminants within the pond sediments.

- Macroinvertebrate assemblages were sampled in a single year (2009). Changes in physical and chemical characteristics within an urban pond are likely to change year to year and result in shifts in macroinvertebrate community composition. The National Pond Survey methodology was labour intensive and time-consuming. Alternative, more rapid, assessment methods e.g. PLOCH (Oertli *et al.* 2005) or the use of indicator taxa (Briers and Biggs 2003) could have been employed in order to sample changes in macroinvertebrate assemblages between years and across more seasons and/or sites. Conversely, whilst rapid assessment techniques may have sufficed for biological assessment, they may not have provided the same amount of biodiversity detail (King and Richardson 2002).

- This study was the first to undertake a leaf decomposition experiment in an urban pond environment and provided some useful results. However, around a quarter of the initial leaf pack outlay was not recovered and four study sites were lost from the analysis completely due to interference from passers-by, wildfowl or site managers (despite close liaison). The use of a more labile leaf species may have provided greater resolution than beech leaf, which was a slow degrader and water quality was sampled on two occasions during the experiment. A higher resolution of water quality sampling may have captured a greater range in water quality than was observed. Similarly, although relevant, dissolved oxygen was measured at the water's surface, a measurement of oxygen conditions at the sediment-water interface would have been more appropriate and shed-light on the presence or absence of anoxia.

7.4. Suggestions for further research

This research is one of few that have ventured to consider a highly disturbed aquatic ecosystem. In doing so, more questions of applicability to urban and exurban pond systems have been highlighted.

- *Is there a need for a paradigm shift when considering urban pond networks?*
  Urban ponds are predominately man made, but the natural processes of pond creation are inhibited by land-use intensification. Nonetheless, whilst some ponds e.g. farmland ponds, are no longer required for their original purpose, new ponds are being created for their aesthetic or amenity purpose e.g. golf courses. Considerable biodiversity has been highlighted in urban ponds (Vermonden *et al.* 2009, this study), highway retention ponds (Le Viol *et al.* 2009), golf course ponds (Colding *et al.* 2009) and old industrial mill ponds (Wood and Barker 2000; Scher and Thiery 2005) amongst others (Gledhill *et al.* 2008; Sayer *et al.* 2008). Furthermore, garden ponds are largely unresearched and may play a pivotal role as stepping stones. Therefore, in our quest to understand aquatic insect dispersal in urban environments we may need to be more broad-minded as to the adaptability of aquatic organisms and take a more all-encompassing approach that includes a wide variety of aquatic habitats).

- *What is the effect of the habitat matrix on aquatic insect dispersal?*

Least-cost-path analyses, amongst others (McRae *et al.* 2008), are being increasingly used to aid decision-making for reserve networks (e.g. Verbeylen *et al.* 2003; Desrochers *et al.* 2011; Gurrutxaga *et al.* 2011). Despite the relevance of ponds to island-biogeography theory and the accepted view that many pond organisms exist in metacommunities, there is a considerable lack of knowledge regarding their dispersal abilities and how they are impacted, if at all, by different land-uses. Considering the well known risk to insects from urban areas e.g. light pollution (Horváth *et al.* 2009), we are still largely unaware of the real cost to aquatic insect populations. Recent stable isotope techniques may be one method to aid future research (Caudill 2003; Briers *et al.* 2004).

- *What is the comparative diversity of different aquatic systems in urban landscapes?*
  Rural ponds are known to contribute significantly to regional biodiversity (Williams *et al.* 2004), however, although this research has again highlighted that urban ponds can support highly diverse macroinvertebrate assemblages it is not clear how valuable they are in the context of other urban aquatic systems or against rural ponds. A comparative study of regional biodiversity and the contributions made by urban ponds, rural ponds, lakes, rivers and streams would be valuable in this regard. Any such study should also be set in the context of the national diversity of macroinvertebrates.

- *What life traits are required by macroinvertebrates to succeed in urban pond networks?*
  This research suggested a high degree of stochastic variability within urban ponds. They are dynamic systems within a dynamic network which has a continual turnover in the number and spatial configuration of pond sites. In order to survive in such a changeable environment the species they support may need a different set of survival traits than in a less disturbed, more stable environment.

- *What is the impact of non-native species in urban ponds?*
  Although it was beyond the study scope, three non-native macroinvertebrate species and ten non-native macrophyte species were recorded, although not all were invasive. *C. pseudogracilis* was the only obligate shredder that was tolerant of the conditions in the most urban sites and coexisted with native shredder species in less urban sites. Under certain circumstances, can non-native species compensate for the functional role, lost with the removal of a native species? Conversely, the site of

highest conservation value was threatened by *C. helmsii,* which if unchecked can choke native macrophyte species and lower macrophyte complexity; an important variable for macroinvertebrate richness.

8

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Appendices

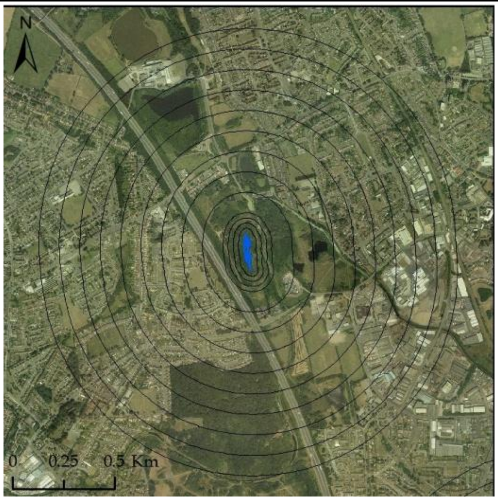
Appendix 1. Priority pond criteria *after* BRIG 2008

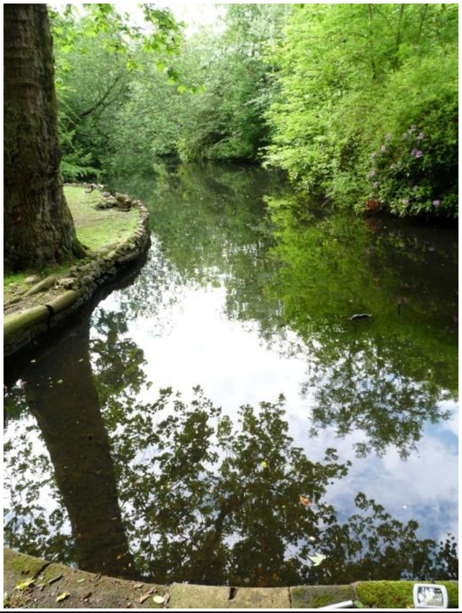
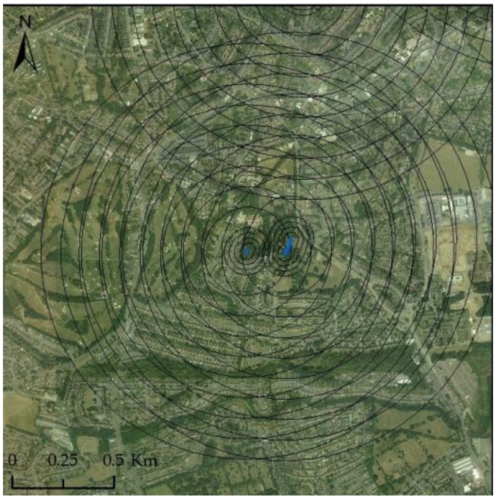
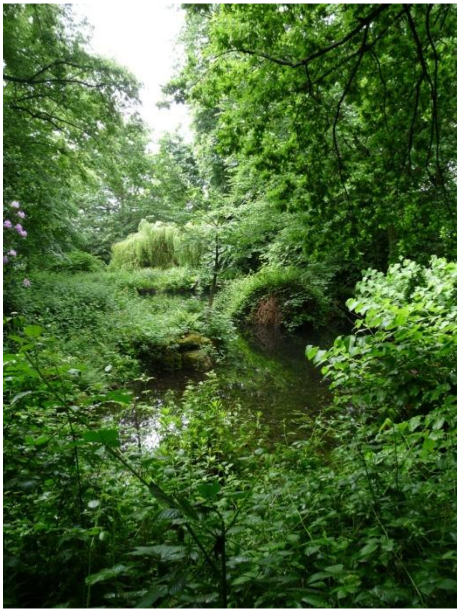
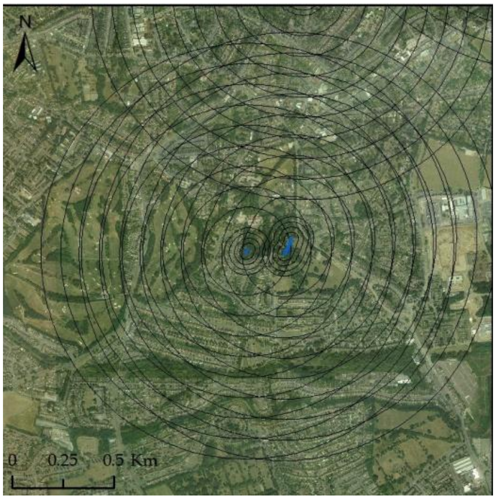
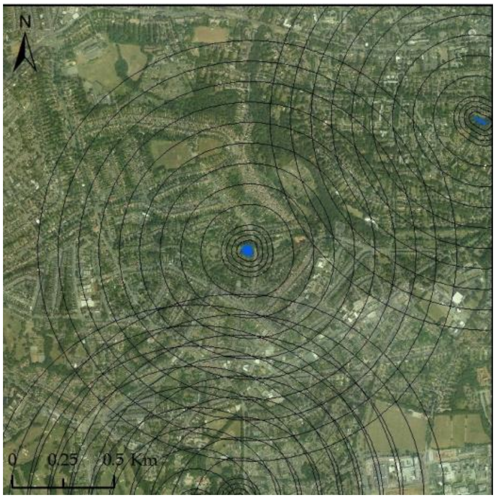
Ponds, for the purpose of UK BAP priority habitat classification, are defined as permanent and seasonal standing water bodies up to 2 ha in surface area, which meet one or more of the following criteria:

- Habitats of international importance: Ponds that meet criteria under Annex I of the Habitats Directive.
- Species of high conservation importance: Ponds supporting Red Data Book species, UK BAP species, species fully protected under the Wildlife and Countryside Act Schedule 5 and 8, Habitats Directive Annex II species, a Nationally Scarce wetland plant species, or three Nationally Scarce aquatic invertebrate species.
- Exceptional assemblages of key biotic groups: Ponds supporting exceptional populations or numbers of key species. Based on (i) criteria specified in guidelines for the selection of biological SSSIs (currently amphibians and dragonflies only), and (ii) exceptionally rich sites for plants or invertebrates (i.e. supporting ≥30 wetland plant species or ≥50 aquatic macroinvertebrate species).
- Ponds of high ecological quality: Ponds classified in the top PSYM category ("high") for ecological quality (i.e. having a PSYM score ≥75%).

  PSYM (the Predictive SYstem for Multimetrics) is a method for assessing the biological quality of still waters in England and Wales; plant species and / or invertebrate families are surveyed using a standard method; the PSYM model makes predictions for the site based on environmental data and using a minimally impaired pond dataset; comparison of the prediction and observed data gives a % score for ponds quality.
- Other important ponds: Individual ponds or groups of ponds with a limited geographic distribution recognised as important because of their age, rarity of type or landscape context e.g. pingos, duneslack ponds, machair ponds.

Appendix 2. Site summaries

Site	Site photo	Aerial image (with concentric rings)
1		
2		
3		

Site	Site photo	Aerial image (with concentric rings)
4		
5		
6		

Site	Site photo	Aerial image (with concentric rings)
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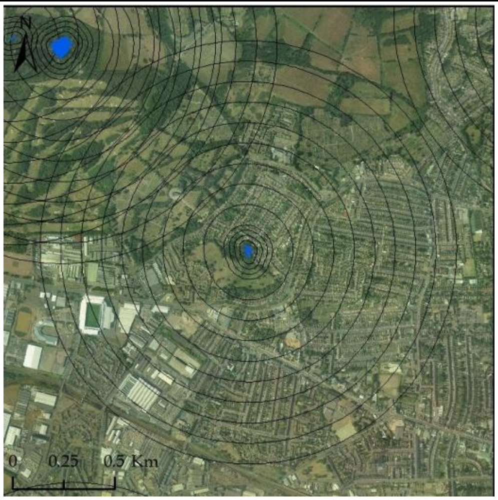
8		
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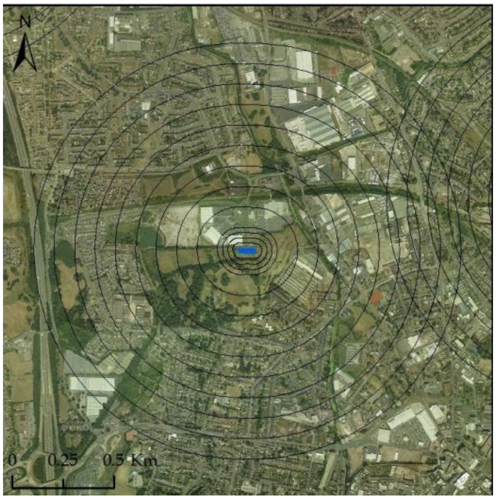
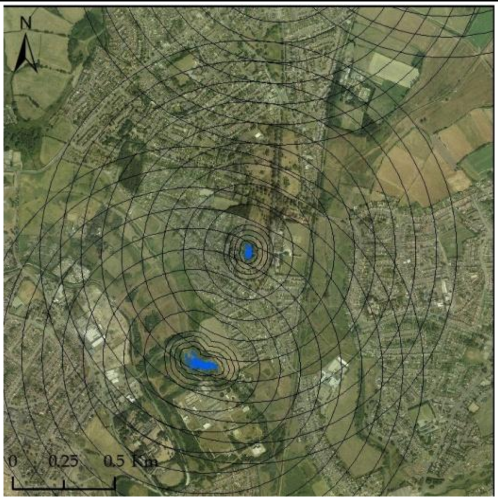
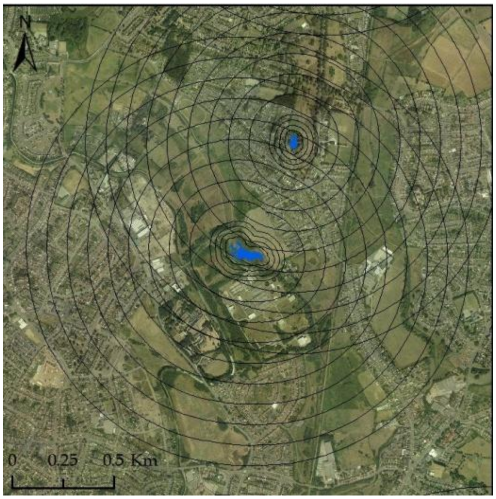
Site	Site photo	Aerial image (with concentric rings)
10		
11		
12		

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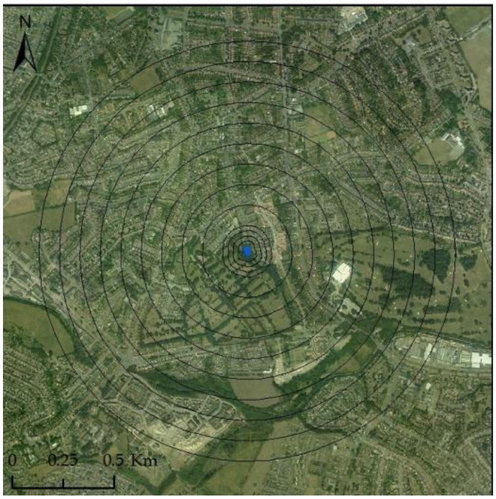
13		
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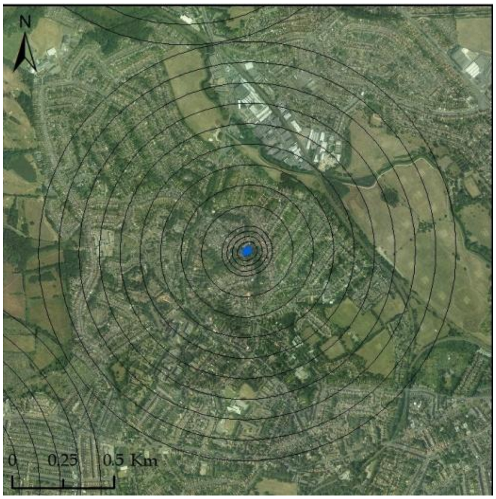
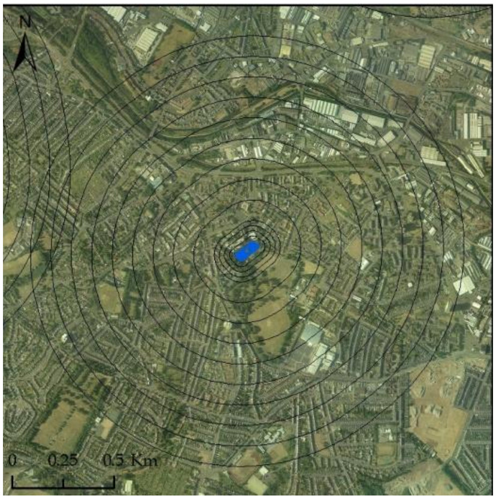
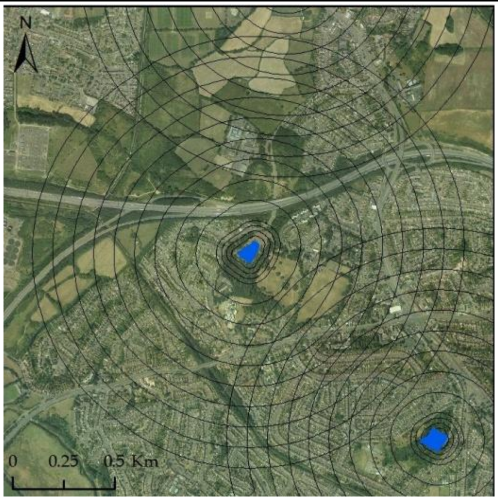
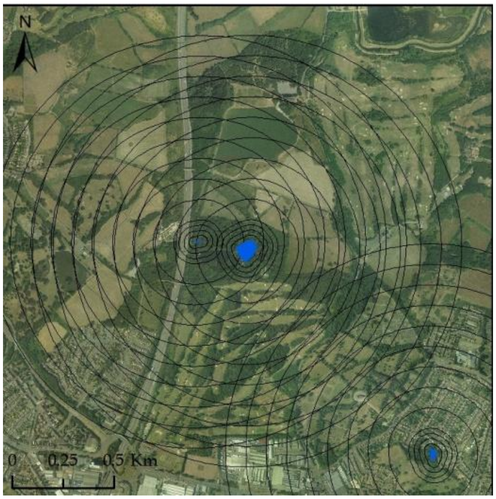
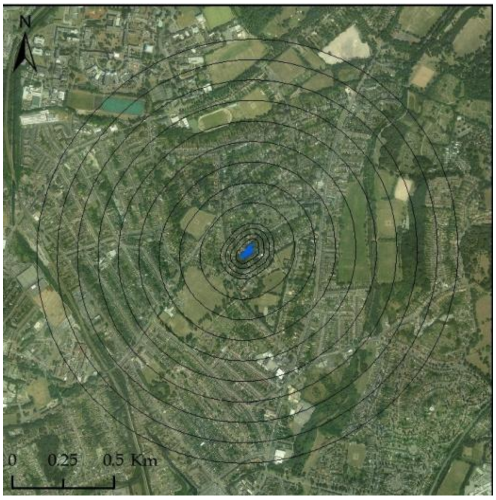
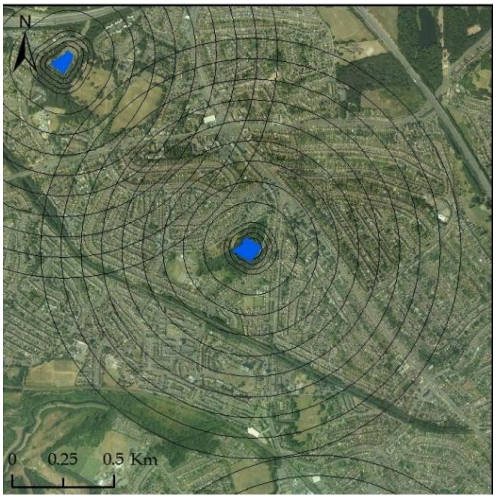
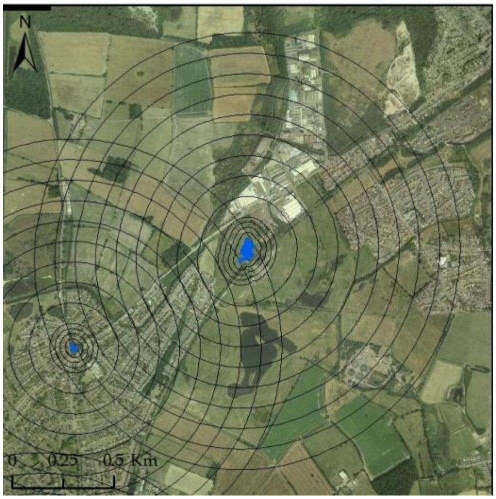
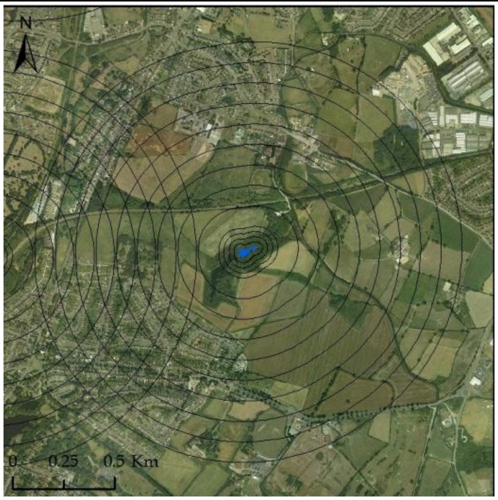
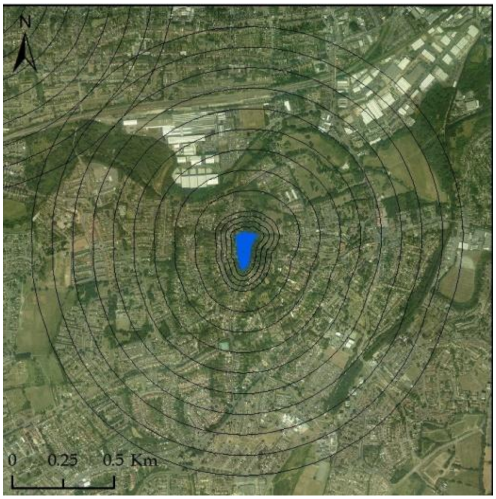
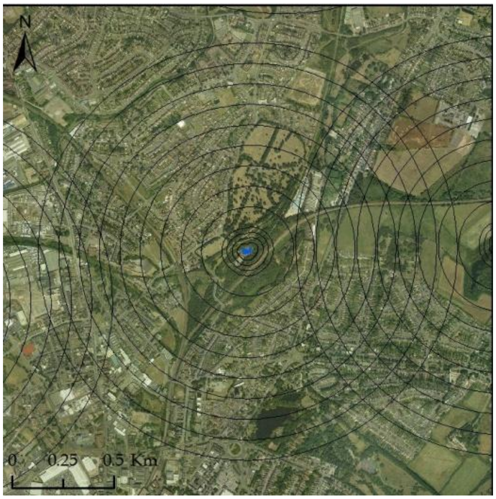
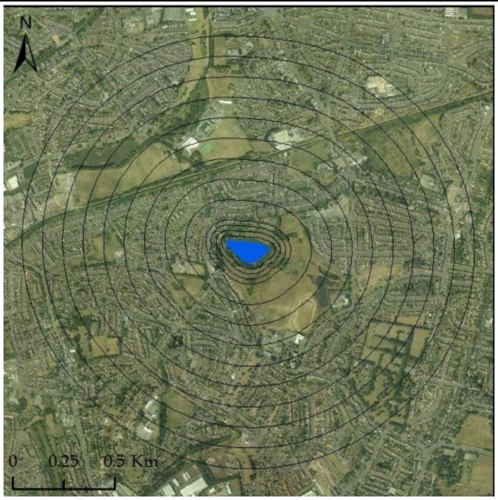
15		

Site	Site photo	Aerial image (with concentric rings)
16		
17		
18		

Site	Site photo	Aerial image (with concentric rings)
19		
20		
21		

Site	Site photo	Aerial image (with concentric rings)
22		
23		
24		

Site	Site photo	Aerial image (with concentric rings)
25		
26		
27		

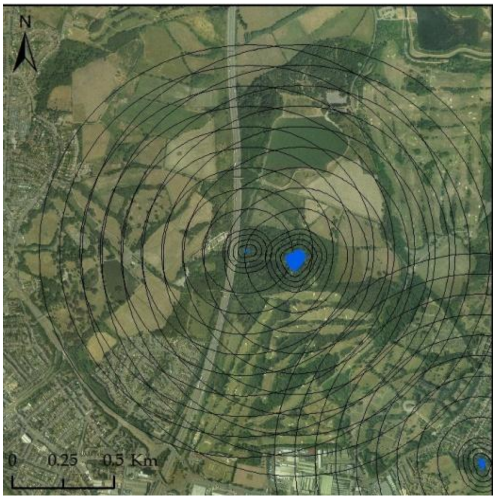
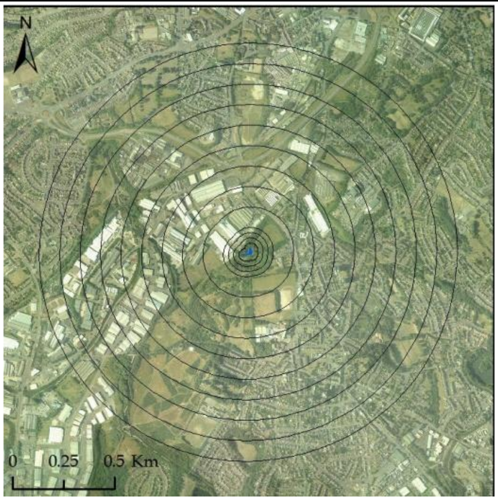
Site	Site photo	Aerial image (with concentric rings)
28		
29		
30		

Table A2.1. Site summary table of main local physical factors. m² Pond surface area, Tree Percentage tree cover, *Spr* Approximate percentage of water sourced from groundwater, *Rain* Approximate percentage of water sourced from rainwater run-off, *RdBld* Approximate percentage of water sourced from road and building run-off, *Stream* Approximate percentage of water sourced from stream input, *Storm* Approximate percentage of water sourced from storm water storage, *Bore* Approximate percentage of water sourced from borehole input, *WLFI* Wetland Fluctuation Index, *Fish* Fish presence / absence.

Site	x	y	m²	Tree	Spr	Rain	RdBld	Stream	Storm	Bore	WLFI	Fish
1	397642	291856	1822	0.03	50	50	0	0	0	0	72.8	1
2	398519	301654	4942	0.17	100	0	0	0	0	0	111.6	1
3	402986	280070	1215	0.38	0	0	0	100	0	0	16.0	0
4	404342	285615	925	0.69	0	0	100	0	0	0	41.2	0
5	402938	283702	408	0.93	75	25	0	0	0	0	24.0	0
6	403200	284982	2224	0.42	75	25	0	0	0	0	28.1	1
7	403140	283725	1499	0.42	0	25	75	0	0	0	14.1	0
8	403313	290445	1327	0.65	75	25	0	0	0	0	15.5	1
9	400512	288765	14923	0.09	75	25	0	0	0	0	12.0	1
10	400095	299680	2281	0.00	0	100	0	0	0	0	154.3	0
11	402168	302673	1774	0.06	0	50	25	0	0	25	20.4	0
12	401939	302131	5987	0.00	100	0	0	0	0	0	353.6	0
13	407231	280055	1226	0.51	0	100	0	0	0	0	119.4	0
14	409639	289193	721	0.00	0	100	0	0	0	0	168.8	1
15	405047	291599	1505	0.32	75	25	0	0	0	25	6.4	1
16	402431	288266	5499	0.20	0	25	0	75	0	0	69.8	1
17	403816	294543	6423	0.03	0	0	0	0	100	0	18.6	0
18	402401	291441	6963	0.15	100	0	0	0	0	0	8.9	1
19	405514	282619	2108	0.50	25	0	75	0	0	0	66.3	0
20	404722	293626	1464	0.75	75	25	0	0	0	0	199.8	0
21	403157	304684	5992	0.00	0	75	25	0	0	0	87.8	0
22	402316	304215	1010	0.27	25	25	50	0	0	0	3558.2	0
23	403073	299968	2023	0.15	0	0	0	100	0	0	18.1	1
24	404369	278777	12078	0.19	0	0	0	100	0	0	29.5	1
25	401801	299936	1123	0.00	100	0	0	0	0	0	17.7	1
26	400203	293942	14967	0.04	0	25	0	75	0	0	15.5	0
27	396817	296552	3784	0.09	3	0	25	0	0	0	4.1	1
28	402162	291482	299	0.95	0	100	0	0	0	0	4698.4	0
29	393963	288745	692	0.06	0	100	0	0	0	0	248.1	0
30	404004	295942	710	1.00	0	0	100	0	0	0	126.9	0

Appendix 3. Water quality limits of detection

Table A3.1. Limits of detection for ion analyses. Values listed are 3 x SD of blank (deionised water) sample within each survey analysis, for each determinand. Values are in mg/l.

	Cl	NO_3	SO_4	PO_4	Na	NH_4	K	Mg	Ca
20th May – 11th June 2009	0.103	0.272	0.032	NA	0.231	0.033	0.206	0.372	0.231
3rd August – 14th August 2009	0.023	0.019	0.026	NA	0.023	0.026	0.041	0.420	0.023
3rd November – 17th November 2009	0.073	0.033	0.117	0.01	0.090	0.069	0.060	0.087	0.090
27th February – 3rd March 2010	0.068	0.281	1.606	0.01	0.039	0.080	0.025	0.133	0.039
9th June – 11th June 2010	0.068	0.007	0.125	0.01	0.021	0.048	0.005	0.110	0.021
18th August – 25th August 2010	0.095	0.529	0.322	0.01	0.186	0.134	0.022	0.191	0.186
10th November – 13th November 2011	0.512	0.119	0.689	0.01	0.036	0.021	0.006	0.190	0.036
21st February – 23rd February 2011	0.548	0.123	0.938	0.01	0.061	0.049	0.075	0.519	0.279

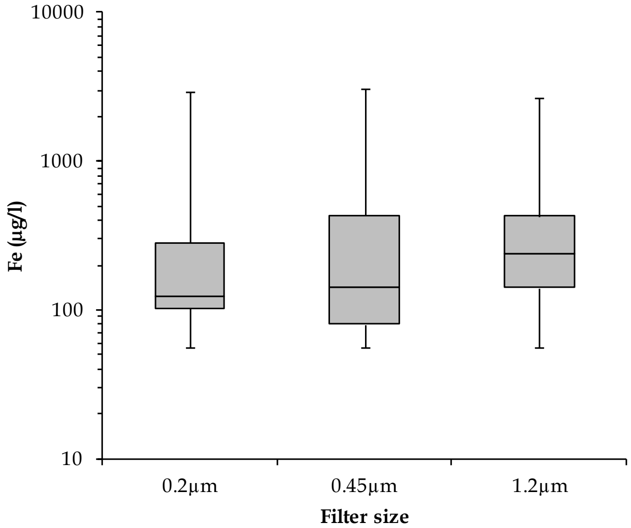
*PO_4 measured using Hanna Instruments Phosphate Photometer, which has a 0.01mg/l resolution and ±0.04mg/l accuracy

Table A3.2. Limits of detection for trace metals analysis (flame atomic absorption spectrometry). Values listed are 3 x SD of blank (deionised water) sample within each survey analysis, for each determinand. Values are in mg/l.

	Co	Cr	Cu	Fe	Mn	Ni	Pb	Zn
20th May – 11th June 2009	0.153	0.113	0.023	0.097	0.092	0.082	0.032	0.078
3rd August – 14th August 2009	0.032	0.054	0.022	0.096	0.019	0.073	0.155	0.034
3rd November – 17th November 2009	0.121	0.038	0.013	0.059	0.023	0.047	0.084	0.012
27th February – 3rd March 2010	0.116	0.025	0.021	0.076	0.020	0.051	0.190	0.052
9th June – 11th June 2010	0.035	0.063	0.016	0.093	0.025	0.081	0.116	0.028
18th August – 25th August 2010	0.035	0.068	0.028	0.047	0.007	0.054	0.088	0.015
10th November – 13th November 2011	0.108	0.089	0.039	0.038	0.121	0.084	0.058	0.012
21st February – 23rd February 2011	0.023	0.033	0.019	0.081	0.006	0.079	0.127	0.014

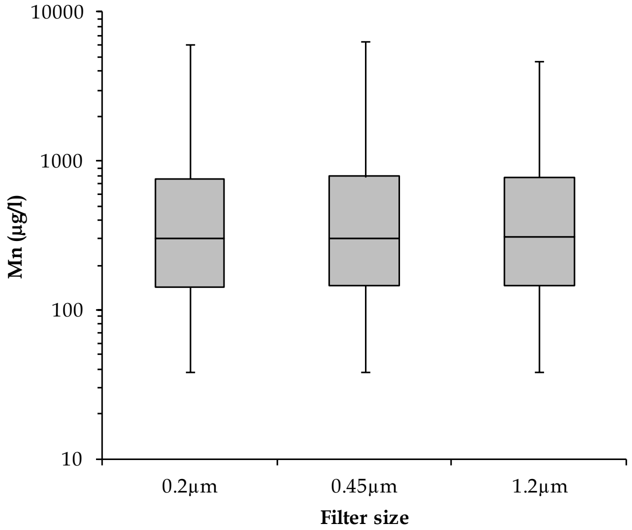
Appendix 4. The effect of filter size upon trace metal concentrations

a) Fe (n = 29) b) Mn (n = 23) c) Zn (n = 7)



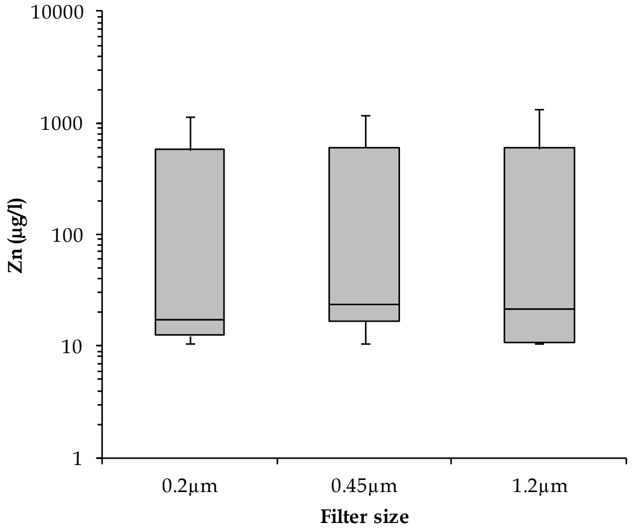
ln(Fe+1) transformation

ANOVA $P > 0.05$



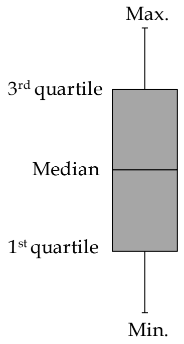
ln(Mn+1) transformation

ANOVA $P > 0.05$



ln(Zn+1) transformation

ANOVA $P > 0.05$



* As per the main study, few samples contained concentrations of Ni, Cu, Co, Cr or Pb above detection limits

Appendix 5. Publication of historical maps

Figure A7.4.1. County Series map publication

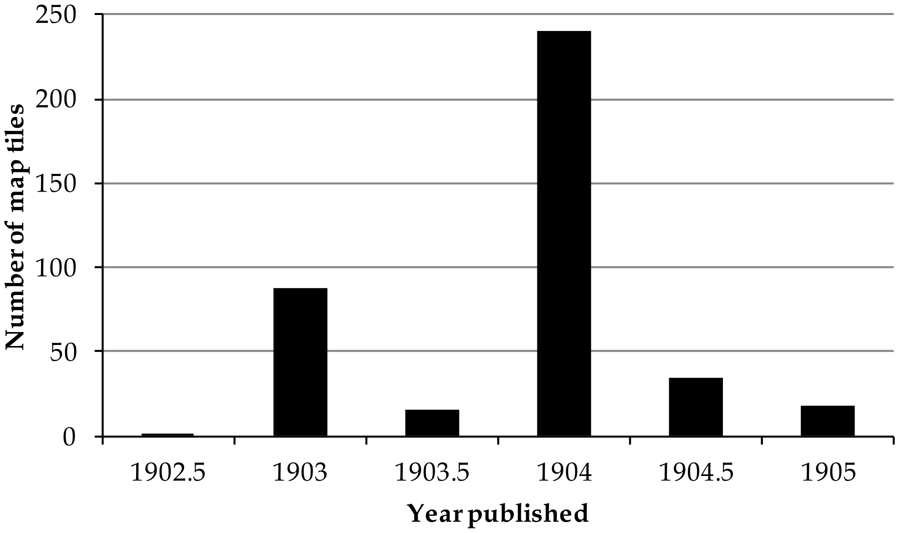
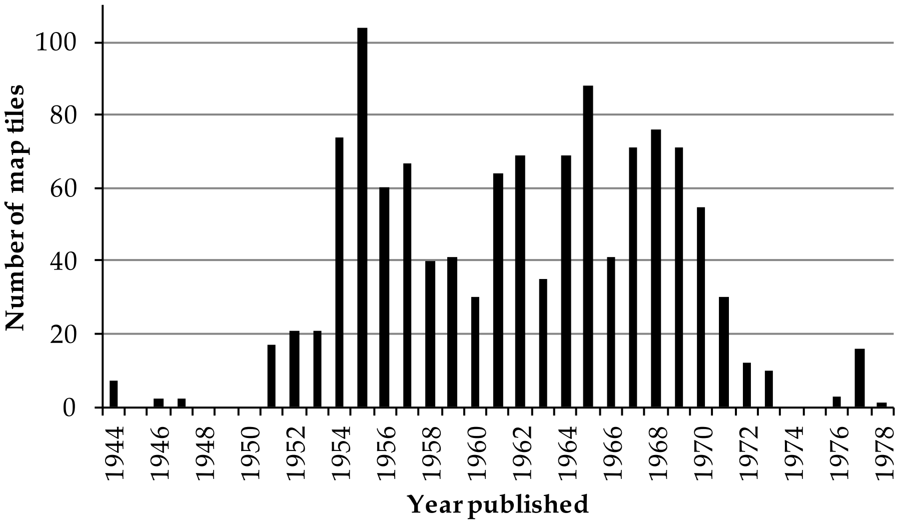


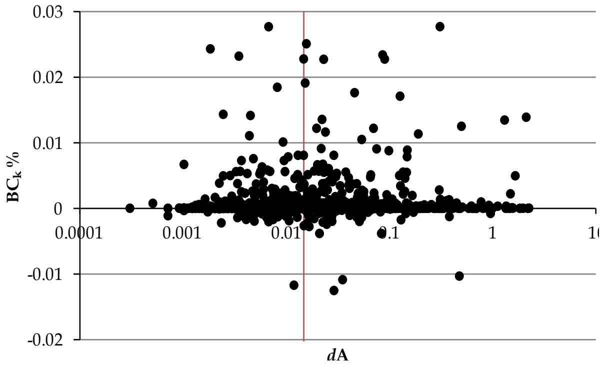
Figure A7.4.2. National Grid series map publication



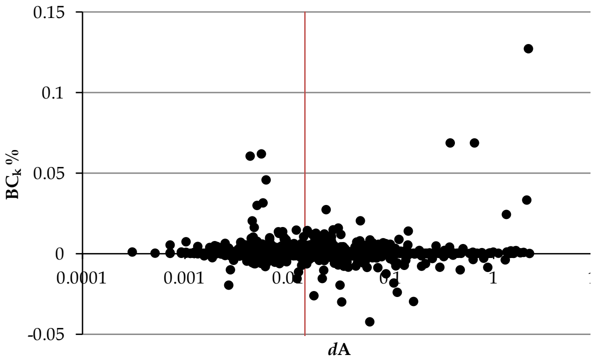
Appendix 6. The effect of retaining larger connected ponds relative to smaller ponds

BC_k % The betweeness centrality metric expressed as a proportion of the sum of BC metrics for the whole network, dA The proportional contribution to the total habitat made by the focal node. Black dots are individual nodes, the red vertical line is the median dA

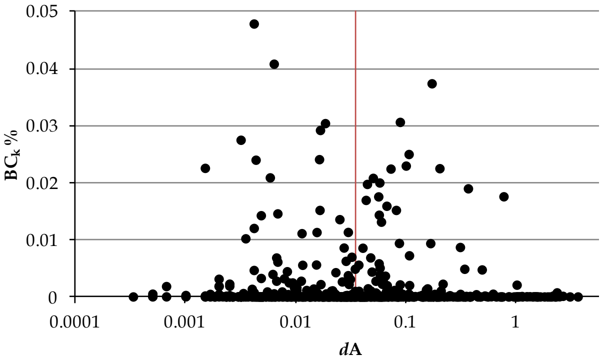
a) *ca*1904, 500m threshold
Median *d*A = 0.0148
$\bar{x}$ BC_k % > 0.0148 = 0.000404
$\bar{x}$ BC_k % > 0.0148 = 0.000611



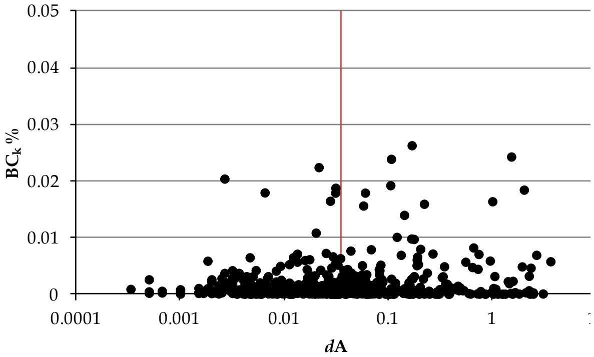
b) *ca*1904, 1500m threshold
Median *d*A = 0.0148
$\bar{x}$ BC_k % < 0.0148 = 0.000549
$\bar{x}$ BC_k % > 0.0148 = 0.000493



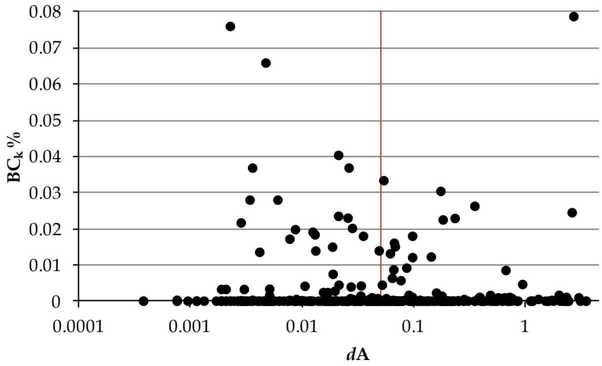
c) *ca*1962, 500m threshold
Median *d*A = 0.0351
$\bar{x}$ BC_k % < 0.0351 = 0.001978
$\bar{x}$ BC_k % > 0.0351 = 0.001838



d) *ca*1962, 1500m threshold
Median *d*A = 0.0351
$\bar{x}$ BC_k % < 0.0351 = 0.001598
$\bar{x}$ BC_k % > 0.0351 = 0.002219



e) 2009, 500m threshold
Median dA = 0.0509
$\bar{x}$ BC_k % < 0.0509 = 0.003538
$\bar{x}$ BC_k % > 0.0509 = 0.002343



f) 2009, 1500m threshold
Median dA = 0.0509
$\bar{x}$ BC_k % < 0.0509 = 0.002555
$\bar{x}$ BC_k % > 0.0509 = 0.003327

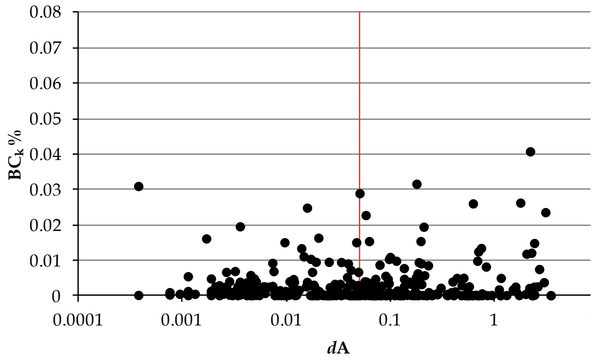
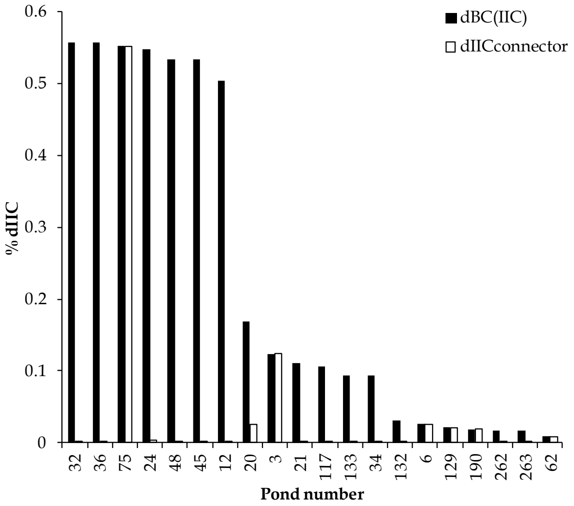


Table A6.1. Average BC_k values for ponds greater than, and smaller than median surface area between ca1904, ca1962 and 2009. *d*A is the pond surface area expressed as a proportion of total habitat area.

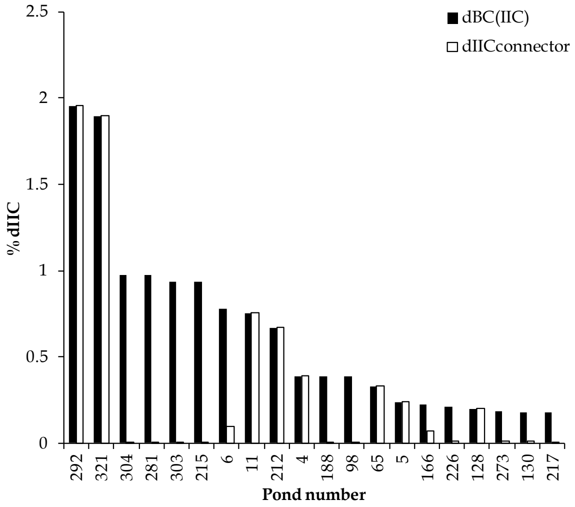
	1500m		500m	
Period	>Med. *d*A	<Med. *d*A	>Med. *d*A	<Med. *d*A
1904	0.48	**0.53**	**0.61**	0.42
1962	**0.58**	0.42	0.48	**0.52**
2009	**0.57**	0.43	0.40	**0.60**

Appendix 7. Identifying priority pond sites for connectivity in the Birmingham pondscape

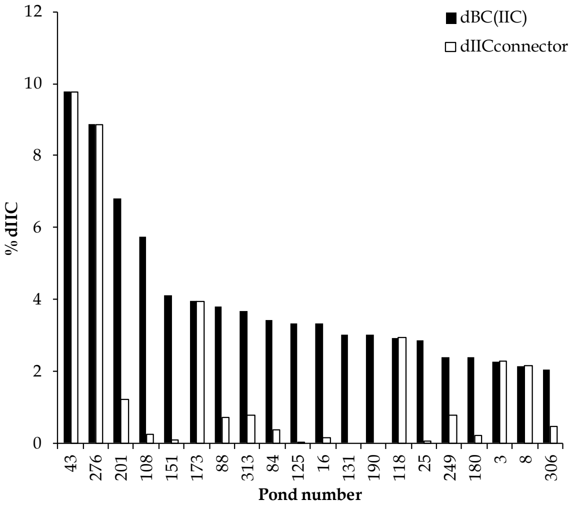
a) 250m



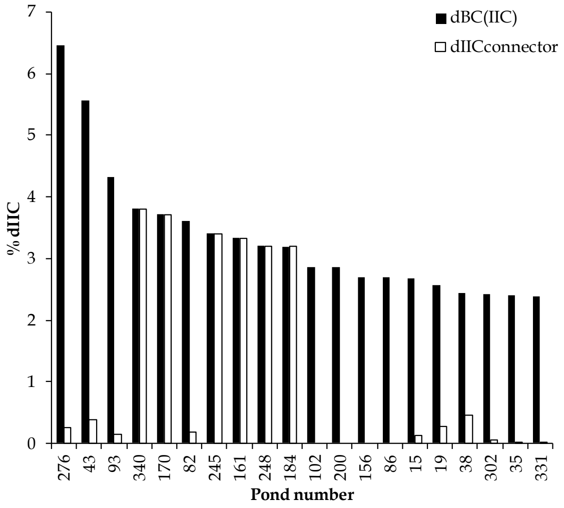
b) 500m



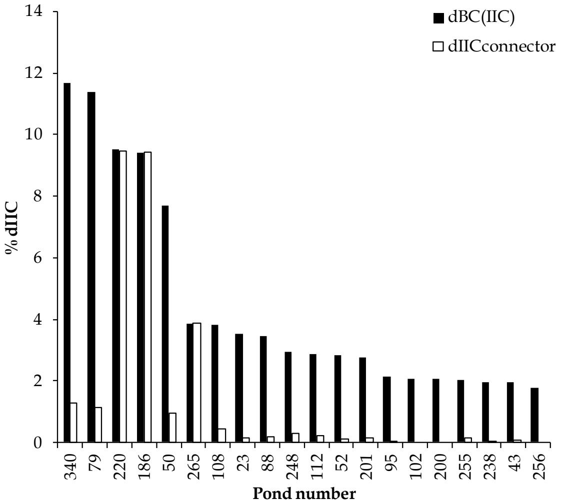
c) 1000m



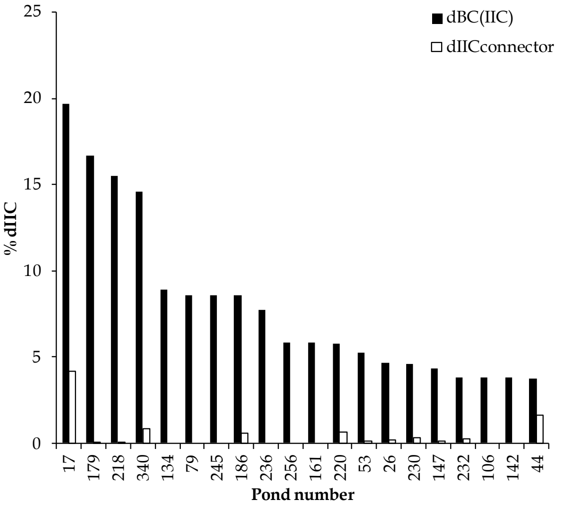
d) 1500m



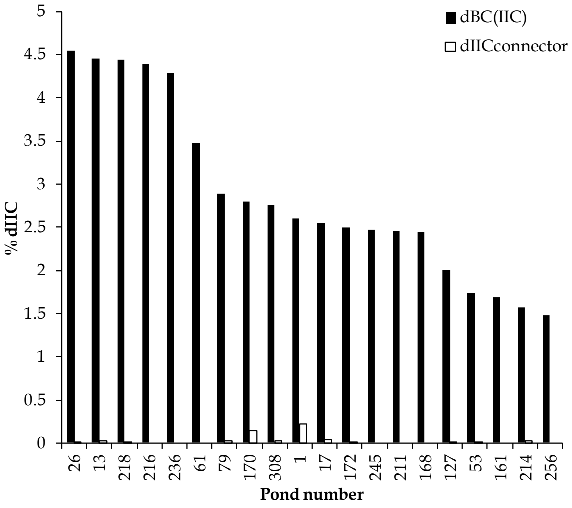
e) 2000m



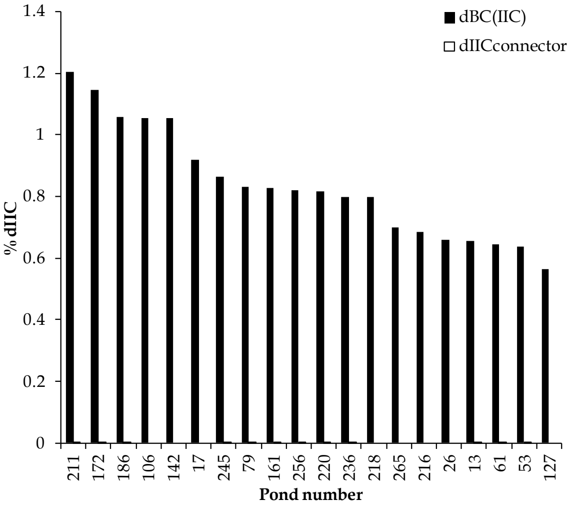
f) 2500m



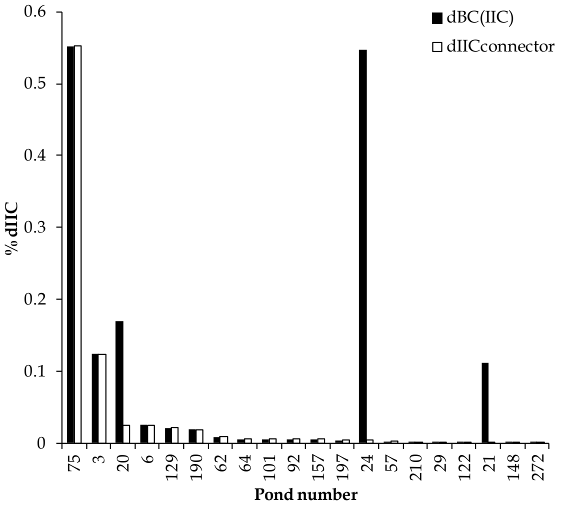
g) 5000m



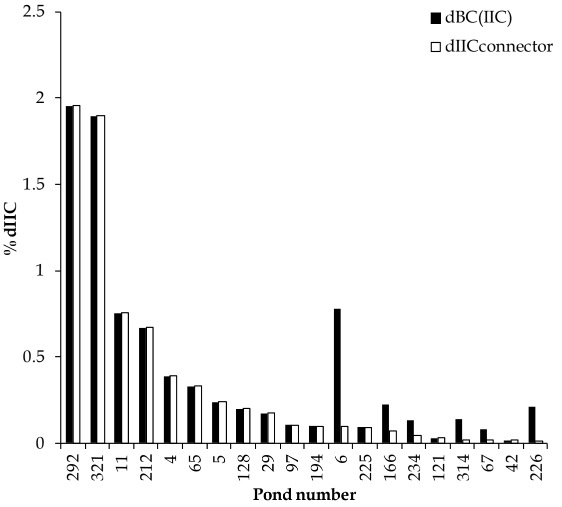
h) 10000m



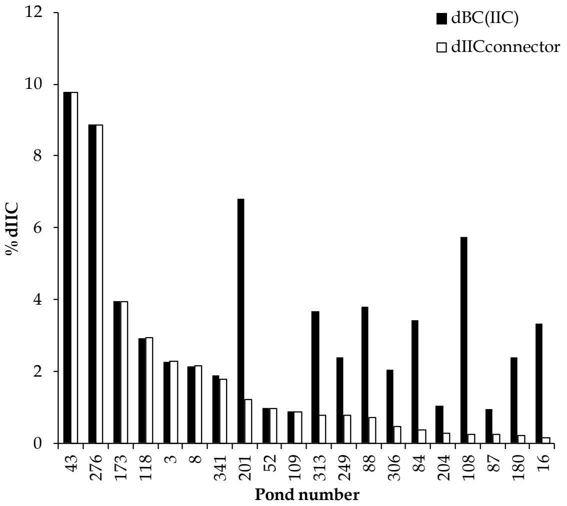
i) 250m



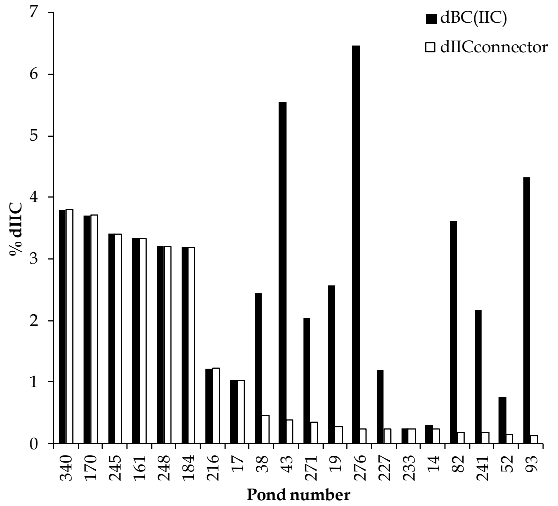
j) 500m



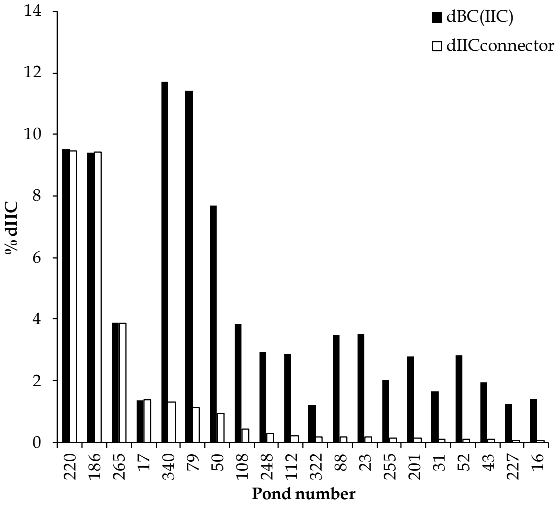
k) 1000m



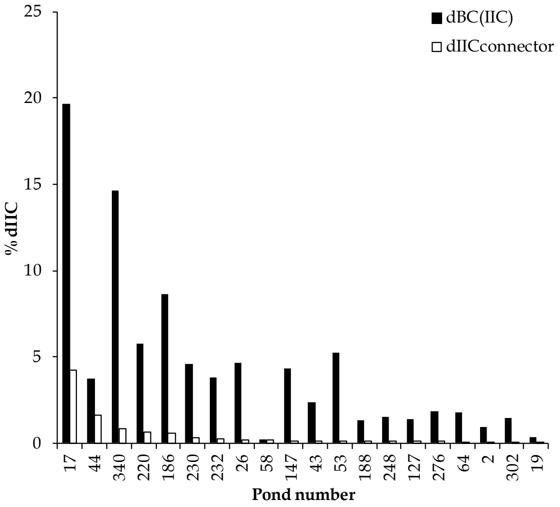
l) 1500m



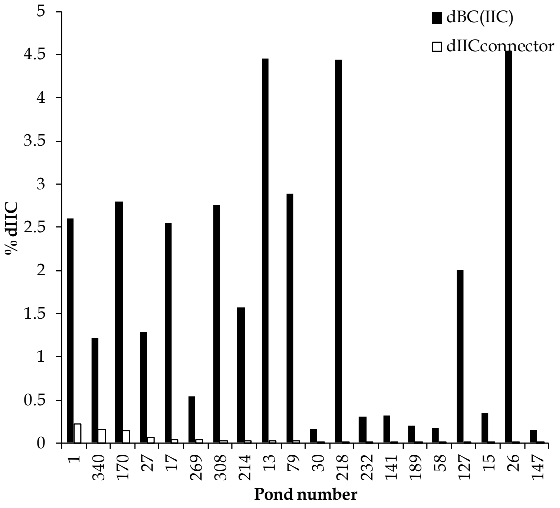
m) 2000m



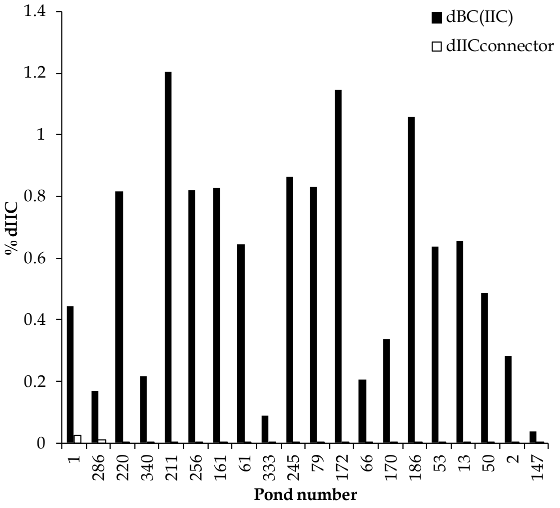
n) 2500m



o) 5000m



p) 10000m



Appendix 8. The potential for garden ponds to reconnect the 'pondscape' following pond loss

The possible presence of garden ponds was modelled based upon the results of a study by Davies *et al.* (2009), which suggests that, on average, 10% of U.K. households have a garden pond, with a surface area of 0.99m^2. OS MasterMap delineates private gardens but does not make explicit to which building each garden belongs, and one garden may be represented by several polygons. Consequently, permanent roofed constructions (OS MasterMap definition) were selected within Birmingham which shared a boundary with a private garden. An initial 587379 features were selected, which at this stage included duplicates. Small and large buildings such as garages and warehouses were removed by imposing a lower limit of 29m^2 and an upper limit of 300m^2 derived from aerial image inspections, nonetheless it is acknowledged that some none residential buildings remained. Subsequently, 347314 such buildings remained to represent households after duplicates were removed. A random selection of 10% (34731) of households were selected using Hawths Tools ArcGIS extension (Beyer 2004) and their centroids were used to represent garden pond potential. Unfortunately, due to the computing power it was not possible to include garden size as a weighting factor toward pond presence, as suggested by Loram *et al.* (2011).

The density of possible garden ponds, and ponds within the public realm (i.e. Section 2.1), was extracted from a GIS for each 1 km^2, as was the proportional cover of impermeable surface. An inverse correlation between the two types of urban ponds was revealed up until the 50 – 60% impermeable surface cover, after which, numbers of both decline (Figure A7.4.1). A non-linear, unimodal relationship was exhibited between the proportion of impermeable surface and the number of possible garden ponds, the peak density in which occurs at around 50% impermeable surface cover. Plotting the number of public ponds directly against possible garden ponds reveals the inverse relationship more clearly (Figure A7.4.2).

The peak in possible garden ponds, around 50%, is broadly consistent with the percentage cover of impermeable surface found within LCM2007 suburban 'habitats' (Section 2.3.1.3) in Birmingham (36.1%). The 50% figure likely represents densely populated residential areas. The inverse relationship between the number of possible garden ponds and public ponds suggests the potential for garden ponds to act as stepping stones to enable aquatic organisms to penetrate deeper into urban areas than public ponds alone. Nonetheless, the modelling methodology leaves much room for improvement and field validation is necessary in order to form any reliable conclusions.

Figure A7.4.1. Mean counts of possible garden ponds and public ponds within 1km² regions of Birmingham

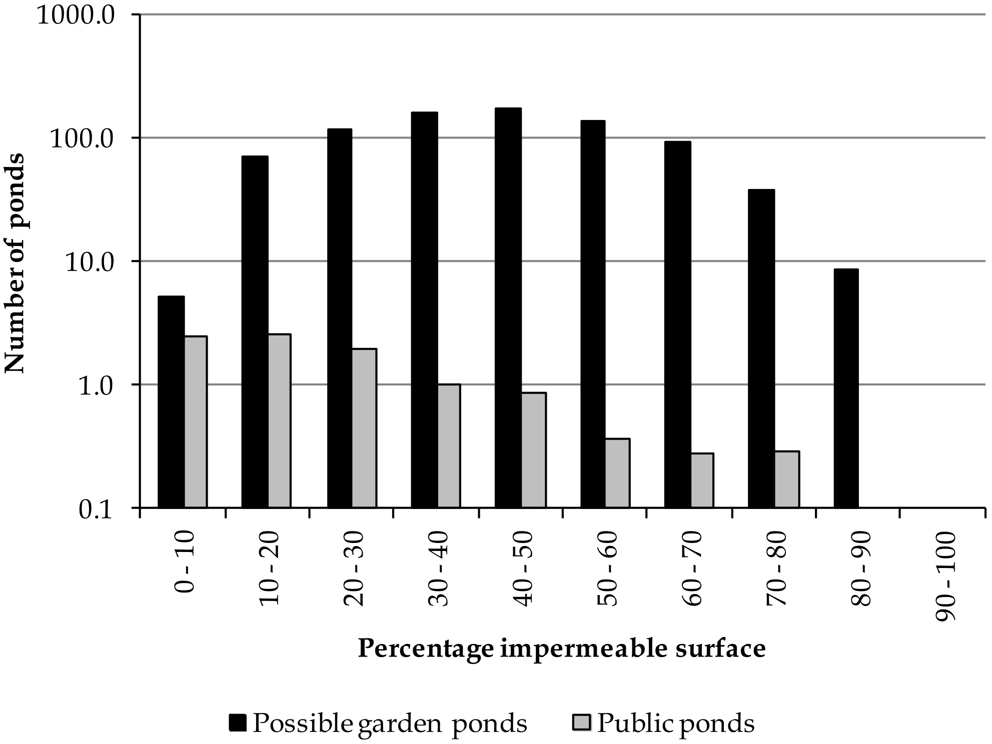
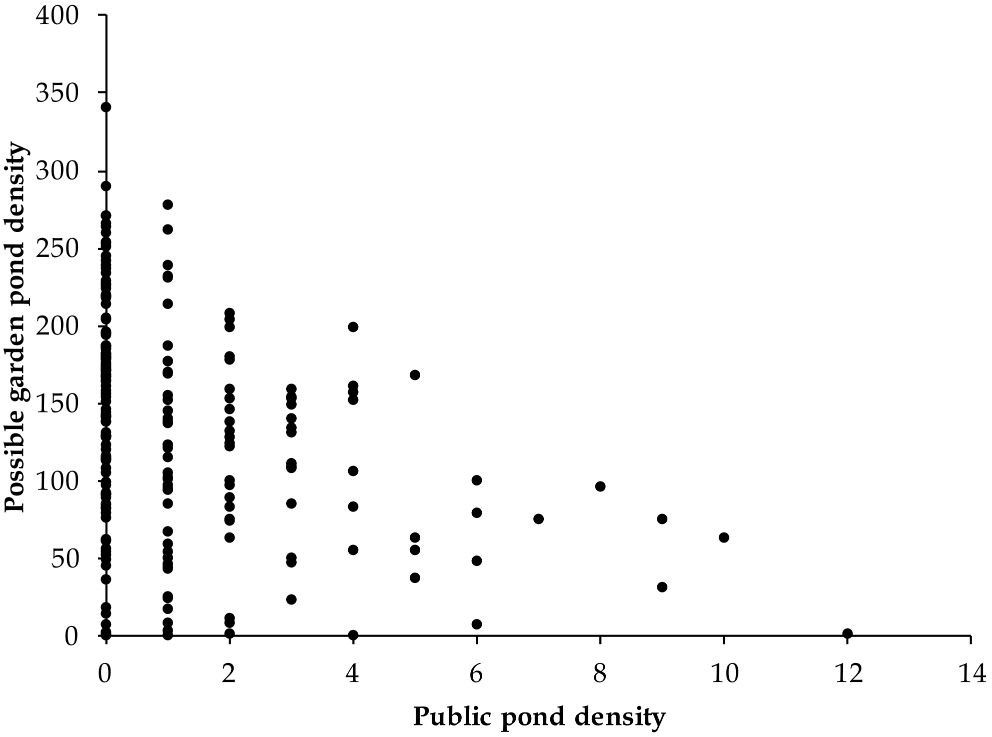


Figure A7.4.2. Public pond density vs. possible garden pond density within 1km² regions of Birmingham



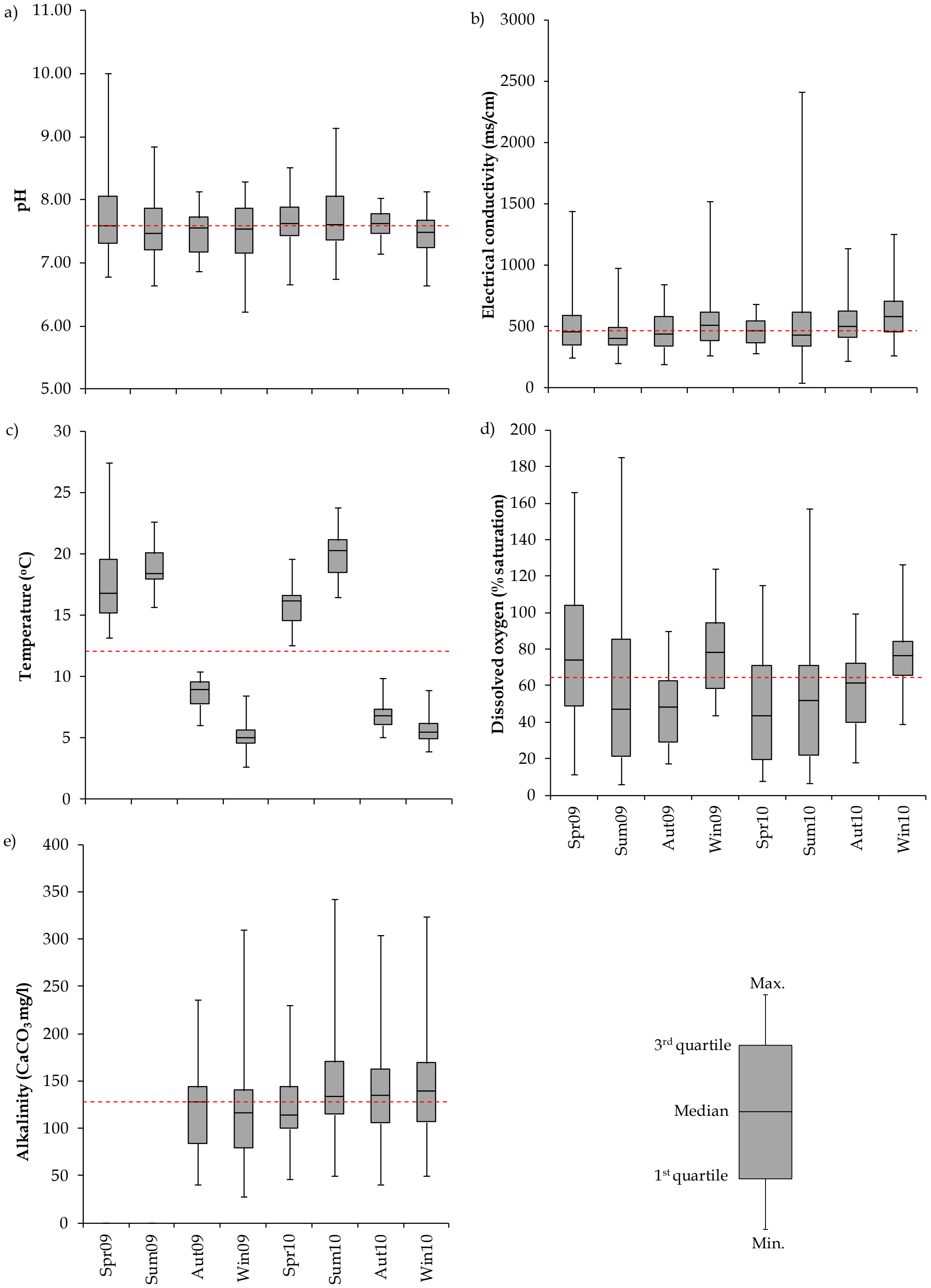
Appendix 9. Water quality summary table for all study sites and overall study (8 samples per site unless otherwise stated, average ± 1SD)

Site	pH	EC	Temp.	DO	Cl	NO_3	PO_4	SO_4	$CaCO_3$*	Na	NH_4	K	Mg	Ca	Fe**	Mn**	Zn**	Chl-*a****	SS***
	-	µS/cm	^{o}C	%	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	µg/l	µg/l	µg/l	µg/l	mg/l
1	8.0±0.3	431±60	13.4±8.0	84.0±14	9.7±3	0.99±1.8	0.04±0.03	47.2±21	172±52	9.6±2.8	0.07±0.07	8.0±0.8	8.2±2.5	58.7±23	90±8.4	128±75.1	101±-	12.6±10	8.7±6
2	7.6±0.2	695±57	12.3±7.1	71.3±21	131.3±57	2.06±2.7	0.02±0.01	39.7±16	72±14	74.3±7.6	0.18±0.33	3.4±0.8	5.8±1.1	36.7±4	229±161	95±44.7	87±-	25.6±35	9.4±9
3	7.2±0.2	352±103	12.5±6.6	44.1±28	24.6±11	5.69±5.6	0.81±0.99	22.9±6	100±26	14.7±5.4	0.80±0.63	2.5±1.1	5.8±2.6	42.7±22	572±285	225±201	71±105	9.4±15	20±14
4	7.4±0.2	357±64	11.3±4.9	35.6±23	23.8±12	2.31±3.7	1.30±0.92	29.5±14	95±24	12.6±5.4	2.23±2.16	4.7±0.9	3.9±0.8	38.5±8	120±35.2	165±82.0	116±135	8.6±10	6.1±2
5	6.8±0.3	349±22	11.2±5.4	33.7±28	30.0±8	1.29±2.3	0.44±0.34	56.6±16	46±11	14.7±1.4	0.83±0.94	12.6±0.7	7.0±1.8	23.3±3	369±132	547±255	219±339	38.2±57	11±16
6	8.2±0.5	411±48	12.8±6.5	88.5±35	29.3±8	1.07±1.8	0.63±0.81	27.6±10	139±16	15.4±2.1	0.50±0.54	5.2±1.1	4.3±1.2	51.4±5	547±-	104±-	72±84.9	139±196	27±22
7	7.3±0.3	524±90	11.8±6.5	47.7±33	64.5±25	1.86±3.0	0.71±0.57	40.1±17	117±39	31.0±16	1.49±2.19	6.7±2.2	6.4±1.7	43.8±5	540±411	446±286	63±57.4	60.4±80	10±8
8	7.3±0.1	419±27	11.6±5.7	42.5±24	44.1±9	2.31±3.3	0.14±0.08	46.7±7	91±18	21.2±2.1	0.32±0.44	4.2±0.7	4.5±0.7	42.5±4	377±329	965±596	63±88.5	49.5±50	12±8
9	7.8±0.4	550±62	12.4±6.9	87.1±22	37.7±10	10.8±18.0	0.28±0.22	54.4±19	176±33	13.0±3.8	0.21±0.26	3.8±3.0	7.7±1.8	70.7±10	158±110	139±106	803±-	25.9±41	7.5±5
10	8.1±0.4	333±35	12.3±7.0	82.2±20	18.7±17	14.5±26.2	0.08±0.13	43.4±21	105±15	6.7±2.0	0.14±0.14	4.1±0.8	7.4±2.1	39.3±6	336±418	75±-	63±84.1	38.1±59	47±31
11	7.3±0.4	382±117	12.5±7.3	65.0±30	39.0±23	2.13±3.9	0.12±0.11	28.6±10	106±30	14.6±3.8	1.93±1.29	8.3±1.6	5.5±1.9	36.9±14	2405±3283	155±102	160±191	320±535	30±23
12	7.8±0.8	586±70	13.9±9.4	88.9±38	46.7±19	4.08±8.0	0.19±0.19	76.0±46	146±25	26.6±5.5	0.19±0.33	4.5±0.8	9.3±1.7	61.2±8	225±56.4	143±84.9	21±1.3	6.7±11	7.0±4
13	7.6±0.4	422±37	11.4±6.2	46.7±33	15.0±9	1.27±2.1	1.72±0.88	8.6±12	191±30	6.1±1.1	1.48±1.94	6.9±1.6	10.2±2.0	47.9±11	257±279	550±497	45±20.5	162±262	16±10
14	7.7±0.3	334±33	13.6±6.5	63.1±10	12.4±7	2.21±3.9	0.03±0.02	8.9±18	142±24	6.6±1.3	0.10±0.08	3.0±1.4	4.9±1.0	45.5±4	126±48.3	83±27.4	31±14.8	26.7±30	13±7
15	8.1±0.7	459±72	13.1±6.5	88.1±65	29.2±14	9.82±19.3	0.69±0.92	37.8±26	112±18	17.4±5.2	0.66±0.93	4.6±0.6	6.9±1.3	49.4±7	225±122	158±138	34±17.2	22.7±37	14±12
16	7.5±0.3	449±71	12.9±6.4	37.4±18	37.8±21	2.84±3.7	0.11±0.09	29.6±20	145±20	17.0±7.2	1.47±0.98	5.0±0.9	5.7±1.2	49.2±11	365±252	557±332	27±-	84.7±120	13±12
17	8.1±1.0	573±441	12.5±6.9	62.0±25	95.6±121	3.60±6.3	0.10±0.08	25.1±14	99±27	56.8±66	0.24±0.25	2.8±1.2	3.9±1.1	36.2±16	248±69.9	58±25.7	40±24.3	152±340	15±12
18	7.8±0.3	488±66	12.5±6.9	72.2±28	21.8±9	1.31±1.8	0.08±0.02	41.2±21	175±25	20.4±32	0.16±0.13	1.9±0.9	5.2±1.4	60.4±18	198±-	69±38.7	134±-	22.3±33	7.0±6
19	7.2±0.1	274±23	11.0±6.0	28.9±16	18.1±15	6.44±14.5	1.38±0.62	11.1±24	112±19	7.1±1.6	1.83±1.41	6.2±1.1	4.5±0.7	28.2±2	224±72.2	407±195	125±89.1	164±241	21±14
20	7.4±0.3	539±57	11.4±5.2	54.4±8	60.4±19	5.36±3.4	0.19±0.15	58.3±22	112±16	34.1±8.4	0.21±0.23	3.2±0.4	9.8±1.8	49.0±9	832±1076	548±577	21±-	5.8±5.3	19±22
21	7.9±0.8	603±253	13.7±7.8	100±27	89.0±52	2.31±3.5	0.04±0.02	92.8±66	129±69	43.0±17	0.17±0.21	6.2±2.8	13.8±11	46.7±20	584±227	354±161	82±111	27.2±32	24±15
22	7.3±0.4	679±171	12.4±7.4	35.2±25	124.5±67	2.56±4.7	0.12±0.11	5.7±3	129±17	76.1±34	0.23±0.37	9.5±4.3	3.3±1.0	31.3±6	1447±403	265±229	34±8.3	127±118	64±17
23	7.8±0.2	719±94	11.6±5.9	82.6±16	51.5±36	9.69±8.0	0.09±0.04	127±105	183±53	27.1±12	0.14±0.13	4.7±2.0	15.6±7.0	79.2±17	94±-	458±715	-	23.6±35	7.3±4
24	7.6±0.4	297±96	11.9±6.8	65.9±25	39.8±23	1.29±1.5	0.50±0.62	13.5±6	72±13	20.8±13	0.11±0.15	2.6±0.7	4.7±1.1	20.9±6	220±123	85±58.0	110±149	59.8±65	13±3
25	7.6±0.3	629±33	12.3±6.9	65.1±16	57.9±14	9.18±9.0	0.04±0.02	98.6±22	119±17	41.8±6.4	0.12±0.16	3.7±1.0	8.6±1.4	56.5±8	168±113	161±54.9	-	49.8±93	12±8
26	7.4±0.1	1100±258	14.0±5.3	55.7±14	65.5±31	9.98±6.5	0.12±0.06	138.9±36	265±53	51.1±17	5.54±1.48	10.7±2.2	16.1±3.6	107.3±23	167±81.5	843±232	52±34.4	10.5±12	17±6
27	7.9±0.3	461±85	13.0±6.7	67.0±30	37.6±8	13.2±28.5	0.07±0.04	35.8±10	138±22	20.5±5.5	0.10±0.19	3.4±2.1	10.7±1.9	47.6±9	460±565	334±-	322±-	36.0±48	6.6±5
28	7.3±0.3	822±659	11.0±5.4	28.1±17	25.6±14	1.37±2.1	0.46±0.39	225±440	180±113	12.3±5.2	0.67±1.23	7.8±6.4	8.9±5.7	147±152	556±472	2863±3028	48±29.3	46.0±65	33±10
29	7.8±0.2	809±260	12.4±6.4	75.3±32	31.2±17	2.76±3.7	0.09±0.07	143±169	218±100	17.0±9.0	0.08±0.09	9.0±3.5	16.1±7.7	87.3±46	167±51.3	134±73.4	34±14.1	58.5±125	7.7±5
30	7.5±0.2	800±167	10.5±5.3	33.3±22	123.2±115	3.17±5.4	0.40±0.30	28.3±21	149±47	80.6±51	0.86±0.76	6.2±1.5	4.8±1.3	63.8±18	213±83.3	394±292	38±25.7	7.6±8.9	25±28
Av.	7.61	528.0	12.31	61.1	47.82	4.58	0.37	54.74	134	27.00	0.77	5.50	7.65	53.3	480	473	80	92.3	17.4
Min	6.23	44.5	2.67	6.0	0.73	0.00	0.00	0.93	28	4.24	0.00	0.58	1.56	12.0	81	40	20	0.0	0.0
Max	10.01	2418.0	27.5	185.2	377.78	77.50	2.63	1312.18	342	209.32	8.21	21.39	41.36	516.4	8139	9211	803	1495	109.8

*Analysis began autumn 2009; 6 samples analysed **Figures do not include concentrations below the limit of detection ***Analysis began summer 2009; 7 samples analysed

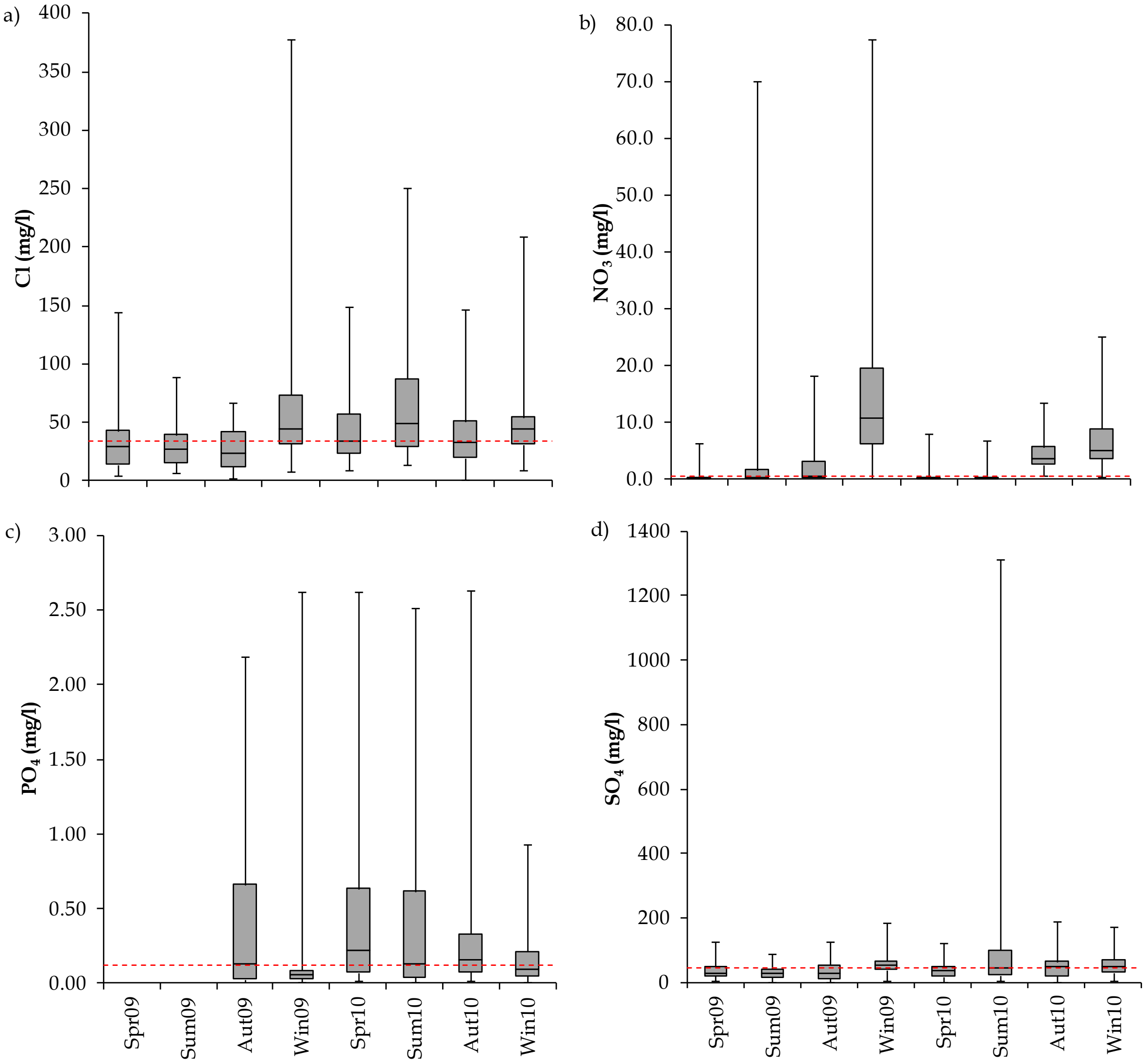
Appendix 10. Temporal variation of In situ measurements

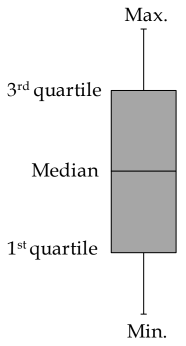
a) pH b) electrical conductivity c) temperature d) dissolved oxygen e) alkalinity and study median (red dashed line)



Appendix 11. Temporal variation of major anion concentrations

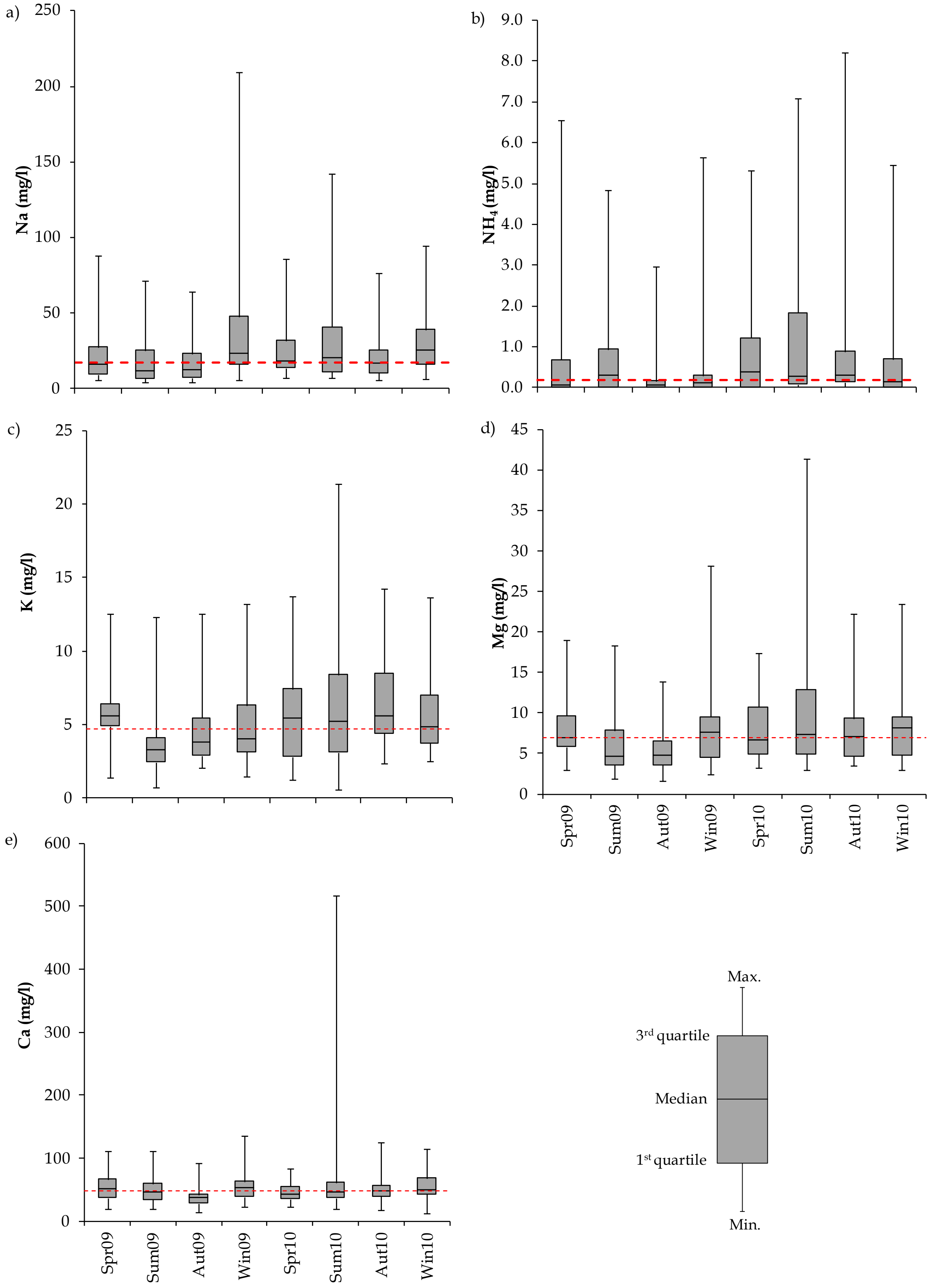
a) Cl b) NO_3 c) PO_4 d) SO_4 e) $CaCO_3$ and study median (red dashed line)





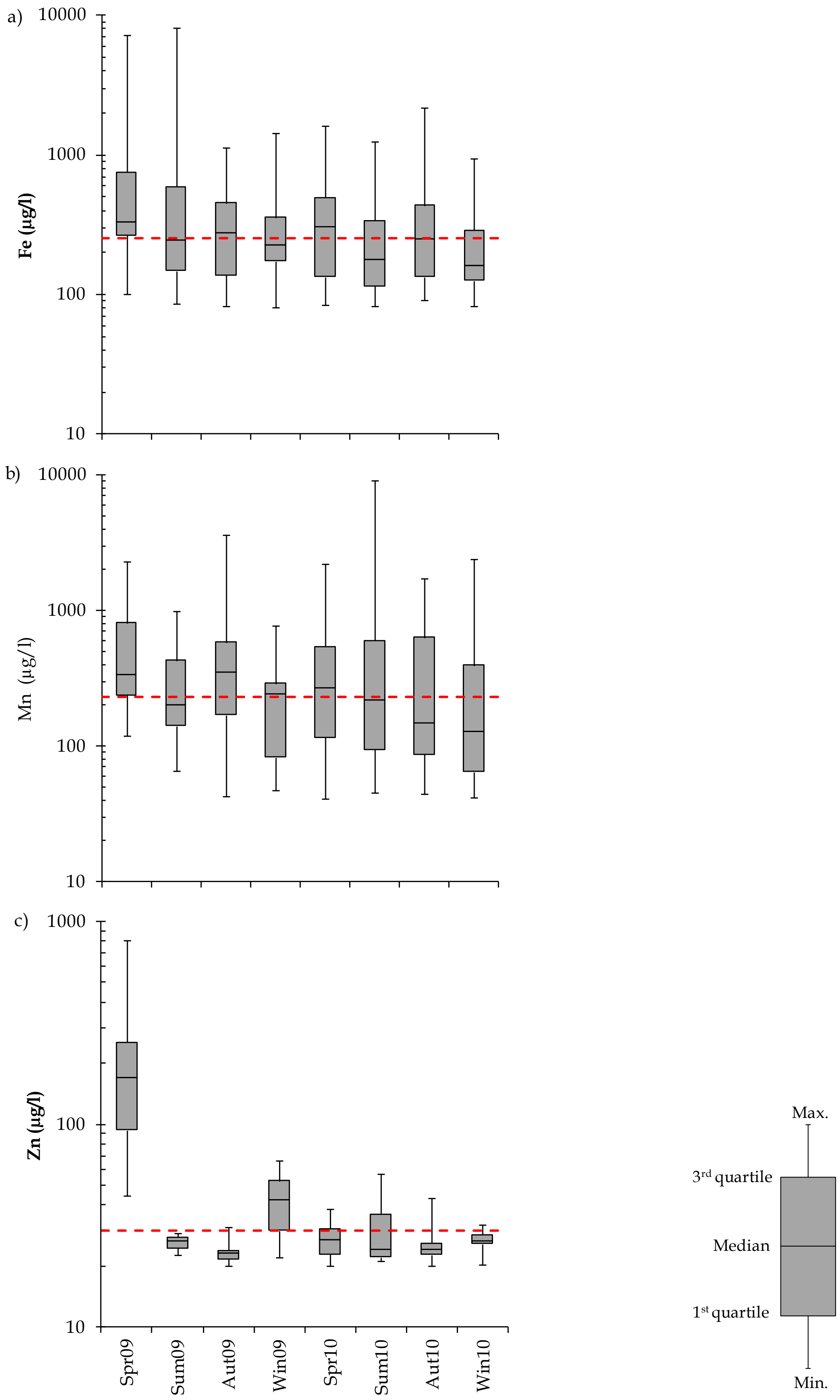
Appendix 12. Temporal variation of major cation concentrations

a) Na b) NH$_4$ c) K d) Mg e) Ca and study median (red dashed line)



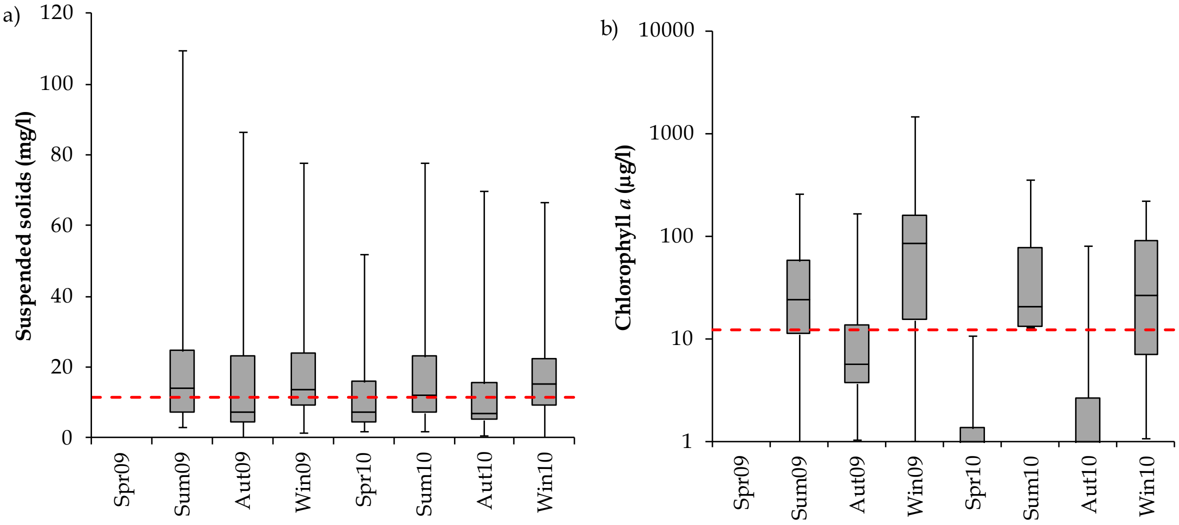
Appendix 13. Temporal variation of trace metal concentrations (for samples >LOD)

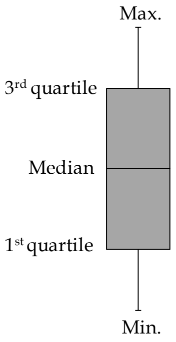
a) Fe b) Mn c) Zn and study median (red dashed line)



Appendix 14. Temporal variation of for chlorophyll *a* and suspended solids

a) Suspended solids b) Chorophyll *a* and study median (red dashed line)





Appendix 15. Pooled macrophyte species list from sampling periods, 20th May – 11th June / 2nd August – 14th August 2009.

Species	Common name	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	∑
Alisma plantago-aquatica	Water plantain														•									•		•			•			4
Apium nodiflorum	Fools watercress		•										•								•											3
Azolla filiculoides	Water fern																						•									1
Berula erecta	Water parsnip												•											•				•		•		4
Callitriche spp.	Water-starwort	•	•																		•			•	•						•	6
Callitriche stagnalis	Common water-starwort		•																		•	•	•	•					•			6
Carex spp.	Sedge	•									•								•							•			•			5
Carex riparia	Greater pond sedge																				•								•			2
Ceratophyllum spp.	Hornwart	•	•												•			•	•									•			•	7
Crassula helmsii	New Zealand Pygmyweed														•							•				•						3
Eleocharis spp.	Spike rush										•													•								2
Elodea canadensis	Canadian waterweed	•	•												•			•			•					•						6
Elodea nuttallii	Nuttalls waterweed	•													•																	2
Glyceria fluitans	Floating sweet-grass										•										•											2
Hippuris vulgaris	Mares' tail		•																			•		•						•		4
Hydrocotyle ranunculoides	Floating pennywort		•																													1
Hydrocotyle vulgaris	Marsh pennywort		•																													1
Iris pseudocorus	Yellow flag iris	•		•			•		•			•		•						•	•				•		•	•		•		12
Iris setosa	Blue flag iris								•																							1
Juncus effusus	Soft rush										•		•										•	•								4
Juncus articulatus	Jointed rush										•																					1
Lagarasiphon major	Curly water-thyme																				•					•						2
Lemna minor	Common duckweed						•			•	•	•	•					•		•	•	•	•	•				•	•	•	•	15
Lemna triscula	Ivy-leaved duckweed		•										•		•				•	•			•					•	•	•		9
Mentha aquatica	Wild mint																							•					•	•		3
Myosotis sp.	Water forget-me-not																		•					•								2
Myriophyllum aquaticum	Parrots feather																			•												1
Myriophyllum spicatum	Water milfoil		•												•							•	•									4
Nuphar lutea	Yellow pond-lily								•			•				•			•													4
Nymphaea alba	White water-lily	•	•																	•				•				•				5

Species	**Common name**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**	∑
Nymphoidea peltatus	Fringed water-lily																		●									●		●		3
Persicaria amphibia	Amphibious bistort	●	●																					●								3
Phragmites australis	Common reed	●								●			●														●			●		5
Potamogeton crispus	Curly-leaf pond weed		●															●	●						●							4
Potamogeton natans	Broad-leaved pond weed	●									●				●								●	●		●						6
Potamogeton berchtoldii	Small pondweed										●																					1
Potamogeton pusillus	Small pondweed										●																					1
Potamogeton perfoliatus	Perfoliate pondweed										●																					1
Ranunculus spp.	Water crowfoot		●																			●								●		3
Ranunculus circinatus	Fan-leaved water crowfoot		●																													1
Ranunculus peltatus	Pond water-crowfoot		●																			●	●							●		4
Ranunculus repens	Creeping buttercup																						●									1
Riccia fluitans	Liverwort																						●									1
Rorippa nasturtium aquaticum	Water cress														●						●											2
Schoenoplectus lacustris	Bulrush										●	●	●													●				●		5
Typha sp.	Reedmace		●								●		●		●							●		●		●			●	●		9
Utricularia sp.	Bladderwort										●																			●		2
V. beccabunga/ M. aquatica																								●								1
Zanichellia palustris	Horned pondweed												●														●					2
	Bryophyte 1 (small)	●	●		●	●	●		●		●	●	●		●	●		●	●		●		●	●	●					●	●	19
	Bryophyte 2 (large)					●														●				●								3
	Bryophyte 3		●																													1
	Emergent unk.					●																										1
	Unknown small broadleaved																	●														1
	Unknown #1		●							●												●										3
Veronica spp. (*Veronica anagallis-aquatica*?)	Unknown #2														●																	1
	Unknown #5																													●		1
	Unknown #6																												●			1
	Unknown #7																		●				●									2
	Unknown #8																				●	●										2
	Taxa richness:	11	20	1	1	3	3	0	4	3	14	5	10	1	12	2	0	6	9	6	12	10	12	17	4	8	3	7	9	15	4	

Appendix 16. Creation of a resistance landscape

Landscapes are rarely homogeneous and aquatic insect flights paths are likely to reflect this. The 'effective distance' (Spear *et al.* 2010), therefore, incorporates the ease at which landscape features are to traverse. Least cost path (LCP) analysis is one means by which resistance in the landscape can be incorporated into dispersal. For aquatic insect dispersal, this has rarely been considered (Fahrig 2007). However, the lack of studies incorporating LCP analysis into aquatic insects is probably due to the lack of empirical evidence to suggest the relative cost of crossing differing land-uses. Expert opinion is typically the main method employed to assign resistance scores (Spear *et al.* 2010), which may be incorporated within this study in the future. Presently, resistance scores were based on a review of the potential influence of broad land-use categories on aquatic insect dispersal, derived from scientific literature. The objective is not to delineate exact flight paths; however, the resulting non-linear LCPs provide a suggestion of how the connectivity of a landscape could change if the inter-habitat terrestrial matrix is considered.

16.1.1. Applying resistance to the landscape for aquatic insects

16.1.1.1. Surface water

Areas of surface water are intrinsically linked to aquatic insect presence, as such they were attributed a low resistance score. Many features of riparian stream corridors, such as the presence of surface water, greater soil moisture, greater humidity, taller, more dense vegetation and/or more abundant plant resources, could be predicted to contribute to greater terrestrial arthropod abundances in these areas (Lynch *et al.* 2002). Furthermore, some authors have suggested a safety in numbers aspect to surface water areas whereby the increased abundance of potential predators such as bats, birds and Odonates may be negated by mass emergence (Sweeney and Vannote 1982).

16.1.1.2. Urban areas

Urban areas comprise of a number of land-use features, such as buildings, structures, developing land, manmade open space and landforms. High resistance was attributed to urban areas, although the values were reduced where features were associated with vegetation. Direct physical pressure of traversing a landscape of tall buildings may increase, mayflies and caddisflies, for example have rarely been observed at heights of 60 - 1525m (Johnson 1969). During their upstream compensation flight *P. Longicauda* females were recorded at a maximum height of 20 – 25m and 5 – 15m (Malnas *et al.* 2011) and in an experimental setting the same species were recorded at heights of between 15 – 30m whilst in 'water-seeking' flight (Kriska *et al.* 2007). In any case, the extra energy required to scale tall buildings likely makes them a less favourable land–use to traverse, with adjacent gardens or vegetated features a more likely prospect in this regard. Tall buildings, close to small ponds, may also hinder detection of a pond habitat, as with mature vegetation (Remsburg *et al.* 2008).

Perhaps the greatest concern is the phenomenon known as 'polarized light pollution' (Horvath *et al.* 2009). Many insects identify habitats by means of a horizontal polarization of water-reflected light (Schwind 1991). Plastics, glass buildings, green metal roofs, oil lakes, and solar panels may all act as polarized ecological traps for a range of polarotactic insects including, but not restricted to mayflies, beetles and dragonflies (Horvath *et al.* 1998; Larson *et al.* 2000; Kriska *et al.* 2007; Stevens *et al.* 2007; Malik *et al.* 2008; Horvath *et al.* 2010).

Furthermore, many insects are attracted to artificial light (Schwind 1991; Kovats *et al.* 1996). Consequently street lighting may also create an ecological trap. This may be considerable in dawn emergences such as some *Caenis spp.* (Bradbeer and Savage 1980). In the study by Kovats et al. (1996), low catches in light traps at one location was related to the traps proximity to a crossroads illuminated by brighter streetlights than that of the light trap itself. Greater attraction of females by ultra-violet light has been reported for *Hexagenia spp.* mayfly adults (Hunt 1953), which may have significant implications for egg dispersal.

16.1.1.3. Roads

Similar to the surfaces noted above, interpretation of road surfaces as habitat for mayflies has long been recognised (Ladócsy 1930), for example, oviposition on wet and dry asphalt (Malnas *et al.* 2011). Similar behaviour has been exhibited by Coleoptera (Larson *et al.* 2000) and the metal exteriors of cars, and cars of dark and red colours, have been found to attract aquatic insects (Larson *et al.* 2000; Kriska *et al.* 2006a). The effect of the construction of a bridge over a stream was found to act as a barrier to compensation flight in a mayfly species (*Palingenia longicauda*) due to its polarizing properties. The impact was particularly notable to females of the species, of which 86% were impacted (Malnas *et al.* 2011).

Traffic may also be a direct source of mortality, particularly for Odonates (Rao and Girish 2007). Research by Soluk *et al.* (2011) found fatality following collision with road traffic of two species of Odonates (*Plathemis lydia* and *Libellula luctuosa*) accounted for 14 and 31% of Odonate activity. In a similar study in Austria, collision risk was calculated to be between 5.9 to 7.6% for *S. vulgatum* and *S. striolatum* at a motorway feeder road (Staufer 2010).

16.1.1.4. Vegetated areas

Vegetation can provide food, shelter from the elements, protection from predators and resting places, for example for mayfly maturation (Kovats *et al.* 1996). Where trees occur, the atmospheric boundary layer (ABL) is thicker (Jackson and Resh 1989) and may consequently aid insect movement. Although, thickening of the ABL may also occur around buildings and other structures.

Some stonefly species have been found more numerous in deciduous woodland than on open heath land (Petersen *et al.* 1999) and dense riparian forest revealed the highest densities of caddis fly (Collier and Smith 1998). This suggests the importance of riparian forest for Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa but mostly refer to the riparian vegetation in close proximity to the stream edge rather than between habitats. Various caddis fly families were found up to 200m inland within riparian forest (Collier and Smith 1998), relatively small by comparison to open cropland (Kovats *et al.* 1996). However, Petersen *et al.* (2004) found no differences in the decline of catch with distance from stream between a coniferous plantation forest, cleared forest and moorland, although fewer individuals dispersed a substantial distance in forested catchments than in the cleared or moorland one. In densely vegetated landscapes, adult chironomids appeared to be mainly confined to the stream from which they emerged (Delettre and Morvan 2000). Relatively open forest, however, was more likely to result in large dispersal distance by caddisflies (Jackson and Resh 1989). Large dispersal distances, by fewer individuals of caddis fly across cropland has also been found (Kovats *et al.* 1996). Furthermore, in open landscapes, chironomid species assemblages were maintained when compared to those in dense riparian vegetation (Delettre and Morvan 2000).

The degree of vegetation appears to be influential in structuring species distribution in the landscape (Collier and Smith 1998; Delettre and Morvan 2000). Dense woodland may provide shelter, food and protection, however, this appears suited for those engaged in mating and maturation processes rather than in the act of dispersal. Greater dispersal distances, although lower densities, have typically been noted in more open environments. Logically, a compromise between fully cleared grassland and dense woodland would appear to be the more suited land-use to dispersal providing food, shelter and protection, but less physical obstruction, for example, the presence of hedgerows for chironomidae (Delettre and Morvan 2000). Higher temperatures from reduced cover may also promote greater dispersal distances due to metabolic effects.

16.1.1.5. Agricultural land

Intensive agricultural land has been shown to increase resistance between two patches functionally connected by distance alone, i.e. known to be within dispersal abilities of a focal species (*C. mercuriale*, Rouquette and Thompson 2007). Furthermore, the presence of pesticides and floral monocultures across much agricultural landscape may hinder insect dispersal. Nonetheless, field margins may provide more naturalised dispersal corridors, although overall habitat heterogeneity may be reduced (Curado *et al.* 2011).

16.1.2.Creating a land-use mosaic for Birmingham

The first step towards LCP analysis is the formulation of a land-use mosaic for the study area to which resistance scores can be attributed. This was achieved by combining the four GIS layers (refer to Section 2.3), in raster format, within a GIS using the raster calculator function within ArcGIS 9.3 (Figure A7.4.1) and a sequence of spatial calculations and reclassifications. Despite a complicated process, a simple objective was retained throughout, whereby each land-use combination was kept unique, such that it was possible to reclassify according to the desired resistance scores.

16.1.2.1. Land-use resistance scores

Forty-five different land-use categories, each with its own characteristics and potentially differing resistance to adult aquatic insects dispersal were established. The subsequent costs applied are not absolute values, but are better served as relative costs to aquatic insect dispersal.

Due to computational limitations, it was not possible to compute least-cost paths across the entire Birmingham landscape unless a coarse resolution was used. Therefore, LCP analysis was undertaken within the maximum spatial extent required for connectivity analyses (Section 5.2.4.2). To improve computational efficiency the resistance raster was clipped to the maximum spatial extent and resampled from a 2m x 2m resolution to a 4m x 4m using bilinear interpolation, by which four 2m x 2m cells were averaged within a single 4m x 4m cell. This remains a resolute representation of the terrestrial matrix when compared to other studies (e.g. 250m, Gurrutxaga *et al.* 2011), although it was accepted that an element of data smoothing would occur.

Effective distances between ponds were calculated in Pathmatrix 1.1 (Ray 2005) within ArcGIS 3.2. In doing so, a LCP is plotted as well and the accumulated cost along that path is calculated.

Figure A7.4.1 Land-use mosaic creation process in ArcGIS for resistance landscape mapping

Key to raster groups

R1 – OS MasterMap

R1a
builds & gspace = 0
Other values as specified[1]
R1b
gspace =1
other values = 0
R1c
builds =1
other values = 0

R2 – Photogrammetry

R2a
<7m = 500
>7m = 1000
R2b
0m = 1
0≤3m = 2
>3 ≤60m = 3
>60m = 4

R3 – NDVI

R3a
vegetation = 2
non-vegetation = 1

R4 – LCM2007

R4a (trees)
coniferous = 3
broadleaved = 2
none = 1
R4b
woodland = 1000
other values as specified[1]

[1] Unique values attributed to each OS MasterMap or LCM2007 feature prior to processing

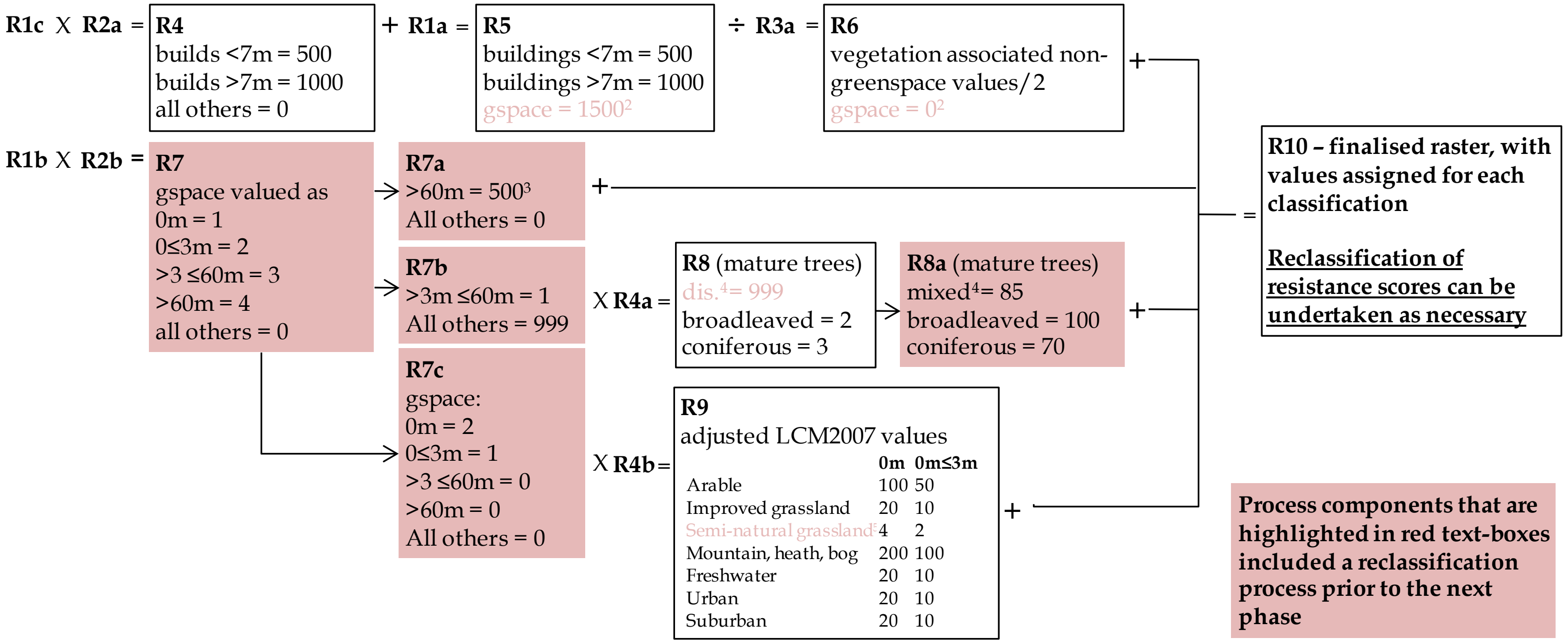
[2] Greenspace reclassified as 1500 to avoid loss of definition against other land-use classes and a NoData result following division by R3a. Values of 1500 and 750 within R6 then reclassified to 0.

[3] Here where NDVI registered green at a height >60m (UK max. tree height), it was presumed green atop a building and classified accordingly

[4] Where discrepancies arose between LCM2007 not identifying a wooded area and a combination of NDVI and photogrammetry suggested woodland, the latter was preferred and classified as mixed woodland

[5] Where LCM2007 suggests woodland, but not by NDVI and height combined, classified to low level and ground level semi natural grassland. Also includes rough low productivity grassland

Land-use mosaic creation process undertaken in raster calculator



R1c X **R2a** = **R4**
builds <7m = 500
builds >7m = 1000
all others = 0

+ **R1a** = **R5**
buildings <7m = 500
buildings >7m = 1000
gspace = 1500[2]

÷ **R3a** = **R6**
vegetation associated non-greenspace values/2
gspace = 0[2]

R1b X **R2b** = **R7**
gspace valued as
0m = 1
0≤3m = 2
>3 ≤60m = 3
>60m = 4
all others = 0

R7a
>60m = 500[3]
All others = 0

R7b
>3m ≤60m = 1
All others = 999

X **R4a** = **R8** (mature trees)
dis.[4]= 999
broadleaved = 2
coniferous = 3

R8a (mature trees)
mixed[4]= 85
broadleaved = 100
coniferous = 70

R7c
gspace:
0m = 2
0≤3m = 1
>3 ≤60m = 0
>60m = 0
All others = 0

X **R4b** = **R9**
adjusted LCM2007 values

	0m	**0m≤3m**
Arable	100	50
Improved grassland	20	10
Semi-natural grassland[5]	4	2
Mountain, heath, bog	200	100
Freshwater	20	10
Urban	20	10
Suburban	20	10

= **R10 – finalised raster, with values assigned for each classification**

Reclassification of resistance scores can be undertaken as necessary

Process components that are highlighted in red text-boxes included a reclassification process prior to the next phase

Table A16.1 Resistance scores applied to 45 classes of land-use

Land-use	Resistance score
Non-residential buildings (>7m)	1000
Structures	1000
Motorway	600
Residential buildings (<7m)	500
A road	500
Vegetated non-residential buildings (>7m)	500
Vegetated structures	400
B road	300
Minor road	300
Vegetated motorway	300
Developing land	250
Steps	250
Pavements	250
Paths	250
Manmade open space	250
Manmade landforms	250
Vegetated residential buildings	250
Vegetated A road	250
Railway	200
Vegetated B road	200
Vegetated minor road	150
Vegetated developing land	125
Vegetated steps	125
Vegetated pavements	125
Vegetated paths	125
Vegetated manmade open space	125
Vegetated manmade landforms	125
Vegetated railway	100
Broadleaved woodland	100
Arable (0m)	100
Mixed woodland	85
Coniferous woodland	70
Gardens	50
Arable (<3m)	50
Vegetated gardens	25
Improved grassland (0m)	20
Freshwater	20
Semi-natural grassland (0m)	16
Improved grassland (<3m)	10
Semi-natural grassland (scrub, <3m)	10
Freshwater (riparian vegetated, <3m)	8
Canals and rivers	2
Still waters	2
Vegetated canals and rivers	1
Vegetated still water	1

Appendix 17. JNCC threat category definitions and criteria, after Chadd and Extence (2004)

Red Data Book 1 (RDB1): Endangered

Taxa in danger of extinction, the survival of which is unlikely if causal factors to its decline continue to operate. Taxa whose numbers have been reduced to a critical level or whose habitats have been so dramatically reduced that they are deemed to be in immediate danger of extinction. Included are taxa that are known only as a single population in only one 10 km square, taxa that only occur in habitats known to be especially vulnerable, or taxa that have shown a continuous decline over the last 20 years and now exist in five or fewer 10 km squares.

Red Data Book 2 (RDB2): Vulnerable

Taxa believed likely to designated RDB1 in the near future. Included are taxa of which most or all of the populations are decreasing because of overexploitation, extensive destruction of habitat or other environmental disturbance. Also included are taxa with populations that have been seriously depleted, the ultimate security of which is not yet assured. Finally, taxa with populations that may still be abundant but which are under threat from serious adverse factors throughout their range.

Red Data Book 3 (RDB3): Rare

Taxa with small populations, which are risk, but not at present designated as RDB1 or RDB2. RDB3 taxa are usually localized within restricted geographical areas or habitats, or are thinly scattered over a more extensive range. Usually, such taxa are not likely to exist in more than 15 x 10 km squares of the National Grid. This criterion may be relaxed where populations are likely to exist in more than 15 x 10km squares but occupy small areas of especially vulnerable habitat.

Notable A (NA) and Notable B (NB)

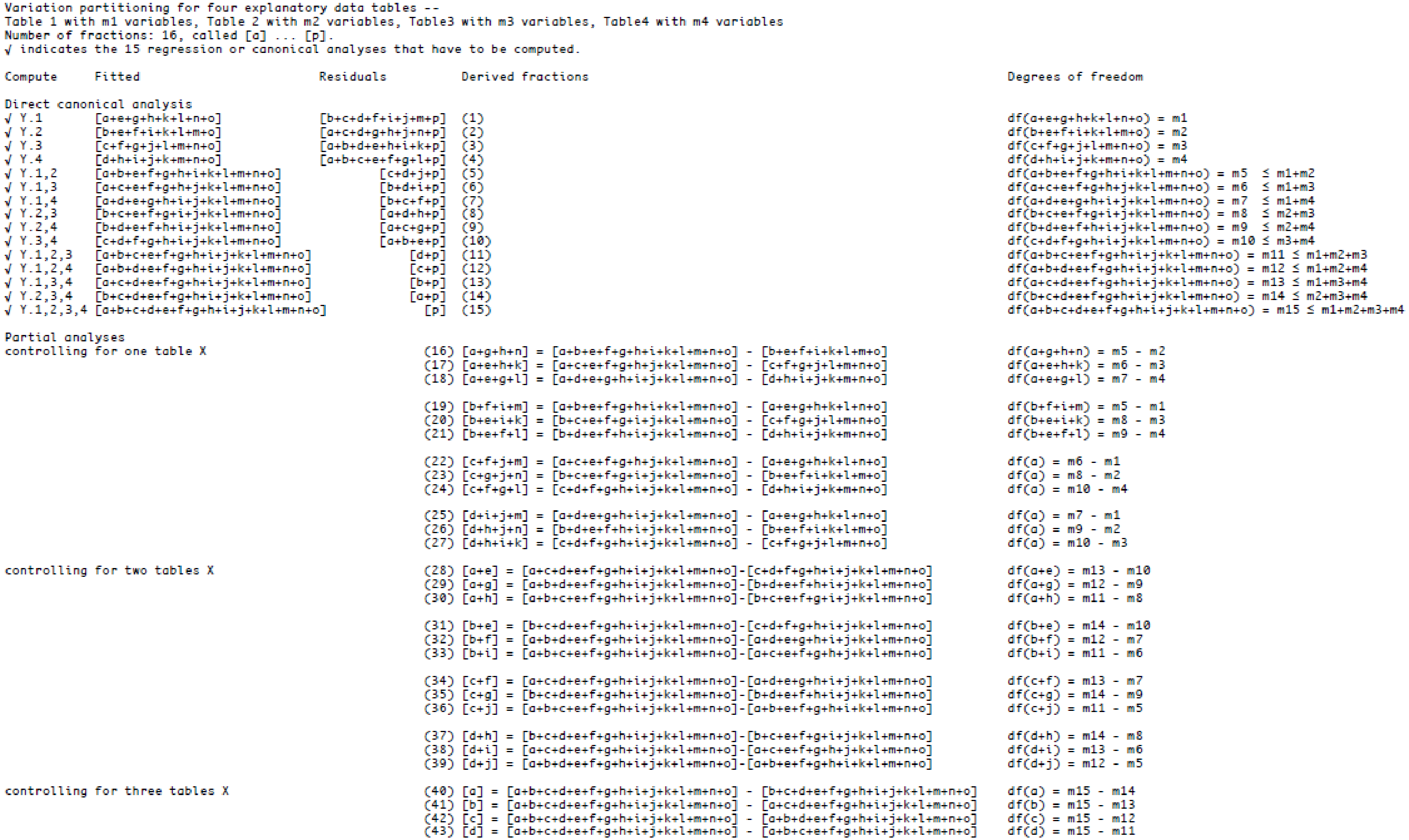
Taxa that do not fall within RDB categories 1–3 but which are nonetheless scarce in the United Kingdom. For some well-recorded groups of invertebrates (e.g. Coleoptera) this category has been subdivided; Notable A taxa are present in 30 or fewer 10 km squares, Notable B taxa are present in 31 to 100 x 10 km squares nationwide

Notable Regional (NR)

Taxa that are too common nationally to fall within the Notable category, but which are uncommon in some parts of the country. ‘Uncommon’, in this case, means found in five or fewer localities. The region to which this status applies is described for individual species.

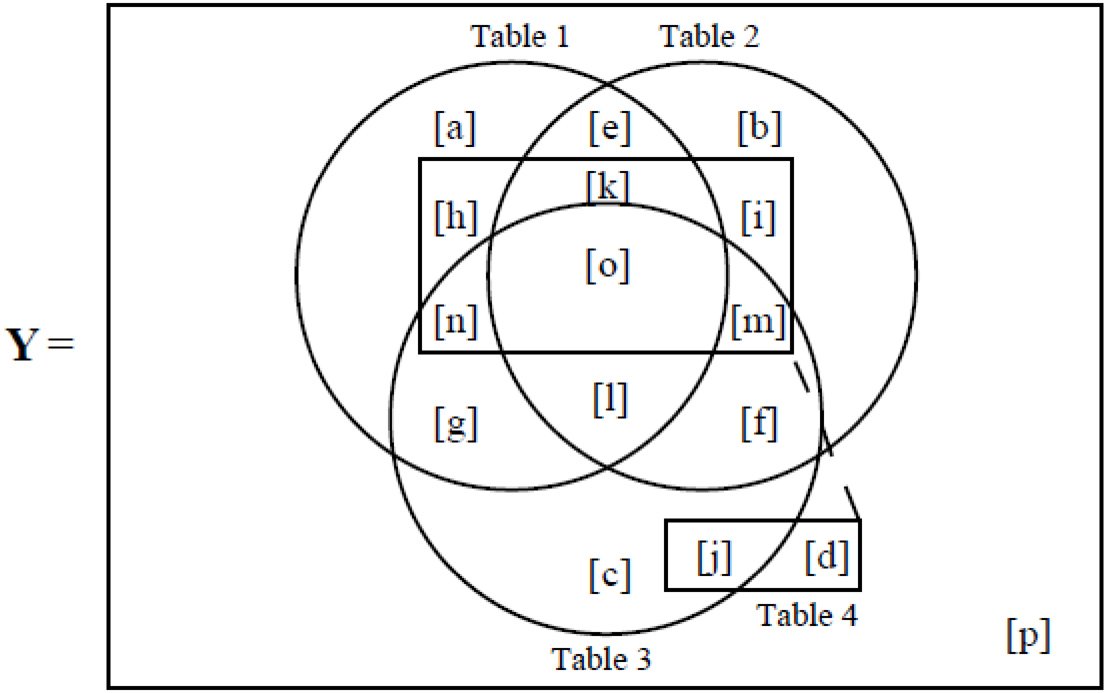
Appendix 18. Formulas for calculation of explained variance fraction (varpart procedure Oksanen et al. 2012)

Matrix Y = Assemblage structure, a = water quality variables, b = habitat characteristics, c = land-use, d = connectivity metrics



```
Variation partitioning for four explanatory data tables --
Table 1 with m1 variables, Table 2 with m2 variables, Table3 with m3 variables, Table4 with m4 variables
Number of fractions: 16, called [a] ... [p].
√ indicates the 15 regression or canonical analyses that have to be computed.

Compute     Fitted                        Residuals         Derived fractions                                                        Degrees of freedom

Direct canonical analysis
√ Y.1       [a+e+g+h+k+l+n+o]             [b+c+d+f+i+j+m+p] (1)                                                                       df(a+e+g+h+k+l+n+o) = m1
√ Y.2       [b+e+f+i+k+l+m+o]             [a+c+d+g+h+j+n+p] (2)                                                                       df(b+e+f+i+k+l+m+o) = m2
√ Y.3       [c+f+g+j+l+m+n+o]             [a+b+d+e+h+i+k+p] (3)                                                                       df(c+f+g+j+l+m+n+o) = m3
√ Y.4       [d+h+i+j+k+m+n+o]             [a+b+c+e+f+g+l+p] (4)                                                                       df(d+h+i+j+k+m+n+o) = m4
√ Y.1,2     [a+b+e+f+g+h+i+k+l+m+n+o]             [c+d+j+p] (5)                                                                       df(a+b+e+f+g+h+i+k+l+m+n+o) = m5  ≤ m1+m2
√ Y.1,3     [a+c+e+f+g+h+j+k+l+m+n+o]             [b+d+i+p] (6)                                                                       df(a+c+e+f+g+h+j+k+l+m+n+o) = m6  ≤ m1+m3
√ Y.1,4     [a+d+e+g+h+i+j+k+l+m+n+o]             [b+c+f+p] (7)                                                                       df(a+d+e+g+h+i+j+k+l+m+n+o) = m7  ≤ m1+m4
√ Y.2,3     [b+c+e+f+g+i+j+k+l+m+n+o]             [a+d+h+p] (8)                                                                       df(b+c+e+f+g+i+j+k+l+m+n+o) = m8  ≤ m2+m3
√ Y.2,4     [b+d+e+f+h+i+j+k+l+m+n+o]             [a+c+g+p] (9)                                                                       df(b+d+e+f+h+i+j+k+l+m+n+o) = m9  ≤ m2+m4
√ Y.3,4     [c+d+f+g+h+i+j+k+l+m+n+o]             [a+b+e+p] (10)                                                                      df(c+d+f+g+h+i+j+k+l+m+n+o) = m10 ≤ m3+m4
√ Y.1,2,3   [a+b+c+e+f+g+h+i+j+k+l+m+n+o]             [d+p] (11)                                                                      df(a+b+c+e+f+g+h+i+j+k+l+m+n+o) = m11 ≤ m1+m2+m3
√ Y.1,2,4   [a+b+d+e+f+g+h+i+j+k+l+m+n+o]             [c+p] (12)                                                                      df(a+b+d+e+f+g+h+i+j+k+l+m+n+o) = m12 ≤ m1+m2+m4
√ Y.1,3,4   [a+c+d+e+f+g+h+i+j+k+l+m+n+o]             [b+p] (13)                                                                      df(a+c+d+e+f+g+h+i+j+k+l+m+n+o) = m13 ≤ m1+m3+m4
√ Y.2,3,4   [b+c+d+e+f+g+h+i+j+k+l+m+n+o]             [a+p] (14)                                                                      df(b+c+d+e+f+g+h+i+j+k+l+m+n+o) = m14 ≤ m2+m3+m4
√ Y.1,2,3,4 [a+b+c+d+e+f+g+h+i+j+k+l+m+n+o]             [p] (15)                                                                      df(a+b+c+d+e+f+g+h+i+j+k+l+m+n+o) = m15 ≤ m1+m2+m3+m4

Partial analyses
controlling for one table X                             (16) [a+g+h+n] = [a+b+e+f+g+h+i+k+l+m+n+o] - [b+e+f+i+k+l+m+o]                df(a+g+h+n) = m5 - m2
                                                        (17) [a+e+h+k] = [a+c+e+f+g+h+j+k+l+m+n+o] - [c+f+g+j+l+m+n+o]                df(a+e+h+k) = m6 - m3
                                                        (18) [a+e+g+l] = [a+d+e+g+h+i+j+k+l+m+n+o] - [d+h+i+j+k+m+n+o]                df(a+e+g+l) = m7 - m4

                                                        (19) [b+f+i+m] = [a+b+e+f+g+h+i+k+l+m+n+o] - [a+e+g+h+k+l+n+o]                df(b+f+i+m) = m5 - m1
                                                        (20) [b+e+i+k] = [b+c+e+f+g+i+j+k+l+m+n+o] - [c+f+g+j+l+m+n+o]                df(b+e+i+k) = m8 - m3
                                                        (21) [b+e+f+l] = [b+d+e+f+h+i+j+k+l+m+n+o] - [d+h+i+j+k+m+n+o]                df(b+e+f+l) = m9 - m4

                                                        (22) [c+f+j+m] = [a+c+e+f+g+h+j+k+l+m+n+o] - [a+e+g+h+k+l+n+o]                df(a) = m6 - m1
                                                        (23) [c+g+j+n] = [b+c+e+f+g+i+j+k+l+m+n+o] - [b+e+f+i+k+l+m+o]                df(a) = m8 - m2
                                                        (24) [c+f+g+l] = [c+d+f+g+h+i+j+k+l+m+n+o] - [d+h+i+j+k+m+n+o]                df(a) = m10 - m4

                                                        (25) [d+i+j+m] = [a+d+e+g+h+i+j+k+l+m+n+o] - [a+e+g+h+k+l+n+o]                df(a) = m7 - m1
                                                        (26) [d+h+j+n] = [b+d+e+f+h+i+j+k+l+m+n+o] - [b+e+f+i+k+l+m+o]                df(a) = m9 - m2
                                                        (27) [d+h+i+k] = [c+d+f+g+h+i+j+k+l+m+n+o] - [c+f+g+j+l+m+n+o]                df(a) = m10 - m3

controlling for two tables X                            (28) [a+e] = [a+c+d+e+f+g+h+i+j+k+l+m+n+o]-[c+d+f+g+h+i+j+k+l+m+n+o]          df(a+e) = m13 - m10
                                                        (29) [a+g] = [a+b+d+e+f+g+h+i+j+k+l+m+n+o]-[b+d+e+f+h+i+j+k+l+m+n+o]          df(a+g) = m12 - m9
                                                        (30) [a+h] = [a+b+c+e+f+g+h+i+j+k+l+m+n+o]-[b+c+e+f+g+i+j+k+l+m+n+o]          df(a+h) = m11 - m8

                                                        (31) [b+e] = [b+c+d+e+f+g+h+i+j+k+l+m+n+o]-[c+d+f+g+h+i+j+k+l+m+n+o]          df(b+e) = m14 - m10
                                                        (32) [b+f] = [a+b+d+e+f+g+h+i+j+k+l+m+n+o]-[a+d+e+g+h+i+j+k+l+m+n+o]          df(b+f) = m12 - m7
                                                        (33) [b+i] = [a+b+c+e+f+g+h+i+j+k+l+m+n+o]-[a+c+e+f+g+h+j+k+l+m+n+o]          df(b+i) = m11 - m6

                                                        (34) [c+f] = [a+c+d+e+f+g+h+i+j+k+l+m+n+o]-[a+d+e+g+h+i+j+k+l+m+n+o]          df(c+f) = m13 - m7
                                                        (35) [c+g] = [b+c+d+e+f+g+h+i+j+k+l+m+n+o]-[b+d+e+f+h+i+j+k+l+m+n+o]          df(c+g) = m14 - m9
                                                        (36) [c+j] = [a+b+c+e+f+g+h+i+j+k+l+m+n+o]-[a+b+e+f+g+h+i+k+l+m+n+o]          df(c+j) = m11 - m5

                                                        (37) [d+h] = [b+c+d+e+f+g+h+i+j+k+l+m+n+o]-[b+c+e+f+g+i+j+k+l+m+n+o]          df(d+h) = m14 - m8
                                                        (38) [d+i] = [a+c+d+e+f+g+h+i+j+k+l+m+n+o]-[a+c+e+f+g+h+j+k+l+m+n+o]          df(d+i) = m13 - m6
                                                        (39) [d+j] = [a+b+d+e+f+g+h+i+j+k+l+m+n+o]-[a+b+e+f+g+h+i+k+l+m+n+o]          df(d+j) = m12 - m5

controlling for three tables X                          (40) [a] = [a+b+c+d+e+f+g+h+i+j+k+l+m+n+o] - [b+c+d+e+f+g+h+i+j+k+l+m+n+o]    df(a) = m15 - m14
                                                        (41) [b] = [a+b+c+d+e+f+g+h+i+j+k+l+m+n+o] - [a+c+d+e+f+g+h+i+j+k+l+m+n+o]    df(b) = m15 - m13
                                                        (42) [c] = [a+b+c+d+e+f+g+h+i+j+k+l+m+n+o] - [a+b+d+e+f+g+h+i+j+k+l+m+n+o]    df(c) = m15 - m12
                                                        (43) [d] = [a+b+c+d+e+f+g+h+i+j+k+l+m+n+o] - [a+b+c+e+f+g+h+i+j+k+l+m+n+o]    df(d) = m15 - m11
```

```
Fractions estimated by subtraction          (44) [e] = [a+e] - [a]                                        df(e) = m1-m1 = 0
(cannot be tested)                          (45) [f] = [b+f] - [b]                                        df(f) = m2-m2 = 0
                                            (46) [g] = [a+g] - [a]                                        df(g) = m1-m1 = 0
                                            (47) [h] = [a+h] - [a]                                        df(h) = m1-m1 = 0
                                            (48) [i] = [b+i] - [b]                                        df(i) = m2-m2 = 0
                                            (49) [j] = [c+j] - [c]                                        df(j) = m3-m3 = 0
                                            (50) [k] = [a+e+h+k] - [a+e] - [h]                            df(k) = m1-m1-0 = 0
                                            (51) [l] = [a+e+g+l] - [a+e] - [g]                            df(l) = m1-m1-0 = 0
                                            (52) [m] = [b+f+i+m] - [b+f] - [i]                            df(m) = m2-m2-0 = 0
                                            (53) [n] = [a+g+h+n] - [a+g] - [h]                            df(n) = m1-m1-0 = 0
                                            (54) [o] = [a+e+g+h+k+l+n+o] - [a+e+h+k] - [g] - [l] - [n]    df(o) = m1-m1-0-0-0 = 0
                                            (55) [p] = residuals = 1 - [a+b+c+d+e+f+g+h+i+j+k+l+m+n+o]    df2(p) = n-1-m15
```



The contributions of Table 4 are represented by the two rectangles

15 regression/canonical analyses and 27 subtraction equations are needed to estimate the 16 (= 2^4) fractions.

[a] to [d] and subsets containing [a] to [d] can be tested for significance (5 canonical analyses per permutation to test [a] to [d]). Fractions [e] to [o] cannot be tested singly.

Appendix 19. Selection probabilities of explanatory variables for macroinvertebrate orders within local and regional factors sets

Table A19.1. Model selection probabilities for water quality variables for predicting macroinvertebrate richness, calculated as the sum of GLM model weights (*w*AIC$_c$) in which each explanatory variable occurs. Selection probabilities >0.7 are highlighted in bold.

Order	DO	K	Mg	Ca	PO_4	SO_4	NH_4	$CaCO_3$	Fe	Mn
Gastropoda	0.07	0.63	0.09	0.07	**1.00**	0.05	0.17	0.06	**0.71**	0.06
Coleoptera	0.16	0.15	0.23	0.16	0.59	0.23	0.32	0.42	0.14	0.14
Trichoptera	0.06	0.04	0.03	0.03	**0.98**	0.03	**0.98**	0.04	**0.74**	0.04
Odonata	0.27	0.11	0.07	0.08	**1.00**	0.14	0.62	0.28	0.32	0.03
Hemiptera	0.49	0.09	0.11	0.35	0.12	0.10	0.16	**0.76**	0.38	0.20
Taxa	0.06	0.08	0.13	0.06	**0.96**	0.05	**0.84**	0.60	0.06	0.06

Table A19.2. Model selection probabilities for local physical factors for predicting macroinvertebrate richness, calculated as the sum of GLM model weights (*w*AIC$_c$) in which each explanatory variable occurs. Selection probabilities >0.7 are highlighted in bold.

Order	%Con	%Tree	%Rip	%Flt	Mphyte	Spr.	RunOff	RdBld	Str.	WLFI	Fish	Chl*a*	SS
Gastropoda	0.07	0.11	0.06	0.07	**0.99**	0.06	0.09	0.09	0.06	0.10	**0.97**	0.06	0.09
Coleoptera	0.40	0.05	0.04	0.07	**0.99**	0.05	0.10	0.06	0.06	0.26	**0.71**	0.05	0.05
Trichoptera	0.17	0.35	0.03	0.25	**0.95**	0.07	0.08	0.10	0.04	0.04	**0.78**	0.04	0.04
Odonata	0.24	0.53	0.18	0.08	**1.00**	0.06	0.02	0.29	0.40	0.03	0.02	0.04	0.02
Hemiptera	0.13	**0.92**	0.19	0.12	0.12	0.12	0.13	0.18	0.14	0.12	0.23	0.12	0.12
Taxa	0.30	0.36	0.10	0.19	**0.99**	0.30	0.08	0.07	0.07	0.08	0.09	0.08	0.08

%Con Percentage of concrete edge, *%Tree* Percentage tree cover over pond, *%Rip* Percentage of riparian vegetation cover over pond, *%Flt* Percentage of floating vegetation cover over pond, *Mphyte* Macrophytes species richness, *Spr.* Spring or groundwater source, *RunOff* Surface water run-off source, *RdBld* Road or building run-off source, *Str.* Stream water source, *WLFI* Wetland Fluctuation Index, *Fish* Fish presence/absence, *Chla* Average chlorophyll *a* concentration, *SS* Average suspended solids load.

Table A19.3. Model selection probabilities for land-use variables for predicting macroinvertebrate richness, calculated as the sum of GLM model weights ($w\mathrm{AIC}_c$) in which each explanatory variable occurs. Selection probabilities >0.7 are highlighted in bold. Distance (m) is the spatial extent at which the most parsimonious (by AIC_c) model was found.

Order	Distance (m)	TH	Gard.	IG	Tree	PCA1	PCA2	$E_{1/D}$	Grass
Gastropoda	200m	0.16	-	0.23	0.38	**1.00**	0.15	0.16	0.36
Coleoptera	500m	0.14	-	0.09	**0.80**	**0.95**	0.13	0.40	0.26
Trichoptera	200m	0.14	-	0.16	0.52	**1.00**	0.27	0.16	0.24
Odonata	100m	0.03	-	0.00	**0.98**	**1.00**	0.01	0.07	**0.82**
Hemiptera	25m	0.20	0.21	0.18	**0.95**	0.42	0.18	0.20	-
Taxa	200m	0.06	-	0.07	**0.98**	**1.00**	0.05	0.05	0.64

TH Distance to nearest town hall, *Gard.* Proportional coverage of gardens, *IG* Proportional coverage of improved grassland, *Tree* Proportional coverage of mature vegetation, *PCA1* Urbanisation, *PCA2* Social gradient (housing density, percentage people in rural employment), $E_{1/D}$ Index of land-use heterogeneity, *Grass* Proportional coverage of all ground level vegetation.

Table A19.4. Model selection probabilities for connectivity metrics for predicting macroinvertebrate richness, calculated as the sum of GLM model weights ($w\mathrm{AIC}_c$) in which each explanatory variable occurs. Selection probabilities >0.7 are highlighted in bold. Distance (m) is the spatial extent at which the most parsimonious (by AIC_c) model was found.

Order	Distance (m)	Pond	Water	APNN	CNN	AC2km
Gastropoda	250m	0.23	**0.89**	0.24	0.55	0.53
Coleoptera	2500m	**0.99**	0.20	0.20	0.37	0.37
Trichoptera	500m	-	**0.87**	0.29	0.48	0.60
Odonata	250m	0.44	**1.00**	0.09	0.08	**0.95**
Hemiptera	2500m	**0.75**	0.28	0.27	0.29	0.27
Taxa	200m	**0.88**	0.19	0.19	0.43	**0.76**

Pond Proportional coverage of pond habitat, *Water* Proportional coverage of water, *APNN* Distance to nearest neighbour weighted by neighbour surface area, *CNN* Effective cost to nearest neighbour, *AC2km* Effective cost weighted by surface area of all pond habitat within 2km of focal pond

Appendix 20. Pooled macroinvertebrate species list from sampling periods, 20th May – 11th June / 2nd August – 14th August 2009.

Symbols represent proportional abundance **● >25% ○ > 1 < 25% ▪ <1% Blank or dash = not present**

Order/Family	Species Site no:	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
Amphipoda																															
Crangonyctidae	*Crangonyx pseudogracilis*	○/○	○/○	○/○			▪/▪	○/▪	▪/○	○/○	▪/▪	▪/▪	●/○	○/●	○/○		▪/-	○/○	○/○	○/○	▪/○	○/▪	○/○	○/○	○/▪	▪/▪	▪/○	○/▪	○/○	○/○	○/○
Gammaridae	*Gammarus lacustris*																				▪/-			▪/▪		-/▪					▪/-
Gammaridae	*Gammarus pulex*																	▪/○	▪/-		▪/○	-/▪		▪/▪		▪/▪					▪/▪
Annelida																															
Oligochaeta	*Oligochaeta*	○/●	○/○	●/○	○/○	▪/●	●/○	▪/○	○/○	○/○	○/○	○/●	-/▪	○/○	○/○	○/●	○/▪	▪/▪	○/○	○/●	●/○	○/○	▪/○	○/○	○/●	●/●	○/●	●/○	○/▪	○/○	○/-
Bivalvia																															
Sphaeriidae	*Sphaeriidae*	▪/○	▪/▪	○/-		▪/-	▪/-		▪/○	○/▪	-/▪	-/▪	▪/-	-/▪	▪/-	-/▪	▪/●	○/○	○/○	○/○	○/○	○/○	○/○	○/▪	○/▪		-/▪	▪/▪	▪/-	○/○	○/-
Coleoptera																															
-	*Larvae*	-/○	▪/-		-/○	○/▪		-/▪	▪/▪	▪/▪	○/▪	▪/○	▪/▪	▪/▪	▪/-		-/▪	▪/▪	▪/▪	▪/▪	○/▪	○/▪	▪/○	○/-	▪/○	▪/▪	-/▪	▪/○	○/▪	▪/▪	▪/-
Dytiscidae	*Acilius sulcatus*					▪/-								-/▪						-/▪			-/▪								
Dytiscidae	*Agabus bipustulatus*		-/▪								▪/-												▪/-				▪/-		▪/▪		
Dytiscidae	*Agabus sturmii*																			▪/-	▪/▪			-/▪					-/▪		
Dytiscidae	*Colymbetes fuscus*													-/▪						▪/▪									-/▪		-/▪
Dytiscidae	*Dytiscus marginalis*																									-/▪					-/▪
Dytiscidae	*Hydroglyphus geminus*						▪/-	▪/-																							
Dytiscidae	*Hydroporus angustatus*																			▪/-									▪/-		
Dytiscidae	*Hydroporus palustris*				▪/-	▪/▪								▪/▪									▪/-	▪/▪					▪/-	▪/▪	▪/-
Dytiscidae	*Hydroporus planus*											-/▪									▪/-								-/▪		
Dytiscidae	*Hydroporus pubescens*																												▪/-		

Order/Family	**Species** **Site no:**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**
Dytiscidae	*Hygrotus impressopunctatus*																					▪/▪									
Dytiscidae	*Hygrotus inaequalis*						▪/-					▪/-	▪/▪							▪/▪		○/▪	-/▪							▪/▪	
Dytiscidae	*Hygrotus versicolor*		-/▪																												
Dytiscidae	*Hyphydrus ovatus*												▪/▪							▪/-				▪/▪						▪/▪	
Dytiscidae	*Ilybius ater*												-/▪							-/▪	-/▪										
Dytiscidae	*Ilybius fuliginosus*																				-/▪								▪/-		
Dytiscidae	*Ilybius quadriguttatus*												-/▪									-/▪									
Dytiscidae	*Laccophilus hyalinus*																							-/▪							
Dytiscidae	*Laccophilus minutus*		▪/-										▪/▪										▪/▪								
Dytiscidae	*Rhantus exsoletus*																						▪/▪								
Dytiscidae	*Rhantus suturalis*																												▪/-		
Dytiscidae	*Suphrodytes dorsalis*												▪/-																		
Gyrinidae	*Gyrinus substriatus*								-/▪																						
Haliplidae	*Haliplus confinis*		-/▪										▪/-																		
Haliplidae	*Haliplus immaculatus*												-/○						▪/▪					▪/▪				▪/-			
Haliplidae	*Haliplus lineatocollis*		-/○																											-/▪	
Haliplidae	*Haliplus lineolatus*																		▪/-					▪/-							
Haliplidae	*Haliplus ruficollis*	▪/▪	▪/○										○/○	-/▪	▪/▪				▪/▪	○/▪		▪/▪	-/▪	▪/▪		-/▪		▪/▪	▪/-	▪/▪	
Helophoridae	*Helophorus aequalis*										▪/-												▪/-			▪/-				▪/-	
Helophoridae	*Helophorus brevipalpis*			▪/-							▪/-	▪/-	▪/-					▪/-		-/▪	▪/-	▪/-	▪/-	▪/-		○/-		▪/-	▪/-		
Helophoridae	*Helophorus flavipes*										-/▪		▪/-								▪/-	▪/-									
Helophoridae	*Helophorus grandis*																			▪/-									▪/-		
Helophoridae	*Helophorus minutus*																							▪/-		▪/-	▪/-			▪/-	
Helophoridae	*Helophorus spp.*					-/▪																	▪/-								
Hydraenidae	*Hydraena riparia*		-/▪																							-/▪					
Hydraenidae	*Ochthebius minimus*																													▪/-	

Order/Family	**Species** **Site no:**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**
Hydrochidae	*Hydrochus elongatus*																					▪/▪									
Hydrophilidae	*Anacaena globulus*								▪/-																						
Hydrophilidae	*Anacaena limbata*										-/▪												-/▪	-/▪					▪/-		
Hydrophilidae	*Anacaena lutescens*										▪/-				-/▪							▪/-	▪/▪	▪/-		-/▪					
Hydrophilidae	*Enochrus testaceus*		-/▪										-/▪		-/▪				-/▪			▪/-		▪/-				▪/▪			
Hydrophilidae	*Helochares lividus*										▪/-											-/▪									
Hydrophilidae	*Hydrobius fuscipes*												▪/-								▪/-	▪/-	-/▪			▪/▪			▪/▪		
Hydrophilidae	*Laccobius colon*												▪/-											▪/▪							
Hydrophilidae	*Laccobius minutus*		▪/-									▪/-										-/▪	▪/-								
Noteridae	*Noterus clavicornis*		-/▪	-/▪			▪/-				-/▪	▪/▪	▪/○	-/▪	▪/▪				▪/▪			▪/▪	▪/▪	▪/-		▪/▪		▪/▪		▪/-	
Paelobiidae	*Hygrobia hermanni*												-/▪					▪/▪												-/▪	
Sphaeridiidae	*Cercyon convexiusculus*	-/▪																													
Diptera	Family level only																														
Ceratopogonidae	-	▪/○	○/-			-/○			-/▪	▪/○	▪/▪	▪/▪		-/▪	▪/▪	▪/-		▪/-	▪/○	▪/-	-/▪	○/▪	▪/○	▪/-	▪/○	○/-		▪/-	-/▪	▪/-	▪/-
Chaoboridae	-		-/▪		-/•	○/-	-/▪				-/•	•/▪	○/•	•/▪	▪/-			-/▪	▪/○	○/-	▪/-	-/○	○/▪					-/▪	▪/○	○/○	○/-
Chironomidae	-	•/○	○/○	•/•	•/○	○/○	○/○	○/•	•/○	•/○	○/○	○/○	-/▪	○/○	○/•	•/•	•/○	○/○	○/○	•/○	○/○	○/○	○/○	○/○	•/○	○/•	○/○	○/▪	○/○	○/•	•/-
Culicidae	-			-/▪						-/▪	-/▪		▪/▪	▪/-	▪/-				-/▪	-/▪		▪/▪	-/○		-/▪	▪/▪	-/▪	▪/▪	○/▪	-/▪	▪/-
Dixidae	-	-/▪		-/▪	▪/-	-/▪	▪/-		▪/-	-/○		-/○	▪/-	-/▪	▪/▪				▪/○	▪/▪	▪/▪	▪/▪	-/○	-/▪	-/○	○/▪	-/▪	▪/-	-/▪		▪/-
Empididae	-																								-/○	▪/-					
Ephydridae	-	▪/-							-/▪																			-/▪			
Limonidae/Pedicidae	-						-/▪		-/▪	-/▪	▪/▪		-/▪						-/▪	-/▪	▪/-				-/▪	-/▪					
Psychodidae	-	▪/-				-/▪	▪/-						-/▪			▪/-		-/▪	-/▪	▪/▪	▪/▪	-/▪	▪/-	▪/-	-/▪		-/▪	▪/○	-/▪	-/▪	
Ptychopteridae	-	○/-	▪/-																		▪/-								-/▪		
Sciomyzidae	-	-/▪																							-/▪	▪/-					
Stratiomyidae	-												-/▪	-/▪					-/▪				-/▪	▪/-				▪/-			
Syrphidae	-			▪/-	▪/-	-/▪	▪/-						▪/-	▪/-						▪/-	-/▪		▪/-		-/▪			-/▪	▪/-		▪/-

Order/Family	Species Site no:	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
Tabanidae	-	-/○	▪/-							-/▪												▪/▪									
Tipulidae	-	▪/▪	▪/-				▪/▪		-/○	-/▪	▪/▪	-/▪	-/▪	-/▪		-/▪		-/▪		-/▪	▪/▪	▪/-	▪/▪	▪/-	-/○	▪/-	▪/▪	▪/○	▪/-	▪/-	
Ephemeroptera																															
Baetidae	*Cloeon dipterum*	▪/○	○/○	-/▪	▪/○	▪/▪	▪/○		○/▪	-/▪	○/▪	▪/▪	-/○	▪/▪	▪/○	▪/○		▪/○	○/○	○/○	▪/▪	▪/○	▪/▪	▪/○	○/▪	-/▪	-/▪	▪/○		▪/○	
Baetidae	-												▪/-																		▪/-
Caenidae	*Caenis horaria*	▪/○	○/▪							▪/▪			○/▪					▪/-	○/▪					○/▪	○/▪	○/▪					
Caenidae	*Caenis luctuosa*																									▪/-					
Caenidae	*Caenis robusta*		-/▪															▪/-	-/▪			▪/-						▪/-			
Gastropoda																															
Acroloxidae	*Acroloxus lacustris*	-/▪	▪/▪				▪/○		▪/▪							▪/▪	▪/-		○/○	▪/▪			-/▪	▪/○	○/▪	-/▪		-/▪			▪/-
Bithyniidae	*Bithynia leachii*									-/▪																▪/-					
Bithyniidae	*Bithynia spp.*																											▪/-			
Bithyniidae	*Bithynia tentaculata*		▪/○							○/○					○/▪	○/○		▪/○			-/▪	▪/○			▪/○	-/○	▪/▪	▪/○			
Hydrobiidae	*Potamopyrgus antipodarum*	○/-	▪/•						▪/-								▪/-		○/▪		-/▪	▪/○		▪/○	-/▪	-/▪		▪/-			○/○
Lymnaeidae	*Lymnaea peregra*	▪/▪	▪/▪						-/▪	-/▪		-/▪	○/○		○/○	▪/▪		○/○	-/▪			○/○	•/○	○/○			•/○	▪/▪	▪/▪	▪/-	
Lymnaeidae	*Lymnaea stagnalis*	○/○	-/▪							▪/-	▪/-		○/○		•/○			-/▪	○/○					○/○		○/○		○/▪	▪/-	-/▪	
Lymnaeidae	*Radix auricularia*	-/▪																										-/▪			
Lymnaeidae	*Stagnicola palustris*									-/▪																					
Physidae	*Physa fontinalis*	○/○	-/▪		▪/-	▪/▪						▪/▪						○/○	▪/○	-/▪	-/▪	▪/○		▪/▪			○/○				▪/▪
Planorbidae	*Anisus vortex*									○/○						○/▪			▪/○			○/○		○/○	▪/-				▪/▪		
Planorbidae	*Armiger crista*	▪/▪	○/○		▪/-	-/▪	-/▪						-/▪				-/▪		○/▪	▪/▪	▪/○	▪/○		○/○	▪/▪	-/▪				▪/▪	▪/▪
Planorbidae	*Gyraulus albus*	○/○	○/○						▪/-	▪/▪	-/▪	-/▪	-/○		-/▪			▪/○	○/▪	-/▪		○/○		○/○	▪/▪	○/○		-/○	▪/▪	▪/-	
Planorbidae	*Hippeutis complanatus*		○/○				○/○		▪/▪	▪/▪		▪/▪		▪/▪	▪/-	○/○	-/○	▪/▪	○/▪	▪/▪				▪/▪	○/▪		-/▪	-/▪			
Planorbidae	*Planorbarius corneus*	○/▪										-/▪	▪/▪		▪/▪		○/-	▪/○			○/○		▪/▪	▪/-		▪/-	▪/○	○/▪	▪/▪	-/▪	
Planorbidae	*Planorbis carinatus*														-/▪				▪/-				-/▪					▪/-	-/▪		
Planorbidae	*Planorbis planorbis*	-/▪	-/▪										-/▪		○/○				▪/○			▪/○	▪/▪	-/▪		▪/-		-/▪	○/○		

Order/Family	Species Site no:	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
Valvatidae	*Valvata piscinalis*		-/▪		▪/-					○/○								▪/○						▪/○							
Hemiptera																															
Corixidae	*Callicorixa praeusta*							▪/▪		▪/-	▪/-	▪/▪	-/▪	▪/▪									-/▪							-/▪	
Corixidae	*Callicorixa wollastoni*																			-/▪											
Corixidae	*Corixa dentipes*							▪/-				▪/-	-/▪																		
Corixidae	*Corixa panzeri*									▪/-	-/▪		-/▪	▪/-				-/▪													
Corixidae	*Corixa punctata*											-/▪	▪/○	▪/○						-/▪											
Corixidae	*Cymatia bonsdorffii*						▪/-						▪/▪					▪/-													
Corixidae	*Cymatia coleoptrata*												○/○											○/○			-/▪				
Corixidae	*Hesperocorixa linnaei*				▪/-								▪/▪		▪/▪					▪/-		▪/-		-/▪							
Corixidae	*Hesperocorixa sahlbergi*			▪/-		▪/▪								▪/▪						▪/▪	▪/▪		▪/-		▪/-						-/○
Corixidae	*Micronecta poweri*										-/▪								▪/-					▪/▪							
Corixidae	*Micronecta scholtzi*	-/▪						○/-		▪/○	○/▪											○/○		▪/▪	○/○	▪/▪					
Corixidae	*Sigara concinna*							▪/▪					▪/▪																		
Corixidae	*Sigara distincta*			▪/-			-/○	-/▪		-/▪	▪/▪	▪/-	-/▪	-/▪				▪/-	▪/-			▪/▪								▪/▪	
Corixidae	*Sigara dorsalis*	▪/-	-/▪	-/▪			▪/▪	▪/▪		▪/-	▪/▪	▪/▪	-/▪	▪/○	▪/▪	▪/-		▪/▪	▪/-			▪/○	-/▪	▪/▪						▪/▪	
Corixidae	*Sigara falleni*			-/▪			○/▪	▪/▪		-/▪	○/▪	▪/▪	-/▪	-/▪	▪/-							▪/▪		-/▪	▪/-						
Corixidae	*Sigara fossarum*											▪/-	-/▪											▪/○							
Corixidae	*Sigara lateralis*			▪/▪			▪/-	▪/▪		▪/▪		▪/▪		-/▪			▪/▪					-/▪					-/▪				
Corixidae	*Sigara limitata*		-/▪									▪/-										-/▪					-/▪			▪/▪	
Corixidae	*Sigara nigrolineata*			▪/-																											▪/-
Corixidae	(nymphs)	○/○	○/-		○/▪	•/○	•/▪	○/-	▪/○	○/○	○/○	○/○	○/○	•/○	○/▪	▪/▪	▪/○	•/▪	○/○	○/▪	▪/○	○/▪	○/○	○/○	○/▪	▪/-			▪/▪	○/-	▪/-
Gerridae	*Gerris lacustris*						▪/▪		-/▪	-/▪	-/▪	▪/▪			-/▪						▪/▪			-/▪		-/▪		-/▪			
Gerridae	*Gerris odontogaster*												▪/-		▪/-									-/▪							
Gerridae	(nymphs)	-/▪				-/▪	▪/-	-/▪	-/▪	-/▪	▪/-	-/○	○/-	-/▪	○/-				-/▪	-/▪	-/▪	▪/▪	-/▪	▪/-	-/○	○/-	-/▪	○/-	-/▪	-/▪	
Hydrometridae	*Hydrometra stagnorum*								-/▪		-/▪	▪/▪		-/▪					-/▪	-/▪	-/▪		▪/-	-/▪		▪/-		▪/▪	▪/▪		

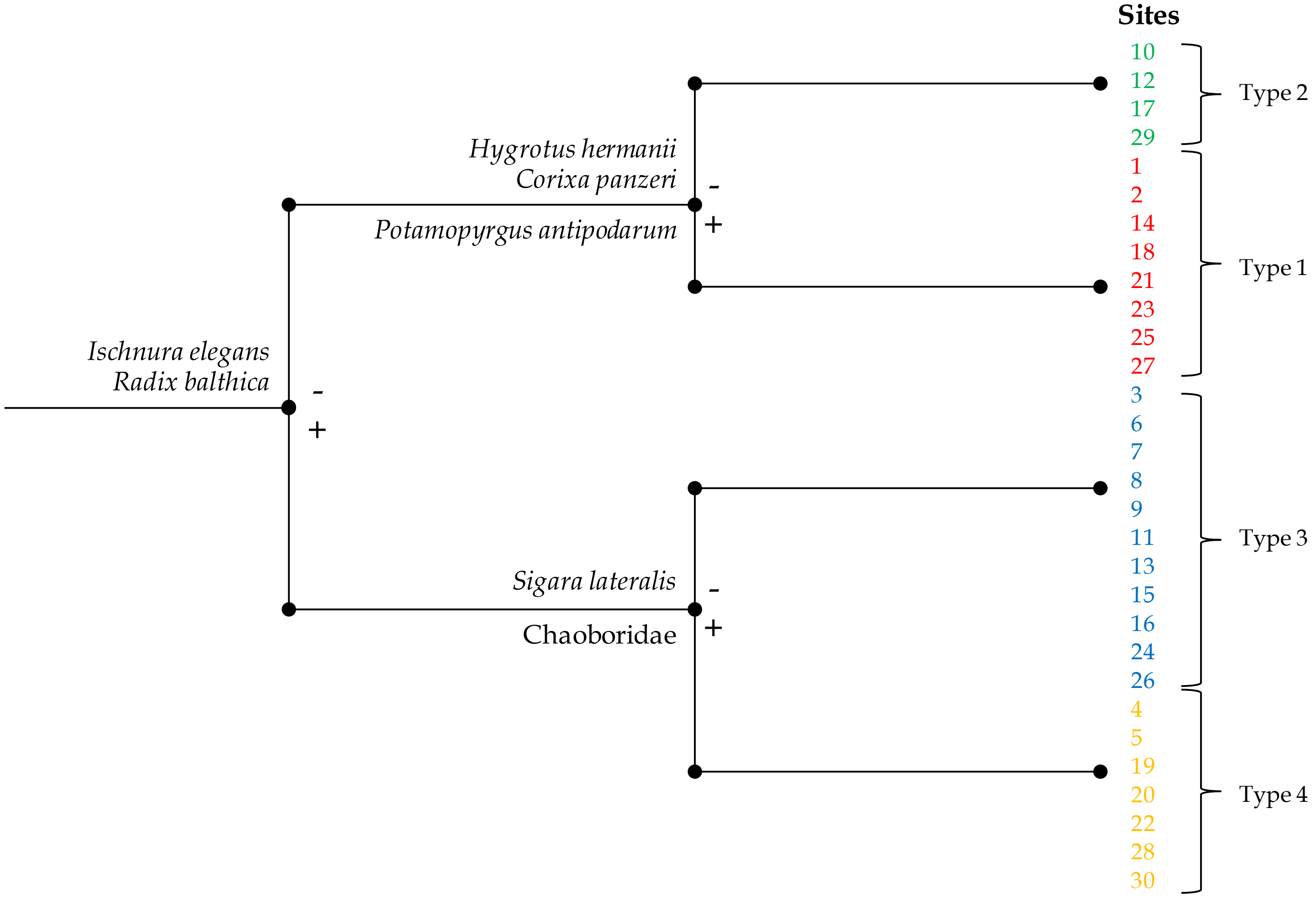
Order/Family	**Species** **Site no:**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**
Hydrometridae	(nymphs)	-/▪						-/▪	-/▪		-/▪		-/▪				-/▪	-/▪	-/○	-/▪		-/▪	-/○	-/▪	-/▪		-/▪	-/▪			
Mesoveliidae	*Mesovelia furcata*																									-/▪					
Naucoridae	*Ilyocoris cimicoides*												-/▪		▪/-							-/▪						▪/-			
Naucoridae	(nymphs)											-/▪		-/▪							-/○	○/-	-/▪					▪/-			
Nepidae	*Nepa cinerea*								▪/-												-/▪					▪/-		-/▪			
Nepidae	*Ranatra linearis*	-/▪											-/▪		-/▪							-/▪				-/▪		▪/▪			
Nepidae	(nymphs)	-/▪							-/▪	-/▪				-/▪						-/▪							-/▪	-/▪			
Notonectidae	*Notonecta glauca*		-/▪	▪/-			▪/▪		▪/-		-/▪	▪/▪	-/○	-/▪	-/▪			-/▪	-/▪	-/▪			-/▪	-/○		▪/-		▪/-		-/▪	
Notonectidae	*Notonecta maculata*											-/▪																			
Notonectidae	(nymphs)		▪/-		-/▪	▪/▪	○/-	▪/-	▪/-	▪/▪	○/▪	▪/▪	▪/▪		○/-				○/▪	▪/-		▪/-	▪/▪	○/-		▪/-		▪/-	-/▪	○/▪	
Pleidae	*Plea minutissima*																											▪/-			
Pleidae	(nymphs)		▪/-																		-/▪										
Veliidae	*Microvelia reticulata*		-/▪										▪/-													▪/-		▪/-			
Veliidae	(nymphs)					▪/-			▪/-			-/▪								▪/▪			-/▪	▪/-	-/▪	▪/-		-/▪	▪/-		▪/-
Veliidae	*Velia caprai*																			▪/-	-/▪										
Hirudinea																															
Erpobdellidae	*Erpobdella octoculata*		-/▪				▪/○	▪/-	-/▪	▪/-				▪/▪					-/▪	▪/-	▪/○	▪/▪		▪/▪	▪/▪	-/▪	-/▪				
Erpobdellidae	*Erpobdella testacea*		-/▪				-/○			-/○				-/▪	-/▪					▪/○	-/▪	-/▪					-/▪	-/▪			
Erpobdellidae	*Erpobdella spp.*					-/▪																	▪/-								
Glossiphoniidae	*Glossiphonia complanata*													-/▪				-/▪	▪/▪		▪/▪			-/▪			-/▪	▪/▪			
Glossiphoniidae	*Glossiphonia heteroclita*		▪/▪				○/○						-/▪		-/▪		-/▪	-/▪				▪/▪		▪/○			-/▪	▪/○		▪/-	
Glossiphoniidae	*Helobdella stagnalis*		▪/○	○/○	-/▪	▪/○	○/○	○/○	▪/○	○/○		○/○	▪/▪	○/○		▪/▪	○/○	▪/▪	▪/○	○/○	▪/○	▪/▪	▪/-	▪/▪	○/▪	▪/▪	▪/▪	▪/▪		○/▪	▪/▪
Glossiphoniidae	*Hemiclepsis marginata*						○/○		-/▪			-/▪					▪/-		▪/▪					-/▪		-/▪	-/▪	▪/▪			
Glossiphoniidae	*Glossiphonia spp.*		-/○																												
Glossiphoniidae	*Theromyzon tessulatum*	-/▪		-/▪			-/○			-/▪		-/▪	-/▪	-/▪	-/▪	-/▪		▪/▪	-/▪	-/▪	-/▪	▪/○	-/▪	-/▪	-/▪		-/▪	○/▪		-/▪	
Piscicolidae	*Piscicola geometra*																		▪/▪			▪/-		▪/▪		-/▪		▪/▪			

Order/Family	**Species** **Site no:**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**
Isopoda																															
Asellidae	*Asellus aquaticus*	○/○	○/○		▪/-	○/○	○/○	•/-	•/•	○/○		○/▪	○/○	○/○	○/○	○/▪	○/▪	○/○	○/○	○/○	○/○	○/○	○/○	○/○	○/○	○/▪	○/•	•/•	•/•	○/○	•/•
Megaloptera																															
Sialidae	*Sialis lutaria*	▪/▪	-/▪						▪/-	-/▪	▪/-	-/▪				-/▪			▪/▪		-/▪	▪/▪	-/▪	▪/▪	-/▪	-/▪		▪/▪			
Odonata																															
(Anisoptera)	*(early instar/damaged)*	▪/▪	▪/-				▪/-		▪/-	▪/▪	○/-		○/-		▪/-	▪/-		▪/-		-/▪	-/▪	▪/-		▪/-		▪/-		▪/-			
(Zygoptera)	*(early instar/damaged)*	○/○	▪/-			-/▪		-/▪		-/○	○/-	-/○	○/-	-/○	▪/-		-/▪	-/○	○/-	-/▪	▪/▪	○/○	-/○	▪/▪	-/○	○/-	-/○	▪/-	-/▪		
Aeshnidae	*Aeshna cyanea*										-/▪									-/▪			-/▪						-/▪		
Aeshnidae	*Aeshna grandis*								-/▪		-/▪		-/▪		-/▪				-/▪		-/▪		-/▪	▪/▪	▪/-	-/▪		▪/-		-/▪	
Aeshnidae	*Anax imperator*																													-/▪	
Coenagrionidae	*Coenagrion puella*	▪/-							▪/-		-/▪		-/▪		○/▪				▪/-		▪/▪					▪/-					
Coenagrionidae	*Enallagma cyathigerum*	▪/▪											▪/-									▪/-		▪/-							
Coenagrionidae	*Erythromma najas*	○/○	▪/▪										-/▪		▪/▪				▪/○				▪/-					▪/▪			
Coenagrionidae	*Ischnura elegans*	○/○	○/▪						▪/-		○/○		▪/▪		○/▪			▪/▪	▪/▪		▪/-	▪/▪	▪/-	○/▪	▪/▪	▪/○	▪/-	▪/▪		▪/-	
Coenagrionidae	*Pyrrhosoma nymphula*																				▪/▪					-/▪					
Lestidae	*Lestes sponsa*																													▪/-	
Libellulidae	*Libellula depressa*										▪/▪																				
Libellulidae	*Libellula quadrimaculata*																					-/▪									
Libellulidae	*Orthetrum cancellatum*																									-/▪					
Libellulidae	*Sympetrum fonscolombii*										-/▪																				
Libellulidae	*Sympetrum sanguineum*	▪/-	▪/-								▪/-		▪/-		▪/-							▪/-		▪/-							
Libellulidae	*Sympetrum striolatum*	-/▪									○/▪				-/▪																
Trichoptera																															
Hydroptilidae	*Agraylea multipunctata*	-/▪	-/▪							-/▪	-/▪		-/▪						▪/-	-/▪		-/▪				-/▪		▪/▪		-/▪	
Hydroptilidae	*Agraylea sexmaculata*		-/▪								-/○		-/▪									▪/-									

Order/Family	**Species** **Site no:**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**
Hydroptilidae	*Hydroptila spp.*	-/▪							-/▪	-/○				-/▪									-/▪	▪/-	-/▪	▪/-					
Hydroptilidae	*Oxyethira spp.*	○/○	○/-																▪/-						-/○		-/▪	○/-			
Leptoceridae	*Athripsodes aterrimus*																		-/▪					▪/▪							
Leptoceridae	*Leptocerus tineiformis*	▪/-	▪/▪																▪/-									▪/-			
Leptoceridae	*Mystacides azurea*																									▪/-					
Leptoceridae	*Mystacides longicornis/nigra*	▪/○	▪/▪				▪/▪			▪/▪			▪/▪					▪/▪				-/○		-/▪	○/▪	-/▪					
Leptoceridae	*Oecetis lacustris*												○/-													▪/-					
Leptoceridae	*Oecetis ochracea*																				-/▪										
Leptoceridae	*Triaenodes bicolor*	-/▪	○/-									-/▪		-/▪	▪/-				▪/-		-/▪	-/○	▪/▪	▪/-	-/▪	▪/-			-/▪	▪/-	
Limnephilidae	*Anabolia nervosa*																		▪/-							▪/-					
Limnephilidae	*Limnephilus decipiens*		▪/-																▪/-												
Limnephilidae	*Limnephilus flavicornis*								▪/-						▪/-				▪/-									▪/-	▪/-		
Limnephilidae	*Limnephilus lunatus*		▪/-						○/-	▪/-	○/-				▪/-				○/-		○/▪	▪/▪		▪/-	▪/-	○/-		▪/-	▪/-	▪/-	▪/-
Limnephilidae	*Limnephilus marmoratus*												▪/-		▪/-													▪/-			
Limnephilidae	*(early instar/damaged)*																														
Molannidae	*Molanna angustata*																		-/▪					-/▪	-/▪						
Phryganeidae	*Agrypnia pagetana*	▪/▪											-/▪					▪/-	-/▪												
Phryganeidae	*Phryganea bipunctata*		-/▪								-/▪		▪/▪		-/▪				-/▪					-/▪		-/▪		-/▪			
Polycentropodidae	*Cyrnus flavidus*	-/▪																	-/▪					-/▪		-/▪					
Polycentropodidae	*Cyrnus trimaculatus*										-/▪																				
Polycentropodidae	*Holocentropus dubius*	▪/-																	▪/-												
Polycentropodidae	*Holocentropus picicornis*	-/▪											▪/○		-/▪	-/▪			▪/▪							-/▪					
Psychomyiidae	*Lype reducta*		▪/-																												
Tricladida																															
-	*(damaged during preservation)*	▪/-				▪/-											-/▪	▪/▪	-/▪				▪/▪	▪/-	-/▪		▪/-	▪/-		▪/-	▪/-
Dendrocoelidae	*Dendrocoelum lacteum*							-/▪	-/▪	-/▪			-/▪	-/▪				-/▪				-/▪	-/▪				-/▪	-/▪	-/▪		

Order/Family	**Species** **Site no:**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**
Dugesiidae	*Dugesia lugubris/polychroa*					-/▪	-/▪	-/○					▪/-	-/▪		-/○	-/○		▪/▪	-/▪	-/▪	-/▪	-/▪	-/▪		-/○			-/▪	▪/-	
Dugesiidae	*Dugesia spp.*	▪/-																													
Dugesiidae	*Dugesia tigrina*	▪/▪				-/▪		-/▪	-/○	-/○		-/○		-/▪			-/▪	-/▪	-/○	-/▪	-/▪	-/▪			-/▪	-/▪					
Planariidae	*Planaria torva*																										▪/-				
Planariidae	*Polycelis nigra/tenuis*				-/○	▪/-		-/▪	-/▪											-/○	-/▪	-/▪	-/▪	▪/-					-/▪		

Appendix 21. Two Way Indicator Species (TWINSPAN) dendrogram, indicator taxa and urban pond types.



Appendix 22. Mean (min and max) values of local (local physical and water quality) variables from 20th May -11th June and 3rd – 14th August 2009 sampling periods.

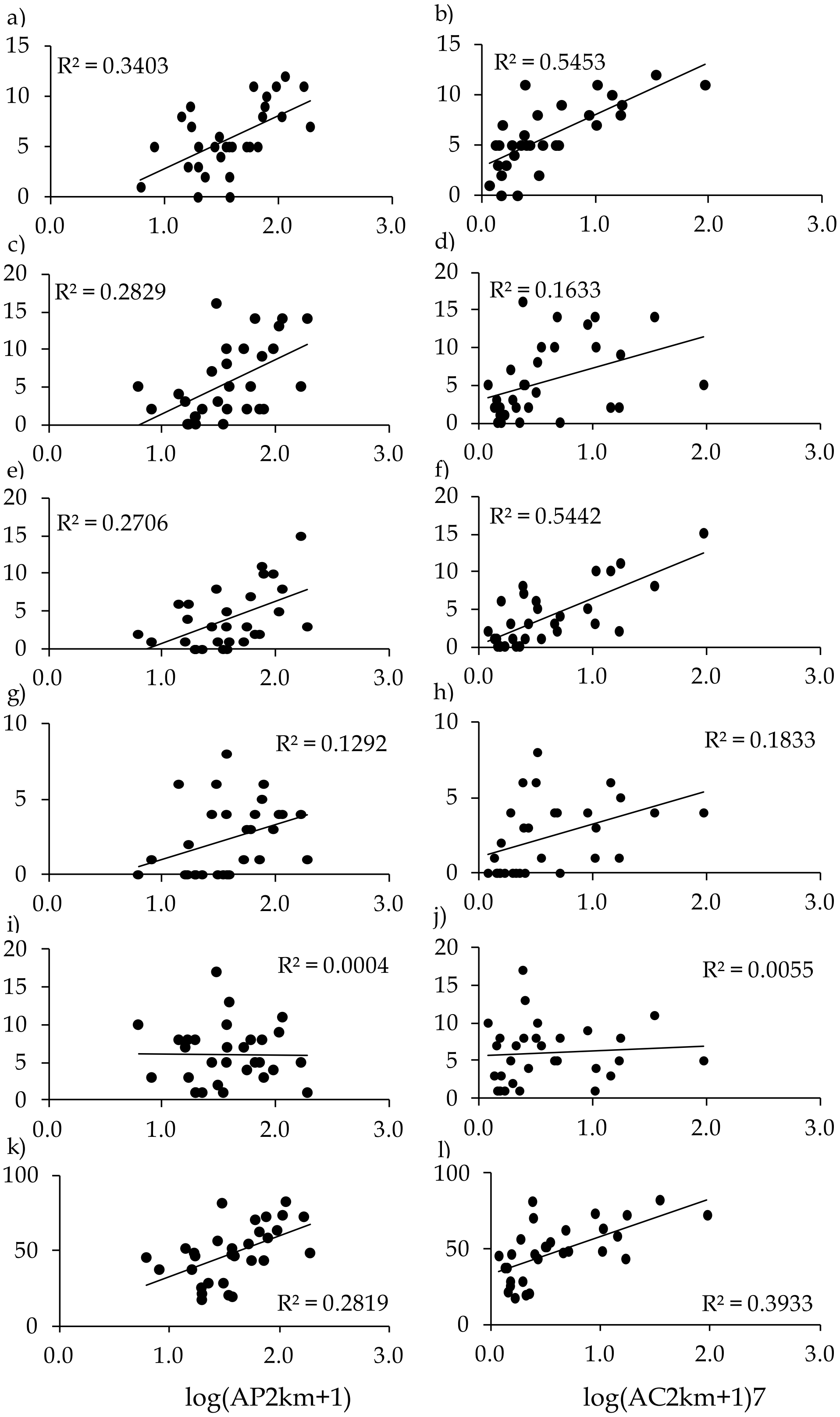
	Unit	20th May -11th June	3rd – 14th August	Study mean
Water quality				
pH	-	7.86 (6.8 – 10.0)	7.51 (6.6 – 8.8)	7.68 (6.6 – 10.0)
EC	µS/cm	505.4 (247 – 1442)	453.8 (205 – 981)	479.6 (205 – 1442)
Temp.	°C	17.5 (13.1 – 27.5)	18.8 (15.7 – 22.7)	18.2 (13.1 – 27.5)
DO%	%	75.2 (11.2 – 166.0)	56.6 (6.0 – 185.2)	65.9 (6.0 – 185.2)
$CaCO_3$	mg/l	123.9 (46 – 230)*	148.1 (50 – 342)	136.0 (46 – 342)
Na	mg/l	23.1 (6.1 – 88.3)	19.8 (4.5 – 71.3)	21.5 (6.1 – 88.3)
NH_4	mg/l	0.77 (0.0 – 6.6)	0.78 (0.0 – 4.8)	0.77 (0.0 – 6.6)
K	mg/l	5.75 (1.4 – 12.5)	3.85 (0.7 – 12.3)	4.80 (0.7 – 12.5)
Mg	mg/l	8.20 (3.0 – 19.0)	6.36 (1.9 – 18.3)	7.28 (1.9 – 19.0)
Ca	mg/l	53.86 (20.0 – 110.5)	50.56 (20.0 – 111.9)	52.21 (20.0 – 111.9)
Cl	mg/l	36.88 (4.2 – 144.8)	29.93 (6.3 – 88.4)	33.41 (4.2 – 144.8)
NO_3	mg/l	0.25 (0.01 – 6.43)	3.45 (0.01 – 70.26)	1.85 (0.01 – 70.26)
SO_4	mg/l	38.32 (2.7 – 126.6)	30.08 (1.2 – 87.6)	34.2 (1.2 – 126.6)
PO_4	mg/l	0.51 (0.02 – 2.62)	0.46 (0.01 – 2.51)	0.48 (0.01 – 2.62)
Fe	µg/l	862.6 (101 – 7204)	774.4 (87 – 8139)	818.5 (87 – 8139)
Mn	µg/l	627.3 (118 – 2283)	309.3 (66 – 984)	468.3 (66 – 2283)
Zn	µg/l	209.7 (45 – 803)	26.1 (23 – 29)	117.9 (23 – 803)
Local physical				
Habitats	Count	3 (1 – 6)	4 (1 – 7)	3.6 (1 – 6.3)
Surface area	m^2	-	-	3597 (299 – 14967)
Concrete	%	-	-	15 (0 – 100)
Open water	%	-	-	52 (0 – 96)
Tree cover	%	-	-	30 (0 – 100)
Riparian veg.	%	-	-	13 (0 – 41)
Floating veg.	%	-	-	5 (0 – 47)
Macrophytes	Count	4.4 (0 – 16)	4.4 (0 – 15)	7 (0 – 20)*
Spring	Binary	-	-	0.47 (0 – 1)
RainRun	Binary	-	-	0.60 (0 – 1)
RdBldRun	Binary	-	-	0.27 (0 – 1)
Stream	Binary	-	-	0.17 (0 – 1)
WLFI	-	-	-	344 (4.0 – 4698.4)
Fish	Binary	-	-	0.43 (0 – 1)
Chlorophyll *a*	µg/l	1.5 (0.0 – 11)**	55.8 (0.6 – 259)	28.6 (0.0 – 259)
Suspended solids	mg/l	14.0 (2 – 52)**	21.7 (3 – 110)	17.9 (2 – 110)

* Sum total from the two sampling periods

**Data from the equivalent period(s) in the 2010 hydrological year (as described in Section 2.2)

Appendix 23. Comparison of the performance of the area informed, Euclidean (area-proximity, AP) and effective distance metrics (AC) for predicting macroinvertebrate richness at 2km distance

a,b) Gastropoda, c&d) Coleoptera, e,f) Trichoptera, g,h) Odonata, i,j) Hemiptera, k,l) Taxa



Appendix 24. Composition of mature vegetation (>3m) within 10m of study pond margin

Species	Common name	%	Species	Common name	%
Crataegus spp.	Hawthorn	11.1	*Sambucus nigra*	Elder	2.62
Betula pendula	Silver birch	9.97	*Rhododendron ponticum*	Rhododendron	2.14
Salix fragilis	Crack willow	8.00	*Corylus avellana*	Hazel	1.80
Salix cinerea	Grey willow	7.99	*Prunus spp.*	Laurel	1.71
Ilex aquifolium	Holly	7.93	*Taxus baccata*	Yew	1.50
Alnus glutinosa	Alder	7.92	*Populus tremula*	Aspen	1.29
Acer pseudoplatanus	Sycamore	5.21	*Pinus nigra*	Corsican pine	1.21
Fraxinus excelsior	Ash	4.66	*Acer platanoides*	Norway maple	1.11
Quercus robur	English Oak	3.59	*Acer campestre*	Field maple	1.09
Salix caprea	Goat willow	3.15	*Tilia x europaea*	Lime	1.07
Salix × sepulcralis	Weeping willow	2.86	*Aesculus hippocastanum*	Horse chestnut	0.92
			Fagus sylvatica*	**Beech**	**0.83**
				Others	10.3

*with reference to Chapter 6

Appendix 25. Summary of taxon richness and abundance with functional feeding group, after Tachet *et al.* (2002)

Taxa	**FFG**	1	2	3	4	5	6	7	8	9	10	12	13	15	16	17	19	20	21	22	23	24	25	27	28	29	30
CRUSTACEA																											
A. aquaticus	SHR	●	●	▪		○	●	●	●	●	○	●	○	○	○	●	○	○	●	●	●	●		●	○	○	●
C. pseudogracilis	SHR	○	○		○					○	○	○	▪			○		○	○	○	○	○		○	○		○
G. pulex	SHR															○					▪						
TRICHOPTERA																											
A. sexmaculata	PIE									▪		▪															
A. multipunctata	PIE									○		▪									▪		○				
A. varia	SHR											○															
P. bipunctata	SHR	○																			▪			▪			
M. longicornis	SHR									▪						▪			▪		▪	▪					
T. waeneri	SCR																				▪						
C. flavidus	PRED		▪																		▪						
C. trimaculatus	PRED										○											▪					
H. dubius	PRED																						▪				
H. piccicornis	PRED											▪									▪						
A. aterimmus	SHR	○																									
T. bicolor	SHR		▪																								
GASTROPODA																											
B. tentaculata	FF		○							○				○		○			○			○		○			
B. leachi	FF	○								▪													▪				
P. carinatus	SCR	○																							○		
P. corneus	SHR											▪				▪					▪						
H. complanata	SCR		○				▪		▪	○				●	○	○	○				▪	▪					
G. albus	SCR	▪	▪							▪		○			▪	○			▪	▪	○		○	○			
P. fontinalis	SCR	○		○		○									▪				▪		▪						○
R. balthica	SCR		▪									○								▪	▪						
L. stagnalis	SCR											○											▪				
A. vortex	SCR						▪			▪									▪					▪	○		
A. crista	SCR		○									▪						○	▪		○	○	○	○			
A. lacustris	SCR	○	○				▪															○	●	○			
P. antipodarum	SHR	▪	▪	▪																	▪	▪					○
BIVALVIA																											
Sphaeridae	FF		○							▪					▪	○		○	▪	○	○	▪		○			○
TRICLADIDA																											
D. tigrina	PRED									▪																	
D. lacteum	PRED									○																	
*D. polychroa/lugribris**	PRED			▪						○			▪											▪			
Tricladida*	PRED																							▪			
HIRUDINEA																											
H. stagnalis	PRED		▪	○	○	●	●	●	○	○		○	●	○	▪		●	○	○	▪	▪	○	▪	○			○
E. octoculata	PRED		▪				▪			○								○	▪		▪						
E. testacea	PRED						▪										○										
T. tessulatum	APRA			▪		▪	▪	▪	▪	▪		○			▪			○	▪	▪				▪			
Taxa	**FFG**	1	2	3	4	5	6	7	8	9	10	12	13	15	16	17	19	20	21	22	23	24	25	27	28	29	30

H. marginata	APRA	○					▪												▪					○			
G. heteroclita	PIE						▪																				
G. complanata	PIE									▪																	
*E. octoculata/testacea**	PRED						▪	▪		▪									▪			▪	▪				
HEMIPTERA																											
S. dorsalis	SHR																				▪						
H. stagnorum	PRED																								○		
EPHEMEROPTERA																											
C. horaria	DF		○																		○	▪	○		○		
C. dipterum	DF	▪	▪						○			▪		○							○	▪	▪				
ODONATA																											
A. grandis	PRED										○	▪				▪						▪	▪				
I. elegans	PRED	○	○								○	○				○					○	▪	○	▪			○
E. cyathigerum	PRED									▪												▪					
C. puella	PRED	○	○									○															
E. najas	PRED	○	▪								○								▪								
P. nymphula	PRED																				▪						
Zygoptera*	PRED		○								○	○									▪	▪		▪			
MEGALOPTERA																											
S. lutaria	PRED																	○			▪						
COLEOPTERA																											
A. limbata	SHR																								○		
Larvae	-									▪		▪							▪		▪						
DIPTERA																											
Chironomidae	DF	○	○	●	●	●	○	▪	○	○	●	○	○	●	●	○	○	○	○	○	○	●	○	○	○	○	○
Ceratopogonidae	DF		○									▪		▪				○				▪	○			○	
Chaoboridae	PRED											▪						○									
Tipulidae	SHR																								○		
Psychodidae	SHR																								○		
OLIGOCHAETA	DF	○	▪	○	○		○			▪			●	▪	○	○	○	○	○	○		▪		○	●	●	○

●	**Taxa contribute >25% to total macroinvertebrate abundance**
○	**Taxa contribute between 1 and 25% to total macroinvertebrate abundance**
▪	**Taxa contribute <1% to total macroinvertebrate abundance**
*	**Individuals too immature or poorly preserved to be identified further**

SHR Shredder, PIE Piercer, SCR Scraper, PRED Predator, FF Filter feeder, DF Deposit feeder

Appendix 26. Intercorrelates with pond surface area (Pearson's correlation)

P value of relationships >0.1 = I, <0.1 = *I*, <0.05 = **I**, <0.01 = ***I***

	Area	TreeCov	Fish	pH	°C	DO	K	NH$_4$
Area	-							
TreeCov	***-0.59***	-						
Fish	**0.41**	0.23	-					
pH	***0.60***	***-0.69***	0.20	-				
°C	***0.52***	***-0.69***	*0.38*	**0.48**	-			
DO	***0.61***	***-0.70***	0.32	***0.82***	***0.54***	-		
K	***-0.52***	0.29	*-0.35*	-0.32	-0.08	-0.22	-	
NH$_4$	**-0.46**	***0.68***	-0.29	***-0.60***	**-0.42**	***-0.75***	*0.35*	-

TreeCov Percentage coverage of vegetation >3m height, *Fish* Fish presence