Local and landscape scale patterns of macroinvertebrate diversity across ponds in Greater Kuala Lumpur, Malaysia

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Abstract

Ponds and wetlands are increasingly recognized as valuable ecosystems in the urban landscape, mainly for their benefits to residents but increasingly for their capacity to support biodiversity. Their capacity to support diversity is critical for freshwater species and populations that continue to decline globally due to habitat loss and degradation. Conservation of these species is contingent on an understanding of the interactions between the built environment and the quality of urban wetland habitats. Research on urban biodiversity patterns from tropical regions that are experiencing rapid urbanization remains limited, and this is especially the case for ponds and wetlands. The aim of this thesis was therefore to characterize biodiversity patterns in ponds across Greater Kuala Lumpur (GKL), a rapidly expanding, tropical urban area, and understand the factors that influence these patterns. In so doing, the broader goals of the work were to highlight the biodiversity value of these abundant ecosystems and provide data to help direct management and conservation.

With aquatic macroinvertebrates as the focal group, biological, environmental and spatial data were collected for 28 ponds across GKL. Data were collected in both the wet and dry seasons (2021 - 2022). Data on pond chemistry and physical habitat characteristics (e.g. presence of macrophytes, size) were collected, while a range of broader, landscape-scale data were obtained from existing GIS and remote sensing sources. Landscape data included information on surrounding land cover and pond distributions across GKL. These data were used to characterize and examine variations in taxonomic and trait diversity and community composition, and understand the local and landscape-scale factors that influence ecological differences between ponds. Generalized Additive Modelling and Redundancy Analysis were performed to identify factors influencing the variation between ponds. To assess the importance of dispersal and connectivity between the ponds, functional connectivity metrics were derived using geospatial resistance surfaces based on land use and taxa dispersal distance information. In addition, landscape-scale pond network analysis was carried out to characterize structural connectivity for ponds across GKL.

Ponds in GKL exhibited considerable variation in taxonomic and functional alpha diversity, mainly explained by variations in pond characteristics, namely aquatic vegetation cover and suspended solids concentration. Surrounding land use, characterized by extent of built (impervious) land cover and road density explained variation in taxonomic and functional community composition (beta diversity). High taxonomic beta-diversity was recorded among pond invertebrate communities, primarily driven by species turnover (replacement). This was different to the patterns observed for trait-based beta-diversity which was primarily driven by richness difference. These findings are broadly consistent with findings from non-tropical regions, with similarities such as the positive relationship between aquatic

vegetation cover and diversity measures and the high, turnover-driven beta-diversity. Some notable patterns found for the GKL ponds were the relative importance of ponds with smaller surface areas for diversity, the negative relationship between alpha diversity measures (taxonomic and functional) and suspended sediment concentration, and the association between communities with abundant gastropod taxa such as *Physa* and non-native *Pomaceae* and pond water conductivity.

Clustering analysis revealed two types of pond that exhibited statistically significant differences in taxonomic richness, diversity and community composition. The two types were primarily distinguished by differences in their size and distance from roads, further indicating the importance of both pond characteristics and surrounding land use. Increasing connectivity, quantified as the number of neighbouring ponds, was associated with increasing alpha diversity among sampled ponds but there were no significant associations between diversity and other structural or functional connectivity metrics such as Euclidean distance to nearest ponds. On the other hand, cost distance (representing functional connectivity) to neighbouring ponds explained some variation in community composition among the sample ponds. The landscape-scale pond network connectivity analysis showed that greater concentrations of highly connected pond clusters were present in two districts, with pond densities generally higher in more urban commercial/residential regions (Shah Alam, Putrajaya) than in the peri-urban or industrial (Hulu Langat, Klang) parts of GKL.

This study is one of the first to systematically examine local and landscape scale diversity patterns in tropical urban ponds. The work helps provide some points of guidance to support improved management of these habitats as well as to direct future research. These include (i) developing pond quality assessment criteria that integrate both local and landscape scale information, (ii) designing or redesigning ponds to have earth banks with greater aquatic vegetation cover, and (iii) coordinating pond management among various stakeholders in different administrative districts, to reflect the importance of connectivity across the whole city. Key to implementing these measures will be raising awareness of the ecological value of ponds in GKL that, as urban habitats, can be important in achieving the urban biodiversity conservation goals outlined in the Malaysian national policy on biodiversity. To support these goals, expanding the scope of research to other taxonomic groups using ponds and integrating ecosystem services as a critical component of pond research and management strategy are recommended. More research on the landscape-scale responses of tropical freshwater species to the urban environment will also be necessary to determine whether the patterns recorded in this study are consistent across other tropical cityscapes and to inform management strategies at regional scales.

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Introduction

1. INTRODUCTION

Over half of the world's human population now resides in urban areas, with the proportion of urban residents expected to continue rising (to ~70%) over the next thirty years (UNCTAD, 2023). Urban expansion impacts natural environments directly (through habitat loss, fragmentation and degradation) and indirectly (as urban populations consume resources and produce pollution) (McDonald et al., 2020). The rate of urbanisation is particularly high in South-East Asia, which include major cities such as Kuala Lumpur, Bangkok, and Jakarta that are located within regions of high biological diversity (Myers et al., 2000; Simkin et al., 2022; UNCTAD, 2023). Urban expansion is associated with loss or degradation of terrestrial ecosystems (Tee et al., 2018) as well as changes in water quality and the hydrological regimes of river systems, with detrimental impacts on aquatic biodiversity (Ramírez et al., 2009; Yule et al., 2015). Nevertheless, while numerous efforts to preserve and restore natural habitats are underway, researchers are increasingly drawing attention to the important biodiversity within urban areas, especially in relation to providing ecosystem services (Aronson et al., 2014; Beninde et al., 2015; Soanes et al., 2020).

Characterization of urban biodiversity reveals different patterns for terrestrial and aquatic ecosystems, and for plant and animal species. Generally, areas with the highest degree of urbanization have the lowest biodiversity and areas with low urbanisation have higher biodiversity (McKinney, 2002); however, for some groups such as plants, intermediate levels of urbanisation exhibit high diversity (McKinney, 2002; McKinney, 2008). Although there is evidence for local diversity preservation in urban habitats (Aronson et al., 2014), for certain urban ecosystems, species composition can become dominated by more tolerant species or non-native species (Sinclair et al., 2020) reflecting differences in species responses to urban environmental conditions (Yule et al., 2015). In addition, areas where endemic populations exist in patches within an urban matrix, habitat fragmentation (for terrestrial ecosystems) threatens population viability (Tee et al., 2018).

In recent years, the relative importance of lentic water bodies for biodiversity in urban areas has been the subject of growing interest and research. Changes in land use, especially for urban or industrial development, have historically resulted in the loss of lentic waterbodies such natural or rural man-made ponds (Huq, 2017; Wood et al., 2003). But lentic water bodies, including ponds, lakes and wetlands are systems which can increase in abundance when urbanisation occurs. This is because new lakes, settlement ponds or reservoirs are constructed as part of the urban water management infrastructure (Forman, 2014; Hale et al., 2015; Hassall, 2014; Sinclair et al., 2020; Teo et al., 2021). However, these systems often have primary infrastructure purposes (e.g. stormwater retention) and

are subject to management regimes that may not be beneficial to wildlife; hence, their long-term value as habitats for biodiversity conservation and ecosystem services provision remains unclear, and is still being examined (Alves et al., 2020; Briers, 2014; Forman, 2014). This is especially important for tropical cities within regions of high biological diversity, but there is both geographic and taxonomic bias in knowledge of such matters, with tropical regions relatively understudied (Beninde et al., 2015; McDonald et al., 2020; Oertli & Parris, 2019). As awareness of the importance of natural features such as trees, vegetation, ponds – collectively referred to as green and blue spaces – for ecosystem services (e.g. regulating temperature, flood water, pollution), it will be important to address gaps in knowledge of local diversity and community assemblage patterns and their responses to a range of environmental conditions that characterize tropical urban areas, including higher year-round temperature and rainfall that influence ecosystem processes (Yong & Yule, 2004; Yule et al., 2015). Establishing the ecological value of lentic ecosystems will also be necessary if they are to be recognized, integrated and managed as habitats capable of supporting freshwater biodiversity and ecosystem services within planning policies and practices related to towns and cities.

1. 1 Urban pond biodiversity

1.1.1 Definitions of urban pond

Ponds are generally described as small, shallow lentic water bodies. Within pond research, the most widely accepted definition is that of Biggs et al. (2005) - 'water bodies between 1m^2 and 2ha in area which may be permanent or seasonal, including both man-mad and natural water bodies' (Richardson et al., 2022). Thus, ponds are a subset of standing water habitats that extend to include large wetlands, lakes and reservoirs. Ponds are formed through natural or anthropogenic processes that create land depressions that then fill up with rain- or ground-water (Oertli et al., 2005). They can be ephemeral, with periods of dryness, and eventually accumulate sediment that fills up the depression (Biggs et al., 1994).

The distinction between ponds and other standing water habitats is often unclear as different regions apply different terminologies with overlaps in some but not other characteristics. Pond size is often the main criterion, although there are often variations in the size ranges among researchers and policy-makers (Oertli et al., 2010; Richardson et al., 2022). Differences in surface area, littoral zone to area ratio, depth, extent of pelagic-benthic exchanges, macrophyte and fish assemblages have also been used to delineate ponds and larger lakes (Søndergaard et al., 2005). Nevertheless, pond size—which has implications for ecosystem processes such as metabolism and nutrient exchange—remains key to distinguishing ponds from other lentic waterbodies (Richardson et al., 2022; Søndergaard et al., 2005), even if a precise definition currently remains elusive. A more functionally resolved definition, however,

may be necessary as the value of freshwater ponds for biodiversity and ecosystem services is increasingly recognized (Biggs et al., 2017; Cantonati et al., 2020; Moorhouse et al., 2021; Richardson et al., 2022).

Ponds in urban areas comprise a range of freshwater bodies including natural, relic freshwater ponds (i.e. those neither drained nor built on) as well as constructed ponds such as storm water retention ponds, ornamental ponds, repurposed industrial ponds and garden ponds (Forman, 2014; Hassall, 2014). The abundant ponds in urban areas are more likely to be man-made and often have distinct characteristics compared to their natural counterparts, with some types being completely novel habitats and ecosystems (Hill et al., 2016; Oertli, 2018). They can vary in their morphometry and water quality, which will vary depending on their function, the characteristics of the surrounding land use and shore, and the nature of the water runoff (or source of water) they receive (Forman, 2014). Urban ponds are also subject to different management interventions and different levels of anthropogenic disturbance (Blicharska et al., 2016; Forman, 2014; James S. Sinclair et al., 2020). All of these factors can affect the water quality in terms of water clarity, pollution, nutrient content, oxygen levels and aesthetic appeal (Forman, 2014). In addition, urban residents' uses and perceptions of ponds, both within and between cities, can vary considerably, and affect the distribution of ponds and increase local variability in pond conditions (Forman, 2014; Moorhouse et al., 2021; Ngiam et al., 2017). As cities adopt ever more sustainable practices in design and management, different types of ponds are constructed among other solutions for addressing flooding within cities and limiting contaminated water flowing into natural water bodies (DID, 2012; Forman, 2014; Oertli & Parris, 2019). This suggests that pond networks are becoming important features of the urban landscape, adding 'blue spaces' and potentially providing multiple regulating and cultural ecosystem services in the same manner that green spaces do (Oertli & Parris, 2019).

1.1.2 Ecology of urban ponds

The ecological value of urban ponds has also become the subject of biodiversity research in recent years (Oertli & Parris, 2019). To date studies have shown the conservation value of urban ponds for wetland plants, dragonflies, damselflies and amphibian species, including threatened species (Bounas et al., 2020; Holtmann et al., 2019; Soanes et al., 2020). Johansson et al. (2019) showed that urban ponds can support up to 61% of regional Odonata species. Similarly, Hill et al. (2016) documented macroinvertebrate species with a national conservation designation being present in urban ponds in the UK. On the other hand, Noble and Hassall (2014) showed that urban ponds can be species poor when management priorities are not favourable for biotic community establishment. In addition, Hale et al. (2015) warn about the potential threat to species or population fitness where urban habitats become ecological traps rather than refuges as a result of management activities. As lentic (standing

water) habitats, urban ponds and wetlands also raise concerns of pest or invasive species occurrence. Hanford et al. (2019) emphasized the need for more species-specific studies to mitigate potential threats associated with mosquitoes in urban wetlands, while Sinclair et al. (2020) highlighted the prevalence of invasive plant species in urban ponds.

Urban-rural gradient studies and comparisons with regional species pool data suggest that the quality of the urban habitat determines the proportion of regional freshwater diversity it can support (Hill & Wood, 2014; Johansson et al., 2019). For multiple taxonomic groups (including amphibians and aquatic insects) species richness tends to decrease with greater urbanization (Blicharska et al., 2017; Hamer & Parris, 2011; Thornhill et al., 2017). However, urban ponds tend to show more variation in community composition than non-urban counterparts, and this is probably driven by local variation in pond conditions within a given urban area (Hill et al., 2016). In addition, different taxonomic groups, or even species of the same group, can exhibit contrasting responses to urbanization (Blicharska et al., 2017; Hamer & Parris, 2011). Notably this is contrary to findings on urban lotic (rivers, streams) diversity which tend to support lower diversity and more homogeneous composition than non-urban ones (Forman, 2014; Pickett et al., 2011; Ramírez et al., 2009; Walsh et al., 2005; Yule et al., 2015).

The potential value of urban ponds in supporting biodiversity might be increasingly relevant as freshwater populations decline globally (Hill et al., 2016; Oertli & Parris, 2019; Reid et al., 2019). This will require understanding of the general patterns in community responses to common urban features as well as diversity patterns of local taxa within local urban and geographical contexts. However, only a small fraction of existing research on urban pond ecology is based on research from outside North America and Europe (Oertli & Parris, 2019). There is a need to address this gap as urban areas are rapidly growing in regions such as tropical South-East Asia, that are characterized by (i) naturally high ecological diversity and different local species to those areas studied to date (ii) different environmental conditions (for example, year round high temperature and thus different rates of ecosystem processes) and (iii) urban development that may be different in form and management (Gálvez et al., 2023; Grêt-Regamey et al., 2020; Marcotullio et al., 2021; Olmo et al., 2022; Simkin et al., 2022; Yule et al., 2015). Research approaches that are based on key concepts and frameworks related to biodiversity and connectivity can address data and knowledge gaps and provide information that will be necessary for evidence-based management and monitoring of tropical urban pond habitats.

1.2 Key concepts: Understanding biodiversity patterns and drivers

1.2.1 Taxonomic diversity, functional diversity and community composition

Among the most common means of quantifying biodiversity and describing a community are taxonomic richness and sets of diversity metrics that incorporate information on species presence and relative

abundance in a given habitat. Taxonomic richness is the number of species or taxa in a given sample whereas diversity metrics integrate information on total number of species and the number of individuals of each species. Examples of diversity metrics include the Shannon-Weiner index, Simpson's index, Berger Parker dominance index, and Pielou's evenness index. However, because of information loss, one measure is not enough to definitively describe a community in a given habitat. Thus, multiple and complementary diversity metrics are often used for better descriptions of a given community while also making it possible to compare multiple communities (Morin, 2009). These metrics that correspond to a local scale (individual habitat level) assessment of biodiversity are also referred to as alpha diversity (Whittaker, 1972)

Biodiversity can also be described at a landscape or regional scale. Beta-diversity describes the extent of differences in the species that are present among multiple communities or sites within a given area, and how they add to overall biodiversity (Anderson et al., 2011; Whittaker, 1972). Beta diversity metrics include measures that quantify the extent of dissimilarity between a pair of sites (or samples) and calculation of total variance of a given community matrix (i.e. site x species table) (Legendre & De Cáceres, 2013). Several (dis)similarity measures or coefficients are available for quantitative pairwise comparison of sites from which total beta diversity for a set of sites (or region) can be derived. These coefficients include the Jaccard and Sorenson indices which based on counts or the presence/absence of species in a given pair of sites, calculate an index ranging 0 – 1 with 1 reflecting maximum dissimilarity, i.e. no species in common (Gotelli & Ellison, 2004; Legendre & De Cáceres, 2013). Beta diversity is comprised of two components: richness difference and turnover (Legendre, 2014). Richness difference refers to variation in assemblages that is attributed to the difference in species number between sites, i.e., one site has more species than the other (Legendre, 2014). A pair of communities or sites can have different numbers of species with all, some or none that are shared. For example, nestedness is a type of richness difference and refers to when the species present at sites with lower species richness are subsets of those species present at sites with greater richness, i.e. all the species in the smaller community are present in the larger one (Almeida-Neto et al., 2008; Legendre, 2014). On the other hand, turnover (also referred to as species replacement) refers to variation driven by presence of new or different species between sites (Baselga, 2012; Legendre, 2014). These two components (Figure 1.1) contribute to community heterogeneity in varying proportions, and this may also vary by region or species groups (Soininen et al., 2018). It is also possible to calculate the contribution of individual sites to total beta diversity. This is known as Local Contribution to Beta diversity (LCBD) and measures how unique an individual site or community is (in terms of species composition or even environmental characteristics) relative to others within a given region (Legendre & De Cáceres, 2013). The Local Contribution to Beta Diversity (LCBD) makes it possible to assess the relative importance of a given

habitat based on its contribution to overall beta diversity or the uniqueness of its environmental conditions (Heino et al., 2022; Heino & Grönroos, 2017; Legendre & De Cáceres, 2013).

Total biodiversity encompassing all of the species from all study sites in a defined geographical area is referred to as gamma diversity (Whittaker, 1972). It is often calculated as the aggregate of alpha diversity (e.g. species richness) calculated for all of communities within the area (Whittaker, 1960).

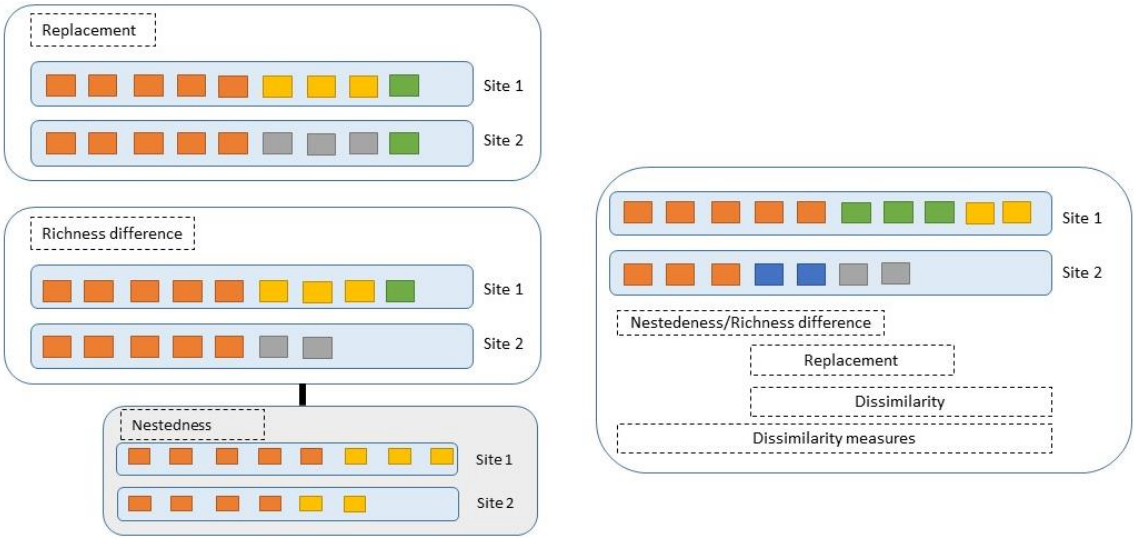


Figure 1.1 Components of beta diversity and partitioning of dissimilarity measures for a pair of sites (adapted from Legendre (2014)).

In addition to the above metrics that rely on species or taxa identity, there are also trait-based metrics that incorporate information about the species – that is their biological traits (morphology, physiology, behaviour) and ecological preferences – making it possible to quantify functional diversity of a community (Petchey & Gaston, 2006). Trait-based metrics include trait richness and functional diversity which are based on the traits of species present, their relative abundance, and the extent of dissimilarity among traits present (Champely & Chessel, 2002) in a given sample or community. Characterizing communities in terms of the presence and prevalence of given traits can potentially provide more information about community responses to environmental variables (McGill et al., 2006). This is because environmental variables like temperature, altitude, fine sediment concentration act on or impact traits like body size, forms of locomotion, respiration or dispersal which can make them more useful for understanding how certain environmental conditions select for certain species or structure a community (Buendia et al., 2013; McGill et al., 2006). Furthermore, descriptions of relationships between traits and environmental factors are more likely to lead to generalizations of community patterns across habitat types and scales than species identity focused descriptions (McGill et al., 2006; Pollard & Yuan, 2010).

1.2.2 Factors influencing urban pond biodiversity

1.2.2.1 Local and regional environmental factors

Species distribution and abundance patterns are the result of internal dynamics within a given community (e.g. predation, competition), and the influences of external factors such as environmental constraints and spatial factors (Morin, 2009). Generally, the environmental factors that influence pond community characteristics (diversity, richness, density, composition) are classified into local and regional factors (Oertli & Parris, 2019; Thornhill et al., 2017). Local environmental factors refer to pond features such as area, water quality, and morphometry, whereas regional factors refer to variables associated with the spatial distribution of habitats and surrounding landscape features, including presence and distances of other ponds, and proportion of impervious surfaces or vegetation(Oertli & Parris, 2019; Thornhill et al., 2017). The impact of surrounding land use can affect urban pond biodiversity in several ways – for example, by determining the quality of water entering and the pond, or by presenting barriers that can impede successful dispersal of organisms for processes such as colonizing new habitats (Oertli & Parris, 2019)

Local environmental factors such as pond size, depth, presence of aquatic vegetation, water quality, and hydroperiod are some of the factors found to be important drivers of species richness and community composition (Goertzen & Suhling, 2012; Heino, Bini, et al., 2017; Holtmann et al., 2018; Thornhill et al., 2017). Regional environmental variables such as the proportion of impervious land cover and number of waterbodies surrounding ponds can also influence community composition and alpha diversity (Holtmann et al., 2018; Thornhill et al., 2017). Both sets of factors can affect community characteristics to different extents or interact in ways that determine community patterns at multiple spatial and temporal scales (Hill et al., 2019; Holtmann et al., 2017; Thornhill et al., 2017). Research findings, however, are sometimes contradictory and this has been attributed to differences in responses among taxonomic groups, the spatial scale of studies as well as specific characteristics of a given urban area (Oertli & Parris, 2019). For example, macroinvertebrate diversity responses to urban conditions have been shown to vary according to their dispersal capabilities and sensitivity to habitat conditions (Blicharska et al., 2016; Liao et al., 2020). The relationships between a given environmental variable and a community characteristic can also vary, depending on the diversity metric examined (for example, species richness, functional diversity, beta diversity, LCBD). This variation reflects the different ways that environmental gradients affect communities (Gallardo et al., 2011; Hill et al., 2019); for example, regional environmental variables such as surrounding land use may be more important for LCBD than alpha diversity metrics (Heino, Bini, et al., 2017). A potential explanation for this is that barriers to dispersal in the surrounding landscape create isolated habitats where community composition is not supported by migration of species from habitats that may be otherwise similar (Heino, Bini, et al., 2017).

1.2.2.2 Ecological connectivity

Community structure and variation patterns are influenced by the movement of (and interaction) of species among a set of communities (Leibold et al., 2004; Wilson, 1992). The ability of organisms to find, select and move to, or disperse to, suitable habitats also play a role in structuring communities (Morin, 2009). Dispersal is a critical ecological process that not only ensures populations greater access to habitat resources but is also necessary for maintaining or increasing genetic diversity, colonization of new habitats and preventing local extinctions (Baguette et al., 2013).

The extent to which the landscape supports or impedes the movement of organisms from one suitable habitat to another is the subject of ecological connectivity research that has become increasingly relevant in landscapes subject to fragmentation and anthropogenic land uses (Taylor et al., 1993). Connectivity is broadly classified as structural and functional. Structural connectivity refers to landscape characteristics (composition and arrangement) that are conducive or present barriers to movement from one suitable habitat to another (Watson et al., 2017). Functional connectivity, in addition to landscape features, considers the biological and behavioural capacities of organisms to traverse a landscape in order to move between habitat patches (Baguette et al., 2013; Taylor et al., 2006; Tischendorf & Fahrig, 2000). Differences in macroinvertebrate community composition between ponds within an urban area have been shown to be not only associated with conditions of individual ponds but also the functional connectivity (availability and access to other ponds) among ponds in a region – i.e. 'the pondscape' (Hyseni et al., 2021; Thornhill et al., 2017). Measures applied to quantify connectivity and examine the effect of connectivity (or isolation) on diversity include pond density within given distances, distance to nearest ponds, availability of green spaces (favourable for the movement of organisms between ponds) and cost distance (distance integrating relative ease of movement across different land cover types) although connectivity remains relatively less explored for urban macroinvertebrate groups despite the importance of dispersal for pond communities (Bounas et al., 2020; Gledhill et al., 2008; Hill et al., 2019; Liao et al., 2020; Parris, 2006).

1.2.3 Importance for conservation and management

Where ponds are abundant freshwater ecosystems in urban landscapes, understanding their biodiversity patterns and responses to environmental factors is key to assessing their relative importance for freshwater biodiversity and conservation. Globally, freshwater biodiversity declines at rates faster than terrestrial or marine but receive relatively less conservation and policy attention (Reid et al., 2019; WWF, 2018). The primary cause of this decline is habitat loss or degradation associated with anthropogenic use of freshwater resources worldwide (WWF, 2018). Climate change associated threats such as increasing temperatures and changes in rainfall patterns that affect the life processes and survival of organisms are also emerging threats, further worsening the freshwater biodiversity crisis

that is expected to continue into the near future (Reid et al., 2019; Strayer & Dudgeon, 2010). Among the most diverse and important groups of freshwater organisms are macroinvertebrates (including aquatic insects, snails and bivalves, and crustaceans) that form an integral part of freshwater biodiversity, food webs and are key to functioning ecosystems (Wallace & Webster, 1996). Macroinvertebrates comprise multiple taxonomic groups that occupy different trophic levels (predators, secondary producers) and different substrates or microhabitats within waterbodies (Yule and Yong). Their life strategies exhibit great diversity in reproduction, movement, and behaviour with their responses to key water parameters making them useful bioindicators of water or habitat quality (Dolédec & Statzner, 2008).

To date however, there remain critical knowledge gaps in the taxonomy, distribution and ecology of tropical species (Sundar et al., 2020). These knowledge gaps have implications for conservation with the lack of data leading to uncertainty in the effects of multiple current and emerging threats on tropical freshwater macroinvertebrate. Knowledge on biodiversity patterns, responses to anthropogenic and climate stressors (pollutants, warming temperatures, changing rainfall patterns, biological invasions) are necessary to devise policy and research strategies that predict and mitigate the impacts on biodiversity. Liew et al., (2020) further emphasize the need for freshwater biodiversity and ecological data from southeast Asia to develop precise predictive models that can be used to specifically address conservation needs in the region. Beyond the tropics, macroinvertebrate ecology data such as functional diversity patterns will also be relevant across biogeographical regions where urbanization and climate change related changes to temperature and hydrology are predicted to affect freshwater biodiversity.

Complementary diversity measures (taxonomic and trait-based, local and regional scale) and their relationships with environmental factors are often investigated to discern mechanisms underlying species distribution and provide the underpinning information for biodiversity monitoring and to help improve management strategies (Goertzen & Suhling, 2012; Heino, Bini, et al., 2017; Oertli & Parris, 2019; Perron & Pick, 2020). The proximity of urban ponds to anthropogenic stressors such as pollution, increased sediment loads, disturbances (e.g. dredging) and higher urban temperatures make it necessary to understand how each of these acts, often together, on different species, ecological processes and ultimately ecological functions (Gallardo et al., 2011; Thornhill et al., 2018). Characterizing functional diversity (species trait presence and distribution patterns) has become increasingly important in assessing pond biodiversity as the roles of species within a community are associated with important ecosystem functions including water purification and nutrient cycling (Thornhill et al., 2017). Functional composition in turn is associated with ecosystem stability as when for example, multiple species with similar functional roles or traits are present within a community and

buffer it against the loss of one species (Biggs et al., 2020). Taxonomic diversity patterns may not necessarily correspond to functional diversity patterns and understanding the links between them, as well as differential responses to environment constraints, has implications for strategies in conservation and management (Devictor et al., 2010), especially with urban ecosystems such as ponds which need to balance the needs of wildlife with the needs of people (e.g. water regulation, aesthetics, etc.). Similarly, incorporating landscape-scale beta diversity measures is important for management efforts where the goal is to maximize biodiversity over larger areas rather than an individual pond (Socolar et al., 2016).

Ecological connectivity research is often applied in terrestrial conservation efforts. In cases where natural landscapes have become fragmented through anthropogenic land use, e.g. farming and transportation, the focus is often on preserving corridors through which wildlife movement can occur (Epps et al., 2018). Ecological connectivity is also relevant in habitat rehabilitation efforts where the aim is not only to improve degraded habitats but also ensure organisms can move into or colonize the new habitats (Baguette et al., 2013). Urban landscapes, however, present a predominantly built environment with natural or semi-natural land covers (forest parks, ponds, rivers, gardens) often occurring in patches among an inhospitable matrix of anthropogenic structures and activities, reducing freshwater connectivity in these landscapes. Nevertheless, as understanding of urban ecosystems' potential for biodiversity and ecosystem services provisioning grows, it becomes necessary to consider factors such as connectivity that are critical to ecological processes.

Studies of landscape connectivity in urban areas tend to focus on mammals and birds, with research applying techniques such as GPS tracking and landscape genetics (LaPoint et al., 2015). These methods allow assessment of the degree of connectivity among urban habitat patches by tracking individual movements or species distribution. The findings of such research have practical implications for urban habitat management and land use decisions, since they allow identification of urban features (under road passages, tree planting locations) that can be modified to improve connectivity for specific taxa (LaPoint et al., 2015; Tremblay & St. Clair, 2009). Similarly, connectivity modelling can be applied to characterize the degree of connectivity across different areas within an urban region, and allow prediction of the impact of urban development or the effectiveness of urban habitat creation (Kirk et al., 2018). Studies of the structural and functional connectivity of urban habitats for arthropods and molluscs are relatively scarce (LaPoint et al., 2015) but studies that explore connectivity among urban pond macroinvertebrates have suggested that structural connectivity metrics such as pond isolation and proximity for species distribution and diversity (Hill et al., 2019; Liao et al., 2022) can have small but important effects on species distribution and diversity. Other studies have explored the functional connectivity of pond and aquatic habitats by relying on metrics that consider surrounding land use and

known biological or behaviour traits (e.g. dispersal distances, responses to different land covers) of multiple taxa. Characterizing the extent of habitat connectivity in a given urban area can inform urban planning and help assess the impact of development scenarios on biodiversity (Hyseni et al., 2021; Kirk et al., 2018; Thornhill, 2013).

1.3 Urban biodiversity management in Greater Kuala Lumpur

1.3.1 Regional importance of urban biodiversity

Urban areas in Southeast Asia host considerable terrestrial and freshwater biodiversity, native and non-native, in patches of remnant or constructed habitats (Corlett, 2010; Oh et al., 2018; Tan & Abdul Hamid, 2014; Tee et al., 2018; Teo et al., 2003). Ponds and wetlands in particular remain cultural and materially important within urban landscapes (Moorhouse et al., 2021). In addition, there are increasing examples of urban policies and practices that incorporate 'green' or 'blue' spaces, so semi-natural features like retention ponds and wetlands that are expected to provide recreational and educational benefits (Linh et al., 2023; PUB, 2013; Vojinovic et al., 2021). However, the biodiversity patterns and conservation value for freshwater species of these aquatic urban ecosystems remains poorly understood.

1.3.2 Current trends in urban biodiversity conservation in Greater Kuala Lumpur

Greater Kuala Lumpur (GKL) represents a tropical urban landscape with a heterogeneous land use configuration that is changing rapidly. Past and current urban biodiversity research in Malaysia has focused on diversity of terrestrial fauna (Lim et al., 2018; Rahman et al., 2018; Tee et al., 2018), perceptions and attitudes towards remnant urban forests and urban parks (Hassan et al., 2019; Ibrahim et al., 2020; Nath & Magendran, 2020; Norhuzailin & Norsidah, 2015), urban tree composition (Abdullah et al., 2018)and urban green space connectivity (Danneck et al., 2023). Local community and environmental groups play a key role in public education on the conservation value of forest patches within the highly urban regions in GKL. Despite their abundance and structural variation (Teo et al., 2021), few studies have characterized diversity of pond (or lentic) habitats in GKL (Lee et al., 2019; Razak & Sharip, 2019). Lee et al. (2019) had limited spatial scope but demonstrated Chironomidae response to surrounding land use whereas Razak et al., 2019 characterized zooplankton community responses to urbanization.

1.3.3 Urban pond management in Greater Kuala Lumpur

While understudied, ponds and wetlands are increasingly constructed in GKL to aid with water management (DID, 2012); the federal city of Putrajaya is an example of a city-wide planned wetland network constructed for ecosystem service benefits such as flood mitigation but with biodiversity enhancement explicitly included as a key target of the wetland network (Majizat et al., 2016; Moser, 2010; Noordin et al., 2017). With an abundance of natural and constructed wetlands, as well as various

park lakes and retention ponds (Teo et al., 2021), set against the rapidly changing city-scape, GKL provides an opportunity to explore the distribution of freshwater organisms in urban tropical ponds and wetlands. The complex mosaic of land uses also allows for study of the effects of various landscape factors on wetland ecosystems, notably biological community assembly and dynamics. In GKL, ponds are usually constructed habitats and often designed with specific functions which may lead to homogenous conditions, although there is much variation habitat structure, history and management intensities.

1.4 Research problems and questions

1.4.1 Research problem

This thesis addresses the substantial gap in biodiversity data from ponds in tropical urban landscapes. There is limited evidence for urban pond management and conservation from regions beyond Australia and the geographical north (Oertli & Parris, 2019). Quantifying and characterizing the diversity patterns of pond communities and their responses to environmental factors and urban stressors (across multiple scales) is necessary to assess their value for local species conservation, develop benchmarks and indicators for monitoring and management, and understand their relative importance for and relationship with multiple ecosystem benefits. The present study asks key questions that can inform policy, research and management priorities for GKL as well as other cities in the region. Second, studies of the role of habitat connectivity in structuring pond communities are equivocal (Hill et al., 2019; Hyseni et al., 2021) and there is a need for data from landscapes where different urban or development patterns can elucidate the relative importance of structural and functional connectivity as well as context dependent patterns in urban pond ecology. Finally, across the tropical region, freshwater macroinvertebrate species are experiencing threats from climate change and other stressors with conservation efforts challenged by broad knowledge gaps in how multiple macroinvertebrate groups respond to stressors (Sundar et al., 2020). This study addresses key ecological knowledge gaps that can further inform research directions and strategies for tropical freshwater species conservation globally, especially the potential importance of urban habitats.

1.4.2 Research questions and hypotheses

The following questions guided the systematic literature review and analyses.

1. What is the extent and ecological focal point of research on freshwater biodiversity in tropical urban landscapes?

   This question directs guided the systematic literature review (Chapter 2)

2. How do macroinvertebrate taxonomic alpha and gamma diversity patterns vary among ponds in Greater Kuala Lumpur and which habitat and surrounding land use factors influence variation?

Aquatic vegetation cover, area, pH, conductivity, proportion of impervious and vegetated surrounding land cover are variables that influence habitat quality of urban ponds. ***Therefore, I hypothesize that there is high variation in taxonomic patterns among ponds in GKL in response to variation in physical habitat characteristics and surrounding land cover.***

3. What factors are associated with variation in taxonomic community composition (beta diversity) of ponds in Greater Kuala Lumpur?

   Ponds across landscapes may exhibit community heterogeneity driven by taxonomic turnover in response to differences in environmental conditions across the urban matrix. ***Therefore, I hypothesize that there is high community heterogeneity driven by taxonomic turnover and influenced by landscape scale variation in pond habitat characteristics as well as surrounding land cover.***

4. How does functional (trait-based) alpha diversity vary among ponds in Greater Kuala Lumpur and what are the habitat and land use factors affecting it?

   Functional diversity may be positively correlated with taxonomic alpha diversity of freshwater habitats (Gallardo et al., 2011). ***Therefore, I hypothesize that trait based (functional) diversity patterns reflect taxonomic diversity and show high variation in response to variation in local habitat characteristics and surrounding land cover.***

5. How do these factors drive variation in functional beta diversity (compositional variation) of macroinvertebrate communities among ponds in Greater Kuala Lumpur?

   Functional beta diversity may be driven primarily by richness difference in heterogeneous landscapes in response to isolation and urban environmental constraints (Gianuca et al., 2017; Hill et al., 2019). ***Therefore, I hypothesize that pond communities exhibit high functional beta diversity driven by richness difference in response to local habitat characteristics and surrounding land cover****.*

6. How does pond connectivity influence community composition and diversity patterns in Greater Kuala Lumpur?

   Proximity to neighbouring freshwater habitats and number of neighbouring habitats increase the total habitat area available and promote dispersal that supports greater diversity (Gledhill et al., 2008; Hyseni et al., 2021). ***Therefore****, **I hypothesize that both structural and functional connectivity among ponds increase macroinvertebrate alpha diversity whereas isolation leads to community heterogeneity.***

1.5 Research aim & objectives

1.5.1 Research aim

The aim of this research is to characterize macroinvertebrate biodiversity patterns among pond habitats in GKL, and to understand their environmental and spatial or regional drivers. The heterogeneous land use matrix and diversity of pond types in GKL (Teo et al., 2021) suggests that urban ponds may be supporting diverse ecological assemblages which at present remain largely unknown, and have yet to be examined systematically. This research was conducted to address this fundamental gap. The focal taxa in this study are macroinvertebrates. These organisms comprise multiple taxonomic groups with different functional roles and cultural value within *pond* ecosystems: (i) as grazers or shredders processing organic matter and recycling nutrients, (ii) as food sources for higher organisms such as fishes and birds (Sundar et al., 2020), (iii) they also include culturally important taxa such as dragonflies (Ngiam et al., 2017) and potentially threatening taxa such as mosquitoes (Adnan et al., 2021). The research approach here applies a range of community analyses tools to characterize different components of urban pond biodiversity and their responses to environmental conditions within the Greater Kuala Lumpur metropolitan landscape. The findings provide baseline data on pond macroinvertebrate distribution patterns and a starting point for future research and planning that aim to design and manage urban ponds as habitats for freshwater biodiversity.

1.5.2 Research objectives

The following are the specific objectives of the research.

Objective 1: Describe alpha and beta diversity patterns and identify drivers of macroinvertebrate diversity at the local scale (richness, diversity, abundance) and landscape scale (beta diversity and its components) in Greater Kuala Lumpur (GKL).

Research questions 2 and 3 address this objective by quantifying taxonomic alpha and beta diversity and statistically testing the relationship between key environmental and land use factors and diversity measures.

Objective 2: Describe trait-based (functional) pond community patterns and identify environmental and spatial drivers of trait distribution patterns. i

Research questions 4 and 5 address this objective by quantifying functional (trait-based) alpha and beta diversity and using statistical tests to determine the role of key environmental and land use factors in structuring functional composition.

Objective 3: Map ecological connectivity for pond macroinvertebrates in GKL and assess the influence of connectivity on diversity patterns.

Research question 6 addresses this objective by calculating connectivity measures for ponds across GKL and constructing connectivity maps.

1.6 Thesis outline

There are seven chapters in this thesis (Figure 1.2). Chapter 2 is a systematic review of literature on the ecology and ecosystem services of lentic habitats in tropical urban landscapes. This chapter has already been published (in the journal *Freshwater Biology*) in the form presented in the thesis (Gebreselassie et al., 2022). It summarizes current research findings, describes trends in research theme, identifies knowledge and research gaps and recommends directions for future research. The findings were important for highlighting areas of urban lentic habitat research that remain underrepresented or unexplored in tropical urban landscapes.

Chapter 3 describes the data collection methods and analyses used for Chapters 4, 5, 6. These are summarised in Figure 1.2. In addition to water quality and pond characteristics, remotely sensed pond and land use maps were used to derive environmental and connectivity metrics. Multiple measures quantifying different aspect of biodiversity were calculated and a wide range of statistical analyses where then undertaken to address research objectives.

Chapter 4 quantifies taxonomic diversity and describes community composition patterns based on macroinvertebrate occurrence and abundance in ponds. It identifies the factors associated with taxonomic diversity and those that structure communities or assemblages. It highlights the importance of trait based as well as taxa-specific assessments of biodiversity and shows the relative importance of pond (habitat) design elements in determining diversity patterns.

Chapter 5 explores macroinvertebrate diversity and assemblages in terms of the biological and ecological traits of species and taxonomic groups. This chapter quantifies trait-based diversity and the extent of variation in distribution of traits among urban ponds. It also highlights environmental factors that influence functional diversity and trait distribution.

Chapter 6 characterizes and quantifies ecological connectivity of ponds in Greater Kuala Lumpur. It identifies areas with high connectivity and ponds important for overall pond network connectivity. It also calculates functional connectivity metrics for sampled ponds and examines the relationship between diversity patterns and the connectivity levels of the urban ponds.

Chapter 7 is the final chapter and synthesises the main scientific findings of this research. The findings have implications for conservation and/or management strategies, highlighting the importance of aquatic vegetation, addressing water quality and pond distribution. The limitations of this study and

future directions for urban pond biodiversity research in Greater Kuala Lumpur are also discussed in this chapter.

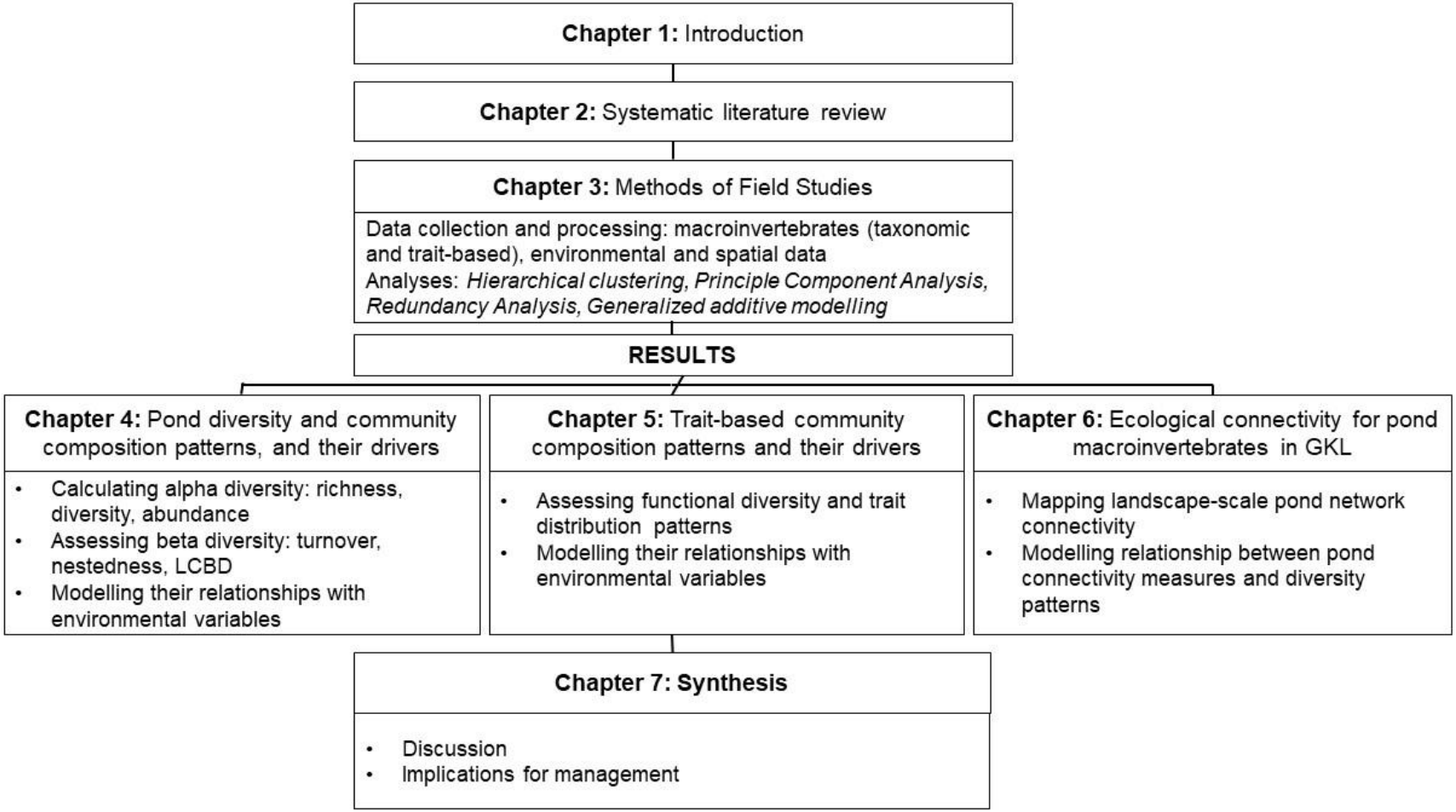


Figure 1.2 Thesis structure.

Chapter Two

A review of current knowledge and research priorities for conservation of lentic biodiversity in tropical wet and monsoonal urban landscapes

2. A REVIEW OF CURRENT KNOWLEDGE AND RESEARCH PRIORITIES FOR CONSERVATION OF LENTIC BIODIVERSITY IN TROPICAL WET AND MONSOONAL URBAN LANDSCAPES

2.1 Introduction

Freshwater species and populations are declining globally at rates higher than their terrestrial counterparts, with a decline in the annual population index (WWF Living Planet Index) of 3.9% compared to 1.1% for terrestrial species (Reid et al., 2019; WWF, 2018). Urbanisation is one of the major threats to freshwater biodiversity, especially in tropical regions where urban expansion is rapid and there is limited consideration of the ecological implications of urban growth (Cantonati et al., 2020; Sundar et al., 2020).

Urbanisation may impact biodiversity through the loss and modification of natural habitats (Grimm et al., 2008; McDonald et al., 2020). Increasingly, research is highlighting the potential of urban areas to support terrestrial and freshwater biodiversity (Beninde et al., 2015; Ives et al., 2016; Oertli & Parris, 2019; Seto et al., 2012). New habitats (e.g., stormwater ponds) are often created as cities expand, forming new ecosystems that may contribute to freshwater diversity (Briers, 2014; Hassall, 2014; Holzer, 2014). The sustainable management and conservation of freshwater diversity requires an approach that considers the processes governing species distribution patterns at multiple spatial and temporal scales, as well as their responses, in terms of diversity and function, to environmental factors (Geist, 2011). This may be particularly important in urban environments where freshwater habitats are often managed to cater for societal needs (Noble & Hassall, 2014).

Tropical freshwater ecosystems are characterized by high endemism and species richness (Barlow et al., 2018; Cantonati et al., 2020; Dudgeon et al., 2006) attributed to geographical isolation and specialization (Boyero et al., 2021; Cantonati et al., 2020). The high degree of specialisation in the tropics is facilitated by environmental conditions (notably high solar energy), including their temporal stability (Brown, 2014; Fine, 2015). Changes to these conditions, especially temporal patterns and the magnitude of variability, therefore has great potential to impact tropical freshwater ecosystems (Cantonati et al., 2020; Jardine et al., 2015; Liew, Lim, Low, Mowe, Ng, Zeng, et al., 2020; Wohl et al., 2012). In urban areas, changes to temporal patterns arise as a result of management regimes and infrastructure that can alter the dynamics of runoff (e.g. concrete surfaces) and channel flow (e.g. weirs,

canalisation of water courses), and may be accompanied by deterioration in physical and chemical quality (Grimm et al., 2008; McGrane, 2016).

The ecological communities of ponds and wetlands (lentic systems) differ in their responses to urban conditions to those of streams and rivers (lotic systems) (Hill et al., 2017; Prescott & Eason, 2018). Lotic systems are generally characterized by impoverished communities, and this is attributed to poor water quality, alteration to flow regime and loss of habitat heterogeneity and variability in urban areas (Allan, 2004; Beavan et al., 2001; Jesús-Crespo & Ramírez, 2011; Reid & Tippler, 2019). While, lentic systems exhibit inconsistent patterns, with some research highlighting urban ponds and wetlands as important refuges for aquatic and semi-aquatic species in urban areas, including invertebrates, amphibians, birds and bats (Ancillotto et al., 2019; Hamer & Parris, 2011; Hill et al., 2019; Holtmann et al., 2017; Johansson et al., 2019; O'Brien, 2014; Prescott & Eason, 2018) but other studies finding urban lentic habitats to be ecologically impaired or dominated by invasive species (Noble & Hassall, 2014; J. S. Sinclair et al., 2020)

Current research suggests that with management strategies that promote biodiversity, including maintaining multiple wetlands that vary in characteristics th

at are favourable to different species, small wetlands and ponds may be of particular ecological value (Blicharska et al., 2016; Hassall, 2014). Urban wetlands may also be an important component of Water Sensitive Urban Design approaches, providing Nature-based Solutions such as flood alleviation, local climate regulation, and retention of both sediment and nutrients (Hamel & Tan, 2021; Wong & Brown, 2009). In addition, they provide provisioning services such as freshwater, food and fuel and cultural services linked to urban ponds and wetlands include well-being and aesthetic benefits, as well as opportunities for education and recreation (Manuel, 2003; Ngiam et al., 2017; Thornhill et al., 2019).

Although the physical structure of urban landscapes tends to be broadly similar globally (Wu, 2014), biogeographic contexts, socioeconomic circumstances and socioecological settings differ markedly from country to country. Also, in the tropics the demands placed on urban infrastructure by the climate differ from those in temperate regions (Lechner et al., 2020; Lourdes et al., 2021; Muñoz-Erickson et al., 2014; Ramírez et al., 2009). These factors may influence the nature, diversity and perceived value of urban wetlands. Intense rainfall events in tropical cities lead to frequent flooding, with floodwaters that are high in nutrients and suspended sediment, and may be highly polluted (Parkinson et al., 2010; Rivard et al., 2006). These issues require different approaches to management to those needed in cities where such floods are less frequent or intense. In tropical Kuala Lumpur (KL), for example, all new housing developments require sediment retention ponds designed to receive overland flow, resulting in a proliferation of new wetland habitats across the city. However, while these and other novel habitats

(as well as remnant natural ones) may contribute positively to pond diversity in tropical urban areas, negative social or environmental impacts including health risks associated with standing water, notably mosquitos (Rivard et al., 2006), may create social pressures that run counter to the desire to conserve or create new urban wetlands.

The potential significance of urban wetlands in tropical regions, in terms of their diversity and the services they provide, raises questions about how much we currently know about these habitats. As an evidence base is needed to support the conservation and management of urban wetlands (Ehrenfeld, 2000; McInnes, 2014), and the threats posed by the ongoing and rapid expansion of tropical cities are increasing, a review of current knowledge is necessary and timely. In this paper, we present a systematic review of literature on the ecology and diversity of lentic freshwater in tropical urban landscapes. We focus on urban areas in tropical wet and monsoonal climate regions, as defined by the Koppen climate classification scheme (Peel et al., 2007); hereafter these regions are simply referred to as 'tropical.' The review addresses the following questions: 1) How much research has been conducted on ponds and wetlands in tropical urban areas? 2) What are the ecological focal points of published literature, and which countries does it come from? 3) What patterns of diversity (species richness, community composition) do tropical urban ponds and wetlands exhibit? 4) What are the factors influencing this diversity? and 5) What ecosystem services are provided by urban ponds and wetlands in tropical regions? Identifying limitations in our understanding of the ecology and diversity of tropical urban wetlands is a key focus of the review, and so we provide recommendations for future research needed to improve this understanding and to guide their conservation.

2.2 Methods

2.2.1 Data collection

A literature search was conducted to find original research articles that focused on inland, lentic freshwater bodies (ponds and wetlands) within tropical urban landscapes. The scope was limited to studies of their distribution, ecology and ecosystem service provision, so excluded purely hydrological or water quality studies. The search was conducted through the online publication databases Web of Science and Scopus. The following search terms were used, adapted from Oertli & Parris (2019):

Web of Science: TOPIC: ((urban* OR cit*) AND (pond OR wetland*)) AND TOPIC: (flora OR plant* OR macrophyte* OR vertebrate* OR invertebrate* OR mammal* OR fish* OR bird OR insect OR amphibi* OR frog* OR macroinvertebrate* OR crustac* OR dragonfl* OR damselfly* OR odonat* OR reptilian OR mollusc* OR beetle* OR coleopter* OR butterfly* OR turtle* OR fung* OR biodiversity* OR diversity) NOT TOPIC: (marine OR coast*)

Scopus: (TITLE-ABS-KEY (urban*) OR TITLE-ABS-KEY (city) OR TITLE-ABS-KEY (cities) AND TITLE-ABS-KEY (pond*) OR TITLE-ABS-KEY (wetland*) OR TITLE-ABS-KEY (lentic) AND TITLE-ABS-KEY (biodiversity) OR TITLE-ABS-KEY (diversity) OR TITLE-ABS-KEY (richness) OR TITLE-ABS-KEY (ecosystem*)) AND PUBYEAR > 1989 AND (LIMIT-TO (DOCTYPE , "ar")) AND (LIMIT-TO (LANGUAGE, "English"))

Web of Science returned 3,131 and Scopus 2,531 publications (Figure 2.1). We screened these based on several criteria. All publications had to be studies of inland freshwater lentic, pond or wetland habitats within an urban landscape. The title, abstract and key words were read to determine whether an article met these criteria. In some cases, the methods section of the paper was read to ascertain landscape and waterbody type. Reviews and studies that focused solely on the chemical assessment of water quality, constructed waste-water treatment wetlands (unless they are also managed or evaluated as biodiversity habitats), performance of wetlands (constructed or natural) in water treatment, aquaculture or coastal wetlands or brackish water habitats were excluded. The resulting set of publications was then categorized by the climate of the study region. Only studies undertaken in urban areas with tropical wet and monsoon climates were included in the review.

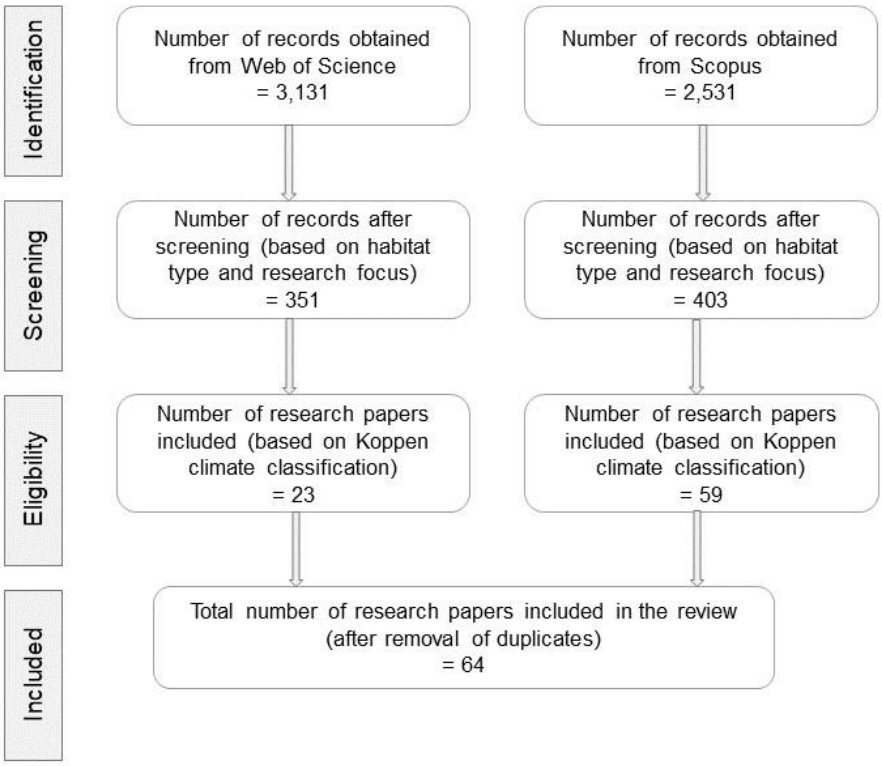


Figure 2.1 Summary of research paper screening and selection process (adapted from Moher et al., 2009).

2.2.2 Data analysis

For each paper, the geographical location (country and city name) of the study, and the type(s) of freshwater habitat (Table 2.1) were recorded. We also recorded the research objectives and spatial scale of each study to determine research scope, identify research focal points and understand the extent to which studies dealt with single or multiple wetlands. For studies that focused on understanding relationships between environmental conditions and taxonomic diversity, subject taxa and habitat type were recorded. Studies that were primarily concerned with ecosystem service provision or habitat distribution were categorized based on their research objectives.

Table 2.1 Research attributes used to characterize papers on urban ponds and wetlands in tropical urban areas (n=64).

Research attribute	Category	Description
Year	Pre-2000	Published before the year 2000
	2000-2004	Published between 2000-2004
	2005-2009	Published between 2005-2009
	2010-2014	Published between 2010-2014
	2015-2019	Published between 2015-2019
	2020 onwards	Published in the years 2020 and 2021
Spatial scale	1-2 sites	1-2 habitats sampled
	3-5 sites	3 -5 habitats sampled
	6-15 sites	6 -15 habitats sampled
	More than 15 sites	More than 15 habitats sampled
	City wide	All pond/wetland water bodies across entire city mapped with remotely-sensed data
Habitat type	Wetland	Includes remnant floodplain wetlands, park wetlands, habitats primarily described as such by authors, natural and constructed
	Lakes	Lakes as described by authors, natural or constructed
	Ponds	Ponds as described by authors, natural or constructed
	Reservoirs	Reservoirs as described by authors, constructed
	Multiple	More than one of the above

Research attribute	Category	Description
Taxonomic group	None	No subject taxonomic group
	Aquatic vegetation	Includes emergent, submerged, and/or floating plants
	Plankton	Includes phytoplankton and/or zooplankton
	Macroinvertebrate	Includes aquatic and semi-aquatic macroinvertebrate groups
	Vertebrate	Includes birds, amphibians, fish
	Multiple	More than one of the above groups
Habitat origin	Natural	Natural, modified habitats within an urban landscape
	Constructed	Ancient or modern constructed pond and wetland habitat
	Mixed	Both natural and constructed pond and wetland habitat
	Not mentioned (non)	Origin of the habitat not discussed in the paper
Research focus	Diversity	Habitat diversity or biodiversity
	Habitat cover/loss/expansion	Quantification of habitat cover within an urban landscape, or habitat loss or gain over years
	Bio-indicators of water quality	Assessing water quality with organisms as indicators
	Ecosystem services, disservices	Includes assessing value perception, disservices, provisional, regulating or supporting services
	More than one of the above	More than one of the above
	None	Other than the above, unique focus
Temporal scale	Single season	One season, one year
	Single year	Different seasons, one year
	Multiple years	Same/different seasons, multiple years

Research attribute	Category	Description
Biodiversity measures	None	No biodiversity assessment was conducted
	Inventory	Checklist of a specific taxonomic group occurrence
	Single measure	Single metric used (for e.g. species richness only)
	Multiple measures	Multiple metrics used (for e.g. richness, abundance, diversity)
	Multivariate analyses	Multiple diversity measures used and relationships with environmental or landscape variables examined with multivariate analyses or modelling
Variables examined as correlates of biodiversity patterns in wetlands	None	Biodiversity and/or correlates not assessed
	Water quality measures	For example pH, nutrient concentration levels
	Physical habitat characteristics	For example depth, substrate type, bank material
	Landscape composition	For example presence of roads, built structures, surrounding land cover type
	Multiple	More than one of the above
Ecosystem services	None	Ecosystem services not assessed
	Mapping & Inventory	Mapping and/or inventory of ecosystem services provided by habitat
	Use or function assessment	Assesses habitat use by fauna or human communities, or the performance of ecosystem function
	Perception of value assessment	Documents attitudes and perceptions of human communities toward the habitat documents
Urban population % (of total country population)	Under 20	Less than 20%
	21-50	21-50%
	51-80	51-80%
	81-100	81-100%

To summarize research approaches and detect trends in the literature, each paper was characterized using eleven research attributes (Table 2.1). These data were then analysed using multiple correspondence analysis (MCA), using the package FactoMineR (Husson et al., 2016). MCA is a form of correspondence analysis for categorical datasets (Abdi & Williams, 2010) and enables the assessment

of the multivariate similarity/dissimilarity of samples (in this case each paper was a sample), and the identification of attributes accounting for variation among samples (Abdi & Williams, 2010).

2.3 Results

2.3.1 Characteristics and focal points of tropical urban pond and wetland research

The systematic search identified 64 papers (see Appendix Table A1) on ponds and wetlands that considered the biodiversity and distribution of lentic habitats in tropical urban areas from fifteen countries (Figure 2.2a). The earliest publication was from 1995 and the number of papers published in subsequent years ranged from three to eight until 2020 (Figure 2.2b). The year 2020 had the highest number of publications for a single year (n=13). An increasing trend in publications in the past decade is notable (about 70% of studies were published after 2011), reflecting global trends in wetland research (Oertli & Parris, 2019).

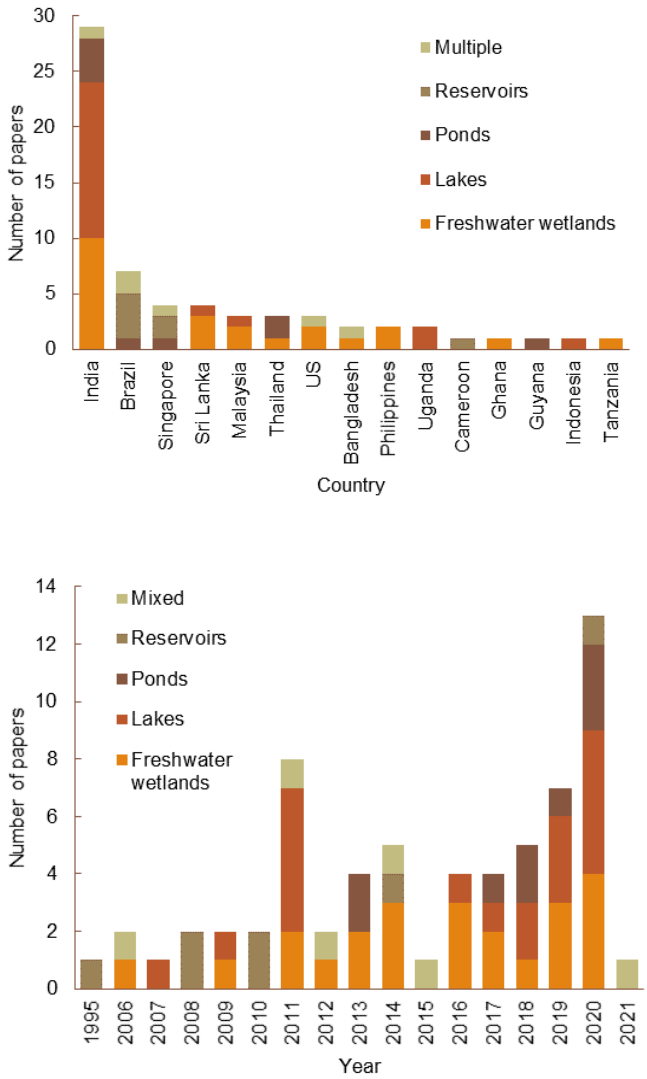


Figure 2.2 Types of urban ponds and wetlands studied in tropical urban areas by (a) country and (b) year.

The majority of the 64 studies were from urban areas in India (45%, n=29). This is notable as the proportion of the national population that is urban is one of the lowest among the tropical countries included in the review (World Development Indicators, 2020). Studies from Brazil contributed 11% (n=7) of the articles reviewed. Sri Lanka and Singapore, with the highest and lowest urban populations respectively, contributed only four papers (6%) each; the remaining countries each contributed one to three papers (Figure 2a). Overall, the review indicates that a high proportion of published studies are from relatively few countries; there is a large number from India but very few from countries in Southeast Asia (n=11), tropical Africa (n=5) and South America (n=8) are represented.

In places with higher urban populations such as Singapore (n=4) and Brazil (n=7), larger, constructed waterbodies such as reservoirs (n=7) were often the focal point of studies, whereas in cities where urban populations are smaller (e.g., Sri Lanka (n=4)), studies tended to focus on remnant natural wetlands (Figure 2b). Overall, there was no bias towards larger metropolises. Although several megacities, including Manila, Dhaka and Kolkata, with urban populations greater than 10 million, were the focus of 37.5% (n=24) of the papers, cities with populations under 10 million (n=11) and under 5 million (n=29), consisting of either principle cities or smaller urban areas, were also well represented.

Based on habitat or ecosystem descriptions and classifications used by authors, the types of ponds and wetlands studied consisted of floodplain and park wetlands, lakes and lake wetlands, reservoirs and their littoral zones, and ponds. These included natural, modified, restored, and constructed systems (Table 2.2). Ecosystems described as 'wetland habitats' were the most widely studied (23 papers), followed by lakes (19 papers). Research focused on the ecology of reservoirs (7 papers) was almost exclusively limited to Brazil and Singapore. Notably, several papers included wetlands as part of multiple habitats in studies of urban blue-green spaces (Gandhi et al., 2015; Hayes et al., 2020; Mukhopadhyay & Mazumdar, 2019; Vallejo Jr et al., 2009; Zinia & McShane, 2021). Habitats categorized as freshwater wetlands were mostly natural wetlands, whereas lakes and ponds tended to include constructed waterbodies.

Table 2.2 Description of systems covered in the reviewed literature (the proportion of papers that provide information on depth and area are shown in brackets)

Category	General characteristics	Total number of papers (percentage of total publications)	Depth range (m)		Area range (ha)	
Freshwater wetlands	Natural floodplain or constructed wetlands. Included perennial and seasonal wetlands within and around city borders. The term often encompassed different types of waterlogged or lentic habitats.	23 (36%)	-		0.3 - 12500	(10/23)
Lake and lake wetlands	Natural or constructed lakes, with the term sometimes used interchangeably with reservoirs. Some study sites are also described as wetlands surrounding major lakes.	19 (30%)	2 - 8	(2/19)	2.5 - 62.5	(7/19)
Ponds	Mostly constructed lentic waterbodies. Included those built within temples, parks, and ponds with economic functions.	9 (14%)	1.5 - 8	(2/9)	0.029 - 104	(2/9)
Reservoirs	Constructed water bodies. Term also used interchangeably with pond and lake	7 (11%)	2 - 5	(6/7)	0.5 - 59	(6/7)
Multiple	More than one of the above categories	6 (9%)	1.67-4.81	(1/6)	0.06-488	(1/6)

The papers included assessments of extent and/or change of pond and wetland distribution, biodiversity, bio-indicators of water quality, and ecosystem services provided by the ponds and wetlands (Figure 2.3a). The spatial scale of the study area was limited to one or two sites per urban area in half of the papers (n=32). The number of sites for the rest of studies ranged from 3-57 (Figure 2.3b), except for seven studies that mapped all ponds or wetlands within an urban area with aerial or satellite data. A common theme in the papers was the impact of formal and unregulated urban development on natural wetlands, especially in terms of loss in pond/wetland area to impervious land cover, and deterioration of water quality with increasing human use (Athapaththu et al., 2020; Campion & Venzke, 2011; Das & Basu, 2020; Das et al., 2020; Hettiarachchi, Athukorale, et al., 2014; Isunju & Kemp, 2016; Naigaga et al., 2011).

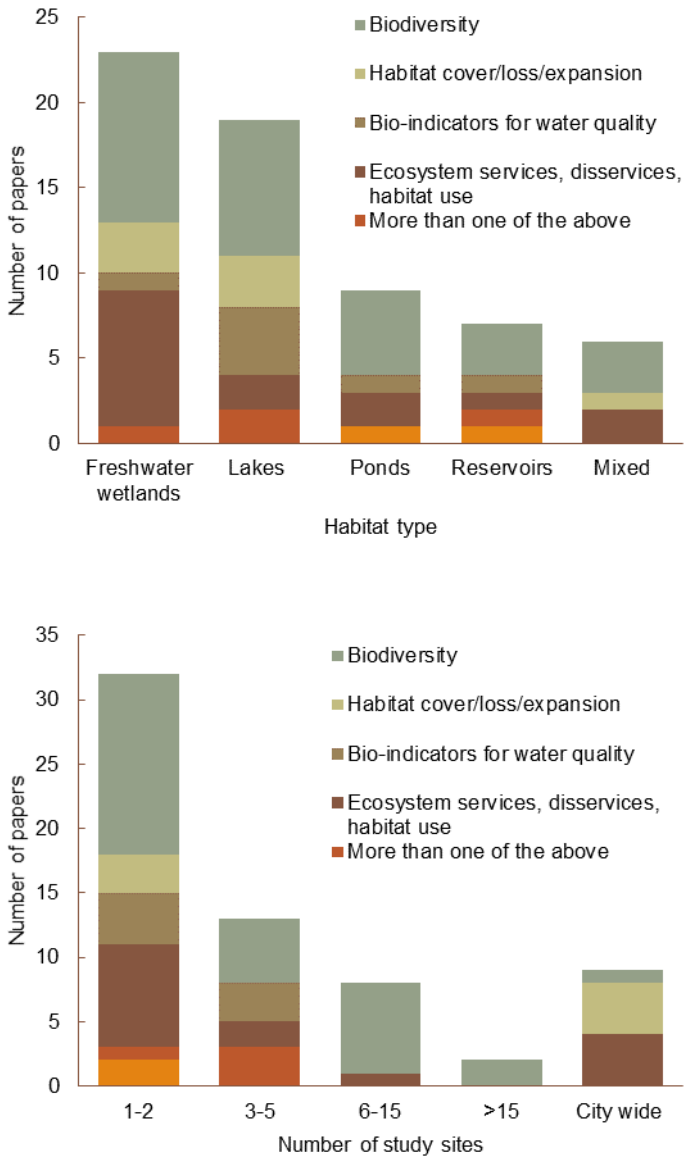


Figure 2.3 Primary research focal points of papers by type of study ecosystem (top) and number of study sites (bottom) (n=64).

Multiple correspondence analysis (MCA) revealed some similarities in approaches and trends in the research. Papers on diversity (dv) tended to have positive values for dimension 1 and 2, while papers on ecosystem services, disservices and habitat use (es) and habitat cover/loss/expansion (hb) have positive values in dimension 1 and negative values in dimension 2. These suggest that there are few studies that integrate biological components of habitats (in terms of taxonomic composition and richness) with data on perceptions and value of habitats or habitat distribution (Figure 2.4a). The papers were arranged along dimension 2 according to the percentage of urban population (pop) in the study and type of pond or wetland (type). The variable-dimension correlation plot (Figure 2.4b) shows that type and pop are both correlated with dimension 2. This corresponds to the observation made above where studies from cities with greater urban populations tended to focus on constructed

ponds and wetlands. There was no clear temporal ordering of studies (as defined using the 11 metrics) across the scatterplot (Figure 2.4-top).

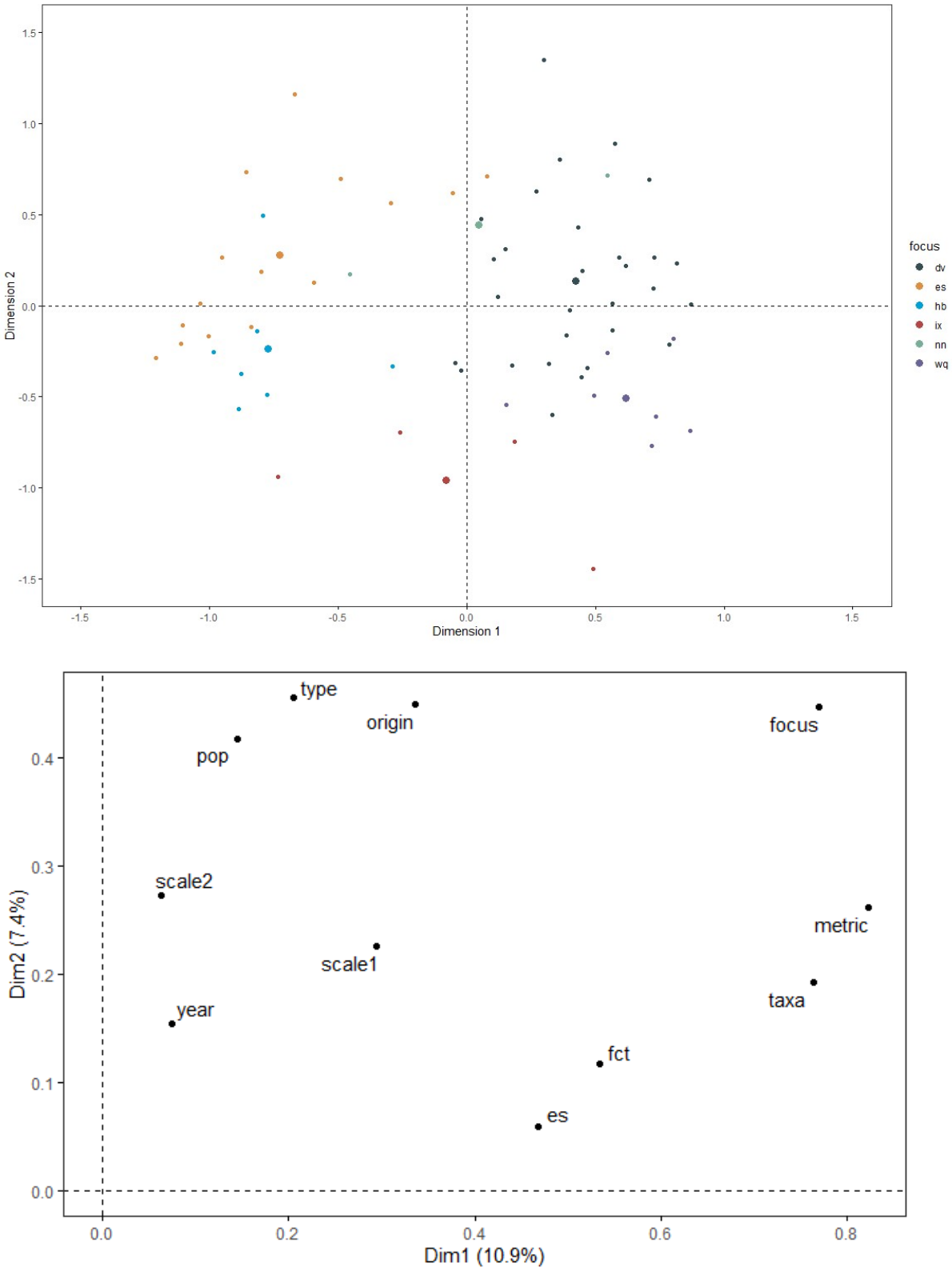


Figure 2.4 (top) Multiple correspondence analysis (MCA) plot showing distribution pattern of papers (dots) based on similarities in research attributes. Papers are colour coded by main research focus (dv-diversity; es-

ecosystem services, disservices, habitat use; hb-habitat cover/loss/expansion; ix-more than one; nn-none; wq-bio-indicators of water quality) (bottom) Dimension-variable correlation plot showing how eleven research attributes (variables) of studies are correlated to the first (Dim1) and second (Dim2) dimensions. The eleven research attributes are year (year), spatial scale (scale1), habitat type (type), taxonomic group (taxa), habitat origin (origin), research focus (focus), temporal scale (scale2), biodiversity measures (metric), variables examined as correlates of biodiversity (fct), ecosystem services (es) and urban population percentage (pop).

2.3.2 Patterns of biodiversity

Of the 64 papers reviewed, 40 addressed taxonomic diversity. Of these, 33 documented taxonomic diversity with a species inventory or the calculation of biodiversity metrics. The remaining seven assessed species diversity as a bio-indicator of water quality. Vertebrates were the most studied group (Figure 2.5a), with assessment of avian diversity a common focal point (n=9), followed by fish (n=3) and amphibians (n=2).

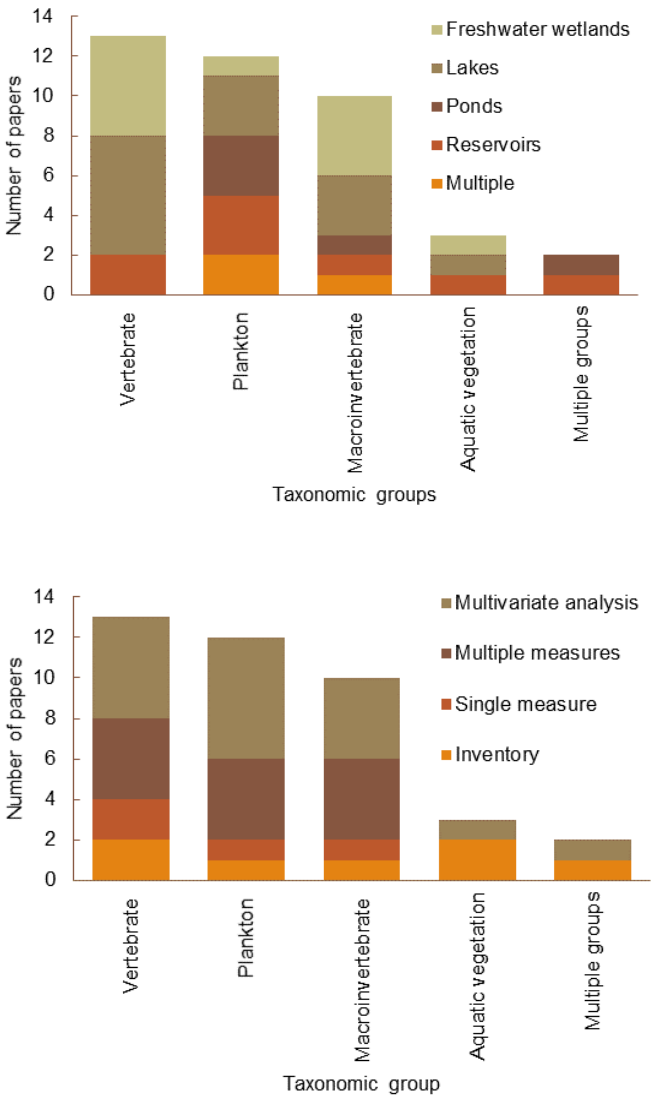


Figure 2.5 Taxonomic groups examined in studies by (a) type of study ecosystem and (b) extent of analyses carried out for each taxonomic group (n=40).

In total, 27.5% (n=11) of the 40 papers documented species presence/absence or relied on a single diversity metric, specifically richness and/or abundance of a taxonomic group. Another 30% of studies (n=12) used multiple metrics, mainly combining Shannon-Wiener and Simpson's indices for taxonomic diversity and density, in addition to species richness measures. Five studies documented functional diversity. Studies that carried out multivariate analyses and/or statistical modelling to describe relationships between diversity metrics and environmental factors made up 42.5% (n=17) of these 40 papers (Figure 2.5b). Besides these, there were another nine papers that focused on the feeding habits of birds in urban areas (Murray et al., 2018; Varner et al., 2014), introduced fish species presence (Kwik et al., 2013), and wetland plant species composition change (Hettiarachchi, Morrison, et al., 2014).

Several studies found urban habitats to be species-rich and characterized communities in terms of species composition, endemism, rarity or vulnerability status. For example, Clements et al. (2006) showed that distinct mollusc assemblages occupied different types of lentic water bodies in Singapore. Wakhid et al. (2020) found variation in aquatic insect assemblages among small lakes associated with differences in water quality and macrophyte cover. Razak and Sharip (2019) found that zooplankton diversity showed taxa-specific variation with degree of urban development, defined in terms of built environment density and human population density around the study site. Three studies assessed functional feeding group (aquatic macroinvertebrates) and dietary guild composition (urban wetland and non-wetland birds) and found that taxa distribution was associated with organic pollution (Wakhid et al., 2020) and type of urban blue-space (Hayes et al., 2020; Mukhopadhyay & Mazumdar, 2019). Phytoplankton functional group compositions and their correlation with water nutrient concentrations were examined in another three studies (Crossetti & Bicudo, 2008a, 2008b; Fonseca & Bicudo, 2010).

The relative contribution of urban wetlands and ponds to regional diversity was calculated in 10 of the 64 studies. Hayes et al. (2020), found about 10% of regional species (comprising wetland and non-wetland avian species) were present in wetlands within urban green spaces in Guyana. Similarly, Mukhopadhyay and Mazumdar (2019) found around 12% of regional bird species were represented in sub-urban areas, with wetland habitats supporting greater species richness than purely green spaces. Ansari (2017) produced an inventory of resident and migrant water bird species occurring at Surajpur Lake, and reported an occurrence of 95 species that included eight listed as Vulnerable or Near Threatened by IUCN. In contrast, Seshadri et al. (2008) reported low anuran species richness in urban freshwaters with only 14 species recorded in wetlands throughout urban Puducherry, India.

There was a limited number of studies (n=4) on problem or nuisance species (Kwik et al., 2020; Kwik et al., 2013; Reis et al., 2018; Sareein et al., 2019). Urban stormwater ponds in Singapore were found

to be populated with tolerant, non-native fish species that were considered to potentially endanger native fish species if they spread to natural sites (Kwik et al., 2013). Sareein et al. (2019) examined the correlation between mosquito species, Culex, and predatory insect density; their findings suggest that predator diversity could provide a biological control for nuisance species in urban areas.

2.3.3 Factors structuring biological communities

Some of the studies addressed spatial and temporal variation in species distribution and composition, primarily in relation to water quality. Figure 6 summarizes relationships between urban wetland diversity measures and the environmental variables considered in the published studies. Physical habitat characteristics of the wetlands were not commonly included as potential explanatory variables, nor were wider urban landscape characteristics. Generally, water quality variables and one or two physical characteristics at the site-level were the main correlates examined. Physical habitat characteristics examined included the area of the waterbody, substrate type, and presence of aquatic vegetation. The findings suggest species-specific responses to habitat features such as area (Clements et al., 2006; Razak & Sharip, 2019) and the importance of macrophyte density for odonate species richness and abundance (Wakhid et al., 2020). A positive correlation between plants and waterbird diversity was found in one study from India (Rajashekara & Venkatesha, 2018). Only one study looked at interspecies dynamics such as predator-prey relationships (Sareein et al., 2019). No papers assessed the influence of habitat origin or management practices on diversity.

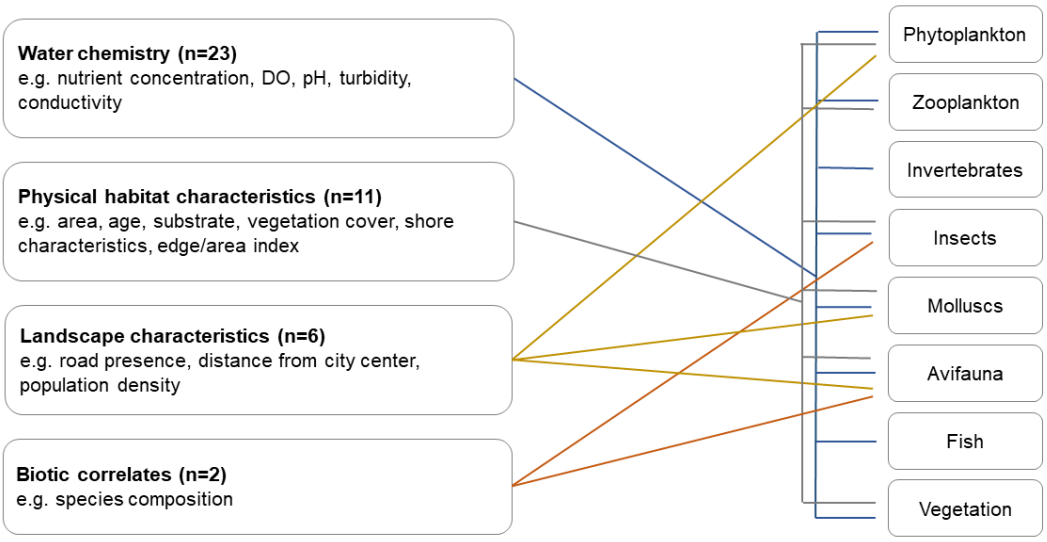


Figure 2.6 The number of studies that examined environmental factors influencing biodiversity patterns, by variable category and the specific environmental variable-taxonomic group relationships examined (n=40). For physical habitat characteristics and landscape characteristics, a maximum of one metric was measured in most studies (Note: total number of studies does not add up to 40 as some studies looked at more than one category of variables).

Empirical studies of the relative effects of dispersal and connectivity in structuring biodiversity were absent from the literature. Similarly, quantitative assessment of local environmental conditions and spatial factors at a landscape-scale and their relationship with species diversity was largely absent. The three papers that included surrounding urban land use measures indicate that the importance of landscape variables varies between taxonomic groups (Clements et al., 2006; Rajashekara & Venkatesha, 2018; Razak & Sharip, 2019). Razak and Sharip's (2019) findings reveal negative correlations between the degree of urban development within a 1km radius of lentic waterbodies and zooplankton diversity. On the other hand, Clements et al. (2006) did not find isolation from areas of human development to be a predictor for mollusc richness. These findings highlight the need for more research before generalizable patterns in habitat biodiversity-environment relationships can be determined.

Few studies examined the impact of seasonal variation on diversity (n=4), especially responses to monsoon periods (Ansari, 2017; Koparde, 2016; Razak & Sharip, 2019; Yardi et al., 2019), but there were only two long-term investigations of diversity patterns and these focused on phytoplankton (see Crossetti et al, 2008a; Crossetti et al 2008b). While land cover studies reported wide-scale wetland habitat loss to urban expansion over time (Athapaththu et al., 2020; Mondal et al., 2017), there were no studies documenting changes in biodiversity as regions urbanised (but see Hettiarachchi et al. 2014).

2.3.4 Biodiversity and ecosystem services

Fifteen of the 64 studies addressed ecosystem services associated with urban ponds and wetlands. These included assessment of the provision of regulating or supporting services (n=12), perceptions of value (n=2), and inventoried ecosystem services (n=1). The study sites included urban parks (Baharuddin et al., 2017; Shafaghat et al., 2019), and natural wetlands that provided services for peri-urban and urban communities (Das & Basu, 2020; Hara et al., 2018; Hettiarachchi, Athukorale, et al., 2014). Research approaches often integrated land use and land cover changes with stakeholder surveys and interviews to assess changes in use for recreational and economic activities over time (D'Souza & Nagendra, 2011; Das & Basu, 2020; Hara et al., 2018; Hettiarachchi, Athukorale, et al., 2014). A combination of GIS and statistical tools were also applied to quantify ecosystem services use and analyse factors influencing stakeholders' preferences and attitudes (Bandyopadhyay et al., 2006; Das & Basu, 2020).

An inventory of urban ecosystems in Dhaka identified ponds and wetlands as providing food and water supply, regulating water flow, space for recreation and habitats for migratory birds (Zinia & McShane, 2021). Hara et al. (2018) reported that new ponds (from excavation activities) in urbanizing

landscapes around Bangkok allow for economic activities (fishing and recreation) and increase wetland bird abundance. Other studies documented changes in wetland ecosystem use, and ecosystem service impairment, as a direct or indirect result of urbanization and over-exploitation (Hettiarachchi, Morrison, et al., 2014; Mombo et al., 2014). The extent to which urban communities rely on wetlands and ponds within the urban landscape are emphasized in studies that evaluate perceptions of the value of these habitats (Das & Basu, 2020; Mombo et al., 2014). Das and Basu (2020) showed that residents' satisfaction with the delivery of wetland ecosystem services varies with proximity to the wetland with those living nearer to the wetland area perceiving a greater need for habitat improvement. While all studies emphasized the importance of pond and wetland ecosystems for urban and peri-urban residents, quantitative or direct measurements of the relationship between ecological characteristics of urban ponds and wetlands and ecosystem functions or services were minimal. Thus, it remains unclear how important diversity might be in supporting ecosystem service provision in tropical urban areas.

2.4 Discussion

Our review found that most urban tropical wetland research was concerned with the impact of urban growth on the extent of individual wetland areas, or changes to their water quality or diversity, rather than on patterns and trends across whole urban areas. Nevertheless, studies suggest that rapid and unplanned urban development is a major threat to freshwater habitats in tropical urban areas, and potentially undermines the provisioning and flood protection services they provide to urban residents (Brinkmann et al., 2020; D'Souza & Nagendra, 2011; Hettiarachchi, Athukorale, et al., 2014; Isunju & Kemp, 2016). There remain significant gaps in our fundamental understanding of the structure and functioning of urban lentic habitats at different spatial scales, limiting our capacity to develop effective and evidence-based conservation measures.

The findings of this review reveal geographical disparity among tropical regions, with over half of the research coming from a relatively small number of countries. This limits attempts to discern broad geographic patterns in tropical urban pond and wetland diversity. Oertli and Parris' (2019) review of the global urban ponds literature describes a range of design and management practices to support freshwater biodiversity. However, almost all the examples given in the review are from non-tropical countries and it remains unclear whether these practices are suitable for tropical urban ecosystems. Given the unique and different conditions prevailing in wetlands in tropical urban areas, establishing appropriate design and management practices is critical.

The total number of publications retrieved with our search terms suggests the tropical literature is growing, but is still small in comparison to temperate regions. Though the number of papers analysed

as part of this review remains limited, the systematic nature of our search means that we have been able to address our questions. However, two caveats need to be emphasised. Firstly, we intentionally excluded brackish standing water systems. These systems are unique and important, and these things, together with the largely coastal nature of their distribution, brings a wider set of pressures that warrants its own review (Barnes, 1999; Basset et al., 2013). Secondly, our search excluded terms explicitly related to diseases and disease vectors, which are important issues in tropical cities. In some instances the control of vectors may influence the design, management practices and diversity of urban ponds and wetlands (Walton, 2012), and may affect public support for their conservation. This warrants some consideration and so discussed below (see point 3).

Discerning trends in biodiversity patterns and the conservation value of different types of lentic habitat in tropical urban areas will require baseline data on all components of biodiversity. Moreover, further studies of species responses to the environmental and social characteristics of tropical urban environments are needed, and of the relationships between biodiversity and ecosystem service provision. To help direct future research on tropical urban ponds and wetlands, we provide a series of recommendations in the section that follows.

2.4.1 Key limitations in research approaches and recommendations for future research

1. Determining spatial distribution and characteristics of urban ponds and wetlands in tropical cities

Research limitations: Detailed ecological or biodiversity studies of tropical cities tend to be limited to one or two major lakes, wetlands or reservoirs (Figure 3b), while landscape-scale studies tend to focus on basic mapping or assessment of the spatial distribution of wetlands. These approaches limit urban freshwater biodiversity assessments in two ways. Firstly, the ecological value of non-surveyed, often smaller habitats, is overlooked. For instance, cities such as Kuala Lumpur (Teo et al., 2021) and Singapore (Lim & Lu, 2016) have numerous constructed ponds as part of flood mitigation measures yet these are rarely the subject of ecological studies. This is significant since research findings suggest that small retention or storm-water ponds can be important ecosystems for freshwater species conservation and ecosystem services provision (Hassall, 2014; Hill et al., 2016; Holtmann et al., 2017; Johansson et al., 2019). Other anthropogenic standing water bodies that are common in urban areas such as fountains, golf course ponds and drainage ditches, with their distinctive environmental conditions, may also support taxa of conservation (Čerba & Hamerlík, 2022). Chester and Robson (2013) provide an inventory of the various types of anthropogenic water body found in urban areas, many of which have yet to receive significant research attention, especially in tropical cities. Secondly, there are limited field-based data and studies for characterization of urban wetlands at a level

required to adequately understand the extent to which local and regional processes influence biodiversity. This is especially important as previous research has suggested that small lentic waterbodies contribute most to biodiversity at the landscape-scale, reflecting their wide environmental heterogeneity and connectivity (Hill et al., 2018). Moreover, along with the interchangeable use of terms for habitats, incongruent characterization limits the extent to which comparisons with other tropical cities or geographical regions can be drawn.

Recommendations: It is important to inventory and characterize all types of ponds and wetlands across an urban area to maximize ecological benefits and opportunities for biodiversity conservation. This will also help in the development of a typology of urban ponds and wetlands that can allow for targeted ecological studies and effective management for biodiversity. For example, if a given type of pond or wetland is found to support a taxonomic group of interest, it can be selectively managed or prioritised for biodiversity support. In heterogeneous urban landscapes, standardized collection of data on the extent of impervious surfaces, open areas, building and road type and density, and land use surrounding study sites may also be useful in classifying pond types, and provide insight into mechanisms underlying species assemblages (Jeanmougin et al., 2014). Additionally, this review found that even though land-use data derived from remote sensing methods are applied in mapping distributions of habitats, these are seldom applied in biodiversity studies that focus on specific sites. Detailed landscape-scale data can be combined with thorough physico-chemical characterization of sites (Table 2.3) to support ecological studies that aim at understanding how spatial distribution influences diversity patterns (Heino, Bini, et al., 2017). Finally, considering the range of differences in urban profiles among tropical urban areas, consistent description of both wetland habitat characteristics and urban features (in terms of regulation and patterns of development, build-up density) will also be critical for building baseline data on ponds and wetlands in tropical urban areas, and allow for useful knowledge transfer among cities. Oertli and Parris (2019) note that this has been overlooked in pond studies globally as well.

Table 2.3 Suggested metrics to aid characterization of tropical urban ponds and wetlands and identification of variables important for freshwater biodiversity. Adapted from Oertli and Parris (2019), Biggs et al. (1998), Biggs et al. (1998), Ehrenfeld (2000) and Briers (2014). (Metrics that are already consistently present in reviewed literature, like water quality variables, are not included).

	Metrics	Justification
Urban setting, and sampling scales and location	Study area population size and density Road density Proportion and nature of built environment (regulated or unregulated urban development) Function of ponds and wetlands Pond and wetland density (proximity and number of ponds and wetlands around the site), and hydrological connectivity Proportion and nature (natural, lawn park) of green spaces	Urban development patterns and features will vary among cities, and these metrics provide background information for comparable ecological studies. Studies can then compare diversity patterns among and between mega cities and smaller cities. Furthermore, the presence and proximity of other wetland and pond habitats can be important factors influencing community assembly in ponds, and along with information about site quality may help understand mechanisms driving biodiversity among multiple taxonomic groups and at multiple scales.
Physical habitat characteristics	Age, origin (constructed or natural) and type Area, perimeter and margin complexity Vegetation composition and structure (in and around site) Source and depth of water Type of bank and substrate Degree of shading Type and intensity of management Intensity of use by urban residents	Depending on the taxonomic or functional group that is being examined, a range of factors will be significant in determining occurrence and distribution. Used consistently, these metrics can describe overall characteristics for comparing wetlands and, with additional physical variables specific to target taxa (e.g. margin slope, shade, microclimate, nesting trees, etc.) can help assess the quality of ponds and wetlands as habitats for freshwater taxa

2. Assessing the ecological value of tropical urban ponds and wetlands

Research limitations: Biodiversity measures used in the published papers were mostly restricted to taxonomic richness and alpha diversity metrics, with minimal compositional or trait-based assessments. Similarly, temporal variation in diversity and habitat conditions have yet to be examined in-depth. Studies of non-tropical regions suggest that urban pond habitats could be subject to temporal changes in quality as environmental conditions respond to urban pressures like nutrient

loads and sedimentation (Briers, 2014). Long-term monitoring is needed to understand the responses of different types of ponds and wetlands to disturbances related to their intended anthropogenic roles (the use of wetlands as sedimentation ponds, for runoff retention or treatment, as recreational sites). In addition, natural and constructed wetland habitats are subject to rapid, inter-annual variability in habitat conditions as well as biological community structure (Jeffries, 2005; Ruhí et al., 2013). This warrants attention in tropical urban systems subject to frequent flooding events and concomitant surges in sediment and nutrient loads that continue to challenge conventional storage and treatment structures in tropical cities. In addition, without data from long-term monitoring of species populations or community structure, it is difficult to assess the risk of urban habitats becoming ecological traps (Hale et al., 2015).

Recommendations: Future research should focus on assessing multiple components of diversity for a range of taxonomic groups to determine the ecological value of urban habitats. All components of tropical freshwater biodiversity need more attention, including taxonomic and functional compositional variation (beta-diversity) which are important to understand mechanisms driving species distribution patterns in urban landscapes, and their role in maintaining these ecosystems, respectively (Petchey et al., 2009; Socolar et al., 2016). In addition, quantifying the variation in community composition among habitats (e.g., determining the contribution of nestedness and turnover to total beta-diversity, and the ecological uniqueness of individual sites) and the relative contribution of particular sites to broader-scale biodiversity is important for conservation prioritisation at the landscape-scale (Heino, Bini, et al., 2017; Hill et al., 2021; Socolar et al., 2016). Taxonomic and functional diversity patterns are not necessarily congruent, and the predominant focus on taxonomic richness found in most literature potentially overlooks components of biodiversity (functional and phylogenetic) that are relevant to critical ecosystem processes, functioning, and resilience (Devictor et al., 2010; Hill et al., 2019; Strecker et al., 2011). For example, Heino, Bini, et al. (2017) report a negative correlation between species richness of a habitat and the uniqueness of its species, demonstrating the importance of multiple measures of diversity in ponds in temperate regions. These assessments and monitoring of species distribution can be carried out periodically to obtain data over longer time periods, and assess long-term habitat viability and resilience to frequent hydro-meteorological disturbances. It may also be necessary and useful to rely on multiple tools, including environmental DNA (e-DNA) analyses, for documenting biodiversity. For many freshwater taxa in tropical regions, especially macroinvertebrates, ecological and taxonomic knowledge remains limited and this constrains diversity assessments and monitoring (Sundar et al., 2020). Developing the capacity and reference databases for e-DNA analyses take time, but this approach offers great potential for more efficient biodiversity assessments, surveys, and species mapping (Belle et al.,

2019). Tropical cities may have abundant freshwater habitats (Teo et al., 2021) but the dramatic changes in these systems due to the pace of urban development means that tools such as e-DNA able to rapidly assess diversity for the purpose either of conservation prioritisation or monitoring would be particularly significant.

3. Identifying environmental correlates of biodiversity and community structure for multiple taxonomic groups

Research limitations: Besides water quality, there was limited documentation and examination of habitat characteristics at a site level (Figure 6). The nature and relative importance of physical habitat characteristics and surrounding land-use have not been examined in depth for different faunal and floral groups in the tropical urban context. Physical habitat characteristics and surrounding landscape can facilitate or impede species establishing populations in urban ponds and wetlands (Hamer & Parris, 2011; Hamer et al., 2011; Liao et al., 2020) and affect ecological processes such as feeding, reproduction, dispersal or shelter (Goertzen & Suhling, 2012; Thornhill et al., 2017). Surrounding land use characteristics can also play a role in determining community structure (Holtmann et al., 2018), especially where it facilitates or impedes dispersal. Dispersal mechanisms and habitat connectivity may be particularly important in urban ecosystems where built structures, along with species dispersal and colonization capabilities, can limit an organisms' ability to move to and establish populations in suitable freshwater habitats (Oertli & Parris, 2019; Parris, 2006; Ruhí et al., 2013; Smith et al., 2009).

Recommendations: Research will need to focus on environment-taxa relationships that assess the influence of local environmental and spatial variables (land-use, dispersal, connectivity) on target species populations or whole communities at larger scales. This is key to effective and targeted management of wetland habitats in different urban areas. For example, several design and management recommendations are available for the support of urban populations of amphibians and dragonflies in non-tropical regions (Goertzen & Suhling, 2012; Hamer et al., 2011). Similar approaches to biodiversity research are needed for tropical species. Urban habitats may provide opportunities for ecological studies of taxonomic groups such as freshwater macroinvertebrates that remain underrepresented in conservation literature (Sundar et al., 2020). The relevance and influence of environmental variables will vary depending on the taxonomic group in question (endemism, tolerance, dispersal capabilities), the component of diversity examined and the spatial scale of the research. Among the environmental variables that warrant further attention from tropical urban research are measures related to microhabitat conditions in tropical urban environments. Tropical

urban environments are subject to higher temperatures (a combination of both climate and urban heat island effects) and greater volumes of surface runoff characterized by high sediment loads that can settle and alter substrate properties within ponds and wetlands. Thus, variables like shade availability, and substrate type may be important for understanding the distribution of taxa like macroinvertebrates and amphibians with life processes that are vulnerable to heat stress and fine sediment or debris in water or substrates. Furthermore, in lentic habitats from non-tropical regions, potential 'master' variables have been identified that have a large influence over the richness and composition of aquatic taxa, including surface area, hydroperiod, connectivity and aquatic macrophyte coverage (Hill et al., 2019; Parris, 2006; Scheffers & Paszkowski, 2013). Studies are needed from tropical regions that consider the importance of these variables for multiple taxonomic groups, to identify any congruency (or lack of) in lentic habitat biodiversity-environment relationships among tropical and non-tropical regions.

Examining the relationships between environmental conditions and nuisance species is necessary to address health risks associated with disease vector proliferation in water bodies in tropical urban areas. As part of such research, it will be important to also consider how established practices or design features that aim at discouraging vector proliferation. Hanford et al. (2019) found that mosquito species vary in their responses to specific aspects of urban wetland habitats and suggested that identifying specific design features that promote or discourage target vector species occurrence may be key to managing urban habitats for biodiversity while mitigating health risks associated with them. Management such as water level regulation, bank gradient, plant choice and growth control, and the use of larvicides (Knight et al., 2003; Zakaria et al., 2004) may impact non-target species and wetland community composition or diversity. In a review of wetlands and mosquito research, Dale and Knight (2008) note that ecological studies rarely include assessments of vector prevalence or competence and vice versa. Urban vector research in the medical or vector entomology literature tends to focus either on larval microhabitats and oviposition sites in buildings and residential areas, or the influence of social factors such as population density and infrastructure on disease prevalence (Carbajo et al., 2006; Li et al., 2014; Mint Mohamed Lemine et al., 2017; Samson et al., 2015), rather than how wetland habitats contribute to vector abundance. Constructed wetlands (for wastewater treatment or runoff management) are better represented in mosquito research but the effect of vector control design features and practices on non-target species still requires research attention (Dale & Knight, 2008). Overall, there is a need to quantify and assess the potentially differing vector risks associated with the various types of ponds and wetlands in urban areas (see Crocker et al., 2017) and to improve data available for assessing trade-offs when multiple functions are expected from urban ponds and wetlands. Integrating understanding of infectious disease vectors and the risks they

pose in urban areas with evidence from medical or public health literature, as well as constraints faced by city managers, in terms of maintenance costs and barriers to practical implementation (for example, number and sizes of wetland) will be integral to developing effective strategies and garnering support for urban freshwater biodiversity conservation.

Similarly, while connectivity may be important for maintaining biodiversity in urban habitats (Oertli & Parris, 2019), abundant and linked drainage systems may facilitate spread of invasive species capable of exploiting conditions in novel pond and wetland habitats or thriving in the warm, nutrient- rich waters (Kwik et al., 2020; Mansor, 1996). Research is needed to assess the risks posed by urban ponds and wetlands and to determine the specific species traits, connectivity factors and habitat characteristics that can be monitored or managed to control for invasion threats without compromising opportunities for biodiversity improvement (James S. Sinclair et al., 2020). Table 3 presents some potentially important environmental variables at local and landscape scales that require study to determine their importance for biodiversity patterns and function in tropical urban ponds and wetlands.

4. Identifying relationships between biodiversity and ecosystem services

Research limitations: The links between freshwater biodiversity and ecosystem services have gained increased research attention recently, and evidence suggests that species loss, especially within fragmented environments, compromises services (Durance et al., 2016). However, published studies from tropical urban areas examining the interaction between biodiversity, ecosystem service provision and urbanization are limited in number (but see Hettiarachchi, 2014). Major gaps in the current literature also exist for key urban ecosystem services which are important for tropical urban environments including mitigating hydro-meteorological disasters such as flooding and addressing urban heat island effects which are particularly prevalent in tropical cities and likely to increase in frequency and intensity with climate change (Lechner et al., 2020).

Recommendations: Integrate biodiversity and ecosystem service research. Assessing the capacity of urban lentic systems to undertake their primary function (e.g., stormwater retention or recreation) alongside an assessment of their biodiversity value, will enable management strategies to be developed that ensure these systems support both society and wildlife. For example, soil surveys of urban ponds and wetlands can quantify carbon content and assess their potential for atmospheric carbon sequestration (Moore & Hunt, 2012). Several studies in this review highlighted the role of governance and public perceptions and practices on the state of urban wetlands (D'Souza & Nagendra, 2011; Das & Basu, 2020; Hettiarachchi, Athukorale, et al., 2014). Many urban systems are primarily built for purposes other than biodiversity, and as a result management of these systems

rarely considers the inhabiting fauna and flora. Management activities vary in methods (including vegetation selection and removal, water level manipulation, dredging) and intensity among pond and wetland types, and are directed by intended functions, landscaping practices or aesthetic choices (Holtmann et al., 2019; Schad et al., 2020). However, in many cases, small ecologically-focused changes to current management plans can maximise the biodiversity that is supported in urban lentic habitats, while not reducing the efficacy of their primary function, e.g., storm water/pollutant collection (Rosenzweig, 2003).

Urban ponds and wetlands are very often associated with parks or remnant areas of natural vegetation. This creates opportunities for their incorporation in the planning and design of urban blue-green spaces (Ahn & Schmidt, 2019; el-Baghdadi & Desha, 2017). This is typically accomplished within the framework of nature-based solutions, and allows the multiple functions of lentic systems to be explored (Lafortezza et al., 2018) and their cost-effectiveness relative to conventional 'grey' infrastructure to be assessed. The feasibility of using blue-green spaces for such purposes depends on the nature of the existing urban landscape, and the willingness and/or capacity of cities to adopt them plays a major role in their inclusion in city plans. Lechner et al. (2020) argued that the lack of data on the benefits of blue-green spaces from tropical areas limits their inclusion. The financial viability of replacing conventional built structures with natural systems is still a subject of research and debate, especially in terms of methods for economic valuation and ecosystem services assessment (el-Baghdadi & Desha, 2017; Wild et al., 2017). Fundamental biodiversity assessments and monitoring are essential to improve understanding of ecological processes and functions that underpin ecological services and value (Reid et al., 2019). Decision makers using nature-based solutions to tackle problems faced by cities will need this ecological knowledge base, in addition to measures of social and economic values of wetlands (Durance et al., 2016; el-Baghdadi & Desha, 2017).

Finally, direct relationships between biodiversity patterns and management practices associated with social attitudes and urban societal needs and priorities also warrant exploration. This may be especially important for reconciling social preferences with management for conservation (Blicharska et al., 2017; Ngiam et al., 2017). Opportunities for education and public engagement can also be explored by assessing the efficacy of educational infrastructure like signage, and conducting participatory research and monitoring projects with urban residents (Simpson & Newsome, 2017; Soanes et al., 2020).

2.5 Conclusion

As tropical urban areas expand, ponds and wetlands can provide refuge for freshwater organisms and a range of ecosystem services for urban residents. In order to determine the conservation value of these ecosystems and ensure management and/or design that promotes biodiversity, research will have to move beyond focus on single, prominent wetlands to an approach that examines large-scale patterns of biodiversity across urban areas. It will also be important to determine the response of different aspects of urban pond and wetland diversity to the distinct climate and hydrology of tropical urban areas, as well as the diverse range of lentic habitats that occur there, and how they respond to different management practices. While the number of publications on tropical urban ponds and wetlands is growing, there remains a need for more consistent descriptions of habitat and urban landscape characteristics to enable knowledge transfer among tropical cities. As the importance of green infrastructure for sustainable urban development becomes more apparent, sound ecological data are needed to maximize the potential of new and remnant pond and wetland habitats for biodiversity conservation. This is especially important for tropical freshwater taxa, long challenged by taxonomic and ecological knowledge gaps. Expanding the scope of tropical freshwater biodiversity research to urban areas and assessing links between biodiversity and ecosystem services can contribute to addressing these gaps and also aid tropical cities in creating, managing or restoring natural habitats for the benefits they provide to society and biodiversity.

Chapter Three

Field Study Methods

3. FIELD STUDY METHODS

3.1 Study area

3.1.1 Greater Kuala Lumpur

Greater Kuala Lumpur (GKL) is the conurbation comprising Kuala Lumpur, the federal capital of Malaysia, and surrounding districts of Petaling, Putrajaya, Sepang, Gombak, Klang and Hulu Langat. GKL covers an area of 2950km^2 and is characterized by high density commercial zones and sprawling residential developments (32.2% of GKL area), green spaces which exist in the form of remnant forest patches and public parks (Ahmad et al., 2014; Maryanti et al., 2017) and agricultural land, predominantly oil palm and rubber plantations (16.2% of GKL area). Extensive forest reserves are also present, primarily along the north-eastern boundaries of the city (Figure 3.1). Around 25% (over 7 million people) of the Malaysian population now resides in GKL: populations of the districts of Petaling (2.3 million) and Hulu Langat (1.5 million) are the highest in the city while population densities vary from 7863 people/km^2 in the central Kuala Lumpur district (population: 1.9 million) to about 600 people per km^2 in the Sepang district (DOSM, 2023; Koya, 2023). Kuala Lumpur and the immediately surrounding urban and suburban areas underwent rapid, successive transformations after the mid-1800s: originally tin mining settlements in the mid to late 19th century, followed by extensive rubber plantations in the first half of the 20th century which were then replaced by the development of commercial, industrial and residential centres in latter half of the century (Wong, 2023). The expansion of urban land use into formerly agricultural or forested land is ongoing, with detrimental impacts on terrestrial and aquatic ecosystems(Abdullah & Hezri, 2008; Lechner et al., 2021; Naeem et al., 2016; Yong & Yule, 2004).

GKL is situated within what historically was lowland tropical forest and swamp forest landscapes of tropical Malaysia, a country recognised for its high biological diversity. It has a warm lowland, tropical humid climate with mean annual precipitation of 2500-3000 mm and mean daily temperature of 28 ^{o}C (METMalaysia, 2022). There are two monsoon seasons, the northeast (November to March) and the relatively drier southwest monsoon (May to September) as well as inter-monsoon periods (METMalaysia, 2022). The Klang and Langat river catchments are two major catchments in GKL. The Klang River runs through Kuala Lumpur with the confluence of many of its major and minor tributaries (Gombak, Ampang, Batu, Damansara, Penchala) within the city, then flows through the heavily populated districts of Petaling and Klang where it reaches the Straits of Malacca (LUAS, 2017). The urban areas of Hulu Langat (including Kajang municipality) and Putrajaya districts are part of the Langat River basin, south of the Klang River basin. Its major tributaries include the Semenyih and Labu rivers, with an upper catchment that comprises a hill forest reserve (LUAS, 2021).

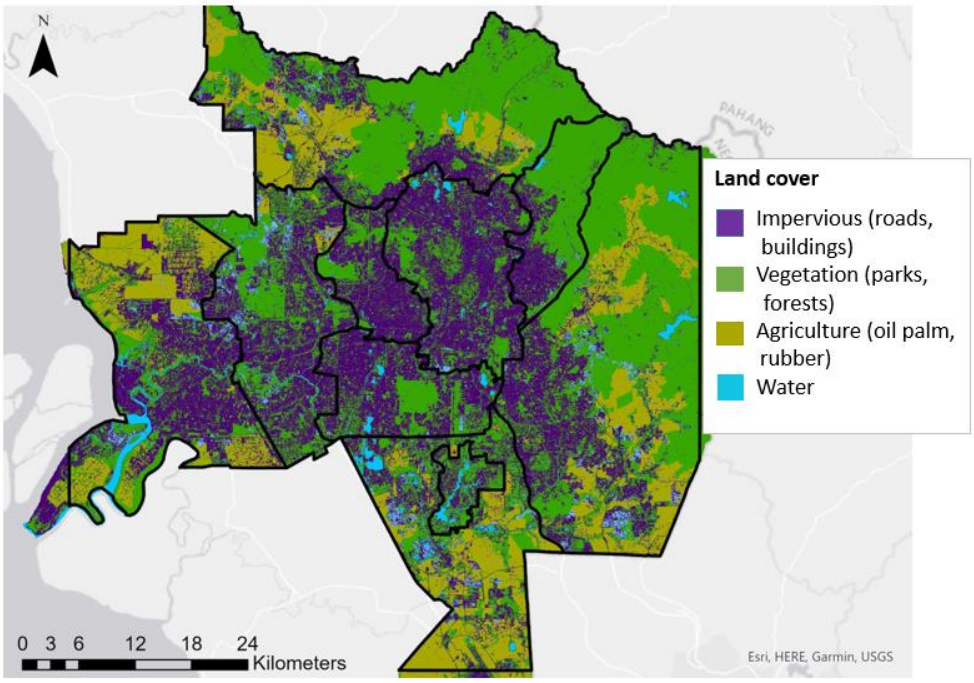
While rivers and streams are perennial and abundant, natural ponds and lakes are typically (for tropical regions) few and relatively small (Yong & Yule, 2004). However,, anthropogenic inland water bodies including reservoirs, lentic water bodies are abundant, constructed as part of water supply and storm water management systems, for aesthetic or recreational purposes; a large number of lakes and ponds were formed by historical tin mining which left depressions or quarries once mining ceased (DID, 2012; Low et al., 2016; Teo et al., 2021; Yong & Yule, 2004). Lentic waterbodies across GKL exhibit variation in water quality, management, surrounding land use and ecology (Razak & Sharip, 2019; Teo et al., 2021; Yong & Yule, 2004).

3.1.2 Site selection

The focus of the present study were ponds located within the GKL region. Ponds are defined here as lentic water bodies with surface areas between 1 m^2 and 2 ha, holding water for at least four months in a year and either natural or anthropogenic in origin, following accepted delineation in urban pond research (Biggs et al., 2005; Hill et al., 2016). A recent map of lakes and ponds produced using remotely sensed data (see Section 3.2.2 for details) inventoried 1,013 standing waterbodies in GKL (Teo et al., 2021). This map was modified with ArcGIS tools to exclude waterbodies greater than 2ha – the remaining ponds (777 ponds) were the pool of potential study sites.

Ponds were identified from the core urban centre of the study area, Kuala Lumpur, and from surrounding municipalities (Petaling Jaya, Subang Jaya and Kajang) broadly representing a geographical spread of decreasing urban population density and mixed urban/suburban land uses, with sampling sites selected randomly from these. Since it is possible that some ponds may have been overlooked (e.g. temporary ponds, garden ponds), additional ponds were identified through field visits and visual inspection of maps in Google Earth (GoogleEarth, 2021). These included smaller ponds found within parks with greater canopy cover. The locations of these ponds were then added to the GIS pond dataset using ArcGIS tools. All ponds were subsequently checked for accessibility through field visits and permission for sampling was requested from respective municipal authorities and/or pond managers. A total of 30 ponds were selected for study of which two had to be excluded because of access restrictions (Figure 3.1). These ponds represented a wide variation in key variables such as impervious land cover, surface areas, aquatic vegetation cover for a representative sample of ponds in GKL. The majority of the ponds were located within public parks and gardens (n=22), that are surrounded by lawn grass or forested parks, while the remaining were roadside or residential area retention ponds (n=6) (Figure 3.2).

(a)



(b)

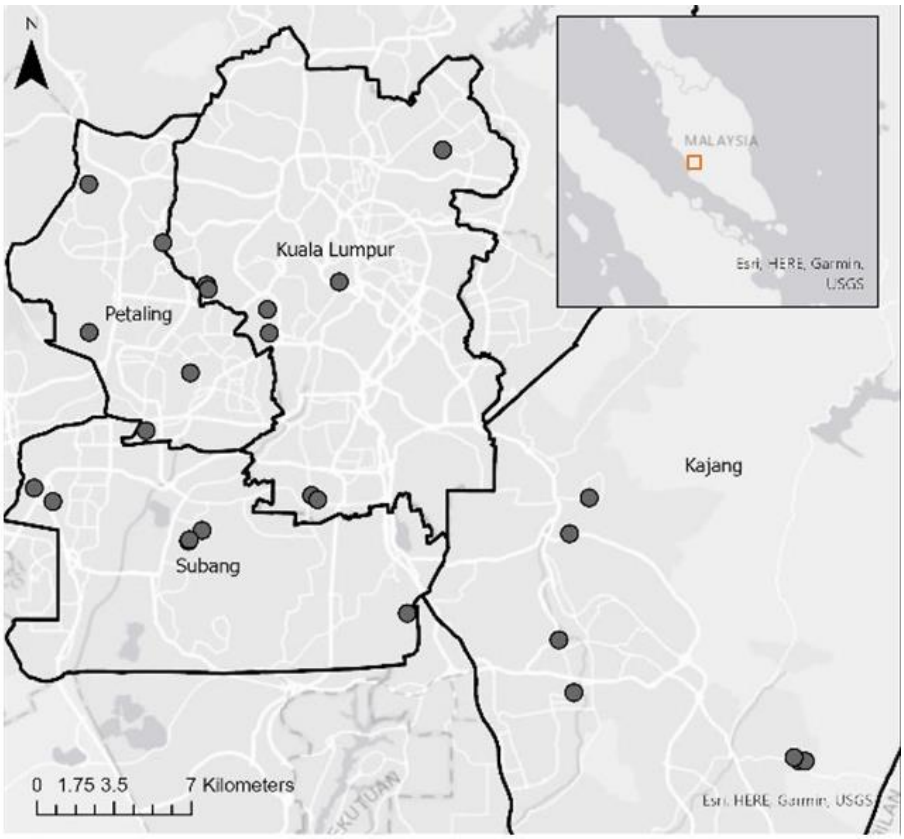


Figure 3.1 (a) Land use map of Greater Kuala Lumpur with borders of nine municipalities and (b) the location of the study sites (ponds) within Kuala Lumpur, the capital, and three of the municipalities namely Petaling (Petaling district), Subang (Petaling district) and Kajang (Hulu Langat district). Inset map shows location of GKL in Malaysia.

Figure 3.2: Selected ponds from (top left) Kuala Lumpur– park pond surrounded by dense urban commercial and residential areas; (top right) retention pond in Subang between urban residential area and forest reserve; (bottom left) pond in Kajang surrounded by urban residential area and some agriculture; (bottom right) park pond in Petaling Jaya surrounded by dense urban commercial area.

3.2 Data collection

Environmental and macroinvertebrate data collection from sample ponds was carried out twice: first toward the end of a monsoon season (January – March 2022) and then early in a monsoon season (October – December 2022). All of the sample ponds were permanent but seasonal variation in surface water quality parameters and zooplankton ecology have been recorded for the region (Loi et al., 2022; Razak & Sharip, 2019).

3.2.1 Environmental data: Water parameters and physical habitat characteristics

At each pond site physicochemical characteristics, physical habitat characteristics, and spatial factors were recorded. A YSI Pro Plus multi-parameter probe was used to record water temperature (^{o}C), pH, salinity (ppt), conductivity (us/cm), and dissolved oxygen (%). Water samples were collected for analysis of total suspended solids (photometric method) and hardness (calmagite colorimetric method) with a laboratory spectrophotometer. In addition, the following physical habitat characteristics were recorded based on visual inspection: percentage of pond water surface shaded, percentage of pond margin shaded, nature of bank material, and percentage of pond covered by aquatic vegetation. Finally, geospatial software (ArcGIS) tools were used calculate the area and perimeter of each pond with the geographic information for each pond recorded in the pond GIS dataset.

3.2.2 *Land cover mapping and surrounding land use data for GKL*

Analysis of pond network connectivity was based on GIS datasets produced by Teo et al. (2021) and Danneck et al. (2023) that inventoried lentic waterbodies and classified land use/land cover for GKL. Briefly, remotely sensed satellite data for GKL were used to identify waterbodies, applying multiple indices to improve automatic identification of water bodies from the images, and integrating publicly available high-resolution imagery to improve resolution to 2.4m. Non-lentic waterbodies (rivers) were then removed manually, creating a vector shapefile of the lentic waterbodies from which aquaculture ponds were also manually excluded considering that they are subject to high intensity management.

Similarly, remotely sensed satellite images (10m resolution), as well as secondary sources of land use—Open Street Map data (OSM) and agricultural land use data—were used to classify and map nine land cover categories. The land cover classes were impervious land cover, major and minor roads, non-agricultural vegetation, oil palm plantations, rubber plantations, other agricultural vegetation, waterbodies (running or standing) and bare soil. The percentage cover of each land-use variable surrounding each of the 28 ponds, within radii of 100m, 250m, 500m and 2km, was calculated using ArcGIS processing tools and R. Thus, for each pond, there were up to seven land use variables per given radius. The land cover map was also used to calculate road density within 100m, 250m, 500m and 2km radius of each pond. Finally, the distance to the nearest road was calculated for each pond.

3.2.3 Macroinvertebrate data collection

The macroinvertebrate sampling method used here followed the protocol developed by Biggs et al. (1998). Available mesohabitats were first identified (e.g. vegetation stands, open water, shaded areas) and macroinvertebrates were collected from each using the sweep technique with a standard pond net (30cm x 30cm net, with a 1mm mesh size). Each pond was sampled for a total 3 minutes, with the time divided equally among the mesohabitats. An additional one minute was allocated for manual collection in areas that could not be sampled by the net (e.g., fallen tree logs, the water surface and on large boulders). Macroinvertebrate samples were immediately preserved in 70% ethanol and transported to the laboratory for sorting and identification. Individuals were identified to family, genus or species level based on available keys (Yule and Yong, 2010) and counted to create two site x species datasets: one with count (abundance) values and another with presence or absence values (that is 1 for present or 0 for absent).

3.2.4 Macroinvertebrate functional traits

Data for 60 biological traits organized into 10 grouping features (terminology follows Schmera et al. (2015) and Hill et al. (2019)) was obtained from the trait affinities database of Tachet et al. (2010) for the sampled freshwater macroinvertebrates (Table 3.1). In the database, taxa are assigned scores (ranging from 0 to 3 or 0 to 5) for each trait that indicate their affinity for a trait within a grouping feature. Scores are calculated using a fuzzy coding procedure, and this method accounts for differences in traits in different life stages of some species and variations among different species of the same genus (Chevene et al., 1994). Table 3.2 shows the 60 traits analysed in this study and their ten grouping features.

The database was developed for European taxa so several pond taxa from the GKL ponds for which information was not available in the database had to be excluded from the functional analyses, e.g. Ampullariidaes spp, Thiaridae spp and *Diplonychus rusticus* (see full list in Appendix C1). For taxa identified to genus level but for which data was not available in the database, scores were assigned at family level (mean of score for taxa in the family). Examples include *Agriocnemis*, *Pseudagrion*, *Amerianna*, *Filopaludina*, *Rhagadotarsus*. Finally, taxa identified to family level only, e.g. Gomphidae, Acentropinae, Tanypodinae, were assigned family level scores.

Table 3.1: Trait and trait categories selected for analyses based on the database of trait scores for freshwater macroinvertebrates (Tachet et al., 2010; Usseglio-Polatera et al., 2000).

Grouping feature	Trait	Code in analyses
Maximal potential size	<0.25	A1
	0.25-0.5	A2
	0.5-1	A3
	1-2	A4
	2-4	A5
	4-8	A6
	>8	A7
Life cycle duration	</=1 year	B1
	>1 year	B2
Aquatic stages	Egg	C1
	Larva	C2
	Nymph	C3
	Adult	C4
Dispersal	aquatic passive	D1
	aquatic active	D2
	aerial passive	D3
	aerial active	D4
Food	fine sediment & microorganism	E1
	detritus <1mm	E2
	dead plants > = 1mm	E3
	living microphytes	E4
	living macrophytes	E5
	dead animals >1mm	E6
	living microinvertebrates	E7
	living macroinvertebrates	E8
	Vertebrates	E9
Feeding habits	Absorber	F1
	deposit feeder	F2
	Shredder	F3
	scraper	F4
	filter-feeder	F5
	piercer	F6
	predator	F7
	parasite	F8
Respiration	tegument	G1
	gill	G2
	plastron	G3
	spiracle	G4
	hydrostatic vesicle	G5

Grouping feature	Trait	Code in analyses
Substrate	flags/boulders/cobbles/pebbles	H1
	gravel	H2
	sand	H3
	silt	H4
	macrophytes, filmentous algae	H5
	microphytes	H6
	twigs/roots	H7
	organic detritus/litter	H8
	mud	H9
Locomotion and substrate relation	flier	I1
	surface swimmer	I2
	full water swimmer	I3
	crawler	I4
	burrower	I5
	intersitital	I6
	temporarilty attached	I7
	permanently attached	I8
Current velocity	null	J1
	slow (<25 cm/s)	J2
	medium (25-50 cm/s)	J3
	fast (>50 cm/s)	J4

Each taxon was assigned a score for each of the traits which was then multiplied with the scaled abundance, log(x+1), of the taxon in each pond (Buendia et al., 2013; Feio & Dolédec, 2012; Larson et al., 2016). Thus, each pond was characterized based on the traits present, abundance weighted. The following illustrates the process of obtaining trait data for each pond based on the method described in Buendia et al. (2013).

Step 1: assign score to each recorded taxon for each trait based on the Tachet et al. (2010) database; scores range from no affinity to high affinity, 0-3 or 0-5 **(Species x Trait table)**

Taxa	Grouping feature A			Grouping feature B		Grouping feature C		
	A1	A2	A3	B1	B2	C1	C2	C3
Taxon 1	0	0	3	1	3	5	0	0
Taxon 2	1	3	0	0	5	3	2	0
…								

Step 2: scale scores so the sum for traits within a grouping feature is 1 (Buendia et al., 2013; Dolédec & Statzner, 2008) **(Species x Trait table - scaled)**

Taxa	Grouping feature A			Grouping feature B		Grouping feature C		
	A1	A2	A3	B1	B2	C1	C2	C3
Taxon 1	0	0	1	0.25	0.75	1	0	0
Taxon 2	0.25	0.75	0	0	1	0.6	0.4	0
…								

Step 3: For each pond, assign taxa the trait score multiplied by its abundance log(x+1) transformed, then add up the total for each trait. **(Species x Trait table of each pond)**

Taxa	Grouping feature A			Grouping feature B	
	A1	A2	A3	B1	B2
Taxon 1	0	0	1	0.25	0.75
Taxon 2	0.25	0.75	0	0	1
…					
TOTAL	0.2	0.75	1	0.25	1.75

Pond_01

Taxa	Grouping feature A			Grouping feature B	
	A1	A2	A3	B1	B2
Taxon 1	0	0	1	0.25	0.75
Taxon 5	0.5	0.5	0	1	0
…					
TOTAL	0. 5	0. 5	1	1.25	0.75

Pond_02

Step 4: Compile the total score for each trait in every pond to obtain the trait dataset for subsequent analyses **(Site x Trait dataset)**

Taxa	Grouping feature A			Grouping feature B	
	A1	A2	A3	B1	B2
Pond_01	0.2	0.75	1	0.25	1.75
Pond_02	0. 5	0. 5	1	1.25	0.75
…					

3.3 Data analysis

All analyses and visualizations were performed or generated using R Statistical Software (RCoreTeam, 2022).

3.3.1 Correlation among environmental variables

Correlations among all environmental variables were assessed with Pearson's correlation. Where environmental variables showed high correlation (Pearson's r>0.7), one was removed to avoid instability issues associated with collinearity in subsequent modelling, as collinearity can lead to incorrect regression parameter estimates (Legendre & Legendre, 2012; Thornhill et al., 2017).

3.3.2 Variation in environmental conditions among the ponds

Principal component analysis (PCA) was carried out to assess the variation in environmental conditions and identify the main variables driving differences among ponds in terms of water parameters, habitat structure and surrounding land use. PCA is an unconstrained ordination method applied to multivariate datasets to reduce the number of variables (dimensions) that describe the data objects. Where many variables are used to describe objects (or samples), PCA allows for the identification of the main trends in data variation and summarize relationship among variables and among data objects (Legendre & Legendre, 2012). PCA methods involve determining the centroid of the concentration ellipsoid (cloud) formed by the position of the data objects in multidimensional space (equal to the number of variables) and rotating the original axes so that each axis goes in the direction of maximum variation. The first principal axis is the line the goes through the ellipsoid in the direction of maximum variation. Each subsequent axis is then drawn perpendicular to the preceding one. Data objects have new positions (scores) on these new axes that describe their how they relate to each other and the most important directions of change (gradients) in the dataset (Legendre & Legendre, 2012)

PCA was performed with the function *rda()* from the 'vegan' package in R (Oksanen et al., 2022). All of the environmental variables were first standardized and a pair-wise Euclidean distance matrix was calculated for the site x environmental variable matrix before performing PCA.

3.3.3 Taxonomic and trait-based diversity measures

3.3.3.1 Gamma diversity

Gamma diversity is a measure of diversity that includes all of the species present in a given landscape (Whittaker, 1960). Gamma diversity was calculated as the total number of macroinvertebrate taxa recorded from the 28 ponds. Furthermore, the non-parametric Chao2 estimator was applied to the taxa presence-absence data to calculate estimated gamma diversity. This is a robust model for estimating the lower bound or minimum total richness within an area based on the incidence of the rarest taxa (present in only 1 or 2 sites in a sample (Gotelli & Colwell, 2011; Karsdorp, 2022). The calculation was carried out in R with the function *specpool()* in the R package 'vegan'. The following are

the formulae for Chao2 estimated richness (gamma diversity), where S_P is estimated Chao richness, S_O is observed richness, a1 is the number of species with occurring in one site only, a2 is the number of species occurring in two and N is the number of sites. The bias corrected formula is applied if there are no taxa limited to two sites.

Equation 1. Chao2 estimator

$$S_P = S_o + a1^2/(2(a2)) \times (N-1)/N$$

Equation 2. Chao2 estimator with bias-corrected

$$S_P = S_o + a1((a1-1)/(2(a2+1))) \times (N-1)/N$$

3.3.3.2 Alpha diversity

Alpha diversity is defined here as the diversity within individual sample sites (Whittaker, 1960). The following alpha diversity metrics were calculated for each pond as response variables.

- Taxonomic richness - the total number of taxa recorded from each individual pond site.
- Abundance - the total number of individuals of a taxon or major taxonomic group recorded from each pond site.
- Shannon's diversity index, H' (Shannon & Weaver, 1949) - a measure of diversity that takes into account both the taxa present and their abundances. Index values range from 0 to 5.

Equation 3. Shannon's diversity index. N represents the total number of individuals and n_i is the total number of i species.

$$H' = -\Sigma (n_i/N \times \ln n_i/N)$$

- Pielou evenness, J (Pielou, 1966)– the ratio of observed diversity to maximum diversity. It measures how evenly distributed taxa are within a community. Values range from 0 to 1 with 1 indicating equal abundances for all taxa.

Equation 4. Pielou eveness. H' is Shannon's diversity index and S is the total number of species present.

$$J = H'/\ln S$$

- Berger-Parker dominance index – the extent to which the most abundant taxon dominates over other taxa in a community (Berger & Parker, 1970). The higher the value, the more dominated a pond is by a few species, implying lower overall diversity for a given pond.

Equation 5. Berger parker dominance index. n_{max} is the count of the most abundant species and N is the total number of individuals of all species present.

$$D = n_{max}/N$$

- Trait richness – the total number of trait categories present in each pond (Buendia et al., 2013).
- Rao's functional diversity coefficient – this was calculated to quantify functional (trait-based) diversity, with the R package 'FD' (Laliberté & Legendre, 2010; Laliberté et al., 2014). This metric is calculated weighting dissimilarity (Euclidean) in traits among ponds (species x trait dataset in Step 2) with taxa abundance within each pond (Champely & Chessel, 2002).

Equation 6. Rao's diversity coefficient. S is the total number of species in each pond; d_{ij} is the dissimilarity of species among ponds and p_{ij} is the proportion of each species.

$$H_p = \Sigma^{s}_{i=1} \Sigma^{s}_{j=1} d_{ij} p_i p_j$$

3.3.4 Seasonal differences in environmental and diversity data

The distribution of values for each environmental variable and the alpha diversity measures were first checked for normality through visual inspection of histograms and the Shapiro-Wilk test. The function *shapiro.test*() was run to perform the Shapiro-Wilk test to determine if the distribution of each of the variables and measures was significantly different from a normal distribution. Then, for environmental and diversity variables that showed normal distribution, parametric two-way t-tests were performed to determine if there were statistically significant differences in mean values between the two seasons. For variables that showed non-normal distribution, the non-parametric Kruskal Wallis H test was performed.

3.3.5 Clustering analysis and differences among pond groups

Clustering analysis was applied in order to classify ponds based on local environmental conditions and subsequently examine the relationship between pond type and macroinvertebrate diversity. First, agglomerative clustering was performed with the R function *hclust()* and the number of optimal clusters was determined with visual inspection of the dendogram produced. To assess the validity of the number of clusters, the results of multiple clustering methods were compared for different number of clusters with the R function *clValid()* (Brock et al., 2008), which also calculates cluster internal validation measures.

The pond groups determined by the clustering analysis were then examined for differences in diversity. To test for statistically significant differences between the groups in taxonomic and trait based diversity measures, t-tests were performed. Prior to performing the statistical tests, the groups' distributions

were checked for normality and their variances compared by means of the Shapiro-Wilk Test and Bartlett's Test, respectively.

3.3.6 Spatial autocorrelation analysis

Spatial autocorrelation describes the similarity or differences between response variables that are due to increasing or decreasing geographic distances between sites (Legendre & Legendre, 2012). Positive or negative spatial autocorrelation can indicate the importance of organism dispersal in the study region or spatial structuring of suitable habitats. Spatial autocorrelation violates the assumption that samples are independent and leads to bias in statistical analyses (Heino, Bini, et al., 2017; Peres-Neto & Legendre, 2010). Thus, to test for spatial autocorrelation in the response variables (e.g., taxa richness, Shannon's diversity index and Local Contribution to Beta Diversity), Moran's I was calculated and correlograms were constructed with the function *correlog()* from R package pgirmess (Giraudoux et al., 2018). To test for spatial patterns in community composition, Mantel's correlograms were constructed using the *mantel.correlog()* function (from R package 'vegan') for the macroinvertebrate (presence-absence data) pairwise distance matrix (Euclidean) and the geographical distances between the ponds. The geographical data input was the XY coordinates of the pond sites (UTM zone 47N). The graphs constructed with both functions show the extent of autocorrelation at given distance intervals and whether the calculated correlations are significant.

3.3.7 Modelling relationships between alpha diversity metrics and environmental variables

Following Buendia et al. (2013) and Gallardo et al. (2011) the relationship between diversity metrics and environmental variables were examined with Generalised Additive Models (GAM). GAM allow modelling with non-normal distribution families, similar to Generalised Linear models, but also allow modelling of non-linear, response-predictor relationships. This is implemented with smoothing function f() applied to the predictor variables, controlled by penalized regression (smoothing parameter λ) and basis dimension κ (Wood, 2008)

Where y_i is the response variable, x_i the predictor variable, α the intercept and ε_i the residual, the basic GAM equation is:

Equation 7. Basic Generalised Additive model.

$$y_i = a + f(x) + e_i$$

Poisson, negative binomial and Gaussian distributions were used for modelling residuals variation based on the nature of the response variable (taxon richness – Poisson; Shannon's diversity index, Rao's functional diversity coefficient – Gaussian; abundance – negative binomial). Analyses were performed with the function *gam()* from the 'mgcv' package = (Wood & Wood, 2015)with thin plate regression spline selected as the smooth function. The *gam()* automatically selects smoothness parameter by

REML method (residual maximum likelihood estimation). It also allows assessment of goodness of fit by determining percentage of deviance explained by the model. Model validation was carried out through inspection of residual plots and k value.

GAMs were constructed for each of the taxonomic diversity measures (richness, Shannon's diversity index, abundance) and the functional diversity measure (Rao's functional diversity coefficient). Initially, the association between each diversity measure (response variable) and each of the environmental variables was modelled individually (4 response variables x 11 environmental variables). This was done in order to minimise overfitting in model construction. Environmental variables that showed a statistically significant relationship with the response variable were then integrated into a synthesis model for each response variable.

3.3.8 Beta diversity: variation in macroinvertebrate community composition (taxonomic) among ponds

Beta diversity represents the extent of compositional variation among habitats, within a given region or area (Whittaker, 1960). This variation is expressed with dissimilarity measures (in a dissimilarity matrix) that quantify the extent of (dis)similarities in species presence (or abundance) between all pairs of samples or sites (Legendre & De Cáceres, 2013). The two most used indices for ecological dissimilarity are Jaccard's coefficient and Sorenson's coefficient effectively quantify dissimilarity between two sites or communities based on species shared and those present in one community and not the other. Jaccard's dissimilarity is calculated as follows (Gotelli & Ellison, 2004)

Equation 8. Jaccard's dissimilarity. *d* is dissimilarity, *a* is the number of species present only in site *i*, *b* is the those only present in site *j* and *c* is the number of species present in both sites.

$$d_{i,j} = a + b/a+b+c$$

The dissimilarity matrix (pairwise dissimilarity coefficient for every pair of communities) can be partitioned into the two components of beta diversity: turnover and nestedness (Baselga, 2012; Legendre & De Cáceres, 2013). Turnover refers to the replacement of one or more species by another species from one community to the next (Legendre, 2014). Nestedness refers to where one community may have more species than another and all of the species in the smaller community may be the same as those found in the larger community (Legendre, 2014). A pairwise dissimilarity matrix can be decomposed into a pairwise turnover component matrix and a pairwise nestedness matrix (the two components add up to the total dissimilarity) making it possible to further understand how different ecological processes respond to environmental variables. Here, we used decomposition of pairwise dissimilarity coefficient matrix to calculate the relative contributions of turnover and nestedness to

total beta diversity, and applied ordination methods to pairwise turnover and nestedness matrices to examine their responses to environmental variables.

Ordination methods can also be applied to multivariate community composition and environmental data to examine patterns in community variation along compositional or environmental gradients (Whittaker, 1972). Here, beta diversity was examined with constrained and unconstrained ordination methods applied to multivariate community composition data. Unconstrained ordination methods like PCA and PCoA (principle coordinate analysis) are applied here to reduce the dimensionality of multivariate data while preserving maximum information (variation in data) so that data can be examined for patterns also be represented in two-dimensional space. Constrained ordination or canonical analyses (Redundancy Analysis) is applied to perform regression like analysis for the multivariate data. All of these allow description of community variation among all pond sites and the identification of environmental/spatial factors driving variation in community composition.

3.3.8.1 Differences in community composition between seasons

To determine if there were statistically significant differences in the community composition of the two seasons, two features of community data were calculated and compared: within-season variances (group dispersion), and centroids (group median). Anderson's (2006) test of homogeneity approach which allows for non-Euclidean dissimilarity measures more suitable for ecological data was implemented to calculate group dispersion. Permutational multivariate analysis of variance (PERMANOVA) was applied to examine differences in the centroids of the two seasons' community composition data.

Within season-variance (group dispersion) is measured as the average distance of a samples/sites (described by their species composition) to the season's (group) centroid (Anderson, 2006). Anderson's (2006) test of homogeneity was performed as follows. First, the function *betadisper()* from the 'vegan' package was applied to calculate the distances to group centroid (dispersion) for each season. This was followed by analysis of variance (ANOVA) to test for statistically significant differences in their dispersions. The analyses were applied to Sorenson dissimilarity matrix of the macroinvertebrate community data.

PERMANOVA is a non-parametric statistical test for multivariate data (Anderson, 2001). It tests the null hypothesis that the group centroids of the two seasons' data are equivalent. PERMANOVA was conducted with the function *adonis2()* from the R package 'vegan' on the Sorenson dissimilarity matrix of the macroinvertebrate community data. Statistical significance, the *p-value*, was determined with permutation tests implemented by *adonis2()*.

Similarly, PERMANOVA was carried out to test for statistically significant differences in water quality parameters between the two seasons. The input data for environmental analyses was the Euclidean distance matrix of water parameter data.

3.3.8.2 Calculating total beta diversity and the relative contribution of turnover and nestedness

Total beta diversity for the pond communities was calculated based on pairwise Jaccard dissimilarity coefficients (i.e. the dissimiliarity calculated for each pair of ponds with presence-absence data). The maximum value possible for the total beta-diversity is 0.5 (Legendre, 2014). The calculation follows the approach described in Legendre (2014) as implemented by the *beta.div.comp()* function in the 'adespatial' package (Dray et al., 2018). The method described in Legendre (2014) calculates total beta diversity as the total variance of a community dissimilarity matrix (first described in Legendre and De Cáceres (2013)) and also further partitions total variance into turnover and nestedness: each component's contribution to total beta diversity is then calculated as a fraction of total beta diversity. The function also produces pairwise turnover and nestedness matrices which can be subjected to further multivariate analyses to explore the response of each component to environmental variables (Legendre, 2014)

3.3.8.3 Unconstrained ordination - Patterns in macroinvertebrate community composition

Unconstrained ordination analysis was used to examine variation in community composition among the ponds. The ordination method used was transformation-based PCA in which species composition data is Hellinger-transformed first and Euclidean distance matrix is then calculated for this data, measuring the pairwise dissimilarities (similarities) among all the ponds (Legendre & Legendre, 2012). The ordination plot produced illustrates the similarities and differences in composition among the ponds and highlights the species associated with the ponds.

3.3.8.4 Environmental drivers of total taxonomic beta diversity, turnover and nestedness

The constrained ordination method, redundancy analysis (RDA), was performed to examine the relationship between pond community composition and environmental variables. RDA is a form of canonical analysis that is similar to regression in that allows mathematical description (modelling) of the relationship between response and predictor variables where response and predictor variables are multivariate. It also incorporates ordination methods that allow representation of these relationship in reduced dimensional space (Legendre & Legendre, 2012). RDA models can be tested for statistical significance with a permutations-based test and their explanatory power assessed with adjusted R^2 values. An adjusted R^2 refers to the redundancy statistic and estimates the proportion of the variation in the response variable that is explained by the predictors (Legendre & Legendre, 2012).

To perform RDA, the pooled abundance data (data from both sampling periods were combined for each site) was first Hellinger transformed to address the high frequency of zeroes in the data (Legendre &

Gallagher, 2001). Environmental parameters were averaged across the two seasons then standardized to address differences in measurement units. Then, the R function *rda()* (from R package 'vegan') was used to implement RDA to generate a full model explaining the relationship between composition variation and all the environmental variables. The R function *ordiR2step()* was subsequently used to apply a variable selection approach to identify most important explanatory variables explaining variation while maintaining maximum amount of variation explained by a model. Forward model selection is a variable selection procedure that begins modelling with a null model and adds predictors one at a time until stopping criteria for the modelling are met (Legendre & Legendre, 2012). The function *ordiR2step()* implements forward model selection for constrained ordination with permutation tests and adjusted R^2 values. The model selection parameters were set up to stop the selection when the following occurred – the adjusted R^2 started to decrease, or when the full model's adjusted R^2 was exceeded, or the permutation-based p-value exceeded the significance level of $p = 0.05$ (Oksanen et al., 2022). The final model thus comprised the selected variables of the model selection procedure. Permutations based significance tests (with the R function *anova.cca()*) were then used to determine the statistical significance of the final model, the predictor variables and the ordination axes. The ordination axes of the RDA, similar to a PCA axis, represents the direction of maximum variation in multivariate space (Legendre & Legendre, 2012). The adjusted R-squared value was also determined for the final model with the function *RsquareAdj()*. The functions *ordiR2step(), anova.cca()* and *RsquareAdj()* are all functions in the 'vegan' package.

RDA was also performed to examine the effect of environmental variables on the turnover and nestedness components of beta-diversity, separately. First, the R function *beta.div.comp()* (adespatial package) was used to derive triangular pairwise dissimalirity matrices (Jaccard dissimilarity coefficient) for nestedness-based dissimilarity and turnover-based dissimilarity for all pond pairs based on community presence-absence data. Then, using the *pcoa* function (package 'ape', Paradis et al., 2016), principal co-ordinate analysis (PCoA) was performed to derive eigenvectors for each of the distance matrices (that is one set each for turnover and nestedness). The PCoA eigenvectors were then used as the response variable in two separate RDAs (one model for turnover, one for nestedness), with the environmental and spatial variables as predictors. The function *ordiR2step* was used to implement forward model selection and identify the most important predictors and build the final model. The model selection parameters were set up to stop the selection when the following occurred – the adjusted R^2 started to decrease, or when the full model's adjusted R^2 was exceeded, or the permutation-based p-value exceeded the significance level of $p = 0.05$ (Oksanen et al., 2022). This was followed by permutation-based significance tests (with the function *anova.cca()*) to determine the

statistical significance of the final model, the predictor variables in the model and the ordination axes. The adjusted R^2 value for the model was also calculated with the function *RsquareAdj().*

Additionally, PERMANOVA was also performed to test for statistically significant differences in community composition between the pond clusters obtained from the clustering analysis. Subsequently, indicator species analysis was carried out with the function *multipatt()* from the package 'indicspecies' in R (De Caceres et al., 2016). The function calculates and ranks associations (Indicator Value index, Dufrêne and Legendre (1997)) between taxa and each predefined group (in this case, a pond group or type), then tests the statistical significance (applying a permutations-based test) of the association between a taxa and the group it has a high association with (De Cáceres, 2013). This makes it possible to identify taxa that can be potential indicators of a given type of site.

3.3.9 Beta diversity: variation in macroinvertebrate functional composition among ponds

3.3.9.1 Total functional beta-diversity, richness difference and turnover (replacement)

Differences in community composition in terms of the functional traits present was assessed by calculating total functional beta diversity and partitioning it into turnover (replacement) and richness difference. The procedures for calculating functional beta diversity with the R 'BAT' package (Cardoso et al., 2015) tools are based on the functional beta diversity partitioning framework where $\beta_{total} = \beta_{replacement} + \beta_{richness\ difference}$ as demonstrated in Cardoso et al. (2014).

To calculate the total functional beta diversity and the contribution of richness difference and species replacement to total beta-diversity, the site x species matrix (describing taxa abundance in each site) and the species x trait matrix (describing traits for each taxa) were used. Firstly, the R function *gowdis()*, from the 'FD' package (Laliberté & Legendre, 2010) was used on the scaled species x trait matrix to calculate trait distances (Gower's dissimilarity coefficient) between each pair of recorded taxa. The resulting trait matrix was then subjected to agglomerative clustering, specifically unweighted arithmetic average clustering, with the R function *upgma()* ('phangorn' package; Schliep (2011)) to generate a functional distance tree, clustering taxa based on trait dis(similarities). The R function *beta.multi* from the 'BAT' package was then used (with functional distance tree and site x species table as input) to calculate total functional beta diversity and the proportion of replacement and richness difference contributions to functional beta diversity.

To generate pairwise dissimilarity matrices for total functional beta-diversity and for species replacement and richness difference, the functional distance tree calculated above and the site x species matrix were used again. The R function *beta()* ('BAT' package) was employed to calculate pairwise distance matrices quantifying dissimilarities (total functional beta diversity) among ponds as

well as proportions of functional beta diversity explained by replacement and richness difference. PCoA was undertaken on each of the three pairwise dissimilarity matrices (total beta-diversity, species replacement and richness difference). The PCoA eigenvectors for total beta diversity, richness difference and species replacement were used as response variables in subsequent RDAs, following the approach developed by Legendre (2014).

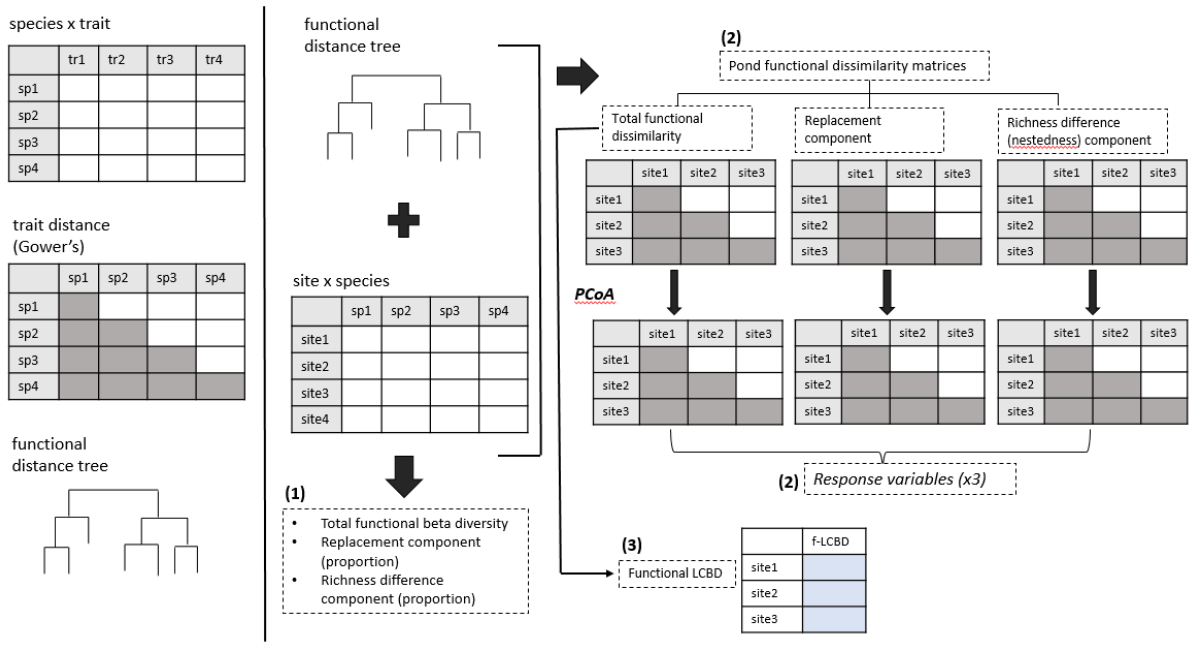


Figure 3.3 Diagram showing steps to calculating functional beta diversity values (1), pairwise functional beta diversity matrices (response variables) (2) and functional LCBD (3).

3.3.9.2 Environmental drivers of functional beta diversity, replacement and richness difference

RDA was also undertaken to examine the relationship between the macroinvertebrate functional composition and the measured environmental variables. The R function *rda()* was used to implement three separate analyses where the response variables were the PCoA eigenvectors of total beta diversity, replacement and richness difference described above. The constraints or explanatory variables were the environmental variables. The R function *ordiR2step()* was then used to apply forward model selection identifying the most important explanatory variables and build the final models. for each of the response variables. The model selection parameters were set up to stop the selection when the following occurred – the adjusted R^2 started to decrease, or when the full model's adjusted R^2 was exceeded, or the permutation based p-value exceeded the significance level of $p = 0.05$ (Oksanen et al., 2022). This was followed by permutations based significance tests (with the function *anova.cca()*) to determine the statistical significance of the final model, the predictor variables in the model and the ordination axes. The adjusted R^2 value for the model was also calculated with the function *RsquareAdj()*.

3.3.10 Local Contribution to Beta Diversity

3.3.10.1 Local Contribution to taxonomic Beta Diversity (LCBD)

The Local Contribution to Beta Diversity (LCBD) is a measure of ecological uniqueness, decomposing total beta diversity into individual sites contributions in a given area (Legendre & De Cáceres, 2013). LCBD values are comparative and involve decomposition of total beta diversity thus each pond is assigned a value representing the relative extent to which the pond contributes to total beta-diversity (Legendre & De Cáceres, 2013). To calculate LCBD, the R function *beta.div.comp()* was first used to calculate pairwise dissimilarity (Jaccard's coefficient for presence-absence data) among all ponds. The pairwise dissimilarity matrix produced (total beta diversity) was then used to calculate a total LCBD value for each pond with the R function *LCBD.comp()* ('adespatial' package). The function implements partitioning of total variation in community datasets as described in Legendre and De Cáceres (2013). To determine if there were statistically significant differences in the LCBD values between the two seasons, Mann Whitney's U test was performed with the R function *wilcox.test()*.

Similarly, the contribution of individual sites to the nestedness (LCBD-nest) and (LCBD-turn) components of total beta diversity was calculated. Total pairwise dissimilarity (calculated above) was decomposed into two matrices – pairwise dissimilarity for turnover and nestedness components. LCBD-nest and LCBD-turn values for each pond were obtained with the R function *LCBD.comp()* with the two matrices as input data. The associations between LCBD-nest and LCBD-turn, as well as the relationship with LCBD-total was determined with Spearman's correlation.

Linear regression models were used to examine the response of LCBD values (total, nestedness and turnover) to taxonomic richness. Following inspection of scatter plots, quadratic statistical functions were applied for LCBD-nest and LCBD-turn and linear function for LCBD-total.

Finally, relationships between LCBD values and environmental variables were modelled with regression analysis. The response variable LCBD-total was modelled against each environmental predictor individually. Similarly, the response of each of LCBD-turn and LCBD-nest to the environmental variables was determined with linear regression models.

The uniqueness of ponds was also considered in terms of their environmental characteristics. Firstly, a multivariate dataset comprising the following environmental variables was standardized: pH, water temperature, conductivity, TSS, pond area, proportion of shade over pond, extent of aquatic plant cover in pond, proportion of impervious land cover (within 500m radius), proportion of non-agricultural vegetation (within 2km radius), proportion of water cover (within 500m radius) and road density (within 2km radius). Then, a pairwise Euclidean distance matrix was calculated with the function *vegdist()* from the R package 'vegan'. The R function *lcbd.comp* was subsequently employed (with the Euclidean

distance matrix as input data) to determine environmental LCBD values for each of the ponds. Spearman's correlation was then calculated with the R function *cor()* to examine associations between environmental LCBD values and each of compositional LCBD values, taxonomic richness and Shannon's diversity index.

3.3.10.1 Local Contribution to functional Beta Diversity (f-LCBD)

The contribution of individual sites to total functional beta diversity (f-LCBD) was calculated to determine ecological uniqueness of ponds in terms of macroinvertebrate trait composition. Calculation f-LCBD, following the approach described in (Heino et al., 2022), required a functional (pairwise) community dissimilarity matrix produced with the steps described in section 3.3.8.1: Gower's dissimilarity coefficient, implemented with the R function *gowdis(),* was calculated for the scaled species x trait matrix to calculate trait distances between each pair of recorded taxa and reduce the dimensionality of the trait dataset. The resulting trait matrix underwent agglomerative clustering (unweighted arithmetic average clustering), with the R function *upgma()* to generate a functional distance tree, clustering taxa based on trait dis(similarities). Based on this functional distance tree and the site x species dataset pairwise functional dissimilarity among the ponds was calculated using the R function *beta.multi*(), producing a functional dissimilarity matrix. f-LCBD value for each pond was then calculated using the function *beta.div()* based on the functional community dissimilarity matrix (Figure 3.3).

Following inspection of scatterplots, linear regression models were performed to determine the response of fLCBD to (i) functional diversity, (ii) environmental LCBD and (iii) each of the environmental variables individually.

3.3.10 Ecological connectivity

While ponds are abundant in GKL, their physical arrangement such as distance to neighbouring ponds (structural connectivity), and surrounding land use may influence the extent to which they are accessible (functional connectivity) and hence able to support meta-community interactions. In order to answer questions about the relative importance of connectivity for diversity and community composition, structural and functional connectivity were quantified and their relationship with macroinvertebrate diversity patterns examined. Here, both aspects of connectivity were considered by using proximity metrics and cost distance metrics. A graph-based approach is adopted to quantify and characterize the links among ponds at a landscape scale. In graph based approaches, habitats are represented as nodes and the connectivity between habitats are described by links (Urban et al., 2009). Spatial data and species dispersal information (for example, maximum distance that a species can travel) are combined to describe distances between ponds, the importance of each pond and the overall

connectivity of the pond network in a given area (Heino, Alahuhta, et al., 2017). Figure 3.4 summarizes the analysis workflow.

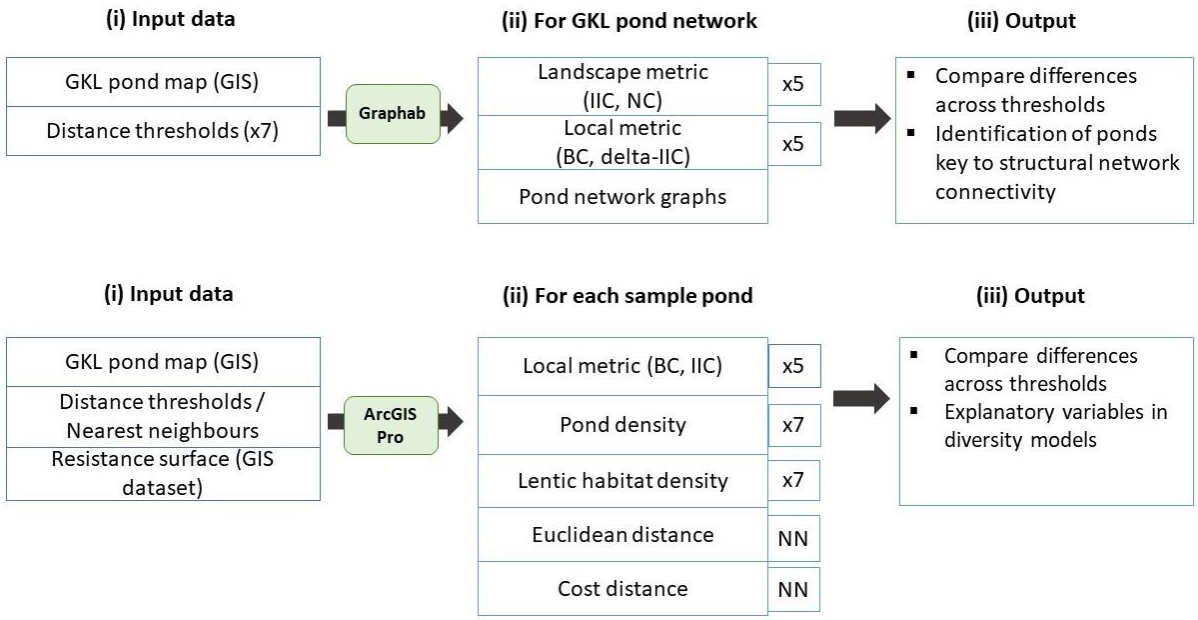


Figure 3.4 Connectivity analysis workflow including (i) preparing input data: spatial dataset (GIS) and ecological information on potential dispersal (distance threshold); (ii) lists of connectivity metrics calculated for the whole pondscape and for selected sample ponds (n=28); (iii) Output and subsequent analyses for connectivity metrics.

To identify areas within GKL that are characterized by a high degree of pond network connectivity, landscape-scale connectivity (Section 3.3.10.1) metrics were first calculated and pond network graphs produced for multiple distance thresholds, using the Graphab 2.8 software (Foltête et al., 2012). To account for the varying dispersal distance capabilities among macroinvertebrates, the metrics were calculated at multiple distance thresholds (100m, 250m, 500m, 1000m, 1500m, 2000m, 2500m) following Thornhill (2013). To identify ponds that are integral to pond network connectivity within the area, individual pond (local scale) connectivity metrics were calculated.

To examine the effects of structural and functional connectivity on macroinvertebrate diversity and community composition, six connectivity metrics were calculated (as above, at multiple distance thresholds) for each of the sample ponds in the study (section 3.3.10.2). Potential functional connectivity metrics include cost distance to nearest ponds and cost distance to nearest ponds weighted by their area. Cost distance refers to the accumulative distance between habitats along surfaces that represent varying resistance to movement (Heino, Alahuhta, et al., 2017; Kärnä et al., 2015). The accumulative cost for the optimal path was calculated along a cost surface where the landscape between ponds (urban matrix) has a resistance score representing relative ease or difficulty of movement for species (Heino, Alahuhta, et al., 2017). Resistance multipliers ranging from 1 (low resistance to movement) to 100 (high resistance to movement) were assigned to nine land use classes

for the land use land cover map of GKL (cell size = 10). Macroinvertebrates represent a range of organisms that have varying movement methods including passive dispersal along water routes (rivers, streams) or hosts (birds), and active dispersal by flight. Moreover, species within the same taxonomic group can show different flight capacities. These characteristics of macroinvertebrates and the limited available empirical data on movement patterns for most species in urban areas, mean that assigning resistance scores is often based on expert opinion (but see Hyseni et al. (2021)) and/or representative taxonomic groups. For the present study, the resistance scores were set (Appendix D6) based on the values assigned in Thornhill (2013) pond macroinvertebrates in a UK urban landscape (Thornhill, 2013).

3.3.11.1 Calculating landscape scale pond connectivity in GKL

The landscape metrics calculated for the GKL pond network was the Integral Index of Connectivity (IIC). This is a widely applied index of landscape connectivity that summarizes the degree of connectivity based on patch availability, the number of links among them and the total area of study region (Pascual-Hortal & Saura, 2006). The index value ranges from 0 to 1 with higher value indicating higher connectivity. The IIC formula is as follows

Equation 9. IIC. A is the area of the study region, i and j are patches, a_i a_j their areas, and nl_{ij} the number of links between them

$$\text{Integral Index of Connectivity (IIC)} = \frac{1}{A^2}\sum_{i=1}^{n}\sum_{j=1}^{n}\frac{a_i a_j}{1+nl_{ij}}$$

This was calculated at five distance thresholds to describe the potential change in connectivity for species of different dispersal capabilities. It was also calculated with two surface rasters – one with a resistance (movement cost) value assigned to each cell of the urban matrix and another without.

At a local scale, the metrics delta-IIC (connector) and the weighted, IIC-based Betweenness Centrality (BC) were calculated for each pond to assess their importance for overall connectivity in the landscape. The delta IIC metric achieves this by examining the effect of the removal of a node (in this case, a pond) on the overall pond network connectivity. The BC metric on the other hand quantifies the degree to which a given node, or pond, maintains connectivity among other nodes or ponds by measuring how many other links pass through it (or are connected through it) relative to others in the network (Freeman, 1977). That is, the node or pond is one through which many individuals would pass through more often as they move around the landscape (Bodin & Norberg, 2007). IIC-based BC is weighted by the area and inter node distances (Bodin & Saura, 2010). All of the above metrics were calculated with the software Graphab (Foltête et al., 2012).

3.3.11.2 Calculating pond connectivity for each of the sample ponds

To assess the effects of the availability and access to ponds on diversity and community structure, the following connectivity variables, altogether 34, were calculated at multiple distance thresholds (Thornhill 2013) for the 28 sample ponds.

1. Pond density (PD)
2. Wetland habitat density (WD)
3. Euclidean distance to nearest pond and average distance to nearest 5 and 15 and all within 1km and 2km (ED)
    a. Euclidean distance (area weighted) (ED-A)
4. Cost distance to nearest ponds (CD) and average distance to nearest 5 and 15 and all within 1km and 2km (ED)
    a. Cost distance (area weighted) (CD-A)

In addition, BC and delta-IIC values for the 28 sample ponds were also extracted from the output of the analysis described in the previous section (3.3.11.1) to be included as explanatory variables.

Pond density and aquatic habitat density were calculated in ArcGIS by creating buffers at radii distances of 100m, 250m, 500m, 1000m, 1500m, 2000m, 2500m and counting the number of ponds (1) and total aquatic habitats including lakes, reservoirs (2). ED, CD, ED-A and CD-A were based on calculations done in Excel and ArcGIS Pro tools, and included distance to nearest pond, average distance to nearest 5 and 15 ponds, average distance of ponds within 1km and average distance of ponds within 2km. Euclidean distance (ED, ED-A) to nearest ponds were obtained with the Nearest Neighbor tool in ArcGIS Pro. The cost distance for the least cost (optimal) path was obtained with the ArcGIS tools Distance Accumulation and Optimal Path as Line. The Distance Accumulation tool calculates the accumulated distance to a given source location; in the present study, cost distance accumulation raster layers were created for each sample pond (source) representing cost distance to the pond given different costs to movement presented by land use-land cover type (resistance scores outlined above in 3.3.10). To calculate the cost distance of the path to each pond from the nearest ponds, the Optimal Path as Line tool was used with the cost distance accumulation rasters (for each sample pond) and the location of the nearest ponds as input. The method for calculating area weighted variables was based on Thornhill (2013) where for a given sample pond, the area of each of its nearest ponds is divided by the distance (Euclidean or cost) between the pond and the sample pond.

3.3.11.3 Modelling relationships between diversity patterns and pond connectivity

Correlations among the connectivity variables were assessed with Spearman's correlation to address potential collinearity in subsequent modelling (Legendre & Legendre, 2012). Where the correlation coefficient was greater 0.7, only one of the variables was retained for further analyses.

Linear regression models were used to examine the response of the diversity measures taxonomic richness, Shannon's index, functional diversity, taxonomic-LCBD and functional-LCBD to each of the connectivity variables. The models for the latter measures were fitted with Gaussian distributions set as the error distribution, whereas the Poisson distribution was used for modelling taxonomic richness (discrete response variable) with the function *glm()* in R. Where model validation reveals the Poisson distribution to be inadequate to model variance in a discrete/count response variable, the negative binomial distribution is applied with the function *glm.nb()* in R package 'MASS' (Ripley et al., 2013; Smith & Warren, 2019).

Redundancy analysis (RDA) was performed to examine the relationship between community composition variation and the connectivity variables. First, the connectivity variables were standardized and the community data was Hellinger-transformed (Legendre & Gallagher, 2001). Then the R function *rda()* was used to build a model with the connectivity variables as explanatory variables. This was followed by the variable selection procedure, to identify the variables that explain the greatest amount of variation in community composition. This variable selection procedure was implemented with the function *ordiR2step()*. The model selection parameters were the same as those set up for modelling with environmental variables (3.3.7.4), that is, to stop the selection when the adjusted R^2 started to decrease, or when the full model's adjusted R^2 was exceeded, or the permutation-based p-value exceeded the significance level of $p = 0.05$ (Oksanen et al., 2022). The resulting model was then subjected to permutations-based significance tests to determine whether the full model, the connectivity variables and the ordination axes were statistically significant.

Chapter Four

Pond diversity patterns, community composition variation and drivers

4. POND DIVERSITY PATTERNS, COMMUNITY COMPOSITION VARIATION AND THEIR DRIVERS

This chapter presents the results of analyses quantifying diversity and characterizing macroinvertebrate community composition in GKL ponds. First, environmental characteristics of the urban ponds are described. Then multiple (alpha) diversity metrics are calculated for ponds and their responses to environmental variables are described. Finally, variation in community composition (beta-diversity) is quantified and main environmental drivers structuring communities are identified.

4.1 Environmental characteristics

4.1.1 Summary of pond environmental characteristics

Mean pond surface area was 7,893 m2 but pond areas ranged from 35.2m^2 to 19,692.5m^2. In general, pond surfaces tended to not be shaded but shading around pond margins was more varied. Most ponds had natural banks although some were partially bordered by concrete or stone walls while some had very steep, high earth slopes. Larger ponds tended to have scarce or no aquatic vegetation cover (Pearson's r = -0.53). Aquatic vegetation recorded was primarily *Hydrilla* or *Nymphaea* species. Mean percentage coverage of submerged or floating vegetation was 12.3 % (SD=18.1) and 12.9 % (SD=27.7) respectively, whereas mean open water percentage cover was 67.1% (SD=30.3). Large variations were evident in water conductivity and total TSS (Table 4.1). None of the ponds were ephemeral. Table 4.1 presents a summary of pond physical features and mean water quality parameters (see Appendix Table B1 for each season's summary).

4.1.2 Surrounding land use

Figure 4.1 shows the proportion of different land use (land cover) types for all of the ponds studied. Most of the ponds were surrounded by impervious land cover (including buildings and roads) that made up more than 40% of the surrounding land cover (range: 40-87%) at 500m and 2km radii. Only 4 ponds had less than 40% impervious land cover at 500m and 2km radius). Median distance to a road was 34m (minimum: 0.64m; maximum: 356m). Surrounding land use data at 100m and 250m radii were highly correlated with land use data within 500m radius and were thus excluded from further analysis (see Appendix Figure B4).

Table 4.1 Descriptive statistics of pond physical features and water parameters

Variable	Minimum	Maximum	Mean	SD	Median
Area (m^2)	35.2	19692.5	7893.0	6336.3	7432.65
Perimeter (m)	22.1	955.3	425.7	263.5	446.35
Shade (%)					
Water overhung (%)	0.0	40.0	8.2	10.2	5
Pond margin overhung (%)	0.0	80.0	32.1	29.8	20
Bank type (%)					
Natural earth	10.0	100.0	85.5	27.1	100
Stone	0.0	75.0	8.4	22.5	0
Concrete	0.0	90.0	5.0	17.7	0
Lined	0.0	30.0	1.1	5.7	0
Macrophyte cover (%)					
Submerged	0.0	70.0	12.3	18.1	10
Emergent	0.0	60.0	5.9	12.3	0
Floating	0.0	90.0	12.9	27.7	0
Open water	10.0	100.0	67.1	30.3	75
Floating algae	0.0	30.0	1.1	5.7	0
Woody/leafy debris	0.0	20.0	0.7	3.8	0
Shoreline vegetation (%)					
Grass	0.0	100.0	67.7	33.5	80
Shrub	0.0	80.0	13.8	23.7	0
Tree	0.0	80.0	5.5	17.5	0
Stone	0.0	70.0	9.1	21.3	0
Water parameters					
pH	6.7	8.7	7.7	0.4	7.6
Conductivity (us/cm)	14.4	298.5	103.1	64.95	107.15
Salinity (ppt)	0.0	0.1	0.1	0.1	0.05
DO (mg/l)	1.0	9.5	4.7	2.6	4.85
DO (%)	12.8	123.5	61.0	34.25	60.05
Total dissolved solids (TDS)	9.4	182.2	62.9	39.5	66.8
Temperature (˚C)	25.4	32.6	28.35	1.4	28.4
Hardness	7.4	79.2	33.7	19.27	33.75
Total suspended solids (TSS)	5.0	93.5	29.4	26.97	17.25

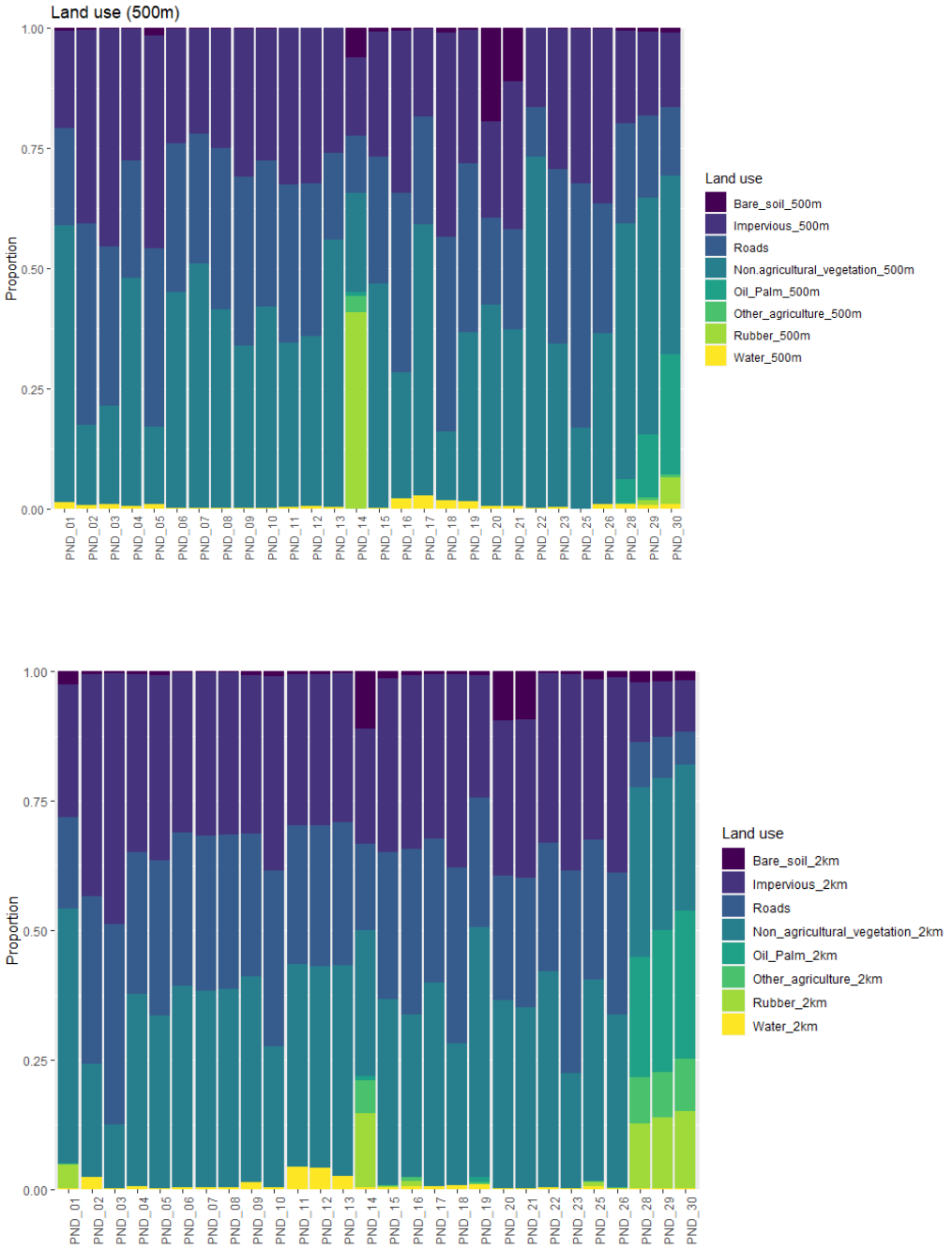


Figure 4.1 Relative proportions of land use – land cover surrounding each pond (n=28) within a 500m radius (top) and a 2km radius (bottom).

4.1.3 Seasonal differences in water quality parameters

There were no differences in pond structure variables (area, aquatic vegetation cover, shade) recorder over the two seasons except for a decrease in extent of vegetation cover in two of the ponds. Excluding pH, which showed a small difference in means between late monsoon and early season (Table 4.2), there were no significant seasonal differences among any of the recorded variables. There was no

significant difference (PERMANOVA: *Pseudo* $F_{1,51}$ = 0.03, p = 0.85) in overall water quality between the two seasons (Appendix B7).

Table 4.2 Two-way t-test results comparing water quality variables for two seasons (S1 late monsoon season, S2 early monsoon season). *Remaining water quality variables (non-normal distributions, Kruskal-Wallis tests) showed no significant differences (see Appendix Table B3).*

Parameter	S1 mean	S2 mean	T	df	P value
pH	7.8	7.5	2.11	50.796	**0.03**
Temperature	28.41	28.07	0.87	50.496	0.3
Hardness	33.77	32.74	0.17	42.01	0.85

4.1.4 Correlations among the ponds' environmental variables

Salinity, nature of bank material, water cover (within 2km radius) and agricultural land cover variables were excluded from regression analyses as there was minimal variation between ponds in their respective values. Distance to road was excluded as it had extreme values. Hardness and TDS were excluded as they are highly correlated with conductivity (Pearson's r >0.7). Similarly, impervious land cover (within 2km radius) was highly correlated with road density at 2km (Pearson's r = 0.86) and thus removed. Road density (500m) was highly correlated with impervious cover at 500m (Pearson's r = 0.76) and non-agricultural vegetation was negatively correlated with impervious cover at 500m (Pearson's r=0.75) and only impervious cover at 500m was retained (see Appendix Figure B4). Thus, the final set of variables retained for analyses comprised water quality variables: pH, Temperature, Conductivity, TSS; pond physical features: Shade, Aquatic vegetation cover, Area; and surrounding land use variables: proportion of non-agricultural vegetation (2km radius); water (500m); impervious cover (500m radius); road density within 2km radius.

4.1.5 Variation in environmental conditions among the ponds

PCA shows loose grouping mainly along vectors related to surrounding land use, surface area and extent of aquatic vegetation cover. The two PC axes explain 44.2 % of variation in ponds in terms of environmental conditions. Along PC1 (Figure 4.2), ponds can be grouped into those characterized by greater extent of aquatic vegetation cover, smaller surface areas and lesser proportions of impervious land cover (within 500m). Conversely, ponds positioned in the opposite direction have greater proportions of surrounding impervious cover (within 500m) and road density (within 2km) as well as greater surface areas. Notably, conductivity is positively and highly correlated with impervious cover. The ordination also reflects the distribution of ponds in GKL where the larger ponds are found in highly built up residential or commercial areas with narrower buffer or park zones (ponds 2, 3,5, 18, 19, 26). On the other hand, the smaller (ponds 6, 8, 17, 22) are more common within public gardens or forest parks with greater amounts of surrounding vegetation. The latter set of ponds were also more likely to have greater amounts of shade because of the presence of mature trees.

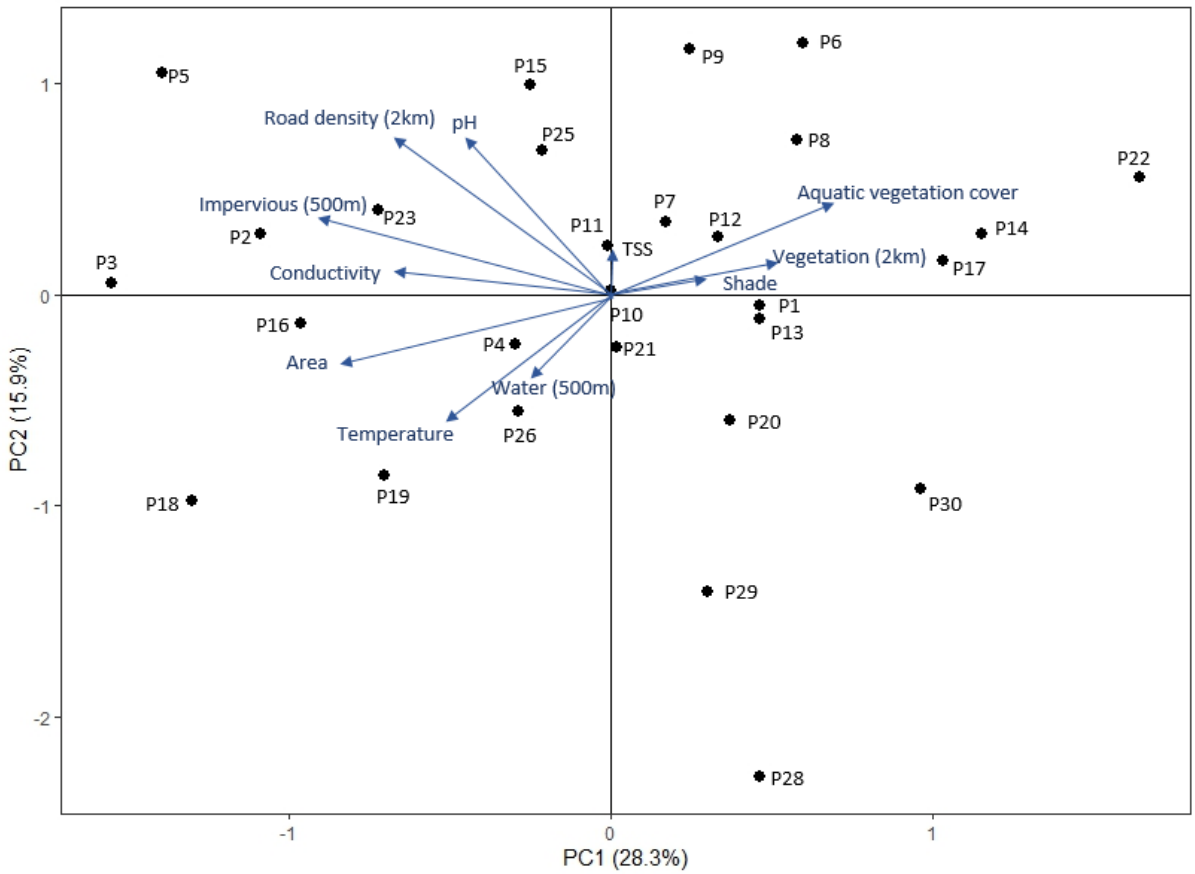


Figure 4.2 PCA showing distribution of ponds in terms of environmental variables. Closed circles are ponds described in terms of environmental variables and arrows indicate strength and direction of each variable.

Clustering analysis yielded 2 groups as the optimal number of clusters based on similarities in local environmental characteristics. The two groups primarily differed in surface area, extent of aquatic vegetation cover, water conductivity and total suspended solids (Figure 4.3). The two groups were uneven with twenty ponds in cluster 1 and eight ponds in cluster 2.

Figure 4.3 Comparison of the local environmental characteristics of the two pond groups.

4.2 Diversity patterns

4.2.1 Taxonomic diversity patterns

The number of macroinvertebrate individuals collected from 28 ponds over two seasons was 11,081 with a total of 99 taxa from nine orders. Estimated gamma diversity was 111.05 (SE 7.08) taxa (Table 4.3). Across both seasons, taxa richness within individual ponds ranged from 8 to 38 (mean=21.36, SD = 8.21) and Shannon's diversity index ranged from 0.79 to 2.94 (mean = 2.06, SD = 0.51). Table 4.4 presents a summary of alpha diversity for ponds across two seasons. Only six ponds had communities with 30 or more taxa; 9 ponds had a taxa richness of 20 - 29 while 13 ponds recorded less than 20 taxa each. The most taxa rich orders were Odonata (25 taxa) and Heteroptera (24 taxa), while only one Palaemonidae taxa was recorded, present in 18 of 28 ponds.

Table 4.3. Chao estimate of gamma diversity based on pooled macroinvertebrate abundance data.

Gamma	No. of taxa	Chao	95% CI	n
	99	111.05	97.15 – 124.95	28
Total abundance	11,081			

Table 4.4 Summary of diversity metrics for pooled macroinvertebrate data diversity metrics summary.

Parameter	Minimum	Maximum	Mean	SD
Taxonomic richness	8	38	21.36	8.21
Abundance	46	1766	395.8	324.95
Pielou's evenness	0.29	0.86	0.68	0.12
Shannon's diversity index	0.79	2.94	2.06	0.51
Berger-Parker dominance	0.16	0.83	0.36	0.16

A total of 20 taxa were only found in one pond and these were primarily comprised of Odonata (35%, n=7) and Coleoptera (30%, n=6) species. Approximately 32% (n=9) of ponds had taxa that were only found in one pond. Nearly half of those occurring in a single pond were found in pond 14. The rest were found among eight other ponds that did not exhibit distinct patterns in their environmental or diversity metrics otherwise.

Heteroptera and Diptera taxa were the most abundant (Figures 4.3, 4.4), with 2994 and 2252 individuals recorded respectively, and were present in all ponds. Coleoptera (104 individuals) and Lepidoptera (279 individuals) had limited distributions (Figure 4.4). Species of the Heteroptera genus *Nychio* and the families Pleidae and Gerridae were present in over half the ponds; Diptera were primarily represented by Chironomidae including subfamilies Chironominae and Tanypodinae. Lepidoptera were primarily found in ponds 10 and 17, and Coleoptera mostly in pond 14.

A total of 5 of the 11 Coleoptera taxa recorded were present in only one pond (pond 14) while other Coleoptera species were also recorded among 13 other ponds. However, while the rest of the ponds had an average abundance of about one Coleoptera individual, Coleoptera abundance in pond 14 was

higher (mean = 8 individuals per taxon). Similarly, aquatic Lepidoptera (sub-family Acentropinae) had limited distribution (present in 7 ponds) with most of the occurrence distributed among pond 10, and ponds 6, 8 and 17 (high diversity ponds).

The range in taxonomic diversity reported here is comparable to values found from non-tropical urban ponds. While number of taxa (richness) reported from a town and a major conurbation in the UK are higher – Hill et al. (2016) reported 170 taxa among urban ponds in the town of Loughborough (UK) and Thornhill et al. (2017) reported 194 taxa from a major conurbation – Hassall and Anderson (2014) reported 55 taxa from ponds across Ottawa (n=30). However, the range of taxon richness and Shannon's diversity reported here are similar to those of urban, high nutrient pond types reported in Hassall and Anderson (2014). The relatively higher proportions of Diptera, Hemiptera and Gastropoda has previously been recorded among urban ponds although the proportion of Coleoptera recorded in this study is relatively smaller (Hill et al., 2016; Thornhill et al., 2017).

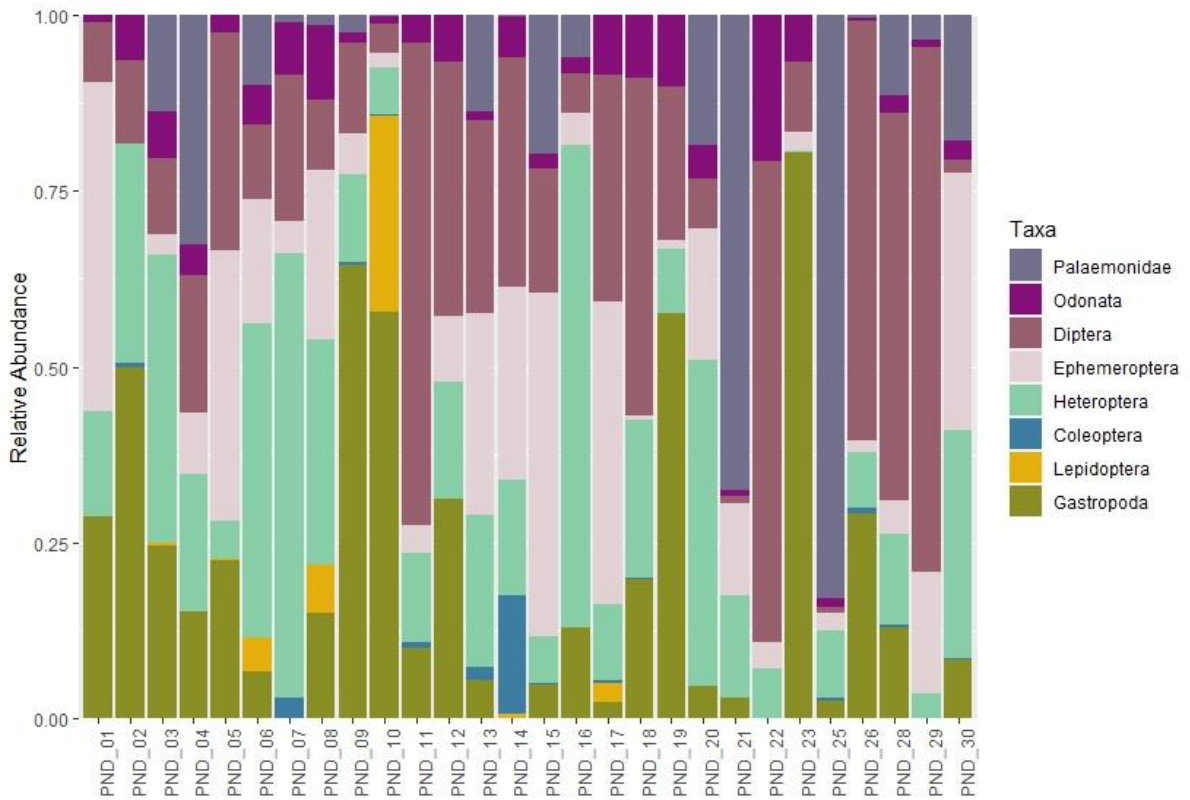


Figure 4.3. Relative abundance of macroinvertebrate groups in the ponds studied (n=28).

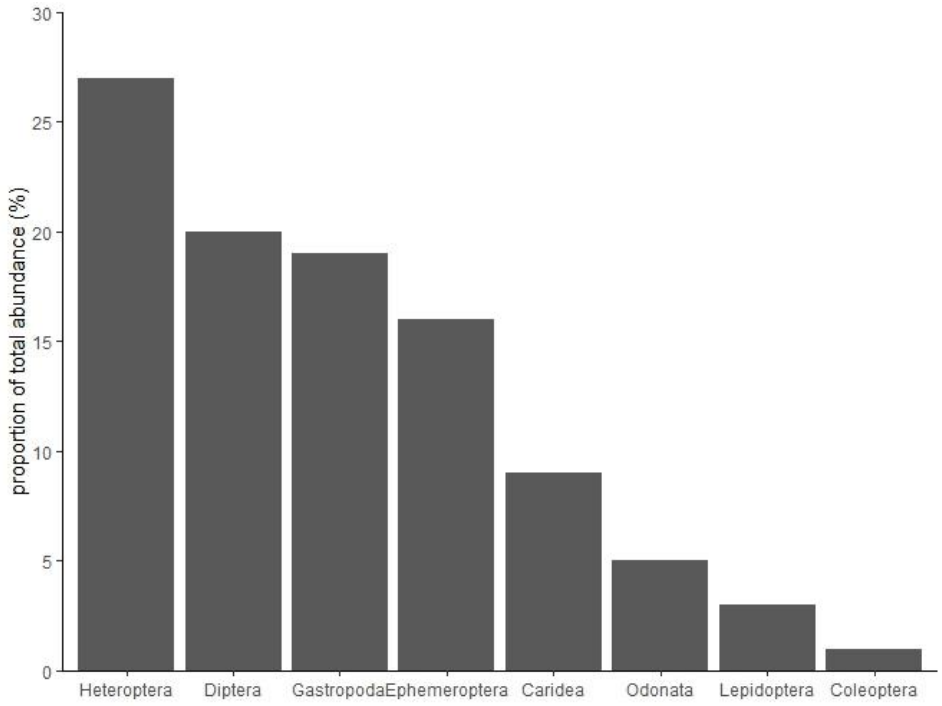


Figure 4.4. Relative abundance of taxonomic groups among the ponds (n=28).

4.2.2 Seasonal differences in alpha diversity metrics

Figure 4.5 illustrates differences in diversity metrics between two seasons. t-test for taxonomic richness, Shannon's diversity index and Pielou's evenness showed no significant seasonal differences (Table 4.5). Similarly, Kruskal-Wallis test showed no significant difference ($p>0.05$) in abundance between the two seasons (*Chi-sq*=2.8*, p-value*=0.09).

Table 4.5 Summary of two-way t-test results comparing diversity values for the two seasons.

Parameter	S1 mean	S2 mean	t	df	P value
Shannon's index	1.82	1.71	0.74	51.9	0.45
Taxonomic richness	14.82	13.5	0.74	49.8	0.45
Pielou's Evenness	0.71	0.67	1.06	52.25	0.29

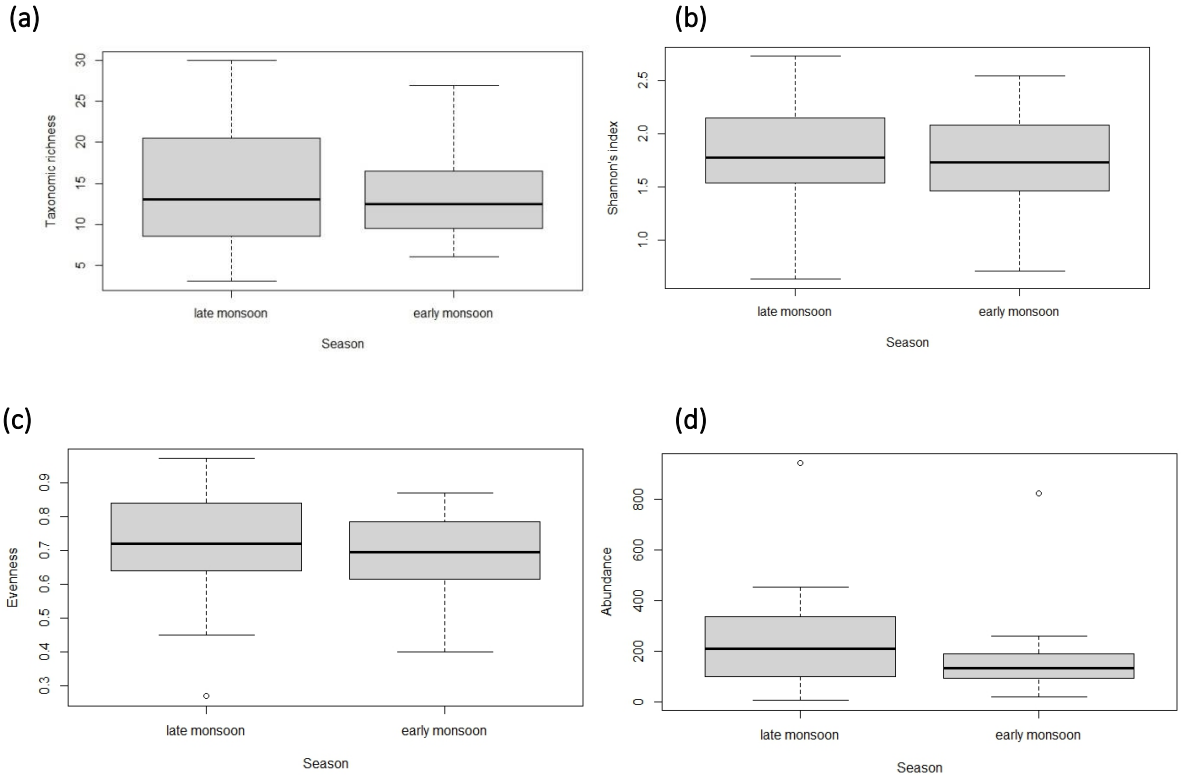


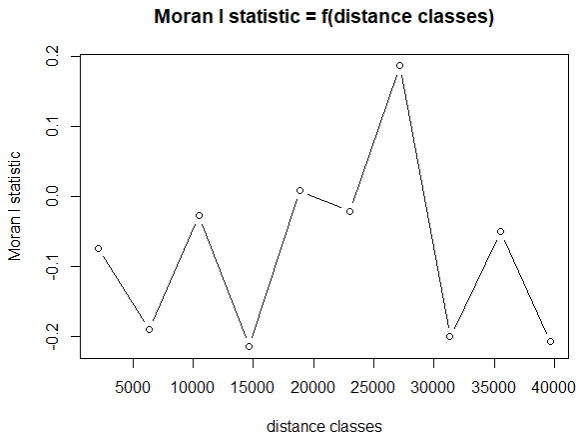
Figure 4.5 Boxplots of diversity metrics a) taxonomic richness, b) Shannon's diversity index, c) Pielou's evenness and d) abundance recorded among the 28 sampled ponds across the two seasons. The boxes represent the interquartile range of the metric for each season. The thick black lines represent median value for each season. The whiskers represent the 1.5xIQR lower and upper bound.

4.2.3 Spatial autocorrelation analysis

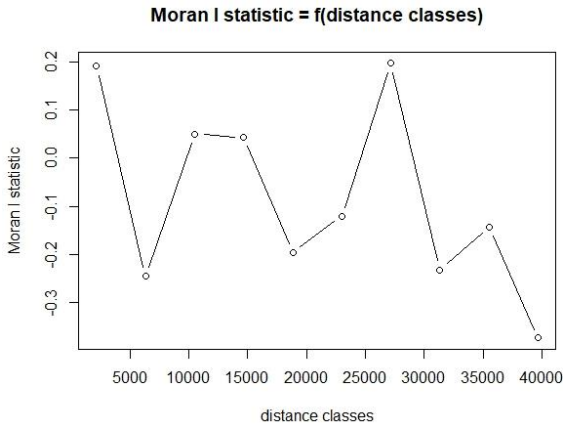
No significant spatial autocorrelation (p >0.05) was evident for either taxonomic richness or Shannon's diversity index (Figure 4.6).

Figure 4.6. Moran's I correlograms showing spatial autocorrelation values for (a) taxonomic richness and (b) Shannon's diversity index at 5km geographic distance intervals.

(a) Richness



(b) Shannon's diversity index



4.3 Drivers of diversity patterns

4.3.1 Taxonomic richness and environmental variables

GAMs were fitted for the response variable taxon richness (pooled) and each of the environmental variables separately. Richness was significantly related to five of the variables. These were area (*edf* = 2.2, *adj.*R^2 = 0.06, *p* = 0.01), aquatic vegetation cover (*edf* = 1.5, *adj.*R^2 = 0.36, *p* < 0.0001), TSS (*edf* = 2.5, *adj.*R^2 = 0.03, *p* = 0.03), water cover within 500m radius (*edf* = 3.69, *adj.*R^2 = 0.229, *p* = 0.0001) and road density within 2km radius (*edf* = 2.19, *adj.*R^2 = 0.05, *p* = 0.017). These were integrated into a single synthesis model. Only three of the variables were statistically significant (p <0.05) in the synthesis model, namely aquatic vegetation cover, TSS, and water cover within 500m radius. Figure 4.7 presents partial effect plots for the statistically significant variables in the model.

Table 4.6 Summary of synthesis GAM for taxon richness.

Response variable	Predictors (p value)	edf	p-value	Model R-sq. (adj)	Deviance explained
Taxonomic richness	Aquatic vegetation cover	2.1	0.004	0.57	70.5%
	TSS	1.00	0.034		
	Water cover (within 500m)	2.6	0.02		
	Road density (within 2km)	1.4	0.10		
	Area	1.00	0.53		

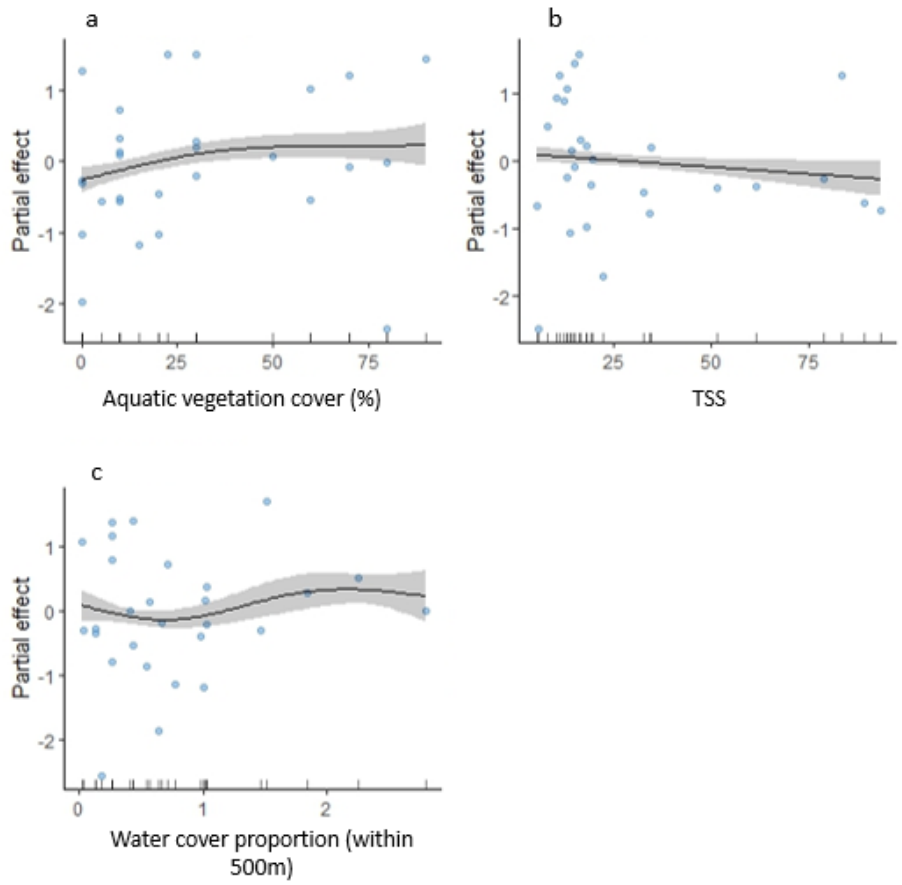


Figure 4.7 Partial effect plots showing the relationship between taxonomic richness and (a) aquatic vegetation cover (b) TSS and (c) water cover (500m radius). Blue circles represent partial residuals. The shaded area represents 95% confidence intervals.

4.3.2 Shannon's diversity index and environmental variables

GAMs were fitted for the response variable Shannon's diversity index (pooled) and each of the environmental variables separately. Shannon's diversity values were significantly related to two of the variables – aquatic vegetation cover (*edf* = 1, *adj.*R^2 = 0.27, *p* = 0.002) and TSS (*edf* = 1, *adj.*R^2 = 0.216, *p* = 0.007). The synthesis GAM showed both of the variables were statistically significant (Table 4.7). Figure 4.8 presents partial effect plots for variables in the model.

Table 4.7 Summary of final GAM for Shannon's diversity index.

Response variable	Predictors	edf	p-value	Model R-sq. (adj)	Deviance explained
Shannon's diversity index	Aquatic vegetation cover	1.00	0.0036	0.43	48.2%
	TSS	1.41	0.026		

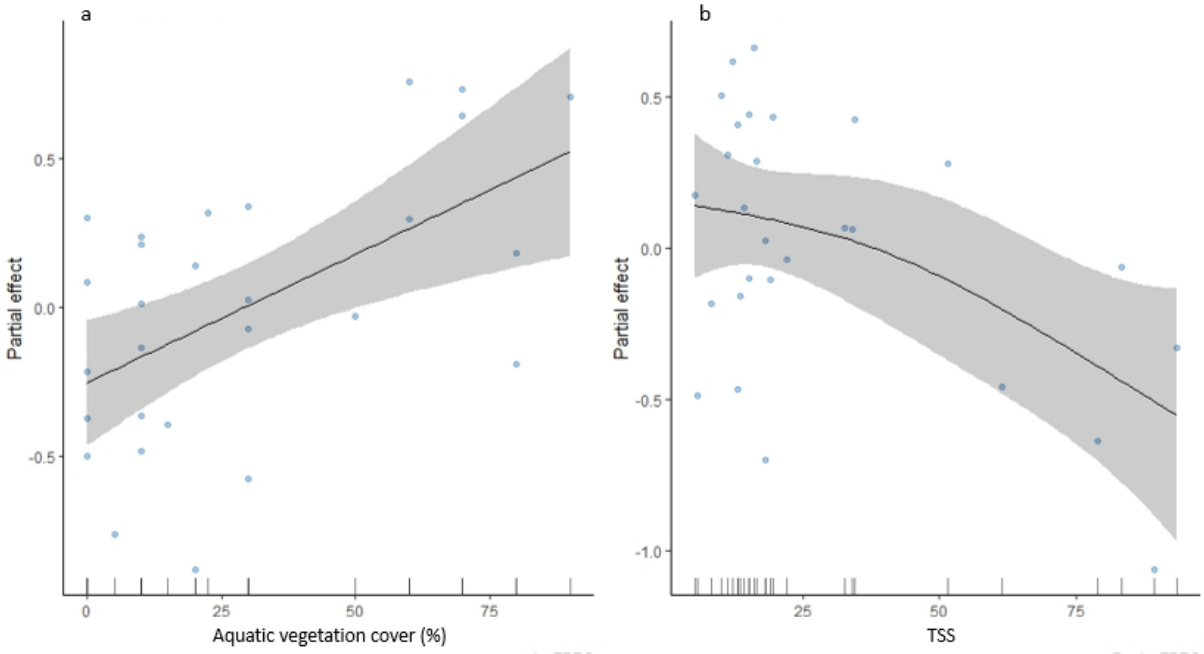


Figure 4.8 Partial effect plots showing the relationship between Shannon's diversity index and a) aquatic vegetation cover and b) TSS. Blue circles represent partial residuals. The shaded area represents 95% confidence intervals.

4.3.3 Abundance and environmental variables

GAMs were fitted for the response variable abundance (pooled) and the each of the environmental variables separately (Appendix B9). Two of the variables were significantly related with abundance – area (*edf* = 1, *adj.*R^2 = 0.127, *p* = 0.028) and temperature (*edf* = 1, *adj.*R^2 = 0.127, *p* = 0.039). Both variables were included as predictors in a synthesis GAM model for abundance. Deviance explained by the synthesis model but neither variable was statistically significant in the model (Table 4.8). Figure 4.9 presents partial effect plots for area and temperature.

Table 4.8 Summary of synthesis GAM for abundance (n=27).

Response variable	Predictors	edf	p-value	Model R-sq. (adj)	Deviance explained
Abundance	Area	1.00	0.15	0.17	19.1%
	Temperature	1.00	0.16		

4.3.4 Differences in alpha diversity between two pond clusters

Two sample t-tests show statistically significant differences between the two clusters of ponds in terms of taxonomic richness ($t_{10.4}$ = -2.3, *p* = 0.03) and Shannon's diversity index ($t_{10.4}$ = -2.6, *p* = 0.02). Mean taxonomic richness was greater for cluster 2 (mean = 27.25, SD = 26.5) than cluster 1 (mean = 19, SD = 6.8) (Figure 4.9). Similarly, Shannon's diversity index was higher in cluster 2 (mean = 2.4, SD = 0.49) than in cluster 1 (mean = 1.9, SD = 0.45). There were no significant differences in abundance between the two groups (Mann-Whitney: *W*=54, *p*=0.19).

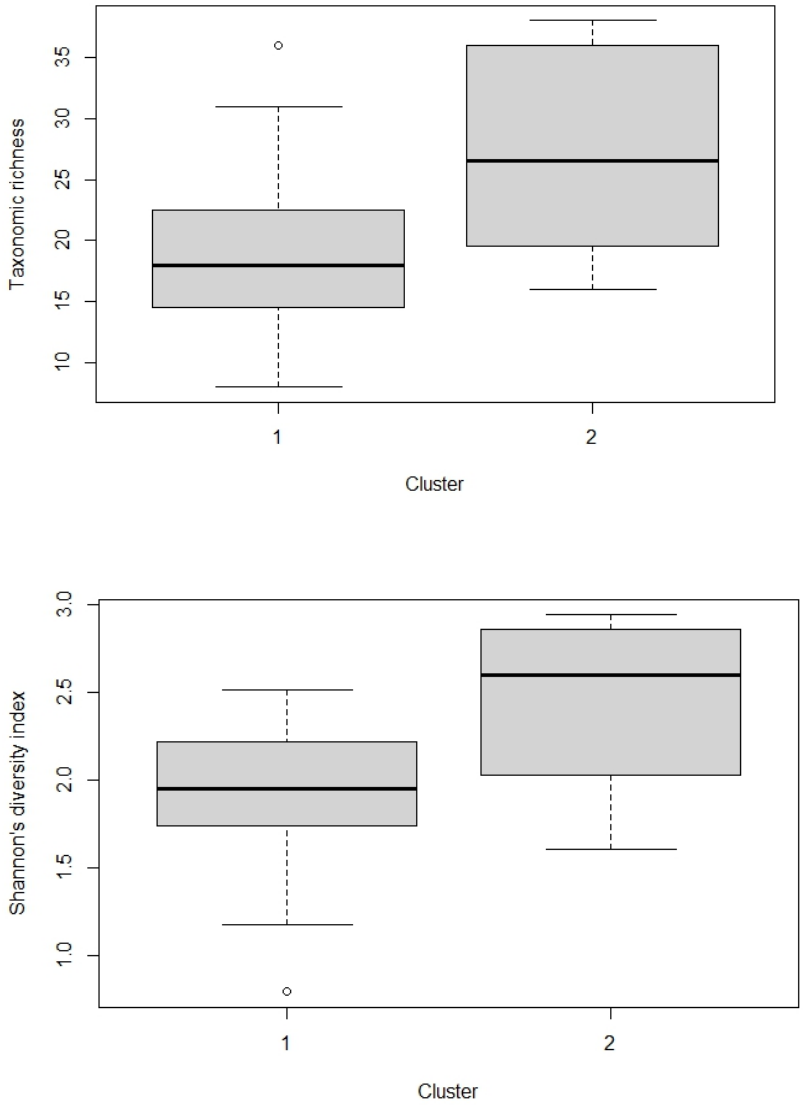


Figure 4.9 Boxplots comparing (top) taxonomic richness and (bottom) Shannon's diversity index for the two pond clusters. The boxes represent the interquartile range of distances to centroid for each cluster. The thick black lines represent median distance to centroid for each season. The whiskers represent the 1.5xIQR lower and upper bound.

The findings support the hypothesis that variation in taxonomic alpha diversity patterns is driven by variation in local physical or habitat characteristics although the support for the role of surrounding land cover is weak. Aquatic vegetation cover has a positive relationship with richness and diversity whereas increasing levels of TSS negatively influence richness and diversity. The strong relationships between physical habitat characteristics are consistent with findings from other authors (Hill et al., 2016; Oertli & Parris, 2019). These factors are strongly linked to habitat quality (resource availability, optimum water quality) and suggest that physical habitat characteristics are relatively more important than surrounding land cover in supporting greater diversity in ponds.

4.4 Beta diversity: variation in macroinvertebrate community composition among ponds

4.4.1 Seasonal differences in taxonomic community composition

There were no significant seasonal differences (ANOVA: F_1=0.99, p=0.3) in dispersion (variance) of community composition (Figure 4.10). Table 4.11 presents the average distance to group centroid for community composition for two seasons. However, PERMANOVA results show small but significant differences in community composition (distance index – Sorenson) between the two seasons (PERMANOVA: *Pseudo* F_1=2.28, p = 0.001). Figure 4.11 presents a two-dimensional visualization comparing the centroid and dispersion of community composition for the two seasons.

Table 4.11 Average distance of each site's community composition to community centroid.

	Late monsoon season	Early monsoon season
Average distance to centroid	0.5937	0.5700

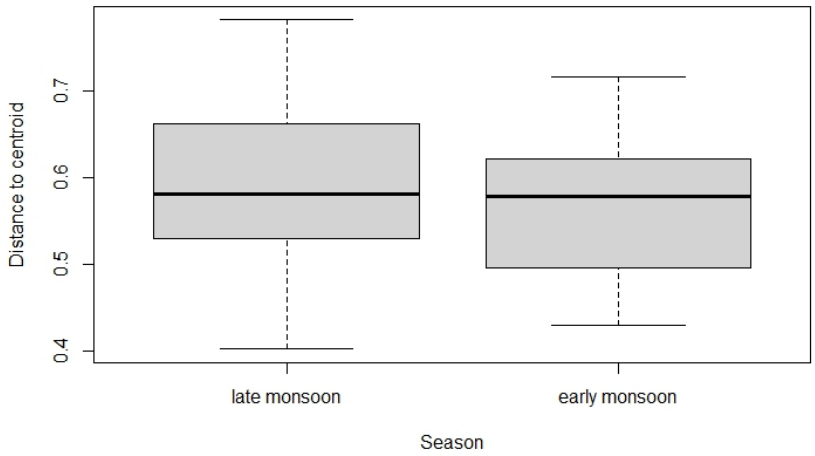


Figure 4.10 Boxplot comparing variance and mean distance to centroid for community composition data. The boxes represent the interquartile range of distances to centroid for each season. The thick black lines represent median distance to centroid for each season. The whiskers represent the 1.5xIQR lower and upper bound.

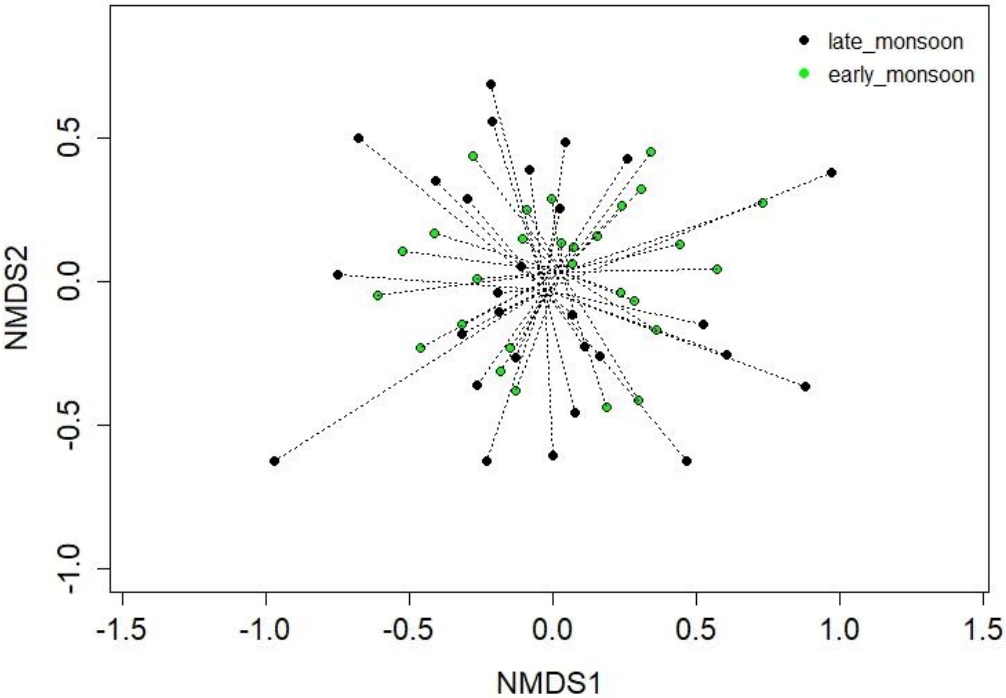


Figure 4.11 Non-Metric Dimensional Scaling (NMDS) comparing the centroid and dispersion of community composition for the two seasons. *Stress value = 0.25.*

4.4.2 Variation in total macroinvertebrate composition

Total beta-diversity for pond macroinvertebrate communities was 0.37 indicating moderate to high beta-diversity. The relative contribution of the turnover component to total beta diversity was much greater than that of nestedness (Table 4.12).

Table 4.12 Total beta diversity and contribution of beta diversity components, turnover and nestedness, to total beta diversity.

Total beta diversity	Turnover (%)	Nestedness (%)
0.37	0.315 (85)	0.054 (15)

Figure 4.12 presents the PCA ordination plot for macroinvertebrate community data. The first two axes explain 30.3% of variation in community composition (Figure B.2). Most of the ponds form a loose group and a few ponds are isolated. Generally, while the larger group (lower half of plot) includes the most species rich and diverse ponds (ponds 6, 8, 14) the isolated ponds are characterized by taxa poor ponds (ponds 3, 21, 25 and 29) whereas ponds 23, 26, 10 and 19 are characterized by high gastropod abundance or richness. This is also evident in the position of Physidae, *Melanoides* and *Filopaludina* (Gastropoda) species. The position of Palaemonidae suggests that this genus is more typical of taxa poor ponds.

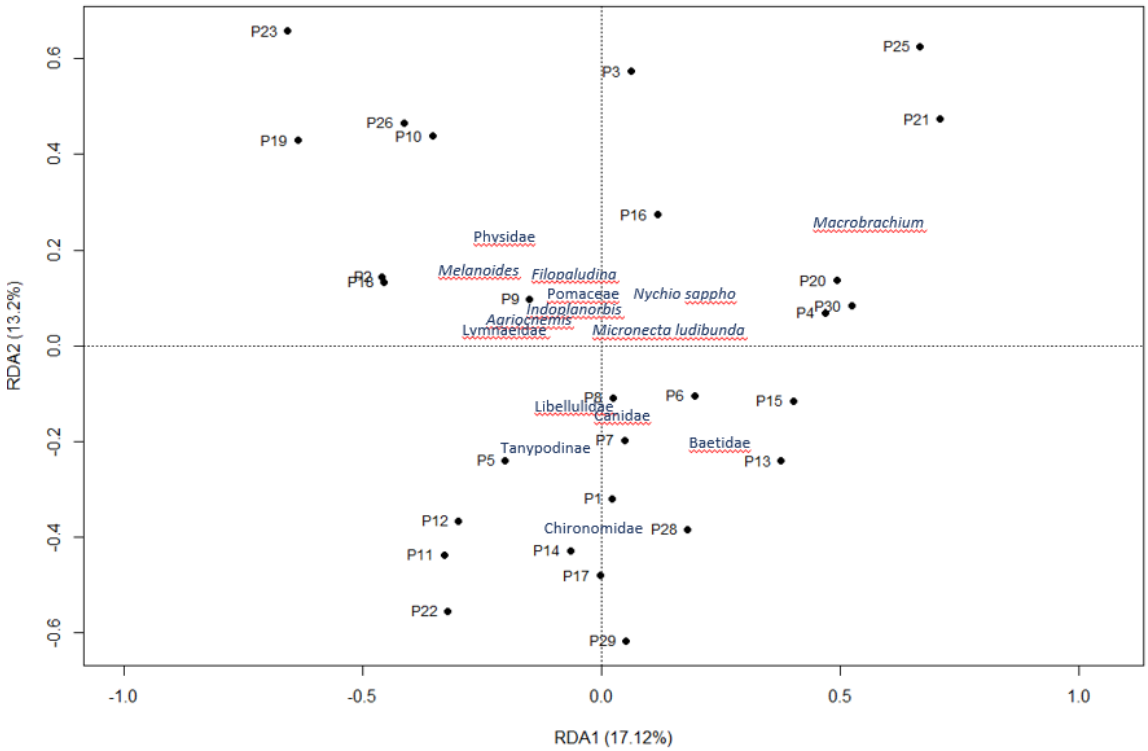


Figure 4.12 Transformation-based PCA of showing distribution of ponds based on community composition and key taxa. For clarity, only species with high scores (>0.1) are shown.

4.4.4 Differences in community composition between two pond clusters

PERMANOVA results show significant differences in community composition (distance index – Sorenson) between the two pond clusters (PERMANOVA: *Pseudo* $F_{1,26}$=1.72, p = 0.03). Figure 4.13 presents a two-dimensional visualization comparing the centroid and dispersion of community composition for the two clusters. Indicator species analysis identified Gastropoda: Physidae sp. as a potential indicator of ponds that belong in cluster 1 whereas indicators for cluster 2 comprised *Synaptonecta* sp. (p = 0.005), Diptera: Culicinae (p = 0.04), *Ceriagrion* sp. (p = 0.01), and *Elophila* sp (p = 0.016).

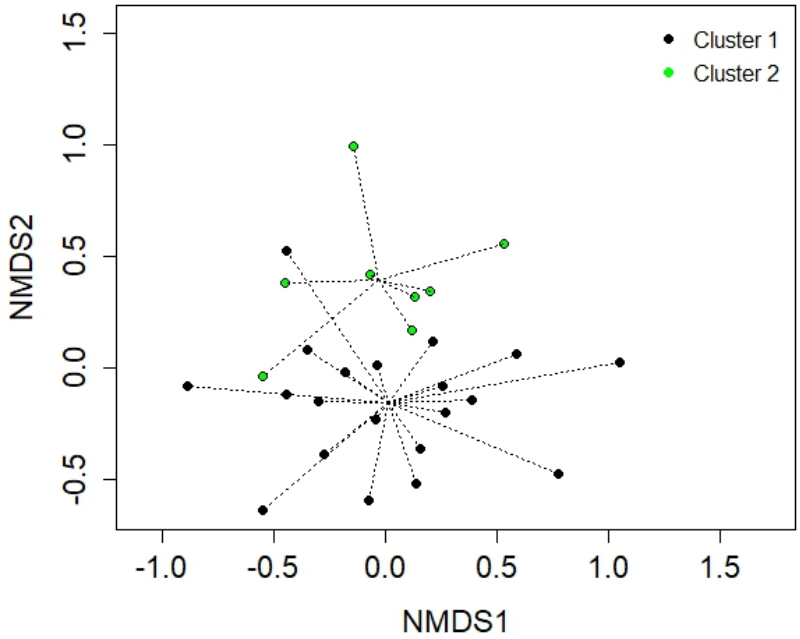


Figure 4.13 Non-Metric Dimensional Scaling (NMDS) comparing the centroid and dispersion of community composition for the two clusters. *Stress value = 0.22*.

4.5 Drivers of community variation

4.5.1 Relationship between community composition variation and environmental variables

There was no significant spatial autocorrelation (Figure 4.14) in community composition among the ponds.

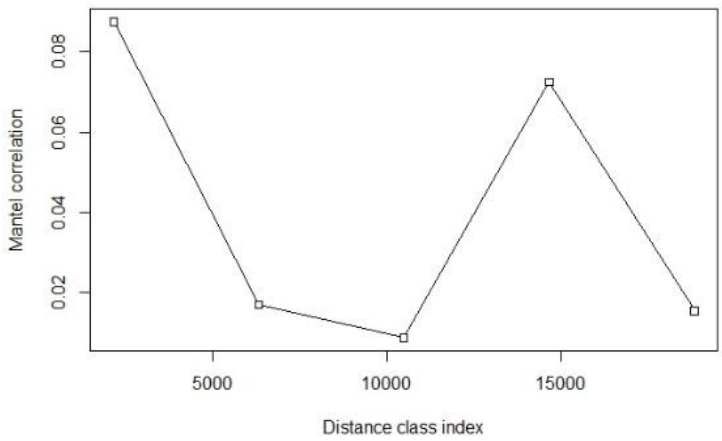


Figure 4.14 Mantel correlogram for macroinvertebrate composition along 5km distance intervals.

RDA with forward selection yielded a final model with aquatic vegetation cover and impervious cover (within 500m radius) as statistically significant predictors ($p<0.05$). The p value for the final model was 0.001 and the adjusted R^2 value was 0.07. The two RDA axes explained 13.9% of variation in community composition (RDA1 – 7.64%, RDA 2 - 6.29%). Both RDA axes and variables were significant ($p<0.05$). Figure 4.15 presents the RDA plot for the final model.

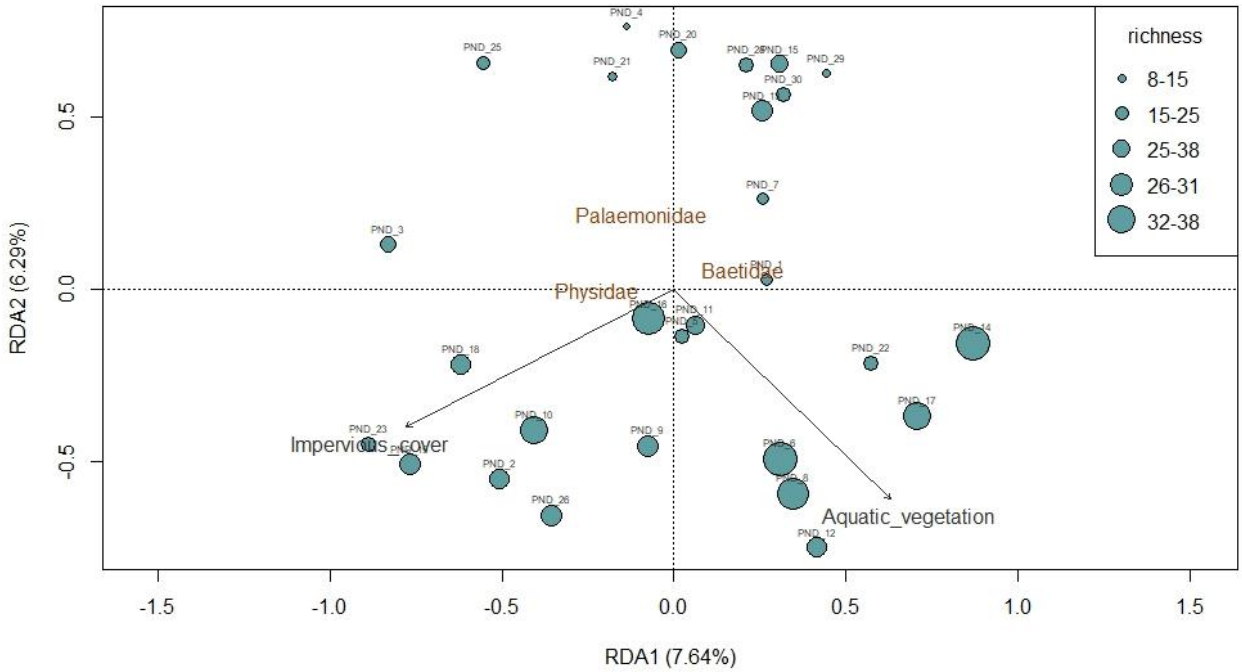


Figure 4.15 RDA plot showing distribution of pond communities with environmental constraints. Arrows represent the strength and direction of environmental variables. Circles represent ponds with size corresponding to taxonomic richness recorded for the pond. Key species names are presented, and for clarity, only species with high scores (>0.2) are shown.

The RDA plot shows the most diverse ponds (ponds 6,8,14, 17) were separated from the relatively species poor ponds along the first RDA axis. Ponds characterized by high gastropod abundance or diversity form a loose group along the impervious cover gradient and those with high taxa richness and diversity are associated with a gradient of aquatic vegetation cover. The ponds forming a cluster across the RDA 2 axes (top) include those with little to no aquatic vegetation cover. The species scores indicate taxa associated with each grouping. Most of the species cluster around the centre but some families and genera show distinct patterns. Palaemonidae is associated with larger ponds with little vegetation cover; these are also taxa-poor ponds (example ponds 4, 21, 25). The gastropod family Physidae is associated with ponds with greater impervious surrounding land cover.

4.5.2 Relationships between turnover and nestedness components of beta diversity and environmental variables

Constrained ordination RDA with forward selection method yielded a best model ($F_{1,26}=3.12$, $adj.R^2=0.07$, $p=0.001$) for the turnover component of beta diversity with proportion of impervious cover (within 500m radius) as the only significant predictor. The same method yielded a model for the nestedness component of beta diversity with extent of aquatic vegetation cover as the most important explanatory variable ($F_{1,26}=4.8978$, $adj.R^2=0.12$, $p=0.005$)

The findings support the hypothesis that there is high beta diversity driven by taxonomic turnover in response to both physical habitat characteristics and surrounding land cover. This shows that greater beta diversity across an urban landscape is supported by differences in the taxa present among ponds rather than differences in number of taxa and this agrees with findings from Hill et al. (2018) who found that turnover drives beta diversity within cities and across cities. Contrastingly, examining Odonata species only, Johansson et al. (2019) report richness difference drivers a greater proportion of beta diversity among ponds which suggests that there may be different patterns across taxonomic groups. Ponds being discrete waterbodies, distributed across the mosaic of urban land use, can be of different ages, subject to different uses and undergo disturbances (stochastic events) leading to environmental conditions that support different species establishing in them or colonizing them over time (Hill et al., 2018). The findings also show that beta diversity varies along a gradient of impervious or built-up land cover which can structure pond community either by presenting barriers to movement that limits the species that are able to establish in ponds or by affecting habitat conditions in ponds such as water quality (e.g. nutrient load). The significant relationship between beta diversity and aquatic vegetation agrees with research to date that emphasize the role of plant cover (which provides resources, shelter and substrate) in supporting multiple taxonomic groups in urban ponds (Hill et al., 2019; Hill et al., 2015; Johansson et al., 2019).

4.6 Local Contribution to Beta Diversity

4.6.1 LCBD, LCBD-turnover and LCBD- nestedness patterns

Figure 4.16 compares LCBD values for both seasons. There were no significant differences in LCBD values between the two seasons datasets. (Mann-Whitney U test: *W*= 360, p-value = 0.6). The mean LCBD value for pooled macroinvertebrate data was 0.035 (SD=0.004) and ranged from 0.029 to 0.044.

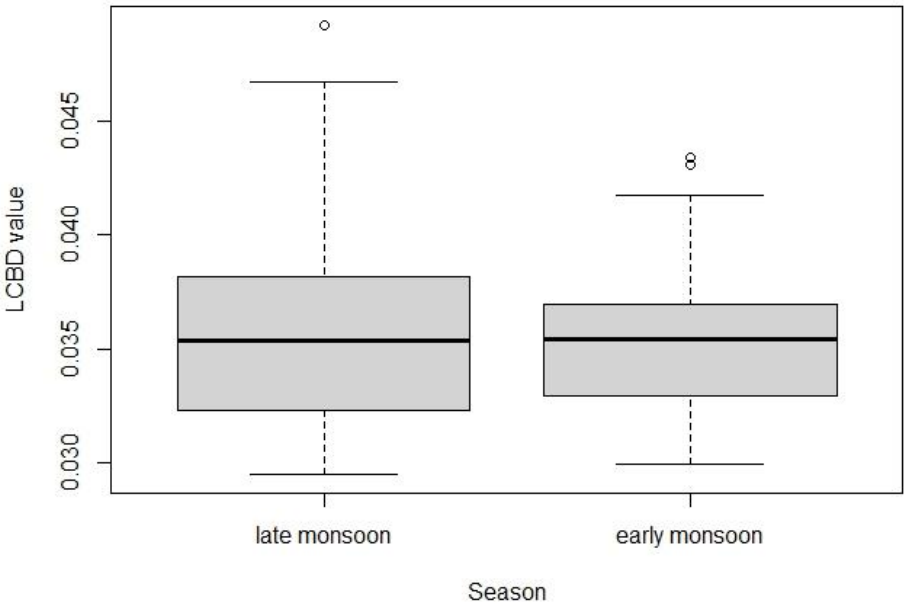


Figure 4.16 Boxplot comparing LCBD values for two seasons. The boxes represent the interquartile range of LCBD values for each season. The thick black lines represent median LCBD value for each season. The whiskers represent the 1.5xIQR lower and upper bound.

No significant spatial autocorrelation was evident for LCBD values (Figure 4.17).

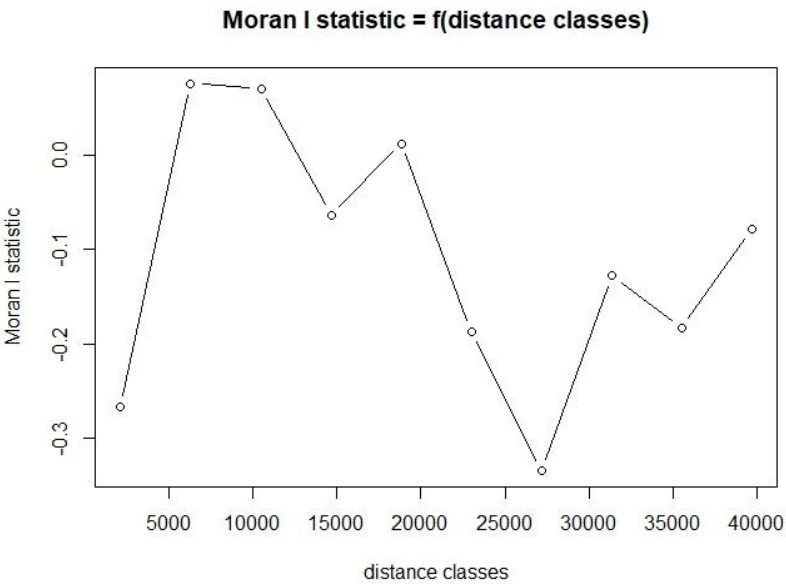


Figure 4.17 Moran's I correlogram for autocorrelation in LCBD values along 5km distance intervals.

Higher total LCBD values are associated with higher LCBD-turn (Spearman's r = 0.63, p <0.001). LCBD-nest and LCBD-turn show an inverse relationship (Spearman's r = -0.75, p<0.001). There was no strong pattern between LCBD and LCBD-nest (Spearman's r = -0.14, p>0.05) (Appendix B14).

There was a statistically significant relationship between LCBD-total and richness (F_{26} = 3.686, *adj.R* ^{2}= 0.16, *p* = 0.039) but not with Shannon's diversity index (F_{26} = 1.764, *adj.R* ^{2}= 0.02, *p* = 0.19). Figure 4.18 presents the relationships between LCBD and taxonomic richness.

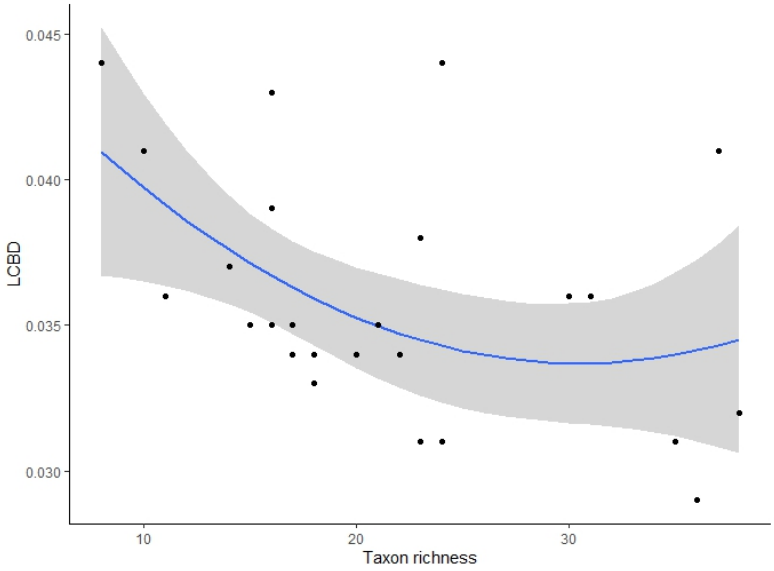


Figure 4.18 Relationship between LCBD values and richness. Shaded area represents 95% confidence interval.

Table 4.15 presents the summary of the relationship between taxonomic richness and each of LCBD-turnover and LCBD-nestedness values. Both components of LCBD were significantly associated with taxonomic richness. Figure 4.19 represents the quadratic relationship between taxonomic richness and the LCBD components.

Table 4.15 Summary of regression models (quadratic function) for LCBD-nestedness and LCBD-turnover with taxonomic richness.

Response variable	Statistical function	F statistic	df	P value	adj.R^2
LCBD-nest	Quadratic	5.475	26	0.02	0.14
LCBD-turn	Quadratic	7.923	25	0.002	0.33

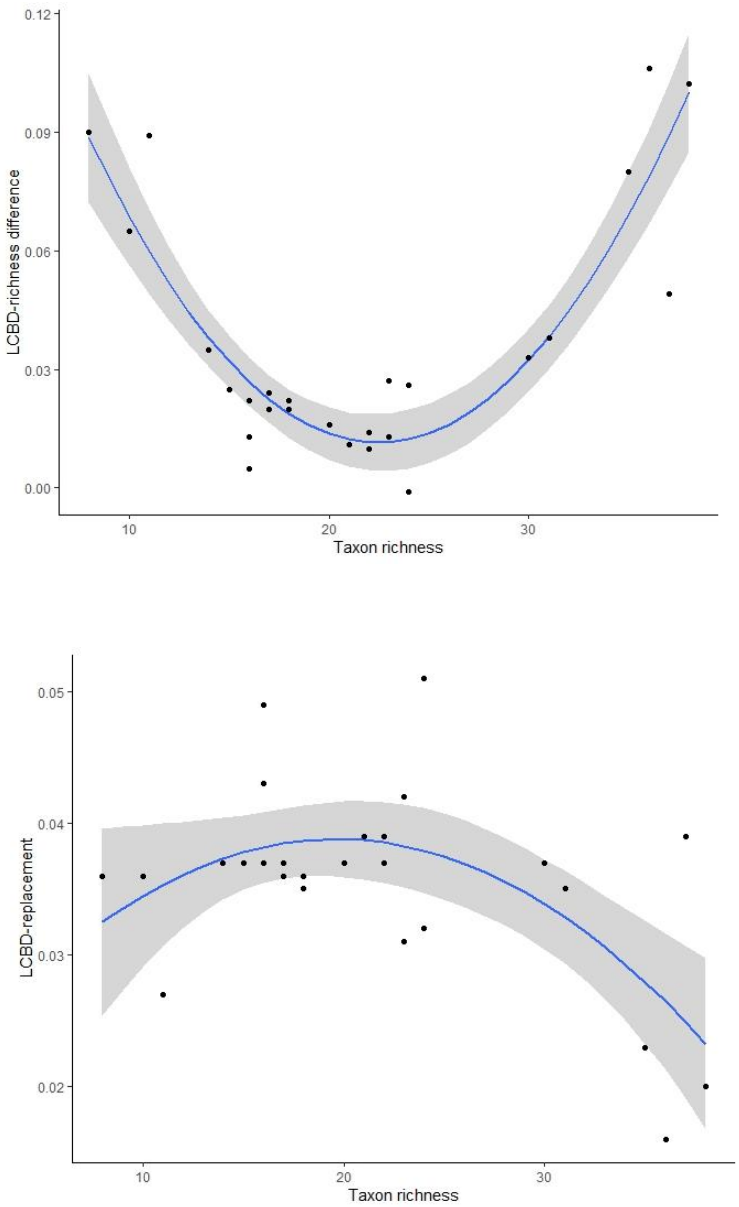


Figure 4.19 Relationship (quadratic function) between richness and (top) LCBD-nestedness vs (bottom) LCBD-replacement.

4.6.2 Relationship between LCBD values and environmental variables

LCBD did not show any statistically significant relationship with any of the environmental variables, modelled individually (Appendix B11). LCBD-nest exhibited a significant (F_{26}= *7.31, adj.R* ^{2}= *0.19, p = 0.01*) and positive relationship with shade over pond margin (Table 4.13) (Appendix B13). Figure 4.20 presents the relationship between LCBD-nest and pond shade. On the other hand, no statistically significant relationships were observed between LCBD-turn and the environmental variables, modelled individually (Appendix B11).

Table 4.13 Summary of linear model with LCBD-nestedness values as response variable.

	Estimate	Std. Error	t value	p value
Intercept	0.02	0.0077488	2.622	0.014
Shade	0.00048	0.0001784	2.704	0.011

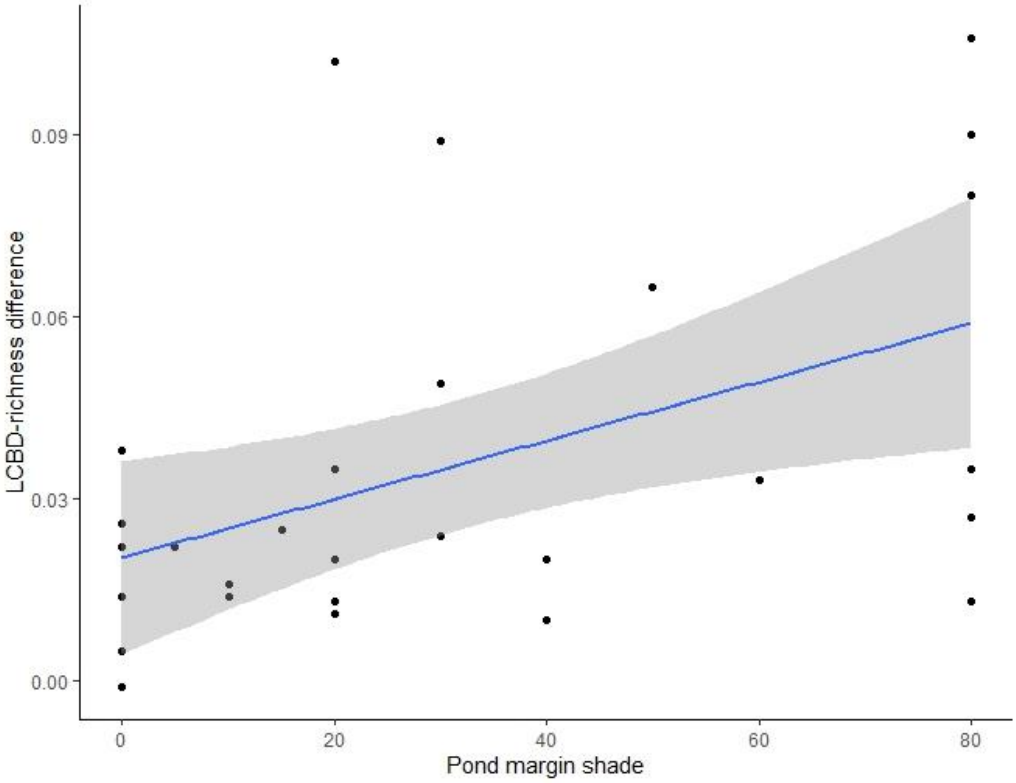


Figure 4.20. Relationship between LCBD-nestedness and pond margin shade. Shaded are represents 95% confidence intervals.

4.6.3 Relationship between richness, taxonomic LCBD and environmental LCBD.

Environmental LCBD values ranged from 0.01 to 0.07 (mean = 0.035, SD=0.017). However, there were no significant associations between environmental LCBD and any of the diversity (richness, Shannon's index) or total taxonomic LCBD (Appendix B15).

4.7 Summary

The present chapter characterized pond communities in terms of taxonomic diversity and community composition. The analyses here quantified alpha, beta and gamma diversity and examined how diversity patterns vary among ponds. They also identified environmental variables driving variation at multiple scales. Pond habitat characteristics such as extent of aquatic vegetation cover and TSS were the most important environmental variables influencing macroinvertebrate alpha diversity. Aquatic vegetation cover was also important in structuring macroinvertebrate community composition. The findings in this chapter also suggest that the proportion of impervious surrounding land cover may also be structuring macroinvertebrate communities at landscape scale. Interestingly, analyses following partitioning of the components of beta diversity reveals that extent of aquatic vegetation cover may be driving the

nestedness component of beta diversity and that LCBD values of ponds have a negative relationship with taxonomic richness.

The analyses presented in this chapter characterized pond communities and their responses to environmental drivers in terms of species or taxa presence and distribution. The following chapter describes pond communities in terms of functional or biological traits of macroinvertebrates. Descriptions of functional biodiversity and their relationship with environmental variables can provide information that explains patterns of community composition as it highlights specific biological traits or ecological preferences that are responding to environmental variation. Findings on the nature of the relationship between taxonomic richness and functional diversity for lentic waterbodies are inconclusive - there is evidence of independence or low correlation (Heino and Tolonen, 2017a) as well as findings where functional diversity was found to decrease with increasing taxonomic diversity, possibly due to similarities in biological traits or ecological preferences among different species (Hill et al., 2019). The next chapter explores functional diversity patterns among GKL ponds, describing its distribution patterns, how it relates to taxonomic diversity, and its relationship with pond characteristics and surrounding land use.

Chapter Five

Trait-based pond community patterns and their drivers

5. TRAIT-BASED POND COMMUNITY PATTERNS AND DRIVERS

This chapter presents the results of analyses characterizing ponds in terms of the traits of their macroinvertebrate communities. The analyses compare trait-based diversity (functional diversity) metrics with taxonomic diversity metrics, and examines the relationship between functional diversity and environmental/spatial variables. The variation in functional composition among ponds in the study area is also quantified and its response to environmental variables described.

5.1 Trait-based diversity patterns

5.1.1 Trait based richness and functional diversity

The total number of trait categories present, trait richness, in ponds ranged from 52 to 56 out of a possible maximum of 60. Mean trait richness was 54.5 (SD=1.14) with 46.4 % (n=13) ponds having a trait richness value of 55. Functional diversity values ranged from 30.24 – 65.11 with mean value 48.28 (SD=9.35) whereas Functional evenness values ranged from 0.29 to 0.75 with mean value 0.48 (SD=0.11).

5.1.2 Correlation among diversity metrics

The following describes the relationship between the taxonomic diversity metrics and functional diversity. It must be noted that the taxonomic diversity metrics, Shannon's diversity index and taxonomic richness, were recalculated for comparison here. The same taxa that were excluded from the traits analyses for lack of information were excluded from the taxonomic metrics calculation.

There was moderate correlation (Pearson's r: 0.69) between Shannon's diversity index (taxonomic) and functional diversity (Figure 5.1). Taxonomic richness and functional diversity showed a low to moderate correlation (Pearson's r: 0.30). The simple regression model with Shannon's diversity index as a predictor of functional diversity was statistically significant (R^2(adj) = 0.45, $F_{(1,26)}$ = 23.25, p <0.0001).

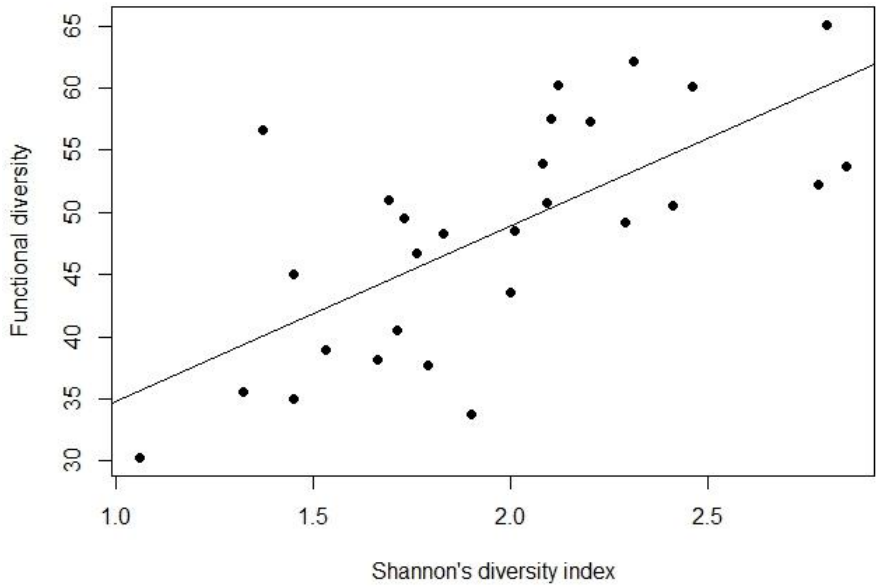


Figure 5.1 Relationship between functional diversity index and Shannon's diversity index (n=28).

5.1.3 Relationship with environmental variables

GAMs were fitted for the response variable functional diversity and the each of the environmental variables separately. One variable, total suspended solids (TSS), showed a significant negative relationship (*edf* = 1, *adj.*R^2 = 0.11, *p* = 0.04) and road density within 2km showed a non-linear relationship with p value <0.1 (*edf* = 2.1, *adj.*R^2 = 0.17, *p* = 0.09). These were integrated into a single synthesis model. Inspection of the synthesis GAM showed that both of the variables were significant (Table 5.1; Figure 5.2).

Table 5.1 Summary of final GAM for functional diversity.

Response variable	Predictors (p value)	edf	p-value	Model R-sq. (adj)	Deviance explained
Functional diversity	TSS	1.00	0.008	0.37	44.8%
	Road density (within 2km)	2.3	0.024		

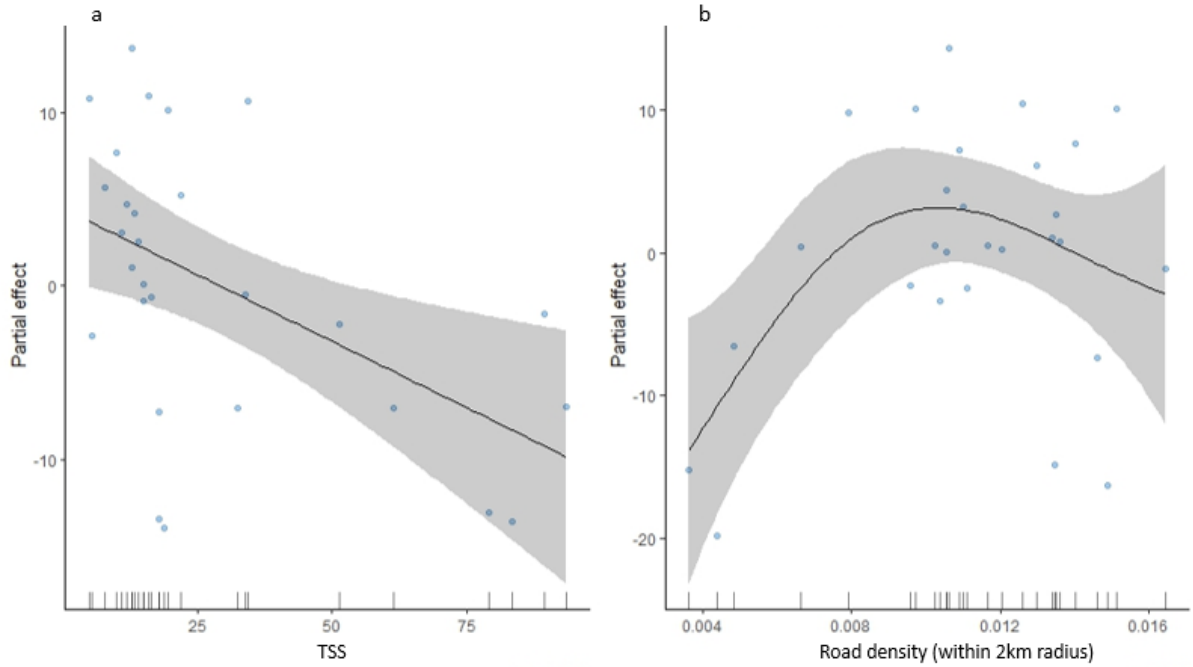


Figure 5.2 Partial effect plots showing the relationship between functional diversity and (a) suspended solids concentration and (b) road density within a 2km radius. Blue circles represent partial residuals. The shaded area represents 95% confidence intervals.

5.1.4 Differences in functional diversity between two pond clusters

Comparing the two pond clusters, cluster 2 showed slightly higher functional diversity (mean = 51.24, SD = 9.51) than cluster 1 (mean = 47.09, SD = 9.27). However, the t-test did not show statistically significant differences in functional diversity between the two clusters ($t_{12.6}$ = -1.05, p = 0.3).

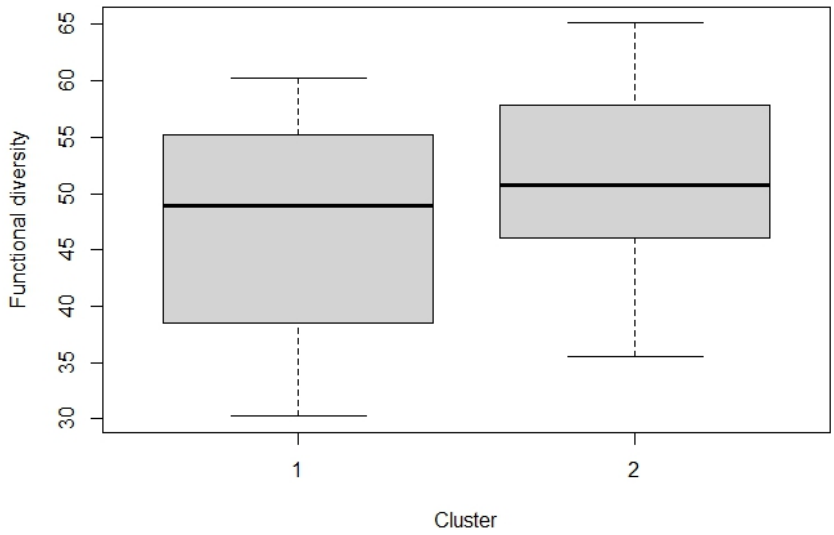


Figure 5.3 Boxplots comparing functional diversity for the two pond clusters. The boxes represent the interquartile range of distances to centroid for each season. The thick black lines represent median distance to centroid for each cluster. The whiskers represent the 1.5xIQR lower and upper bound.

The results support the hypothesis that functional diversity increases with taxonomic diversity in GKL ponds and is influenced by both physical habitat features and surrounding land use. Higher TSS values

are associated with poorer water quality and this could lead to ponds that only support taxa capable of tolerating higher TSS conditions. Variables associated with water quality have been associated with functional diversity in studies of lakes macroinvertebrates but with different effects. Heino and Tolonen (2017) found that functional diversity exhibited a positive relationship with conductivity and phosphorus levels. They suggest that the high productivity in the water supported different kinds of resources, increasing the types of food and feeding mechanism-associated traits and therefore, functional diversity in habitats. However, the generalizability of the relationship between taxonomic and functional diversity, especially among urban pond habitats, remains unclear as conflicting findings, i.e., a negative relationship between taxonomic and functional diversity was reported by Hill et al. (2019).

5.2 Trait distribution patterns and main drivers

5.2.1 Trait-based community variation

Total functional beta diversity based on trait composition of pond communities was 0.79 indicating high beta-diversity. Richness difference explained 61% of total functional beta diversity (Table 5.2). Note that total functional beta diversity was partitioned into replacement and richness difference components, where total functional beta diversity = beta diversity explained by replacement + beta diversity explained by richness difference. While replacement (turnover) reflects differences in traits present among communities, richness difference reflects differences in numbers of traits among communities.

Table 5.2 Total functional beta diversity and contribution of beta diversity components, turnover (replacement) and richness difference.

Total functional beta diversity	Replacment (turnover) (%)	Richness difference (%)
0.79	0.31 (39)	0.48 (61)

5.2.2 Drivers of trait-based community variation

The full RDA model for functional composition (total functional beta diversity) did not explain variation well and was not statistically significant (F_{11}=1.04, $adj.R^2$=0.01, p=0.3) although there were site clusters associated with increasing conductivity, aquatic vegetation cover and TSS gradients (Appendix C2, C3). RDA with forward selection did not identify any of the environmental variables as a statistically significant predictor of total functional beta diversity.

Similarly, the RDA model for the turnover component of functional based beta diversity was not statistically significant and none of the environmental variables were statistically significant predictors (Appendix C4). RDA of the richness difference component of trait-based beta diversity yielded a model (Figure 5.4) where the first two RDA axes explained 39.5% of total variation. However, the full model was not statistically significant (F_{11}=1.3, p=0.18) Forward selection yielded a model with only total suspended solids as a predictor variable ($F_{1,26}=3.73$, $adj.R^2=0.09$, $p=0.019$).

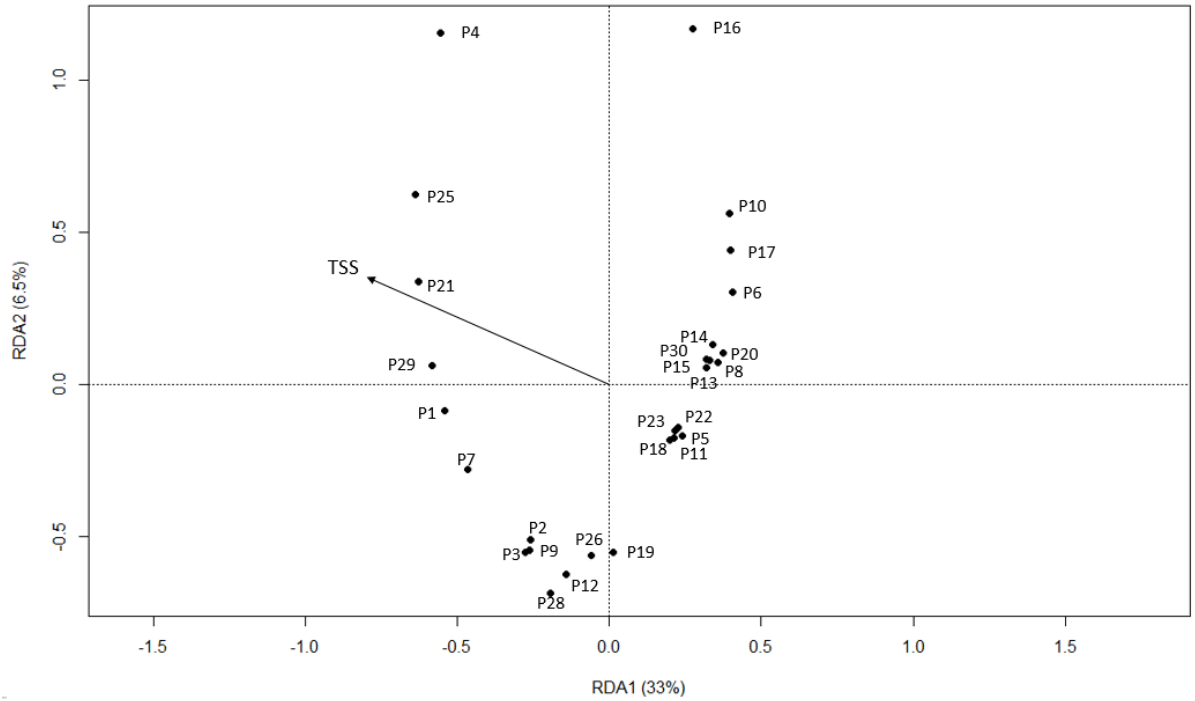


Figure 5.4 RDA plot showing variation in richness difference component of trait-based beta diversity with statistically significant environmental variable shown. Circles represent pond communities and arrow represents strength and direction of the constraining variable.

The findings support the hypothesis that functionally diversity is primarily driven by richness difference and partially support the hypothesis that functional beta diversity is influenced by both physical habitat characteristics and surrounding land use. They do not provide evidence for the role of surrounding land use but suggest that variations in TSS levels could be structuring trait distribution across ponds by filtering out taxa based on traits associated with respiration or feeding – suspended solids can clog feeding or breathing structure (Greenway, 2017). Hill et al. (2019) identify multiple other physical habitat characteristics (aquatic vegetation, surface area, pH, conductivity, shade) as key variables influencing functional beta diversity among ponds but no significant relationship with these variables was found for ponds in GKL. A potential explanation is differences in the types of ponds included in this study, which was limited to urban pond, while Hill et al. (2019) examined functional beta diversity across both urban and non-urban landscapes which may influence the relative importance of environmental

variables. Nevertheless, both findings highlight the relatively greater role of individual pond conditions in influencing functional beta diversity.

5.3 Local contribution to functional beta diversity (f-LCBD)

Functional LCBD (f-LCBD) values for ponds ranged from 0.03 to 0.04 (mean f-LCBD = 0.036; SD = 0.002). There was a negative and weak correlation between f-LCBD and functional diversity (Spearman's r = -0.34). The regression model for f-LCBD with functional diversity as predictor was not statistically significant ($adj.R^2$= 0.05, $F_{(1,26)}$ = 2.465, p = 0.1). Similarly, no strong pattern was detected between f-LCBD and environmental-LCBD (Spearman's r = -0.35). The regression model for f-LCBD with environmental-LCBD as predictor was not statistically significant ($adj.R^2$= 0.07, $F_{1,26}$ = 3.288, p = 0.08).

Finally, none of the environmental variables recorded were statistically significant predictors of f-LCBD, although TSS yielded $adj.R^2$ = 0.08 (p = 0.07) (Appendix C5).

5.3 Summary

In the present chapter, urban pond biodiversity was quantified based on the biological traits of macroinvertebrates (functional diversity). Road density within 2km and TSS were identified as important environmental variables influencing functional diversity. Increasing values of TSS had a negative relationship with functional diversity and were also identified as statistically significant drivers of community variation. Specifically, TSS was identified as a driver of the richness difference component of functional beta-diversity. Furthermore, decomposition of functional community variation revealed that richness difference explained a greater proportion of functional beta diversity than did turnover. Analysis of functional LCBD indicated that none of the ponds were important contributors to functional beta-diversity nor is functional LCBD strongly associated with any of the recorded environmental variables.

The present chapter examined variation in functional traits among pond communities. A key ecological trait or process of macroinvertebrates is dispersal or movement from one community or suitable habitat to another and this is known to affect community dynamics across a given landscape. While methods of quantifying connectivity vary, findings to date suggest pond density and distances to other ponds may influence species richness and community composition (Gledhill, 2008; Liao et al., 2020) as well as functional alpha diversity (Hill 2019) in urban landscapes. The following chapter quantifies extent of connectivity among ponds in GKL based on available land use data and a pond inventory map. It also examines the extent to which connectivity drives diversity and community variation among ponds.

Chapter Six

Ecological connectivity for pond macroinvertebrates in GKL

6. ECOLOGICAL CONNECTIVITY FOR POND MACROINVERTEBRATES IN GKL

This chapter presents the results of analyses characterizing structural and functional connectivity for ponds in Greater Kuala Lumpur. First, connectivity across the entire landscape (i.e. all ponds, rather than only the 28 detailed in previous chapters) is described by means of graph network analysis, and potential connectivity for macroinvertebrates is quantified. Then, for each of the 28 sample ponds, seven connectivity variables are derived based on pond distribution patterns, inter-pond distances and the nature of the urban matrix. The relationships between connectivity and diversity measures are then examined to understand the relative importance of ecological connectivity for community composition and diversity patterns.

6.1 Description of the GKL pondscape

There were altogether 777 ponds, defined as lentic waterbodies with a surface area less than 2ha, distributed across Greater Kuala Lumpur (Section 3.1.2). The total surface area of ponds was 5,844,471.88 m^2 and the mean pond area was 7521.84 m^2. The average number of ponds per km^2 was 0.3, ranging from 0.1 to 0.52 per km^2. Shah Alam had the highest pond density (0.52 per km^2) and Klang the lowest (0.1 per km^2).

Excluding Kuala Lumpur, Petaling district (including Subang, Shah Alam and Petaling Jaya municipalities) had the highest proportions of impervious land cover as well the highest pond densities (0.52-0.32 per km^2). Putrajaya was the exception, with a lower proportion of impervious land cover (34%) but higher pond density (0.34 per km^2). (Appendix D1). Figure 6.1 shows the distribution of ponds across GKL. Areas with high pond density (>5 ponds per km^2) are present in eastern Kuala Lumpur, Petaling district (Shah Alam and Petaling Jaya) and Sepang–Putrajaya border, and these tend to be golf course ponds or wetland botanic gardens.

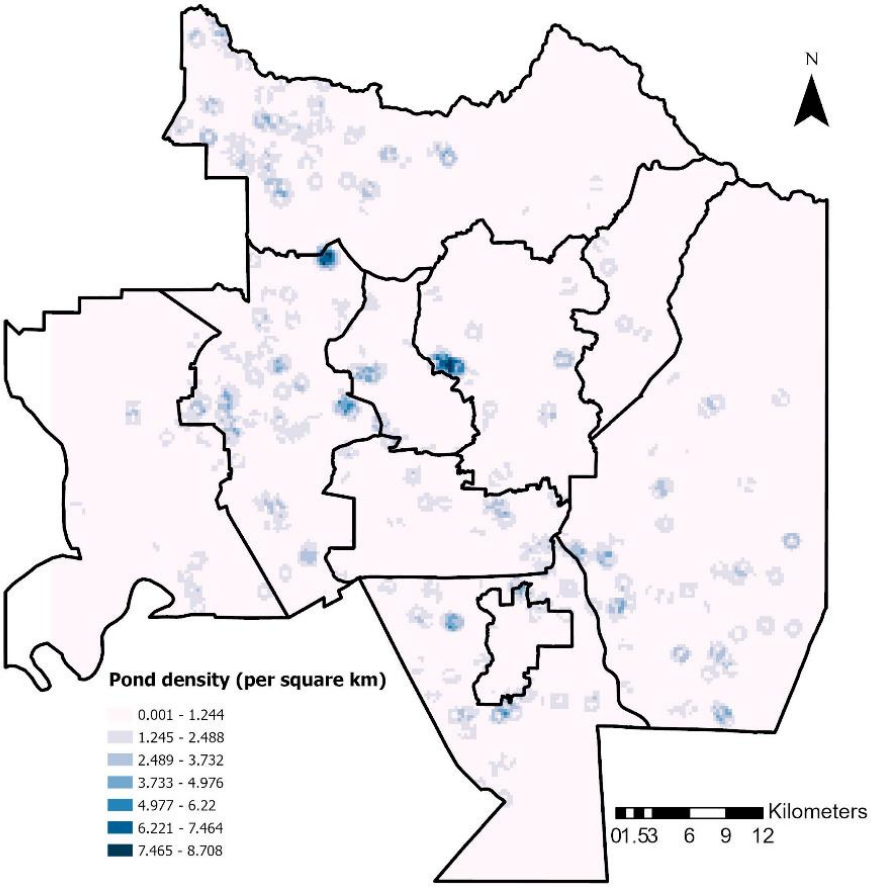


Figure 6.1 Distribution and density of ponds per 1 km^2 across Greater Kuala Lumpur.

6.2 Landscape scale connectivity

Landscape connectivity metrics, based on the arrangement of ponds and inter-pond Euclidean distances, were assessed for multiple maximum dispersal distance thresholds (Table 6.1, APPENDIX D2). The highest IIC value was at 2500m threshold, increasing by 120% from the value at 2000m. The percentage increase in number of links is greatest between the 500m and 1000m (102%) and least between the 2000m and 2500m thresholds (38%). The number of components, which represents an isolated pond (or an isolated group of ponds), decreased by an average of 44% at every 500m increment in distance threshold. At 500m threshold, there are 426 small components showing isolated ponds throughout the region (Figure 6.2a) but this decreases to 42 at 2500m thresholds where there are also large components encompassing larger areas of connected ponds. At 2500m, ponds in Selayang, Shah Alam and Petaling Jaya and western KL in the north-west are well connected (Figure 6.2c). Similarly, Subang, Putrajaya and Kajang (Hulu Langat) show high structural connectivity. At all thresholds, Klang district, most of KL and northern Hulu Langat show greater more isolated ponds or small, isolated pond clusters (Figure 6.2a-c).

Table 6.1 Changes in connectivity levels for ponds across distance thresholds.

Landscape scale metric	500m	1000m	1500m	2000m	2500m
IIC	2.50E-09	4.10E-09	1.04E-08	1.80E-08	3.96E-08
No. of components	426	249	145	76	42
No. of links	762	1538	2368	3398	4701

The level of connectivity for individual ponds in the network was quantified with Between Centrality (Figures 6.3 a - c) and delta-IIC (Figures 6.4 a - c) at each of the maximum dispersal distance thresholds (Appendix D3). Ponds that exhibited high values across both measures, and across distance thresholds, were primarily located within areas of high pond density, namely Selayang, Shah Alam and the borders of Sepang and Putrajaya (Table 6.2). They included ponds ranging in size from 1,772 m^2 to 19,075.3m^2.

Table 6.2 Most important ponds for pond network based on high values for both BC and delta-IIC measures.

Threshold	Pond ID	Area (m^2)	Municipality	BC	Delta-IIC
500m	50	18,241.37	Selayang	1.34E+08	0.015
	35	17,615.95	Selayang	1.54E+08	0.016
	279	9,485.51	Kuala Lumpur	2.72E+08	0.009
	170	11,882.95	Shah Alam	1.24E+08	0.009
	105	13,238.02	Selayang	1.33E+08	0.010
	792	8,443.15	Sepang	2.15E+08	0.012
1000m	64	6,358.4	Selayang	3.23E+08	0.011
	364	12,508.37	Shah Alam	6.10E+08	0.011
	405	12,404.14	Shah Alam	5.56E+08	0.011
	425	12,821.05	Shah Alam	6.04E+08	0.014
	670	15,843.94	Putrajaya	4.18E+08	0.016
	689	13,342.26	Putrajaya	3.81E+08	0.013
	777	13,759.21	Sepang	4.58E+08	0.011
1500m	90	3,960.985	Selayang	1.63E+09	0.056
	338	10,423.6	Shah Alam	1.52E+09	0.092
	395	4,794.87	Shah Alam	2.43E+09	0.089
	402	3,231.3	Shah Alam	2.51E+09	0.088
	587	12,091.43	Subang	2.78E+09	0.052
2000m	23	3,231.3	Selayang	3.17E+09	0.043
	51	2,397.4	Selayang	2.82E+09	0.030
	54	16,990.5	Selayang	4.09E+09	0.044
	286	12,925.32	Shah Alam	5.22E+09	0.012
2500m	209	3,022.85	Shah Alam	4.91E+09	0.017
	399	19,075.3	Shah Alam	5.02E+09	0.011
	526	3,127.1	Subang	6.28E+09	0.031
	551	1,772.02	Subang	6.52E+09	0.031
	621	4,690.6	Sepang	4.42E+09	0.019

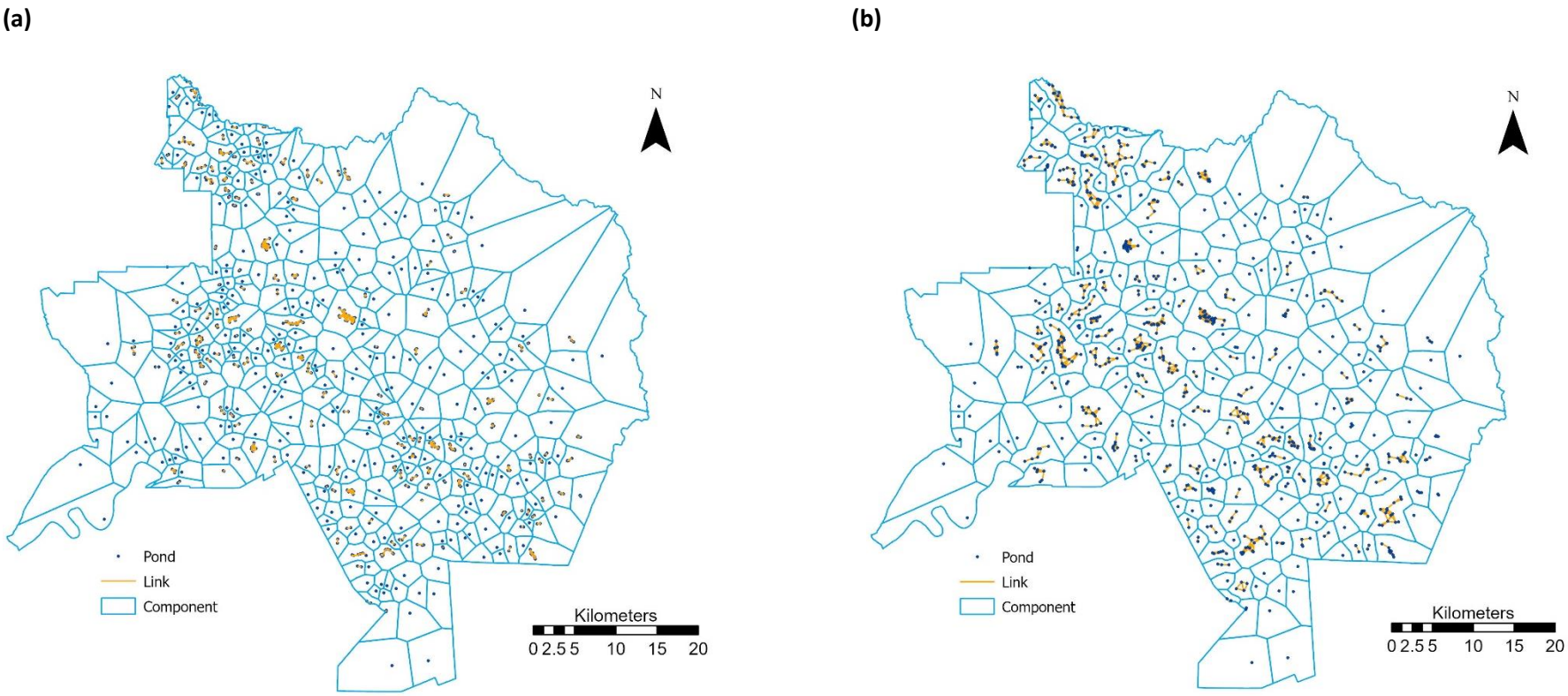
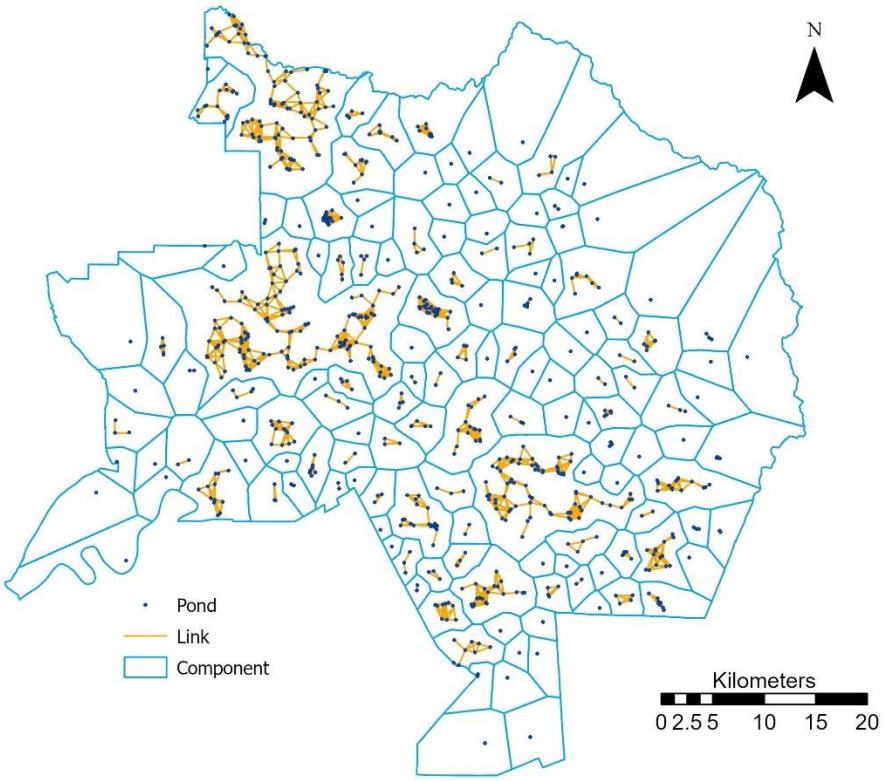


Figure 6.2a Pond habitats, components and linkages between ponds at 500m (left) and 1000m (right). Connectivity metrics based on arrangement and Euclidean distances between ponds.

(c)



(d)

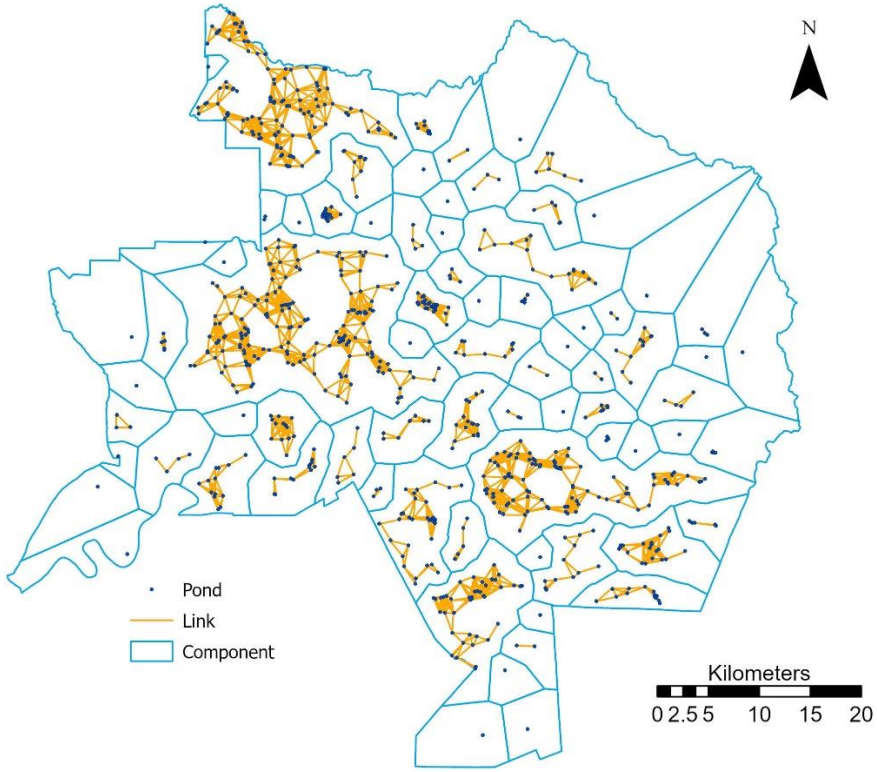


Figure 6.2b Pond habitats, components and linkages between ponds at 1500m (left) and 2000m (right). Connectivity metrics based on arrangement and Euclidean distances between ponds.

(e)

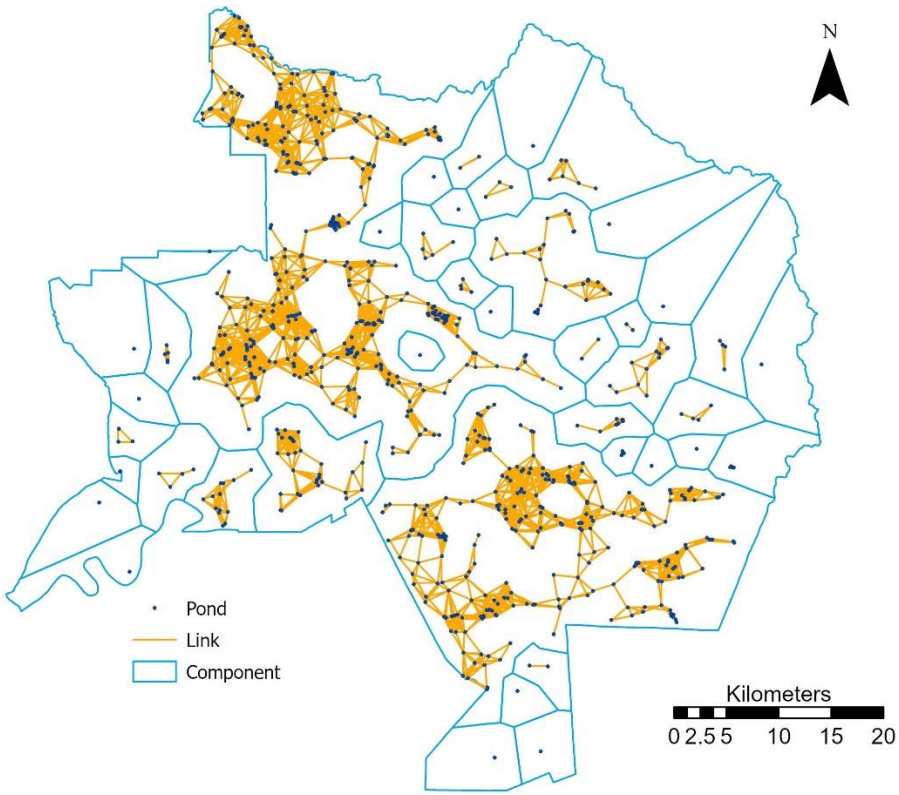
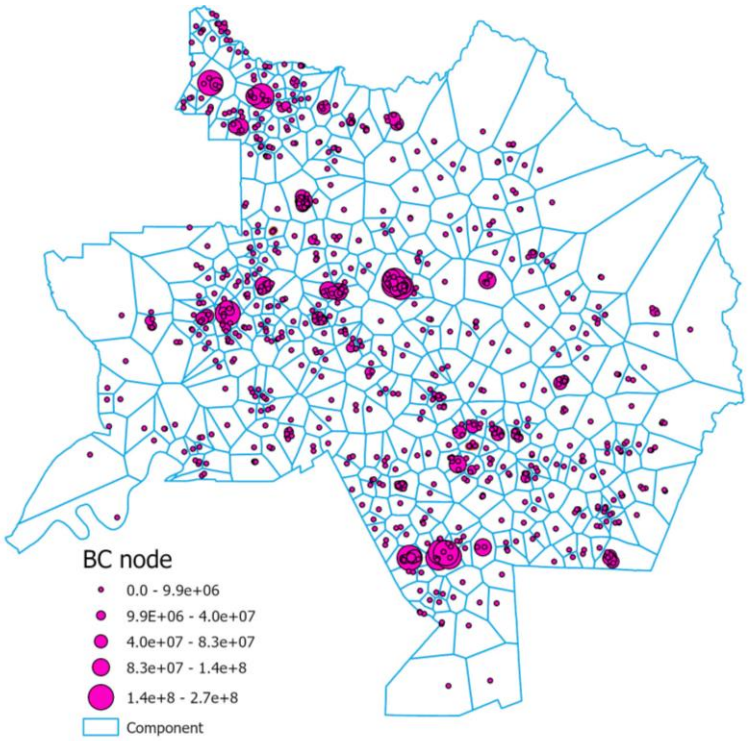


Figure 6.2c Pond habitats, components and linkages between ponds at 2500m. Connectivity metrics based on arrangement and Euclidean distances between ponds.

(a)



(b)

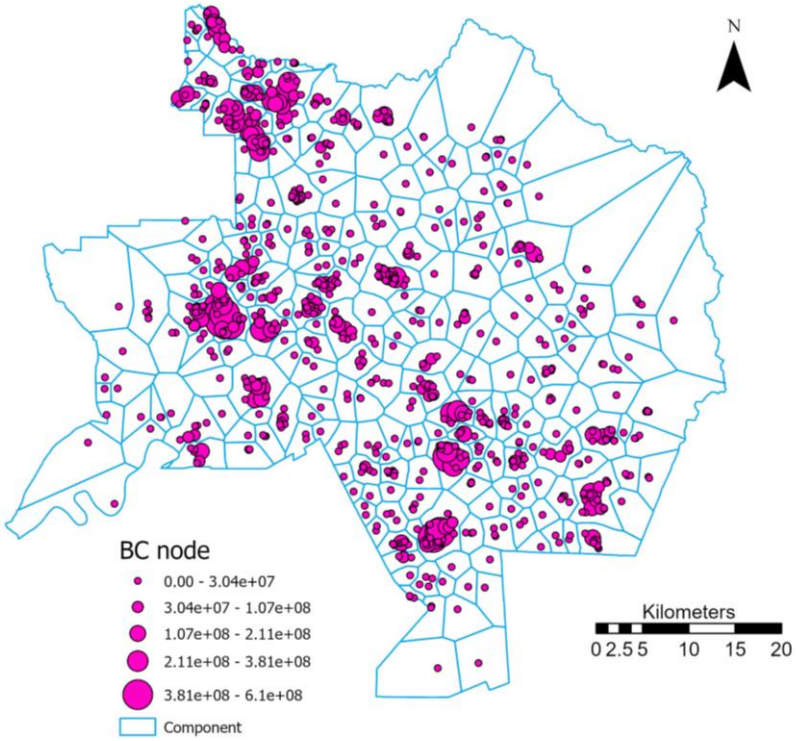
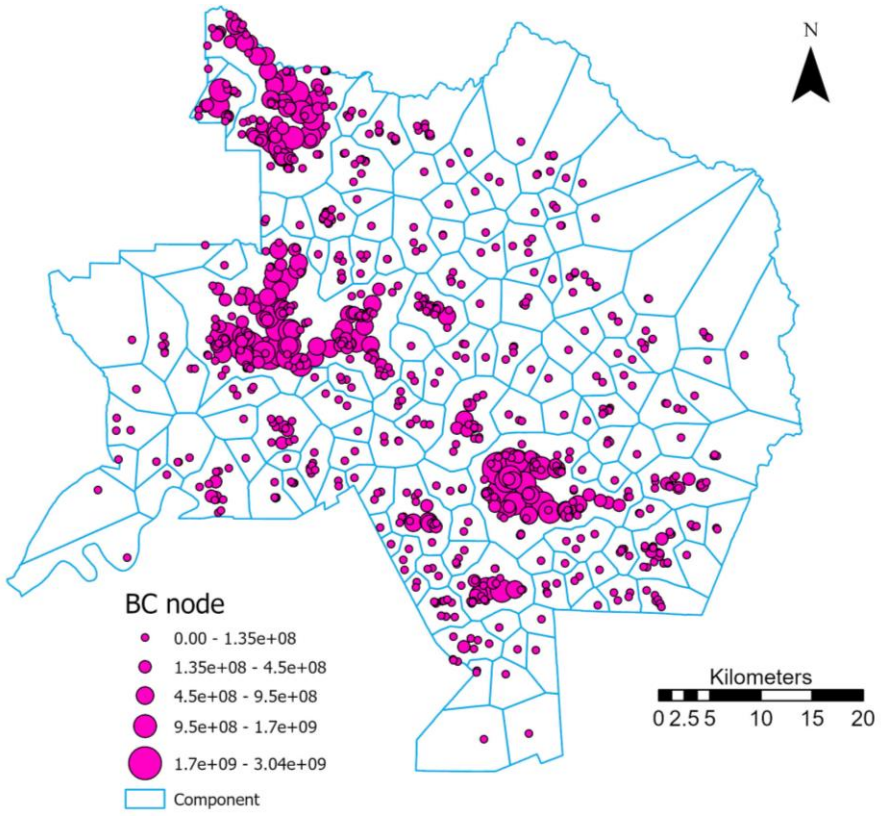


Figure 6.3a BC values for each pond and important linkages in the network for distance thresholds of 500m (left) and 1000m (right).

(c)



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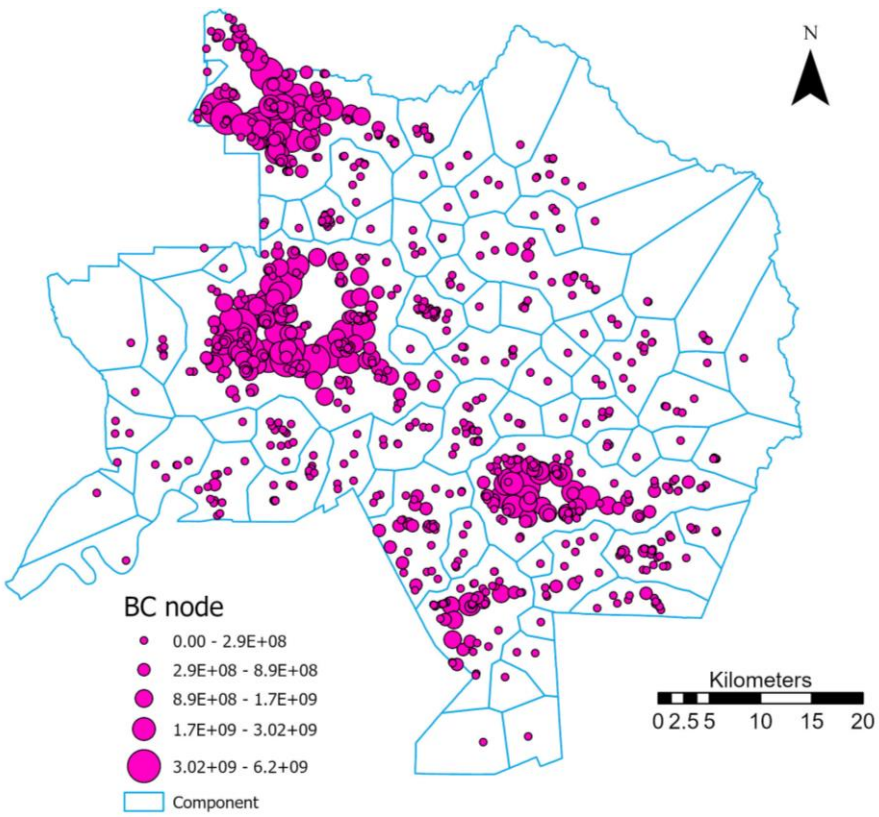


Figure 6.3b BC values for each pond and important linkages in the network for distance thresholds of 1500m (left) and 2000m (right).

(e)

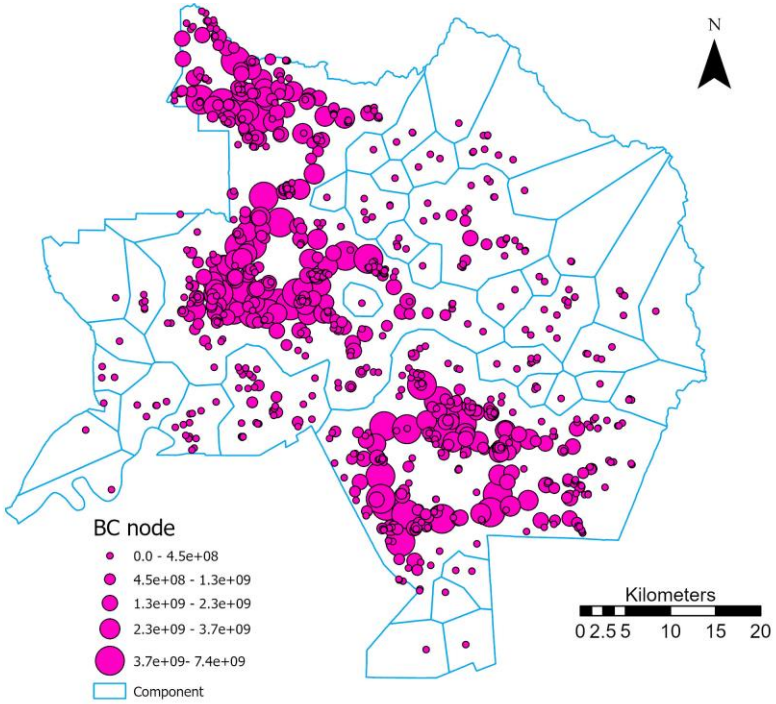
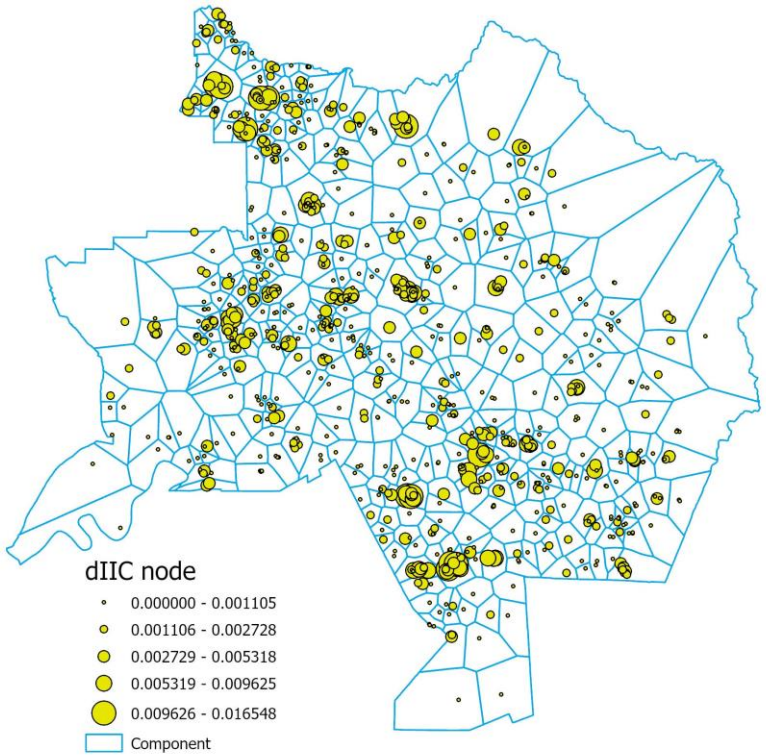


Figure 6.3c BC values for each pond and important linkages in the network for distance threshold of 2500m.

(a)



(b)

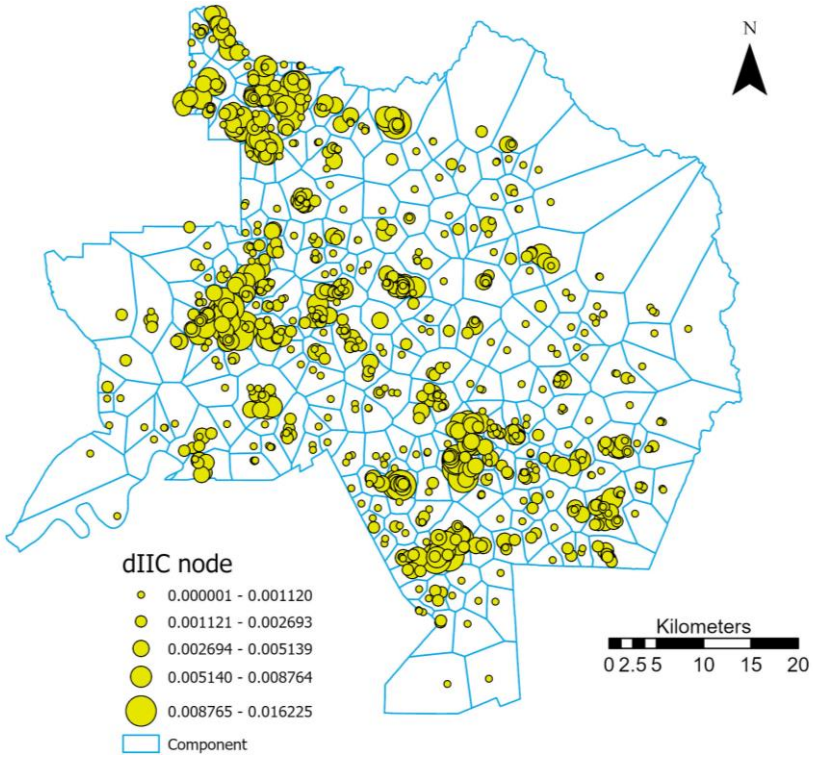
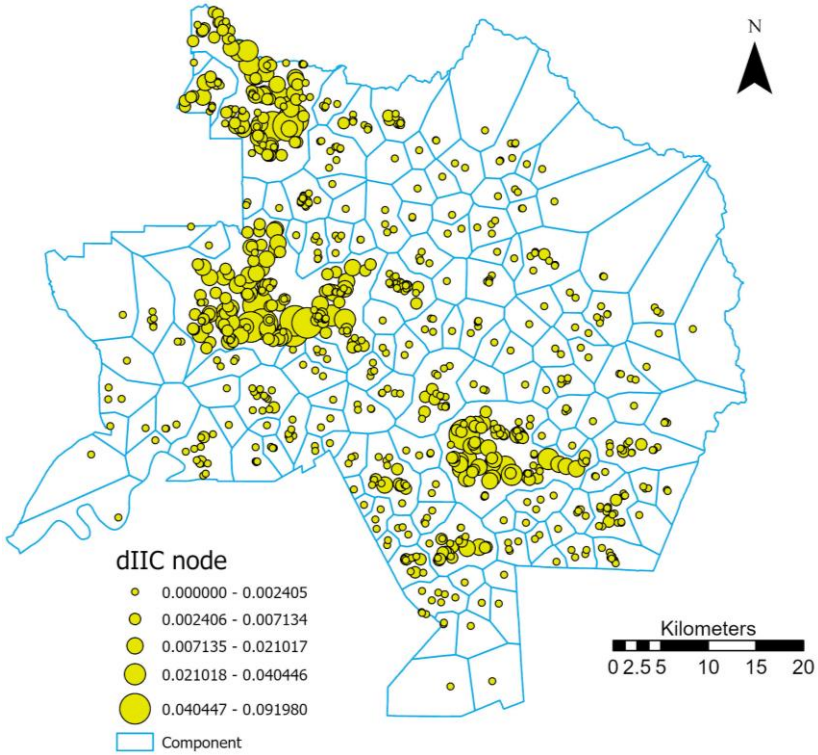


Figure 6.4a delta IIC values for each pond and important linkages in the network for distance thresholds of 500m (left) and 1000m (right)

(c)



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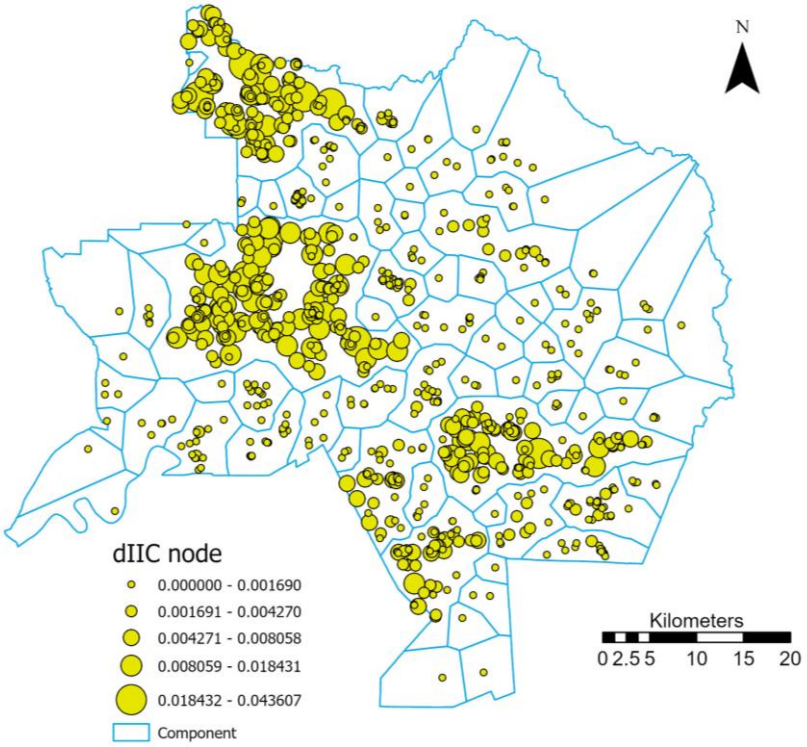

Figure 6.4b delta IIC values for each pond and important linkages in the network for distance thresholds of 1500m (left) and 2000m (right)

(e)

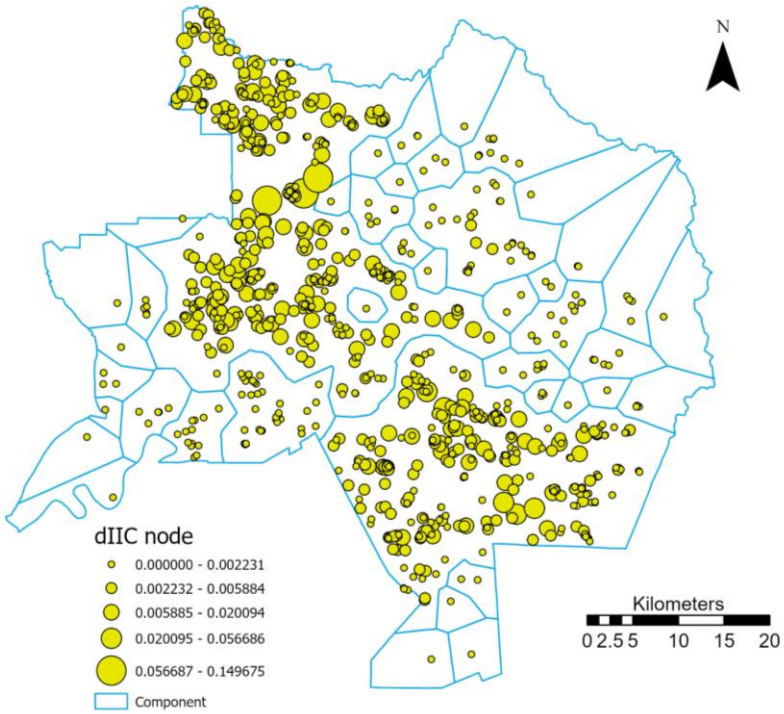

Figure 6.4c delta IIC values for each pond and important linkages in the network for distance thresholds of 2500m.

6.3 Relationships between connectivity variables and macroinvertebrate diversity

6.3.1 Connectivity variables

Using the geospatial dataset that comprised all of the mapped ponds in GKL, 34 connectivity metrics were calculated for the 28 ponds for which ecological data was collected. As such, ponds that were not sampled for ecological data but were within the distance thresholds set were included in count and distance measures. In addition, the BC and delta-IIC values (at 5 distance thresholds) for the 28 ponds were extracted from the landscape-scale pond analysis above. Among the sampled ponds, there was little to no variation in BC values below the 2000m threshold, thus only BC2500 was retained for further analysis. On the other hand, the delta-IIC for the 28 ponds did not show much variation beyond 1500m threshold, and the 500 m and 1000 m thresholds showed high correlation (Spearman's r=0.89) among the values for different thresholds and thus only IIC500 was retained.

The number of ponds surrounding each sample pond up to a 2.5km radius ranged from 0 to 24, with the average number of ponds only exceeding 1 beyond the 500m radius (Table 6.3). The average number of waterbodies larger than ponds (lakes, reservoirs) was also less than 1 within a 500m radius. The mean distance to nearest pond was 655.6m while the maximum distance to nearest pond recorded for a sample pond was 2676.2m (Table 6.3). For all sample ponds, the mean distance to nearest 5 ponds was 1333.3m, ranging from 333.7m to 3243.7m.

Spearman's correlation analysis showed high correlation ($r > 0.7$) among area-weighted variables. Euclidean and cost distance variables also showed high correlation with Spearman r values ranging from 0.83 to 0.93 for corresponding pairs of variables (e.g. EuD_N — CD_N), except for average distances within 2km radius ($r = 0.56$) (Appendix D4). The number of ponds within 2500m showed a negative correlation with Euclidean distance to nearest 15 ponds and thus the latter was removed from subsequent analysis. Similarly, number of ponds and other lentic waterbodies (PD and WD variables) were highly correlated as well. Consequently, the following variables were retained for further analysis: PD_2500m, ED_N1, EuDA_N and CD_2km (Appendix D5).

Table 6.3 Summary of retained connectivity variables.

Variable	Minimum	Maximum	Mean	SD	Median
Number of ponds within 2500m **(PD_2500m)**	0	24	9.7	7.1	7.5
Euclidean distance to nearest pond, m **(EuD_N)**	8	2676.2	655.6	711.3	449.6
Euclidean distance to nearest pond-area weighted **(EuDA_N)**	0.100	1134.4	66.48	213.09	11.4
Cost distance to ponds within 2km (CD_2km)	1121	48,623	12,258	9006.5	11,312
Betweeness Centrality at 2500 threshold **(BC2500)**	0	2.7E+09	3.94E+08	7.05E+08	2.48E+07
delta-IIC, Integral Index of Connectivity, at 500m threshold **(IIC500)**	0	0.00453	0.001318	0.001417	0.000711

6.3.2 Response of alpha diversity to pond connectivity

A linear model fitted for the response variable Shannon's diversity index was statistically non-significant ($F=1.6_{6,20}$; $adj.R^2$=0.12; p=0.19), only the variable PD_2500 had p value <0.05 (Table 6.4). A separate model with only PD_2500 as a predictor yielded a statistically significant, positive relationship ($F=5.74_{1,26}$; $adj.R^2$=0.15; p=0.02) (Figure 6.5a). The full model for taxonomic richness showed a similar statistically significant relationship for PD_2500. The connectivity models did not perform well for functional diversity ($F_{6,20}$=1.336, $adj.R^2$=-0.07, p=0.286), taxonomic LCBD ($F_{6,20}$=0.43, $adj.R^2$=-0.15, p=0.8) or functional LCBD ($F_{6,20}$=0.738, $adj.R^2$=-0.06, p=0.6) and no patterns were evident between LCBD measures and any of the connectivity variables.

Table 6.4 Summary of full linear model for Shannon's diversity index, taxonomic richness and functional diversity (p-value < 0.5 in bold; p-value <0.1 in italics).

Response variable	Parameter	Estimate	S.E	p-value
Shannon's diversity index	(Intercept)	1.450	0.2813	**0.000078**
	PD_2500m	0.04412	0.01682	**0.016**
	EuD_N	0.00017	0.00018	0.347
	EuDA_N	0.00058	0.00047	0.235
	CD_2km	0.000008	0.00001	0.574
	BC2500	2.28e-11	1.44e-10	0.848
	IIC500	-45.26	77.17	0.564
Taxonomic richness	(Intercept)	2.833	0.1964	**0.0000**
	PD_2500m	0.0245	0.0135	**0.0479**
	EuD_N	0.00016	0.0001	0.188
	EuDA_N	0.00006	0.0003	0.832
	CD_2km	0.0000003	0.00001	0.972
	BC2500	-2.06e-11	9.67e-11	0.831
	IIC500	-69.9	54.46	0.199
Functional diversity	(Intercept)	37.12	5.305	**0.000**
	PD_2500m	0.5995	0.3171	*0.07*
	EuD_N	0.00045	0.0001	0.89
	EuDA_N	0.01526	0.0003	0.10
	CD_2km	0.00042	0.000009	0.13
	BC2500	-7.150e-10	2.726e-09	0.79
	IIC500	-509.3	1455	0.73

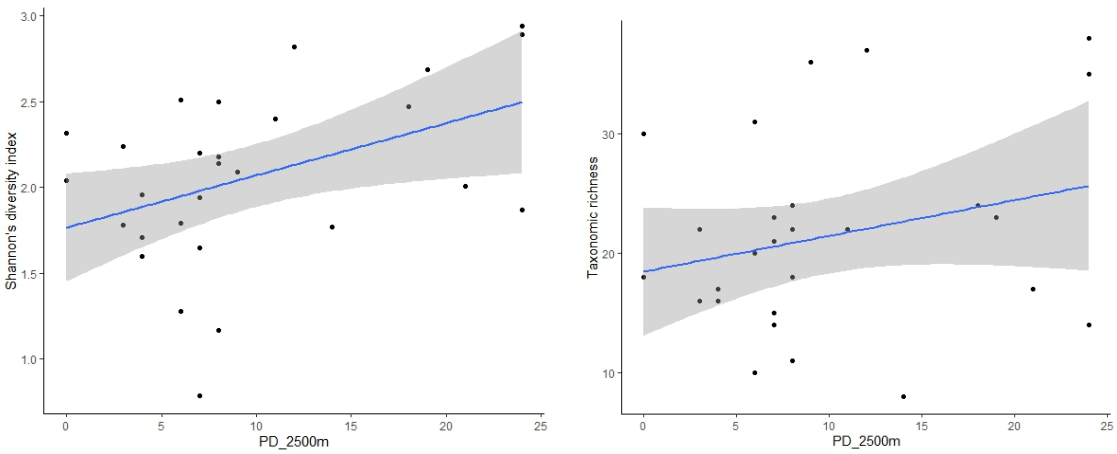

Figure 6.5 Relationship between the number of ponds (PD_2500) within 2500m of a given pond and Shannon's diversity index (left) and taxonomic richness (right). The shaded area represents 95% confidence intervals.

6.3.3 Relationship between pond connectivity and community composition variation

RDA with forward model selection yielded a model for community composition with only CD_2km (cost distance to nearest ponds within 2km radius) as a statistically significant predictor ($F_{1,25}$ *=1.619,* adj.R^2=0.023, p=0.02) of macroinvertebrate community variation (Figure 6.6).

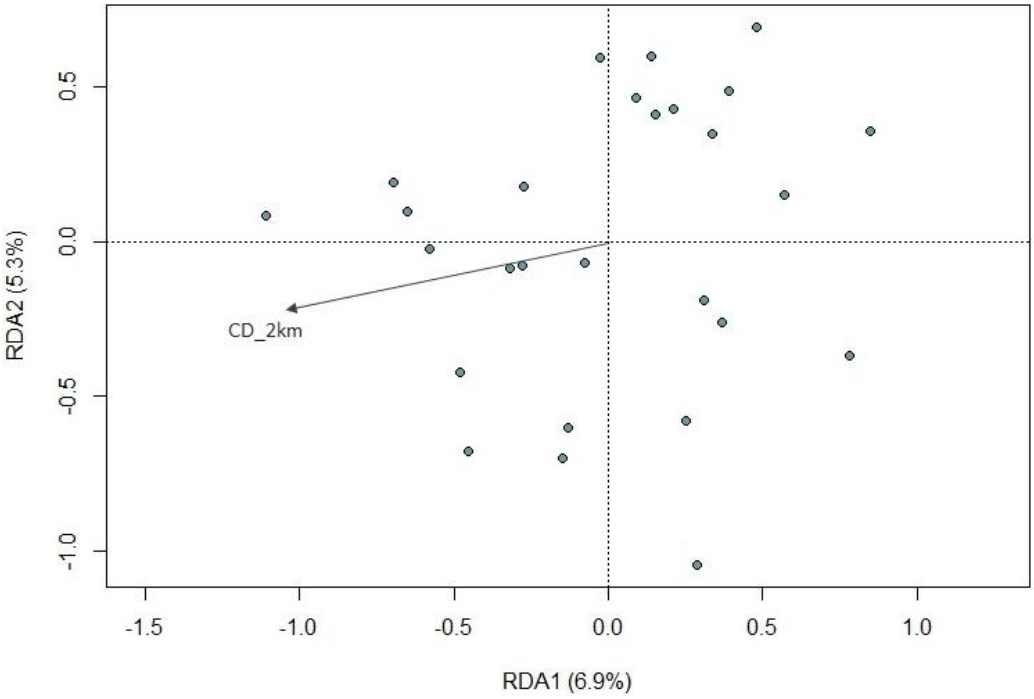

Figure 6.6 RDA plot showing distribution of pond communities (blue circles) and statistically significant connectivity variable CD_2km – cost distance to nearest ponds within 2km radius. The arrow represent the strength and direction of the variables.

The findings support the hypothesis that connectivity variables influence alpha and beta diversity patterns but the findings do not suggest strong relationships and only two of the variables examined showed significant relationships. Pond density (or the number of ponds within fixed distances of a focal pond) influenced taxonomic richness but only at 2500m radius. Gledhill et al. (2008) found greater pond density to be key to greater invertebrate richness especially across greater distance intervals. This could be attributed to greater connectivity among ponds increasing the available habitat available for taxa to utilize. Functional connectivity (cost distance) showed a significant relationship with variation in community composition and suggests that increasing distance and barriers to movement (presented by land cover type) play a role in structuring macroinvertebrate communities through isolation that limits dispersal and leads to formation of unique (or impoverished) pond communities (Hill et al., 2018; Thornhill et al., 2017).

6.4 Summary

This chapter described structural connectivity for the Greater Kuala Lumpur pond network, applying graph network analysis to highlight areas of highly connected ponds as well as ponds that are important for maintaining connectivity across the network. Using multiple measures of structural and functional connectivity (with increments of distance thresholds to reflect differences in species dispersal capacities), the relationship between diversity and connectivity was characterized. The results suggest that the number of neighbouring ponds rather than proximity is more important for alpha diversity. On the other hand, cost distance (to nearest ponds within 2km radius) was the only statistically significant variable associated with variation in community composition. This suggests that ponds separated from neighbouring ponds by barriers in the urban matrix and longer distances may have different community structures than ponds that are functionally connected with their neighbours.

This chapter concludes the results of data analysis. The next chapter discusses the key findings, how they compare with urban pond ecology knowledge to date as well as their implications for pond biodiversity management in Greater Kuala Lumpur.

Chapter Seven

Discussion and Synthesis

7. DISCUSSION AND SYNTHESIS

7.1 Key scientific findings

Ponds are widely distributed in the urban landscape of Greater Kuala Lumpur and serve multiple aesthetic and water management functions. Despite growing awareness of the importance of urban green spaces in Malaysia, very few studies have examined the value of blue spaces (particularly ponds) for supporting biodiversity (Lee et al., 2019; Razak & Sharip, 2019). Globally, recent studies have demonstrated the considerable habitat value of urban ponds for biodiversity (Oertli & Parris, 2019) but knowledge and data from tropical cities remains limited. While the Malaysian stormwater pond management guidelines include recommendations for aquatic plant management to support biodiversity (DID, 2012), this is the first study to examine ponds across the GKL region and look at multiple species to assess diversity. Such studies provide critical data necessary for local management, monitoring and conservation decisions. Thus, this study focused on characterizing pond biodiversity patterns in GKL and their relationships with the local and spatial variables to provide baseline data and information for management as well as determine potential directions for future research that targets urban pond management for biodiversity conservation and ecosystem services. The following sections summarise the main findings of the research with respect to each of its specific objectives.

Objective 1: Describe taxonomic alpha and beta diversity patterns and identify drivers of macroinvertebrate diversity at the local scale (richness, diversity, abundance) and landscape scale (beta diversity and its components) in Greater Kuala Lumpur (GKL)

In total 99 macroinvertebrate taxa were recorded over two seasons from ponds across the study region, with mean pond richness of 21.3 taxa and Shannon's diversity index values that ranged from 0.79 to 2.94. Clustering analyses revealed two groups of ponds with the most species rich ponds characterized by smaller surface areas, greater extents of aquatic vegetation cover and low water conductivity. The differences in alpha diversity between the two groups were statistically significant, highlighting the importance of pond-scale habitat structure and water quality for diversity. The data also indicated that ponds in the two groups had different community composition. Extent of aquatic vegetation cover and TSS were important variables for taxonomic alpha diversity. Both richness and Shannon's diversity index showed a negative response to an increase in TSS. Aquatic vegetation was also consistently important for both taxonomic alpha and beta diversity. Only one surrounding land use variable – proportion of impervious land cover within 500m – was important for taxonomic diversity, and only for explaining variation in community composition (beta diversity). Decomposition of beta diversity showed that community variation was primarily driven by replacement of taxa. LCBD analyses, however, did not

show any one site to have a statistically significant unique contribution to beta diversity. Nevertheless, findings suggest a non-linear relationship between LCBD metrics and taxon richness.

Objective 2: Describe trait-based (functional) pond community patterns and identify environmental and spatial drivers of trait distribution patterns

Pond communities were characterized in terms of 60 trait categories to determine functional richness and diversity. Mean trait richness was 54.5 (SD=1.14), with maximum richness possible being 60, and showed little variation, but functional diversity and functional evenness showed considerable variation among ponds, ranging from 30.24 – 65.11 and 0.29 to 0.75, respectively. Functional diversity increased with an increase in taxonomic diversity but there are some differences in the responses of these alpha diversity metrics to environmental variables. Functional diversity showed a similar negative response to increased TSS as taxonomic diversity, but no clear relationships were found between this metric and other environmental variables. On the other hand, functional diversity showed a non-linear response to road density within a 2km radius. Unlike taxonomic alpha diversity, functional diversity did not show a statistically significant difference between the pond types identified by clustering analysis. Furthermore, decomposition of beta diversity showed that richness difference contributed more to variation in functional community composition (beta diversity) than did turnover (replacement), suggesting redundancy of functional groups despite high taxonomic turnover. TSS was a statistically significant predictor of the richness difference component of functional diversity, but the RDAs did not yield a statistically significant model for total functional beta diversity, or for the turnover (replacement) component. There was little variation in functional LCBD and analyses did not yield statistically significant responses to functional diversity or any of the measured environmental variables.

Objective 3: Map ecological connectivity for pond macroinvertebrates in GKL and assess the influence of connectivity on diversity patterns

Geospatial analysis yielded insights into pond connectivity across GKL. Analyses used multiple distance thresholds that accounted for the differences in movement capabilities of macroinvertebrates. These allowed identification of areas of high and low connectivity pond networks, and also derivation of connectivity metrics based on distances among ponds and land use-land cover types between ponds. Contrary to expectations, Euclidean distance (proximity) measures did not explain diversity but pond density and cost distance measures had some influence on variation in alpha diversity and community composition, respectively.

7.2 Discussion

7.2.1 High variation in alpha and beta diversity

The variation in alpha and beta diversity, both taxonomic and functional, is broadly in line with current knowledge on urban pond macroinvertebrate biodiversity. Urban ponds in GKL show high variation in taxonomic and functional alpha diversity in response to habitat conditions, surrounding land use and pond connectivity. The predominance of aquatic insects in the samples was similar to findings from other regions (Hill et al., 2015; Thornhill et al., 2017), although Coleoptera taxa distribution was relatively rarer. In addition, Gastropoda distribution showed distinct patterns. Previous research on freshwater gastropod distribution among urban habitats in Singapore had suggested that urban reservoirs may be important habitats for local molluscan diversity (Clements et al., 2006). It must be noted, however, that among the most abundant Gastropoda recorded in this study was the non-native *Pomaceae canaliculata.* Similarly, Clements et al. (2006) had highlighted the proliferation of this large snail species in urban reservoirs. *P. canaliculata* is listed globally as a major invasive species of wetland ecosystems and occurs widely among different types of freshwater habitats (especially ponds and lakes) in Peninsular Malaysia, across urban and agricultural landscapes (Hah et al., 2022). The presence of *Pomaceae* populations affects community structure and ecosystem functioning through herbivory that reduces aquatic vegetation cover or alters aquatic vegetation composition (Horgan et al., 2014). This could indirectly lead to changes in pond community compositions through inter species competition or through altering habitat conditions (for example from macrophyte rich ecosystems to turbid habitats dominated by phytoplankton) which would also have implications for ecosystem functions such as nutrient cycling (Horgan et al., 2014).

The high level of taxonomic beta diversity driven by a greater proportion of species turnover than nestedness reported in this study agrees with findings from several urban and non-urban pond macroinvertebrate studies (Hill et al., 2017; Hill et al., 2018). Scheffer et al. (2006) list stochastic events, high variability in local conditions and isolation as factors explaining high variation in community composition among ponds (small habitats). The findings of the present study are in line with these to some extent – three variables (discussed further below) were found to influence beta diversity (taxonomic), although the proportion of variation explained by these was small (environmental and land use variables - $adj.R^2$=0.07; connectivity variable – $adj.R^2$=0.02). While the findings in this study suggest that structural connectivity (the number of surrounding ponds) has significant but weak influence on alpha diversity, there was no significant relationship with variation in community composition. Only cost distance (here, a measure for functional connectivity) was a statistically significant predictor of beta-diversity. This suggests that isolation or dispersal limitation in the urban landscape may contribute to variation in community composition. Components of the urban matrix

such as buildings, roads and traffic, as well as the lack of stepping stones between habitats in the form of riparian vegetation or green spaces, may prevent successful movement or dispersal of species to and from neighbouring ponds (Hyseni et al., 2021). Without a consistent exchange of individuals of different species, the community structure of neighbouring ponds may vary over time in response to different stochastic events (nutrient inputs, disturbance) or community dynamics (inter-species competition, predation) (Scheffer, 2006).

None of the of LCBD values calculated for the ponds were statistically significant. This suggests that although beta diversity is characterized by high turnover for the study area, no single pond has a significant unique composition. LCBD values did show variation, however, and the relationships observed among the LCBD variables with taxonomic richness were similar to what has been documented elsewhere. Negative relationships between richness and LCBD have been reported in previous studies (Heino, Bini, et al., 2017; Hill et al., 2021) but a weak non-linear relationship was observed in this study, where LCBD was lowest at intermediate richness. This could be due to the similarities in composition among the more species rich ponds which also tended to be those that supported the rarer taxa. Previous investigations found somewhat contradicting results for urban ponds; for example, Heino, Bini, et al. (2017) found no significant relationships with local environmental variables and suggested that landscape scale variables may be acting on LCBD by limiting dispersal (Heino, Bini, et al., 2017). On the other hand, Hill et al. (2021) found limited but significant effects of two local scale variables (including aquatic vegetation) across multiple cities but none with surrounding land use highlighting the importance of local pond conditions. The present study does not provide adequate evidence to support either of these suggestions, although weak correlations ($r = 0.32 – 0.34$) were observed for variables associated with land use variables within 500 m and 2km of ponds.

7.2.2 Main drivers of diversity patterns: aquatic vegetation cover and TSS

The extent of aquatic vegetation cover was consistently important for taxonomic richness and diversity (alpha) and community composition variation (beta-diversity) of ponds in GKL. This agrees with most urban pond research that reports higher diversity with greater or intermediate extents of aquatic vegetation cover, for multiple taxonomic groups (Oertli & Parris, 2019; Thornhill et al., 2017). Aquatic vegetation in not only a key food source for pond communities, but also provides shelter from predators and sites for oviposition, with different parts of a plant used for different purposes or by different species (Biggs et al., 1994). The structural complexity provided by aquatic vegetation also increases the mesohabitats available for pond communities and improves conditions (for example, oxygenation and sediment stability) for macroinvertebrate richness (Bazzanti et al., 2010; Biggs et al., 1994; Waters & San Giovanni, 2002).

The findings of this study, specifically the relationship between aquatic vegetation cover and the nestedness component of beta diversity, also suggest that while aquatic vegetation supports greater diversity, community composition may be similar among the more species rich ponds. To some extent, this could be because of the similarities in aquatic plant communities noted among urban ponds. It is notable that relatively rarer aquatic beetles (Coleoptera) and moths (Acentropinae) were recorded at high abundances in ponds with high aquatic plant cover but with plant species that were uncommon among pond habitats. While most ponds tended to be dominated by *Hydrilla* spp or *Nymphaea* spp, ponds that supported high Acentropinae (P10, P22) or Coleoptera (P14) diversity had mesohabitats with different submerged plant and emergent aquatic plant (grass) communities, respectively. Similarly, a pond (P9) with a primarily *Nelumbo* (lotus) aquatic plant community had greater gastropod density. Findings from previous studies on urban pond species also suggest diversity of aquatic vegetation influences invertebrate species richness (Goertzen & Suhling, 2013; Law et al., 2019; Thornhill et al., 2017). Another potential explanation for the relatively rarer presence of some taxa is the variation in substrate type, which was not analysed in this study but recorded for some of the sites. Substrate type may directly influence pond habitat quality for macroinvertebrates that require specific substrates for life stages such as pupation or indirectly by determining the amount and type of plants that can establish (Oertli & Parris, 2019). Pond P9, for example, had a shallow muddy substrate mesohabitat supporting the high density lotus plant community.

Total suspended solids concentration (TSS) was an important variable across taxonomic and functional diversity metrics and was also associated with variation in functional composition among ponds. Both taxonomic and functional diversity declined with increasing TSS. The RDA for functional composition showed relatively taxa poor communities positioned along the increasing TSS gradient. TSS is not often considered explicitly in studies of urban pond macroinvertebrates but comparing diversity between flood retention and water treatment ponds, Manzo et al. (2020), found higher diversity among the former which were partly characterized by low levels of TSS. Similarly, studies of urban river diversity found an inverse relationship between diversity measures and TSS for a GKL river (Azrina et al., 2006).

TSS is often used in water quality monitoring for urban wetlands and ponds, especially in determining the efficacy of various pond management in treating urban runoff (Greenway, 2017). High levels of TSS have been associated with decreasing habitat quality, physically and chemically, for both plant and invertebrate populations, by clogging feeding or breathing structures or inhibiting access to plant food sources (Greenway, 2017). In addition, high TSS concentrations can facilitate contamination with other pollutants or metals through adsorption of both organic and inorganic materials (Bilotta & Brazier, 2008). The mean TSS value recorded for ponds in this study (30mg/L) is much lower than those recorded for Klang and Langat rivers, 62mg/L and 152mg/L, respectively (Loi et al., 2022), although the effect of

duration of exposure and geochemical composition (or highly correlated variables such as chlorophyll a, turbidity) may give further insight into the relationship between TSS and species responses to urban ponds where pollutant loads may have longer durations (Bilotta & Brazier, 2008). It is also worth noting that low background concentrations of TSS (as well as other pollutants like nitrogen and phosphorus) are to be expected in urban ponds and wetlands (Greenway, 2017) and that some aquatic organisms depend on suspended organic matter for feeding (Bilotta & Brazier, 2008).

7.2.3 Drivers of functional (trait-based) variation across GKL ponds

Previous studies have shown that local habitat variables (including water quality and aquatic vegetation) influenced functional (trait-based) alpha and beta diversity in urban and non-urban ponds and lakes (Heino & Tolonen, 2017; Hill et al., 2019). Although similar local variables (conductivity, area, shade, aquatic vegetation cover) were examined in this study, similar patterns were not detected among ponds in GKL. Among pristine lake habitats, Heino and Tolonen (2017) found that both functional alpha diversity and to some extent functional community assembly was explained by conductivity and phosphorus, and suggested that habitat productivity may be more important than the physical characteristics (such as area). On the other hand, among ponds across various land uses, Hill et al. (2019) found that in addition to conductivity, both area and aquatic vegetation cover influenced functional alpha diversity and the latter two variables were also important for functional beta diversity. While the present study did not yield the same result for conductivity, another water parameter, TSS, best explained variation in functional alpha and beta diversity. These variations in responses perhaps reflect differences in the context of these studies (pristine habitats and disturbed habitats, or subject to different management practices) and in the present study key environmental drivers or covariates may have been overlooked and thus, a more targeted research design examining specific functional traits and potential drivers may be more useful in explaining the variation in functional traits observed among urban ponds in GKL. Nevertheless, the different responses of taxonomic and functional diversity measures to environmental variables is in line with previous research and highlight the importance of both taxonomic and functional diversity measures.

7.2.4 The role of impervious (built-up) surrounding land cover in structuring pond communities

None of the surrounding land use variables were found to be statistically significant predictors of taxonomic alpha diversity (richness and Shannon's diversity index) for ponds across GKL. This is contrary to findings which suggest that surrounding land use, specifically increased built up areas (such as greater proportion of buildings or sealed areas around a pond) negatively influences pond insect diversity (Blicharska et al., 2016; Heino, Bini, et al., 2017). However, the finding agrees with Goertzen and Suhling (2012) who found no relationship with dragonfly diversity and suggested results could

reflect species-specific responses to barriers to movement presented by land use, or the relatively greater importance of local habitat conditions. Nevertheless, water cover within 500 m and, as discussed below, the number of ponds in the surrounding area, were important for alpha diversity. This could be explained by the greater availability of total habitat presented by greater number of ponds supporting a greater number of species. The findings suggest that the influence of pond density may be relatively more important than the potential barriers presented by built urban components, either by increasing habitat available for colonization or also by presenting stepping stones connecting distant ponds. Nevertheless, targeted research covering a wider variation of land use/land cover proportion than covered in this study may be needed to capture the effects of land use variation on pond diversity.

On the other hand, the present study found that proportion of impervious cover within 500m radius influenced taxonomic community composition, suggesting direct (e.g. affecting movement between ponds) or indirect effects (e.g. affecting water quality) on community assembly (Akasaka et al., 2010; Hyseni et al., 2021; Thornhill et al., 2017). The proportion of impervious cover in the surrounding land use also best explained the turnover component of beta diversity. While high aquatic vegetation cover is associated with nestedness patterns (possibly because of similarities in habitat conditions), it is possible that ponds within built-up areas are subject to different environmental conditions due to differences in type of anthropogenic disturbance. For example, both commercial (traffic-heavy) and residential land use are categorized as impervious land cover but type and frequency of pond management and runoff water quality of these land uses may vary. Interestingly, greater proportion of impervious land cover was associated with communities characterized by a greater abundance or richness of Gastropoda including invasive *Pomacea* species and poor water quality indicators such as Physidae: *Physa*. The gastropod abundance model showed conductivity, which tended to increase with increasing impervious cover in the surrounding land use (Figure 4.2), positively influenced gastropod abundance.

Clements et al. (2006) showed that calcium ion concentration may be a predictor of mollusc richness in tropical urban lentic waterbodies, suggesting that higher concentrations found in modified landscapes are more conducive to molluscan diversity than the more acidic waters of natural habitats. Generally, higher conductivity is typical in urban waterbodies and this is because of built-up surface runoff carrying salts, ions or heavy metal pollutants (Paul & Meyer, 2001). It is thus often associated with lower diversity and abundance, for multiple taxonomic groups including amphibians, macroinvertebrates and reptiles (Hamer & Parris, 2011; Hassall & Anderson, 2014; Stokeld et al., 2014). This could be due to the direct, toxic impact of pollutants on species survival or by indirect impact through subsequent effects on food web structures (Oertli & Parris, 2019). Oertli and Parris (2019), however, note that the certain taxonomic groups such as mosquitos (Yadav et al., 2012) may respond

positively to high conductivity levels. The findings in the present study suggest that gastropods respond positively as well however, it is also possible that other organic or inorganic input into the water, or anthropogenic disturbance, associated with proximity to or increase in impervious surfaces or built structures are also influencing other water parameters including trophic state. For example, Razak and Sharip (2019) found that development density within 1 km of waterbodies, but not distance from the city centre, influenced trophic conditions and zooplankton communities among GKL waterbodies.

Pond connectivity measures explored in this thesis were found to influence structural and functional connectivity to some extent, although models with multiple connectivity measures for alpha diversity and total beta diversity did not explain variation in alpha diversity and total beta diversity (community composition) well. The findings suggest that the number of ponds, or available habitats, within a 2500m radius are more important than Euclidean or cost distances to nearest ponds for taxonomic and functional alpha diversity. This is in line with Gledhill et al. (2008) who indicated that the impact of the number of surrounding ponds on alpha diversity varied with spatial scale and that it was most important at the larger landscape scale. This could potentially be explained by greater flight dispersal capacities of taxa belonging to Hemiptera, Coleoptera and Odonata groups (McCauley et al., 2014; Thornhill, 2013) and perhaps also by the presence of components within the urban landscape, such as streams and riparian vegetation, that could facilitate movement (Kirk et al., 2018).

Hill et al. (2018) suggest that increasing connectivity among ponds in urban areas increases dispersal and colonisation opportunities. Measures of pond connectivity (or isolation) used in urban pond studies have been distance to nearest pond, number of hydrological connections, or percent isolation by barriers, and these generally explain variation in alpha diversity to a lesser degree than local environmental conditions or pond characteristics (Heino, Bini, et al., 2017; Hill et al., 2019; Hill et al., 2015; Holtmann et al., 2018). However, in the above studies, cost distances between ponds (reflecting the varying degrees to which specific urban components can potentially facilitate or impede movement) were not considered and the findings of the present study suggest that cost distance may be important for community assembly. This finding is in line with Liao et al. (2022) whose research revealed differences in aquatic beetle assemblages between isolated and clustered urban ponds. They showed that communities in isolated ponds were more dissimilar and characterized by beetles that were strong dispersers, whereas clustered ponds included weak dispersers as well, highlighting the importance of functional traits in assessing connectivity among urban ponds. Similarly, Hyseni et al. (2021) found that low functional connectivity (estimated by applying electric circuit theory) explains considerable proportion of community differentiation among urban ponds.

7.2.5 Differences among pond types

Contrary to other findings in urban landscapes (Heino, Bini, et al., 2017; Hill et al., 2015; Holtmann et al., 2018; Oertli & Parris, 2019), the present study did not find pH or pond area to be important predictors of macroinvertebrate diversity. This could be due to the limited variation recorded in pH among ponds and the uniformity in pond design, despite the variation in surface area. Previous studies have suggested the importance of area in increasing available habitat and supporting habitat heterogeneity and therefore sustaining higher diversity (Heino, Bini, et al., 2017; Hill et al., 2015). Goertzen and Suhling (2012) on the other hand, did not find area to be an important predictor for richness in urban ponds, and suggested that this reflects the importance of the quality of available habitat rather than size alone. Studies conducted in non-urban landscapes also show varying responses of pond diversity to pond area (Biggs et al., 2005; Hassall et al., 2011) and Scheffer et al. (2006) posited that second-order effects from the interaction of species (predation) may lead to biodiversity patterns that deviate from classic species-area relationships.

The output of the clustering analysis suggests that area may be influencing pond diversity across GKL to some extent, with larger areas being one of the habitat characteristics differentiating the two clusters. Ponds in Cluster 1 (which had lower alpha diversity values) had greater surface areas, and were closer to roads. It is possible that as the more common, larger ponds in this study serve as permanent retention ponds for a greater catchment area, they are managed differently, subject to more disturbance and receive more runoff water from the surrounding area than the ponds that made up Cluster 2, which were smaller or unmanaged ponds. In addition, small and medium sized fish were more likely to be present in the larger ponds, with informal fishing activities observed in several of them. Notably, the potential indicator taxa for Cluster 1 was Physidae: *Physa*, a taxa associated with nutrient rich waters. The limited aquatic vegetation, fish presence and urban water input may together be impacting habitat quality and leading to lower diversity or altered community composition.

None of the of LCBD values calculated for the ponds were statistically significant. This suggests that although beta diversity is characterized by high turnover for the study area, no single pond has a significant unique composition. LCBD values did show variation, however, and the relationships observed among the LCBD variables with taxonomic richness were similar to what has been documented elsewhere. Negative relationships between richness and LCBD have been reported in previous studies (Heino, Bini, et al., 2017; Hill et al., 2021) but a weak non-linear relationship was observed in this study, where LCBD was lowest at intermediate richness. This could be due to the similarities in composition among the more species rich ponds which also tended to be those that supported the rarer taxa. Previous investigations found somewhat contradicting results for urban ponds; for example, Heino, Bini, et al. (2017) found no significant relationships with local environmental

variables and suggested that landscape scale variables may be acting on LCBD by limiting dispersal (Heino, Bini, et al., 2017). On the other hand, Hill et al. (2021) found limited but significant effects of two local scale variables (including aquatic vegetation) across multiple cities but none with surrounding land use highlighting the importance of local pond conditions. The present study does not provide adequate evidence to support either of these suggestions, although weak correlations (r = 0.32 – 0.34) were observed for variables associated with land use variables within 500 m and 2km of ponds.

The literature review (Chapter 2) highlighted a number of important ecological knowledge gaps in lentic waterbodies research for tropical urban landscapes, several of which have been addressed in this thesis.

- One of the major limitations identified was the focus on major natural wetlands or lakes, overlooking abundant, smaller and man-made ponds in urban areas. The present study explored ecological diversity in ponds across GKL, documenting diversity in their environmental and ecological characteristics.
- The present study also addressed the lack of characterization that combined both environmental (including surrounding land use) and spatial characteristics (proximity, density, connectivity) in ecological studies of urban lentic habitats. Ponds were characterized by urbanization and spatial measures, that can facilitate knowledge transfer and comparative studies among tropical cities.
- Limitations in ecological characterizations were also highlighted with reviewed studies focusing on taxonomic richness. The present study demonstrated the importance of measuring multiple components of urban pond biodiversity, which show variations in their responses to environmental/design factors, to understand the mechanisms structuring pond communities and develop effective design and management practices for biodiversity conservation.

Finally, the findings of this study are broadly comparable with findings from temperate regions. Key among these is the importance of aquatic vegetation cover for supporting species-rich ponds, the high variation in community composition (beta diversity) across urban ponds, primarily driven by species turnover and the importance of pond density for macroinvertebrate diversity, all of which align with most findings from non-tropical regions (Gledhill et al., 2008; Hill et al., 2015; Oertli & Parris, 2019). In contrast, pond size did not significantly influence diversity and the results of clustering analysis suggested that ponds with smaller surface areas may be supporting greater macroinvertebrate diversity. Nevertheless, further research is needed to disentangle the influence of potential co-variables (for example, management type, function). Notably, the findings of the present study highlighted the importance of TSS for both taxonomic and functional diversity among ponds in GKL. Many ponds in GKL are part of storm water management systems and, considering the frequency of intense rainfall events

in the region (Bhuiyan et al., 2022), TSS may present a challenge for pond management practices that aim to improve biodiversity.

7.3 Implications for management

The present research is among the first to describe biodiversity patterns in tropical urban ponds, and the first comprehensive study in Malaysia. It highlights key patterns that align with global pond research findings to date, as well as key variations and local characteristics, all of which have implications for pond management that targets biodiversity conservation. These particular findings are relevant for underpinning evidence-based management of urban ponds in GKL. The following recommendations are based on the findings of this study and, where possible, are mapped to practices discussed in the Urban Stormwater Management Manual for Malaysia, MSMA (DID, 2012).

First, ponds in GKL show considerable variation in alpha and beta diversity, habitat conditions, management types and responses to surrounding land use. The high variation in pond conditions and high taxonomic turnover in macroinvertebrate diversity is consistent with research findings from non-tropical regions and suggests that ponds in GKL, despite mostly being man-made, can be important refuges for urban biodiversity. Urban biodiversity research and education in GKL has thus far focused on terrestrial taxonomic groups or ecosystem services of green spaces, while on the other hand, research on urban waterbodies have focused on water quality or lotic systems. The findings of the present study therefore demonstrate the potential biodiversity value of ponds that are abundant across GKL, functioning as water management or aesthetic components in the urban landscape. Therefore, it will be necessary to raise awareness of the value of urban pond as biodiversity habitats among urban stakeholders.

- Recommendation: raise awareness among pond managers and urban research organizations on the potential of urban ponds to act as refuges for local freshwater biodiversity.
  A useful step forward would be developing criteria for assessing public pond quality based on parameters besides water quality including, for example, ecological habitat quality or local species richness. These could be the basis for identifying regionally important taxa and engaging stakeholders in planning for pond management. Communicating both the ecological value and potential ecosystem disservices (e.g. invasive species proliferation) will also be importance, along with introducing strategies for routine monitoring of urban ponds for detection of invasive taxa known to alter or impoverish community composition and ecosystem functioning.

Second, as is well established for ponds in non-tropical regions, aquatic vegetation cover (and potentially aquatic vegetation) is important for improving habitat value. The present study did not explore factors that encourage planting and plant choices among public park ponds although extensive recommendations are provided in the Urban Stormwater Management Manual for Malaysia (DID, 2012). But the findings suggest that both the amount of vegetation cover and the type of aquatic vegetation are key to improving diversity. Among the sampled ponds, variation in aquatic vegetation cover was found among both managed and unmanaged ponds, and both types of ponds were included in the pond cluster associated with higher taxonomic diversity. In addition, although not explicitly analyzed in the present study, variation in communities with different species observed as well as findings from previous studies suggest that increasing the diversity of aquatic plants in ponds will be necessary to promote pond diversity for a wide range of taxonomic groups.

- Recommendation: add or increase proportion of aquatic vegetation cover in ponds.
  Promoting diverse aquatic vegetation cover in urban ponds, especially where no significant disruptions to primary functions are expected, would improve their habitat quality. Replacing concrete walls with earth margins and gentle slopes will be necessary for aquatic vegetation establishment, improving access to ponds for semi-aquatic organisms such as amphibians and increasing pond margin complexity (to create more heterogeneous mesohabitats). Appendix 1 of the Urban Stormwater Management for Malaysia lists plant species with specific properties (bank protection, aesthetic, indigenous) for stormwater ponds and surrounding landscape in urban and suburban areas. Referring to the manual for the implementation of planting strategies including submerged and floating plants is recommended for improving the habitat value of urban ponds. There is also room for research to understand and address the potential barriers to implementation or widespread adoption of recommended planting strategies

Third, the findings in this study show that water quality influences both alpha diversity and community composition, and thus monitoring pollutant levels will be important not only for improving habitat quality but also potentially limit the proliferation of non-native species. As discussed earlier, the proportion of impervious land cover within 500m was found to be highly correlated with conductivity and influenced community composition. Similarly, TSS was consistently found important for taxonomic and functional diversity. Considering the water management, sediment removal or pollution treatment functions of ponds, this may require consideration of costs and trade-offs but improvements in designs may help improve some water parameters like suspended solids concentration (Greenway, 2017).

Notably, even among ponds located within well managed public parks, litter and unpleasant odors were encountered when sampling for this study.

- Recommendation: monitor and mitigate the effects of high concentrations of suspended solids (TSS) and high levels of conductivity.
  Greater attention to improving water quality is needed not only for its effects on wildlife but also for potential impact on urban stakeholders' perceptions of the value of ponds. However, improving water quality will not be easy in tropical cities such as GKL where poorly maintained storm overflow systems lead to contaminated water entering urban water courses and ponds during frequent heavy rainfall events. Implementing preventative and mitigating measures for addressing TSS and salt/ion input (associated with conductivity) such as (i) integrating buffer strips of selected plants around ponds to filter soluble pollutants from water entering ponds and stabilize banks (ii) improving implementation of water quality controls recommended by MSMA such as sediment forebays and (iii) planting wetland vegetation for regulating nutrient levels and trapping sediment are recommended. Documenting the efficacy of these practices and their impact on pond biodiversity would also be valuable for developing best practices for reconciling water management and biodiversity conservation for ponds in GKL. Finally, ponds receive water from drains beyond their immediate surroundings and thus highlighting the impacts of drain pollution on pond life in ongoing awareness campaigns on river pollution will also be necessary.

Fourth, clustering analysis yielded two pond types based on a small set of water and pond structure characteristics. Most of the ponds, however, were included in Cluster 1, characterized by relatively greater surface area, low vegetation cover and closer distance to roads (also higher levels of TSS and conductivity). This perhaps reflects the greater proportion of public park ponds sampled. These were more likely to have similar designs—simple margins, greater proportion of open water, often steeper slopes and receiving water from multiple stormwater drains. While there is variation in the structure of park ponds and several of the ponds in Cluster 2 were also found in public parks (others were found in botanical gardens, roadsides or campus locations), the larger ponds are often central features of a park landscape. This may also be a trend among ponds within residential estates with all new developments required to build wet retention ponds. As reported by Sinclair et al., (2020) for Florida, urban ponds in GKL are also owned and managed by a number of different stakeholders (e.g. local municipalities, real estate developers, botanic gardens, universities, and the Department of Irrigation and Drainage) each of which will have different priorities with pond management. This may not necessarily be a hindrance to promoting biodiversity since preserving heterogeneity in urban pond structure and conditions,

including among ponds that are not subject to intensive or regular management, improves habitat heterogeneity and overall diversity (Hill et al., 2019). Nevertheless, where practices such as those recorded for ponds Cluster 2 prevail, it may be necessary to raise awareness of the value of approaches to pond design and management that maximize benefits for biodiversity.

- **Recommendation: establish small ponds (with surface areas less than 1000m^2) where possible.**

  Promoting the establishment of smaller ponds with greater vegetation cover around these central ponds (as was the case for some of the sampled ponds) may be one way of increasing the availability of ponds that meet the habitat requirements of freshwater species while also providing additional aesthetic benefits. For example, many ponds are located within lawn parks where there is potentially space to construct smaller ponds that can be managed differently (for example, shallower or with different macrophyte plant groups) adding to the heterogeneity of both freshwater habitat and green space available.

Finally, the findings in this study highlighted the importance of both pond connectivity (number of surrounding ponds) and increasing cost distance (isolation) for alpha diversity and community heterogeneity. Hill et al. (2018) suggested that both high pond density and isolation may be necessary; the former ensures increased dispersal and colonization whereas some extent of isolation would prevent homogenization of pond communities. The pond connectivity maps shown in Chapter 6 make it possible to identify pond clusters or patches with varying levels of connectivity. These show regions like central Kuala Lumpur and Shah Alam with low and high pond connectivity, respectively. While the scope of the present research limits evidence for the relative value of ponds in each region, similar analysis can be used to identify priorities for each region: for example, either increasing the number of ponds in a region, or prioritizing rehabilitation or maintenance of existing ponds for their unique community composition.

- **Recommendation: develop a regional framework for pond management integrating landscape ecology knowledge.**

  Pond management should have a landscape scale element rather than focusing only on individual ponds. This includes measures that consider city-wide pond habitat availability including (i) identifying regions that have low pond density or structural connectivity for increasing pond availability where possible, (ii) identifying key ponds that support functional connectivity (or act as stepping stones) and ensuring their conservation, (iii) improving green cover or riparian vegetation cover (corridors) to facilitate movement of organisms between ponds where greater proportions of built structures or human activity present barriers to

movement, (iv) conservation of areas that support highly connected pond clusters, (v) designating high conservation value status to ponds that support rich or unique species composition in the urban landscape. In the case of GKL, Teo et al. (2021) estimated a network of at least 1500 ponds and wetlands across the city. The challenge for management is that these are spread across multiple administrative districts each of which has its own development plans and priorities. There is a need for the national government to provide an overarching framework for pond conservation that district authorities are required to adhere to which however, as discussed below, may present some challenges.

As waterbodies created for water management purposes or aesthetic landscape components, urban ponds in Malaysia are not generally recognized as biodiversity resources. Although there is a growing movement to support terrestrial urban biodiversity, much less research attention has been given to the ecology of urban ponds. An example of a city-wide approach to managing wetlands for multiple benefits including biodiversity is present in Malaysia for the Federal Territory of Putrajaya but no similar initiatives have extended to the whole of GKL. This study presents the first step in demonstrating the opportunities for increasing wildlife habitat in urban areas across GKL through targeted research and management of urban ponds. The Malaysian National Policy on Biological Diversity (2016-2025) includes protecting and maintaining urban biodiversity as a key target and indicator for Target 6 which is concerned with protection of ecosystems through area-based conservation measures (NRE, 2016). However, there is no explicit mention of urban freshwater bodies within the policy. This omission likely reflects the historic lack of awareness in the biodiversity of urban ponds.

Ponds in GKL represent novel habitats constructed for a range of anthropogenic uses. With water retention as a primary function of many GKL ponds, pond biodiversity is currently not a key research or conservation priority. Increasing attention to the benefits of urban blue-green spaces is drawing research and conservation attention to habitats in urban areas across tropical cities (Jaturas et al., 2020; Lugo, 2010; Wong et al., 2023) but the knowledge of the relative ecological and social importance of ponds remains limited. Most conservation effort and attention primarily focus in Malaysia focus on natural habitats including coastal wetlands and mangroves, natural forests, upstream rivers and peat swamps threatened by land use changes. Nevertheless, there are is emerging research on urban habitats and their benefits in GKL that spans studies of social and ecological importance. Studies of Putrajaya and other repurposed wetland recreational parks in peri-urban areas show their importance for avian biodiversity (Martins et al., 2019; Rajpar & Zakaria, 2014) and recreation (Siew et al., 2015). Emerging studies of urban parks and green spaces in Malaysia suggest favorable attitudes of urban residents toward protecting and supporting management of green spaces (Jamean & Abas, 2023; Nath

et al., 2018). Despite awareness of the potential multiple ecosystem services of urban blue-green spaces, there are considerable challenges to introducing more ecologically directed or biodiversity friendly design/management approaches (Ibrahim et al., 2020). These include concerns associated with perceptions of threat to safety (e.g., physical safety with complex vegetation structure that prevents visual permeability, preferences for 'manicured lawns', mosquitoes in vegetated ponds) that can limit the relative biodiversity or conservation role of urban habitats such as ponds (Ibrahim et al., 2020). Considering the multiple uses of ponds and their proximity to anthropogenic activities, diverse ownerships, and concerns about diseases (mosquito-borne), there is a call for interdisciplinary research (e.g. epidemiologists, ecologists, social scientists and practitioners) to systematically identify challenges and opportunities (that can vary within GKL (Ibrahim et al., 2020)) for conservation and provide evidence-based policy recommendations within the framework of Target 6.

To support recognition of urban ponds as biodiversity habitats, there is a need for local authorities and researchers to first build baseline datasets and a typology of urban ponds based on their functions *and* ecological characteristics. This will require developing systematic sampling procedures for multiple taxa, followed by identification of priority taxa for conservation. This can then be the basis for assessing and monitoring overall pond habitat quality in addition to water quality and stormwater retention or water management criteria. As mentioned above, urban ponds are owned and managed by many different public and private stakeholders. Thus, in order to generate representative datasets and integrate them, greater efforts are need to raise awareness about their potential conservation value and encourage participation in regional surveys. In addition, urban ponds can also provide an opportunity for freshwater species research providing accessible ecosystems for nature education and harnessing citizen science to tackle some of the challenges to conservation including data paucity and limited funding.

7.4 Research limitations

The present research presented a systematic study of ponds in a tropical lowland urban landscape of Greater Kuala Lumpur, contributing to knowledge of urban pond research and addressing a geographic bias in research to date. This thesis also demonstrated the variation in biodiversity and habitat value of ponds across GKL and this can serve as a starting point for further applied scientific research. This will be necessary for urban ponds to gain recognition for their ecological value as freshwater habitats in GKL and for developing policies that support management practices discussed above. Nevertheless, there were several limitations in the present research which should be acknowledged and addressed.

First, the relatively small sample size studied here, while representative of many types of ponds observed in GKL was limited primarily to ponds open to the public and accessible. Although there was

considerable variation among these public and/or unmanaged ponds, there remain different types of ponds either within private parks or residential estates that make up a considerable proportion of the pondscape (new housing developments are required to have wet retention ponds). It is possible that some of these exhibit different features due to different management and aesthetic preferences, as well as age and spatial features. Hence, the types of ponds determined in this study and the extent of variation in key parameters recorded may be limited. Nevertheless, the sampled ponds varied in key ecologically relevant parameters (area, surrounding land cover, aquatic vegetation cover, water chemistry) and key patterns reported in this study are in line with literature on both urban pond and pond knowledge in general.

Second, several key environmental parameters such as pond depth, age, management, slope, substrate type and nutrient content may need to be examined with specific taxonomic groups in order to further explore the impact of local design and management practices on pond biodiversity. As suggested by the findings in this study and in previous research, responses to environmental conditions or urban stressors vary according to the taxonomic group subject, and even within taxonomic groups, variation in responses among species have been observed (Clements et al., 2006; Goertzen & Suhling, 2012; Liao et al., 2020). Thus, while the findings in this study indicate the importance of several environmental and spatial variables for macroinvertebrate diversity, a more thorough approach investigating targeted species group may be more informative. This will be especially relevant for taxa such as non-native *Pomaceae* species and pest mosquito species distribution. In a review of freshwater macroinvertebrate conservation challenges, Sundar et al. (2020) highlight the critical distribution, taxonomic and ecological knowledge shortfalls that persist for tropical species. They recommend addressing these shortfalls and greater documentation of species distributions as well as assessing their extinction risks without which conservation planning against threats like climate change and pollution is limited (Sundar et al., 2020). Addressing these shortfalls will be necessary to adequately assess the value of urban ponds for conservation and inform research priorities, especially by (i) contributing to existing trait databases (such as the one used here to assign trait scores, developed by Tachet et al. (2010) for European freshwater taxa) and (ii) prioritizing species for conservation.

Third, several taxa were necessarily excluded from the trait-based analyses because of inadequate information on their ecological traits while others were assigned trait scores at family level. This may have limited the degree to which functional diversity was represented in this study.

Fourth, the cost distances calculated in this study (representing functional connectivity) were based on methods in assessing cost distance along resistance scores assigned to nine land use categories. Although the application of cost distance has been found to perform well in explaining variation in

stream communities (Heino, Bini, et al., 2017; Kärnä et al., 2015), the methods can be further improved. More nuanced approaches that take into consideration the variations within each land use-land cover, (for example, the height of buildings or whether access to a low resistance surface like a stream is limited by steep concrete margins) would improve connectivity mapping and calculation of connectivity metrics. In addition, Teo et al. (2021)note that their inventory of GKL's lentic waterbodies misses smaller ponds and these include several of the smaller sample ponds in this study that were ecologically important (exhibited high taxa richness, rare taxa). Thus, future studies of pond connectivity may have to combine multiple sources (e.g. local knowledge, municipal or town maps) to better represent habitat presence and distribution. While the approach applied in this study was necessary for the wide geographical area covered in this study (and available land use datasets) more accurate assessments can be achieved with targeted species (or species groups) studies and higher resolution land use datasets for the region.

Finally, the research design could be further improved by experimental approaches that directly measure the impact of key design or water chemistry on diversity, allowing for findings that are interpretable, or practical knowledge, for pond design or restoration. For example, controlled studies of the interaction between aquatic vegetation cover and other variables such as water chemistry or surface area, may yield insights for pond management practices that maximize multiple benefits (e.g. flood mitigation and biodiversity). Furthermore, research shows that taxonomic groups vary in their responses to environmental variables and pond design. There is room for research design that targets specific taxonomic groups (for e.g. weak vs strong fliers, groups occupying different niches, non-native taxa) to compare their responses to pond characteristics. This can be useful for effective pond management or conservation initiatives that targets species of conservation concern, or aims to increase the heterogeneity of pond design at a landscape scale.

7.5 Suggestions for future research

Considering the findings of this research, as well as the limitations listed above, the following directions and approaches for future research are suggested to further contribute to pond biodiversity knowledge and improve local management and monitoring practices. Potential contributions to urban biodiversity research in GKL are also discussed.

1. Expand research to aquatic and semi-aquatic vertebrate groups common in urban ponds in GKL.

The present study focused on macroinvertebrates but ponds are also habitats or resources for vertebrates including fish, amphibians, monitor lizards and aquatic birds. Assessments of the conservation value of urban ponds would be more representative with knowledge of the distribution

patterns of these organisms. It is also important to understand the extent to which interactions among these different groups affect diversity and community composition. For example, the presence of fish has been found to influence community structure in small freshwater habitats and urban ponds, either through predation or by affecting nutrient concentrations in ponds (Liao et al., 2020; Oertli & Parris, 2019; Scheffer et al., 2006). The findings of this study suggest that fish and other vertebrate presence may also be a key factor – for example, dense populations of monitor lizards and red-eared sliders were observed in P20 and P21 which may have interacted with the local environmental parameter to affect diversity measures. In addition, fish were also observed in many of the larger ponds in this study and with introduced species common in tropical urban waterways (Kwik et al., 2020; Saba et al., 2020; Yap et al., 2005) future research focus on determining their impact on local biodiversity patterns.

2. Document pond distribution among urban forest parks and develop a typology of ponds based on multiple physical and ecological criteria

A number of the smaller ponds within forested areas here were included in the sampling pool based on field visits. There are a number of intact forests and forested parks across GKL that are habitats for large terrestrial mammals of conservation importance as well as terrestrial insects (Danneck et al., 2023; Mbugua et al., 2020; Sing et al., 2016; Tee et al., 2018). However, knowledge on freshwater habitat availability and quality compared with urban habitats remains limited. Significant differences between urban and forested urban lake water chemistry and Chironomidae communities have previously been reported for GKL (Lee et al., 2019) but further research is needed to develop a typology of urban freshwater habitats based on their physical, chemical and biological community characteristics. Teo et al. (2021) developed a typology of GKL ponds based on a range of environmental characteristics. Extending the characterization of pond types to include their ecology will be important in facilitating a biodiversity-oriented management on a landscape scale.

3. Conduct long-term ecological research examining temporal variations in ecological and environmental conditions and their implications for long-term biodiversity management

The present study sampled across two seasons but future work should also focus on developing long term studies that explore succession patterns as well as responses of biological communities to environmental changes over extended periods of time. There is evidence of changes temporal changes in diversity in urban ponds habitats over a longer observation period in response to increasing nutrient and pollutant loads (Briers, 2014) and thus, moving forward, it will be important to characterize and quantify the patterns in environmental changes and their impact on pond biodiversity. Notably, of the more important variables for diversity identified in this study was TSS and suspended sediment concentration which has effects that act over long exposure (Bilotta & Brazier, 2008). Furthermore,

suspended sediment can release nutrients and pollutants through adsorption (Bilotta & Brazier, 2008) and have also been found to correlate with microplastic concentrations in GKL rivers (Chen et al., 2021). Although the potential long-term effects of the latter remain unclear (Stanković et al., 2022; Windsor et al., 2019); these highlight the need for studies that can inform long term management for biodiversity in urban habitats that are subject to consistent anthropogenic disturbances (Briers, 2014). In addition, as more reference databases become available, more cost-effective and rapid methods of biodiversity assessment such as environmental DNA analyses can be adopted for long-term monitoring (Belle et al., 2019).

4. Design controlled studies to further describe and explain the influence of functional connectivity on alpha and beta diversity patterns

The connectivity mapping carried out in the present study showed areas of high and low structural connectivity, as well as methods for identifying important ponds based on connectivity metrics derived from the maps. Future research can extend this approach by designing controlled studies that explore differences in diversity between ponds in these areas (e.g. Balbi et al. (2021)) as well as the relative contribution of connectivity and environmental factors to the habitat value of urban ponds. In their study, Balbi et al., (2021) model areas of low and high moth habitat connectivity for moths in an urban landscape. They then employ an experimental mark-release-recapture approach to determine whether movement patterns detected with this method converge with modelled pathways thereby validating the relevance of connectivity modeling. Similar studies combining approaches can be carried out to validate connectivity metrics and describe their relationship with diversity patterns.

5. Identify ecosystem services and disservices associated with ponds in GKL and the relationships between perceptions of ponds, preferences for pond design and biodiversity

Finally, evaluating the ecosystem services provided by urban ponds will be important to identifying the constraints and opportunities that might influence their management and conservation. Understanding urban stakeholders' perceptions of urban ponds and addressing concerns will be key to developing effective management strategies and garnering support for them. This will be especially relevant where conflict arises between the infrastructure or aesthetic functions of ponds and habitat conditions critical for freshwater wildlife. For example, while urban residents may appreciate ponds for their cultural value, their preferences for pond appearance and characteristics may not be aligned with habitat qualities (such as less managed, complex vegetation structure) required for wildlife (Ngiam et al., 2017; Qiu et al., 2013). These preferences however, may need only apply to public park ponds and less actively managed ponds may still provide diverse habitat types.

In recent years, several organizations and community groups dedicated to urban green space experience, protection and education have been established in GKL. In addition, an increasing number of social studies have reported positive perceptions and tolerant attitudes toward urban wildlife among urban GKL residents (Lim & Wilson, 2019; Tan et al., 2020) although it should be noted that there are variations and taxonomic biases in these results and there is need for further research on attitudes toward urban wildlife more likely to be encountered more frequently, at closer proximity, and belonging to less popular or visible taxonomic groups like freshwater species. Nevertheless, these initiatives can also provide platforms for extending this recognition to the ecological role of freshwater habitats, supported by biodiversity research.

Research on urban biodiversity patterns suggests that besides environmental factors, socio-economic factors indirectly play a role in determining the amount and quality of urban habitats. This is especially evident with the 'luxury effect', where more affluent areas of a given urban landscape or city often support greater biodiversity than less affluent areas (Hope et al., 2003; Leong et al., 2018). The luxury effect has been detected for terrestrial species in tropical landscapes, with many non-native species being introduced by urban habitats (Bigirimana et al., 2012). Contrastingly, Meléndez-Ackerman et al. (2014) highlight the absence of the luxury effect and the greater importance of other demographic factors such as residents' age and ownership. Nevertheless, studies have focused on terrestrial plant and animal species while evidence of the 'luxury effect' for aquatic ecosystems remaining limited (Marques et al., 2024). There is room for research investigating how socio-economic factors such as median household income of district, ownership (public or private), greater availability of green spaces —associated with greater wealth within cities as well as across cities (Richards et al., 2017)— directly or indirectly influence pond habitat distribution, water quality, design preferences, and management practices.

Exploring current and potential ecosystem disservices associated with ponds will also be critical to maximizing the benefits of urban ponds. Concerns associated with pest or invasive species have to be systematically researched and addressed as well. In a study of stormwater ponds across the state of Florida, Sinclair et al., (2020) reported high frequencies of invasive species occurrence ponds across both managed and unmanaged ponds. Specific patterns in trait compositions of invasive plant communities were also identified Sinclair et al. (2020b) and understanding these patterns allows for targeted management and monitoring activities as well as assessing potential risks of invasive species spread into natural waterbodies. Hanford et al. (2019) also highlight the public health threats associated with wetland species such as mosquitoes that may affect perceptions of urban habitats, yet they also show that identifying the responses of targeted species to urban environmental factors to targeting

these for management can help mitigate risks to public health while preserving habitats for urban pond biodiversity.

7.6 Conclusion

Greater Kuala Lumpur is a tropical metropolis within a region of rich biodiversity but there remains limited systematic research on the ecological value of remnant and novel natural spaces within the city. This is especially true for ponds that, although common, are often novel habitats with mainly water management functions and so receive less attention than terrestrial or lotic habitats. In this thesis, macroinvertebrate diversity patterns in ponds across Greater Kuala Lumpur were assessed to characterize local and landscape scale diversity patterns and identify key environmental factors structuring communities. The findings presented in this thesis demonstrate the ecological habitat value of ponds in Greater Kuala Lumpur, highlighting the need for raising awareness of their contribution to urban biodiversity. The high variation in alpha and beta diversity demonstrated in this thesis, as well as the results of analyses examining their relationship with environmental and spatial factors, suggest that ecological patterns in tropical urban ponds are comparable with those of non-tropical regions, and although mostly man-made, can provide refuge for local freshwater communities when conditions such as high aquatic vegetation cover and pond density are met. In addition, this thesis also demonstrated the importance of urbanization related factors, namely suspended solids and conductivity in pond waters that have significant impacts on the taxonomic and functional composition of macroinvertebrate communities. By incorporating spatial analysis of pond distribution and connectivity, this thesis has also demonstrated the importance of considering the influence of the surrounding landscape in structuring communities, highlighting how adding a landscape scale approach to assessing ponds can be valuable for prioritizing regions or ponds for biodiversity management. Finally, this thesis has contributed to the emerging research and policy interest in urban biodiversity in Malaysia, and outlined key findings and information that can help inform management strategies and vital research directions for pond ecology in the region.

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APPENDICES

APPENDIX A

Table A1 List of research articles included in the review analyses

	Author	Title	Year	Publication
1	Abd Razak, SB; Sharip, Z	Spatio-temporal variation of zooplankton community structure in tropical urban waterbodies along trophic and urban gradients	2019	Ecological Processes 8(1): 1-12.
2	Adhikari S., Roy Goswami A., Mukhopadhyay S.K.	Diversity of zooplankton in municipal wastewater-contaminated urban pond ecosystems of the lower Gangetic plains	2017	Turkish Journal of Zoology 41(3): 464-475.
3	Amin A.K.M.K., Haque M.A., Alamgir M.	Analysis of the wetland degradation around the vicinity of Dhaka city in Bangladesh	2013	Asian Journal of Water, Environment and Pollution 10(2): 19-26
4	Ansari, NA	Monitoring of water bird population with an account of heronry at Surajpur lake, an urban wetland in National Capital Region, India	2017	Indian Journal of Animal Sciences 87(12): 1504-1512
5	Ansari, NA	Assessment of scope for fish biodiversity conservation in relation to environmental variables at Surajpur Lake, an urban wetland of the Upper Gangetic Plain, Northern India	2018	Indian Journal of Fisheries 65(3)
6	Atekwana, EA; Agendia, PL; Atekwana, EA; Fonkou, TH	Wetland vegetation colonization and expansion in small impoundments in Yaounde, Cameroon, West Africa	1995	Wetlands 15(4): 354-364
7	Athapaththu A.H.L.C.M., Wickramasinghe D., Somachandra M.G.M.C.	Hotspots of land use/land cover change around bolgoda wetland, Sri Lanka	2020	Journal of the National Science Foundation of Sri Lanka 48(3)
8	Baharuddin Z.M., Rusli N., Ramli L., Othman R., Yaman M.	The diversity of birds and frogs species at perdana botanical lake Garden, Kuala Lumpur, Malaysia	2017	Advanced Science Letters 23(7): 6256-6260
9	Bandyopadhyay S., Narayanan K., Ramanathan A.	Determinants of Willingness to Pay for	2006	Studies in Regional Science 35(4): 983-996

		Wetland Conservation: A Study of Kolkata Wetland		
10	Bareuther M., Klinge M., Buerkert A.	Spatio-temporal dynamics of algae and macrophyte cover in urban lakes: A remote sensing analysis of bellandur and varthur wetlands in Bengaluru, India	2020	Remote Sensing 12(22): 3843
11	Basu A., Sengupta S., Mandal D., Kundu G., Roy S.	Primary productivity and alpha-radioactivity in selected managed and unmanaged ponds at East-Calcutta Wetlands	2013	Indian Journal of Environmental Protection 33(9): 729-736
12	Biswas, S; Banerjee, A	Studies on Avifaunal Diversity of Santragachi Wetland, West Bengal, India	2016	Asian Journal of Water, Environment and Pollution 13(3): 29-36
13	Brinkmann K., Hoffmann E., Buerkert A.	Spatial and temporal dynamics of Urban Wetlands in an Indian Megacity over the past 50 years	2020	Remote Sensing 12(4): 662
14	Campion, BB; Venzke, JF	Spatial patterns and determinants of wetland vegetation distribution in the Kumasi Metropolis, Ghana	2011	Wetlands Ecology and Management 19(5): 423
15	Chowdhury S., Soren R.	Butterfly (Lepidoptera: Rhopalocera) fauna of east calcutta wetlands, West Bengal, India	2011	Check List 7(6): 700-703
16	Clements, R; Koh, LP; Lee, TM; Meier, R; Li, DQ	Importance of reservoirs for the conservation of freshwater molluscs in a tropical urban landscape	2006	Biological Conservation 128(1): 136-146
17	Crossetti, LO; Bicudo, CED	Adaptations in phytoplankton life strategies to imposed change in a shallow urban tropical eutrophic reservoir, Garcas Reservoir, over 8 years	2008	Hydrobiologia 614(1): 91-105
18	Crossetti, LO; Bicudo, CED	Phytoplankton as a monitoring tool in a tropical urban shallow reservoir (Garcas Pond): the assemblage index application	2008	Hydrobiologia 610(1): 161-173

19	da Costa D.F., Dantas E.W.	Diversity of phytoplankton community in different urban aquatic ecosystems in metropolitan João Pessoa, state of Paraíba, Brazil [Diversidade da comunidade fitoplactônica em diferentes ecossistemas aquáticos urbanos da região metropolitana de João Pessoa, PB, estado da Paraíba, Brasil]	2011	Acta Limnologica Brasiliensia 23(4): 394-405
20	Das S., Pradhan B., Shit P.K., Alamri A.M.	Assessment of wetland ecosystem health using the pressure-state-response (PSR) model: A case study of Mursidabad District of West Bengal (India)	2020	Sustainability 12(15): 5932
21	Das, A; Basu, T	Assessment of peri-urban wetland ecological degradation through importance-performance analysis (IPA): A study on Chatra Wetland, India	2020	Ecological Indicators 114: 106274
22	de Oliveira, FHPC; Ara, ALSCE; Moreira, CHP; Lira, OO; Padilha, MDF; Shinohara, NKS	Seasonal changes of water quality in a tropical shallow and eutrophic reservoir in the metropolitan region of Recife (Pernambuco-Brazil)	2014	Anais da Academia Brasileira de Ciências 86(4): 1863-1872
23	Dey D., Banerjee S.	Land-use Change and Vocational Transition in East Kolkata Wetlands: Evidence from Time Diary	2016	Environment and Urbanization Asia 7(2): 243-266
24	D'Souza, R; Nagendra, H	Changes in Public Commons as a Consequence of Urbanization: The Agara Lake in Bangalore, India	2011	Environmental management 47(5): 840
25	Evans B.A., Gawlik D.E.	Urban food subsidies reduce natural food limitations and reproductive costs for a wetland bird	2020	Scientific reports 10(1): 1-12
26	Fonseca, BM; Bicudo, CED	How important can the presence/absence of macrophytes be in	2010	Journal of plankton research 32(1): 31-46

		determining phytoplankton strategies in two tropical shallow reservoirs with different trophic status?		
27	Hara, Y; Yamaji, K; Yokota, S; Thaitakoo, D; Sampei, Y	Dynamic wetland mosaic environments and Asian openbill habitat creation in peri-urban Bangkok	2018	Urban Ecosystems 21(2): 305-322
28	Hayes W.M., Fisher J.C., Pierre M.A., Bicknell J.E., Davies Z.G.	Bird communities across varying landcover types in a Neotropical city	2020	Biotropica 52(1): 151-164
29	Hettiarachchi M., Athukorale K., Wijekoon S., de Alwis A.	Urban wetlands and disaster resilience of Colombo, Sri Lanka	2014	International Journal of Disaster Resilience in the Built Environment 5(1): 79-89
30	Hettiarachchi M., Morrison T.H., Wickramsinghe D., Mapa R., De Alwis A., McAlpine C.A.	The eco-social transformation of urban wetlands: A case study of Colombo, Sri Lanka	2014	Landscape and Urban Planning 132: 55-68
31	Isunju J.B., Kemp J.	Spatiotemporal analysis of encroachment on wetlands: A case of Nakivubo wetland in Kampala, Uganda	2016	Environmental monitoring and assessment 188(4): 203
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APPENDIX B

B1 Descriptive statistics for pond physical features and water parameters for seasons 1 and 2. The remaining variables did not show change across seasons and are presented in Table A1.

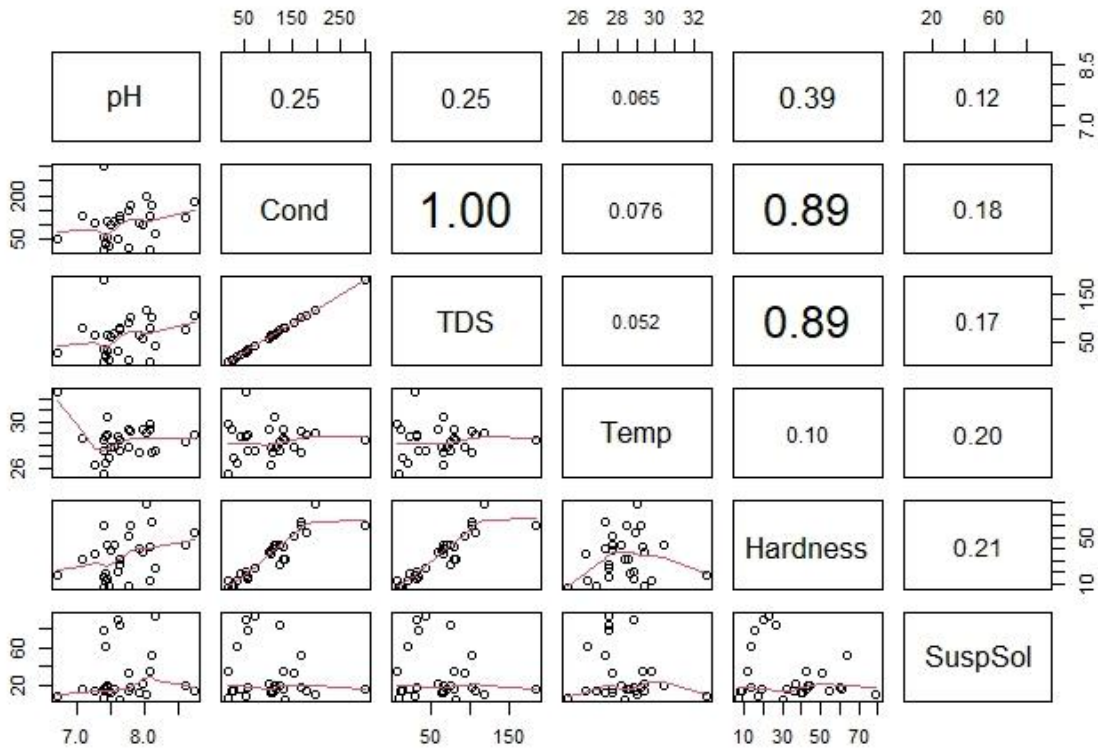
	Late monsoon season			Early monsoon season		
Variable	Min - Max	Mean (SD)	Median	Min - Max	Mean (SD)	Median
Aquatic vegetation cover %						
Submerged	0.0 – 70.0	12.3(18.1)	10	0.0 – 70.0	12.3(18.1)	10
Emergent	0.0 – 60.0	5.9(12.3)	0	0.0 – 60.0	5.9(12.3)	0
Floating	0.0 – 90.0	12.9(27.7)	0	0.0 – 90.0	9.10(22.9)	0
Open water	10.0 – 100.0	67.1(30.3)	75	10.0 – 100.0	70.9(28.8)	80
Floating algae	0.0 – 30.0	1.1(0)	0	0.0 – 30.0	1.1(0)	0
Woody/leafy debris	0.0 – 20.0	0.7(0)	0	0.0 – 20.0	0.7(0)	0
Water parameters						
pH	6.7 – 8.9	7.8(0.5)	7.8	6.88 – 8.57	7.53(0.51)	7.3
Conductivity (us/cm)	14.9 – 282.9	100.79 (63.4)	102.9	11.7 – 314.2	104.26(71.8)	104.5
Salinity (ppt)	0.0 – 0.1	0.1(0.1)	0.05	0.1	0.1(0.1)	0.05
DO (%)	12.8 – 123.5	61.0(34.3)	60.05	-	-	-
Total dissolved solids (TDS)	9.1 – 174.2	61.5(38.7)	63.75	7.80 – 189.8	63.80(43.5)	65
Temperature (˚C)	25.4 – 32.6	28.41(1.5)	28.3	25.5 – 30.0	28.08(1.23)	28.2
Hardness	7.6 – 66.4	33.7(16.54)	31.6	3.6 – 92.0	32.75(23.9)	34.2
Total suspended solids(mg/l)	2.0 – 129.0	32.0(34.1)	18.5	2.0 – 125.0	28.2(31.73)	16

B3 Kruskal Wallis test for water quality variables with non-normal distributions

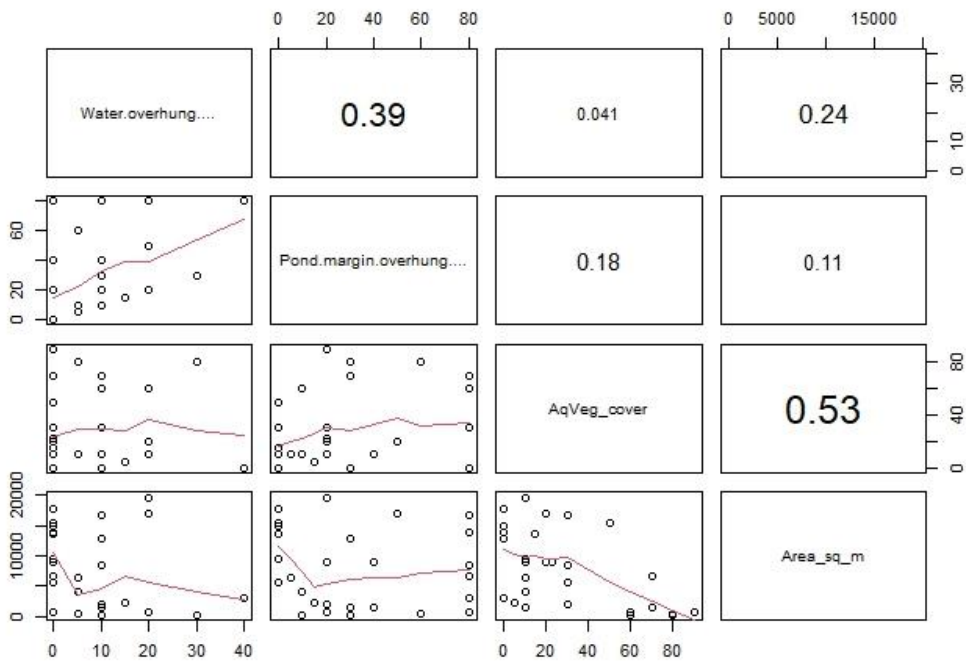
Parameter	Chi-squared	df	P value
Conductivity	0.0003	1	0.98
TDS (total dissolved solids)	0.0007	1	0.97
TSS	0.616	1	0.43

B4 Correlations among environmental variables

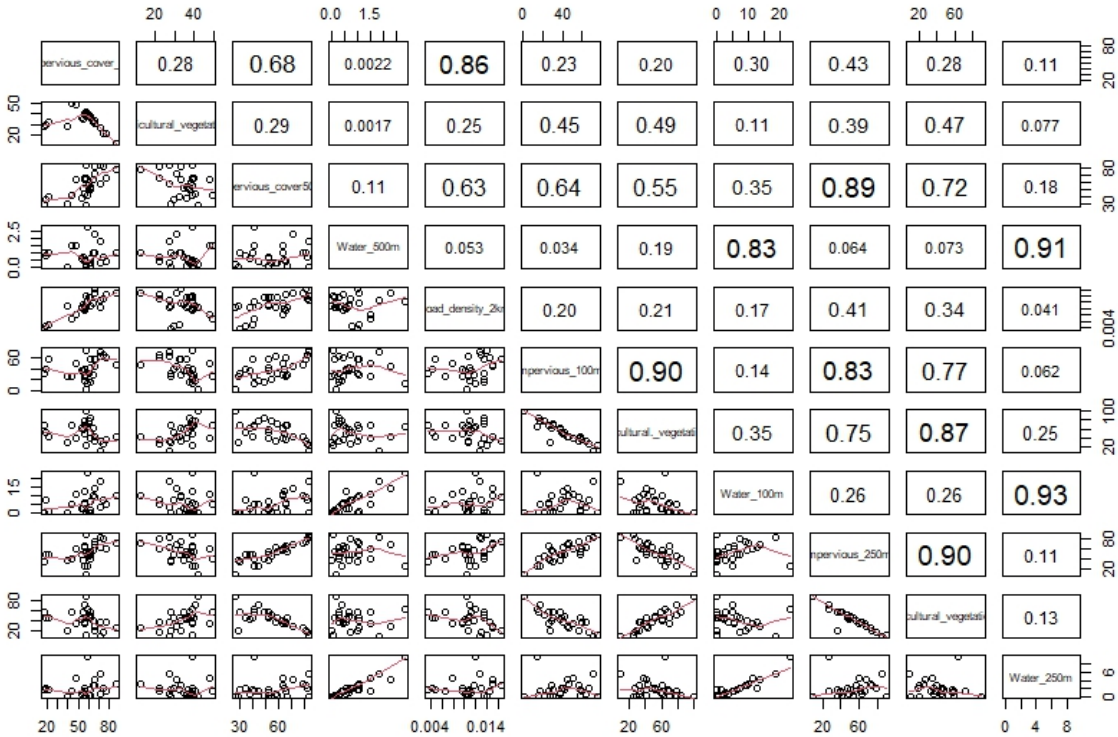
a. Water parameters (mean)



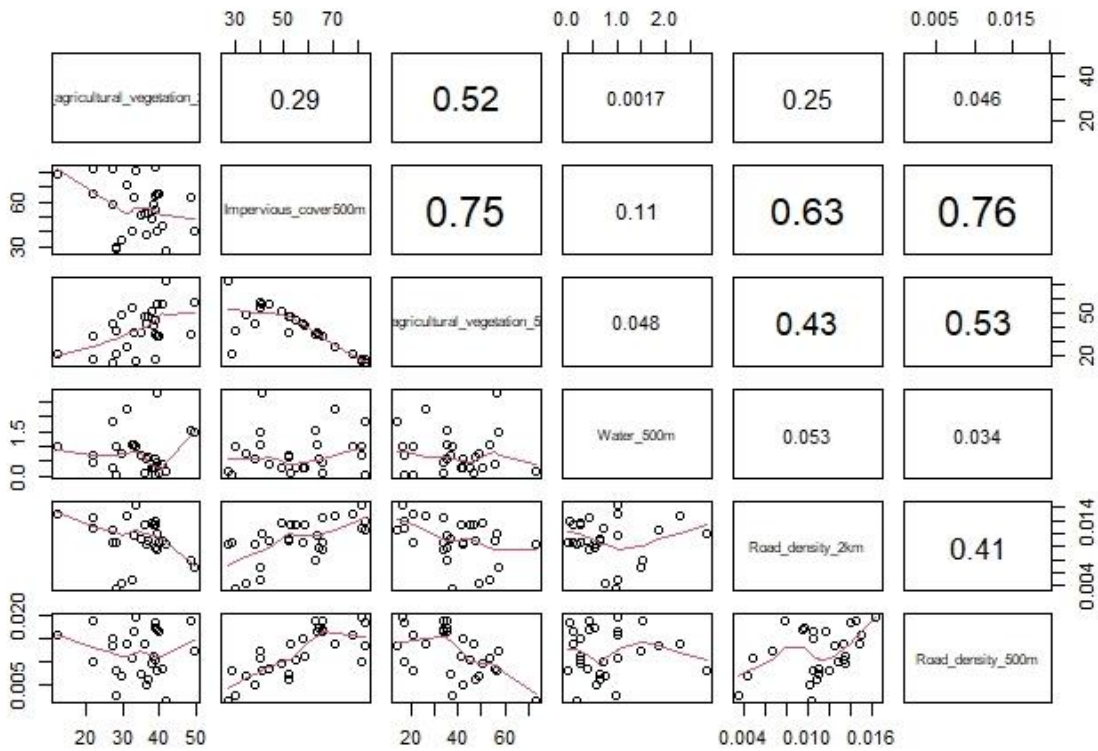
b. Physical features



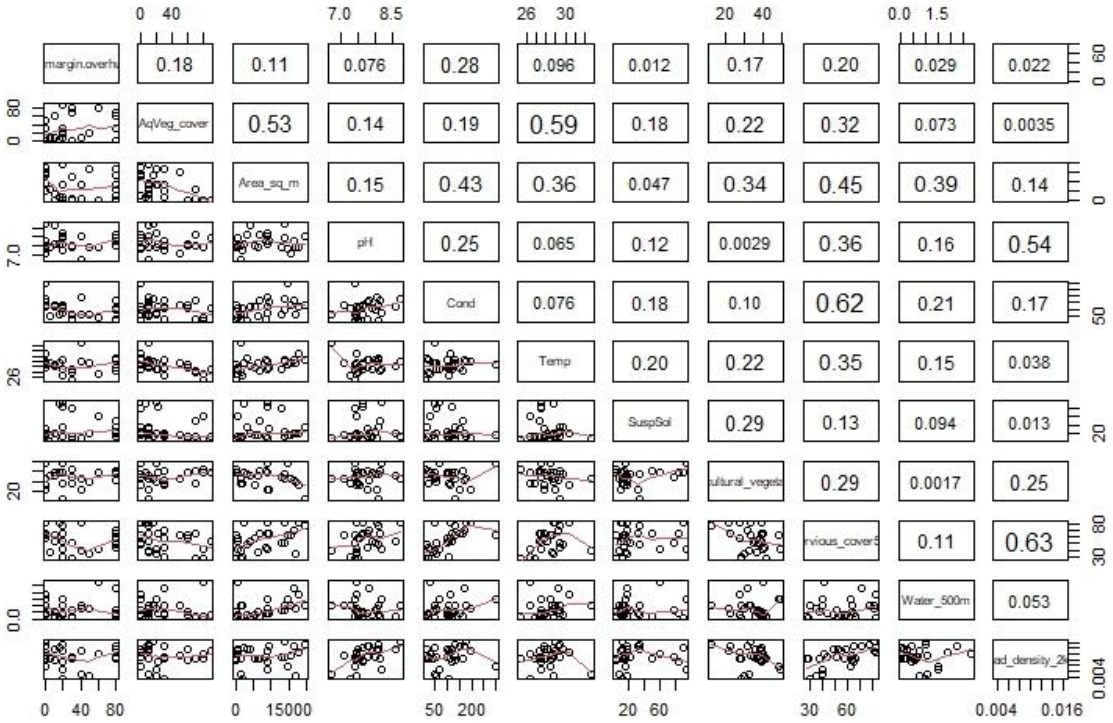
c. Surrounding land use (2km, 500m, 250m and 100m)



d. Surrounding land use (2km and 500m)



e. All retained predictors



B5 Summary of PCA for environmental data (standardized variables, variables with outliers removed, and highly correlated variables, Pearson's r>0.7 removed)

	PC1	PC2	PC3	PC4	PC5	PC6	PC7	PC8	PC9	PC10
Eigenvalue	3.266026	1.751789	1.633737	1.353586	1.064776	0.841381	0.637841	0.465048	0.458196	0.296527
Proportion Explained	0.272169	0.145982	0.136145	0.112799	0.088731	0.070115	0.053153	0.038754	0.038183	0.024711
Cumulative Proportion	0.272169	0.418151	0.554296	0.667095	0.755826	0.825941	0.879095	0.917849	0.956032	0.980742

B6 Relative abundance of taxonomic orders across two seasons

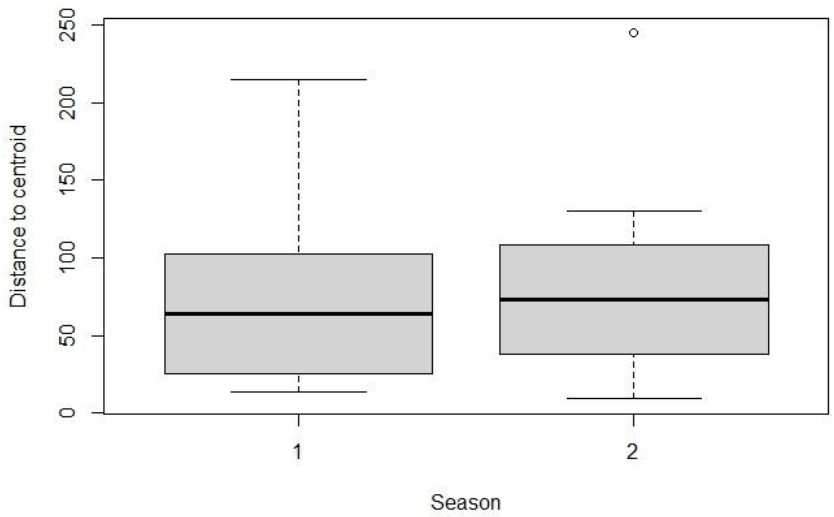
Order	Abundance	Relative abundance
Caridea	1028	0.09
Odonata	525	0.05
Diptera	2252	0.2
Ephemeroptera	1757	0.16
Heteroptera	2994	0.27
Coleoptera	104	0.01
Lepidoptera	279	0.03
Gastropoda	2142	0.19

B7 Group variance for multivariate seasonal differences in water parameters: a) average for each season; b) boxplot comparing multivariate dispersion for the two seasons

a)

	Late monsoon	Early monsoon
Average distance to median	68.72	74.33

b)



B8 Transformation based PCA (unconstrained) results for community composition data (Chapter 4)

	PC1	PC2	PC3	PC4	PC5	PC6	PC7	PC8	PC9	PC10
Eigenvalue	0.101253	0.078091	0.057658	0.053604	0.038358	0.034875	0.032331	0.029514	0.025176	0.022167
Proportion Explained	0.171184	0.132025	0.097479	0.090625	0.064849	0.058961	0.05466	0.049898	0.042564	0.037477
Cumulative Proportion	0.171184	0.303209	0.400687	0.491312	0.556162	0.615123	0.669784	0.719682	0.762246	0.799723

B9- Summary of GAM with abundance as response variable modelled separately with each predictor. GAM was performed with negative binomial distribution and REML method

Response variable: Abundance (n=27)				
Predictor	**edf**	**p value**	**adj.R^2**	**Deviance explained**
Shade	1	0.13	0.07	7.72%
Aquatic vegetation cover	1	0.1	0.07	8.08%
Area	1	0.02	0.12	14.3%
pH	3.1	0.15	0.13	24.7%
Conductivity	1	0.9	-0.04	0.002

Temperature	1	0.03	0.12	13.4%
TSS	1	0.2	0.01	4.63%
Vegetation 2km	1	0.9	-0.03	0.08%
Impervious cover	1	0.4	-0.01	2.04%
Water 500m	1.9	0.3	0.1	12.8%
Road density	1	0.4	-0.01	1.83%

B10 Summary of LCBD-nestedness and LCBD-turnover values for all ponds (n=28)

Response variable	Range	Mean	SD
LCBD-nestedness	-0.0014 – 0.106	0.035	0.03
LCBD-replacement	0.01 – 0.05	0.035	0.007

B11 Summary of regression models with LCBD (taxonomic) as response variable modelled separately with each environmental variable (df = on 1 and 26).

Predictor	**F**	**p value**	**Adj.R^2**
Shade	0.1497	0.7	-0.3
Aquatic vegetation cover	0.0087	0.9	-0.3
Area	0.34	0.5	-0.02
pH	0.18	0.6	-0.3
Conductivity	0.75	0.39	-0.009
Temperature	0.77	0.38	-0.008
TSS	0.25	0.6	-0.02
Vegetation 2km	1.25	0.2	0.009
Impervious cover	1.5	0.2	0.01
Water 500m	0.7962	0.3	-0.007
Road density	1.218	0.27	0.008

B12 Summary of regression models with LCBD-replacement as response variable modelled separately with each environmental variable

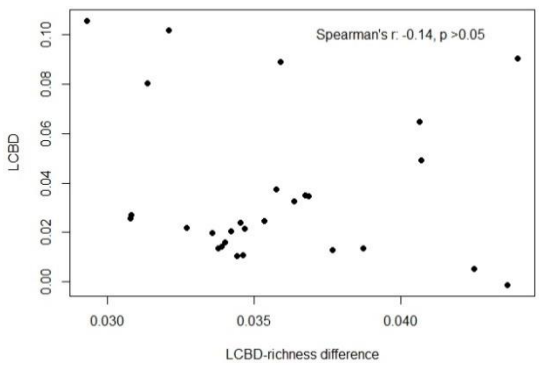
Predictor	**$F_{1,26}$**	**p value**	**Adj.R^2**
Shade	4.131	0.05	0.1039
Aquatic vegetation cover	0.40	0.5	-0.02
Area	0.004	0.9	-0.03
pH	0.16	0.6	-0.03
Conductivity	0.0005	0.9	-0.03
Temperature	0.07	0.7	-0.03
TSS	0.17	0.6	-0.03
Vegetation 2km	1.347	0.2	0.01
Impervious cover	0.007	0.9	-0.03
Water 500m	0.492	0.4	-0.019
Road density	1.062	0.3	0.002

B13 Summary of regression models with LCBD-richness difference as response variable modelled separately with each environmental variable (df = on 1 and 26).

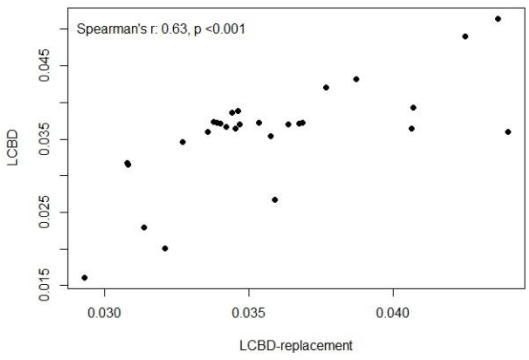
Predictor	F	p value	Adj.R^2
Shade	7.31	0.01	0.18
Aquatic vegetation cover	0.5372	0.47	-0.017
Area	0.119	0.73	-0.03
pH	0.03	0.86	-0.03
Conductivity	0.48	0.49	-0.01
Temperature	0.06	0.8	-0.03
TSS	0.0048	0.9	-0.03
Vegetation 2km	0.44	0.5	-0.02
Impervious cover	0.643	0.4	-0.01
Water 500m	0.103	0.75	-0.003
Road density	0.29	0.59	-0.02

B14 Scatterplots showing relationships between LCBD and (a) LCBD-nestedness, (b) LCBD-richness and between (c) LCBD-replacement and LCBD-nestedness

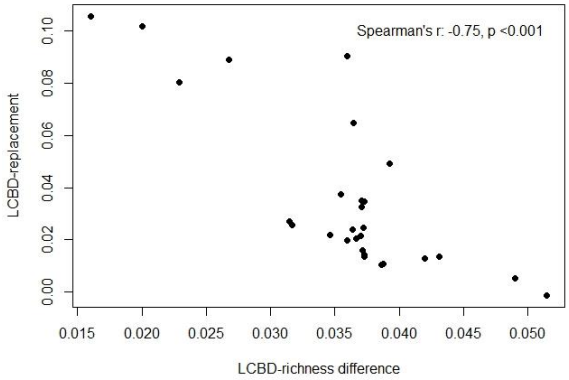
(a)



(b)

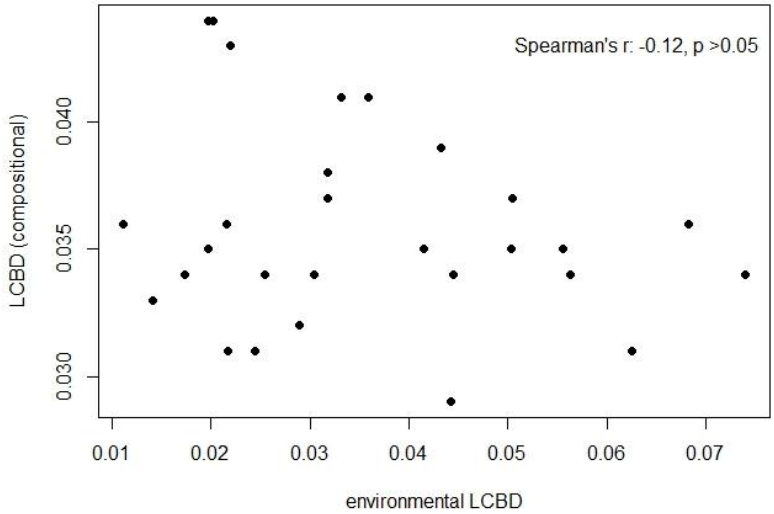


(c)

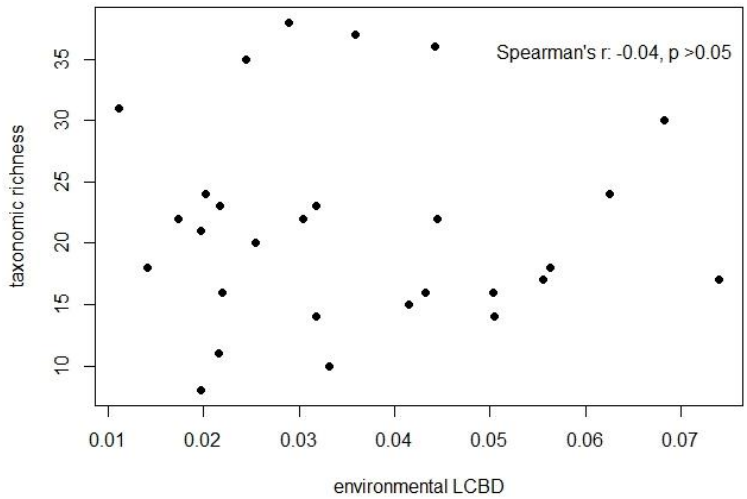


B15 Scatter plots of environmental LCBD and (a) taxonomic LCBD, (b) taxonomic richness and (c) Shannon's diversity index

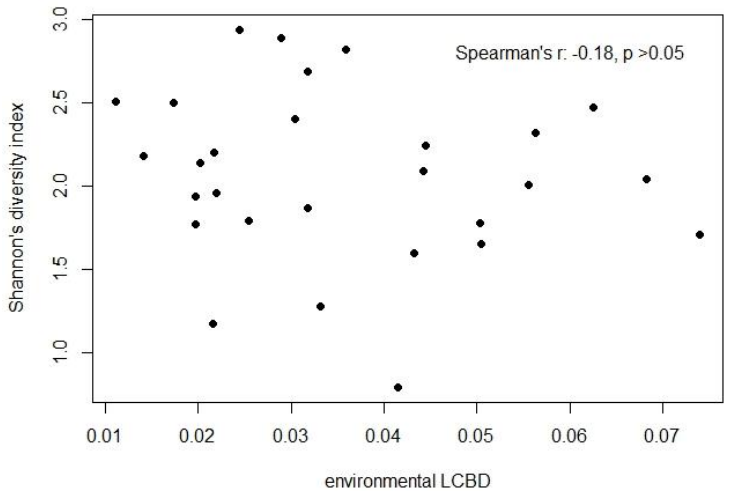
(a)



(b)



(c)


Spearman's r: -0.18, p >0.05
Shannon's diversity index
environmental LCBD
1.0
1.5
2.0
2.5
3.0
0.01
0.02
0.03
0.04
0.05
0.06
0.07

APPENDIX C

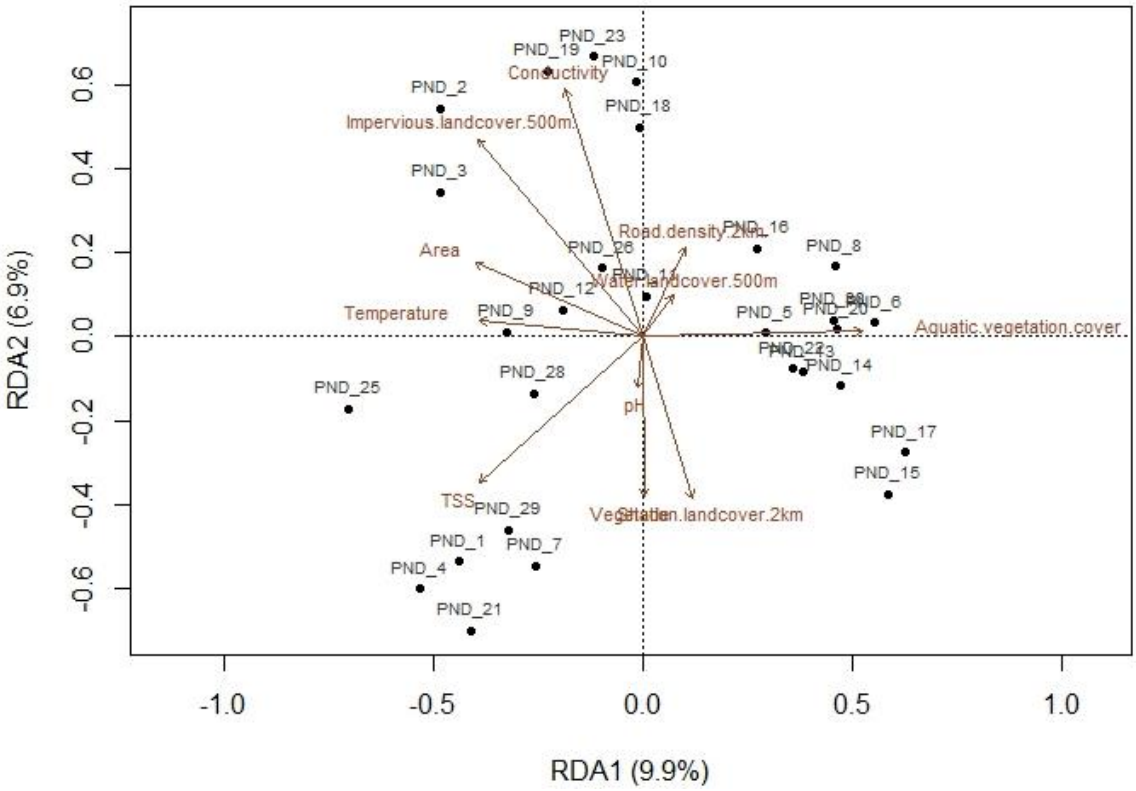
C1 Lists of taxa excluded or aggregated for trait-based (functional) diversity analyses

Taxa excluded from trait-based (functional) analysis for lack of trait information or low resolution identification				
Palaemonidae	Unidentified Ephemeroptera	Unidentified Diptera	Unidentified Gastropoda	Ampullariidae spp
Unidentified Zygoptera	Unidentified Heteroptera	Unidentified Coleoptera	Thiaridae spp	*Diplonychus rusticus*
Genera for which information was not available and so were assigned family level scores				
Agriocnemis	*Pseudagrion*	*Amerianna*	*Filopaludina*	*Rhagadotarsus*
Trithemis	*Urothemis*	*Hydrobasileus*	*Brachythemis*	*Acisoma*
Tramea	*Copera*	*Ctenipocoris*	*Synaptonecta*	*Dineutus*

C2 RDA summary for total functional beta diversity with 11 environmental variables as constraints

Response variable	adj.R^2	Df	F value	Pr (>F)
Total beta diversity	0.01	11	1.04	0.32

C3 RDA plot showing distribution of pond communities with environmental constraints. Arrows represent the strength and direction of environmental variables. Circles represent pond communities characterized by trait composition.



C4 RDA summary for replacement component of functional beta diversity with 11 environmental variables as constraints

Response variable	adj.R^2	Df	F value	Pr (>F)

Turnover component of beta diversity	-0.027	11	0.93	0.7

C5 Summary of simple linear regression models for functional LCBD with environmental variables as explanatory variables

Explanatory variable	Estimate	adj.R^2	$F_{(1,26)}$	p value
Shade	-0.00001	0.03	2.016	0.16
Aquatic vegetation cover	-0.00001	-0.01	0.698	0.4
Area	-0.0000	-0.03	0.008	0.9
pH	0.001	0.009	1.264	0.27
Conductivity	-0.000006	0.007	1.19	0.285
Temperature	-0.00015	-0.02	0.27	0.6
TSS	0.00002	0.08	3.462	**0.07**
Vegetation 2km	-0.00001	-0.03	0.04	0.8
Impervious cover	-0.0000	-0.03	0.0005	0.98
Water 500m	-0.0003	-0.02	0.4357	0.5
Road density	0.13	0.004	1.11	0.3

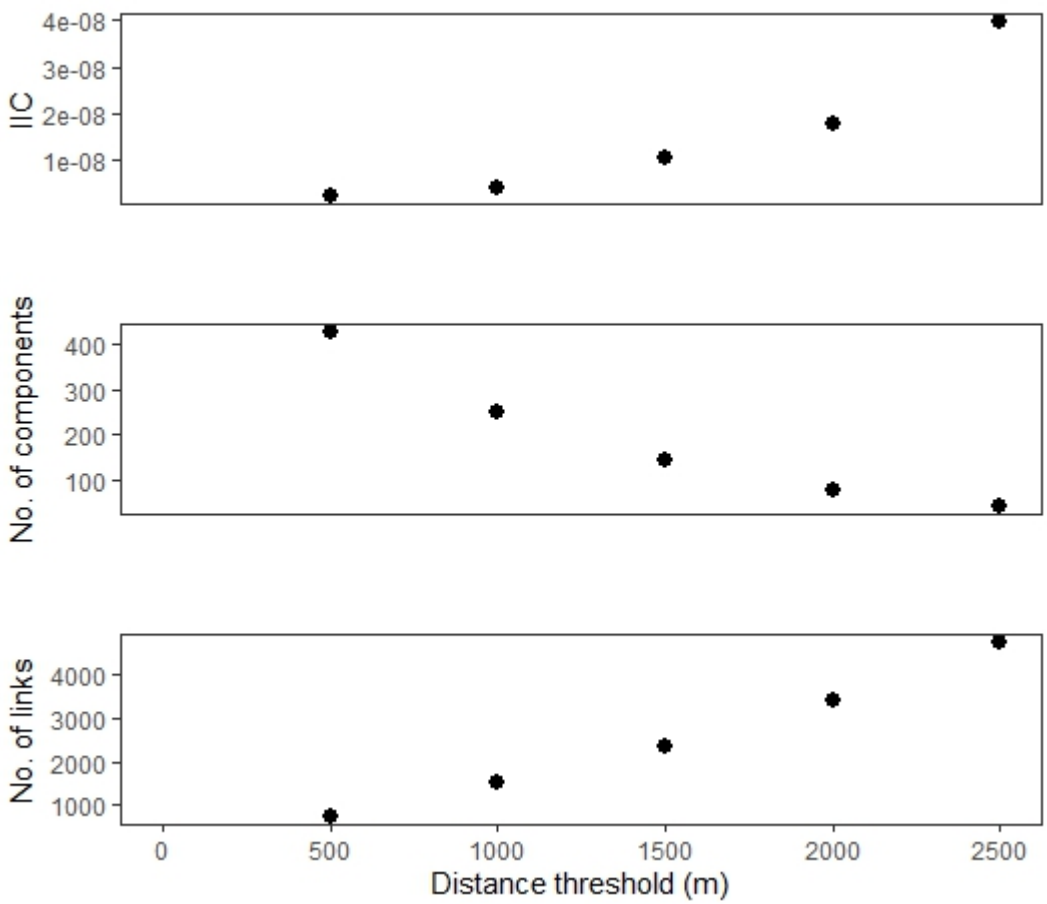
APPENDIX D

D1 Summary of Greater Kuala Lumpur land cover type and pond distribution by district/municipality

District/Municipality		Proportion of total land cover	Total no. of ponds	Pond density per km2
Kuala Lumpur	Non-agricultural vegetation	0.33	67	0.27
	Impervious	0.64		
	Agricultural	0.000024		
	Bare soil	0.01285		
	Water	0.014		
Petaling Jaya	Non-agricultural vegetation	0.34	36	0.36
	Impervious	0.64		
	Agricultural	0.001		
	Bare soil	0.007		
	Water	0.005		
Shah Alam	Non-agricultural vegetation	0.454	158	0.52
	Impervious	0.47		
	Agricultural	0.02		
	Bare soil	0.044		
	Water	0.01		
Klang	Non-agricultural vegetation	0.28	42	0.1
	Impervious	0.385		
	Agricultural	0.267		
	Bare soil	0.02		
	Water	0.0381		
Subang	Non-agricultural vegetation	0.4	53	0.32
	Impervious	0.56		
	Agricultural			
	Bare soil	0.0123		
	Water	0.023		
Sepang	Non-agricultural vegetation	0.32	103	0.327
	Impervious	0.24		
	Agricultural	0.37		
	Bare soil	0.037		
	Water	0.03		
Putrajaya	Non-agricultural vegetation	0.55	15	0.34
	Impervious	0.346		
	Agricultural	0.0008		
	Bare soil	0.013		
	Water	0.09		
Hulu Langat	Non-agricultural vegetation	0.55	140	0.2
	Impervious	0.21		
	Agricultural	0.2		

	Bare soil	0.025		
	Water	0.009		
Ampang	Non-agricultural vegetation	0.7		
	Impervious	0.28		
	Agricultural		16	0.114
	Bare soil	0.007		
	Water	0.011		
Selayang	Non-agricultural vegetation	0.6		
	Impervious	0.178		
	Agricultural	0.19	147	0.28
	Bare soil	0.015		
	Water	0.009		

D2 Change in landscape connectivity measures across increasing distance thresholds

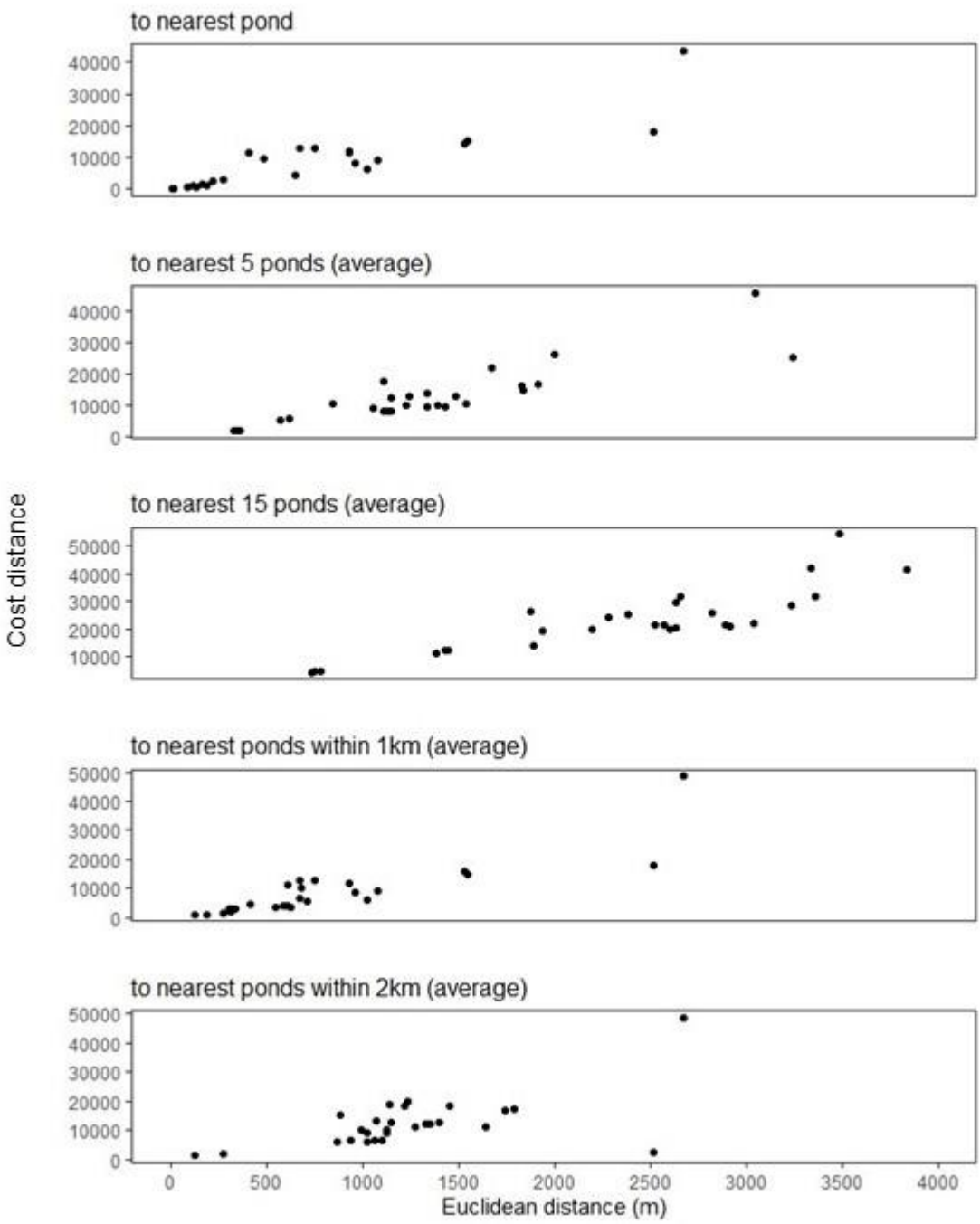


D3 Summary of connectivity measures individual ponds (n=777) in the GKL pond network, calculated for multiple maximum dispersal distance thresholds

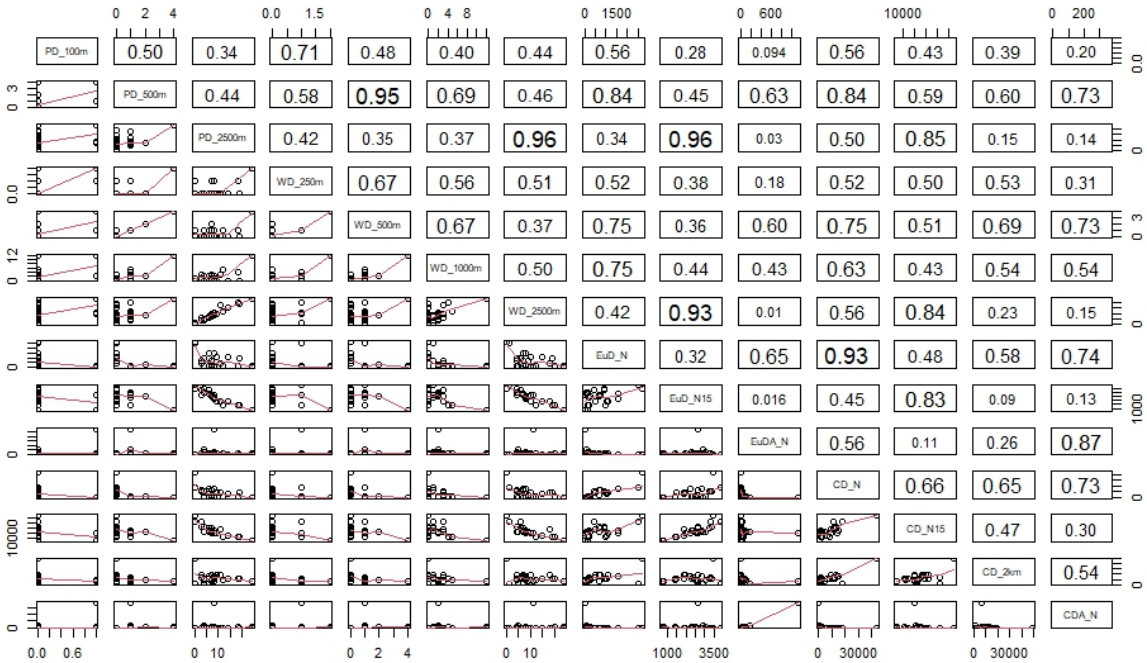
	500m	1000m	1500m	2000m	2500m
BC					

Mean (SD)	7.4e+06 (2.8e+07)	2.98E+07 (7.78E+07)	1.64E+08 (3.88E+08)	3.95E+08 (7.79E+08)	7.93e+08 (1.19e+08)
Range	0.00 -2.72E+08	0.00 - 6.10E+08	0.00 - 3.04E+09	0.00 - 6.29E+09	0.00 - 7.40E+09
d-IIC					
Mean (SD)	0.0018 (0.002)	0.002 (0.002)	0.0035 (0.008)	0.0028 (0.0039)	0.0034 (0.009)
Range	1.07e-07 - 0.0165	1.10e-07 - 0.016	4.23e-07 - 0.09	2.42e-07 - 0.04	1.10e-07 - 0.1496

D4 Relationship between Euclidean distance and Cost distance for distance to nearest pond, average to nearest 5 ponds, average to nearest 15 ponds and average for ponds within 1km and 2km.



D5 Correlation coefficients for selected connectivity variables



D6 Resistance multipliers assigned to each of the nine land use – land cover class of the Greater Kuala Lumpur area raster (The highly defined land use categories available in Thornhill (2013) were not available for the present study. Reference range refers to the range of values used in Thornhill (2013) for a range of subcategories in land use-land cover class

No.	Land use class	Reference range (Thornhill et al, 2013)	Multiplier adapted in present study
1	Impervious	500-1000	100
2	Major roads	300-500	50
3	Minor roads	300	30
4	Bare soil	250	15
5	Rubber plantation	100	10
6	Oil palm plantation	100	10
7	Other agricultural	100	10
8	Non-agricultural vegetation	10-85	5
9	Water	1-2	1