The ecology of constructed ponds on the lower Waikato River floodplain: implications for waterfowl management

A thesis submitted in partial fulfilment

of the requirements for the degree

of

Master of Science

in Biological Sciences

at

The University of Waikato

by

Jeremy Garrett-Walker

_____

2014

THE UNIVERSITY OF

WAIKATO

Te Whare Wānanga o Waikato

Abstract

Numerous ponds have been constructed in recent years on the lower Waikato River floodplain yet many waterfowl populations are in decline. Overseas research highlights the importance of constructed ponds for waterfowl populations; however, no comprehensive research has been carried out in New Zealand. The overarching aim of this research was to investigate how the ecology of constructed ponds on the lower Waikato River floodplain influences waterfowl densities, community composition and juvenile productivity. The study involved 34 constructed ponds which were predominantly found around Lakes Waikare and Whangape, and the internationally significant Whangamarino wetland. Data were collected on the physicochemical, landscape and vegetation characteristics of the ponds, as well as macroinvertebrate and fish assemblages to determine relationships between and within abiotic and biotic pond factors. Waterfowl communities were observed four times between September and December 2013 to determine mean densities of waterfowl per hectare and the mean community composition of each pond. The waterfowl data were used to explore key relationships with abiotic and biotic factors.

Site hydrology was found to have a significant influence on macroinvertebrate and fish community composition. The relative abundance of macroinvertebrates was predominantly lower in degraded ponds which were characterised by water supplied by swamps that frequently flood. Benthic macroinvertebrate abundance was lower in temporary ponds, indicating pond permanence was important. Fish communities of frequently flooded ponds were characteristic of pest fish as a result of increased connectivity. Ponds with high pest fish biomass, especially koi carp (*Cyprinus carpio*), tended to be more turbid with relatively low macroinvertebrate abundance. The percentage of pond margin fenced had consistently high explanatory power for differences in community composition of macroinvertebrate and fish communities.

Food availability and physical pond characteristics were important for waterfowl. Higher waterfowl abundances were found on ponds with high food availability, larger areas, and high pond complexity. Waterfowl densities were higher on ponds with lower fish biomass which is likely a result of decreased competition for

macroinvertebrates as food. The suitability of a pond for waterfowl appeared to be species-specific. Broods were often encountered on ponds with large areas, high complexities, and increased marginal fencing.

This study has allowed for the development of a conceptual model of the relationships between pond attributes and waterfowl communities. The findings of this study indicate habitat heterogeneity at the landscape scale is important for waterfowl. Providing a network of heterogeneous ponds across the landscape will provide enough varied habitat to support diverse and abundant waterfowl communities, and should include ponds of varying sizes, shapes, depths, vegetation and hydrology. It is also important to construct permanent ponds with limited flood frequency. Excluding pest fish and livestock from ponds will likely increase waterfowl use, and brood occupancy, of ponds as a result of improved water quality and reduced competition for food with fish.

Acknowledgements

opportunities to take my mind off uni and relax. You are a great bunch of people to have in my life; the constant banter and laughter keeps me honest. Feet are firmly planted on the ground thanks to all you guys.

And the flatmates who have come and gone along the way. Courtney, your work ethic (and goofiness), inspired me to work hard and not leave it all to the last minute. Kris, sorry for beating you at FIFA so much. At least you tried. And good luck with Lilly-Rose! Martin, you were another terrific flatmate. However, I think it best to keep details of our antics away from these pages. Sarah, Josh, Vishal, and Suprya, thanks for being a great current crop of flatmates.

Finally, Amanda. Herewith are a thank-you and an apology. Sorry for being virtually absent over the past month or so of my thesis. But thank you so much for putting up with it. And thank you for helping me out on fieldwork when no one else was available, I'm sure it wasn't your first choice of activity to do, but you never complained. Very much appreciated. I'll buy you dinner.

It's currently gone 1 am. And it's the final week I have before I get this thesis printed. So if I've forgotten anyone, I apologise. Blame the lack of sleep and terrible memory. Hit me up and I'll shout you a beer.

Table of contents

List of figures

List of tables

Chapter 1

General Introduction

1.1 Wetland biodiversity

Approximately 6% of the Earth is covered by wetlands (OECD/IUCN, 1996) of which only 0.8% are freshwater systems (Dudgeon et al., 2006; Stendera et al., 2012). Wetlands play a key role in the survival of over 100,000 faunal species globally (Dudgeon et al., 2006; Keddy et al., 2009). They are particularly important freshwater ecosystems as they often have high biodiversity and species abundance (Cereghino et al., 2008; Karaus et al., 2005; Oertli et al., 2002; Scheffer et al., 2006; Williams et al., 2004; Williams et al., 2008), with many species of plants, birds, fish, and macroinvertebrates relying on wetlands for breeding and habitat provision (Beard, 2010).

Freshwater wetlands can have high levels of endemism (Benstead et al., 2003; Dudgeon, 2000; Dudgeon et al., 2006; OECD/IUCN, 1996; Beard, 2010). For example, the Yangtze and Mekong Rivers are each home to over 100 endemic Mollusca species (Dudgeon, 1999), while the Madagascar wetlands support numerous freshwater endemic organisms (Benstead et al., 2003). Endemic species can be vulnerable to extinction when exotic, invasive species are introduced into wetlands; for example, around 200 endemic species of the cichlid fish have been lost from Lake Victoria, Africa, as a result of the introduction of the Nile perch (*Lates niloticus*).

The Ramsar Convention on Wetlands definition and classification of wetlands includes artificial wetlands such as ponds and seasonally flooded agricultural land (Ramsar, n.d.). Ponds include farm dams and stock ponds, while agricultural land includes managed or grazed wet pasture. Ponds are considered important for maintaining biodiversity at a landscape scale (Scheffer et al., 2006) because the biodiversity they support is often disproportionately high for their size (Cereghino et al., 2008; Oertli et al., 2002; Scheffer et al., 2006). Oertli et al. (2002) found varying positive relationships between pond area and faunal diversity, with Odonata showing a positive relationship, whereas Gastropoda, Amphibia, and Coleoptera showed no association. Oertli et al. (2002) also found smaller ponds

had higher proportions of rare species, highlighting their importance for the survival of endangered species. It was also found one large pond had less biodiversity then a collection of small ponds totalling to the same area; similar relationships have been found between aquatic vegetation diversity and pond area (Helliwell, 1983).

Williams et al. (2004) compared the biodiversity of ponds, lakes, ditches, streams, and rivers in Oxfordshire and Wiltshire, England, and found that, although rivers had a greater mean number of invertebrate and wetland plant species, ponds had greater species abundance. It was also found that rare species were more common in ponds compared to all other wetland types (Williams et al., 2004).

1.1.1 Environmental services

Baron et al. (2002) defined different ecosystem services as either short-term or long-term. Short-term services include drinking water, food supply, flood control, purification of waste, and habitat for plants and animals; while long-term services involve the adaptive capacity of the ecosystem, as well as the sustained provision of the current ecosystem services (Baron et al., 2002). Wetlands are important for the role they play in a number of environmental services, such as: stabilisation of sediment, recharging water supplies, purifying water, carbon sinks, mitigating floods, as well as recreational resources such as gamebird hunting and maintaining habitat and biodiversity (Beard, 2010; Keddy et al., 2009). With water becoming one of the potential limiting factors to humans in the future, Keddy et al. (2009) determined that the most important service of wetlands is the provision of water.

Wetlands can purify a number of water types as plants absorb nutrients and reduce organic matter, and filter suspended solids and pathogens (Gottschall et al., 2007), reducing the potential for eutrophication (Kadlec and Knight, 1996; Mitsch et al., 2001). For example, Comin et al. (1997) found nearly 100% of nitrogen input from rice field irrigation networks was retained in restored wetlands in north-east Spain; dissolved organic nitrogen was the only form of nitrogen which still had high levels leaching into the Ebro River delta. The nitrogen uptake of plants and

sediment improves water quality, contributing to enhanced biodiversity in wetlands (Comin et al., 1997).

Wetlands, especially those on peat, are also effective carbon sinks (Beard, 2010; Keddy et al., 2009). The magnitude of carbon stored in wetlands over a multi-annual scale is unclear (Roulet, 2000). Short-term studies in Canada indicate the degree to which carbon is stored in wetlands varies widely, with some peat bogs releasing carbon in to the atmosphere (Burton et al., 1996; Jarvis et al., 1997; Neumann et al., 1994; Suyker et al., 1997). It is estimated that wetlands store 60% more carbon than forests, and over 2,500% more than agricultural lands (Roulet, 2000).

Wetlands also abate floods by retaining water (Keddy et al., 2009; Zedler, 2003; Potter, 1994). The storage of water in upstream wetlands can minimise the effects of floods by reducing and delaying downstream flood peaks (Keddy et al., 2009; Potter, 1994). Intact wetlands have a greater potential to store water; short-term water retention in drained wetlands results in water flowing downstream at greater volumes (Potter, 1994). For example, the flood in the US Midwest in 1993 would likely have been mitigated by the presence of restored wetlands (Hey and Philippi, 1995).

1.1.2 Threats and stressors

Environmental services help shape the human use of wetlands which have become key recreational areas supporting a range of activities including game hunting (Keddy et al., 2009; National Wetland Trust of New Zealand, 2005). However, humans are also the main cause of the large majority of wetland loss which has occurred at alarming rates over the last two centuries (Cui et al., 2012). Between 56-65% of wetlands in Europe and North America had been drained by 1985, compared with around 27% in Asia, 6% in South America and 2% in Africa (OECD/IUCN, 1996). This averages to a 26% loss in wetlands globally by 1985; although Cui et al. (2012) estimate less than 50% of global wetlands remain today. Between the 1780s and 1980s approximately 53% of wetlands were lost in the United States of America (USA; excluding island states) (Dahl, 1990). Little is known on the loss of wetlands in Pacific islands nations; however, losses are

estimated at 90% for New Zealand, and 27 and 89% in the Australian states of Victoria and South Australia, respectively (Moser et al., 1996).

One of the most significant anthropogenic threats to global wetlands is their drainage to increase land availability (OECD/IUCN, 1996). Drainage of wetlands can have negative impacts on local and regional ecosystems. Regional impacts include, reducing the water storage capabilities of wetlands in flooding events, as well as increased nutrient loading in the watershed (Steinman et al., 2003). For example, drainage of 40% of wetlands in the Lake Okeechobee watershed, Florida, resulted in the doubling of phosphorus concentrations entering the lake over a 30 year timescale (Steinman et al., 2003). Local impacts include decreases in biotic diversity which can rely on connectivity for gene flow and life-cycle completion (Dudgeon et al., 2006; Stendera et al., 2012). Wetlands have also been drained in an attempt to improve public health, in particular to reduce the incidence of malaria and schistosomiasis (OECD/IUCN, 1996).

The introduction of exotic faunal species can negatively alter wetland biodiversity and endemism (Benstead et al., 2003; Kelly and Dick, 2005; Rahel, 2002). Exotic fauna can be difficult to remove from wetlands as they are often highly successful generalists (Baron et al., 2002). Kelly and Dick (2005) sampled the Erne River catchment to determine the effects of the introduced predatory macroinvertebrate *Gammarus duebeni celticus* on benthic macroinvertebrate diversity. In sites where *G. d. celticus* had invaded, diversity decreased, including fewer pollution sensitive species. Introducing exotic fish to wetland ecosystems is likely to lead to biotic homogenisation (Rahel, 2002) due to replacement of native fauna (McKinney and Lockwood, 1999; Rahel, 2002). In the United States introduced species, notably the common carp (*Cyprinus carpio*), have increased the turbidity in wetlands which has led to extinctions of indigenous fish (Pimentel et al., 2000). It is estimated that introductions of exotic fish in the United States have caused 44 native fish species to become threatened or endangered, and a further 27 species to be significantly harmed (Wilcove and Bean, 1994).

1.2   Wetland re-creation

Davies et al. (2004) acknowledges the importance of pond construction as part of the conservation and enhancement of biodiversity in wetlands. Construction of ponds allows for the design of suitable landscape and habitat features aimed at increasing particular ecological values or services (Williams et al., 1998). At a landscape level, biodiversity is optimised when there is a high concentration of ponds with varying habitat types and physical attributes (Cereghino et al., 2008).

In the United Kingdom three key design features have been identified to increase biodiversity potential of constructed ponds (Davies et al., 2004). Firstly, water supply to the pond needs to be clean and unpolluted; 25% more species are found in clean, unpolluted ponds compared to ponds with low water quality (Davies et al., 2004). Secondly, ponds should be created near existing waterbodies to increase the chance of colonisation (Davies et al., 2004; Williams et al., 1998). Lastly, good physical structure and dynamic hydrology are important to promote species diversity (Davies et al., 2004; Williams et al., 1999).

Sanchez-Zapata et al. (2005) studied key design features of irrigation ponds that increased breeding waterfowl populations. Vegetation was an important feature of constructed ponds as brooding waterbirds use it as habitat, especially emergent vegetation which is used for nest building. The area of the pond was also a key design feature as larger ponds were able to accommodate more breeding waterbird pairs from a greater number of breeding species compared with smaller ponds (Sanchez-Zapata et al., 2005).

1.3   Wetlands in New Zealand

Historically, wetlands covered 10% of New Zealand's land area (Ausseil et al., 2011); however, only around 2% remain (Keddy et al., 2009; Moser et al., 1996; Hunt, 2007). Wetland loss in New Zealand has been greatest in the North Island where there are more pressures from population growth (Ausseil et al., 2011; Myers et al., 2013). Agricultural development and run-off are the primary drivers of the declining health of New Zealand wetlands (Parliamentary Commissioner for the Environment, 2004). This has resulted in many wetlands, especially in

lowland areas, becoming highly enriched with increased nutrient loads (McGlone, 2009), and decreased ecological health (Myers et al., 2013).

Small wetlands are particularly susceptible to drainage in New Zealand as dairy farming continues to expand and intensify (Ausseil et al., 2008). Since 1995, 63 small wetlands in the Taranaki region alone have been drained and remaining wetlands have experienced decreased ecological health (Taranaki Regional Council, 2009). Drainage of wetlands without council consent has been a significant issue in recent years in New Zealand (Myers et al., 2013); rules around wetland drainage in the Tasman District Plan had to be tightened after three wetlands of significance were drained without consent (Parliamentary Commissioner for the Environment, 2001). Recent advances in statutory acknowledgement of the importance of wetland ecosystems is hoped to decrease the rate of wetland loss nationally (Beard, 2010).

1.3.1   Wetlands in the Waikato

Wetlands in the Waikato River catchment covered 14% of land area at the time of European settlement. However, this has declined significantly to wetlands presently covering approximately 1% of the region, and wetlands have decreased in size (Beard, 2010).

Chapman (1996) identified wetland loss as one of the three major anthropogenic impacts on the lower Waikato River catchment. The Whangamarino wetland and Lake Waikare (Figure 2-1) showcase the delicate interaction between natural and human processes in a wetland. The Whangamarino wetland is one of six wetlands in New Zealand on the Ramsar list of protected sites (National Wetland Trust of New Zealand, 2005) due to its highly diverse and nationally significant flora and fauna. The Australasian bittern (*Botaurus poiciloptilus*), the swamp helmet orchid (*Anzybas carsei*), and the giant cane rush (*Sporodanthus ferrugineus*) are at risk of being displaced from continuing drainage and land use change (Beard, 2010). However, it is a remnant of a once much larger wetland ecosystem which was drained to make way for productive farmland (Hunt, 2007). Despite this loss of habitat, the Whangamarino wetland is still home to an estimated 20,000 waterbirds, 239 wetland plant species and 18 species of fish (Beard, 2010).

The introduction of exotic vegetation has altered the ecological integrity of wetlands in the lower Waikato River catchment (Beard, 2010), with the invasive crack willow (*Salix fragilis*) now dominating riparian vegetation (Champion, 1997; Clarkson, 2002). Willow trees rapidly colonise disturbed areas and out-compete native species by excluding low-growing native wetland plants through shade (Beard, 2010; Champion and Clayton, 2010). Wetlands have also been extensively invaded by macrophytes such as reed sweet grass (*Glyceria maxima*) and alligator weed (*Alternanthera philoxeroides*) which both proliferate where willows are absent (Beard, 2010). Less invasive exotic plants that have become abundant around the riparian zones of wetlands in the Waikato include primrose willow (*Ludwigia peploides* subsp. *montividensis*), parrot's feather (*Myriophyllum aquaticum*) and Mercer grass (*Paspalum distichum*) (Beard, 2010).

A diverse range of fish, birds, reptiles and invertebrates utilise Waikato's wetlands for spawning, nurseries, foraging, roosting and nesting (Beard, 2010). For example, the nationally threatened black mudfish (*Neochanna diversus*) is specialised for wetland conditions and was once widespread throughout the Waikato catchment (Lake, 2010). These fish are now in constant decline as wetland habitat is lost through drainage and flooding of ephemeral wetland habitat (Lake, 2010). It has been estimated that 25% of New Zealand's Australasian bittern population lives in Whangamarino wetland, and other rare fauna include the North Island fernbird (*Bowdleria poiciloptilus*) and spotless crake (*Porzana punctate*) (Beard, 2010). A newly discovered species of moth (*Houdinia flexilissima*) is only found living inside the stems of giant cane rush (Hoare et al., 2006). With the giant cane rush only being found naturally in four wetlands around the Waikato (Beard, 2010), it is significantly at risk, subsequently becoming a species of high conservation status (Hoare et al., 2006).

Native fauna, as well as flora, are at risk from exotic fauna through competition and predation (Beard, 2010). There are some introduced species which are seen as desirable for hunting and fishing purposes, such as mallard ducks (*Anas platyrhynchos*); however, the majority of introduced species are perceived as pests (Beard, 2010). The invasive gambusia (*Gambusia affinis*) is understood to outcompete and displace the threatened black mudfish, especially in constructed wetlands (Ling and Willis, 2005). Koi carp (*Cyprinus carpio*) is arguably the most

detrimental pest fish species found in the Waikato River catchment, and poor water quality and habitat degradation in the lower Waikato lakes and wetlands is partly attributable to their benthic feeding behaviour which causes sediment resuspension and increased turbidity of the water (Chapman, 1996; Hayes et al., 1992; Scheffer, 2004). Parkos et al. (2003) found koi carp were positively related to turbidity and suspended solids, and negatively associated with the abundance of macrophytes and invertebrates. Koi carp can also reduce invertebrate abundances by directly feeding on them and changing their habitat; while decreases in macrophyte abundance are likely due to the reduction in light penetration in the water column resulting from increased turbidity (Parkos et al., 2003).

1.4   Objective of thesis

Although considerable overseas literature has been published on wetland pond construction practices (Davies et al., 2004; Sanchez-Zapata et al., 2005; Williams et al., 2008), no comprehensive literature is available from New Zealand. Since most information comes from the northern hemisphere (Cereghino et al., 2008; Davies et al., 2004; Sanchez-Zapata et al., 2005; Williams et al., 1998; Williams et al., 1999; Williams et al., 2008), it has only limited relevance due to its differing climate, stressors and threats in comparison to New Zealand. There is also no certainty that optimum design features for maximising biodiversity in New Zealand wetland ponds will be the same as overseas. A better understanding of the relationships within and between biotic and abiotic features is required.

In recent years Fish and Game New Zealand waterfowl surveys and gamebird hunter reports have indicated some waterfowl species are declining in Waikato wetlands, highlighting a need to better understand factors limiting the productivity of waterfowl. Therefore, a key aim of this research was to determine why some ponds appear to be much more productive for waterfowl than others by identifying factor(s) potentially limiting adult and juvenile abundances in constructed ponds. It is hoped this research can be used to guide the future development/restoration of high quality wetlands in a cost-effective way. With Auckland/Waikato Fish and Game administering this project, and funding coming

from the Waikato Catchment Ecological Enhancement Trust (WCEET), the focus of this research needed to be within the lower Waikato River catchment.

The following general objectives were developed for this thesis:

1. Determine the influences of landscape, physicochemical and biological factors, including aquatic vegetation, on macroinvertebrate communities as an indicator of waterfowl food supply in constructed ponds;
2. Investigate the links between fish abundance and biomass, especially pest fish, and environmental conditions in constructed ponds;
3. Determine environmental and biotic factors associated with differences in adult and juvenile waterfowl abundance among constructed ponds.

1.5   Outline of thesis

This thesis comprises five chapters with the two main results chapters set out in the style of manuscripts for submission to scientific journals. Therefore, there is some repetition with other parts of this thesis, especially within the methods sections and study site descriptions. Chapter 1 reviews what is already known about wetland biodiversity and productivity, both globally and in New Zealand and sets out the objectives of the thesis. Chapter 2 presents an outline of the study area, including detailed physical descriptions of all sampling sites. Chapter 3 examines the linkages between, and influences of, landscape variables (including potential pond design features), physicochemical conditions, and aquatic and riparian vegetation on the macroinvertebrate and fish assemblages of constructed wetland ponds. Chapter 4 identifies the factors potentially influencing waterfowl productivity in these ponds and how the variables in Chapter 3 may influence this. The final discussion chapter summarises the main findings from Chapters 3 and 4, and discusses possible methods for constructing highly productive wetlands in the future. Raw data summaries are presented as appendices.

Chapter 2

Study area

2.1 The lower Waikato River catchment

The Waikato River is New Zealand's longest river, cutting a 442 km channel that drains 13% of the North Island (Daniel et al., 2011; Collier et al., 2010). Chapman (1996) identified a range of human induced impacts on the Waikato River system including power generation, flood control, agricultural development, exotic forest development, waste disposals, and exotic species introductions.

The lower Waikato River section, from the Karapiro hydropower dam to Tuakau, has experienced greater human impact compared with the upper sections of the river (Chapman, 1996). Large meat and dairy factories discharge into the river a few kilometres north of Hamilton City which is the largest contributor of urban runoff into the river (Chapman, 1996). Impacts also include the substantial loss of wetlands, spread of exotic plants and animals (both within the water, and along the river banks), and thermal discharge from coal-fired power generation (Chapman, 1996).

Lakes Waikare and Whangape, and the nearby 7,100 ha Whangamarino wetland, are located on the lower floodplains of the Waikato River catchment (Chisnall and Hayes, 1991; Daniel et al., 2011). Lakes Waikare and Whangape have surface areas of 34.4 km^2 and 14.5 km^2, respectively, although Lake Waikare's catchment (215 km^2) is roughly two-thirds the size of Lake Whangape's (300 km^2) (Chisnall and Hayes, 1991). Lake Waikare and the Whangamarino wetland are connected by the Pungarehu Canal.

Both lakes have agricultural catchments dominated by high-intensity dairy farming which has, along with the introduction of koi carp (*Cyprinus carpio*), resulted in the lake water quality degrading to a hypertrophic state (Daniel et al., 2011). Exotic fish have been introduced into the Waikato River catchment since 1871 and now account for 41% of the fish species in the catchment (Hicks et al., 2010). The introduced goldfish (*Carassius auratus*), catfish (*Ameiurus nebulosus*), and koi carp are all benthivorous feeders and are known to increase water turbidity through the resuspension of sediments (Chapman, 1996; Hayes et al.,

1992; Scheffer, 2004). These benthivorous fish are efficient at feeding on macroinvertebrates, such as Chironomidae, which reduces the food stores for native fishes and waterfowl (Hicks et al., 2010).

Most of the ponds constructed on the lower Waikato River floodplain have been developed for waterfowl habitat and hunter opportunities. There are also ponds in the region which are used for stock watering or treatment ponds for various water treatment facilities.

2.2 Selection of sites

All study sites were 'constructed ponds', considered to be any wetland or shallow pond that had been developed through engineering water retention, as well as any pond that has been extended, enlarged, or deepened in some way by humans. Thirty-one sites were selected in the Lake Waikare, Lake Whangape, or the Whangamarino wetland areas, along with an additional three sites on the northern outskirts of Huntly or near the township of Rangiriri (34 sites in total) (Figure 2-1). The sites covered ca. 27 km north-to-south distance, and ca. 20 km east-to-west. The additional Huntly and Rangiriri sites are still closely influenced by the Waikato River, but are ca. 9 km south of the western Lake Waikare sites, and ca. 11 km southeast of the Lake Whangape sites.

The following criteria were considered in site selection to enable investigation of the roles of these factors on waterfowl productivity:

Age, size and shape of wetlands

Representative coverage of ponds of different (i) ages (time since construction or most recent major modification where it at least doubled in size), (ii) sizes in terms of water surface area when fully inundated, and (iii) shapes (ranging from roughly circular to highly convolute) were selected.

Influence of fish

With pest fish being found extensively throughout the lower Waikato River catchment, the influence they have on wetland productivity was another factor of interest. Originally it was intended that equal numbers of sites would be selected

12

with and without large-bodied pest fish. However, this proved difficult to achieve as a result of the drought in the summer of 2012/13 when a number of proposed sites with pest fish dried out.

Waterfowl suitability

Four key waterfowl habitat identifiers were used as part of the initial site selection process: food availability, nesting habitat, loafing habitat, and overhead cover, as used by McDougall et al. (2009) in a similar study carried out in the East Coast region of New Zealand. Each of the habitat identifiers was given a ranking of 0 – 4; loosely translating to providing poor to good habitat (McDougall et al., 2009).

All sites used have been given a code which indicates their geographical position. For example, LWK_1 refers to site number 1 from near Lake Waikare (LWP = Lake Whangape; WGM = Whangamarino wetland; HUN = Huntly; and RAN = Rangiriri).

Table 2-1 shows a summary of their physical properties. For a complete table of the physical properties of the 34 sites included in this study refer to Appendix 1.
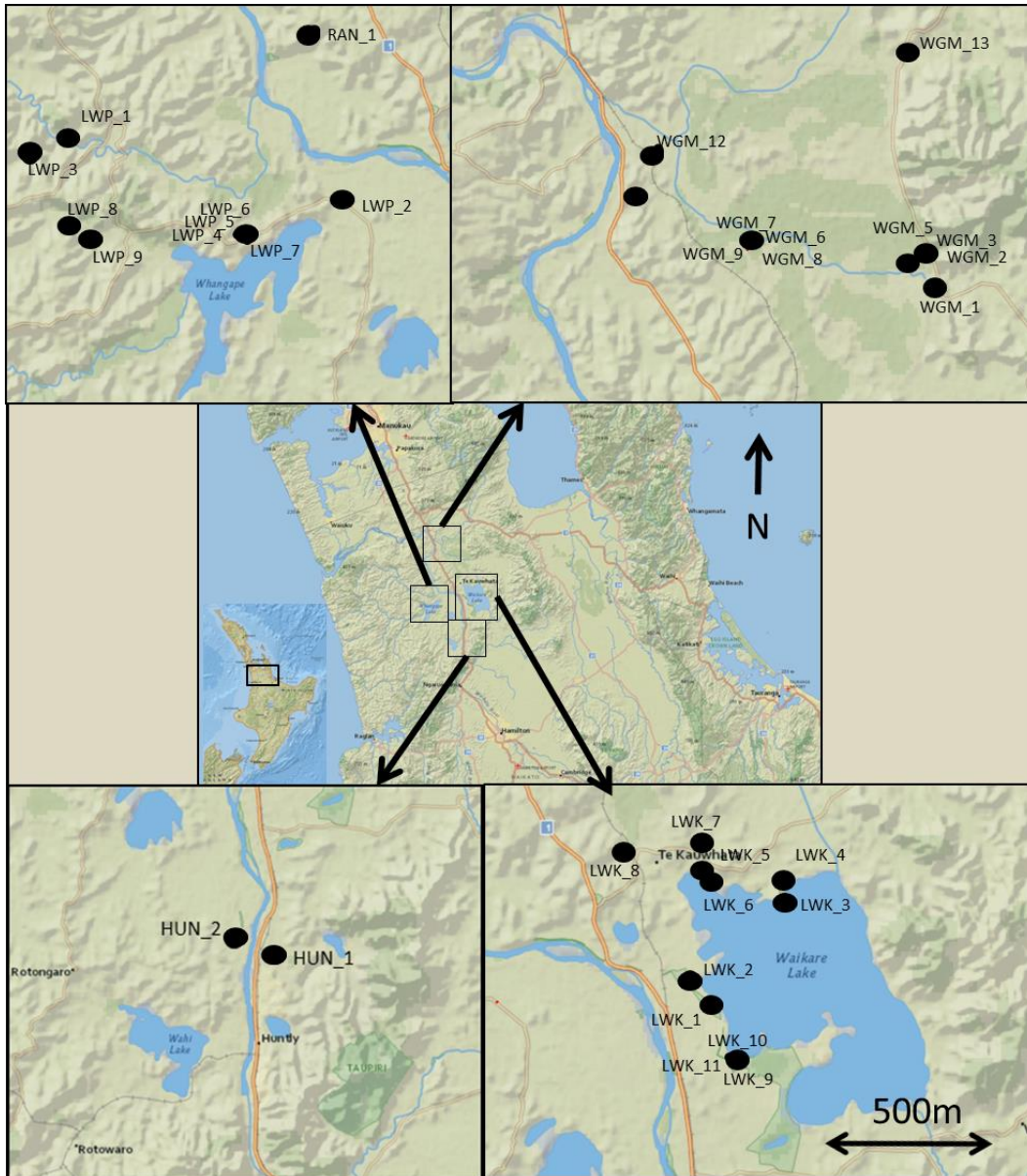
Figure 2-1: Location of sites in the Waikato region (centre image). Clockwise from top left: Lake Whangape (LWP; number of ponds $(n) = 9$) and Rangiriri (RAN; $n = 1$); Whangamarino wetland (WGM; $n = 11$); Lake Waikare (LWK; $n = 11$); and Huntly (HUN; $n = 2$). In some cases, where sites are in close proximity, a single black dot is used.

2.2.1 Lake Waikare (LWK)

For this research 11 ponds were selected from around the edges of Lake Waikare (Figure 2-1), ranging from ca. 12 m to 1,800 m away from the lake edge, with five on the western shoreline and six along the northern edge (Figure 2-2). The Lake Waikare sites ranged in size from 0.21 ha (LWK_9) to 4.06 ha (LWK_2) in area; LWK_9 and LWK_2 also had the shortest and longest shoreline lengths (218 m and 1156 m), respectively. Six of the ponds surrounding Lake Waikare were constructed from grazed farmland, whereas the remaining ponds were developed on marginal wetlands and existing wetlands. The pond sediment varied between sites, with peat being the dominant sediment in seven of the ponds, and silt/clay dominating in five of the ponds. One pond was younger than five years old, seven ponds were 6 – 10 years old, one pond was 11 – 20 years old, and two ponds were older than 20 years at the time of sampling.

2.2.2 Lake Whangape (LWP)

The nine sites around Lake Whangape (Figure 2-3) extend as far as ca. 5 km away from the lake edge (Figure 2-1). These sites included the largest and smallest of all sites used in this research; LWP_1 at 5.12 ha and LWP_5 with a surface area of 0.07 ha (Table 2-1). LWP_1 also had the longest edge length (1838 m) of all 34 ponds, and LWP_5 had the shortest (111 m). Seven of the Lake Whangape sites were constructed from existing grazed land, one from a marginal wetland, and one on the southern edge of the Opuatia Stream wetland. The pond found next to the Opuatia Stream was the only one originating from an existing wetland. At the time of sampling four of the ponds around Lake Whangape were younger than five years, three were 6 – 10 years old, and one was 11 – 20 years old. Only one pond was older than 20 years.

2.2.3 Whangamarino wetland (WGM)

In total 11 sites were used around the Whangamarino wetland (Figure 2-4), of which only three were on private land (Figure 2-1). The range of pond sizes was smaller than the Lake Waikare and Lake Whangape sites with the smallest and largest being 0.11 ha (WGM_3) and 0.64 ha (WGM_6), respectively (Table 2-1).

WGM_12 had the longest edge length at 669 m, while the shortest was 149 m (WGM_3). Four of the WGM sites (WGM_6, WGM_7, WGM_8, and WGM_9) were found on the same piece of Fish & Game land, and all were of a similar horseshoe shape. The rest of the sites ranged in shoreline complexity from virtual rectangles (WGM_3) to highly convolute (WGM_12). Nine of the Whangamarino wetland sites had peat as their dominant substrate. WGM_13 was the only pond which was not originally wetland; it was previously grazed land before pond construction. Only two Whangamarino wetland ponds were no older than five years, as well as only two being older than 20 years. Three ponds were 6 – 10 years old, while the remaining four ponds were 11 – 20 years old at the time of sampling.

2.2.4   Huntly (HUN) and Rangiriri (RAN)

The site north-west of Rangiriri (RAN_1; Figure 2-1) was the settling ponds used for the Te Kauwhata water treatment plant and consisted of two ponds separated by a small bund (for the purposes of this research considered as one site) and consequently had a high perimeter compared with area (330 m and 0.3 ha respectively) (refer to Table 2-1). This site was surrounded by grazed land; with stock being able to access the ponds in places. This site was older than 20 years.

HUN_1 was situated on the eastern side of the river, and HUN_2 was found on the western side (Figure 2-1). The Huntly sites had very similar edge lengths, with HUN_1 having an edge length of 1,212 m and HUN_2 being only 65 m longer at 1,277 m; however, HUN_1 had a surface area nearly double that of HUN_2 (5.2 ha and 2.8 ha, respectively). HUN_1 was located on Solid Energy land and HUN_2 was found on a private farm; both ponds were originally grazed land. HUN_1 was older than 20 years and HUN_2 was 6 – 10 years old.

Both HUN sites and RAN_1 were ponds surrounded by grazed pasture, and have a silt/clay base. Table 2-1 summarises the physical characteristics of the two HUN sites and RAN_1.

Figure 2-2: Photo images of a selection of Lake Waikare ponds. A = LWK_5; B = LWK_10; C = LWK_2; D = LWK_4.

Figure 2-3: Photographic images of a selection of Lake Whangape sites. A = LWP_3; B = LWP_6; C = LWP_2; D = LWP_8.

Figure 2-4: Photographic images of a selection of sites around the Whangamarino wetland. A = WGM_2; B = WGM_4; C = WGM_6; D = WGM_12.

Table 2-1: Basic physical properties of 34 constructed ponds on the lower Waikato River floodplain. Pond age: 1 = 0-5 years, 2 = 6 − 10 years, 3 = 11 − 20 years, 4 = 21+ years.

Site code	X coordinate (NZTM 2000)	Y coordinate (NZTM 2000)	Pond age (1-4)	Pond Area (ha)	Edge length (m)	Pond origin	Dominant substrate
LWK_1	1791670	5853730	2	1.87	1018	wetland	peat
LWK_2	1791080	5854530	2	4.06	1156	wetland	peat
LWK_3	1794050	5856920	2	0.81	511	grazed land	peat
LWK_4	1794010	5857550	2	3.12	757	grazed land	silt/clay
LWK_5	1791540	5857860	4	1.40	789	grazed land	peat
LWK_6	1791860	5857510	2	0.32	247	grazed land	silt/clay
LWK_7	1791460	5858660	4	0.49	416	grazed land	peat
LWK_8	1789100	5858500	3	0.31	312	grazed land	peat
LWK_9	1792440	5852080	2	0.21	218	marginal wetlands	peat, silt/clay
LWK_10	1792250	5852130	2	0.51	408	marginal wetlands	silt/clay
LWK_11	1792360	5852070	1	0.59	474	marginal wetlands	silt/clay
LWP_1	1775640	5856390	1	5.12	1838	marginal wetlands	silt/clay
LWP_2	1785170	5854020	2	0.27	273	grazed land	peat
LWP_3	1774470	5855850	4	3.57	1523	grazed land	silt/clay
LWP_4	1781610	5853030	2	0.27	258	grazed land	silt/clay
LWP_5	1781490	5853000	1	0.07	111	grazed land	silt/clay
LWP_6	1781730	5853110	2	0.36	293	wetland	peat
LWP_7	1781810	5852680	3	0.18	288	grazed land	silt/clay
LWP_8	1775850	5853540	1	1.51	874	grazed land	silt/clay
LWP_9	1776450	5852970	1	2.09	1016	grazed land	silt/clay
WGM_1	1793250	5864120	4	0.56	653	wetland	silt/clay
WGM_2	1793090	5865180	2	0.22	294	wetland	peat
WGM_3	1793000	5865290	2	0.11	149	wetland	silt/clay
WGM_4	1792540	5864910	2	0.25	364	wetland	peat
WGM_5	1792920	5865430	1	0.35	404	wetland	peat
WGM_6	1787490	5865800	3	0.64	422	wetland	peat
WGM_7	1787350	5866010	3	0.48	366	wetland	peat
WGM_8	1787190	5865560	3	0.26	259	wetland	peat
WGM_9	1787350	5865780	3	0.46	370	wetland	peat
WGM_12	1784350	5868770	4	0.29	669	wetland	peat
WGM_13	1790770	5855290	1	0.26	265	grazed land	peat
HUN_1	1791240	5844240	4	5.20	1212	grazed land	silt/clay
HUN_2	1790050	5844710	2	2.75	1277	grazed land	silt/clay
RAN_1	1784200	5859890	4	0.33	330	grazed land	silt/clay

Chapter 3

Influence of physicochemical conditions on macroinvertebrate and fish assemblages in constructed ponds of the lower Waikato River floodplain

3.1 Introduction

Environmental and physicochemical conditions, especially pond area and water chemistry, vegetation structure, and habitat heterogeneity, have been widely reported as having a significant impact on lentic macroinvertebrate abundances (Hannigan and Kelly-Quinn, 2012; Kelly and Dick, 2005; Nicolet et al., 2004). Fish populations are also driven by similar pond conditions and impact on macroinvertebrate communities (Batzer, 1998; Batzer et al., 2000; Gilinsky, 1984; Hanson and Riggs, 1995; Nummi et al., 2012; Wellborn et al., 1996).

Sanderson et al. (2005) identified water permanence as the driving factor explaining macroinvertebrate abundances, and others have identified frequency and duration of floods as key determinants of species richness in temporary and semi-permanent ponds (Ebert and Balko, 1987; Neckles et al., 1990). Neckles et al. (1990) manipulated water levels in three marshes and found there was a significant decline in macroinvertebrate abundances in areas of semi-permanent flooding compared with seasonal flooding. However, it was only the dominant macroinvertebrate taxa (Cladocera, Ostracoda, and Culicidae) that were affected; Dytiscidae, Corixidae, Chironomidae, Ceratopogonidae, and Ephydridae remained unaffected by flooding regime. The Neckles et al. (1990) study reinforced the conceptual model presented by Ebert and Balko (1987) where maximum species richness is found in waters where flooding and drying was frequent.

More recent studies have looked at multivariate links between hydroperiod, landscape and physicochemical factors, and macroinvertebrate abundances (Bazzanti et al., 1996; Bischof et al., 2013; Hannigan and Kelly-Quinn, 2012; Jeffries, 2003; Sanderson et al., 2005; Smith et al., 2003). A study examining the relative effects of different periods of water permanency on aquatic macroinvertebrate communities found only truly temporary ponds had a reduced

number of macroinvertebrate species (Collinson et al., 1995). Permanent and semi-permanent ponds supported, on average, 35 and 37 species respectively, compared to just 17 species in temporary ponds. However, Collinson et al. (1995) suggest the small datasets in many published studies cannot identify large-scale real-life gradients which would likely show a continuum between invertebrate community richness and pond permanency.

Nicolet et al. (2004) discussed water permanence as the dominant factor affecting macroinvertebrate composition only when there is a lack in pH variation in a pond.

> *"The most important environmental factor influencing biotic assemblages in temporary ponds was the water chemistry, and in particular alkalinity and pH. For macroinvertebrates, these results differed from a number of studies (Brooks, 2000; Collinson et al., 1995; Schneider and Frost, 1996; Williams, 1997), which showed that the most important environmental influence on the macroinvertebrate assemblage composition of temporary ponds is hydroperiod."* *(Nicolet et al. 2004, pp. 272)*

However, this comparison is between studies that did not include pH as an analysed variable (Brooks, 2000; Collinson et al., 1995; Schneider and Frost, 1996; Williams, 1997), and the quoted study (Nicolet et al., 2004) did not quantify pond permanence.

There is debate as to the respective significance of species-area relationships and habitat heterogeneity (Kallimanis et al., 2008; Ricklefs and Lovette, 1999; Shi et al., 2010). Habitat heterogeneity indicators vary for each organism (Ricklefs and Lovette, 1999); however, all describe the diversity of habitat at a site. Habitat heterogeneity is suggested to increase the overall macroinvertebrate abundance in ponds (Hannigan and Kelly-Quinn, 2012). For example, macroinvertebrate density can be influenced by pond area (Gee et al., 1997; Sanderson et al., 2005), and depth (Bischof et al., 2013; Hannigan and Kelly-Quinn, 2012). Gee et al. (1997) found pond age as having no significant influence on macroinvertebrate communities. The role of substrate size is argued both for (Nicolet et al., 2004) and against (Bischof et al., 2013) as a factor influencing the macroinvertebrate communities of ponds. There may also be impacts on macroinvertebrate assemblage from the surrounding land-use; for example cattle stocking rates did

not have a significant impact on aquatic invertebrate communities, whereas the type of surrounding pasture did in the study of Steinman et al. (2003).

Riparian vegetation can also influence habitat heterogeneity and it is often included in wider studies to identify factors affecting macroinvertebrate assemblages (Bischof et al., 2013; Gee et al., 1997; Hannigan and Kelly-Quinn, 2012; Jeffries, 2003; Nicolet et al., 2004; Sanderson et al., 2005). In a recent study, Bischof et al. (2013) found overhead vegetation, in the form of percentage canopy cover, was significant in explaining variations in macroinvertebrate composition. The accumulation of abscised leaves from riparian vegetation, along with other organic matter in ponds, may also impact macroinvertebrate communities as the decaying material can cause a decrease in dissolved oxygen (Gee et al., 1997; Neckles et al., 1990). Accordingly, Neckles et al. (1990) inferred from their results that a low level of detritus was required for high population growth.

At the taxon level, however, increasing canopy cover did not indicate a significant positive relationship with any taxa (Bischof et al., 2013). These results are similar to those of Gee et al. (1997) who found a negative relationship between canopy cover and a select few macroinvertebrate taxa (Odonata, Ephemeroptera, and Trichoptera). Sanderson et al. (2005) also found three taxonomic groups of macroinvertebrates strongly related to vegetation (Trichoptera, Mollusca, and Crustacea), and further narrowed it down to five species within the three groups. All of the five species were also strongly related to water permanence (Sanderson et al., 2005). Of the three studies mentioned above (Bischof et al., 2013; Gee et al., 1997; Sanderson et al., 2005), only Trichoptera was detected in more than one study as having a significant relationship with vegetation.

It has also been proposed that some vegetation may act as cover for invertebrate species from predation (Hannigan and Kelly-Quinn, 2012). A number of studies supports this proposal by reporting positive relationships between invertebrate biomass and the availability of macrophytes (Cyr and Downing, 1988; Hornung and Foote, 2006; Lillie and Budd, 1992; Jeffries, 1993). The difference in macroinvertebrate assemblages between open-water pools and small hollows was attributed to the cover of *Sphagnum* in the hollows (Hannigan and Kelly-Quinn, 2012). Aquatic vegetation can also act as protection for macroinvertebrates from

fish predation (Perrow et al., 1999; Stansfield et al., 1997). Studies have shown Cladocera actively seek the shelter of macrophytes in the presence of fish predation (Perrow et al., 1999; Stansfield et al., 1997). However, this positive relationship between fish predation and macrophyte-use as protection may cease at high fish biomasses (Schriver et al., 1995).

Invertebrate populations and communities can be significantly impacted by the presence of fish (Batzer, 1998; Batzer et al., 2000; Gilinsky, 1984; Haas et al., 2007; Hanson and Riggs, 1995; Nummi et al., 2012; Wellborn et al., 1996). Fish predation was found to have a different effect on macroinvertebrate densities depending on the season (Gilinsky, 1984). During autumn and winter the number of benthic macroinvertebrates increased in the presence of fish, whereas there were no significant differences during the spring and summer months. It was also noted that during the summer months, high fish predation decreased the number of epiphytic Chironomidae, while benthic Chironomidae increased during autumn and winter (Gilinsky, 1984). This finding is supported in more recent research carried out by Batzer (1998) and Batzer et al. (2000) who noted that Chironomidae can comprise around 61% of total food consumed by pumpkinseed sunfish (*Lepomis gibbosus*), brown bullhead catfish (*Ameiurus nebulosus*), black crappie (*Pomoxis nigromaculatus*), and common carp (*Cyprinus carpio*) in marshes (Batzer et al., 2000). The ability for epiphytic Chironomidae to increase in population size in the presence of fish may be down to the life-cycle and/or behavioural characteristics of the species (Batzer et al., 2000; Hershey, 1987; McPeek, 1990). Nummi et al. (2012) found the macroinvertebrate community abundance and the average size of dytiscid beetles decreased in the presence of perch (*Perca fluviatilis*). Other research indicates invertebrate abundance decreases in the presence of brook stickleback (*Culaea inconstans*) (Hornung and Foote, 2006).

3.1.1   Aims and objectives

The aim of this chapter is to determine which factors influence macroinvertebrate and fish community composition in constructed ponds of the lower Waikato River floodplain. There have been many studies carried out around the world looking at

the possible factors regulating macroinvertebrate and fish communities in ponds, but limited research in New Zealand where ponds are often constructed to enhance waterfowl productivity. Specific objectives of the study were to determine:

- The roles that pond permanence, flooding frequency and water supply have on macroinvertebrate and fish community composition and abundance in constructed ponds. Many constructed ponds serve as farm dams that are not permanent features of the landscape and dry out annually, while others are created for duck shooting in existing wetlands (e.g. the Whangamarino wetland) which can flood annually.

- The role played by vegetation, both aquatic and terrestrial, in shaping macroinvertebrate and fish communities. The literature suggests both overhead cover and the abundance of macrophytes have significant impacts on invertebrate abundances.

- Which other physicochemical characteristics of ponds are associated with changes in macroinvertebrate and fish community composition and whether findings on the lower Waikato River floodplains support the overseas research.

- Whether fish, in particular pest fish, can influence macroinvertebrate abundances in constructed Waikato ponds.

Based on the findings of the overseas literature, I hypothesise that flood frequency and pond permanency will significantly influence macroinvertebrate community composition, with flood frequency having a stronger effect on macroinvertebrates colonising substrates in the water column, and permanency having a stronger effect on benthic communities (Hypothesis 1). I also hypothesise that flooding and permanence will have similar effects on fish communities as changes in macroinvertebrate communities are likely to affect fish feeding habits (Hypothesis 2). I hypothesise that habitat characteristics, including landscape and physicochemical, will have a stronger influence on macroinvertebrate communities than they will on fish communities (Hypothesis 3).

3.2   Methods

3.2.1   Study sites

Sites for this study were selected to incorporate a wide spread of physicochemical conditions, aquatic vegetation diversity and abundance, and anticipated fish biomass. All sites are located on the lower Waikato River floodplain near Lakes Waikare and Whangape, the Whangamarino wetland, Huntly, and the Rangiriri Pa (Figure 2-1). The ponds were constructed from one year to over 20 years ago and range from locations that were originally grazed, to dammed springs/seepages, to sites that were excavated from an existing swamp area. All 34 sites are waterlogged for at least eight months of the year; 18 sites can dry out over the warmer summer months. Further details on the study sites are available in Chapter 2.

3.2.2   Physicochemical sampling

Water quality

Water temperature (^{o}C), dissolved oxygen (DO; as % and mg/L) conductivity (µS/cm) (all with YSI Pro2030) and pH (Eutech Instruments Waterproof pHTestr 10) were measured, and turbidity was visually assessed, in all ponds during winter (late June - early August, 2013), and then again in summer (December-January, 2013-14). All measurements were taken at three separate locations at each pond, and then averaged across locations and seasons.

In some cases where water depth was very low, the measurements (especially conductivity and DO) would begin to increase as a result of the pond sediment being disturbed; therefore, the lowest reading was recorded. The pH of the water was measured after a stabilisation period of at least 30 seconds.

A ranking system was used to determine the observed turbidity of the water based on McDougall et al. (2009): 1 = clear; 2 = moderately clear; 3 = moderately turbid and; 4 = turbid. Turbidity observations were made before entering the water, thus eliminating the influence of disturbed sediment.

Physical characteristics

Pond area (m^2) and edge length (m) were calculated using ArcGIS (ArcMap v10). The latest Waikato Regional Aerial Photograph Service (WRAPS 2012) was used as the base map for the Geographic Information Systems (GIS) calculations as this provided the most up-to-date images of the Waikato region. These images were taken over summer when water levels were expected to be low. Polygons were constructed around the edge of each pond on the WRAPS base layer to determine pond area and edge length. Island area (m^2) and island edge length (m) were also measured; island area was removed from the pond area measurements, whilst island edge length was added to the total pond edge length to provide an estimate of the extent of shorezone habitat. An area:perimeter ratio was calculated for each pond as an indicator of edge complexity.

Personal interviews with landowners and/or land administrators provided information on original land use prior to pond construction, how often the pond flooded and dried out, as well as the age of the pond. Flood inundation was determined according to the following rank: 1 = never; 2 = occasionally; 3 = annually. The pond permanence ranking systems was: 1 = never dries; 2 = rarely dries (20 year event); 3 = sometimes dries (5 year event) and; 4 = dries annually. The age of the ponds was organised into categories for the purpose of analysis, as follows: 1 = 0-5 years old; 2 = 6-10 years; 3 = 11-20 years and; 4 = 21+ years.

The percentage of pond margin, including islands, overhung by vegetation in late spring-early summer was estimated visually to the nearest 5%. Livestock access was rated as present/absent and the percentage of pond margin fenced off was visually recorded to the nearest 5%. On-site observations were used to determine whether there was a riparian buffer strip present around the perimeter of each pond as it was often unclear on the WRAPS images. Width of the buffer strip was approximated on-site at four different locations (northern, eastern, southern, and western sides of the pond) and then averaged to provide a single measurement of width. The percentage of open water with visible surface-reaching aquatic macrophytes was visually estimated to the nearest 5%. Dominant pond substrate was visually assessed and classified as peat or silt/clay.

3.2.3 Macroinvertebrate sampling

Field collection

Two sampling techniques were used to collect macroinvertebrates at three sites in each pond between July and August 2013. A 250 µm mesh net was used to sweep macroinvertebrates in the water column and on vegetation, while a mini Ekman grab (area = 273.6 cm^2; volume = 1217.52 cm^3) was used to sample macroinvertebrates in the pond benthos. All samples collected within habitats at each site were pooled for analysis as within-site variation was not considered in the analyses (i.e., for each site there was one sediment and one sweep sample).

The sweep sample was carried out according to Biggs et al. (1998), and involved sweeping a net for three minutes in proportion to mesohabitat area (e.g. open water, submerged macrophytes, near-shore, etc.) at each site to ensure a constant effort across all ponds. For example, if three different mesohabitats were identified then a 1-minute sweep was taken at each; if a mesohabitat took up a larger proportion of the pond than the others then the time was split to reflect this. The entire sweep sample was then placed in a 500 µm mesh sieve bucket, to sieve out as much fine material as possible. All samples from each pond were pooled and then placed into containers and preserved with 95-100% ethanol.

Sediment grab samples were collected in water no deeper than knee-depth (50 cm) and as evenly spread around the pond as allowed by access. Three grabs were collected at each site (i.e., nine samples per pond), and were immediately placed in a 500 µm mesh sieve bucket. After in-the-field sieving, the sample was placed in a container and preserved with 95-100% ethanol.

Laboratory processing

In the laboratory each sample was washed through 4 mm, 1 mm and 500 µm sieves. All invertebrates too large to go through the 4 mm sieve were removed and placed in a vial. Material caught in the 1 mm and 500 µm sieves was preserved together in 70% ethanol for future sorting.

To avoid bias, all samples were sorted by the same person following sample processing protocol P2 (Stark et al., 2001). Protocol P2 describes the 200-fixed count + scan for rare taxa macroinvertebrate sorting method. The sieved sample

was spread evenly across a white sorting tray and topped up with enough water to just cover the sample. The sorting tray was divided into 24, (6 x 4) 55 mm by 55 mm squares. A random number generator was used to randomly select a square to subsample. A cardboard square, the size of the subsample squares, was then placed around the selected subsample square, and material in this square was transferred to a petri dish via a plastic pipette to be sorted. This subsample was methodically searched and all macroinvertebrates were removed, identified to the taxonomic level shown in Table 4-3, and counted. Oligochaete worms often became fragmented; therefore, only whole worms were counted, resulting in an underestimation of abundance. The sorting process was repeated, square by square, until at least 200 individuals were counted; the counting continued past 200 until the corresponding square was completely sorted. The total number of squares counted was noted to determine the percentage of the entire sample sorted so that total abundances per sample could be estimated. After the 200-count was complete the sorter scanned the remainder of the sample for any rare taxa that had not been previously encountered to provide a complete list of taxa present.

3.2.4 Fish sampling

Two single wing coarse fyke nets and three fine mesh Gee minnow traps were set at each site overnight in autumn 2013. All fyke nets were 8 m in length, including a 5 m long leader, and 3 m long funnel, made with 4 mm mesh size. The leader was 0.6 m high, and the funnel had a 0.65 m opening. Gee minnow traps consisted of two halves which joined together to create a small cylindrical trap with inward facing funnels at each end. Each opening was 2.5 cm in diameter, made from the same 3 mm mesh as the main body of the net. All fish captured were measured for length (total length for eels, fork length for all other species; to the nearest mm), and then total weight (in grams) by species. All fish caught were kept alive during the processing and then released back into the pond in accordance to the Standard Operating Procedure's 6 and 7 of the University of Waikato Animal Ethics Committee. Ten minutes of electrofishing was also undertaken at each site, and all fish stunned were captured and measured (total length for eels, fork length for all other species; to the nearest mm).

The three fishing methods stated above were not effective at catching koi carp; therefore, where koi carp were observed the entire pond margin was walked and all visible fish were counted. This was undertaken in November and December 2013, when water levels had decreased, enhancing the visibility of koi carp around the margins. Estimates of koi carp numbers were then transformed into biomass using the mean weight of all fish previously caught by the University of Waikato's electrofishing boat until February 2010 (Hicks, 2014). The square root mean was used to calculate an estimated biomass of the koi carp visibly counted at ponds.

3.2.5 Statistical analysis

Environmental factors

Principal Component Analysis (PCA) was used to explore relationships in environmental factors, with the first three Principal Component (PC) axes being used to determine factor loadings of the variables. Three PC axes were used as they collectively explained > 50% of the variation among sites. All landscape and physicochemical variables were grouped together based on which PC axis they had the strongest coefficient with. Variables with coefficients < 0.3 were not considered in this assessment.

Community analyses

PRIMER 6 (v 6.1.15) software was used to create non metric multi-dimensional scaling (MDS) ordination plots. Separate MDS plots were created for macroinvertebrate abundance and fish abundance and biomass to indicate relative dissimilarity between communities. Communities with similar compositions are denoted by points relatively closer together in ordination space than communities which are dissimilar. Macroinvertebrate abundance data were converted into relative abundance prior to analysis so that sweep and benthic samples could be directly compared, and then fourth root transformed. Fish abundance was calculated as catch per unit effort (CPUE), while fish biomass was calculated as g CPUE^{-1}; these data were log(x + 1) transformed. Resemblance matrices were created using Bray-Curtis similarity coefficients. Five ponds with no fish present,

and one outlier were omitted from the analysis as Bray-Curtis resemblance matrices rely on samples with data greater than zero; LWK_9, LWK_10, LWK_11, LWP_8, and LWP_9 had no fish, and WGM_13 had three black mudfish (*Neochanna diversus*). Associations with landscape, water quality, and individual macroinvertebrate (for macroinvertebrate MDS plots) and fish taxa (for fish MDS plots) were assessed using vector overlays. A Spearman correlation coefficient of $r_s > 0.2$ was used for determining which variables were included in the vector overlays.

One-way PERMANOVA was used to test for significant variation in abundance and biomass community composition explained by different categorical factors. Factors tested were pond location, water supply, pond age class, flood frequency, and pond permanence. Pair-wise tests were carried out on factors which returned a $p < 0.05$ to determine which pairs of categories had significant differences in community composition. For the one-way and pair-wise testing 9,999 permutations of residuals were run under a reduced model method. Refer to Appendix 7 for the factors used in PERMANOVA and the groups used in pair-wise tests.

Step-wise distance-based linear models (DistLM) were used to test which predictor variables best explained the dissimilarities between macroinvertebrate relative abundance, and fish abundance or biomass. Step-wise DistLM adds variables which, at each step, improves the selection criteria. At each step it also tests whether removing variables improves the overall model, and stops running when adding and/or removing variables leads to no further improvements. R^2 was used as the selection criteria for the DistLMs, which indicates the proportion of variation explained by the cumulative fit of predictor variables. All variables used in the DistLM tests were $\log(x + 1)$ transformed excluding the rank variables. DistLMs were run using only landscape variables, and then all predictor variables to determine if landscape design features gave similar explanatory power as all variables. Each predictor variable included in the final DistLM model has a corresponding p-value, individual R^2 value, and a cumulative R^2 value. Each run of the DistLM analysis included 9,999 permutations and marginal tests. Refer to Appendix 8 for a list of the variables included in the DistLM tests.

Taxa abundances   32

Non-parametric Kruskal-Wallis tests were used to test for differences in abundance of macroinvertebrate and abundance and biomass of fish across factor classes (STATISTICA v11). This analysis tested the null hypothesis that the median of all taxa was the same, without the necessity of homogeneity of variance or normally distributed data (Dytham, 2005). All data were untransformed for this analysis as Kruskal-Wallis converts data into ranks to reduce the effects of outliers (Dytham, 2005); therefore, ponds which were removed for the MDS analyses were included. The same factors used in the PERMANOVA tests were used. Macroinvertebrate or fish taxa which returned significant ($p < 0.05$) differences in medians were displayed in boxplots.

3.3 Results

3.3.1 Environmental factors

A summary of environmental factors across the different locations is shown in Table 3-1. Principal Component Analysis (PCA) was carried out to determine which landscape/environmental features were the primary variables responsible for variation between sites. The first two axes accounted for 43% of the variation, with a total of 86% accounted for by seven axes (Table 3-2). The PCA plot with environmental overlay indicated ponds from around the Whangamarino wetland were positively influenced by pond temperature and turbidity, as well as flood frequency and buffer width (Figure 3-1). The Whangamarino wetland ponds were negatively associated with the distance to nearest waterbodies, a measure of connectivity. These ponds were predominantly found towards the lower right of the ordination, indicating they were positively associated to PC1, while negatively associated with PC2. The Lake Whangape ponds were positively associated with dissolved oxygen, ambient conductivity, and pond permanence. The two Huntly ponds, on the left hand side of the PCA ordination space, were positively associated with distance to the five nearest waterbodies, and pond area, edge length and number of islands. The Rangiriri pond had no association with either PC axis, while the Lake Waikare ponds were found throughout the PCA plot suggesting a high degree of heterogeneity. For the environmental characteristics of each pond refer to Appendix 1 and Appendix 2.

Table 3-1: Mean ± standard error of landscape and water quality variables of constructed ponds on the lower Waikato River floodplain. LWK = Lake Waikare, LWP = Lake Whangape, WGM = Whangamarino wetland, HUN = Huntly, RAN = Rangiriri. n = number of ponds. Probability value indicates significance of differences among site groups using Kruskal-Wallis test.

Variable	LWK (n = 11)	LWP (n = 9)	WGM (n = 11)	HUN (n = 2)	RAN (n = 1)	Overall (n = 34)	p
Area (m²)	11483 ± 3731	14869 ± 5971	3481 ± 492	37041	3245	11052 ± 2409	0.101
Edge length (m)	808 ± 195	737 ± 217	416 ± 49	1591	330	694 ± 97	0.221
Area:perimeter	14 ± 3	15 ± 3	9 ± 1	23	10	12 ± 1	0.108
No. of islands	2.6 ± 1.2	0.4 ± 0.3	0.8 ± 0.6	3	0	1.4 ± 0.5	0.069
Island area (m²)	962 ± 519	54 ± 50	49 ± 43	2719	0	501 ± 221	0.019
Island edge length (m)	235 ± 122	18 ± 14	23 ± 19	347	0	109 ± 44	0.027
Overhanging vegetation (%)	36 ± 11	19 ± 8	39 ± 13	35	0	32 ± 6	0.391
Fenced (%)	63 ± 12	62 ± 10	5 ± 5	58	15	42 ± 7	0.002
Buffer width (m)	30 ± 10	8 ± 4	161 ± 49	0	0	64 ± 20	0.009
Distance to 5 nearest waterbodies (m)	2051 ± 426	3058 ± 818	1052 ± 103	2194	3851	2056 ± 287	0.065
Age class	2 ± 0	2 ± 0	3 ± 0	3	4	2 ± 0	0.265
Temperature (°C)	17 ± 0	16 ± 1	18 ± 0	16	17	17 ± 0	0.114
Ambient conductivity (μS/cm)	160 ± 16	159 ± 23	185 ± 17	152	131	168 ± 10	0.809
Dissolved oxygen (mg/L)	7 ± 1	8 ± 1	7 ± 1	6	6	7 ± 0	0.43
pH	7.5 ± 0.1	7.1 ± 0.4	7.7 ± 0.1	7.4	7.8	7.5 ± 0.1	0.185
Turbidity (rank)	2 ± 0	2 ± 0	3 ± 0	3	1	3 ± 0	0.115

Table 3-2 Eigenvalues and amount of variation for each principal component (PC) axis for 34 constructed ponds on the lower Waikato River floodplain.

PC	Eigenvalues	% Variation	Cumulative % Variation
1	3.90	26	26
2	2.56	17	43
3	1.65	11	54
4	1.47	10	64
5	1.27	9	72
6	1.13	8	80
7	0.93	6	86
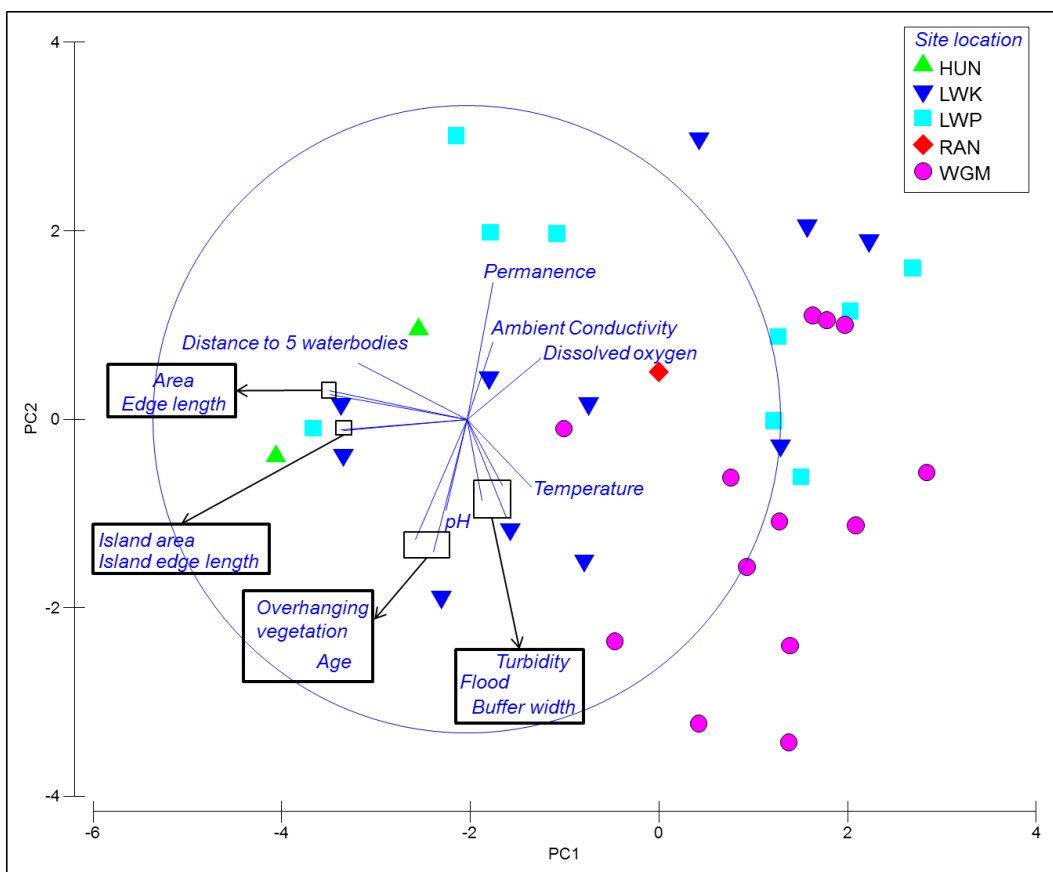
Figure 3-1: Plot of ponds along principal component (PC) axes 1 and 2 showing correlated environmental factors. HUN = Huntly ponds; LWK = Lake Waikare; LWP = Lake Whangape; RAN = Rangiriri, and; WGM = Whangamarino wetland.

The first PCA axis appeared to represent pond area and complexity, followed by distance to the five nearest waterbodies (Table 3-3). Permanence and pond age had the highest factor loading for PC axis 2 followed by two riparian variables, the extent of overhanging vegetation and the buffer width. Axis 3 was most strongly represented by the water quality variables ambient conductivity, dissolved oxygen and turbidity, and also by frequency of flooding.

Table 3-3: Factor loadings of each environmental variable for the first three principal component (PC) axes. The closer the number is to +1 or -1 the stronger the relationship to that corresponding PC axis. Values >0.3 are shown in bold.

Variable	PC1	PC2	PC3
Temperature ($^{\circ}$C)	0.205	-0.215	0.087
Ambient Conductivity (μS/cm)	0.083	0.245	**-0.581**
Dissolved oxygen (mg/L)	0.236	0.194	**0.320**
pH	-0.066	-0.291	-0.062
Turbidity rank	0.113	-0.210	**-0.414**
Area (ha)	**-0.437**	0.092	0.002
Edge length (m)	**-0.436**	0.078	-0.023
Permanence score	0.083	**0.437**	-0.185
Flood frequency score	0.050	-0.258	**-0.331**
Overhanging vegetation (%)	-0.163	**-0.382**	-0.029
Age class	-0.106	**-0.422**	0.291
Island area (m^2)	**-0.400**	-0.033	-0.178
Island edge length (m)	**-0.394**	-0.036	-0.192
Distance to nearest 5 waterbodies (m)	**-0.345**	0.179	0.233
Buffer width (m)	0.125	**-0.312**	-0.155

3.3.2 Macroinvertebrate communities

Initial analysis of the macroinvertebrate data indicated a clear difference between the sweep and benthic macroinvertebrate communities based on relative abundance (Figure 3-2). All the benthic communities were situated towards the left-hand side of the ordination plot, whilst the majority of the sweep (water-column) samples were on the right-hand side of the plot. The assumption the two community types were different was confirmed through ANOISM analysis; there was a highly significant difference (Global $R = 0.621$; p = 0.01).

A macroinvertebrate taxa vector overlay showed strong positive associations between Chironomidae and Oligochaeta taxa for benthic macroinvertebrate communities (Figure 3-2). Sweep macroinvertebrate communities were positively associated with Notonectidae, Corixidae, Odonata, Dytiscidae, and Acari. The mean abundance of macroinvertebrates per CPUE was nearly 24 times greater in sweep than in benthic samples which on average had two fewer taxa present per sample (Table 3-4).
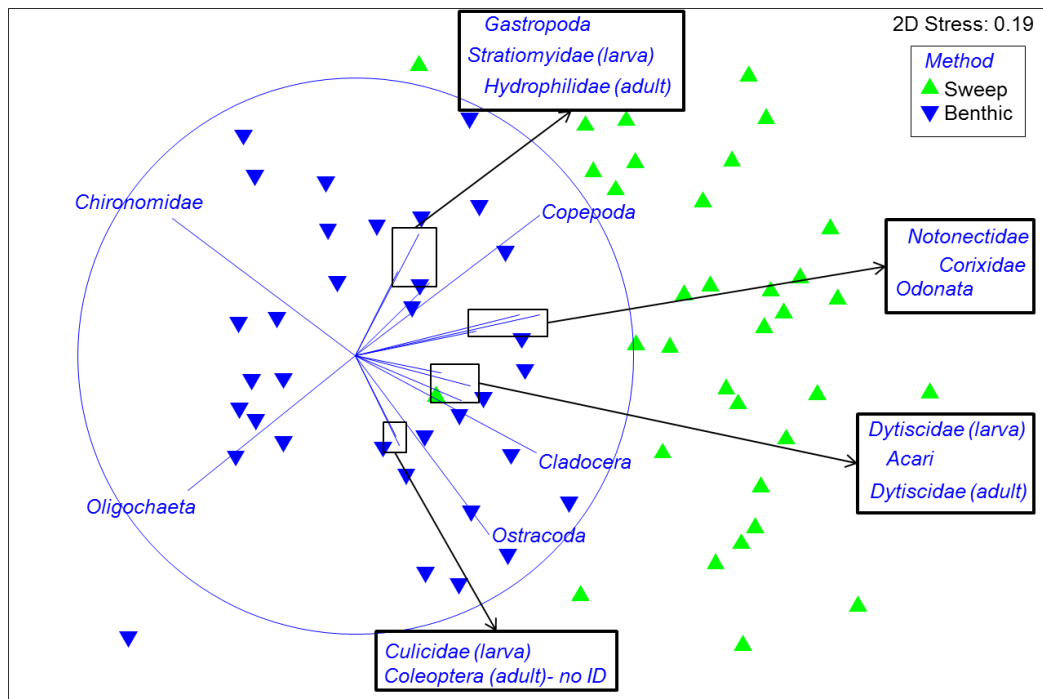
Figure 3-2: MDS ordination plot of macroinvertebrate communities, denoted by their collection method. Sweep = collected from substrates in the water column; Benthic = collected from pond sediment. Macroinvertebrate taxa vector overlay determined by a Spearman rank correlation coeffient r_s >0.3.

Table 3-4: Mean ± standard error of absolute abundance of sweep and benthic macroinvertebrate groups from 34 constructed ponds on the lower Waikato River floodplain. LWK = Lake Waikare, LWP = Lake Whangape, WGM = Whangamarino wetland, HUN = Huntly, RAN = Rangiriri. n = no. of ponds. Probability value indicates significance of differences among site groups using Kruskal-Wallis test.

Macroinvertebrate groups	LWK (n = 11)	LWP (n = 9)	WGM (n = 11)	HUN (n = 2)	RAN (n = 1)	Overall (n = 34)	p
Sweep samples			**No. per 3-minute sweep**				
Crustacea	7714 ± 4962	8354 ± 4513	775 ± 315	21888	28	6246 ± 2135	0.024
Coleoptera	16 ± 6	31 ± 19	7 ± 5	0	7	16 ± 6	0.722
Hemiptera	350 ± 230	255 ± 99	35 ± 11	24	22	194 ± 80	0.268
Diptera	364 ± 202	712 ± 248	385 ± 170	312	15	450 ± 108	0.213
Odonata	96 ± 54	8 ± 5	10 ± 4	48	120	42 ± 19	0.46
Other insects	6 ± 3	2 ± 2	3 ± 2	0	2	4 ± 1	0.318
Gastropoda	46 ± 38	19 ± 14	49 ± 34	0	264	44 ± 18	0.11
Oligochaeta	66 ± 38	13 ± 6	4 ± 2	0	0	26 ± 13	0.482
Other	63 ± 34	23 ± 13	7 ± 4	0	0	29 ± 12	0.136
Total	8721 ± 5054	9414 ± 4439	1276 ± 431	22272	458	7050 ± 2155	0.009
Benthic samples			**No. per 9 grabs**				
Crustacea	172 ± 165	63 ± 33	2 ± 1	1	7	73 ± 54	0.036
Coleoptera	0.5 ± 0.3	1 ± 1	0 ± 0	0	0	0.3 ± 0.2	0.535
Hemiptera	1.3 ± 1.1	0.2 ± 0.2	0 ± 0	0	0	0.5 ± 0.4	0.382
Diptera	115 ± 67	344 ± 96	29 ± 14	10	2	138 ± 39	0.007
Odonata	0.1 ± 0.1	0.5 ± 0.5	0.5 ± 0.3	0	1	0.3 ± 0.2	0.227
Other insects	0 ± 0	0.7 ± 0.5	0.2 ± 0.1	0	0	0.2 ± 0.1	0.516
Gastropoda	0.3 ± 0.2	3.7 ± 3.7	0.3 ± 0.1	0	0	1.2 ± 1	0.872
Oligochaeta	79 ± 40	218 ± 78	11 ± 2	2	1	87 ± 28	0.006
Other	1 ± 1	8 ± 4	1 ± 1	1	0	3 ± 1	0.067
Total	369 ± 224	637 ± 155	43 ± 14	13	11	303 ± 91	>0.001

3.3.2.1   Sweep samples

In total 26 macroinvertebrate taxa were identified in the sweep samples, including unidentifiable Coleoptera and Diptera adults and larva. Some taxa (e.g. Dytiscidae) with adult and larval individuals present in the sample were kept separate, taking the total number of taxa (adult and unidentified larvae) to 33. Four Crustacea taxa were identified, of which Cladocera, Copepoda, and Ostracoda were widely distributed, while Isopoda was only found at LWP_8. Coleoptera (Curculionidae, Dytiscidae, Scirtidae, and Hydrophilidae) were less abundant than the Crustacea. Dytiscidae were the most frequently collected Coleoptera, being found at 13 sites, and Curculionidae was present at one site. Three Hemiptera taxa were found in the sweep samples; Corixidae, Notonectidae, and Gerridae. Corixidae and Notonectidae were found at 29 and 25 ponds, respectively, whilst Gerridae was only present in two ponds. Of the Diptera taxa, Chironomidae was the most widely distributed, found in 31 ponds. Diptera relative abundances ranged from 0.4% in LWP_8 to 95% in WGM_12. The remaining Diptera taxa were located in no more than seven ponds each. Odonata were dominated by Zygoptera and found in 22 ponds; this taxon had a relative abundance below 6% at most sites. LWK_1, LWK_9, and RAN_1 had Odonata relative abundances of 42%, 29%, and 25%, respectively. Lepidoptera and Trichoptera were the other insects identified from the sweep samples, as well as five unknown insect larva found at LWK_1.

Gastropoda were also commonly found in ponds, with individuals being identified in 16 sites. The macroinvertebrate community in RAN_1 was over 50% Gastropoda, compared with less than 1% in five ponds. Oligochaeta were found in 15 ponds; their relative abundance was never above 7% in any pond but, as noted above, numbers were under-estimated due to fragmentation. Acari, Aranae, Collembola, and Hirudinea were the other macroinvertebrate taxa found in sweep samples. Acari were found in eight ponds, with the relative abundance never reaching above 4%. Aranae relative abundance was always below 1% in the five ponds where it was found. Of the six ponds with Collembola, only one had relative abundance above 1% (1.4%). Hirudinea was commonly encountered; it was found in 14 ponds with a relative abundance always below 4%. Refer to Appendix 5 for the abundance of sweep macroinvertebrates at each site.

Environmental relationships

Rain-fed ponds were positively associated with Acari, Ostracoda, Culicidae, and Cladocera, and negatively with Stratiomyidae, Hydrophilidae, Corixidae, and Diptera pupa (Figure 3-3C). There was a positive association between rain-fed ponds and Acari, while a number of macroinvertebrate taxa were less common in these ponds, including Chironomidae, Hirudinea, Notonectidae, and Gastropoda. Copepoda had a strong positive association with the swamp-fed ponds WGM_3, WGM_6, and WGM_7. Gastropoda was strongly positively associated with WGM_2 and LWP_1. Ponds with strong positive associations with Ostracoda and Cladocera were negatively associated to Diptera pupa, Corixidae, Stratiomyidae, and Hydrophilidae.
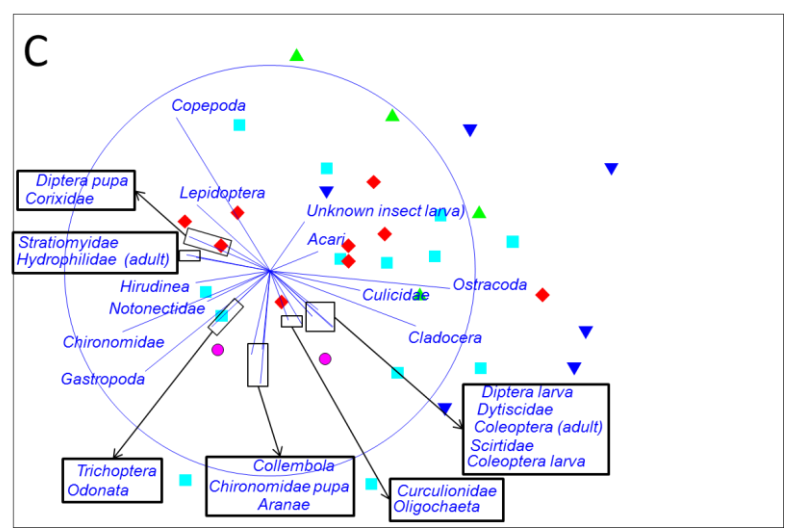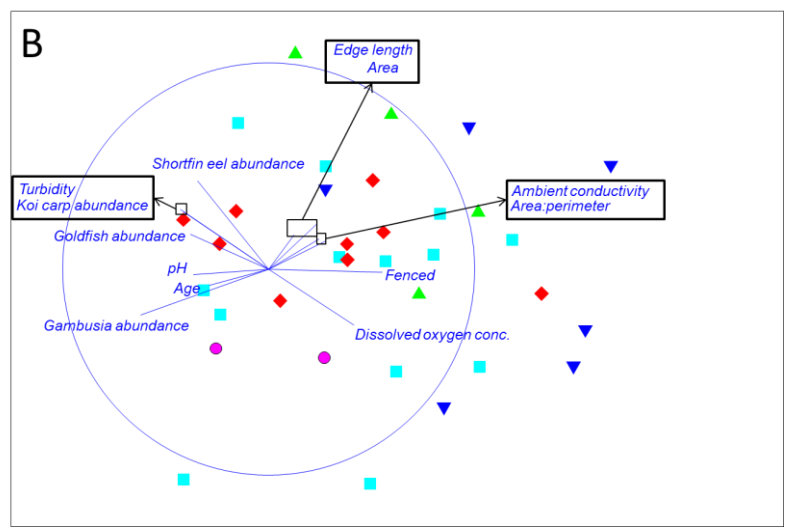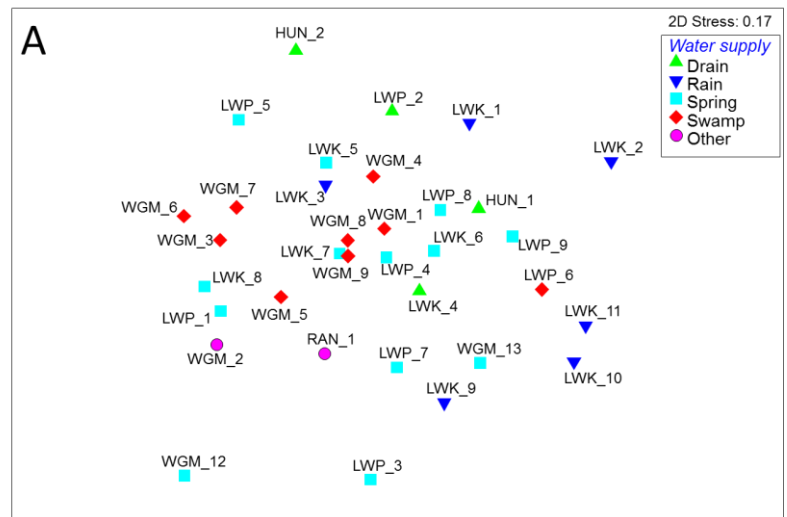
Figure 3-3: MDS plots of sweep macroinvertebrate community data with data point symbols determined by water supply class to the ponds. A = data points labelled by location (see Table 3-1). B = landscape, water quality, and fish abundance vector overlay. C = macroinvertebrate taxa overlay. Vector overlays calculated using a Spearman rank correlation coefficient r_s >0.2.

Initial PERMANOVA analysis indicated there was a significant difference between sweep macroinvertebrate communities and water supply, location, and flood frequency (Table 3-5). The sites around the Whangamarino wetland were significantly different from those around Lake Waikare and Lake Whangape. For water supply, there were significant differences between rain-fed and spring-fed ponds, and rain-fed and swamp-fed ponds (Figure 3-3A). Lastly, the macroinvertebrate communities of sites that never flood were significantly different from sites that annually flood.

Table 3-5: PERMANOVA results table for factors affecting sweep macroinvertebrate communities. P-values for significant factors shown, followed by pairwise analysis of significant within group dissimilarities.

Source	Pseudo-F value	p-value
Location	1.7537	0.0238
Water supply	1.9402	0.0121
Flood frequency	2.5751	0.0071
Pairwise differences	t statistic	p-value
WGM, LWK	1.8105	0.0123
WGM, LWP	1.5641	0.0338
Rain, Spring	1.6836	0.0221
Rain, Swamp	2.0472	0.0123
Never, Annually	1.929	0.0022

A MDS plot with landscape, water quality, and fish biota vector overlays indicated rain-fed ponds were positively associated with ambient conductivity, area:perimeter ratio, percentage of perimeter fenced, and the dissolved oxygen concentration (Figure 3-3B). Gambusia abundance was negatively associated with swamp-fed ponds while drain-fed ponds were positively associated with edge length and area. Edge length, area, pH and age had weak associations with the macroinvertebrate communities. Turbidity and koi carp abundance had a very similar influence on communities, and were positively associated with three swamp-fed ponds (WGM_3, WGM_6, and WGM_7).

A total of 33% variation in macroinvertebrate community composition was accounted for in a Distance Linear Model (DistLM) using only landscape variables (Table 3-6). The amount of the pond fenced was the only landscape variable having a significant impact on macroinvertebrate communities, accounting for 14% of the variation among sweep samples. The remaining landscape variables explained less than 4% variation each. All landscape variables were included in the most parsimonious model.

Table 3-6: DistLM analysis of the cumulative effect of landscape variables on sweep macroinvertebrate communities of 34 constructed ponds on the lower Waikato River floodplain. Significant p-values are in bold. Only variables contributing >3% variation are included in the table.

Variable	Pseudo-F value	P-value	Proportion	Cumulative R^2
Fenced (%)	5.015	**0.001**	0.135	0.135
Buffer Width (m)	1.438	0.187	0.038	0.174
Area:perimeter	1.359	0.222	0.036	0.210
Age class	1.359	0.227	0.035	0.245
Area (m^2)	1.265	0.268	0.033	0.278
Edge length (m)	1.397	0.213	0.036	0.313

When all measured variables were included in the DistLM analysis, the abundance of gambusia was found to be the most influential variable determining sweep macroinvertebrate communities, accounting for 18% of the variation (

Table 3-7). The abundances of the four fish species (gambusia, shortfin eel, goldfish, and koi carp) were all significant in explaining variation among sites. Temperature and pH were the only water quality variables associated with macroinvertebrate community composition. The percentage of the pond margin which is fenced, and the age of the ponds were the only other variables which individually explained more than 3% of the variation among sites. The best-fit model included all landscape, water quality, and fish variables to explain 71% of the variation between sweep macroinvertebrate communities.

Table 3-7: DistLM analysis of the cumulative effect of all variables on sweep macroinvertebrate communities of 34 constructed ponds on the lower Waikato River floodplain. Significant p-values are in bold. Only variables contributing >3% variation are included in the table.

Variable	Pseudo-F value	P-value	Proportion	Cumulative R^2
Gambusia abundance (CPUE^{-1})	7.055	**<0.001**	0.181	0.181
Fenced (%)	3.942	**0.001**	0.092	0.273
Shortfin eel abundance (CPUE^{-1})	2.565	**0.013**	0.057	0.330
Age class	1.909	0.061	0.041	0.372
Temperature (^{o}C)	2.122	**0.042**	0.044	0.416
Goldfish abundance (CPUE^{-1})	2.079	**0.046**	0.042	0.458
Koi carp abundance (CPUE^{-1})	2.429	**0.019**	0.046	0.504
pH	2.231	**0.037**	0.041	0.545

Sweep taxa abundances

Kruskal-Wallis testing on invertebrate abundances revealed there were no significant differences between macroinvertebrate abundances (CPUE^{-1}) and flooding regime or location classes. The only significant difference found was for Diptera abundances between spring-fed and rain-fed ponds (p = 0.004) (Figure 3-4). Median Diptera abundance was lowest in rain-fed ponds and highest in spring-fed ponds. One swamp-fed pond also supported high Diptera abundance but interquartile ranges were similar to drain-fed ponds.
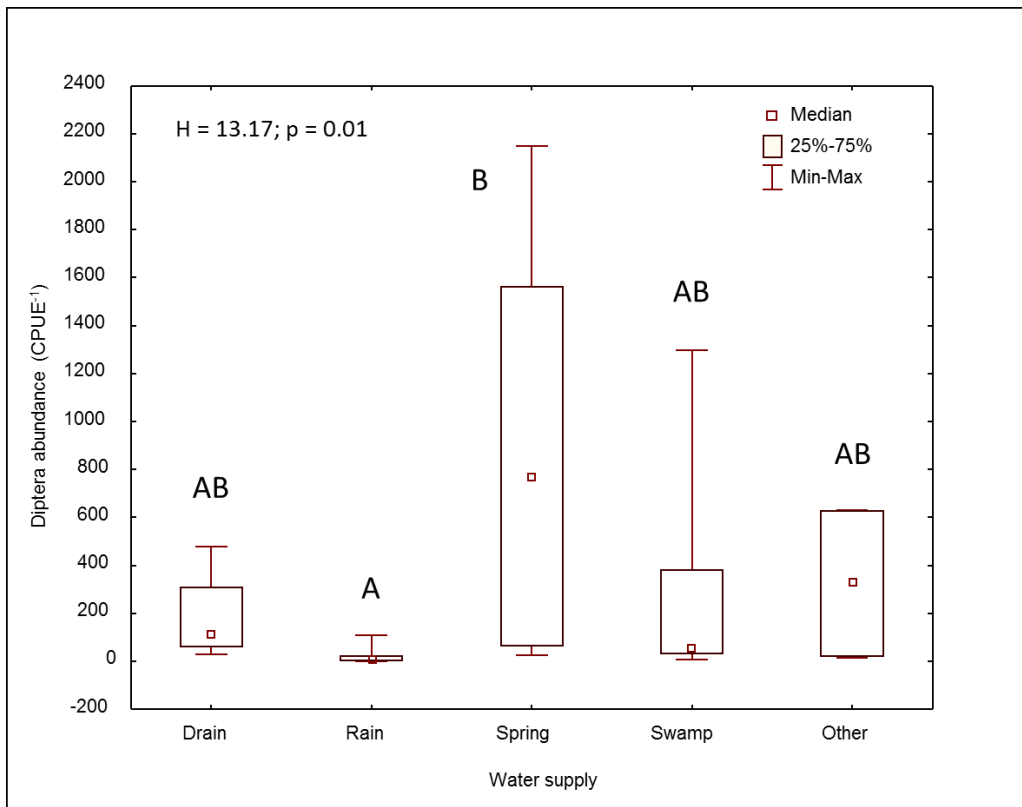
Figure 3-4: Box plots of sweep Diptera abundances (CPUE^{-1}) between water supply classes of 34 constructed ponds on the lower Waikato River floodplain. Classes with the same letter above boxplots are not significantly different.

3.3.2.2 Benthic samples

Twenty-three macroinvertebrate taxa were identified from the benthic samples. Chironomidae and Oligochaeta were the two most frequently encountered taxa, being found at 33 and 25 ponds, respectively. Chironomidae relative abundance ranged from 0% in LWK_2 to 99% in LWP_3; Chironomidae made up over half the macroinvertebrate abundance at 15 ponds. In ponds where Oligochaeta were found, relative abundance ranged from 2% in LWP_1 and WGM_12 to 98% in LWK_2. Ostracoda were found in 18 ponds, and ranged in relative abundance from 1% in LWK_5 to 72% in LWK_6. Eight macroinvertebrate taxa were only identified in one pond each. Isopoda made up 4% of the benthic macroinvertebrate community in WGM_13. Adult and larval Dytiscidae were only found in LWP_5 and LWK_6, respectively, where they made up less than 1% of the community composition. Of the two Hemiptera taxa identified, Corixidae was found in four benthic samples, while Notonectidae was only found in one. Lepidoptera larvae and Trichoptera were the other insect taxa only found in one pond each. Unidentifiable insect larvae were found in WGM_8, and unidentifiable insect pupa was found in LWP_7. Refer to Appendix 6 for the benthic macroinvertebrate abundances (CPUE^{-1}) of each site.

Environmental relationships

A MDS plot of benthic community composition with landscape, water quality and fish biota vector overlays indicated Chironomidae, Gastropoda, and Copepoda were positively associated with the abundances of gambusia, koi carp, goldfish, and shortfin eel while Isopoda were negatively associated with these fish species (Figure 3-5 B&C). As the percentage of pond perimeter fenced increased, the relative abundances of benthic Cladocera, Odonata, and Chironomidae pupa increased, while Aranae decreased. Oligochaeta and pH had a negative relationship, as did Hirudinea and pond area. Ostracoda was negatively associated with pond edge length. Corixidae, Diptera larva, and Dytiscidae larva were all negatively associated with buffer width, temperature, and edge length of a pond.
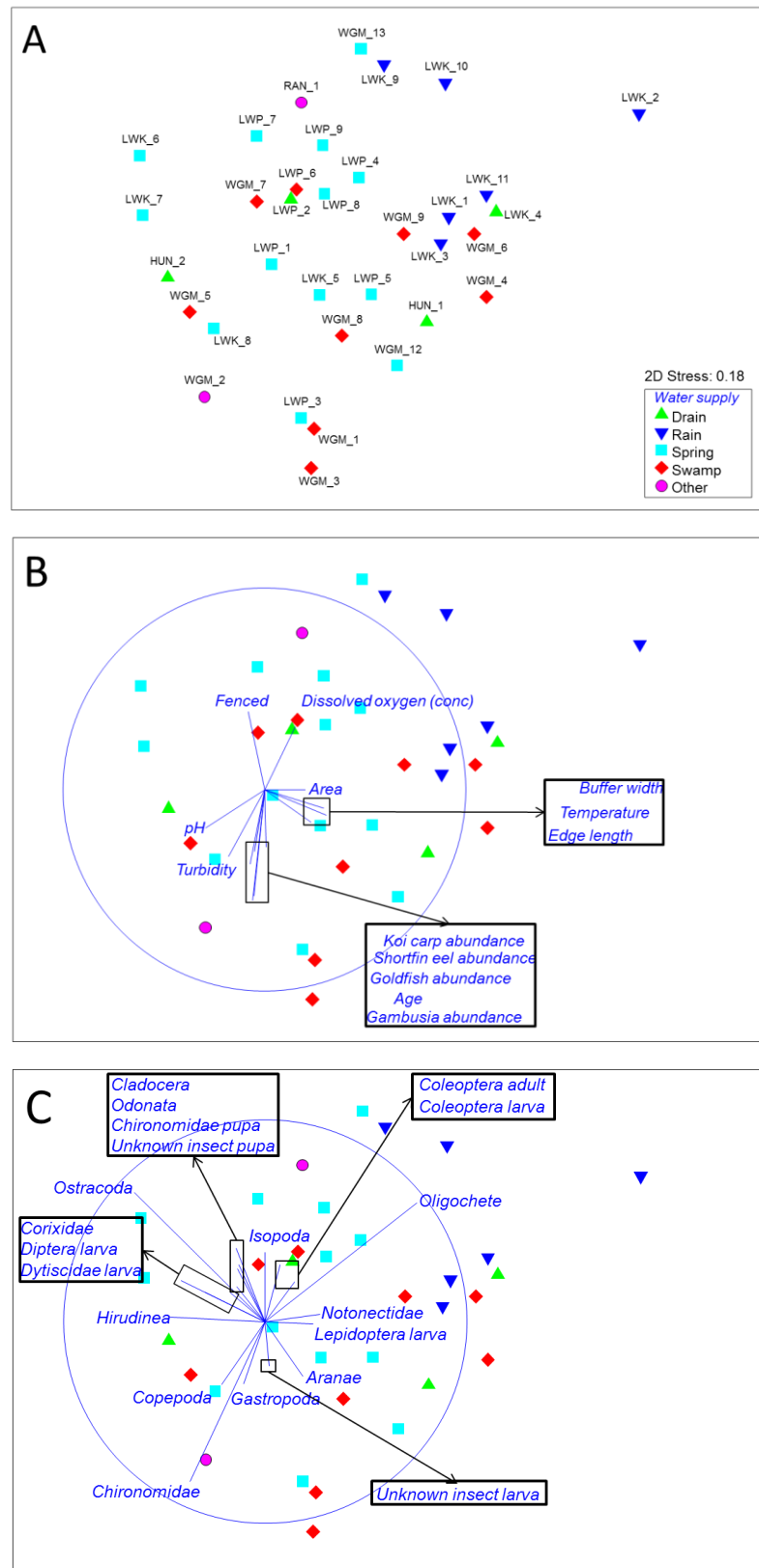
Figure 3-5: MDS plots of benthic macroinvertebrate community data with data point symbols determined by water supply to the ponds. A = data points labelled by location (see Table 3-1). B = landscape, water quality, and fish vector overlay. C = macroinvertebrate trajectory overlay. Overlays have been calculated using a Spearman rank correlation coefficient $r_s >0.2$

47

For benthic communities, pond permanence and water supply were the only two factors with significant influences on community composition (Table 3-8). The significance of pond permanence was evident between those sites that rarely and sometimes dry out, rarely and annually or sometimes dry out, and those that dry out sometimes and never. Within water supply classes, there were significant differences in benthic macroinvertebrate communities between rain-fed ponds and all other pond types. No other pairs of permanence or water supply classes had significant differences in macroinvertebrate communities.

Table 3-8: PERMANOVA results for factors affecting benthic macroinvertebrate communities of 34 constructed ponds on the lower Waikato River floodplain. p-values for significant factors shown, followed by pairwise analysis of significant within group dissimilarities.

Source	Pseudo-F value	p-value
Permanence	2.194	0.012
Water supply	2.468	0.002
Pairwise differences	**t statistic**	**p-value**
Rarely, Sometimes	1.863	0.012
Rarely, Annually	1.727	0.035
Sometimes, Never	1.760	0.019
Rain, Drain	1.603	0.024
Rain, Spring	2.627	<0.001
Rain, Swamp	2.297	0.001
Rain, Other	2.155	0.037

Pond fencing had the most significant influence on benthic communities when factors were considered alone (Table 3-9). Pond fencing could explain 8% of the variation among benthic communities. Buffer width and area explained 6 and 5% variation, respectively. The most parsimonious fit of variables to the model included all variables; explaining 32% of the variation among macroinvertebrate community compositions.

Goldfish abundance was the most significant variable explaining differences in benthic macroinvertebrate community composition when landscape, water quality,

and fish species were included in the DistLM (Table 3-10). These variables were followed by three landscape variables; pond fencing, buffer width, and pond area which each contributed to more than 5% of the variation among benthic macroinvertebrate communities. Two water quality variables, dissolved oxygen and turbidity were significant contributors to the individual variation among macroinvertebrate communities. All variables were included in the best-fit DistLM model, which accounted for 60% variation in macroinvertebrate community composition among ponds.

Table 3-9: DistLM analysis of landscape features and benthic macroinvertebrate communities of 34 constructed ponds on the lower Waikato River floodplain. Significant p-values are in bold.

Variable	Pseudo-F value	p-value	Proportion	Cumulative R^2
Fenced (%)	2.831	**0.024**	0.081	0.081
Buffer width (m)	2.233	0.063	0.062	0.143
Area (m^2)	1.874	0.104	0.050	0.193
Age (class)	1.698	0.135	0.045	0.238
Overhanging vegetation (%)	1.381	0.236	0.036	0.274
Edge length (m)	0.541	0.745	0.014	0.288
Area:perimeter	1.331	0.256	0.035	0.323

Table 3-10: DistLM analysis of the cumulative effect of all predictor variables on benthic macroinvertebrate communities of 34 constructed ponds on the lower Waikato River floodplain. Significant p-values are in bold. Only variables contributing >2% variation are included in the table.

Variable	Pseudo-F value	p-value	Proportion	Cumulative R^2
Goldfish abundance (CPUE^{-1})	3.866	**0.003**	0.108	0.108
Fenced (%)	2.243	0.050	0.060	0.168
Buffer width (m)	2.107	0.070	0.055	0.223
Area (m^2)	2.055	0.074	0.051	0.274
Dissolved oxygen (mg/L)	2.328	**0.044**	0.056	0.330
Turbidity	2.346	**0.039**	0.054	0.383
Ambient conductivity (µS/cm)	1.462	0.219	0.033	0.416
Temperature (^{o}C)	1.070	0.397	0.024	0.440
pH	1.728	0.131	0.038	0.478
Koi carp abundance (CPUE^{-1})	1.551	0.182	0.033	0.511
Overhanging vegetation (%)	1.148	0.356	0.024	0.535

Benthic taxa abundances

Kruskal-Wallis analysis indicated recorded Oligochaeta abundance varied significantly among different permanence classes (Figure 3-6A). There was a significant difference ($p < 0.05$) between ponds that dry out rarely compared with annually, although variation among types was high. The range of Oligochaeta numbers recorded at each rarely-drying site was larger than the Oligochaeta abundances of ponds that annually dry out, the median number was much higher at sites that annually dry out. Benthic Diptera was the only macroinvertebrate group whose abundances were affected by the different water supply classes (Figure 3-6B). There was a significant difference between spring-fed and rain-fed ponds ($p = 0.027$). No other macroinvertebrate groups had significantly different abundances between permanence or water supply classes.
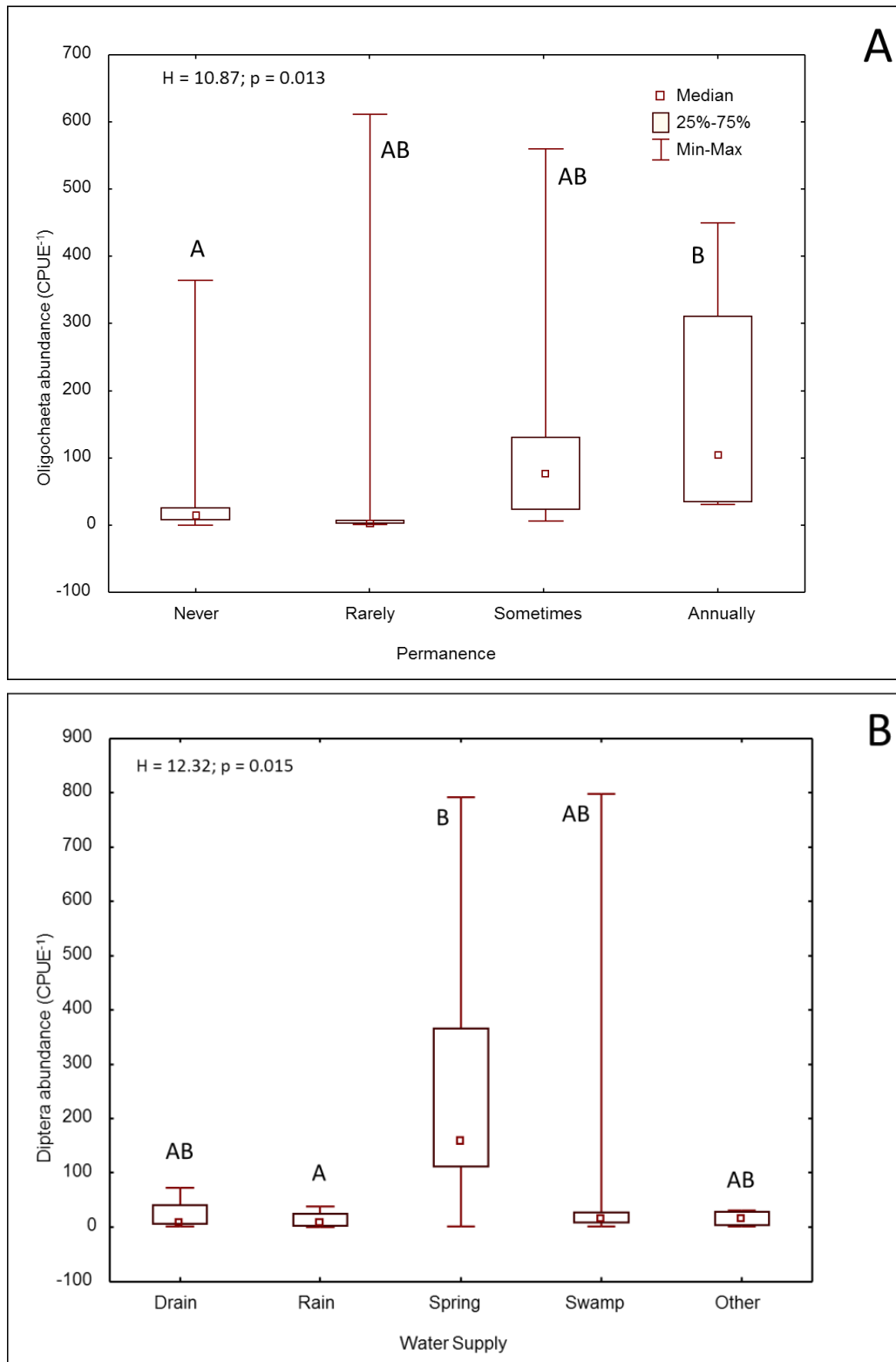
Figure 3-6: Boxplots of benthic Oligochaeta abundance (CPUE^{-1}) between permanence classes (A), and benthic Diptera abundances (CPUE^{-1}) between water supply classes (B) of 34 constructed ponds on the lower Waikato River floodplain. Classes with the same letter above boxplots are not significantly different.

3.3.3   Fish communities

In total, 2,629 fish were caught in the study ponds from the fyke nets, Gee minnow traps, and 10 minutes electrofishing (CPUE^{-1}). Of the 29 ponds where fish were caught, abundance ranged from three individuals to 474 (CPUE^{-1}). This included 10 fish species, with five native species and five exotic species.

The exotic gambusia (*Gambusia affinis*) was the most abundant fish, with 1,688 individuals caught across 17 ponds. Koi carp and goldfish (*Carassius auratus*) were the next most abundant exotic fish with 145 and 111 individuals caught, respectively. Koi carp were caught in eight ponds, six of which were in Whangamarino wetland ponds. Nine ponds had goldfish in them, six ponds in the Whangamarino wetland, and one pond each from around Lakes Whangape and Waikare, and Huntly. Brown bullhead catfish were caught in ten ponds, where abundance ranged from one in LWK_3 and LWK_4 to 25 in WGM_6, for a total of 84 individuals. The final exotic fish, perch, was only found in WGM_2, where two were caught.

Only 739 of the total 2,629 fish caught were native fish. Shortfin eel (*Anguilla australis*) was the most abundant native fish, with 675 individuals caught across 27 ponds. WGM_1 had the highest shortfin eel abundance at 214, with the next highest catch being 65 individuals at LWK_5. Forty-three common bully (*Gobiomorphus cotidianus*) were caught across seven sites, with over half (23) found in WGM_7. Only nine common bully individuals were caught outside of the Whangamarino wetland ponds. Sixteen longfin eel (*Anguilla dieffenbachii*) were caught across four sites, with half being caught at LWK_3. WGM_9 and WGM_13 were the only sites with black mudfish, with one and three individuals being caught in the ponds, respectively. One common smelt (*Retropinna retropinna*) was caught in LWP_1.

Total fish biomass across the 29 ponds with fish was 366,238 g CPUE^{-1}. The exotic fish biomass, (205,501 g CPUE^{-1}) was greater than the native fish biomass (160,737 g CPUE^{-1}). Koi carp had the highest species biomass of 180,525 g CPUE^{-1}, followed by shortfin eel with 148,710 g CPUE^{-1}. Catfish, longfin eel, and goldfish had similar total species weights (12,980, 11,930, and 11,356 g CPUE^{-1}, respectively), while the rest of the fish species had total weights below 400 g

CPUE^{-1}. Over half the total fish abundance consisted of gambusia; however, it made up just 300 g CPUE^{-1} of the total fish biomass. LWP_1 had the highest fish biomass for a pond, 82,796 g CPUE^{-1}, which largely consisted of koi carp (74,700 g CPUE^{-1}). HUN_2 had the next highest fish biomass at 44,170 g CPUE^{-1}, of which nearly 31,000 g CPUE^{-1} was of exotic fish. RAN_1 and LWP_2 had the lowest fish biomasses at 2 and 3 g CPUE^{-1}, respectively.

The average abundance of fish from ponds in the Whangamarino wetland and around Lake Whangape was higher than for Lake Waikare ponds; however, there was not a significant difference in fish abundance between locations (Table 3-11). There was also no significant difference in the average fish biomass in ponds between locations. Koi carp was the only fish species which had significantly different average abundance and biomass between locations. Refer to Appendix 4 for the abundance (CPUE^{-1}) and biomass (g CPUE^{-1}) of fish species in each pond.

Table 3-11: Mean ± standard error of fish abundance and biomass (CPUE^{-1}) in 34 constructed ponds on the lower Waikato River floodplain. LWK = Lake Waikare, LWP = Lake Whangape, WGM = Whangamarino wetland, HUN = Huntly, RAN = Rangiriri. n = no. of ponds. Probability value indicates significance of differences among site groups using Kruskal-Wallis test.

Species	LWK (n = 11)	LWP (n = 9)	WGM (n = 11)	HUN (n = 2)	RAN (n = 1)	Overall (n = 34)	p
No. fish (CPUE^{-1})							
Mudfish	0 ± 0	0 ± 0	0.4 ± 0.3	0	0	0.1 ± 0.1	0.366
Bully	0 ± 0	0.8 ± 0.5	3.1 ± 2.1	0	2	1.3 ± 0.7	0.104
Smelt	0 ± 0	0.1 ± 0.1	0 ± 0	0	0	0.02 ± 0.02	0.596
Longfin eel	1.1 ± 0.8	0.1 ± 0.1	0.3 ± 0.3	0	0	0.5 ± 0.3	0.899
Shortfin eel	12.3 ± 5.5	8.9 ± 3.8	34.6 ± 18.9	39	2	19.9 ± 6.6	0.205
Catfish	0.2 ± 0.1	0.9 ± 0.9	5 ± 2.4	10	0	2.5 ± 1	0.127
Perch	0 ± 0	0 ± 0	0.2 ± 0.2	0	0	0.1 ± 0.1	0.719
Gambusia	32.8 ± 27.7	79 ± 50.9	56 ± 20.9	0	0	49.6 ± 17.4	0.121
Goldfish	0.3 ± 0.3	6.2 ± 6.2	4.3 ± 3.3	3	0	3.3 ± 1.9	0.148
Koi carp	0 ± 0	6.7 ± 6.7	5.9 ± 2.4	10	0	4.3 ± 2	0.047
Total	46.6 ± 28.1	102.7 ± 49.8	109.6 ± 27.8	61	4	81.4 ± 18.5	0.258
Biomass of fish (g CPUE^{-1})							
Mudfish	0 ± 0	0 ± 0	2.2 ± 2	0	0	0.7 ± 0.6	0.366
Bully	0 ± 0	1.5 ± 1	5.3 ± 3.4	0	2.3	2.2 ± 1.2	0.102
Smelt	-	<1	-	-	-	-	<1
Longfin eel	930 ± 838.5	0 ± 0	154.6 ± 154.6	0	0	350.9 ± 276.1	0.683
Shortfin eel	6012.7 ± 2817.5	1274.4 ± 733.3	4494.6 ± 2060.2	10830	0	4373.8 ± 1198.1	0.068
Catfish	96.4 ± 89.6	140.0 ± 140.0	610.9 ± 285.5	1970	0	381.8 ± 152.9	0.173
Perch	0.0 ± 0.0	0.0 ± 0.0	30.9 ± 30.9	0	0	10 ± 10	0.719
Gambusia	3.9 ± 3.2	15.5 ± 11.0	10.7 ± 3.8	0	0	8.8 ± 3.3	0.114
Goldfish	5.5 ± 5.5	129.4 ± 129.4	743.8 ± 564.0	975	0	334 ± 194.5	0.091
Koi carp	0.0 ± 0.0	8300.0 ± 8300.0	7356.8 ± 2953.1	12450	0	5309.6 ± 2461.2	0.047
Total	7048.4 ± 2945.0	9860.8 ± 9120.3	13409.7 ± 4389.8	26225	2.29	10771.7 ± 3064.5	0.075

3.3.3.1 Fish abundance

MDS plots show ponds which annually flood had different fish community compositions than ponds that occasionally and never flood (Figure 3-7). Fish communities of ponds which never flood also seemed to be slightly different than ponds which occasionally flood.

Annually flooded ponds were positively associated with overhanging riparian vegetation, pond area and edge length, ambient conductivity, and the area:perimeter ratio (Figure 3-7B). Ponds which never flood had a positive relationship with pond fencing and the abundance of sweep invertebrates (CPUE), while being negatively associated with turbidity. Fish communities with a positive association to benthic invertebrates had negative relationships with overhanging vegetation, ambient conductivity, and pond area, edge length, and area:perimeter ratio.

The large-bodied exotic fish, koi carp, goldfish, and catfish, were positively associated with annually flooded ponds, as was the native shortfin eel (Figure 3-7C). Gambusia was positively related with water temperature and benthic invertebrate abundance. Shortfin eel was strongly positively associated with ponds with high areas and edge lengths, and negatively with benthic invertebrate abundance. Overhanging vegetation and turbidity were positively associated with fish communities dominated by large-bodied pest fish. The common bully was positively associated with pond temperature and negatively with sweep invertebrate abundance.
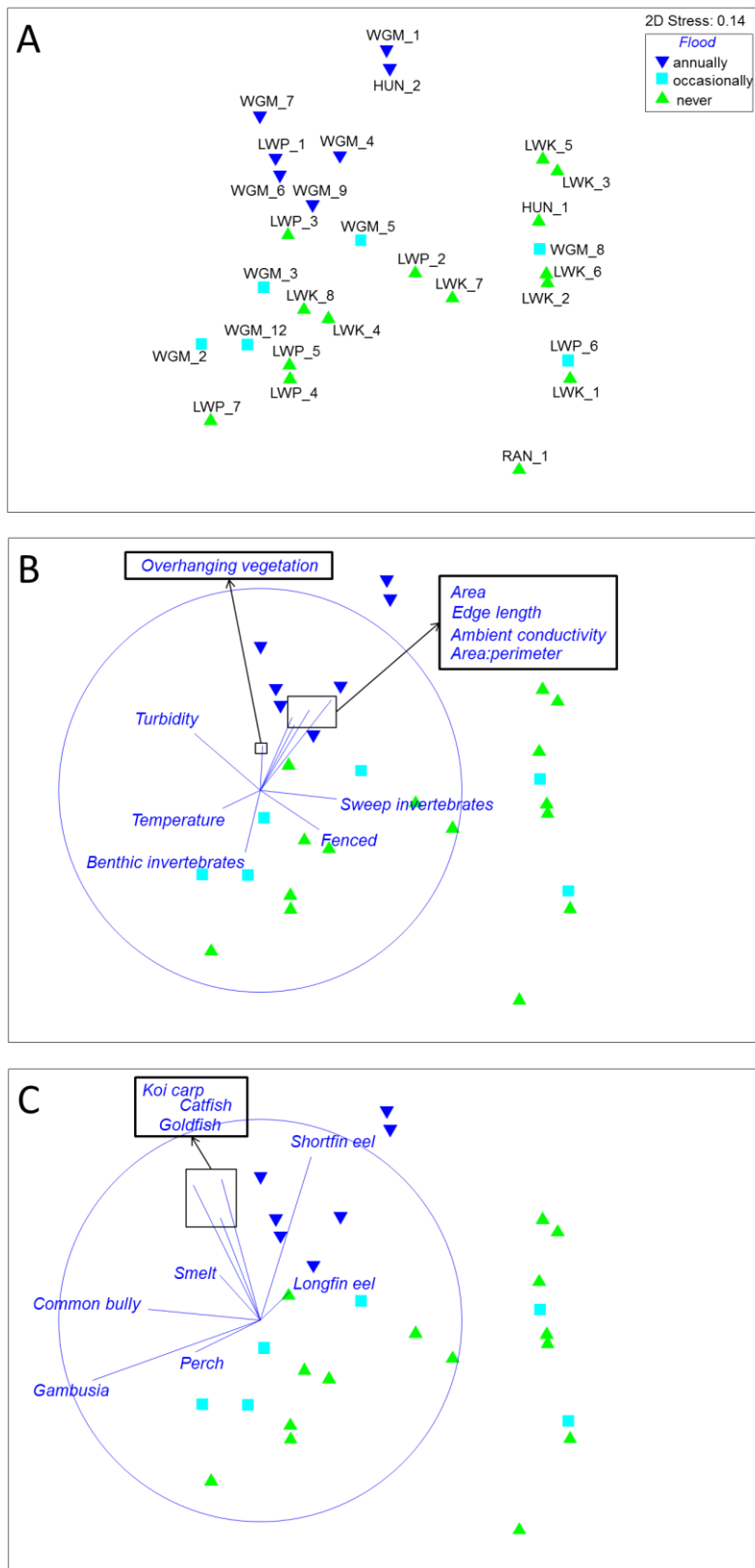
Figure 3-7: MDS plots of fish community composition based on abundance data (CPUE^{-1}), with data point symbols denoting pond flood frequency. A = data points labelled by location (see Table 3-1). B = landscape, water quality, and invertebrate abundance overlay. C = fish species taxa overlay. Overlays have been calculated using Spearman rank correlation coeffient r_s >0.2.

Fish community composition, based on abundances (CPUE-1), was significantly affected by location, water supply and flood frequency (Table 3-12). The permanence of a pond had no significant impact. There were significant differences between fish communities of the Lakes Waikare and Whangape sites, and Lake Waikare and Whangamarino wetland ponds. Significant differences in water supply were found between rain-fed and spring-fed ponds, and rain-fed and swamp-fed ponds. Fish communities also differed between ponds that never and annually flood, and annually and occasionally flood. The difference between the fish abundances of ponds that occasionally and never flood was highly insignificant (p = 0.87).

Table 3-12: PERMANOVA results table for factors affecting fish abundances in 34 constructed ponds on the lower Waikato River floodplain. P-values for significant factors shown, followed by pairwise analysis of significant within group dissimilarities. LWK = Lake Waikare ponds; LWP = Lake Whangape ponds; WGM = Whangamarino wetland ponds.

Source	Pseudo-F value	p-value
Location	2.286	0.011
Water supply	1.938	0.024
Flood frequency	3.168	0.003
Groups	**t-statistic**	**p-value**
LWK, LWP	1.817	0.050
LWK, WGM	1.986	0.005
Rain, Spring	1.699	0.047
Rain, Swamp	1.953	0.026
Annual , Never	2.434	<0.001
Annual, Occasional	2.152	0.002

Total pond area had the highest influence on fish community composition, accounting for 10% of the variation among ponds when landscape variables were considered alone (Table 3-13). Edge length of the pond was also significant, contributing over 10% variation among fish communities; however, it was only considered the fourth most important variable in the stepwise DistLM model. In total the landscape variables explained 39% of the variation between sites.

When all variables were included, 61% of the variation in fish communities was explained. Sweep macroinvertebrate abundance was the only significant variable, explaining 11% of the variation. Pond area was no longer significant; however, it still explained 8% of the variation. The most parsimonious model included all variables.

Table 3-13: DistLM analysis of the cumulative effect of landscape variables on fish abundances in 34 constructed ponds on the lower Waikato River floodplain. Significant p-values are in bold. Only variables contributing >3% variation are included in the table.

Variable	Pseudo-F value	P-value	Proportion	Cumulative R^2
Area (m^2)	2.948	**0.032**	0.102	0.102
Fenced (%)	2.273	0.079	0.075	0.177
Area:perimeter	1.054	0.369	0.035	0.211
Edge length (m)	3.560	**0.018**	0.106	0.317
Buffer width (m)	1.359	0.259	0.040	0.357

Table 3-14: DistLM analysis of the cumulative effect of all variables on fish abundances in 34 constructed ponds on the lower Waikato River floodplain. Significant p-values are in bold. Only variables contributing >2.8% variation are included in the table.

Variable	Pseudo-F value	P-value	Proportion	Cumulative R^2
Sweep macroinvertebrates ($CPUE^{-1}$)	3.157	**0.027**	0.108	0.108
Area (m^2)	2.388	0.066	0.078	0.186
Buffer width (m)	1.910	0.128	0.060	0.246
Fenced (%)	1.829	0.130	0.056	0.302
Turbidity	1.554	0.200	0.046	0.348
pH	1.065	0.378	0.031	0.379
Area:perimeter	0.966	0.430	0.029	0.408
Edge length (m)	2.179	0.085	0.061	0.469
Age (class)	1.350	0.271	0.037	0.506
Ambient conductivity (µS/cm)	1.249	0.303	0.034	0.540

Kruskal-Wallis analysis indicated flood frequency was the only factor which significantly influenced fish abundances; catfish, shortfin eel, and koi carp abundances varied significantly among different flood frequency classes. The catfish abundance in ponds which annually flood was significantly different to ponds which never flood ($p < 0.001$) and occasionally flood ($p = 0.013$) (Figure 3-8A). Shortfin eel abundance was only significantly different between ponds which annually and never flood ($p = 0.013$; Figure 3-8B). Koi carp abundance was significantly different between ponds which annually and never flood ($p < 0.001$), and annually and occasionally flood ($p = 0.021$) (Figure 3-8C). For all three fish species, the range of abundance was greater in annually flooded ponds, with median abundances near zero in ponds which never and occasionally flood. No other species had significantly different median abundances between flood frequency classes.
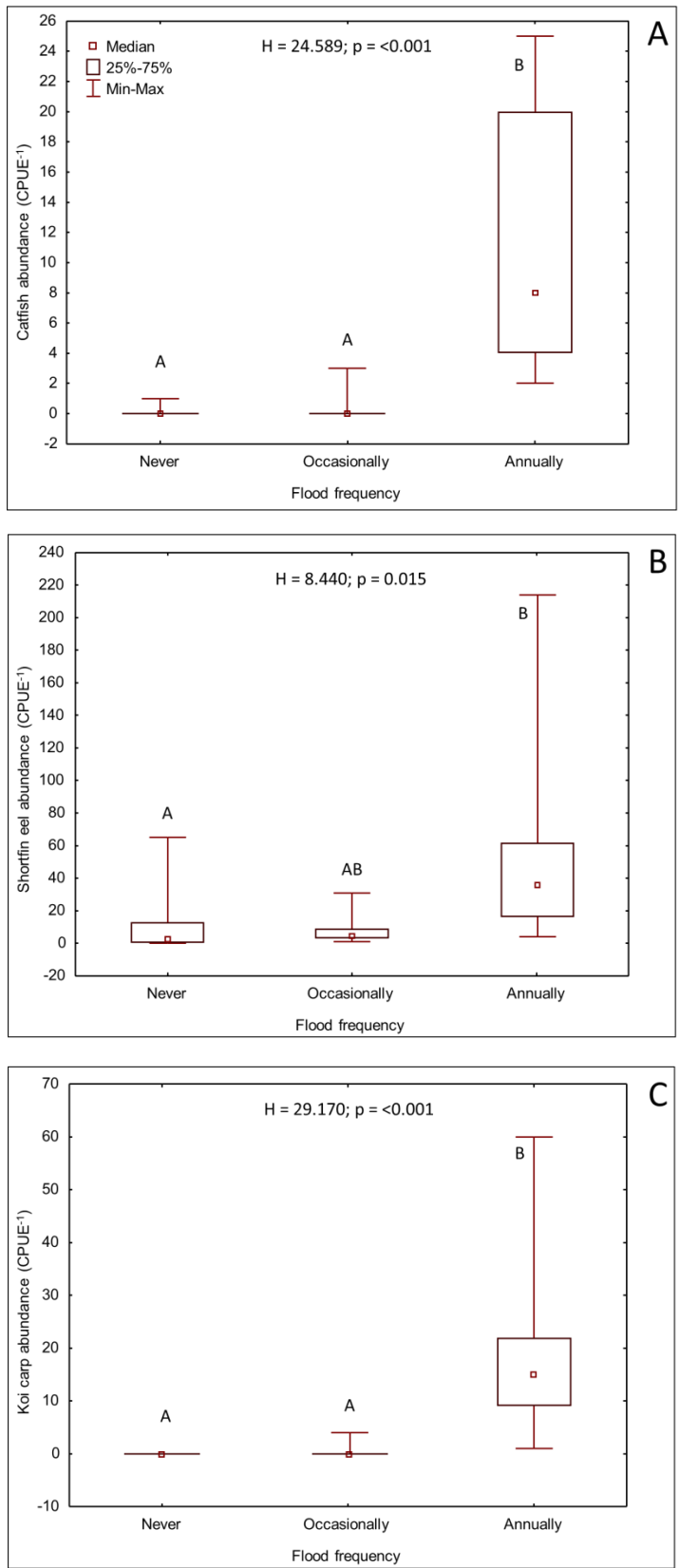
Figure 3-8: Boxplots of catfish abundance (A), shortfin eel abundance (B), and koi carp abundance (C) between flood frequency classes in 34 constructed ponds on the lower Waikato River floodplain. Classes with the same letter above boxplots are not significantly different.

3.3.3.2 Fish biomass

The fish communities based on biomass in ponds which annually flood were similar to each other, and different from ponds which occasionally and never flood (Figure 3-9A; Table 3-15). There appeared to be a less distinct difference in fish community composition between ponds which never and occasionally flood. The fish community composition based on biomass for RAN_1, WGM_12, LWP_2, and LWP_7 was distinctly different to all other fish communities.

Fish community composition based on biomass for annually flooded ponds was positively associated with pond area and edge length landscape variables (Figure 3-9B). The fish communities of ponds which occasionally and never flood were loosely positively associated with pond fencing and sweep invertebrates, and negatively associated with pH.

Annually flooded pond fish communities were positively associated with shortfin eels, as were most ponds which occasionally and never flood (Figure 3-9C). Large-bodied pest fish were positively associated with ponds which annually flood, and negatively to ponds which occasionally and never flood. Longfin eel were loosely positively associated with ponds which occasionally and never flood, whilst being negatively related to annually flooded ponds. Longfin eel were positively associated with sweep invertebrate abundance, while sites with greater area and edge length also had fish communities characterised by greater koi carp, goldfish, catfish, and shortfin eel biomass.
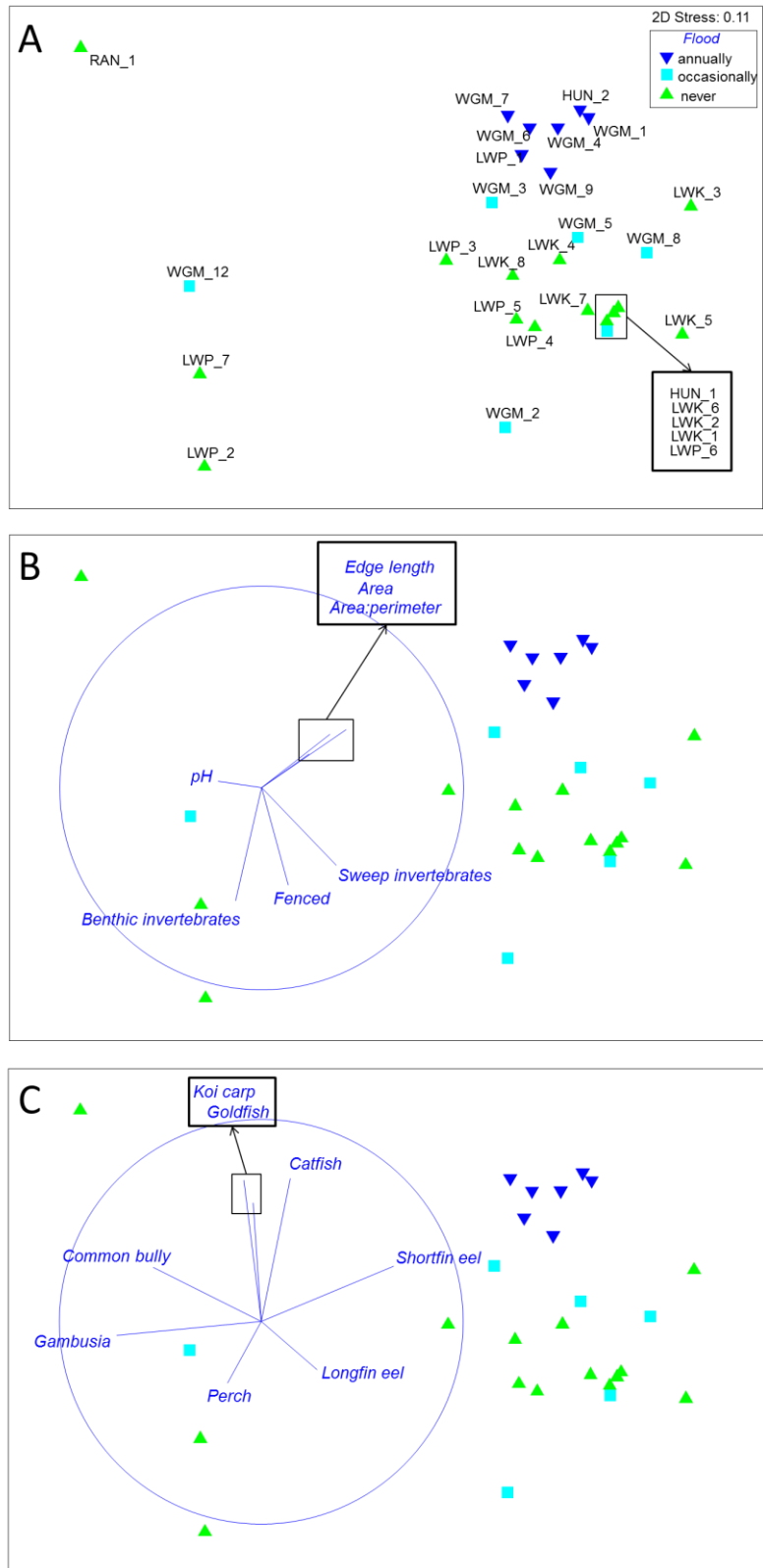
Figure 3-9: MDS plots of fish community composition in 34 constructed ponds on the lower Waikato River floodplain based on biomass with data point symbols denoted by flood frequency. A = data points labelled by location. B = landscape, water quality, and inverterbate abundance overlay. C = fish biomass overlay. Overlays have been calculated using a Spearman rank correlation coefficient $r_s > 0.2$.

The biomass of fish communities in ponds was best described by location, water supply, and flood frequency (Table 3-15). There was a significant difference in fish communities based on biomass between sites from Lakes Waikare and Whangape, and Lake Waikare and the Whangamarino wetland. The significant influence of water supply on fish biomass composition reflected significant differences between rain-fed and swamp-fed ponds, spring-fed and swamp-fed ponds, and ponds fed by swamp compared to other sources. The difference in community biomass between sites that never and annually flood was highly significant, as was the difference between annually and occasionally flooded ponds.

The area:perimeter ratio of a pond was the only landscape variable to have a significant influence on fish communities based on biomass, explaining 13% of the variation between communities in the DistLM analysis (Table 3-16). However, pond fencing was considered the most important explanatory variable in the stepwise DistLM model, where it explained the first 8% variation between fish communities. Collectively, the landscape variables alone explained 40% of the variation between fish communities based on biomass; all landscape variables were used in the final best-fit model.

Table 3-15: PERMANOVA results table for factors affecting fish biomass in 34 constructed ponds on the lower Waikato River floodplain. Significant p-values are shown, followed by pairwise analysis of significant within group dissimilarities. LWK = Lake Waikare communities; LWP = Lake Whangape communities; WGM = Whangamarino wetland communities.

Source	Pseudo-F value	p-value
Location	2.694	0.004
Water supply	2.021	0.025
Flood frequency	3.190	0.004
Pairwise differences	**t-statistic**	**p-value**
LWK, LWP	1.888	0.014
LWK, WGM	2.072	0.001
Swamp , Rain	1.889	0.039
Spring, Swamp	1.866	0.005
Swamp, Other	2.435	0.018
Annually, Never	2.551	<0.001
Annually, Occasionally	2.204	0.001

Table 3-16: DistLM analysis of the cumulative effect of landscape variables on fish biomasses in 34 constructed ponds on the lower Waikato River floodplain. Significant p-values are in bold.

Variable	Pseudo-F value	p-value	Proportion	Cumulative R^2
Fenced (%)	2.343	0.060	0.083	0.083
Area:perimeter	4.144	**0.005**	0.130	0.213
Age (year class)	1.612	0.165	0.050	0.263
Buffer width (m)	0.928	0.443	0.029	0.291
Area (m^2)	0.465	0.761	0.015	0.306
Edge length (m)	2.542	0.057	0.075	0.381
Overhanging vegetation (%)	0.486	0.747	0.015	0.396

No variables were significant explainers of variation in fish community composition based on biomass when landscape, water quality, and invertebrate abundance variables are included in the DistLM (Table 3-17). However, the abundance of sweep invertebrates was considered the most important variable, explaining 9% of the variation. Turbidity was the only water quality variable explaining more than 5% of the variation, with pond area being the most important landscape variable. All variables were included in the most parsimonious model, which explained 61% of the variation between fish communities based on biomass.

Table 3-17: DistLM analysis of the cumulative effect of all variables on fish biomasses in 34 constructed ponds on the lower Waikato River floodplain. Only variables contributing >4% variation are included in the table.

Variable	Pseudo-F value	p-value	Proportion	Cumulative R^2
Sweep invertebrates ($CPUE^{-1}$)	2.405	0.051	0.085	0.085
Turbidity (rank)	2.012	0.096	0.068	0.153
Area (m^2)	2.338	0.060	0.075	0.228
Fenced (%)	1.708	0.146	0.053	0.281
Area:perimeter	1.835	0.118	0.055	0.337
Edge length (m)	1.569	0.181	0.046	0.383
Buffer width (m)	1.600	0.166	0.046	0.429

Flood frequency was the only factor which significantly influenced fish biomass. Catfish biomass (g $CPUE^{-1}$) was significantly different between ponds which never and annually flood (p = 0.001), as well as between occasionally and annually flooded ponds (p = 0.018) (Figure 3-10A). The biomass of catfish in ponds which annually flood was never zero, compared with the median biomass being zero in ponds which never and occasionally flood. Goldfish (Figure 3-10B) and shortfin eel biomass (Figure 3-10C) (g $CPUE^{-1}$) were significantly different between ponds which annually and never flood (p = 0.030 and 0.046, respectively). Koi carp biomass (g $CPUE^{-1}$) in ponds which annually flood was significantly different to ponds which never (p < 0.001) and occasionally (p = 0.021) flood (Figure 3-10D). Flood frequency did not affect the biomass of any other fish taxa.
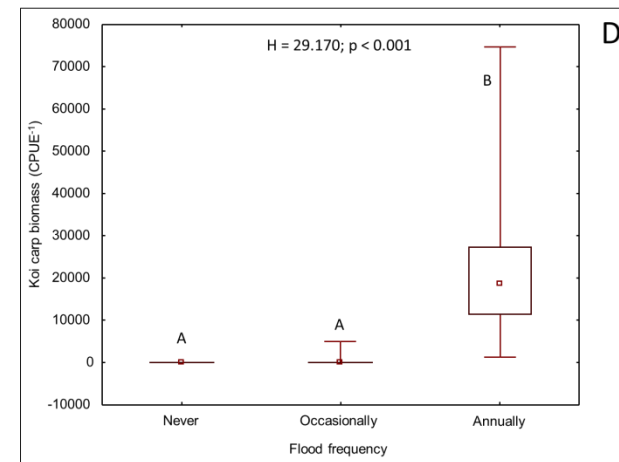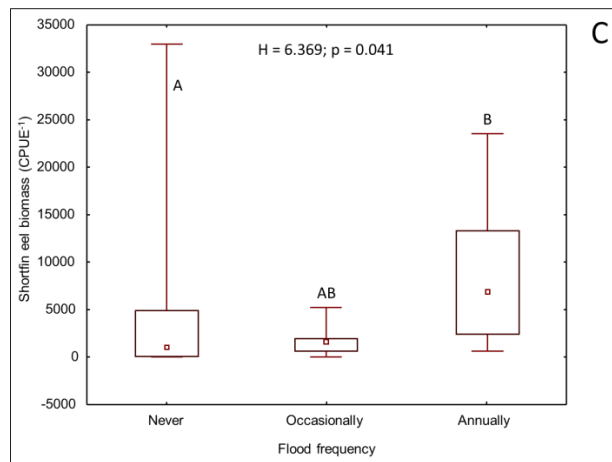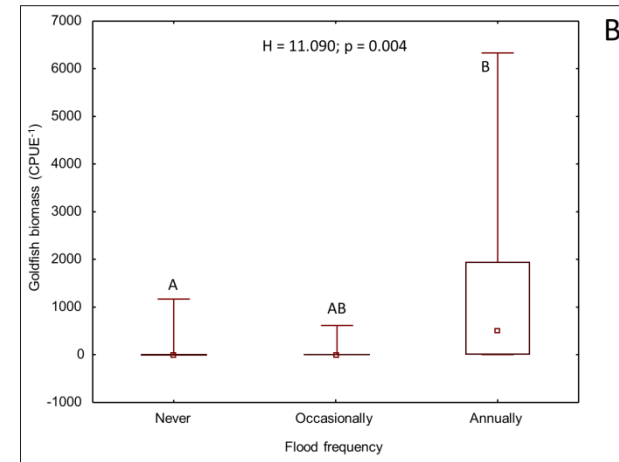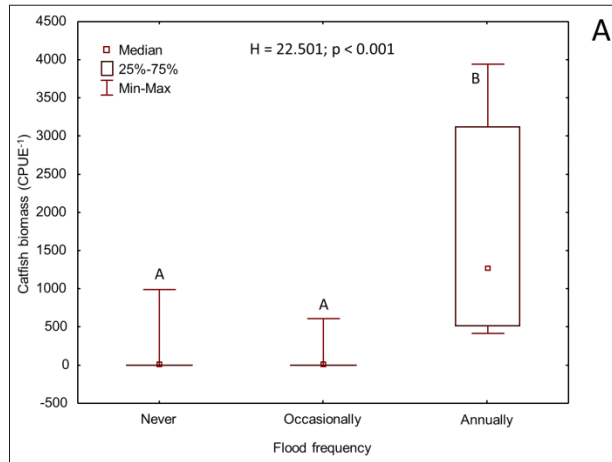
Figure 3-10: Boxplots of catfish (A), goldfish (B), shortfin eel (C), and koi carp (D) biomass (CPUE^{-1}) between flood frequency classes in 34 constructed ponds on the lower Waikato River floodplain. Classes with the same letter above boxplots are not significantly different.

3.4 Discussion

Ponds are often neglected aquatic habitats in the landscape; however, several studies have now revealed their importance to biodiversity (Oertli et al., 2002; Wood et al., 2003). The focus of this chapter is on factors that influence macroinvertebrate and fish communities in constructed ponds of the lower Waikato River floodplain, specifically, to ascertain the hydrological and physicochemical features that significantly influence the distribution and abundance in communities across the landscape.

3.4.1 Environmental factors

A number of key factors was found to structure environmental differences between ponds. Factors identified included pond complexity, maturity and vegetation, and water quality and hydrology. Pond complexity included area and shoreline length, while vegetation was considered related to pond maturity as well-developed vegetation was predominantly found at older ponds. Hydrology impacted the water quality of ponds as permanence was associated with conductivity and dissolved oxygen concentration, while flood frequency appeared related to water turbidity.

Ponds associated with the Whangamarino wetland (WGM) were relatively homogenous, and characterised by smaller area and edge lengths, higher turbidity and higher water temperature than other ponds. Four of the ponds are part of a larger complex of waterfowl hunting ponds which were all constructed with the same horseshoe shape design; consequently they have low physical complexity. Another small horseshoe-shaped pond was part of a private complex of small hunting ponds on the eastern side of the Whangamarino wetland. The Whangamarino ponds are located in the wetland so are characterised by frequent flooding. The retention of flood waters causes annual flooding in the Whangamarino wetland and subsequently the associated ponds, controlled by the Lower Waikato Flood Protection Scheme. This scheme involves the diversion of flood waters from the Waikato River into Lake Waikare and then the Whangamarino wetland where water is retained long enough for flood waters in the river to subside (Brown, 2010). The high concentration of ponds and waterbodies in the Whangamarino wetland accounts for these ponds displaying

high connectivity with the river network, as indicated by the negative association with the distance to the nearest five waterbodies.

Permanence was an important positive factor characterising some ponds, notably several associated with Lake Whangape (LWP) indicating they are permanent features of the landscape, largely attributable to being fed by springs that provide a year-round water supply. Pond maturity and vegetation buffering also accounted for some of the environmental heterogeneity among sites. Notably, several LWP ponds, only two of which were older than 10 years, were negatively associated with age and vegetation factors. Therefore, younger ponds were likely to have less developed and diverse vegetation communities surrounding them. Land around Lake Whangape is subject to intensive farming, meaning riparian vegetation is often limited to small patches around ponds as the availability of grazing land is maximised.

The two Huntly (HUN) sites had large areas and edge lengths, and were largely isolated from other water bodies. The most heterogeneous group of ponds was those associated with Lake Waikare (LWK) which showed no discernable groupings in the two-dimensional principal components plot. The ponds found on the western side of Lake Waikare were constructed on existing wetlands and are fed by rain, while ponds on the northern side of Lake Waikare are predominantly farm dams constructed on grazed land. The LWK ponds also range in age, and therefore, the vegetation structure varied accordingly. The ponds along the western side of Lake Waikare have been developed by organisations including Auckland Waikato Fish and Game, and the Department of Conservation for hunting and biodiversity purposes, whereas many of the northern LWK ponds are farm dams on private land; purpose of construction is likely to be one reason for the observed habitat heterogeneity.

Flooding and permanence have important influences on the habitat heterogeneity of a landscape. The flooding regime also influences water turbidity and is a function of connectivity, while the water supply to ponds can dictate the level of permanence. To maximise habitat heterogeneity it is important to have a range of pond types across the landscape with varying levels of connectivity, water supplies, and ages to increase overall habitat diversity.

3.4.2 Macroinvertebrate communities

Macroinvertebrate communities were analysed based on high-level taxonomic differences because an over-arching aim of this thesis was to understand factors affecting waterfowl food supplies. The sampling habitat had an important influence on the composition and abundance of macroinvertebrates. Abundance of macroinvertebrate groups was greater in sweep samples compared with benthic samples based on sampling effort or area, except for Oligochaeta which was considerably more abundant in benthic samples. Sweep samples were heavily dominated by Crustacea in most ponds while benthic samples were dominated by Oligochaeta. However, it is hard to compare the two habitats due to the difference in sampling method.

There was a significant difference in macroinvertebrate abundances between locations with the mean macroinvertebrate abundance of WGM ponds consistently lower than the overall mean abundance across sweep and benthic samples. The significant differences in mean total macroinvertebrate abundances between the three main locations (LWK, LWP, and WGM) were consistent in benthic and littoral samples. The lower abundances of Crustacea, Hemiptera, and Oligochaeta in WGM ponds compared to LWK and LWP may reflect the more turbid water associated with the Whangamarino wetland.

Diptera and Oligochaeta dominated the benthic samples across all locations which is likely a result of their ability to survive in a wide range of water qualities. Williams et al. (2007) found crustaceans to be early and passive colonisers of ponds, which may account for their much higher relative abundances in LWK and LWP ponds with a mean age of 5-10 years compared to 10-20 years old for WGM ponds. Insects require more time to colonise ponds so Crustacea may be able to flourish in the absence of predation by predatory insects such as Odonata and some Coleoptera and Hemiptera. Vandekerkhove et al. (2012) found Ostracoda survival rates significantly decreased in the presence of predatory macroinvertebrates such as Hemiptera.

Water supply significantly influenced sweep and benthic macroinvertebrate communities, with rain-fed pond communities consistently different. Notably, sweep and benthic Diptera abundances were significantly lower in rain-fed ponds

compared with spring-fed ponds. New Zealand's rain-fed wetlands have low nutrient levels, and thus, are less fertile (Beard, 2010; Johnson and Gerbeaux, 2004), while spring-fed and swamp-fed habitats can have moderate to high nutrient concentrations. For ponds in this study, water from springs flowed overland before entering the ponds. While this water may be low in nutrients at its source, it would inevitably collect nutrients and sediment as it flowed over the land, ensuring it is sufficiently enriched when it enters a pond (Johnson and Gerbeaux, 2004). Biggs (1990) suggests conductivity can be used as a surrogate for nutrient concentrations; WGM ponds had higher mean conductivities than other ponds which indicates it is likely to have higher nutrient concentrations than ponds from other locations.

Overseas studies indicate the ability for Chironomidae to tolerate varying nutrient levels is genera dependent (Brodersen and Anderson, 2002; Langdon et al., 2010; Medeiros and Quinlan, 2011). In those studies, some individual Chironomidae taxa responded positively to increasing total nitrogen or phosphorus concentrations, whereas some responded negatively. With rain-fed ponds in my study likely to have low nutrient status, and swamp and spring-fed ponds likely to have increased nutrients, it is possible Chironomidae abundances were limited by nutrient levels through effects on the growth of algal food supplies. However, in this context my study was limited by not identifying macroinvertebrates beyond family, or measuring total nutrient concentrations in the ponds.

The effects of flooding and permanence on macroinvertebrate communities have been widely discussed in the literature (Collinson et al., 1995; Jeffries, 2003; Neckles et al., 1990; Nicolet et al., 2004; Robson and Clay, 2005; Sanderson et al., 2005; Williams, 1997). In my study, there were significant differences in benthic macroinvertebrate communities between permanence classes but no differences were found in sweep samples. This difference may be due to colonisation rates after each dry phase; most species adapted to life in shallow temporary ponds are unaffected by short droughts as individuals of most species can survive in the mud and repopulate ponds during the following wet-phase (Biggs et al., 1994; Collinson et al., 1995; Palmer, 1981). Significant differences in community composition were found between ponds which annually, sometimes, and rarely dry out, supporting the findings of Collinson et al. (1995) who reported

macroinvertebrate species richness increased as pond permanence increased. However, in the present study no significant difference in benthic macroinvertebrate community composition was detected between permanent and temporary ponds. On a taxonomic level, differences were found between permanent and annually drying ponds where median benthic Oligochaeta abundance was significantly different. This finding is consistent with Brooks (2000) who found Oligochaeta were present in all ponds regardless of hydroperiod, but abundance was significantly lower in ponds inundated for short periods, compared to medium and long-term inundation. There is likely to be an increasing continuum in species richness from temporary to permanent ponds (Collinson et al., 1995).

Flood frequency had a significant influence on sweep communities, notably between ponds which never flood and are inundated annually. Neckles et al. (1990) found macroinvertebrate abundance was significantly greater in temporary, Northern European ponds with frequent, long-lasting floods compared to seasonally flooded sites. In my study, the WGM sites that were characterised by annual flooding predominantly had lower mean sweep abundances than the less frequently flooded locations such as LWP and LWK. However, as discussed above, flood waters enter the Whangamarino wetland via Lake Waikare as a result of the Lower Waikato Flood Protection Scheme. Lake Waikare has very poor water quality, with high nutrient levels and suspended solids (Hamilton et al., 2010; Hicks et al., 2013). High volumes of these degraded waters will be entering the Whangamarino wetland during flood events which may contribute to the lower macroinvertebrate abundances in WGM ponds.

The DistLM analysis that included water quality variables and fish abundances, along with landscape features, provided approximately twice the predictive power than landscape features alone. The percentage of pond fenced off from livestock access featured highly in the sweep and benthic models when all variables were included. According to the MDS plots, sites that were fenced tended to have lower turbidity, suggesting that fencing may influence pond water quality, although this could be over-ridden by the turbidity of inflows and activities of benthic-feeding fish such as koi carp and goldfish. Livestock access in ponds may not impact directly on macroinvertebrate communities (Cereghino et al., 2008; Ranganath et

al., 2009); however, it is well documented that livestock access to streams decreases water quality (Kauffman and Krueger, 1984; Miller et al., 2011; Trimble and Mendel, 1995). The benthic feeding koi carp is also proven to increase turbidity and suspended solid concentrations as it stirs up sediment (Parkos et al., 2003; Scheffer, 2004) which can have an impact on macroinvertebrate communities through direct predation and habitat changes (Parkos et al., 2003). The apparent relationship between koi carp abundance, turbidity and fencing appeared to influence benthic macroinvertebrate communities more so than for sweep samples. Chironomidae were most strongly and positively associated with degraded, turbid ponds in the benthic samples, whereas Oligochaeta showed the opposite association.

The DistLM analysis indicated sweep macroinvertebrate community composition was significantly explained by abundance of gambusia, while benthic community composition was explained by goldfish abundance. Gambusia are known pelagic feeders, consuming anything from small zooplankton to large invertebrates and terrestrial fauna (Mansfield and McArdle, 1998). Consequently, Ling (2004) found gambusia were more likely to impact macroinvertebrate communities than other fishes. Goldfish are benthivorous and, similar to koi carp, stir up the sediment causing increased turbidity and nutrient levels (Rowe, 2007), and are therefore, likely to influence degradation of shallow ponds. The reason behind goldfish abundance explaining more of the variation between macroinvertebrate communities may be that goldfish were more widely spread across the ponds than koi carp.

Based on the DistLM models when all predictor variables were included, it appears water quality and exotic fish abundances played a larger role in structuring macroinvertebrate communities than landscape features. Disregarding fencing, which is likely to moderate pond degradation, pond age was the only landscape feature explaining more than 3% of the variation between sweep macroinvertebrate communities. Pond age, along with buffer width (which was important in explaining benthic communities), potentially influence water quality; older ponds without buffering vegetation are likely to be more degraded in an agricultural setting as a result of nutrient leaching, while fenced and vegetated riparian buffer zones are likely to absorb runoff nutrients from farmland before

they enter the pond (Beard, 2010) and also provide some shelter from wind stirring up sediments.

3.4.3   Fish communities

Fish communities based on biomass and abundance largely consisted of exotic species. The high numbers of exotic fish in WGM ponds is likely a result of high connectivity with Lake Waikare and the Waikato River. Koi carp were absent from the Whangamarino wetland until approximately 30 years ago (Strickland, 1980) but now dominate total fish biomass there and in the Waikato River (Hicks et al., 2008; Hicks et al., 2010). My results confirm that koi carp have the highest biomass, accounting for over half the total fish biomass in Whangamarino wetland ponds. Koi carp have been shown to regularly migrate between the Waikato River and adjacent lakes and wetlands (Daniel et al., 2011). In particular they found the Te Onetea Stream to be heavily used by koi carp migrating between the Waikato River and Lake Waikare; the Pungarehu Stream draining Lake Waikare provides a direct route for fish migrating into the Whangamarino wetland. This suggests the easy access between the Waikato River and the Whangamarino wetland could explain the high koi carp biomasses in WGM ponds. As discussed above, these are also the ponds which frequently flood, increasing the chance of colonisation by invasive fish such as koi carp.

The mean koi carp abundance and biomass in LWP ponds was greater than for WGM ponds but this reflects only one pond having koi present. This pond, LWP_1, was constructed immediately adjacent to the Opuatia Stream where koi carp were actively observed in high numbers. LWP_1 was openly connected to the Opuatia Stream in times of flood, allowing for easy access for fish. Koi carp were also only detected in one of the HUN ponds, meaning the Whangamarino wetland was the only location where koi carp were detected in more than one pond. No ponds around Lake Waikare had koi in them, likely due to their lack of connectivity with the lake and the construction of raised culverts and relatively high banks to prevent exotic fish species from invading.

While koi dominated total fish community biomass where they were present, gambusia was consistently the most abundant fish. Gambusia was originally

introduced to New Zealand to reduce mosquito numbers; however, this proved ineffective (Chapman, 1996). The species has subsequently spread and invaded many freshwater systems around the North Island. Gambusia comprised over half the mean abundance of fish at all locations, except HUN and RAN. Although Hicks et al. (2008) did not quantify gambusia in the Whangamarino wetland, they did note their high abundance in 2008; observations similar to this were also made by Strickland (1980). There is concern gambusia cause a decline in native fish abundances, including the critically threatened black mudfish found in the Whangamarino wetland (Ling, 2004). However, Ling and Willis (2005) found gambusia had little effect on adult black mudfish, but did appear to cause poor health in post-larval mudfish in constructed wetlands. Despite there being no indication that gambusia are effective at reducing mosquito densities, some landowners still actively introduce the fish into their private ponds.

Fish communities appeared to be influenced by the same hydrological factors as macroinvertebrates. Fish communities were significantly influenced by location, with LWK fish communities being consistently different to LWP and WGM ponds. Possible reasons for this have been discussed above; high connectivity between WGM ponds and streams and rivers compared to the more isolated LWK ponds. Furthermore, LWK ponds are predominantly fed by rainwater and springs, whereas WGM ponds are fed by the Whangamarino swamp water. Swamp-fed ponds and rain-fed ponds had significantly different fish community types based on abundances which indicate location and water supply may be interrelated.

Flood frequency appeared to significantly affect abundances of shortfin eel and koi carp, and catfish abundances and biomasses, as well as goldfish biomass. On all occasions abundance and biomass were higher in annually flooded ponds, indicating hydrological connectivity was a key factor influencing fish community composition. The flooding of ponds provides an opportunity for the shortfin eel to migrate to new habitats in search of food (Chisnall, 1987; Chisnall and Hayes, 1991). The relationship between increased water levels and carp movement has been studied by Daniel et al. (2011), Gorski et al. (2014), and Jones and Stuart (2009) whose findings suggest carp will actively migrate from main channels to new submerged habitat to spawn during periods of high water levels and floods.

Pond fencing and turbidity featured highly on the DistLM models for fish communities as well as macroinvertebrate communities, and fish communities based on abundance and biomass appeared to be distinguished by sweep macroinvertebrate abundances. However, koi carp are known to increase the turbidity in ponds, as supported by the MDS plots for abundance, leading to resuspension of nutrients (Chapman, 1996; Hayes et al., 1992; Pimentel et al., 2000). Goldfish and catfish are also known benthivorous feeders (Rowe, 2007); therefore, it is possible that turbidity in annually flooded ponds is a result of the high exotic fish biomass rather than landscape or hydrological features. The benthic feeding habits of koi carp, goldfish, and catfish is a likely reason benthic macroinvertebrate abundance was negatively associated with annual flooding of ponds.

3.4.4   Summary

Hypothesis 1 was supported as flooding and permanency affected sweep and benthic macroinvertebrate communities, respectively. However, only flooding appeared to affect fish communities (Hypothesis 2). Hypothesis 2 was supported as it appeared macroinvertebrate were more sensitive to changes in environmental and biotic conditions. The findings of this study support the management implications proposed by Gorski et al. (2014) for floodplain environments. Their findings suggested the health of wetland areas in the Waikato floodplain could be enhanced by exclusion of exotic species and they recommended the creation of 'controlled connectivity' to restrict exotic species while allowing native species to migrate into floodplain habitats. My findings suggest connectivity caused by flood frequency plays a large role in the distribution of exotic fish species in constructed ponds. The large-bodied pest fish (koi carp, goldfish, and catfish) were consistently associated with degraded ponds which had lower macroinvertebrate abundances. Limiting the access of exotic fish into shallow constructed ponds should result in better ecological health and increased food supplies for other biota such as waterfowl. However, exotic fish, koi carp especially, use flooding events to migrate into new ponds, although this may not be an issue for ponds that occasionally flood. Ponds which occasionally flood did not have significantly different fish communities than ponds which never flood. Management of

livestock accessibility to ponds also appears to influence pond health as reflected in macroinvertebrate community composition. Ponds with more fencing, and thus greater livestock exclusion, had greater macroinvertebrate abundances for several taxa. Minimising the potential for exotic fish species and livestock to enter ponds, through adequate fish barriers (for example raised culverts or raised stop banks) and fencing around the pond perimeter, should result in improved ecological health for these floodplain ponds.

Chapter 4

Factors influencing waterfowl communities on constructed ponds of the lower Waikato River floodplain

4.1 Introduction

4.1.1 Environmental factors

Wetlands and ponds are a necessity for waterbirds (Liang et al., 2002), including 48 threatened Anatidae species (Green, 1996). The high biodiversity and productivity of ponds (Gibbs, 1993) sustain the macroinvertebrate communities that waterfowl feed on to gain important nutrients (Liang et al., 2002; Cox et al., 1998). The ability of waterfowl to feed, along with resting/loafing habitat and nesting success, is influenced by direct and indirect effects of physicochemical factors (Liang et al., 2002), such as: pond area, shoreline complexity, pond depth, and distance to nearest waterbodies (Austin, 2002; Austin and Buhl, 2009; Flake et al., 1977; Kloskowski et al., 2009; Mack and Flake, 1980; Paracuellos, 2006; Sanchez-Zapata et al., 2005; Soulliere and Monfils, 1996; Walker et al., 2013). There are also reports that the surrounding land-use can influence waterfowl community composition (Kuczynski and Paszkowski, 2012; Sanchez-Zapata et al., 2005). Austin (2002) concluded year-to-year variation in waterfowl abundances was driven by water conditions and wetland habitat.

Pond area has been identified as an important factor in determining waterfowl composition and abundance. Waterfowl populations, such as mallard, teal, and shoveler, can have a positive relationship with pond area (Cowardin et al., 1998). Larger ponds are able to hold more breeding pairs than smaller ponds, and therefore, can maintain greater species richness and abundances (Sanchez-Zapata et al., 2005). A number of other studies are consistent with this finding (Austin and Buhl, 2009; Flake et al., 1977; Soulliere and Monfils, 1996; Svingen and Anderson, 1998). Paracuellos (2006) suggested larger ponds hold larger waterfowl populations as they have the ability to accommodate area-independent as well as area-dependent species, whereas smaller ponds are likely to only hold area-independent species. However, small ponds are still important for other

waterfowl because they are used during the breeding phase while larger ponds are used for the brooding phase (Austin and Buhl, 2009). During the breeding phase, the average size of ponds used by waterbirds was 0.71 – 0.77 ha, while brooding waterbirds occupied ponds sized 0.72 – 2.23 ha (Austin and Buhl, 2009). This is in contradiction to Walker et al. (2013) who found, given equal water area, numerous small to mid-sized ponds were more important for brooding waterbirds than larger ponds. Numerous small to mid-sized ponds provide a larger amount of brooding habitat than fewer larger waterbodies with the equivalent surface area (Walker et al., 2013).

Shoreline complexity has been found to be important during the brooding phase of waterfowl. Ponds containing mallard, blue-winged teal, and pintail had, on average, a shoreline twice as long as ponds where broods were not found (Mack and Flake, 1980). Mack and Flake (1980) suggested this was due to the increased number of bays provided by greater shore length, which is likely to increase the amount of shallow foraging sites. This hypothesis was supported by Austin and Buhl (2009) who found increased shoreline complexity provided the habitat diversity brooding waterfowl find appealing; brooding waterfowl require ponds with good foraging habitat, loafing sites, and escape cover. They have also discussed the possibility that greater shoreline complexity allows for greater brood densities as numerous bays provide visual isolation between broods.

Nearby waterbodies can be important as alternative habitats, especially during extreme events such as droughts and severe cold snaps (Kirby, 1995; Kloskowski et al., 2009). This significant relationship between duck numbers and proximity of surrounding waterbodies has long been studied (Austin, 2002; Austin and Buhl, 2009; Evans and Black, 1956; Flake et al., 1977; Johnson and Grier, 1988; Kloskowski et al., 2009). When Kloskowski et al. (2009) compared waterbird populations between the Veta la Palma (VLP) marshland and the Donana National Park (DNP) in Spain they found there was a positive relationship in waterbird numbers between the two locations. As flooding in the DNP increased, duck numbers gradually decreased in the VLP while during dry periods duck numbers declined in both locations. Austin and Buhl (2009) found no broods on isolated wetlands and unmodified seasonal wetlands, indicating the importance of suitable habitats distributed throughout the landscape. While some waterbirds have been

negatively associated to distance between waterbodies (e.g. blue-winged teal, pintail) (Lokemoen, 1973), other studies have found mallard to have a positive association with distance between ponds (Flake et al., 1977; Lokemoen, 1973). Flake et al. (1977) identified mallard as most likely to occupy ponds distant from other ponds.

4.1.2 Biotic factors

Macroinvertebrates, especially those found in benthic habitats, are a significant food source for all waterbirds (Rundle, 1982). Macroinvertebrates are, almost exclusively, the dominant food source during the first fortnight after waterbirds hatch (Cox et al., 1998; Street, 1977; Sugden, 1973). Street (1977) compared the food content of ducklings younger than 13 days and those 13 days and older, and found the diet of <13 day old ducklings consisted of 60% macroinvertebrates, compared with just 5% for 13 – 45 day old ducklings which fed predominantly on plant matter. Accordingly, duckling growth can be positively related to macroinvertebrate density (Cox et al., 1998; Street, 1978). In support of this, decreased abundances of macroinvertebrates in pesticide-treated wetlands has been shown to result in low growth and survival rates in ducklings (McCarthy, 1995).

Adult waterbird behaviour is also influenced by macroinvertebrate densities as waterbirds actively seek waterbodies with high benthic densities (Safran et al., 1997). Correlations have been found between the abundance of waterbirds on a pond and the density of macroinvertebrates (Joyner, 1980; Murkin and Kadlec, 1986; Murkin et al., 1982). Murkin and Kadlec (1986), for example, found a significant correlation between invertebrate abundances and total duck, total dabbling duck, and blue-winged teal abundances on ponds during spring. Relationships were also found between benthic invertebrates and diving duck, and total duck densities during the brooding season (Murkin and Kadlec, 1986).

4.1.3   Aims and objectives

The broad aim of this chapter is to determine what factors influence waterfowl densities and communities during the brooding season on constructed ponds of the lower Waikato River floodplains. Studies from Europe and North America indicate which factors may be important for these regions; however, limited studies have been completed in New Zealand where results may differ. Numerous ponds in the Waikato are constructed for waterfowl productivity and hunting purposes yet knowledge is limited on what makes ponds highly productive for waterfowl. Specific aims of the study were to determine:

- Which landscape feature(s) best explain variation in waterfowl densities. Literature indicates features such as pond proximity and area are drivers of waterfowl abundance overseas; however, New Zealand conditions may result in other landscape factors being important.

- The role that different vegetation types play in affecting waterfowl communities and densities. The available literature suggests individual waterfowl taxa select for different vegetation types; however, this has not been documented in New Zealand.

- Whether conditions suitable for adult waterfowl are different to those for juvenile waterfowl in constructed ponds on the lower Waikato River floodplain. I wanted to determine if there are specific pond features, including landscape, biotic and vegetation variables, which influence the likelihood of juveniles being present on a pond.

Based on the findings of the overseas literature I hypothesise that pond features that strongly characterise high waterfowl communities in the Waikato will be pond area and edge length, and pest fish presence. I hypothesise that waterfowl abundances per hectare will increase as pond area and edge length increase. I also expect pest fish to reduce food availability for waterfowl, and thus waterfowl broods to be predominantly found on pest-fish free ponds.

4.2 Methods

4.2.1 Study sites

Sites for this study were selected to incorporate a wide spread of physicochemical conditions, aquatic vegetation diversity and abundance, and anticipated fish biomass. All sites are located on the lower Waikato River floodplains near Lakes Waikare and Whangape, the Whangamarino wetland, Huntly, and the Rangiriri Pa (Figure 2-1). The wetlands were constructed from one year to over 20 years ago, and range from locations that were originally grazed, dammed springs/seepages, to sites that were excavated from an existing swamp area. All 34 sites are waterlogged for at least eight months of the year; 18 ponds can dry out over the warmer summer months. More details on study sites can be found Chapter 2.

4.2.2 Pond features

Table 4-1 summarises the habitat and physicochemical features measured for the study. For features measured using a ranking system, the ranks were designed to incorporate an even spread of sites, unless otherwise stated. Further details on the methods are found in Chapter 3.

.

Table 4-1: Table of pond features measured at each of 34 constructed ponds on the lower Waikato River floodplain, and their units, with a brief description.

Feature	Unit	Explanation
Location	N/A	Based on the location of the pond. LWK, LWP, WGM, RAN, and HUN = ponds near Lake Waikare, Lake Whangape, Whangamarino wetland, Rangiriri, and Huntly, respectively.
Pond area	m^2/ha	Measured in both square meters and hectares from aerial photos.
Edge length	m	Measured based on average spring/summer water extent from aerial photos.
Area:perimeter ratio	N/A	Calculated by dividing pond area (m^2) by edge length (m).
Island area	m^2	Measured from aerial photos
Island edge length	m	Measured based on average spring/summer water extent from aerial photos
Permanence	1-4	Rank system based on how often the pond completely dries based on landowner interviews: 1 = never dries, 2 = rarely dries, 3 = sometimes dries, 4 = annually dries.
Flood frequency	1-3	Rank system based on how often the pond floods beyond spring-summer water extent based on landowner interviews: 1 = never floods, 2 = occasionally floods, 3 = annually floods.
Water supply	N/A	The pond's dominant source of water.
Overhanging vegetation	%	Visual assessment of the percentage of pond margin overhung by vegetation. Vegetation included trees and shrubs with roots outside the pond margin.
Pond age	1-4	Rank system based on the time since last earthworks which resulted in a >50% increase in size and/or depth based on landowner interviews: 1 = 0-5 years, 2 = 6-10 years, 3 = 11-20 years, 4 = 21+ years.
Pond origin	N/A	Land-use prior to original pond construction based on landowner interviews.
Number of maimai	N/A	Count of the maimai found on or within 20 m of the pond. Used as a measure of hunting intensity
Fenced	%	Visual assessment of the percentage of the pond margin fenced off from livestock.
Dominant substrate	N/A	Visual assessment of the dominant substrate in the pond.
Temperature	^{o}C	Taken from three locations and averaged.
Ambient conductivity	μS/cm	Taken from three locations and averaged.
Dissolved oxygen	mg/L	Taken form three locations and averaged.
pH	N/A	Taken from three locations and averaged
Turbidity	1-4	Rank system of observed water clarity in the pond. Based on the ranking system used by McDougall et al. (2009): 1 = clear, 2 = moderately clear, 3 = moderately turbid, 4 = turbid.

4.2.3 Vegetation descriptions

A sketch of each site recorded the location and extent of riparian and aquatic vegetation as vegetation may influence nesting, brooding, resting, and foraging behaviours. The sketch included aquatic vegetation in the water, aquatic vegetation around the pond margin, and all riparian vegetation within 20 m of open water. Vegetation was identified down to genus where possible and described by assessing the percentage of open water, pond margin, or surrounding land a given vegetation type covered. It was often found that the newer ponds (less than two years old) had a limited degree of vegetation around them, whereas some of the older wetlands had substantial vegetation in and around them. The percentage of pond margin, including islands, overhung by vegetation in late spring-early summer was estimated visually to the nearest 5%. On-site observations were used to determine whether there was a buffer strip of ungrazed vegetation present around the perimeter of each pond. Width of the buffer strip was approximated using WRAPS imagery at four different locations (northern, eastern, southern, and western sides of the pond) and then averaged. Percentage of open water with no visible aquatic macrophytes on the surface was also visually estimated to the nearest 5%.

4.2.4 Waterfowl habitat suitability

Four categories, adapted from McDougall et al. (2009), were used as indicators of habitat suitability for waterfowl; food availability, nesting habitat, loafing habitat, and overhead cover. The rank system used for each category is summarised in Table 4-2. The scores for each category were determined through visual assessment of the pond and riparian margin.

Table 4-2: Waterfowl habitat suitability indicators and description of how each was calculated, based on McDougall et al. (2009). Ranks were determined by visual assessment for 34 constructed ponds on the lower Waikato River floodplain.

Suitability indicator	Description
Food availability	Rank system used based on a visual assessment of seed and macroinvertebrate abundance: 1 = poor, 2 = poor to average, 3 = average, 4 = average to good, 5 = good.
Nesting habitat	Rank system based on the proportion of ungrazed margin (including rank grass and vegetation) to pond area: 0 = 0%, 1 = 1-20%, 2 = 21-40%, 3 = 41-60%, 4 = 61-80%, 5 = 81-100%.
Loafing habitat	Rank system based on the proportion of grazed pond margin to ungrazed loafing areas, including bank topography: 0 = 0%, 1 = 1-20%, 2 = 21-40%, 3 = 41-60%, 4 = 61-80%, 5 = 81-100%.
Overhead cover	Rank system based on the percentage of riparian zone, including pond margin which has overhead cover, including trees and bushes: 0 = 0%, 1 = 1-20%, 2 = 21-40%, 3 = 41-60%, 4 = 61-80%, 5 = 81-100%.

4.2.5   Macroinvertebrate sampling

Field collection

Two sampling techniques were used to collect macroinvertebrates at three sites in each pond to determine the biomass of potential food availability. A 250 μm mesh net was used to sweep macroinvertebrates in the water column and on vegetation following Biggs et al. (1998). It involved a three minute sweep at each site to ensure a constant effort across all ponds in proportion to mesohabitat area (e.g. open water, submerged macrophytes, near-shore, etc.) For example if three different mesohabitats were identified then a minute sweep was taken at each; if a mesohabitat took up a larger proportion of the pond than the others then the time was split to reflect this. The entire sweep sample was then placed in a 500 μm mesh-bucket, to sieve out as much fine material as possible. All samples from each pond were pooled and then placed into containers and preserved with 95-100% ethanol.

Three benthic grab samples were collected at each site (i.e., 9 samples per pond) using a mini Ekman grab (area = 273.6 cm^2; volume = 1217.52 cm^3). Grab

samples were collected in water no deeper than 50 cm as evenly spread around the pond as possible, and were immediately placed in a 500 µm mesh sieve-bucket. After in-the-field sieving the sample was placed in containers and preserved with 95-100% ethanol.

Laboratory processing

In the laboratory each sample was washed through 4 mm, 1 mm and 500 µm sieves. All invertebrates too large to go through the 4 mm sieve were removed and placed in a separate vial. Material caught in the 1 mm and 500 µm sieves was preserved in ethanol for future sorting.

To avoid human bias all samples were sorted by the same person following sample processing protocol P2 which involves a 200 fixed count (Stark et al., 2001). The sieved sample was spread evenly across a white sorting tray and topped up with enough water to just cover the sample. The sorting tray was divided into 24, (6x4) 55 mm by 55 mm, squares. Once the sample was spread across the tray a random number generator was used to select a square to subsample. A cardboard square, the size of the subsample squares, was then placed around the selected subsample square and material in this square was then transferred to a petri dish via a plastic pipette to be sorted. This subsample was methodically searched and all macroinvertebrates were removed and then placed in a vial with 70% ethanol. This process was repeated, square by square, until 200 individuals were counted. The counting continued past 200 until the corresponding square was completely sorted. Once the square was completed the total number of squares counted was noted to determine the percentage of the entire sample sorted.

Invertebrates were sorted into nine groups to determine macroinvertebrate biomass in the ponds (Table 4-3). Invertebrates in each of the nine groups were placed in small aluminium trays and dried overnight at 60 $^{\circ}$C in an oven. The following morning the trays were reweighed using a fine balance which measured down to 0.0001 grams, and dry weights of the specimens were determined.

Table 4-3: Macroinvertebrate taxa collected from 3-minute sweep and benthic grab samples groups and their components used for biomass analysis.

Group	Taxa included
Crustacea	Cladocera, Copepoda, Isopoda, Ostracoda
Coleoptera	Curculionidae, Dytiscidae, Scirtidae, Hydrophilidae
Hemiptera	Corixidae, Notonectidae, Gerridae
Diptera	Chironomidae, Culicidae, Stratiomyidae
Odonata	Zygoptera, Anisoptera
Other insects	Lepidoptera, Trichoptera, Unknown insects
Gastropoda	Gastropoda
Oligochaeta	Oligochaeta
Other	Acari, Aranae, Collembola, Hirudinea

4.2.6 Fish sampling

Two single wing coarse fyke nets and three fine mesh Gee minnow traps were set at each site overnight in autumn 2013. All fyke nets were 8 m in length, including a 5 m long leader, and 3 m long funnel, made with 4 mm mesh size. The leader was 0.6 m high, and the funnel had a 0.65 m opening. Gee minnow traps consisted of two halves which join together to create a small cylindrical trap with inward facing funnels at each end. Each opening is 2.5 cm in diameter, made from the same 3 mm mesh as the main body of the net. All fish captured were measured for length (total length for eels, fork length for all other species; to the nearest mm), and then total species weights (in grams) were taken using hanging bonso scales, accurate to 0.01g. All fish caught were kept alive during the processing and then released back into the pond in accordance to the Standard Operating Procedure's 6 and 7 of the University of Waikato Animal Ethics Committee; pest fish were released back as the fish were needed in the pond to determine their effect. Ten minutes of electrofishing was undertaken at each site as a standard measure of effort, along with the netting and trapping. All fish stunned were captured and measured (total length for eels, fork length for all other fish to the nearest millimetre and total species weight in grams).

The three fishing methods stated above were not effective at catching koi carp (*Cyprinus carpio*), so where koi carp were observed the entire pond margin was walked and all visible koi carp were counted. This was undertaken in November

and December, 2013 where water levels had decreased, enhancing the visibility of koi carp around the margins. These estimates of koi numbers were then transformed into biomass by using the average weight of all koi carp caught by the University of Waikato's electrofishing boat up to February 2010 (Hicks, 2014). The square root mean was used to calculate an estimated biomass of the koi carp visibly counted at ponds.

4.2.7 Waterfowl observations

Counts of waterfowl numbers were conducted at each site four times at monthly intervals throughout the study; the first observations made between 5/9/13 and 17/9/13, while the last observations were made between 2/12/13 and 6/12/13. Observations were made within the first four hours and the last four hours of daylight ensuring that there were two observations at each site in the morning and in the evening to minimise any bias in the time of day observations were made.

The method used for each pond was a 15-minute timed observation followed by a flush-out procedure. For the timed count, the observer used a pair of binoculars (10x42mm) or spotting scope (20-60x80mm) to help with the identification of waterfowl. Where possible, a vantage point was found on a hill near the pond where as much open water as possible could be seen. If there were no suitable hills then the observer would quietly approach the pond until a suitable location could be found which maximised the visibility of the open water. Finally, if only small patches of open water could be seen, then throughout the 15 minutes the observer moved quietly between suitable vantage points to maximise the percentage of pond that could be observed. During this time, all visible birds were counted, including those that flew in or flew off the pond.

After the 15-minute observation was completed the observer(s) would walk around the entire margin of the pond attempting to flush out any birds that had not previously been spotted. Where possible the observer(s) would keep ~10 m away from the water's edge in the hope of flushing birds onto the water. During the flushing process only the birds not previously seen in the 15-minute observation were counted to get an indication of total waterfowl abundance. Binoculars were used to help identify waterfowl.

Total abundance of waterfowl species was determined by counting all waterfowl seen using the pond, including those loafing in the near vicinity. Any broods on the pond were identified, juveniles counted, and the age of the young was determined (Gollop and Marshall, 1954). Having an idea of the age class of the juvenile waterfowl, and the number in each brood allowed broods to be identified as "new" or "old" when subsequent visits to sites were made.

4.2.7.1 Waterfowl density

Waterfowl density was calculated by converting the observational data into waterfowl per hectare to allow for comparison across sites. This was done for adult waterfowl and juvenile waterfowl. Due to the inability to visit the sites more than once a month only new ducklings were considered to provide an estimate of productivity as few ducklings were found at the fledgling stage.

4.2.8 Statistical analyses

Waterfowl communities

Non metric multi-dimensional scaling (MDS) analysis was carried out using PRIMER 6 (v 6.1.15) software to analyse dissimilarities between waterfowl communities. Separate MDS ordination plots were created using adult waterfowl and juvenile waterfowl abundance per hectare data. Waterfowl data were $\log(x + 1)$ transformed in PRIMER before resemblance matrices were created using Bray-Curtis similarity coefficients. Thirteen ponds with no juvenile waterfowl were omitted from the analysis, including four Lake Waikare ponds, one Lake Whangape pond, and eight Whangamarino wetland ponds. Waterfowl adult and juvenile communities and their associations with individual waterfowl species, landscape and water quality variables, vegetation types, and biotic species were assessed using vector overlays. A Spearman correlation coefficient of $r_s > 0.2$ was used for determining which variables were included in the four vector overlays.

One-way PERMANOVA was used to test for significant variation in adult or juvenile waterfowl community composition explained by different categorical factors. Factors tested include pond location, water supply, dominant substrate, age, origin, permanence, flood frequency, and presence of waterfowl hunting

activity. Where the PERMANOVA indicated $p < 0.1$, pair-wise testing was performed using $p < 0.05$ to determine significance. For the one-way PERMANOVA and pair-wise testing, 9,999 permutations were run under the permutation of residuals under a reduced model method. For a complete list of categories within factors, refer to Appendix 7.

Step-wise distance-based linear models (DistLM) were used to test which predictor variables best explained the dissimilarities between the adult and juvenile waterfowl communities. All variables used in the DistLM tests were $\log(x + 1)$ transformed. DistLMs were run for adult and juvenile waterfowl communities to identify which landscape variables could be used to predict variations in composition, and which other variables were identified in full models. Each run of the DistLM analysis included 9,999 permutations and marginal tests. For a complete list of variables included in the DistLM tests refer to Appendix 8.

Species abundances

Non-parametric Kruskal-Wallis tests were used to test for differences in the abundance of adult and juvenile waterfowl species across factor classes (STATISTICA v11). This analysis tested the null hypothesis that the median of all taxa was the same, without the necessity of homogeneity of variance or normally distributed data (Dytham, 2005). All data were untransformed for this analysis as Kruskal-Wallis converts data into ranks to reduce the effects of outliers (Dytham, 2005); therefore, ponds which were removed for the MDS analyses were included. The same factors used in the PERMANOVA tests were used. Adult or juvenile waterfowl species which returned significant ($p < 0.05$) differences in medians were displayed in boxplots.

4.3 Results

4.3.1 Waterbirds

In total 15 avian species and 3,046 individual adult birds were counted. Of the 15 species, eight were waterfowl taxa; non-waterfowl taxa included the Australasian bittern, black shag (*Phalacrocoraz carbo novaehollandiae*), dabchick (*Poliocephalus rufopectus*), pied stilt (*Himantopus himantopus*), pukeko (*Porphyrio melanotus*), little shag (*Phalacrocorax melanoleucos*), and white-faced heron (*Egretta novaehollandiae*). Waterfowl comprised 90% of the total birds observed.

4.3.2 Adult waterfowl

Waterfowl species in this study included black swan (*Cygnus atratus*), Canada goose (*Branta Canadensis*), grey duck (*Anas superciliosa*), grey teal (*A. gracilis*), mallard (*A. platyrhynchos*), New Zealand scaup (*Aythya novaeseelandiae*), paradise shelduck (*Tadorna variegate*), and shoveler (*Anas rhynchotis*). The New Zealand scaup was only encountered on two different occasions at LWK_2. Over 1,000 individual adult waterfowl were encountered on the first visit, compared to fewer than 500 on the second.

Grey teal had the greatest overall densities followed by mallard, while paradise shelduck had the lowest (Table 4-4). Canada goose and black swan densities were highest on LWK and LWP ponds. Grey duck and grey teal densities were greatest on LWP and WGM ponds. All waterfowl species, except paradise shelduck, had densities greater than 1 individual per hectare on average. Canada goose and paradise shelduck had significantly different densities between locations; however, the difference in densities between locations was not significant.

Table 4-4: Mean and standard error of adult waterfowl abundance per hectare at different locations. LWK = Lake Waikare, LWP = Lake Whangape, WGM = Whangamarino wetland, HUN = Huntly, RAN = Rangiriri. n = number of ponds. p value calculated using Kruskal-Wallis analysis.

Waterfowl	LWK	LWP	WGM	HUN	RAN	Overall	p
	$n = 11$	$n = 9$	$n = 11$	$n = 2$	$n = 1$	$n = 34$	
Black swan	1.3 ± 0.4	2.2 ± 0.9	0.3 ± 0.2	0.7	0	1.1 ± 0.3	0.082
Canada goose	3.3 ± 0.8	2.0 ± 0.9	0.5 ± 0.2	0.5	0	1.8 ± 0.4	0.030
Grey duck	1.3 ± 0.7	3.2 ± 0.8	2.8 ± 1.0	0.2	0.8	2.2 ± 0.5	0.227
Grey teal	2.8 ± 0.8	8.7 ± 3.6	6.0 ± 2.6	0.7	1.5	5.2 ± 1.3	0.853
Mallard	2.9 ± 1.1	4.3 ± 1.1	3.1 ± 1.0	2.3	0	3.2 ± 0.6	0.507
Paradise shelduck	0.1 ± 0.1	2.5 ± 1.5	0.0 ± 0.0	0	1.5	0.7 ± 0.4	0.012
Shoveler	0.5 ± 0.3	0.7 ± 0.3	1.8 ± 0.5	0.5	0	1.0 ± 0.2	0.174
Total	12.1 ± 2.6	23.6 ± 5.8	14.4 ± 4.4	4.9	3.9	15.2 ± 2.4	0.084

4.3.2.1 Species abundances

Kruskal-Wallis analysis revealed a significant difference in median Canada goose abundance between ponds, notably those associated with Lake Waikare and the Whangamarino wetland. There were no other significant differences in the median Canada goose abundance per hectare between locations (the one Rangiriri site never had Canada geese on it). Waikare and Whangape ponds had the largest range in Canada goose abundances; the median Waikare abundance was three individuals per hectare, and Whangape's near 0.5/ha; however, there was no statistically significant difference between locations. No other significant differences in median abundance of waterfowl species and location were found through the Kruskal-Wallis analysis; however, there was a significant difference in the mean abundance of paradise shelduck between locations reflecting the high numbers on Lake Whangape.

No significant differences in species abundances were detected for water supply, pond origin, and flood frequency factors. There was, however, a difference in the abundance of paradise shelduck and the dominant pond substrate (Figure 4-1). Abundances on ponds with a silt/clay base ranged from 0 – 12 individuals per hectare; the range on peat was between 0 and <1 individuals per hectare. However, the median number of individuals was similar between the two substrate classes (near zero). No other waterfowl species had a significant difference in abundance between a silt/clay and peat substrate. Refer to Appendix 3 for the abundance per hectare of adult waterfowl species on each pond.
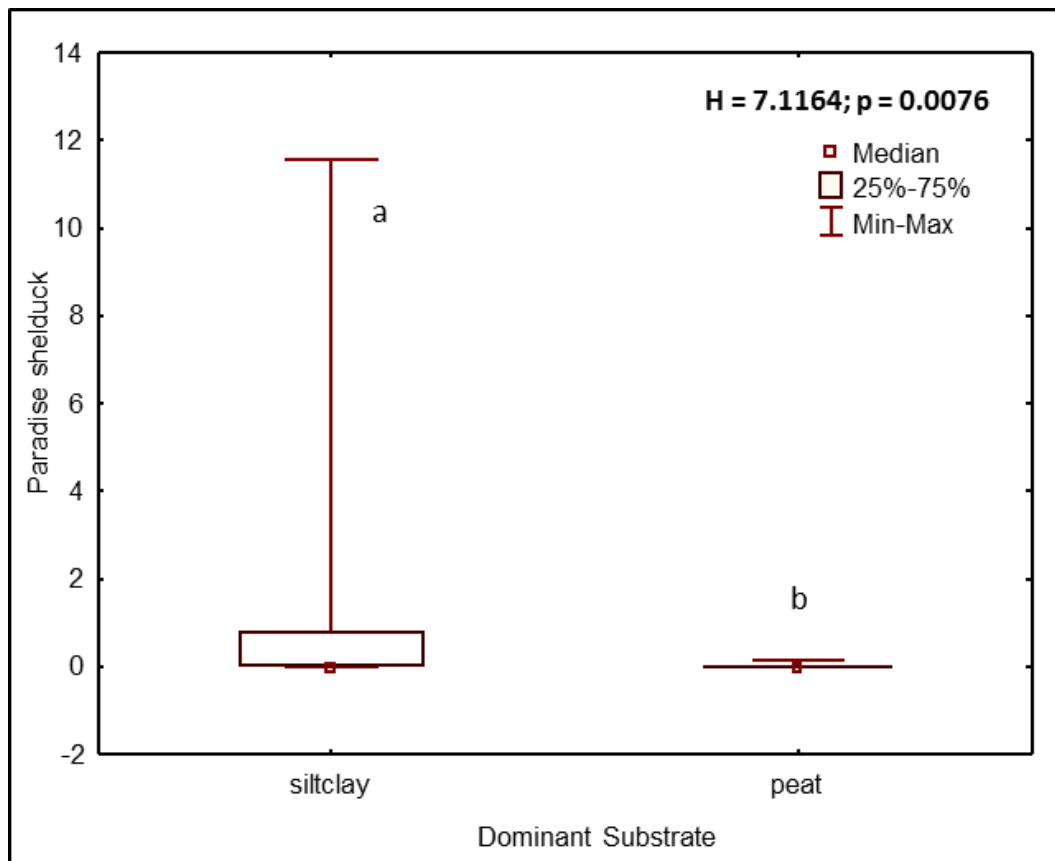
Figure 4-1: Boxplots of differences in median paradise shelduck abundance and dominant substrate composition across 34 constructed ponds on the lower Waikato River floodplain. Classes with the same letter above boxplots are not significantly different.

Kruskal-Wallis analysis identified two waterfowl species as having significantly different abundances between sites that are hunted on, and sites that are not. Mallard abundances were higher on sites where hunting occurred (Figure 4-2A). The median abundance of mallards on sites that are not hunted was near zero, whereas there were around four individuals per hectare on sites that are hunted on. There was also a significant difference in shoveler abundances with more individuals located on sites that are hunted on (Figure 4-2B). The median abundance of shoveler individuals per hectare was near zero on sites which had no hunting and around one per hectare on sites which were hunted on.
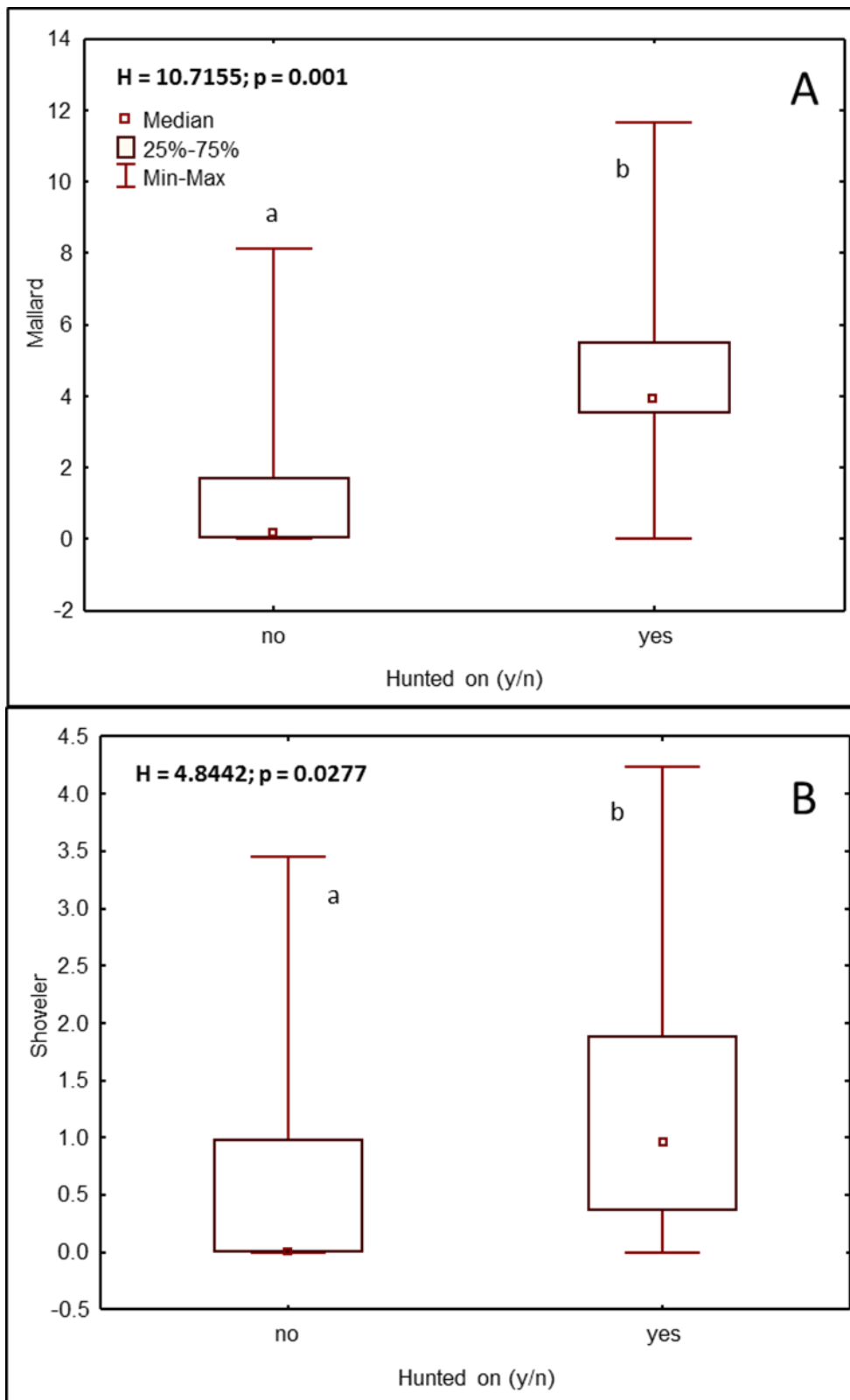
Figure 4-2: Kruskal-Wallis analysis of differences in median mallard abundance per hectare (A) and shoveler abundances (B) between sites that are hunted on, and those that are not. Sites include 34 constructed ponds on the lower Waikato River floodplain. Classes with the same letter above boxplots are not significantly different.

Mallard ducks were the only waterfowl species to show a significant response to the food availability score (Figure 4-3). A food score of 1 had a median mallard abundance of zero, ranging up to four individuals per hectare, while a score of 4 ranged between zero and ten individuals.

In terms of the overhead cover score, shoveler ducks were the only waterfowl species to show differences in abundance, with a significant difference between cover scores of 0 and 5 (Figure 4-4). Where there was no overhead cover (score of 0), shoveler abundance ranged between zero and 0.5 individuals per hectare, while abundances at the highest cover score ranged between one and four individuals per hectare, with a median of 2.5.

Figure 4-3: Box plot of differences in median mallard abundance per hectare relative to the food availability score of 34 constructed ponds on the lower Waikato River floodplain. Classes with the same letter above boxplots are not significantly different.

Figure 4-4: Box plot of differences in media shoveler abundance per hectare and the overhead cover score of 34 constructed ponds on the lower Waikato River floodplain.

Two waterfowl species were identified through Kruskal-Wallis analyses as having significant differences in abundances between nesting habitat scores. Median mallard abundances differed between nesting scores 1 and 2 (Figure 4-5A). The median number of mallards per hectare for a nesting score of 1 was four, compared to zero individuals for a nesting score 2. There were no other significant differences in abundances between other pairings of nesting scores. Nesting score 2 had the narrowest range of mallard abundance (0 – 3.5 mallards/ha), whereas nesting score 1 ranged between zero and 12 individuals per hectare. Nesting score 4 was the only class which the lower quartile did not reach zero mallards per hectare.

There was also a difference in shoveler abundances between nesting scores 1 and 2 (Figure 4-5B). No shoveler ducks were found on ponds with a nesting score of 2, whereas the median abundance on ponds with scores 1, 3, or 4 was between 0.5

and one individual per hectare. Shoveler abundances on ponds with nesting scores 1 and 4 ranged from zero – four individuals per hectare, and zero – 3.5 per hectare for ponds with nesting score 3.

Kruskal-Wallis analyses found no significant differences between loafing habitat score and waterfowl species abundances.
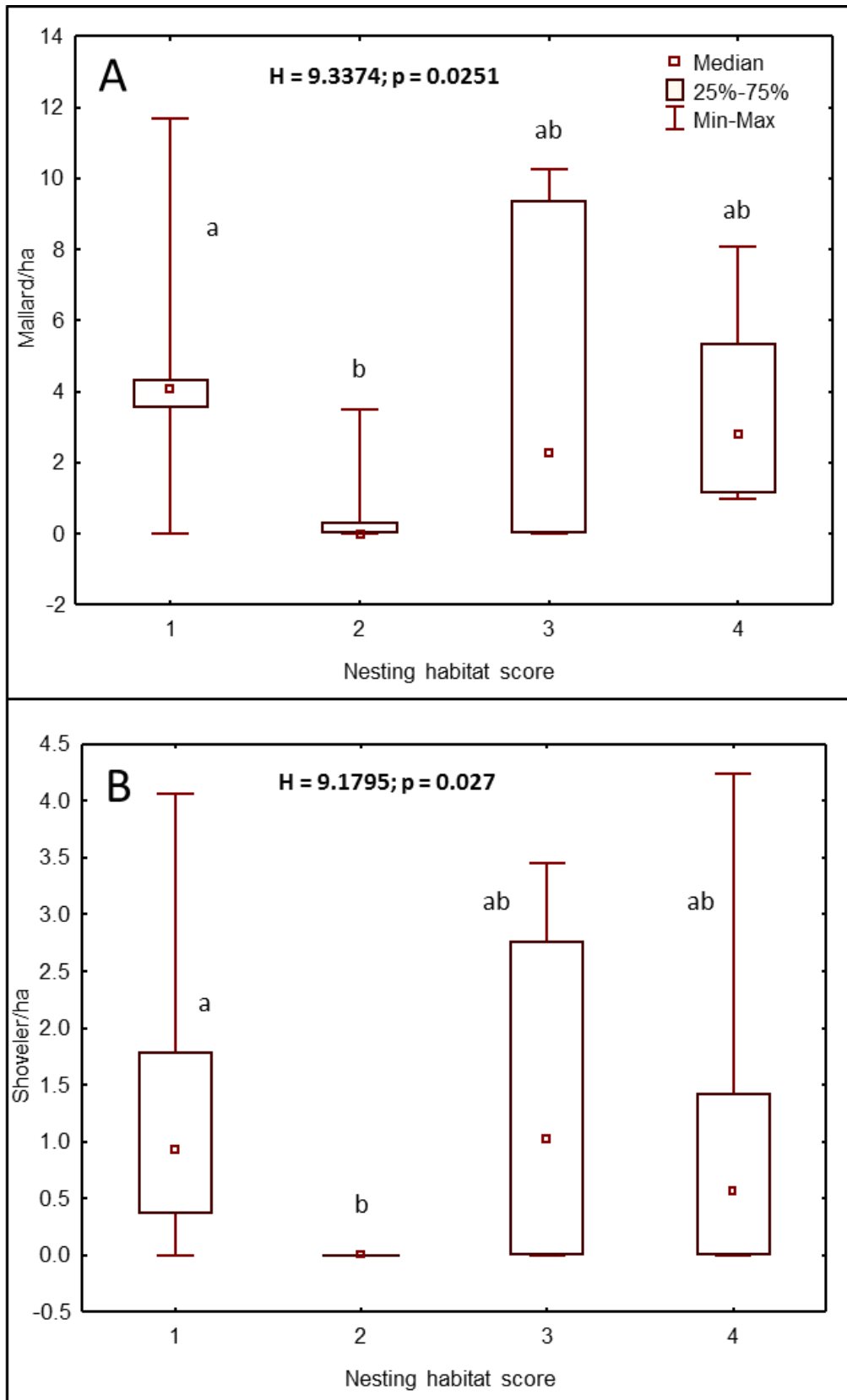
Figure 4-5: Box plot of differences in nesting habitat scores and mallard ducks per hectare (A), and shoveler ducks per hectare (B) across 34 constructed ponds on the lower Waikato River floodplain. Classes with the same letter above boxplots are not significantly different.

4.3.2.2 Waterfowl communities

MDS plots of the waterfowl communities indicated Canada goose and black swan were associated with Lake Waikare ponds, while paradise shelduck populations were closely related to Lake Whangape ponds (Figure 4-6A). Grey duck appeared to be associated with a selection of Whangamarino wetland ponds. Grey teal were loosely associated with Lake Whangape and some Whangamarino wetland ponds. No waterfowl vector trajectories were associated with the communities in the bottom right ordination space, including the two Huntly sites, indicating there was a mix of species at these ponds. The NZ scaup was not included on the vector overlay as it had a Spearman rank correlation $r_s < 0.2$.

Canada goose and black swan appeared to be positively associated with area and edge length of both ponds and islands (Figure 4-6B). The percentage of open water, number of islands, fenced margin, nesting habitat, and percentage of pond 20 – 50 cm deep were also positively associated with Canada goose and black swan. No landscape or water quality vectors were positively associated with paradise shelduck. Shoveler and mallard were positively associated with the number of maimai present on a pond, as well as, the amount of overhead vegetation. The percentages of pond shallower than 20 cm and deeper than 1 m were positively associated with shoveler and grey duck populations. Shoveler and grey duck were also positively associated with pond age class and the pH of the water. Grey teal appeared to have no strong positive or negative associations with landscape and water quality variables.

Pasture grass was the only vegetation type positively associated with paradise shelduck, while riparian flax and aquatic *Glyceria* showed the opposite association (Figure 4-6C). Canada goose and black swan had positive associations with riparian *Carex* and *Juncus*. Hornwort (*Ceratophyllum demersum*), riparian willow weed (*Polygonum persicaria*) and toi toi (*Austroderia* sp.) were positively associated with grey duck. Parrot's feather (*Myriophyllum aquaticum*) was the only aquatic plant positively associated with mallard, along with the riparian mixed bush, *Glyceria*, mixed shrub, and sedge.

Benthic Oligochaeta was the only macroinvertebrate group positively associated with Canada goose and black swan abundances (Figure 4-6D). Grey teal

populations were positively associated with ponds that supported higher benthic Diptera biomass.

PERMANOVA indicated there were significant differences between the waterfowl communities of ponds associated with Lakes Waikare and Whangape, as well as a highly significant difference between Lake Waikare and the Whangamarino wetland ponds (Table 4-5). Five different pairs of water supply classes were identified as significantly influencing waterfowl communities; rain-fed ponds were significantly different to swamp-fed and spring-fed ponds, as well as ponds with other types of water supplies. There was also a difference between drain-fed ponds and swamp-fed ponds, as well as swamp-fed ponds and ponds sourced from other water supplies. Ponds on marginal wetlands had different waterfowl communities than ponds found in wetlands. Flooding had a significant impact on waterfowl communities, with differences occurring between all flood frequency classes (never, occasionally, and annually). Significant differences were also found between ponds which are and are not hunted on, as well as between silt/clay and peat substrates.

DistLM analysis found the landscape variables explained 65% of the variation in waterfowl communities among sites (Table 4-6). Pond area and edge length were both important explanatory variables; yet they were less important than the combined area:perimeter ratio which explained 11% of the variation. The percentage of pond deeper than 1 m was the only other variable which individually explained over 10% of the variation. The top three variables, collectively, explained over 25% of the variation. Four variables were significant ($p = < 0.05$), indicating they made stand-alone contributions to the model. Seven variables contributed more than 3% of the variation explained. All variables were included in the final best-fit model.
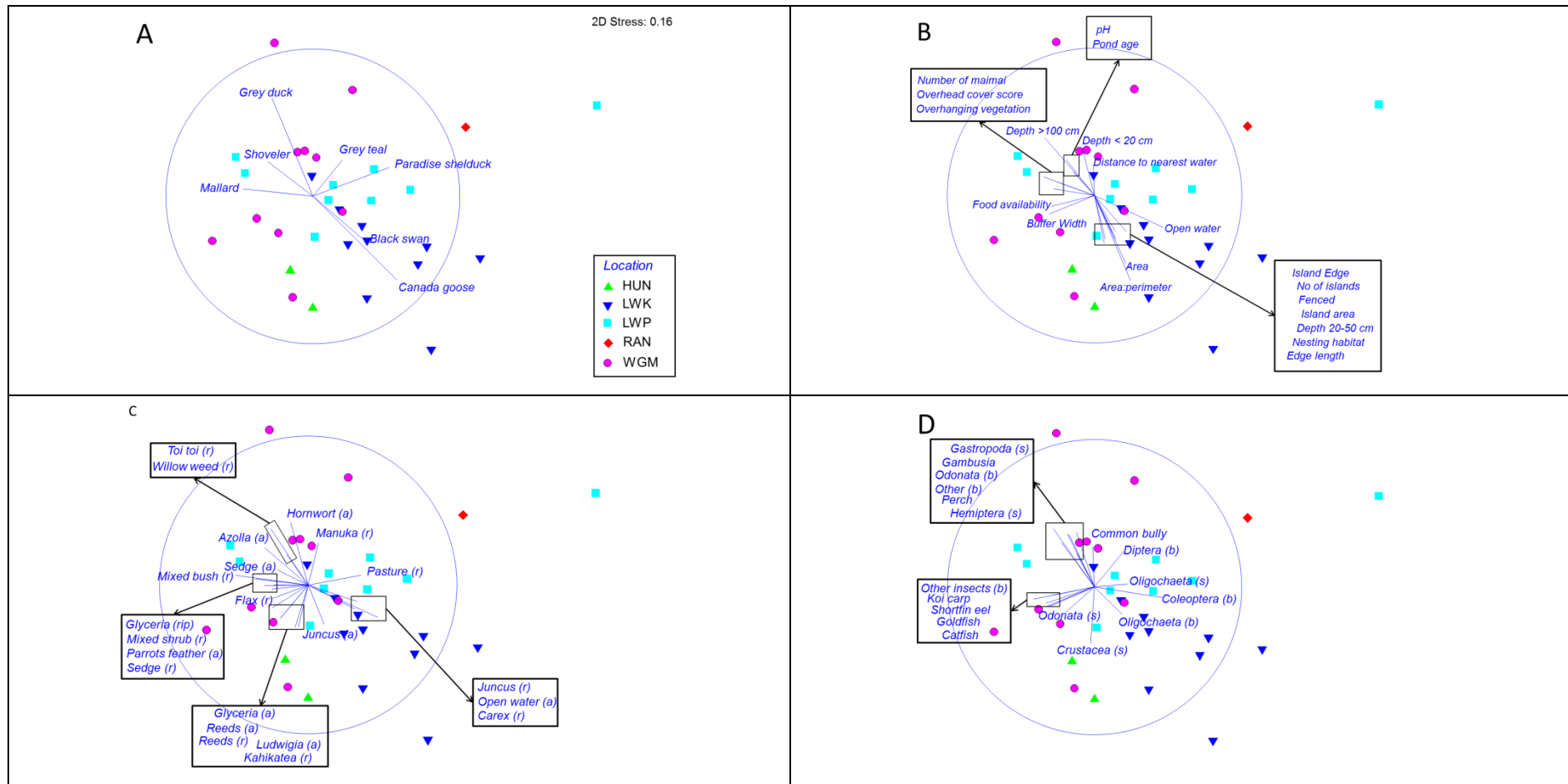
Figure 4-6: MDS plots of the waterfowl community composition of 34 constructed ponds on the lower Waikato River floodplain. A= waterfowl vector overlay (for locations refer to Table 4-4). B= landscape and water quality overlay. C= vegetation overlay (r = riparian; a = aquatic.) D= biotic overlay (b = benthic invertebrates; s = sweep invertebrates). Overlays have been calculated using Spearman rank correlation coefficient r_s >0.2.

Table 4-5: PERMANOVA results of factors significantly influencing the waterfowl community composition of 34 constructed ponds on the lower Waikato River floodplain. Only significant (p < 0.05) results are shown. LWK = Lake Waikare, LWP = Lake Whangape, WGM = Whangamarino wetland.

Source	Pseudo-F value	p-value
Location	2.753	0.001
Water supply	2.681	0.001
Pond origin	2.111	0.035
Flood frequency	2.994	0.003
Hunted	4.172	0.001
Substrate type	2.301	0.049
Pairwise differences	**t statistic**	**p-value**
LWK, LWP	1.847	0.011
LWK, WGM	2.460	0.001
Drain-fed, Swamp-fed	1.577	0.049
Rain-fed, Spring-fed	1.516	0.039
Rain-fed, Swamp-fed	2.669	0.001
Rain-fed, Other	2.200	0.046
Swamp-fed, Other	2.275	0.023
Wetland, Marginal wetland	1.868	0.018
Never floods, Annually floods	1.789	0.015
Never floods, Occasionally floods	1.616	0.029
Annually floods, Occasionally floods	1.935	0.013

Table 4-6: DistLM of landscape features influencing variability among the waterfowl communities on 34 constructed ponds on the lower Waikato River floodplain. Significant p-values are in bold. Only variables explaining > 3% of the variation are included in the table.

Variable	Pseudo-F value	p-value	Proportion	Cumulative R^2
Area:perimeter	3.813	**0.003**	0.113	0.113
Depth >1 m (%)	3.906	**0.003**	0.105	0.218
Area (ha)	2.307	0.054	0.060	0.278
Distance to 5 nearest waterbodies (m)	2.650	**0.027**	0.065	0.342
Edge length (m)	2.497	**0.042**	0.058	0.400
Buffer width (m)	2.054	0.095	0.046	0.445
Depth 51 – 100 cm (%)	1.818	0.135	0.039	0.484

The area:perimeter ratio was still the most important variable contributing to differences between waterfowl communities when all variables were included in the DistLM analysis (Table 4-7). The perceived availability of food and koi carp abundance were also highly significant, with all three variables collectively explaining over 30% of the variation. The percentage of the pond deeper than 1 m was the only other variable which explained more than 5% of the variation. Of the waterfowl habitat identifiers, only the food availability score had any significance and ambient conductivity was the most important water quality variable explaining differences between sites. All other water quality variables individually explained less than 2% of the variation. The top seven variables collectively explained over half the variation among sites. To create the most parsimonious fit, three variables were not included (area, percent of margin fenced, and shortfin eel abundance); all other explanatory variables were included to produce a model which explained 97% of the variation among ponds.

Table 4-7: DistLM of landscape, water quality, and biotic variables which influence the variability among waterfowl communities on 34 constructed ponds on the lower Waikato River floodplain. Significant p-values are in bold. Only variables explaining > 3% of the variation are included in the table.

Variable	Pseudo-F value	p-value	Proportion	Cumulative R^2
Area:perimeter	3.813	**0.003**	0.113	0.113
Food availability score	3.928	**0.003**	0.106	0.219
Koi carp abundance (CPUE^{-1})	4.390	**<0.001**	0.106	0.325
Depth >1 m (%)	3.154	**0.008**	0.071	0.395
Ambient conductivity (μS/cm)	2.291	0.054	0.049	0.444
Sweep invertebrate biomass (g CPUE^{-1})	1.616	0.190	0.034	0.478
Buffer width (m)	2.107	0.087	0.042	0.520
Benthic invertebrate biomass (g CPUE^{-1})	1.769	0.154	0.034	0.554
Depth 51 – 100 cm (%)	1.852	0.145	0.035	0.589
Number of maimai	1.670	0.188	0.030	0.619

Riparian vegetation alone could explain 58% of the variation in waterfowl communities (Table 4-8). Flax and willow bush were the only vegetation types with significant individual effects ($p < 0.05$), and along with mixed bush, explained the first 20% of variation among waterfowl communities. Mixed shrub, *Glyceria*, and pine trees were excluded as explanatory variables in the final step-wise model. Of the tree-like vegetation types, willow bush and manuka each explained over 5% of the variation, while kahikatea bush and oaks explained less than 3% each.

Of the 55% variation in waterfowl communities explained by aquatic vegetation, hornwort accounted for the largest proportion (10%; Table 4-9). Cane rush was the only other aquatic vegetation type that was significant, accounting for 7% of the variation. The percentage of the pond free of vegetation (open water) was the third most important variable influencing waterfowl communities. Hornwort, cane rush, and open water explained the first 22% of the variation, followed by marginal aquatic species taking the cumulative R^2 to 0.4.

Table 4-8: DistLM of influence of different riparian vegetation types on the variability among waterfowl communities of 34 constructed ponds on the lower Waikato River floodplain. Significant p-values are in bold. Only variables explaining > 3% of the variation are included in the table.

Variable (% cover)	Pseudo-F value	p-value	Proportion	Cumulative R^2
Flax	2.729	**0.023**	0.083	0.083
Mixed bush	2.288	0.059	0.067	0.150
Willow bush	2.521	**0.040**	0.070	0.221
Carex	1.919	0.112	0.052	0.272
Manuka	2.003	0.099	0.052	0.324
Toi toi	1.849	0.148	0.047	0.371
Reeds	1.460	0.203	0.036	0.407
Rank grass	1.212	0.321	0.030	0.437

Table 4-9: DistLM of the influence different aquatic vegetation types on the variability among waterfowl communities of 34 constructed ponds on the lower Waikato River floodplain. Significant p-values are in bold. Only variables explaining > 3% of the variation are included in the table.

Variable (% cover)	Pseudo-F value	P-value	Proportion	Cumulative R^2
Hornwort	3.444	**0.016**	0.103	0.103
Cane rush	2.538	**0.048**	0.072	0.175
Open water	1.854	0.122	0.051	0.226
Willow weed	2.151	0.069	0.057	0.283
Parrots feather	1.869	0.115	0.048	0.332
Ludwigia	1.268	0.292	0.032	0.364
Glyceria	1.560	0.201	0.039	0.403
Carex	1.570	0.184	0.038	0.441

4.3.3 Juveniles

In total 592 juvenile waterfowl were encountered over the course of the four visits, of which 451 were only counted once. Three ducklings observed at LWP_3 were unable to be identified to species level so were classified as dabbling ducklings. In total 263 dabbling ducklings were produced across 17 sites, with none being encountered on the remaining ponds. Thirteen sites did not produce any juvenile waterfowl, including eight sites from the Whangamarino wetland; as a result these sites were omitted from further analyses. LWP_3 produced the most juvenile waterfowl (67), closely followed by LWP_8 which produced 66. A single mallard duckling and one grey teal duckling were counted on LWK_8 and WGM_5, respectively. Twelve sites produced more than 10 juvenile waterfowl; the number of juvenile waterfowl remained in single figures for nine sites.

Mallard produced the most juveniles, with 104 ducklings being counted across 10 sites. Seventy-two black swan cygnets were also found at ten different sites. Canada goose goslings were the second most abundant juvenile birds with 80 found, of which 34 were encountered on LWK_4. Grey duck and grey teal produced 59 ducklings apiece, while 38 shoveler and 36 paradise shelduck ducklings were counted.

Mallard had a mean juvenile abundance of 2.5 ha^{-1}; paradise shelduck and shoveler had the lowest average abundance at 0.7 juvenile's ha^{-1} (Table 4-10). Black swan and paradise shelduck had higher mean abundances in Lake Whangape ponds; in both cases no juveniles were encountered in Whangamarino wetland ponds. Canada goose had a higher mean abundance in Lake Waikare ponds, with Huntly being the only other location where Canada goose was present. Lake Whangape ponds produced, on average, 19.3 juveniles ha^{-1}, with all other locations remaining in single figures. Refer to Appendix 3 for the abundance per hectare of juvenile waterfowl species on each pond.

Mann-Whitney U tests revealed factors affecting the probability of juvenile presence on a pond varied for different species (Table 4-11). Grey duck and mallard were the only waterfowl species which had more than one factor influencing the probability of juveniles being present. Juvenile grey ducks were more likely to be present on ponds with high sweep Oligochaeta biomass, and a lower relative abundance of riparian rank grass. Mallard juveniles were more likely to be present on larger ponds with a high area:perimeter ratio. Sites with juvenile mallard present had greater distance to the five nearest waterbodies, more pond fencing, higher abundance of pasture grass, and lower buffer width, percentage of pond shallower than 20 cm, sweep Odonata biomass, and cover by willow bush. Overall, ponds that were larger, more isolated from other waterbodies and with more fencing, narrower buffer widths and depths between 51 and 100 cm, were more likely to have juvenile waterfowl present.

Table 4-10: Mean ± standard error of juvenile waterfowl abundances per hectare by location. LWK = Lake Waikare, LWP = Lake Whangape, WGM = Whangamarino wetland, HUN = Huntly, RAN = Rangiriri. *n* = number of ponds. p-values were calculated using Kruskal-Wallis analysis.

Waterfowl	LWK	LWP	WGM	HUN	RAN	Overall	p
	n = 11	*n* = 9	*n* = 11	*n* = 2	*n* = 1	*n* = 34	
Black swan	1.0 ± 0.5	4.6 ± 2.1	0.0 ± 0.0	0.5	0	1.6 ± 0.6	0.027
Canada goose	3.9 ± 1.3	0.0 ± 0.0	0.0 ± 0.0	0.4	0	1.3 ± 0.5	0.008
Grey duck	0.7 ± 0.7	3.8 ± 2.1	2.3 ± 1.6	0	0	2.0 ± 0.8	0.159
Grey teal	1.2 ± 0.6	1.1 ± 0.6	3.2 ± 2.9	0	0	1.7 ± 1.0	0.766
Mallard	0.4 ± 0.3	7.1 ± 4.4	0.7 ± 0.7	4.9	0	2.5 ± 1.3	0.071
Paradise shelduck	0.0 ± 0.0	1.7 ± 1.2	0.0 ± 0.0	0	9.2	0.7 ± 0.4	0.007
Shoveler	0.7 ± 0.6	1.0 ± 0.7	0.8 ± 0.8	0	0	0.7 ± 0.4	0.899
Total	7.8 ± 2.8	19.3 ± 5.0	7.0 ± 5.7	5.7	9.2	10.5 ± 2.5	0.047

Table 4-11: Z values of factors affecting juvenile presence on ponds where adults are present. Only significant variables included in table. ↑/↓ indicate factor is increased or decreased when juveniles are present. n = number of ponds with adults present. * $p < 0.05$, ** $p < 0.01$. (s) sweep sample, (b) benthic sample.

Variable	Black swan	Canada goose	Grey duck	Grey teal	Mallard	Paradise shelduck	Shoveler	Combined
	n = 18	n = 20	n = 24	n = 24	n = 25	n = 8	n = 19	n = 32
Landscape								
Area (m^2)	-	-	-	-	↑ 2.247*	-	-	↑ 2.301*
Area:perimeter	-	-	-	-	↑ 2.025*	-	-	↑ 2.301*
Distance to 5 waterbodies (m)	-	-	-	-	↑ 2.413*	-	-	↑ 1.984*
Fenced (%)	-	↑ 2.219*	-	-	↑ 2.247*	-	-	↑ 2.420*
Buffer width (m)	-	-	-	-	↓ -2.690**	-	-	↓ -2.837**
Depth <20 cm (%)	-	-	-	-	↓ -2.552*	-	-	-
Depth 51-100 cm (%)	-	-	-	-	-	-	-	↑ 2.063*
Depth >1m (%)	-	-	-	-	-	↑ 2.087*	-	-
Biota (g CPUE^{-1})								
Oligochaeta (s)	-	-	↑ 2.174*	-	-	-	-	-
Odonata (s)	-	-	-	-	↓ -2.052*	-	-	-
Hemiptera (b)	↑ -1.983*	-	-	-	-	-	-	-
Other (b)	-	-	-	-	-	-	↓ -2.011*	-
Riparian Vegetation (%)								
Rank grass	-	-	↓ -3.001**	↓ -2.534*	-	-	-	-
Pasture grass	-	-	-	-	↑ 2.302*	-	-	-
Willow bush	-	-	-	-	↓ -2.080*	-	-	-

Where juvenile waterfowl were present, communities on Lake Waikare ponds were characterised by Canada goose, grey teal, and shoveler, while paradise shelduck ducklings were associated with the Rangiriri pond (Figure 4-7A). Grey duck, mallard, and black swan juveniles appeared to not be associated with ponds from specific locations.

Ponds that supported higher numbers of mallard ducklings tended to have greater area:perimeter ratio, shallower water and be further away from other waterbodies. In contrast, ponds characterised by higher abundances of black swan cygnets tended to have more overhead/hanging vegetation, maimai, and available food. There were strong associations between ponds supporting more grey duck ducklings and the age, and percentage of water shallower than 20 cm. Juvenile communities dominated by Canada goose goslings tended to be characterised by high ambient conductivity, number of islands, island edge length and area, area of macrophyte-free open water, nesting habitat, and turbidity. In contrast, higher water pH and longer distance to the nearest five waterbodies were characteristic of ponds with more grey teal ducklings.

Ponds with juvenile waterfowl communities dominated by Canada goose goslings were characterised by more riparian *Glyceria* and raupo, as well as a higher percentage of open water free of macrophytes. Toi toi was the only vegetation type strongly associated with grey teal ducklings, while aquatic *Glyceria* tended to be characteristic of ponds dominated by paradise shelduck and shoveler ducklings. Grey duck duckling populations had a positive relationship with aquatic sedge, willow trees, and *Azolla*, and negative associations with riparian flax. Kahikatea and duckweed, and riparian and aquatic reeds were positively associated with mallard ducklings; aquatic *Glyceria* had a negative relationship with mallard ducklings. Aquatic *Carex* and *Ludwigia* positively characterised ponds with black swan cygnets, while *Glyceria* and raupo were negatively associated.

Mallard duckling populations had a positive association with benthic Oligochaeta and sweep Crustacea. Black swan cygnets were positively associated with benthic Gastropoda and Diptera, and common smelt, gambusia and koi carp fish species. Benthic Hemiptera was the only biota with a positive association to Canada goose goslings. Shoveler and paradise shelduck ducklings had a positive relationship

with benthic Odonata, sweep Oligochaeta, Coleoptera, Odonata, and Gastropoda macroinvertebrates, and the common bully fish species. No biota taxa were positively associated with grey teal ducklings.
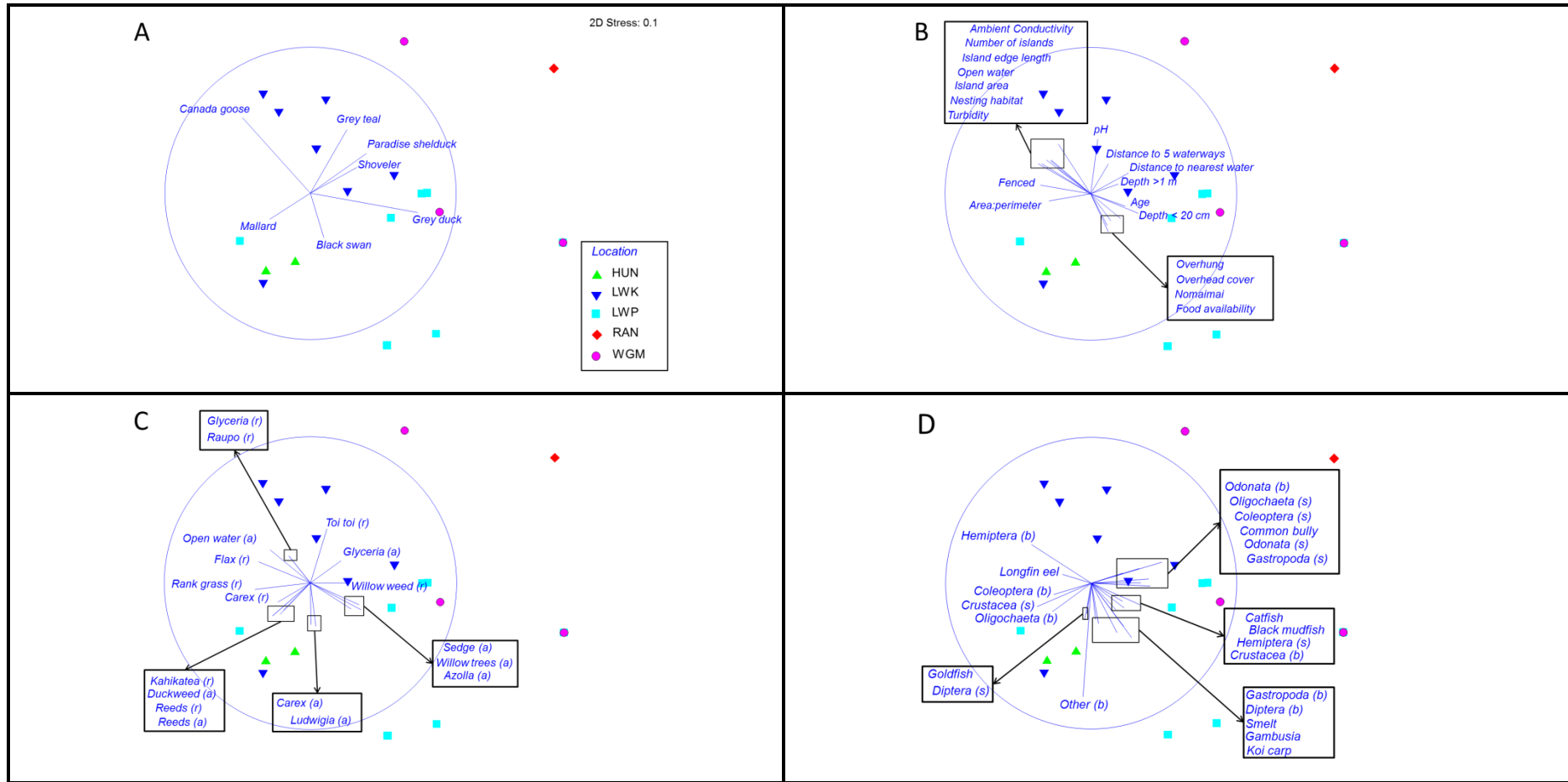
Figure 4-7: MDS plots of the juvenile waterfowl community composition of 21 constructed ponds on the lower Waikato River floodplain; with: A = waterfowl, B = landscape and water quality, C = vegetation, and D = biotic vector overlays. Refer to Table 4-4 for locations. Overlays have been calculated using Spearman rank correlation coefficient r_s >0.2.

Of the eight factors tested in PERMANOVA, only two were found to have a significant impact on juvenile waterfowl community composition (Table 4-12). Two pairs of location groups had significantly different juvenile waterfowl communities; Lake Waikare pond communities were significantly different to the communities of Lake Whangape and the Whangamarino wetland ponds. Pond water supply also had a significant influence on juvenile waterfowl communities, with drain-fed ponds having different communities to both spring-fed and swamp-fed ponds.

Table 4-12: PERMANOVA results of factors significantly influencing the juvenile waterfowl community composition of 21 constructed ponds on the lower Waikato River floodplain. Only significant ($p < 0.05$) results are shown. LWK = Lake Waikare, LWP = Lake Whangape, WGM = Whangamarino wetland.

Source	Pseudo-F value	P-value
Location	2.790	<0.001
Water supply	1.991	0.011
Pairwise differences	**t statistic**	**P-value**
LWK, LWP	2.151	0.002
LWK, WGM	1.756	0.028
Drain-fed, Spring-fed	1.600	0.030
Drain-fed, Swamp-fed	1.559	0.027

DistLM analysis found the percentage of pond deeper than 1 m to be the only significant variable, accounting for 10% of the variation, although the full model explained over 90% (Table 4-13). Including all variables in the DistLM analysis raised the R^2 value to 0.999, with the percentage of pond deeper than 1 m and nesting habitat score being the only significant variables, explaining 11 and 12% of the variation, respectively (Table 4-14). Fourteen variables were dismissed as explanatory variables from the most parsimonious model.

Table 4-13: DistLM of landscape variables explaining variation among juvenile waterfowl communities of 21 constructed ponds on the lower Waikato River floodplain. Significant p-values are in bold.

Variable	Pseudo-F value	P-value	Proportion	Cumulative R^2
Depth >1 m (%)	2.264	**0.039**	0.106	0.106
Area:perimeter	1.326	0.257	0.061	0.168
Area (ha)	1.587	0.156	0.071	0.239
Overhanging vegetation (%)	1.297	0.278	0.057	0.296
Number of islands	1.487	0.206	0.063	0.359
Distance to 5 nearest waterbodies (m)	1.268	0.283	0.053	0.413
Macrophyte-free open water (%)	1.683	0.152	0.067	0.479
Depth 51 - 100cm (%)	1.335	0.274	0.052	0.532
Age (class)	0.899	0.488	0.035	0.567
Buffer width (m)	0.939	0.453	0.037	0.604
Number of maimai	1.170	0.336	0.046	0.650
Edge length (m)	0.694	0.574	0.028	0.678
Distance to nearest waterbody (m)	0.418	0.762	0.018	0.696
Fenced (%)	1.101	0.379	0.047	0.743
Island edge length	0.578	0.629	0.027	0.770
Depth <20 cm (%)	1.376	0.297	0.059	0.829
Depth 20-50 cm (%)	1.688	0.263	0.062	0.890
Island area (m^2)	0.353	0.713	0.016	0.907

Table 4-14: DistLM of landscape, water quality, and biotic variables which influence the variability among juvenile waterfowl communities of 21 constructed ponds on the lower Waikato River floodplain. Significant p-values are in bold. Only variables explaining > 3% of the variation are shown in the table.

Variable	Pseudo-F value	P-value	Proportion	Cumulative R^2
Depth >1 m (%)	2.264	**0.043**	0.106	0.106
Food availability score	1.807	0.103	0.082	0.188
Nesting habitat score	2.995	**0.009**	0.122	0.310
Turbidity (rank)	1.654	0.156	0.065	0.374
Goldfish abundance (CPUE^{-1})	1.583	0.174	0.060	0.434
Area:perimeter ratio	1.462	0.227	0.054	0.488
Distance to 5 nearest waterbodies (m)	2.307	0.069	0.077	0.565
Macrophyte-free open water (%)	1.274	0.305	0.042	0.607
Overhanging vegetation (%)	1.202	0.335	0.039	0.645
Number of islands	1.862	0.147	0.056	0.701
Ambient conductivity (µS/cm)	1.010	0.418	0.030	0.731
Island edge length (m)	1.021	0.408	0.030	0.762
Depth <20 cm (%)	1.083	0.373	0.032	0.794
Fenced (%)	1.902	0.161	0.050	0.843
Island area (m^2)	1.933	0.157	0.044	0.887
Number of maimai	1.822	0.218	0.035	0.922
Shortfin eel abundance (CPUE^{-1})	3.094	0.127	0.039	0.962

Riparian vegetation alone could explain 79% of the variation in juvenile waterfowl communities, although none of the individual vegetation types had statistically significant influences (Table 4-15). *Glyceria* was omitted from the final step-wise model as it did not add to the variance explained.

Aquatic vegetation alone accounted for 63% of the variation among juvenile waterfowl communities (Table 4-16). *Juncus*, hornwort, and cane rush were not included in the analysis as they were only found at sites with no juvenile waterfowl. Duckweed was considered the most influential in determining juvenile waterfowl communities and was the only vegetation type that was significant, explaining more than 10% of the variation.

Table 4-15: DistLM of influence of different riparian vegetation types on the variability among juvenile waterfowl communities of 21 constructed ponds on the lower Waikato River floodplain. Only variables explaining > 2% of the variation are included in the table.

Variable (% cover)	Pseudo-F value	p-value	Proportion	Cumulative R^2
Rank grass	1.667	0.146	0.081	0.081
Mixed bush	2.072	0.065	0.095	0.176
Toi toi	1.975	0.072	0.086	0.261
Reeds	1.534	0.202	0.065	0.326
Flax	1.336	0.273	0.055	0.381
Willow bush	0.988	0.426	0.041	0.422
Kahikatea stand	1.322	0.299	0.053	0.475
Willow weed	1.048	0.447	0.042	0.517
Pasture grass	1.088	0.379	0.043	0.561
Oak trees	1.387	0.259	0.054	0.614
Carex	0.524	0.716	0.021	0.636
Mixed shrub	0.517	0.716	0.022	0.658
Juncus	2.634	0.059	0.094	0.751
Raupo	0.977	0.447	0.035	0.786

Table 4-16: DistLM of the influence different aquatic vegetation types on the variability among juvenile waterfowl communities of 21 constructed ponds on the lower Waikato River floodplain. Significant p-values are in bold. Only variables explaining > 2% of the variation are included in the table.

Variable (% cover)	Pseudo-F value	P-value	Proportion	Cumulative R^2
Duckweed	2.354	**0.030**	0.110	0.110
Azolla	1.835	0.096	0.082	0.193
Sedge	1.720	0.126	0.074	0.267
Willow weed	1.308	0.274	0.055	0.322
Willow trees	1.212	0.325	0.051	0.373
Open water	1.153	0.346	0.048	0.421
Reeds	1.102	0.358	0.045	0.466
Carex	0.684	0.598	0.029	0.495
Ludwigia	0.793	0.534	0.034	0.529
Ricciocarpus natans	1.323	0.290	0.055	0.584
Glyceria	0.458	0.749	0.020	0.604
Parrots feather	0.589	0.674	0.027	0.631

4.4 Discussion

The results of this chapter were used to determine (i) which environmental factors best explained variations in adult waterfowl communities; and (ii) whether these factors differed for juvenile waterfowl in constructed ponds. Numerous ponds are being created on the lower Waikato River floodplains to increase wetland habitat and enhance duck shooting opportunities, but some waterfowl populations are still declining (D. Klee, Fish and Game, Auckland-Waikato Region, pers. comm). This research has assisted in identifying which features could be included in constructed pond design to increase waterfowl abundance and productivity.

4.4.1 Land-use

Kuczynski and Paszkowski (2012) found a significant difference in waterfowl assemblages with regards to surrounding land use that included a gradient from agricultural, mixed habitat, and forested ponds. They found waterfowl assemblages on agricultural ponds were significantly different from the assemblages on both mixed-habitat and forested ponds; also there was no difference in waterfowl assemblage between mixed habitat and forested ponds. Constructed ponds in the Waikato region displayed a similar gradient, with pond origin being used as a surrogate for surrounding land use comprising grazed land (agricultural; 100% of pond margin is grazed), marginal wetlands (mixed habitat; <80% of the pond margin is grazed or the pond margin can only be grazed during the dry season), and forested wetlands dominated by willow scrub. However, I found the only difference in waterfowl community composition to be between ponds originating in wetlands, and those on marginal wetlands. This significant difference may be due to underlying differences in habitat across the three groups. Kuczynski and Paszkowski (2012) noted forested ponds in their study tended to be larger and shallower than the agricultural ponds which had less vegetation than other pond types. In my study, marginal wetland and wetland ponds were spatially isolated. Ponds originating from marginal wetlands were only found around Lake Waikare (LWK), whereas ponds of wetland origin tended to be located in the Whangamarino wetland (WGM). Co-varying factors associated with location or hydrology may also explain the significant difference found between waterfowl communities of LWK and WGM. Although water supply and pond substrate had

significant influences on waterfowl community composition it is likely these are functions of location. Swamp-fed ponds were predominantly located in the Whangamarino wetland, rain-fed ponds around Lake Waikare, and spring-fed and drain-fed ponds around Lakes Waikare and Whangape. WGM ponds mostly consisted of peat substrate, whereas silt/clay based ponds were found in all other locations.

4.4.2 Hydrology

Waterfowl in Manitoba, Canada, showed a negative response to flooding due to changes in food supply (Murkin and Kadlec, 1986). Their study indicated heightened water levels during floods caused benthic macroinvertebrates to become inaccessible and potentially allowed pelagic macroinvertebrates to transfer to waters too deep for waterfowl to feed in. Flooding may also negatively impact waterfowl during the breeding season as water levels cover emergent vegetation, thus reducing potential areas of vegetation cover for juveniles (Murkin et al., 1982). I found significant differences in waterfowl community composition between all flood classes except ponds which annually and occasionally flood, predominantly in the Whangamarino wetland. However, this study was conducted the year after a severe drought which may explain some of these findings. The LWK ponds never flood and had more Canada goose and black swan on them, while the WGM ponds flood annually and had considerably more grey teal. The LWP sites were a mixture of ponds which never, occasionally and annually flood, and this hydrological diversity may have contributed to them having, on average, more waterfowl than LWK and WGM ponds. It would appear a landscape of ponds with a mixture of flooding frequencies will support a higher density and diversity of waterfowl than landscapes with a homogenous flooding regime.

4.4.3 Landscape conditions

Physical complexity

Numerous studies in the Northern Hemisphere have indicated that factors such as pond complexity, proximity to other ponds, and habitat heterogeneity influence

waterfowl communities (Austin, 2002; Austin and Buhl, 2009; Flake et al., 1977; Locky et al., 2005; Mack and Flake, 1980; Soulliere and Monfils, 1996; Walker et al., 2013). I found similar results for constructed ponds on the lower Waikato River floodplain. Pond area, edge length and area:perimeter ratio were considered measures of pond complexity, while the distance to the nearest five waterbodies indicated the influence of proximity to alternative habitats.

Larger ponds have been shown to maintain larger waterfowl communities (Austin and Buhl, 2009; Cowardin et al., 1998; Sanchez-Zapata et al., 2005) while a number of studies indicate shoreline complexity is an important factor for brooding waterfowl (Austin and Buhl, 2009; Mack and Flake, 1980). It has been suggested small ponds are still important, especially during the breeding phase as numerous small ponds provide more nesting habitat than a few large ponds (Walker et al., 2013). Waterfowl observations for my study were conducted during the breeding/brooding phase, which may explain the higher abundance of waterfowl in LWP ponds as there was a wide range of pond sizes across this landscape. The HUN ponds were also relatively large yet had lower mean abundance of waterfowl than LWK, LWP and WGM ponds; however, only two ponds were sampled near Huntly which reduces the sample size.

I found area:perimeter ratio consistently explained a large portion of variation in waterfowl community composition between ponds. During the brooding phase, waterfowl likely select ponds with a high shoreline complexity for a number of behavioural reasons, including increased shallow foraging sites, loafing sites, escape cover, and visual isolation from other broods (Austin and Buhl, 2009; Mack and Flake, 1980). Similarly, it seems that for ponds in the Waikato region shoreline complexity can influence waterfowl community composition, but does not increase waterfowl abundances.

Cowardin et al. (1998) found the number of waterfowl pairs was positively associated with pond area in North America; waterfowl then relocated to smaller ponds for the brooding phase (Walker et al., 2013). Shoreline length can be an important landscape factor for broods; Mack and Flake (1980) found duck broods occupied ponds with a shore length twice that of ponds with no broods. I found ponds supporting waterfowl broods had greater pond area, larger percent of

margin fenced and increased area:perimeter ratio compared to ponds where adults were present but there were no broods. However, mallard was the only species to show this response, and also to indicate an effect of water depth of brood occurrence. Increased shoreline length and complexity creates more bays and shallow waters which ducks, such as mallard, utilise for foraging and loafing (Mack and Flake, 1980). However, in my study the percent of shallow water (<20 cm depth) was lower in ponds with mallard broods suggesting juvenile feeding and foraging is not limited to shallow waters. Mallard broods in my study may be positively associated with increased shoreline complexity for other reasons, such as increased loafing sites, escape cover, or the use of embayments as visual cover from other broods (Austin and Buhl, 2009).

Pond area is assumed to be reason for the significant influence hunting had on waterfowl communities. Hunting predominantly occurred on ponds which had relatively high waterfowl densities. However, waterfowl hunting occurred less on ponds with high shoreline complexity which is likely due to lack of visibility of waterfowl.

Pond proximity

It is widely suggested waterfowl abundance is negatively related to the distance to nearby waterbodies because proximity to other waterbodies may be important as alternative habitats during extreme events such as droughts, floods and cold snaps (Austin, 2002; Austin and Buhl, 2009; Evans and Black, 1956; Flake et al., 1977; Johnson and Grier, 1988; Kirby, 1995; Kloskowski et al., 2009). These studies tended to refer to wintering waterfowl populations. My waterfowl observations began in late winter 2013, which may explain why proximity to nearby water appeared to strongly affect the variation in waterfowl communities of my study.

I found more broods as distance from other waterbodies increased, which was contrary to the literature. For example, pond proximity and density were identified as important for waterfowl brood use of ponds in North America (Austin and Buhl, 2009; Lokemoen, 1973; Mack and Flake, 1980). Flake et al. (1977) identified mallard as a potential exception to this rule, which may explain why I found a

positive effect of increasing distance to the nearest five waterbodies on mallard broods. The LWP ponds had a higher mean brood occupancy than the other locations, and also had some of the highest distances to the nearest five waterbodies, although distance was often small to the single nearest waterbody (<250 m on average).

4.4.4   Vegetation structure

Waterfowl communities and abundances have a positive association with vegetation, especially emergent macrophytes and riparian cover (del Hoyo et al., 1992; Kuczynski and Paszkowski, 2012; Soulliere and Monfils, 1996; Walker et al., 2013). Bélanger and Couture (1988) indicated ponds with ≥30% of the surface area covered by emergent vegetation were used more by dabbling ducks in the brooding phase; however, no such relationship was found in the Hawke's Bay region, New Zealand, by McDougall et al. (2009). In that study, there was no relationship between emergent vegetation and pond use by mallard and grey duck. I found hornwort and cane rush were the only aquatic vegetation types significantly influencing variation in waterfowl community composition; however, both species were only found in two ponds which limits the inferences that can be made. Constructed ponds in the Waikato region appeared to support the findings of McDougall et al. (2009) rather than the overseas studies.

4.4.5   Food availability

Invertebrates are an important food source for adult ducks and ducklings; adult ducks require a high energy diet for egg production (Alisauskas and Ankney, 1992) and ducklings exclusively eat invertebrates for the first fortnight of their lives (Cox et al., 1998; Street, 1977). Invertebrates are still an important food source after the first two weeks as they grow towards the fledgling stage (Sedinger, 1992); therefore, food availability is important for all life stages (Gunnarsson et al., 2004). Dabbling ducklings were associated with shallow waters (<20 cm depth) and deeper waters (>1 m depth) which may provide different feeding and foraging habitats as they feed on benthic invertebrates in shallow waters and emergent invertebrates in deeper waters. Diets of adult ducks

and ducklings are likely to overlap during the brooding phase, but predominantly adults feed on benthic invertebrates and ducklings feed on emerging invertebrates (Dessborn et al., 2011) suggesting the feeding niche is different. Therefore, shallower waters are likely required for adult feeding behaviour and deeper water may be used by broods foraging on the surface. The brooding season coincides with the emergence of Diptera, a dominant food source for mallard (Dessborn et al., 2011), which suggests ducks are able to take advantage of increases in previously inaccessible Diptera near the surface of deeper waters.

Studies have shown Diptera, especially Chironomidae, as the dominant food source for ducklings, as well as Corixidae, Coleoptera, Mollusca, and Asellidae (Danell and Sjoberg, 1980; Lees and Street, 1974; Street, 1977). In my study, ponds which had broods on them did not have obvious increased abundances of invertebrate taxa highlighted in the literature compared to ponds with no broods; Diptera, Corixidae, and Coleoptera were not significantly different in ponds with and without broods. However, there were strong associations between mallard broods and Diptera, grey duck broods and Hemiptera, and Coleoptera and the broods of shoveler, paradise shelduck and grey teal. This suggests food supply may affect juvenile abundance but not presence. Presence and hatching success may be more affected by nesting cover and predation; however, there were no strong associations with riparian buffer vegetation and nest predation was not monitored. The literature cited above refers to invertebrate abundance whereas I only looked at the relationships between duckling abundances and invertebrate biomass (refer to Chapter 5 for further discussion)

Although young ducklings feed almost exclusively on macroinvertebrates, seeds are still an important component of their diet (Cox et al., 1998; Dessborn et al., 2011; Lees and Street, 1974; Sedinger, 1992; Street, 1977; Sugden, 1973). Seeds become an ever increasing component of a duckling's diet as they grow; Street (1977) found 95 percent of a duckling's diet can comprise seeds immediately prior to fledgling. The importance of seeds in the diet of adult and juvenile ducks could explain why the food availability score featured higher than invertebrate abundances for adults and ducklings, since it represents the importance of invertebrates and seeds as food stores. Dessborn et al. (2011) found adults and ducklings fed heavily on *Carex* seeds. This likely explains the positive association

between mallard ducklings and riparian *Carex*; however, an association was only evident for adult ducks.

The abundance, habitat use and survival of waterfowl have been negatively associated to fish presence in ponds (Bouffard and Hanson, 1997; Haas et al., 2007). Fish affect the food source of waterfowl by consuming benthic and planktonic invertebrates before they are readily accessible to the birds (Eriksson, 1983). However, there are also reports of fish having no such effect on waterfowl (Eriksson, 1983; Paszkowski and Tonn, 2000). In my study, adult waterfowl communities appeared to be significantly affected by koi carp, possibly because the feeding behaviour of koi carp had a negative influence on macroinvertebrate abundance. Hanchet (1990) has noted koi carp can significantly reduce macrophyte abundance in New Zealand waterbodies. Therefore, koi carp may also be reducing the seed abundance in the ponds.

4.4.6   Summary

Habitat heterogeneity at the landscape scale appears to increase the likelihood of pond occupancy by both adult and juvenile waterfowl. To increase the number of waterfowl on a landscape, a range of habitats are important. Locations with a range of pond sizes and flooding regimes, such as LWP, had the highest waterfowl abundances. Shoreline complexity is often discussed in the literature alongside pond area as important for waterfowl abundances; however, I found shoreline complexity to have negligible effect on waterfowl abundance but a notable effect on community composition. Therefore, in the Waikato, large ponds with complex shorelines will likely hold more waterfowl, more species and more juveniles. There was no indication of reliance of juvenile waterfowl on macroinvertebrate biomass in the Waikato; although, most past studies identified invertebrate abundances as influential for waterfowl. While koi carp may have a significant impact on waterfowl communities, it is not known whether this is due to impacts on macrophytes or macroinvertebrates as food sources. However, it is likely the exclusion of exotic fish will increase the macroinvertebrate component of the food source for waterfowl in ponds. Future studies should focus around temporal variation in macroinvertebrate communities throughout the brooding

phase. To better understand waterfowl productivity in terms of the number of juveniles' fledgling, more frequent waterfowl observations will increase the chance of witnessing fledged birds on ponds, as well as selecting ponds known to produce large numbers of waterfowl broods.

Chapter 5
General Discussion

The overarching aim of this thesis was to identify factors which potentially limit waterfowl productivity and density in constructed wetlands on the lower Waikato River floodplain. While there is extensive literature indicating the pond features waterfowl respond positively and/or negatively to (Austin and Buhl, 2009; Gibbs, 1993; Lokemoen, 1973; Mack and Flake, 1980; Sanchez-Zapata et al., 2005; Walker et al., 2013; Williams et al., 2008), there are no comprehensive studies from New Zealand. Overseas studies have highlighted the importance of constructed wetlands for increasing waterfowl abundance (Davies et al., 2004; Sanchez-Zapata et al., 2005; Williams et al., 1998; Williams et al., 1999; Williams et al., 2008). Construction of ponds on the Waikato floodplains may help to reverse the recently-observed decline of some waterfowl populations. This thesis aimed to determine a set of key design features which can be implemented when constructing ponds for this purpose. To identify the key features, I firstly investigated the influence of landscape and physicochemical factors on macroinvertebrate communities as an indicator of waterfowl food supply. Secondly, I investigated the links between fish abundance and biomass, especially pest fish, and macroinvertebrate communities to identify if fish can directly or indirectly limit waterfowl food supply. Finally, I determined the environmental and biotic factors associated with differences in waterfowl abundance and productivity.

5.1   Hydrology

The flooding regime of ponds was consistently identified as a significant factor influencing macroinvertebrate, fish and waterfowl communities. Increased flooding appeared to reduce macroinvertebrate abundances and increase fish biomass. Waterfowl seemed to have a more species-specific response to flood regime; grey teal abundances were highest in annually flooded ponds, whereas black swan and Canada goose abundances were highest in ponds that never flood. Waterfowl may respond to the water conditions presented by different flooding

regimes; therefore, to increase waterfowl diversity it would likely be beneficial to include a range of flooding conditions within a landscape of constructed ponds.

Water turbidity was positively associated with increasing flood frequency, which was likely a function of the increased pest fish abundance in frequently flooded ponds. The pest fish in the Waikato region, notably koi carp, catfish, and goldfish, are known benthivorous feeders (Rowe, 2007), and subsequently cause increased turbidity through the resuspension of sediment and nutrients (Chapman, 1996; Hayes et al., 1992; Pimentel et al., 2000). During flood events, koi carp have been observed migrating from main channels into newly inundated areas (Daniel et al., 2011; Gorski et al., 2014; Jones and Stuart, 2009).

In my study, the flooding regime of ponds impacted their accessibility for fish. Ponds which never flood were often characterised by raised culverts, floodgates and stop-banks which limited water intrusion from the river and decreased the likelihood of pest fish entering. Limiting the connectivity of constructed ponds, and thus, access for pest fish, will likely increase the ecological health and integrity of the pond. Water clarity should be relatively good, and food availability for waterfowl should increase as a result. No differences in fish density was detected between ponds which occasionally and never flood; therefore, to improve the health of a pond, limiting the frequency of floods is more important than attempting to eliminate floods altogether.

The permanence of constructed ponds on the lower Waikato River floodplain appeared to be controlled by the water supply to the pond. Spring-fed ponds had consistent water supplies so tended to be permanent, whereas rain-fed ponds were temporary features due to evaporation during summer. Diptera are a known significant food source for mallard ducklings (Danell and Sjoberg, 1980; Lees and Street, 1974; Street, 1977); however, their benthic abundance decreased in temporary, rain-fed ponds. Macroinvertebrate communities of temporary ponds can be characterised by taxa that are able to survive in mud, whereas communities in permanent ponds can support a mixture of different taxa (Collinson et al., 1995). Thus, the construction of permanent ponds, will, in theory, increase the diversity of food supplies potentially available for waterfowl. Permanence can be achieved

by constructing ponds which capture spring flow to provide a year-round water supply.

5.2   Pond complexity

Waterfowl were most abundant in Lake Whangape (LWP) ponds which varied in size. While the larger LWP ponds held the most waterfowl, the literature indicates smaller ponds may be important during the breeding phase as they likely provide more nesting cover (Walker et al., 2013). This theory could explain the greater number of juvenile waterfowl in LWP. The Whangamarino wetland (WGM) ponds had the most juvenile grey teal, which is likely a function of the high adult grey teal abundance. The same is true for mallard adults and juveniles in LWP ponds and Canada goose on Lake Waikare (LWK) ponds. Community composition varied between pond locations, with larger ponds having higher waterfowl abundances and species richness even when adjusted for water area. This finding is consistent with studies which have found larger ponds have greater waterfowl abundance and diversity (Austin and Buhl, 2009; Cowardin et al., 1998; Sanchez-Zapata et al., 2005). Area was also important in determining variation among benthic macroinvertebrate communities, indicating higher benthic macroinvertebrate abundances expressed as catch per unit effort in larger ponds. My results support the pond construction recommendations of Soulliere and Monfils (1996) who identified pond size and shoreline complexity as key design features, as well as the availability of various depth zones.

Water depth influences the feeding behaviour of waterbirds. Liang et al. (2002) found it was the only variable that affected all wading-feeders, surface-feeders, and underwater-feeders in the Sitsao wetlands of southern Taiwan. In all cases, feeding activity decreased with increasing water depths. It has been recommended, when constructing a pond, that only 10% of area is deeper than one meter; with the remaining 90% evenly split between <0.3 m, 0.3 - 0.7 m, and 0.7 – 1.0 m (Soulliere and Monfils, 1996). Dabbling duck populations appeared to respond positively to the percentage of pond shallower than 20 cm in my study; however, they also responded positively to the percentage of the pond deeper than one meter. For ponds in the Waikato region, the Soulliere and Monfils (1996) model

should be amended to include a larger percentage of pond deeper than one meter which may yield more emerging insects. In my study, the percentage of pond shallower than 20 cm was positively associated with adult duck numbers, potentially because larger ponds with more shallow water have more accessible benthic macroinvertebrates. During the brooding season the diet of adult waterfowl largely consists of benthic macroinvertebrates (Dessborn et al., 2011), suggesting an importance in maintaining water depths less than 20 cm to optimise waterfowl accessibility to macroinvertebrates.

Shallow feeding habitat could be further increased in ponds by increasing the length of shoreline. Increasing the shoreline complexity of a pond will increase the availability of loafing sites, escape cover, visual isolation, and foraging sites for waterfowl (Austin and Buhl, 2009; Mack and Flake, 1980). The area:perimeter ratio was positively associated with the abundance of sweep Coleoptera, biomass of sweep Diptera, and biomass of benthic Coleoptera and Hemiptera, which are all known food sources of ducklings (Danell and Sjoberg, 1980; Lees and Street, 1974; Street, 1977). Soulliere and Monfils (1996) identified that waterfowl were attracted to ponds with varying shapes and depths distributed across the landscape in response to increased diversity of habitats suitable for birds throughout different life stages. For juvenile waterfowl it is the same; Mack and Flake (1980) found the probability of brood occupancy on ponds increased as the shoreline length increased relative to pond area. In the Waikato, shoreline complexity appeared to influence waterfowl community composition, but had no effect on abundances, suggesting constructing ponds with convoluted shorelines increases the waterfowl species richness, whereas pond area increases waterfowl densities.

5.3   Vegetation

The vegetation structure around ponds was often representative of their age; older ponds had more mature riparian vegetation, while young ponds had very little vegetation. This is an important relationship as riparian vegetation can absorb runoff nutrients before they enter the pond and potentially cause eutrophication (Gottschall et al., 2007; Kadlec and Knight, 1996; Mitsch et al., 2001). The retention of nutrients in plants results in improved water quality and increases the

ecological health of the pond (Comin et al., 1997). Increasing the width of the vegetation buffer will not only reduce nutrient loading in the ponds, but it also limits the direct impact livestock can have, as well as reducing the influence of wind stirring bottom sediments. Livestock are known to degrade the water quality in stream ecosystems when allowed access (Kauffman and Krueger, 1984; Miller et al., 2011; Trimble and Mendel, 1995). Some studies have reported that livestock access does not have a direct impact on macroinvertebrate communities (Cereghino et al., 2008; Ranganath et al., 2009), but it does appear to impact on the probability of brood occupancy on ponds. I found brood occupancy increased as pond fencing increased, specifically with Canada goose and mallard broods, suggesting small riparian margins with dense vegetation and fencing improve nesting and brood rearing habitat in waterfowl, such as Canada goose and mallard.

Waterfowl densities have often been positively associated with emergent aquatic vegetation as well as riparian vegetation (del Hoyo et al., 1992; Kuczynski and Paszkowski, 2012; Soulliere and Monfils, 1996; Walker et al., 2013). However, McDougall et al. (2009) suggested this relationship might not be present in constructed ponds in New Zealand, which may be a result of limited ponds with >30% emergent vegetation cover.

5.4 Food supply

Seeds are an important component of adult and juvenile waterfowl diets (Cox et al., 1998; Dessborn et al., 2011; Lees and Street, 1974; Sedinger, 1992; Street, 1977; Sugden, 1973). The food availability score used in my research was a function of macroinvertebrate and seed abundances, and it consistently featured as an important driver of variations among waterfowl community composition. This finding highlights the likely importance of seed and macroinvertebrate abundances in constructed ponds. *Carex* is a dominant vegetation food source for adult and juvenile ducks, especially mallard (Dessborn et al., 2011); however, I found it was only associated with adult mallard ducks. The lack of significant individual riparian vegetation types explaining variation among waterfowl communities is likely because waterfowl prefer ponds with mixed vegetation, for both riparian and aquatic habitats.

Fish are efficient feeders and consume macroinvertebrates before waterfowl can access them (Eriksson, 1983). There was a marked decrease in sweep macroinvertebrate abundance in ponds with large-bodied pest fish in my study. Fish can further reduce waterfowl food availability by reducing macrophyte abundance (Hanchet, 1990). Adult waterfowl communities were negatively affected by the presence of koi carp in Waikato ponds which is likely due to competition for food.

There were species-specific relationships between macroinvertebrate taxa and waterfowl species in the ponds I studied. Mallard ducks were associated with ponds that supported higher Diptera biomass, whereas grey duck were associated with Hemiptera, and shoveler, paradise shelduck, and grey duck were associated with ponds high in Coleoptera. Diptera abundance and biomass were significantly greater in spring-fed ponds than rain-fed ponds indicating they prefer high-nutrient environments. Elsewhere Diptera have been shown to be an important food source for adult and juvenile waterfowl (Danell and Sjoberg, 1980; Lees and Street, 1974; Street, 1977). Invertebrates occupying the water column sampled by sweep-netting were important, especially for juvenile waterfowl, who cannot feed on benthic macroinvertebrates (Dessborn et al., 2011).

5.5   Pest fish

Gambusia are efficient predators on macroinvertebrates and zooplankton (Mansfield and McArdle, 1998), and impact on macroinvertebrate communities more than other fishes (Ling, 2004). Gambusia was an important predictor variable for sweep macroinvertebrate communities with invertebrate abundances being lower in ponds where gambusia were present. Gambusia tend to inhabit the macrophyte dominated areas of the shallow (<1 m deep) littoral zone (Hicks et al., 2010). With macroinvertebrates the main food source for adult and juvenile waterfowl during the brooding season (Alisauskas and Ankney, 1992), keeping gambusia out of new ponds will likely increase macroinvertebrate abundances and potentially lead to greater juvenile waterfowl production.

Benthivorous fish, such as koi carp, catfish, and goldfish, were negatively associated with benthic macroinvertebrates. Koi carp decrease the abundance of

macrophytes and invertebrates through their feeding behaviour (Parkos et al., 2003), as well as increasing water turbidity and sediment resuspension (Chapman, 1996; Hayes et al., 1992; Scheffer, 2004). There were signs koi carp have a similar impact in constructed ponds in the Waikato, with turbidity being closely associated with the large-bodied pest fish. The exclusion of these fish, especially koi carp, from newly constructed ponds will likely lead to better water quality and more macrophyte and macroinvertebrate communities dominated by taxa that are more readily accessible by waterfowl.

5.6   Pond construction in the Waikato

Based on my results and overseas research I was able to develop conceptual models of how the ecology of constructed ponds influences adult (Figure 5-1) and juvenile (Figure 5-2) waterfowl communities and abundances. There are ten features which directly influence adult waterfowl communities. Benthic macroinvertebrates do not have a strong influence on juvenile waterfowl; therefore, juvenile communities and abundances are influenced by nine pond features. In both models, hydrology directly influences waterfowl, as well as influencing fish and macroinvertebrates communities. Vegetation structure influences water quality and waterfowl behaviour, as well as directly influencing waterfowl communities. These conceptual models can be used to assist with future pond construction to potentially increase waterfowl diversity and abundance.

Habitat heterogeneity is desirable in constructed ponds (Creighton et al., 1997; Locky et al., 2005; Paracuellos, 2006). When constructing wetlands in the Hilliardton Marsh, Locky et al. (2005) observed a net increase in mesohabitat types as the habitat changed through time. Paracuellos (2006) acknowledged habitat heterogeneity is important for waterbird ponds, with heterogeneity increasing as pond size increases; larger ponds are able to accommodate a wider range of habitat types. Therefore, time, size and complexity are likely crucial players in a ponds potential to reach maximum heterogeneity.

Habitat heterogeneity is important for maximising waterfowl diversity and density in newly constructed wetlands. The lower Waikato River floodplain supports a

wide range of pond habitats within the landscape, including ponds of different sizes, edge lengths, water supplies, flooding regimes, and vegetation structure. Based on my analysis of these ponds and the overseas literature the following recommendations can be drawn:

- Large ponds have the highest waterfowl abundances per hectare; however, small ponds are still important in a landscape. Therefore, it is important to design ponds at the landscape scale so that multiple ponds provide a range of habitats.
- A high degree of shoreline complexity leads to increased foraging and loafing habitat for waterfowl. Ponds with high shoreline complexities are also more likely to have broods on them.
- Depth is important when constructing new ponds. During the brooding season this is likely important in ponds so there is enough habitat for the benthic feeding adult ducks, and the pelagic feeding ducklings.
- The water source and frequency of flooding of the pond are important. Often the water source dictates the permanence and flooding regime of the pond. Understanding the hydrology is important as it has significant impacts on macroinvertebrate communities.
- Pest fish, such as koi carp, use flooding events to migrate into new water bodies due to the higher degree of connectivity. Reducing the chance of new ponds flooding by constructing raised culverts, floodgates, or stop banks, limits the opportunities for the destructive pest fish to invade.
- Rain-fed, drain-fed and spring-fed ponds allow for more control over flooding events; however, rain-fed ponds tend to dry out over the course of summer. Spring-fed ponds with limited connectivity to adjacent waterbodies are best for waterfowl.
- Vegetation is significant as a source of nesting habitat and food. Having a wide range of vegetation around a pond will increase the ecological health of the pond by absorbing run-off nutrients, as well as providing cover for the waterfowl. Seeds are also an important food source for adult and juvenile waterfowl.
- Fencing around ponds is important. It reduces the impact of livestock on ponds, as well as providing an opportunity for ungrazed vegetation to

grow. Waterfowl are more likely to nest in fenced off areas, where there is less interaction with surrounding agricultural practices.

5.7   Future work

This thesis gives an insight into the limiting factors of waterfowl productivity and density on constructed wetlands. This is the first comprehensive study in New Zealand looking at the ecology of constructed ponds, and is limited to the lower Waikato River floodplains, so should only be used as a guide for constructing ponds. To gain more spatial knowledge, replicate studies should be carried out in other regions of New Zealand to determine if waterfowl communities respond in similar ways to various pond features. This study was also temporally limited as it was carried out during a single brooding season which followed a severe drought in the Waikato region. Further studies should be carried out over the course of multiple brooding seasons to better understand how waterfowl communities respond to multiple environmental stimuli. Finally, future studies should be carried out to monitor newly constructed ponds from age zero to a set age, to determine which sets of variables are best included in initial pond design to encourage suitable successional habitats.
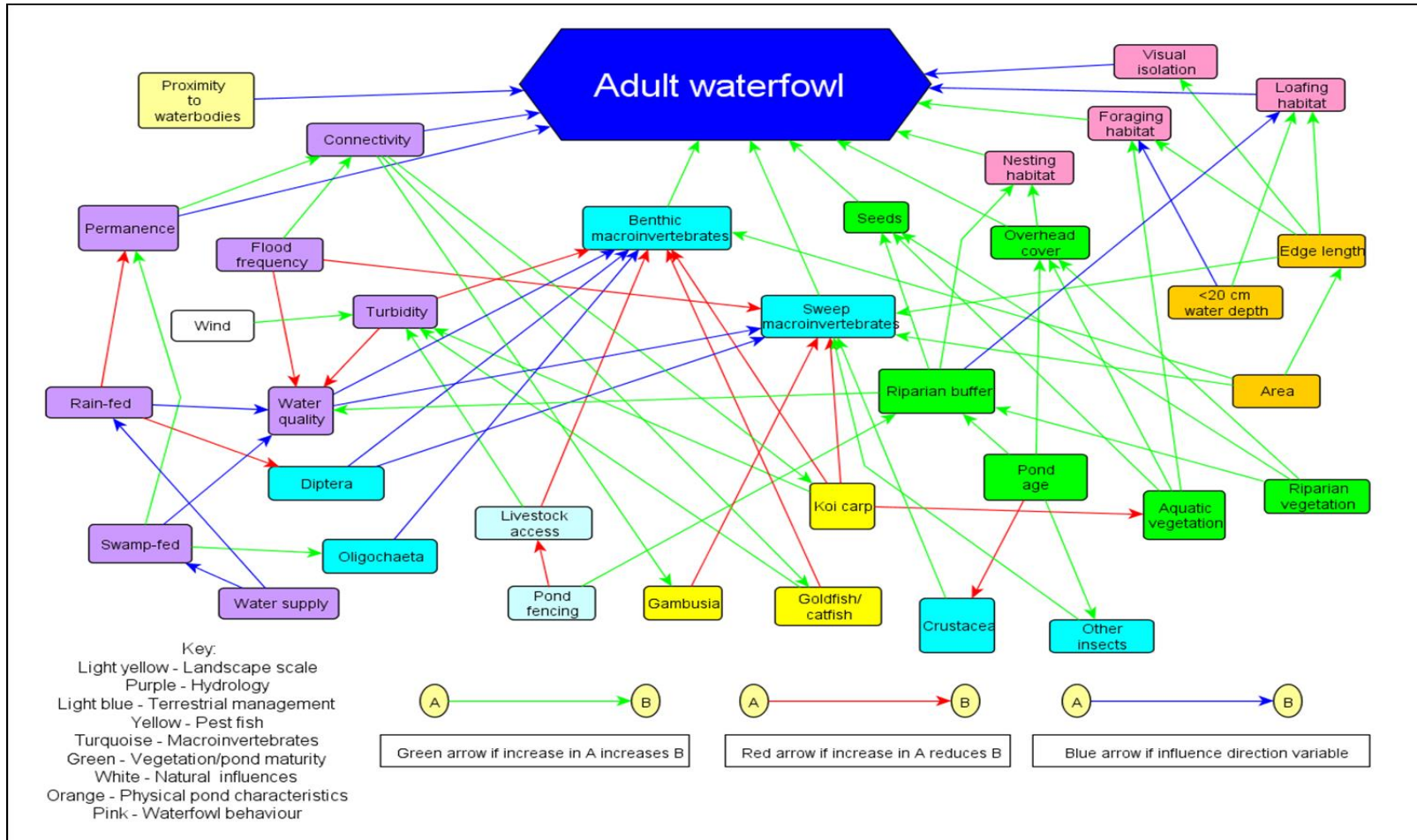
Figure 5-1: Conceptual model of the abiotic and biotic pond features that influenced adult waterfowl communities and abundances per hectare in 34 constructed ponds on the lower Waikato River floodplain
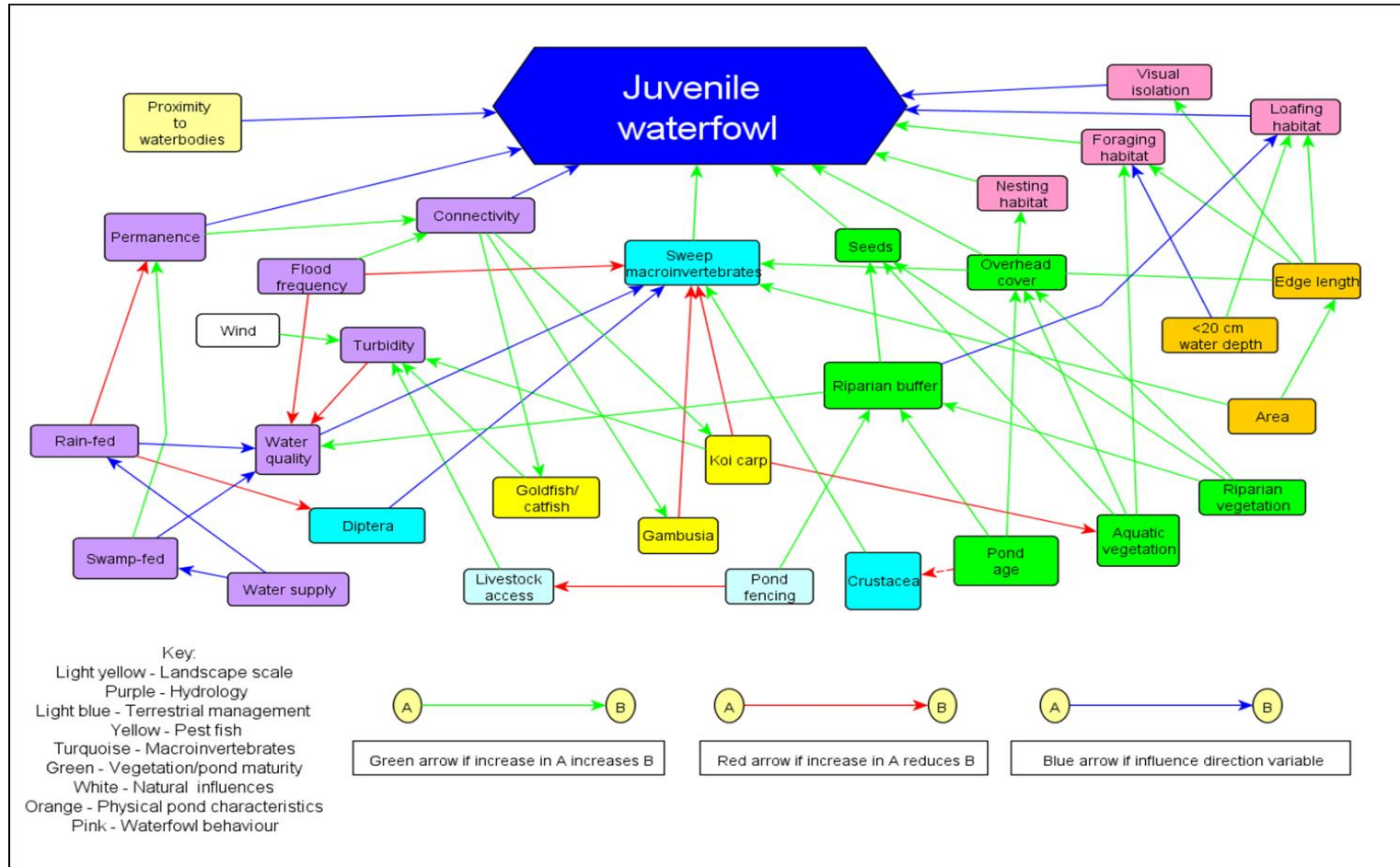
Figure 5-2: Conceptual model of the abiotic and biotic pond features that influenced juvenile waterfowl communities and abundances per hectare in 34 constructed ponds on the lower Waikato River floodplain

References

ALISAUSKAS, R. T. & ANKNEY, C. D. 1992. The cost of egglaying and its relationship to nutrient reserves in waterfowl. *In:* BATT, B. D. J., AFTON, A. D., ANDERSON, M. G., ANKNEY, C. D., JOHNSON, D. H., KADLEC, J. A. & KRAPU, G. L. (eds.) *Ecology and Management of Breeding Waterfowl.* Minnesota: University of Minnesota Press.

AUSSEIL, A. G. E., CHADDERTON, W. L., GERBEAUX, P., STEPHENS, R. T. T. & LEATHWICK, J. R. 2011. Applying systematic conservation planning principles to palustrine and inland saline wetlands of New Zealand. *Freshwater Biology,* 56**,** 142-161.

AUSSEIL, A. G. E., GERBEAUX, P., CHADDERTON, W. L. S., T., BROWN, D. & LEATHWICK, J. R. 2008. Wetland ecosystems of national importance for biodiversity. Wellington, New Zealand: Landscare Research.

AUSTIN, J. E. 2002. Responses of dabbling ducks to wetland conditions in the Prairie Pothole region. *Waterbirds,* 25**,** 465-473.

AUSTIN, J. E. & BUHL, D. A. 2009. Factors associated with duck use of impounded and natural wetlands in western South Dakota. *The Prairie Naturalist,* 41**,** 1-27.

BARON, J. S., POFF, N. L., ANGERMEIER, P. L., DAHM, C. N., GLEICK, P. H., HAIRSTON, N. G., JACKSON, R. B., JOHNSTON, C. A., RICHTER, B. D. & STEINMAN, A. D. 2002. Meeting ecological and societal needs for freshwater. *Ecological Applications,* 12**,** 1247-1260.

BATZER, D. P. 1998. Trophic interactions among detritus, benthic midges, and predatory fish in a freshwater marsh. *Ecology,* 79**,** 1688-1698.

BATZER, D. P., PUSATERI, C. R. & VETTER, R. 2000. Impacts of fish predation on marsh invertebrates: Direct and indirect effects. *Wetlands,* 20**,** 307-312.

BAZZANTI, M., BALDONI, S. & SEMINARA, M. 1996. Invertebrate macrofauna of a temporary pond in central Italy: Composition, community parameters and temporal succession. *Archiv Fur Hydrobiologie,* 137**,** 77-94.

BEARD, C. 2010. Wetlands and lowland floodplains. *In:* COLLIER, K., HAMILTON, D., VANT, W. & HOWARD-WILLIAMS, C. (eds.) *The Waters of the Waikato.* Hamilton, New Zealand: Environment Waikato and the Centre for Biodiversity and Ecology Research, The University of Waikato.

BÉLANGER, L. & COUTURE, R. 1988. Use of Man-Made Ponds by Dabbling Duck Broods. *The Journal of Wildlife Management,* 52**,** 718-723.

BENSTEAD, J. P., DE RHAM, P. H., GATTOLLIAT, J. L., GIBON, F. M., LOISELLE, P. V., SARTORI, M., SPARKS, J. S. & STIASSNY, M. L. J. 2003. Conserving Madagascar's freshwater biodiversity. *Bioscience,* 53**,** 1101-1111.

BIGGS, B. J. F. 1990. Periphyton communities and their environments in New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research,* 24**,** 367-386.

BIGGS, J., CORFIELD, A., WALKER, D. W., M. & WILLIAMS, P. 1994. New approaches to pond management. *British Wildlife,* 5**,** 273-287.

BIGGS, J., FOX, G., NICOLET, P., WALKER, D., WHITFIELD, M. & WILLIAMS, P. 1998. A guide to the methods of the National Pond Survey. Oxford, England.

BISCHOF, M. M., HANSON, M. A., FULTON, M. R., KOLKA, R. K., SEBESTYEN, S. D. & BUTLER, M. G. 2013. Invertebrate Community Patterns in Seasonal Ponds in Minnesota, USA: Response to Hydrologic and Environmental Variability. *Wetlands,* 33**,** 245-256.

BOUFFARD, S. H. & HANSON, M. A. 1997. Fish in waterfowl marshes: waterfowl managers' perspective. *Wildlife Society Bulletin***,** 146-157.

BRODERSEN, K. P. & ANDERSON, N. J. 2002. Distribution of chironomids (Diptera) in low arctic West Greenland lakes: trophic conditions, temperature and environmental reconstruction. *Freshwater Biology,* 47**,** 1137-1157.

BROOKS, R. T. 2000. Annual and seasonal variation and the effects of hydroperiod on benthic macroinvertebrates of seasonal forest ("vernal") ponds in central Massachusetts, USA. *Wetlands,* 20**,** 707-715.

BROWN, E. J. 2010. Flow regime and water use. *In:* COLLIER, K., HAMILTON, D., VANT, W. & HOWARD-WILLIAMS, C. (eds.) *The Waters of the Waikato: Ecology of New Zealand's longest river.* Hamilton: Environment Waikato and the Centre for Biodiversity and Ecology Research, The University of Waikato.

BURTON, K. L., ROUSE, W. R. & BOUDREAU, L. D. 1996. Factors affecting the summer carbon dioxide budget of subarctic wetland tundra. *Climate Research,* 6**,** 203-213.

CEREGHINO, R., RUGGIERO, A., MARTY, P. & ANGELIBERT, S. 2008. Biodiversity and distribution patterns of freshwater invertebrates in farm ponds of a south-western French agricultural landscape. *Hydrobiologia,* 597**,** 43-51.

CHAMPION, P. 1997. An overview of the Lower Waikato/Hauraki Plains wetlands and issues relating to their management. *NIWA consultancy report.* Hamilton, New Zealand: National Institute of Water and Atmospheric Research.

CHAMPION, P. & CLAYTON, J. 2010. Box 14.1: Riparian vegetation. *In:* COLLIER, K., HAMILTON, D., VANT, W. & HOWARD-WILLIAMS, C. (eds.) *The Waters of the Waikato: Ecology of New Zealand's longest river.* Hamilton: Environment Waikato and the Centre for Biodiversity and Ecology Research, The University of Waikato.

CHAPMAN, M. A. 1996. Human impacts on the Waikato River system, New Zealand. *GeoJournal,* 40**,** 85-99.

CHISNALL, B. L. 1987. *Juvenile eel biology in the backwaters of the Waikato River.* MSc thesis, University of Waikato.

CHISNALL, B. L. & HAYES, J. W. 1991. Age and growth of shortfinned eels (Anguilla australis) in the lower Waikato basin, North Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research,* 25**,** 71-80.

CLARKSON, B. R. 2002. Swamps, fens and bogs. *In:* CLARKSON, B. D., MERRETT, M. & DOWNS, T. (eds.) *Botany of the Waikato.* Hamilton, New Zealand: Waikato Botanical Society/Department of Biological Sciences, The University of Waikato.

COLLIER, K., WATENE-RAWIRI, E. & MCCRAW, J. 2010. Geography and history. *In:* COLLIER, K., HAMILTON, D., VANT, W. & HOWARD-WILLIAMS, C. (eds.) *The Waters of the Waikato: Ecology of New Zealand's longest river.* Hamilton: Environment Waikato and teh Centre for Biodiversity and Ecology Research (The University of Waikato).

COLLINSON, N. H., BIGGS, J., CORFIELD, A., HODSON, M. J., WALKER, D., WHITFIELD, M. & WILLIAMS, P. J. 1995. Temporary and permanent ponds - an assessment of the effects of drying out on the conservation value of aquatic macroinvertebrate communities. *Biological Conservation,* 74**,** 125-133.

COMIN, F. A., ROMERO, J. A., ASTORGA, V. & GARCIA, C. 1997. Nitrogen removal and cycling in restored wetlands used as filters of nutrients for agricultural runoff. *Water Science and Technology,* 35**,** 255-261.

COWARDIN, L. M., PIETZ, P. J., LOKEMOEN, J. T., SKLEBAR, H. T. & SARGEANT, G. A. 1998. Response of Nesting Ducks to Predator Exclosures and Water Conditions during Drought. *The Journal of Wildlife Management,* 62**,** 152-163.

COX, R. R., HANSON, M. A., ROY, C. C., EULISS, N. H., JOHNSON, D. H. & BUTLER, M. G. 1998. Mallard duckling growth and survival in relation to aquatic invertebrates. *Journal of Wildlife Management,* 62**,** 124-133.

CREIGHTON, J. H., SAYLER, R. D., TABOR, J. E. & MONDA, M. J. 1997. Effects of wetland excavation on avian communities in eastern Washington. *Wetlands,* 17**,** 216-227.

CUI, B. S., ZHANG, Z. M. & LEI, X. X. 2012. Implementation of Diversified Ecological Networks to Strengthen Wetland Conservation. *Clean-Soil Air Water,* 40**,** 1015-1026.

CYR, H. & DOWNING, J. A. 1988. The abundance of phytophilous invertebrates on different species of submerged macrophytes. *Freshwater Biology,* 20**,** 365-374.

DAHL, T. E. 1990. Wetland losses in the United States 1780's to 1980's. Washington, D.C.: Department of the Interior, Fish and Wildlife Service.

DANELL, K. & SJOBERG, K. 1980. Foods of wigeon, teal, mallard and pintail during the summer in the Northern Swedish Lake. *Viltrevy,* 11**,** 141-167.

DANIEL, A. J., HICKS, B. J., LING, N. & DAVID, B. O. 2011. Movements of Radio- and Acoustic-Tagged Adult Koi Carp in the Waikato River, New Zealand. *North American Journal of Fisheries Management,* 31**,** 352-362.

DAVIES, B. R., BIGGS, J., LEE, J. T. & THOMPSON, S. 2004. Identifying optimum locations for new ponds. *Aquatic Conservation-Marine and Freshwater Ecosystems,* 14**,** 5-24.

DEL HOYO, J., ELLIOT, A. & SARGATAL, J. 1992. *Handbook of the Birds of the World, Vol. 1,* Barcelona, Spain, Lynx Edicions.

DESSBORN, L., BROCHET, A. L., ELMBERG, J., LEGAGNEUX, P., GAUTHIER-CLERC, M. & GUILLEMAIN, M. 2011. Geographical and temporal patterns in the diet of pintail Anas acuta, wigeon Anas penelope, mallard Anas platyrhynchos and teal Anas crecca in the Western Palearctic. *European Journal of Wildlife Research,* 57**,** 1119-1129.

DUDGEON, D. 1999. *Tropical Asian Streams: Zoobenthos, Ecology and Conservation,* Hong Kong, Hong Kong University Press.

DUDGEON, D. 2000. The ecology of tropical Asian rivers and streams in relation to biodiversity conservation. *Annual Review of Ecology and Systematics,* 31**,** 239-263.

DUDGEON, D., ARTHINGTON, A. H., GESSNER, M. O., KAWABATA, Z. I., KNOWLER, D. J., LEVEQUE, C., NAIMAN, R. J., PRIEUR-RICHARD, A. H., SOTO, D., STIASSNY, M. L. J. & SULLIVAN, C. A. 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews,* 81**,** 163-182.

DYTHAM, C. 2005. *Choosing and Using Statistics: A Biologist's Guide,* Oxfor, United Kingdom, Blackwell Publishing.

EBERT, T. A. & BALKO, M. L. 1987. Temporary pools as islands in space and in time - the biota of vernal pools in San-Diego, southern California, USA. *Archiv Fur Hydrobiologie,* 110**,** 101-123.

ERIKSSON, M. O. 1983. The role of fish in the selection of lakes by nonpiscivorous ducks: mallard, teal and goldeneye. *Wildfowl,* 34**,** 27-32.

EVANS, C. D. & BLACK, K. E. 1956. Duck production studies on the prairie potholes of South Dakota. *Special Scientific Report -- Wildlife.* U.S. Fish and Wildlife Service.

FLAKE, L. D., PETERSEN, G. L. & TUCKER, W. Habitat relationships of breeding waterfowl on stock ponds in northwestern South Dakota. Proc. South Dakota Acad. Sci, 1977. 135-151.

GEE, J. H. R., SMITH, B. D., LEE, K. M. & GRIFFITHS, S. W. 1997. The ecological basis of freshwater pond management for biodiversity. *Aquatic Conservation-Marine and Freshwater Ecosystems,* 7**,** 91-104.

GIBBS, J. P. 1993. Importance of small wetlands for the persistence of local-populations of wetland-associated animals. *Wetlands,* 13**,** 25-31.

GILINSKY, E. 1984. The role of fish predation and spatial heterogeneity in determining benthic community structure. *Ecology,* 65**,** 455-468.

GOLLOP, J. B. & MARSHALL, W. H. 1954. A guide for aging duck broods in the field. *Mississippi Flyway Council Technical Section.*

GORSKI, K., COLLIER, K. J., HAMILTON, D. P. & HICKS, B. J. 2014. Effects of flow on lateral interactions of fish and shrimps with off-channel habitats in a large river-floodplain system. *Hydrobiologia,* 729**,** 161-174.

GOTTSCHALL, N., BOUTIN, C., CROLLA, A., KINSLEY, C. & CHAMPAGNE, P. 2007. The role of plants in the removal of nutrients at a constructed wetland treating agricultural (dairy) wastewater, Ontario, Canada. *Ecological Engineering,* 29**,** 154-163.

GREEN, A. J. 1996. Analyses of globally threatened anatidae in relation to threats, distribution, migration patterns, and habitat use. *Conservation Biology,* 10**,** 1435-1445.

GUNNARSSON, G., ELMBERG, J., SJOBERG, K., POYSA, H. & NUMMI, P. 2004. Why are there so many empty lakes? Food limits survival of mallard ducklings. *Canadian Journal of Zoology-Revue Canadienne De Zoologie,* 82**,** 1698-1703.

HAAS, K., KOHLER, U., DIEHL, S., KOHLER, P., DIETRICH, S., HOLLER, S., JAENSCH, A., NIEDERMAIER, M. & VILSMEIER, J. 2007. Influence of fish on habitat choice of water birds: A whole system experiment. *Ecology,* 88**,** 2915-2925.

HAMILTON, D., VANT, W. & NEILSON, K. A. 2010. Lowland lakes. *In:* COLLIER, K., HAMILTON, D., VANT, W. & HOWARD-WILLIAMS, C. (eds.) *Waters of the Waikato: Ecology of New Zealand's longest river.* Hamilton: Environment Waikato and the Centre for Biodiversity and Ecology Research, The University of Waikato.

HANCHET, S. 1990. The effects of koi carp on New Zealands aquatic ecosystems. *New Zealand freshwater fisheries report No. 117.* Rotorua: MAF Fisheries.

HANNIGAN, E. & KELLY-QUINN, M. 2012. Composition and structure of macroinvertebrate communities in contrasting open-water habitats in Irish peatlands: implications for biodiversity conservation. *Hydrobiologia,* 692**,** 19-28.

HANSON, M. A. & RIGGS, M. R. 1995. Potential effects of fish predation on wetland invertebrates - a comparison of wetlands with and without fathead minnows. *Wetlands,* 15**,** 167-175.

HAYES, J. W., RUTLEDGE, M. J., CHISNALL, B. L. & WARD, F. J. 1992. Effects of elevated turbidity on shallow lake fish communities. *Environmental Biology of Fishes,* 35**,** 149-168.

HELLIWELL, D. R. 1983. The conservation value of areas of different size - Worcestershire ponds. *Journal of Environmental Management,* 17**,** 179-184.

HERSHEY, A. E. 1987. Tubes and foraging behavior in larval Chironomidae - implications for predator avoidance. *Oecologia,* 73**,** 236-241.

HEY, D. L. & PHILIPPI, N. S. 1995. Flood reduction through wetland restoration: the upper Mississippi River Basin as a case study. *Restoration Ecology,* 3**,** 4-17.

HICKS, B. J. 2014. Summary of the average weight of koi carp caught by the University of Waikato's electrofishing boat prior to 2011. Unpublished data: University of Waikato.

HICKS, B. J., BRIJS, J., BELL, D. G. & POWRIE, W. 2008. Electrofishing survey of the fish community in the Whangamarino Wetland. *CBER Contract Report 67.* Hamilton: Centre for Biodiversity and Ecology Research, The University of Waikato.

HICKS, B. J., LING, N. & WILSON, B. J. 2010. Introduced fish. *In:* COLLIER, K., HAMILTON, D., VANT, W. & HOWARD-WILLIAMS, C. (eds.) *The Waters of the Waikato.* Hamiton: Waikato Regional Council.

HICKS, B. J., STICHBURY, G. A., BRABYN, L. K., ALLAN, M. G. & ASHRAF, S. 2013. Hindcasting water clarity from Landsat satellite images of unmonitored shallow lakes in the Waikato region, New Zealand. *Environment Monitoring and Assessment,* 185**,** 7245-7261.

HOARE, R., DUGDALE, J. & WATTS, C. 2006. The world's thinnest caterpillar? A new genus and species of Batrachedridae (Lepidoptera) from Sporadanthus ferrugineus (Restionaceae), a threatened New Zealand plant. *Invertebrate Systematics,* 20**,** 571-583.

HORNUNG, J. P. & FOOTE, A. L. 2006. Aquatic invertebrate responses to fish presence and vegetation complexity in western boreal wetlands, with implications for waterbird productivity. *Wetlands,* 26**,** 1-12.

HUNT, J. 2007. *Wetlands of New Zealand: A bitter-sweet story,* Auckland, New Zealand, Random House New Zealand.

JARVIS, P. G., MASSHEDER, J. M., HALE, S. E., MONCRIEFF, J. B., RAYMENT, M. & SCOTT, S. L. 1997. Seasonal variation of carbon dioxide, water vapor, and energy exchanges of a boreal black spruce forest. *Journal of Geophysical Research-Atmospheres,* 102**,** 28953-28966.

JEFFRIES, M. 1993. Invertebrate colonization of artificial pondweeds of differing fractal dimension. *Oikos,* 67**,** 142-148.

JEFFRIES, M. J. 2003. Idiosyncratic relationships between pond invertebrates and environmental, temporal and patch-specific predictors of incidence. *Ecography,* 26**,** 311-324.

JOHNSON, D. H. & GRIER, J. W. 1988. Determinants of Breeding Distributions of Ducks. *Wildlife Monographs***,** 3-37.

JOHNSON, P. & GERBEAUX, P. 2004. *Wetland types in New Zealand,* Wellington, Department of Conservation.

JONES, M. J. & STUART, I. G. 2009. Lateral movement of common carp (Cyprinus carpio L.) in a large lowland river and floodplain. *Ecology of Freshwater Fish,* 18**,** 72-82.

JOYNER, D. E. 1980. Influence of invertebrates on pond selection by ducks in Ontario. *Journal of Wildlife Management,* 44**,** 700-705.

KADLEC, R. H. & KNIGHT, R. L. 1996. *Treatment Wetlands,* New York, Lewis.

KALLIMANIS, A. S., MAZARIS, A. D., TZANOPOULOS, J., HALLEY, J. M., PANTIS, J. D. & SGARDELIS, S. P. 2008. How does habitat diversity affect the species-area relationship? *Global Ecology and Biogeography,* 17**,** 532-538.

KARAUS, U., ALDER, L. & TOCKNER, K. 2005. "Concave islands": Habitat heterogeneity of parafluvial ponds in a gravel-bed river. *Wetlands,* 25**,** 26-37.

KAUFFMAN, J. B. & KRUEGER, W. C. 1984. Livestock impacts on riparian ecosystems and streamside management implications - a review. *Journal of Range Management,* 37**,** 430-438.

KEDDY, P. A., FRASER, L. H., SOLOMESHCH, A. I., JUNK, W. J., CAMPBELL, D. R., ARROYO, M. T. K. & ALHO, C. J. R. 2009. Wet and Wonderful: The World's Largest Wetlands Are Conservation Priorities. *Bioscience,* 59**,** 39-51.

KELLY, D. W. & DICK, J. T. A. 2005. Effects of environment and an introduced invertebrate species on the structure of benthic macroinvertebrate species at the catchment level. *Archiv Fur Hydrobiologie,* 164**,** 69-88.

KIRBY, J. S. 1995. Winter population estimates for selected waterfowl species in Britain. *Biological Conservation,* 73**,** 189-198.

KLOSKOWSKI, J., GREEN, A. J., POLAK, M., BUSTAMANTE, J. & KROGULEC, J. 2009. Complementary use of natural and artificial wetlands by waterbirds wintering in Donana, south-west Spain. *Aquatic Conservation-Marine and Freshwater Ecosystems,* 19**,** 815-826.

KUCZYNSKI, E. C. & PASZKOWSKI, C. A. 2012. Constructed Borrow-Pit Wetlands as Habitat for Aquatic Birds in the Peace Parkland, Canada. *ISRN Ecology,* 2012**,** 13.

LAKE, M. D. 2010. Box 14.2: Black mudfish. *In:* COLLIER, K., HAMILTON, D., VANT, W. & HOWARD-WILLIAMS, C. (eds.) *The Waters of the Waikato: Ecology of New Zealand's longest river.* Hamilton, New Zealand: Environment Waikato and the Centre for Biodiversity and Ecology Research, The University of Waikato.

LANGDON, P. G., RUIZ, Z., WYNNE, S., SAYER, C. D. & DAVIDSON, T. A. 2010. Ecological influences on larval chironomid communities in shallow lakes: implications for palaeolimnological interpretations. *Freshwater Biology,* 55**,** 531-545.

LEES, P. R. & STREET, M. 1974. The feeding ecology of young Mallard (*Anus platyrhynchos*) and Tufted Duck (*Aythya fuligula*) in wet grave quarries. *In:* LUNDSTROM, S. (ed.) *Transactions of the 11th Congress of the International Union of Game Biologists.* Stockholm: International Union of Game Biologists.

LIANG, S. H., SHIEH, B. S. & FU, Y. S. 2002. A structural equation model for physiochemical variables of water, benthic invertebrates, and feeding activity of waterbirds in the Sitsao wetlands of southern Taiwan. *Zoological Studies,* 41**,** 441-451.

LILLIE, R. A. & BUDD, J. 1992. Habitat architecture of myriophyllum-spicatum l as an index to habitat quality for fish and macroinvertebrates. *Journal of Freshwater Ecology,* 7**,** 113-125.

LING, N. 2004. Gambusia in New Zealand: really bad or just misunderstood? *New Zealand Journal of Marine and Freshwater Research,* 38**,** 473-480.

LING, N. & WILLIS, K. 2005. Impacts of mosquitofish, Gambusia affinis, on black mudfish, Neochanna diversus. *New Zealand Journal of Marine and Freshwater Research,* 39**,** 1215-1223.

LOCKY, D. A., DAVIES, J. C. & WARNER, B. G. 2005. Effects of wetland creation on breeding season bird use in boreal eastern Ontario. *Canadian Field-Naturalist,* 119**,** 64-75.

LOKEMOEN, J. T. 1973. Waterfowl Production on Stock-Watering Ponds in the Northern Plains. *Journal of Range Management,* 26**,** 179-184.

MACK, G. D. & FLAKE, L. D. 1980. Habitat Relationships of Waterfowl Broods on South Dakota Stock Ponds. *The Journal of Wildlife Management,* 44**,** 695-700.

MANSFIELD, S. & MCARDLE, B. H. 1998. Dietary composition of Gambusia affinis (Family Poeciliidae) populations in the northern Waikato region of New Zealand. *New Zealand Journal of Marine and Freshwater Research,* 32**,** 375-383.

MCCARTHY, W. F. 1995. *Direct and indirect effects of esfenvalerate on wetland biota.* University of Minnesota.

MCDOUGALL, M., MAXWELL, I., CHEYNE, J. & WINLOVE, T. 2009. Mallard (*Anus platyrhynchos*) and Grey/Parera (*A. superciliosa*) presence at ponds in the East Coast Region: a pilot study to identify influential pond characteristics. Rotorua: Fish and Game New Zealand, Eastern Region.

MCGLONE, M. S. 2009. Postglacial history of New Zealand wetlands and implications for their conservation. *New Zealand Journal of Ecology,* 33**,** 1-23.

MCKINNEY, M. L. & LOCKWOOD, J. L. 1999. Biotic homogenization: a few winners replacing many losers in the next mass extinction. *Trends in Ecology & Evolution,* 14**,** 450-453.

MCPEEK, M. A. 1990. Determination of species composition in the Enallagma damselfly assemblages of permanent lakes. *Ecology,* 71**,** 83-98.

MEDEIROS, A. S. & QUINLAN, R. 2011. The distribution of the Chironomidae (Insecta: Diptera) along multiple environmental gradients in lakes and ponds of the eastern Canadian Arctic. *Canadian Journal of Fisheries and Aquatic Sciences,* 68**,** 1511-1527.

MILLER, J., CHANASYK, D., CURTIS, T., ENTZ, T. & WILLMS, W. 2011. Environmental quality of Lower Little Bow River and riparian zone along an unfenced reach with off-stream watering. *Agricultural Water Management,* 98**,** 1505-1515.

MITSCH, W. J., DAY, J. W., GILLIAM, J. W., GROFFMAN, P. M., HEY, D. L., RANDALL, G. W. & WANG, N. M. 2001. Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River Basin: Strategies to counter a persistent ecological problem. *Bioscience,* 51**,** 373-388.

MOSER, M., PRENTICE, C. & FRAZIER, S. A Global Overview of Wetland Loss and Degradation. Preecedings of the 6th Meeting of the Conference of Contracting Parties of the Ramsar Convention, 1996.

MURKIN, H. R. & KADLEC, J. A. 1986. Relationships between waterfowl and macroinvertebrate densities in a northern prairie marsh. *Journal of Wildlife Management,* 50**,** 212-217.

MURKIN, H. R., KAMINSKI, R. M. & TITMAN, R. D. 1982. Responses by dabbling ducks and aquatic invertebrates to an experimentally manipulated cattail marsh. *Canadian Journal of Zoology-Revue Canadienne De Zoologie,* 60**,** 2324-2332.

MYERS, S. C., CLARKSON, B. R., REEVES, P. N. & CLARKSON, B. D. 2013. Wetland management in New Zealand: Are current approaches and policies sustaining wetland ecosystems in agricultural landscapes? *Ecological Engineering,* 56**,** 107-120.

NATIONAL WETLAND TRUST OF NEW ZEALAND 2005. *Our Wet & Wild Places: Celebrating New Zealand's Internationally Significant Wetlands*, National Wetland Trust of New Zealand.

NECKLES, H. A., MURKIN, H. R. & COOPER, J. A. 1990. Influences of seasonal flooding on macroinvertebrate abundance in wetland habitats. *Freshwater Biology,* 23**,** 311-322.

NEUMANN, H. H., DENHARTOG, G., KING, K. M. & CHIPANSHI, A. C. 1994. Carbon-dioxide fluxes over a raised open bog at the Kinosheo lake tower site during the northern wetlands study (NOWES). *Journal of Geophysical Research-Atmospheres,* 99**,** 1529-1538.

NICOLET, P., BIGGS, J., FOX, G., HODSON, M. J., REYNOLDS, C., WHITFIELD, M. & WILLIAMS, P. 2004. The wetland plant and macroinvertebrate assemblages of temporary ponds in England and Wales. *Biological Conservation,* 120**,** 261-278.

NUMMI, P., VAANANEN, V. M., RASK, M., NYBERG, K. & TASKINEN, K. 2012. Competitive effects of fish in structurally simple habitats: perch, invertebrates, and goldeneye in small boreal lakes. *Aquatic Sciences,* 74**,** 343-350.

OECD/IUCN 1996. Guidelines for aid agencies for improved conservation and sustainable use of tropical and sub-tropical wetlands. Paris: OECD.

OERTLI, B., JOYCE, D. A., CASTELLA, E., JUGE, R., CAMBIN, D. & LACHAVANNE, J.-B. 2002. Does size matter? The relationship between pond area and biodiversity. *Biological Conservation,* 104**,** 59-70.

PALMER, M. 1981. Relationship between species richness of macrophytes and insects in some waterbodies in the Norfolk Breckland. *Entomologist's Monthly Magazine,* 117**,** 35-46.

PARACUELLOS, M. 2006. How can habitat selection affect the use of a wetland complex by waterbirds? *Biodiversity and Conservation,* 15**,** 4569-4582.

PARKOS, J. J., SANTUCCI, V. J. & WAHL, D. H. 2003. Effects of adult common carp (Cyprinus carpio)on multiple trophic levels in shallow mesocosms. *Canadian Journal of Fisheries and Aquatic Sciences,* 60**,** 182-192.

PARLIAMENTARY COMMISSIONER FOR THE ENVIRONMENT 2001. Boggy patch or ecological heritage? Valuing wetlands in Tasman. Wellington, New Zealand: Parliamentary Commissioner for the Environment.

PARLIAMENTARY COMMISSIONER FOR THE ENVIRONMENT 2004. Growing for good: Intensive farming, sustainability and New Zealand's environment. Wellington, New Zealand.

PASZKOWSKI, C. A. & TONN, W. M. 2000. Community concordance between the fish and aquatic birds of lakes in northern Alberta, Canada: the relative importance of environmental and biotic factors. *Freshwater Biology,* 43**,** 421-437.

PERROW, M. R., JOWITT, A. J. D., STANSFIELD, J. H. & PHILLIPS, G. L. 1999. The practical importance of the interactions between fish, zooplankton and macrophytes in shallow lake restoration. *Hydrobiologia,* 395**,** 199-210.

PIMENTEL, D., LACH, L., ZUNIGA, R. & MORRISON, D. 2000. Environmental and economic costs of nonindigenous species in the United States. *Bioscience,* 50**,** 53-65.

POTTER, K. W. 1994. Estimating potential reduction flood benefits of restored wetlands. *Water Resource Update,* 97**,** 34-38.

RAHEL, F. J. 2002. Homogenization of freshwater faunas. *Annual Review of Ecology and Systematics,* 33**,** 291-315.

RAMSAR. n.d. *Classification System for Wetland Type* [Online]. The Ramsar Convention on Wetlands. Available: http://www.ramsar.org/cda/en/ramsar-documents-guidelines-classification-system/main/ramsar/1-31-105%5E21235_4000_0__ [Accessed 18 June 2014].

RANGANATH, S. C., HESSION, W. C. & WYNN, T. M. 2009. Livestock exclusion influences on riparian vegetation, channel morphology, and benthic macroinvertebrate assemblages. *Journal of Soil and Water Conservation,* 64**,** 33-42.

RICKLEFS, R. E. & LOVETTE, I. J. 1999. The roles of island area per se and habitat diversity in the species-area relationships of four Lesser Antillean faunal groups. *Journal of Animal Ecology,* 68**,** 1142-1160.

ROBSON, B. J. & CLAY, C. J. 2005. Local and regional macroinvertebrate diversity in the wetlands of a cleared agricultural landscape in south-

western Victoria, Australia. *Aquatic Conservation-Marine and Freshwater Ecosystems,* 15**,** 403-414.

ROULET, N. T. 2000. Peatlands, carbon storage, greenhouse gases, and the Kyoto Protocol: Prospects and significance for Canada. *Wetlands,* 20**,** 605-615.

ROWE, D. K. 2007. Exotic fish introductions and the decline of water clarity in small North Island, New Zealand lakes: a multi-species problem. *Hydrobiologia,* 583**,** 345-358.

RUNDLE, W. D. 1982. A case for esophageal analysis in shorebird food studies. *Journal of Field Ornithology,* 53**,** 249-257.

SAFRAN, R. J., ISOLA, C. R., COLWELL, M. A. & WILLIAMS, O. E. 1997. Benthic invertebrates at foraging locations of nine waterbird species in managed wetlands of the northern San Joaquin Valley, California. *Wetlands,* 17**,** 407-415.

SANCHEZ-ZAPATA, J. A., ANADON, J. D., CARRETE, M., GIMENEZ, A., NAVARRO, J., VILLACORTA, C. & BOTELLA, F. 2005. Breeding waterbirds in relation to artificial pond attributes: implications for the design of irrigation facilities. *Biodiversity and Conservation,* 14**,** 1627-1639.

SANDERSON, R. A., EYRE, M. D. & RUSHTON, S. P. 2005. Distribution of selected macroinvertebrates in a mosaic of temporary and permanent freshwater ponds as explained by autologistic models. *Ecography,* 28**,** 355-362.

SCHEFFER, M. 2004. *Ecology of shallow lakes,* Dordrecht, The Netherlands, Kluwer Academic.

SCHEFFER, M., VAN GEEST, G. J., ZIMMER, K., JEPPESEN, E., SONDERGAARD, M., BUTLER, M. G., HANSON, M. A., DECLERCK, S. & DE MEESTER, L. 2006. Small habitat size and isolation can promote species richness: second-order effects on biodiversity in shallow lakes and ponds. *Oikos,* 112**,** 227-231.

SCHNEIDER, D. W. & FROST, T. M. 1996. Habitat duration and community structure in temporary ponds. *Journal of the North American Benthological Society,* 15**,** 64-86.

SCHRIVER, P., BOGESTRAND, J., JEPPESEN, E. & SONDERGAARD, M. 1995. Impact of submerged macrophytes on fish-zooplankton-phytoplankton interactions - large-scale enclosure experiments in a shallow eutrophic lake. *Freshwater Biology,* 33**,** 255-270.

SEDINGER, J. S. 1992. Ecology of prefledging waterfowl. *In:* BATT, B. D. J., AFTON, A. D., ANDERSON, M. G., ANKNEY, C. D., JOHNSON, D. H., KADLEC, J. A. & KRAPU, G. L. (eds.) *Ecology and Management of Breeding Waterfowl.* Minnesota: University of Minnesota Press.

SHI, J. M., MA, K. M., WANG, J. F., ZHAO, J. Z. & HE, K. 2010. Vascular plant species richness on wetland remnants is determined by both area and habitat heterogeneity. *Biodiversity and Conservation,* 19**,** 1279-1295.

SMITH, G. R., VAALA, D. A. & DINGFELDER, H. A. 2003. Distribution and abundance of macroinvertebrates within two temporary ponds. *Hydrobiologia,* 497**,** 161-167.

SOULLIERE, G. J. & MONFILS, M. J. 1996. Waterbird use of a constructed wetland complex in eastern Upper Michigan. Michigan Department of Natural Resources Wildlife Division.

STANSFIELD, J. H., PERROW, M. R., TENCH, L. D., JOWITT, A. J. D. & TAYLOR, A. A. L. 1997. Submerged macrophytes as refuges for grazing Cladocera against fish predation: Observations on seasonal changes in relation to macrophyte cover and predation pressure. *Hydrobiologia,* 342**,** 229-240.

STARK, J. D., BOOTHROYD, I. K. G., HARDING, J. S., MAXTED, J. R. & SCARSBROOK, M. R. 2001. Protocols for sampling macroinvertebrates in wadeable streams. *New Zealand Macroinvertebrate Working Group Report No. 1.* Ministry for the Environment.

STEINMAN, A. D., CONKLIN, J., BOHLEN, P. J. & UZARSKI, D. G. 2003. Influence of cattle grazing and pasture land use on macroinvertebrate communities in freshwater wetlands. *Wetlands,* 23**,** 877-889.

STENDERA, S., ADRIAN, R., BONADA, N., CANEDO-ARGUELLES, M., HUGUENY, B., JANUSCHKE, K., PLETTERBAUER, F. & HERING, D. 2012. Drivers and stressors of freshwater biodiversity patterns across different ecosystems and scales: a review. *Hydrobiologia,* 696**,** 1-28.

STREET, M. 1977. The food of Mallard ducklings in a wet gravel quarry, and its relation to duckling survival. *Wildfowl,* 28**,** 113-125.

STREET, M. 1978. The role of insects in the diet of Mallard ducklings - an experimental approach. *Wildfowl,* 29**,** 93-100.

STRICKLAND, R. R. 1980. Fisheries Aspects of the Whangamarino Swamp. *Fisheries Environmental Report No. 7.* Turangi: NZ Ministry of Agriculture and Fisheries.

SUGDEN, L. G. 1973. Feeding ecology of pintail, gadwall, American widgeon, and lesser scaup ducklings. *Canadian Wildlife Service Report Series.* Canadian Wildlife Service.

SUYKER, A. E., VERMA, S. B. & ARKEBAUER, T. J. 1997. Season-long measurement of carbon dioxide exchange in a boreal fen. *Journal of Geophysical Research-Atmospheres,* 102**,** 29021-29028.

SVINGEN, D. & ANDERSON, S. H. 1998. Waterfowl management on grass-sage stock ponds. *Wetlands,* 18**,** 84-89.

TARANAKI REGIONAL COUNCIL 2009. Taranaki Where We Stand: State of the environment report 2009. New Plymouth, New Zealand: Taranaki Regional Council.

TRIMBLE, S. W. & MENDEL, A. C. 1995. The cow as a geomorphic agent - a critical-review. *Geomorphology,* 13**,** 233-253.

VANDEKERKHOVE, J., NAMIOTKO, T., HALLMANN, E. & MARTENS, K. 2012. Predation by macroinvertebrates on Heterocypris incongruens (Ostracoda) in temporary ponds: impacts and responses. *Fundamental and Applied Limnology,* 181**,** 39-47.

WALKER, J., ROTELLA, J. J., SCHMIDT, J. H., LOESCH, C. R., REYNOLDS, R. E., LINDBERG, M. S., RINGELMAN, J. K. & STEPHENS, S. E. 2013. Distribution of duck broods relative to habitat characteristics in the Prairie Pothole Region. *Journal of Wildlife Management,* 77**,** 392-404.

WELLBORN, G. A., SKELLY, D. K. & WERNER, E. E. 1996. Mechanisms creating community structure across a freshwater habitat gradient. *Annual Review of Ecology and Systematics,* 27**,** 337-363.

WILCOVE, D. S. & BEAN, M. J. 1994. *The Big Kill: Declining Biodiversity in America's Lakes and Rivers,* Washington, D.C., Environmental Defense Fund.

WILLIAMS, D. D. 1997. Temporary ponds and their invertebrate communities. *Aquatic Conservation-Marine and Freshwater Ecosystems,* 7**,** 105-117.

WILLIAMS, D. D., HEEG, N. & MAGNUSSON, A. K. 2007. Habitat background selection by colonizing intermittent pond invertebrates. *Hydrobiologia,* 592**,** 487-498.

WILLIAMS, P., BIGGS, J., BARR, C. J., CUMMINS, C. P., GILLESPIE, M. K., RICH, T. C. G., BAKER, A., BAKER, J., BEESLEY, J., CORFIELD, A., DOBSON, D., CULLING, A. S., FOX, G., HOWARD, D. C., LUURSEMA, K., RICH, M., SAMSON, D., WHITE, R. & WHITFIELD, M. 1998. Lowland Pond Survey 1996. Oxford, England: Pond Action.

WILLIAMS, P., BIGGS, J., WHITFIELD, M., THORNE, A., BRYANT, S., FOX, G. & NICOLET, P. 1999. *The Pond Book. A Guide to the Management and Creation of Ponds,* Oxford, England, Ponds Conservation Trust.

WILLIAMS, P., WHITFIELD, M. & BIGGS, J. 2008. How can we make new ponds biodiverse? A case study monitored over 7 years. *Hydrobiologia,* 597**,** 137-148.

WILLIAMS, P., WHITFIELD, M., BIGGS, J., BRAY, S., FOX, G., NICOLET, P. & SEAR, D. 2004. Comparative biodiversity of rivers, streams, ditches and ponds in an agricultural landscape in Southern England. *Biological Conservation,* 115**,** 329-341.

WOOD, P. J., GREENWOOD, M. T. & AGNEW, M. D. 2003. Pond biodiversity and habitat loss in the UK. *Area,* 35**,** 206-216.

ZEDLER, J. B. 2003. Wetlands at Your Service: Reducing Impacts of Agriculture at the Watershed Scale. *Frontiers in Ecology and the Environment,* 1**,** 65-72.

Appendices

Appendix 1: Landscape characteristics of 34 ponds on the lower Waikato River floodplain.

Site	Area (ha)	Edge length (m)	Area:perimeter ratio	Overhanging vegetation (%)	Age (class)	Number of islands	Island area (m2)	Island edge length (m)	Distance to 5 nearest waterbodies (m)	Fenced pond margin (%)	Vegetation buffer width (m)	Number of maimai	Water depth < 20 cm (%)	Water depth 20 - 50 cm (%)	Water depth 51 - 100 cm (%)	Water depth > 1 m (%)
HUN_1	5.20	1212	43	70	4	3	4734	486	2141	100	0	0	1	7	70	23
HUN_2	2.75	1277	22	0	2	3	705	207	2247	15	0	0	6	46	45	2
LWK_1	1.87	1018	18	80	2	13	5669	1335	3246	0	106	1	3	24	73	0
LWK_2	4.06	1156	35	85	2	5	1711	426	2537	100	48	1	1	3	96	0
LWK_3	0.81	511	16	5	2	5	2069	495	4817	0	81	0	2	63	35	0
LWK_4	3.12	757	41	5	2	0	0	0	4013	100	14	0	3	19	64	13
LWK_5	1.40	789	18	95	4	1	239	60	1692	100	33	0	9	45	28	18
LWK_6	0.32	247	13	5	2	0	0	0	1070	100	10	0	8	46	46	0
LWK_7	0.49	416	12	65	4	1	160	48	1717	100	13	1	9	25	54	12
LWK_8	0.31	312	10	25	3	2	120	59	754	40	0	1	3	25	51	20
LWK_9	0.21	218	10	30	2	0	0	0	1006	50	11	0	6	9	84	0
LWK_10	0.51	408	13	5	2	0	0	0	884	50	12	0	4	88	8	0
LWK_11	0.59	474	12	0	1	2	611	163	824	50	0	0	3	96	1	0
LWP_1	5.11	1838	28	30	1	0	0	0	2953	25	0	2	11	83	5	2
LWP_2	0.27	273	10	5	2	0	0	0	1589	100	12	1	2	8	19	71
LWP_3	3.57	1523	23	30	4	2	452	129	6580	90	23	4	2	9	64	25
LWP_4	0.27	258	10	0	2	0	0	0	892	60	0	0	9	68	23	0
LWP_5	0.06	111	6	2	1	0	0	0	973	10	0	0	29	51	20	0
LWP_6	0.36	293	12	70	2	0	0	0	1289	50	32	0	9	18	53	20
LWP_7	0.18	288	6	25	3	0	0	0	1219	60	4	0	8	21	51	20
LWP_8	1.51	874	17	5	1	2	32	33	6533	90	0	0	12	31	29	28
LWP_9	2.09	1016	21	5	1	0	0	0	5496	75	0	2	5	21	56	17
RAN_1	0.32	330	10	0	4	0	0	0	3851	15	0	0	3	7	40	50
WGM_1	0.56	653	9	5	4	0	0	0	1479	0	0	1	0	74	18	8
WGM_2	0.21	294	7	10	2	0	0	0	1435	0	117	1	7	19	32	42
WGM_3	0.11	149	7	2	2	0	0	0	1177	0	99	1	7	27	66	0
WGM_4	0.25	364	7	90	2	0	0	0	1258	0	286	1	7	33	55	6
WGM_5	0.35	404	9	5	1	6	477	209	993	0	0	0	10	13	43	34
WGM_6	0.64	422	15	25	3	0	0	0	673	0	132	1	4	13	16	66
WGM_7	0.48	366	13	95	3	0	0	0	520	0	108	1	5	15	38	42
WGM_8	0.26	259	10	5	3	2	45	34	1433	0	504	1	10	90	0	0
WGM_9	0.46	370	13	90	3	1	13	14	753	0	383	1	6	18	70	6
WGM_12	0.29	669	4	95	4	0	0	0	1122	0	14	0	23	44	21	12
WGM_13	0.27	364	8	7.5	1	0	0	0	725	50	136	0	12	71	17	0

Appendix 2: Physicochemical characteristics of 34 ponds on the lower Waikato River floodplain.

Site	Temperature (°C)	Ambient conductivity (µS/cm)	Dissolved oxygen (mg/L)	pH	Turbidity (rank)
HUN_1	17.38	103.37	5.03	7.42	1.67
HUN_2	15.53	201.13	6.13	7.32	3.50
LWK_1	17.81	143.36	4.42	7.53	2.50
LWK_2	17.77	183.38	4.82	7.58	1.50
LWK_3	18.20	123.28	7.44	7.63	2.50
LWK_4	17.62	109.32	8.22	7.55	3.00
LWK_5	16.52	133.05	5.68	7.63	2.83
LWK_6	16.03	113.40	8.26	7.65	3.00
LWK_7	14.78	152.73	4.91	7.48	3.00
LWK_8	17.85	110.33	3.77	8.00	2.00
LWK_9	17.66	202.88	11.58	7.08	1.50
LWK_10	17.52	215.78	9.64	7.15	2.00
LWK_11	18.68	271.02	10.32	6.97	3.50
LWP_1	6.40	101.25	2.57	3.62	2.00
LWP_2	15.85	317.38	7.01	8.10	3.00
LWP_3	15.43	188.53	8.49	7.93	2.00
LWP_4	18.85	98.10	10.07	7.25	2.50
LWP_5	15.60	172.24	7.07	7.20	3.50
LWP_6	17.33	152.48	8.72	7.14	2.00
LWP_7	19.05	81.60	11.05	7.43	1.50
LWP_8	16.22	155.22	9.48	7.35	2.75
LWP_9	16.19	167.80	9.65	7.72	2.00
RAN_1	17.19	130.74	6.19	7.83	1.00
WGM_1	18.83	181.75	8.26	7.55	3.50
WGM_2	19.40	141.62	9.29	7.85	4.00
WGM_3	19.77	315.18	5.13	7.45	4.00
WGM_4	17.40	214.65	7.87	7.07	1.75
WGM_5	16.87	280.87	3.40	8.18	3.00
WGM_6	18.02	128.63	8.28	7.77	4.00
WGM_7	18.90	143.88	6.71	7.85	4.00
WGM_8	19.42	157.80	10.39	7.78	2.00
WGM_9	17.85	153.63	6.39	7.73	4.00
WGM_12	16.00	101.42	4.07	7.63	2.50
WGM_13	17.09	196.12	7.27	7.68	2.00

Appendix 3: The average abundance per hectare of adult and juvenile waterfowl on 34 constructed ponds in the lower Waikato River floodplain. Abundances have been average across four observations. Values to 1 d.p.

Site	Black swan	Canada goose	Grey duck	Grey teal	Mallard	Paradise shelduck	Shoveler	Black swan	Canada goose	Grey duck	Grey teal	Mallard	Paradise shelduck	Shoveler
	Adult waterfowl per hectare							**Juvenile waterfowl per hectare**						
HUN_1	0.5	0.8	0.4	0.2	1.2	0.0	0.6	1.0	0.8	0.0	0.0	2.1	0.0	0.0
HUN_2	0.9	0.2	0.0	1.2	3.5	0.0	0.4	0.0	0.0	0.0	0.0	7.6	0.0	0.0
LWK_1	2.1	2.8	0.7	3.2	2.3	0.1	0.4	2.7	9.6	0.0	1.6	0.0	0.0	0.0
LWK_2	4.1	5.9	0.3	4.9	4.2	0.0	2.8	3.4	2.5	0.0	1.7	0.7	0.0	0.7
LWK_3	1.8	4.3	0.6	1.2	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
LWK_4	0.2	8.1	0.0	0.0	0.3	0.0	0.0	0.0	10.9	0.0	0.0	0.3	0.0	0.0
LWK_5	1.4	2.1	1.6	8.1	5.4	0.0	1.4	5.0	5.0	7.9	6.4	0.0	0.0	6.4
LWK_6	1.6	5.5	1.6	0.8	0.0	0.0	0.0	0.0	6.3	0.0	3.2	0.0	0.0	0.0
LWK_7	2.0	0.0	7.7	7.2	10.2	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
LWK_8	0.0	4.8	1.6	1.6	8.1	0.0	0.0	0.0	0.0	0.0	0.0	3.2	0.0	0.0
LWK_9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
LWK_10	0.5	1.0	0.0	2.4	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
LWK_11	0.0	2.1	0.0	1.3	0.0	0.8	0.0	0.0	8.5	0.0	0.0	0.0	0.0	0.0
LWP_1	0.2	0.5	0.7	1.9	4.4	0.2	0.4	2.0	0.0	0.8	0.0	0.0	0.0	0.0
LWP_2	0.0	0.0	3.7	0.0	5.5	0.0	0.9	0.0	0.0	0.0	0.0	40.6	0.0	0.0
LWP_3	2.7	6.7	1.0	33.6	9.4	0.8	1.9	2.8	0.0	0.6	1.4	5.0	4.8	3.4
LWP_4	4.6	6.5	3.7	13.9	0.0	0.0	0.0	18.6	0.0	0.0	0.0	0.0	0.0	0.0
LWP_5	0.0	0.0	0.0	0.0	0.0	11.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
LWP_6	0.7	1.4	5.6	6.3	3.5	0.0	0.0	11.2	0.0	0.0	0.0	0.0	0.0	0.0
LWP_7	0.0	0.0	7.0	0.0	4.2	0.0	0.0	0.0	0.0	16.8	0.0	0.0	0.0	0.0
LWP_8	7.8	2.7	4.3	12.3	8.1	9.6	2.8	3.3	0.0	11.9	3.3	8.6	10.6	6.0
LWP_9	4.1	0.2	2.5	9.9	3.7	0.1	0.7	3.3	0.0	3.8	4.8	10.0	0.0	0.0
RAN_1	0.0	0.0	0.8	1.5	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0	9.2	0.0
WGM_1	0.0	0.0	11.2	25.6	11.7	0.0	1.8	0.0	0.0	14.4	32.3	7.2	0.0	9.0
WGM_2	0.0	0.0	4.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
WGM_3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
WGM_4	0.0	0.0	0.0	3.0	4.1	0.0	4.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
WGM_5	0.0	0.0	5.6	15.5	2.8	0.0	4.2	0.0	0.0	0.0	2.8	0.0	0.0	0.0
WGM_6	0.0	0.0	0.0	0.0	4.3	0.0	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0
WGM_7	1.0	2.1	1.0	11.9	3.6	0.0	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
WGM_8	0.0	2.0	2.9	0.0	3.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
WGM_9	0.0	0.0	3.2	9.2	1.6	0.0	1.1	0.0	0.0	10.8	0.0	0.0	0.0	0.0
WGM_12	0.0	0.0	1.7	0.9	0.0	0.0	3.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0
WGM_13	1.9	0.9	0.0	0.0	2.3	0.0	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Appendix 4: Fish abundance and biomass from 34 constructed ponds on the lower Waikato River floodplain. CPUE = catch per unit effort (2 fyke nets, 3 Gee minnow traps and 10 minutes backpack electrofishing). Common smelt and Black mudfish have been omitted as they were only found at 1 and 2 sites, respectively.

Site	Common bully	Longfin eel	Shortfin eel	Gambusia	Goldfish	Catfish	Koi carp	Common bully	Longfin eel	Shortfin eel	Gambusia	Goldfish	Catfish	Koi carp
	Fish abundance (CPUE^{-1})							Fish biomass (g CPUE^{-1})						
HUN_1	0	0	41	0	0	0	0	0	0	8280	0	0	0	0
HUN_2	0	0	36	0	5	20	20	0	0	13,380	0	1,950	3,940	24,900
LWK_1	0	0	3	0	0	0	0	0	0	2,740	0	0	0	0
LWK_2	0	0	13	0	0	0	0	0	0	4,960	0	0	0	0
LWK_3	0	8	12	0	0	1	0	0	9,270	2,850	0	0	990	0
LWK_4	0	0	6	52	0	1	0	0	0	3,040	7	0	70	0
LWK_5	0	4	65	0	0	0	0	0	960	32,960	0	0	0	0
LWK_6	0	0	16	0	0	0	0	0	0	4,990	0	0	0	0
LWK_7	0	0	10	3	0	0	0	0	0	8,640	1	0	0	0
LWK_8	0	0	10	306	3	0	0	0	0	5,960	35	60	0	0
LWK_9	0	0	0	0	0	0	0	0	0	0	0	0	0	0
LWK_10	0	0	0	0	0	0	0	0	0	0	0	0	0	0
LWK_11	0	0	0	0	0	0	0	0	0	0	0	0	0	0
LWP_1	3	0	29	39	0	8	60	9	0	6,820	8	0	1,260	74,700
LWP_2	0	1	20	7	0	0	0	0	0	0	3	0	0	0
LWP_3	4	0	22	22	56	0	0	5	0	250	4	1,164	0	0
LWP_4	0	0	2	53	0	0	0	0	0	1,060	6	0	0	0
LWP_5	0	0	3	116	0	0	0	0	0	1,500	16	0	0	0
LWP_6	0	0	4	0	0	0	0	0	0	1,840	0	0	0	0
LWP_7	0	0	0	474	0	0	0	0	0	0	103	0	0	0
LWP_8	0	0	0	0	0	0	0	0	0	0	0	0	0	0
LWP_9	0	0	0	0	0	0	0	0	0	0	0	0	0	0
RAN_1	2	0	2	0	0	0	0	2	0	0	0	0	0	0
WGM_1	0	0	214	0	37	4	9	0	0	23,560	0	6,330	620	11,205
WGM_2	3	3	1	40	0	0	0	15	1,700	540	7	0	0	0
WGM_3	0	0	5	153	1	0	4	0	0	1,470	34	2	0	4,980
WGM_4	0	0	16	14	3	2	22	0	0	2,350	3	500	500	27,390
WGM_5	0	0	31	24	0	3	0	0	0	5,210	4	0	610	0
WGM_6	1	0	62	138	2	25	15	1	0	7,510	28	610	3,130	18,675
WGM_7	23	0	4	5	1	6	14	37	0	610	3	130	410	17,430
WGM_8	0	0	9	0	3	0	0	0	0	2,020	0	610	0	0
WGM_9	0	0	36	57	0	14	1	0	0	6,170	13	0	1,450	1,245
WGM_12	7	0	3	185	0	0	0	5	0	0	26	0	0	0
WGM_13	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix 5: The relative abundance and biomass (g CPUE^{-1}) of macroinvertebrates sampled from one 3-minute sweep in each of 34 constructed ponds on the lower Waikato River floodplain.

Site	\<-- Relative abundance --\> Crustacea	Coleoptera	Hemiptera	Diptera	Odonata	Other insects	Gastropoda	Oligochaeta	Other	\<-- Biomass (g CPUE^{-1}) --\> Crustacea	Coleoptera	Hemiptera	Diptera	Odonata	Other insects	Gastropoda	Oligochaeta	Other
HUN_1	99.2	0.0	0.2	0.6	0.0	0.0	0.0	0.0	0.0	1.12	0.00	0.18	0.02	0.00	0.00	0.00	0.00	0.00
HUN_2	97.2	0.0	0.0	2.4	0.5	0.0	0.0	0.0	0.0	0.60	0.00	0.00	0.27	0.02	0.00	0.00	0.00	0.00
LWK_1	49.1	3.0	5.2	0.0	42.1	0.7	0.0	0.0	0.0	0.02	0.01	0.03	0.00	0.11	0.00	0.00	0.00	0.00
LWK_2	99.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LWK_3	89.5	0.0	3.5	3.9	0.0	0.9	0.0	0.4	1.7	0.01	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00
LWK_4	80.9	0.0	11.8	3.3	1.4	0.0	0.1	0.0	2.4	0.03	0.00	0.12	0.00	0.00	0.00	0.00	0.00	0.01
LWK_5	56.2	0.4	17.4	23.2	0.0	0.9	0.0	0.4	1.3	0.04	0.00	0.16	0.07	0.00	0.02	0.00	0.00	0.00
LWK_6	76.2	0.0	13.1	10.7	0.0	0.0	0.0	0.0	0.0	0.84	0.00	1.05	0.27	0.00	0.00	0.00	0.05	0.00
LWK_7	76.0	1.7	14.5	2.1	0.4	0.0	0.8	0.0	4.5	0.07	0.02	1.03	0.00	0.01	0.00	0.00	0.00	0.02
LWK_8	28.8	0.0	2.8	45.6	0.0	1.1	18.6	0.0	3.2	0.02	0.00	0.11	0.04	0.00	0.02	0.48	0.00	0.09
LWK_9	53.7	5.0	1.9	0.9	29.2	0.0	6.0	0.5	2.8	0.05	0.03	0.01	0.01	0.19	0.00	0.19	0.01	0.04
LWK_10	98.6	0.0	0.2	0.0	0.3	0.0	0.0	0.2	0.7	3.41	0.00	0.11	0.00	0.23	0.00	0.00	0.14	0.11
LWK_11	90.2	0.7	0.0	2.9	0.0	0.0	0.0	6.2	0.0	0.13	0.03	0.00	0.03	0.00	0.00	0.00	0.01	0.00
LWP_1	18.5	0.4	0.8	72.2	1.6	0.0	4.0	0.4	2.0	0.02	0.00	0.02	0.15	0.14	0.00	0.19	0.02	0.10
LWP_2	61.2	0.0	31.5	3.2	0.0	0.0	0.0	0.0	4.1	0.06	0.00	0.21	0.03	0.00	0.00	0.00	0.00	0.07
LWP_3	14.5	0.0	0.0	80.5	0.4	0.0	0.0	4.0	0.5	0.01	0.00	0.00	0.06	0.01	0.00	0.00	0.01	0.00
LWP_4	38.6	3.6	16.5	40.7	0.0	0.0	0.0	0.0	0.6	0.21	0.15	0.52	0.15	0.00	0.00	0.00	0.00	0.03
LWP_5	68.4	0.0	22.5	6.3	0.0	0.0	0.0	2.7	0.1	0.01	0.00	0.28	0.01	0.00	0.00	0.00	0.01	0.00
LWP_6	96.6	0.0	1.4	1.9	0.0	0.0	0.0	0.0	0.0	0.96	0.00	2.07	0.13	0.00	0.00	0.00	0.00	0.00
LWP_7	48.4	7.1	13.3	24.2	0.9	0.9	3.2	1.4	0.5	0.05	0.07	3.43	0.03	0.00	0.01	0.02	0.01	0.00
LWP_8	98.7	0.0	0.9	0.4	0.0	0.0	0.0	0.0	0.0	0.58	0.00	0.38	0.05	0.00	0.00	0.00	0.00	0.00
LWP_9	98.1	0.0	0.0	1.9	0.0	0.0	0.0	0.0	0.0	1.97	0.22	0.00	0.19	0.00	0.00	0.00	0.00	0.00
RAN_1	6.0	2.0	4.6	3.3	25.3	0.5	58.4	0.0	0.0	0.00	0.02	0.05	0.00	0.30	0.00	0.36	0.00	0.00
WGM_1	63.0	0.0	1.3	35.3	0.0	0.0	0.0	0.4	0.0	0.02	0.00	0.08	0.02	0.00	0.00	0.00	0.00	0.00
WGM_2	7.7	0.5	2.4	76.0	0.5	0.0	10.6	0.0	2.4	0.02	0.01	0.13	0.04	0.00	0.00	0.23	0.01	0.02
WGM_3	65.5	0.0	4.6	19.1	5.7	0.0	1.5	3.1	0.5	0.01	0.00	0.11	0.01	0.02	0.00	0.01	0.01	0.00
WGM_4	94.4	0.0	1.3	3.4	0.9	0.0	0.0	0.0	0.0	0.02	0.00	0.07	0.01	0.01	0.00	0.00	0.00	0.00
WGM_5	62.4	0.0	2.4	26.3	0.0	0.0	7.8	0.0	1.0	0.07	0.00	0.35	0.17	0.00	0.00	0.26	0.00	0.11
WGM_6	52.8	1.9	5.7	15.1	3.8	7.5	13.2	0.0	0.0	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.00	0.00
WGM_7	56.6	0.9	9.4	25.5	2.8	1.9	2.8	0.0	0.0	0.00	0.00	0.03	0.05	0.01	0.00	0.00	0.00	0.00
WGM_8	82.7	2.4	6.2	5.8	2.4	0.0	0.5	0.1	0.0	0.03	0.19	0.17	0.01	0.02	0.00	0.00	0.00	0.00
WGM_9	66.1	1.0	22.5	4.3	5.1	0.0	0.5	0.0	0.5	0.01	0.00	0.26	0.00	0.01	0.00	0.00	0.00	0.00
WGM_12	0.0	0.0	0.5	94.6	0.0	1.5	2.9	0.0	0.5	0.00	0.00	0.03	0.12	0.00	0.00	0.10	0.00	0.03
WGM_13	93.9	1.9	1.8	0.9	1.4	0.0	0.1	0.0	0.0	0.40	0.03	0.50	0.02	0.02	0.00	0.00	0.00	0.00

Appendix 6: The relative abundance and biomass (g CPUE^{-1}) of benthic macroinvertebrates captured in nine grabs in each of 34 constructed ponds on the lower Waikato River floodplain.

Site	Relative abundance									Biomass								
	Crustacea	Coleoptera	Hemiptera	Diptera	Odonata	Other insects	Gastropoda	Oligochaeta	Other	Crustacea	Coleoptera	Hemiptera	Diptera	Odonata	Other insects	Gastropoda	Oligochaeta	Other
HUN_1	0.0	0.0	0.0	77.8	0.0	0.0	0.0	11.1	11.1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
HUN_2	7.7	0.0	0.0	92.3	0.0	0.0	0.0	0.0	0.0	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00
LWK_1	0.0	0.0	0.0	28.6	0.0	0.0	0.0	71.4	0.0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00
LWK_2	0.0	0.0	0.0	1.5	0.0	0.0	0.0	98.5	0.0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00
LWK_3	0.0	0.0	0.0	51.5	0.0	0.0	0.0	48.5	0.0	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.00
LWK_4	0.0	0.0	14.3	28.6	0.0	0.0	0.0	57.1	0.0	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00
LWK_5	1.7	0.0	0.0	92.5	0.0	0.0	0.8	5.0	0.0	0.00	0.00	0.00	0.07	0.00	0.00	0.01	0.00	0.00
LWK_6	72.0	0.0	0.5	27.5	0.0	0.0	0.0	0.0	0.0	0.27	0.00	0.03	0.27	0.00	0.00	0.00	0.00	0.00
LWK_7	18.8	0.0	6.3	50.0	0.0	0.0	0.0	0.0	25.0	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00
LWK_8	3.1	0.0	0.0	93.9	0.0	0.0	0.5	0.0	2.6	0.01	0.00	0.00	0.06	0.00	0.00	0.01	0.01	0.02
LWK_9	45.9	0.0	0.0	1.6	1.6	0.0	0.0	50.8	0.0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00
LWK_10	4.1	0.5	0.0	2.7	0.0	0.0	0.0	92.8	0.0	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.66	0.00
LWK_11	0.0	1.9	0.0	22.3	0.0	0.0	0.0	75.8	0.0	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.19	0.00
LWP_1	3.3	0.0	0.0	67.5	0.0	0.0	26.0	2.4	0.8	0.00	0.00	0.00	0.02	0.00	0.00	0.06	0.01	0.00
LWP_2	8.6	0.0	1.9	69.5	0.0	0.0	0.0	17.1	2.9	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.02	0.00
LWP_3	1.0	0.0	0.0	98.9	0.0	0.0	0.0	0.0	0.0	0.00	0.00	0.00	0.10	0.00	0.00	0.00	0.00	0.00
LWP_4	8.0	0.5	0.0	35.7	0.0	0.0	0.0	54.9	0.9	0.03	0.05	0.00	0.08	0.00	0.00	0.00	0.26	0.03
LWP_5	0.0	1.0	0.0	66.0	0.0	0.0	0.0	10.7	22.3	0.00	0.00	0.00	0.05	0.00	0.00	0.00	0.03	0.02
LWP_6	5.4	0.0	0.0	79.1	0.0	0.0	0.0	14.3	1.2	0.01	0.00	0.00	0.20	0.00	0.00	0.00	0.12	0.01
LWP_7	26.3	0.0	0.0	61.5	3.4	0.6	0.0	7.6	0.6	0.00	0.00	0.00	0.02	0.01	0.00	0.00	0.01	0.00
LWP_8	7.7	0.0	0.0	75.7	0.0	0.0	0.0	16.7	0.0	0.01	0.00	0.00	0.50	0.00	0.00	0.00	0.13	0.00
LWP_9	36.7	0.1	0.0	36.2	0.0	0.6	0.0	26.0	0.6	0.06	0.00	0.00	0.30	0.00	0.01	0.00	0.26	0.00
RAN_1	63.6	0.0	0.0	18.2	9.1	0.0	0.0	9.1	0.0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
WGM_1	0.0	0.0	0.0	100.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
WGM_2	2.6	0.0	0.0	76.9	5.1	0.0	0.0	0.0	15.4	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
WGM_3	0.0	0.0	0.0	100.0	0.0	0.0	0.0	0.0	0.0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
WGM_4	14.3	0.0	0.0	28.6	0.0	0.0	14.3	42.9	0.0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00
WGM_5	16.3	0.0	0.0	81.6	0.0	0.0	2.0	0.0	0.0	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00
WGM_6	0.0	0.0	0.0	40.0	0.0	10.0	0.0	50.0	0.0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00
WGM_7	20.0	0.0	0.0	63.3	0.0	0.0	0.0	10.0	6.7	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00
WGM_8	2.9	0.0	0.0	85.3	0.0	2.9	0.0	5.9	2.9	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.00
WGM_9	0.0	0.0	0.0	42.1	0.0	0.0	0.0	52.6	5.3	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.00
WGM_12	0.0	0.0	0.0	94.7	1.8	0.0	0.6	2.4	0.6	0.00	0.00	0.00	0.01	0.00	0.00	0.02	0.00	0.00
WGM_13	23.1	0.0	0.0	7.7	11.5	0.0	0.0	57.7	0.0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00

Appendix 7: Factors used in each PERMANOVA test and the categories used in the pair-wise PERMANOVA tests.

Factor	Categories	Description
Location	HUN; LWK; LWP; WGM; RAN	Based on the geographic region where the pond was located. HUN = Huntly; LWK = Lake Waikare; LWP = Lake Whangape; WGM = Whangamarino wetland; RAN = Rangiriri.
Water supply	Drain; Rain; Spring; Swamp; Other	The pond's dominant source of water.
Permanence	Never; Rarely; Sometimes; Annually	Rank system based on how often the pond dries.
Flood frequency	Never; Occasionally; Annually	Rank system based on how often the pond floods beyond spring-summer water extent.
Pond origin	Grazed land; Marginal wetland; Wetland	Dominant land-use prior to original pond construction. Marginal wetland = land was inundated during wet months, and grazed during dry months.
Dominant substrate	Silt/clay; Peat	Visual assessment of the dominant pond substrate.
Hunted on	Yes; No	Based on landowner interviews, irrespective of the number of maimai present at the pond.

Appendix 8: Variables included in the DistLM tests.

Variables	Unit	Variable	Unit
Landscape		**Riparian vegetation $^\delta$**	
Area*$^\delta$	m^2	Rank grass	% cover
Edge length*$^\delta$	m	Pasture grass	% cover
Area:perimeter*$^\delta$	Ratio	Reeds	% cover
Overhanging vegetation*$^\delta$	%	*Carex*	% cover
Age *$^\delta$	Class	Kahikatea stand	% cover
Number of islands$^\delta$	Count	Mixed bush	% cover
Island area$^\delta$	m^2	Willow bush	% cover
Island edge$^\delta$	m	Mixed shrub	% cover
Distance to five nearest waterbodies$^\delta$	m	Oaks	% cover
Fenced margin*$^\delta$	%	Raupo	% cover
Riparian buffer width*$^\delta$	m	Flax	% cover
Number of maimai$^\delta$	Count	*Juncus*	% cover
Depth < 20 cm$^\delta$	%	*Glyceria*	% cover
Depth 20 – 50 cm$^\delta$	%	Willow weed	% cover
Depth 51 – 100 cm$^\delta$	%	Toi toi	% cover
Depth > 1 m$^\delta$	%	*Cyperaceae*	% cover
Macrophyte-free open water$^\delta$	%	Pine trees	% cover
		Manuka	% cover
Water quality/biota		**Aquatic vegetation $^\delta$**	
Food availability$^\delta$	Score	Willow weed	% cover
Nesting habitat$^\delta$	Score	*Ludwigia*	% cover
Loafing habitat$^\delta$	Score	Parrots feather	% cover
Overhead cover$^\delta$	Score	Duckweed	% cover
Temperature*$^\delta$	^{o}C	Reeds	% cover
Ambient conductivity*$^\delta$	µS/cm	*Glyceria*	% cover
Dissolved oxygen*$^\delta$	mg/L	*Ricciocarpus*	% cover
pH*$^\delta$		*Azolla*	% cover
Turbidity*$^\delta$	Rank	*Juncus*	% cover
Koi carp abundance*$^\delta$	CPUE^{-1}	Willow trees	% cover
Shortfin eel abundance*$^\delta$	CPUE^{-1}	*Carex*	% cover
Gambusia abundance*$^\delta$	CPUE^{-1}	Hornwort	% cover
Goldfish abundance*$^\delta$	CPUE^{-1}	*Cyperaceae*	% cover
Sweep macroinvertebrate abundance*$^\delta$	CPUE^{-1}	Cane rush	% cover
Benthic macroinvertebrate abundance*$^\delta$	CPUE^{-1}		

* = used in Chapter 3 DistLMs; $^\delta$ = used in Chapter 4 DistLMs. All riparian and aquatic vegetation variables only used in Chapter 4 DistLMs.