Are aquatic invertebrates useful for assessing wetland condition?

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General Abstract

Freshwater wetlands are one of the most biodiverse ecosystems and at the same time of the most threatened globally. New Zealand has lost 90% of its wetlands and of those remaining, 60% are considered degraded. Establishing accurate wetland inventories and assessing wetland condition are priorities for the management and conservation of these important ecosystems. Aquatic invertebrates are used worldwide to assess the condition of other aquatic ecosystems such as rivers and lakes; however, their use for assessing wetland condition has not been extensive.

A wetland's hydroperiod is considered one of the most important environmental variables affecting wetland biota and one that has also been most altered by anthropogenic stresses. The second chapter of this thesis analyses the effect of hydroperiod on the macroinvertebrate communities of the Ō Tū Wharekai (Ashburton lakes) wetland system in New Zealand. A total of 40 taxa from 11 orders were recorded from 4 permanent lakes, 3 semi-permanent ponds, and 7 temporary ponds in September 2016. The macroinvertebrate assemblages in lakes were distinct to those in semi-permanent and temporary ponds. Overall, temporary ponds were slightly more diverse than the semi-permanent ponds and lakes. Semi-permanent and temporary ponds were most similar to each other in macroinvertebrate composition. They host more species of small crustaceans such as cladocerans and ostracods, while species belonging to the Trichoptera, Odonata and Hirudinea orders were only present at permanent sites. The results emphasize the need to include small and seasonal wetlands in freshwater conservation efforts since they often hold unique biotic communities.

In the third chapter, the potential to use macroinvertebrate communities in wetland assessment is evaluated. The macroinvertebrate communities of 14 freshwater wetlands in the lower North Island were sampled. The sites represent a gradient of wetland condition and include urban lagoons, agricultural swamps and lacustrine wetlands with recognized ecological value. A total of 63 invertebrate taxa were identified, of which crustaceans were the most abundant. There appeared to be no link between the

composition and diversity of macroinvertebrate communities and wetland condition. However, of the habitat characteristics measured at each site, nutrient enrichment appeared to be the most important variable in determining macroinvertebrate assemblages. On the other hand, macrophyte communities appear to be more reflective of wetland condition. There are considerable knowledge gaps regarding invertebrate response to environmental change in freshwater wetlands and this limits their suitability as a biomonitoring tool.

Assessing wetland condition accurately is one of the greatest challenges for the management and conservation of these threatened ecosystems. Aquatic invertebrates are used as biomonitoring tool for many freshwater ecosystems but not wetlands. This is because the way wetland invertebrates respond to environmental change remains unclear. So far, in New Zealand, there appears to be no link between wetland condition scores and invertebrate communities. Thus, the final section of this thesis proposes a simple dichotomous wetland condition scoring system exemplified with information from the 14 freshwater wetland sampled in the North Island. The method has limitations, but allows the integration of biotic data into wetland condition assessment.

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Chapter 1:

The Conservation Value of Wetlands

Wetlands associated to Lake Kohangapiripiri

Wetlands are ecosystems that represent the transition between aquatic and terrestrial landscapes and are considered some of the most biodiverse and productive in the world (Mitsch and Gosselink 2015). The most widely used definition stems from the first article of the Ramsar Convention on Wetlands:

"areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres"

This definition encompasses a wide variety of habitats that are permanently or intermittently inundated. According to Batzer and Boix (2016) wetlands are mostly defined by their climate, hydrology and vegetation, while Mitsch and Gosselink (2015) suggest that wetlands are distinguished by three factors: the presence of water, unique soil characteristics and the presence of biota specialized for wet conditions. Jackson et al. (2014) point out that these variables are often interdependent, giving rise to a circular description of their physical characteristics. In New Zealand, wetlands are defined by the Resource Management Act (1991) as: "permanently or intermittently wet areas, shallow water, and land water margins that support a natural ecosystem of plants and animals that are adapted to wet conditions".

Assessing a wetland's hydrology is a logical first step to characterizing these habitats. The hydrology of a wetland can be described through its inflows and outflows of water (water budget) and through the geomorphology of its basin (Mitsch and Gosselink 2015). Hydroperiod, or the amount of time standing (surficial) water is present in a wetland is also considered a defining characteristic (Batzer and Boix 2016). Based on a wetlands' association with water, the Ramsar Convention of Wetlands recognizes five major types: marine, estuarine, riverine, lacustrine and palustrine. In an effort to provide a classification scheme for wetlands in New Zealand, Johnson and Gerbeaux (2004) add four other types (inland saline, plutonic, geothermal and nival) to the previous list (Fig. 1).

Table 1. Classification of major New Zealand wetland types (Johnson and Gerbeaux 2004, RAMSAR 2016)

Type of Wetland	Characteristics
Marine	Coastal wetlands including coastal lagoons, rocky shores, seagrass beds and coral reefs
Estuarine	River deltas, tidal marshes, mudflats, and mangrove swamps
Lacustrine	Wetlands associated with lakes
Riverine	Wetlands along rivers or streams
Palustrine	Marshes, swamps and bogs
Inland Saline	Inland wetlands where strong evaporation results in high concentrations of salts
Plutonic	Wetlands that occur as caves or underground systems
Geothermal	Wetlands with geothermal-derived water
Nival	Frozen hydrosystems such as snowfields and glaciers

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Man-made water bodies such as dams, reservoirs, rice paddies, urban and agricultural ponds and lagoons, waste water treatment canals and others are also considered wetlands (Johnson and Gerbeaux 2004, RAMSAR 2016). Over the last 38 years the proportion of human made wetlands around the world has increased (RAMSAR 2015). There are potential physical and psychological health benefits associated with urban wetlands as well as an increase of public interest in urban conservation (Carter 2015). Nevertheless the gains in man-made wetlands do not compensate for the loss of natural wetlands and their associated ecosystem services (RAMSAR 2015).

Wetland soils are characterized by saturation and are also called hydric soils (Mitsch and Gosselink 2015). Air in the soil is displaced by water and the remaining oxygen is depleted by microbes, this leads to anaerobic conditions which are exploited by facultative and anaerobic microorganisms through oxidation-reduction reactions (Jackson et al. 2014). Wetland soils can be of mineral or organic origin (Mitsch and Gosselink 2015). Soils are classified as organic if the organic matter is above 17%, while the term 'peat' refers to soils with more than 50% organic content (Johnson and Gerbeaux 2004). Environmental gradients result in many transformations of nitrogen, sulphur, iron, carbon, phosphorus and manganese occurring in wetland ecosystems (Mitsch and Gosselink 2015). Wetlands act as both sources and sinks of nutrients (Kayranli et al. 2010, Mitsch and Gosselink 2015). Nutrient status or fertility can be used to classify wetlands as oligotrophic (nutrient poor), mesotrophic (moderately fertile) or eutrophic (nutrient rich) (Johnson and Gerbeaux 2004). Nutrient status and reduction of water quality in lentic environments and rivers has been linked to land-use in the surrounding catchment (Galbraith and Burns 2007, Julian et al. 2017).

The flora of wetlands have developed a number of adaptations to flooding such as pore space in cortical tissues, fluted trunks, prop and adventitious roots (Mitsch and Gosselink 2015). Dominant vegetation is also used to describe wetlands and can be categorized into emergent annual macrophytes, emergent perennial macrophytes, submersed macrophytes, woody trees and shrubs and algae (Batzer and Boix 2016). Wetland plant communities such as flax swamps of *Phormium,* reed estuaries with *Leptocarpus* and *Juncus,* and kahikatea (*Dacrycarpus dacrydioides*) swamp forests are unique to New Zealand (Cromarty and Scott 1995). Aquatic vegetation can increase habitat complexity and associated biodiversity (Hornung and Foote 2006). Nevertheless, invasive species have been recognized as a major threat to freshwater ecosystems (Saunders et al. 2002); in New Zealand there are over 70 introduced aquatic plants (Elston et al. 2015). Invasive water weeds can have severe consequences for human activities and health as well as reduce biodiversity (Howard and Harley 1998).

Junk et al. (2006) in a comparative study of globally significant wetlands show that these ecosystems are very diverse and usually support threatened or rare species adding to their conservation value. Overall, wetlands are mostly recognized for their value as bird habitat (RAMSAR 2016). In New Zealand wetlands provide feeding grounds for migratory birds that come to the country during the northern winter (Weeks et al. 2016). Some iconic wetland birds in New Zealand include the Australasian bittern, the Australasian crested grebe, the black stilt, the blue duck, the brown and subantartic teal, the wry bill, among others (Cromarty and Scott 1995). Notable are also five species of endemic mudfish that are specifically adapted to wetland habitats (Weeks et al. 2016).

Considerable taxonomic bias exists across the biological sciences. Although wetland invertebrates make up the food supply of most other species (Batzer and Wissinger 1996), research on this group remains scarce (Clark and May 2002). Strayer (2006) states that even the best-known and legally protected invertebrates are only 1% as well studied when compared to vertebrate species. Batzer and Ruhi (2013) analyzed taxa lists from 447 wetlands around the world to determine the invertebrates species commonly found in these ecosystems. They found that wetlands are dominated by 40 species of widespread invertebrates, but also that wetlands are highly variable and invertebrate occurrence is hard to predict. Batzer and Ruhi (2013) also found that the invertebrate species in Australia and New Zealand deviate from that main core. Some freshwater invertebrate groups are poorly represented in New Zealand while other groups have their southernmost representatives; other distinctive features include a large number of primitive groups and a high proportion of endemic species (Collier 1993). There is no national data-base for invertebrates in New Zealand and information remains incomplete, however 295 freshwater invertebrate species are recognized under some status of threat (Joy and Death 2014).

Wetlands provide a number of ecosystem services, many of which are important to humans. The interactions between its biotic and abiotic characteristics give wetlands a unique water storage capacity: they mitigate both floods and droughts, they retain nutrients and sediment purifying the water, they play a role in recharging aquifers, they protect the shoreline and control erosion, and they are important in maintaining climate stability (Mitsch and Gosselink 2015, RAMSAR 2016). These highly diverse ecosystems provide a large variety of resources for the human population and are often associated with cosmological and spiritual beliefs, and play a role in society's traditions. For the

Maori, wetlands are named "repo" or "ngaere" and are associated with some of the body's organs such as the liver and the kidneys, which signifies their cleansing role (Harmsworth 2002). Furthermore, their historical and present importance for resource and food gathering is widely recognized (Cromarty and Scott 1995, Harmsworth 2002, GWRC 2003, Ausseil et al. 2008). Wetlands provide a variety of useful plants: harakeke, raupo and toetoe for weaving, kuta for carving and others as medicine. Their importance as mahinga kai (food gathering sites) is undeniable; examples include raupo roots and pollen to make bread and porridge, fish such as tuna (eels) and kākahi (mussels).

Freshwater ecosystems are among the most threatened in the world (Saunders et al. 2002); monitored freshwater populations have declined 76% over the last 40 years (RAMSAR 2016). Today, around 5-8% of the Earth is covered by wetlands (Mitsch and Gosselink 2015) and it is estimated that their extent has declined between 64% and 72% over the last century (RAMSAR 2016). Wetland drainage and conversion for agriculture is an obvious driver of decline and one that was largely encouraged before the 1970's (Mitsch and Gosselink 2015). This acts as a hydrological stressor (Craft 2016a), but chemical and biological factors also stress wetlands. For wetlands, some of the most important drivers of decline include: hydrological and geomorphic modifications, urbanisation, nutrient enrichment through agricultural intensification, pollution with heavy metals and pesticides, invasive species, salinization, poaching and climate change (Brinson and Malvárez 2002, Saunders et al. 2002, Jackson et al. 2014, Mitsch and Gosselink 2015, Craft 2016a, Weeks et al. 2016).

New Zealand's oceanic climate provides heavy rainfalls and facilitates wetland development (Brinson and Malvárez 2002). Nevertheless, a 90% wetland loss in New Zealand is cited as one of the most dramatic examples worldwide (Mitsch and Gosselink 2015). Although this figure is recurrently used, attempts at delineating wetland extent in New Zealand are limited. Cromarty and Scott (1995) compiled a wetland directory with an expert panel approach but lacked biological data and detailed wetland maps. More recently, as part of the Waters of National Importance (WONI) project, Ausseil et al. (2008) delineated wetland extent and ecological status in the country including a classification system as was proposed by Johnson and Gerbeaux (2004). A total of 7032 wetlands were mapped, and the 90% loss confirmed. The loss has been greater in the North Island, which only retains 4.9% of the original extent (Fig. 2). Furthermore the Index of Ecological Integrity developed by Ausseil et al. (2008) indicates that more that 60% of the country's wetlands are moderately to severely degraded. Furthermore, the distribution of loss vs protection of wetland ecosystems is uneven. Although 63% of inland palustrine wetlands are now within protected areas (Robertson 2015) most of the wetlands in low-land fertile areas are not and are among the most degraded in the country (Ausseil et al. 2008).

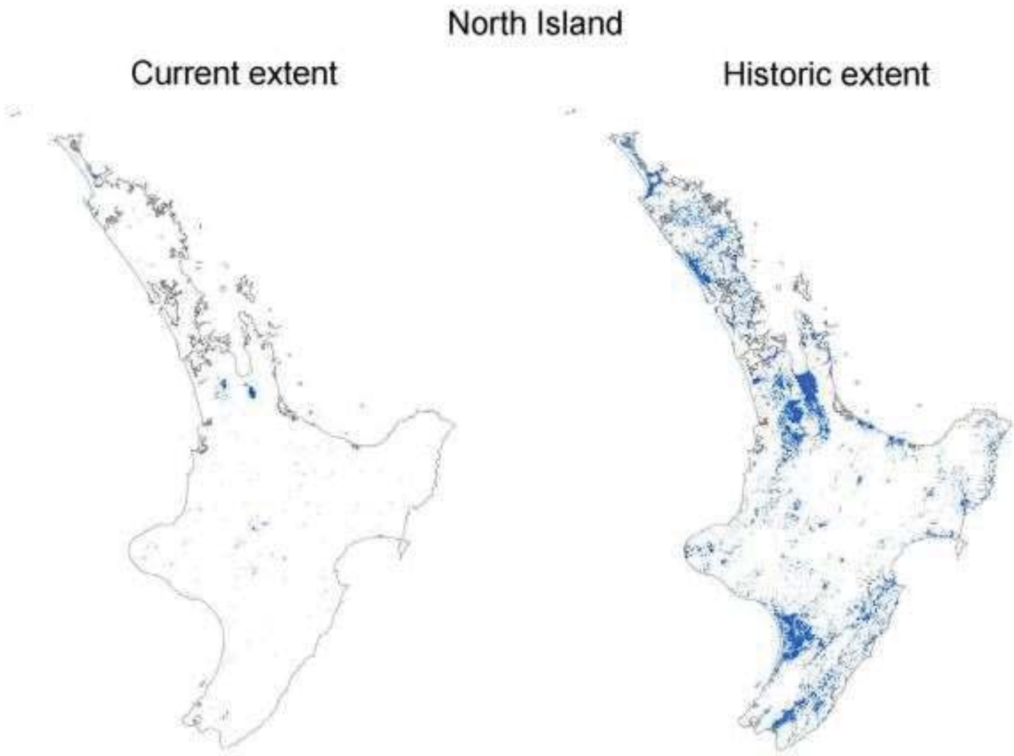

Figure. 2 Current and historic extent of wetlands in New Zealand, reproduced from Ausseil et al. (2008).

While the uniqueness and importance of wetlands is recognized, so are the imminent threats these ecosystems face. Wetland studies have only recently emerged as a discipline in its own right (Mitsch and Gosselink 2015) and filling in knowledge gaps about these ecosystems can only be beneficial for their appropriate management. Invertebrates in wetlands are among the least studied organisms but they are also the staple biomonitoring tool of many freshwater ecosystems (Bonada et al. 2006), which is why they are the main focus of this study. There is little consensus about how wetland invertebrates respond to environmental changes (Batzer 2013). In wetlands, hydroperiod is an obvious and important environmental driver, thus the second chapter of this thesis assesses the effect of hydroperiod on macroinvertebrate communities of the Ashburton Lakes complex in the South Island of New Zealand. In the third chapter, I examine the effect of ecological condition and environment on determining the macroinvertebrate communities of 14 palustrine and lacustrine wetlands in the lower North Island of New Zealand and whether they are a useful indicator of wetland condition. Finally, I provide some ideas about how wetland condition could be assessed.

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Chapter 2:

Macroinvertebrate communities of the Ō Tū Wharekai (Ashburton lakes) wetland system in New Zealand: the effect of hydroperiod

Temporary Pond in the Ashburton Lakes Complex

Abstract

The littorial macroinvertebrate assemblages of 4 lakes, 3 semi-permanent ponds, and 7 temporary ponds in the Ō Tū Wharekai (Ashburton lakes) wetland system in Canterbury, New Zealand were sampled in September 2016. A total of 40 taxa from 11 orders were recorded. Overall, temporary ponds were slightly more diverse than the semi-permanent ponds and lakes. Semi-permanent and temporary ponds were most similar to each other in macroinvertebrate composition, while permanent lakes were distinct. Semi-permanent and temporary sites host more species of small crustaceans such as cladocerans and ostracods, while species belonging to the Trichoptera, Odonata and Hirudinea orders were only present at permanent sites. Seasonal ponds have unique environmental and biological characteristics and should be given extra consideration in freshwater conservation.

Introduction

Whole catchment management has been recognized as an effective strategy for conservation of freshwater ecosystems (Saunders et al. 2002). To best achieve this goal the individual contribution of different water bodies to biodiversity must be assessed and should include both small and seasonal sites. Temporary wetlands, such as ponds, are common across temperate ecosystems (Batzer and Boix 2016). A pond is defined as a water body between $1m^2$ and 2ha in area which might be permanent or seasonal (dries out naturally at some stage during the year), natural or man-made (Biggs et al. 2005). Ponds are recognized as a threatened and ecologically distinct landscape feature (Oertli et al. 2005), harbouring more uncommon species than other waterbodies (Biggs et al. 2005).

There are few studies comparing macroinvertebrate biodiversity across waterbody types particularly those involving temporary sites, such as ponds, which seems an oversight in the face of increasing drought under climate change scenarios (Death et al. 2016, Roberston et al. 2016). The proportion of pond-related publications remains below 10% when compared with other water bodies (Oertli 2009). Dense pondscapes coupled with high costs for surveys and problems of land access (Boothby 1997) contribute to a generalized lack of monitoring of potentially highly sensitive ecosystems (Oertli et al. 2005). Williams et al. (2003) compared biodiversity values of rivers, streams, ponds and ditches of the lowland British country side and found that ponds contribute the most to biodiversity at a regional scale. Hill and Wood (2014) studied the macroinvertebrate biodiversity and conservation values of garden and field ponds. Their results show that garden ponds usually represent a subset of macroinvertebrate taxa but also contain unique taxa. Hill et al. (2016) compared macroinvertebrate diversity between perennial and ephemeral ponds and found that although perennial ponds are more diverse, ephemeral ponds support distinct invertebrate communities.

Although freshwater is considered one of New Zealand's most valuable assets, the rapid decline of biodiversity in rivers, lakes and wetlands in the country is higher than in many other countries (Joy and Death 2014, Weeks et al. 2016).There are 638 endemic aquatic invertebrate species in the country and there is still no coordinated monitoring plan (Elston et al. 2015). Of these species, only a small portion are present in wetlands. With only 10% remaining, wetlands are among the most threatened ecosystems in New Zealand (Ausseil et al. 2008) . Ō Tū Wharekai (Ashburton lakes, Canterbury) is an example of a montane wetland system with high conservation value and is part of the Arawai Kākāriki wetland restoration programme which aims to restore three of New Zealand's most important wetlands. The area is considered one of the best remaining high-country freshwater wetlands but is under threat from run-off from agricultural intensification, water abstraction and the expansion of invasive plants (Sullivan et al. 2012). This study examines the macroinvertebrate biodiversity values of 4 lakes, 3 semi-permanent ponds, and 7 temporary ponds in the Ō Tū Wharekai (Ashburton lakes) wetland system.

Study Area

The Ō Tū Wharekai wetland complex includes the Ashburton Lakes and the upper Rangitata River in the highlands of Canterbury in New Zealand. It is formed by wetlands, lakes, braided rivers, ephemeral ponds and interconnecting streams and is surrounded by sub-alpine mountain ranges. There are a number of lakes and kettle holes (bowl shaped depressions formed by a retreating glaciar (Johnson and Gerbeaux 2004)) that vary in size and support a high diversity of wetland plants. Vegetation is dominated by *Schoenus pauciflorus* and *Carex secta,* but includes red tussock, sphagnum and a number of threatened turf and plant species such as the native lily *(Iphigenia novae-zelandie).* Invasive plant species include broom, Russell lupin, grey and dark willows. The wetlands are also home to a wide range of birds including the largest population of the endemic wrybill (*Anarhynchus frontalis)* and threatened species of fish such as the upland longjaw galaxias (*Galaxias prognathous*) and the longfin eel (*Anguilla dieffenbachii*) (Sullivan et al. 2012). The area is a mixture of conservation land managed by the Department of Conservation, unallocated crown land and pastoral leases. Farming practices have changed from low intensity merino sheep farming to high intensity sheep, cattle and deer farming (Sullivan et al. 2012). This study includes lakes, semi-permanent ponds and temporary ponds (Fig. 1)

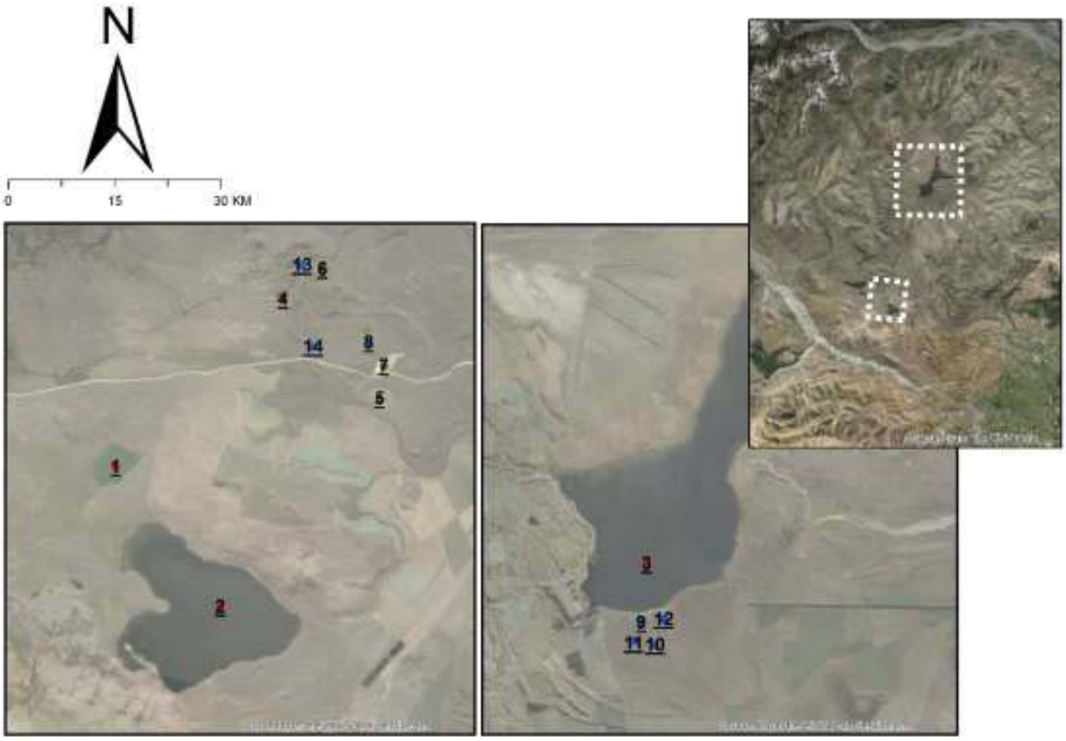

Figure 1: Location of sampling sites from the Ashburton Lakes in Canterbury, New Zealand. Permanent lakes are shown in red (1-4), semi-permanent ponds green (4-7) and temporary ponds in blue (8-14).

Methods

Habitat characteristics such as water permanence, area, and depth were recorded at the Ashburton Lakes in September 2016. Physio-chemical characteristics of water such as temperature, pH and conductivity were also recorded on site with an Oaktron Conductivity and pH metre.

Macroinvertebrates were collected by sweeping with a D-net (frame size: $0.09m^2$, mesh: 250µm) for 3 minutes at each water body (Biggs 1998). The 3 minutes where divided into three 1-minute sampling efforts covering all mesohabitats present (Fig. 2). Samples were preserved in 70% alcohol. Samples were sorted and taxa enumerated using the key by Winterbourn et al. (2006). Abundant taxa were subsampled by counting the individuals found in one randomly chosen square of a 16 square grid. Macroinvertebrate taxa were identified to species level where possible except for Oligochaeta, Diptera larvae and small crustaceans which were identified to family or genus level.

Species richness and species diversity (calculated with the Simpson's Index (Simpson 1949)) results were compared with a one-way analysis of variance (ANOVA) using R (R-Core-Team, 2015). Analysis of Similarity (ANOSIM) was used to examine if hydroperiod (permanence) affects pond and lake macroinvertebrate community composition using Primer 7.01 (Clarke and Warwick 2001). Differences in species composition between sites were also examined using the Nonmetric Multidimensional Scaling (NMDS) ordination technique and a Bray-Curtis dissimilarity matrix on log (x+1) transformed data also using Primer. Similarity percentage analysis (SIMPER) was used to determine the taxa contributing to differences between hydroperiod categories.

Figure 2: Sampling sites in the Ashburton Lakes wetland complex: temporary ponds on the left, semi-permanent ponds in the centre, permanent lakes on the right.

Results

A total of 40 taxa from 11 orders were collected in the Ashburton Lakes wetland complex. Temporary ponds are slightly more diverse, although ANOVA indicated no significant differences (Fig. 3). The richest site with 22 taxa was the semi-permanent pond Tiny Spider. ANOSIM indicated there was a significant difference in macroinvertebrate composition between the three pond/lake types ($R = 0.402$, $p = 0.006$). This difference was due to lakes being different from the semi-permanent and temporary sites (Fig. 4). Semi-permanent and temporary sites host more species and larger numbers of small crustaceans such as cladocerans and ostracods, while species belonging to the Trichoptera, Odonata and Hirudinea were absent from the pond sites (Fig. 5).

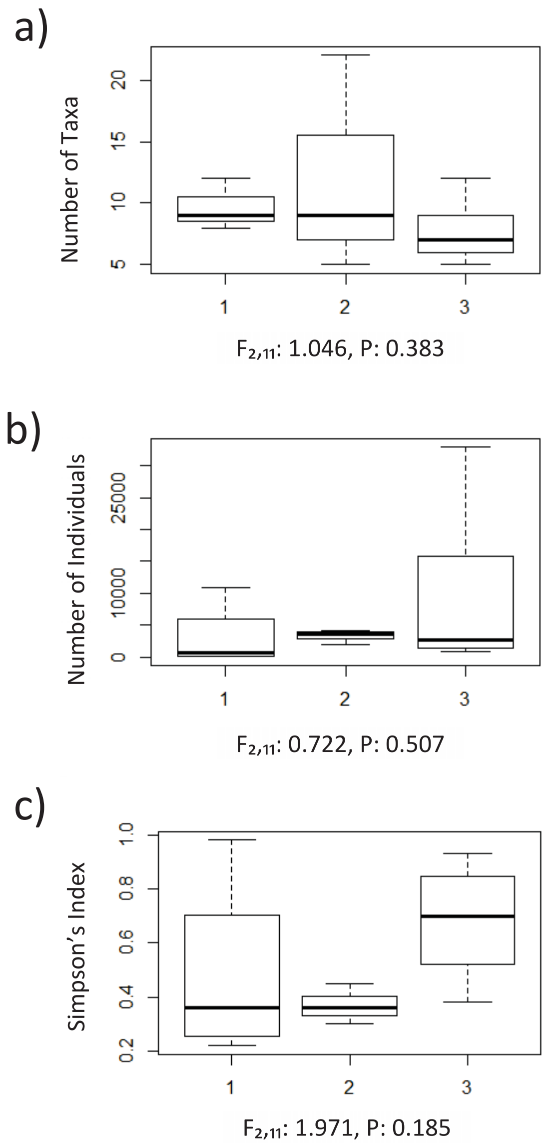

Figure 3: a) Total number of taxa, b) Total number of individuals, c) Simpsons Index for sweep net collections of invertebrates collected in September 2016 at 14 Ashburton ponds and lakes differing in hydroperiod permanence (1= Permanent, 2= Semi Permanent, 3= Temporary). The mid line represents the median, the box 50% of the data and the whiskers 95%.

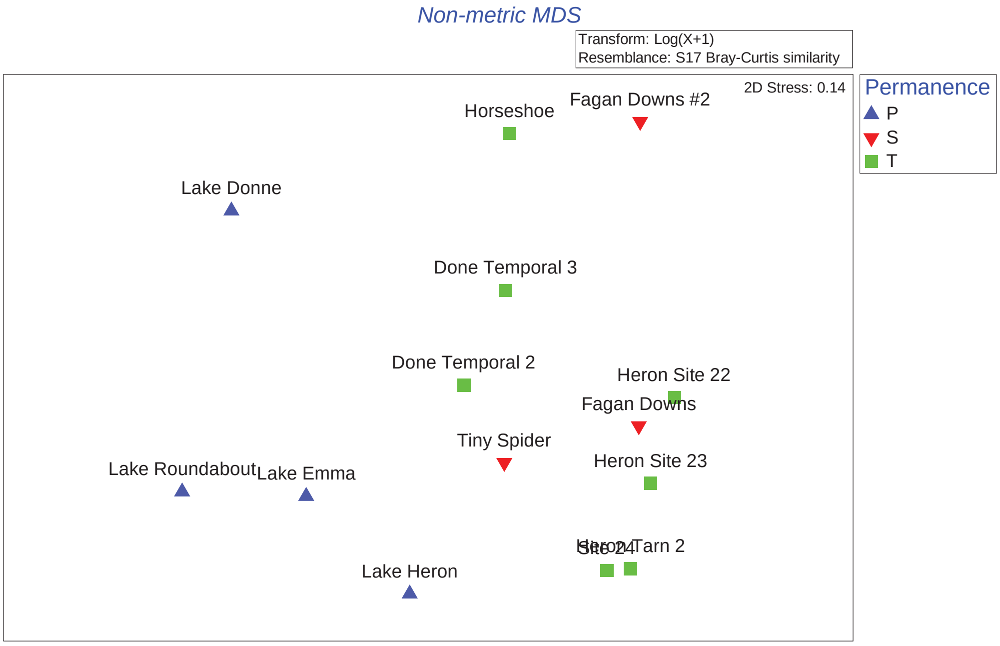

Figure 4: Ordination of sweep net samples of invertebrates collected in September 2016 at 14 Ashburton ponds and lakes differing in hydroperiod permanence (P: Permanent Lakes, S: Semi-temporary ponds, T: Temporary ponds).

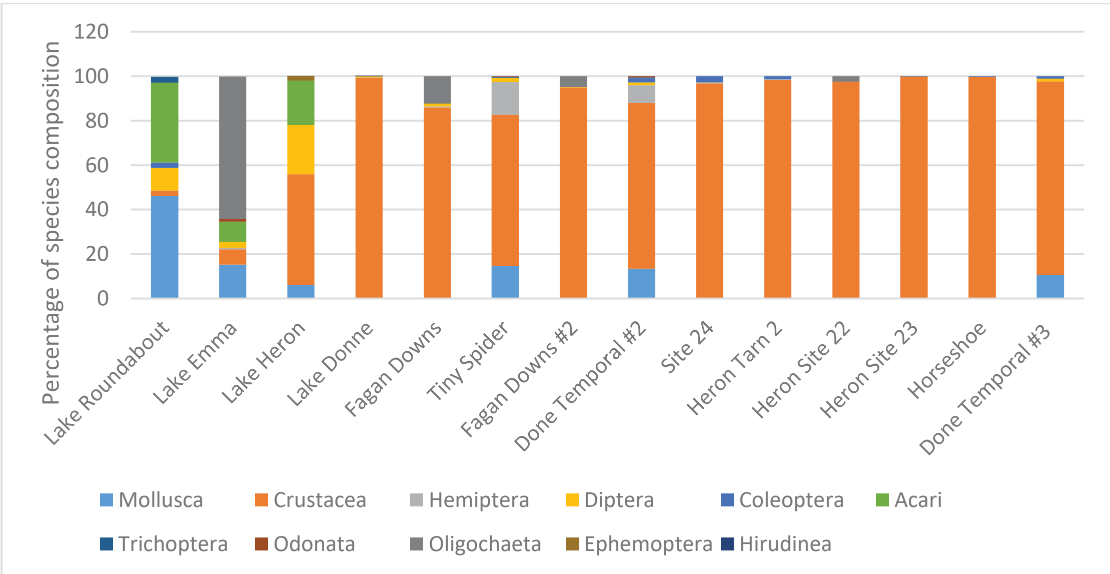

Figure 5: Faunal composition of permanent, semi-permanent and temporary water bodies based on sweep net samples of invertebrates collected in September 2016 within the Ashburton wetland complex.

Discussion

Hydroperiod (hydroregime/permanence) has long been recognized as an important influence on pond invertebrate communities (Batzer and Wissinger 1996, Batzer and Boix 2016). There were small differences in the diversity of invertebrates with ponds (both semi-permanent and temporary) having more taxa than lakes in the Ashburton Lakes complex (Tarr et al. 2005, Cristina Stenert 2007, Greig 2008, Seminara et al. 2015, Hill et al. 2016). However, as size is strongly linked with hydroperiod (smaller ponds dry quicker than larger ponds and lakes) it is hard to conclusively link the observed patterns with hydroperiod rather than pond/lake area. Tarr et al. (2005) report that invertebrate genera richness and abundance increases linearly along the hydrological gradient: sites that are inundated for a longer amount of time are more diverse. Nevertheless, Batzer and Ruhi (2013) analyzed taxa lists from 447 wetlands around the world and concluded that it would be unwise to generalize about hydrology effects since climate and geography are also important.

Williams et al. (2003) in a comparative study of rivers, streams, ditches and ponds concluded that ponds can contribute significantly to regional biodiversity but there is considerable variation in species richness between ponds. In the Ashburton wetlands, the highest mean biodiversity value was in the temporary ponds. Furthermore, semi-permanent and temporary ponds had a distinct macroinvertebrate community composition to permanent lakes. Both the highest and poorest species richness sites were not permanent ponds. This contrasts with work in Leicestershire, UK, where perennial ponds supported nearly twice the macroinvertebrate taxa of ephemeral ponds (Hill et al. 2016) and in New Hampshire, USA, where pond invertebrate diversity also increased with hydroperiod length (Tarr et al. 2005). Hydroregime and habitat size are closely related and have been shown to have unique but also shared effects on shaping community structure and diversity (Vanschoenwinkel et al. 2009). As highlighted by Batzer (2013) there is a variety of factors that contribute to diversity and composition of pond invertebrate communities including area, hydrology, land use and climate.

In the Ashburton Lakes complex, semi-permanent and temporary sites host more species of small crustaceans such as Cladocera and Ostracoda, while species belonging to the Trichoptera, Odonata and Hirudinea orders occurred only in the lakes. The fact that there is a difference between the macroinvertebrate communities of lakes and non-permanent ponds reflects the difference in aquatic invertebrate life history strategies. In seasonal wetlands, invertebrates recur either from drought resistance strategies or adult migration and oviposition (Batzer and Wissinger 1996). In the same study area, Greig (2008) found that community composition and species richness was influenced by pond permanence but that species in temporary ponds were a subset of generalists also found in permanent sites. However, crustaceans comprised 40% of the species limited to temporary ponds. Galatowitsch (2014) studied the life cycles of the generalist species in those wetlands *Xanthocnemis zealandica* and *Sigara arguta* and found that both species have alternating life history strategies to cope with the change in water; either quick development, dispersion or desiccation tolerance. Hill et al. (2016) identified several gastropod taxa and juvenile stages of Dysticidae and Corixidae as indicator taxa of perennial ponds, while Seminara et al. (2015) reported that microcrustacean diversity of small temporary water bodies is higher than those with permanent flooding. Seminara et al. (2015) believed that hydroperiod was the main driving force characterizing microcrustacean assemblages in their study ponds; with cladoceran and copepod groups discriminating between ponds with different wet phase durations (some species were exclusive to ponds with a short hydroperiod while others only occurred at permanent sites). The difference in macroinvertebrate community composition in the Ashburton lakes with microcrustaceans in temporary and semi-permanent ponds and insects in lakes supports the same view. Batzer and Ruhi (2013) conclude that there is a core of invertebrate taxa existing in wetlands, and that the absence of certain groups indicates some unique environmental characteristics.

From a conservation perspective, managing small and seasonal water bodies is an essential although often overlooked strategy for maximizing the preservation of overall maximum diversity. This is even more so the case as seasonal water bodies are usually perceived by developers and agriculturalists as irrelevant to the preservation of waterway biodiversity; resulting in draining and/or development of these wetlands. Furthermore, changes in precipitation from climate change are going to alter the distribution of temporary ponds; today´s permanent ponds will become tomorrow´s temporary ponds. Previous studies (Williams et al. 2003, Tarr et al. 2005, Hill and Wood 2014, Seminara et al. 2015, Hill et al. 2016) agree that conserving a wide array of ponds with distinct environmental characteristics provides protection for the biggest range of invertebrate taxa. This fits with modern ideas in ecology of metacommunity theory determining species sorting (Leibold et al. 2004). This perspective considers that species and populations vary across a gradient of abiotic factors. This relationship causes complex population dynamics that are often cyclical (Batzer and Wissinger 1996, Leibold et al. 2004). Although there is an agreement on the inherent conservation value of seasonal ponds, corresponding protection and legislation is generally lacking or non-existent. Investigating the biodiversity values of small and seasonal water bodies provides a useful reference point for conservation efforts.

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Chapter 3:

How good are macroinvertebrates for assessing wetland condition?

Lake Kohangatera: a lacustrine wetland in the southern North Island.

Abstract

To assess the potential for macroinvertebrate communities to be used to assess wetland condition, 14 freshwater wetlands were sampled in the lower North Island of New Zealand. The wetlands represent a gradient of condition in freshwater habitats, and include urban lagoons, agricultural swamps and protected lacustrine wetlands. A total of 63 invertebrate taxa were identified, of which crustaceans were the most abundant. There appeared to be no link between the composition or diversity of macroinvertebrate assemblages and wetland condition. Macrophyte communities were however, more reflective of wetland condition. Of habitat characteristics measured at each wetland, nutrient enrichment appeared to be most strongly linked with macroinvertebrate community structure in these wetlands. In contrast to lotic systems, there is limited information on how lentic invertebrates respond to environmental change in freshwater wetlands. This limits their suitability as a biomonitoring tool for assessing wetland condition.

Introduction

Freshwater ecosystems are among the most threatened in the world; they are inherently and increasingly rare patches of habitat in the terrestrial landscape (Strayer 2006). New Zealand has abundant freshwater resources, but much of that freshwater is being lost and/or degraded as a result of river diversions, water abstraction, pollution (particularly diffuse agricultural pollution), overharvesting, invasive species and climate change (Elston et al. 2015, Weeks et al. 2016). Nowhere is the loss more obvious than in wetlands. New Zealand has lost 90% of its wetlands (Ausseil et al. 2008) and most of those remaining are under 10ha in size and of unknown ecological health (Myers et al. 2013).

Invertebrates represent the bulk of the wetland fauna, reaching densities of $10^6/m^2$ (Strayer 2006). They are the principal link between primary production and larger organisms (Hornung and Foote 2006). Furthermore, they have important roles in regulating decomposition, water clarity, thermal stratification and nutrient cycling (Strayer 2006). Originally it was their importance as waterfowl food that provoked interest in their research (Batzer and Wissinger 1996) but their ubiquitous presence, high species richness and range of environmental responses makes these organisms suitable for study (Bonada et al. 2006). Thus, invertebrates are the most widely used biomonitoring tool for freshwater management, an activity that has a worldwide annual budget of US$ 100 billion (Bonada et al. 2006). In spite of this, the study of wetland invertebrates is far from extensive. Batzer and Ruhi (2013) reviewed taxa lists for 447 wetlands worldwide and found that most wetlands share a number of widespread taxa but that their occurrence is unpredictable. Mostly generalist species make up wetland invertebrate communities but the way these respond to environmental variables remains largely unclear (Batzer 2013).

New Zealand is a signatory to the Ramsar Convention on Wetlands and in legislation the protection of wetlands is considered a matter of national importance, yet protection remains inadequate, overlooking smaller or more degraded sites (Myers et al. 2013). Furthermore, there are 638 known endemic invertebrate species in New Zealand (Elston et al. 2015) but there is still no coordinated monitoring plan (Weeks et al. 2016).

This study examines the macroinvertebrate communities of 14 lacustrine and palustrine wetlands in the lower North Island of New Zealand that differ in their ecological condition. The study also assesses whether the invertebrate communities in a wetland reflect the ecological condition of that wetland. The sampling sites represent a range of freshwater wetlands, from urban lagoons to ecologically significant lake marshes.

Study Area

Fourteen freshwater wetlands in the southern half of the North Island were sampled in this study (Fig. 1). All of them are located in lowland areas but differ markedly in the surrounding land use and consequently ecological condition. Sampling sites could be grouped into categories of land use (urban, bush and pasture), wetland type (lacustrine and palustrine), origin (natural or man-made) or management regime (private, protected). The result is a gradient of freshwater wetlands; ranging from small, urban, fabricated lagoons, to large, lake associated wetlands with protected ecological value. This gradient reflects wetland loss in New Zealand, with well conserved and severely degraded sites, but also with relatively new potential freshwater refuges.

Figure 1. Map of the sampled wetlands in the North Island.

Zealandia Upper Dam

Zealandia Lower Dam

Lake Kohangatera

Lake Kohangapiripiri

Nga Manu Top Pond

Nga Manu Main Pond

Boggy Pond

Lake Waitawa

Omahu Farm

Pharazyn Reserve

Te Hapua

Queen Elizabeth Park

Fig. 2 Photographs of the 14 freshwater wetlands sampled in this study.

Methods

Macroinvertebrates were sampled and associated habitat characteristics recorded at 14 wetlands in the lower North Island of New Zealand between September 2016 and May 2017 (Fig. 2). Because wetlands vary greatly in size and accessibility, an equal intensity sampling protocol was applied (Biggs 1998). Mesohabitat composition, including substrate type, was determined visually by a perimeter walk of the wetland. Physio-chemical characteristics including water temperature and conductivity were recorded on site with an Oaktron conductivity metre. Further environmental characteristics including area, proportion of surrounding natural and impervious cover, nitrate leaching risk, introduced fish, presence of gorse and willows, and proportion of the wetland under protection were obtained from the FENZ Geodatabase (Leathwick et al. 2010). A measure of wetland condition (Ausseil et al. 2008) based on the naturalness of the catchment cover, the proportion of artificial impervious cover, nutrient enrichment, introduced fish, woody weeds and drainage was also extracted from this database and used in the analysis as an assessment of ecological condition.

In order to obtain the most representative sample for wetlands, macroinvertebrates were collected by sweeping (Cheal et al. 1993) with a D-net (frame size: $0.09m^2$, mesh: 250µm) for 3 minutes at each wetland (Biggs 1998). The 3 minutes where divided into three 1-minute sampling efforts covering represented mesohabitats. Where the substrate was stone or gravel, it was lightly disturbed before sweeping. Samples were preserved in 70% alcohol. Samples were sorted and species enumerated using the key of Winterbourn et al. (2006). Abundant taxa were subsampled by counting the individuals found in one randomly chosen square of a 16 square grid. Macroinvertebrate taxa were identified to species level where possible. Macrophytes were photographed on site and recorded in a presence/absence matrix using the Macrophyte ID guides by NIWA (Champion and Reeves).

Species richness and species diversity (Simpson 1949) was calculated for all sites. Sites were placed into three categories of land use surrounding the wetland: bush, pasture and urban, and diversity results compared using a one-way analysis of variance (ANOVA) using R (R-Core-Team 2015). Differences in species composition between sites were examined using the Vegan package to perform the Nonmetric Multidimensional Scaling (NMDS) ordination technique with a Bray-Curtis dissimilarity matrix (Oksanen et al. 2017) on log (x+1) transformed data and on a presence/absence matrix. Similarities between the macrophyte composition of the wetland sites was examined in the same way.

The relationship between macroinvertebrate diversity, wetland condition and the aforementioned environmental variables obtained from the FENZ Geodatabase (Leathwick et al. 2010) was analysed through a multivariate plot also on R (R-Core-Team 2015). A regression tree model (Ripley 2016) was used to establish the most important variable in determining species richness and species diversity for freshwater wetlands in the lower North Island. Individual linear regressions were carried out to establish whether these relationships are statistically significant and to further explore the specific nature of the relationship between other variables such as area/condition and area/proportion protected.

Results

A total of 63 taxa were collected from the 14 wetlands (Table 1). Invertebrate abundance was dominated by Crustacea (74%), followed by Diptera (7.2%), Mollusca (4%) and Hemiptera (4%). There were no significant differences in any aspect of macroinvertebrate diversity between sites based on land-use categories (Fig. 3). Both high and low diversity sites were found in each category (Fig. 4). Ordination similarly did not reveal any strong differences in the composition of macroinvertebrate assemblages based on land use surrounding the freshwater wetlands (Fig. 5). However, land use is better reflected when wetland sites are grouped based on their macrophyte community (Fig. 6). In this figure, group A includes some of the most biodiverse sites: wetlands 6, 10 and 11 are located within reserves and number 14 is an urban lake. The Whitby lakes and Aotea Lagoon, form group B; both are urban sites without a macrophyte community. Group C encompasses all other wetlands and includes protected and private sites. These sites (except number 7) are located in highly modified lowland agricultural areas. A regression tree indicated taxa richness was predominantly determined by the area of the wetland (Fig. 7), but the classical species-area relationship did not occur ($F_{1,12}$=1.19, P=0.34). There was no relationship between the number of species in a wetland and its condition score (Fig. 8). Another regression tree model of Simpson's diversity indicated that the wetland's risk of nitrate leaching was the strongest predictor (Fig. 7). However, there was no significant linear relationship (Fig. 9) ($F_{1,12}$=1.40, P=0.25).

Table 1: Number of macroinvertebrate taxa, total number of individuals, Rarefied species richnesss, Simpson's Index, land use in the immediate surroundings, presence of exotic and invasive macrophytes for 14 North Island wetlands sampled between September 2016 and June 2017.

#	Site	Total Number of Species	Total number of individuals	Rarefied Species Richness	Simpson Index (1-D)	Buffer Land Use	Presence of Exotic Macrophytes	Presence of Invasive Macrophytes
1	Boggy Pond	8	115	13.56	0.676	Pasture	x	
2	Nga Manu main pond	3	136	21.98	0.044	Bush	x	
3	Aotea Lagoon	4	913	28.25	0.305	Urban		
4	Pharazyn Reserve	11	1536	33.14	0.689	Urban	x	
5	Whitby Lakes	12	852	37.19	0.569	Urban	x	
6	Zealandia Upper Dam	19	345	40.73	0.736	Bush		
7	Zealandia Lower Dam	26	684	43.93	0.624	Bush		
8	Lake Waitawa	19	553	46.90	0.572	Pasture	x	x
9	Omahu Farm	22	1290	49.69	0.494	Pasture	x	
10	Lake Kohangatera	9	249	52.35	0.392	Bush	x	x
11	Lake Kohangapiripiri	15	917	54.90	0.264	Bush		
12	Te Hapua	5	110	57.35	0.572	Pasture	x	
13	Nga Manu Top Pond	11	667	59.71	0.314	Bush	x	
14	Queen Elizabeth Park Lake	24	633	62	0.737	Urban	x	

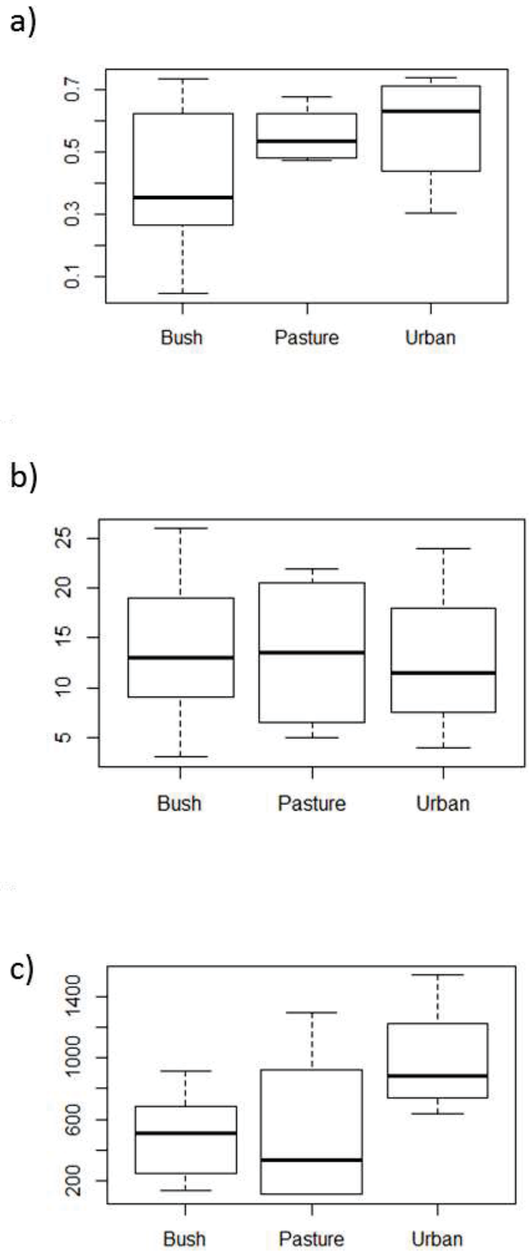

Figure 3. Invertebrate diversity collected in 14 wetlands in the lower North Island differing in surrounding land use. a): Simpson's Index ($F_{2,11}$=1.19, P=0.34) , b): Total number of species ($F_{2,11}$=0.02, P=0.97), c): Total number of Individuals ($F_{2,11}$=1.95, P=0.18). The mid line represents the median, the box 50% of the data and the whiskers 95%.

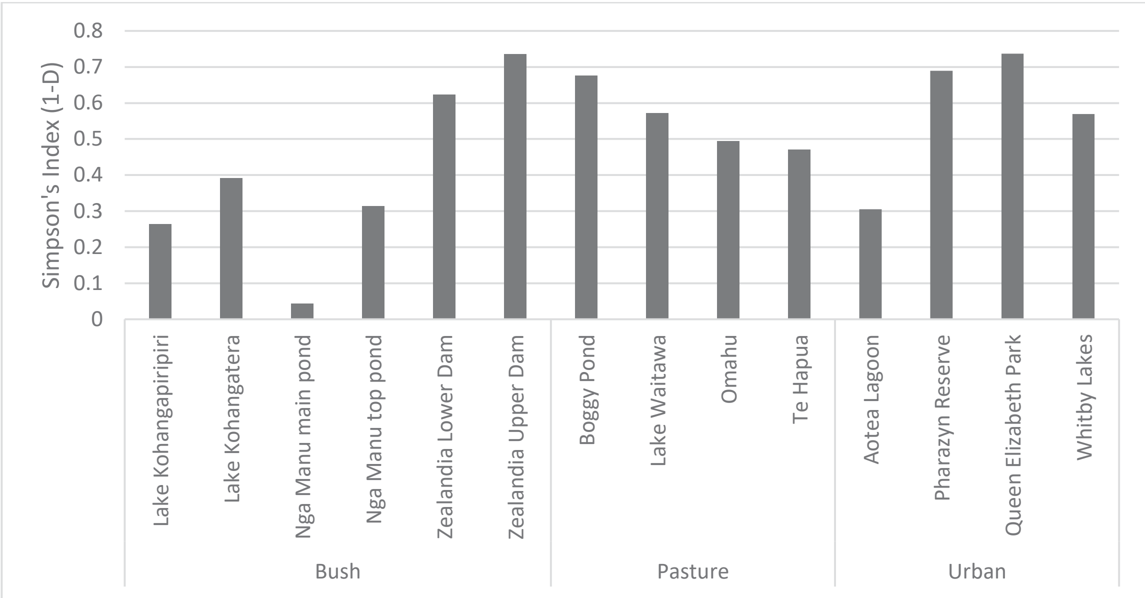

Figure 4. Simpson's Index for each of 14 wetlands sampled between September 2016 and June 2017 divided into three land use categories: Bush, Pasture and Urban.

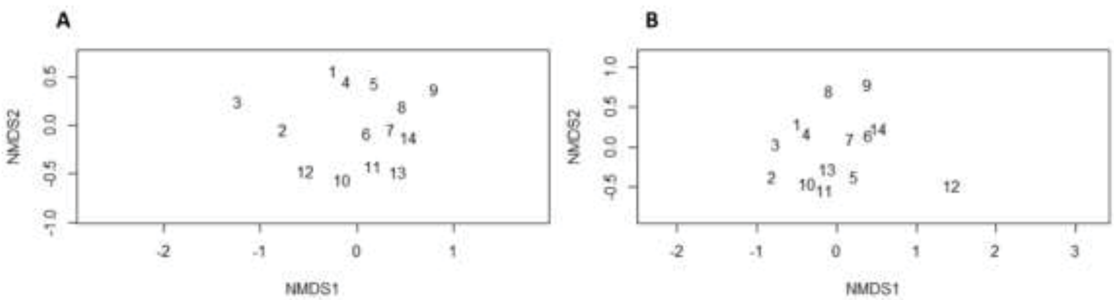

Figure 5. NMDS plot of invertebrate communities collected in 14 wetlands sampled between September 2016 and June 2017. A) species presence/absence data B) log (x+1) transformed data.

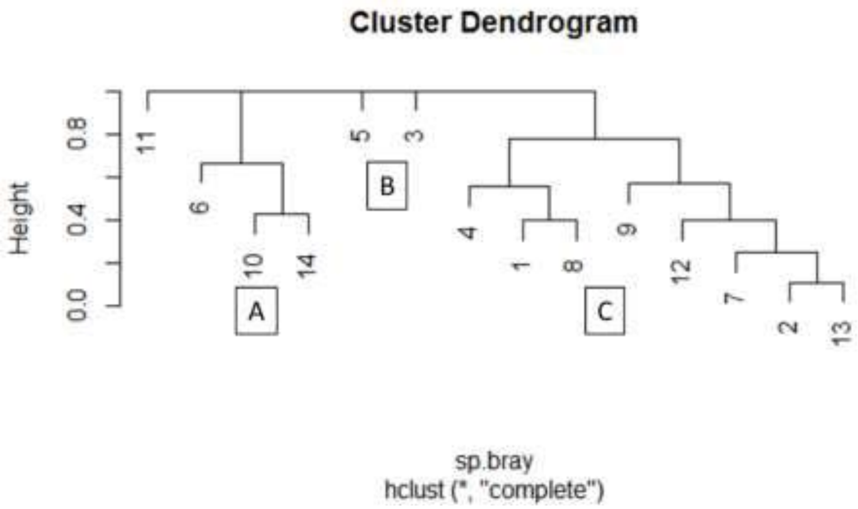

Figure 6. Cluster dendogram showing the similarities between the macrophyte communities collected in 14 wetlands sampled between September 2016 and June 2017.

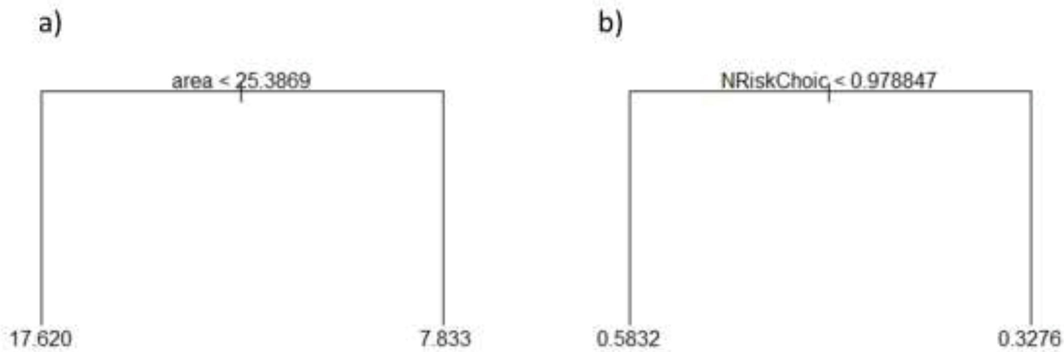

Figure 7: Regression trees indicate that the number of invertebrate species is largely determined by area (a) and that the strongest predictor of species diversity is the risk of nitrate leaching (b). The threshold value for each variable is shown on top and mean low and high values at each of the fork's sides.

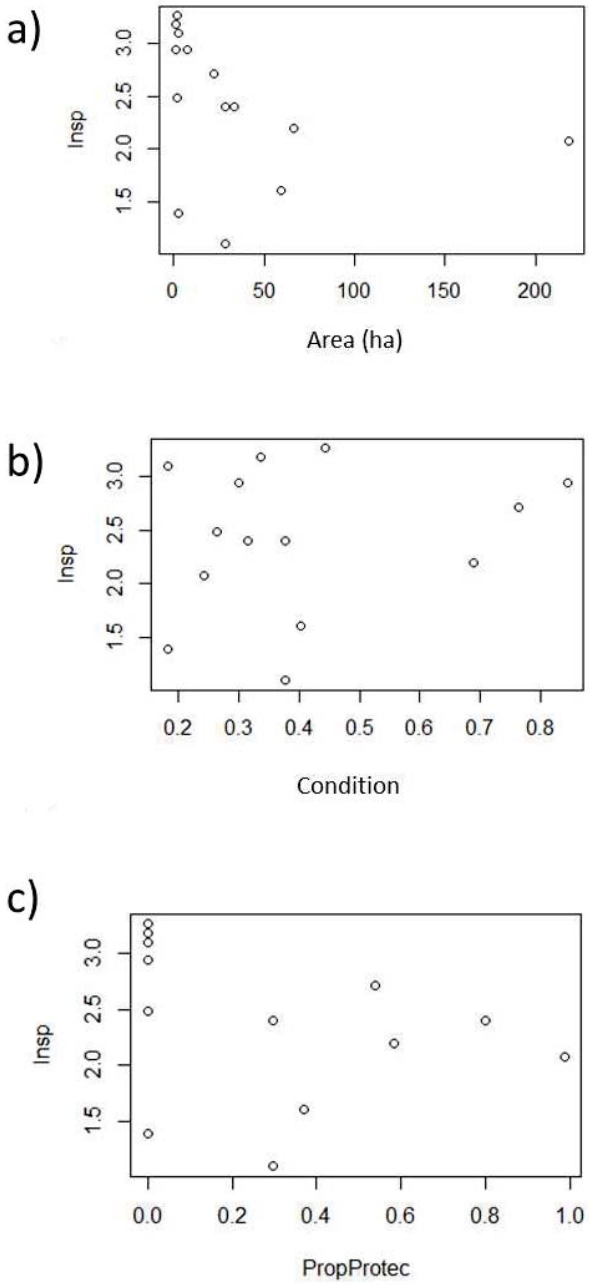

Figure 8. Number of taxa (Log(x+1)) plotted against (a) area ($F_{1,12}$:1.33, P:0.27), (b) wetland condition ($F_{1,12}$:0.48, P:0.50) and (c) proportion of the wetland protected ($F_{1,12}$:1.50, P:0.24).

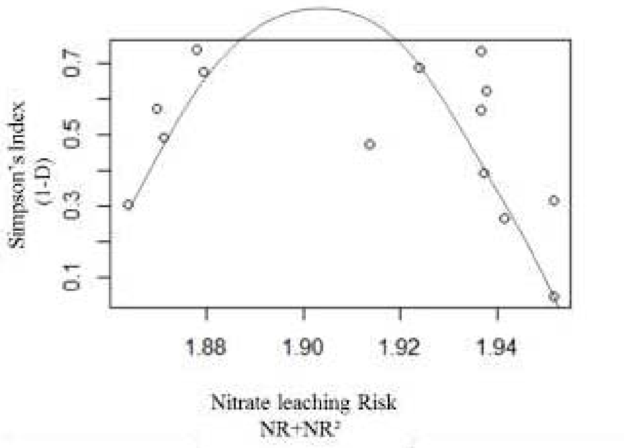

Figure 9. Relationship between the Nitrate leaching Risk and wetland macroinvertebrate diversity calculated through the Simpson's Index ($F_{1,12}$=1.40,P=.025).

Discussion

Macroinvertebrate communities collected in these wetlands did not reflect the condition of the wetland as measured by Ausseil et al. (2008), however, macrophyte composition does seem to reflect ecological condition. The presence of native and exotic macrophytes has been used in New Zealand previously to assess ecological condition of lakes (Clayton and Edwards 2006). Aquatic plants often constitute the basis of food webs for wetland ecosystems and provide habitat (Batzer and Wissinger 1996). It seems that the most productive wetlands are those in which open spaces of water mix with patches of emergent macrophytes (Voigt 1976). Invertebrate feeding groups have been associated with aquatic plant architecture (Howard and Harley 1998) and wetland birds are associated with invertebrate abundance (Voigt 1976). On the other hand, macrophytes can also have negative effects on biodiversity. Floating aquatic weeds can form dense mats that prevent sunlight from penetrating the water, which limits photosynthesis and results in decreased dissolved oxygen with consequent effects on the fauna (Howard and Harley 1998). In this study, three of the less diverse sites (Nga Manu main and top ponds and Te Hapua) were covered in a mat of *Azolla pinnatta,* an exotic free-floating waterweed (Fig. 9).

Fig 9. Nga Manu Top Pond on the left and Te Hapua on the right. A dense mat of *Azolla pinnatta* covers these wetlands.

What environmental variables drive invertebrate assemblages in wetlands remains unclear. A review of wetland invertebrate research by Batzer (2013) reports that a lack of response to obvious environmental drivers like land use is common (Tangen et al. 2003, Batzer et al. 2004, Scheffer et al. 2006). Furthermore, Batzer and Ruhi (2013) reviewed taxa lists for wetland invertebrates from around the world and although a core set of 40 widespread generalist species exists, patterns were in general idiosyncratic. In New Zealand, freshwater wetland invertebrates are only beginning to be studied. Suren and Sorrell (2010) in a large scale survey of invertebrate biodiversity in lowland wetlands of New Zealand found a total of 133 taxa but did not find any major environmental drivers; concluding that invertebrate communities are regulated by a combination of variables acting together. In this study, no significant relationships between environmental variables and invertebrate biodiversity were found. Suren et al. (2011) found a similar result: a lack of association between two landscape-based indices of wetland condition (including the Ausseil et al. (2008) index used in this study) on the invertebrate community. This frustrating lack of a relationship between invertebrates and wetland condition indicates they are highly resilient to environmental change or conversely, that they are highly sensitive and their responses hard to trace, causing a cascade of responses (Batzer 2013).

In this study, wetlands placed into different categories of land use (bush, pasture, urban) did not differ significantly in biodiversity. Each land use type has rich and impoverished sites. Studies comparing biodiversity contributions of different wetland types are few. Thornhill et al. (2016) studied ponds across an urban land use gradient and found that invertebrate assemblages reflected local factors such as macrophyte structure, nutrient concentration and surrounding urban land. Williams et al. (2003) and Hill and Wood (2014) concluded that small water bodies such as agricultural and urban ponds can contribute significantly to regional biodiversity and have the potential to enhance invertebrate conservation. Biodiversity loss in New Zealand has been greatest in lowland areas; thus lowland environments are commonly represented within cities and less so within protected areas (Clarkson et al. 2007). Because little is known about the potential biodiversity value of anthropogenic water bodies (Chester and Robson 2013), wetland research and management must consider anti-urban bias (the notion that only non-urban ecosystems are worthy of conservation efforts) (Cavin 2013).

Part of the present gradient of wetlands in the lower North Island is formed by sites located in agricultural and pastoral surroundings. Agricultural intensification and diffuse pollution is currently considered to be the worst threat to these freshwater ecosystems in New Zealand (Elston et al. 2015, Weeks et al. 2016). During the last 40 years, agriculture in New Zealand has steadily intensified (McLeod and Moller 2006) resulting in escalating dissolved nitrogen levels and the decline in water quality of rivers (Julian et al. 2017) and lakes (Galbraith and Burns 2007). In this study, the main environmental driver behind macroinvertebrate diversity was the wetland's risk of nitrate leaching, a surrogate measurement for nutrient enrichment. Similarly, Gascón et al. (2009) found conductivity as the key factor influencing invertebrate diversity in Mediterranean wetlands and Suren et al. (2011) reported nutrient status had the strongest influence on invertebrate communities in wetlands of the western South Island in New Zealand. Nutrient enrichment affects wetland invertebrates indirectly (Batzer and Wissinger 1996) and favours the growth of aquatic weeds (Howard and Harley 1998).

Considerable evidence exists on nutrient enrichment provoking a subsidy-stress gradient response (Odum et al. 1979). A usable input of nutrients initially enhances biodiversity but it declines soon afterwards as the input increases, and the invertebrate community changes to one characteristic of eutrophy (Odum et al. 1979, Batzer 2013). The correlation between nitrate leaching risk and biodiversity in this study might reflect such a pattern (Fig. 8). Nevertheless, this effect is often complicated by the interaction of other variables (Odum et al. 1979). For example, Liston et al. (2008) found that invertebrate densities increased with nutrient enrichment only until periphyton mats were lost, after which they declined.

So far, it appears that invertebrates are of limited use in assessing wetland condition (Batzer 2013). Some examples exist such as the currently used macroinvertebrate IBI adapted for the Great Lakes (Uzarski et al. 2004). Another is the Community conservation Index developed by Richard Chadd (2004). Nevertheless this method is based on the rarity of the species and may not be suitable for wetland invertebrate fauna which often comprises common and generalist species (Batzer 2013, Batzer and Ruhi 2013). In biomonitoring, a solid knowledge of the unaltered control conditions is essential (Bonada et al. 2006). Thus, in wetland monitoring, finding appropriate reference sites could be a challenge since freshwater wetlands are very diverse but also increasingly rare and degraded ecosystems.

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Chapter 4:

How should wetland condition be assessed?

Invertebrate sampling, Lake Heron, 2016.

Introduction

Wetlands are among the most threatened ecosystems in the world (Mitsch and Gosselink 2015). In New Zealand, 90% of wetlands have been lost and of those left 60% are considered severely degraded (Ausseil et al. 2008). Biodiversity losses are caused by both a reduction in habitat area and a deterioration of condition (Brinson and Malvárez 2002). The rapid loss and degradation of wetlands has generated a need to accurately and efficiently monitor these changes (Nichols and Williams 2006). Finlayson (2003) differentiates between three aspects of wetland management: inventory (extent), assessment (status or condition), and monitoring (to track management results). Most countries lack accurate records of wetland area and condition (Brinson and Malvárez 2002) and this remains one of the greatest challenges for wetland conservation (RAMSAR 2015).

The first attempt at creating a wetland inventory for New Zealand (Cromarty and Scott 1995) was based on an expert panel approach but lacked detailed maps and biological data. More recently, Ausseil et al. (2008) mapped the country's wetlands through combining existing GIS databases. Two main methods for assessing the condition of wetlands have developed in New Zealand. Clarkson et al. (2004) developed a field-based method, which considers hydrological integrity, physicochemical parameters, ecosystem intactness, browsing, predation and harvesting regimes, and dominance of native plants. Ausseil et al. (2008) propose a GIS-based measure of wetland condition with a score between zero and one; based on the naturalness of the catchment cover, artificial impervious cover, nutrient enrichment, introduced fish, woody weeds and drainage. However, Suren et al. (2011) report that neither the field-based approach (Clarkson et al. 2004) nor the GIS-based approach (Ausseil et al. 2008) to wetland condition scoring was strongly associated with the invertebrate and diatom communities of 29 lowland wetlands in New Zealand. They concluded that these indices do not include variables that influence wetland biota.

Biotic information is thought to provide one of the best pieces of information for ecosystem assessment (Craft 2016b). Because invertebrates represent the trophic link between primary production and top predators and have important roles in regulating decomposition, water clarity, thermal stratification and nutrient cycling (Strayer 2006) they are considered the staple biomonitoring tool for many freshwater systems (Bonada et al. 2006). However, how wetland invertebrates respond to environmental change is not well established (Batzer 2013). Examples of their use as indicators of wetland condition are scattered and the suitability of these methods is still debated (Chessman et al. 2002, Richard Chadd 2004, Boix et al. 2005, Davis et al. 2006, Suren and Sorrell 2010). This study proposes a simple, dichotomous wetland scoring system that integrates the environmental characteristics, macroinvertebrate and macrophyte communities recorded in 14 freshwater wetlands of the lower North Island in New Zealand.

Methods

In order to exemplify the proposed scoring system, the environmental characteristics and biodiversity data collected from 14 palustrine and lacustrine wetlands in the southern tip of New Zealand's North Island were used. The sites include a variety of wetlands, from small urban lagoons to large lake marshes with recognized ecological value. Environmental characteristics were assessed in the field, and include whether the wetland is natural or artificial, the naturalness of the immediate buffer zone, and the presence of concrete edges. Habitat measurements used in Chapter 3 such as substrate composition, temperature and conductivity were not included because they were highly variable among wetlands and do not appear to reflect wetland condition. The biodiversity data includes: macroinvertebrate richness and Simpson's diversity and the presence/absence of exotic and invasive macrophytes.

For each characteristic, a score of 1 or 0 is assigned. For numerical values such as invertebrate diversity, a score of 1 is assigned if the value is greater than the median (middle value) of that particular data set, and a 0 is given if the value is lower than the median. For presence and absence data such as whether a wetland is natural or artificial, a 1 is given to the characteristic that is considered ideal; in this case a natural wetland receives a score of 1 while an artificial wetland equals 0. Section A in Table 1, shows the raw data, while section B shows the conversion of the values to a dichotomous (1/0) score. The example given is a simple one, however this method allows the integration of more environmental characteristics and biotic data. A total score for each wetland is obtained by summing all individual scores. Thus, the ideal reference site would have a score of 1 for each environmental and biotic characteristic. In this example, seven characteristics are evaluated; therefore, the highest possible score is 7. Lake Kohangapiripiri is the best rated wetland of this data set with a score of 6. This score can be converted to other scoring systems (for example: 0-1) with a rule of three (Table 2). This allows for a comparison (Fig. 1) with the GIS score (Ausseil et al. 2008) obtained from the FENZ database (Leathwick et al. 2010).

A

Site	Natural vs Artificial	Concrete edges?	Natural cover in buffer?	Macroinvertebrate species richness	Macroinvertebrate diversity (Simpson's Index)	Exotic Macrophytes?	Invasive Macrophytes?
Lake Waitawa	Natural	No	No	19	0.572	yes	Yes
Te Hapua	Natural	No	No	5	0.471	yes	No
Nga Manu main pond	Natural	Yes	No	3	0.044	yes	No
Nga Manu top pond	Natural	No	Yes	11	0.314	yes	No
Omahu	Natural	No	No	22	0.494	yes	No
Aotea Lagoon	Artificial	Yes	No	4	0.305	No	No
Whitby Lakes	Artificial	Yes	No	12	0.569	Yes	No
Boggy Pond	Natural	No	No	8	0.676	Yes	No
Zealandia Lower Dam	Artificial	Yes	Yes	26	0.624	No	No
Zealandia Upper Dam	Artificial	Yes	Yes	19	0.736	No	No
Lake Kohangatera	Natural	No	Yes	9	0.392	Yes	Yes
Lake Kohangapiripiri	Natural	No	Yes	15	0.264	No	No
Pharazyn Reserve	Natural	No	No	11	0.689	Yes	No
Queen Elizabeth Park	Artificial	Yes	No	24	0.737	Yes	No

B

Site	Natural vs Artificial	Concrete edges?	Natural cover in buffer?	Macroinvertebrate species richness	Macroinvertebrate diversity (Simpson's Index)	Exotic Macrophytes?	Invasive Macrophytes?
Lake Waitawa	1	1	0	1	1	0	0
Te Hapua	1	1	0	0	0	0	1
Nga Manu main pond	1	0	0	0	0	0	1
Nga Manu top pond	1	1	1	0	0	0	1
Omahu	1	1	0	1	0	0	1
Aotea Lagoon	0	0	0	0	0	1	1
Whitby Lakes	0	0	0	1	1	0	1
Boggy Pond	1	1	0	0	1	0	1
Zealandia Lower Dam	0	0	1	1	1	1	1
Zealandia Upper Dam	0	0	1	1	1	1	1
Lake Kohangatera	1	1	1	0	0	0	0
Lake Kohangapiripiri	1	1	1	1	0	1	1
Pharazyn Reserve	1	1	0	1	1	0	1
Queen Elizabeth Park	0	0	0	1	1	0	1

Table 1: A) Environmental characteristics and biodiversity measures for 14 freshwater wetlands in the lower North Island, B) Dichotomous score assigned to each characteristic

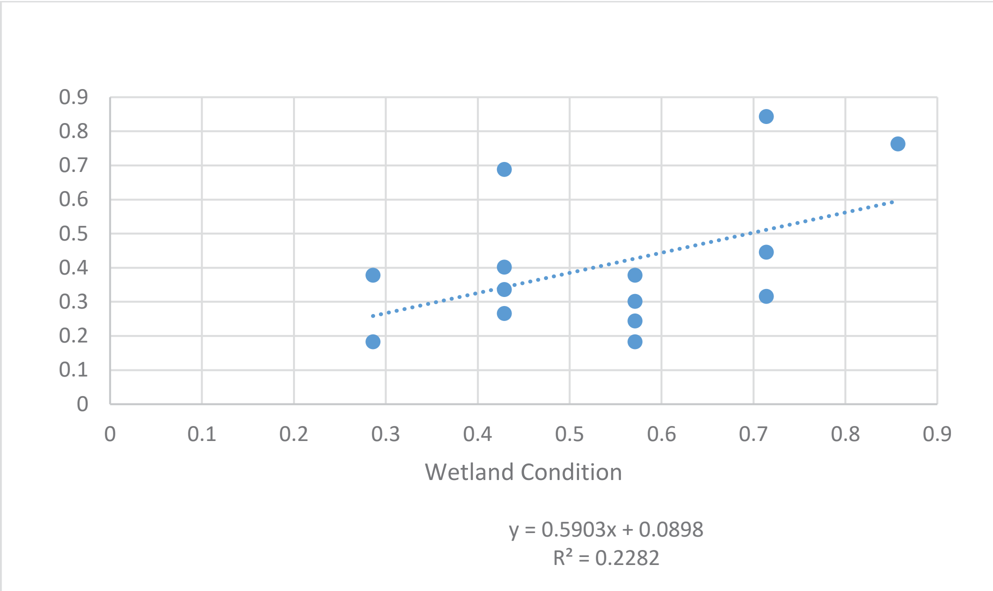

Figure 1: Comparison of wetland scores with the proposed method and the GIS based score for 14 North Island Wetlands

Table 2: Total score for wetland condition of the 14 wetlands, the same score converted to a 0-1 scale and the GIS based wetland condition score (Ausseil et al. 2008).

Site	Total Score	Score (0-1)	Score by Ausseil et al 2008 (0-1)
Lake Waitawa	4	0.571	0.301
Te Hapua	3	0.429	0.402
Nga Manu main pond	2	0.286	0.378
Nga Manu top pond	4	0.571	0.378
Omahu	4	0.571	0.183
Aotea Lagoon	2	0.286	0.183
Whitby Lakes	3	0.429	0.266
Boggy Pond	4	0.571	0.244
Zealandia Lower Dam	5	0.714	0.445
Zealandia Upper Dam	5	0.714	0.843
Lake Kohangatera	3	0.429	0.688
Lake Kohangapiripiri	6	0.857	0.763
Pharazyn Reserve	5	0.714	0.316
Queen Elizabeth Park	3	0.429	0.336

Discussion

Aquatic invertebrates are used successfully to monitor the ecological condition of many freshwater ecosystems, but not wetlands (Chapter 3). Although they have been studied extensively, the way they respond to environmental change remains unclear (Batzer 2013). Therefore, how best to incorporate invertebrates into wetland assessment is unclear.

Recently, wetland invertebrates have received more attention in New Zealand (Suren et al. 2008, Suren and Sorrell 2010, Suren et al. 2011, Galatowitsch 2014, O. Ball 2015). Suren and Sorrell (2010) published an inventory of invertebrate species of lowland wetlands in New Zealand; it was part of the project to establish tolerance values for wetland taxa and a WMCI (Wetland Macroinvertebrate Community Index). However, their results showed that invertebrate communities are different in fens and bogs, which means a different index would be necessary for each wetland type. Similar attempts exist for Australian (Chessman et al. 2002) and Mediterranean wetlands (Boix et al. 2005). In Australia, Davis et al. (2006) argue that it is necessary to tailor wetland bio assessment models for each region and it might be too costly to be implemented. Richard Chadd (2004) proposed a Community Conservation Index for inland flowing and still waters of Great Britain, based on the richness and rarity of the species present. This method allows the comparison of conservation value of different water body types. However, wetland invertebrate communities are mostly comprised of generalist species (Batzer and Ruhi 2013). This might be a severe limitation for the application of a biological method using invertebrates in the assessment of wetland condition. In the North Island of New Zealand for example, Suren and Sorrell (2010) recorded no unique wetland invertebrate species.

So far, there is no clear answer on how to integrate aquatic invertebrates into wetland assessment. Thus, other kinds of information need to be used. Macrophyte structure has been linked to wetland productivity (Voigt 1976) and is the main focus of the field-based method of wetland assessment proposed by Clarkson et al. (2004). They argue that macrophytes cover a large area of the wetland, are a permanent feature, and integrate environmental stress over long periods of time. There is certainly a trend to use macrophytes as environmental indicators of lake associated wetlands (Uzarski et al. 2004, Clayton and Edwards 2006). In, New Zealand, the other method for assessing wetland condition is GIS-based (Ausseil et al. 2008). Its greatest advantage for managers is that a wetland condition score is readily available through the FENZ database (Leathwick et al. 2010). On the other hand, the evaluation lacks a present-day biotic component.

Adaptive management entails making decisions while still learning and acquiring more information (Westgate et al. 2013). The method outlined in this study places an equal value to all information types. This allows the addition of as many variables as wanted; giving managers the possibility to integrate whatever information is available. However, this also means that no variable is considered more important than another when in reality this might not be true. It is most useful as a simple ranking tool for wetlands within a certain area, which allows to place a numerical value on what could be considered a pristine site.

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Appendix

Table 1: Community characteristics of pond invertebrates collected in the Ashburton Lakes, September 2016. Values are totals of the 3 sweep nets.

	SITE	Hydroperiod	Total Number of Species	Total number of individuals	Percentage of Total species richness (%)	Simpson Index (D)
1	Lake Roundabout	Permanent	9	39	22.5	0.22
2	Lake Emma	Permanent	12	176	30	0.43
3	Lake Heron	Permanent	8	1187	20	0.29
4	Lake Donne	Permanent	9	10804	22.5	0.98
5	Fagan Downs	Semi permanent	9	1958	22.5	0.36
6	Tiny Spider	Semi permanent	22	4092	55	0.30
7	Fagan Downs #2	Semi Permanent	5	3627	12.5	0.45
8	Done Temporal 2	Temporary	12	1718	30	0.38
9	Site 24	Temporary	7	1137	17.5	0.70
10	Heron Tarn 2	Temporary	5	2684	12.5	0.93
11	Heron Site 22	Temporary	10	32893	25	0.52
12	Heron Site 23	Temporary	7	29115	17.5	0.76
13	Horseshoe	Temporary	5	2586	12.5	0.93
14	Done Temporal 3	Temporary	8	872	20	0.52

Table 2: Raw macroinvertebrate species data for the Ashburton Lakes complex, 2016.

Sites:	Lake Round about	Lake Emma	Fagan Down s	Tiny Spider	Done Temp oral 2	Site 24	Heron Tarn 2	Heron Site 22	Heron Site 23	Lake Heron	Lake Donne	Fagan Down s #2	Horse shoe	Done Temp oral 3
Acari	14	16	0	1	0	0	0	0	0	237	0	0	0	0
Amphipoda	0	0	1088	0	0	0	0	0	0	0	0	0	0	0
Anisops wakefieldi	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Antiporus strigosulus	0	0	0	6	0	0	0	0	0	0	0	0	0	0
Chironomidae	0	1	0	0	0	1	0	0	0	100	31	5	0	12
Cladocera	1	0	208	416	992	0	0	171	11	0	0	1644	2578	124
Coleoptera (adult undetermined)	0	0	0	0	0	0	0	0	32	0	8	0	5	7
Copepoda 1	0	0	0	0	0	0	0	7728	0	0	0	0	0	0
Copepoda 2	0	0	320	2064	0	944	2592	22304	25072	561	0	0	0	0
Copepoda 3	0	0	0	0	0	0	0	0	0	0	10728	0	0	0
Copepoda 4	0	0	0	0	0	0	0	0	0	0	0	1802	0	0
Deleatidium (Ephemeroptera)	0	0	0	0	0	0	0	0	0	26	8	0	0	0
Diptera (pupae)	1	2	0	15	12	0	0	0	0	0	0	0	0	0
Dysticidae (adult undetermined)	0	0	0	0	0	0	0	11	1	0	0	0	0	0
Dysticidae (adult undetermined)	0	0	0	0	0	0	0	0	0	0	3	0	0	0
Dysticidae (larvae undetermined)	0	0	1	1	22	28	26	3	1	0	0	0	0	2
Gyraulus sp	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Hirudinea	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Lancetes lanceolatus	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Lepidurus	0	0	0	1	0	0	0	2	0	0	0	0	0	18
Liodessus deflectus	0	0	0	1	10	3	12	0	0	0	0	0	0	0
Lymnaea sp	0	0	0	0	0	0	0	0	0	0	0	0	0	83

Microvelia mcgregori	0	0	0	592	120	0	0	0	0	0	0	0	0	0
Muscidae (pupae)	0	0	0	7	0	0	0	0	0	0	11	0	0	0
Neolimnia	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Oligochaeta	0	113	240	0	0	0	0	224	0	0	0	16	0	0
Oligochaeta (branchiura)	0	0	0	0	0	0	0	544	0	0	0	160	0	0
Orthoclad (larvae)	0	0	0	0	0	0	0	0	0	164	0	0	0	0
Ostracoda	0	12	69	304	288	157	48	1901	3996	30	0	0	0	617
Paralimnophila skusei	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Physa sp.	7	12	0	120	211	0	0	0	0	27	9	0	1	9
Pisidium	1	7	1	2	0	0	0	0	0	0	0	0	0	0
Polypedilum	1	1	0	1	0	0	0	0	0	0	0	0	0	0
Potamopyrgus antipodarum	10	8	0	471	21	0	0	0	0	42	0	0	0	0
Ranthus (larvae)	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Sigara sp	0	1	7	9	16	3	6	5	1	0	5	0	2	0
Stratiomyidae	2	0	0	54	10	1	0	0	0	0	0	0	0	0
Tipulidae	0	0	24	0	0	0	0	0	0	0	0	0	0	0
Triplectides cephalotes	1	0	0	12	6	0	0	0	0	0	0	0	0	0
Xantochemis zealandica	0	2	0	11	10	0	0	0	0	0	0	0	0	0
Total number of individuals	39	176	1958	4092	1718	1137	2684	32893	29115	1187	10804	3627	2586	872

Table 3: Raw Macroinvertebrate data of 14 freshwater wetlands in the lower North Island, 2017.

Site	Boggy Pond	Nga Manu main pond	Aotea Lagoon	Pharazyn Reserve	Whitby Lakes	Zealandia Upper Dam	Zealandia Lower Dam	Lake Waitawa	Omahu	Lake Kohangatera	Lake Kohangapiripiri	Te Hapua	Nga Manu Top Pond	Queen Elizabeth Park
Acari	4	2	0	1	216	3	2	37	0	7	2	0	0	4
Aeshna brevistyla	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Amphipoda	36	133	127	337	513	2	67	20	0	191	784	1	550	0
Anisops wakefieldi	15	0	0	391	0	165	3	0	9	0	0	0	35	7
Austrolestes colensonis	1	0	0	4	0	0	4	0	0	0	0	0	0	0
Austrosimulium	0	0	0	0	0	0	3	0	0	0	0	0	0	0
Antiporus (adult)	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Antiporus (larvae)	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Barbronia	0	0	0	0	0	0	0	0	2	0	0	0	5	2
Berosus	0	0	0	0	0	0	0	0	9	0	0	0	0	2
Ceratopogonidae	0	0	0	0	4	0	0	0	0	0	0	0	0	0
Chironomous	0	0	0	0	16	0	0	25	8	0	1	0	3	9
Cladocera	0	0	0	1	45	0	55	0	0	0	0	0	0	289
Culex pervigilous	0	0	0	0	2	0	0	0	180	0	0	0	0	0
Diaprepocoris zealandiae	0	0	0	0	0	5	0	0	0	2	32	0	0	1
Diptera (larvae)	0	0	0	0	0	0	4	1	10	0	0	0	0	3
Empipidae larvae	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Ferrissia neozelanica	0	0	0	0	0	1	0	0	0	1	0	0	0	0
Glossiphonidae	0	0	0	0	0	0	0	35	0	0	0	0	0	0
Gyraulus corinna	4	0	0	0	0	7	1	2	27	0	0	0	0	14
Hirudinea	1	0	0	9	4	0	3	0	1	0	2	0	0	0
Hydra	0	0	0	0	0	0	0	2	0	0	0	0	0	0
Hydraenidae	0	0	0	0	0	0	0	0	12	0	0	0	0	0

Hydrobiosidae	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Hydrophilidae (larvae)	0	0	0	0	0	0	0	6	0	0	0	0	0	0
Hydroptilidae	0	0	0	0	4	5	2	1	0	1	9	0	0	4
Hygraula nitens	0	0	0	0	0	0	0	0	0	0	6	0	0	0
Ischnura aurora	0	0	0	0	21	2	17	0	1	2	2	11	3	3
Isopoda	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Isopoda (sphaeromatidae)	0	0	0	0	0	0	0	0	0	7	0	0	0	0
Laccobius	0	0	0	0	2	0	0	0	0	0	0	0	0	0
Lancetes lanceolatus	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Liodessus	0	0	0	0	0	0	0	0	4	0	0	0	0	0
Maoridiamesa	0	0	0	0	0	0	3	0	0	0	0	0	4	0
Microvelia mcgregori	0	0	0	1	0	2	7	355	16	0	0	0	5	1
Muscidae (pupae)	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Polychaete (undetermined)	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Nepia	0	0	0	0	0	0	0	0	0	0	0	0	0	3
Oecetis unicolor	0	0	0	0	0	0	0	0	0	0	0	0	0	5
Oeconesus	0	0	0	0	0	0	2	0	0	0	0	0	0	0
Ostracoda	0	0	750	675	0	19	408	0	896	0	0	0	0	18
Oligochaeta	0	1	0	3	0	10	0	0	1	0	0	20	0	88
Oxyethira	0	0	0	0	0	0	0	0	0	0	0	0	10	0
Paroxyethira hendersoni	0	0	0	0	0	11	11	2	0	3	60	0	11	8
Paratya	0	0	35	0	0	0	0	0	0	0	0	0	0	0
Physa	0	0	0	0	0	45	16	1	69	0	7	1	5	16
Pisidium	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Plectrocnemia	0	0	0	0	0	0	4	0	0	0	0	0	0	0
Polypedilum	0	0	0	0	23	20	34	11	16	0	3	0	0	27

Potamopyrgus antipodarum	0	0	0	0	0	9	11	0	0	35	6	0	36	3
Rhantus (larvae)	0	0	0	0	0	0	0	0	12	0	0	0	0	0
Rhantus suturalis	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Scirtidae	0	0	0	0	0	0	0	0	0	0	0	77	0	0
Sigara	53	0	0	107	0	36	14	3	0	0	0	0	0	8
Stratiomyidae	0	0	0	0	0	0	0	0	7	0	0	0	0	0
Tanypod	0	0	0	0	0	0	2	0	0	0	0	0	0	4
Tanytarsini	0	0	0	0	0	0	0	36	7	0	0	0	0	113
Tipulidae larvae	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Triplectides cephalotes	0	0	0	0	0	1	8	0	0	0	0	0	0	0
Xantocnemis zealandica	1	0	0	7	2	0	0	13	0	0	0	0	0	0
Zelandoptila moseleyi	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Zelandoperla	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Dysticidae (undetermined)	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Total Number of Individuals	115	136	913	1536	852	345	684	553	1290	249	917	110	667	633

Table 4: Presence/absence of macrophytes of 14 freshwater wetlands in lower North Island, 2017.

Site	Bog gy Pond	Nga Manu main pond	Aot ea Lag oon	Pharaz yn Reserv e	Whit by Lakes	Zealan dia Upper Dam	Zealan dia Lower Dam	Lake Wait awa	Omahu	Lake Kohan gatera	Lake Kohanga piripiri	Te Hapua	Nga Manu Top Pond	Queen Elizab eth Park
Ricciocarpus natans	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Lemna minor	1	1	0	1	0	0	1	1	1	0	0	1	1	0
Myriophyllum sp	1	0	0	0	0	1	0	1	0	0	0	0	0	0
Spirodella punctata	1	0	0	1	0	0	0	1	1	0	0	1	0	1
Azolla Rubra	0	0	0	0	0	0	0	1	1	0	0	0	0	0
Azolla Pinnata	0	1	0	0	0	0	0	0	0	0	0	1	1	0
Nymphaea sp	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Ranunculus sp	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Ludwigia palustris	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Baumea rubiginosa	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Ceratophyllum demersum	0	0	0	0	0	0	0	1	0	1	0	0	0	0
Elodea Canadensis	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Thypha orientalis	0	0	0	1	0	0	0	0	0	0	0	1	0	0
Carex secta	0	1	0	0	0	0	1	0	1	1	0	1	1	1
Phormium tenax	0	1	0	0	0	0	1	1	1	0	1	1	1	0
Austroderia toetoe	0	0	0	0	0	1	0	0	0	1	0	1	1	1
Isolepsis sp	0	0	0	0	0	0	0	0	1	0	1	0	0	0