

# **Community composition and distribution of macroinvertebrates on natural and artificial substrates in North Island lakes and ponds.**

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# Abstract

Artificial structures in freshwater ecosystems are becoming increasingly common. The introduction of these structures in marine systems lead to changes in the physical structure and the biotic assemblages. Even though artificial structures provide hard substrates for species to colonize and settle on, they are commonly shown to support a smaller number of taxa than natural substrates and are also known to facilitate the settlement of disproportionate numbers of non-native species. Nevertheless, little is known of their effects in freshwaters.

In the second chapter of my thesis, I compare macroinvertebrate communities on three substrate types, macrophytes, sediments and artificial structures, among twenty-two ponds, to identify differences in community composition and distribution of macroinvertebrates between natural and artificial substrates. In addition, I investigated whether non-native macroinvertebrates prefer artificial substrates more than natural substrates. The similarities between artificial substrates and macrophytes can be attributed to the habitat structure, i.e., hard surfaces and the provision of refugia for macroinvertebrates from their predators. The differences between sediments and macrophytes, and artificial substrates were attributed to the difference in habitat structure and the fauna they supported (i.e., sediments contained large proportions of infauna). Non-native species that were collected in my study did not show preference for any of the habitats that were sampled.

In the third chapter of my thesis, I identified important environmental factors that affect the distribution of macroinvertebrates communities across the twenty-two ponds sampled. The most important factor identified was temperature, nutrient gradients, pH and dissolved oxygen levels.

Both chapters of my thesis offer an understanding of how existing artificial substrates and environmental variables in lakes and ponds on the North Island, New Zealand, affect macroinvertebrate community composition and distribution in relation to natural substrates.

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# **Chapter 1**

## **Thesis Introduction**



### *Anthropogenic effects on aquatic systems*

Human activities such as agricultural land use, economic activities and urban expansion with rapid human population growth have caused drastic changes to aquatic ecosystems like rivers, lakes, coastal marshes and estuaries (Maybeck, 2004). These changes have affected the quality of water as well as the species that live in these aquatic environments. In regions such as North American and Europe, much riparian vegetation has been lost due to the expansion of urban areas and as result of economic development (Naiman, 1993). Climatic changes, diseases and invasive species, nutrient run-off from agricultural and farming activities into aquatic systems, have all caused changes in water quality, community composition and distribution (Rosenzweig et al., 2008). For example, industrial effluent flows in Portage Creek, Michigan, resulted in decreased abundances of macroinvertebrate communities (Nedeau et al., 2003), while the distribution and abundance of zooplankton in marine waters were shown to have changed in response temperature and acid increase (Donney et al., 2009; Polivina et al., 2008).

### *Artificial structures in marine and freshwater ecosystems*

The modification of coastlines is an anthropogenic impact of growing importance in marine areas, due to the demand of the growing human population to support economical, recreational or research activities. These structures include pontoons, drifting buoys, groynes, pier pilings, seawalls and jetties. Many of these structures are installed to protect shorelines from degradation due to wave action, and to support human activities such shipping ports, aquaculture farms and oil rigs (Asif & Muneer, 2007).

### *Changes in marine and freshwater ecosystems*

Artificial structures are introduced constructions that are constructed from man-made or natural material (e.g., concrete, wood). These structures have been found to affect species assemblages within ecosystems. Some effects of artificial structures include shading, provision of artificial lighting, vertical

and floating surfaces (Vaselli et al., 2008; Dafforn et al., 2015), as well as the facilitation of invasions by non-native species in introduced habitats (Cohen & Carlton, 1995; Wasson et al., 2005). The establishment of these man-made constructions has been demonstrated to affect species composition and distribution. The effects on species assemblages from shading by artificial substrates have been demonstrated in an experiment by Glasby (2009), at Mitchell's Marina, Sydney, Australia; this study showed differences in communities, with abundant filamentous algae and spirorbid polychaetes, on unshaded pilings, while bryozoans, serpulid polychaetes, ascidians and sponges were more common on shaded pilings. Artificial lighting is also associated with man-made constructions (e.g. piers) in aquatic systems and can affect the ecology of many species. For example, the behaviour of hatchlings of loggerheads (*Caretta caretta*) and green turtles (*Chelonia mydas*) were found to be affected by the brightness of artificial light, as natural lighting provides cues for hatchlings to crawl towards the sea (Tuxbury & Salmon, 2005). In addition, introduced structures may alter sediments, especially in soft-bottomed habitats, and taxa composition in these habitats. Coates et al. (2013), for example, found higher proportions of finer sediments near the foundations of a gravity-based wind turbine in the North Sea, Belgium. There, sediment particle sizes increased further away from the structures, and taxa changed, with the polychaete *Lanice conchilega* more common in sediments near the turbine foundation (Coates et al., 2013). Artificial reefs were shown to decrease sea pen *Stylatula elongata* densities a month after installation due to the presence of predator fish that were attracted to the reefs in the San Diego-La Jolla Underwater Park Reef, California, USA (Davies et al., 1982).

Introduced hard substrates can provide surfaces for organisms to live on. An experimental study by Attila & Fleeger (2000) showed high abundances of taxa, dominated by the copepods *Harpacticus* sp. and *Parategastes* sp., on artificial substrates near Grand Isle, Louisiana, USA. In addition, five scyphozoan species (*Aurelia aurita*, *Cyanea capillata*, *Cyanea lamarckii*, *Chrysaora hysoscella*, and *Rhizostoma octopus*) were observed to prefer artificial substrates (e.g. concrete, glass) over natural substrates (shells) for settlement in a laboratory experiment (Holst & Jarms, 2007).

Artificial structures are commonly colonized by non-native species. Many studies on colonization of artificial structures by non-native species have shown marked preferences of artificial structures by these species (Cohen & Carlton, 1995; Lambert & Lambert, 2003; Fofonoff et al., 2008). For example, in Sydney Harbour, Australia, Glasby et al. (2007) found higher taxa richness of non-native species on artificial structures (pontoons and pilings) than on natural rocky reefs. Similarly, changes in habitats caused by the presence of artificial structures encourage non-native species to thrive. In a study by Bulleri & Airoidi (2005), it was shown that the non-native alga *Codium fragile* spp. *tomentosoides* had high density and cover in the sheltered, landward side provided by breakwaters on the Adriatic coast of Italy. In Lajes Reservoir, Brazil, the non-native fish *Cichla kelberi* and *Tilapia rendalli* were observed to be associated with artificial habitats than other fish species.

#### *Artificial structures in New Zealand lakes and ponds*

New Zealand has more than 3000 lakes of varying sizes (from 1ha - >1000 ha) and types (e.g. peat, riverine, glacial) that have artificial structures built in them (Bay of Plenty Regional Council, 2019). These include walkways, bridges and jetties that are mainly used for recreational purposes. There have been limited studies undertaken to understand the effects of artificial structures on species composition and distribution in New Zealand lakes as well as in lakes elsewhere. Bright et al. (2004) found a positive correlation between the number of man-made structures and the frequency of a native bird, the dabchick (*Poliiocephalus rufopectus*), in three lakes in the North Island. The effects of artificial shading was tested in an experiment using a polythene sheet to mimic canopy cover in the Waitakere River resulted in no epiphyton growth, leading to changes in macroinvertebrate communities in the stream (Townes, 1981). Other researchers have used artificial material to investigate changes in communities due to water pollution (Biggs, 1989), for biomonitoring purpose of macroinvertebrates (Boothroyd & Dickie, 1989; Suren, 1991; Suren, 1992; Collier et al., 2009), periphyton biomass (Biggs, 1988; Suren, 1993), and to investigate relationships among periphyton,

nutrients and macroinvertebrates (Towns, 1981; Biggs & Lowe, 1994; Zimmerman & Death, 2002).

Studies undertaken to examine the species composition on existing artificial structures in lake ecosystems are have not been done at all. In New Zealand, related studies to this question includes one done by Banks & Duggan (2009), where they found that almost half the species found were non-native and restricted to artificially constructed water bodies. Knowledge of the impacts of artificial structures in coastal areas have indicated increased changes in species composition and distribution, as well as provision of habitats for the colonization of non-native species. This suggests further research into the impacts of artificial structures in lakes is warranted, to investigate whether freshwater macroinvertebrate communities are affected by artificial substrates.

## **Thesis outline**

In this thesis I investigated; i) whether invertebrates on artificial structures in lakes are different from those on natural substrates; and ii) important environmental variables that influence the distribution and composition of sampled macroinvertebrate communities among ponds Auckland, Waikato and Bay of Plenty Regions.

In Chapter 2, I compare macroinvertebrate communities on three substrate types; macrophytes, sediments and artificial substrates, among ponds. The aim of this chapter is to identify whether there are differences in community composition and distribution of macroinvertebrates between natural and artificial substrates. In addition, I investigate whether non-native macroinvertebrates prefer artificial substrates more than natural substrates.

In Chapter 3, I identify environmental variables that affect the composition and distribution of macroinvertebrates among North Island ponds. The aim of this chapter is to identify important environmental variables that influence the distribution and composition of sampled macroinvertebrate communities among ponds.

In my concluding chapter, Chapter 4, I summarize the findings of Chapter 1 and 2 of my research.

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## **Chapter 2**

# **Differences in macroinvertebrate community composition between natural and artificial substrates in lakes and ponds in Auckland, Waikato and Bay of Plenty Regions, North Island**

## Introduction

Artificial structures in marine and freshwater ecosystems are constructed from materials including concrete, wood, steel, plastic and metals (Bulleri & Chapman, 2010). Such structures range from pontoons, drifting buoys, groynes and piers, built to assist economic and boating activities, to walkways and boat ramps for leisure, and retaining walls and seawalls to reduce the threat of shoreline erosion (e.g. Duvat, 2013). Growth in coastal (Bulleri & Chapman, 2015) and freshwater (Leigh et al., 2012, Vorosmarty et al., 2010) urbanization has seen an increase in the construction of these structures off- and on-shore.

Marine ecosystems can be affected when environmental conditions are altered by the addition of artificial structures. For example, the position of structures such as groynes reduce the movement of sediments on beaches (Bulleri & Chapman, 2015). Further, if groynes are placed at right angles to the shore, fine sediments can accumulate on their landward side (Bulleri & Airoidi, 2005). Retaining walls and seawalls result in sediment loss due to wave action on these hard surfaces (Kraus & McDougal, 1996). Other effects, such as shading and artificial lighting, cause changes in reproductive patterns and alterations in the distributions of fish. For example, low abundances and reduced species richness of juvenile fish have been found under piers compared to open waters (Able et al., 1998). Alternatively, these structures can act as novel habitats, whereby the constructed objects introduce hard substrates suitable for epibenthic organisms to colonize, including invertebrates and fish (Davis et al., 2002), ascidians (Lambert & Lambert, 2003), polychaetes and brown algae (Glasby et al., 2007). For example, an experiment by Feng et al. (2017) found that polyps of the jellyfish *Aurelia aurita* had a high preference for artificial substrates constructed from rigid polyvinyl chloride plates and nets than the shells of two scallop species (*Azumapecten farreri* and *Patinopecten yessoensis*).

Environmental conditions created by artificial substrates are commonly unsuitable for native species and facilitate invasions by non-native species (Bulleri & Airoidi, 2005, Page et al., 2007). In marine environments,

populations of the same species that are restricted to particular geographical areas or habitats have been assumed to result from natural processes rather than from human causes. As such, Chapman and Carlton (1991) developed ten criteria that could be used to infer whether the isopod *Synidotea laevidorsalis* (and other marine species) constituted a non-indigenous species in North and South America; one of their proposed criteria was the species live in association with new or artificial environments, such as pontoons and pilings. This pattern has been observed for non-native invertebrates including bryozoans, ascidians and polychaetes, where greater percentage coverage and richness of these invertebrates has been observed on pontoons and pilings than natural reefs in Port Jackson, Australia (Dafforn, 2012). Bulleri et al. (2006) found the establishment and spread of the invasive green alga species *Codium fragile* ssp. *tomentosoides* on the coast of Cesenatico, Italy, to be a result of the introduction of hard substrates (breakwaters), which acted as sites for colonization. Differences have been reported in the community structure and assemblages between artificial infrastructure and natural habitats in marine environments, where non-native species such as brown alga (Farrell & Fletcher, 2006), kelp (Thornber et al., 2004), polychaetes (Holloway & Keough, 2002), arthropods and bryozoans (Glasby et al., 2009) were observed to favour man-made structures.

While there are a number of studies of the effects of artificial structures in coastal and estuarine environments (Connell & Glasby, 1999; Connell, 2001), their effects in freshwater ecosystems are less well understood. In lakes, the effects of artificial structures have been investigated for fish, which have been found to have higher species richness and density in shorelines adjacent to areas with less artificial modification in a lake in Iowa, USA (Bryan & Scarnecchia, 1992). Artificial structures such as bridges have also been shown to contribute to river mussel declines, as a result of the damage and sedimentation in river bed habitats when the bridges were built (Cooksley et al., 2012). Nevertheless, studies examining the species composition of benthic communities on artificial structures in lake or pond ecosystems are few. Most studies of man-made introduced objects in freshwater systems have been those that have examined the use of artificial

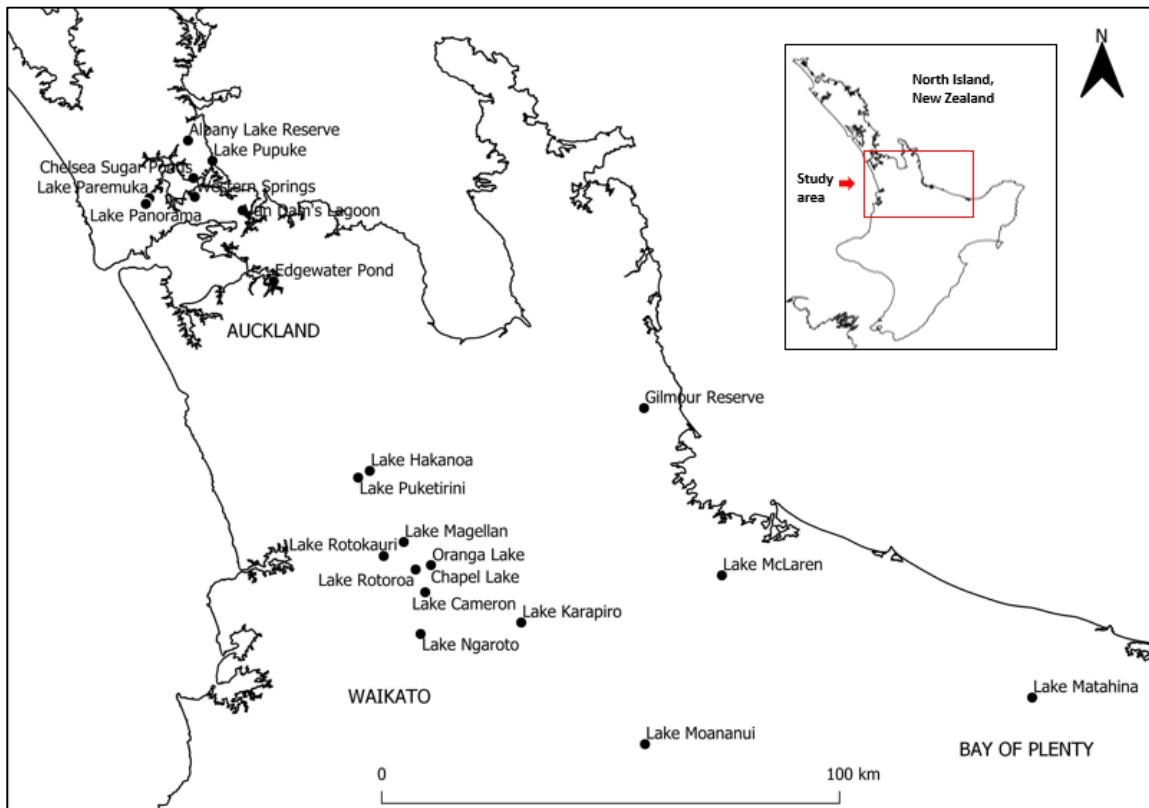
substrates, temporarily placed in waterbodies, as tools for sampling benthic communities. Lamberti & Resh (1985), for example, compared natural rocks with clay tiles submerged in Big Sulphur Creek, USA, and found that microbial communities were similar on the tiles, sterilized rocks and natural rocks sampled instream. These authors also found macroinvertebrate communities were similar between substrates during the study and concluded that up to two months of submerging tiles was adequate to accurately represent benthic communities. Lowe & Gail (1980) also showed that frosted glass was colonized by similar algal communities as natural rocks in the Susquehanna River, Pennsylvania, showing that using glass as an artificial substrate can reduce variability in sampling periphyton when left to settle for a month, and sampled monthly for nine months. Polyurethane foam units were sampled for two weeks to assess protozoan community composition in the eutrophic Chaohu Lake, China. Here, the polyurethane units had more species than natural substrates (Xu et al., 2005).

Despite mounting evidence that species assemblages differ between natural and artificial habitats in marine systems, and that they facilitate invasions, little research has been undertaken to understand how permanent human-made structures affect species assemblages in freshwater systems. The aim of this study was to investigate whether: 1) macroinvertebrates on artificial structures in lakes are different from those on natural substrates; and 2) non-native macroinvertebrate species are more prevalent on artificial structures relative to natural habitats.

## Methods

### *Sampling sites*

Twenty-two lakes in the Waikato, Auckland and Bay of Plenty regions, North Island, New Zealand (Figure 1), were selected as sampling sites which had artificial structures (e.g., jetties, retaining walls, pier pilings, culverts, etc.), macrophytes and exposed bottom sediments. A search of lakes and ponds using satellite imagery on Google Maps, as well as personal knowledge, were used to evaluate and select waterbodies based on the presence of these substrates. The geographical location (latitude and longitude coordinates) of each lake was recorded using a handheld GPS (GPSMAP 64). Altitude information was also recorded on the GPS when coordinates were saved but in instances where there were errors in altitudinal data, cross-referencing of lake sites were done using information from Land Information New Zealand (LINZ) Data service online and Google Earth® (2019) to correct these.



**Figure 1.** Map showing lakes in the Waikato, Auckland and Bay of Plenty Regions, North Island, New Zealand, where sampling was carried out during this study.

**Table 1.** The location and altitude of lakes and ponds that were sampled in this study. Site identification numbers (ID) represent numbers used in the non-metric Multidimensional Scaling ordination plot. Lakes are listed by sampling order.

Site ID	Lake Name	Latitude	Longitude	Altitude (m)
1	Albany Lake Reserve	36°43'34.8"S	174°42'31.1"E	36
2	Lake Pupuke	36°46'37.6"S	174°46'11.5"E	9
3	Chelsea Sugar Ponds	36°49'12.6"S	174°43'20.1"E	9
4	Lake Hakanoa	37°33'05.7"S	175°09'47.9"E	10
5	Lake Moananui	38°14'03.1"S	175°51'07.0"E	336
6	Lake Ngaroto	37°57'32.5"S	175°17'26.1"E	36
7	Lake Cameron	37°51'17.2"S	175°18'07.4"E	61
8	Lake Rotokauri	37°45'51.6"S	175°11'53.6"E	47
9	Western Springs	36°52'01.2"S	174°43'31.8"E	13
10	Lake Paremuka	36°52'12.8"S	174°36'43.2"E	41
11	Lake Panorama	36°53'04.8"S	174°36'11.0"E	41
12	Van Dammes Lagoon	36°54'02.8"S	174°50'43.1"E	21
13	Edgewater Pond	37°04'40.8"S	174°55'23.2"E	16
14	Lake Gilmour	37°23'41.2"S	175°50'58.2"E	93
15	Lake Matahina	38°07'04.0"S	176°49'14.2"E	67
16	Lake McLaren	37°48'44.9"S	176°02'39.5"E	93
17	Lake Karapiro	37°55'49.4"S	175°32'32.5"E	59
18	Lake Magellan	37°43'45.5"S	175°14'54.2"E	40
19	Lake Rotoroa	37°47'52.8"S	175°16'41.3"E	38
20	Lake Puketirini	37°34'06.6"S	175°08'03.5"E	10
21	Chapel Lake	37°47'17.4"S	175°18'53.2"E	51
22	Oranga Lake	37°47'14.0"S	175°18'58.3"E	50

### *Invertebrate sampling*

Sampling took place in July and August 2018 (austral winter). Macroinvertebrates were collected from three substrate types; artificial substrates, macrophytes and sediments. Seasonal differences do not affect invertebrate sampling in New Zealand freshwater systems. McCord & Kuhl (2012) found no seasonal variance in community assemblage in a two-year study in the Mississippi River, Minnesota. Most macroinvertebrates (e.g.

crustaceans, worms, molluscs) are not very mobile and remain in ponds all year round.

Collection was undertaken by wading from the shore to a depth of 1 m; three minutes was spent sampling from each substrate type for macroinvertebrates using a 450  $\mu\text{m}$  dip net. The contents of the net were emptied after collection from each substrate into 2 L polypropylene wide-mouth storage containers and were sorted on return to the laboratory.

### *Laboratory analysis*

In the laboratory, total macroinvertebrate counts were undertaken after sieving the contents of the containers through a 450  $\mu\text{m}$  sieve. Contents retained on the sieve were emptied into a white sorting tray (29.2 x 45.1 x 1.9 cm) where all visible invertebrates were picked out using a pipette or tweezers and stored in 90% isopropyl alcohol. Individuals were identified to the lowest practical taxonomic level, using Nikon E200 compound and Nikon SMZ25 stereo microscopes and relevant taxonomic keys (e.g., Chapman & Lewis, 1976; Winterbourn & Gregson 1989; online guides such as NIWA Invertebrate ID Guides and Landcare Research Freshwater Invertebrate Guide). Individual invertebrates that were difficult to identify to species level were left at subfamily (e.g., Oligochaetes) or genus level (e.g., *Polypedilum* sp.).

### *Statistical analyses*

Total abundances and taxonomic richness of invertebrates between substrate types were compared using one-way ANOVA in Statistica v.13 (Statsoft Inc., Tulsa USA). Data were tested for normality using the Shapiro-Wilks test in Statistica, and in cases where data were not normally distributed, these were  $\log(x+1)$  transformed prior to analysis to improve normality. ANOVA results that indicated significant differences were examined using a Tukey HSD post-hoc test, to identify specific differences between pairs of means. Patterns in community composition of invertebrates between substrates among lakes were assessed using non-

metric Multidimensional Scaling (nMDS) and Analysis of Similarities (ANOSIM) (PRIMER v.6.6.13; Primer E Ltd 2009). To reduce the influence of rare species that may have been sampled by chance, taxa that did not contribute at least 10% of the total abundances in any sample, and sites containing <3 individuals, were excluded from the multivariate analyses. This left 29 species (of 43 taxa collected) and 57 samples (of 66 collected) included in the analyses. nMDS was conducted on the Bray-Curtis similarity coefficient calculated on  $\log(x+1)$  transformed species data. This produced a 2-dimensional plot of all samples based on their similarity to each other. ANOSIM was conducted on the Bray-Curtis similarity matrix to test for statistically significant differences in invertebrate community composition among substrate types. ANOSIM produces *P*-values and an R-statistic for between group comparisons, where the R-statistic is usually between 0 and 1; when the R-statistic is closer to 0, this indicates that samples between groups are very similar in composition, but if it is closer to 1, the samples are highly dissimilar. When ANOSIM indicated significant differences among groups, Similarity Percentages Analyses (SIMPER) were carried out to determine the species that contributed most greatly to the similarity or dissimilarity between each substrate type.

One-way Analysis of Variance (ANOVA) was undertaken in Statistica (Statsoft Inc., Tulsa USA) to test if there were differences in the total abundance of non-native species between each substrate. Results that indicated significant differences were examined using a Tukey HSD Test to identify specific differences between pairs of mean abundances.

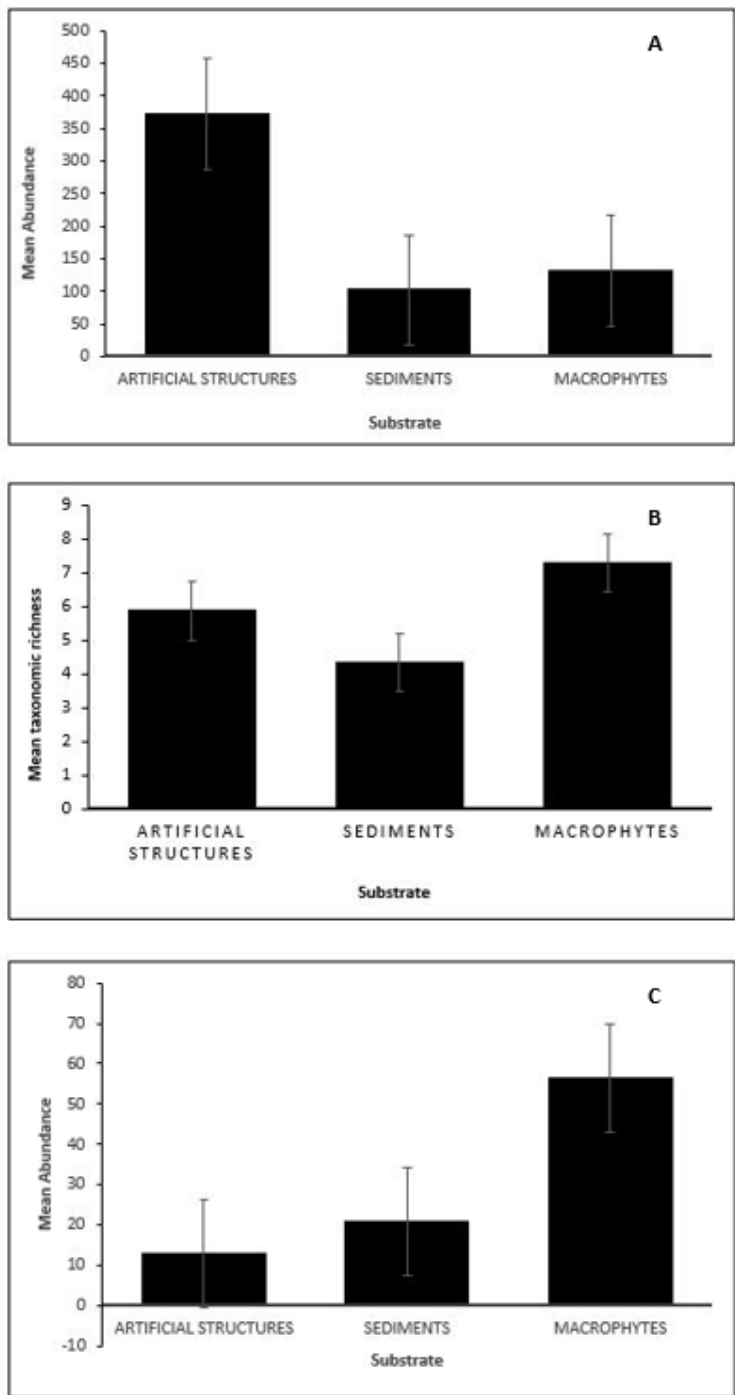


## Results

### *Invertebrate abundances and community structure*

A total of 25 911 invertebrate individuals were collected, with 43 invertebrate taxa identified across all substrate types (Table 1). Of these taxa, 35 taxa were found on artificial substrates, 24 in sediments and 34 on macrophytes. A significant difference was found for taxonomic richness (Figure 2b) of invertebrates between substrate type (one-way ANOVA; ( $F(2) = 5.702, p = 0.005$ )). Post-hoc comparisons using a Tukey HSD Test indicated that the mean richness of taxa in sediments was significantly lower than that on macrophytes, with a mean of 4.7 taxa in sediments and a mean of 7.4 taxa on macrophytes. There were no differences in mean taxa richness between artificial substrates (5.9 taxa) and macrophytes, or between artificial substrates and sediments ( $p < 0.05$ ).

A one-way ANOVA conducted to compare total abundances of all invertebrates between each substrate type indicated that there were no significant differences among substrate types ( $F(2) = 0.668, p = 0.536$ ) (Figure 2a).



**Figure 2. a)** Total abundances of invertebrates found in each substrate; artificial structures (N=22, mean=335 SD=965.926), sediments (N=22, mean=359, SD=999.709) and macrophytes (N=22, mean=164, SD=468.474) in all sites, **(b)** Taxonomic richness of invertebrates found associated with all substrates; artificial structures (N=22, mean=5.863, SD=3.796), sediments (N=22, mean=4.318, SD=2.885) and macrophytes (N=22, mean=7.272, SD=2.815), and **(c)** Total abundance of non-native invertebrate species (N=12, mean=30, SD=55.258) found in all the substrates.

**Table 2** List of invertebrate taxa sampled from lakes and ponds in this study. Mean abundances of taxa are shown per sampling effort for all substrates in all sites and mean abundances for each substrate type across all 22 sampled sites.

Invertebrate species	No. of lakes taxa present	Mean abundance/ minute of sampling across all substrates and sites	Mean abundance on artificial substrates across all sites	Mean abundance on macrophytes across all sites	Mean abundance in sediments across all sites
<b>CRUSTACEANS</b>					
<b>Cladocera</b>					
<i>Ilyocryptus sordidus</i> Liévin, 1848	1	0.11	0.33	0.00	0.00
<i>Simocephalus vetulus</i> Müller, 1776	6	332.00	966.67	17.67	11.67
<b>Copepods</b>					
<i>Acanthocyclops robustus</i> Sars 1863	11	142.44	376.67	37.00	13.67
<i>Macrocyclops albidus</i> Jurine 1820	4	5.56	16.00	0.67	0.00
<i>Skistodiaptomus pallidus</i> Herrick 1879	1	0.111111	0.333333	0	0
<b>Amphipod</b>					
<i>Paracalliope fluviatilis</i> Thomson 1879	4	592.89	1533.00	122.67	123.00
<b>Ostracod</b>					
<i>Herpetocypris pascheri</i> Brehm 1929	6	15.11	34.67	5.33	5.33
<b>Isopod</b>					
<i>Styloniscus</i> sp.	7	1.67	0.33	4.67	0.00
<b>Shrimp</b>					
<i>Paratya curvirostris</i> Heller 1862	2.22	0.67	6.00	0.00	2.22

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**INSECTS****Dragonflies**

<i>Hemicordulia australiae</i> Rambur 1842	3	0.78	0.33	2.00	0.00
<i>Procordulia grayi</i> Selys 1871	2	0.22	0.00	0.67	0.00
<i>Procordulia smithi</i> White 1846	2	3.33	1.00	9.00	0.00
<i>Aeshna brevistyla</i> Rambur 1842	3	0.67	0.00	2.00	0.00

**Damselflies**

<i>Xanthocnemis zealandica</i> McLachlan1873	10	11.44	2.67	28.33	3.33
<i>Ishnura aurora</i> Brauer 1865	2	1.89	1.33	4.33	0.00
<i>Austrolestes colenonis</i> White 1846	1	0.44	0.00	1.00	0.33

**Mayflies**

<i>Nesameletus ornatus</i> Eaton 1883	1	0.44	0.00	1.33	0.00
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**Dipterans**

<i>Limonia nigrescens</i> Lackschewitz1928	2	0.78	0.33	2.00	0.00
<i>Chironomus zealandicus</i> Hudson 1892	17	256.56	62.33	176.00	531.33
<i>Polypedilum</i> sp.	3	27.22	1.00	23.00	57.67
<i>Tanytarsus vespertinus</i> Hutton 1902	1	0.11	0.00	0.33	0.00
<i>Tanytarsus funebris</i> Freeman 1959	1	0.67	2.00	0.00	0.00
<i>Tanytarsus</i> sp.	3	3.22	6.67	2.33	0.67
Orthoclaadiinae	1	10.22	28.67	2.00	0.00

**Acari**

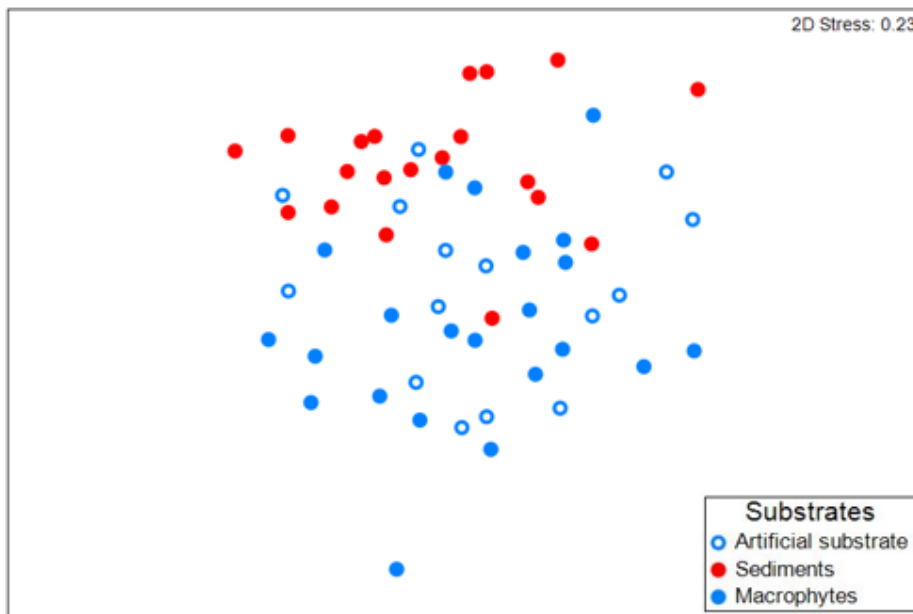
<i>Piona uncata exigua</i> Viets 1949	18	44.33	76.67	36.67	19.67
<b>Corixidae</b>					
<i>Sigara</i> sp.	4	8.00	16.33	0.00	7.67
<b>Notonectidae</b>					
<i>Anisops wakefieldi</i> White	6	38.78	13.67	100.33	2.33
<b>Gerridae</b>					
<i>Microvelia macgregori</i> Kirkaldy, 1899	3	0.44	0.33	0.67	0.33
<b>Dytiscidae</b>					
<i>Antiporus strigosulus</i> Broun	6	14.22	6.67	35.33	0.67
<b>Trichoptera</b>					
<i>Oxyethira albiceps</i> McLachlan, 1862	9	14.67	11.00	27.33	5.67
<i>Paroxyethira</i> sp.	1	0.78	2.33	0.00	0.00
<i>Triplectides obsoletus</i> McLachlan, 1862	1	1.11	0.00	0.00	3.33
<b>Molluscs</b>					
Sphaeriidae sp.	2	20.568.41	0.000.00	0.000.00	61.6761.67
<i>Gyraulus corinna</i> Gray, 1850	5	2.000.82	0.670.67	4.674.67	0.670.67
<i>Physa acuta</i> Draparnaud, 1805	13	31.5612.91	13.0013.00	65.3365.33	16.3316.33
<i>Potamopyrgus antipodarum</i> J. E. Gray, 1843	13	589.33241.09	1647.001647.00	49.6749.67	71.3371.33
<b>Annelids</b>					
<i>Barbronia weberi</i> Blanchard, 1897	2	0.220.09	0.000.00	0.000.00	0.670.67
<i>Albiglossiphonia</i> sp.	1	0.110.05	0.330.33	0.000.00	0.000.00

<i>Helobdella</i> sp.	9	8.113.32	3.673.67	10.0010.00	10.6710.67
Oligochaetes	19	590.56241.59	370.33370.33	899.67899.67	501.67501.67
<b>Platyhelminthes</b>					
<i>Cura</i> sp.	16	85.1134.82	80.0080.00	163.33163.33	12.0012.00
<i>Neppia</i> sp.	3	6.672.73	8.008.00	12.0012.00	0.000.00
<b>Cnidaria</b>					
<i>Hydra</i> sp.	2	0.330.14	0.670.67	0.330.33	0.000.00

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### *Community composition between substrates*

The non-metric multidimensional scaling ordination indicated that invertebrate communities among the sediment samples were different to those among the macrophyte and artificial substrate samples, with sediment samples distributed primarily towards the top of the ordination plot, while communities associated with macrophytes and artificial structures were distributed primarily towards the lower half of the ordination (Figure 1). Communities on artificial substrates and macrophytes overlap on the ordination, with no difference in communities between these substrates apparent. Analyses of Similarities (ANOSIM) suggested that there were significant differences in community composition between sediments and macrophytes (Global  $R=0.261$ ,  $p=0.001$ ) and between sediments and artificial substrates (Global  $R=0.204$ ,  $p=0.004$ ). Communities between artificial substrates and macrophytes did not differ (Global  $R=0.04$ ,  $p=0.18$ ) (Table 3).



**Figure 3** Non-metric Multidimensional Scaling (nMDS) plot representing the differences in community composition of invertebrates between substrate types; artificial structures, sediments and macrophytes at each site.

A SIMPER analysis showed that species that contributed the most (>5%) to the dissimilarity between sediment and macrophyte communities were Oligochaetes (13.33%), *Chironomus zealandicus* (12.56%), *Cura* sp. (9.99%), *Potamopyrgus antipodarum* (6.76%), *Physa acuta* (6.45%), *Piona uncata exigua* (5.36%) and *Acanthocyclops robustus* (5.07%) (Table 4). Oligochaetes and *Chironomus zealandicus*, which had the greatest contributions to the differences between the two substrates, along with *Potamopyrgus antipodarum*, had higher average abundances among sediments than on macrophytes. Conversely, *Cura* sp., *Physa acuta* and *Acanthocyclops robustus* had higher average abundances on macrophytes than sediments.

Species that contributed most greatly (>5%) to the differences between artificial substrates and sediments were, similarly, Oligochaetes (12.71%), *Chironomus zealandicus* (11.42%), *Potamopyrgus antipodarum* (10.78%), *Acanthocyclops robustus* (9.37%), *Piona uncata exigua* (7.56%), *Cura* sp. (5.96%) and *Simocephalus vetulus* (5.38%) (Table 4). Oligochaetes and *Chironomus zealandicus* made the greatest contributions to the differences between the two substrates and had higher average abundances in sediments than on artificial substrates. The remaining species contributing more than 5 % to the dissimilarity had higher average abundances on the artificial substrates than sediments (e.g., *P. antipodarum*, *A. robustus*, *Piona uncata exigua*, *Cura* sp. and *S. vetulus*).



**Table 3** Results of a SIMPER analysis showing species contributing >5% to the dissimilarity between (A) artificial substrates and sediments and, (B) sediments and macrophytes. Species are listed in order of higher to lower percentage contribution.

**A: Artificial substrates & Sediments**

	<b>Artificial substrates</b>	<b>Sediments</b>			
	<b>Av. Abundance</b>	<b>Av. Abundance</b>	<b>Av. Diss.</b>	<b>Contrib.%</b>	<b>Cum.%</b>
Oligochaetes	2.03	2.77	9.86	12.71	12.71
<i>Chironomus zealandicus</i>	1.18	2.15	8.86	11.42	24.13
<i>Potamopyrgus antipodarum</i>	1.76	0.94	8.36	10.78	34.91
<i>Acanthocyclops robustus</i>	2.17	0.34	7.27	9.37	44.27
<i>Piona uncata exigua</i>	1.55	0.45	5.86	7.56	51.83
<i>Cura</i> sp.	1.23	0.35	4.62	5.96	57.79
<i>Simocephalus vetulus</i>	1.45	0.23	4.17	5.38	63.17

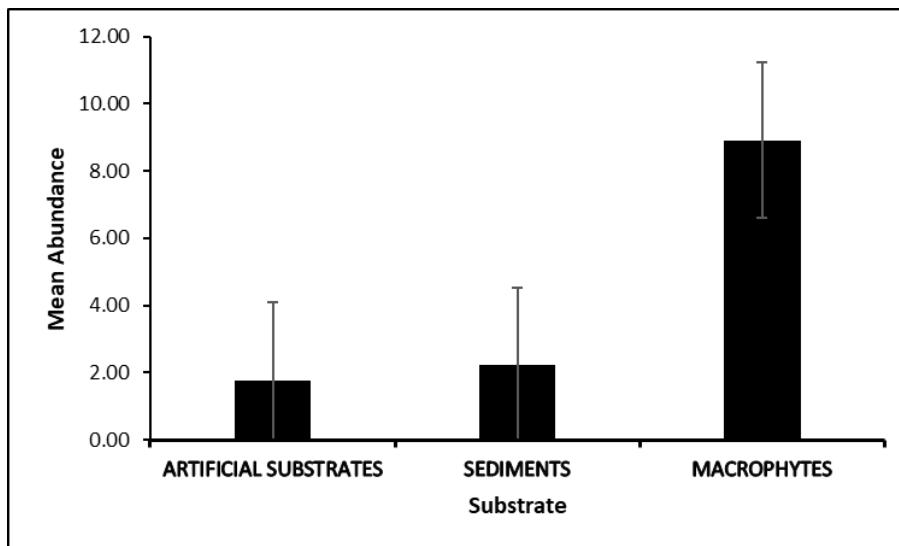
**B: Macrophytes & Sediments**

	<b>Macrophytes</b>	<b>Sediments</b>			
	<b>Av. Abundance</b>	<b>Av. Abundance</b>	<b>Av. Diss.</b>	<b>Contrib.%</b>	<b>Cum.%</b>
Oligochaetes	1.21	2.77	10.65	13.33	13.33
<i>Chironomus zealandicus</i>	1.08	2.15	10.03	12.56	25.89
<i>Cura</i> sp.	1.77	0.35	7.98	9.99	35.89
<i>Potamopyrgus antipodarum</i>	0.83	0.94	5.40	6.76	42.65
<i>Physa acuta</i> .	1.11	0.49	5.15	6.45	49.10
<i>Piona uncata exigua</i>	1.01	0.45	4.28	5.36	54.46
<i>Acanthocyclops robustus</i>	0.86	0.34	4.05	5.07	59.53

### *Non-native invertebrate species*

Four non-native species were collected in this study; the copepod *Skistodiaptmus pallidus*, gastropod *Physa acuta*, and the leeches *Barbronia weberi* and *Helobdella* sp. One-way ANOVA indicated that there were no differences in the total abundances of non-native taxa between substrate types ( $F(2) = 0.668$ ,  $p = 0.536$ ).

The SIMPER analysis also indicated that among all non-native species, only the gastropod *Physa acuta* was shown to contribute >5% (6.45%) to the average dissimilarities between macrophytes and sediment samples, but not between artificial substrates and sediments (Figure 4).



**Figure 4** Mean abundances of *Physa* sp. collected from each substrate type from the 22 sampling sites.

## Discussion

Similarities in invertebrate community composition were identified between artificial substrates and macrophytes, but differences in community composition between macrophytes and sediments in sampled ponds and lakes. Further, species richness was observed to differ significantly between the three substrate types with higher number of invertebrate species found in sediments than that found in macrophytes. The differences in community compositions and species richness can be explained as an indication of the differences in habitat types and resources (such as food, space and shelter from predation) that are available and suitable for species in these substrates to utilize for their survival (Collier & Death, 2010). There were different species that dominated each of the substrate types. The copepod *Acanthocyclops robustus* was the abundant taxa on the artificial substrates while Oligochaetes were abundant in sediments and *Cura* sp. contributed more to the abundances found on the macrophytes.

### *Community composition on artificial substrates versus macrophytes*

Invertebrate communities on artificial structures were similar to communities found among macrophytes. This could be because both habitats offer epiphyton for grazers to use. Macrophytes are important for epiphytic invertebrates as they provide a food source from the epiphytic algae that grow on them, they also offer protection from predators and support for these invertebrates (Becerra-Muñoz & Schramm, 2007).

The species that contributed to similar communities found between artificial substrates and macrophytes include *Acanthocyclops robustus*, *Piona uncata exigua*, *Simocephalus vetulus* and *Cura* sp. The first three species are mainly planktonic species and *Cura* sp. is a grazer. As such they have been found to be common on both human-made structural surfaces as well as plant material which support the growth of epiphyton. The epiphytic algae are used as a food source for these species.

Artificial structures are known to provide suitable habitats for species in areas with limited suitable habitats and because they could have reduced predation and more available space (Alroidi & Bulleri, 2005). Thus, communities may be similar due to these substrates offering similar ecological functions to support these communities. Similar results were seen by Collier (2010), who found that communities on artificial structures were similar to those found on natural habitats and suggested that his results were due to the different times that sampling was done and compared rather than differences from the substrates that were compared in the study.

In addition, mean species richness between artificial substrates and macrophytes were similar. On the other hand, differences in richness of macroinvertebrates were found to decrease from vegetated areas towards a culvert in a highway crossing in two forested wetlands (King et al., 2000). Gerrish and Bristow (1979) reported varied results where a low number of macroinvertebrate species were on the *Myriophyllum exalbescens* plant than artificial (plastic) substrates while more taxa were found on *Vallisneria americana* than the artificial substrates in Lake Opinicon, Ontario.

The similarity between communities and species richness in artificial substrates and macrophytes in my study could be due to the period of time the artificial structures been in the lakes. The structures that were sampled in my study have been in the water for a long period of time (i.e., many years). These structures included walkways, retaining walls, jetties and moorings. Other studies have only sampled structures that were installed and sampled for the duration of the study to answer their research questions. These experiments were designed and sampled periodically, for instance; daily for a month, on a monthly basis for a year or so, or once after a certain number of months after installation (Aikins & Kikuchi, 2001, Uwadiae & Felix 2015, Boothroyd & Dickie, 1989). For example, Anderson et al. (2010) and Guerra-Garcia et al. (2004) sampled temporarily installed artificial structures and reported differences in invertebrate community compositions between artificial substrates and adjacent natural substrates. As artificial substrates are left to stand for lengthier periods, communities between both substrate types may then begin to become similar as stable

communities are developed after species immigration and emigration equalizes. This supports studies that who discussed that artificial structures were colonized based on their complex surface areas and fixed period in the environment allowing the structures to accumulate epiphyton biomass and other materials to support their colonizing communities as macrophytes do (King & Richardson, 2000; Saliu & Ovuorie, 2006; Meier et al. 1979).

#### *Community composition on artificial substrates versus sediments*

Differences were found in community composition between artificial substrates and the surrounding sediments. These differences can be attributed to the artificial structures supporting primarily free-living and planktonic species like *Acanthocyclops robustus*, while sediments supported infaunal species such as oligochaetes and *Chironomus zealandicus*. Zooplankton are important species in aquatic food webs (Brookes & Dodson, 1965) because they are a food source for adult and juvenile fish (Guest et al., 1990) and other larger organisms. Zooplankton are also known to control the growth of phytoplankton communities due to their grazing behaviour (Benndorf et al., 1988). Man-made constructions in freshwater systems support the growth of epiphytic algae which these planktonic communities feed on. Therefore, the planktonic species are found on the surface of these structures. On the other hand, the sediments support benthic communities include oligochaetes and chironomids. These species are usually detritivorous, consuming decaying matter as well as bacteria that live in the soil (Brinkhurst et al., 1972). Thus, these benthic macroinvertebrates process the organic matter at the bottom of freshwater systems, playing a significant role in nutrient cycling in the ecosystem (Galdean et al., 2001). In addition, other species in sediments (e.g. bivalves) contribute to organic enrichment disturbing the sediment chemistry which in turn results in changes in species diversity (Dauer, et al., 2000). This study supports those such as Casey & Kendall (1996) who also found differences in abundance and community composition of colonizing macroinvertebrates

between artificial substrates (clay bricks) and natural substrates in the McLeod River, Canada.

In addition, mean species richness between artificial substrates and sediments were similar. These results were very different other studies that were done on these two substrate type. Casey and Kendall (1997, 1996) found more taxa had colonized natural benthic substrates than artificial substrates in studies in the Battle and McLeod Rivers, Canada. On the other hand, differences in richness of macroinvertebrates were seen with higher number of taxa on artificial substrates in the Ohinemuri River, New Zealand (Boothroyd & Dickie, 1989). These varied results in taxa richness could be due to the differences in sediments that were sampled; soft-bottomed vs hard substrates (Benke et al., 1984, Collier, 2010).

Much of the data provided on comparison of the effects of artificial structures on community assemblages against natural substrates were done on marine and estuaries where many studies show species on pilings were different to those in the surrounding habitats (Glasby & Connell, 1999). For example, Glasby (1999) found that species such as polychaetes and sponges were more abundant on artificial structures than on neighbouring rocks near marinas in Sydney, Australia. Moreover, there have been studies done that looked at using artificial substrates to compare the efficiency of different sampling techniques for monitoring freshwater benthic communities (Meier et al., 1979, Lamberti & Resh, 1985, Boothroyd & Dickie, 1989, Modde & Drewes, 1990, Collier et al., 2009) while others carried out experiments that observed different communities that colonize different types of artificial substrate types (Saliu & Ovuorie, 2006, McCabe & Gotelli, 2003). This demands for further research to be undertaken to investigate community differences between artificial substrates and sediments near them in freshwater systems. Most of the lakes that I sampled were soft-bottomed and were found to have similar communities with artificial substrates. More investigation can be done to find out if invertebrate communities are similar to artificial substrates if the lakes or ponds have hard substrata. A study by Benke et al. (1984) reported that taxa were more diverse on snags then soft-bottomed sediment types where the sandy substrates consisted of mostly midges and while a lot of

oligochaetes in muddy substrates where collected in a subtropical river in USA. Further, a study that looked at introduced stones into a lake showed an increased abundance in crayfish in these sites than the control sites with soft-bottomed substrates (Johnsen & Taugbol, 2008). Differences between different structural characteristics of substrate types may support different community assemblages.

#### *Community composition on macrophytes versus sediments*

Differences were also observed in communities between macrophytes and sediments. As discussed above, the difference in these substrates contribute to the dissimilarities between the species that inhabit them. Macrophytes encourage epiphytic algae that is used as a food source for grazers. In addition, these plants offer a refuge from predators and available space away from competition (Becerra-Muñoz & Schramm, 2007). Thus, planktonic species like *Acanthocyclops robustus*, *Piona uncata exigua* and *Simocephalus vetulus* were found on macrophytes. Burrowers and detritivorous invertebrates such as oligochaetes are usually associated with sediments ecosystems. Sediments contain decaying organic matter that these species feed on and play an immense role in nutrient cycling due to their feeding behaviour (Brinkhurst et al., 1972). A study done by Findlay et al. (1989) found that the colonizing abundance of epiphytic invertebrates on water chestnuts was higher than those in sediments in the Hudson River, New York. This emphasizes the difference in the supported communities and the importance of macrophytes to provide habitats with large surface areas for large numbers of invertebrates to use (Soszka, 1975).

In addition, there was a difference observed in species richness between macrophytes and sediments where the mean taxa richness was higher on macrophytes than sediments. This result supports studies that show more species richness in macrophytes than in benthic substrates. Phillips (2003) found that macroinvertebrate taxa richness was higher than those found in bedrock. The same results were reported by Shupryt & Stelzer (2009) who also recorded higher species richness in plant beds than benthic substrates

(sand/gravel) and Thorp et al., (1997) who found more taxa in vegetated areas than in habitats without vegetation. The high species richness of invertebrates found on macrophytes is attributed to the epiphytes that use the plants to grow on and provide a food source for these invertebrates (Riis & Biggs, 2003). Also, the high surface area provided by macrophytes allows invertebrate species to hide from predators (Warfe & Barmuta, 2006) and utilize surfaces for colonization (Huryn & Wallace, 2000).

#### *Non-native invertebrate species*

There were four non-native species that were collected from the study sites. These included a copepod, a gastropod and two leech species. There were no similarities seen in the abundance of these non-native invertebrate species between the substrates; macrophytes, sediments and artificial structures. This indicated that the non-native species don't prefer a habitat type.

My results did not produce the same results as other studies done in marine environments that determined that artificial structures facilitate invasions by non-native species (Page et al., 2007; Bulleri & Airoidi, 2005). The presence of artificial structures such as walkways, boat ramps and retaining walls constructed in aquatic ecosystems change the environmental conditions and habitats while creating new, suitable habitats for exotic species to thrive (Alroidi & Bulleri, 2005). In the same study by Alroidi and Bulleri (2005), they reasoned that artificial walls had supported an invasive alga to prevail and spread. Glasby et al. (2007) also found that there were more non-native species found on pontoons and pilings than on nearby natural habitats in Sydney Harbour, Australia.

These studies have suggested that the invasions could be due to the man-made constructions installed in areas that have changed the natural substrates and introduced new hard surfaces which make these ecosystems vulnerable to biological invasions. In marine areas, studies have shown biological invasions are mainly associated with shipping



activities where non-native species are carried on hulls or ballast water to new areas where they establish themselves on hard substrata such as pontoons and pilings (Glasby et al., 2007, Lewis et al., 2004). The same has been reported for the Rhine River in Europe. This huge river and its tributaries are connected to the Caspian, European seas and Atlantic Ocean by canals that ships use. This network to access ports has made the introduction and spread of 45 non-native macroinvertebrate species possible in the Rhine River (Leuven et al., 2009). In another example, the golden mussel (*Limnoperna fortunei*) is native to rivers in China. This species has now spread to South America via attachment to ships. The mussel was then dispersed to large rivers in South America through human activities like fishing or water currents (Darrigran, 2002). Interestingly, a study on the invasive amphipod, *Dikerogammarus villosus*, in the Schipbeek River, Netherlands, reported that this species spread and became abundant because of fish passages. Fish passages were made using concrete structures that were found to facilitate the spread of the amphipod. Most introductions of non-native species as such discussed above have been accidental while some are intentional (Ricciardi & Maclsaac, 2000). For example, the mysid shrimp, *Hemimysis anomala*, was one of the crustacean species that were deliberately put into lakes and reservoirs to increase fish production in Eastern Europe in the 20<sup>th</sup> century (Marty, et al., 2010). This species is now so widely spread that it is an invasive species in the Great Lakes in North America as well (Audzijonyte & Vainola, 2008).

In this study, non-native species were found in very low abundances. Therefore, no other statistical tests were performed to make any more inferences about substrate preferences and explain why these results are different to many other studies that agree non-native species are found on artificial substrates. It may be possible that the non-native species found in this study may not be in competition with or predate heavily on native species to cause a preference in substrate use and that all sampled substrates are suitable for the recruitment and proliferation of non-native and native invertebrates species alike.

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## **Chapter 3**

# **Environmental variables influencing macroinvertebrate taxa in lakes and ponds in the Auckland, Waikato and Bay of Plenty Regions, North Island**



## Introduction

Human activities have long affected freshwater ecosystems. Agricultural and farming practices (Matson et al., 1997), industrial wastes, climate change and habitat degradation are some of the activities known to cause changes in the physical, chemical and ecological attributes of freshwater ecosystems (Rapport & Whitford, 1999). In freshwater systems, macroinvertebrates are sensitive to changes in environmental conditions (Verdonschot, 1990). This has led to macroinvertebrates being utilised as bioindicators of the impacts of anthropogenic activities on water quality (Rossaro & Pietrangelo, 1993).

Changes in freshwater systems are mostly quantified using climatic, geographical and other environmental variables that influence species assemblages. Some of the environmental variables that affect aquatic macroinvertebrate species assemblages include water temperature (Jacobsen et al., 1997), sediment composition (Miserendino 2001), availability of food (Dobson, 1999), pH (Townsend et al., 1984), conductivity, algal concentrations, nutrient levels (total nitrogen and phosphorus) and water transparency (Boieng & Ramcharan, 2003). The relationship between these factors and accompanying species assemblages has been used to understand the health of aquatic systems and ecological processes (De Shon, 1995; Resh, 1995). Physical and chemical variables in streams near urban areas, and in conserved areas, were compared by Souto et al. (2012) in Uberlândia, Brazil. Souto et al. (2002) found that factors such as dissolved oxygen, turbidity and conductivity were important variables that explained differences in macroinvertebrate distribution between urban streams and streams in conserved areas. Similar environmental variables were also found to be responsible for differences in community composition in stream sites within the Qinjiang River watershed, Guangxi, China (Zhang et al., 2014). Zhang et al. (2014) reported that dissolved oxygen, conductivity, total phosphorus as well as physical aspects (habitat quality) were important for macroinvertebrate assemblages. Conductivity, altitude and stream gradient were also found to be important for macroinvertebrates in streams in national parks and reserves in Poland (Lewin et al., 2015). Community

composition of invertebrates sampled from forty lakes sampled in central Ontario, Canada, was reported to be associated with pH (Jackson & Harvey, 1993). In New Zealand, many studies have been undertaken on the distribution of lotic macroinvertebrate communities and associated environmental factors. Some have looked at water chemical factors affecting macroinvertebrate distributions. Duggan et al. (2007), for example, compared distributions of macroinvertebrates between non-geothermal and geothermal sites and reported that distributions were associated with temperature and pH. Collier (1995) observed taxa richness across 29 streams in Northland, New Zealand, and found it to be associated with factors such as temperature and conductivity. Taxa richness was found to be negatively associated with temperature in the Fox and Waiho Rivers, South Island (Milner et al. (2002). Other studies have examined physical factors that affect macroinvertebrate communities. For instance, Quinn & Hickey (1990) surveyed benthic communities in 88 rivers throughout New Zealand and found preferences in habitat particulate sizes in different feeding guilds. For example, Naididae, and the chironomids *Eukiefferiella* sp. and *Tanytarsus vespertinus* were common on boulders and cobbles, while Orthoclaadiinae were abundant in sandy substrates (Quinn & Hickey, 1990).

Fewer studies have also been conducted on the distribution and community composition of invertebrate groups among lakes and ponds. Cladocerans and copepods were found to be correlated to macrophyte biomass in Jack Lake, Nova Scotia, Canada (Paterson, 1993). In Narrow Lake, Alberta, Canada, Hanson (1990) found a higher biomass of macroinvertebrates like chironomids and gastropods in macroalgal beds than on rooted macrophytes, which supported mostly amphipods. In New Zealand, the distribution rotifer species across the North Island was found to be primarily determined by trophic state (Duggan et al., 2001). Others have identified common taxa that are associated with macrophyte species among lakes (Biggs & Malthus, 1982; Talbot & Ward, 1987). Species composition has also been studied in benthic communities, where dominant species have been recorded. For example, molluscs and chironomids were found to be abundant in the benthos of 22 lakes in the South Island. Forsyth (1978)

surveyed and recorded benthic communities in the seven Rotorua Lakes, North Island, showing that chironomids, molluscs and oligochaetes were common invertebrates across these ponds. Similar benthic communities were recorded from Lake Taupo (Forsyth & McCallum, 1981) and in lakes in Canterbury, South Island (Timms, 1983). Weatherhead & James (2001) found that macroinvertebrate composition was related to habitat characteristics; coarse sediments, submerged macrophytes, and fine sediments, among nine lakes throughout New Zealand. Further, water chemical variables affecting macroinvertebrate distributions have also been examined. Harding (1992) reported that macroinvertebrate species composition in Lakes Kaniere, Mapourika and Wahapo, South Island, was affected by temperature in addition to other physical factors.

Research has been undertaken in New Zealand on macroinvertebrate communities in ponds to understand important variables that affect assemblages, and to understand changes that occur in freshwater systems because of anthropogenic activities such as land use. The purpose of this chapter is to assess the relationship between macroinvertebrate communities in North Island ponds containing existing man-made constructions (jetties, retaining walls, pier pilings, culverts etc.) and environmental variables.

## **Methods**

### *Sampling sites*

A total of twenty-two lakes in the Waikato, Auckland and Bay of Plenty regions, North Island, New Zealand (Table 1), which had artificial structures (e.g. jetties, retaining walls, pier pilings, culverts etc.), macrophytes and exposed bottom sediments were selected as sampling sites. Lakes and ponds were evaluated and selected using Google Maps, as well as personal knowledge, based on the presence of these substrates.

### *Invertebrate sampling*

Sampling took place in July and August 2018 (austral winter). Macroinvertebrates were collected from three substrate types; artificial substrates, macrophytes and sediments using a 450 µm dip net. Macroinvertebrates were collected by wading from the shore to a depth of one meter and sampled for three minutes from each substrate type. Contents of the net were emptied after collection from each substrate into 2 L polypropylene wide-mouth storage containers and were sorted and identified on return to the laboratory

**Table 4.** The location and altitude of lakes and ponds that were sampled in this study. The Site ID are keys to the non-metric Multidimensional Scaling plot. Lakes are listed in sampling order.

Site ID	Lake Name	Latitude	Longitude	Altitude (m)
1	Albany Lake Reserve	36°43'34.8"S	174°42'31.1"E	36
2	Lake Pupuke	36°46'37.6"S	174°46'11.5"E	9
3	Chelsea Sugar Ponds	36°49'12.6"S	174°43'20.1"E	9
4	Lake Hakanoa	37°33'05.7"S	175°09'47.9"E	10
5	Lake Moananui	38°14'03.1"S	175°51'07.0"E	336
6	Lake Ngaroto	37°57'32.5"S	175°17'26.1"E	36
7	Lake Cameron	37°51'17.2"S	175°18'07.4"E	61
8	Lake Rotokauri	37°45'51.6"S	175°11'53.6"E	47
9	Western Springs	36°52'01.2"S	174°43'31.8"E	13
10	Lake Paremuka	36°52'12.8"S	174°36'43.2"E	41
11	Lake Panorama	36°53'04.8"S	174°36'11.0"E	41
12	Van Dammes Lagoon	36°54'02.8"S	174°50'43.1"E	21
13	Edgewater Pond	37°04'40.8"S	174°55'23.2"E	16
14	Lake Gilmour	37°23'41.2"S	175°50'58.2"E	93
15	Lake Matahina	38°07'04.0"S	176°49'14.2"E	67
16	Lake McLaren	37°48'44.9"S	176°02'39.5"E	93
17	Lake Karapiro	37°55'49.4"S	175°32'32.5"E	59
18	Lake Magellan	37°43'45.5"S	175°14'54.2"E	40
19	Lake Rotoroa	37°47'52.8"S	175°16'41.3"E	38
20	Lake Puketirini	37°34'06.6"S	175°08'03.5"E	10
21	Chapel Lake	37°47'17.4"S	175°18'53.2"E	51
22	Oranga Lake	37°47'14.0"S	175°18'58.3"E	50

### *Environmental data*

For total nitrogen and phosphorus, water samples were collected in 50 mL containers. These were frozen and analysed within a month of collection. For chlorophyll *a* analysis, a syringe was used to collect 50-60 mL of water filtered through a 0.2 µm Whatman GF/C glass microfiber filter at low vacuum. Each filter was then folded in half with the residue placed inwards. These were then wrapped in foil to prevent exposure to sunlight and placed in a chilly bin containing ice before return to the laboratory, and then frozen until chlorophyll *a* analysis was undertaken. Dissolved oxygen, pH, temperature and conductivity measurements were obtained using Yellow Springs Instruments (YSI) Model 650 Multiparameter Display System (MDS). Measurement of water transparency was done using a 25 cm Secchi disk.

### *Laboratory analyses*

For chlorophyll *a* analysis, extraction of chlorophyll was undertaken by grinding the filters in buffered acetone in 15 mL centrifuge tubes using a D-Lab: D160 Handheld Homogenizer at low speed. Each centrifuge tube was then shaken and left to rest in the dark for 24 hours at 4 °C. These were shaken once during this time. After 24 hours, the tubes were shaken again vigorously and centrifuged for 10 minutes at 1647 G at high brake using Hettich Universal 320R centrifuge. They were left to stand in the dark for 30 minutes at room temperature before reading were taken using Turner Designs 10-AU fluorometer and chlorophyll *a* was calculated using pre-programmed MS Excel® spreadsheet. Total nitrogen and total phosphorus samples were analysed by RJ Hill Laboratories Limited. Nitrate-N and Nitrite-N were measured using the Automated Cadmium Reduction Test (APHA 4500-NO<sub>3</sub>- I) and TKN was calculated by Flow Injection Analysis (APHA 4500-N<sub>org</sub> D. and the Phenate Method 4500 NH<sub>3</sub> F). Total nitrogen was calculated for each lake summing Total Kjeldahl Nitrogen (TKN), Nitrate-N and Nitrite-N. Total phosphorus was measured using the Ascorbic Method (APHA 4500-P B & E).

### *Statistical analyses*

Canonical Correspondence Analysis (CCA) was undertaken to identify important environmental variables, physical and chemical, that may be important for variability of invertebrate communities among the sampled ponds. CCA was performed on measured environmental data and the averages for all species composition data across three sampled substrates (artificial, sediments and macrophytes), which were sampled for three minutes each, totalling nine minutes of sampling effort per pond.

For abundant species data, the influence of rare species and species that may have been sampled by chance were reduced by excluding taxa that contributed <10% of total abundance and sites containing <3 individuals from the analysis. These left 28 species (from 43 total taxa) that were included in the analyses. In cases where data were not normally distributed, these were transformed using  $\log(x + 1)$  transformations to reduce any effect from species that may be rare. Data for environmental variables was standardized to zero mean and unit variance by determining the means and standard deviation for all environmental variables and then subtracting each data point with its mean and dividing by its standard deviation. This was done for temperature, Secchi depth, chlorophyll a, total nitrogen and total phosphorus measurements. Oxygen saturation (in %) was not included in the CCA as it was highly correlated to DO concentration. Altitude also was not included in CCA.

## Results

### *Invertebrate abundances*

The total of 25 911 invertebrate individuals were collected, with 43 invertebrate taxa identified across all substrate types. Of these taxa, 35 taxa were found on artificial substrates, 24 in sediments and 34 on macrophytes. A mean of 4.7 taxa was found in sediments, a mean of 7.4 taxa on macrophytes and a mean of 5.9 taxa was found on artificial substrates.

### *Multivariate Analyses*

*Macrocyclus albidus*, *Paracalliope fluviatilis* and *Cura* sp. were most strongly positively associated with Axis 1, and *Sigara* sp., *Neppia* sp. and *Antiporus strigosulus* were strongly negatively associated. *Ilyocryptus sordidus*, *Anisops wakefieldi* and *Simocephalus vetulus* were positively associated with Axis 2, and *Sphaeriidae* sp., *Polypedilum* sp. and *Oxyethira albiceps* were negatively associated. Temperature, conductivity and latitude were strongly positively associated with Axis 1. Total phosphorus and total nitrogen were strongly positively associated with Axis 2, and pH and DO were strongly negatively associated. Overall, Axis 1 is influenced by a temperature gradient and Axis 2 by a productivity gradient. Results of forward selection and Monte Carlo permutation tests from CCA indicate that temperature explained the largest proportion of variation in species composition ( $\lambda A = 0.17$ ,  $p = 0.02$ ) and total nitrogen and pH explained any significant amount of the remainder of variation ( $\lambda A = 0.15$  and  $0.12$ ,  $p = 0.03$  and  $0.09$ , respectively).

Comparisons of species composition with the environmental variables showed species such as the copepod *M. albidus* and amphipod *P. fluviatilis* were associated with warmer lakes and high conductivity, while *Sigara* sp. and *Neppia* sp. were associated with colder waters. Species such as the cladoceran *I. sordidus* and back swimmer *A. wakefieldii* were associated with high nutrient levels (total phosphorus and nitrogen) while *Sphaeriidae*

sp. was strongly associated with low levels of nutrients. *Physa acuta* and *Potamopyrgus antipodarum* were associated with high levels of DO and lakes that were alkaline.

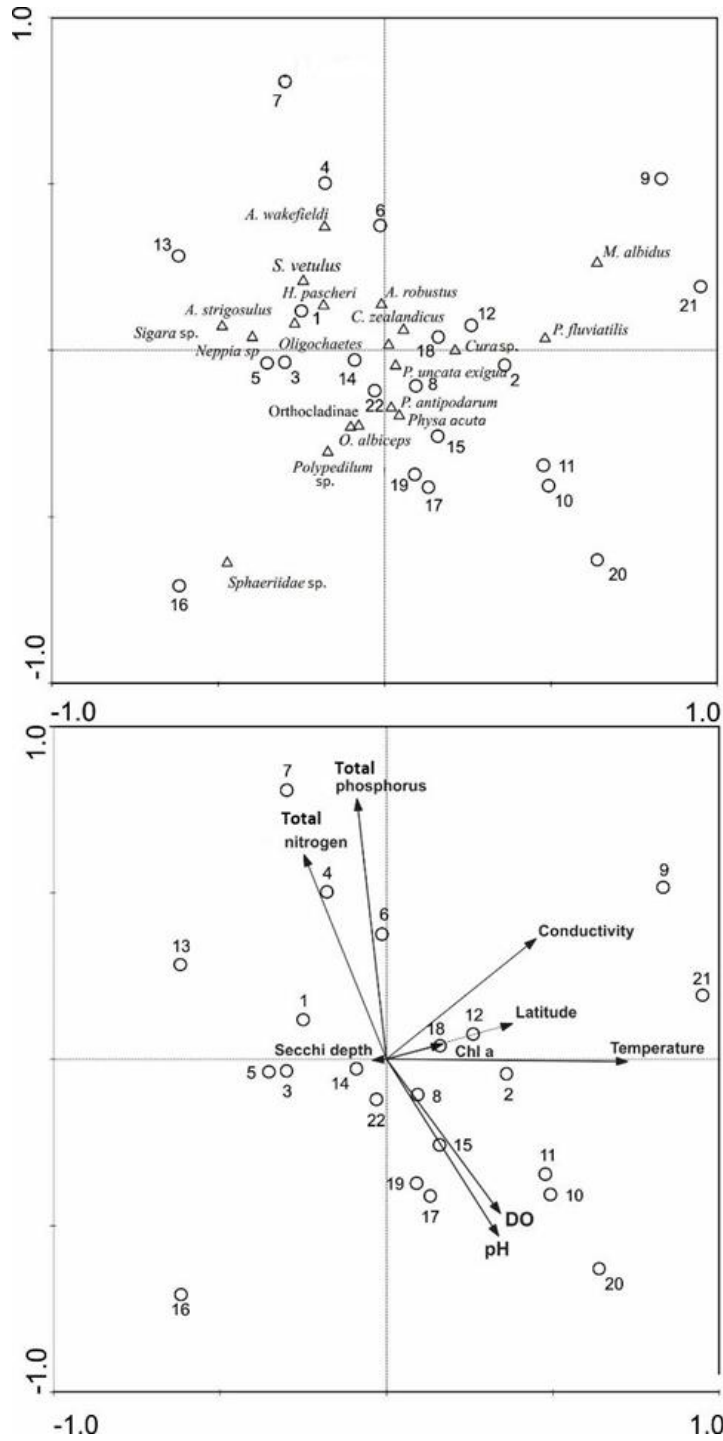
**Table 5** The mean and range of physico-chemical values calculated from readings taken from all 22 sampling lakes and ponds.

Physico-chemical factor	Mean	Range
Temperature (°C)	12.45	10.27 - 17.10
pH	6.74	5.13 - 8.20
DO (mg/L)	15.62	7.61 - 98.60
Secchi depth (m)	1.00	0.22 -2.62
Specific conductivity (µS/cm)	185.41	65 - 375
Chlorophyll <i>a</i> (ug/l)	26.55	1.72 - 102.6
Total nitrogen (g/m <sup>3</sup> )	2.13	0.3 - 16.8
Total phosphorus (g/m <sup>3</sup> )	0.08	0.012 - 0.600

**Table 6** Results of forward selection and Monte Carlo permutation tests from canonical correspondence analysis (CCA) environmental variables in 22 ponds sampled in the Waikato, Auckland Region and Bay of Plenty macroinvertebrate species. Lambda 1 values represent the variation associated with each environmental variable when variables are treated separately. Lambda A shows the list of the environmental variables in the order of inclusion into the CCA model with the important variable listed first. *P*-values indicate the significance of each environmental variable at time of inclusion.

	Lambda 1	Lambda A	<i>P</i>
Temperature	0.17	0.17	0.020
Total nitrogen	0.15	0.15	0.030
pH	0.15	0.12	0.094
Chl <i>a</i>	0.15	0.10	0.178
Total phosphorus	0.13	0.11	0.208
Latitude	0.13	0.08	0.314
Conductivity	0.11	0.09	0.318
DO	0.09	0.10	0.168
Secchi depth	0.08	0.06	0.560





**Figure 5** Ordination biplots based on canonical correspondence analysis (CCA) of Waikato and Auckland region (North Island, New Zealand) invertebrate species with respect to environmental variables. Eigenvalues for the first two axes are 0.222 and 0.204.

## Discussion

The most important factors that contributed to the distribution of macroinvertebrate species assemblages in the lakes sampled in the North Island, New Zealand, were temperature and productivity gradients. In aquatic ecosystems, temperature is an important factor that influences macroinvertebrate communities. Temperature has been shown to affect metabolic activities (Knight & Gauffin, 1963), growth and development rates of aquatic insects (Voelz et al., 1994; Chadwick & Feminella, 2001), as well as influence primary productivity within freshwater systems (Richardson, 1992). In my study, the copepod *Macrocyclus albidus* and the amphipod *Paracalliope fluviatilis* were associated with warmer temperatures, higher than the mean temperatures of 12.4°C recorded among the lakes (Table 2). These results could to some degree be due to the temperature measurements being a factor of time. Many lakes on the left of the CCA plot (Figure 1) were sampled during colder days, while the lakes on the right of the ordination were sampled during warmer days. Warmer lakes associated with the collected copepods and amphipods were Chapel Lake, Western Springs and Van Damms Lagoon. These ponds have a depth of 1.8 meters. These lakes are shallow and so mix well and are productive. These species would be associated with such lakes that have high algal growths (Wissel & Ramacharan, 2003). In the current study, sampling was done in austral winter with temperatures ranging from 10 - 17°C, it is sensible that *M. albidus* and *P. fluviatilis* were collected from lakes that were sampled on warm days with temperatures ranging from 11-17°C. Warmer lakes usually allow the proliferation of periphytic algae (primary production) (Schindler et al., 1996), which is important for planktonic species as food sources. Temperature is also known to influence survival rates in copepods. For example, the survival of *Pseudodiaptomus pelagicus* from early nauplii to adult was highest between 28°C and lowest between 32 - 34°C. Temperatures of 20°C was observed to be optimum for reproductive stages of the amphipod *Hyaella azteca* in two Ontario lakes in Canada (Panov & McQueen, 1998). It is commonly reported that temperature affects species distribution (Hogg & Williams, 1996; Daufresne et al., 2004). Kamps (1978) reported that development of the copepod *Skistodiaptomus pallidus* was affected by temperature variations where at each development stage,

broods were smaller with increasing temperatures. In addition, Abdullahi (1990) also found that as experimental temperatures were, clutch size and embryo development time, as well as other reproductive aspects, were seen to respond inversely in *Megacyclops viridis*, *Macrocyclus albidus* and *Acanthocyclops vernalis*. Temperature is also an important factor for growth rates in copepods as temperature affects their respiration. Therefore, temperature is an important factor for the copepod and amphipod collected from warm lakes in the current study.

On the other hand, the water boatman *Sigara* sp. and the flatworm *Neppia* sp. were associated with colder lakes, sampled on days that had temperatures ranging from 10-13°C. Water boatmen are mostly found in shallow water and are associated with plants where they feed on algae and detritus, while *Neppia* sp. are sediment dwelling species that prey on other invertebrates such as snails. Sweeney & Schnack (1977) investigated the reproduction of *Sigara alternata* in an experiment and found that the optimum temperature range for egg laying was 11 – 12 °C. A near similar result was found by Barahona et al. (2005) in *Sigara selecta*, where their egg laying was prompted at 10°C. In addition, the development and growth of nymphs were negatively correlated to increasing temperature in *Sigara selecta* at experimental temperatures up to 26°C (Barahona et al., 2005). Temperature is also observed to be important for dispersal of flying water bugs like Hemipterans. Boda and Csabai (2009) did an observation of dispersal activities of *Sigara lateralis* in Hagyma's basin marsh, Hungary, and found that dispersal was higher in warmer days mostly in the months of June and July (summer). Adult *Sigara lateralis* was not collected in traps at temperatures below 12°C (Boda & Csabai, 2009). Overall, this indicates that the distribution of *Sigara* sp. in my study may have been limited by temperature where larvae are found in cooler waters and emerge and move away from ponds to terrestrial ecosystems when waters were warmer (Coccia et al., 2013). *Neppia* sp. were also collected in cooler lakes. Temperature is important in the distribution, behaviour and ecology of aquatic flatworms. In an experiment that tested the reactions of planarians to changes in temperature changes, Mast (1903) reported that *Planaria dorotocephala* responded to warm thermal stimuli by moving away from it,

and showed lower responses to colder thermal stimuli. In addition, *P. dorotocephala* was also found in temperatures as low as 10°C (Mast, 1903). This is similar to my study where *Neppia* sp. was collected in lakes with temperatures between 10-13°C. Claussen et al. (2003) found that the locomotion *Planaria dorotocephala* was affected by temperature changes where movement became quicker at temperatures starting at 10°C but slowed once temperature was raised to 30°C. Besides temperature, the CCA plot also shows that latitude is positively correlated to axis 1, enough to explain community differences. In this study, the differences in water surface temperature between lakes were related to latitude.

Results for this study also show that the cladocerans *Ilyocryptus sordidus* and *Simocephalus vetulus* were associated with high nutrient levels (total phosphorus and nitrogen), and low DO and pH. As such, cladoceran species were found to be associated with lakes that had higher algal concentrations due to high total nitrogen and phosphorus levels. Similar results were reported from data analysed by Gyllstrom et al. (2005) from 81 lakes across Europe, which showed that increases of TP were correlated with increases in biomass of cladocerans. Phosphorus and nitrogen are important for algal growth, which are food sources for zooplankton for the production of amino and nucleic acids and proteins which are essential for growth and reproduction (Fileto et al., 2010). Cladocerans are important filter feeders who selectively feed on particles such as algae, bacteria and detritus. In my study, high abundances of cladocerans appear to be associated with peat lakes (e.g., Lake Hakanoa, Lake Ngaroto and Lake Cameron) that are characterized by high productivity, more acidic water and low levels of dissolved oxygen. The cladocerans *I. sordidus* and *S. vetulus* are species primarily associated with macrophytes and sediments (Frey, 1987; Lauridsen et al., 1996). Peat lakes have high organic matter that may encourage algal growth, detritus and suspended particles that are food source for these invertebrates (Dole-Olivier et al., 2000; Hutchinson 1967). High nutrient run off from agricultural activities near Lake Vela, Portugal, encouraged algal blooms (mostly Cyanobacteria), which showed an increase in species density of *Ceriodaphnia pulchella* but a decrease in the abundance of *Daphnia longispina* (Abrantes et al., 2006). This mostly

indicates that larger cladocerans have a limited food source (specific algae they feed on) while it is the opposite for smaller crustaceans having been able to effectively capture and consume bacteria and so they increase in abundance with increased nutrient loads (Sommer et al., 1986) and using bacteria as a food source as well (Gliwicz, 1969). Cladocerans obtain these nutrients from algae and the importance of obtaining fatty acids in their diet was shown in a feeding and growth experiment by Demot & Muller-Navarra (1997) who reported *Daphnia gessneri*, collected from Lake Monte Alegre, Brazil, did not grow well and had small clutch sizes when they were fed cultured alga *Scenedesmus spinosus* that was deficient in nitrogen and phosphorus. An experiment by Demot & Muller-Navarra (1997) also reported *Ceriodaphnia cornuta* had low clutch sizes when fed *Scenedesmus spinosus* that was nitrogen deficient. The results of the current study show that productivity explains much of the variation in community composition. Despite total phosphorus and total nitrogen affecting the distribution of the cladocerans collected, chlorophyll *a* levels did not coincide with these increased nutrient levels. Similar results were observed by Filstrup & Downing (2017) from 139 lakes in Iowa, USA, where 107 of these lakes were man-made. Increasing TP and TN resulted in low chlorophyll *a* levels prompting four hypothesis to explain this pattern. Filstrup & Downing (2017) suggested a zooplankton grazing hypothesis as one of the four hypothesis that explained the high TP and TN, and low chlorophyll *a* pattern. This hypothesis could also explain why chlorophyll *a* was not related to increased productivity in my study. Cladocerans were associated with increased productivity levels in ponds I sampled; grazing by these could have reduced algal biomass resulting in low chlorophyll *a* levels.

*Physa acuta* and *Potamopyrgus antipodarum* were found in alkaline lakes with moderately high levels of DO. These species were most greatly associated with Lake Puketirini and Lake Paremuka, with pH measurements 8.2 and 7.0, respectively. Acidic lakes have been demonstrated to correlate with low numbers of snails and phytoplankton (Okland, 1992). In addition, pH conditions have been shown to influence detrital matter (Frieberg, 1980) and algal growth (Hall, 1980), which are important for these grazers. Thus, as periphytic algae increases with pH, so will snail populations. Acidification

of freshwater environments through human activities into naturally acidic systems can occur from chemicals and fertilizers that are used in agricultural activities which run off into these ecosystems as well as through atmospheric CO<sub>2</sub>. Many studies have examined levels of acidification that affects invertebrate communities. It has been shown that some gastropods such as *Lymnea peregra* and *Planorbis acronicus* were sampled in abundance in waters of pH greater than 5.5 (Raddum & Fjellheim, 1984), which is similar to the results obtained in my study. On the other hand, Raddum & Fjellheim (1984) sampled freshwater benthic invertebrates in Norway and reported that most species were found in a pH range of 4.5 and 7.0. In my study, no snails were collected in Lake Cameron, although it had a high abundance of *Chironomus zealandicus*, and the cladocerans *Ilyocryptus sordidus* and *Simocephalus vetulus*. The pH in Lake Cameron was the lowest of all lakes (5.13) and was associated with high nutrient levels. Although pH did not differ between sampled lakes ( $p=0.09$ ), CCA results show that pH was the third environmental factor (after temperature and nutrient levels) that explained remaining variation of species composition in this study. This could be because Lake Cameron was the only lake that was the more acidic ( $pH < 5$ ) compared to other lakes that were sampled in this study. Thus, pH levels can affect species assemblages of snails. Acidic freshwater systems have been shown to affect the shells of gastropods when carbonate that is used to build shells is lost after hydrogen ions, from carbonic acid formed from CO<sub>2</sub> and water, bond with existing carbonate to form more bicarbonate in the water (Feely et al., 2012). This rise in acidification can cause gastropod shells to corrode (Fabry et al., 2008). Hoverman et al. (2011) reported that pH was one of the important factors that affected snail species richness was higher in ponds with alkaline conditions. In a study by Spyra (2017) in forest ponds in Poland, most gastropods were collected in ponds with pH levels  $>6.0$ . *Physa acuta* is characteristic of alkaline ponds with pH between 7.7 and 9.0 (Spyra, 2017). Ntonifor & Ajayi (2007) sampled ponds in Central Nigeria and found snails in ponds with pH with a range of 7-10.9. The results of my study are similar to other studies where *Physa acuta* and *Potamopyrgus antipodarum* were found was found in alkaline water bodies. Shield et al. (2014) found these two gastropods (*P. acuta* and *P. antipodarum*) in streams with pH greater

than 7.0 in the Georges River catchment, Sydney. These snails physiologically tolerate alkaline lakes that support the growth of algae that serve as food source for the snails to graze on.

Dissolved oxygen (DO) was also closely associated with the lakes where the snails *Physa acuta* and *Potamopyrgus antipodarum* were collected in my study. The lakes where this species were found had DO measured between 14 – 16 mg/L. DO is important for molluscs and influences species composition and abundances (Harman & Berg, 1971). Snails do better in ponds with increased DO levels. This is conducive for respiration, reproduction and food source availability. The snails *Succinea cleopatra*, *Lanistes carinatus*, *Lymnaea natalensis* and *Cleopatra bulimoides* were found in high abundances in the River Nile and its branches (EL-Khayat, et al., 2017). CCA results in the study by EL-Khayat et al. (2017) indicated that differences in abundance between sampling sites could be explained by dissolved oxygen (range: 2 – 6 mg/L) to describe the relationship between snail and environmental factors. In an experiment on the juveniles of the freshwater clam *Elliptio complanata* reported behaviour exhibited by the juvenile clams indicating stress at DO levels of 2 and 4 mg/L compared to when they were exposed to DO levels of 8mg/L (Sparks & Strayer, 1998). In the current study, the fingernail clams *Sphaeriidae* sp. were collected from Lake McLaren and Lake Rotoroa. Lake Rotoroa (Site 19) was associated with slightly increased DO levels.

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## **Chapter 4**

### **Thesis Summary and Conclusion**

The introduction of artificial structures in marine and freshwater ecosystems has been shown to cause changes to the physical and biological characteristics of these areas, including sediment changes, changes in community composition and support of non-native species (Bulleri & Airoldi, 2005). In my thesis, I investigated; i) invertebrates on artificial structures in lakes are different from those on natural substrates; ii) the important environmental variables that influence the distribution and composition of sampled macroinvertebrate communities among ponds in lakes in the Auckland, Waikato and Bay of Plenty Regions.

In Chapter 2 of my research, I quantified the community composition and distribution of macroinvertebrates on sediments, macrophytes and artificial substrates across twenty-two ponds. I found that there were similarities in species richness and composition between artificial substrates and macrophytes, but differences in species richness and community composition between sediments and macrophytes and artificial substrates and sediments. The similarities between artificial substrates and macrophytes can be attributed to the habitat structure, i.e., hard surfaces and the provision of refugia for macroinvertebrates from their predators. The differences between sediments and macrophytes were attributed to the difference in habitat structure and the fauna they supported (i.e., sediments contained large proportions of infauna). In addition, non-native species that were collected in my study did not show preference for any of the habitats that were sampled.

In Chapter 3, I identified important environmental factors that affect the distribution of macroinvertebrates communities across the twenty-two ponds that were sampled. The most important factor identified was temperature, showing community differences between warm and cold lakes. The copepod *Macrocyclus albidus* and amphipod *Paracalliope fluviatilis* were associated with warm lakes while water boatman *Sigara* sp. and the flatworm *Neppia* sp. were found in cooler lakes. Nutrient gradients also explained a significant proportion of the variation in community composition among ponds. The cladocerans *Ilyocryptus sordidus* and *Simocephalus vetulus* were found to be associated with high nutrient levels.

Snails (*Physa acuta* and *Potamopyrgus antipodarum*) were found to be associated with high pH and dissolved oxygen levels.

Conclusions drawn from both these chapters are that habitat structure is important in habitat preferences for macroinvertebrate communities in lakes. Environmental factors such as temperature, nutrients, pH and dissolved oxygen are important in the distribution of macroinvertebrate communities as they are linked the ecology of these species.



## Future Research

The results of my study revealed that existing artificial structures in lakes have similar numbers of macroinvertebrate taxa and community composition with macrophytes. This shows how long-standing man-made constructions are able to support macroinvertebrate communities and the importance of artificial structures in lakes which may not have macrophytes growing in them.

Further study could be done to explore the macroinvertebrate communities in lakes with and without macrophytes, and artificial substrates. In addition, other variables such as macrophyte type and cover, characterisation of sediments and estimation of when the man-made structures were built in the lakes could also be included in further research to understand the interactions between invertebrate communities and habitat preferences among natural and existing artificial substrates.

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