THE IMPACT OF HEAVY METALS ON BENTHIC MACROINVERTEBRATE COMMUNITIES IN CHRISTCHURCH'S URBAN WATERWAYS.

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Abstract

The urbanisation of a catchment results in substantial changes to associated waterways. These effects, known as the “urban stream syndrome” can include flashier hydrographs due to stormwater inflows, altered geomorphology, and increased inputs of sediment, nutrients, and toxicants. Metal pollution of rivers and streams is an area of significant concern for management of freshwaters, and urban runoff is recognised as an increasingly relevant source of metals. Heavy metals can to be toxic to aquatic invertebrates, and can impact community structure and abundance. To investigate the influence of heavy metals on macroinvertebrate community composition, I compared invertebrate community composition over a gradient of heavy metal pollution within Christchurch City’s urban waterways. I also investigated the survival of three taxa, the mayfly *Deleatidium* spp., the caddisfly *Pycnocentria* spp., and the snail *Potamopyrgus antipodarum* in short-term *in situ* mesocosm experiments in six streams of varying metal contamination. CCA analysis identified that sediment bound metals, dissolved metals, and impervious surface area were the three most significant environmental factors explaining invertebrate community structure. Stepwise regression analysis of invertebrate community metrics and indices identified metals bound to the sediment to be among the prevailing factors in explaining invertebrate community composition across my study sites. The results of my mesocosm experiments suggest that heavy metal contamination could be rendering more impacted streams uninhabitable to relatively sensitive taxa (such as *Deleatidium* and *Pycnocentria*). However, over the seven day time frame of my mesocosm experiment, conditions in moderately polluted streams did not appear to directly affect survival of *Deleatidium* significantly more than conditions in streams containing natural populations of the mayfly. Knowledge of relevant stressors is key to the management and rehabilitation of urban streams. My results suggest that heavy metals are likely a key stressor on many invertebrate communities in Christchurch’s urban waterways. While rehabilitation of streams in Christchurch’s heavily urbanised areas can improve attractiveness and societal value, unless stormwater inputs and associated pollutants are mitigated an improvement in biological communities seems unlikely.

CHAPTER 1

URBANISATION AND WATERWAYS

1.1 - Urban growth in New Zealand

In 2008, for the first time ever, more than half of the world's population lived in urban centres. In the next 50 years it is predicted that 95% of the net global population increase will occur in cities in the developing world (Heilig 2012).

In New Zealand in the late 1800's, just under 60% of New Zealand's population lived in rural areas (Statistics New Zealand 2009). Over the following century however, New Zealand underwent a transformation from an agrarian population to an urbanised one (Figure 1.1), with 85% of the population now residing in urban areas, and less than 10% of the New Zealand work force now involved in fishing, forestry and agriculture (Statistics New Zealand 2009). This shift is demonstrated by the contrast in rural vs urban population growth between 1881 and 2001; rural area populations have increased by 83% in this period, while urban area populations have increased by 1500% (Statistics New Zealand 2009).

Due to their historical use for exploration and transport by both Maori and early European settlers, several New Zealand settlements were constructed around rivers (Young 2012). Notable examples of significant urban waterways in New Zealand are Whanganui with the Whanganui River, Hamilton with the Waikato River, and Christchurch with the Avon and Heathcote Rivers.

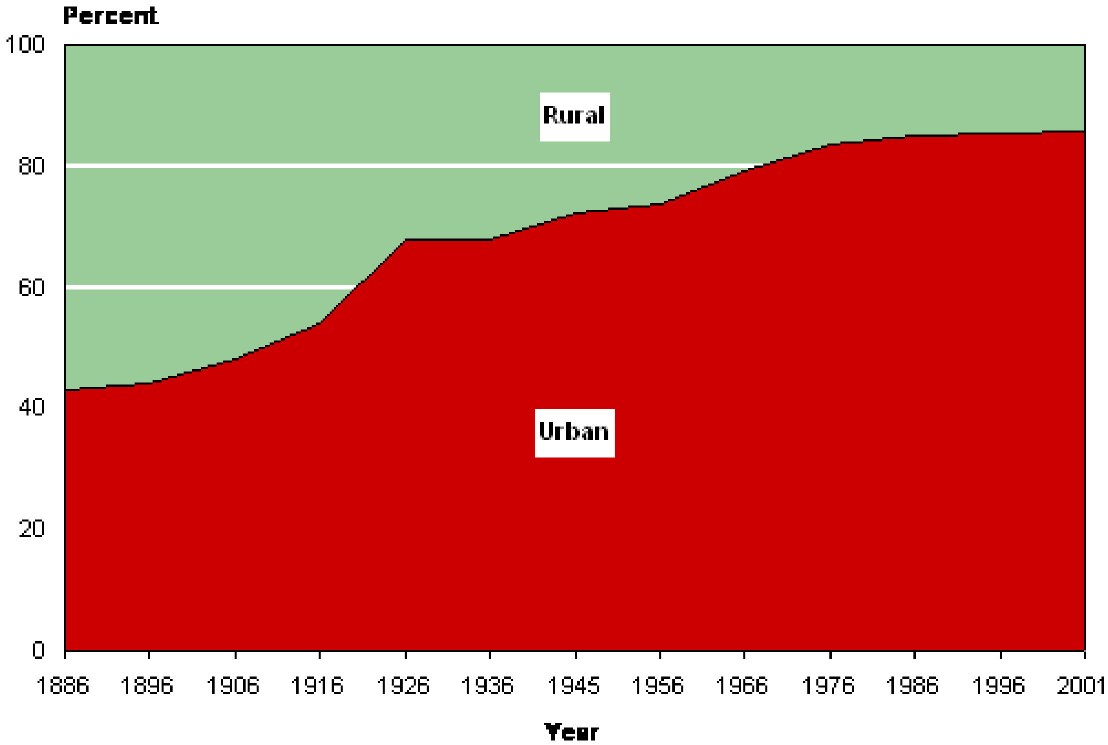


Figure 1.1. Proportion of New Zealanders living in rural and urban areas. (Statistics New Zealand 2009)

1.2 - Effects of urbanisation on waterways

The impacts of urbanisation on waterways have been studied globally (Meyer et al. 2005; Suren et al. 2005a; Wenger et al. 2009). Urbanisation of a catchment affects waterways in a range of ways, including increased hydrological variability, increased inputs of contaminants, and a decrease of biotic diversity and ecosystem health (Paul et al. 2001; Walsh et al. 2005b). These multiple effects have resulted in urban waterways suffering from what has been termed "the urban stream syndrome" (Meyer et al. 2005; Walsh et al. 2005b).

Much of the effects of urbanisation on waterways can be attributed to increases in impervious surface cover. For example, increases in impervious surfaces such as roofs, carparks, and roads can have significant effects on the water cycle (Marsalek et al. 2008). In a forested catchment, the majority of water entering waterways is via either subsurface flow, or percolation into deeper soils as groundwater (Lull et al. 1966; Leopold 1968). An increased proportion of impervious surface cover in a catchment results in an increased volume of water entering waterways via overland flow and a significantly smaller volume infiltrating the soil during rainfall events (Walsh 2004a) (Figure 1.2). As a result, streams draining largely impervious catchments are often characterised as having an increased frequency of erosive flows, increased magnitudes of high flows, higher peaks of storm hydrographs, and decreased lag times between rainfall events and peak flow (Walsh et al. 2005b).

The connectivity of impervious catchment areas and streams is also often increased by storm water systems (Walsh 2004a). Conventional stormwater drainage systems are designed to prevent flooding and waterlogging of foundations in constructions. However, they commonly discharge this water via pipes directly into streams. In addition to the hydrological changes resulting from these direct storm water connections and the substantial increase in overland flow in urbanised areas, these changes often result in an increase in sediment and pollutant input into waterways (Leopold 1968).

Increased sediment input, flash flooding, and a lack of riparian vegetation often seen in urbanised catchments can result in changes in the geomorphology of urban streams. While many urban stream channels are deliberately augmented by channelisation, concreting, and battering of stream banks for drainage, transport, and flood management purposes, stream channel dimensions are also altered over time by long term changes in sediment supply and bankfull discharge (Leopold 1968; Hammer 1972; Brookes et al. 1983). As a result, urban waterways are often found to have increased channel width, pool depth, incidence of scour, and decreased channel complexity (Walsh et al. 2005b; Violin et al. 2011).

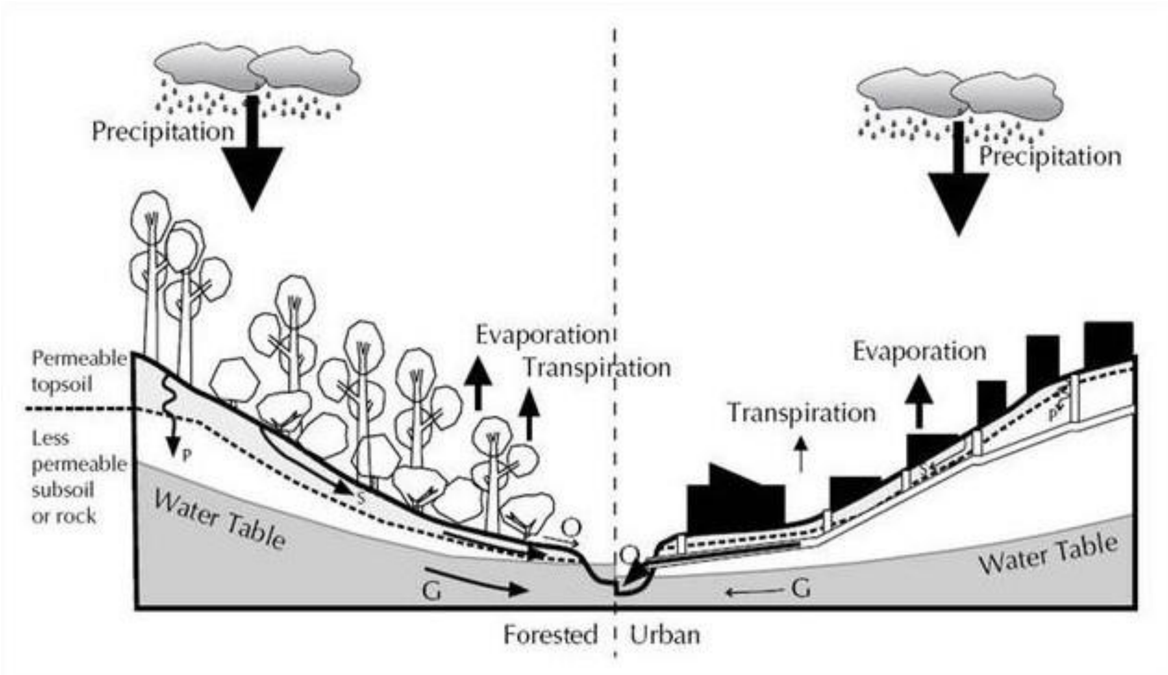


Figure 1.2. Increased impervious surface cover associated with urbanisation influences overland flow (O), subsurface flow through permeable topsoil (S), and percolation (P) into groundwater flow (G) (Leopold et al. 1978; Walsh 2004a).

Urbanisation of a catchment can also result in significant changes in stream water chemistry. Increases in water temperature, nutrients, pathogens, heavy metals, hydrocarbons, and pharmaceuticals and personal care products (PPCPs) are some common characteristics of urban waterways (summarised in Paul et al. 2001; Walsh et al. 2005b; Ellis 2006; Surbeck et al. 2009). These effects can be variable, and can be influenced by land use (residential/industrial), wastewater treatment facilities, and the extent of stormwater drainage within a catchment (Paul et al. 2001).

Water temperature is an important factor influencing ecological processes in streams, and can directly affect metabolic rates, invertebrate life history, and nutrient cycling (Jacobsen et al. 1997; Huryn et al. 2000; Poole et al. 2001). Removal of riparian vegetation, and the urban "heat island" effect can result in water temperatures in urban waterways that are higher than those in forested catchments (Pickett et al. 2008). However, riparian vegetation not only affects stream water temperature by shading streams during the day, but can also reduce heat loss by insulating streams during colder weather

(Klein 1979). For example, in New York, USA, Pluhowski (1970) found that while stream temperatures in urban streams were between 5-8^{o}C warmer in summer when compared to undisturbed streams, urban streams were also 1.5-3^{o}C colder in winter.

Though elevated nitrogen and phosphorus concentrations are generally associated with waterways draining agricultural catchments, similar levels have been found in urban waterways (Hoare 1984, Nagumo and Hatano 2000). Elevated nutrient levels in streams can promote algal blooms, and can also create odour and water clarity problems in streams (Carpenter et al. 1998). Sources of nutrients in waterways draining urban catchments can include wastewater and septic tanks, residential fertiliser use, and household detergents (La Valle 1975; Hoare 1984).

Wastewater treatment plants, combined sewer overload, and animal faeces entering into waterways can result in increased levels of pathogens in urban waterways (Young et al. 1999; Mallin et al. 2009). Faecal bacteria, viruses, and protozoa are among the most common pathogens, and can be a direct threat to human health via ingestion of contaminated fish, shell fish or water, as well as though skin contact (Arnone et al. 2007).

One of a suite of water chemistry contaminants which distinguish urban streams from other land use related contaminants are heavy metals. Heavy metals, originating from industrial activities and construction materials for roofs, cars, and pipes, usually enter urban streams via stormwater runoff (Horowitz et al. 1987; Batcheler et al. 2006). Heavy metals are found in both the water column and sediment of urban streams (Rainbow 2002). Heavy metals can be toxic to a number of aquatic organisms, and can have significant effects on aquatic communities (Atchison et al. 1987; Hickey et al. 1998; Medley et al. 1998; Clements et al. 2000; Jezierska et al. 2009). Heavy metals have also been shown to bioaccumulate in a range of organisms including biofilm, macrophytes, invertebrates, fish, and birds (Bryan et al. 1992; Gundacker 2000; Widianarko et al. 2000; Moldovan et al. 2001; Morin et al. 2008; Bonanno et al. 2010). Metal accumulation can occur both via direct exposure to metals (dissolved and particulate) in the water column, sediment, and biofilm; and also by ingestion of metals associated with small size particles of sediment, organic matter, or prey in the case of predatory animals (Medeiros et al. 1983; Ellis et al. 1996; Beltman et al. 1999; Gundacker 2000; Widianarko et al. 2000; Bonanno et al. 2010). Trace metals in aquatic environments partition among various compartments, with a portion associating with dissolved inorganic and organic ligands in solution, and another portion being associated with particulate matter via uptake by planktonic organisms, precipitation, coprecipitation, and adsorption (Tessier et al. 1987). Due to the capacity of sediments to accumulate heavy metal contaminated particles over time, concentrations in sediment are often viewed as important indicators of metal contamination (Dixit et al. 1983; Duzzin et al. 1988). Additionally, concentrations of metals in sediments can often be higher than that in the water column, and can act as a "storage reservoir" for heavy metals in streams (Van Hassel et al. 1980).

Polycyclic aromatic hydrocarbons (PAH), petroleum-based aliphatic hydrocarbons, and other organic contaminants are commonly found at elevated concentrations in urban streams (Yamamoto et al. 1997; Paul et al. 2001; Hwang et al. 2006). PAHs are a group of organic compounds comprising of two or more fused benzene rings and are of both natural and synthetic origins (Yamamoto et al. 1997). PAHs can occur naturally from volcanic eruptions, microbial synthesis and forest fires, though the majority of PAHs enter the environment via forest and prairie fires, the burning of fossil fuels, industrial activities, and oil spills (Baek et al. 1991). However, in urban streams PAHs typically enter waterways via stormwater runoff. PAHs can have toxic effects on aquatic invertebrates and fish (Eisler 1987; Barron et al. 1999; Engraff et al. 2011). The toxicity of PAHs can be highly variable, and can be increased by UV radiation, which can often be elevated in urban streams due to a lack of shading (Landrum et al. 1987; Ireland et al. 1996; Meyer et al. 2005).

PPCPs are among the most studied group of emerging contaminants in urban waterways, comprising of chemicals such as cosmetics, fragrances, sunscreens, and human and veterinary drugs such as antibiotics and tranquilisers (Ellis 2006; Peng et al. 2008). PPCPs enter urban streams primarily through hospital discharges, disposal of drugs and cosmetics, and excretion after house-hold use (Ellis

2006). The environmental effects of such a diverse group of chemicals can be difficult to ascertain (Ellis 2006). While acute toxicity at environmental relevant concentrations seems unlikely (Fernández et al. 2013; Watanabe et al. 2016), chronic effects of PPCP's such as increased embryo mortality and developmental abnormalities in fish have been exhibited (Galus et al. 2013).

Ecologically, urban waterways are typically characterised as having reduced biotic richness, an increase of tolerant invertebrates, and a decrease in sensitive species of invertebrates and fish (Walsh et al. 2005b).

The response of algae to conditions in urban waterways appears to be inconsistent. While several studies have found elevated algal biomasses in urban streams, likely a response to higher concentrations of nutrients (Taylor et al. 2004; Busse et al. 2006; Catford et al. 2007; O'Brien et al. 2010), others have found reduced algal biomass, potentially due to increased contaminants, sediment, flow disturbance and stream depth (Paul et al. 2001; Potapova et al. 2005). Algal community composition also shows mixed responses to catchment urbanisation. While studies have reported a shift in diatom composition from oligotrophic to eutrophic species (Winter et al. 2000; Sonneman et al. 2001; Newall et al. 2005), species diversity showed no consistent response to urbanisation across these studies and appears inconsistent across geographic regions (Walsh et al. 2005b).

Macrophytes are an important component of many ecosystems, though the response of macrophytes to urbanisation has not been widely studied (Paul et al. 2001). Vermonden et al. (2010) found that in tributaries of the Rhine River in the Netherlands, species diversity of macrophytes in was lower in urban streams relative to semi-natural streams bordering urban areas. King et al. (2000) found that in Sydney, the number of macrophyte species was higher in urban waterways than non-urban waterways, though the number of native species did not differ.

The response of benthic invertebrates to conditions in urban waters has been shown to be relatively consistent, making them an important indicator for impacts of land-use change (Walsh et al. 2005b). Highly degraded streams are typically species poor and dominated by contamination tolerant taxa (oligochaetes and chironomids), while sensitive taxa such as Ephemeroptera, Plecoptera, and Trichoptera (EPT) are either absent or found in lesser abundance (Hall et al. 2001; Paul et al. 2001; Suren et al. 2005b; Collier et al. 2009). Exceptions do exist however, tolerant caddisfly such as hydroptilids can often be found in higher numbers in moderately impacted waterways, likely due to their higher tolerance to pollutants than other EPT taxa (Clements et al. 1994). Further to this, some seepage habitats in urban gullies in Hamilton, New Zealand have been found to be dominated by sensitive taxa, which is likely a consequence of the lack of upstream development and stormwater connections in these environments (Collier et al. 2009).

Fish communities in urban streams follow a similar pattern to macroinvertebrate communities, showing a reduction in species richness and abundance of sensitive taxa, and assemblages dominated by more tolerant species (Onorato et al. 2000; Walsh et al. 2005b; Miserendino et al. 2008; Alexandre et al. 2010). In New Zealand, however, a number of fish species are frequently recorded in urban streams. Collier et al. (2009) sampled urban and peri-urban sites in Hamilton, finding urban streams supported a similar range of species to peri-urban. Urban streams in Hamilton were also found to contain longfin eel and giant kōkopu, two species which are considered threatened and in gradual decline (Hitchmough et al. 2005). Similarly, McEwan et al. (2009), found that in Auckland, while species richness was lower in urban streams than in forested reference streams, urban streams still contained native species such as banded kōkopu and longfin eel.

1.3 - Rehabilitation

The widespread occurrence of the urban stream syndrome has resulted in urban stream rehabilitation projects becoming common worldwide (Bernhardt et al. 2007). Rehabilitation projects have been typically focused on a single reach or reaches throughout a catchment (Roni et al. 2008). Enhancement techniques including streambed boulder and log additions, sediment traps, culvert removals/additions, channel shaping, riparian plantings, and macrophyte management are common in urban stream rehabilitation projects (Rutherfurd et al. 2000; Blakely et al. 2005; Collier et al. 2008; Roni et al. 2008). The theoretical basis for these rehabilitation programmes is based on the "Field of Dreams" hypothesis, or "If you build it they will come" (Palmer et al. 1997). That is to say, that following this paradigm the rehabilitation aims are to reconfigure reaches of impacted streams to recreate the instream and riparian conditions of a reference stream, under the assumption that this will trigger recolonisation of sensitive taxa these reaches, restoring biotic integrity (Palmer et al. 1997).

However, most rehabilitation efforts have been met with mixed results, with little difference between rehabilitated vs degraded stream biological communities (Parkyn et al. 2003; Suren et al. 2005a; Blakely et al. 2006; Palmer et al. 2010; Violin et al. 2011). Several studies have investigated the inability of urban stream rehabilitation programmes to improve the biological condition of streams beyond that common of urban degraded streams (Booth 2005; Blakely et al. 2006; Palmer et al. 2010; Violin et al. 2011). Factors thought to inhibit biological community recovery in rehabilitated streams include physical barriers to insect movement and ovipositioning (culverts, lack of connectivity), lack of suitable substrate for ovipositioning (due to anthropogenically induced changes in hydrology and morphology), and increases in sediment and pollutants such as polycyclic aromatic hydrocarbons (PAHs) and heavy metals associated with high impervious surface cover and storm water connections (Parkyn et al. 2003; Walsh 2004a; Blakely et al. 2005; Suren et al. 2005a; Blakely et al. 2006).

1.4 - Urban waterways in Christchurch City

Ōtautahi, Christchurch City, built upon the Avon/Ōtākaro and Heathcote/Ōpāhawo Rivers, is the largest city in the South Island of New Zealand. Since its European settlement and founding in 1856, the former wetlands, streams, and ponds that made up Christchurch have undergone continued urbanisation to the point of the modern city of the current day (Wilson 1989). An increase in industrial land use, population density, impervious surface cover, storm water systems, and associated runoff have caused heavy metals in urban streams to reach levels which are of concern, with several found to be exceeding ANZECC guidelines in Christchurch City Council's periodical water and sediment quality monitoring (Gadd et al. 2014; Margetts 2014; Gadd 2015).

High concentrations of heavy metals can be a key factor in the structure of invertebrate communities (Clements et al. 2000; Beasley et al. 2003), and are thought to be a potential barrier to recolonisation of sensitive taxa to rehabilitated reaches in Christchurch (Blakely et al. 2005; Suren et al. 2005a; Winterbourn et al. 2007). Whilst heavy metal concentrations are monitored periodically by Environment Canterbury and CCC, pairing of wide scale surveys of metal contamination of urban waterways with ecological assessments are lacking, particularly in lower order streams (Golder Associates 2012).

1.5 - Thesis Structure

The objective of my thesis is to investigate the effects of heavy metals on the benthic invertebrate fauna of Christchurch City.

Chapter Two describes the results of a survey of 20 urban waterways in Christchurch City. Specifically, the survey aimed to investigate the changes in aquatic macroinvertebrate communities over a gradient of metal contamination. Chapter Three is an investigation of the survival of selected benthic taxa in mesocosms in six urban streams over a gradient of metal contamination. Chapter Four contains a conclusion of findings and implications to future management of urban waterways.

Chapter 2

The response of stream invertebrates to a gradient of heavy metals.

2.1 Introduction

Aquatic invertebrates commonly accumulate heavy metals in their tissues (Rainbow 2002). These metals become toxic to organisms once the concentration of the metabolically available portion has reached a threshold, this threshold varies greatly between invertebrate species and metals (Rainbow et al. 2011). Due to this variation, a threshold is often independent of total accumulated metal concentrations in organisms as trace metals are often stored in detoxified forms, which do not cause toxic effects (Rainbow et al. 2011). In addition, individual species of invertebrates can accumulate and store heavy metals in different concentrations in their tissues and organs (Rainbow et al. 2011). As a result, invertebrates living in the same habitat can vary greatly in concentrations of metals in their tissues, even between species of the same genus (Moore et al. 1987; Hare 1992; Rainbow 1995). Though metal toxicity has been detected throughout the cell, tissue, individual, population, and community level (Hare 1992), the variation of toxic effects between invertebrate species and different metals at these levels can make predicting the effects of metals on invertebrate communities challenging.

Generally, the response of invertebrate communities to heavy metals is decreased total taxonomic richness and decreased richness and abundance of the pollution sensitive orders Ephemeroptera (mayflies), Plecoptera (stoneflies) and Tricoptera (caddisflies) (EPT) (Hickey et al. 1998; Clements et al. 2000; Ruse et al. 2000). EPT taxonomic richness and abundance can be a good indicator of metal pollution in water (Lenat et al. 1994; Gray 2004). However, Clements et al. (1994) showed that EPT metrics can be insensitive to moderate levels of heavy metal contamination due to mayflies being replaced by metal-tolerant caddisfly taxa at metal impacted sites. For example, the mayfly *Deleatidium* can be replaced by the hydroptilid caddisflies *Oxyethira* and *Paraoxethira* in New Zealand streams. Clements et al. (2000) found that the impact of heavy metals on Colorado mountain stream invertebrate communities varied depending on metal concentrations. At moderate metal concentrations, EPT richness was significantly lower, while taxonomic richness, total abundance, and EPT abundance were significantly lower only at sites with high metal concentrations. Clements et al. (2000) also investigated metal impacts on functional feeding group composition (i.e. their feeding strategies), finding that scrapers and shredders were significantly less abundant at medium and high metal sites, while shredders and collectors were relatively tolerant of metals, and only showed significantly reduced abundance at high metal concentrations. A similar pattern was found in New Zealand streams, with abundance and species richness of mayflies, EPT richness, and total taxonomic richness the best indicators of metal contamination (Hickey et al. 1998).

Sources of metals in the aquatic environment can be both natural and anthropogenic. The metals most commonly attributed to urban waterways include zinc, copper, lead, chromium, and cadmium (Wilber et al. 1979). Zinc is primarily sourced from vehicle tires, metal production processes, coal combustion, and galvanised piping and roofing (Frassinetti et al. 2006). Wicke et al. (2014) analysed storm water runoff from metal roofs of varying age, finding galvanised iron roofing contributes considerable amounts of dissolved zinc (runoff contained between 200 μg L^{-1} and 450 μg L^{-1} zinc over an entire 16 hour rain event), 98% of which was in bioavailable dissolved form. In contrast, copper contamination in urban streams typically originates from car brake linings, treated timber, and also copper roofing, building facades, and copper guttering and downpipes (Pennington et al. 2008; Wicke et al. 2014). Elevated lead is often attributed to legacy contamination from lead additions to paint and fuel, however, Kennedy et al. (2008) found precipitation, soils, roof materials and lead tipped nails contributed a significant amount of lead into Auckland’s urban streams. Chromium contamination can occur from leaching from concrete, and vehicle wear and tear (Göbel et al. 2007; Kayhanian et al.

2009), while electro-plating, batteries, cadmium lined brakes, and phosphorous fertilisers are among the sources of cadmium to waterways (Davis et al. 2001; Abrahim et al. 2002). In Christchurch, the metals of particular concern include dissolved copper and zinc, and sediment bound zinc and lead (Gadd et al. 2014; Margetts 2014; Gadd 2015). Ecological studies of Christchurch's urban waterways have found them to be relatively depauperate, particularly with respect to EPT taxa (Suren et al. 2005a; Boffa Miskell Limited 2014, 2015). While early surveys of Christchurch's urban waterways found EPT taxa at many sites throughout the city (Robb 1992), many of these same sites no longer contain these more pollution sensitive taxa (McMurtrie et al. 2003).

The aims of this chapter were to identify sites along a gradient of metal contamination across Christchurch's urban waterways, and to investigate the response of macroinvertebrate communities to any heavy metal gradient.

2.2 METHODS

2.2.1 - Site Selection

Initially, 33 urban waterways in Christchurch were selected to be surveyed. The majority of the sites were chosen to represent a metal contamination gradient (both in the water and sediment) from reference sites with low metals to highly industrial sites with high metal content. Waterways were selected based on existing literature (Environment Canterbury 2008, 2009, 2012; Christchurch City Council 2013), expert opinion, and spatial coverage across the city. All waterways were visited prior to sampling, however, of the 33 sites, 13 had either dried up or were found to be stagnant due to a particularly dry summer. As a result only 20 sites were considered suitable for sampling, consisting of 18 rivers and streams within 10 catchments (Figure 2.1) (CMAPS / Canterbury Regional Council 2015). In each waterway an approximately 20m sampling reach was selected, containing a riffle and run where possible (Figure 2.2).

2.2.2 - Physico-chemical characteristics

In each reach, a suite of physical, chemical and biological measurements and samples were taken. Spot water quality parameters (dissolved oxygen, pH, specific conductivity, temperature) were measured using a HACH 40d pH-Cond-DO meter. Physical parameters measured included velocity (at five points on a transect) using Global Water TX100 velocity meter; wetted width (at three transects) and depth (at five points along a transect) using a 1 metre ruler; and shading using a densitometer (Robert E. Lemmon, Spherical Densiometer, Model A). The substrate composition and size was determined by visual estimates of the percentage of the streambed dominated by each of the substrate size classes of Wentworth (1922) along three transects. In addition to these parameters, each site was scored for streambed and bank stability using the method of Pfankuch Pfankuch (1975).

A substrate index (SI) was calculated by averaging the SI score for each substrate estimate taken across the three transects at each site, using a modification of the method of Jowett et al. (1990):

<u>Substrate index</u> = (0.08% boulder) + (0.07% cobble) + (0.06% pebble) + (0.05% gravel) + (0.04% sand) + (0.03% silt).

Impervious surface area was also obtained for each stream from the 'Freshwater Ecosystems of New Zealand Geodatabase' (FENZ) geodatabase (Leathwick et al. 2010), in which the proportion of impervious surface area for the immediate catchment was calculated from topographic map data and traversed downstream and an area weighted average for the upstream catchment was calculated.

2.2.3 - Trace metal analysis

At each reach a composite fine sediment sample (<2mm) was collected to determine longer-term metal concentrations. At each site, up to three inorganic sediment deposits were collected by scraping approximately 200 grams of the top layer of oxidised fine sediment from the bed of the stream. Samples were collected in into a clean plastic container, excess water in the container was drained, and the container was sealed to avoid contamination. Samples were then transported to the laboratory for analysis.

In the laboratory, sediment samples were dried in a drying oven for 3 days at 50°C. Dried samples were double bagged in clean sealable bags and disaggregated by light tapping with a rolling pin. Samples were then sieved through a 2mm nylon sieve. One gram of the fine sediment was then weighed and placed in acid washed digestion tubes. Acid digestion involved adding 4 ml of 1:1 nitric acid (HNO_3) and 10 ml of a 1:4 hydrochloric acid (HCl) to each tube and allowing to sit for overnight. The solutions were then heated to 85°C and refluxed for 40 minutes before being cooled to room temperature and made up to 40 ml with milliQ water. Duplicate samples of certified reference material (U.S National Institute of Standards and Technology Reference Material 2702) and experimental blanks were included in the digestion process. All digested samples were left to stand overnight. Samples were then diluted by a factor of 10 by adding 0.5 ml of the digested sediment solution to 4.5 ml of 2% HNO_3 and 0.5% HCl solution. A duplicate sample was included every 10 samples to test for quality control (Table 2.1). Additionally, a triplicate sample was spiked and included every 20 samples to measure the recovery of trace metals. Analysis of trace metal content was then performed by inductively coupled plasma mass spectroscopy (ICP-MS) as per Method 200.8 from the U.S. Environmental Protection Agency (Brockhoff et al. 1999).

A single grab water sample (250 ml) was also collected from each sampling reach for analysis of dissolved metals. Samples were filtered through a Whatman 0.45 µm membrane filter into an acid washed polyethylene bottle. Dissolved metal samples were then transferred to the University of Canterbury ICP-MS Facility where they were acidified and analysed by ICP-MS. All ICP-MS analysis was done on an Agilent 7500 Series ICP-MS with an octopole reaction system at the University of Canterbury by Rob Stainthorpe.

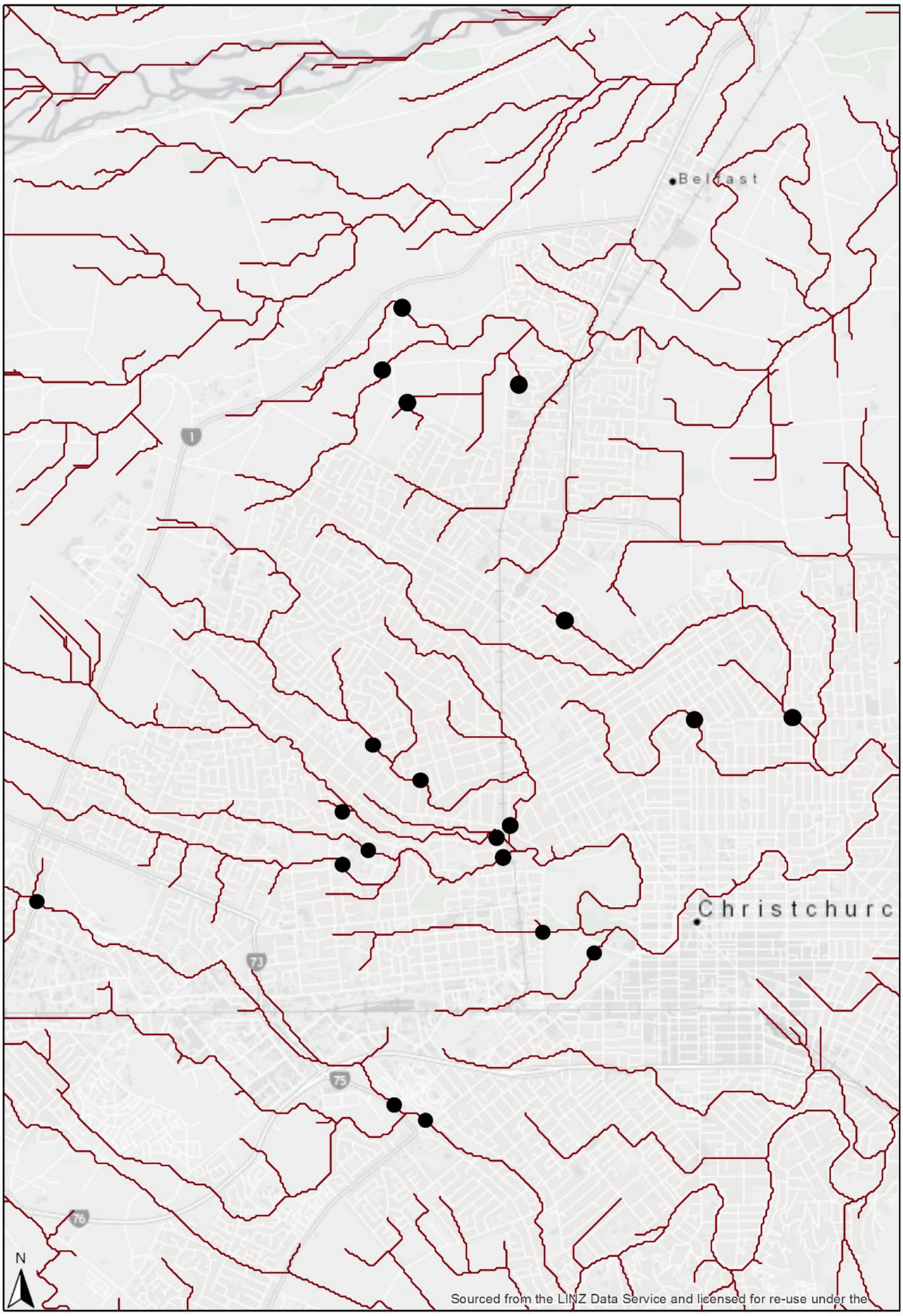


Figure 2.1. Christchurch's urban stream network with my 20 sites (black dots), sampled between January and March 2015.

Figure 2.2. Photos showing Site 2, a rehabilitated reach of Smack's Creek (top left), Site 3, a highly channelised section of Gardiner's Road Stream (top right), Site 5, a highly channelised section of Curlett's Road Stream draining a highly industrial catchment (bottom left), Site 9, Addington Brook, situated in Hagley Park (bottom right).

2.2.4 - Macroinvertebrate sampling

Benthic invertebrates were sampled using both qualitative and quantitative methods. A composite kick-net sample (0.5 mm mesh) was collected from a range of micro-habitats at each sampling reach in order to determine overall taxonomic richness. Samples are collected by kicking and scuffing the substrate for 45 seconds to free any macroinvertebrates present, allowing them to drift into the downstream net. To examine invertebrate densities, three qualitative Surber Samples (0.5mm mesh, 0.0625m^2 area) were collected randomly in riffles and runs. Surber samples are collected by placing the Surber sampler on the bed, and disturbing the substrate by hand to a depth of 5cm. All invertebrate samples are preserved in the field with 70% ethanol. Invertebrate samples were returned to the laboratory and identified to the taxonomic level used to calculate an MCI (Winterbourn 1973; Winterbourn et al. 2000). This was generally genera, though Diptera were identified to family level, and Oligochaetes to class level. Taxonomic richness was determined by using data from kicknet samples and any additional taxa found in Surber Samples. Surber Samples were counted and converted to densities per m^2.

2.2.5 - Biotic indices

Macroinvertebrate community indices used included the Macroinvertebrate Community Index (MCI) (Stark 1985), the Quantitative Macroinvertebrate Community Index (QMCI) (Stark 1993). These indices are based on tolerance of taxa to organic enrichment in hard bottom streams, in which each species is allocated a score ranging from 1-10, with 10 indicating low tolerance to organic enrichment. Margalef's Index (Margalef 1958) was also used, along with other metrics including total number of taxa, the number of Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa, percent EPT abundance, percent Mollusca abundance, percent Chironomid abundance, and percentage abundance of functional feeding groups (Predators, Shredders, Grazers, Collector-gathers, and Filter-feeders).

2.2.6 - Statistical Analysis

The effect of dissolved metals, metals in sediment, water chemistry and physical characteristics on macroinvertebrate community metrics were analysed using a combination of stepwise model building using permutation tests, and constrained ordination. Macroinvertebrate community abundance data from the three Surber samples were averaged and ln(x+1) transformed to meet assumptions of normality. Environmental variables were also log transformed to meet assumptions of normality. Due to a strong covariance between pH and SMI (R^2 = 0.59, P = <0.001) and pH not being at levels which are likely to affect macroinvertebrate communities (pH range: 6.0-7.2), pH was removed from the model. These data were then entered into a Canonical Correspondence Analysis

(CCA) using R Statistical package (R Statistical Package 2009). CCA is a multivariate constrained ordination technique that extracts gradients among combinations of environmental variables, and arranges biological assemblages along these gradients (Ter Braak 1987). Significant explanatory variables that explain macroinvertebrate abundance and community composition were then selected using the R function *Ordistep* from the 'vegan' package, an automatic stepwise model builder for constrained ordination methods (Oksanen et al. 2007; Blanchet et al. 2008; R Statistical Package 2009). Ordistep alternates between dropping and adding variables using permutation P-values to determine significance. Ordistep stops when the model is not changed within one step.

Stepwise regression was conducted on univariate community indices against environmental variables using R (R Statistical Package 2009). Stepwise regression is a semi-automated model builder which operates by successively adding or removing variables based on Akaike Information Criterion (AIC). Variables were log transformed when necessary to meet assumptions of normality.

Non-Metric Dimensional Scaling (NMDS) was used to give an indication of similarity between sites for physical characteristics. NMDS ordination ranks sites such that distance in ordination space represents dissimilarity of input variables, in this case using Euclidean distance. These ordination scores allow sites to be represented on an x-y scatterplot, with sites closer together being more similar than those further apart (Minchin 1987). Goodness of fit of the NMDS ordination is assessed by the 'stress value. A stress value of 0 indicates perfect fit, while a stress value of <0.2 is considered to represent a good interpretation of data (Quinn et al. 2002).

Dissolved and sediment metals which were at concentrations likely to affect biological communities (exceeding guidelines), were scaled and centred and reduced to a "Sediment Metal Index" (SMI) and a "Dissolved Metal Index" (DMI) by using principal component analysis (PCA). Metals were entered into a PCA, and the scores from the first PCA axis were used as the metal indices. Due to only having a single site above detection limits for dissolved aluminium, this metal was excluded from the PCA. For other metal concentrations below detection limits, values of half the detection limit were used.

Table 2.1. Quality control data showing sample detection limits, blanks, and CRM recoveries. ICP-MS recovery of ICP-MS CRM (IV SRM 1643 standard reference material), and digest recovery for marine sediment CRM (U.S NIST Standard Reference Material 2702).

		Na	Mg	Al	K	Ca	V	Cr	Mn	Fe	Co	Ni	Cu	Zn	As	Cd	Sb	Pb
Water Quality Control																		
Detection Limit	µg/l	100	10	10	100	1000	0.1	0.1	0.1	10	0.1	0.1	0.1	1.0	1.0	0.1	0.1	1.0
ICP-MS CRM recovery[1] (n=1)	%	98.1	100.9	112.7	105.0	111.3	111.6	105.5	107.2	102.0	107.0	104.8	103.9	100.0	107.7	105.7	109.5	100.0
Sediment Digest Quality Control																		
Detection Limit	µg/l	100	10	10	1.0	100	0.1	0.1	10	10	0.1	0.1	0.1	1.0	0.1	0.1	0.1	0.1
ICP-MS CRM recovery[1] (n=1)	%	90.0	93.5	105.6	99.1	99.7	114.5	107.5	102.6	142.9	119.6	106.6	99.1	112.5	114.5	110.0	119.1	115.5
Sediment CRM recovery[2] (n=2)	%	-	-	-	-	-	74.1	74.3	87.6	-	93.5	63.8	73.6	89.6	97.8	100.0	23.2	90.5
Digest blank	µg/l	<100	<10	<10	13	<100	0.2	0.5	<10	10	<0.1	<0.1	<0.1	<1.0	0.1	<0.1	<0.1	<0.1

[1]certified standard IV SRM 1643, [2]U.S NIST Standard Reference Material 2702

2.3 Results

2.3.1 - Physical Characteristics

The mean stream width of my 20 sites was 2.83 m (± 0.17). The narrowest stream was 0.7m (wetted width), and the widest 6.5m (Table 2.2). Stream depth varied widely between sites with the shallowest depth 0.04-0.07m and the deepest 0.26-0.55 m. Current velocity ranged from 0.05 m/s to a moderate 0.46 m/s, while substrate index ranged from 3.8 (dominated by silt and sand) to 7.13 (dominated by cobbles). Shading also varied widely, ranging from very poorly shaded sites at 4% to almost completely shaded sites at 88%. Mean shading was 47.6% (± 4.8). Pfankuch scores indicated a wide range in habitat stability across sites, from a good score of 44 to fair score of 87. The mean Pfankuch score was 62.3 (± 2.9). Impervious surface cover ranged from 15% to 96%, with a mean of 66.7 % (± 5.0).

A non-metric dimensional scaling (NMDS) ordination indicated that there was variation of physical characteristics between sites (Figure 2.3).

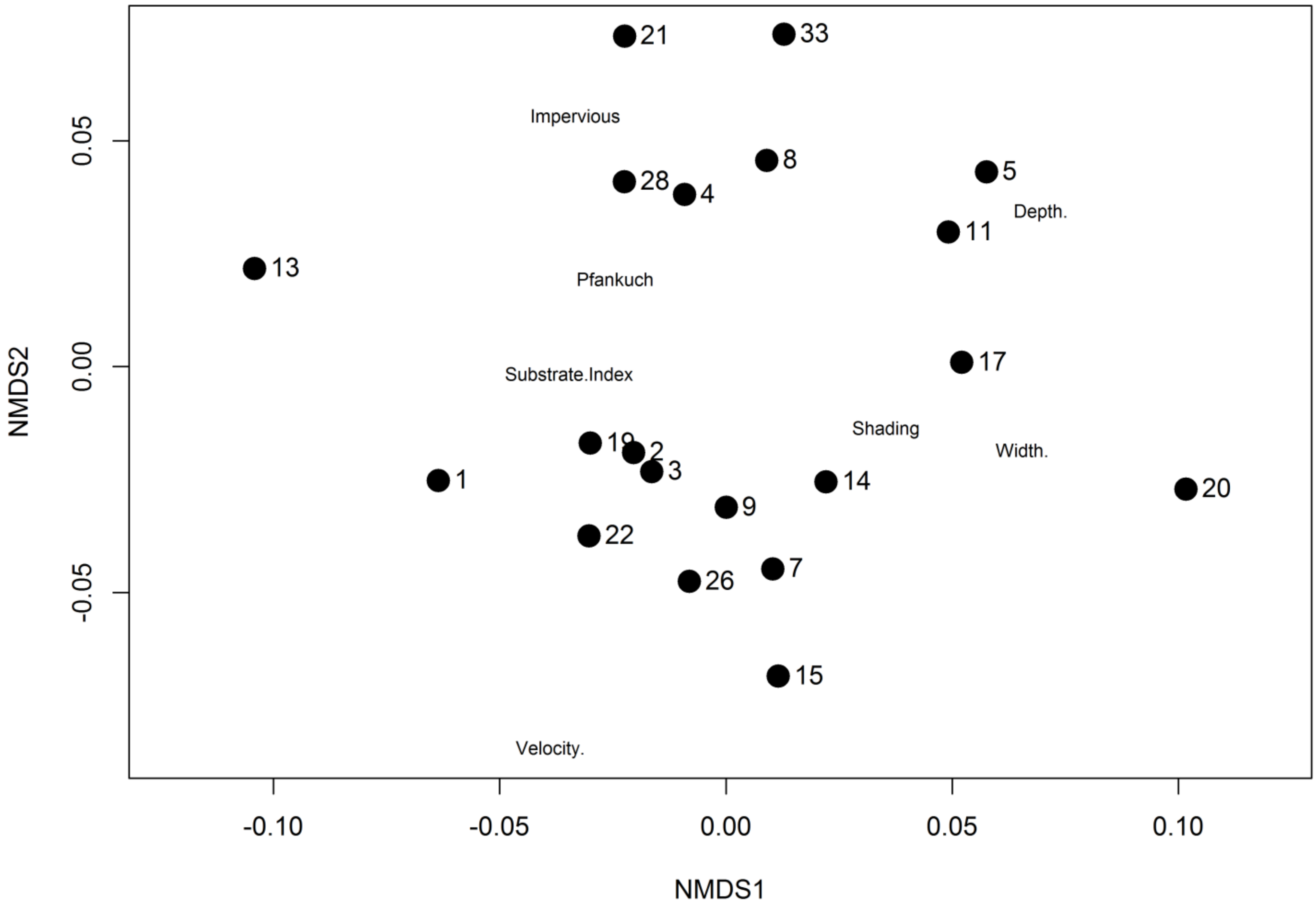


Figure 2.3. Non-metric dimensional scaling (NMDS) ordination based on a Euclidean distance matrix of dissimilarities calculated from physical characteristics (wetted width, depth, velocity, shading, substrate index, % impervious surface area) measured at each of the 20 sampling reaches (two-dimensional stress = 0. 12).

Table 2.2. Physical Characteristics (mean ± 1 SE) of the 20 sampling reaches, as measured on one occasion between January-March 2015. *Channel stability from Pfankuch (1975)

Site Number	Site Name	Easting	Northing	Shading	Width	Depth	Velocity	Substrate Index	Impervious Surface Area	Channel Stability*
				%	M	m	m/s-1		%	
1	Styx River	1566532.44	5186940.14	68	3.2 ± 0.4	0.42 ± 0.05	0.16 ± 0.05	3.8 ± 0.4	26	85
2	Smacks Creek	1566843.19	5187921.69	52	2.0 ± 0.3	0.09 ± 0.01	0.38 ± 0.12	6.5 ± 0.2	53	63
3	Gardiners Road	1566994.61	5186651.48	40	0.9 ± 0.0	0.15 ± 0.01	0.1 ± 0	5.9 ± 0.4	67	68
4	Cavendish Stream	1568411.39	5186956.50	20	2.2 ± 1.1	0.06 ± 0.01	0.22 ± 0.05	6.5 ± 0.2	69	49
5	Curletts Road	1565595.32	5178726.44	24	1.7 ± 0.1	0.08 ± 0.02	0.06 ± 0.04	5.4 ± 1.1	94	70
7	Paparua Stream	1562297.83	5180464.98	4	0.8 ± 0.0	0.08 ± 0.01	0.24 ± 0.04	7.1 ± 0.2	29	54
8	Riccarton Stream	1568721.63	5180027.49	48	1.2 ± 0.0	0.15 ± 0.01	0.46 ± 0.04	6.6 ± 0.1	95	82
9	Addington Brook	1569373.32	5179768.75	64	2.1 ± 0.1	0.15 ± 0.01	0.05 ± 0	3.9 ± 0.2	67	87
11	Papanui Stream	1568942.34	5183943.15	88	2.2 ± 0.2	0.14 ± 0.01	0.4 ± 0.07	6.6 ± 0.1	84	44
13	Wairarapa Stream (a)	1568248.14	5181198.26	36	6.5 ± 0.0	0.28 ± 0.02	0.16 ± 0.02	4.9 ± 1.0	74	58
14	Avon River (a)	1568276.10	5181035.25	40	5.1 ± 0.1	0.4 ± 0.05	0.26 ± 0.07	6.7 ± 0.1	66	53
15	Fendalton Stream	1568225.57	5181177.60	72	4.7 ± 0.1	0.22 ± 0.04	0.4 ± 0.08	6.6 ± 0.1	31	45
17	Wairarapa Stream (b)	1567148.52	5181935.39	44	5.3 ± 0.0	0.26 ± 0.01	0.1 ± 0	6.6 ± 0.2	69	59
19	Wairarapa Stream (c)	1566610.79	5182267.54	44	3.5 ± 0.4	0.17 ± 0.02	0.08 ± 0.04	6.3 ± 0.1	65	50
20	Okeover Stream	1566662.24	5181006.92	60	2.9 ± 0.2	0.08 ± 0.01	0.26 ± 0.05	5.9 ± 0.1	15	66
21	Avon River (b)	1566161.25	5180889.71	64	3.4 ± 0.1	0.14 ± 0.03	0.46 ± 0.01	5.4 ± 0.9	93	61
22	Waimairi Stream	1566290.66	5181551.35	40	2.3 ± 0.2	0.05 ± 0.01	0.26 ± 0.09	6.3 ± 0.1	85	51
26	Dudley Creek	1571877.77	5182685.11	24	1.9 ± 0.1	0.07 ± 0.01	0.3 ± 0.03	5.5 ± 0.6	82	61
28	Heathcote River	1567229.55	5177635.31	84	3.8 ± 0.1	0.06 ± 0.01	0.42 ± 0.09	7 ± 0.3	48	62
33	St Albans Stream	1570620.67	5182694.29	36	0.7 ± 0.1	0.1 ± 0.02	0.06 ± 0.02	4.9 ± 0.8	96	79

2.3.2 - Water chemistry

The pH, water temperature, DO, and specific conductivity measured across the survey sites were generally within the ranges expected from streams draining urban catchments (Table 2.3). Stream water pH ranged from 6.0 to 7.2, while spot water temperature ranged from 12.4-17.5 °C with a mean of 14.6 °C (± 0.3). No sites were above the 20 °C maximum guideline stipulated in the LWRP for springfed (plains) urban waterways.

Dissolved oxygen ranged from 5.7 to 11.8 mg/L, while specific conductivity ranged from 77 to 336 µS/cm with a mean of 149 µS/cm (± 12). Conductivity was consistent across sites, though Site 4 was notably lower than the remainder of sites, while Site 9 was notably higher.

Table 2.3. Water quality parameters of site as measured on one occasion between January and March 2015. Site names followed with a.) b.) and c.) denote different sites along stream continuum.

Site Number	Site Name	pH	DO		Conductivity	Temperature
			DO%	DOmg/L	$\mu S/cm^{-1}$	°C
1	Styx River	6.0	55.9	5.9	121	12.7
2	Smacks Creek	6.1	64	6.7	116	13.3
3	Gardiners Road	6.4	88.5	9.3	111	12.4
4	Cavendish Stream	6.3	67.8	7.0	77	13.6
5	Curletts Road	7.2	106.6	10.6	152	14.4
7	Paparua Stream	6.3	73.8	7.4	163	15.1
8	Riccarton Stream	6.7	95.8	9.6	200	15
9	Addington Brook	7.2	77.9	7.8	336	14
11	Papanui Stream	6.6	90.1	8.7	97	15.9
13	Wairarapa Stream (a)	6.4	91.1	9.3	137	14.7
14	Avon River (a)	6.4	93	9.4	158	14.7
15	Fendalton Stream	6.5	95.9	9.6	130	14.7
17	Wairarapa Stream (b)	6.5	106.3	10.8	143	14.7
19	Wairarapa Stream (c)	6.5	118.1	11.8	113	14.7
20	Okeover Stream	6.6	96	9.3	170	16.2
21	Avon River (b)	6.2	74.8	7.5	166	13.8
22	Waimairi Stream	6.3	97.6	9.5	148	17.5
26	Dudley Creek	6.8	86	8.3	108	16.9
28	Heathcote River	6.5	87.6	8.9	188	14.1
33	St Albans Stream	7.2	54.9	5.7	138	13.2

2.3.3 - Heavy metals

Dissolved metal concentrations in my spot water samples were similar across most sites, with the exception of three sites which had significantly higher concentrations (Table 2.4) (Figure 2.4). Site 2, 7, and 33 had the highest dissolved metal concentrations. Sediment metal concentrations followed a similar pattern to dissolved metals, with most sites having relatively similar sediment metal concentrations but for five sites which were more heavily contaminated (Table 2.5). These sites were Site 5, 8, 9, 28, and 33. Dissolved metals found to be breaching guidelines were Al, As, Cd, Cr, Cu, and Zn, while sediment metals found breaching guidelines were As, Cu, Pb, and Zn (Table 2.4, 2.5) (Anzecc 2000).

2.3.4 - Metal indices

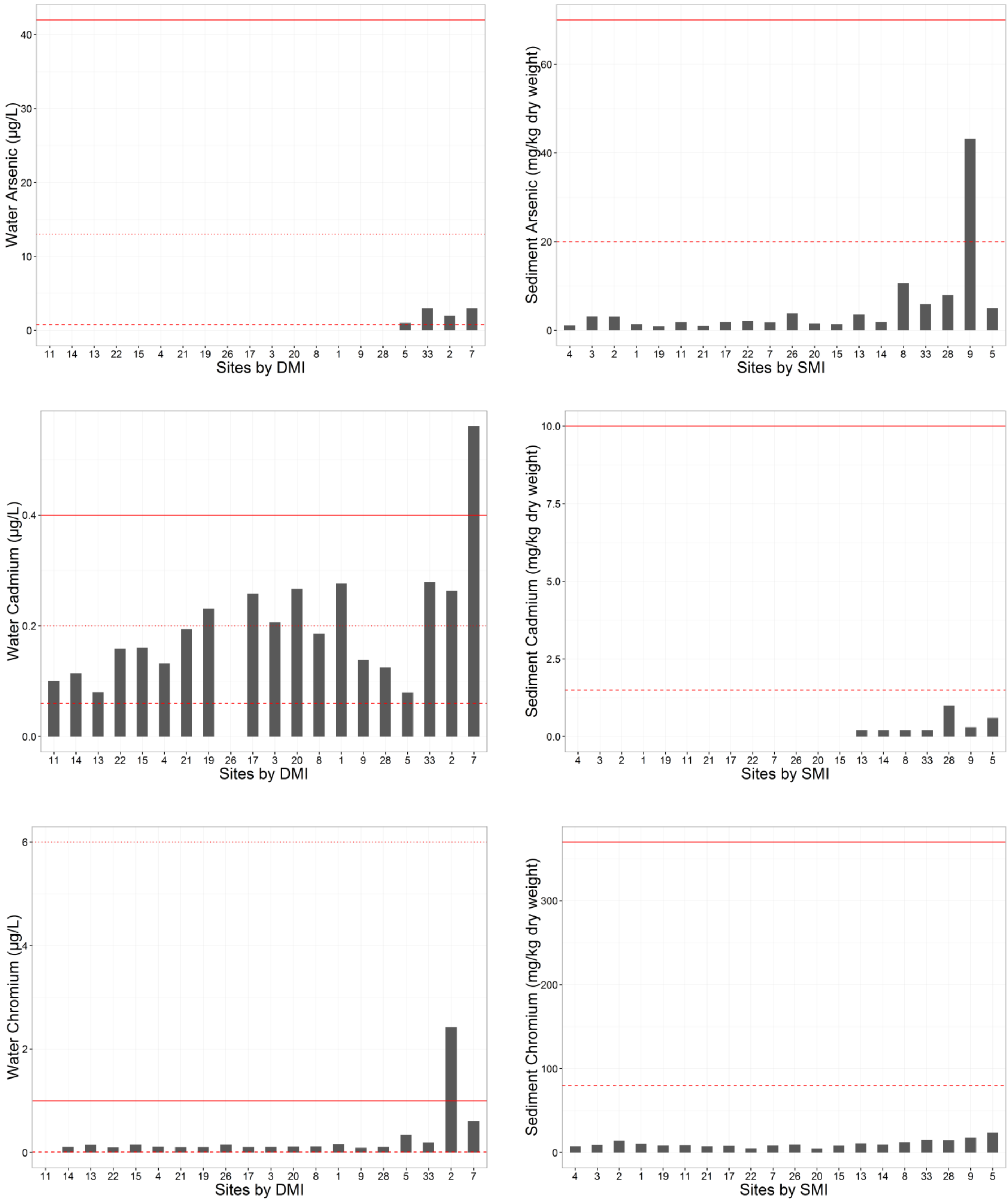
My aim was to sample sites along a metal gradient. PCA analysis of both dissolved metals and sediment metals of ecological concern (exceeding guidelines) revealed two significant axes which were used as a Dissolved Metal Index (DMI) and Sediment Metal Index (SMI). The first axis of the dissolved metal PCA explained 64 % of the variation of the five metal variables (Cr, Cu, Zn, As, Cd), while the first axis of the sediment metal PCA explained 71% of the variance of the four metals (As, Cu, Pb, Zn). The DMI index shows a high number of sites with moderate metal concentrations, with few with higher concentrations, whereas a more even spread of concentrations in evident in the SMI (Figure 2.5). The relative contributions of each metal to the PCAs are listed in Table 2.6. Dissolved metal concentrations and sediment metal concentrations were not strongly correlated (DMI and SMI regression: R^2 = 0.05 P = 0.82), with sites containing high sediment metal concentrations not necessarily having high metal concentrations in the water (Figures 2.4, 2.6).

Table 2.4. Dissolved metals as measured on one occasion between Jan-Mar 2015. ANZECC trigger values for alternative levels of protection are included (Anzecc 2000).

Site Number	Na µg/L-1	Mg µg/L-1	Al µg/L-1	K µg/L-1	Ca µg/L-1	V µg/L-1	Cr µg/L-1	Mn µg/L-1	Fe µg/L-1	Co µg/L-1	Ni µg/L-1	Cu µg/L-1	Zn µg/L-1	As µg/L-1	Cd µg/L-1	Sb µg/L-1	Pb µg/L-1
1	5900	1800	<10.0	900	15000	0.2	0.2	1.3	<10	<0.1	0.1	0.1	35	<1.0	0.3	<0.1	<1.0
2	5400	1630	<10.0	1200	14000	0.2	2.4	1.3	10	<0.1	0.1	0.3	51	2.0	0.3	<0.1	<1.0
3	6500	2200	<10.0	1000	18000	0.2	0.1	3.6	10	<0.1	<0.1	0.2	17	<1.0	0.2	<0.1	<1.0
4	5300	2250	<10.0	800	15000	0.2	0.1	7.0	10	<0.1	<0.1	0.2	21	<1.0	0.1	<0.1	<1.0
5	17300	3390	<10.0	1200	27000	0.8	0.3	1.3	20	<0.1	0.2	1.3	18	1.0	0.1	0.3	<1.0
7	7000	1200	50	1900	10000	0.5	0.6	27.9	70	<0.1	0.6	2.4	156	3.0	0.6	0.3	<1.0
8	13400	5220	<10.0	1500	33000	0.2	0.1	22.2	20	<0.1	0.1	0.2	14	<1.0	0.2	<0.1	<1.0
9	17700	6570	<10.0	2300	34000	0.1	0.1	132.4	30	0.2	0.5	0.9	16	<1.0	0.1	0.2	<1.0
11	5500	2410	<10.0	800	18000	0.1	<0.1	8.0	10	<0.1	0.2	0.2	15	<1.0	0.1	<0.1	<1.0
13	7800	2690	<10.0	900	21000	0.3	0.2	1.1	10	<0.1	<0.1	0.1	20	<1.0	0.1	<0.1	<1.0
14	9400	2920	<10.0	1000	23000	0.2	0.1	1.6	10	<0.1	0.1	0.1	17	<1.0	0.1	<0.1	<1.0
15	8100	2710	<10.0	900	21000	0.3	0.1	0.9	10	<0.1	<0.1	0.1	17	<1.0	0.2	<0.1	<1.0
17	7300	2340	<10.0	900	19000	0.2	0.1	4.2	10	<0.1	<0.1	0.1	14	<1.0	0.3	<0.1	<1.0
19	6800	2110	<10.0	900	19000	0.2	0.1	1.0	10	<0.1	<0.1	0.1	18	<1.0	0.2	<0.1	<1.0
20	8800	2500	<10.0	1000	21000	0.20	0.1	0.4	10	<0.1	<0.1	0.1	18	<1.0	0.3	<0.1	<1.0
21	9300	2900	<10.0	1100	23000	0.2	0.1	2.6	10	<0.1	0.1	0.1	22	<1.0	0.2	<0.1	<1.0
22	8500	2600	<10.0	900	21000	0.1	0.1	0.6	10	<0.1	<0.1	0.1	15	<1.0	0.2	<0.1	<1.0
26	7700	3680	<10.0	1600	21000	0.2	0.2	57.4	40	0.2	0.5	0.7	19	<1.0	<0.1	<0.1	<1.0
28	17800	5320	<10.0	2200	30000	0.3	0.1	46.1	20	0.1	0.4	1.1	25	<1.0	0.1	0.2	<1.0
33	9700	2590	<10.0	1300	14000	1.3	0.2	13.8	20	<0.1	0.4	1.0	33	3.0	0.3	0.2	<1.0
ANZECC 99% Protection			27				0.01	1200			8	1.0	2.4	1	0.06		1.0
ANZECC 95% Protection			55				1	1900			11	1.4	8	24	0.2		3.4
ANZECC 90% Protection			80				6	2500			13	1.8	15	94	0.4		5.6
ANZECC 80% Protection			150				40	3600			17	2.5	31	360	0.8		9.4

Table 2.5. Sediment metal concentrations as measured on one occasion between Jan-Mar 2015. ANZECC interim sediment quality guideline (ISQG) values are included (Anzecc 2000).

Site Number	Na mg/kg	Mg mg/kg	Al mg/kg	K mg/kg	Ca mg/kg	V mg/kg	Cr mg/kg	Mn mg/kg	Fe mg/kg	Co mg/kg	Ni mg/kg	Cu mg/kg	Zn mg/kg	As mg/kg	Cd mg/kg	Sb mg/kg	Pb mg/kg
1	<100	2690	6570	572	3200	13.6	10.4	130	9870	3.9	7.7	7.8	46	1.4	<0.1	0.1	16.1
2	<100	1640	3820	393	3400	8.7	14.1	100	6410	2.3	5.4	6.8	47	3.1	<0.1	0.1	10.1
3	<100	2320	5180	437	2100	11.3	9.3	130	9540	3.6	6.5	4.3	53	3.1	<0.1	<0.1	7.2
4	<100	2070	4620	400	1900	9.9	7.3	180	8090	4.0	6.3	3.6	63	1.1	<0.1	<0.1	6.3
5	100	2070	5090	472	4300	14.7	23.7	130	9600	4.2	11.5	101.7	827	5.0	0.6	1.9	100.7
7	<100	2190	4910	380	2100	11.5	8.4	120	8000	3.6	6.3	6.6	175	1.8	<0.1	0.2	24.9
8	<100	2470	4980	410	2500	13.2	12.2	410	13240	6.1	9.2	15.8	293	10.6	0.2	0.5	63.6
9	200	3120	8690	648	4200	34.0	17.7	1190	39810	25.7	14.4	18.9	540	43.2	0.3	0.4	74.6
11	<100	2310	5590	513	2800	12.1	9.0	140	10800	5.3	7.1	10.9	137	1.9	<0.1	0.2	16.2
13	<100	2240	5220	543	2800	11.8	11.0	120	8620	3.6	7.1	15.4	136	3.6	0.2	0.3	52.4
14	<100	2070	4910	453	2500	10.8	9.6	110	7900	3.5	6.4	14.2	148	1.9	0.2	0.3	55.9
15	<100	1970	4550	392	2100	9.6	8.3	110	7650	3.3	5.9	12.3	140	1.4	<0.1	0.1	40.2
17	<100	2070	4660	372	2000	10.5	8.0	100	7530	2.7	6.1	11.9	84	1.9	<0.1	0.1	23.9
19	<100	2240	4740	417	2300	10.8	8.4	120	7920	3.3	6.3	7.4	90	0.9	<0.1	0.1	21.3
20	<100	1150	2800	236	1300	6.6	4.7	80	5280	2.1	3.5	22.3	72	1.6	<0.1	0.2	37.1
21	<100	1850	3980	313	1700	9.3	7.3	110	6960	2.5	5.3	12.7	98	1.0	<0.1	<0.1	22.3
22	<100	1040	2250	227	1200	5.9	4.9	70	4970	1.8	3.3	7.1	54	2.1	<0.1	<0.1	33.6
26	<100	2600	5750	622	2800	12.2	9.6	160	10560	4.9	7.7	10.2	157	3.8	<0.1	0.2	29.9
28	<100	2180	4820	483	2600	11.0	14.8	160	9460	4.5	7.3	60.7	394	8.0	1.0	0.2	64.0
33	109.57	3050	7120	692	3300	15.9	15.2	200	11600	5.7	9.9	23.8	339	5.9	0.2	0.4	74.8
Anzecc Low							80				21	60	200	20	1.5	2	50
Anzecc High							370				52	270	410	70	10	25	220


Water Arsenic (µg/L)
Sites by DMI
11 14 13 22 15 4 21 19 26 17 3 20 8 1 9 28 5 33 2 7
Sediment Arsenic (mg/kg dry weight)
Sites by SMI
4 3 2 1 19 11 21 17 22 7 26 20 15 13 14 8 33 28 9 5
Water Cadmium (µg/L)
Sites by DMI
Sediment Cadmium (mg/kg dry weight)
Sites by SMI
Water Chromium (µg/L)
Sites by DMI
Sediment Chromium (mg/kg dry weight)
Sites by SMI

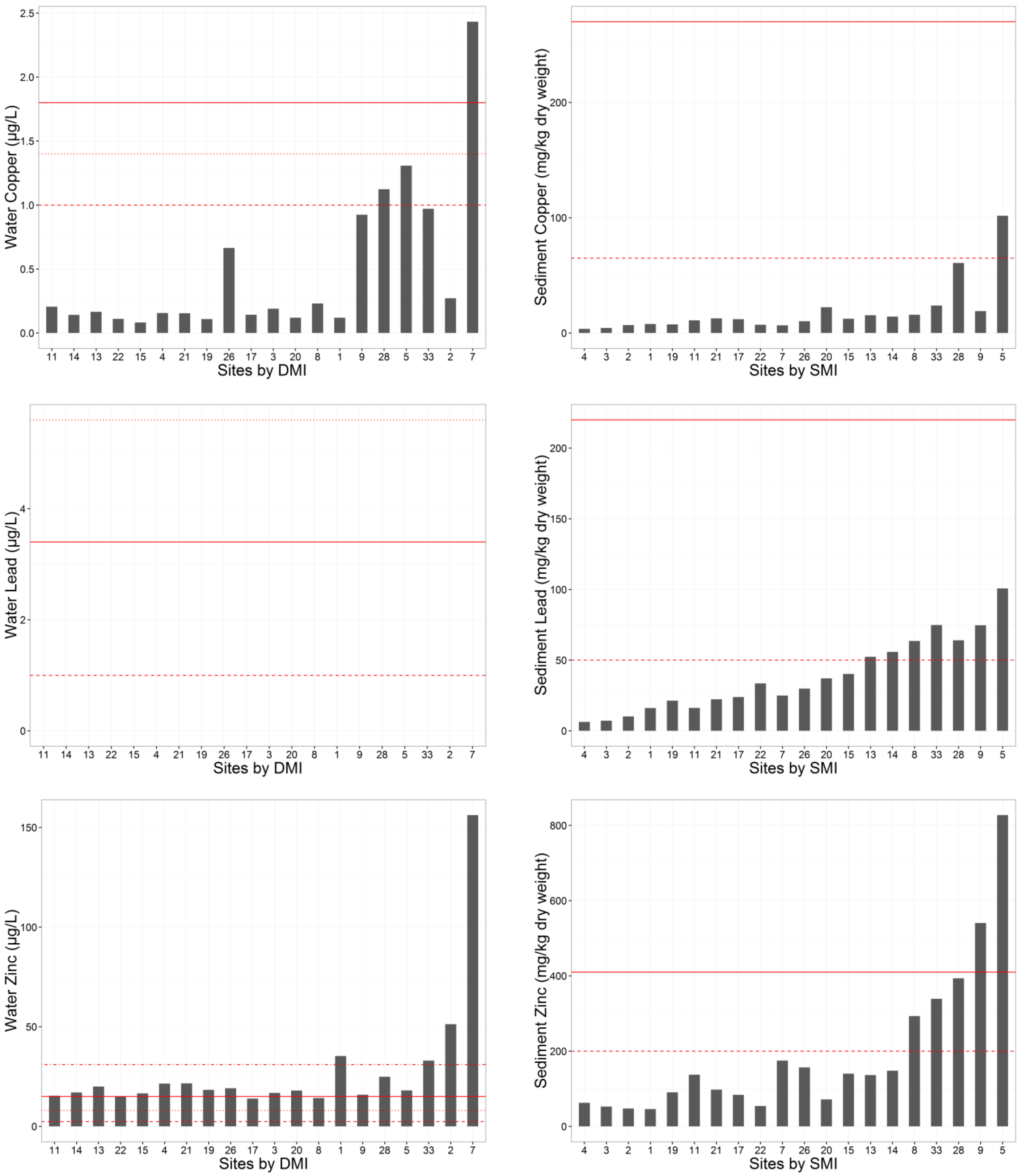


Figure 2.4. Dissolved arsenic (previous page, top left), sediment arsenic (previous page, top right) dissolved cadmium (previous page, middle left), sediment cadmium (previous page, middle right) dissolved chromium (previous page, bottom left), sediment chromium (previous page, bottom right), dissolved copper (top left), sediment copper (top right) dissolved lead (middle left), sediment lead (middle right), dissolved zinc (bottom left), sediment zinc (bottom right), as measured on one occasion at each survey site. For dissolved metal plots, dotted-dashed line represents ANZECC guideline for protection of 80% of species, solid red line indicates ANZECC guideline for protection of 90% of species, dotted red line indicates ANZECC guideline for protection of 95% of species, and dashed red line indicates ANZECC guideline for protection of 99% of species. For sediment plots, solid red line indicates ANZECC ISQG-High guideline values, dotted red line indicates ISQG-Low trigger values (Anzecc 2000).

Table 2.6. Relative contributions to principle component analysis (PCA) used to obtain metal indices for dissolved and sediment metals.

Dissolved metal PCA loadings				
Cr	Cu	Zn	As	Cd
0.274	0.457	0.523	0.496	0.444
Sediment metal PCA loadings				
Pb	Cu	Zn	As	
0.492	0.423	0.515	0.321	

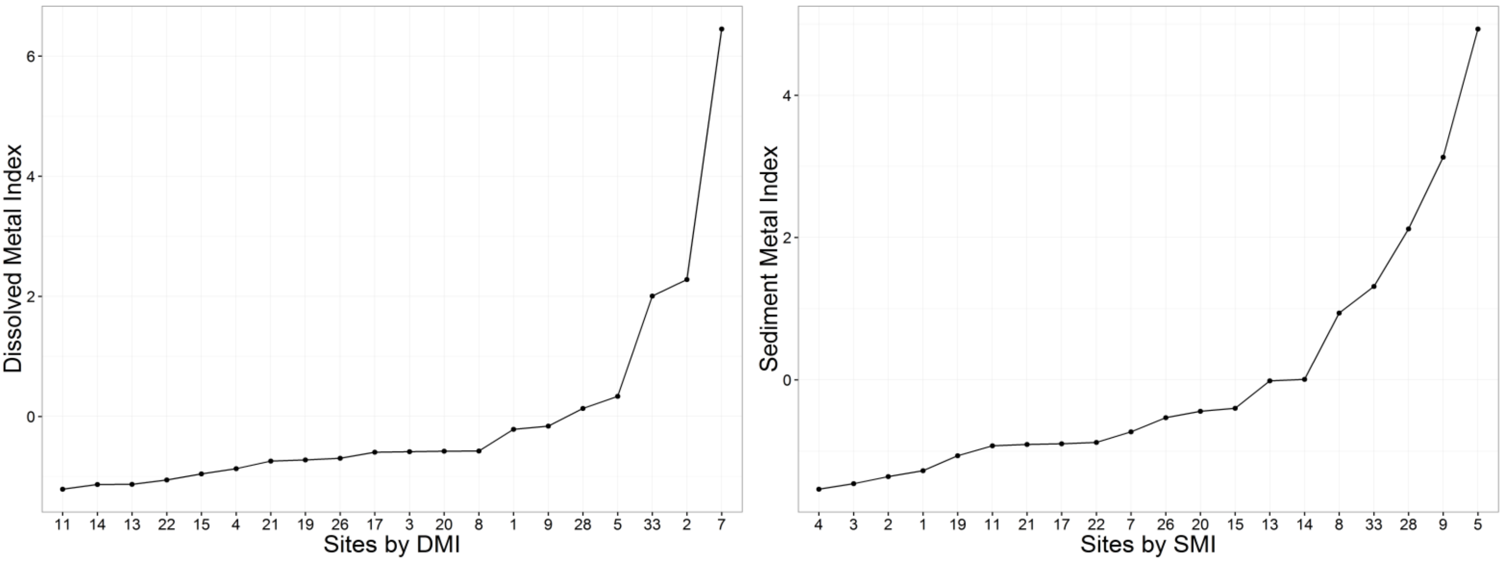


Figure 2.5. Gradients obtained from PCA analysis of dissolved (left) and sediment (right) metals across my 20 study sites.

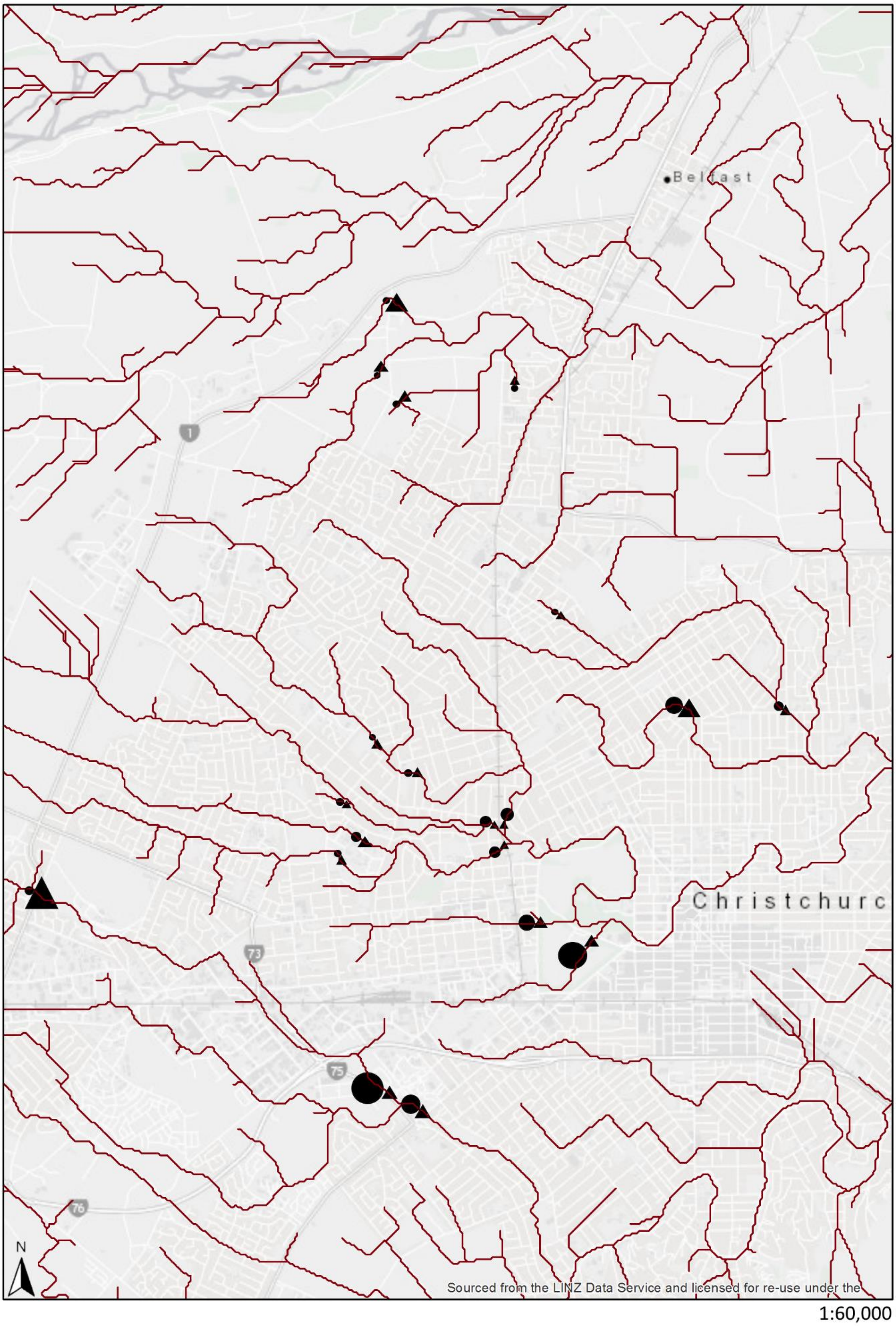


Figure 2.6. Christchurch City's urban waterways. SMI (●) and DMI (▲) are shown for my 20 study sites. Size of symbols represent approximate scores for metal indices, with a large symbol indicating high metal concentrations and a small symbol indicating low metal concentrations as measured between Jan-Mar 2015.

2.3.5 - Invertebrate communities

A total of 71369 macroinvertebrates were collected in Surber samples. Combining qualitative and quantitative samples resulted in 31 different taxa across my 20 sites. Caddisflies (Trichoptera) and true flies (Diptera) showed the most diversity, with 9 different taxa each. Crustaceans, and snails and bivalves (Mollusca) were the next most diverse group with 5 taxa, while mayflies (Ephemeroptera), worms (Annelida), and flat worms (Platyhelminthes) only had 1 taxa each. *Potamopyrgus antipodarum* snails were the most abundant taxa (33% of numbers across all sites), followed by oligichaete worms (27%) and the orthclad midge (7.5%). No stoneflies, beetles, odonata, or true bugs were collected in any of my 20 sites.

2.3.6 - Invertebrate community metrics

Invertebrate community metrics indicated that invertebrate communities across my 20 study sites were generally degraded (Figure 2.6). Total taxonomic richness ranged from 26 at Site 2, to 10 at Site 9, while the number of EPT taxa ranged from 9 at Site 2 to 0 at Site 33, though Sites 5, 9, 19, and 26 only contained 1 EPT taxa each. Percentage abundance of EPT taxa ranged from 43% at Site 1, to 0% at Site 33, while percentage abundance of Mollusca ranged from 84% at Site 9, to 7% at Site 13. MCI scores ranged from 95 (fair) to 62 (poor), while QMCI scores ranged from a fair 4.88 to a poor 1.36. The mean MCI score was 77 (± 2.5) across all sites, while the mean QMCI was 2.99 (± 0.2), both of which are considered poor and indicitive of sites with probable severe pollution (Stark et al. 2007).

2.3.7 - Drivers of macroinvertebrate community composition

Regression analysis between metal indices and invertebrate metrics (total taxa, EPT taxa, percentage EPT abundance, percentage mollusca abundance, MCI, and QMCI) revealed a significant negative relationship between SMI and total taxa, EPT taxa and MCI. In contrast there was no significant relationship between DMI and invertebrate metrics (Figure 2.7).

Stepwise regression identified the environmental variables that best explained a range of macroinvertebrate community variables (Table 2.7). SMI was the most important environmental variable for five of the community metrics, more than any other environmental variable. In total, SMI was selected as a predictor in 9 of the 13 community metrics, and was statistically significant ($p<0.05$) in 7 of those. DMI was a predictor variable in 2 of the 13 models, and was statistically significant ($p<0.05$) in both. SMI was the sole predictor variable for both the MCI and EPT Taxa models. Impervious surface area was the most important predictor for percent EPT abundance, which was also the only community metric for which DMI and SMI were selected in the same model. After SMI at nine, width was the predictor variable selected second most frequently in models at eight times, followed by impervious surface area at six, and shading at five.

CCA analysis of community composition revealed two significant axis which cumulatively explained 26% of the variation (Figure 2.8). Stepwise selection identified three environmental variables (DMI, SMI, and percent impervious surface cover) that strongly influenced community composition. DMI and SMI were most strongly correlated with Axis 1 (R^2 = 0.42, $P < 0.005$; R^2 = 0.23, $P < 0.05$ respectively), while impervious surface area was most strongly correlated with Axis 2 (R^2 = 0.44, $P< 0.001$). Highly impacted streams (high SMI, DMI, and impervious surface area) tended to occur to the right side of axis 1 and the top of axis 2, while less impacted streams (low SMI, DMI, and impervious surface area) occurred to the left of axis 1 and the bottom of axis 2.

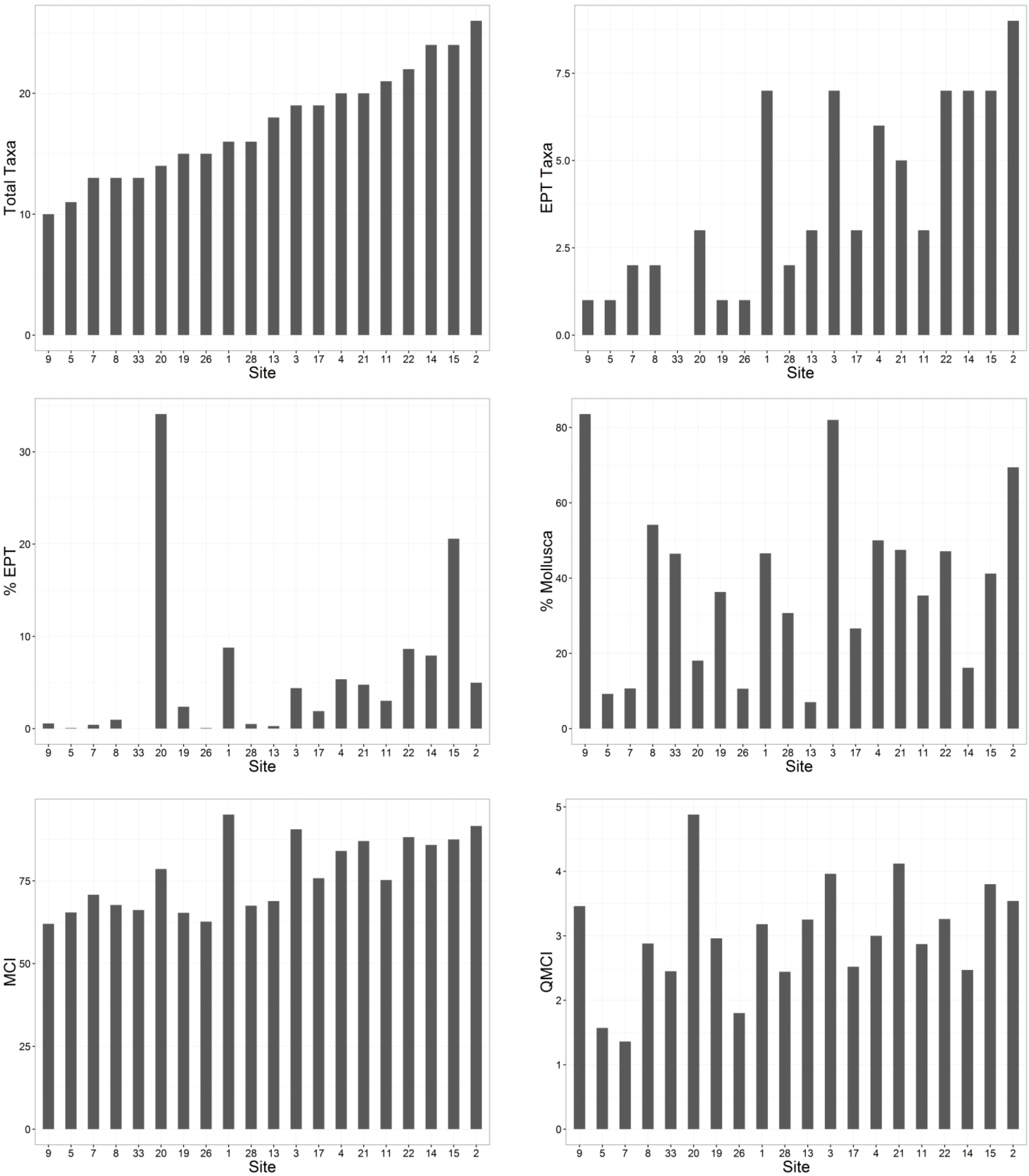
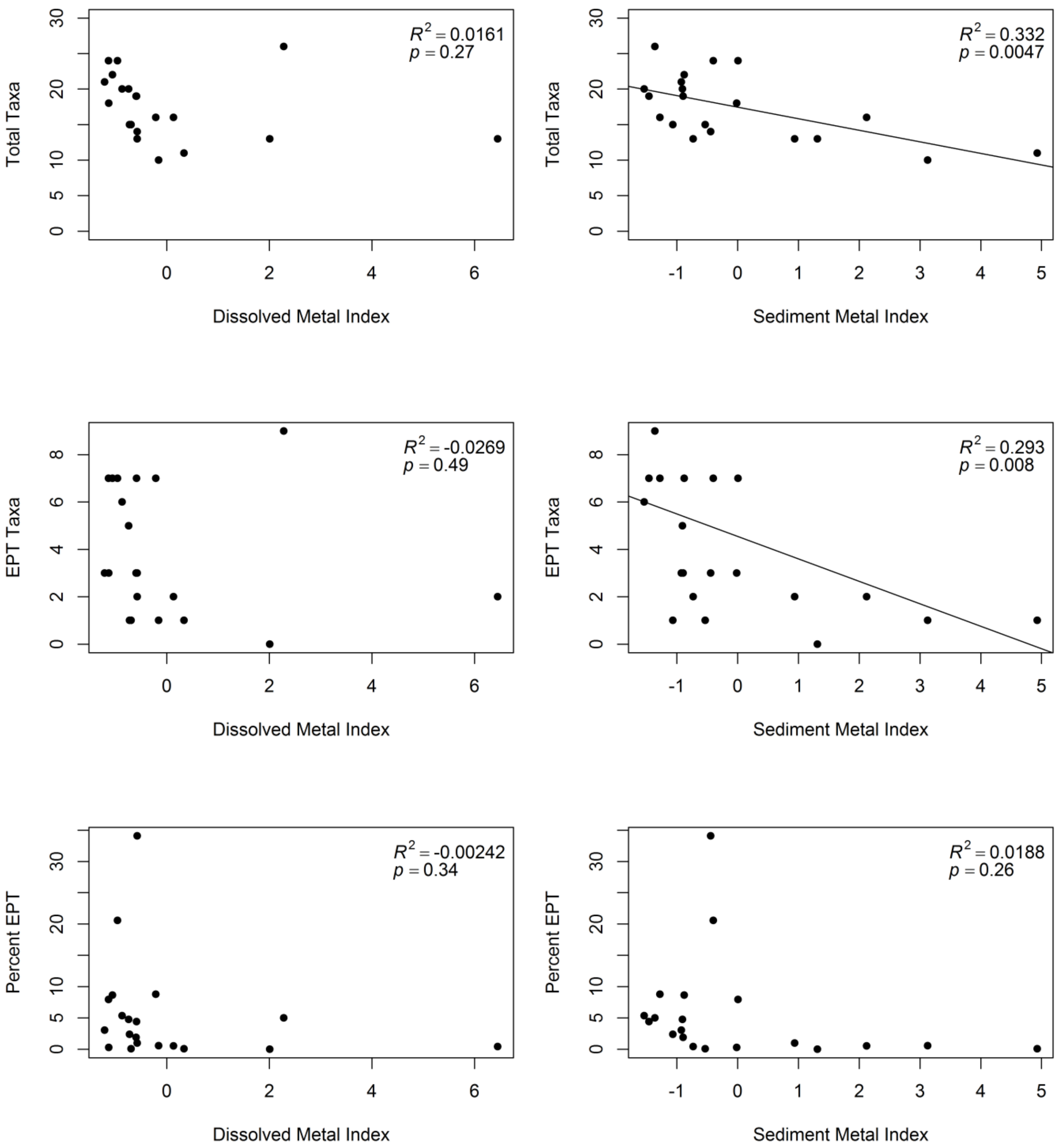


Figure 2.7. Total taxa (top left), EPT taxa (top right), percentage EPT abundance (middle left), percentage Mollusca abundance (middle right), MCI scores (bottom left), and QMCI scores (bottom right) across my 20 study sites sampled between Jan-Mar 2015.



Total Taxa
Dissolved Metal Index
$R^2 = 0.0161$
$p = 0.27$
Total Taxa
Sediment Metal Index
$R^2 = 0.332$
$p = 0.0047$
EPT Taxa
Dissolved Metal Index
$R^2 = -0.0269$
$p = 0.49$
EPT Taxa
Sediment Metal Index
$R^2 = 0.293$
$p = 0.008$
Percent EPT
Dissolved Metal Index
$R^2 = -0.00242$
$p = 0.34$
Percent EPT
Sediment Metal Index
$R^2 = 0.0188$
$p = 0.26$

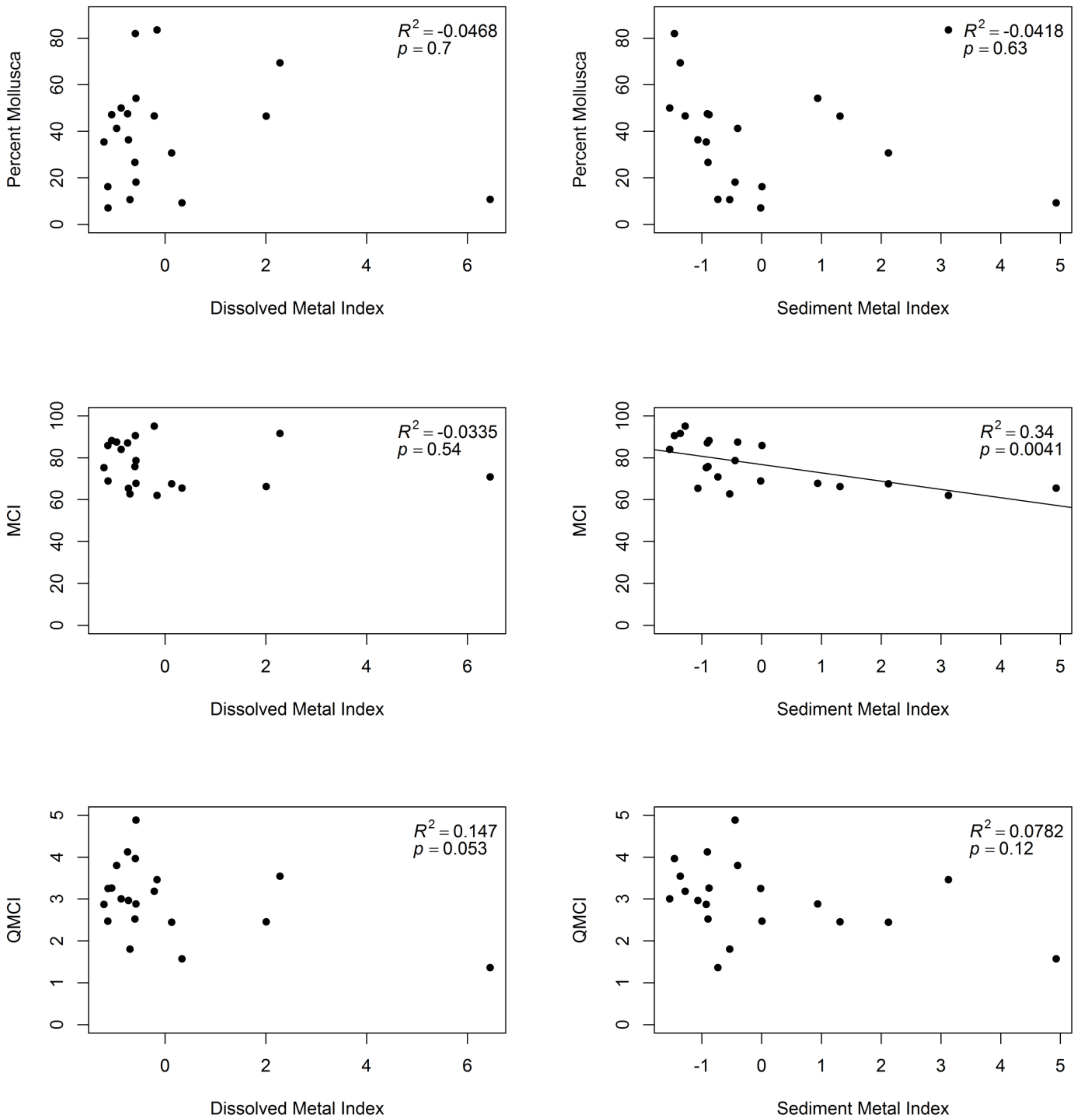


Figure 2.8. Scatter plots showing the relationships between SMI and DMI on total taxa (previous page top row), EPT taxa (previous page middle row), percentage EPT abundance (previous page bottom row), percentage Mollusca abundance (top row), MCI (middle row) and QMCI (bottom row) across my 20 study sites sampled Jan-Mar 2015. Relationships were significant at p<0.05.

Table 2.7. Results of stepwise regression between the physical and chemical condition of waterways and macroinvertebrate community response variables for 20 sites across Christchurch's urban waterways.

Dependent Variable	Predictors in Model	Partial R^2	Standardised Coefficient	Predictor p value	Predictor F	Model Adjusted R^2	Model F	Model p value
Taxonomic Richness	SMI	0.43	-0.47	<0.001	27.60	0.69	$F_{6,13} = 8.19$	<0.005
	Substrate Index	0.43	0.51	<0.05	4.88			
	Width	0.35	0.44	<0.05	7.58			
	DO		-0.36	0.08	3.52			
	% Impervious		0.24	0.08	3.67			
	Shading		0.21	0.19	1.93			
Margalef's Index	Width	0.36	0.49	<0.05	6.55	0.56	$F_{5,14} = 5.84$	<0.005
	SMI	0.33	-0.46	<0.005	16.73			
	Velocity		0.27	0.16	2.15			
	DO		-0.26	0.2	1.85			
	% Impervious		0.21	0.21	1.74			
EPT Taxa	SMI	0.30	-0.55	<0.05	7.64	0.26	$F_{1,18} = 7.64$	<0.05
% EPT	% Impervious	0.50	-0.61	<0.001	16.48	0.63	$F_{3,16} = 11.77$	<0.001
	DMI	0.40	-0.47	<0.005	12.95			
	SMI	0.27	-0.35	<0.05	5.87			
QMCI	SMI	0.35	-0.62	<0.05	6.31	0.57	$F_{4,15} = 7.06$	<0.005
	Shading	0.12	0.31	<0.005	16.01			
	Conductivity		0.53	0.06	4.14			
	% Impervious		-0.26	0.12	2.76			
MCI	SMI	0.37	-0.61	<0.005	10.8	0.34	$F_{1,18} = 10.80$	<0.005
% Mollusca	Shading	0.38	0.67	0.06	4.26	0.37	$F_{2,17} = 6.59$	<0.01
	Width	0.34	-0.62	<0.01	8.93			
% Chironomid	Width	0.30	0.47	<0.05	6.94	0.42	$F_{3,16} = 5.60$	<0.01
	Substrate Index	0.26	0.42	<0.05	5.62			
	% Impervious		0.43	0.06	4.24			
% Shredder	SMI	0.30	-0.51	<0.01	8.72	0.34	$F_{2,17} = 5.85$	<0.05
	Width		0.33	0.10	2.30			
% Filter Feeders	Width	0.49	0.68	<0.05	6.21	0.48	$F_{5,14} = 4.47$	<0.05
	Substrate Index	0.41	-0.57	<0.05	5.15			
	Conductivity	0.41	-0.74	<0.05	5.96			
	SMI		0.36	0.09	3.25			
	% Impervious		0.23	0.20	1.77			
% Grazers	Width	0.48	-0.70	<0.005	10.82	0.50	$F_{4,15} = 5.73$	<0.01
	Shading	0.19	0.45	<0.05	6.25			
	SMI		-0.57	0.13	2.60			
	Conductivity		0.52	0.09	3.27			
% Collectors-Browsers	Shading		-0.61	0.06	4.21	0.22	$F_{2,17} = 3.68$	<0.05
	Width		0.41	0.09	3.15			
% Predators	Conductivity	0.32	-0.54	<0.05	7.68	0.53	$F_{3,16} = 8.24$	<0.005
	DMI	0.19	-0.42	<0.005	15.53			
	Shading		0.60	0.24	1.50			

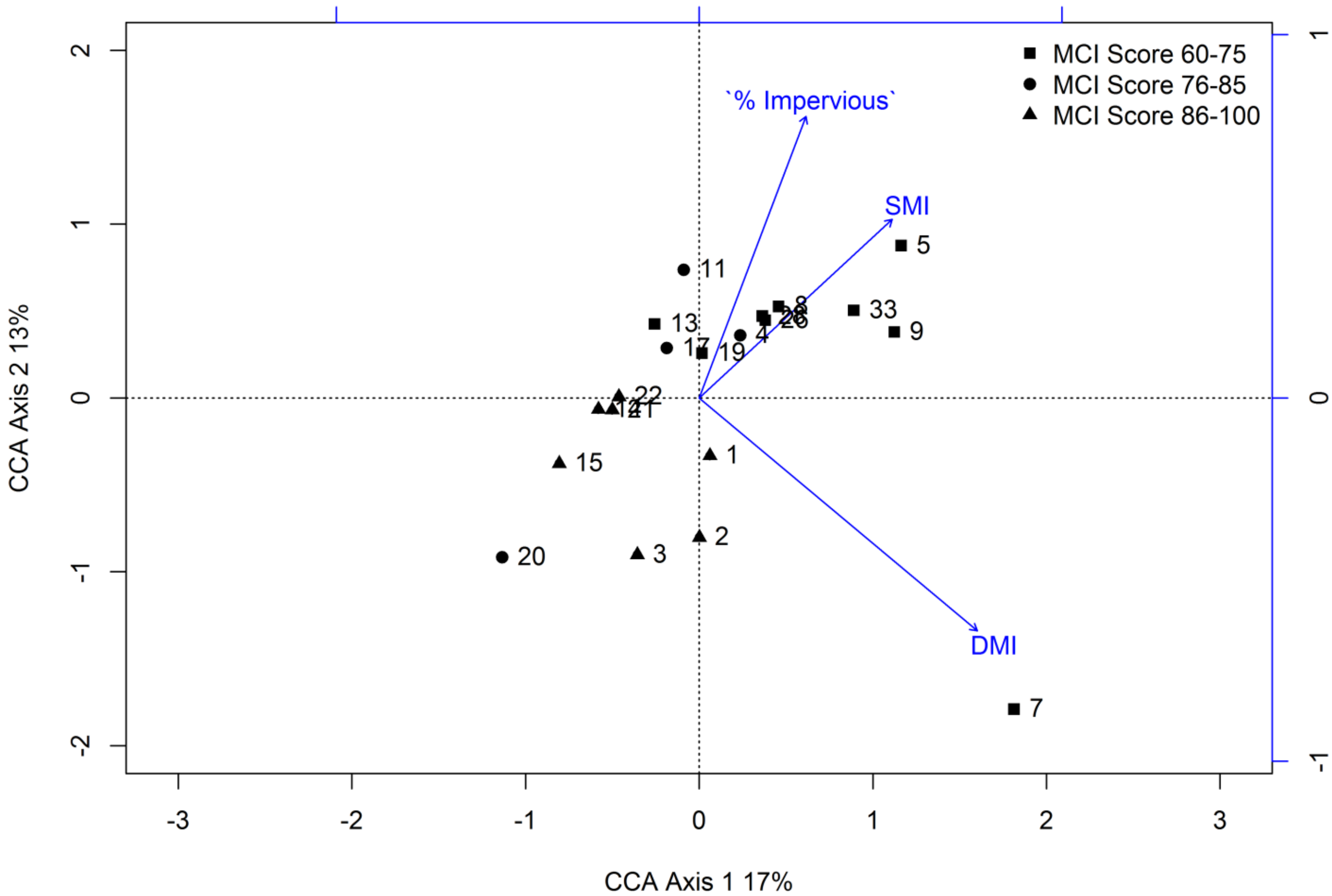


Figure 2.9. Canonical correspondence analysis (CCA) for macroinvertebrate community composition in 20 reaches in Christchurch's urban waterways and three significant (P <0.05) environmental predictors (SMI, percent impervious surface cover, and DMI) that were selected by stepwise model building. Sites are divided into groups based on MCI scores to give an indication of invertebrate community health at each site.

2.4 Discussion

Storm water infrastructure in urban areas is designed to collect water during rain events and convey it out of cities, usually via stream and river networks. Pollutants which collect on impervious surfaces in these catchments are frequently transported to urban waterways (Williamson 1986; Hatt et al. 2004; Brown et al. 2006). As a result, heavy metals are frequently found in high concentrations in the water column and sediment of urban waterways (Wilber et al. 1979; Bryan et al. 1992; Neal et al. 1997; Horowitz et al. 1999).

Studies of invertebrates in New Zealand's urban streams have found that the response of invertebrates to urbanisation is similar to that found internationally (Suren et al. 2005b; Walsh et al. 2005b). New Zealand urban streams tend to be species poor, and dominated by tolerant taxa such as the snail *Potamopyrgus antipodarum,* chironomid midges, and oligochaete worms, with sensitive taxa such as mayflies and stoneflies rarely found (Suren 2000; Hall et al. 2001; Collier et al. 2009).

Studies in Christchurch have found metals in urban waterways at levels of environmental concern. Blakely et al. (2005) measured sediment metal concentrations in Okeover Stream, Waimairi Stream and the Avon River while conducting a study to quantify the biological response of stream rehabilitation efforts. Mean Pb concentrations were found to exceed ISQG low guidelines in Waimairi Stream (50 mg/kg), while sampling of sediment along several reaches of Okeover Stream found concentrations of Zn at up to 332 mg/kg, Pb at up to 180 mg/kg, and Cu at up to 200 mg/kg, all exceeding ISQG guidelines. These metals are thought to be a contributing factor to the lack of ecological response to rehabilitation efforts in Okeover Stream (Blakely et al. 2005; Winterbourn et al. 2007), and also an underlying factor in the lack of response in other stream rehabilitation projects in Christchurch (Suren et al. 2005a).

The metal concentrations measured in my study were similar to the results of previous Christchurch City Council surveys (Gadd et al. 2014; Gadd 2015). I found Zn and Pb were the primary metals which occurred in high concentrations in the sediment, while Zn, Cd and Cr occurred at high concentrations in the water (Table 2.7). Elevated Zn, Cu and Pb have been found commonly in Christchurch in City Council surveys (Margetts 2014; Gadd 2015) and past studies (Blakely et al. 2005; O'Sullivan et al. 2012). Blakely et al. (2005) measured Cd concentrations in the sediment of Okeover Stream, Waimairi Stream, and the Avon River, which were similar to concentrations found in my study. However, the study of dissolved Cd and Cr concentrations in other urban waterways in Christchurch is limited, and these metals are typically not included in Christchurch City Council water quality monitoring. Though several studies show that Cd and Cr are commonly found at elevated levels in urban streams internationally (Wilber et al. 1979; Van Metre et al. 2003).

Three sites in my study had particularly high concentrations of metals, Site 5 (Curlett's Road Stream), Site 7 (Paparua Stream), and Site 9 (Addington Brook). These sites also contained some of the most degraded invertebrate communities, and were among the sites with lowest taxonomic richness, EPT taxa, and EPT abundance (Figure 2.7). At Curlett's Road Stream, sediment Zn exceeded ISQG-high guidelines by 2-fold with a concentration of 827 mg/kg (Anzecc 2000). High concentrations of Cu and Pb were also found in the sediment at Curlett's Road Stream. This is likely a result of the close proximity to major roads and high levels of industrial land use in the catchment, including a bronze and brass smelter. Addington Brook had sediment Zn concentrations of 540 mg/kg, as well as concentrations exceeding ISQG-high guidelines for Pb and As in the sediment. Arsenic was historically used in sheep dip, and high As concentrations at Addington Brook could be a result of historical contamination from a sheep dip at the former Canterbury Saleyard site on Deans Ave, while historical contamination from the Addington Railway Workshop could also contribute to high metal concentrations at this site (Gadd et al. 2014). Dissolved Zn at Paparua Stream was particularly high at 156 µg/L, while dissolved Cu was 2.4 µg/L. High concentrations in Paparua Stream could be due to the close proximity of the stream to the highly used State Highway 1 and stormwater runoff from residential land use in the upper catchment, including major works on a sub-division under development. The high heavy metal concentrations in these streams in close proximity to high traffic roads and industrial land use are similar to results of other studies. In Sydney, Davis et al. (2011) found that heavy metals in bulk atmospheric deposition were strongly correlated with road proximity and traffic density , likely due to vehicle engine emissions, tire wear, and brake lining abrasion which several studies have recognised as a major source of deposited metals (Davis et al. 2001; Zanders 2005; Kadi 2009). Additionally, metal roofing, a common impervious surface in industrial and residential urban areas, can both accumulate deposited contaminated particles and contribute metals through the weathering of roofing materials (Van Metre et al. 2003).

2.4.1 - Effects of metals in water and sediment on macroinvertebrate communities

Invertebrate communities across my streams showed relatively low diversity, with a paucity of sensitive EPT taxa, particularly at more degraded sites. Urban sites were dominated by sensitive taxa such as *Potamopyrgus* snails and oligochaete worms, though sensitive taxa such as *Deleatidium* were found in peri-urban streams. These findings are similar to other studies both internationally and in New Zealand (Paul et al. 2001; Suren et al. 2005a; Collier et al. 2009). In this study I found that heavy metal concentrations explained the greatest amount of the variation in macroinvertebrate communities across my study sites. In particular, metals bound to the sediment were better indicators than metals in the water, and were the primary variable in structuring macroinvertebrate communities. This result is consistent with that of Beasley et al. (2003) in Yorkshire, UK, who used partial Canonical Correspondence Analysis (pCCA) analysis to investigate the effect of sediment bound metals on invertebrate communities across 62 urban stream sites. Metals in sediment were found to explain 24% of the variation in macroinvertebrate community composition, and EPT taxa were absent from streams with metal polluted sediment. Medeiros et al. (1983) in Greenfield, Massachusetts, investigated the toxicity of urban runoff to macroinvertebrates in an urban river reach, storm drain channel, and a control site. During rain events, invertebrate diversity was reduced across all stations. However, during periods of no runoff invertebrate community diversity remained constant at the control site and storm drain channel, but was reduced further in the urban river section. Additionally, uptake of heavy metals by invertebrates in the urban river reach also increased during dry-weather periods, suggesting that during dry-weather, sediment was a major source of metals.

A potential reason for sediment bound metals being more important than dissolved metals in my study could be the limited ability of smaller tributaries to flush sediment (Winterbourn et al. 2007). Many of Christchurch's urban streams are springfed, and this, in combination with the flat topography of Christchurch, results in low gradient streams with relatively stable flows (Suren 2000; Suren et al. 2005b). Despite increased flows during rain events and increased flashiness of stream hydrographs due to high levels of impervious surface area in the urbanised catchments, the capacity of Christchurch's urban streams to flush sediment appears to be relatively low (Harding et al. 2015). As sediment can accumulate many times the amount of metals as in the water column at any given time, it seems likely that this would contribute to an environment where heavy metal concentrations remain for extended periods of time at levels with the potential for chronic toxic effects (Peijnenburg et al. 2003). Furthermore, due to the tendency of sediment to accumulate metals, sediment metal concentrations are likely to be a longer-term measurement of metal contamination sites (Rhoads et al. 1999).

2.4.2 - Effects of other environmental variables on macroinvertebrate community composition

Stream width was also identified as an important variable by univariate model selection methods. Stream width is probably a surrogate for several characteristics of habitat condition, including a reflection of habitat size. The summer of 2015, when my invertebrate sampling occurred, was particularly dry, likely resulting in a reduction of flow and wetted width in many streams. Several studies have found that habitat size can limit invertebrate diversity, density, and community structure (McIntosh et al. 2002; Dewson et al. 2007; McHugh et al. 2015). Brönmark et al. (1984) for example, found that stream size was the most important factor determining species richness over 22 streams in Bornholm, Denmark, with species number increasing with stream area. They surmised that an increase in stream heterogeneity with stream size, and the possibility of immigration rates and food diversity scaling with stream size could be key factors in this phenomenon.

Impervious surface area was also selected in several models. Catchments with high impervious surface area might represent systems with significant car parking and roading infrastructure, with possible high levels of PAHs in stream sediments, a variable not accounted for in this study. Surveys of sediment quality undertaken for Christchurch City Council found that impervious surface cover was linked with PAH contamination (Gadd et al. 2014). Suren et al. (2005a) also found several Christchurch streams had high PAH concentrations, exceeding ANZECC guidelines, which could cause reduced invertebrate diversity through direct toxicity and potential interaction effects with metal contaminants (Beasley et al. 2002).

Another explanation for impervious surface area being selected in several models could be an interaction between disturbance, invertebrate body-size, and tolerance to heavy metals. High amounts of impervious area within a catchment can create flashy stream hydrographs due to increased efficiency of water transport from surfaces to streams via storm water pipes (Walsh 2004a). In the prealpine Neckar River in Switzerland, Matthaei et al. (1997) found that disturbance from floods can result in communities with increased proportions of small-sized species and young age classes. Body size measurements of the three most common taxa (Chironimidae, and the two mayflies *Baetis* spp. and *Rithrogena* spp.) revealed that post-flood, the relative contribution of <1mm size class to the total abundance of each taxa was increased. Changes in macroinvertebrate size distributions could alter the tolerance of these communities to heavy metals. In a laboratory experiment in Cardiff, UK, Green et al. (1986) found that life history stages had a significant impact on cadmium resistance in the isopod *Asellus aquaticus,* with larger juveniles and adults significantly more resistant to cadmium than smaller specimens. In the USA, Kiffney et al. (1994) found that small size classes of the mayfly *Drunella grandis* were more sensitive to a metal mixture (Cd, Cu, and Zn) than large mayfly instars from the same population. Kiffney et al. (1996) found a similar relationship across three mayflies (*Baetis* *tricaudatus*, *Ephemerella infrequens*, and *Rhithrogena hageni)*, and the plecopteran *Pteronarcella badia*, confirming that larger invertebrate animals were more tolerant to a combination of metals.

Imperviousness was not closely correlated with either SMI (Pearson correlation coefficient, ρ = 0.29) or DMI (Pearson correlation coefficient, ρ = -0.28). This could be because the proportion of impervious surface area in a catchment, or total imperviousness, does not account for drainage connections. Total impervious area within a catchment includes all impervious surfaces, including those which do not contribute run off to waterways, for example a shed in the middle of a parkland (Booth et al. 1997). Several studies have found drainage connection (the proportion of impervious surface area directly connected to waterways by pipes or drains) is a better predictor of pollutant loads, and ecological condition of streams (Booth et al. 1997; Hatt et al. 2004; Taylor et al. 2004; Walsh 2004b). Furthermore, it is important to note that the FENZ database information for the impervious surface data (for the upstream catchment of streams) used in my study are from estimates (Leathwick et al. 2010). For example, Okeover Stream is attributed a value of 15%, where other studies have found impervious cover in the Okeover Stream catchment at closer to 40% (Charters et al. 2014). However, as the FENZ database has values for all waterways in my study, and to keep consistent values, the estimates from the FENZ database were used.

Shading was also identified as an important factor. Shading could reflect potential effects of riparian vegetation in urban streams. Hession et al. (2002,2003) found that the presence of riparian forest affected geomorphology, algal biomass and concentrations of bioavailable nutrients independent to an urban gradient. Riparian buffers can act as a barrier to overland flow of stormwater and associated pollutants, though the direct connection between impervious surfaces and streams via piped stormwater systems could negate this effect (Walsh et al. 2005b). Overhanging trees and riparian vegetation can also be an important source of organic carbon to streams through leaf litter deposition (Bilby et al. 1992), while large branches and logs can increase stream habitat heterogeneity (O'connor 1991; Reid et al. 2010). Organic carbon can act as a food source in streams and is an important factor in the bioavailability of trace metals (Schlesinger et al. 1981; Tessier et al. 1987). Metal speciation and binding to organic carbon can greatly influence uptake and toxicity of metals to invertebrates, with organic carbon correlating negatively to metal bioavailability in several studies (Luoma et al. 1982; Aiken et al. 2011). While temperature and shading are not strongly correlated in this study, shading of a stream can also reduce temperatures of streams, and act as a thermal buffer (Poole et al. 2001).

2.4.3 - The structuring of invertebrate communities in Christchurch's urban streams

The aims of this chapter were to identify a gradient of metal contamination across Christchurch's urban streams and to investigate the response of macroinvertebrates communities to a heavy metal gradient. Urbanisation can have major effects on urban waterways, degrading stream environments and impairing ecological communities (Walsh et al. 2005b). Due to the multivariate nature of stressors to urban stream biotic communities (Paul et al. 2001), the investigation of specific mechanisms leading to invertebrate community degradation can be challenging. However, stormwater inputs, including contaminants such as heavy metals, are thought to be among the key drivers of the ecological degradation commonly seen in urban waterways (Walsh 2004a). This study has demonstrated that in Christchurch's urban landscape, heavy metals are likely to be among the prevailing influences on stream macroinvertebrate community structure. With increasing proportions of human population growth occurring in cities in New Zealand (Statistics New Zealand 2009), urbanisation of catchments is likely to increase. Mitigation of stormwater borne contaminants is essential if the ecological values of Christchurch's urban streams are to be maintained or improved.

CHAPTER 3

TESTING *IN SITU* INVERTEBRATE SURVIVAL.

3.1 INTRODUCTION

Invertebrate communities in urban waterways are subjected to multiple stressors (see review by Paul et al. 2001). In particular, water and sediment in urban waterways contaminated by high heavy metal concentrations are known to impact invertebrate communities (Medeiros et al. 1983; Beasley et al. 2003). However, other constraints including barriers to adult dispersal and life cycle completion and biotic interactions are also important in controlling the composition and perpetuation of aquatic insect communities (Smith et al. 2009). Understanding the interactions between biotic, local habitat, and dispersal constraints, and their relationship with overarching landscape-scale constraints is necessary when determining limiting factors to anthropogenically-impacted aquatic macroinvertebrate communities (Parkyn et al. 2011) (Figure 3.1).

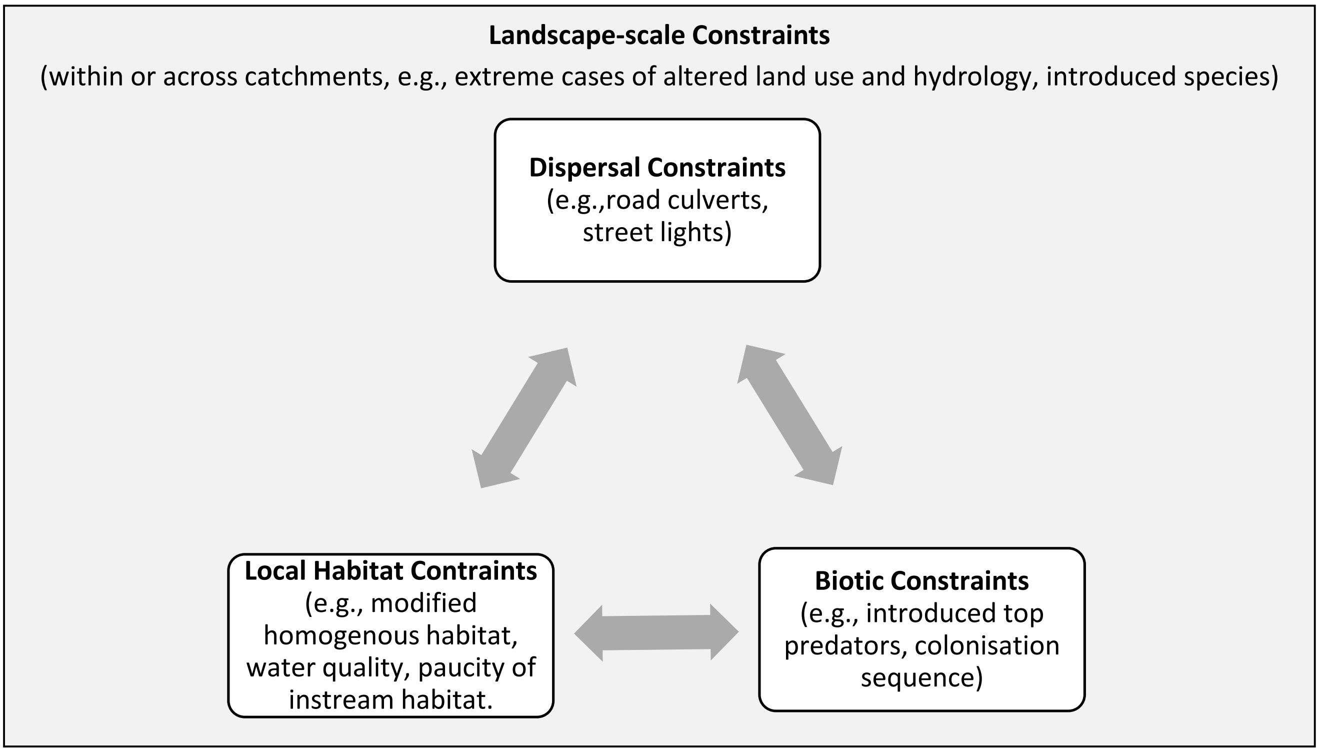


Figure 3.1. Conceptual framework for constraints to stream invertebrate communities. Dispersal, local habitat, and biotic constraints interact with each other, and also landscape constraints, to influence invertebrate communities. Adapted from Parkyn et al. (2011).

Sub-lethal chronic levels of heavy metals have the potential to promote metal tolerance in invertebrates (Klerks et al. 1987), with potential trade-offs and susceptibilities to other stressors. In the USA, Clements (1999) found that mayflies (*Rhithrogena hageni*) from a metal polluted river showed higher tolerance to metals (Cd, Cu, Zn), but a higher susceptibility to predation than mayflies from a reference river. This was thought to be a result of chronic metal exposure affecting the ability of mayflies to detect or avoid predators. This suggests that metal pollution can influence the outcomes of biotic interactions in waterways, further complicating invertebrate community patterns and dynamics in metal impacted waterways.

Dispersal constraints are also likely to be an important factor in urban waterways due to highly modified stream networks, riparian conditions, and instream habitat. Urban et al. (2006) found that the best predictors of local invertebrate community diversity and abundance in streams along a rural-urban gradient in Connecticut, USA were riparian vegetation and watershed landscape structure. Remnant forest cover alone explained 70% of the variation in invertebrate richness at the catchment level, while increases in watershed human density from 1 to 10 households per ha were associated with an approximately 50% reduction of species richness in local waterways. A key mechanism by which a loss of connectivity can impact local invertebrate communities in urban waterways are barriers to adult insect movement and colonisation (Fagan 2002; Parkyn et al. 2011). Road culverts, bridges, and piped sections are common in urban streams and have the potential to act as physical barriers to movement along streams (Smith et al. 2009). For example, Blakely et al. (2006) used malaise trapping to investigate adult caddisfly numbers above and below three culverts in Okeover Stream, Christchurch, finding that adult caddisfly diversity decreased upstream of each road culvert. They concluded that road culverts acted as a barrier to upstream dispersal of adult caddisflies. Blakely et al. (2005) found that the distribution of caddisfly larvae in Okeover Stream reflected the longitudinal patterns of adults. Further to this, the number of egg masses deposited in Okeover Stream increased greatly when suitable ovipositioning substrata were added (Blakely et al. 2006). These results suggested that caddisfly recruitment was reduced by a lack of oviposition sites such as emergent boulders in the stream. New Zealand's urban streams are often categorised by a low level of habitat heterogeneity (Suren 2000), and a lack of emergent boulders and other suitable ovipositioning substrata in conjunction with physical barriers to adult movement could be creating dispersal constraints for colonisation of urban streams by aquatic invertebrates.

In this chapter, my aim was to investigate the survival of invertebrates in urban streams without the potential dispersal constraints and biotic interactions which affect invertebrates on the population and community level. I used *in situ* mesocosm to test this aim. *In situ* mesocosms containing stream invertebrates can act as an intermediate step between laboratory bioassays and natural systems. Mesocosms allow for studies of survival in naturally polluted environments and over longer time periods than laboratory experiments typically allow (Clements 1991), while confounding factors such as prey-predator interactions and colonisation constraints can be widely excluded.

Larvae of the Ephemeropteran *Deleatidium* spp. and the Trichopteran *Pycnocentria* spp., and the Mollusc *Potamopyrgus antipodarum* were chosen to represent varying degrees of sensitivity to urban pollution. *Deleatidium* have been shown to be highly sensitive to be chronic metal pollution (Hickey et al. 1992; Hickey et al. 2002). *Deleatidium* were found in only three of my sites from Chapter 2, while *Pycnocentria* were found at ten and have shown moderate tolerance to Cd and Zn (Hickey 2000). *Potamopyrgus* are relatively tolerant to metals and are able maintain viable populations in metal polluted urban waterways (Dorgelo et al. 1995; Blakely et al. 2003), and were present at all but one of my 20 streams in Chapter 2.

3.2 METHODS

3.2.1 - Site selection

In situ survival experiments were carried out in six of Christchurch's urban streams, each lasting seven days between October 2015 and January 2016. Sites were selected to reflect metal contamination, using data from Chapter 2, ensuring sites had adequate width and depth to accommodate mesocosms. Sites selected were Smacks Creek (least impacted), Waimairi Stream, Wairarapa Stream at Waiwetu Street, Okeover Stream, St Albans Creek, and Addington Brook (most impacted, high metals) (Table 3.1). Three separate experiments were conducted using the three separate macroinvertebrate taxa (*Deleatidium* spp., *Pycnocentria* spp., *Potamopyrgus antipodarum)*.

3.2.2 - Mesocosm design

A randomised block design was used to assess survival of macroinvertebrates in *in situ* mesocosms. 36 enclosures were constructed from 100 mm PVC pipe, using a similar mesocosm design to Blakely (2003). The ends were covered by fine nylon mesh (300 x 600 µm), attached by zip ties and rubber bands at both ends. Six enclosures were riveted to a 600mm length of PVC, with zip ties attached on both ends and in the middle to anchor the enclosures to the stream bed with metal stakes (Figure 2). For each mesocosm, similar sized cobbles colonised with algae were taken from each riffle. Cobbles were cleaned of any invertebrates, and one was placed in each of the six enclosures to provide substrate and food for invertebrates. Invertebrates were added and the mesocosms were placed in riffles with enough depth to be fully immersed and with one end tilted upwards slightly to allow entry of light. Mesocosms were secured to the stream bed with metal stakes.

Table 3.1. Dissolved metal concentrations (Cd, Cu, Zn, As, Cr), sediment metal concentrations (Cu, Zn, As, Pb), pH, and DO over the six sites selected for my mesocosm experiment.

Site	**Dissolved Cd**	**Dissolved Cu**	**Dissolved Zn**	**Dissolved As**	**Dissolved Cr**	**Sediment Cu**	**Sediment Zn**	**Sediment As**	**Sediment Pb**	**pH**	**DO**
	µg/l	µg/l	µg/l	µg/l	µg/l	mg/kg	mg/kg	mg/kg	mg/kg		%
Smacks	0.26	0.27	51.31	1.64	2.43	6.77	47.49	3.11	10.11	6.1	64
Waimairi	0.16	0.11	15.15	0.15	0.10	7.12	54.06	2.08	33.55	6.3	98
Okeover	0.27	0.12	18.01	0.16	0.11	22.31	71.64	1.55	37.11	6.6	96
Wairarapa	0.26	0.14	13.91	0.26	0.11	11.91	83.80	1.89	23.91	6.5	106
St Albans	0.28	0.97	33.01	2.56	0.19	23.77	339.11	5.93	74.84	7.2	55
Addington	0.14	0.92	15.86	0.68	0.09	18.93	540.46	43.15	74.63	7.2	78

Deleatidium mayflies and *Pycnocentria* caddisfly larvae were collected from Cust Main Drain near Rangiora (Easting: 1569833.35, Northing: 5197953.45), and *Potamopyrgus* were collected from Gardiner's Road Stream (Easting: 1566994.61, Northing: 5186651.48). For each experiment 10 randomly selected individuals were added to each of the six mesocosms. Individuals were of varying size, however those with black wing pads (therefore close to emergence) were not used. Invertebrates were collected using a kicknet (mesh 0.5mm). Mesocosms were left in-stream for seven days, after which the number of surviving individuals were counted from each enclosure.

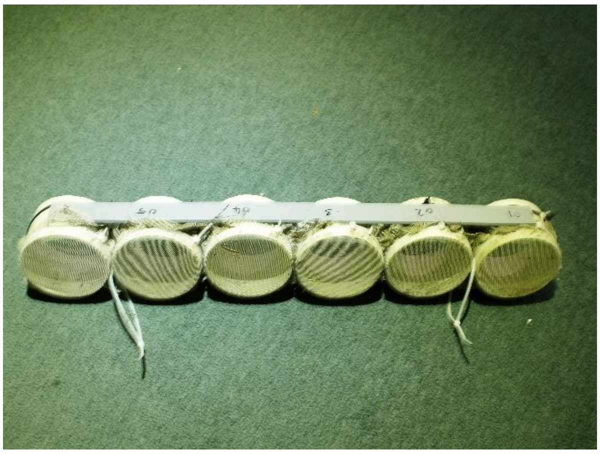


Figure 3.2. Instream mesocosm enclosures used at six urban waterways between October 2015 and January 2016.

3.2.3 - Statistical Analysis

Differences in survival at each site for each invertebrate were determined by using a two-way Analysis of Variance (ANOVA), blocking by enclosure to control for any flow differences across the six enclosures. Differences between sites were determined by post hoc Tukey tests. Statistical analysis was performed in R statistical software package (R Statistical Package 2009).

3.3 Results

3.3.1 - Survival of Deleatidium mayflies

In all streams mayfly mortality was relatively high, even in my least impacted stream approximately 50% of mayflies did not survive. However, the number of *Deleatidium* nymphs alive after seven days differed significantly between sites (ANOVA stream $F_{5,25} = 6.608$, $p<0.001$) (Figure 3.3). A post-hoc Tukey test showed that significantly fewer ($p<0.05$) nymphs survived in St Albans Creek and Addington Brook than in Smacks Creek and Waimairi Stream. No significant difference was found between individual enclosures across the stream channel (ANOVA enclosure $F_{5,25} = 0.815$, $p = 0.551$).

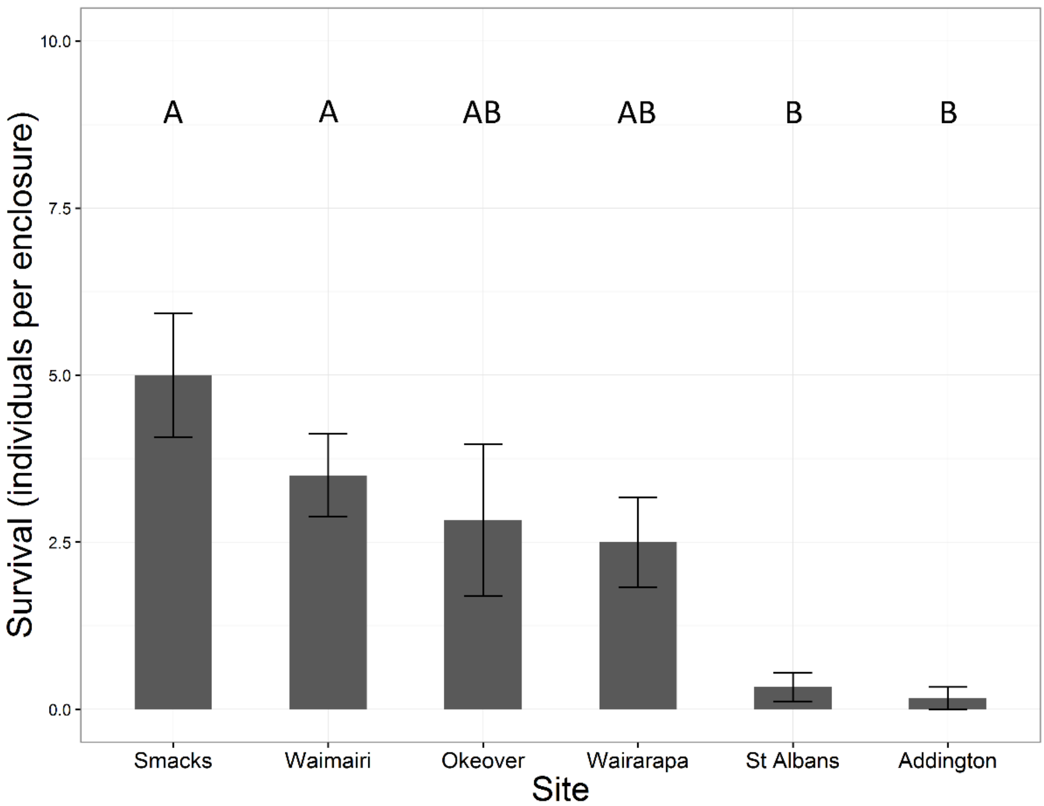


Figure 3.3. Mean (± 1SE; n=6) survival of *Deleatidium* mayfly nymphs in in situ mesocosms in six of Christchurch's urban waterways.

3.3.2 - Survival of Pycnocentria caddisflies

Survival of caddisflies across all my streams was generally high. In my least impacted stream, mortality was less than 20%, however, as with mayflies, the number of *Pycnocentria* nymphs alive after seven days also differed significantly between sites (ANOVA stream $F_{5,25} = 6.497$, $p<0.001$) (Figure 3.4). Survival patterns across streams, however, differed from *Deleatidium*. A post-hoc Tukey test showed that significantly fewer ($p<0.05$) nymphs survived in St Albans Creek, Wairarapa Stream, and Addington Brook than in Smacks Creek, with no other significant differences in survival between streams. Again, no significant difference was found between individual enclosures across the stream channel (ANOVA enclosure $F_{5,25} = 0.217$, $p = 0.952$).

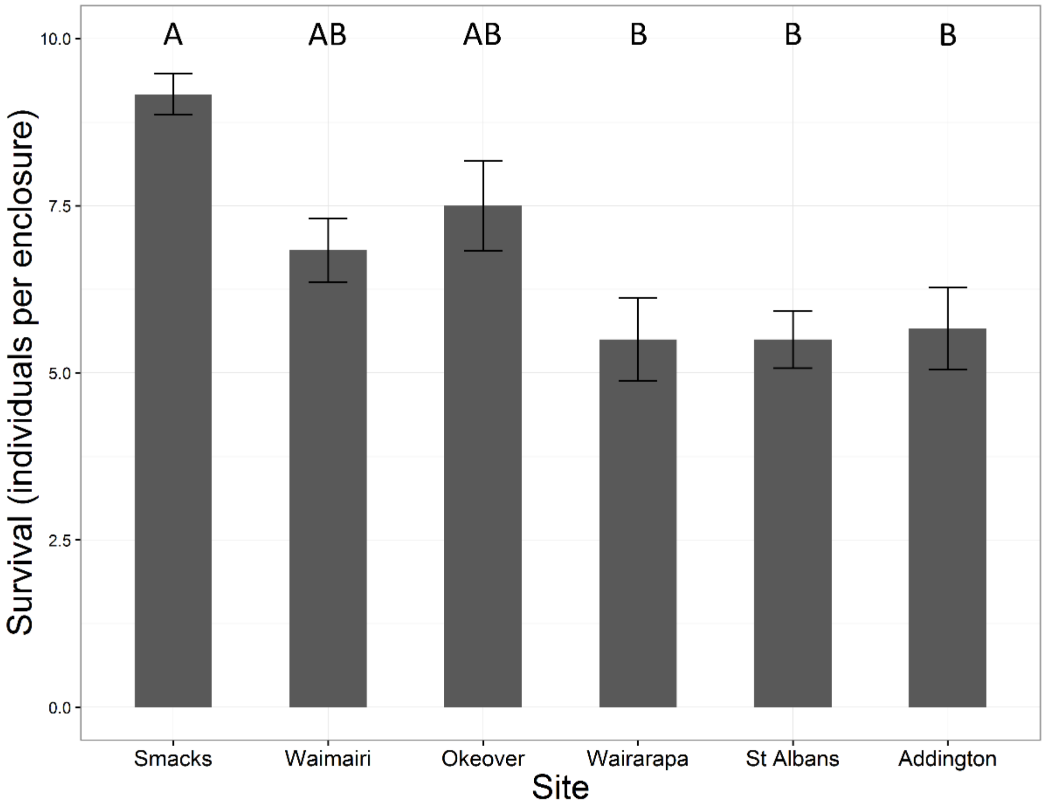


Figure 3.4. Mean (± 1SE; n=6) survival of *Pycnocentria* caddisfly larvae in in situ mesocosms in six of Christchurch's urban waterways.

3.3.3 - Survival of Potamopyrgus snails

Potamopyrgus snails showed very high survival in all streams, and was generally greater than 80%. Unlike *Deleatidium* and *Pycnocentria*, the number of snails alive after seven days did not differ significantly between sites (ANOVA stream $F_{5,25}$ = 0.683, p = 0.683) (Figure 3.5), and no significant difference was found between individual enclosures across the stream channel (ANOVA enclosure $F_{5,25}$ = 0.175, p = 0.970).

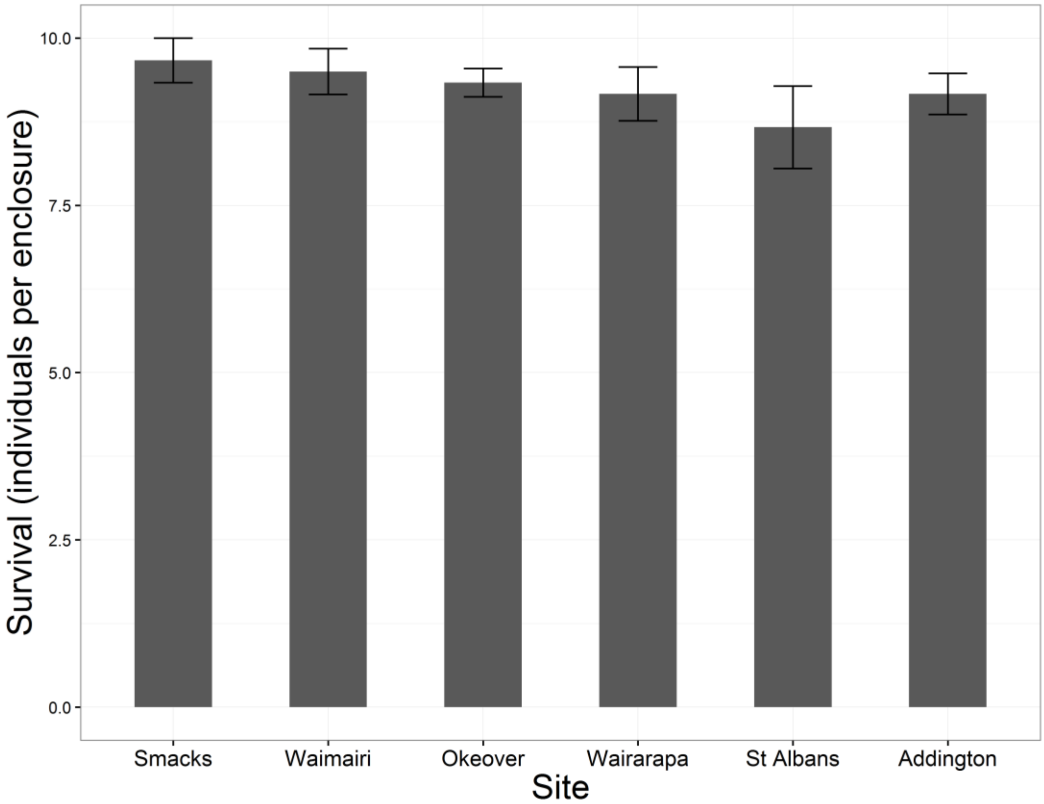


Figure 3.5. Mean (± 1SE; n=6) survival of *Potamopyrgus* snails in in situ mesocosms in six of Christchurch's urban waterways.

3.4 Discussion

The diversity of benthic invertebrates is often greatly reduced in urban waterways (Walsh et al. 2005b). This low diversity is due to species loss from a number of potential factors including physical habitat conditions, dispersal constraints, and pollution (Beasley et al. 2002; Suren et al. 2005a; Urban et al. 2006; Miserendino et al. 2008; Parkyn et al. 2011). In this study I tested the mortality of three common New Zealand stream invertebrates in six urban waterways of varying metal pollution. *In Situ* mesocosm results suggest that the water quality of my study streams significantly affected the survival of some invertebrate taxa.

Across my six streams, only Smacks Creek had naturally occurring populations of the mayfly *Deleatidium*. However, survival in *in situ* mesocosms showed that only St Albans Creek and Addington Brook showed significantly lower survival rates than Smacks Creek for *Deleatidium* nymphs. Heavy metal concentrations in the sediment appear to be an important factor influencing survival of invertebrates across the six streams in my mesocosm experiment. The two sites with the lowest mesocosm survival, St Albans Creek and Addington Brook, have the highest levels of sediment bound metals of the six streams, as well as the lowest flow velocity and highest proportion of their beds made up of silt and sand. Studies investigating the effects of metals on *Deleatidium* have shown that relative to other New Zealand freshwater invertebrates, *Deleatidium* has a high sensitivity to chronic metal contamination, especially over the longer timeframes associated with sediment contamination (Hickey et al. 1992; Hickey et al. 1998; Hickey et al. 2002). For example, Hickey et al. (1992) found that in a laboratory toxicity experiment EC_{50} values (the concentration resulting in a response in 50% of the exposed organisms, in this case immobilisation of the mayfly) decreased markedly after each 24 hour assessment. They found that 24hr EC_{50} values (Cd=>1000 µg/l; Cr=4495 µg/l; Cu=325.4 µg/l; Zn=>25000 µg/l), were substantially higher than 96hr EC_{50} values (Cd=135.2 µg/l; Cr=54.7 µg/l; Cu=38.8 µg/l; Zn=9036 µg/l). Hickey et al. (2002), using a mesocosm trial over 34 days, found that

Deleatidium abundances were significantly reduced at both low (Cu= 2.3 µg/l; Zn= 72 µg/l) and high concentrations of a combination of metals (Cu=13 µg/l; Zn=570 µg/l). They also found that *Pycnocentria* showed no significant response to the low or high metal treatment. This could explain the higher mortality rates of *Deleatidium* compared to *Pycnocentria* seen in my study.

The caddisfly *Pycnocentria* displayed a different pattern of survival than that of *Deleatidium,* where two of my streams which had natural populations of *Pycnocentria* (Okeover Stream and Waimairi Stream) did not have significantly higher mesocosm survival than those streams in which *Pycnocentria* populations are absent (Addington Brook, St Albans Creek, and Wairarapa Stream). This could be because *Pycnocentria* found in Okeover Stream and Waimairi Stream have developed a higher tolerance to heavy metals than those sourced from Cust Main Drain. Due to the often high mortality rates attributed to metal contamination of streams, strong selective pressure can be applied to populations of invertebrates in metal impacted waterways (Mulvey et al. 1991). Several studies have shown the ability of aquatic invertebrates to adapt to high concentrations of heavy metals (Klerks et al. 1989; Postma et al. 1995; Barata et al. 1998; Clements 1999; Groenendijk et al. 2002). Although no studies on New Zealand caddisfly have investigated this phenomenon, pH tolerance has been shown in New Zealand mayflies (O'Halloran et al. 2008). In that study, laboratory toxicity tests showed that mayflies sourced from naturally acidic streams (pH ~ 5.7-6.5) had a distinctly higher tolerance to acid mine drainage and low pH (3.5-4.0) than those sourced from circumneutral streams (pH ~ 7.0-7.4).

In contrast, the common New Zealand mud snail *Potamopyrgus* showed substantially higher survival rates than both *Deleatidium* and *Pycnocentria*, with no significant difference in survival between the least polluted and most polluted waterways. This is not particularly surprising, as during the experiments *Potamopyrgus* were regularly found living on the outside of mesocosms in even the most polluted streams. Several studies investigating the impact of metals on *Potamopyrgus* have found that the snail is relatively tolerant of metal pollution (Brown 1980; Dorgelo et al. 1995; Jensen et al. 2001). For example, in France, Gust et al. (2011) used *in situ* mesocosms to investigate the response of Potamopyrgus snails to a metal gradient. They found no significant difference in snail mortality between reference sites and sites with very high dissolved Zn (900-1600 µg/l) and Cd (26-30 µg/l) concentrations.

While metal concentrations across my six study streams likely contributed to the high mortality at my most impacted sites, several other factors could have influenced mesocosm mortality. Many of my mesocosms accumulated large amounts of detritus such as sticks, leaves and plastic bags during the seven day experiments. While mesocosms were cleared of detritus every two days, it is likely the accumulation between clearings would decrease flow entering the mesocosms. Mayflies and caddisflies have been shown to be relatively sensitive to low flow conditions (Suren et al. 2003), and this could have contributed to mortality rates in my mesocosms. Sand and silt accumulation also occurred inside my mesocosms, particularly at Addington Brook and St Albans Creek. Broekhuizen et al. (2001) showed that *Deleatidium* are particularly sensitive to fine sediment contamination of their periphyton food source, which could further compound potential impacts of sediments with high levels of bound metals. Further to this, during periodical sediment quality monitoring by Christchurch City Council, Gadd et al. (2014) found high concentrations of PAHs in the sediment of St Albans Creek (total PAHs: 32 mg/kg), which could contribute to high mesocosm mortality rates at this site.

The results of my mesocosm experiments suggest that heavy metal concentrations could be rendering streams such as Addington Brook and St Albans Creek uninhabitable to relatively sensitive taxa (such as *Deleatidium* and *Pycnocentria)*. However, metal toxicity at Okeover Stream, Waimairi Stream, and Wairarapa Stream does not appear to directly affect survival of *Deleatidium* nymphs significantly more than that of Smacks Creek over the seven day time frame of mesocosm experiments. This could suggest that populations of sensitive taxa in these streams are absent due to dispersal constraints, that indirect effects of pollution are rendering these environments uninhabitable, or that mortality from chronic levels of these metals occurs over a longer timeframe than my seven day experiment. These findings are likely another reflection of the multiple interacting stressors on multiple spatial scales which are common in urban waterways (Meyer et al. 2005; Urban et al. 2006; Parkyn et al. 2011). Investigating the relevant stressors on invertebrate communities at local levels is critical to understanding constraints on communities and guiding any potential rehabilitation efforts. Typically, rehabilitation efforts of urban stream focus on improving instream habitat and riparian zones (Roni et al. 2008). Sites with high concentrations of heavy metals such as St Albans Creek and Addington Brook are unlikely to be good candidates for these types of rehabilitation projects (Suren et al. 2005a), as water quality conditions in these waterways appear to be inhospitable to more sensitive taxa. However, rehabilitation of sites such as Waimairi Stream, where *Deleatidium* showed significantly greater survival, may result in an environment able to support populations of more sensitive taxa if potential instream habitat or dispersal constraints were overcome.

CHAPTER 4

CONCLUSIONS AND IMPLICATIONS FOR MANAGEMENT

4.1 INTRODUCTION

The urbanisation of a catchment results in substantial changes to waterways within the urban area (Paul et al. 2001). With an increasing proportion of the human population living in urban areas (Heilig 2012), urbanisation is an increasingly relevant pressure on waterways worldwide. Flashier hydrographs due to stormwater inflows, altered geomorphology, and increased inputs of sediment, toxicants, and nutrients are all synonymous with the "urban stream syndrome" (Walsh et al. 2005b; Wenger et al. 2009). As a result of these changes, biological communities in urban streams typically have reduced biotic richness and an increased dominance of tolerant species (Meyer et al. 2005).

Metal pollution of rivers and streams is an area of major concern for management of freshwaters, and urban runoff is recognised as an increasingly significant source of metals (Williamson 1986; Pitt et al. 1993; Brown et al. 2006). Heavy metals can to be toxic to aquatic invertebrates, and can impact community structure and abundance (Clements et al. 2000; Rainbow et al. 2011; Mebane et al. 2012). However, identification of cause-and-effect relationships between metals and aquatic invertebrate communities in urban waterways can be confounded by numerous other stressors associated with urbanisation. For example, dispersal and ovipositioning constraints (Blakely et al. 2005; Blakely et al. 2006; Urban et al. 2006; Parkyn et al. 2011), legacy effects of historic land-use (Harding et al. 1998), habitat size constraints (Brönmark et al. 1984; Dewson et al. 2007; McHugh et al. 2015), and other contaminants (Yamamoto et al. 1997; Beasley et al. 2002; Millward et al. 2004) have the potential to act in an additive, synergistic, or antagonistic nature with metal pollution to influence invertebrate communities in urban streams.

Toxicity experiments in laboratory situations can assess the toxicity of specific metals or metal mixtures to specific invertebrate species, but come with many limitations. Laboratory conditions often don't represent naturally polluted environments, and hence can neglect synergies with other stressors, indirect effects, and biotic interactions (Kimball et al. 1985; Persoone et al. 1989; Preston 2002; Mebane 2010).

An underlying challenge in determining the effects of metals on aquatic macroinvertebrates is that toxicity varies between metals and metal species, as do biological responses to these metals between taxa (Rainbow 2002). The partitioning of dissolved and particulate metals, complexation by dissolved organic matter, metal speciation associated with pH and alkalinity, and hardness can all affect the toxicity of heavy metals to macroinvertebrates (de Schamphelaere et al. 2002). Despite these challenges, several studies have found good agreement between laboratory tests and field responses (Birge et al. 1989; Eagleson et al. 1990; Dickson et al. 1992; Hickey et al. 1998).

4.2 Key Findings

My study indicated that invertebrate communities in Christchurch's urban waterways are generally depauperate, which is consistent with the findings of other studies (Robb 1992; Suren et al. 2005a; Boffa Miskell Limited 2015). Christchurch City Council surveys have found "at risk, declining" taxa such as kākahi, the freshwater mussel, and koura, the freshwater crayfish, in some urban waterways (Grainger et al. 2014; Boffa Miskell Limited 2015). However, other pollution sensitive invertebrates such as the mayfly *Deleatidium* spp. have not been found in central city waterways within Christchurch since 1989 (Robb 1992). Only waterways in peri-urban areas such as Styx River and Smacks Creek now contain these sensitive taxa.

The results of my study suggest that heavy metals could be a key variable shaping aquatic macroinvertebrate communities in Christchurch's urban waterways. CCA analysis identified that sediment bound metals, dissolved metals, and impervious surface area were the three most significant environmental factors explaining invertebrate community structure. These three variables are likely to be interrelated, as a key source of contaminants to urban waterways are deposited pollutants on impervious surfaces which are subsequently washed into waterways during rain events (Walsh 2004a).

Stepwise regression revealed that sediment bound metals were the most dominant environmental factor influencing diversity and community indices. While EPT abundance most closely reflected the results of my CCA analysis, taxonomic richness was the strongest indicator of sediment bound metal pollution in my study. MCI was also a good indicator of metal pollution, which is in contrast to other studies on metal pollution in New Zealand (Hickey et al. 1998; Hickey et al. 2002). The multivariate nature of stressors in urban streams could explain this difference. The MCI was developed to measure the effects of organic enrichment on invertebrate communities, with tolerance scores attributed to taxa accordingly (Stark 1985). Other New Zealand studies on the impacts of metal on invertebrate communities were not performed in urban streams. Hickey et al. (1998) used mesocosms in mine drainage impacted streams while Hickey et al. (2002) used experimental additions of metals in a stream draining a largely forested catchment to investigate the response of metal pollution on macroinvertebrates. The tolerance of invertebrates to the multiple stressors found in urban waterways could more closely reflect the tolerance values assigned to taxa in the MCI than the tolerance of these invertebrates to metal pollution alone. For example, Collier et al. (2009) found that in Hamilton, while MCI scores were not significantly different between urban and peri-urban streams, MCI was significantly correlated with riparian buffering and the degree of channel alteration.

The results of my short-term mesocosm experiment identified that metal pollution could be contributing factor to rendering some sites uninhabitable to pollution sensitive species. While metal concentrations in my most impacted streams breached several ANZECC guidelines for both sediment and water metal concentrations, sediment build up in the mesocosm chambers, sediment and metal contamination of periphyton, and detritus blocking flow to mesocosms likely impacted mesocosm survival in these streams. However, the survival of sensitive species at other less contaminated sites was not significantly different from sites containing natural populations of sensitive species, suggesting water quality conditions could be viable for colonisation by more sensitive species.

4.3 REHABILITATION

Rehabilitation of urban waterways has proven to be a challenging task (Bernhardt et al. 2007; Collier et al. 2008). Urban stream rehabilitation projects tend to focus on instream habitat improvements and riparian management at the reach-scale, following the "Field of Dreams" hypothesis, where improvements in instream habitat will trigger recolonization by sensitive species which were previously unable to persist in the degraded stream (Palmer et al. 1997). While instream habitat quality and riparian vegetation are important factors in stream rehabilitation, such measures do not address catchment scale impacts such as the influx of sediment and contaminants and hydrological and geomorphic changes attributed to the connection of stormwater systems to urban waterways (Hatt et al. 2004).

Restoration efforts in Christchurch have shown limited signs of success. In the 1990's Christchurch City Council selected reaches (50-100m) of five streams for rehabilitation projects. Nottingham Stream, Steamwharf Stream and Smacks Creek were selected for riparian planting, and Papanui Stream and Jacksons Creek underwent a process of channel naturalisation and riparian planting. The response of invertebrate communities to enhancements were mixed (Suren et al. 2005a) (Table 4.1). Papanui Stream exhibited beneficial signs of enhancement, with increases in EPT richness and QMCI scores and a decrease in relative abundance of pollution tolerant oligochaetes post enhancement. Smacks Creek and Jacksons Stream exhibited mixed responses, with increases in taxonomic and EPT richness, and QMCI respectively, but both showed increased relative abundance of pollution tolerant *Potamopyrgus* snails. These inconsistent responses to enhancement are typical of rehabilitation projects in urbanised areas (Bernhardt et al. 2007; Collier et al. 2008; Roni et al. 2008). It is posited that high concentrations of heavy metals could be an underlying factor to the mixed responses to rehabilitation projects of these waterways (Suren et al. 2005a). The results of my study suggest that for some waterways this could be the case.

Table 4.1. Biological responses to enhancement activities in Christchurch waterways. Enhancement effects are either positive (+) or negative (-). Responses were compared temporally (between sites before and after enhancement) and spatially (between controlled and enhanced sections). Effects were either highly significant ($P < 0.001$) both spatially and temporally (+ + +); a mixture of highly significant ($P < 0.001$) and significant ($P < 0.05$) differences spatially and temporally (++); or significant ($P < 0.05$) differences both spatially and temporally (+) (Suren et al. 2005a).

Biotic Index	Jacksons	Nottingham	Papanui	Smacks	Steamwharf
MCI					
QMCI	+ + +		+ +		
Community Metric					
Taxonomic Richness				+ +	
Species Evenness			- - -	- - -	
Species Diversity			- -	- - -	
EPT Richness			+	+ +	
%EPT					
Invertebrate Taxa					
Oligochaeta	- -		- -		
Ostrocoda		- -		-	
Oxyethira albiceps					
Physa		+ +			
Potamopyrgus antipodarum	+ + +		+ +	+ + +	

Elevated heavy metal concentrations in urban streams, along with a significant proportion of factors attributed to urban stream syndrome, are largely a result of inputs from stormwater systems, infrastructure which is essential to the functionality of modern cities (Walsh et al. 2005b; Collier et al. 2009; Davis et al. 2009). Effective impervious cover, the proportion of impervious surface area in a catchment directly connected to streams via storm water systems, effectively increases the connectivity between impervious urban surfaces and streams within the catchment. Effects of stormwater entering streams, particularly when coupled with depleted riparian vegetation often found in urban catchments, can elicit changes in stream hydrology, water chemistry, and geomorphology (Paul et al. 2001). For example, metal concentrations in stormwater runoff entering Okeover Stream at the University of Canterbury have been found to exceed ANZECC guidelines for protection of 90% of species by 18-fold for Zn, and 9- and 5-fold respectively for Cu and Pb (O'Sullivan et al. 2012). Additionally, several studies have found that as little as 10% impervious surface area in a catchment can result in adverse effects on biological communities in waterways (May et al. 1999; Paul et al. 2001; Morse et al. 2003).

While stormwater systems can be pinpointed as a key vector for a significant portion of stressors to urban lotic communities, steps to mitigate these effects by disconnecting existing stormwater systems from urban stream networks can be both difficult and costly (Davis et al. 2009). Measures for reducing stormwater pollutant loads, or best management practices, include gross pollutant traps, sedimentation ponds, infiltration, source controls, and artificial wetlands (Villarreal et al. 2004). Walsh et al. (2005a) found that employing allotment scale (i.e at the scale of a single dwelling) remediation techniques including rainwater tanks, porous pavement and rain gardens could reduce effective impervious area to 2% in catchments below 50% total impervious surface area. While these measures could have good results in rural and low impact urban developments, high costs and low availability of land in highly urbanised catchments with high pollutant loads and impervious surface area can mean high funding requirements without guaranteed results (Davis et al. 2009).

Knowledge of relevant stressors is key to the management and rehabilitation of urban streams. My results suggest that heavy metals bound to sediments are likely a key stressor on many invertebrate communities in Christchurch's urban waterways. While rehabilitation of streams in Christchurch's heavily urbanised areas can improve attractiveness and societal value, unless stormwater inputs and associated pollutants are mitigated an improvement in biological communities seems unlikely. Cost requirements of disconnecting pre-existing stormwater systems from streams are very high. Therefore, a triage like approach is perhaps the best use of funding for urban stream rehabilitation. Identification of streams with low effective impervious area in close proximity to waterways with potential colonising populations of sensitive taxa would make good candidates for rehabilitation. These streams are likely in peri-urban areas where stormwater infrastructure is less intensive. Greenfield developments in these areas should attempt to be constructed with effective stormwater management practices in mind if protection and rehabilitation of these streams is to be possible.

REFERENCES

Abrahim G, Parker R 2002. Heavy-metal contaminants in Tamaki Estuary: impact of city development and growth, Auckland, New Zealand. Environmental Geology 42: 883-890.

Aiken GR, Hsu-Kim H, Ryan JN 2011. Influence of dissolved organic matter on the environmental fate of metals, nanoparticles, and colloids. Environmental science & technology 45: 3196-3201.

Alexandre CV, Esteves KE 2010. Analysis of fish communities along a rural–urban gradient in a neotropical stream (Piracicaba River Basin, São Paulo, Brazil). Hydrobiologia 641: 97-114.

Anzecc A 2000. Australian and New Zealand guidelines for fresh and marine water quality. Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand, Canberra: 1-103.

Arnone RD, Walling JP 2007. Waterborne pathogens in urban watersheds. Journal of Water and Health 5: 149-162.

Atchison GJ, Henry MG, Sandheinrich MB 1987. Effects of metals on fish behavior: a review. Environmental Biology of fishes 18: 11-25.

Baek S, Field R, Goldstone M, Kirk P, Lester J, Perry R 1991. A review of atmospheric polycyclic aromatic hydrocarbons: sources, fate and behavior. Water, air, and soil pollution 60: 279-300.

Barata C, Baird DJ, Markich SJ 1998. Influence of genetic and environmental factors on the tolerance of Daphnia magna Straus to essential and non-essential metals. Aquatic toxicology 42: 115-137.

Barron M, Podrabsky T, Ogle S, Ricker R 1999. Are aromatic hydrocarbons the primary determinant of petroleum toxicity to aquatic organisms? Aquatic Toxicology 46: 253-268.

Batcheler L, Bolton-Ritchie L, Bond Jet al. 2006. Healthy estuary and rivers of the city: water quality and ecosystem health monitoring programme of Ihutai. Environment Canterbury, Christchurch City Council, and the Ihutai Trust, Christchurch.

Beasley G, Kneale P 2002. Reviewing the impact of metals and PAHs on macroinvertebrates in urban watercourses. Progress in Physical Geography 26: 236-270.

Beasley G, Kneale PE 2003. Investigating the influence of heavy metals on macro-invertebrate assemblages using Partial Cononical Correspondence Analysis (pCCA). Hydrology and Earth System Sciences Discussions 7: 221-233.

Beltman DJ, Clements WH, Lipton J, Cacela D 1999. Benthic invertebrate metals exposure, accumulation, and community-level effects downstream from a hard-rock mine site. Environmental Toxicology and Chemistry 18: 299-307.

Bernhardt ES, Palmer MA 2007. Restoring streams in an urbanizing world. Freshwater Biology 52: 738-751.

Bilby RE, Bisson PA 1992. Allochthonous versus autochthonous organic matter contributions to the trophic support of fish populations in clear-cut and old-growth forested streams. Canadian journal of fisheries and aquatic sciences 49: 540-551.

Birge WJ, Black JA, Short TM, Westerman AG 1989. A comparative ecological and toxicological investigation of a secondary wastewater treatment plant effluent and its receiving stream. Environmental toxicology and chemistry 8: 437-450.

Blakely T, Harding J, McIntosh A 2003. Impacts of urbanisation in Okeover stream, Christchurch. Freshwater Ecology Research Group, University of Canterbury, Christchurch.

Blakely TJ 2003. Factors influencing benthic communities and colonization in a Christchurch urban stream. B. Sc.(Hons.) dissertation, Department of Zoology, University of Canterbury, Christchurch.

Blakely TJ, Harding JS 2005. Longitudinal patterns in benthic communities in an urban stream under restoration. New Zealand journal of marine and freshwater research 39: 17-28.

Blakely TJ, Harding JS, Mcintosh AR, Winterbourn MJ 2006. Barriers to the recovery of aquatic insect communities in urban streams. Freshwater Biology 51: 1634-1645.

Blanchet FG, Legendre P, Borcard D 2008. Forward selection of explanatory variables. Ecology 89: 2623-2632.

Boffa Miskell Limited 2014. Ecological Values of the Avon River Catchment: An ecological survey of the Avon SMP catchment. Report prepared by Boffa Miskell Limited for the Christchurch City Council.

Boffa Miskell Limited 2015. Aquatic Ecology of sites within the Heathcote, Estuary & Coastal, and Avon SMP catchments: Informing the Comprehensive Discharge Consent. Report prepared by Boffa Miskell Limited for the Christchurch City Council.

Bonanno G, Giudice RL 2010. Heavy metal bioaccumulation by the organs of Phragmites australis (common reed) and their potential use as contamination indicators. Ecological Indicators 10: 639-645.

Booth DB 2005. Challenges and prospects for restoring urban streams: a perspective from the Pacific Northwest of North America. Journal of the North American Benthological Society 24: 724-737.

Booth DB, Jackson CR 1997. Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation1. Wiley Online Library.

Brockhoff C, Creed J, Martin T, Martin E, Long S 1999. EPA Method 200.8, Revision 5.5: Determination of trace metals in waters and wastes by inductively coupled plasma-mass spectrometry. EPA-821R-99-017.

Brönmark C, Herrmann J, Malmqvist B, Otto C, Sjöström P 1984. Animal community structure as a function of stream size. Hydrobiologia 112: 73-79.

Brookes A, Gregory K, Dawson F 1983. An assessment of river channelization in England and Wales. Science of the Total Environment 27: 97-111.

Brown JN, Peake BM 2006. Sources of heavy metals and polycyclic aromatic hydrocarbons in urban stormwater runoff. Science of the total environment 359: 145-155.

Brown L 1980. The use of Hydrobia jenkinsi to detect intermittent toxic discharges to a river. Water Research 14: 941-947.

Bryan G, Langston W 1992. Bioavailability, accumulation and effects of heavy metals in sediments with special reference to United Kingdom estuaries: a review. Environmental pollution 76: 89-131.

Busse LB, Simpson JC, Cooper SD 2006. Relationships among nutrients, algae, and land use in urbanized southern California streams. Canadian Journal of Fisheries and Aquatic Sciences 63: 2621-2638.

Carpenter SR, Caraco NF, Correll DL, Howarth RW, Sharpley AN, Smith VH 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological applications 8: 559-568.

Catford JA, Walsh CJ, Beardall J 2007. Catchment urbanization increases benthic microalgal biomass in streams under controlled light conditions. Aquatic Sciences 69: 511-522.

Charters F, Cochrane T, O'Sullivan A 2014. Modelling storm water management options in established urban areas. Retrieved December 16: 2015.

Christchurch City Council 2013. Christchurch Rivers Water Quality Monitoring Annual Results Summary May 2012 – April 2013. Christchurch City Council.

Clements WH 1991. Community responses of stream organisms to heavy metals: a review of observational and experimental approaches. Metal ecotoxicology: Concepts and applications: 363-339.

Clements WH 1999. Metal tolerance and predator-prey interactions in benthic macroinvertebrate stream communities. Ecological Applications 9: 1073-1084.

Clements WH, Kiffney PM 1994. Assessing contaminant effects at higher levels of biological organization. Environmental Toxicology and Chemistry 13: 357-359.

Clements WH, Carlisle DM, Lazorchak JM, Johnson PC 2000. Heavy metals structure benthic communities in Colorado mountain streams. Ecological Applications 10: 626-638.

CMAPS / Canterbury Regional Council 2015. 7492 - Catchments All Levels (GIS Data).

Collier K, Clarkson B, Aldridge B, Hicks B 2008. Can urban streams be restored? Linking vegetation restoration with urban stormwater mitigation. Proceedings of the NZWWA Stormwater Conference, Rotorua, New Zealand Water and Wastes Association. Pp. 2.

Collier KJ, Aldridge BM, Hicks BJet al. 2009. Ecological values of Hamilton urban streams (North Island, New Zealand): constraints and opportunities for restoration. New Zealand Journal of Ecology: 177-189.

Davis AP, Shokouhian M, Ni S 2001. Loading estimates of lead, copper, cadmium, and zinc in urban runoff from specific sources. Chemosphere 44: 997-1009.

Davis B, Birch G 2009. Catchment-wide assessment of the cost-effectiveness of stormwater remediation measures in urban areas. Environmental Science & Policy 12: 84-91.

Davis BS, Birch GF 2011. Spatial distribution of bulk atmospheric deposition of heavy metals in metropolitan Sydney, Australia. Water, Air, & Soil Pollution 214: 147-162.

de Schamphelaere KA, Janssen CR 2002. A biotic ligand model predicting acute copper toxicity for Daphnia magna: the effects of calcium, magnesium, sodium, potassium, and pH. Environmental science & technology 36: 48-54.

Dewson ZS, James AB, Death RG 2007. A review of the consequences of decreased flow for instream habitat and macroinvertebrates. Journal of the North American Benthological Society 26: 401-415.

Dickson KL, Waller WT, Kennedy JH, Ammann LP 1992. Assessing the relationship between ambient toxicity and instream biological response. Environmental Toxicology and Chemistry 11: 1307-1322.

Dixit S, Witcomb D 1983. Heavy metal burden in water, substrate, and macroinvertebrate body tissue of a polluted River Irwell (England). Environmental Pollution Series B, Chemical and Physical 6: 161-172.

Dorgelo J, Meester H, van Velzen C 1995. Effects of diet and heavy metals on growth rate and fertility in the deposit-feeding snail Potamopyrgus jenkinsi (Smith)(Gastropoda: Hydrobiidae). Hydrobiologia 316: 199-210.

Duzzin B, Pavoni B, Donazzolo R 1988. Macroinvertebrate communities and sediments as pollution indicators for heavy metals in the River Adige (Italy). Water Research 22: 1353-1363.

Eagleson KW, Lenat DL, Ausley LW, Winborne FB 1990. Comparison of measured instream biological responses with responses predicted using the Ceriodaphnia dubia chronic toxicity test. Environmental Toxicology and Chemistry 9: 1019-1028.

Eisler R 1987. Polycyclic aromatic hydrocarbon hazards to fish, wildlife, and invertebrates: a synoptic review. US fish and wildlife service biological report 85: 81.

Ellis J, Hvitved-Jacobsen T 1996. Urban drainage impacts on receiving waters. Journal of Hydraulic Research 34: 771-783.

Ellis JB 2006. Pharmaceutical and personal care products (PPCPs) in urban receiving waters. Environmental pollution 144: 184-189.

Engraff M, Solere C, Smith KE, Mayer P, Dahllöf I 2011. Aquatic toxicity of PAHs and PAH mixtures at saturation to benthic amphipods: Linking toxic effects to chemical activity. Aquatic toxicology 102: 142-149.

Environment Canterbury 2008. Water quality of the Avon and Heathcote rivers. Environment Canterbury. R09/82.

Environment Canterbury 2009. Haytons Stream catchment water quality investigation. Environment Canterbury,. R09/105.

Environment Canterbury 2012. Canterbury regional urban stream sediment and biofilm quality survey. Environment Canterbury. R12/5.

Fagan WF 2002. Connectivity, fragmentation, and extinction risk in dendritic metapopulations. Ecology 83: 3243-3249.

Fernández C, Carbonell G, Babín M 2013. Effects of individual and a mixture of pharmaceuticals and personal-care products on cytotoxicity, EROD activity and ROS production in a rainbow trout gonadal cell line (RTG-2). Journal of Applied Toxicology 33: 1203-1212.

Frassinetti S, Bronzetti GL, Caltavuturo L, Cini M, Della Croce C 2006. The role of zinc in life: a review. Journal of environmental pathology, toxicology and oncology 25.

Gadd J 2015. Sediment Quality Survey for Heathcote River Catchment, City Outfall Drain and Estuary Drain. NIWA Client Report: AKL-2015-021 for Christchurch City Council.

Gadd J, Sykes J 2014. Avon River Sediment Survey. NIWA Client Report: AKL2014-004 for Christchurch City Council.

Galus M, Jeyaranjaan J, Smith E, Li H, Metcalfe C, Wilson JY 2013. Chronic effects of exposure to a pharmaceutical mixture and municipal wastewater in zebrafish. Aquatic toxicology 132: 212-222.

Göbel P, Dierkes C, Coldewey W 2007. Storm water runoff concentration matrix for urban areas. Journal of contaminant hydrology 91: 26-42.

Golder Associates 2012. Canterbury regional urban stream sediment and biofilm quality survey (Report No. R12/5 prepared for Environment Canterbury by Golder Associates). Environment Canterbury.

Grainger N, Collier K, Hitchmough R, Harding J, Smith B, Sutherland D 2014. Conservation status of New Zealand freshwater invertebrates, 2013. New Zealand Threat Classification Series 8: 28.

Gray L 2004. Changes in water quality and macroinvertebrate communities resulting from urban stormflows in the Provo River, Utah, USA. Hydrobiologia 518: 33-46.

Green DW, Williams KA, Pascoe D 1986. The acute and chronic toxicity of cadmium to different life history stages of the freshwater crustaceanAsellus aquaticus (L). Archives of Environmental Contamination and Toxicology 15: 465-471.

Groenendijk D, Lücker SM, Plans M, Kraak MH, Admiraal W 2002. Dynamics of metal adaptation in riverine chironomids. Environmental Pollution 117: 101-109.

Gundacker C 2000. Comparison of heavy metal bioaccumulation in freshwater molluscs of urban river habitats in Vienna. Environmental Pollution 110: 61-71.

Gust M, Buronfosse T, Geffard Oet al. 2011. Comprehensive biological effects of a complex field poly-metallic pollution gradient on the New Zealand mudsnail Potamopyrgus antipodarum (Gray). Aquatic toxicology 101: 100-108.

Hall MJ, Closs GP, Riley RH 2001. Relationships between land use and stream invertebrate community structure in a South Island, New Zealand, coastal stream catchment. New Zealand Journal of Marine and Freshwater Research 35: 591-603.

Hammer TR 1972. Stream channel enlargement due to urbanization, Regional Science Research Institute.

Harding J, Jellyman P 2015. Earthquakes, catastrophic sediment additions and the response of urban stream communities. New Zealand Journal of Marine and Freshwater Research 49: 346-355.

Harding J, Benfield E, Bolstad P, Helfman G, Jones E 1998. Stream biodiversity: the ghost of land use past. Proceedings of the national academy of sciences 95: 14843-14847.

Hare L 1992. Aquatic insects and trace metals: bioavailability, bioaccumulation, and toxicity. Critical reviews in toxicology 22: 327-369.

Hatt BE, Fletcher TD, Walsh CJ, Taylor SL 2004. The influence of urban density and drainage infrastructure on the concentrations and loads of pollutants in small streams. Environmental management 34: 112-124.

Heilig GK 2012. World urbanization prospects: the 2011 revision. United Nations, Department of Economic and Social Affairs (DESA), Population Division, Population Estimates and Projections Section, New York.

Hession W, Pizzuto J, Johnson T, Horwitz R 2003. Influence of bank vegetation on channel morphology in rural and urban watersheds. Geology 31: 147-150.

Hession W, Charles D, Hart Det al. 2002. Riparian reforestation in an urbanizing watershed: effects of upland conditions on instream ecological benefits. NCERQA Grant Final Report.

Hickey CW 2000. Ecotoxicology: laboratory and field approaches. In: Collier, KJ, Winterbourn, MJ. eds. New Zealand stream invertebrates: ecology and implications for management. New Zealand Limnological Society, Christchurch.

Hickey CW, Vickers ML 1992. Comparison of the sensitivity to heavy metals and pentachlorophenol of the mayflies Deleatidium spp. and the cladoceran Daphnia magna. New Zealand journal of marine and freshwater research 26: 87-93.

Hickey CW, Clements WH 1998. Effects of heavy metals on benthic macroinvertebrate communities in New Zealand streams. Environmental toxicology and chemistry 17: 2338-2346.

Hickey CW, Golding LA 2002. Response of macroinvertebrates to copper and zinc in a stream mesocosm. Environmental Toxicology and Chemistry 21: 1854-1863.

Hitchmough R, Bull L, Cromarty P 2005. New Zealand threat classification system lists. Wellington, Department of Conservation. www.doc.govt.nz Accessed online on 9/05/2016.

Hoare R 1984. Nitrogen and phosphorus in Rotorua urban streams. New Zealand journal of marine and freshwater research 18: 451-454.

Horowitz AJ, Elrick KA 1987. The relation of stream sediment surface area, grain size and composition to trace element chemistry. Applied Geochemistry 2: 437-451.

Horowitz AJ, Meybeck M, Idlafkih Z, Biger E 1999. Variations in trace element geochemistry in the Seine River Basin based on floodplain deposits and bed sediments. Hydrological Processes 13: 1329-1340.

Huryn AD, Wallace JB 2000. Life history and production of stream insects. Annual review of entomology 45: 83-110.

Hwang H-M, Foster GD 2006. Characterization of polycyclic aromatic hydrocarbons in urban stormwater runoff flowing into the tidal Anacostia River, Washington, DC, USA. Environmental Pollution 140: 416-426.

Ireland DS, Burton GA, Hess GG 1996. In situ toxicity evaluations of turbidity and photoinduction of polycyclic aromatic hydrocarbons. Environmental Toxicology and Chemistry 15: 574-581.

Jacobsen D, Schultz R, Encalada A 1997. Structure and diversity of stream invertebrate assemblages: the influence of temperature with altitude and latitude. Freshwater Biology 38: 247-261.

Jensen A, Forbes VE, Parker ED 2001. Variation in cadmium uptake, feeding rate, and life-history effects in the gastropod Potamopyrgus antipodarum: Linking toxicant effects on individuals to the population level. Environmental Toxicology and Chemistry 20: 2503-2513.

Jezierska B, Ługowska K, Witeska M 2009. The effects of heavy metals on embryonic development of fish (a review). Fish Physiology and Biochemistry 35: 625-640.

Jowett IG, Richardson J 1990. Microhabitat preferences of benthic invertebrates in a New Zealand river and the development of in-stream flow-habitat models for Deleatidium spp. New Zealand journal of marine and freshwater research 24: 19-30.

Kadi MW 2009. "Soil Pollution Hazardous to Environment": A case study on the chemical composition and correlation to automobile traffic of the roadside soil of Jeddah city, Saudi Arabia. Journal of hazardous materials 168: 1280-1283.

Kayhanian M, Vichare A, Green PG, Harvey J 2009. Leachability of dissolved chromium in asphalt and concrete surfacing materials. Journal of environmental management 90: 3574-3580.

Kennedy P, Sutherland S 2008. Urban sources of copper, lead and zinc., Prepared by Organisation for Auckland Regional Council. Auckland Regional Council Technical Report 2008/023.

Kiffney PM, Clements WH 1994. Effects of heavy metals on a macroinvertebrate assemblage from a Rocky Mountain stream in experimental microcosms. Journal of the North American Benthological Society: 511-523.

Kiffney PM, Clements WH 1996. Size-dependent response of macroinvertebrates to metals in experimental streams. Environmental Toxicology and Chemistry 15: 1352-1356.

Kimball KD, Levin SA 1985. Limitations of laboratory bioassays: the need for ecosystem-level testing. BioScience 35: 165-171.

King SA, Buckney RT 2000. Urbanization and exotic plants in northern Sydney streams. Austral Ecology 25: 455-461.

Klein RD 1979. Urbanization and stream quality impairment. Wiley Online Library.

Klerks P, Levinton J 1989. Rapid evolution of metal resistance in a benthic oligochaete inhabiting a metal-polluted site. The Biological Bulletin 176: 135-141.

Klerks PL, Weis JS 1987. Genetic adaptation to heavy metals in aquatic organisms: a review. Environmental Pollution 45: 173-205.

La Valle PD 1975. Domestic sources of stream phosphates in urban streams. Water Research 9: 913-915.

Landrum P, Giesy J, Oris J, Allred P 1987. Photoinduced toxicity of polycyclic aromatic hydrocarbons to aquatic organisms. Oil in freshwater: chemistry, biology, countermeasure technology. Elmsford, NY: Pergamon Press: 304œ318.

Leathwick J, West D, Gerbeaux Pet al. 2010. Freshwater ecosystems of New Zealand (FENZ) geodatabase. Department of Conservation, Hamilton, New Zealand.

Lenat DR, Barbour MT 1994. Using benthic macroinvertebrate community structure for rapid, cost-effective, water quality monitoring: rapid bioassessment. Biological monitoring of aquatic systems. Lewis Publishers, Boca Raton, Florida: 187-215.

Leopold LB 1968. Hydrology for urban land planning: A guidebook on the hydrologic effects of urban land use, US Government Printing Office.

Leopold LB, Dunne T 1978. Water in environmental planning. New York, 818p.

Lull H, Sopper W 1966. Hydrologic effects from urbanization of forested watersheds in the northeast: Upper Darby, Pa., Northeastern Forest Expt. Sta 24.

Luoma SN, Bryan G 1982. A statistical study of environmental factors controlling concentrations of heavy metals in the burrowing bivalve Scrobicularia plana and the polychaete Nereis diversicolor. Estuarine, Coastal and Shelf Science 15: 95-108.

Mallin MA, Johnson VL, Ensign SH 2009. Comparative impacts of stormwater runoff on water quality of an urban, a suburban, and a rural stream. Environmental Monitoring and Assessment 159: 475-491.

Margalef R 1958. Temporal succession and spatial heterogeneity in phytoplankton, University of California press.

Margetts B 2014. Surface Water Quality Monitoring Report for Christchurch City Waterways: January – December 2014 Christchurch City Council.

Marsalek J, Cisneros BJ, Karamouz M, Malmquist P-A, Goldenfum JA, Chocat B 2008. Urban Water Cycle Processes and Interactions: Urban Water Series-UNESCO-IHP, CRC Press.

Matthaei C, Uehlinger U, Frutiger A 1997. Response of benthic invertebrates to natural versus experimental disturbance in a Swiss prealpine river. Freshwater Biology 37: 61-77.

May CW, Horner RR, Karr JR, Mar BW, Welch EB 1999. Effects of urbanization on small streams in the Puget Sound ecoregion. Watershed Protection Techniques 2: 79.

McEwan AJ, Joy MK 2009. Differences in the distributions of freshwater fishes and decapod crustaceans in urban and forested streams in Auckland, New Zealand.

McHugh PA, Thompson RM, Greig HS, Warburton HJ, McIntosh AR 2015. Habitat size influences food web structure in drying streams. Ecography 38: 700-712.

McIntosh MD, Benbow ME, Burky AJ 2002. Effects of stream diversion on riffle macroinvertebrate communities in a Maui, Hawaii, stream. River Research and Applications 18: 569-581.

McMurtrie S, Taylor M 2003. Ecological Assessment of the Avon River Mainstem, from Fendalton Road to Fitzgerald Avenue. EOS Ecology report for the Christchurch City Council, Christchurch.

Mebane CA 2010. Relevance of risk predictions derived from a chronic species sensitivity distribution with cadmium to aquatic populations and ecosystems. Risk analysis 30: 203-223.

Mebane CA, Dillon FS, Hennessy DP 2012. Acute toxicity of cadmium, lead, zinc, and their mixtures to stream-resident fish and invertebrates. Environmental Toxicology and Chemistry 31: 1334-1348.

Medeiros C, Leblanc R, Coler RA 1983. An in situ assessment of the acute toxicity of urban runoff to benthic macroinvertebrates. Environmental Toxicology and Chemistry 2: 119-126.

Medley CN, Clements WH 1998. Responses of diatom communities to heavy metals in streams: the influence of longitudinal variation. Ecological Applications 8: 631-644.

Meyer JL, Paul MJ, Taulbee WK 2005. Stream ecosystem function in urbanizing landscapes. Journal of the North American Benthological Society 24: 602-612.

Millward RN, Carman KR, Fleeger JW, Gambrell RP, Portier R 2004. Mixtures of metals and hydrocarbons elicit complex responses by a benthic invertebrate community. Journal of Experimental Marine Biology and Ecology 310: 115-130.

Minchin PR 1987. An evaluation of the relative robustness of techniques for ecological ordination. Theory and models in vegetation science, Springer. Pp. 89-107.

Miserendino M, Brand C, Di Prinzio C 2008. Assessing urban impacts on water quality, benthic communities and fish in streams of the Andes Mountains, Patagonia (Argentina). Water, air, and soil pollution 194: 91-110.

Moldovan M, Rauch S, Gómez M, Palacios MA, Morrison GM 2001. Bioaccumulation of palladium, platinum and rhodium from urban particulates and sediments by the freshwater isopod Asellus aquaticus. Water Research 35: 4175-4183.

Moore P, Rainbow P 1987. Copper and zinc in an ecological series of talitroidean Amphipoda (Crustacea). Oecologia 73: 120-126.

Morin S, Duong T, Dabrin Aet al. 2008. Long-term survey of heavy-metal pollution, biofilm contamination and diatom community structure in the Riou Mort watershed, South-West France. Environmental Pollution 151: 532-542.

Morse CC, Huryn AD, Cronan C 2003. Impervious surface area as a predictor of the effects of urbanization on stream insect communities in Maine, USA. Environmental Monitoring and Assessment 89: 95-127.

Mulvey M, Diamond SA 1991. Genetic factors and tolerance acquisition in populations exposed to metals and metalloids. Ecotoxicology of metals: concepts and applications (MC Newman and AW McIntosh, Editors). Lewis Publishers, Inc., Chelsea, Michigan, USA: 301-321.

Neal C, Robson AJ, Jeffery HAet al. 1997. Trace element inter-relationships for the Humber rivers: inferences for hydrological and chemical controls. Science of the Total Environment 194: 321-343.

Newall P, Walsh CJ 2005. Response of epilithic diatom assemblages to urbanization influences. Hydrobiologia 532: 53-67.

O'connor N 1991. The effects of habitat complexity on the macroinvertebrates colonising wood substrates in a lowland stream. Oecologia 85: 504-512.

O'Halloran K, Cavanagh JA, Harding JS 2008. Response of a New Zealand mayfly (Deleatidium spp.) to acid mine drainage: implications for mine remediation. Environmental Toxicology and Chemistry 27: 1135-1140.

O'Brien PJ, Wehr JD 2010. Periphyton biomass and ecological stoichiometry in streams within an urban to rural land-use gradient. Hydrobiologia 657: 89-105.

O'Sullivan A, Wicke D, Cochrane T 2012. Heavy metal contamination in an urban stream fed by contaminated air-conditioning and stormwater discharges. Environmental Science and Pollution Research 19: 903-911.

Oksanen J, Kindt R, Legendre Pet al. 2007. The vegan package. Community ecology package 10.

Onorato D, Angus RA, Marion KR 2000. Historical changes in the ichthyofaunal assemblages of the upper Cahaba River in Alabama associated with extensive urban development in the watershed. Journal of Freshwater Ecology 15: 47-63.

Palmer MA, Ambrose RF, Poff NL 1997. Ecological theory and community restoration ecology. Restoration ecology 5: 291-300.

Palmer MA, Menninger HL, Bernhardt E 2010. River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? Freshwater biology 55: 205-222.

Parkyn SM, Smith BJ 2011. Dispersal constraints for stream invertebrates: setting realistic timescales for biodiversity restoration. Environmental management 48: 602-614.

Parkyn SM, Davies-Colley RJ, Halliday NJ, Costley KJ, Croker GF 2003. Planted riparian buffer zones in New Zealand: do they live up to expectations? Restoration ecology 11: 436-447.

Paul MJ, Meyer JL 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics: 333-365.

Peijnenburg W, Jager T 2003. Monitoring approaches to assess bioaccessibility and bioavailability of metals: matrix issues. Ecotoxicology and environmental safety 56: 63-77.

Peng X, Yu Y, Tang C, Tan J, Huang Q, Wang Z 2008. Occurrence of steroid estrogens, endocrine-disrupting phenols, and acid pharmaceutical residues in urban riverine water of the Pearl River Delta, South China. Science of the total environment 397: 158-166.

Pennington SL, Webster-Brown JG 2008. Stormwater runoff quality from copper roofing, Auckland, New Zealand. New Zealand journal of marine and freshwater research 42: 99-108.

Persoone G, Van de Vel A, Van Steertegem M, De Nayer B 1989. Predictive value of laboratory tests with aquatic invertebrates: influence of experimental conditions. Aquatic Toxicology 14: 149-167.

Pfankuch DJ 1975. Stream reach inventory and channel stability evaluation. US Department of Agriculture Forest Service, Region 1.

Pickett ST, Cadenasso ML, Grove JMet al. 2008. Urban ecological systems: linking terrestrial ecological, physical, and socioeconomic components of metropolitan areas. Urban Ecology, Springer. Pp. 99-122.

Pitt R, Lalor M, Field R, Brown M 1993. The investigation of source area controls for the treatment of urban stormwater toxicants. Water Science and Technology 28: 271-282.

Pluhowski EJ 1970. Urbanization and its effect on the temperature of the streams on Long Island, New York. 2330-7102.

Poole GC, Berman CH 2001. An ecological perspective on in-stream temperature: natural heat dynamics and mechanisms of human-causedthermal degradation. Environmental management 27: 787-802.

Postma JF, Davids C 1995. Tolerance induction and life cycle changes in cadmium-exposed Chironomus riparius (Diptera) during consecutive generations. Ecotoxicology and Environmental Safety 30: 195-202.

Potapova M, Coles JF, Giddings EM, Zappia H 2005. A comparison of the influences of urbanization in contrasting environmental settings on stream benthic algal assemblages. American Fisheries Society Symposium. Pp. 333-359.

Preston BL 2002. Indirect effects in aquatic ecotoxicology: implications for ecological risk assessment. Environmental Management 29: 311-323.

Quinn GP, Keough MJ 2002. Experimental design and data analysis for biologists, Cambridge University Press.

R Statistical Package 2009. R: A language and environment for statistical computing. Vienna: R Foundation for Statistical Computing.

Rainbow P, Luoma S 2011. Metal toxicity, uptake and bioaccumulation in aquatic invertebrates—Modelling zinc in crustaceans. Aquatic Toxicology 105: 455-465.

Rainbow PS 1995. Biomonitoring of heavy metal availability in the marine environment. Marine Pollution Bulletin 31: 183-192.

Rainbow PS 2002. Trace metal concentrations in aquatic invertebrates: why and so what? Environmental Pollution 120: 497-507.

Reid H, Brierley G, Boothroyd I 2010. Influence of bed heterogeneity and habitat type on macroinvertebrate uptake in peri-urban streams. International Journal of Sediment Research 25: 203-220.

Rhoads BL, Cahill RA 1999. Geomorphological assessment of sediment contamination in an urban stream system. Applied Geochemistry 14: 459-483.

Robb J 1992. A biological re-evaluation of the Avon River catchment 1989-90. Christchurch Drainage Board, Christchurch.

Roni P, Hanson K, Beechie T 2008. Global review of the physical and biological effectiveness of stream habitat rehabilitation techniques. North American Journal of Fisheries Management 28: 856-890.

Ruse L, Herrmann S 2000. Plecoptera and Trichoptera species distribution related to environmental characteristics of the metal-polluted Arkansas River, Colorado. Western North American Naturalist: 57-65.

Rutherfurd ID, Marsh N, Jerie K 2000. A rehabilitation manual for Australian streams, Land and Water Resources Research and Development Corporation and Cooperative Research Centre for Catchment Hydrology Canberra, Australia.

Schlesinger WH, Melack JM 1981. Transport of organic carbon in the world's rivers. Tellus 33: 172-187.

Smith RF, Alexander LC, Lamp WO 2009. Dispersal by terrestrial stages of stream insects in urban watersheds: a synthesis of current knowledge. Journal of the North American Benthological Society 28: 1022-1037.

Sonneman JA, Walsh CJ, Breen PF, Sharpe AK 2001. Effects of urbanization on streams of the Melbourne region, Victoria, Australia. II. Benthic diatom communities. Freshwater Biology 46: 553-565.

Stark J, Maxted JR 2007. A user guide for the Macroinvertebrate Community Index. Cawthron Report 1166: 58.

Stark JD 1985. Macroinvertebrate community index of water quality for stony streams. National Water and Soil Conservation Authority, Wellington, New Zealand.

Stark JD 1993. Performance of the Macroinvertebrate Community Index: effects of sampling method, sample replication, water depth, current velocity, and substratum on index values. New Zealand journal of marine and freshwater research 27: 463-478.

Statistics New Zealand 2009. New Zealand: an urban/rural profile update. Wellington.

Surbeck CQ, Jiang SC, Grant SB 2009. Ecological control of fecal indicator bacteria in an urban stream. Environmental science & technology 44: 631-637.

Suren A 2000. Effects of Urbanisation. In: New Zealand Stream Invertebrates: Ecology and Implications for Management (Eds Collier K. J. & Winterbourn M. J.), pp. 260–288. New Zealand Limnological Society, Christchurch, New Zealand.: 260-88.

Suren AM, McMurtrie S 2005a. Assessing the effectiveness of enhancement activities in urban streams: II. Responses of invertebrate communities. River Research and Applications 21: 439-453.

Suren AM, Biggs BJ, Duncan MJ, Bergey L, Lambert P 2003. Benthic community dynamics during summer low-flows in two rivers of contrasting enrichment 2. Invertebrates. New Zealand journal of marine and freshwater research 37: 71-83.

Suren AM, Riis T, Biggs BJ, McMurtrie S, Barker R 2005b. Assessing the effectiveness of enhancement activities in urban streams: I. Habitat responses. River Research and Applications 21: 381-401.

Taylor SL, Roberts SC, Walsh CJ, Hatt BE 2004. Catchment urbanisation and increased benthic algal biomass in streams: linking mechanisms to management. Freshwater Biology 49: 835-851.

Ter Braak CJ 1987. The analysis of vegetation-environment relationships by canonical correspondence analysis. Theory and models in vegetation science, Springer. Pp. 69-77.

Tessier A, Campbell P 1987. Partitioning of trace metals in sediments: relationships with bioavailability. Hydrobiologia 149: 43-52.

Urban MC, Skelly DK, Burchsted D, Price W, Lowry S 2006. Stream communities across a rural–urban landscape gradient. Diversity and distributions 12: 337-350.

Van Hassel JH, Ney JJ, Garling Jr DL 1980. Heavy metals in a stream ecosystem at sites near highways. Transactions of the American Fisheries Society 109: 636-643.

Van Metre PC, Mahler BJ 2003. The contribution of particles washed from rooftops to contaminant loading to urban streams. Chemosphere 52: 1727-1741.

Vermonden K, Leuven RS, van der Velde Get al. 2010. Species pool versus site limitations of macrophytes in urban waters. Aquatic Sciences 72: 379-389.

Villarreal EL, Semadeni-Davies A, Bengtsson L 2004. Inner city stormwater control using a combination of best management practices. Ecological Engineering 22: 279-298.

Violin CR, Cada P, Sudduth EB, Hassett BA, Penrose DL, Bernhardt ES 2011. Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. Ecological Applications 21: 1932-1949.

Walsh CJ 2004a. Urban stormwater and the ecology of streams, Cooperative Research Centre for Freshwater Ecology and Cooperative Research Centre for Catchment Hydrology.

Walsh CJ 2004b. Protection of in-stream biota from urban impacts: minimise catchment imperviousness or improve drainage design? Marine and Freshwater Research 55: 317-326.

Walsh CJ, Fletcher TD, Ladson AR 2005a. Stream restoration in urban catchments through redesigning stormwater systems: looking to the catchment to save the stream. Journal of the North American Benthological Society 24: 690-705.

Walsh CJ, Roy AH, Feminella JW, Cottingham PD, Groffman PM, Morgan RP 2005b. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society 24: 706-723.

Watanabe H, Tamura I, Abe Ret al. 2016. Chronic toxicity of an environmentally relevant mixture of pharmaceuticals to three aquatic organisms (alga, daphnid, and fish). Environmental Toxicology and Chemistry.

Wenger SJ, Roy AH, Jackson CRet al. 2009. Twenty-six key research questions in urban stream ecology: an assessment of the state of the science. Journal of the North American Benthological Society 28: 1080-1098.

Wentworth CK 1922. A scale of grade and class terms for clastic sediments. The Journal of Geology: 377-392.

Wicke D, Cochrane TA, O'Sullivan AD, Cave S, Derksen M 2014. Effect of age and rainfall pH on contaminant yields from metal roofs. Water Science & Technology 69: 2166-2173.

Widianarko B, Van Gestel C, Verweij R, Van Straalen N 2000. Associations between trace metals in sediment, water, and guppy, Poecilia reticulata (Peters), from urban streams of Semarang, Indonesia. Ecotoxicology and Environmental Safety 46: 101-107.

Wilber WG, Hunter JV 1979. The impact of urbanization on the distribution of heavy metals in bottom sediments of the Saddle River. JAWRA Journal of the American Water Resources Association 15: 790-800.

Williamson R 1986. Urban stormwater quality II. Comparision of three New Zealand catchments. New Zealand Journal of Marine and Freshwater Research 20: 315-328.

Wilson J 1989. Christchurch: swamp to city: a short history of the Christchurch Drainage Board, 1875-1989, Te Waihora Press for the Christchurch Drainage Board.

Winter JG, Duthie HC 2000. Export coefficient modelling to assess phosphorus loading in an urban watershed. JAWRA Journal of the American Water Resources Association 36: 1053-1061.

Winterbourn M 1973. A guide to the freshwater Mollusca of New Zealand. Tuatara 20: 141-159.

Winterbourn MJ, Gregson KL, Dolphin CH 2000. Guide to the aquatic insects of New Zealand, Entomological Society of New Zealand Auckland.

Winterbourn MJ, Harding JS, McIntosh AR 2007. Response of the benthic fauna of an urban stream during six years of restoration. New Zealand Natural Sciences 32: 1.

Yamamoto K, Fukushima M, Kakutani N, Kuroda K 1997. Volatile organic compounds in urban rivers and their estuaries in Osaka, Japan. Environmental Pollution 95: 135-143.

Young D 2012. Rivers - The impact of European settlement

Young KD, Thackston EL 1999. Housing density and bacterial loading in urban streams. Journal of Environmental Engineering 125: 1177-1180.

Zanders J 2005. Road sediment: characterization and implications for the performance of vegetated strips for treating road run-off. Science of the Total Environment 339: 41-47.

APPENDIX I

Average abundance data for invertebrates across my 20 sites. Invertebrates were identified to the taxonomic level used to calculate an MCI. This was generally genera, though Diptera were identified to family level, and Oligochaetes to class level

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 7	Site 8	Site 9	Site 11	Site 13	Site 14	Site 15	Site 17	Site 19	Site 20	Site 21	Site 22	Site 26	Site 28	Site 33
Trichoptera																				
Hudsonema	1	28	9	29	0	4	4	0	48	1	39	127	17	36	0	6	28	0	1	0
Triplectides	1	2	9	0	0	1	0	8	0	0	1	0	1	0	0	0	1	1	4	0
H. parumbripennis	0	1	0	0	0	0	7	0	7	0	7	3	0	0	0	0	0	0	0	0
Oeconesus	0	0	0	0	0	0	0	0	0	0	0	1	1	0	3	6	1	0	0	0
Olinga	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oxyethira	0	55	7	11	0	1	173	3	185	1	79	48	4	24	0	5	2	12	101	0
Polyplectropus	1	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	3	0	0	0
Psilochorema	1	7	10	1	0	0	0	0	1	0	11	13	3	0	0	10	2	0	0	0
Pycnocentria	2	4	34	1	0	0	0	0	0	0	23	12	0	0	197	0	5	0	0	0
Pycnocentrodes	0	5	4	0	0	0	0	0	0	1	45	135	0	0	190	1	27	0	0	0
Ephemeroptera																				
Deleatidium	1	14	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diptera																				
Austrosimulium	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chironomus	0	2	1	2	4	0	0	0	49	6	465	44	43	0	1	1	11	100	29	0
Culex	0	0	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Empididae	0	0	0	2	0	0	0	0	5	0	0	12	8	3	0	0	5	0	4	0
Orthoclad	2	65	11	49	38	9	48	11	132	13	319	48	104	73	9	7	87	549	158	71
Paradixa	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Tanyderidae	0	0	0	0	0	0	0	0	3	0	1	0	0	0	0	1	1	0	0	0
Tanypodinae	0	13	5	5	0	0	0	0	0	7	27	21	7	5	2	0	0	11	1	4
Tipulidae	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Muscidae	0	1	0	1	0	0	4	0	8	0	9	3	5	0	0	0	0	0	2	0
Crustacea																				
Amphipoda	7	9	57	1	0	0	0	0	31	39	95	151	57	45	445	160	11	15	5	4
Copepoda	0	0	0	2	9	0	0	1	1	0	4	0	0	0	0	0	1	0	2	5
Ostracoda	10	2	31	48	62	29	1	3	341	129	12	75	173	352	0	7	88	187	99	65
Collembola	0	29	7	0	0	7	0	0	0	0	3	3	0	0	0	0	0	0	0	0
Cladocera	0	0	0	0	0	0	0	0	17	255	4	1	4	32	0	1	1	0	0	0
Snails																				
Gyraulus	0	1	0	60	0	0	1	0	4	1	0	0	3	0	0	0	0	60	0	6
Limnea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Physa	0	19	0	15	4	67	17	3	8	0	16	88	9	19	9	10	8	43	9	46
Potamopyrgus	24	697	1241	213	0	48	576	1060	597	42	203	411	297	521	196	212	354	99	265	886
Sphaerium	17	134	4	6	47	25	8	129	47	7	1	1	9	12	1	3	1	7	25	121
Ferrissia	0	5	1	2	0	0	1	0	0	1	39	84	0	0	1	9	0	0	0	0
Oligochaeta	19	131	77	132	384	1112	264	205	347	219	187	117	448	383	58	41	130	873	261	1047
Flatworms																				
Cura sp.	1	8	11	11	7	3	9	4	23	8	11	20	3	16	32	11	5	5	7	25