Short term impacts of physical instream restoration works on the invertebrate community in Waituna Creek, measured by Surber and kick-net sampling methods.

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Abstract

Due to anthropogenic activity, many rivers have been degraded and have lost structural diversity. Rivers restorations are commonly undertaken in order to reinstate the lost habitat heterogeneity. In New Zealand, freshwater systems are mainly threatened by agricultural land use. Streams running through agricultural land have often being artificially straightened, therefore resulting in a decrease in riparian vegetation and an increase in fine sediment and nutrients. Waituna Creek, a tributary of Waituna Lagoon in Southland, has all these characteristics. Therefore, there have been efforts to reduce the effect of agriculture within the catchment, with the largest effort being a physical instream restoration.

The Waituna Creek restoration project focused on restoring the physical aspects of the stream. This work required the use of a digger to widen the stream banks and install large wooden logs. However, this kind of work can have undesired effects on the stream bed by compacting and/or re-suspending existing fine sediment, or by further increasing fine sediment cover. Stressors such as excess fine sediment often have negative effects on macroinvertebrate communities, but the short term effects of physical instream works that can further stress macroinvertebrate communities are not well studied. In the present study, the first aim was to determine the short term effects of physical instream restoration works on the macroinvertebrate community within Waituna Creek.

In New Zealand, the standard method used for the annual State of the Environment monitoring in streams is quantitative Surber sampling. This method is not often suitable for soft-bottomed, slow-flowing streams such as Waituna Creek. Therefore, the second aim of the present study was to compare the standard Surber sampling approach, with semi-quantitative kick-net sampling, in order to determine the better-suited method for Waituna Creek.

I monitored the macroinvertebrate communities of Waituna Creek using a BACI design, by collecting samples at replicated Control and Impact sites, before (360 days) and shortly after (2 days) the restoration works, using the two different sampling approaches. Surber sampling involved three 0.1-m² samples taken from riffle areas within each 40-m study reach. The kick-net procedure involved sampling 10 locations allocated to equally represent
the different microhabitats present within the 40-m reach. To assess the effect of the restoration works, a variety of stream macroinvertebrate community health indexes were used, as well as the relative abundances of invertebrate taxa commonly found.

For the community index data collected using both sampling methods, the majority of macroinvertebrate community indexes (5 of 9) showed no Year x Site interaction, indicating no effect of the restoration works. The four EPT taxon richness metrics that did show a positive Year x Site effect also had a confounding effect of a drought in the summer of 2017/2018 (the period leading up to the collection of the after samples), so it cannot be confirmed whether the positive trends were caused by the restoration works. The results of the common taxa showed that for the Surber data, five of 10 common taxa found had a Year x Site interaction, with all five implying a negative effect of the restoration works. The kick-net data showed five of 14 common taxa with Year x Site interactions, with three taxa implying a negative effect of the restoration while the other two implied a positive effect. Considering a new taxon-specific sediment tolerance metric developed in 2017, there were also implications that some of these common taxa were being affected by increased sediment cover, a possible consequence of the restoration or the drought.

Overall, there appeared to be little effect of the physical restoration works on the invertebrate community indexes. This may be because the Waituna Creek invertebrate community as a whole is already fairly resistant to the possible additional of fine sediment added by the restoration works. Most of the common invertebrate data did not show any effect of the restoration. However, there were still several taxa that showed predominantly negative effects, which may be due to increased sedimentation from the restoration works or the drought.

The two sampling methods compared well for the community level metrics but differed more often for the common taxa, likely due to the different microhabitats of the creek sampled. The Surber method focused on the riffle areas of the creek whereas the kick-net method also sampled other important microhabitats such as areas with macrophytes and woody debris. Therefore, I recommend the semi-quantitative kick-net sampling method for routine monitoring of macroinvertebrate community in lowland soft-bottomed streams.
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Chapter 1:

Trends in river restoration
1.1 Global trends in river restoration

1.1.1 The problem

We live in a world dominated by humans (Vitousek et al. 1997). With our advances in technology, we have altered our surroundings more than any other animal has in the past (Vitousek et al. 1997; Foley et al. 2005). The human need for food, fibre, water, and shelter has caused global changes to forests, farmlands, waterways, and air (Foley et al. 2005). These changes are causing both direct and indirect effects on ecosystem structure and function (Vitousek et al. 1997; Allan 2004; Foley et al. 2005). Biodiversity is declining worldwide, with some of the highest declines seen in freshwater ecosystems (Sala et al. 2000; Dudgeon et al. 2006).

1.1.2 Direct impacts

Many rivers are directly affected by human impacts and have been redirected and/or artificially straightened for various reasons, including flood control, land drainage, and erosion prevention (Brookes et al. 1983; Grimm et al. 2008). Some rivers have also been dammed to ensure a larger ready source of water and in some cases for the production of hydroelectricity (Nilsson & Berggren 2000). These human activities have caused large changes in the physical aspects of the rivers as well as their surrounding ecosystems. Heavily modified rivers that have been redesigned to ‘efficiently’ transport water through urban or farmed land have often had their natural meanders and banksides removed and their channels cleared of obstructions that would have impeded the passage of water (Walsh et al. 2005b). This can result in available habitats and flow velocity regimes becoming overly homogeneous at the segment and reach scale (Harper et al. 1997; Bunn & Arthington 2002; Pretty et al. 2003), which can reduce local biodiversity (Harper et al. 1997; Pretty et al. 2003).

Many rivers have had their riparian vegetation removed (Rutherford et al. 1999). Riparian vegetation provides a buffer from terrestrial runoff, especially excess fine sediment and
nutrients (Dosskey et al. 2010). Vegetation also provides shade from direct sunlight and heat (Rutherford et al. 1999; Sweeney et al. 2004). Moreover, riparian vegetation shapes stream morphology by altering flow patterns during high flow periods and strengthening stream banks to prevent erosion (Hickin 1984; Rutherford et al. 1999), and provides a primary food source and microhabitats for aquatic fauna (Nakano et al. 1999; Sweeney et al. 2004). The removal of riparian vegetation changes the physical and chemical aspects of the river, and often results in loss of biodiversity (Rutherford et al. 1999; Sweeney et al. 2004; Dosskey et al. 2010).

1.1.3 Non-point source pollutants

Land use can cause drastic changes within river ecosystems (Allan et al. 1997; Allan 2004; Dudgeon et al. 2006). The main way land use affects freshwater systems is through runoff during rain events, which collects fine sediment, nutrients and contaminants from the land and delivers it to the waterways (Paul & Meyer 2001; Walsh et al. 2005; Kail et al. 2015). Excess fine sediment is one of the major problems affecting waterways and is largely due to the clearing of native bush and the conversion of land for primary industries and agriculture (Quinn et al. 1997; Holmes 2007; Quinn et al. 2009a; Matthaei et al. 2010). Excess fine sediment increases water turbidity in streams, blocking light for primary production (Davies-Colley & Smith 2001). Fine sediment can also cover the stream bed, smothering aquatic flora and making it uninhabitable for some benthic organisms (Wood & Armitage 1997; Matthaei et al. 2006; Ramezani et al. 2014). Leaching from fertiliser and livestock excreta have caused increased levels of nitrogen and phosphorus in waterways, causing certain algae or macrophytes to become overly dominant and, in some cases, decreased oxygen levels (Quinn 2000; Holmes 2007). Increased nutrients can cause unnatural phase shifts within the ecosystem which can wipe out sensitive aquatic flora and fauna (Riley et al. 2003; Niyogi et al. 2007b; Robertson & Funnell 2012).

Waterways running through urban and industrial areas are also affected by the surrounding land. A consistent effect of urbanisation on streams is an increase in impervious surface cover within urban catchments (Roy et al. 2008; Booth et al. 2016; Walsh et al. 2016). This
creates stormwater runoff which is often discharged straight into streams (Roy et al. 2008), altering their hydrology and geomorphology (Paul & Meyer 2001; Booth et al. 2016; Walsh et al. 2016). Urban stormwater also carries elevated concentrations of nutrients and contaminants from municipal and industrial discharges, which poison pollutant-sensitive species (Paul & Meyer 2001; Walsh et al. 2005b; Dudgeon et al. 2006). The combined effects of stormwater runoff and historical modifications of urban streams have caused considerable degradation of their physical, chemical and biological condition, and this degradation has been described as the ‘urban stream syndrome’ (Walsh et al. 2005b; Booth et al. 2016).

Once a river ecosystem has been degraded, it also becomes more susceptible to climate-change related stressors (Palmer et al. 2008; Nelson et al. 2009; Palmer et al. 2009). For every river catchment, there is a range of natural flow regimes representative of the unaltered landscape. The local climate has a large effect on regulating this regime by altering the water flow, while extreme climate conditions can cause droughts and floods (Lake 2000; Bunn & Arthington 2002). Due to the impact humans are having on the global environment, extreme climate events will become more frequent and intense in the future (Trenberth 2011), which will in turn affect freshwater ecosystems, with more extreme low and high water flows (Boulton 2003; Lake 2003). While river biota have adapted to be resistant and resilient to occasional extreme conditions, degraded stream ecosystems are less able to cope with, and recover from, extreme climate events. In many areas, the ecological impacts from current human activities still far exceed the impacts from the ongoing but gradual global climate change (Scholze et al. 2006); however, effects of climate change are likely to increasingly interact with other human-induced stressors to worsen the effects of the latter (Nelson et al. 2009; Palmer et al. 2009). This is another reason why we need to restore our freshwater ecosystems.

1.1.4 Aims of restoring a river

In developed countries, there are increasing efforts to restore degraded rivers, with the purpose to restore or improve natural ecosystem functions that are of economic, recreational, cultural or spiritual importance (Roni et al. 2008; Kail et al. 2015). Natural streams and rivers provide many ecosystem goods and services that benefit human
populations (Postel & Carpenter 1997), known as ecosystem services (Costanza et al. 1997). There is a general consensus that the loss of biodiversity within an ecosystem weakens ecosystems function (Hooper et al. 2005; Worm et al. 2006), which in turn reduces ecosystem services (Costanza et al. 1997; Postel & Carpenter 1997). Therefore, the general aim of most river restoration projects is to restore and protect the natural biodiversity of the river.

For projects that monitor the ecological success of stream/river restoration, there are various types of biological and cultural indexes that can be used, with different indexes working better under different conditions (Stark 1993; Wallace et al. 1996; Iliopoulou-Georgudaki et al. 2003; Townsend et al. 2004). The most commonly used biological indexes focus either on species abundance or species richness, and depending on which index is used, different results and conclusions can be reached about the success of a restoration project (Greenwood et al. 2012; Li et al. 2016). River restorations often have a strong effect on abundance or biomass and a weaker effect on species richness and diversity, as in general it is easier to increase the number of individuals in restored reaches than establishing new taxa (Kail et al. 2015). The distance and dispersal barriers between source populations and the restored reach can also limit the speed and effectiveness of species with poor dispersal ability to colonise restored streams (Stoll et al. 2014; Tonkin et al. 2014; Kitto et al. 2015). Accordingly, it can take decades for a stream ecosystem to respond to a restoration project (Meals et al. 2010), especially after instream restoration, as these often cause a large disturbance (Li et al. 2016). Unfortunately, the mid- to long-term effects of restoration projects are typically not monitored, which makes it hard to determine if the initial restoration efforts are successful (Bernhardt et al. 2005; Palmer et al. 2005; Alexander & Allan 2007).

1.1.5 Approaches to restoring rivers

It is generally hypothesised that, as habitat heterogeneity increases, so does biological diversity (MacArthur 1965; Palmer et al. 1997). This general tenant has been adopted by river restoration planers in hopes that by restoring habitat heterogeneity, biodiversity will
naturally return (Harper et al. 1997; Bunn & Arthington 2002; Palmer et al. 2010). This type of restoration generally involves re-configuring the channel to a more natural state, restoring the riparian vegetation, and adding features such as boulders and woody debris, to create a variety of structural habitats and different flow velocity conditions. By adding these features, certain habitats and niches that where lost can be restored within the stream, which are then expected to allow the species that were lost to recolonise the reopened niches. This hypothesis has been popular and widely embraced in restoration projects; however, as noted earlier there is a lack of long-term monitoring which means there is surprisingly little data to support the hypothesis (Palmer et al. 1997; Lepori et al. 2005; Palmer et al. 2010). Some scientists say that restoration projects must target the structural heterogeneity that is relevant to the target organisms in order to be more successful (Lepori et al. 2005). It is also argued that if the sources of the problems are not addressed, such as non-point stressors from land use at the catchment scale, then restoration efforts targeted at structural heterogeneity will fail due to the overwhelming effects of land use still occurring at the reach scale (Lepori et al. 2005; Palmer et al. 2010).

Restoring the riparian vegetation is an effective way of reducing the amount of non-point sediment and nutrients entering the stream (Niyogi et al. 2007a; Greenwood et al. 2012; Collins et al. 2013). This approach is increasingly being used in agricultural and urban streams as it is often considered a cheaper and easier option than instream restoration or fixing the source of the problem. However, riparian restoration has had mixed success, improving the biodiversity of some streams while having no effect on others (Greenwood et al. 2012; Wright-Stow & Wilcock 2017). This is probably because restoring riparian vegetation can fail to reduce continual inputs from catchment scale land use stressors (Scarsbrook & Halliday 1999; Greenwood et al. 2012; Collins et al. 2013).

For projects that aim to restore rivers systems by fixing the source problem, they must look to change the land use practices that are causing the ongoing degradation. For streams in agricultural areas, better farming techniques such as storing effluent during wet conditions, irrigating out the effluent onto pasture during periods without rain, and diverting runoff from waterways, can be implemented to reduce the amount of excess sediment and nutrients running into streams (Wright-Stow & Wilcock 2017). Preventing livestock from entering waterways improves both stream and riparian health (Davies-Colley et al. 2004;
Bewsell et al. 2007). For example, fencing off streams and preventing livestock from river crossings by using bridges reduces sediment input and resuspension by preventing bank and riparian vegetation trampling, and also reduces direct nutrient inputs by preventing livestock defecating in the stream (Davies-Colley et al. 2004; Wright-Stow & Wilcock 2017). In areas where similar practices have been applied, there has been positive change such as reduced fine sediment, nitrogen and phosphorus levels and indicator bacteria (E. coli), with some cases also showing an increase in invertebrate community health (Wilcock et al. 2013; Wright-Stow & Wilcock 2017). However, better land management practices have only been shown to slightly reduce the negative effect primary land use has on streams, leaving the overall effect remaining negative as we are unable to prevent all impacts of land use on waterways. For example, nitrogen may remain a problem because it can leach into underground water where it can bypass riparian restoration efforts and man-made drainage (Rissmann et al. 2012).

In urban areas, stormwater is one of the major issues degrading streams (Walsh et al. 2005a; Walsh et al. 2005b), therefore most management plans aim to redirect the flow of stormwater. This involves redesigning urban infrastructure by disconnecting impervious surfaces from streams and stormwater management approaches that maximize infiltration and water harvesting within watersheds (Walsh et al. 2005a; Roy et al. 2008; Booth et al. 2016). Prevention of degradation by negative effects of stormwater has been demonstrated (Walsh et al. 2012); however, there are still no examples of urban streams that have been successfully ecologically restored (Walsh et al. 2016). Existing infrastructural, institutional or governance contexts often prevent application of the required management actions necessary to achieve effective protection or restoration (Roy et al. 2008; Walsh et al. 2016). On the other hand, some scientists suggest that we should stop trying to achieve the impossible of restoring urban streams to a pure natural state, but instead opt for ‘designed ecosystems’ that optimize services to urban human populations while still containing diverse aquatic biota (Grimm et al. 2008; Dufour & Piégay 2009).
1.1.6 General trends in river restorations

Each river targeted for restoration is unique and has been degraded in different ways, so every river restoration project will need to be slightly different to successfully restore and protect the native ecosystem (Palmer et al. 2010; Pan et al. 2016). There is no easy fix for river restoration projects, and most projects are failing to fully restore native biodiversity (Palmer et al. 2010). Yet there are some general trends and suggestions that have been observed in terms of restoration size and project monitoring. Globally, the scale of ecological restoration projects has become larger in terms of physical size, moving from small reaches up to whole rivers and even entire catchments (Alexander & Allan 2007; Pan et al. 2016). Further, the management of running waters is changing from simply meeting water quality standards to restoring the ecosystem function of the river (Smith et al. 2014). Alongside this, restoration project success measures are changing as well (Pan et al. 2016). Success determination is becoming more complex and is now involving multiple biological indicators to get more accurate measurements of the impacts river restoration projects are having on freshwater ecosystems (Kail et al. 2015; Pan et al. 2016).

1.2 River restoration in New Zealand

1.2.1 The problem of agricultural land use

The main threat to New Zealand’s river ecosystems is agricultural land use. Agriculture is the dominant land use in many catchments of New Zealand freshwater systems (Scarsbrook et al. 2016). Large areas of New Zealand’s native alluvial floodplain forests, fertile wetlands and indigenous grasslands have been converted into pastoral land in order to support the farming industry (MacLeod & Moller 2006; Townsend et al. 2008b; Scarsbrook et al. 2016). Streams running through agricultural land have often been physically modified by channelisation, have modified flow regimes and have riparian vegetation communities dominated by exotic pasture grasses (Matthaei et al. 2010; Burdon et al. 2013; Wright-Stow & Wilcock 2017). A study by Niyogi et al. (2007a), which compared New Zealand streams running through pastoral land and streams running through native bush, found that pastoral
streams had increased nutrients and fine sediment with poor invertebrate communities according to several biotic indices. Therefore, it is no surprise that agricultural land use has been implicated as the single largest cause of water pollution in New Zealand (Winterbourn et al. 1981; Monaghan et al. 2007; Matthaei et al. 2010; Scarsbrook et al. 2016). In 2002, the ‘dirty dairying’ campaign was started by Fish and Game in order to voice their growing concern about the declining ecological health of freshwater systems in New Zealand due to farming (Holland 2015). This was an example of increasing public awareness of the impact farming, and in particular dairy farming, has on streams and rivers. This public interest has grown over the years and now the conservation of freshwater and intensive farming have become major political issues for New Zealand.

1.2.2 Riparian vegetation restoration

In New Zealand there appears to be a strong focus on restoring the native riparian vegetation alongside degraded streams, in order to buffer the agricultural runoff from land use (Parkyn et al. 2003; Greenwood et al. 2012; Collins et al. 2013). It is known that physicochemical water quality and ecological stream health are better in unaltered native bush streams than in those draining pastoral lands (Niyogi et al. 2007a), so it is hoped that by replanting native riparian barriers, stream health and water quality will increase. Some studies have found that replanting riparian vegetation has increased water quality, by reducing the amount of nitrogen (Scarsbrook & Halliday 1999), phosphorus (Wilcock et al. 2009) and fine sediment (Niyogi et al. 2007a; Collins et al. 2013; Wright-Stow & Wilcock 2017) from overland flow entering streams. In terms of measuring stream health using invertebrate community indices, there have been mixed results, particularly in highly degraded streams (Greenwood et al. 2012). Some studies claiming that riparian restoration had a positive effect on invertebrate richness and biotic indices (Scarsbrook & Halliday 1999; Niyogi et al. 2007a), whereas other studies only showed equivocal results of riparian restoration for invertebrate communities (Collins et al. 2013; Wright-Stow & Wilcock 2017). Nevertheless, most researchers argue that there are benefits of restoring the riparian vegetation. They say that with increased restoration area, and allowing more time for invertebrates to colonise, the benefits of riparian restoration to invertebrate communities
will be seen (Meals et al. 2010; Collins et al. 2013; Kitto et al. 2015). However, most researchers suggest that just replanting the riparian vegetation alone will not fully solve the problems of water quality and will not fully restore the instream community (Scarsbrook & Halliday 1999; Greenwood et al. 2012; Collins et al. 2013).

1.2.3 Improving farming practices

Another focus in New Zealand river restoration is improving farming practices to reduce their impact on the environment (Cullen et al. 2006; Scarsbrook et al. 2016; Wright-Stow & Wilcock 2017). Public pressure on the New Zealand dairy industry resulted in some farmers improving their environmental practices (Quinn et al. 2009a; Holland 2015). In New Zealand this has mainly involved planting riparian vegetation, fencing off rivers and mitigation practices such as storing effluent during wet conditions and irrigating it out onto pasture during periods without rain (Bewsell et al. 2007; Wilcock et al. 2007; Wilcock et al. 2013). However, as many of these practices are voluntary there has been mixed implementation among farmers (Deans & Hackwell 2008; Holland 2015). It is expected that, given more time and with more farmers improving their environmental practices, stream health will improve to some degree, but it is accepted that more work needs to be done to protect catchments running through agricultural land (Meals et al. 2010; Wright-Stow & Wilcock 2017).

1.2.4 General trends in New Zealand stream restoration

New Zealand rivers face many similar problems when compared to rivers in other parts of the world in terms of degradation due to human land use activities. However, New Zealand is behind regarding its restoration efforts. New Zealand is still working on small streams at the sub-reach scale and there has been no history of large-scale instream restoration work. Moreover, many smaller-scale restoration projects have been conducted without any monitoring or the monitoring data have remained unpublished (R. Holmes, Cawthron Institute, personal communication).
1.3 The Waituna Lagoon Catchment

The Waituna catchment is located in the New Zealand province of Southland near the city of Invercargill. It comprises approximately 20,000 hectares and includes the Waituna Lagoon and three tributary creeks: Waituna Creek (which contributes most of the flow to the lagoon), Moffat Creek and Currans Creek. The lagoon is an intermittently closed and open lagoon which is closed off from the sea most of the time and takes up approximately 1350 hectares of the catchment (Schallenberg et al. 2010; Robertson & Funnell 2012). The Waituna wetland area is home to many endemic and threatened species and communities and has therefore been recognised by the Ramsar Convention on Wetlands to have international importance and “special value for maintaining the genetic and ecological diversity of the region and provide a habitat for plants and animals at critical stages of their biological cycles” (Thompson & Ryder 2003). The surrounding catchment area is mainly used for intensive agriculture including sheep/beef farming and predominately dairy farming (Johnson & Partridge 1998).

1.3.1 Waituna Catchment history

The Waituna Lagoon catchment historically consisted largely of peat-bog wetland, which gave the lagoon a characteristic tannin stain, low nutrient status, and low pH. The lagoon and its creeks were traditionally used as areas to collect food by Maori, and are still used today by anglers and for other recreational activities (Johnson & Partridge 1998). In the 1950s, the majority of the wetland bog was converted into pastured land to support an increase in farm development in the area. This required the draining of the wetland areas and the clearance of indigenous vegetation (Rissmann et al. 2012). These activities left the majority of the lagoon tributaries artificially straightened. To date most of the Waituna catchment is intensively farmed, with the runoff draining into the creeks of the catchment, which then accumulates into the lagoon, which now has a high nutrient status (Rissmann et al. 2012; Robertson & Funnell 2012). The local council artificially breaches the closed lagoon on average once a year, opening it to the sea to flush out the excess sediment and nutrients
to reduce the chance of the lagoon entering a phytoplankton-dominated state (Schallenberg et al. 2010; Robertson & Funnell 2012).

1.3.2 Waituna Catchment biodiversity

The wetland houses a wide variety of species within the catchment, with more than 80 bird species occurring within the wetland complex. The aquatic plant community within the lagoon is characterised by an extensive population of seagrass, *Ruppia* species, which are considered a keystone species because of their importance as a habitat or food source for herbivorous water fowl, invertebrates and fish (Atkinson 2008; Duggan & White 2010; Robertson & Funnell 2012). These macrophytes are sensitive to reduced light levels from phytoplankton blooms and increased water turbidity, therefore they are heavily reduced when the lagoon becomes eutrophic (Cosgrove 2011; Robertson & Funnell 2012).

Fish are found in high abundance and diversity within the lagoon’s tributary creeks. Sections of the tributary creeks offer plenty of cover for fish with an abundance of overhanging riparian vegetation, undercut banks and instream macrophytes (Atkinson 2008). The creeks house many native fish species with three species, Longfin eel (*Anguilla dieffenbachia*), Giant kokopu (*Galaxias argenteus*) and Inanga (*Galaxias maculatus*), that are currently declining and meet the national criteria of ‘at risk’ as specified by Townsend et al. (2008a) (Atkinson 2008; Dunn et al. 2018). The majority of the fish found are diadromous freshwater fish that migrate between freshwater and the sea at some stage in their life cycle (Atkinson 2008). There are also known species of marine fish that spawn in freshwater, and a few species of marine wanderers; however, the abundance of marine fish is heavily dependent on whether the lagoon is open to provide access (Atkinson 2008).

1.3.3 Restoration work in the Waituna Catchment

In 2014 Environment Southland, working alongside other organisations such as the Department of Conservation (DOC) and Cawthron Institute, began a program aimed at
restoring the modified areas of the Waituna catchment tributaries and reducing the
damaging effects of the current land use (Holmes et al. 2015; Holmes 2017). Individual
farming advice has been given to farmers on nutrient budgeting, and to increase the
awareness of land management issues, particularly for the critical winter grazing period. To
help reduce bank erosion in the tributaries, approximately 14 kilometres of Waituna Creek
were re-battered and fenced off to prevent livestock access to the creeks and banks, which
reduced the trampling and erosion of the banks and riparian strips (Holmes 2017). Waituna
Creek has also had its overgrown macrophytes and built-up sediment taken out by diggers,
in a process termed mechanical clearing, to improve farm drainage (Holmes et al. 2015).
Wetland areas have been constructed to act as filters to attenuate farmland runoff before
reaching the catchment’s waterways. The most recent restoration effort included as a part
of the ongoing Waituna area management is the physical instream restoration project which
occurred within Waituna Creek during March 2018. The aim of the restoration project was
to increase the physical habitat and flow variability, to create new habitat types for
freshwater species. This involved installing large wood structures and rehabilitation of the
stream banks for future riparian planting in two reaches of the creek. This instream
restoration project gave me the opportunity to measure the initial impacts the restoration
works had on the invertebrate community, as short term effects (<1 week) are often not
recorded (Palmer et al. 2005; Alexander & Allan 2007).

1.4 Thesis aims and outline

The primary aim of the present study was to determine if physical instream restoration
works have any short term (< 1 week) negative effects on macroinvertebrate communities
living within the stream being restored, and if so what effect they might have on the future
of the restoration. To this end, my study investigated how the large-scale physical instream
restoration works in Waituna Creek initially affected short term stream health determined
by collecting benthic invertebrate samples shortly after the restoration works. To my
knowledge, there is no research on short term (< 1 week) impacts of physical instream
restoration work in agricultural streams in New Zealand.
The secondary aim of the study was to compare two methods for sampling benthic stream invertebrates, semi-quantitative kick-net sampling and quantitative Surber sampling, to determine if the two methods gave the same conclusions, and which method might be better suited for future sampling in similar studies of macrophyte-rich lowland streams.

To ensure continuity and readability of my thesis, Chapter 2 contains both aims of the study, as well as the two data sets collected and analysed. Thus, this chapter investigates how the restoration works affected stream health by using the community-level invertebrate metrics including MCI (New Zealand’s Macroinvertebrate Invertebrate Community) and EPT (Ephemeroptera, Plecoptera and Trichoptera) metrics, as well as relative abundances of common invertebrate taxa. The chapter also compares two invertebrate sampling methods: standard Surber sampling, and a modified version of the standard semi-quantitative, kick-netting method, which was adapted for this study.

Chapter 3 discusses the overall health of Waituna Creek shortly after the restoration works and makes predictions for how the restoration works will affect the future health of the creek. Further, the chapter makes suggestions for further monitoring work and restoration management for Waituna Creek and similar streams.
Chapter 2:

Short term impacts of the restoration works at Waituna Creek and outcomes of two different invertebrate sampling methods
2.1 Introduction

Human activity continues to degrade streams and rivers. We have channelized and redirected river water flow to suit our needs and continue to pollute the streams with fine sediment and nutrient run-off from agricultural and urban land use (Allan 2004; Dudgeon et al. 2006; Grimm et al. 2008). This degradation has caused a loss of biodiversity in many waterways, which in turn can lead to losses of ecosystem functions (Postel & Carpenter 1997; Hooper et al. 2005). Because of this there are increasing efforts to restore degraded rivers in developed countries, with the aim to restore or improve natural ecosystems that are of economic, cultural, or spiritual importance (Roni et al. 2008; Kail et al. 2015).

Many restorations aim to reduce the amounts of pollutants such as fine sediment and nutrients entering the stream (Walsh et al. 2005b; Greenwood et al. 2012). While this can reduce the effect of urban and agricultural land use, it does not undo the damage already done to the physical instream habitat structure (Palmer et al. 2005; Roni et al. 2008). To return the stream’s range of habitats to a more natural state, physical restoration work may be required within the stream (Roni et al. 2008; Miller et al. 2010). This can involve using heavy machinery to redirect the stream flow, either by reforming the stream bed and banks or inserting large obstacles such as boulders or logs (Shields Jr et al. 2006; Kail et al. 2007; Miller et al. 2010). However, restorations could increase human-mediated stressors, at least in the short term, such as increasing fine sediment inputs (Dudgeon et al. 2006). If these stressors caused by the physical restoration works become too severe, it could threaten the stream ecosystem and may reduce abundances or even wipe out sensitive keystone species (Dudgeon et al. 2006; Wagenhoff et al. 2011; Booth et al. 2016). The loss of keystone species would cripple important stream ecosystem services, which would become lost if those species are unable to re-establish (Postel & Carpenter 1997; Hooper et al. 2005; Palmer et al. 2010). In terms of physical stream restorations, the aim to increase habitat variability is usually achieved; however, there is often a lack of biomonitoring that goes on after the restoration work is completed (Palmer et al. 2005; Alexander & Allan 2007; Roni et al. 2008). As well as this, to my knowledge, the short term effects of physical instream restoration works on the stream invertebrate community are internationally not well studied. Short term effects of restoration on biodiversity of macroinvertebrates are the
focus of the current study using the restoration of Waituna Creek in Southland, New Zealand, as a case study.

2.1.1 The Waituna Creek restoration

In New Zealand, pastoral agriculture is the dominant land use in the catchments of many freshwater systems (Scarsbrook et al. 2016) and is a major contributor to the degradation of stream health in these catchments. To date the main approach to reduce the effects of agricultural land use on streams is to restore the riparian vegetation that has been lost (Greenwood et al. 2012; Collins et al. 2013). However, this leaves the stream with poor instream habitat variability, which has often been overlooked in New Zealand stream restoration projects until the Waituna Creek restoration project. Waituna Creek is a soft-bottom stream which has been degraded due to surrounding agricultural land use (Thompson & Ryder 2003; Cosgrove 2011; Robertson & Funnell 2012). The Waituna Creek project aims to restore riparian and instream functions through combining replanting with channel reconstruction and instream habitat creation. It is among the first New Zealand streams to have a large-scale instream restoration project aimed at changing the physical habitat variability. The restoration required the use of diggers to install large logs and to widen the banks of the stream for riparian planting. While the stream restoration work will likely increase the physical habitat heterogeneity, the instream work will likely also affect the stream bed by compacting and/or re-suspending fine sediment, while the bank widening will likely cause additional fine sediment to be deposited into the stream. Fine sediment in large amounts can cover the stream bed, smothering aquatic flora and make it uninhabitable for some benthic organisms (Matthaei et al. 2006; Scarsbrook et al. 2016). Even streams as rich in deposited fine sediment as Waituna Creek, such as deer farming streams, still show a negative effect of the addition of large amounts of further fine sediment (Matthaei et al. 2006). The restoration project gave me the opportunity to investigate the short term effects of physical instream restoration works on ecological stream health determined using the benthic macro-invertebrate community.
2.1.2 Invertebrates as indicators of stream health

Benthic invertebrates are a key component of stream ecosystems and have a central role in the food web (Quigley 1977; Wallace & Webster 1996). They also have varying tolerances to many known pollutants and are easy to sample, making them a reliable and widely used way to assess the health of stream ecosystems (Wallace & Webster 1996; Everall et al. 2017). By understanding how tolerant each individual invertebrate taxon is, we can determine how degraded a stream is by taking a representative sample of the invertebrates living there (Wallace et al. 1996; Stark & Maxted 2007a). Although invertebrate metrics can be hard to create, once established they offer a relatively affordable means of environmental measurement (Wallace et al. 1996). New Zealand already has organic enrichment tolerance values derived for local invertebrate taxa, and therefore has several macroinvertebrate community indexes (Stark & Maxted 2007a; Clapcott et al. 2017). Designed by Stark (1985), the MCI (Macroinvertebrate Community Index) and QMCI (Quantitative Macroinvertebrate Community Index) are both widely used to assess the health of New Zealand’s streams (Stark 1993; Greenwood et al. 2012; Clapcott et al. 2017). There are two variants of the MCI scores; the original scores, which are based on hard-bottomed streams (the majority of streams in New Zealand), and newer scores which are to be used for soft-bottomed streams which have a surface cover of sediment (Stark & Maxted 2007a; Stark & Maxted 2007b). The taxon richness and abundance of EPT taxa (insect larvae belonging to the orders Ephemeroptera, Plecoptera and Trichoptera) were also calculated, as these taxa are generally sensitive to pollution and EPT metrics are commonly used stream health bio-indicators worldwide (Lenat & Barbour 1994; Wallace et al. 1996; Scarsbrook et al. 2000). My study also focusses on commonly found individual taxa within the sampling sites, as these can give insights into the specific interaction between the common taxa and stressors that would otherwise be overlooked.

2.1.3 Invertebrate sampling method comparison

There are many different ways to sample stream invertebrates. Different methods and equipment are used to collect invertebrates from different stream types and from different
microhabitats within a stream (Stark et al. 2001). Sampling methods can also aim for invertebrates in particular life stages, or from particular functional groups (Williams & Hynes 1976). For the current study, I focused on two commonly used benthic invertebrate sampling methods, Surber sampling and kick-net sampling. These two methods aim to get a representative sample of the whole benthic invertebrate community of the stream at a given site (Surber 1937; Stark et al. 2001). Therefore, both are commonly used in invertebrate biomonitoring in New Zealand and overseas to get an indication of stream health (Storey et al. 1991; Carter & Resh 2001; Stark et al. 2001). Both methods are suitable for studying the effects of the restoration works on Waituna Creek; however, there are differences between the two. Surber sampling is a quantitative method while the kick-net procedure is a semi-quantitative method (Surber 1937; Everall et al. 2017). The specific Surber sampling procedure I used followed the standard protocol ‘C3’ by Stark et al. (2001), which is commonly employed in New Zealand stream biomonitoring. It involved three 0.1-m² Surber samples to be taken from riffle areas within each sampled reach (see Methods section 2.2.2 for details). For the kick-net sampling, the Stark et al. (2001) protocol was modified for this study and involved sampling 10 kick sampling locations, allocated to equally represent the different microhabitats present within the 40-m reach for one minute each (see 2.2.2 for details). The intention of using kick-netting was the hope that by sampling a wider range of microhabitats than the Surber sampling, it would collect a more representative sample of the entire reach’s invertebrate community and potentially collect rare invertebrates that might have otherwise been missed. Therefore, I also investigated whether the two sampling methods gave different numbers of taxa found and whether this changed the community health index scores.

2.1.4 Aims and hypotheses

The primary aim of this study was to determine whether the physical restoration works in Waituna Creek had any short term effects on stream health determined using invertebrate community indexes as well as the abundance of individual common invertebrate taxa within the stream. Invertebrate samples were collected according to a BACI (Before - After, Control – Impact) experimental design (Smith 2002), using the two different sampling methods
(kick-net and Surber sampling). The second aim of my study was to compare these two sampling methods to determine if they gave the same overall conclusions.

Three hypotheses were tested:

1. Shortly after the physical instream restoration works within Waituna Creek, the macroinvertebrate community indexes will show a decrease in stream health in the resorted areas due to the stress caused by the restoration works, such as increased input of fine sediment and increased water turbidity.

2. Shortly after the physical restoration works, the common invertebrate taxa which are known to have a lower pollution tolerance will decrease in abundance in impacted sites due to the stress caused by the restoration works.

3. Kick-net sampling will sample a broader range of microhabitats and will therefore collect a larger number of invertebrate taxa per site and will be more sensitive to changes in invertebrate community richness than Surber sampling.

2.2 Methods

2.2.1 Study sites and experimental design

The invertebrate samples for my study were taken from two stretches (one impact stretch, one control stretch, see below) of Waituna Creek. This stream is located 40-km south-east off the city of Invercargill and has a catchment area of 108-km² with a mean flow of 1.57-m³ and a median flow of 0.829-m³ (Environment-Southland 2018). MCI and QMCI invertebrate stream health data provided by Environment Southland from Waituna Creek at Marshall Road (a site located between the impact stretch and the control stretch) show that the creek’s health has been fluctuating between ‘fair’ and ‘poor’ from 1997 to 2017 (R. Hodson, Environment Southland, personal communication 13/11/18). On its website, Environment Southland states that the creek is in “poor ecological condition, indicating poor water quality and/or poor habitat conditions” (Land-Air-Water-Aotearoa 2018).
The control stretch of Waituna Creek is located about 7-km upstream from Waituna Lagoon and runs through a small section of bush with some overhanging vegetation. The stream channel around the control stretch has not been modified for at least ten years prior to 2014 (Holmes 2017). The impact stretch, which underwent restoration, is located within a 1-km segment of creek, about 3-km upstream from the lagoon. Before the restoration, the stream in this stretch was running straight through pastoral farm land and had very little riparian vegetation other than exotic pasture grasses. Physical instream restoration began in early March 2018. This included logs vanes of macrocarpa and bog pine (~300-mm diameter, 1.5 – 2.5 m length) and wired bundles of manuka poles (~400-500 mm diameter, 1 - 1.2 m length) being pinned to the river bed with 40-mm steel rods and Waratah posts (Fig. 1 & 2). The restoration work also widened and contoured the bank leaving an area for riparian vegetation to be planted after the restoration (Fig. 3).

Figure 1: Original cross-section sketches of: A; a log vane embedded into the stream bank. B; a log vane embedded into the stream bed. C; a bundle of manuka poles wired together and pinned to the river bed. Sketches supplied by Robin Holmes.
Figure 22: A; a log vane embedded to the stream bank, with two manuka pole bundles further downstream and another log pinned to the stream bed in Site 1. B; a log vane pinned to the stream bed in Site 1. C; a submerged bundle of manuka poles wired together and pinned to the stream bed with Waratah posts in Site 3.

Figure 3: Stream banks at Site 3 after being widened and contoured during the restoration works, now ready to be planted with riparian vegetation.
The study used a BACI (Before - After, Control – Impact) design, with samples collected before and after the instream restoration in control and impact study sites (Smith 2002). Six sampling sites were split evenly across the control reach and the impact reaches. The three impact sites were located within an area where the restoration works were expected to occur and three control sites were chosen upstream to mirror it for an even study design. Each sampling site was 40-m long. These six sites were sampled with both Surber and kick-net methods.

On the day of the first sample collection, before the restoration (27th March 2017), two more impact sampling sites were added downstream of the original three sites, in case the restoration works were extended further downstream than currently planned at the time. To mirror this, two more samples sites were added up stream in the control sites, giving an overall total of 10 sampling sites, each 40-m long (Fig. 4). However, as these extra four sites were added on the day, and it was unlikely that the restoration work would extend so far in reality, so only kick-net samples were collected from them.

The regional authorities involved decided which stream sections were restored after the Before sampling had taken place. In practice the restoration ended up being split into two areas: The first area contained two of the original sampling sites (Sites 4 & 5) and the second area further downstream contained one of the extra sampling sites (Site 1) (Fig. 4). As a result of this, one of the sites with both the Surber and kick-net samples was lost. This meant from within the restored area (the impact stretch), there were two Surber sampling sites and three kick-net sampling sites. To match this I used the original three control sites (Sites 6, 7 & 8) which had both Surber and kick-net samples, to compare to the impact samples (Fig. 4). This left me comparing two impact sites and three control sites for the Surber samples, and three impact sites to three control sites for the kick-net samples.
Figure 4: The locations of the impact and control stretches within Waituna Creek, with the sampling sites chosen before restoration (Sites 1-10) and the Marshall Road sampling point (used by Environment Southland for annual State of the Environment monitoring). Due to the actual locations of the restoration works (highlighted in green), which were determined by the regional authorities involved after the Before sampling had taken place, only sites printed in bold were used in this study. Of those Sites, no. 3, 4, 6, 7 & 8 had both Surber and kick-net samples taken, whereas Site 1 only had kick-net samples taken. Map adapted from Holmes (2017).
The post impact invertebrate samples were all collected from the control and impact stretches on 13\textsuperscript{th} March 2018, only two days after the instream restoration work had been completed and while there were still earthworks being done on the banks of Site 1 (Fig. 5). The restoration work was initially planned to be finished before sampling, but the work was delayed because there were issues with obtaining resource consent for the rehabilitation works. The ongoing earthworks involved a digger widening the stream bank for future planting of riparian vegetation, this involved the digger cross the creek each day, disturbing the stream bed (Fig. 5).

![Image](image.png)

**Figure 5:** Digger ruts going through the stream and earthworks being conducted on the other side of the river (the true right) at Site 1.

2.2.2 Invertebrate sampling

Three Surber samples (area 0.1-m\textsuperscript{2}, 0.5-mm mesh size) were taken within each relevant 40-m reach. The samples were collected according to Protocol C3 in Stark *et al.* (2001). This required all the cobbles within the sample to be scrubbed cleaned and removed, with the remaining substratum disturbed down to a depth of 5 – 10 cm. The sample was then placed
in a white sorting tray and elutriated to remove unwanted inorganic matter or terrestrial invertebrates, and the final sample was preserved in the field with 70% ethanol.

Within each 40-m impact site, kick-net or Surber invertebrate samples were taken in and around areas where the digger had operated or had been driven over the streambed. Moreover, for the kick-net method, each impact or control reach was visually divided into the different types of microhabitats present. From these, 10 kick-net sampling locations were allocated to equally represent the different microhabitats present within a given reach. For example, if a reach comprised 30% run habitat with no instream macrophyte cover, 40% run habitat with instream macrophyte cover and 30% riffle habitat, then three of the kick-net sampling locations would be allocated to the bare run, four to the run with macrophyte cover and three to the riffle area. At each of the 10 kick-net locations, a streambed area of 0.5-m² was disturbed by foot-kicking for exactly one minute, while the kick net (0.5-mm mesh size) was held downstream to collect the dislodged invertebrates. If macrophytes were present, the kick-net was swept through the macrophytes as well. Organic material and invertebrates from all 10 sampling locations within one reach were pooled to give a single composite sample per 40-m reach. All invertebrate samples were placed in a white sorting tray and elutriated to remove unwanted inorganic matter or terrestrial invertebrates, then preserved with 70% ethanol onsite.

2.2.3 Physical environmental measures

A Hach turbidimeter (Cat. No 2100Q01, Hach World Company Headquarters PO Box 389 Loveland, CO 80539, USA) and a YSI Professional Plus instrument (Professional Series Instrument 6050000, YSI Incorporated, Yellow Springs, Ohio, USA) were used to collect turbidity, temperature, pH, conductivity, and dissolved oxygen (% and mg/L) data from the impact and control sites. At each sampling site, bank cover and overhanging vegetation cover, as well as mesohabitat type and sediment type, were measured according to the habitat assessment method detailed in Holmes and Hayes (2011), and Holmes et al. (2015). Undercut banks over 0.3-m and any overhanging vegetation touching the water over 0.3-m was recorded. Mesohabitat type was recoded as percentage of habitat sub-unit as described in Holmes and Hayes (2011), and sediment type was recorded as estimated percentages of the stream bed dominated by the main substratum sizes.
2.2.4 Laboratory work

Within two days of their arrival in the laboratory, all Surber and kick-net invertebrate samples were elutriated with a 500-μm sieve to remove any excess inorganic sediment left within the samples. Some samples were topped up with extra ethanol to account for the large amount of organic matter they contained.

After elutriation, each Surber sample was split evenly into quarters using an automated rotating subsampler (Waters 1969). A quarter of the sample was then placed in a sorting tray where all invertebrates visible with a dissecting microscope were counted. For invertebrates that were fragmented, only the bodies with heads were counted. As many of the Oligochaetes were fragmented, each head and tail end found were counted and the total number was halved to give the total Oligochaete count. The invertebrates counted were also measured by being placed in a transparent petri dish with 1-mm² grid paper underneath. Using a dissecting microscope, all whole invertebrates were measured down to 1-mm size brackets (e.g. 0-1 mm, 1-2 mm, etc.), with the exception of Oligochaetes due to their fragmentation. If the total number of invertebrates found in the first quarter of the sample was under 300, another quarter was fully counted, until a minimum total of 300 invertebrates were recorded, or until all four quarters (the whole sample) were counted. For samples that did not have the whole sample counted, the number of individual invertebrates counted was extrapolated so they were comparable to fully counted samples; for example, if only one quarter was sorted, the invertebrates counted where multiplied by four. If 300 invertebrates were found before the whole sample had been processed, the remaining quarters were searched for any invertebrate taxa that had not already been found in the counted quarters. These additional taxa found were included as ‘rare taxa’ in the taxon richness count for that sample.

Each kick-net sample was well shaken before opening to homogenise the sample as best as possible. A small proportion of the shaken sample was sieved and put into a sampling tray, where the first 300 invertebrates were counted. The first 300 invertebrates were also measured within 1-mm size brackets, applying the same technique as used for the Surber-sample invertebrates. The rest of the kick-net sample was sorted, searching for any
invertebrate taxa that had not been found in the first 300 individuals, and these taxa were then recorded as ‘rare taxa’ in the taxon richness count for that sample.

All invertebrates counted from both the kick-net and Surber samples were identified to the taxonomic level recommended level by Stark et al. (2001), with the exception of the taxon Chironomidae which was identified to the family level.

2.2.5 Data analysis

The invertebrate data from the kick-net and Surber samples were analysed in two ways, using several widely applied macroinvertebrate community-level indexes and looking at the densities of the most common taxa within the community.

With the data from kick-net and Surber samples, scores were calculated to indicate the stream health using different variations of the MCI and EPT indexes. As Waituna Creek is mostly a naturally occurring soft-bottomed stream, with > 50% streambed surface cover of fine sediment based on model simulations (J. Clapcott, Cawthron Institute, personal communication 13/8/18; Depree et al. 2017) only the soft-bottom MCI scores from Stark and Maxted (2007b) were used in the present study. The EPT indexes calculated were: EPT taxa richness, percentage EPT taxa richness, EPT taxa abundance and percentage EPT taxa abundance. Some New Zealand researchers prefer to exclude the caddisfly family Hydroptilidae because this family is often associated with degraded habitats (Scarsbrook et al. 2000); therefore, I calculated all four EPT indexes both with and without Hydroptilidae.

The common invertebrate taxa for kick-net and Surber samples were defined as taxa that contributed at least 1.0% to the total number of individuals counted from all sampling sites in both years. There were 14 common taxa for the kick-net samples and 10 common taxa for the Surber samples.

To analyse the community-level invertebrate data for both the kick-net and Surber samples, one-way repeated-measures ANOVAs were used. For the common taxa of the kick-net and Surber samples, a one-way repeated-measures MANOVA was used for each sampling method. The final analyses were run in SPSS (IBM SPSS Statistics, version 24). The results of
primary interest for both the ANOVAs and MANOVAs were the “Time x Site” interaction, a within-subjects result. The “Time x Site” interaction implies whether the instream restoration works had an effect on any of the invertebrate community indexes or any of the common taxa. The categorical predictor variables “Site” and “Time” are both unrelated to the restoration works and were therefore of secondary interest for this study. “Site” is a between-subject factor which shows whether there was a pre-existing difference between the two site categories. “Time” is a within-subject factor that compares all samples collected in 2017 to all samples collected in 2018, and therefore shows whether there was a difference between years.

After exploratory data analysis, the common taxa data for kick-net and Surber samples were both log₁₀-transformed to better meet the assumption of homogeneity of variances across different groups on each sampling date.

Due to the small sample size, the significance level alpha was changed from 0.05 to 0.10, to help avoid type II errors (Quinn & Keough 2002). Standardised effect sizes (partial eta squared values, range 0–1) were also taken into account when interpreting the findings. Using a conservative approach, all results with an effect size of >0.30 were interpreted as biologically relevant (Nakagawa and Cuthill (2007) defined effect sizes <0.10 as trivial or biologically irrelevant, >0.10 as small, >0.30 as moderate and >0.5 as strong).

2.3 Results

2.3.1 Kick-net macroinvertebrate community data

Of the 11 macroinvertebrate metrics examined from the kick-net sampling data, five showed a significant P-value or a biologically relevant effect size (partial eta squared > 0.3; see Methods for details) for the Year x Site interaction (Table 1). EPT richness and Percentage EPT richness were the only metrics with a significant P-value and a strong biological effect size for Year x Site. For both metrics, EPT values at the control sites decreased between years, whereas the impact sites only decreased slightly or not at all after the restoration (Fig. 6). This implies that the restoration works had a positive effect on these
two EPT indexes. Total taxon richness, EPT richness without Hydroptilidae and percentage EPT richness without Hydroptilidae all lacked a significant P-value but all had moderate effects sizes (Table 1). Total taxon richness stayed the same at control sites but increased at impact sites after the restoration, implying that the restoration works had a positive effect on taxon richness (Fig. 6). EPT richness and percentage EPT richness, both without Hydroptilidae, experienced a large decrease at control sites and a lesser decrease at impact sites after the restoration (Fig. 6), again indicating that the restoration works had a positive effect on these two indexes. All four EPT richness indexes also showed significant or biologically relevant Site and/or Year effects (Table 1). Control sites generally had a higher EPT richness/percentage richness than impact sites and EPT richness was higher in 2017 (Fig. 6). Total taxon richness had a biologically relevant effect size for Year, showing that on average more taxa were found in 2018 (Fig. 6).

The remaining six indexes from the kick-net data all showed effects of Site, Time or both (Table 1). The MCI showed a significant and strong effect size for both Site and Time. Mean MCI scores were higher at control sites, and higher in 2017 (Fig. 7). The QMCI showed a moderate effect size for Site and a significant, strong effect for Time (Table 1). Unlike the MCI, mean QMCI scores were higher at impact sites; however, the mean QMCI was still higher in 2017 (Fig. 7). All four EPT abundance indexes had moderate or high effect sizes for both Site and Year (Table 1). Unlike the EPT richness indexes, all four EPT abundance indexes were higher at impact sites (Fig. 7). Like the EPT richness indexes, 2017 still had the higher EPT abundances/percentage abundances (Fig. 7).
Table 4: P-values and partial eta squared values from one-way repeated-measures ANOVAs using invertebrate community indexes calculated from the kick-net sampling data. The community indexes are ordered so that all metrics with significant P-values (P < 0.10) and biologically relevant effect sizes (partial eta squared > 0.3; see Methods for details) for Year x Site are listed first, then indexes that showed overall effects of Site but no Year x Site interaction. ETA = effect size (partial eta squared). All P-values < 0.10 and effect sizes > 0.30 are printed in bold.

<table>
<thead>
<tr>
<th>Kick-net data</th>
<th>Year x Site</th>
<th></th>
<th>Site</th>
<th></th>
<th>Year</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Community index</td>
<td>P-value</td>
<td>ETA</td>
<td>P-value</td>
<td>ETA</td>
<td>P-value</td>
<td>ETA</td>
</tr>
<tr>
<td>EPT Richness</td>
<td>0.026</td>
<td>0.750</td>
<td>0.016</td>
<td>0.800</td>
<td>0.026</td>
<td>0.750</td>
</tr>
<tr>
<td>% EPT Richness</td>
<td>0.027</td>
<td>0.746</td>
<td>0.249</td>
<td>0.313</td>
<td>0.007</td>
<td>0.864</td>
</tr>
<tr>
<td>Total taxon richness</td>
<td>0.116</td>
<td>0.500</td>
<td>0.305</td>
<td>0.257</td>
<td>0.116</td>
<td>0.500</td>
</tr>
<tr>
<td>EPT Richness no Hydroptilidae</td>
<td>0.158</td>
<td>0.429</td>
<td>0.002</td>
<td>0.925</td>
<td>0.007</td>
<td>0.871</td>
</tr>
<tr>
<td>% EPT Richness no Hydroptilidae</td>
<td>0.233</td>
<td>0.330</td>
<td>0.019</td>
<td>0.783</td>
<td>0.002</td>
<td>0.926</td>
</tr>
<tr>
<td>MCI Soft-bottom</td>
<td>0.475</td>
<td>0.134</td>
<td>0.006</td>
<td>0.881</td>
<td>0.006</td>
<td>0.878</td>
</tr>
<tr>
<td>EPT Abundance no Hydroptilidae</td>
<td>0.796</td>
<td>0.019</td>
<td>0.113</td>
<td>0.505</td>
<td>0.254</td>
<td>0.307</td>
</tr>
<tr>
<td>% EPT Abundance no Hydroptilidae</td>
<td>0.796</td>
<td>0.019</td>
<td>0.113</td>
<td>0.505</td>
<td>0.254</td>
<td>0.307</td>
</tr>
<tr>
<td>QMCI Soft-bottom</td>
<td>0.670</td>
<td>0.050</td>
<td>0.132</td>
<td>0.472</td>
<td>0.016</td>
<td>0.803</td>
</tr>
<tr>
<td>% EPT Abundance</td>
<td>0.702</td>
<td>0.041</td>
<td>0.132</td>
<td>0.470</td>
<td>0.194</td>
<td>0.378</td>
</tr>
<tr>
<td>EPT Abundance</td>
<td>0.702</td>
<td>0.041</td>
<td>0.132</td>
<td>0.470</td>
<td>0.194</td>
<td>0.378</td>
</tr>
</tbody>
</table>
Figure 6: Macroinvertebrate community indexes from kick-net samples that showed a significant P-value (<0.1) or biologically relevant effect size (>0.3) for the Year x Site interaction. Samples in 2017 were collected before the restoration works, whereas samples in 2018 were collected shortly after the restoration works.
Figure 7: Macroinvertebrate community indexes from kick-net samples that showed a significant P-value (<0.1) or biologically relevant effect size (>0.3) for effects of Site and/or Year, but no Year x Site interaction. Samples in 2017 were collected before the restoration works, whereas samples in 2018 were collected shortly after the restoration works.
2.3.2 Surber macroinvertebrate community data

Of the Surber sampling data, all four EPT richness indexes were the only metrics to show either a significant P-value or a biologically relevant effect size for the Year x Site interaction (Table 2). Percentage EPT richness without Hydroptilidae was the only metric with significant P-value and a strong effect size for Year x Site, but EPT richness, percentage EPT richness and EPT richness without Hydroptilidae still all had a moderate or strong effect size for Year x Site. All four EPT richness metrics showed a large decrease at control sites and a lesser decrease at impact sites after the restoration, implying that the restoration works had a positive effect on all four indexes (Fig. 8). Further, all four EPT richness indexes showed a significant difference and a strong effect size for the effects of both Site and Year (Table 2). They all showed that EPT taxa richness/percentage richness was higher at control sites and higher in 2017 (Fig. 8).

EPT abundance, EPT abundance without Hydroptilidae, percentage EPT abundance without Hydroptilidae and total taxon richness all showed either a significant P-value or a biologically relevant effect size for Site (Table 2). EPT abundance and EPT abundance without Hydroptilidae both had a significant P-value and a strong significant effect size for Site, where percentage EPT abundance without Hydroptilidae and total taxon richness only had a moderate and strong effect size. EPT abundance, EPT abundance without Hydroptilidae and percentage EPT abundance without Hydroptilidae were all higher at impact sites than at control sites (Fig. 9). Moreover, these three EPT indexes all had either a significant P-value or a biologically relevant effect size for Year, with higher values in 2017 (Fig. 9). Total taxon richness was higher at control sites but similar across years (Fig. 9).

MCI, QMCI and percentage EPT abundance had neither significant Year x Site interactions nor Site effects, but they all showed a significant difference and a strong effect size between the two sampling years (Table 2). As all other community indexes, except for total taxon richness, values of these three indexes were higher in 2017 (Fig. 10).
**Table 5:** P-values and partial eta squared values from one-way repeated-measures ANOVAs using invertebrate community indexes calculated from the Surber sampling data. The community indexes are ordered so that all metric with significant P-values (P < 0.10) and biologically relevant effect sizes (partial eta squared > 0.3; see Methods for detail) for Year x Site are listed first, then indexes that showed overall effects of Site but no Year x Site interaction and finally indexes that showed only effects of Years but no Year x Site interaction or effect of Site. ETA = effect size (partial eta squared). All P-values < 0.10 and effect sizes > 0.30 are printed in bold.

<table>
<thead>
<tr>
<th>Surber data</th>
<th>Year x Site</th>
<th>Site</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>P-value</td>
<td>ETA</td>
<td>P-value</td>
</tr>
<tr>
<td>% EPT Richness no Hydroptilidae</td>
<td>0.045</td>
<td>0.786</td>
<td>0.086</td>
</tr>
<tr>
<td>EPT Richness no Hydroptilidae</td>
<td>0.170</td>
<td>0.519</td>
<td>0.077</td>
</tr>
<tr>
<td>% EPT Richness</td>
<td>0.222</td>
<td>0.441</td>
<td>0.061</td>
</tr>
<tr>
<td>EPT Richness</td>
<td>0.239</td>
<td>0.418</td>
<td>0.086</td>
</tr>
<tr>
<td>EPT Abundance</td>
<td>0.570</td>
<td>0.119</td>
<td>0.008</td>
</tr>
<tr>
<td>EPT Abundance no Hydroptilidae</td>
<td>0.507</td>
<td>0.158</td>
<td>0.024</td>
</tr>
<tr>
<td>% EPT Abundance no Hydroptilidae</td>
<td>0.941</td>
<td>0.002</td>
<td>0.157</td>
</tr>
<tr>
<td>Total taxon richness</td>
<td>0.522</td>
<td>0.149</td>
<td>0.239</td>
</tr>
<tr>
<td>MCI Soft-bottom</td>
<td>0.429</td>
<td>0.217</td>
<td>0.341</td>
</tr>
<tr>
<td>QMCI Soft-bottom</td>
<td>0.371</td>
<td>0.268</td>
<td>0.501</td>
</tr>
<tr>
<td>% EPT Abundance</td>
<td>0.487</td>
<td>0.172</td>
<td>0.680</td>
</tr>
</tbody>
</table>
Figure 8: Macrionvertebrate community indexes from Surber samples that showed a significant P-value (<0.1) or biologically relevant effect size (>0.3) for the Year x Site interaction. Samples in 2017 were collected before the restoration works, whereas samples in 2018 were collected after the restoration works.
Figure 9: Macroinvertebrate community indexes from Surber sampling that showed a significant P-value (<0.1) or biologically relevant effect size (>0.3) for the effects of Site, but no Year x Site interaction. Samples in 2017 were collected before the restoration works, whereas samples in 2018 were collected after the restoration works.
Figure 10: Macroinvertebrate community indexes from Surber samples that showed a significant P-value (<0.1) or biologically relevant effect size (>0.3) for the effect of Year, but no Year x Site interaction or effect of Site. Samples in 2017 were collected before the restoration works, whereas samples in 2018 were collected after the restoration works.
2.3.3 Kick-net common macroinvertebrate taxa

Of the 34 invertebrate taxa found in the kick-net samples, 14 taxa were counted as common (Table 3). The top five taxa, found in every kick-net sampling site, were Oligochaetes (38.6%), Chironomidae (20.1%), Hudsonema spp. (8.4%), Potamopyrgus antipodarum (7.8%) and Oxyethira spp. (4.8%). The remaining common taxa were Xanthocnemis zealandica (3.2%), Ostracoda (3.1%), Amphipoda (2.7%), Austrosimulium spp. (2.0%), Sphaeriidae (1.9%), Hydra spp. (1.6%), Physa spp. (1.5%), Nematoda (1.4%) and Oecetis spp. (1.3%). Together, these 14 taxa made up 98.3% of the total number of individuals found within the kick-net samples.

From the kick-net data, the five taxa that showed a Year x Site interaction were Xanthocnemis zealandica, Amphipoda, Ostracoda, Austrosimulium spp. and Sphaeriidae (Table 3). X. zealandica and Ostracoda both showed significant P-values and strong effect sizes for the Year x Site interaction and for Year. X. zealandica and Ostracoda both increased in relative abundances from before to shortly after the restoration works; however, on average they increased more at the control sites, implying that the restoration works had a negative effect on the two taxa (Fig. 11). Amphipoda had significant P-values and strong effect sizes for Year x Site, Site and Year (Table 3). Amphipoda abundance decreased between years at control sites but even more so at impact sites which had started much higher, implying the restoration works had a negative effect on Amphipoda (Fig. 11). Austrosimulium spp. had significant P-values and strong effect sizes for Year x Site, Site and Year (Table 3). Austrosimulium abundance in both site categories dropped between years but more so in control sites which had started higher, implying this taxon was positively affected by the restoration works (Fig. 11). Sphaeriidae had a moderate effect size for Year x Site and significant P-values and strong effect sizes for both Site and Year (Table 3). Sphaeriidae abundance increased moderately at control sites but even more at impact sites implying that the restoration works had a positive effect (Fig. 11). Abundance of this taxon was also higher at impact sites and increased from 2017 to 2018.

Potamopyrgus antipodarum, Oligochaeta, Oecetis spp., Hudsonema spp. and Hydra spp. all showed a significant P-value or relevant effect size for Site (Table 3). P. antipodarum had a significant P-value and strong effect size for both Site and Year, with higher abundance at
impact sites and in 2018 (Fig. 12). Oligochaeta had a significant P-value and strong effect size for Site, with mean abundance being higher at control sites (Fig. 12). Oecetis spp. had a significant P-value and strong effect size for Site and a moderate effect size for Year. Abundance of this taxon was higher at impact sites and in 2018 (Fig. 12). Hudsonema spp. had a significant P-value and strong effect size for both Site and Year; its abundance was higher at impact sites and lower in 2018 (Fig. 12). Hydra spp. had a moderate effect size for Site and a significant P-value and strong effect size for Year, with abundance being higher at control sites and in 2018 (Fig. 12).

Chironomidae and Physa spp. both showed a significant P-value and strong effect size for Years (Table 3). Chironomidae abundance was higher in 2017, whereas Physa abundance was higher in 2018 (Fig. 13). Finally, Oxyethira spp. and Nematoda showed no significant P-values or relevant effect sizes for any of the predictor terms (Table 3).
Table 6: P-values and partial eta squared values from a one-way repeated-measures MANOVA of the 14 common invertebrate taxa from the kick-net sampling data. The taxa are ordered so that all taxa with significant P-values (P < 0.10) and biologically relevant effect sizes (partial eta squared > 0.3; see Methods for detail) for Year x Site are listed first, then taxa that showed overall effect of Site but no Year x Site interaction. Then taxa that showed effect of Year, but no Year x Site interaction and no effect of Site, and last, taxa that showed no relevant relationships with any of the predictor terms. ETA = effect size (partial eta squared). All P-values < 0.10 and effect sizes > 0.30 are printed in bold.

<table>
<thead>
<tr>
<th>Kick-net data</th>
<th>Year x Site</th>
<th>Site</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Common Taxa</td>
<td>P-value</td>
<td>ETA</td>
<td>P-value</td>
</tr>
<tr>
<td>Xanthocnemis zealandica</td>
<td>0.001</td>
<td>0.962</td>
<td>0.717</td>
</tr>
<tr>
<td>Amphipoda</td>
<td>0.013</td>
<td>0.821</td>
<td>0.023</td>
</tr>
<tr>
<td>Ostracoda</td>
<td>0.059</td>
<td>0.632</td>
<td>0.813</td>
</tr>
<tr>
<td>Austrosimulium spp.</td>
<td>0.085</td>
<td>0.564</td>
<td>0.035</td>
</tr>
<tr>
<td>Sphaeriidae</td>
<td>0.191</td>
<td>0.382</td>
<td>0.081</td>
</tr>
<tr>
<td>Potamopyrgus antipodarum</td>
<td>0.377</td>
<td>0.198</td>
<td>0.003</td>
</tr>
<tr>
<td>Oligochaeta</td>
<td>0.936</td>
<td>0.002</td>
<td>0.032</td>
</tr>
<tr>
<td>Oecetis spp.</td>
<td>1.000</td>
<td>0.000</td>
<td>0.075</td>
</tr>
<tr>
<td>Hudsonema spp.</td>
<td>0.509</td>
<td>0.116</td>
<td>0.077</td>
</tr>
<tr>
<td>Hydra spp.</td>
<td>0.650</td>
<td>0.057</td>
<td>0.226</td>
</tr>
<tr>
<td>Chironomidae</td>
<td>0.935</td>
<td>0.002</td>
<td>0.709</td>
</tr>
<tr>
<td>Physa spp.</td>
<td>0.262</td>
<td>0.298</td>
<td>0.367</td>
</tr>
<tr>
<td>Oxyethira spp.</td>
<td>0.404</td>
<td>0.178</td>
<td>0.265</td>
</tr>
<tr>
<td>Nematoda</td>
<td>0.976</td>
<td>0.000</td>
<td>0.995</td>
</tr>
</tbody>
</table>
Figure 11: Common taxa from kick-net samples that showed a significant P-value (<0.1) or biologically relevant effect size (>0.3) for the Year x Site interaction. Samples in 2017 were collected before the restoration works, whereas samples in 2018 were collected after the restoration works.
Figure 12: Common taxa from kick-net samples that showed a significant P-value (<0.1) or biologically relevant effect size (>0.3) for effect of Site, but without a Year x Site interaction. Samples in 2017 were collected before the restoration works, whereas samples in 2018 were collected after the restoration works.
Figure 13: Common taxa from kick-net samples that showed a significant P-value (<0.1) or biologically relevant effect size (>0.3) for effect of Year, but no Year x Site interaction or effect of Site (Oligochaeta and Physa spp.), and taxa that showed no interaction for any of the predictor terms (Oxyethira spp. and Nematoda). Samples in 2017 were collected before the restoration works, whereas samples in 2018 were collected after the restoration works.

2.3.4 Surber common macroinvertebrate taxa

Of the 32 taxa found in the Surber samples, ten taxa were counted as common (Table 4). The top five taxa, found in every Surber sampling site, were Oligochaetes (40.6%), Chironomidae (20.6%), Potamopyrgus antipodarum (11.4%), Hudsonema spp. (10.1%), and Oxyethira spp. (4.7%). These are the same top five taxa as in the kick-net samples. The remaining taxa were Amphipoda (3.5%), Physa spp. (2.2%), Austrosimulium spp. (1.4%), Ostracoda (1.3%) and Hydra spp. (1.3%). Together these ten taxa made up 97.0% of the total number of individuals found within the Surber samples.
For the Surber sampling data, the five invertebrate taxa that had a Year x Site interaction were Amphipoda, Chironomidae, *Potamopyrgus antipodarum*, *Hydra* spp. and *Physa* spp. (Table 4). Amphipoda had a significant P-value and strong effect size for Year x Site, Site and Year. Before restoration, absolute Amphipoda abundance was higher at impact sites than at control sites. Abundances decreased to zero at both site categories after the restoration works, but as the decrease was greater at impact sites; this implies that the restoration works had a negative effect on Amphipoda (Fig. 14). Chironomidae had a significant P-value and strong effect size for Year x Site, and a strong effect size for Year (Table 4). Abundance of this taxon increased between years at control sites but decreased at impact sites (Fig. 14), implying that the restoration works had a negative effect. Further, on average more Chironomidae were found in 2018. *P. antipodarum* had a significant P-value and strong effect size for Year x Site, Site and Year (Table 4). *P. antipodarum* abundance started low at control sites and high at impact sites. After restoration, abundance increased at impact sites but more at control sites, implying that the restoration works had a negative effect for this taxon (Fig. 14). *Hydra* spp. had a significant P-value and strong effect size for Year x Site and Year, and a strong effect size for Site (Table 4). *Hydra* abundance started low at both site categories. After restoration, abundance greatly increased at control sites but only moderately at impact sites, implying the restoration had a negative effect on *Hydra* (Fig. 14). *Physa* spp. had a moderate effect size for Year x Site and a significant P-value and strong effect size for Year (Table 4). *Physa* abundance also started low in both site categories. After restoration, abundance greatly increased at impact but even more at control sites, implying that the restoration had a negative effect on *Physa* spp. (Fig. 14).

*Hudsonema* spp., *Austrosimulium* spp., Oligochaeta and *Oxyethira* spp. all had a significant P-value or relevant effect size for Site (Table 4). *Hudsonema* spp. had a significant P-value and strong effect size for Site and a strong effect size for Year. *Hudsonema* was more common at impact sites and abundance in both site categories decreased about equally between years (Fig. 15). *Austrosimulium* spp. had a strong effect size for Site and a significant P-value and strong effect size for Year (Table 4). *Austrosimulium* abundance was higher at control sites and lower in 2018 (Fig. 15). Oligochaeta had a strong effect size for Site and a significant P-value and strong effect size for Time (Table 4). Oligochaeta abundance was slightly higher at control sites and increased about equally in both site
categories between years (Fig 15). *Oxyethira* spp. had a moderate effect size for Site with higher abundance at control sites (Fig. 15).

Ostracoda was the only invertebrate taxon with a significant P-value and strong effect size for Year, but no Year x Site interaction or overall differences between sites (Table 4). Ostracoda abundance was higher in 2018 in both site categories (Fig. 15).

**Table 7**: P-values and partial eta squared values from a one-way repeated-measures MANOVA of the 10 common invertebrate taxa from the Surber sampling data. The taxa are ordered so all significant P-values (P < 0.10) and biologically relevant effect sizes (partial eta squared > 0.3) for Year x Site are first, then taxa that showed an effect of Site but no Year x Site interaction. Then taxa that showed effect of Year but no Year x Site interaction or effect of Site, and last, taxa that showed no relevant relationships with any of the predictor terms. ETA = effect size (partial eta squared). All P-values < 0.10 and effect sizes > 0.30 are printed in bold.

<table>
<thead>
<tr>
<th>Surber Year x Site P-value</th>
<th>Surber Site P-value</th>
<th>Surber Year P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Common Taxa</strong></td>
<td><strong>P</strong></td>
<td><strong>ETA</strong></td>
</tr>
<tr>
<td>Amphipoda</td>
<td>0.007</td>
<td>0.937</td>
</tr>
<tr>
<td>Chironomidae</td>
<td>0.012</td>
<td>0.907</td>
</tr>
<tr>
<td><em>Potamopyrgus antipodarum</em></td>
<td>0.014</td>
<td>0.901</td>
</tr>
<tr>
<td>Hydra spp.</td>
<td>0.050</td>
<td>0.771</td>
</tr>
<tr>
<td>Physa spp.</td>
<td>0.307</td>
<td>0.335</td>
</tr>
<tr>
<td><em>Hudsonema</em> spp.</td>
<td>0.980</td>
<td>0.000</td>
</tr>
<tr>
<td><em>Austrosimulium</em> spp.</td>
<td>0.909</td>
<td>0.005</td>
</tr>
<tr>
<td>Oligochaeta</td>
<td>0.508</td>
<td>0.158</td>
</tr>
<tr>
<td><em>Oxyethira</em> spp.</td>
<td>0.409</td>
<td>0.234</td>
</tr>
<tr>
<td>Ostracoda</td>
<td>0.457</td>
<td>0.195</td>
</tr>
</tbody>
</table>
Figure 14: Common taxa from Surber samples that showed a significant P-value (<0.1) or biologically relevant effect size (>0.3) for the Year x Site interaction. Samples in 2017 were collected before the restoration works, whereas samples in 2018 were collected after the restoration works.
**Figure 15:** Common taxa (Hudsonema, Austrosimulium, Oligochaeta and Oxyethira) from Surber samples that showed a significant P-value (<0.1) or biologically relevant effect size (>0.3) for the effect of Site, but without a Year x Site interaction. Also, taxa (Ostracoda), that showed a significant P-value (<0.1) or biologically relevant effect size (>0.3) for the effect of Year, but no Year x Site interaction and no effect of Site. Samples in 2017 were collected before the restoration works, whereas samples in 2018 were collected after the restoration works.
2.4 Discussion

2.4.1 Community-level macroinvertebrate indexes

My first hypothesis was not supported by the results of this study, as no invertebrate community indexes were shown to be negatively affected by the restoration works. In fact, all four EPT taxon richness indexes (EPT Richness, % EPT Richness, EPT Richness without Hydroptilidae and % EPT Richness without Hydroptilidae) from both the kick-net and Surber data, as well as total taxon richness from the kick-net data, implied that they were positively affected by the restoration works. The remaining six invertebrate community indexes (MCI, QMCI, EPT Abundance, EPT Abundance without Hydroptilidae, % EPT Abundance and % EPT Abundance without Hydroptilidae) were unaffected by the restoration works.

The EPT richness and percentage richness indexes did not decrease as much at impact sites as they did at control sites from before to after restoration, implying that the restoration works prevented the EPT richness/percentage richness from dropping as much as seen in control sites. However, it is possible that EPT richness decreased more at control sites because initially there were more EPT taxa to lose, as after restoration EPT richness scores were generally about the same in control and impact sites for all four EPT richness metrics from both kick-net and Surber data. Moreover, I detected no effect of the restoration works on any of the four EPT abundance indexes, which shows that only presence or absence of the EPT taxa was affected, not the number of individual EPT taxa. Data from the kick-net samples implied that the restoration works had a positive effect on total taxon richness, which increased in impact sites while it stayed the same in control sites. This community-level pattern was the only one that differed to the Surber data, which may be a result of the kick-net method collecting a higher invertebrate taxon richness per sample (discussed further below in the section 2.4.3).

MCI and QMCI for soft-bottomed streams both showed no effect of the restoration works. Nevertheless, their values across the two sampling years are worth commenting on. On average, the samples collected from Waituna Creek using both sampling methods in 2017 ranged between fair (MCI = 80–99, QMCI = 4.00–4.99) and poor ecological quality (MCI < 80, QMCI < 4.00), this decreased to only poor quality samples found in 2018. This implies that
Waituna Creek as a whole declined from “probable moderate/severe pollution” in 2017 to “probable severe pollution” in 2018 (Stark & Maxted 2007a). This range of MCI values is consistent with the MCI and QMCI data collected by Environment Southland since 1997. I think the MCI score decline across the catchment between the two years is probably the result of the drought that occurred prior to sampling in 2018 (further discussed in the next paragraph). The generally poor quality of Waituna Creek is very likely due to the agricultural land use (Thompson & Ryder 2003; Cosgrove 2011; Robertson & Funnell 2012), which also has been shown to negatively affect other, similar lowland streams throughout Southland (Hamill & McBride 2003; Monaghan et al. 2007) and more generally in New Zealand (Matthaei et al. 2006; McDowell & Wilcock 2008; Wagenhoff et al. 2011; Scarsbrook et al. 2016). The poor condition of Waituna Creek gives one potential reason why most of the invertebrate community-level metrics where unaffected by the restoration works. The disturbance caused by the restoration project would have mainly caused an increase in suspended fine sediment and shifting of the stream bed in places where the digger crossed the creek. Streams draining agricultural land are often faced with increased fine sediment loading (Matthaei et al. 2006; Scarsbrook et al. 2016), and Waituna Creek is also likely, naturally a soft-bottom stream (J. Clapcott, Cawthron Institute, personal communication 13/8/18) (Depree et al. 2017). Therefore the creek and its invertebrate community are already exposed to periods of high amounts of suspended find sediment during rain events (Matthaei et al. 2006; Stark & Maxted 2007a; Depree et al. 2017). This means that the invertebrate community is likely fairly resistant to disturbances similar to those caused by the instream restoration works.

From December 2017 to January 2018, there was a drought which affected many regions of New Zealand including Southland (NIWA 2018). During this drought, Environment Southland measured the lowest flow on record (0.027 m³/s on 31st January) for Waituna Creek (at Marshall Road) since recording began in 2001 (Environment-Southland 2018). During these extended periods of low flow, Waituna Creek would have shrunk, decreasing the habitat availability, which could have left some individuals stranded in dried-up areas resulting in their death (Boulton 2003; Lake 2003; Everard 2010). As water flow and volumes decreased, the water temperatures of Waituna Creek began to rise. The water temperatures recorded at Marshall Road often rose above 20°C and reached over 25°C on some days in January.
2018 (Environment-Southland 2018). Such high temperatures may have been lethal for some stream invertebrates (Quinn et al. 1994; Caruso 2002; Lake 2003). Consequently, drought conditions such as this would have likely had a negative effect on the invertebrate communities living within Waituna Creek, in both the control and impact stretch (Caruso 2002; Boulton 2003; Lake 2003; Reich & Lake 2015). The findings of my study are in line with this interpretation as all invertebrate community indexes, with the exception of total taxon richness, from both the kick-net and Surber data, decreased from 2017 (before the drought) to 2018 (after the drought) in both site categories. Consequently, these negative effects may have overshadowed any additional negative effects that the restoration works might have had on the invertebrate community. This gives a possible reason why most invertebrate metrics showed no effect of the restoration work. However, several of the common taxa did show an effect of the restoration works, which will be further discussed under the heading ‘Common invertebrate taxa’. The four EPT richness indexes which showed “positive” effects of the restoration still generally decreased from 2017 to 2018. As the control site had more taxa to lose (as discussed above), it may only appear that the restoration works had a positive effect on these metrics, when in fact this apparent effect may be a result of pre-existing differences between the control and impact sites.

Due to the limited number of study sites and samples collected across time within Waituna Creek, the statistical power of this study is low. Low statistical power means that only large effects can be detected, so to account for this the significance level alpha was increased from 0.05 to 0.1 to help avoid type II errors (Quinn & Keough 2002). Note that some of the patterns did not have a significant P-value, but were based on effect sizes above >0.3. Such moderate or large effects were interpreted as biologically relevant using a conservative approach (Nakagawa & Cuthill 2007). However, despite adjusting the significance level and interpreting all moderate and large effects, due to the small sample size of this study it is still possible that there are effects of the restoration works that remained undetected.

To summarise this section, the majority of invertebrate community indexes (5 of 9) showed no Year x Site interaction, so no effect of the restoration works. This is likely because the invertebrate community living within Waituna Creek is already quite resistant to the type of disturbance cause by the restoration works. The four EPT taxon richness metrics did show a positive Year x Site effect; however, with the strong over shadowing effect of the drought in
summer 2017/2018 and the low statistical power of the study, it is hard to say that the restoration works were the cause of the positive trends seen.

2.4.2 Common invertebrate taxa

There is some evidence to support my second hypothesis that the pollution-intolerant invertebrate taxa would decrease in abundance in impacted sites due to the short term impact caused within the creek by the restoration works. For the Surber data, five taxa from a total of 10 common taxa, had a Year x Site interaction (Amphipoda, Chironomidae, Potamopyrgus antipodarium, Hydra spp. and Physa spp.). All five of these taxa showed a negative Year x Site interaction, which implies that the restoration works, had a negative effect. For the kick-net data, five taxa of 14 common taxa showed Year x Site interactions. Three of the five taxa showed a negative Year x Site interaction (Xanthocnemis zealandica, Ostracoda and Amphipoda), while the other two showed positive Year x Site interaction (Austrosimulium spp. and Sphaeriidae). Although less clear, this overall still implies a negative effect of the restoration works.

A recent research project funded by the Ministry for the Environment (final report by Clapcott et al. 2017) aimed to develop new stressor-specific invertebrate metrics for stream biomonitoring in New Zealand. This project, included designing a fine sediment metric based on the response of individual stream invertebrate taxa to increased levels of fine sediment, using statistical models supplemented by leading expert opinions. Of the nine taxa that showed a Year x Site interaction in my study, three taxa (Chironomidae, Sphaeriidae and Austrosimulium spp.) have been classified in this metric as responding to deposited fine sediment (Clapcott et al. 2017). In the new metric, Chironomidae was labelled as ‘decreaser’, implying that they are sensitive to increased sediment cover. In line with this classification, the Surber data in my study indicated that Chironomidae were negatively affected by the restoration works. Sphaeriidae and Austrosimulium spp. were both labelled ‘increasers’ in this new metric, implying that increased sediment cover is favourable to them. Supporting this classification, the kick-net data in my study indicated both taxa had a positive effect of the restoration. Consequently, comparing the trends of these three taxa...
from my study to the metric designed by Clapcott et al. (2017), lends more weight to the interpretation that the Year x Site interactions for these taxa were caused by an increase in sediment cover as a result of the restoration works. However, one should note this interpretation is only based on three of the nine taxa with Year x Site interactions and this sediment metric is still very new so its usefulness for New Zealand streams needs to be determined for many more independent datasets such as mine.

The 2017/2018 drought must also be taken into consideration when interpreting the response patterns of the taxa with Year x Site interactions. However, as the restoration works and the 2017/2018 drought are confounded in the study design, the two factors cannot be separated. When looking at difference between years, Xanthocnemis zealandica, Oecetis spp., Oligochaetes, Ostracoda, Sphaeriidae, Potamopyrgus antipodarum, Physa spp. and Hydra spp. all showed an increase in abundance in both site categories for both kick-net and Surber data. These eight taxa are also commonly found in slow-flowing streams and in stream pools (Quigley 1977; Hawking & Smith 1997; Winterbourn et al. 2006), implying that the low flows caused by the drought conditions would have been favourable to them. By contrast, Hudsonema spp., Austrosimulium spp. and Amphipoda all showed a decrease in abundance between sampling years in both site categories for both kick-net and Surber sampling. Hudsonema spp. is commonly found in faster-flowing rivers, while Austrosimulium spp. are commonly found in cooler, well-shaded streams (Winterbourn et al. 2006), implying that both taxa should be negatively affected by low flows and high water temperatures. Amphipoda, which consisted of the two genera Paraleptamphopus spp. and Paracalliope spp., are known to be found in slow-moving waters (Fenwick 2001). However, Paracalliope spp. was found to have only a moderate thermal tolerance by Quinn et al. (1994), so it is likely that both genera were negatively affected by the consistently high water temperatures during the drought. Chironomidae is the only common taxon which showed a different trend between years for each sampling method. The kick-net method showed a clear decrease in abundance between years for both site categories, whereas the Surber method showed an overall slight positive trend where abundance increased in control and dropped in impact sites. All the common taxa that were affected by the restoration works also showed a significant change between years, which was likely caused by the drought-induced low flows and high water temperature (Lake 2000; Boulton 2003; Lake 2003), as
discussed above. It is likely that the effects of the drought were stronger than the effects of the restoration works, which would have potentially overshadowed any detectable effect of the restoration works. Also, the high water temperatures caused by the drought would have likely been worse in the impact sites, as there was very little riparian cover to shade the stream compared to the control sites (Rutherford et al. 1999; Sweeney et al. 2004; Holmes et al. 2015). This makes it hard to determine whether the response patterns seen for taxa with Year x Site interactions were caused by the restoration works or not.

In summary, of the nine common taxa that did show a Year x Site interaction, all five taxa from the Surber data and three of the five taxa from the kick-net data implied negative short term effects of the restoration works. The new metric by Clapcott et al. (2017) implied that three of the common taxa may have been responding to an increase in sediment cover potentially caused by the restoration works. However, these taxa may also be responding to the effect of the 2017/2018 drought, which also likely increased sediment cover due to low water flows (Boulton 2003; Lake 2003). It is hard to confirm which driver had a larger impact on the invertebrate taxa, as due to the study design, the effects of the restoration works and the drought (whose occurrence was beyond the control of the study) are confounded and cannot be separated. However, I would argue that the drought probably had the larger effect based on the points discussed above, but this does not mean the restoration works had no effect.

2.4.3 Differences between invertebrate sampling methods

My third and final hypothesis was that kick-net sampling would collect a larger richness of invertebrates per sampled site and therefore should be more sensitive to changes in the invertebrate community richness than Surber sampling. When looking at the mean taxon richness per sample, it was indeed higher for kick-net samples than for Surber samples. Moreover, the kick-net sample data detected an effect of the restoration works on total taxon richness (which happened to be positive), whereas the Surber samples did not.

Kick-net samples did collect a larger number of invertebrate taxa per sample than individual Surber samples. This was likely due to one kick-net sample consisting of 10 different
sampling locations, whereas one Surber sample only consisted of one sampling location. With 10 sampling locations, the kick-net method had a higher sampling effort per sample and collected invertebrates from a wider range of microhabitats than one Surber sample (Li et al. 2001). However, when comparing the invertebrate taxa found per sampling reach, which meant one kick-net sample compared to three Surber samples, both sampling methods collected similar numbers of taxa. Of the total invertebrate taxa found in both years and sites; kick-net collected a total of 34 invertebrate taxa, while Surber collected a total of 32 taxa. Of the total invertebrate taxa found, 29 were collected by both sampling methods. The eight taxa that were only found with one sampling method were often very rare. So for this study, both methods collected fairly similar taxa richness based on the whole sampled data set. By contrast, Tubic et al. (2017) concluded that for their study, the kick-net sampling method collected more taxa, whereas Storey et al. (1991) concluded that Surber samples collected more taxa. It is likely that the differences seen in other studies come down to the exact field methods used, the total sampling effort per site and the specific stream environment the samples were taken from.

For the statistical analyses performed in this study, the kick-net and Surber sampling data showed essentially the same trend (i.e. both had or did-not had a significant P-value or biologically relevant effect size, and if so, both showed the same positive or negative trend) for the Year x Site interaction for all community indexes, with the exception of total taxon richness (10 of 11 metrics = 91%). For the main effects of Year and Site for the community indexes, the two sampling methods showed the same trend 82% of the time. In terms of invertebrate community level monitoring, the two sampling methods gave very similar overall results for the effects of the restoration. Other studies say that in terms of biomonitoring with commonly used macroinvertebrate metrics, the scores calculated from semi-quantitative kick-net and quantitative Surber sampling methods compare well and do not often differ significantly (Storey et al. 1991; Brua et al. 2011; Everall et al. 2017; Tubic et al. 2017). However, it is interesting that in the present study, the one difference in the Year x Site outcomes is total taxon richness, where the kick-net samples detected a difference while the Surber samples did not. It is possible that by collecting more invertebrate taxa per sample, the kick-net was able to better detect a change in the invertebrate community richness than Surber samples.
Regarding the commonly found invertebrate taxa in kick-net and Surber samples, they often showed different trends for the Year x Site interaction (4 of 10 taxa = 40.0%). However, for the main effects of Year and Site for the common taxa, the two sampling methods showed essentially the same trend 90% of the time. The differences in the Year x Site outcomes for the common taxa are likely because the two sampling methods focused their sampling efforts on different microhabitats. Surber sampling was restricted to the fastest flowing habitat within a reach. By contrast, the kick-net method sampled the fast-flowing areas as well as microhabitats the Surber method did not sample, such as soft-bottomed areas, slower-flowing runs, pools and also macrophyte beds. This difference in microhabitats sampled would have affected the abundance of each invertebrate taxa collected (MacArthur 1965; Palmer et al. 1997). For example, *Xanthocnemis zealandica* live in the littoral vegetation of ponds and streams (Winterbourn et al. 2006), while Sphaeriidae live in the soft sediment of slow-flowing streams (Quigley 1977). Both *X. zealandica* and Sphaeriidae were found to have a Year x Site interaction in the kick-net data, but neither taxa were found in high enough numbers to be considered a common taxa of the Surber data. This is probably because both the microhabitats that these two taxa live in were in streambed areas that the Surber sampling method would not collect from. On the other hand, although Surber sampling is restricted to fast-flowing areas, it is a more intense sampling method that is said to provide more precise abundance data within the localised area (Everall et al. 2017). This increased precision may have made the Surber data more suitable for detecting negative effects of the restoration works on invertebrate taxa that commonly live within fast-following riffle areas.

The difference in the variation of microhabitats sampled by each sampling method affected the individual abundance per invertebrate taxa found, which became apparent when looking at the common taxa data. Despite this, the differences in individual taxon abundances were not large enough to affect the outcome of any of the community-level indexes used in my study, except for total taxon richness.

In terms of field work, the effort of sampling one reach (three Surber samples and one kick-net sample), both methods took relatively similar amounts of time. In the laboratory, individual kick-net samples took much longer to process than individual Surber samples as the kick-net samples were very large due to the high sampling effort (10 minutes total per
They also often contained a large quantity of organic material as a result of sampling the macrophytes within the creek. The macrophytes in the samples had to be carefully picked through to extract entangled invertebrates, which considerably slowed the process of identifying the first 300 invertebrates as well as searching for rear taxa. The Surber samples were easier to process as they did not contain as much organic material, a result of sampling fast-flowing areas, which were devoid of large macrophyte growth. However, per sampled reach (three Surber samples and one kick-net sample) the time taken to process the samples collected with both methods in the laboratory was similar.

In conclusion my study shows that, when dealing with invertebrate community data, we can use either kick-net or Surber sampling, and that community-level data collected using either sampling method can be compared. However, for total taxon richness, it should be noted that taxon richness per site is highly dependent on sampling effort (Li et al. 2001). Further, when looking at the abundance patterns of individual common invertebrate taxa, the data generated by the two methods are less comparable. Consequently, when aiming to study specific invertebrate taxa, the sampling method selected should reflect its suitability for the specific taxa targeted for collection.
Chapter 3:

General Discussion
3.1 The effect of the physical restoration works on Waituna Creek

3.1.1 Invertebrate response to the restoration works

Overall, there appeared to be little effect of the physical restoration works on the invertebrate community indexes. This is most likely because the Waituna Creek invertebrate community as a whole is already quite resistant to the possible additional stressors caused by the restoration works which were most likely increased levels of suspended and deposited fine sediment. While the majority of the common invertebrate data did not show any effect of the restoration works, there were still nine taxa which showed mixed effects. Based on the new sedimentation tolerance metric developed by Clapcott et al. (2017), there were implications that three of these taxa were being affected by increased sediment cover, a likely effect of the restoration. A possible reason there was no detectable effect of the restoration works on the invertebrate community indexes and most of the common invertebrate taxa is that the effect of the 2017/2018 drought may have over-shadowed the effect of the restoration works. This drought appeared to be a main factor that affected the invertebrates in Waituna Creek, causing considerable changes at the invertebrate community level and to common invertebrate taxa.

3.1.2 Current state of the Invertebrate community health of Waituna Creek

It is clear that Waituna Creek is no longer a natural, un-impacted stream and that the current invertebrate community has been degraded. Nevertheless, it is likely that even before any human impact the Waituna Catchment would have had a relatively low-scoring MCI, although using the soft-bottomed MCI does help alleviate this problem. Waituna Creek was naturally a lowland, soft bottom stream, which are often dominated by taxa such as worms, snails and Chironomids (Collier et al. 1998; Stark & Maxted 2007a). However, this is hard to prove as there is little published data on macroinvertebrate communities in minimally impacted soft-bottomed streams, due to the high degree of human modification of lowland areas in New Zealand (Stark et al. 2001). It is also hard to assess which specific stressors are affecting the invertebrate taxa in Waituna Creek, as we lack indexes that look
in depth at Chironomidae and Oligochaetes, the two main taxon groups dominating the invertebrate communities in soft-bottom streams.

3.1.3 Potential future success of restoration on the invertebrate community

Once the physical restoration work is finished, normally invertebrates would need to recover from the stress caused by the restoration work (Li et al. 2016). While in the present study it appears that the invertebrate community of Waituna Creek did not change because of the restoration works based on the community-level metrics, looking at the common taxa, it becomes more apparent that the abundances of several taxa have been affected. This means the invertebrate community will have to recover before they can show improvement. It is also likely that the drought will slow the restoration efforts as the invertebrates within Waituna Creek and the surrounding streams will have to recover from the disturbance before the invertebrate community within the impact stretch can improve (Reich & Lake 2015).

As my samples were collected shortly after the Waituna Creek restoration works, it was too soon to see if the addition of the wooden structures had any effect on the invertebrate community. In other studies, adding log structures to help restore streams has been shown to have positive effects on stream bed and bank stability, and to increase the diversity of the depth and flow velocity profiles (Kail et al. 2007; Lester & Boulton 2008; Miller et al. 2010). While wood additions to streams often show a positive effect, there are some situations in which they might be less beneficial (Lester & Boulton 2008). For example, streams that have severe channel incision, like Waituna Creek, are less likely to be positively affected by the addition of wood logs, as they might not be sufficient to stabilize the banks causing structures to fail (Larson et al. 2001; Shields Jr et al. 2006). Waituna Creek had quite steep banks before the restoration work; however, these were widened as a part of the restoration project so this should not affect the wood installations.

The potential future effect of wood installation on the invertebrate community of Waituna Creek looks positive, as log structures added to agricultural streams have been shown to increase invertebrate diversity in the USA (Johnson et al. 2003) and Australia (Lester et al. 2008).
2007). In New Zealand, a study done by Collier et al. (1998) in 20 lowland, soft-bottomed Waikato streams found that percentages of mayflies, stoneflies, and caddisflies were higher on submerged wood compared to macrophytes.

3.1.4 Potential macroinvertebrate recolonisation of Waituna Creek

Assuming the Waituna Creek ecosystem does improve to the extent where new, pollution-sensitive invertebrate taxa are able to recolonise, the colonising taxa must be able to disperse to the restored areas (Mackay 1992; Bilton et al. 2001; Li et al. 2016). As there is a slightly healthier invertebrate community upstream (in the control stretch and further upstream), this will help invertebrate taxa colonise the downstream restoration site by drifting with the river flow (Mackay 1992; Bilton et al. 2001; Schriever et al. 2007). However, the upper reaches of Waituna Creek still have a relatively poor invertebrate community (based on MCI values (Land-Air-Water-Aotearoa 2018)). Therefore, for a greater increase in invertebrate community health, the colonisers would have to come from other source populations via the aerial dispersal of adults (Bilton et al. 2001; Milner et al. 2008; Parkyn & Smith 2011).

A potential source population for aerial colonisation is the Mataura River, which has a fair MCI score and a descent percent EPT richness compared to Waituna Creek (Land-Air-Water-Aotearoa 2018). However, this river is located about 15-km east of the Waituna Creek restoration site and aerial colonisation is limited by dispersal barriers and distance between source populations and restoration area (Lake et al. 2007; Tonkin et al. 2014; Kitto et al. 2015). A study by Tonkin et al. (2014) showed there is a gradual decline of colonisation over 10-km and suggested that the most important range for source populations was within the first kilometre from the restored site, although this trend was highly variable between individual invertebrate taxa. The area between the Waituna Creek restoration site and the Mataura River contains multiple small streams, which could act as stepping stones for aerial colonising invertebrate taxa (Saura et al. 2014; Tonkin et al. 2014). However, these stepping-stone habitats would have to also be suitable to house the dispersing invertebrate taxa, which might be a limiting factor for the degraded Waituna Catchment.
Assuming that new sensitive invertebrate taxa are able to colonise, the time it takes to see a change in the MCI score will mainly depend on dispersal constraints (Quinn et al. 2009b; Parkyn & Smith 2011; Wright-Stow & Wilcock 2017). Quinn et al. (2009b) and Jowett et al. (2009) found significant improvements in MCI scores 6–8 years after riparian fencing and planting in, small hill-country agricultural streams, where dispersal constraints were low. However, it is more likely that the Waituna Catchment has medium to high dispersal constraints due to the relatively poor upstream invertebrate community and the distance and fragmentation between the source populations and the restoration site (Parkyn & Smith 2011). This means that the MCI score will likely be slow to increase and it may take more than 20 years to see a significant improvement (Parkyn et al. 2003; Parkyn & Smith 2011; Wright-Stow & Wilcock 2017). It is also possible that while the habitat heterogeneity is increased by the restoration works, the water quality remains poor due to continued agricultural stressors, preventing dispersing invertebrates from settling within Waituna Creek (Lepori et al. 2005; Palmer et al. 2010).

3.2 Scope for further research in Waituna Creek

3.2.1 Further monitoring work in Waituna Creek

Surber sampling is suited for biomass/abundance monitoring in localised places within the stream and, in this study, detected more negative effects of the restoration works on the common taxa. By contrast, the kick-net method is a more time-efficient way to monitor the general invertebrate community within a larger stretch of the stream. For the restored areas in Waituna Creeks, the kick-net method allowed sampling areas such as the instream macrophytes and the newly added woody debris, which the C3 Surber protocol used in this study (a quantitative sampling method for hard-bottomed streams, designed by Stark et al. 2001)) did not. These microhabitats are important parts of the stream and can house important taxa which should be monitored (Collier et al. 1998). Consequently for the continued invertebrate community-level biomonitoring of Waituna Creek, I recommend that the kick-net sampling method should be used, as it samples a wider range of microhabitats,
including the new wooden features added by the restoration works. If researchers are focusing more on abundance data of individual taxa, Surber samples may be more appropriate, but it must be noted that Surber data may give misleading abundances of invertebrates that do not live within fast-flowing riffles, such as *X. zealandica* and Sphaeriidae, which were discussed earlier.

For Waituna Creek, I further recommend that MCI and EPT indexes should be continually used as they are commonly implemented through New Zealand. MCI for soft-bottom streams should be used, as the creek naturally has a high surface cover of sediment. As for whether to include or exclude Hydroptilidae for EPT scores, in this study the exclusion of Hydroptilidae did lower the average EPT score due to the high presence of *Oxyethira* spp. in Waituna Creek. However, the presence or absence of Hydroptilidae did not change the overall conclusion from the EPT score results. Therefore, for monitoring of Waituna Creek using EPT indexes, Hydroptilidae could be either included or excluded, as long as the approach is kept consistent.

3.2.2 Potential future restoration work within the Waituna Creek catchment

The restoration work done for this project in Waituna Creek has increased stream structural heterogeneity, which will in turn affect stream bed and bank stability, increase water flow diversity and create habitat diversity. However, the relatively small scale of the restoration will probably limit the improvement of the stream ecosystem (Wohl *et al.* 2005; Pan *et al.* 2016). Therefore, the restoration works should be extended to restore a larger proportion of Waituna Creek.

Although the restoration project has restored the physical aspects of Waituna Creek, the creek’s poor water quality has not been resolved, and this may override the effect of the additional instream habitat diversity (Lepori *et al.* 2005; Lester & Boulton 2008; Palmer *et al.* 2010). While it is unrealistic to completely restore Waituna Creek to a natural state (Grimm *et al.* 2008), we should aim to reduce the impacts humans are still having on the Waituna Catchment, especially given the high conservation status of the Waituna Lagoon. The water quality problem is a catchment scale issue due to runoff and groundwater leaching from
agricultural land use within the catchment (Thompson & Ryder 2003; Rissmann et al. 2012; Robertson & Funnell 2012). Nutrients and fine sediment are both major threats, particularly to the Waituna Lagoon, so it is important that their continued inputs are reduced (Thompson & Ryder 2003; Rissmann et al. 2012; Robertson & Funnell 2012). The now installed riparian vegetation will help to prevent surface runoff, which carries nutrients and fine sediment, from entering the creek which could help increase water quality (Niyogi et al. 2007a; Wilcock et al. 2009; Collins et al. 2013). However the restoration efforts will always be limited unless the source of the pollutants, being the intensive agriculture, is dealt with at the catchment scale (Wohl et al. 2005; Alexander & Allan 2007; Palmer et al. 2010).

3.2.3 Transplanting invertebrate species

A relatively novel idea to help the macroinvertebrate community recolonise the restored areas of Waituna Creek is the transplanting of aquatic insect species. Macroinvertebrate reintroduction is a promising tool to restore the natural biodiversity in freshwater ecosystems and although it is not greatly implemented at present, it will likely be applied more frequently in the future (Jourdan et al. 2018). Macroinvertebrate reintroduction would help targeted species recolonise the restoration sites faster and more efficiently by overcoming the previously discussed dispersal constraints within the Waituna Catchment.

Careful planning must be taken to pick an appropriate target species, as it should be one that has gone locally extinct and would help restore the natural biodiversity if brought back (Jourdan et al. 2018). However, due to the lack of published data on macroinvertebrate communities in minimally impacted soft-bottomed New Zealand streams (Stark et al. 2001), deciding on a target species would be a challenge. Approximately one third of the published studies on freshwater macroinvertebrate reintroduction fail (Jourdan et al. 2018). Therefore, for a successful reintroduction of the target species, the main abiotic and biotic factors influencing the stream ecosystem within Waituna Creek must be considered. It is likely that more restoration work will be needed at the catchment scale before any transplanting of invertebrates. However, once all the specific habitat requirements of the
target species are met and all factors are carefully considered, then successful reintroduction of aquatic insects is possible (Hoffmann 2000; Jourdan et al. 2018).

### 3.2.4 Chironomids and oligochaetes as metrics for soft-bottomed streams

In soft-bottom stream systems such as Waituna Creek, much of the taxonomic diversity occurs within the typically low-scoring taxa Chironomidae and Oligochaeta (Collier et al. 1998; Stark & Maxted 2007a). As in most other ecologically-focused studies on stream invertebrates, I did not attempt to taxonomically resolve the potential diversity of the chironomids and oligochaetes. While identifying chironomid larvae below the family level is achievable once a person has been trained, the steep learning curve and the extra time required put most researchers off (Heiri & Lotter 2010; Nicacio & Juen 2015). Oligochaete taxonomy is considerably more difficult, and this taxon is therefore almost always only identified to the subclass (Brinkhurst & Kennedy 1965; Learner et al. 1978).

The two large taxon groupings of Chironomidae and Oligochaeta could contain taxa that range from very sensitive to very resistant to different adverse conditions, which would make useful indicator species if we would know enough about them (Saether 1979; Heiri & Lotter 2010). There has already been work done with chironomids to be used as indicators of specific stressor such as sediment (Carew et al. 2007; Beermann et al. 2018) and temperature (Heiri & Lotter 2010). To a far lesser extent, there has also been work done with Oligochaete in the family Naididae (Brinkhurst & Kennedy 1965; Learner et al. 1978), but still Oligochaete remain largely under-researched.

To fully understand soft-bottomed stream communities, greater effort needs to be put into resolving the diversity of Chironomidae and Oligochaeta. This means we need to overcome the current challenges of identifying and creating new metrics for these two taxon groupings, and in doing so, we would open up new opportunities to monitor specific stressors related to freshwater health. There are promising advances in genetic approaches that could help quickly and accurately identify similar looking species (Carew et al. 2007; Thomsen et al. 2012; Mächler et al. 2014). Metabarcoding of invertebrate samples has already proved useful in identifying different chironomid species and operational taxonomic
units (OTUs) (Carew et al. 2007; Beermann et al. 2018). Bulk-sample metabarcoding can identify a whole range of different aquatic taxonomic species/OTUs from a standard stream sample, with much higher taxonomic resolution than traditional identification methods (Mächler et al. 2014; Macher et al. 2018). Metabarcoding of environmental DNA from water samples is another promising technique which could provide a rapid, non-destructive way to identify individual species within a stream ecosystem (Thomsen et al. 2012; Ji et al. 2013; Mächler et al. 2014). While these techniques give different results, they hold great potential for future application in biomonitoring and ecological research in running waters (Macher et al. 2018).

3.3 Implications for management of New Zealand streams

3.3.1 Instream restoration work

The physical instream restoration works within Waituna Creek showed some implications of increased sedimentation. Even though Waituna Creek already has a high sediment cover, further increases in sediment can still negatively affect invertebrate communities (Matthaei et al. 2006). This implies that caution should be taken to avoid increasing sediment loading into waterways while using large machinery to restore stream ecosystems.

3.3.2 Implications of continued monitoring of Waituna Creek

With further monitoring of Waituna Creek, lessons can be learned about the effectiveness of woody debris and stream-bank engineering in restoration works of New Zealand lowland streams that have been degraded by agriculture. With continued monitoring, time will tell if restructuring the stream banks and flow, combined with replanting the riparian vegetation to buffer the agricultural runoff, is enough to help restore the highly degraded ecosystem. Otherwise the monitoring will show that more work needs to be done to address the core issue being agricultural land use within the Waituna Catchment.
3.3.3 Invertebrate sampling methods for lowland soft-bottomed streams

The present study has shown that the Surber and kick-net sampling methods used give comparable community-level trends, implying that they are both suitable for monitoring changes in macroinvertebrate community-level indexes. However, the specific stream type and stream environment plays an important role on the efficiency of a macroinvertebrate sampling method. For lowland soft-bottomed streams, such as Waituna Creek, the standard Surber sampling method used for ‘State of Environment’ monitoring in New Zealand is less effective at sampling all microhabitats present within the stream. This standard Surber sampling protocol generally requires the collector to sample only from fast-following riffle areas, ignoring pools, slow runs, bank margins, woody debris and macrophytes. The kick-net sampling method is able to sample all these areas, thus giving a better overview of the whole stream invertebrate biodiversity. Consequently, I recommend that for routine monitoring of macroinvertebrate communities in lowland soft-bottomed streams, the semi-quantitative kick-net sampling method should be used. There should also be development towards incorporating better taxonomic resolution for common soft-bottom invertebrates such as chironomids and oligochaetes.
Reference list


